An evidence review update on the effects of managed burning on upland peatland biodiversity, carbon and water

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Natural England Evidence Review NEER155

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Report details

Authors

Alice Noble[1], David Glaves[1], Pam Leppitt[2], Alistair Crowle[1], David Key[1] and Adam Rodgers[1].

[1] Natural England, [2] Leppitt Associates.

External expert review group

Gareth Clay (University of Manchester), Scott Davidson (University of Plymouth), Chris Evans (UKCEH), Nick Kettridge (University of Birmingham) and Richard Lindsay (University of East London).

Project Manager

David Glaves, Senior Specialist, Uplands, Natural England.

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Executive summary

Peatland ecosystems make important contributions to biodiversity, carbon storage and water provision in the UK and globally. Many UK upland peatlands have been subject to burning for land management purposes, particularly grouse moor management, with the practice increasing over the 20th and early 21st century. Concerns about harmful impacts have led to recent changes in regulation aimed at reducing burning on peatland habitats. The use of burning on peatlands has remained a source of debate and hence an up-to-date overview of new relevant evidence was necessary to inform future policy and practice.

This evidence review updates a review by Glaves and others (2013, NEER004). It considers evidence from 102 studies published since NEER004 relating to the effects of managed burning on upland peatland biodiversity, carbon balance, water quality and hydrology, which were selected following a comprehensive search. Findings have been compared with those from 123 studies in NEER004 to give an updated overview of the whole evidence base. Combined findings of the two reviews have been synthesised into evidence statements, with high-level highlights of key evidence statements given below.

Taken as a whole, the available evidence shows that burning alters the species composition of blanket bog and upland wet heath **vegetation** in at least the short to medium term. This includes a tendency for initial grass and/or sedge dominance, typically followed by an increase in heather *Calluna vulgaris*. This, along with changes in other species (including bryophytes) and vegetation structure can result in a move away from the characteristic vegetation of these peatland habitats. The creation of bare ground following burning has also been observed and this may persist for several years.

Many studies relating to peatland **fauna** focused on breeding birds, and reported various effects of burning depending on species, though it can be difficult to separate the influence of burning from that of predator control carried out as part of grouse moor management. There is also evidence of effects on other faunal groups including invertebrate communities, which are influenced by changes in vegetation and soil characteristics – caused by burning. As with vegetation, these changes may result in a move away from characteristic peatland faunal communities.

Managed burning also affects various aspects of the **carbon** cycle of upland peatlands, with studies showing a large proportion (76–80%) of aboveground carbon stock lost via combustion, followed by gradual re-accumulation over several decades. There is also evidence that the export of dissolved and particulate organic carbon increase after burning, but inconsistent evidence of effects on some other carbon cycle pathways including CO_2 fluxes and on overall carbon balance. For **water**, there is evidence that burning influences various aspects of chemistry and flow, including fluvial carbon export as mentioned above. There is also evidence of increased flow in watercourses draining burned catchments, potentially impacting downstream river levels.

The **severity and frequency** of burning appear to affect outcomes related to vegetation, carbon and water. Meanwhile, relatively few studies investigated interactions between **burning and grazing**, though there was some evidence of effects on vegetation. Regarding the relationship between **burning and wildfire**, there is evidence that out-of-control burns are a cause of wildfire in the UK, particularly in the uplands. There is evidence from other countries and habitats on biomass management by managed burning to reduce wildfire hazard, but limited evidence from the UK peatland context. Variation in burning **extent and frequency** by UK region and year was apparent, with a long-term increase followed by an indication of a recent decrease since 2016. There was also evidence that designated sites and areas of deep peat have been burned at a similar frequency as other areas.

The evidence from 102 recent studies in addition to 123 reviewed in NEER004 gives a significant volume of evidence on which to draw conclusions on the impacts of burning, and many of the evidence gaps identified in NEER004 have been filled. Though there remain some areas where evidence appears inconsistent, this may often be explained by differences in the scale, location or timing of studies.

In conclusion, the evidence base suggests that burning impacts peatlands, and the ecosystem services they provide, via multiple mechanisms, and though recovery is often observed in the short to medium term, repeated burning risks a sustained departure from the characteristic structure and function of these habitats. Overall, this is consistent with the summary and conclusions of NEER004.

Contents

1. Introduction

Upland peatlands

1.1 Peatlands are among the most extensive terrestrial, semi-natural habitats in England, particularly on unenclosed land in the uplands generally referred to as moorland[1], but they also occur more locally in lowland areas (Natural England, 2010). They support a range of nationally and internationally important mire habitats/vegetation types (bog, fen and wet heath) and associated species (Ratcliffe & Thompson, 1988; Thompson and others, 1995; Averis and others, 2004; Natural England, 2008; Littlewood and others, 2011; Lindsay & Clough, 2017; Crowle and others, 2024, in press). It is widely recognised that they provide a range of important provisioning, supporting, regulatory and cultural ecosystem services (Natural England, 2009, 2010; Van der Wal and others, 2011; Alonso and others, 2012, 2021; Dunn and others, 2021; Gregg and others, 2021). Peatlands have been subject to a range of land management practices and other impacts, including managed burning, which has left much of the UK and English resource in a modified or degraded state (para. 1.6). As a result, peatlands face a number of conservation, environmental and land management challenges. This is particularly the case in relation to understanding the effects of land management practices and other impacts on biodiversity and associated ecosystem services.

Previous Natural England upland evidence reviews

1.2 The challenges set out above led to Natural England's Uplands Evidence Review Programme which drew together the best available science and evidence on the effects of key land management activities on upland biodiversity and ecosystem services. This focused on five key upland land management issues:

- The effects of managed burning on upland peatlands (Glaves and others, 2013, hereafter NEER004);
- Restoration of degraded blanket bog (Shepherd and others, 2013);
- The impact of tracks on blanket peat (Grace and others, 2013);
- The impact of moorland grazing (Martin and others, 2013); and
- Management of upland hay meadows (Pinches and others, 2013).

An assurance report was also produced which provided an overview across the five topics (Galbraith and others, 2013).

[1] In England moorland is defined by the Moorland Line (ADAS, 1993) in the Severely Disadvantaged Areas (SDA), generally above 250 m above ordnance datum.

1.3 The Uplands Evidence Review programme provided a basis for advice and decisions on management of the uplands. Although consideration of other relevant information, such as social and economic factors, landscape and archaeology/historic environment, is an important part of the process that Natural England uses to develop advice, the focus of the Uplands Evidence Review Programme and this burning evidence review update is on biodiversity and ecosystem services.

1.4 Since the Uplands Evidence Review Programme was completed, a series of additional Natural England evidence reviews have been published that are relevant to the wider uplands: the ecology of heather beetle *Lochmaea suturalis* (Gillingham and others, 2016a); burning and other management options for the control of heather beetle (Gillingham and others, 2016b); the historic peat record and implications for the restoration of blanket bog (Gillingham and others, 2016c); the effects of trees and scrub on upland ecosystem services and biodiversity (Gold, 2019); the causes and prevention of wildfire on heathlands and peatlands (Glaves and others, 2020b, hereafter NEER014); the influence of recreational activity on upland ecosystems (Gilchrist and others, 2023); and the impacts of vegetation cutting on peatlands and heathlands (Moody & Holden, 2023). Of these, Gillingham and others (2016b,c) and Glaves and others (2020b) feed directly into this review update.

Managed burning and other impacts

1.5 Burning is widely used in the UK as a tool for the management of a range of moorland and other upland habitats, principally to create new growth for livestock grazing, and age and structural diversity (particularly of heather[2] *Calluna vulgaris*, hereafter *Calluna*), for game (red grouse *Lagopus scotica*[3]) management in the uplands (Defra, 2007). It may also be used to address other objectives including conservation management of some habitats including dry heath, gorse scrub and reedbeds, some species, and as a contribution to targeted wildfire risk management (Chapman and others, 1989; Defra, 2007, 2008b, Hancock and others, 2011). However, burning may be damaging to sensitive species and habitats, and in situations such as on steep slopes. This potential for damage is referenced in UK good practice burning codes and other guidance (Defra, 2007, 2008a; WAG, 2008a,b; NatureScot, 2021; DARDNI, no date), in upland condition assessment guidance (JNCC, 2009) and in burning legislation (see Appendix 5).

1.6 Much of the UK upland peatland habitat resource, especially in England and Wales, is degraded, with 'modification' of the characteristic, varied mire vegetation

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[2] English names of plants are given at first mention in the text or tables followed by scientific names which are used thereafter and follow the *Plant Atlas 2020* (Stroh and others, 2023) for vascular plants and the British Bryological Society's *UK checklist of bryophyte names* (Blockeel and others, 2021) and *Species Finder* (BBS, no date) for bryophytes.

[3] Scientific names of fauna are given at the first mention by English name in the text or tables and thereafter English names are used, most applying to birds which follow *The British List* (BOU, 2024).

communities typical of unmodified peatlands (JNCC, 2011; Glaves, 2017; Artz and others, 2019). In some cases, this is to the point that they have been replaced by dominance of single-species stands, particularly *Calluna*, purple moor-grass *Molinia caerulea* (hereafter *Molinia*), hare's-tail cottongrass *Eriophorum vaginatum* and sometimes common deergrass *Trichophorum germanicum* (Critchley and others, 2011a; Defra, 2011; Natural England, 2015; Glaves, 2016; Moors for the Future Partnership, 2017). As a result, the vast majority of designated and undesignated upland peatland habitats are in unfavourable condition, especially in England (Critchley and others, 2011a,b, 2016; Natural England, 2008).This reflects a variety of past and in some cases continuing impacts, including atmospheric pollution, overgrazing, drainage, burning and wildfire (UK Biodiversity Group, 1999; Natural England, 2010; Defra, 2011).

1.7 Prior to the previous burning evidence review (NEER004), concerns were expressed by some about the possible effects of managed burning on biodiversity and wider associated ecosystem services, especially carbon balance, and water quality, distribution and flow of upland peatlands (for example, Crowle, 2007; Yallop and others, 2009; IUCN, 2011; Worrall and others, 2011; Holden and others, 2011, 2012). These concerns were not new but had been articulated in earlier reviews (Mowforth & Sydes, 1989; Coulson and others, 1992; Shaw and others, 1996; Tucker, 2003; Stewart and others, 2004; and Glaves and others, 2005). In addition, The Heather and Grass Burning Code for England (Defra, 2007) included "peat bog and wet heathland... (including blanket bogs, raised bogs, valley bogs or mires, springs and flushes)" in a list of 'sensitive areas' where "there should be a strong presumption against burning". Similarly, previous versions of the code said, "do not burn peat bog and wet moor (flow ground)" (MAFF, 1992); and that "burning should in principle be avoided" on such areas (MAFF, 1984).

The present burning evidence review update

1.8 Both NEER004 and this update are systematic reviews with a methodology defined in Stone (2013). The need for an update review arose for several reasons. Since NEER004, a substantial number of new studies and some new reviews were published on the effects of burning on upland peatlands, for example, Harper and others (2017, 2018) and Holland and others (2022) (also see other recent reviews listed in para. 2.18). This generated considerable debate and the publication of several opinion pieces and forum papers on the effects of managed burning and wider moorland management, for example, Davies and others (2016b) and Thompson and others (2016), followed by a series of responses and further comments (Brown and others, 2016; Davies and others, 2016c,d; Douglas and others, 2016a; Monbiot, 2016; Southerton and others, 2017).

1.9 In addition, a review by Ashby (2020), published as part of a dossier on peatland protection by The Uplands Partnership (2020), specifically evaluated recent (post-NEER004) studies. This was similar to the NEER004 review, for example in scope (addressing the same overarching review question and sub-questions), but there were differences in review methods, particularly the study quality assessment process. The different methods mean that the assessments are not directly comparable, whereas the assessments of studies in this review update follow the same established, previously published methodology as NEER004 (Stone, 2013). This enables conclusions to be drawn based on evidence from NEER004 and the review update together.

1.10 The increasing body of evidence, reviews, reports and discussions since NEER004 led to identification of a need to update NEER004. This was put to a Natural England Science Advisory Committee (NESAC) meeting in September 2020. The committee supported the proposal, which led to the production of this updated review that integrates recent evidence with that contained in NEER004.

Review scope

1.11 The scope of this review update relates to the NEER004 overarching review question:

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

1.12 This review update sought to capture new evidence published in the intervening years focusing on the eight sub-questions listed below which are in some cases slightly revised from those in NEER004. The revisions reflect: (i) new aspects covered in recent (post-NEER004) evidence; or (ii) items covered in NEER004 but not specifically mentioned in the question, and which external expert review group members asked for greater clarity.

1. What are the effects of managed burning on the maintenance and restoration of the characteristic vegetation composition, structure and function of upland peatland habitats?
2. 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?
3. What are the effects of managed burning of upland peatlands on carbon balance?
4. 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 distribution (including hydrology) and flow (including downstream flood risk)?
5. How do differences in the severity, frequency, scale, location and other characteristics of managed burns (including ‘cool burns’) affect upland peatland biodiversity, carbon and water?
6. How does the interaction of managed burning and grazing affect upland peatland biodiversity carbon and water?

7. Is there a relationship between managed burning of upland peatlands and ‘wildfire’ (risk, hazard, occurrence, severity, extent and habitat resilience)?
8. What is the extent, frequency, practice and type of managed burning on upland peatlands (including in relation to designated sites and watercourses)?

1.13 The NEER004 review and this update cover biodiversity maintenance and restoration, including the effects of burning on modified or degraded upland peatland habitats and the effects on carbon balance, soils, and water quality and hydrology. Biodiversity has many different aspects but both reviews focused on upland peatland habitats and their characteristic associated species of flora and fauna, in particular blanket bog, and associated upland habitats on peat soils, including flushes, fens, swamps and wet heath. Characteristic species are those associated with sites that have not been modified or only partly so. Restorability per se was not covered, although it was considered in the Uplands Evidence Review blanket bog restoration topic report (Shepherd and others, 2013).

1.14 This review summarises the evidence base, but it does not make recommendations about how this should be interpreted and applied to Natural England’s working practices and advice. Consideration of other relevant information, such as practicality of implementation (on which social and economic considerations have a bearing), landscape, archaeology and historic environment, and wider ecosystem services is an important part of the process of developing advice but was beyond the scope of both the original review and this update.

1.15 The focus of the review was largely temperate and boreal peatlands (especially blanket bog, but also including other bog, fen and wet heath habitats), their biodiversity (flora and fauna), carbon balance, water (quality and flow), and (managed) burning (also see para. 2.19).

Structure of this report

1.16 Sections 1–3 give introductory and methodological information. Sections 4–11 review recent evidence for each of eight sub-questions. A comparison of findings from NEER004 and the review update is presented in section 12, while a summary of the combined findings and overall conclusion are given in section 13.

1.17 Technical terms and acronyms used in this report are listed and defined or explained in the Glossary.

2. Methods

2.1 The methodology describing the NEER004 review scope, questions, evidence search, inclusion criteria, study type, quality categorisation, and synthesis process is given in NEER004 (Section 2, pp. 5–11). The approach followed the Natural England evidence review guidance (Stone, 2013). This review update followed a similar process with two additional stages. These involved comparing the findings from recent (post-NEER004) studies with those included in the NEER004, from which revised evidence statements were developed from across the studies evaluated in the two reviews, i.e., the combined evidence base. For the specific wildfire sub-question, they were also compared with relevant findings from NEER014.

General principles

2.2 The general principles and approach adopted in the NEER004 review were followed in this update. The process systematically identified available studies providing evidence for the specific questions posed (para. 1.12). A long preliminary list of documents was sifted to ensure that those that were included met with defined criteria, as explained in para. 2.19.

2.3 The 'PICO' framework (see below) was followed which provided a structured approach to formulating review questions and framing the over-arching search strategy (Stone, 2013, Collins and others, 2015), so that inclusion and exclusion criteria could be objectively set. This framework derives from medical reviews and to some extent the terminology reflects this. It comprises the following four elements:

- **Population**: the population/species/habitat of interest, in this instance, upland peatland habitats in the UK, especially England. In some of the studies evaluated in the review, burning on upland peatlands was also compared with its effects on other related habitats, for example, dry heath.
- **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). In some of the studies evaluated, burning was also compared with alternative interventions, for example, cutting, or other impacts, for example, wildfire.
- **Outcome**: the outcomes or effects that are being considered, in this instance, maintenance and restoration of biodiversity and delivery of carbon and water ecosystem services and aspects of them.

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

Development of review scope and questions

2.5 The development of the scope of the original, wider Uplands Evidence Review programme topics (paras. 1.2–1.3) and framing of the sub-topic questions followed a consultation with key stakeholders on the draft scope and questions across the five review topics.

2.6 Responses on the burning topic and sub-topic questions were received from a range of organisations[4]. The external expert review group for the original review[5] also input to the finalisation of the topic scope and questions which were documented in Natural England (2012). A few further minor revisions were made to the sub-questions in the NEER004 report to standardise terminology. Further, generally minor, revisions were made to the sub-questions in this update to reflect some new aspects addressed by recent research and comments made by the external expert review group for this update (p. 3) to improve clarity.

Evidence searches

Search terms and strategy in NEER004

2.7 In the original NEER004 review, evidence searches were conducted using different combinations of relevant search words or terms and wildcard search terms (allowing for alternate spellings and variations on a root word) (NEER004, paras. 2.5–2.7). These were used in a series of 'search strings' that normally included one or more terms from each or several of the PICO categories, normally including the population and intervention (i.e., peatland habitats and burning or related terms) and terms from one or more of the sub-questions.

2.8 The following online databases were searched: CAB Abstracts, Google Scholar, Scirus, Web of Science and Zoological Record. Online publication searches were also undertaken on: British Library EThOS (PhD and MSc theses), Collaboration for Environmental Evidence, Countryside Council for Wales (CCW, now Natural Resources Wales, NRW) library catalogue, Committee for the Promotion and Advancement of Cooperatives (COPAC), Global peatland restoration manual, Natural England library catalogue, water@leeds, PeatNet, Peatscapes and United Utilities Sustainable Catchment Management Programme (SCaMP).

2.9 Potential additional references were also identified through correspondence with academic and research institutes, and other relevant organisations, and from scrutinising relevant reviews/research: Bain and others (2011), Holden and others

[4] The Heather Trust, IUCN UK Peatland Programme, Moorland Association, NFU, North Pennines AONB, Northumberland NPA, North York Moors NPA, RSPB and the SW Uplands Federation.

[5] Richard Lindsay (University of East London), Rob Marrs (University of Liverpool) and Fred Worrall (University of Durham), chaired by Mike Morecroft and attended by David Glaves (both Natural England).

(2011, 2012), IUCN (2011), Worrall and others (2011), Grant and others (2012) and Heinemeyer & Vallack (2012). In addition, there was an open call to interested stakeholders and other organisations and individuals to submit evidence material for consideration as part of the review which resulted in submissions from 11 organisations and three individuals[6].

2.10 This search strategy resulted in a total of 895 references being identified after duplicates were removed. Screening sequentially on title, abstract and full text reduced this to 170 references (NEER004, Table 1, p. 7). These were grouped into 123 individual evaluated studies after references that were considered to relate to the same study were consolidated. Often these covered different time periods or different but related aspects. Thus, studies relating to the same experimental or survey and monitoring setup were treated as a single study unless subsequent references addressed new aspects and/or sites. For example, the routine vegetation monitoring in the long-running grazing and burning Hard Hill experiment at Moor House National Nature Reserve (NNR) was treated as a single study, but non-routine studies addressing other aspects, such as non-routine more specific vegetation outcomes including particular species or groups or biomass, and carbon and water, were treated as separate studies. This judgement was to some extent subjective but considered necessary to avoid duplication and multiple counting of the same study in summarising findings and especially in synthesising evidence statements and allocating strength categories to them. References that documented study methods and/or included additional information or data from that in the main study reference (for example, resurveys over different time periods) and comments and responses about a study were also included as supplementary references. The main study reference and supplementary references for the NEER004 evaluated studies are listed in Table A1.3 in Appendix 1 of this report. Similarly, additional references to those included in NEER004 and this update included from NEER014 in relation to the wildfire sub-question are listed in Table A1.2 in Appendix 1.

Search terms and strategy in the review update

2.11 The search strategy for the review update aimed to identify post-NEER004 references. This involved searching the Scopus online evidence database. This was initially done in February 2021 and then repeated in September 2023 to identify any later publications. A series of search strings were used (an asterisk denotes a wild card search term allowing for several permutations of the word) to search for English-

[6] The following organisations submitted evidence to the original review: the CLA, Exmoor National Park Authority, the Federation of Yorkshire Commoners, GWCT, Moors for the Future, the National Sheep Association, the National Trust, RSPB, United Utilities, the University of Leeds and Yorkshire Water; and the following individuals: Roy Brown, A.E. Peart and Adrian Yallop.

language studies published since NEER004 (post-2011[7]) based on the following simplified initial general search string:

(fire* OR burn*) AND (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland).

2.12 Separate search strings were developed to cover each of the sub-topic questions, except for fire severity, extent and wildfire which were combined, and the PICO categories, as listed in Appendix 2. This was done in part to help identify which sub-questions references were likely to relate to. In practice, the majority of references and studies related to more than one sub-question (para. 2.38). The results from the Scopus searches were downloaded into EndNote reference manager and then a MS Excel spreadsheet for screening.

2.13 The search strategy included a sequence of a further four stages to identify additional references. In the first stage, potential additional references were identified from a corresponding post-NEER004 review by Ashby (2020, para. 1.9) which itself had adopted a three-stage search strategy:

- A search of the Web of Science and Scopus online databases for relevant publications between 2012 and November 2019.
- The identification of additional references included in other reviews and comment/opinion pieces: Brown and others (2015a), Heinemeyer & Vallack (2015, though a 2012 version of this had already been included in the search strategy for NEER004), Davies and others (2016b), Thompson and others (2016), Sotherton and others (2017), and Harper and others (2018).
- Searching and extracting relevant PhD and MSc theses from the EThOS e-theses database on the British Library website[8].

2.14 All of the studies included in Ashby (2020) were included in this update apart from Worrall and others (2013b) which had already been included in NEER004 based on the version published online in 2012. They were, however, consolidated (see para. 2.10) into fewer studies in this update (53 compared with 61 in Ashby, 2020[9]).

2.15 The further three stages of the update search strategy comprised searching and screening for potential additional recent references from various sources in sequence:

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[7] Though NEER004 included nine studies published in 2012.

[8] https://ethos.bl.uk/Home.do.

[9] When Worrall and others (2013b) is excluded.

- Study reports produced, commissioned or supported by Natural England, the other UK country nature conservation agencies, Defra and/or the Scottish Government.
- References included in other recent reviews either not included in the search strategy of Ashby (2020) or published since: Penny Anderson Associates, (2014); Wentworth & Shotter (2019), NEER014 (2020), Belcher and others (2021), Gregg and others (2021), Holland and others (2022), Wentworth (2022) and Tasker & Wentworth (2024).
- References on a spreadsheet list of burning-related publications and grey literature maintained and regularly updated by Natural England upland specialists. This included additional recent publications and other documents submitted to Natural England since NEER004, and further additions suggested by the external and internal review groups for the update.

2.16 In addition, supplementary references relating to the same individual studies, including comments on studies and authors' responses, were identified through the same search strategy process and in some cases from the main study reference and other sources.

2.17 The sub-question on the relationship between managed burning and wildfire was treated differently, given that Natural England had carried out a wider review on the causes and prevention of wildfire on heathlands and peatlands in England relatively recently (NEER014, 2020). Thus, only aspects relevant to the narrower wildfire sub-question considered in NEER004 and this update were searched for, which resulted in nine additional studies. The findings from these more recent studies are compared with those relevant to the wildfire sub-question from NEER014 in Section 12 and the NEER014 studies involved are listed in Table A1.2 in Appendix 1.

2.18 The focus of the searches was on new primary studies, though in a few cases recent publications also provided updated information or comments on studies previously included in NEER004 (and hence are listed in Table A1.3 in Appendix 1). The search strategy also identified a number of additional relevant reviews and proceedings. In most cases these were only used to provide context and to aid interpretation of, and across, primary studies and/or as supplementary references for other studies (Evans and others, 2014; Penny Anderson Associates, 2014; Gillingham and others, 2016a; Meade, 2016; Chapman and others, 2017; GWCT, 2020; Thompson & Wilson, 2020; Gregg and others, 2021; Stafford and others, 2021; Stewart and others, 2021; Armstrong, 2022; Holland and others, 2022; Gilchrist and others, 2023; Moody & Holden, 2023). Two reviews that also reported additional, new analyses and interpretation were included and evaluated (Gillingham and others, 2016b,c).

Screening inclusion and exclusion criteria

2.19 Inclusion and exclusion criteria were used to systematically screen references to include in the review. The criteria used in the review update were the same as for NEER004 (para. 2.13 in NEER004).

2.20 The inclusion criteria were:

- temperate and boreal peatland habitats (especially blanket bog, but also other peatland habitat types: other bog types, fens and wet heath).
- biodiversity (flora and fauna), carbon, water (quality and flow)
- (managed) burning

2.21 The exclusion criteria were:

- dry heath
- mineral soils
- forest/woodland/trees
- tropical/arctic/tundra
- wildfire (unless related to the effect of management burning)

2.22 References relating to other geographical regions, habitats and management interventions were generally excluded unless they were included in peatland studies as comparators, or were specifically relevant to peatland habitats, species, burning management and fire behaviour. Wider moorland studies that included peatland along with other habitats were included where considered relevant.

2.23 Any 2012 studies that were included in NEER004 were excluded. Most of the screening assessments were done by one person, as in NEER004, though where there was uncertainty, the full text was reviewed by a second person. A small number of wildfire studies were included where these: (i) related to the interaction with burning under the wildfire sub-question (Section 10); (ii) made comparisons with the effects of managed burning; and/or (iii) provided relevant information on the effects of differences in fire severity and other aspects of fire behaviour (Section 8) (as was done in NEER004). Similarly, some studies that included other related habitats, for example, dry heath, especially when used as a comparator with similar peatland habitats, for example, wet heath, were included.

Study type, quality and external validity appraisal

Study type

2.24 Each study was categorised by study type (types 1–4, Table 1). A few studies included more than one study type. In particular, some experimental studies also included observational or correlative aspects and were therefore classed as both types 1 and 2 (1,2).

Table 1. Study types.

Type code	Study type and example sub-types
1	Quantitative experimental, e.g., randomised control trials (RCT), systematic reviews of Type 1 studies
2	Quantitative observational or correlative, e.g., non-randomised trials, survey and monitoring, systematic reviews of Type 2 or mixed type studies
3	Qualitative, e.g., case studies
4	Literature and rapid reviews, expert opinion and formal consensus

Study quality

2.25 Following the established Natural England evidence review method process (full details of which are in Stone, 2013, para. 8.9–8.13, pp. 17–18) studies were categorised for quality (or internal validity) against criteria appropriate for different study types. The criteria included aspects of the study area/population, method of allocation to intervention or comparison/control, outcomes and analyses for quantitative studies, and theoretical approach, study design, data collection, trustworthiness, analyses and ethics for qualitative studies/reviews (see para. 2.26 and Appendix 3). Each criterion and the overall assessment of internal and external validity were classified into one of three categories, [++], [+] or [-] based on the extent to which potential sources of bias had been minimised (Table 2). The assessment was done in relation to the effects of burning rather than any wider aspects covered by a study. In a few cases, assessments of the same study varied between different aspects or sub-questions in terms of type and quality (although a single overall classification is given in Table A1.1 in Appendix 1).

Table 2. Study quality (internal validity) categories.

Quality category	Definition
++	All or most of the methodological criteria have been fulfilled. Any criteria not fulfilled or not adequately described are considered very unlikely to alter the conclusions (low risk of bias).
+	Some of the criteria have been fulfilled. Those criteria that have not been fulfilled or not adequately described are considered unlikely to alter the conclusions (moderate 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).

2.26 The quality assessment was done using three different ‘quality checklist’ forms depending on study type: quantitative experimental (study type 1), quantitative observational/correlative (type 2), and in a few cases, qualitative studies (types 3/4) (see Appendix 3). These were based on the example generic forms given in Stone (2013, Appendix 12, pp. 48–60), which were adapted and tailored to this burning review update topic and sub-questions to try to improve clarity and hence consistency of assessments between assessors. In addition to the quality scores, the quality checklists also included additional categories or responses of ‘not reported’ (study reference[s] do not report how a particular criterion/aspect had been considered) and ‘not applicable’ (aspects not applicable to the study design/approach).

2.27 Most study quality assessments were mostly carried out by a single person, though a proportion were assessed by two (30, 29%) or three (3, 3%) people. This was especially done towards the start of the assessment process to enable comparison between assessors scores. Where there were differences, these were discussed and revised scores agreed, which helped calibrate scoring and improve consistency. The vast majority of the assessments were carried out by three individuals (Alice Noble, Pam Leppitt and David Glaves), though a small number were assessed by others. Assessors who were authors of study references did not assess those studies. In a further quality assurance stage towards the end of the review, overall internal and external validity study scores were compared across similar studies and a small number of adjustments were made to improve consistency based on the scores for individual quality checklist criteria (Appendix 3). This particularly applied to external validity (EV) scores and study type or mixed types (more than one type in the same study), which were considered less subjective than the quality (internal validity) scores. A similar rapid comparison was made between the recent studies and similar NEER004 studies and a small number of adjustments were made to the latter to improve consistency, also mostly to EV scores and study type(s).

External validity and applicability

2.28 External validity (EV) was categorised according to the extent to which the results of studies were considered representative, applicable and generalisable to the target habitat resource (i.e., UK, especially English, upland peatlands), burning and other interventions. This took into account criteria such as the number of study sites, geographic coverage and spread, including whether the study was conducted in the UK, and representativeness of the areas, habitats and interventions included (Table 3; also see corresponding criteria in the quality assessment checklists given in Appendix 3). These were the main factors that informed the assessment of applicability of the evidence for each sub-question by reference to descriptions and guidance on the characteristics and state of the UK and English upland peatland resource (para. 4.18). The results of the EV categorisation for the evaluated studies included in this review update are given in the main text of this report and in Table A1.1 in Appendix 1.

Table 3. External validity (EV) categories.

EV category	Definition
++	Representative and applicable nationally or to multiple UK upland regions.
+	Representative and applicable regionally.
-	Not representative of UK peatlands or generalisable beyond single or a small number of specific study site(s) or local area(s). In some cases, specific location(s) unknown.

2.29 Scoring of external validity is part of the established Natural England evidence review method process, although the guidance states that “the external validity rating may be used when citing documents … but its key purpose is to inform the statement of applicability" (Stone, 2013, para. 8.14, p. 18). Although they were not published for individual studies in the NEER004/014 reports, they were included in internal evidence tables used to inform the descriptions of applicability of the evidence given for each of the sub-questions (for example, NEER004 paras 4.13–4.17, pp. 19–21, for vegetation). They are given for the NEER014 studies relevant to the wildfire sub-question in Table A1.2 and the NEER004 studies in Table A1.3 in Appendix 1 of this report.

Synthesis of evidence

2.30 As in NEER004, narrative syntheses were produced from the evaluated studies that met the inclusion criteria and contained sufficient information for quality assessment. They are presented for a range of outcome measures relevant to each sub-question in Sections 4–11.

2.31 A novel part of this review update was a comparison of the findings from the NEER004 review (and some relevant to the specific wildfire sub-question from NEER014) and with that from recent studies. This is presented in the form of tables and brief summaries on the degree of consistency between the findings of the two reviews across a range of outcome measures in Section 12. This also identified new findings from recent studies.

2.32 'Evidence statements' were developed across the combined evidence base drawing particularly on the comparison tables between NEER004 and recent study findings in Section 12 and reflecting:

- The best available evidence of the effect of burning (and in some cases other interventions) including the study type. Apart from a few cases where reviews contributed (which are noted in the text), this was based on 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, and the size and ecological/ environmental importance of the effect.

2.33 Based on these factors, the strength of the evidence for each statement was classed as: strong, moderate, weak or inconsistent. This is partly a subjective judgment considering the above factors, though the following descriptions were used as guidance:

- **Strong**: many studies (typically >four/five) showing consistent trends or one or two high quality or national, representative studies [1++, 1+ or 2++] and/or [EV++].
- **Moderate**: a smaller number (at least two/three) studies of which at least one was classed as a minimum of [2+] and/or [EV+].

- **Weak**: single or a small number of generally lower quality studies, usually including at least some classed as [-] and/or [EV-].

2.34 Evidence statements were only developed where evidence of specific effects of burning (in some cases in comparison with other interventions) was identified in relation to the individual sub-questions and aspects of them across the combined NEER004 and recent evidence base.

Characteristics of the recent evidence base

Search strategy results

2.35 Overall, 102 recent evaluated studies were included in this review update, with 58 (56%) identified from the Scopus database search (Table 4). The number of references remaining from the Scopus search after each stage of screening is given in Table 4. This includes just six references from four studies added from a repeat search of Scopus carried out in September 2023 using the same search terms and strings.

Table 4. The number of recent (post-NEER004) references remaining at each stage of searching and screening of references from the Scopus database in the review update.

Searching, screening and review stage	Number of references
References captured using search terms (including duplicates)	2,364
References captured using search terms (excluding duplicates)	1,726
References remaining after title filter	210
References remaining after abstract filter	90
References remaining after full text filter	83
Evaluated studies included in the review	58 studies

2.36 The total number of evaluated studies and supplementary study references included in the review update from all the search strategy stages was 102 and 76, respectively (178 in total) (Table 5). In terms of the number and percentage of studies identified from the individual search strategy stages, most were identified from the initial Scopus database search (56%), followed by the Ashby (2020) review, country nature conservation agencies and government study reports (most from Natural England/Defra) and the Natural England burning reference list (all between 15 and 11%), with other reviews contributing just 5% (Table 5). Similarly, the highest percentage of supplementary references for these studies was again identified from Scopus (33%), followed by the Natural England reference list (24%), country nature conservation agencies and government study reports (20%) and the main study reference (13%), with less than 10% from the other two search stages.

Table 5. The number of recent (post-NEER004) studies and associated supplementary references relating to these studies from each stage of the update search strategy, ordered by search strategy stage sequence.

Search strategy stage	Number of studies	Number of supplementary references	Total
Scopus database	58	25	83
Ashby (2020) review	16	6	22
NE and other study reports[1]	15	15	30
Other recent reviews	4	2	6
NE burning reference list	11	18	29
From main study reference	-	10	10
Total	102	76	178

[1] Natural England and the other UK country nature conservation agencies, and government (Defra and Scottish Government).

2.37 The main study references and associated supplementary references for the 102 evaluated studies included in this review update are listed in Table A1.1 in Appendix 1 as an evidence table with basic categorical information. An additional 123 studies from NEER004 and 30 on wildfire from NEER014 are listed in Appendix 1 in Tables A1.3 and A1.2, respectively. Thus, this report brings together information from 255 evaluated studies across the combined evidence base.

Sub-questions addressed

2.38 In terms of the number of recent studies relating to individual sub-questions, most related to vegetation (42 studies), followed by fauna (24), water (23) and carbon (20) with similar numbers, and burning extent (11), wildfire (8), severity (8) and grazing (2) with fewer. Many studies (56, 54%) included elements relating to multiple sub-questions, although the findings were not always reported in sufficient detail to be included under all sub-question summaries of evidence. A total of 48 studies (46%) addressed just one sub-question, 43 (41%) two sub-questions, six (6%) three sub-questions and seven (7%) four sub-questions.

Reference types

2.39 For the 102 recent studies included in this review update, the main refence types comprised 79 (78%) journal articles, 20 (20%) reports (including three reviews), three PhD theses and one MSc dissertation.

Study categorisation

2.40 This section presents an overview of the type, quality, location and duration of the 102 recent studies included in this review update. Similar information is given for each individual study in Table A1.1 in Appendix 1, and in more detail for individual sub-questions, including applicability to UK upland peatlands, in Sections 4–11.

2.41 A summary of study type and quality (internal validity) of the 102 recent burning studies is given in Table 6. Most (73, 72%) were classed as type 2; most of the reminder were classed as type 1 or a combination of types 1 and 2 (13, 13% each); and a few were as type 3, 4 or 2 and 4. Most were classed as [+] for quality (90, 88%), nine as [++] and five as [-].

Table 6. Categorisation of type and quality (internal validity) of all recent (post-NEER004) studies by the number of studies. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	1	12	0	13
1,2: both type 1 and 2	2	11	0	13
2: quantitative observational or correlative	6	61	4	71
2,4: both type 2 and 4	0	2	0	2
3: qualitative	0	0	1	1
4: review	0	2	0	2
Total	9	88	5	102

2.42 A summary of study type and external validity of the 102 recent burning studies is given in Table 7. Most individual studies, 61 (60%), were classed as [EV-], with 24 as [EV+] and 19 as [EV++]. The majority classed as [EV++] were either earth observation (EO, including aerial photographic interpretation, API) studies used to map burning distribution and extent (8) or primarily breeding bird studies (6), which typically covered large geographic areas, in some cases almost a complete census or representative national samples. Three wildfire-related studies and one vegetation study were also classed as [EV++]. Those classed as [EV-] tended to be studies on single or a few often geographically close or otherwise similar sites, or from areas that it was considered may not be representative of UK upland peatlands.

Table 7. Categorisation of type and external validity (EV) of recent studies by the number of studies. EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	1	13	14
1/2: both type 1 and 2	0	1	11	12
2: quantitative observational or correlative	17	21	33	71
2,4: both type 2 and 4	0	1	1	2
3: qualitative	0	0	1	1
4: review	2	0	0	2
Total	19	24	59	102

Location of studies

2.43 Most recent studies (90, 88%) were carried out in the UK, with a further three covering wider geographic areas that included the UK. The remaining studies were from Norway (four), Canada, Denmark, Ukraine/ Belarus and China (all single studies). Within the UK, there was considerable overlap in a few large-scale studies which included sites in several countries, especially GB countries (six including England, Scotland and Wales). Most UK studies were in England (58 studies) or included England (nine more), followed by Scotland (17, plus ten including Scotland), Wales (two, plus seven including Wales) and Northern Ireland (one, plus one UK study including Northern Ireland).

3. Introduction to the summaries of recent evidence

3.1 The evidence relevant to the eight sub-questions is summarised in subsequent sections (4–11) in the form of narrative syntheses produced from the evaluated studies that met the inclusion criteria and contained sufficient information for quality assessment.

3.2 Each section includes (i) an introduction that refers to the corresponding evidence presented in NEER004; (ii) characteristics of the recent studies including study type, quality and reported outcome measures; (iii) applicability including study locations, habitat types, burn types and external validity; and (iv) a narrative synthesis of the recent evidence by outcome measures.

4. Recent evidence on the effects of managed burning on the vegetation of upland peatland habitats

4.1 The full, slightly revised, text of this sub-question is:

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

Introduction

4.2 The characteristic vegetation communities, habitats and associated plant species of upland peatlands are described in NEER004 (paras. 3.1–3.6) and in Appendix 6 of this report which lists National Vegetation Community (NVC) types associated with upland peatlands. NEER004 and this update relate to the range of peatland habitats and associated vegetation communities that occur on peat soils on moorland, normally above the Moorland Line (ADAS, 1993), in the uplands (the Severely Disadvantaged Area, SDA) and generally above 250 m above ordnance datum. These comprise blanket bog (and locally on the upland fringe, intermediate and occasionally raised bogs), wet heath, and upland flushes, fens and swamps.

4.3 NEER004 reviewed relevant studies on the effects of burning on upland peatland vegetation composition, structure and function up to 2012 (see Appendix 3, pp. 87–103 and Appendix 5 pp. 113–117). In NEER004, a summary, synthesis and brief interpretation of the information was given across studies, including as evidence statements (paras. 4.12–4.30) and research recommendations (para. 4.31 and paras. 12.35–12.36). The evidence statements were also given in summary form in the Conclusions (paras. 12.3–12.6) and Summary (pp. v-vi).

4.4 There is some crossover between this sub-question and most other sub-questions but especially those concerning fauna (Section 5) and carbon (Section 6).

Recent studies on the effects of managed burning on vegetation composition, structure and function

4.5 Forty-five recent studies since NEER004 provided evidence on the effects of managed burning on the vegetation composition, structure and function of upland peatlands. Information on the characteristics of individual studies is given in an evidence table (across the eight sub-questions) in Appendix 1 (Table A1.1).

4.6 Twenty-nine of the 45 studies (64%) related primarily to vegetation rather than other sub-questions. Sixteen additional studies relating primarily to other sub-questions, especially fauna, often involved the collection of some botanical data which has also been used in addressing this sub-question, though in some cases some of the vegetation data collected were not specifically reported on.

Study type and quality

4.7 A summary of the type and quality of the 45 recent vegetation studies is given in Table 8. The majority, 29 (64%), were classed as type 2, nine (20%) as type 1 and seven (16%) as a combination of type 1 and 2. Most were classed as [+] for quality (38, 84%), five as [++] and two as [-].

Table 8. Categorisation of recent vegetation studies by study type and quality. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	3	6	0	9
1/2: both 1 and 2	1	6	0	7
2: quantitative observational or correlative	1	26	2	29
3: qualitative or 4 review	0	0	0	0
Total	5	38	2	45

4.8 Although 13 of the vegetation studies involved burning treatments as part of field experiments, eight did so by using the existing, long-established Hard Hill burning and grazing experiment at Moor House NNR. One of these reported on the latest, routine vegetation monitoring programme carried out in 2013 (Milligan and others, 2018 [1++, EV-]), which is part of a single study alongside previous Hard Hill vegetation survey references described and evaluated in NEER004 (para. 4.16, p. 20 and Appendix 3, pp. 87–90). The experiment, described by Marrs and others (1986), lies on a slope in an extensive area of high altitude, modified, but active blanket bog (NVC community M19), with high cover of *Eriophorum vaginatum* and *Calluna*, and frequent *Sphagnum*. It was previously burnt as part of a grouse moor prior to the establishment of the NNR in the 1950s. The vegetation was first surveyed in 1961 following burning of four experiment blocks in 1954 and subsequently at approximately 10-year intervals. Hard Hill is the only long-term experiment that covers multiple burning rotations in the UK (six burns in the ten-year treatment and three in the 20-year treatment at the last time routine vegetation monitoring was reported (Milligan and others, 2017 [1+, EV-]).Some other recent (and past) studies that used the Hard Hill experiment covered specific vegetation aspects and/or other outcomes, but are generally treated as separate studies in describing the recent evidence in the following sections.

4.9 Several authors have suggested that Hard Hill and Moor House in general are not typical in several respects (see NEER004, para. 4.16) with, for example, Gray & Levy (2009) and Baird and others (2019) stating that extrapolation of results more widely should be done with caution, especially to unmodified sites in more natural condition. This is supported by more recent investigations into the Hard Hill experimental set-up and site which have identified a number of issues. These include significant differences between blocks, plots and sub-plots in terms of their physical structure (meso-scale structural features that demonstrably affect the vegetation and microtopography) and burning history (pre-experiment and in the experiment burn treatments since the initial 1954 burns) (Clutterbuck and others, 2020 [1,2+, EV-]), as well as trampling effects from surveyors, also previously suggested by Lindsay (2010). Another major issue is the lack of pre-burn baseline survey or monitoring (for example, Lee and others, 2013a [1++, EV-, NEER004]). This was addressed to some extent by the post-hoc establishment of longer-unburned, grazed 'reference plots' adjacent to each of the four blocks in 1965 (Lee and others, 2013b), though these have only had full surveys on three occasions. In addition, general lists of species occurring outside the burnt areas are given in an initial, unpublished report on the experiment (Forrest,1961).

4.10 The other seven Hard Hill studies covered specific vegetation aspects from single or short-term monitoring periods within the treatment and/or reference plots (in some cases in combination with other sites) which are treated as separate studies here: Ward and others (2012 [1,2+, EV-] mainly on carbon uptake and cycling); Lee and others (2013a [1,2+, EV-] on propagules); Alday and others (2015 [1,2+, EV-] on biomass); Clutterbuck and others (2020 [1,2+, EV-] on microtopography); Noble and others (2018a [1,2+, EV-] on *Sphagnum*); and Noble and others (2019a [1,2++, EV+] on fire temperatures/damage to *Sphagnum* which also included a laboratory element, see para. 4.13). Field experiments tend to be resource intensive, reflecting the need to establish and monitor the effects of replicated treatments over, in this case, relatively long burn rotations. Hence, although a number of burning field experiments have been established, only one, the Hard Hill experiment, has covered multiple burn rotations.

4.11 In addition, four other studies including vegetation aspects took place on, or took samples or used data from, Hard Hill or the wider Moor House NNR: Santana and others (2016 [2+, EV+] using biomass data from Alday and others (2015 [1,2+, EV-]); Noble and others (2017 [1+, EV-]); Robertson and others (2017 [2+, EV+]); and Noble and others (2018b [2+, EV++]).

4.12 Five other studies involved field experiments. Two on Norwegian coastal wet (and dry) heaths (Velle and others, 2012 [1+, EV-]; Velle & Vandvik, 2014 [1+, EV-] involving up to six sites); two linked studies in Scotland (Grau-Andrés and others, 2017b, 2019b/a, both [1+, EV-] at two sites, a raised bog and dry heath) and a single study in England (Heinemeyer and others, 2019c /2023 [1+, EV-] at three Pennine/Bowland blanket bog sites under Peatland-ES-UK). All these covered relatively short, initial post-burn periods: Grau-Andrés and others (2017b, 2019b/a) up to 17 months; Velle and others (2012)/Velle & Vandvik (2014) six/seven years; Heinemeyer and others (2019c) initially four years (following a one-year pre-burn baseline), though the project was extended a further five-years as Peatland-ES-UK (Heinemeyer and others, 2023) giving a total of nine years post-burn.

4.13 In addition to the field experiments listed above, five studies were laboratory experiments or included laboratory elements (as well as other study types) which provided evidence on vegetation response to burning: Noble and others (2017 [1+, EV-]) on bulk density, ash deposition and rainwater chemistry treatments on two *Sphagnum* species and heath star-moss *Campylopus introflexus*); Taylor and others (2017 [1+, EV-]) on temperature treatments on acute-leaved bog-moss *Sphagnum capillifolium*); Worrall and others (2013a [1,2+, EV-]) on char production/biomass loss); Noble and others (2019a [1,2++, EV-]) on temperature treatments on five *Sphagnum* species). The last two also included pre- and initial post-burn field data collection at single sites; and Yusup and others (2022/2023 [1+, EV-]) on the response of spore germination in *Sphagnum* and other bryophytes to fire-related cues.

4.14 Whilst Heinemeyer and others (2019c/2023 [1+, EV-]) covered nine-years post-burn, all of the other studies have so far have only covered shorter post-burn periods. In part in response to this, other types of less resource intensive studies have been used relatively widely to monitor the effects of burning on vegetation and other outcomes. These include 'chronosequence' studies that substitute 'space-for-time' in determining the sequence of the post-burn succession from vegetation stands of differing ages since being burnt, thereby allowing longer post-burn periods to be covered within or between sites over much shorter survey/monitoring periods. Such recent studies include: Lee and others (2013a [1,2+, EV-]); Santana and others (2016 [2+, EV+]); Noble and others (2019b [2+, EV+], with repeated short-term resurveys); Whitehead & Baines (2018 [2+, EV-]); and Whitehead and others (2021 [2+, EV-]). Such studies make an assumption that each plot/site in the sequence differs mainly in age since treatment and each has the same history in both biotic and abiotic components. This has led to some criticism of the chronosequence approach (for example, see NEER004, 2013 para. 4.15) which needs to be considered in interpreting the findings of such studies. On the other hand, they enable the collection of data covering different post-burn periods over much shorter timescales and, hence, potentially from more and often larger sites or areas than typical for experimental studies. Thus, they are more time and resource efficient and potentially more representative of the habitat resource on sites and over larger geographic areas.

4.15 Other observational/correlational study sub-types providing vegetation data were all survey or monitoring field studies. There were 11 of these. Five were primarily vegetation studies: Critchley and others (2016 [2++, EV++]); Swindell (2017 [2+, EV-]); Noble and others (2018b [2+, EV++]); Muñoz and others (2014 [2-, EV-]); and Garnett (2023 [2+, EV-]). Four were primarily bird studies but also recorded vegetation variables: Calladine and others (2014 [2+, EV-]); Douglas and others (2017 [2+, EV-]); Robertson and others (2017 [2+, EV+]); and Ludwig and others, (2018 [2+, EV-]). Newey and others (2020 [2+, EV++]) compared species distribution data with burning intensity on grouse moors. All of these were also classed as case studies, mostly comprising relatively few, often not randomly selected, sites, apart from Newey and others (2020) which comprised a census of Scottish grouse moors and Noble and others (2018b) which included a representative sample of 85 English blanket bog sites. Many of the other evaluated studies listed above in this section were also classed as case studies usually in addition to other study type sub-categories.

4.16 A further eleven studies were palaeoecological investigations which produced information on changes in the abundance of some key vegetation species in relation to fire events over long, millennial time periods since the time of blanket bog initiation, typically dating back around 3,000–10,000 years through the Holocene (Simmons, 2003; Gallego-Sala and others, 2016). These include: Fyfe & Woodbridge (2012 [2+, EV-]); Chambers and others (2013, 2017, both [2+, EV-]); Swindles and others (2015, 2016, both [2-, EV-]); McCarroll and others (2016a,b, 2017, all [2+, EV-]); Blundell & Holden (2015 [2+, EV-]); Fyfe and others (2018 [2+, EV+]); and Rowney and others (2023 [2+, EV+]). It is likely that these represent only a proportion of recently published UK upland palaeoecological studies. However, they include those that were identified in the searches carried out as specifically relating to fire and hence are potentially relevant to current burning management. While these can show correlational relationships between fire and vegetation composition, issues around causality, geographical scale and chronology may hamper interpretation of results (for example, Stewart and others, 2004). Nevertheless, in covering much longer time periods than any other type of study, they provide unique information on the timescales of the effects of, and recovery from, past fires and other drivers (for example, Lindsay and others, 2014). They may also provide a better understanding of study site condition prior to recent management interventions and indicate a potential recovery trajectory. A recent evidence review for Natural England on the historic peat record and its implications for the restoration of blanket bog (Gillingham and others, 2016c [4+, EV++]) also contributed evidence relevant to this sub-question based on 28 UK palaeoecological studies.

Outcome measures

4.17 The 45 recent vegetation studies assessed 20 main outcome measures as shown in Figure 1. Only four outcomes involved ten or more studies: key species/groups, vegetation height/structure, overall species composition and palaeoecological data. There is, however, inevitably some overlap between some of the measures, for example, species abundance in relation to overall composition, key species/groups, species of conservation concern and palaeoecological data.

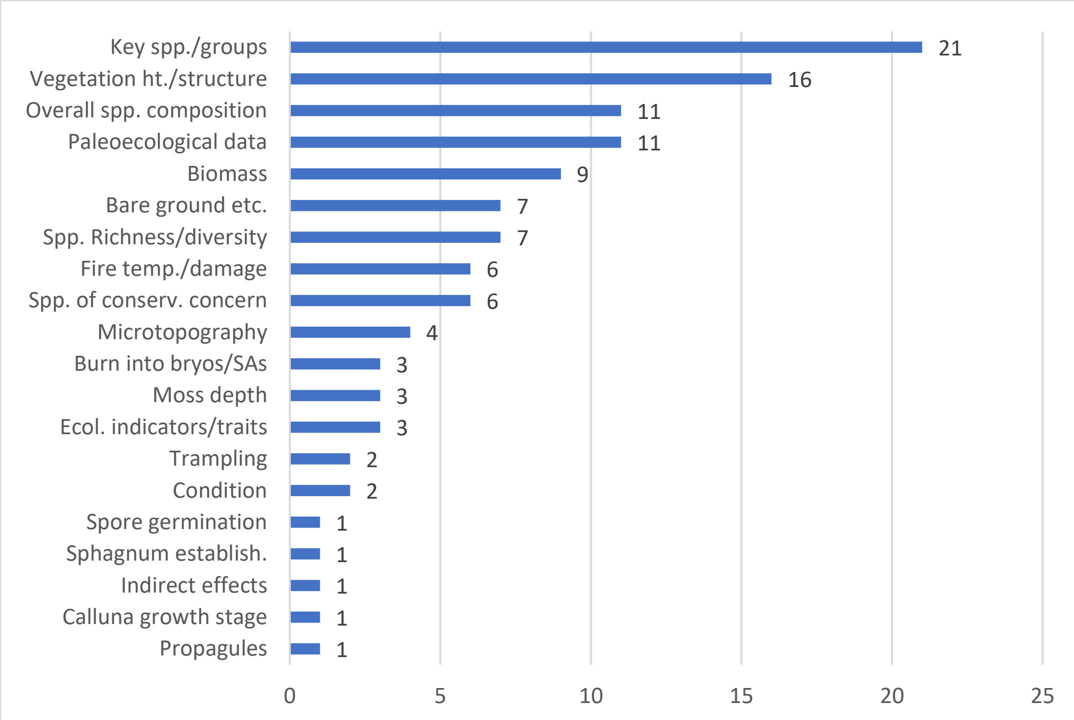

Figure 1. Outcome measures recorded or derived from recent vegetation studies (n = 45 studies). Some studies covered more than one outcome.

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

4.18 This section reviews the applicability of the evidence from the mostly primary, evaluated studies 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 external validity carried out in assessing individual studies and across the studies as a whole. This includes such factors as geographical location and how representative the habitat/vegetation type(s) and intervention(s) were in terms of the characteristics and state of upland peatlands in the study area(s) and especially nationally. These assessments were informed by reviews and guidance on the UK and English upland peatland resource, in particular JNCC (2009, 2011), Natural England (2010), Lindsay & Clough (2017) and Artz and others (2019).

Countries, areas and number of sites

4.19 All but five of the recent vegetation studies were from the UK (40, 89%), with the others from Norway (2), and Canada, China and Spain (one each). Within the UK, the majority were from or included sites in England (32, 80% of UK studies, including one study which also included Scottish sites), followed by Scotland (7, 19%) and single studies in Northern Ireland and Wales. Thus, whilst they are generally geographically and ecologically applicable, they were concentrated in the north of England, especially the Pennines (see Figure 2), which covers the main area of English blanket bog. Other areas and hence the range of variation in the upland peatland resource as a whole were not fully represented. In particular, as a result of the scarcity of study sites in Wales, South-West England and western England and

Scotland, western upland peatlands with their wetter, milder oceanic-influenced climate, are under-represented, as are unmodified or less modified sites which are largely absent (paras. 4.24–4.25).

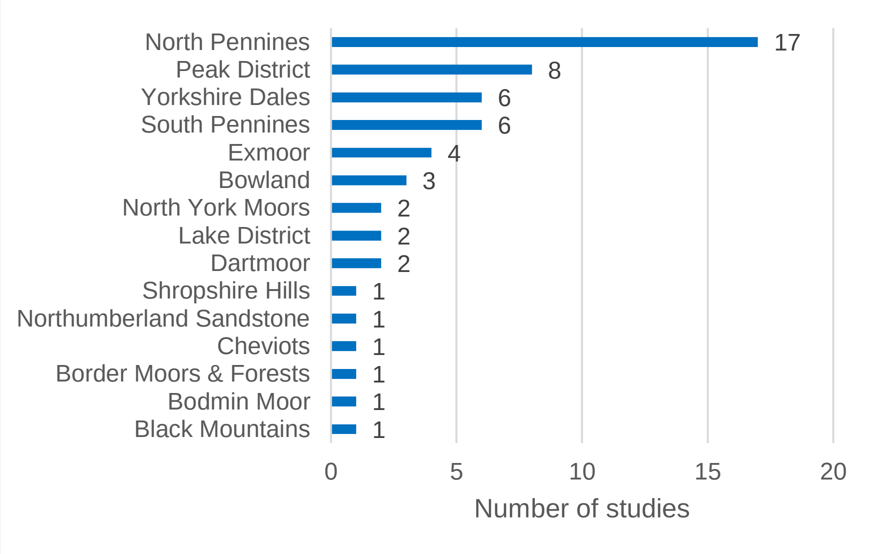

Figure 2. Categorisation of recent vegetation studies by English upland areas (n = 32 studies)

4.20 Within England, there was one representative national English sample of 85 blanket bog sites (Noble and others, 2018b [2+, EV++], comprising a condition data set, which is also reported in terms of habitat condition in Critchley and others (2011a [2++, EV++], NEER004), whilst another comprised 20 long-term moorland monitoring sites scattered across the English uplands (Critchley and others, 2016 [2++, EV++]), and one covered 36 grouse moors across the north of England[10] (Robertson and others, 2017 [2+, EV++]). The remainder included a relatively small number of sites (para. 4.15). Overall, the vast majority were carried out in northern England (26, 93%) most of which included sites in the Pennines/Bowland. Only five studies (17%) included sites south of the Peak District. Of the English upland areas, most studies included the North Pennines (17 or 59% of studies[11]), followed by the Peak District (8, 24%), Yorkshire Dales (6, 21%), South Pennines (6, 21%) and Exmoor (4, 14%), with no other upland areas included in more than three studies (Figure *2*). The large number of sites in the North Pennines in part reflects multiple studies at Moor House NNR (para. 4.8).

4.21 The eight Scottish studies included a census of mapped grouse moors (Newey and others, 2020 [2+, EV++]) in relation to selected species' distributions, with the remainder comprising single (5 studies) or two sites (1 study). Single Welsh and Northern Irish studies comprised (i) a palaeoecological study from a single blanket bog (Brecon Beacons); and (ii) a series of raised bog sites, respectively.

[10] Although only 17 were classed as blanket bog, with the remainder being 'dry heath' (para. 4.91).

[11] Though some studies included sites in multiple English upland areas.

4.22 Across all the remaining vegetation studies, most involved only a small number of sites with the majority comprising single sites (27, 60%); the others included two to five sites (11, 24%) or six to ten sites (2, 4%) (Figure 3). The only additional study, to the four mentioned above (paras. 4.2033–4.21) that involved more than ten sites was a palaeoecological analysis from 16 sites around Exmoor (Fyfe and others, 2018 [2+, EV+]). As might be expected, this was particularly the case for experimental studies which, whilst they provide more evidence on cause-effect relationships, are resource intensive and hence tend to be carried out on relatively few sites (in this case only one involved more than three sites, with six), often in relatively small plots. As a result, they are likely to be less representative of sites, areas and the upland peatland resource as a whole than some other study types which may enable coverage of more sites and areas.

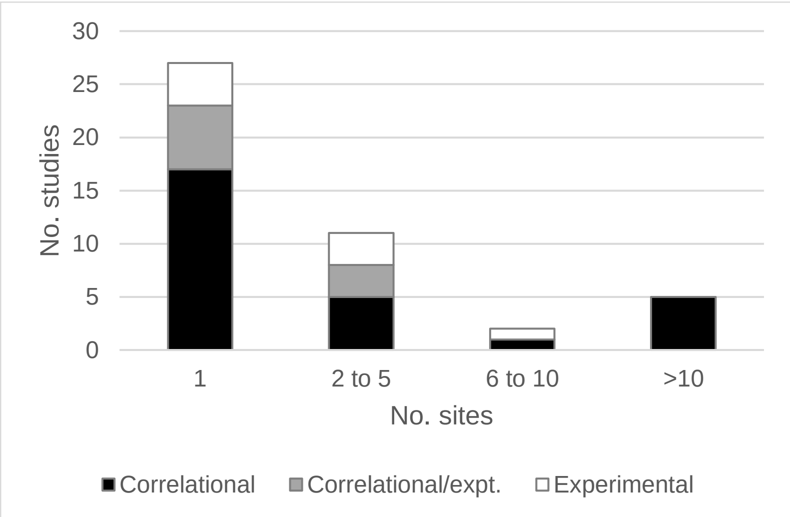

Figure 3. Number of study sites included in recent vegetation studies by main study type (correlational/observational, experimental or a combination) (n = 45 studies)

Habitat and vegetation type

4.23 In many cases, habitats, vegetation types and degree of modification were not well described, in some cases hardly at all. In particular, sites or part sites described as ‘wet heath’ and even ‘dry heath’ or ‘dry heath-type’ vegetation were likely degraded forms of blanket bog and other peatland habitats on deep or shallow peat. By habitat, the majority of recent vegetation studies were carried out on sites reported as blanket bog (35, 78% of studies, a number of which also included other habitats), with other habitats described as wet heath, dry heath, wet/dry heath and undifferentiated ‘moorland’ (each in three studies, 7%), raised bog (two, 4%) and ‘peatland’ (one, 2%).

4.24 Habitat state in terms of degree of modification of vegetation composition and structure was described in 39 studies (several with multiple classes including “various”), though generally not using a standard classification. Eight (21% of studies) included stands classed as ‘severely modified’, 16 each as ‘modified’ or ‘less modified’ (both 36%), and only one was described as “unmodified/intact” (3%), though the basis of this classification was not given. Stands were classed as ‘various’ in seven studies (19%). Growth forms reflecting degree of modification were generally not referred to. For example, *Calluna* growth forms (either in a 'steady state' on less modified, wetter peatland sites with stems reburied by growth of *Sphagnum*, as described in NEER004, paras. 4.2 and 4.22, or the classic heathland pioneer, building, mature and degenerate growth stages, Watt, 1955) were rarely referred to. Similarly, *Eriophorum vaginatum*, *Trichophorum germanicum* and *Molinia* growth forms, either as individual stems scattered through a *Sphagnum* carpet or growing in tussock form in the absence of a *Sphagnum* carpet (Clutterbuck and others, 2020 [1,2+, EV-]), were not referred to.

4.25 NVC type(s) were given or could be derived for 18 studies, most with several community types included. As would be expected, given the northern and especially Pennine distribution of many studies (Figure 2, para. 4.195.14, also see Averis and others 2004), the *Calluna* and *Eriophorum vaginatum*-dominated M19 NVC community was the most frequent type (included in 15 studies, 83%). This was followed by the associated more modified *Eriophorum vaginatum*-dominated M20 community (13, 72%), the less-modified, wetter, often bog-moss *Sphagnum*-dominated M18 community (3, 17%), and the *Molinia*-dominated M25 community[12] and *Calluna*-dominated heath-type vegetation (H2/H10/H12 communities) on peat each in single studies (6%). No examples of the relatively widespread M17 western blanket bog or M15 and M16 wet heath/bog communities were specifically reported, though a few "wet heaths" were.

Types of burn and fire

4.26 The majority of the burns included in the recent vegetation studies were managed events undertaken by land managers (23, i.e., included in 68% of vegetation studies excluding palaeoecological studies [11] where fire type could not be determined). Five studies involved more than one fire type. Other types of burns/fire were classed as experimental (in nine field studies), laboratory (four) and wildfire (three).

4.27 Good practice for managed burning is given in the Heather and Grass Burning Code (Defra, 2007) and corresponding codes for Wales (WAG, 2008) and Scotland (NatureScot, 2021). These recommend 'cool burns' that "aim to remove the dwarf shrub canopy but leave behind a proportion of 'stick'" (Defra, 2007; also see NEER004, para. 3.10). This represents an aim to deliver an outcome rather than necessarily a specific management technique to do so, although it can be achieved using pressurised fuel assisted (PFA) burning[13] (for example, Allen and others, 2016 [2+, EV-], NEER004, para. 8.3, p. 38). Nevertheless, burns are likely to vary in severity both within and between fires and sites depending on moisture content, fuel

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[12] Sometimes regarded as modified blanket bog or wet heath community types, particularly M15.

[13] Where vegetation is sprayed with liquid fuel before ignition and fires can be lit when the vegetation has a higher moisture content (MC) than normal resulting in burning being less limited to rain-free days and low MC.

load and structure, vegetation composition, weather conditions, topography and other factors (for example, Defra, 2007; Legg & Davies, 2009; Robichaud and others, 2000; and para. 8.28 in Section 8 on severity). Burns done as part of research experiments tend to be particularly carefully controlled and carried out in small plots, sometimes over varying rotation lengths, so could differ from typical managed burns carried out by land managers (Lindsay, 2010; NEER004, paras. 3.10 and 4.16).

External validity

4.28 Most recent vegetation studies were classed as [EV-] for external validity (32, 71% of studies), with nine (20%) as [EV+] and four (9%) as [EV++] (Table 9). This reflects the relatively small number of sites included in most vegetation studies (Figure 3, para. 4.22), and relatively limited geographic spread of them in relation to the UK and English upland peatland resource as a whole, with most in the north, especially the Pennines (paras. 4.205.16–4.21, Figure 2). All those classed as [EV++] were type 2 studies (Critchley and others, 2016; Newey and others, 2020; Noble and others, 2018b; Robertson and others, 2017, all [2++, EV++]) involving relatively large numbers of sites and/or samples.

Table 9. Categorisation of recent vegetation studies by external validity (EV) and type of study. EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV++	EV+	EV-	Total
1: quantitative experimental	0	1	8	9
1/2: both type 1 and 2	0	1	6	7
2: quantitative observational or correlative	4	7	18	29
3: qualitative or 4 review	0	0	0	0
Total	4	9	32	45

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

4.29 This section presents a summary, synthesis and brief interpretation of the findings of 45 recent studies on the effects of burning on vegetation composition, structure and function. It covers the 20 outcome measures listed by frequency of inclusion in individual studies in Figure 1, and includes narrative descriptions and summary tables in Appendix 7 that indicate the direction and magnitude of responses in relation to biodiversity (vegetation and habitat composition structure and function) objectives.

Overall species composition

4.30 Although at least 11 recent studies recorded full or near full species/community composition and typically also frequency or cover abundance data, not all reported or interpreted this in full or in some cases at all (other than for a few selected species/groups). When full species data were presented, it tended to be as summary tables by sites and/or treatments or responses, often with limited or in some cases no interpretation at the community level (Velle and others, 2012 [1+, EV-]; Grau-Andrés and others, 2019b [1+, EV-]; Hedley, 2013 [2+, EV+], Appendix 1; Heinemeyer and others, 2019c/2023 [1+, EV-], Appendix 3/Lindsay, 2020, Appendix 3A; Clutterbuck and others, 2020 [1,2+, EV-]). In other cases, community data were presented as ordination plots derived from all or subsets of species (Velle and others, 2012 [1+, EV-]; Muñoz and others, 2014 [2-, EV-]; Velle & Vandvik, 2014 [1+, EV-]; Milligan and others, 2018 [1++, EV-]; Noble and others, 2018b [2+, EV++]).

4.31 Hedley (2013 [2+, EV+]) used full species lists with Domin cover abundance scale categories (Rodwell, 2006) to allocate sample quadrats from the ten EMBER[14] study sites (reported by Brown and others, 2014 [2+, EV+]) to NVC vegetation community and sub-community types using the Tablefit (Hill, 1996) and MAVIS (Smart & Dart Computing, 2000) computer matching programmes arbitrated by expert judgement. This showed expected differences between the main NVC community and sub-community types. All the not-recently-burnt sites were typical blanket mire vegetation types for the Pennines. One was classed as M19a, two as M19b and two as M20b. The recently burnt sites showed more variation with only three classified as mires – one an M6 fen and two M19a blanket bog – and two as heath-types – both H9b. The last is a species-poor community generally the product of degradation of precursor dwarf-shrub heath or mire, but still retaining a few mire species, described by Hedley (2013 [2+, EV+], p. 7) as “sparse but distinct representation of true bog plants, notably the two *Eriophorum* spp.”. The presence of this community has been associated with frequent burning, especially where it occurs on peat soils (Rodwell, 1991; Elkington and others, 2002; Averis and others, 2004; Hedley, 2013 [2+, EV+]). (Also see para. 4.25 regarding NVC community types given or derived for 18 of the recent vegetation studies, with M19 [in 83% of studies] and M20 [72%] blanket bog communities the most frequent, and para. 4.106 regarding habitat condition of the EMBER sites).

4.32 In a review of the Hard Hill experiment (para. 4.8) set up, Clutterbuck and others (2020 [1,2+, EV-]) present a phytosociological sorting (as per Müller-Dombois & Ellenberg, 1974) highlighting ecological groupings of species, similar to NVC floristic or constancy tables (Rodwell, 2006). This is based on presence-absence species data from the Hard Hill experiment routine vegetation monitoring from between 1961 and 2011, and adjacent, longer-unburned ‘reference’ plots in 1965 and 2011, given by Lee and others (2013b [1++, EV-], NEER004, supplementary Table S2). The results show the development of a community characterised by “a relatively species-rich *Sphagnum* assemblage and a low-growth canopy of *Calluna* together with fire-sensitive species such as [lesser twayblade] *Neottia cordata*” in the longer-unburned ‘reference’ plots by 2011. This represents a shift from a "dry heath-like community" in the reference plots (and also the not burnt since 1954 plots) 46 years earlier (in 1965), which changes over time to “dry/damaged bog/damp bare peat with moss and

[14] The ‘Effects of Moorland Burning on the Ecohydrology of River basins’ project (Brown and others, 2014): https://water.leeds.ac.uk/our-missions/mission-1/ember/.

liverworts” to a “slightly damaged/enriched *Sphagnum*/moss community” as a precursor to “damp *Sphagnum*/moss-rich (beneath dwarf shrub)” in the ‘reference’ plots. The dry heath-like vegetation seen in 1965 in the no-burn since 1954 plots was also then evident in the ‘reference’ plots, reflecting a legacy of past burning referred to by Forrest (1961) for the whole of Hard Hill, whereas by 2011 this was much less pronounced in the ‘reference’ plots.

4.33 Ordination analysis plots derived from all or subsets of species also provide information on the response of the vegetation community composition to burning (Velle and others, 2012 [1+, EV-], Norway coast; Muñoz and others, 2014 [2-, EV-], Spain; Velle & Vandvik, 2014 [1+, EV-]/Velle and others, 2014**,** Norway coast; Milligan and others, 2018 [1++, EV-], Hard Hill; and Noble and others, 2018b [2+, EV++], two data sets, EMBER and national CSM condition sample). All apart from Muñoz and others (2014, [2-, EV-], in which only one site was subject to repeated burning) showed an effect of burning on community composition. These included a clear separation of recently burned and not recently burned plots (Velle & Vandvik, 2014 [1+, EV-]/Velle and others, 2014; Noble and others, 2018b [2+, EV++], EMBER data set). Some showed overlap but clear differences in the centres of distribution of burned and not recently burned plots in ordination space (Milligan and others, 2018 [1++, EV-]; Noble and others, 2018b [2+, EV++], condition dataset). Others showed a separation of plots post-burn (greatest in first year) with a gradual transition/ succession back towards the composition of not-recently-burnt plots (Velle and others, 2012 [1+, EV-]; Velle & Vandvik, 2014 [1+, EV-]/Velle and others, 2014; Milligan and others, 2018 [1++, EV-]).

4.34 As might be expected, this was more rapid towards the composition of younger wet heath stands when burnt, than older stands (Velle and others, 2012 [1+, EV-]). In the unburned since 1954 treatment in the Hard Hill experiment, the post-burn transition was not towards the pre-burn position. Rather, it described as “… in a negative direction towards [*Calluna*] *vulgaris* and heath plait-moss [*Hypnum*] *jutlandicum* …” (Milligan and others, 2018 [1++, EV-]/Lee and others, 2013b, NEER004), though this was not to overwhelming *Calluna* dominance and not at the expense of diversity of mire species (NEER004, Appendix 3, pp. 87–90).

4.35 The ordination analyses also showed significant relationships between other factors and vegetation composition: grazing (Milligan and others, 2018 [1++, EV-]; Noble and others, 2018b [2+, EV++], condition data set); geographic location (Velle & Vandvik, 2014 [1+, EV-], northing (Noble and others, 2018b [2+, EV++], both data sets, northing and easting); habitat (Velle and others, 2012 [1+, EV-]; Muñoz and others, 2014 [2-, EV-]; Velle & Vandvik, 2014 [1+, EV-], all between wet and dry heath); and elevation, peat depth and atmospheric pollution (Noble and others, 2018b [2+, EV++]). Of these, Milligan and others (2018 [1++, EV-]) also showed a significant interaction between burning and grazing (also relevant to Section 9) and Noble and others (2018b [2+, EV++]) showed an interaction between post-burn age and nitrogen deposition on *Eriophorum vaginatum* cover, with a significantly more negative relationship on <2 year old burns than unburned plots (‘EMBER data set’), and nitrogen deposition showed a significantly more negative relationship with *Sphagnum* cover on burned than unburned plots ('condition data set').

Species richness and diversity

Species richness

4.36 Only eight recent studies reported on species richness or diversity, with some overlap: five on species richness (three on total plant and *Sphagnum* species richness and one each on vascular plant and propagule species richness) and five on various diversity indices. Species contributing to post-burn change in species-richness and diversity may not be typical mire species and may include species more characteristic of heath, grassland or other habitats, indicative of poor or unfavourable condition, or be colonisers of early successional stages. Thus, species richness and diversity derived from all species present may have limited usefulness for interpreting change in mire vegetation in terms of quality and condition. Indeed, increases in species richness or diversity might be interpreted as negative in relation to the habitat condition of peatlands, depending on the species contributing.

4.37 Effects on total plant species richness varied between three of the studies, perhaps in part reflecting different monitoring timescales. There was an initial post-burn decline across five coastal wet/dry coastal heath sites in Norway followed by an increase back to above pre-burn levels after just two years (Velle & Vandvik, 2014 [1+, EV-]); no significant differences in between (burn, cut, no recent management) treatments four years post-burn across three Pennine/Bowland *Calluna*-dominated blanket bog sites (Heinemeyer and others, 2019c/2023 [1+, EV-]), though over an extended post-burn period to nine years, species richness increased under burn, but not other treatments; and no change across ten wet heath sites in NW Spain on resurvey after 28 years, though not all were affected by burning (Muñoz and others, 2014 [2-, EV-]).

4.38 Milligan and others (2018 [1++, EV-]) reported on vascular plant species richness in the latest resurvey of the Hard Hill experiment, which was relatively low across all treatments (between c.6 and c.9 species), though it showed little temporal change in not burned since 1954 plots compared with increases under the burn treatments, which was greatest in the 10-year plots (though effects on total and moss species richness were not reported, but have been previously, for example, Lee and others, 2013b [1++, EV-, NEER004]). Similarly, in a chronosequence on a North Pennine *Calluna*-dominated blanket bog, vascular plant species richness was lowest in the oldest age class (>17 years) (Whitehead & Baines, 2018 [2+, EV-]). In the latter study, *Sphagnum* species richness was low in the youngest age class (1–2 years), higher between 3–17 years post-burn, and lowest in the longer unburned (>17 years) age class, though areas in the oldest class are likely to vary more markedly in age since burning than in the younger, narrower age classes and potentially also be atypical in differing from the others not just in time since burning. At another North Pennine site, the number of *Sphagnum* species declined initially following burning from a mean of 2.2 species pre-treatment to zero (no cover) for two years, followed by a gradual increase to 2.0 after 13 years, still below the pre-burn level, while following cutting it gradually increased from 2.3 species pre-treatment to 4.3 after 13 years (Garnett, 2023 [2+, EV-]).

4.39 Total species-richness of propagule banks in litter and peat from two Pennine blanket bog areas (both *Calluna*-dominated, but including three, more-modified Peak District sites, than the other, Hard Hill experiment site, in the North Pennines area) was low with only 14 and 12 species in peat, respectively (and 12 in litter which was only assessed at the Peak District site) (Lee and others, 2013a [1,2+, EV-]). There were fewer vascular plants from peat at Hard Hill (three) than at the Peak District site (seven), but slightly more bryophytes (nine and seven, respectively). Some characteristic blanket bog species propagules were not found, for example, *Eriophorum* and cloudberry *Rubus chamaemorus* in either area. At Hard Hill, species richness of the peat propagule bank was highest in the longest unburned 'reference' plots (para. 4.8), followed by the unburned since 1954 plots, and declined further with increasing frequency of burning treatment (a pattern also shown for *Sphagnum* propagule frequency and *Calluna* and haircap moss *Polytrichum* spp. seed density (paras. 4.77–4.78). Thus, burning seems to deplete the propagule bank.

Species diversity

4.40 Burning showed mixed effects on species diversity in five recent studies, though, as with species richness (para. 4.36), the species affected may be species associated with other habitats than mires or with disturbed conditions. Burning had little effect on various diversity indices in three studies over relatively long timescales (Muñoz and others, 2014 [2-, EV-]; Heinemeyer and others, 2019c [1+, EV-]; Whitehead and others, 2021 [2+, EV-]). However, in the Hard Hill experiment diversity (Shannon-Weiner index) declined over time in the unburnt since 1954 treatment (grazed and ungrazed), as it did in the two burn treatments when ungrazed, in contrast to when it was grazed (Milligan and others, 2018 [1++, EV-]). Beta-diversity was higher immediately post-burn in both low and high fire severity plots compared to not-recently-burnt plots at a Scottish raised bog (while at a corresponding dry heath only high fire severity plots had higher beta-diversity) (Grau-Andrés and others, 2019b [1+, EV-]), though this was immediately post-burn so might not be a long-term effect.

Ecological indicators

4.41 Only three recent studies provided information on trends in ecological indicators in response to burning. Such indicators, including Ellenberg indicator values (EIV) (Ellenberg and others, 1992), are estimates of the realised niches of species along various important ecological gradients. The mean EIV for species occurring in a given stand or plot is often used as an estimate of local ecological conditions and a surrogate for measured environmental variables.

4.42 Milligan and others (2018 [1++, EV-]) reported on the effects of burning in the Hard Hill experiment on four weighted Ellenberg ecological indicator scores (Hill and others, 2004/2009): moisture (F), light (L), fertility (N) and reaction/acidity (R). These tended to show declines and then increases for all four scores over time under all three treatments. Overall, there was little change in scores for light or fertility and a slight decline for acidity (reaction) and moisture under the no-burn treatment, compared with increases in all four under 20-year burn treatments, and increases in all but fertility under 10-year burn treatments. Velle & Vandvik (2014 [1+, EV-]/Velle and others, 2014) similarly reported that Ellenberg indicator values for reaction (R) were higher for increasing than for declining species following burning on Norwegian coastal heaths, though no differences were reported for F, L and N scores.

4.43 Lindsay (2020) carried out an ecologically focused re-analysis of the vegetation data from Heinemeyer and others (2019c, [1+, EV-]) which compared burning and cutting treatments at three Pennine/Bowland sites, Mossdale (M), Nidderdale (N) and Whitendale (W). This used a categorisation of species as indicators of bog/fen condition – 'community dominants' (*Calluna* and *Hypnum jutlandicum*) – and species 'showing a tendency to poor-fen/enrichment' likely associated with water movement through revegetated micro-erosion features (for example, flat-topped bog-moss *Sphagnum fallax* and soft-rush *Juncus effusus*), 'associated with drier conditions' (bilberry *Vaccinium myrtillus* and broom fork-moss *Dicranum scoparium*), 'characteristic of active bog' (for example, papillose bog-moss *Sphagnum papillosum* and cranberry *Vaccinium oxycoccus*), and 'other species'. This was done in two parts: a comparison between sites and site sub-catchments in a one-year, pre-treatment baseline; and between treatments and sites over an initial, four-year post-treatment stage. There were differences between the burning and cutting/recently unmanaged treatment plots at baseline at all three sites (note that burning was carried out in a separate sub-catchment to the cutting/unmanaged treatments). There was a tendency for lower cover of poor-fen species (two out of three sites) and higher cover of *Calluna* and *Hypnum jutlandicum* in the burn plots.

4.44 Lindsay (2020) described and interpreted post-treatment trends over time as follows. Background trends over time under the recently unmanaged treatment showed an increase in *Hypnum jutlandicum* cover at two sites (M and N) without an increase *Calluna* cover. This suggests that the bryophyte layer may be re-establishing itself on drier parts at these sites, albeit not yet as peat-forming vegetation. At M, there was also an increase in *Sphagnum fallax* suggesting similar recovery within micro-erosion gullies of a peat-forming sward. W showed a small but steady increase in *Sphagnum papillosum* and common cottongrass *Eriophorum angustifolium* indicating slow recovery of typical bog vegetation. Following burning, the pioneer moss *Campylopus introflexus* became established and increased at all three sites, particularly at the driest site (N). Two sites (M and N) also showed increased frequency of grass and herb species such as wavy hair-grass *Deschampsia flexuosa*, sheep's-fescue *Festuca ovina* and heath bedstraw *Galium saxatile*, which are more typical of dry acid grassland/heath. The third (W) showed a marked increase in leafy liverworts generally associated with dry conditions. In general, the burnt plots had lower cover of typical bog *Sphagnum* species. The most marked effect of cutting was increased cover of *Eriophorum vaginatum* and frequency of poor-fen species at N and W. At M, *Sphagnum fallax* cover increased suggesting enhanced recovery of micro-erosion gullies and/or possible enrichment from brash, together with a marked increase in

Sphagnum capillifolium and *E. vaginatum*, both of which may have benefitted from reduced competition with *Calluna*. It was concluded that "burning appears to be the least beneficial form of management intervention" over the initial post-treatment period.

4.45 The analysis showed that some of the species recorded were not characteristic blanket bog species but rather indicated the presence of other peatland habitats and/or condition states. Thus, differences in responses and successional sequences apparently in progress between sites and treatments were in part reflections of differences in habitats and ecosystem conditions.

Frequency and abundance of key species and groups

4.46 A total of 17 recent studies provided evidence on post-burn changes in the frequency and abundance of key species and groups (excluding palaeoecological studies which are reported separately). Not all studies included pre-burn assessments and/or relied on surveys of nearby not-recently-burnt stands to provide an indication of pre-burn composition and abundance, for example, in chronosequence studies, though such stands may not necessarily differ from burned stands solely in terms of time since burning (para. 4.14).

4.47 The species and groups most frequently reported on were *Calluna* (sometimes lumped with other dwarf shrubs, 16 studies), *Sphagnum* (11 studies), other bryophytes as a group (mostly mosses, sometimes split into acrocarps and pleurocarps, eight studies), *Eriophorum* (in some cases as *Eriophorum vaginatum* or in one case lumped with other graminoids, total of seven studies), and graminoids as a group (sometimes with grasses reported separately, five studies) (Table 10). This indicates a relative dominance in focus on a few species/groups and, in particular, *Calluna* which is predominantly a heath rather than bog species.

Table 10. Post-burn changes in abundance (% cover) of selected key species and groups most frequently reported in recent vegetation studies

Main study reference	Year post-burn	*Sphagnum* spp.	Other bryophytes	*Eriophorum vag.*/spp.	Graminoids/ grasses	*Calluna*/ dwarf shrubs
Calladine *et al*. 2014 [2+, EV-]	10	-	-	-	-	NC
Garnett *et al*. 2019 [2+, EV-]	12	3–0+5	-	-	-	–+80
Grau-Andrés 2018 [1+, EV-]	1+	4+8	68–28	9 NC 10	9 NC 10	64–1
Heinemeyer *et al*. 2020 [1+, EV-]	4	NC <5	*48–20+40*	*5+30*	-	*82–2+25*
Ludwig *et al*. 2018 [2+, EV-]	4 or 5	-	-	-	-	-+
Milligan *et al*. 2018 [1++, EV-]	6,16,59	+–	–+	+	-	–+

Main study reference	Year post-burn	*Sphagnum* spp.	Other bryophytes	*Eriophorum vag.*/spp.	Graminoids/ grasses	*Calluna*/ dwarf shrubs
Noble *et al.* 2018a [1/2+, EV-]	6,16,59, Ref.	+–+	-	-	-	-
Noble *et al*. 2018b [2+, EV++] (EMBER)	2,4,7,10+	ND	-	18–5 *	-	NC
Noble *et al*. 2018b [2+, EV++] (CA)	Unburned, burned	11–8	-	NC	-	11+36
Noble *et al*. 2019b [2+, EV-]	1,5,10+	ND	-	-	[*20*]*+40*	[*75*] *–12+30**
Swindell 2017 [2+, EV-]	Various	–	-	-	-	+
Velle & Vandvik 2014 [1+, EV-]	3	-	*10–2+20**	-	*5–5+8**	*50–5+18*
Velle *et al*. 2012 [1+, EV-]	7	-	*80–2+20**	-	*10–2+10**	*87–2+75*
Whitehead & Baines 2018 [2+, EV-]	up to 17	*–3+10–7*	[*60*] *–12+48*	[*1*]*+16–4*	-	[*95*] *–8+85*
Whitehead *et al.* 2021 [2+, EV-]	up to 10	[*12*]*+30–22*	[*60*] *–43+55*	[*23*]*+55–38*	-	[*60*] *–27+54*
Worrall *et al*. 2013 [1/2+, EV-]*	<1	-	4 NC 5	-	8 NC 10	78–0

Key: arrows indicate direction of change: '+' = an increase; '–' = a decline. NC = no change; ND = no difference. Several arrows indicate changes in direction over time pre- and immediately and longer post-burning. Figures are percentage cover; those in italics are imprecise figures mostly read from graphs and those in square brackets are derived from adjacent not-recently-burnt stands (mostly from chronosequence studies rather than prior to burning followed by monitoring of the same stand post-burn). Ref. = Hard Hill experiment 'reference' plots; CA = condition assessment. Some of the referenced material in this table (as indicated by '*') is cited under a creative commons licence.

Sphagnum abundance

4.48 *Sphagnum* was not identified or reported to species level in most of the 11 recent studies that recorded them (Table 10), though where they were, *Sphagnum capillifolium* which tends to occur in drier conditions than most other bog Sphagna (O'Reilly, 2008), was overwhelmingly the most abundant species (Grau-Andrés and others, 2017b, 2019b/a [both 1+, EV-]; Milligan and others, 2018 [1++, EV-]; and Noble and others, 2018a [1,2+, EV-], the last two both from Hard Hill but using different methods). *Sphagnum fallax*, which tends to occur in conditions with slightly higher nutrient availability than true ombrogenous bog *Sphagnum* species (for example, Kooijman & Kanne, 2013)[15], was equally abundant at another site (Heinemeyer and others, 2019c [1+, EV-]). The generally low frequency and/or

[15] And, unlike other *Sphagnum* species, *S. fallax* is not included as a positive indicator in the upland Common Standards Monitoring guidance for blanket bog (JNCC, 2009).

abundance of other *Sphagnum* species often precluded formal statistical analysis of change in frequency or abundance.

4.49 *Sphagnum* species as a group were reported as showing a range of responses to burning (Table 10), perhaps in part reflecting differences in the presence and abundance of species pre-burn and over time post-burn depending on burn severity (for example, Noble and others, 2019b [2+, EV+]). In addition, the findings are probably influenced by different post-burn resurvey intervals between studies (some lacking the immediate post-burn response and many others only covering relatively short post-burn periods). Some studies suggest small initial declines immediately after burning, often from relatively low pre-burn cover, followed by some initial recovery (Heinemeyer and others, 2019c/2023 [1+, EV-] at two of three sites; Whitehead & Baines, 2018 [2+, EV-]; Garnett, 2023 [2+, EV-]), while others-]). Other studies indicate little or no change, though again generally over relatively short periods (Grau-Andrés and others, 2017b, 2019b [both 1+, EV-]); Heinemeyer and others, 2019c/2023 [1+, EV-] at one of three sites), or similarly no difference between burnt and not-recently-burnt stands (Noble and others, 2018b [2+, EV++], EMBER Pennine data set). Conversely, Noble and others (2018b [2+, EV++]) showed significantly lower *Sphagnum* cover in recently burnt sites than those not-recently-burnt in a second, representative national blanket bog condition data set, though the difference was relatively small. Swindell (2017 [2+, EV-]) mapped *Sphagnum* cover using Earth Observation (EO) (with an unmanned aerial vehicle [UAV] drone) with quadrat ground validation at a Peak District blanket bog site subject to managed burns and affected by a wildfire, which showed highest *Sphagnum* cover in longest-unburned areas (>40 years).

4.50 Findings on *Sphagnum* from the ongoing routine Hard Hill experiment vegetation monitoring were last reported by Milligan and others (2018 [1++, EV-]) for 2013 (two/three years earlier than the Noble and others, 2018a [1,2+, EV-] surveys, para. 4.51). These concentrated on change in probability of occurrence (rather than by ‘abundance’ derived from pin-hits as for other species/groups) over time since 1972 when pin-frames were introduced. *Sphagnum* remained relatively constant under the no-burn since 1954 treatment but increased in the 20- and especially 10-year burn treatments (with a higher probability of occurrence in ungrazed plots but a lower rate of increase). Some of these apparent responses may reflect long-standing differences between the experiment blocks which were described by Hobbs (1984) in relation to *Sphagnum* and other species’ abundance.

4.51 Noble and others (2018a [1,2+, EV-]) reported on a more comprehensive *Sphagnum* survey of the Hard Hill experiment in 2015/16 including the adjacent, longer-unburned ‘reference’ plots only previously surveyed in full in 1965 and 2011. This included frequency from pin-hits, hummock height and maximum patch length/width on transects, and mapping of all individual patches to give location, area and frequency within a square grid, albeit from a single survey. This was eight/nine and 20/21 years after the last burns in the 10-year and 20-year plots, respectively. The findings on frequency/abundance were broadly similar to those from the routine Hard

Hill experiment monitoring reported by Milligan and others (2018 [1++, EV-]), though the survey provides more complete coverage across a wider range of variables and includes the ‘reference’ plots, and hence enables a more nuanced interpretation. A total of 12 *Sphagnum* species was recorded (the same as previously reported by Lee and others, 2013b [1++, EV-], NEER004), though *S*. *capillifolium* was overwhelmingly the most abundant (with pin-hit frequency 27% in the ‘reference’ plots and 18% overall in experiment plots, with no other species >0.5%). The longer unburned ‘reference’ plots and most frequently burned (10-year) treatments had greater *Sphagnum* frequency (and hummock height, para. 4.99) than intermediate treatments (20-year and no-burn-since 1954).

4.52 The abundance of the three most common individual *Sphagnum* species (*S. capillifolium,* lustrous bog-moss *S. subnitens* and *S*. *papillosum*) showed similar patterns. Light grazing had no impact on *Sphagnum*-related variables, nor did it interact with the burning treatments. It is possible that some of the species occurring in <1% of plots, which were not analysed separately, responded differently. For example, fine bog-moss *Sphagnum angustifolium* did not occur in the main experiment plots but was the second most common species in the ‘reference’ plots and conversely Russow’s bog-moss *Sphagnum russowii* occurred only in the experiment plots, but the relative rarity of these species in the plots means that it is difficult to confidently attribute these differences to burning effects. A greater number of species occurred in the 24 experiment plots compared to the four ‘reference’ plots, which would be expected due to the larger size and much greater number of experiment plots and hence total area. Re-analysis of the 1961 data from the experiment plots seven years after the initial 1954 burn showed no significant difference in cover of *Sphagnum* by burn treatment (though only the initial 1954 burn had occurred), grazing or their interaction. However, re-analysis of 1965 data from the (grazed) unburnt since 1954 and longer-unburnt but otherwise comparable ‘reference’ plots found that the latter had significantly greater *Sphagnum* cover. This suggests that the (relatively large, whole block) 1954 burns had a negative effect on *Sphagnum* that was apparent 11 years after burning and which was still observed in the 2015/16 survey over 60 years later. It was concluded that “these results suggest that in some cases fire has a negative impact on *Sphagnum*, and that this can persist for several decades. However, fire return interval and other factors such as atmospheric pollution may alter effects, and in some cases *Sphagnum* abundance may recover. Fire severity and site-specific conditions may also influence effects ..." (Noble and others, 2018a [1,2+, EV-], p. 1).

4.53 Clutterbuck and others (2020 [1,2+, EV-]) also recorded the frequency of *Sphagnum* in 2019 on transects across plots in Block D of the Hard Hill experiment (the highest and perhaps most atypical block) and in two newly established plots outside but adjacent to Block D, one of which was burned in 1954 and the other not (for >30 years previously, c.f. the existing ‘reference’ plots). This showed a similar pattern to the other Hard Hill studies with highest cumulative frequency in the 10-year burn plots and lowest in the not-burned-since 1954 plots, although the new longer-unburned plot had the second highest frequency in sub-plots (c.95+ years since the last burn). Compared with the distribution and cover of *Sphagnum* in the experiment plots at the time of the first survey in 1961, cover was lower in 2019 with it occurring as small, scattered patches rather than a relatively major component of the vegetation cover. This was particularly the case in the 1954 burn sub-plots, which in 2019 had the lowest frequency and was almost absent in the ungrazed sub-plot, though both 1954 burn sub-plots are affected by drainage associated with a meso-scale erosion feature that is likely to have at least contributed to the decline since 1961. Nevertheless, the low frequency in the 1954 burn sub-plots and higher frequency in the new, longer-unburned adjacent plot is consistent with the findings by Noble and others (2018a [1,2+, EV-]) in relation to the similar 'reference' plots, suggesting that the 1954 burn or aftermath was the cause.

4.54 Clutterbuck and others (2020 [1,2+, EV-]) concluded that cessation of burning on an already burnt site had nevertheless resulted in loss of the limited existing *Sphagnum* presence as growth of tussock-forming species, especially *Eriophorum vaginatum*, together with *Calluna*, had proceeded to dominate the bog surface. And that "over a period of several decades, however, mosses such as *Hypnum jutlandicum* have formed continuous swards beneath the *Calluna* canopy, slowing water movement from the site and thereby increasing surface wetness, ultimately providing opportunities for *Sphagnum* to re-establish. In addition, the humid shelter provided particularly by the tussock and/or *Calluna* canopy have also enabled *Sphagnum* to re-establish. Timescales for this process appear to exceed a century".

4.55 Two studies compared the effects of burning with cutting and in one case also no recent management, both of which showed increases in *Sphagnum* following cutting compared with initially little change over time since burning. The first showed a gradual increase at a single North Pennine site following cutting from 4% mean cover pre-management to 5% the year after and 14% after 13 years compared with little change following burning (3% pre-burn, down to 0% after one year and back up to 4% after 13 years) (Garnett, 2023 [2+, EV-]). The second study showed an increase post-cutting after three years (c.11% pre-cutting to c.24% post-cutting), followed by a slight decline to around 20% after eight years. This compared to little change initially following burning (c.5% pre- and a slight decline post-burn) and then a gradual increase (up to c.10% after eight years) on average across three Pennine/Bowland sites (Heinemeyer and others, 2019c /2023 [1+, EV-]). In the second study, *Sphagnum* was much more abundant at one site (Mossdale, the wettest site) where they increased from c.27% pre-treatment to fluctuating up to between c.45 and c.60% up to year nine post-cutting. This compared with an initial slight decline then gradual increase up to c.20% nine years post-burning from the starting cover of c.10% pre-burn (Heinemeyer and others, 2019c /2023 [1+, EV-]). Here, in recently unmanaged plots, *Sphagnum* cover remained high, fluctuating between c.26% initially up to 63% over ten years (also see the ecological interpretation by Lindsay, 2019, of changes across all three sites in para. 4.44).

Other bryophyte abundance

4.56 Most recent studies did not record or report on other (non-*Sphagnum*) bryophytes to species level, though some split them into acrocarpous and pleurocarpous groups and others lumped all species. When individual species were reported or mentioned, several studies noted an abundance of *Hypnum jutlandicum* especially in longer unburned plots.

4.57 Other bryophytes as a group were reported showing initial post-burn declines in cover/frequency in seven studies mostly from high cover pre-burn (Table 10): Velle and others (2012 [1+, EV-], wet heath sites); Velle & Vandvik (2014 [1+, EV-], wet heath sites); Grau-Andrés and others (2019b/a [1+, EV-], raised bog site); Milligan and others (2018 [1++, EV-]); Whitehead & Baines (2018 [2+, EV-]); Heinemeyer and others (2019c [1+, EV-]); and Whitehead and others (2021 [2+, EV-]). In all cases but one (where there was no extended post-burn monitoring (Grau-Andrés and others, 2018 [1+, EV-], raised bog site), the decline was followed by a relatively rapid increase in most cases back up to or towards pre-burn levels. One study showed little difference between pre- and immediate post-burn cover (Worrall and others, 2013a [1,2+, EV-]), probably reflecting very low pre-burn bryophyte cover (4%) on this *Calluna*-dominated, degraded Peak District blanket bog site.

4.58 Only two studies reported on acrocarpous and pleurocarpous moss groups, both showing an increase in acrocarps post-burn: from 4% to 13% immediately post-burn at a Scottish raised bog site (Grau-Andrés and others, 2018/2019b [1+, EV-]) and from c.3% to c.30% at one of three blanket bog sites (Cheviot which had the most bare ground) between around one and three years' post-burn (Noble and others, 2019b [2+, EV+]). Pleurocarps declined immediately post-burn at the Scottish raised bog site from 64% to 20% (with a similar decline at the dry heath site from 41% to 5%) (Grau-Andrés and others, 2018/2019b [1+, EV-]), but remained at relatively high cover at three blanket bog sites and showed no significant relationship with burn age class, though they increased at one site (Cheviot where they were initially at the lowest cover) within the youngest age class between around one-three years post-burn (Noble and others, 2019b [2+, EV+]).Three other individual moss species were reported as showing significant change or differences between treatments. *Campylopus introflexus* was more abundant and increased over time under the burn treatments, especially the shorter ten-year treatment, but declined in the no-burn since 1954 control in the Hard Hill experiment (Milligan and others, 2018 [1++, EV-]). It also occurred at greater cover in the more recently burnt age classes than the 10+ years class across ten Pennine (EMBER) sites (Noble and others, 2018b [2+, EV++]). Conversely, *Hypnum jutlandicum* showed a large increase in the unburned since 1954 Hard Hill plots (especially where ungrazed), a delayed increase in the 20-year burn and no increase in the ten-year burn plots (Milligan and others, 2018 [1++, EV-]), and a large immediate decline from 60% pre- to 19% post-burn at a Scottish raised bog (Grau-Andrés and others, 2019b/a [1+, EV-]). Nodding thread-moss *Pohlia nutans* declined over time in all treatments at Hard Hill but most in the 10-year treatment (Milligan and others, 2018 [1++, EV-]).

4.59 One study comparing the effects of burning with cutting and no recent management showed an initial post-burn decline in other bryophyte cover, followed by an increase back to close to pre-burn levels after four years, compared with relatively little change following cutting and a gradual increase to the highest cover with no management intervention (Heinemeyer and others, 2019c [1+, EV+]).

4.60 Three species of liverwort, Mueller's pouchwort *Calypogeia muelleriana*, two-horned pincerwort *Cephalozia bicuspidata* and tumid notchwort *Lophozia ventricosa*, showed similar declines in the unburned since 1954 plots at the Hard Hill experiment, with abundance higher in the burn treatments where *C. muelleriana* increased over time while the other two species declined (Milligan and others, 2018 [1++, EV-]).

Lichen abundance

4.61 Total lichen cover declined over time in all treatments at the Hard Hill experiment but at different rates (fastest in the unburned since 1954 control; intermediate in the 20-year; and slowest in the ten-year burn treatments) and was slowed by grazing exclusion (Milligan and others, 2018 [1++, EV-]).

Eriophorum and other graminoid species abundance

4.62 Seven recent studies provide data on the response of *Eriophorum,* most of which recorded to species level, with *E. vaginatum* the most abundant species, though *E. angustifolium* was also frequent. In addition, four studies reported on other graminoids that were most likely dominated by *Eriophorum* or other sedges (Table 10).

4.63 Most of the studies showed a post-burn increase in *Eriophorum* cover (Milligan and others, 2018 [1++, EV-]; Heinemeyer and others, 2019c/2023 [1+, EV-]; Whitehead & Baines, 2018 [2+, EV-]; Whitehead and others, 2021 [2+, EV-]), though two studies indicated little change in *Eriophorum* (Grau-Andrés and others, 2019b/a [1+, EV-], raised bog site) or graminoids (Worrall and others, 2013a [1,2+, EV-]) immediately post-burn, likely prior to any post-burn response. The magnitude of the post-burn increase in cover varied between studies, for example, reaching a peak of only c.16% on a *Calluna*-dominated North Pennine blanket bog (Whitehead & Baines, 2018 [2+, EV-]) compared with c.55% on a less-modified, but still *Calluna*-dominated, southern Scottish blanket bog (Whitehead and others, 2021, [2+, EV-]).

4.64 Whitehead & Baines (2018 [2+, EV-]) and Whitehead and others (2021 [2+, EV-]) both recorded lower cover of *Eriophorum* in longer unburned plots, though still higher than in the most recently burned age classes. In the Hard Hill experiment, *Eriophorum vaginatum* was more frequent under the burn treatments, especially the shorter (ten-year) rotation, than in the unburned since 1954 plots where it still occurs at moderate frequency. In addition, Noble and others (2018b [2+, EV++]) showed no difference in *Eriophorum* cover between burnt and not-recently-burnt samples in a national condition assessment data set and higher (though still only moderate, 18%) cover in not recently burned compared with recently burned plots in a second vegetation data set from plots included in the EMBER project.

4.65 Two additional studies on Norwegian coastal wet heaths showed similar initial post-burn declines in grasses/graminoids followed by rapid increases to above pre-burn cover within a few years (Velle and others, 2012 [1+, EV-]; Velle & Vandvik, 2014 [1+, EV-]), though these did not include *Eriophorum* with most of the records being either sedges or grasses.

4.66 One study across three Pennine/Bowland sites, which compared the effects of burning with cutting and no recent management, showed a greater increase in *Eriophorum* cover following cutting (up from c.15% pre- to c.45% five years post-burn followed by a gradual decline) than with burning (c.5% to c.33%), compared to relatively little change (c.20% followed by a slight decline) under no recent management (Heinemeyer and others, 2019/2023 [1+, EV-]). There were some differences between the sites, with a greater increase following burning at the wettest site where it achieved and maintained a similar cover to that following cutting after four years post-burn.

Calluna and other dwarf shrub abundance

4.67 Thirteen recent studies provide data on the response of *Calluna* (11 studies) and dwarf shrubs as a group (two) to burning (Table 10). It is important to note that *Calluna* is "not typically a wetland species, but [is] often found growing on tussocks [and hummocks] of other species on ombrogenous mire and at the edges of soligenous mire" (Grime and others, 1988, pp. 144–155), with its centre of distribution in heaths.

4.68 Ten studies reported an immediate or initial post-burn decline of *Calluna* often down to a few percent cover (excluding dead cover), usually from dominance pre-burn. For example, cover declined from 78% pre- to 0% immediately post-burn at a Peak District blanket bog site (though see para. 4.103 regarding *Calluna* 'stick' and litter) (Worrall and others, 2013a [1,2+, EV-]), and from 64% to 1% at a Scottish raised bog 17 months after burning (Grau-Andrés, 2019b/a [1+, EV-]). In the nine studies that included subsequent monitoring, the decline was followed by a relatively rapid recovery back towards, and in some cases reaching, pre-burn cover typically within around 10–12 years, for example, reaching 80% cover on a North Pennine blanket bog after 12 years (Garnett, 2023 [2+, EV-]) and c.75% on Norwegian coastal wet heaths after just seven years (Velle and others, 2012 [1+, EV-]).

4.69 Noble and others (2018b [2+, EV++]) showed significantly higher cover and frequency of *Calluna* in burned compared to unburned plots in a national blanket bog condition assessment sample across 85 sites (but lower cover in grazed plots which may be the cause when grazing pressure is high. Although there was no difference in cover between burn age categories in a second, smaller Pennine (EMBER) data set covering ten sites.

4.70 A study across three Pennine/Bowland sites comparing the effects of burning with cutting and no recent management showed the same large decline in *Calluna* cover down to a few percent immediately following burning, then a slightly greater initial increase in the two years following cutting compared to burning, followed by a similar increasing trajectory under both treatments up to five years post-burn (up to c.35% cover) (Heinemeyer and others, 2019c/2023 [1+, EV-]). This was then followed by a continued gradual increase under cutting, but a slight decline then gradual recovery under burning which left cover slightly below that under cutting after nine years (c.40%). The authors put this down to heather beetle reducing cover mainly on burnt plots. The pattern differed between sites, with recovery most rapid following burning at the driest site (Whitendale), achieving similar cover (c.70%) to untreated plots after eight years, with slower recovery following cutting (to c.60% after 9 years). The wettest site (Mossdale) showed much slower recovery of *Calluna* under both treatments (to c.22% after nine years post-burn compared to c.40% under cutting). The other site showed intermediate recovery, perhaps reflecting the more recent impact of heather beetle.

4.71 Ludwig and others (2018 [2+, EV-]) reported that *Calluna* cover increased by a third following reintroduction of grouse moor management on a previously heavily grazed Scottish blanket bog site, though this reflected a range of management changes (reduced grazing, *Calluna* reseeding on ‘grass moor’, predator control and cutting) in addition to burning, and it was suggested that the increases were related to grazing reductions rather than burning or cutting. Conversely, Calladine and others (2014 [2+, EV-]) reported no change in cover of *Calluna* (or other selected key species) over ten years following changes in management involving a combination of reduced grazing and localised winter burning (*Calluna*) and cutting (mostly graminoids) on a Scottish blanket bog site that included areas of “degraded” and “intact” bog. These differences in response may reflect the introduction of a different suite of management changes with burning effects not separated out.

4.72 Muñoz and others (2014 [2-, EV-]) reported a decline in dwarf shrub cover (mostly western gorse *Ulex gallii* and cross-leaved heath *Erica tetralix* but also Dorset heath *Erica ciliaris* and *Genista berberidea*) from 96% to 76% (and a corresponding slight increase in herbaceous vegetation) in a resurvey of ten wet heath sites in NW Spain after 28 years. It was concluded that this reflected change from “traditional management” (grazing, cutting and “sporadic” burning) towards more intensive uses, in particular “repeated” burning and tree plantations.

4.73 Few other individual dwarf shrubs were reported as showing significant responses to burning, though single studies reported declines or lower cover associated with burning for crowberry *Empetrum nigrum* (Milligan and others, 2018 [1++, EV-]) and *Vaccinium myrtillus* (Newey and others, 2020 [2+, EV++]). In the latter, *V. myrtillus* was most prevalent across Scottish grouse moors in areas with low to intermediate intensity of burning, with lower prevalence at the highest intensity of burning.

Other species and groups showing significant changes in abundance

4.74 Only a few other individual species or groups were reported as showing significant responses to burning in recent studies. These included declines or lower cover associated with burning for *Rubus chamaemorus* (Milligan and others, 2018 [1++, EV-], only in the 10-year burn treatment) and birch *Betula* (Newey and others, 2020

[2+, EV++]). In the latter, *Betula* was most prevalent across all Scottish grouse moors in areas with low to intermediate intensity of burning, with lower prevalence at the highest intensity of burning. Herbs/forbs were reported as showing a slight increase (associated with a decline in dwarf shrubs) in a resurvey of ten wet heath sites in NW Spain after 28 years reflecting a trend towards more intensive uses, including more frequent “repeated” burning (Muñoz and others, 2014 [2-, EV-], para. 4.72).

Species of conservation concern

4.75 Few recent studies specifically report on species of conservation concern (see NEER004, para. 4.8–4.10, p.18) or other scarce or notable species, though some of the study findings reported under overall vegetation composition and abundance of key species refer to some such species, in particular *Sphagnum* as a group and some individual species within it (see paras. 4.48–4.55).

4.76 Lesser twayblade *Neottia cordata* is a locally scarce orchid in (mostly northern) England, Wales and especially Scotland, associated with *Sphagnum* and sometimes other moss mats, often below an open *Calluna* canopy, on blanket bog and wet heath. It has shown historic declines, sometimes to local extinction, for example, in the southern Pennines by early to mid-last century (Elliott, 1953 [2-, EV+], NEER004]; Preston and others, 2002; Kull & Hutchings, 2006; Ritson & Lindsay, 2023), though it is inconspicuous and may be under-recorded (Kotilínek and others, 2017). It was first recorded at the Hard Hill experiment in the ‘reference’ plots in 2011 (Lee and others, 2013a [1++, EV-], NEER004, Appendix 3, p. 91) and subsequently in other areas adjacent to the experiment plots (Clutterbuck and others, 2020 [1,2+, EV-]; Natural England, unpublished records from 2016 or earlier) but not within them despite repeated vegetation surveys back to 1961. Kotilínek and others (2017) suggest that it is sensitive to repeated burning of moorland based on Elliott (1953 [2-, EV+], NEER004) and supported by Anderson & Shimwell (1981) and Carey and others (2023). Its (re-)occurrence at Hard Hill may reflect an apparent increase in the species across the wider, long-unburned Moor House NNR.

Plant propagules

4.77 Only one recent study has reported on the relationship between burning and plant propagule frequency in peat and litter, based on two Pennine blanket bog areas: the Hard Hill experiment in the North Pennines (propagules in peat) and a burn chronosequence on severely modified, *Calluna*-dominated bog across three sites in the Peak District (propagules in peat and litter) (Lee and others, 2013b [1,2+, EV-]) (also see para. 4.39 on propagule species richness).

4.78 Two analyses were conducted on data from Hard Hill. In the first, from the grazed experiment plots and adjacent grazed 'reference' plots, *Calluna* seed density and *Sphagnum* propagule frequency was highest in the longest unburned 'reference’ plots and lowest in the most recently burned sub-plots, with very low frequency of *Sphagnum* under both burn treatments. In the second analysis from the grazed and fenced experiment plots, *Sphagnum* frequency increased with time since last burn (as in first analysis) as did species-richness (though it was low, para. 4.39). Perhaps surprisingly, *Polytrichum* showed an increase in frequency with time since burning and in grazed treatments. *Calluna* seed density again increased with increasing time since burning (though it was much lower than at the Peak District sites), but also showed an interaction with grazing with greater seed densities in the ungrazed treatments (apart from under the 20-year burn treatment). Thus, species-richness, frequency of *Sphagnum* and *Polytrichum* and density of *Calluna* increased with time since burning suggesting that "continuous production of seeds, spores and plant fragments … were transferred to the surface peat and were accumulated. In contrast, when the fire return-interval was reduced … there was a significant decrease in the propagule bank … reducing the vegetation regeneration potential from the peat bank (Driscoll and others, 2010). Thus, relatively frequent fire return-intervals could produce population extinctions or undesired compositional and structural changes in vegetation (Bradstock and others, 1998)" (Lee and others, 2013b [1,2+, EV-], p. 194).

4.79 The authors suggested "that prescribed burning rotations simultaneously at two temporal scales within a moorland landscape may be needed to conserve *Sphagnum* species: short-rotation burns (every 10-years) to enhance its abundance in the vegetation and long-rotations (>55 years) to maintain *Sphagnum* propagules in the surface peat" (Lee and others, 2013b [1,2+, EV-], pp. 187, 196). But given the relatively high frequency and abundance of *Sphagnum* in the longer-unburned 'reference' plots (paras. 4.51–4.52) and across much of the rest of the unburned Moor House NNR (for example, Eddy and others, 1968; Williams-Mounsey and others, 2023), there does not seem to be a need to burn to maintain this. Apparent post-burn *Sphagnum* recovery under short rotations at Hard Hill may be because the burns are less severe than in the 20-year treatment (Noble and others, 2019a [1,2++, EV-]) and than in normal practice, and/or because of potential rapid ingress from adjacent, frequent *Sphagnum* outside the burn plots. This is less likely to be the case in areas where *Sphagnum* is scarce/absent and/or where rotational burning management occurs across a higher proportion of a site. It might also reflect reduced shading from the *Calluna* canopy, though at this relatively high-altitude site, the regrowth of *Calluna* is relatively slow and the cover and canopy is incomplete under the 10-year burning treatment.

4.80 There was only sufficient data from the Peak District sites to analyse *Calluna* and *Juncus effusus* and some bryophytes in relation to time since burning. For *Calluna*, seed density was much higher than at Hard Hill and greater in peat than litter. While elapsed time since burning had no effect on peat seed density, in the litter layer it increased with time since burning. *J. effusus* increased with time from burning in litter at one site. For bryophytes, most effects related to differences between the three sites, though, as would be expected as an early pioneer, *Campylopus introflexus* declined over time since burning.

Moss depth

4.81 Only two recent chronosequence studies reported on moss depth over time since burning (Noble and others, 2019b [2+, EV+]; Whitehead and others, 2021 [2+, EV-]). Both studies showed lowest moss layer depth in the youngest burn age class: 12–18 months post-burn from two of three Pennine/Cheviot blanket bog sites (Noble and others, 2019b [2+, EV+]) and up to four years compared with unburned for ≥11 years from a Scottish blanket bog site (Whitehead and others, 2021 [2+, EV-]). In another study, moss depth was reported indirectly as immediate moss/litter consumption as a proxy for fire severity which was experimentally manipulated by simulating drought using 'rain-out' shelters (Grau-Andrés and others, 2017a [1+, EV-]; Grau-Andrés and others, 2019b/a). Depth declined by 0.1 cm in 'no-drought' and 1.4 cm in simulated 'drought' (increased severity) burn plots. A similar, likely related, effect has been reported for moss/bryophyte biomass, with increasing biomass over time since burning (Alday and others, 2015 [1,2+, EV-], paras. 4.85–4.86; Ward and others, 2007 [1,2+, EV-], Appendix 3, p. 90, in NEER004).

***Calluna* growth stages**

4.82 Only one recent study reported on the frequency of *Calluna* growth stages from across a sample of 20 long-term English moorland monitoring sites (Critchley and others, 2016 [2++, EV++]/ADAS and others 2017). Recent burning was recorded in mire samples (most blanket bog but some wet heath and valley mire/fen) at eight sites (57% of sites with mires present). The mean per site was 14% of mire samples recently burnt (range 8–26% per site, with 4% burnt 0–2 years and 10% burnt 3–4 years previously; also covered under Section 11 on burning extent). The mean percentage of *Calluna* growth stages in sites with burnt mire samples was 10% pioneer, 54% building, 34% mature and 5% degenerate. This was relatively similar to six not recently burned sites (0.5% pioneer, 72% building, 23% mature and 4% degenerate). The lower pioneer and higher building percentages might reflect recent reduction or cessation of burning on mires at these not recently burned sites, which may also be the case for some of the recently burned sites. This indicates low frequency of older *Calluna* growth stages.

Vegetation biomass

4.83 Eight recent studies provide data on vegetation biomass consumption and/or post-burn production and accumulation of key species/groups following burning and, in one case, additionally cutting. This was usually for *Calluna*, litter, bryophyte and graminoid and in some cases other vascular plant fractions, though in two cases these only covered relatively short, initial post-burn periods (see Table 16 in Section 6 on carbon). The findings from these studies are also relevant to the sub-questions on carbon (Section 6), fire severity/behaviour (Section 8) and the relationship with wildfire (Section 9). All included blanket bog sites, in two cases in Scotland and five in northern England, and one study also included lowland heath sites in Dorset. Three of the studies took place in or used data from the Hard Hill burning and grazing experiment (Ward and others, 2012 [1,2+, EV-]; Alday and others, 2015 [1,2+, EV-]; Santana and others, 2016 [2+, EV+]). The four others measured pre- and post-burn biomass from modified *Calluna*-dominated blanket bog sites (Worrall and others, 2013a [1,2+, EV-]; Clay and others, 2015 [2+, EV-]; Davies and others, 2016a [2+, EV+]; Heinemeyer and others, 2019c [1+, EV+]). Two involved matrix modelling of biomass accumulation (Alday and others, 2015 [1,2+, EV-]; Santana and others, 2016 [2+, EV+]).

4.84 The most comprehensive data for Hard Hill come from Alday and others (2015 [1,2+, EV-]) who sampled all the experiment treatment and 'reference' plots in 2011 (five years post-burn in 10-year plots, 16 years in 20-year plots, 57 years in no-burn since 1954 plots and ≥87 years in the 'reference' plots) (Figure 4). Total aboveground vegetation biomass comprised three main fractions: litter (range 36–67% across treatments), *Calluna* (5–43%) and bryophytes (6–27%). As would be expected, total biomass (and vegetation height) were reduced by burning (but not grazing), being greatest in the no-burn and 'reference' plots.

4.85 Total biomass was significantly lower in the 10- and 20-year burn treatments, though only the shortest (10-year) treatment reduced *Calluna* biomass (and vegetation height). However, modelled growth curves showed an absolute growth rate (AGR) peak in *Calluna* biomass after eight years (and four years for height, para. 4.92), an apparent asymptote (levelling-off) for *Calluna* biomass after 20 years (and vegetation height after 15 years), with no increase and a decline in mean *Calluna* biomass in the longest unburned 'reference' plots compared with the no-burn since 1954 treatment (705 g m^{-2} compared with 834 g m^{-2} under no burn since 1954, data from Marrs, 2015). This decline may be linked to increased occurrence of *Calluna* layering in *Sphagnum* moss within the 'reference' plots. Bryophyte biomass was greatest in the no-burn and 'reference' plots and lowest in the longer (20-year) burn-treatment. This probably reflects an initial post-burn increase and subsequent decline in acrocarps, followed by a gradual increase of pleurocarps over time, peaking in the longest unburned plots. Litter biomass was not affected by burning treatments (Figure 4). The much smaller biomass of graminoids was greatest in the 10-year treatment independent of grazing treatment, but that of 'other vascular plants' was greatest in the 10-year ungrazed plots.

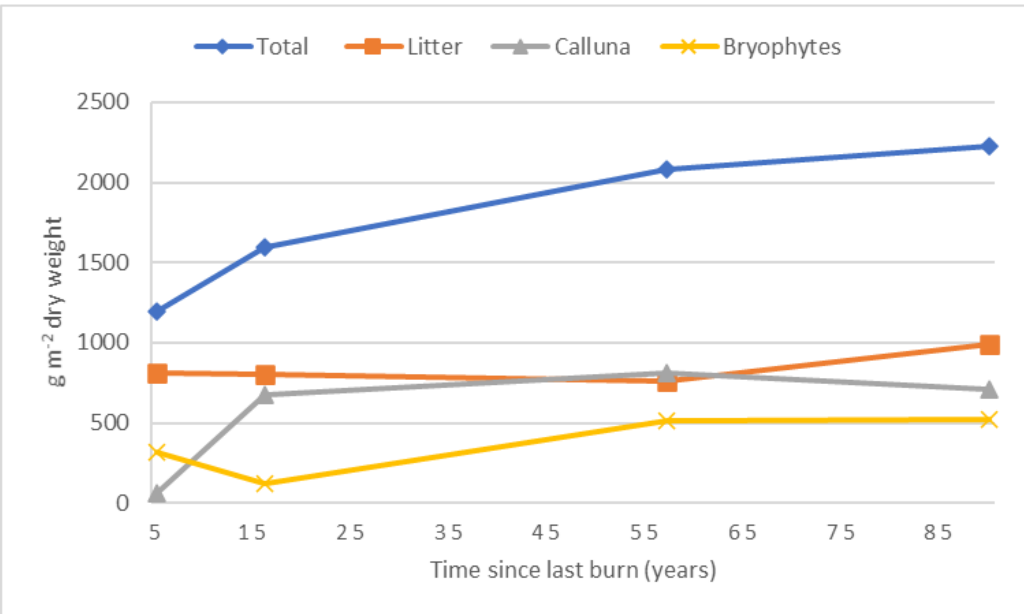

Figure 4. Mean aboveground biomass accumulation for litter, *Calluna* and bryophyte fractions, and in total, against years elapsed since the last burn for the 10-year (at five years), 20-year (16 years), no burn since 1954 (57 years) and longer-unburned 'reference' plot (c.90 years) treatments in the Hard Hill experiment at Moor House NNR, North Pennines, in 2011. Data from Alday and others (2015 [1,2+, EV-], Table 2).

4.86 Ward and others (2012 [1,2+, EV-]) sampled total (aboveground live) vegetation biomass shortly (18 months) after burning in the Hard Hill 10-year treatment plots and the not burned since 1954 plots (after 53 years). Burning reduced total biomass by over 70% reflecting a substantial reduction in dwarf shrubs (down from 661 to 30 g m^{-2} dry weight in samples not burned since 1954 and burned 18 months previously, respectively). This altered the relative contribution of the three plant functional groups by increasing the proportion of total biomass represented by graminoids and bryophytes/lichens. (In contrast, grazing did not affect vegetation biomass and there was no interaction detected between burning and grazing.)

4.87 Santana and others (2016 [2+, EV+]) compared the aboveground biomass accumulation data from Hard Hill blanket bog (Alday and others, 2015 [1,2+, EV-], para. 4.84-4.85) with similar existing biomass accumulation data from three other *Calluna*-dominated sites/areas: a Scottish dry heath (Miller, 1979, at Kerloch), a Peak District more-modified blanket bog (Allen and others 2013 [2+], NEER004, at Howden Moor, para. 4.89) and three lowland dry heath sites in Dorset (Chapman and others, 1975). Modelled accumulation post-burn patterns through time differed between sites, though they were not ordered along the north-south gradient. Hard Hill, one of the sites with colder temperatures and higher precipitation, had the lowest *Calluna* biomass values growing slowly until 20 years after the burn with what was referred to as an asymptote around 8 t ha^{-1}, much lower than the other sites. The two sites at the extremes of the climatic gradient (Kerloch and Dorset) showed similar, intermediate accumulations with growth occurring over the first 20 years until an apparent asymptote of around 20 t ha^{-1} was achieved c.25 years post-burn. These two dry heath sites also regenerated more quickly and reached the greatest biomass values more quickly after burning. *Calluna* biomass at Howden, the site ranked as the second warmest and driest (after Dorset) had the greatest biomass, increasing linearly until c.35 t ha^{-1} 50 years after being burnt.

4.88 Accumulation patterns for litter also differed between sites following a similar pattern, though litter showed different responses at Kerloch and Dorset where *Calluna* accumulation was similar. Litter accumulated faster at Kerloch in the first few years towards an asymptote at approximately 20 years, whereas litter accumulation in Dorset followed a clear sigmoidal curve with an early lag-phase (0–10 years), and a phase of rapid increase (10–30 years) before reaching an asymptote around 30 years. The asymptotes for these sites were also different; Dorset sites reached 29 t ha^{-1} compared to 20 t ha^{-1} at Kerloch. At Howden litter increased linearly until c.35 t ha^{-1} was accumulated 50 years after burning. At Moor House litter accumulated quickly in the first ten years but then reached an asymptote of c.9 t ha^{-1} resulting in it being the site with the lowest litter asymptote. The fate of the litter carbon is likely to be different between different sites/habitats. In a healthy, peat-forming system, the litter is likely to be gradually turned into peat and sequestering carbon. In dry heath, the litter is likely to be decomposing with little or no long-term sequestration.

4.89 Four recent studies provide data on post-burn biomass consumption mostly of aboveground surface fuels following burning, which range from 71% to 93%. Worrall and others (2013a [1,2+, EV-]) reported a pre-burn total aboveground biomass of 880 g m^{-2} (90% from overstory) from a Peak District *Calluna*-dominated, modified blanket bog, c.75% of which was lost through combustion immediately post-burn. Clay and others (2015 [2+, EV-]) estimated mean aboveground biomass of 860 g m^{-2} from not-recently-burnt adjacent *Calluna*-dominated blanket bog across two paired sites in Northumberland, c.80% of which was lost on combustion (with pre-burn biomass not recovered until after 18.6 years). Heinemeyer and others (2019c [1+, EV-]) determined aboveground dry *Calluna* biomass from 30 cm diameter flux chamber-sized circles from three *Calluna*-dominated, modified Pennine/Bowland blanket bog sites of 390 g m^{-2} which was reduced by 93% following burning (and cutting).

4.90 An assessment by Davies and others (2016a [2+, EV+, NEER014]) of five peatland wildfires, three in England and two in Scotland, and 27 managed burns across two Scottish *Calluna*-dominated heath sites (the latter reported in Legg and others, 2007), found that the consumption of surface fuels (heather and graminoids) was a roughly constant proportion of pre-fire fuel load. Modelling indicated a positive linear relationship between pre-fire aboveground biomass and mean fuel consumption. Controlling for Fire Weather Information variables (the dryness of ground conditions as assessed using the Canadian Fire Weather Information System indices) did not improve model fit. No such relationship was found for ground fuels. It was noted that drier moorland community types appear to be at greater risk of severe burns than blanket bog communities.

Vegetation height and structure

Vegetation height

4.91 Nine recent studies reported on the effect of burning on vegetation height. All were on or included blanket bog, though one also included predominantly ‘upland heath’ (19 predominantly <40 cm peat c.f. 17 blanket bog) sites across northern England (Robertson and others, 2017 [2+, EV++]). In most cases, height related to all vegetation (five studies, though in most cases this related to *Calluna* as the dominant tall plant) or *Calluna*/dwarf shrubs (four), in two cases graminoids, in one *Juncus*, and in some studies several of these species/groups. Thus, it seems that in most cases it involved vascular plants and likely was measured from the moss carpet layer. Vegetation height is not normally regarded as a key attribute of peatland habitats. For example, it is not included in the upland CSM guidance (JNCC, 2009). It may be more important for associated species groups such as birds and invertebrates and, indeed, was often recorded in bird studies included in this review. It, together with structure, may also indicate the effect of burning and other management interventions including cutting and grazing.

4.92 As would be expected, most plot-based studies reported a decline in height immediately post-burn (Heinemeyer and others, 2019c [1+, EV-]) or lowest height in the youngest age-class in chronosequence studies (Whitehead & Baines, 2018 [2+, EV-]; Noble and others, 2019b [2+, EV+]; Whitehead and others, 2021 [2+, EV-]); and/or subsequent relatively rapid increases in height (Alday and others, 2015 [1,2+, EV-]; Heinemeyer and others, 2019c [1+, EV-]) or increasing height in increasingly older age-classes (Whitehead & Baines, 2018 [2+, EV-]; Noble and others, 2019b [2+, EV+]; Whitehead and others, 2021 [2+, EV-]). However, all these studies monitored recovery in height over relatively short timescales apart from Alday and others (2015 [1,2+, EV-]) who showed a stabilisation in height (and biomass) with a modelled 'asymptote' of 15 years (and 20 years for *Calluna* biomass, para. 4.85) in the Hard Hill experiment. This reflected an absolute growth rate (AGR) peak in height after four years (and eight years for biomass). Consequently, only the short, ten-year burn rotation treatment significantly reduced vegetation height (and *Calluna* biomass) compared with the longest unburned 'reference' plots.

4.93 Studies monitoring vegetation across larger-scale moorland blocks or compartments reflected the patchwork of rotationally burnt patches and not-recently-burnt areas and, as might be expected, showed less clear responses to burning than smaller plot studies. They also potentially reflect multiple other larger-scale management treatments (for example, cutting, grazing and restoration interventions). Thus, Douglas and others (2017 [2+, EV-]) reported a reduction in mean dwarf-shrub height at Geltsdale Nature Reserve (N Pennines) when a greater percentage area of a compartment was burned between surveys. However, on a SW Scottish blanket bog undergoing habitat restoration management, Calladine and others (2014 [2+, EV-]) reported no overall detectable change in any of three vegetation height/structure metrics across all compartments, though year-on-year variation differed among compartments. Similarly, there was no overall change in *Calluna* height across blocks after reinstatement of either grouse-moor management or habitat restoration at Langholm Moor Demonstration Project (SW Scotland) (Ludwig and others, 2018 [2+, EV-]). This may reflect a similar lack of overall change in an index of *Calluna* management intensity (though vegetation monitoring in chronosequence plots at the same site indicated shorter vegetation height in the youngest burn age-class) (Whitehead and others, 2021 [2+, EV-], para. 4.92). Nevertheless, variance in *Calluna* height decreased (and vegetation density increased) overall across blocks within the site, whereas variation in heather height was positively associated with burning extent across a sample of 36 moors in northern England (Robertson and others 2017 [2+, EV++]), which also relate to vegetation structure (para. 4.95).

4.94 In a comparison between burning and cutting, *Calluna* regrowth across three Pennine/Bowland *Calluna*-dominated blanket bog sites was initially slower on burnt than on cut plots, although after four years, height (and cover) was similar under both treatments (Heinemeyer and others, 2019c [1+, EV-]).

Vegetation structure

4.95 A number of vegetation height variables and derived indices mentioned in part above under vegetation height, also relate to vegetation structure, including variation in vegetation height and density. These were mostly from relatively large-scale vegetation surveys across blocks or compartments subject not just to burning but other management interventions (para. 4.93). Variance in *Calluna* height decreased (and vegetation density increased) overall across blocks within one Scottish site (Ludwig and others, 2018 [2+, EV-]), while variation in heather height increased with increasing burning extent across a sample of 36 moors in northern England (Robertson and others, 2017 [2+, EV++]). On single sites with mixed burning, cutting, grazing and restoration management, indices of vegetation density increased at one site (Ludwig and others, 2018 [2+, EV-]) but showed no effect at the other (Douglas and others, 2017 [2+, EV-]). These mixed responses are likely to reflect the range of different management interventions being applied as well as underlying conditions.

4.96 Milligan and others (2018 [1++, EV-]) reported on the effects of burning and grazing on vegetation structure in the Hard Hill experiment on the basis of multiple pin-hits in four height strata which overall, as would be expected, showed increasing height with increasing age since burning across the 10-year, 20-year and no-burn since 1954 treatments. Modelled responses showed clear differences between burn treatments with most pin-hits in the 10-year burn treatment in the lowest two strata (up to 20 cm), but a curvilinear height profile in the 20-year and no-burn since 1954 treatments with a lower number of hits in the bottom (0–10 cm) and top (>30 cm) layers and a greater number of hits in mid profile (10–30 cm).

Microtopography

4.97 Only three recent studies provide information on the relationship between burning and microtopography (Noble and others, 2018a [1,2+, EV-]/O'Reilly, 2016; Clutterbuck and others, 2020 [1,2+, EV-]/Clutterbuck & Midgley, 2015; Heinemeyer and others, 2019c [1+, EV-]) despite this being a key characteristic of peatlands that relates to composition, structure and function (for example, Lindsay, 2010; Joosten and others, 2017; Crowle and others, in press).

4.98 The most comprehensive data on microtopography come from recent surveys of the Hard Hill experiment plots at Moor House NNR using a variety of techniques including terrestrial laser scanning (TLS), fixed-wing unmanned aerial vehicle (UAV) to collect ultra-high-resolution colour (RGB) imagery UAV, airborne Lidar and derived digital elevation and surface models (DEM, DSM). This was initially carried out as a TLS trial in 2015 (Clutterbuck & Midgley, 2015) followed by a complete survey of Block D between 2017–19 (Clutterbuck and others, 2020 [1,2+, EV-]) which has been extended to the other blocks but not yet been reported (Lindsay & Clutterbuck, in prep.). These included descriptions of the gross morphology of the plots and surrounding areas, assessment of microtopographic variation across the plots and (for plots in Block D, plus two new adjacent, longer not-burned plots) definition of nanotope features (Lindsay, 2010) and associated synusial vegetation, and the background microtopographic network (some aspects of which are covered under Sections 6 and 7 on carbon and water). Microtopographic variation was greater in the not-burned since 1954 plots than in all the 10-year burn sub-plots and the majority of the 20-year burn sub-plots, which it was suggested has maintained the bog surface in “a state of arrested development”. The dominant nanotope features of both the fenced and grazed 10-year burn plots and the fenced 20-year plot in Block D were micro-erosion and tussocks, with the former much scarcer in the 1954-burn treatment plots as were tussocks in one of the new pre-1954-burn plots.

4.99 Noble and others (2018a [1,2+, EV-]/O'Reilly, 2016) also surveyed bog nanotopes, and *Sphagnum* hummock height and patch area across the Hard Hill experiment and ‘reference’ plots in 2015/16, eight/nine and 20/21 years after the last burns in the 10-year and 20-year plots, respectively. O'Reilly (2016) reported on microtope frequency indicating differences between blocks, with blocks B and C having more frequent ‘hummocks’ (T3) and less frequent ‘high ridge’ (T2), as did the 10-year and longer-unburned ‘reference’ plots compared to the not burnt since 1954 (grazed) experiment plots, the last reflecting just the initial 1954 burn over the period of the experiment. ‘Tussocks’ (Tk) showed only minor differences in frequency between blocks and treatments. The unburned ‘reference’ and most frequently burned (10-year) plots had greater *Sphagnum* hummock height (and abundance, para. 4.51) than the intermediate treatments (20-year and no-burn since 1954), though patch area was not significantly associated with burning status (Noble and others, 2018a [1,2+, EV-]).

4.100 Heinemeyer and others (2019a [1,2+, EV-]/2019c) compared the effect of burning and cutting on microtopography after two years based on the mean and standard deviation of height offsets from a 20 cm level measured over plot transects in relation to the plot peat surface level at the start and end points of transects, compared to average peat surface level outside the plot. On average, vegetation was mown to about 12 cm above the peat surface. Cutting reduced the plot height offset by about 2 cm, which was suggested as mostly due to removing the tops of “sedge hummocks” (presumably *Eriophorum vaginatum* tussocks), whereas burnt plots did not differ from recently unmanaged plots (albeit in different sub-catchments), though at one site they showed higher variability in the offset. The assessment was not repeated, so the longevity of the effect is unknown.

Bare ground and litter

Bare ground

4.101 Only five recent studies reported on changes in cover of bare ground, all of which showed an initial post-burn increase, though in only three cases did this specifically relate just to bare ground rather than being combined with ‘duff’[16] and ‘brash’/’burnt’. These three studies showed an increase from 0% pre-burn up to between c.25 and 50% across six Norwegian coastal wet heath sites three years after burning (Velle &

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[16] Decaying and decayed organic matter usually below but sometimes including the litter level.

Vandvik, 2014 [1+, EV-]); a small increase from c.7% pre-burn to c.10% immediately post-burn on a *Calluna*-dominated, modified Peak District blanket bog site (Worrall and others, 2013a [1,2+, EV-]) (though see para. 4.103 regarding post-burn *Calluna* stick and litter cover); and significantly greater cover 12–18 months after burning than in older age classes from chronosequences at two of three northern English (Peak District and Cheviot) blanket bog sites (up to c.20 and c.60% cover, respectively), with a decline recorded in repeated resurveys over the following ten months at the site with highest bare ground down to c.10% and none in the 10+ years category (Noble and others, 2019b [2+, EV+]). Bare ground has previously been reported from the routine, long-term vegetation monitoring of the Hard Hill experiment ([1++, EV-], for example, Lee and others, 2013b, NEER004, Appendix 3, pp.87–90), but not in Milligan and others (2018 [1++, EV-]).

4.102 Other studies reported bare ground amalgamated with 'duff' (Grau-Andrés and others, 2018/2019b [1+, EV-]) and 'brash'/'burnt' (Heinemeyer and others, 2019c [1+, EV-], though bare ground was reported separately in Table 7, p.58, with mean cover 26% over 2012–15), while Worrall and others (2013a [1,2+, EV-]) reported *Calluna* litter and 'stick' (charred stems) separately, which taken together raise issues in relation to definitions and interpretation.

4.103 Grau-Andrés and others (2018/2019a,b [1+, EV-]) reported on bare soil and 'duff' collectively at a Scottish, *Calluna*-dominated raised bog site which showed an increase from 5% in recently unburned stands to 33% 17 months after burning, which was greater (71%) in plots subject to higher severity burns (through a drought treatment) (also see Section 8 on fire severity and other characteristics). Comparable data from a Scottish *Calluna*-dominated upland dry heath site showed a similar but smaller effect, increasing from 2% in recently unburned stands to 20% 15 months after burning (52% with severe burns). On a *Calluna*-dominated Peak District blanket bog site, where *Calluna* cover declined from 78% pre-burn to 0% immediately post-burn, post-burn *Calluna* 'stick' covered 66% and *Calluna* litter 8% (74% combined, similar to pre-burn live cover) (Worrall and others, 2013a ([1,2+, EV-]). Heinemeyer and others (2020 [1+, EV-]) reported on "bare/brash/burnt" collectively from three Pennine/Bowland sites over a longer, four-year, post-treatment period comparing burning with cutting and no recent management. As might be expected, this showed an increase post-treatment which was greater following burning (from c.2% pre-burn to c.66% after one year, declining to c.10% after four years) than cutting (c.2% to c.46%, declining to 8%) compared with little change (c.3%) in recently unmanaged plots. This difference between treatments had been lost by the second post-treatment phase from 2017–21 (Heinemeyer and others, 2023 [1+, EV-]).

Litter

4.104 Only three recent studies specifically reported on the effects of burning on litter cover. On a Scottish raised bog site, Grau-Andrés and others (2018/2019b [1+, EV-]) reported higher litter cover initially (17 months) post-burn (47%), which was lower under a higher severity (drought) treatment (35%), compared with adjacent not-recently-burnt plots (22%) (and similar, slightly higher, post-burn litter cover in a corresponding dry heath site). Conversely, Velle and others (2012 [1+, EV-]) and Velle & Vandvik (2014 [1+, EV-]) reported that reduced litter cover was still apparent after up to seven years in young and old burned Norwegian coastal wet heath plots, compared to adjacent not-recently-burnt stands. Alday and others (2015 [1,2+, EV-]) and Santana and others (2016 [2+, EV+]) reported on litter biomass accumulation following burning (reported earlier in paras. 4.85 and 4.88, respectively).

Habitat condition

4.105 Only three recent studies specifically report on burning in relation to the condition of blanket (and valley) bog and wet heath habitats using the upland Common Standards Monitoring guidance (JNCC, 2009). These studies showed all burned and not recently burned sites to be in unfavourable condition overall. Nevertheless, there were some differences between sites and treatments in relation to individual attribute targets and likely causes of targets not being met.

4.106 Hedley (2013 [2+, EV+]) surveyed the condition (and NVC community types, para. 4.31) of the ten EMBER catchment sites (five recently burned and five with "no recent history of burning") in the Pennines (as reported by Brown and others (2014 [2+, EV+]) in terms of hydrology, water chemistry, soil properties and aquatic invertebrates (also see sections on fauna, carbon and water of this report); and reported by Noble (2018b [2+, EV++]) in terms of burning, atmospheric pollution and grazing effects on peatland vegetation composition. Strictly, all sites were in unfavourable condition overall, though three not-recently-burnt sites and one burnt site were considered close to favourable condition (Hedley, 2013 [2+, EV+]). Three of the five recently burnt sites failed the attribute target overall for burning into the bryophyte and lichen layer and two overall for burning into sensitive areas. Not surprisingly, all but one (affected by a recent wildfire) of the not recently burned sites passed the target for burning into the bryophyte and lichen layer and all for burning into sensitive areas.

4.107 Critchley and others (2016 [2++, EV++]/ADAS and others 2017) also reported on resurveys including condition assessments from across a sample of 20 long-term moorland monitoring sites in England (also see para. 4.82) which included 14 with mires present (most blanket bog but some wet heath and valley mire and other fens). None of the mire features were in favourable condition. Of the eight recently burnt sites, five failed the attribute target for burning into the bryophyte and lichen layer and four for burning into sensitive areas. Unsurprisingly, all six not-recently-burnt sites passed these burning-related attribute targets.

4.108 In addition, Noble and others (2018b [2+, EV++]) also reported on burning and other effects on sites included in another national, condition assessment data set comprising a stratified random sample of blanket bog sites across the English uplands in 2008–09, previously reported in terms of condition by Critchley and others (2011a, [2++, EV++], NEER004, para. 11.14, p. 50 and Appendix 11, p. 164, in relation to burning-related attributes). All of the 85 blanket bog sites (with at least ten samples on ≥30 cm peat) were also in unfavourable condition, with 21% of sites and

11% of all samples failing the attribute target for recent burning into the bryophyte and lichen layer. Similarly, the attribute target for recent burning into sensitive areas was not met in 15% of sites. Though these failure rates are lower than for some other attributes included in the study (such as grazing on *Calluna* shoots which tends to be more widespread across sites), 11% of samples with recent burning into the bryophyte/lichen layer is high for a single survey point in time. It indicates a relatively short rotation and suggests that a higher proportion of the sites would be likely be affected by burning over time.

Indirect effects

4.109 In a controlled, factorial laboratory experiment, Noble and others (2017 [1+, EV-]) investigated indirect effects of burning related differences in peat bulk density (BD, bare peat sourced from Moor House NNR, North Pennines, with a higher BD treatment artificially created by compression), ash deposition and rainwater chemistry treatments on the growth of three moss species: *Sphagnum capillifolium*, *S. fallax* and *Campylopus introflexus* (samples collected from Moor House and the last from the Peak District). Higher peat bulk density limited growth of both *Sphagnum* species and *C. introflexus* responded positively to ash deposition. Less polluted rain[17] limited growth of *C. introflexus*. It was concluded that high peat bulk density typically caused by fire or drainage can limit *Sphagnum* establishment and growth, potentially affecting peatland function. Ash inputs may have direct benefits for some *Sphagnum* species but are also likely to increase competition from other bryophytes and vascular plants which may offset any positive effects. Rainwater pollution may similarly increase competition for *Sphagnum* and could enhance positive effects of ash addition on *C. introflexus* growth.

***Sphagnum* establishment**

4.110 Noble and others (2019b [2+, EV+]) used burn chronosequences, including ongoing short-term (19-month) resurveys, from three relatively widely distributed blanket bog sites with varying levels of modification (Cheviot, North Pennines and Peak District in order of increasing degree of modification) to investigate *Sphagnum* re-introduction success[18]. There was no significant difference in re-introduction success according to burn-age category, suggesting that establishment does not necessarily require burning. However, success decreased over the relatively short resurvey period in the most recent and intermediate burn-age categories at the most *Sphagnum*-poor (Peak District) site, suggesting that recently burnt sites might not provide suitable conditions for *Sphagnum* re-introduction.

[17] Two types of artificial rainwater were produced to represent precipitation chemistry at UK upland sites with relatively low and high atmospheric pollution levels by dissolving compounds (NaCl, $MgSO_4$, $CaSO_4.2H_2O$ and NH_4NO_3).

[18] Survival (‘re-introduction success’) of the added *Sphagnum* was assessed during three subsequent surveys on a semi-quantitative percentage scale.

Bryophyte spore germination

4.111 Yusup and others (2022 [1+, EV-]) caried out laboratory experiments which showed varying effects of fire-related cues on bryophyte germination and spore viability of samples taken from peatlands in the Changbai Mountains in north-east China. They exposed *Sphagnum* spores of four species (*S. angustifolium*, *S. fuscum*, *S. medium and S. squarrosum*) to heat (40, 60 and 100 °C) on its own and combined with smoke-water treatments. A temperature of 100 °C inhibited the spore germination or even killed spores of all species, while spore germination of three species (*S. angustifolium*, *S. fuscum* and *S. squarrosum*) of the four species was promoted by 40 and 60 °C heat compared to the control (20 °C). 'Hollow' species (*S. angustifolium* and *S. squarrosum*) showed a stronger positive responsive to heat than 'hummock' species (*S. fuscum* and *S. medium*). *Sphagnum fuscum* spores responded positively to the combined heat and smoke treatment while the other species did not.

4.112 In a related experiment, Yusup and others (2023 [1+, EV-]) similarly tested the germination of spores of 15 bryophyte species (including nine *Sphagnum* species.) from the same area in China after treatment with 'smoke-water'. Smoke increased the germination percentage for ten of the species: two forest margin species (downy plait-moss *Hypnum callichroum* and dwarf bladder-moss *Physcomitrium sphaericum*), two swamp species (Knieff's hook-moss *Drepanocladus aduncus* and *Sphagnum squarrosum*), and six "open expanse" species (*Physcomitrium strictum*, *Sphagnum fuscum*, *S. medium*, *S. imbricatum*, *S. subnitens* and *S. flexuosum*). For some of these species the effect depended on smoke-water concentration, with varying optima and germination speed for four of these. Smoke-water had a significantly negative impact on germination for one forest margin species, *H. callichroum*, and one open expanse hollow species, *Sphagnum angustifolium*, at high concentration. *S. capillifolium* showed a significantly negative response to all smoke-water treatments. None of the smoke-water treatments had any impact on spore germination in three species: juniper haircap *Polytrichum juniperinum*, common haircap *Polytrichum commune* and *Sphagnum fimbriatum*. Smoke enhanced the germination of 1-year but not 4-year laboratory-stored spores and considerably increased the germination of spores naturally buried in peat for up to c.200 years.

Palaeoecological studies

4.113 Eleven recent palaeoecological studies provide evidence on post-fire changes in the occurrence and abundance of some key plant species and groups (Table 11).

4.114 There is evidence from four studies that indicates that vegetation burning likely contributed to a subsequent change of vegetation community within the peat archive. Rowney and others (2023, [2+, EV-]) and Fyfe and others (2018, [2+, EV-]) both working on Exmoor, found that burning promoted and sustained grass-dominated vegetation. McCarroll and others (2017, [2+, EV-]) working in Yorkshire found a spike in charcoal that indicated forest clearance activity around 1,300 cal. year BP, which was coincident with increases in *Calluna.* This study also found peaks in charcoal in the peat archive from 1870 onwards and more dramatically in 1940 and 1975 that led to the conclusion that burning influenced the vegetation changes in the pollen profile with increased representation of grasses and plantains. Fyfe and others (2018, [2+, EV-]) also reported a case where burning promoted and sustained *Calluna*, at least between 5,500 and 1,500 cal. year BP. Blundell & Holden (2015, [2+, EV-]) also working in Yorkshire on Keighley Moor with multiple cores, found that the vegetation of the last 100 years was atypical of the site since peat formation. They found that *Sphagnum* was an important contributor to the vegetation cover between 1,500 years BP and the early 1900s. However, from the turn of the 20th century, *Sphagnum* levels declined severely, coincident initially with a wildfire event but remaining extremely diminished when the site regularly underwent managed burning, though atmospheric pollution may have at least contributed to this.

4.115 There is evidence that industrialisation and other anthropogenic activity, including though atmospheric pollution especially with sulphur, is associated with vegetation change on peatlands. Chambers and others (2013, [2+, EV-]) working in Wales, and McCarroll and others (2016a, [2+, EV-]) working in Yorkshire, both recorded increases in Spheroidal Carbonaceous Particles, a by-product of some industrial processes, associated with changes in vegetation. Chambers and others (2017, [2+, EV+]) found that the most marked changes within the vegetation communities occurred post industrialisation, with *Sphagnum* highest when charcoal was sparse or absent. McCarroll and others (2017, [2+, EV-]) also found that a decrease in *Sphagnum* spores was associated with an increase in charcoal. McCarroll and others (2016a [2+, EV-]) found that monocots including *Eriophorum vaginatum* increased from the Industrial Revolution onwards, which was attributed to managed burning, although it is likely that sheep grazing pressure was an important co-driver.

4.116 Swindles and others (2015, [2+, EV-]) working in Northern Ireland, and Swindles and others (2016, [2+, EV-]) working in Yorkshire, found moderate evidence that, whilst site specific factors such as fire (determined by changes in charcoal) supported findings from elsewhere that fire is a driver of vegetation change, other factors such as drainage, dust loading, atmospheric pollution and agricultural contamination, combined with the effects of burning, could also lead to vegetation changes as manifest through reductions of *Sphagnum* and increases in *Molinia*. The finding that a range of combined factors can lead to vegetation changes is further supported by Fyfe and Woodbridge (2012, [2+, EV-]) who investigated anthropogenic activity on Dartmoor.

4.117 Gillingham and others (2016c [4+, EV++]) reviewed studies relating to the historic peat record and reported evidence that burning, alongside other factors including grazing, climate and afforestation may have played a role in peatland erosion and losses of or declines in *Sphagnum*.

Table 11. Summary of findings relating to vegetation from recent palaeoecological studies.

Study	Habitat	Date	Drivers of change	Major outcome	Other outcomes	Comments
Blundell & Holden (2015 [2+, EV-]	Blanket bog	c.1900	Wildfire event followed by rotational burn management.	*Sphagnum* went from being an historic important component of vegetation cover (c.1500 years BP to the early 1900s) that fluctuated with recovery from management burning to being extremely diminished.	Prior to 1900, vegetation made up of a range of species including *Sphagnum* spp*.*, *Calluna, Eriophorum* spp. Post-1900, the vegetation became dominated by *Calluna*.	The current vegetation community is atypical of the last 1,500 years.
Chambers and others (2013 [2+, EV-])	Blanket bog	Industrialisation to 20th Century	Post industrialisation associated with increases in atmospheric deposition and localised use of fire as a management tool.	*Sphagnum* appear to decline prior to increases in *Molinia* at one site whilst *Molinia* outcompeted by *Eriophorum vaginatum* at another.	Pollen diagrams indicate decreases in occurrence of *Sphagnum*. But with localised dominance of *Molinia* and *Eriophorum vaginatum.*	Circumstantial evidence suggests that *Eriophorum vaginatum* benefitted from cessation of burning.
Chambers and others 2017 ([2+, EV-])	Blanket bog	Post industrialisation	Charcoal in basal samples indicating periodic fire. Local fire evident at all sites.	*Sphagnum* presence highest where charcoal was sparse or absent. At four sites, increase in ericale rootlets near the surface, suggesting low water tables.	***Loss:*** round-leaved sundew *Drosera rotundifolia* disappeared from three sites. **Increases** in grass and *Empetrum nigrum* pollen in South Pennines likely reflects *Molinia* dominance and drier conditions.	Three regions within north of England. Evidence that two sites had hummock/pool structure until recent centuries.
Fyfe and Woodbridge (2012 [2+, EV-])	Blanket bog	8,000 cal. year BP to present	Periods of increased anthropogenic activity.	Greater heterogeneity in the landscape. Spread of blanket peat	Shift from wooded areas to open mixed landscapes of grassland, bog and scrub.	Fire is recognised as being a driver in landscape change but the relationship

Study	Habitat	Date	Drivers of change	Major outcome	Other outcomes	Comments
				asynchronous, resulting in mosaics of expanding peat.		between fire and vegetation type is spatially variable, with grazing suggested as playing a greater role.
McCarroll and others (2016a [2+, EV-])	Blanket bog	7,000 cal. year BP to present	Anthropogenic activity.	*Sphagnum* present throughout profile but switch from *S. austinii* to *S. papillosum* c.1,000 cal. years BP. *Calluna* shows no clear relationship with human activity and declines progressively through the profile to the current day.	Monocots and *Eriophorum vaginatum* increased from the Industrial Revolution onwards, which is attributed to managed burning. *Sphagnum subnitens* also declines along with *S. austinii.*	The decline in *Calluna* and increase in *Eriophorum vaginatum* may suggest that grazing is a more important driver of vegetation composition at this site than burning.
McCarroll and others (2016b [2+, EV-])	Blanket bog	6700 cal. year BP to present.	Increases in the use of burning associated with anthropogenic activity.	An increase in charcoal from depth of 150 cm that is consistent with a change in species composition.	Increase in *Calluna*, Ericaceae and Poaceae. Disappearance/decrease of *Sphagnum*.	Present vegetation is atypical of the site over time and has only been present for c.200 years.
McCarroll and others (2017 [2+, EV-])	Blanket bog	6000 cal. year BP to present.	Phases of anthropogenic activity including burning and grazing.	Clearance of the landscape and development of peat formation.	Increases in *Calluna* associated with increases in charcoal at 1300 cal. years BP. Shifts in pollen profile from trees, shrubs and *Sphagnum* towards Poaceae*.*	Indicates anthropogenic impact via burning and grazing well before industrialisation.
Rowney and others (2023 [2+, EV-])	Blanket bog	19th Century to present.	Increases in anthropogenic activity.	Use of drainage and burning.	Declines in *Sphagnum* associated with drainage. Increases in Poaceae and Cyperaceae.	Charcoal evidence suggests that burning promoted the increases in grasses and sedges abundance. Charcoal

Study	Habitat	Date	Drivers of change	Major outcome	Other outcomes	Comments
						was not significantly associated with either *Sphagnum* or taxon richness.
Swindles and others (2015 [2+, EV-])	Raised mire	AD 860 to present	Occurrence of fire and soil-derived dust.	Decline in *Sphagnum austinii.*	**Increases** in grass, sedges and agricultural taxa.	Timing of *Sphagnum austinii* decline coincident with landscape scale anthropogenic activity e.g., clearance of scrub and woodland, introduction of arable farming nearby and increases in presence of charcoal.
Swindles and others (2016 [2+, EV-])	Raised mire	c.1950–1975	Increased dust loading from localised quarrying, nutrient loading and heavy metal deposition from agricultural fertilizers and airborne pollutants and localised within-site burning (last reviewed, for example, by Abraham and others, 2017).	Major increase in area of *Molinia caerulea*.	**Decline:** *Calluna vulgaris, Erica tetralix, Carex* spp., *Eriophorum* spp., *Trichophorum germanicum*. **Loss:** *Sphagnum austinii* (formerly *Sphagnum imbricatum)* **Gain:** *Sphagnum papillosum*	Historic episodes of peat cutting and burning prompted substantial changes to vegetation, but recovery occurred through ecosystem resilience.

5. Recent evidence on the effects of managed burning on the fauna of upland peatlands

5.1 The full text of this sub-question 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

5.2 The characteristic fauna of upland peatlands is described in NEER004 (paras. 5.2–5.10). The earlier evidence review also assessed relevant studies on the effects of burning on upland peatland fauna up to 2012 (see Appendix 4 pp. 104–112 and Appendix 5 pp. 117–120). In NEER004 a summary, synthesis and brief interpretation was given across fauna studies, including as evidence statements (paras. 5.11–5.28) and research recommendations (p. 30, paras. 12.35–12.36). The evidence statements were also given in summary form in the Conclusions (paras. 12.7–12.9) and Summary (pp. vi-vii).

5.3 There is some crossover between this sub-question and those concerning vegetation (Section 4) and water (Section 7). Vegetation structure and composition is linked to nesting/breeding sites and food availability, especially of birds, whilst aquatic invertebrate community composition is linked to, and an indicator of, water quality.

Recent studies on the effects of burning on fauna

5.4 Twenty-four recent studies (reported in 46 references) since NEER004 provide evidence on the effects of managed burning on the fauna of upland peatlands. Information on the characteristics of individual studies is given in an evidence table (across the eight sub-questions) in Table A1.1 in Appendix 1.

Study type and quality

5.5 A summary the type and quality of the 24 recent fauna studies is given in Table 12. The majority, 19 (79%) were classed as type 2, followed by two classed as type 1 and a combination of type 1 and 2 (both 8%). For quality, the majority (21, 88%) were classed as [+], two as [++] and one as [-].

Table 12. Categorisation of type and quality of recent fauna studies by number of studies. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	1	1	0	2
1,2: both type 1 and 2	0	2	0	2

2: quantitative observational or correlative	1	17	1	19
3: qualitative	0	0	0	0
4: review	0	1	0	1
Total	2	21	1	24

5.6 Only two studies involved field experiments, both relating to invertebrates. One investigated the effects of burning and cutting on water moisture/levels and cranefly (tipulid) larval emergence on three Pennine/Bowland blanket bog sites, including modelling the effects on breeding birds (Heinemeyer and others, 2019c [1+, EV-], as part of the Peatland-ES-UK project). The other investigated the effects of ash input (as a proxy for burning) in four Pennine upland peatland headwater streams on macroinvertebrate assemblages by depositing ash onto trays filled with natural stream substrata (Johnston & Robson, 2015 [1+, EV+]).

5.7 Two other studies involved field experiment aspects related to the previous two experiments (para. 5.6) along with related survey/monitoring. Burn (2021 [1,2+, EV+]) used samples from the three Peatland-ES-UK experiment sites, and Moor House NNR and three other upland peatland areas (Exmoor, the Peak District and the Scottish Forsinard Flows), to investigate soil microbial community taxonomy and fungal community function in relation to management, climate/geographic location, degree of habitat modification and time since restoration treatments, alongside a novel carbon partitioning mesocosm experiment. Brown and others (2013/2019 [1,2+, EV+], as part of the EMBER project) investigated the effects of burning and sedimentation on aquatic invertebrates in ten Pennine peatland headwaters sampled six times, each over a period of 20 months, along with a riverside mesocosm experiment comprising 24 channels alongside one watercourse.

5.8 Apart from two linked evidence reviews for Natural England on the ecology, burning and other management options for the control of heather beetle (Gillingham and others, 2016a/b [4+, EV+]), all of the other evaluated fauna studies involved survey and/or monitoring, with most also classed as case studies, often incorporating correlative analyses in relation to management interventions, vegetation composition and structure, and other factors. All but one (Heinemeyer and others, 2019c [1+, EV-]) were primarily fauna studies.

5.9 Sixteen studies involved birds (67% of studies), five terrestrial invertebrates (21%), three aquatic invertebrates (13%) and single studies on a mammal (mountain hare), reptile (adder) and soil microbes (each 4%), with some limited overlap (studies including more than one group).

Outcome measures

5.10 The 23 fauna primary studies mostly assessed abundance of individual species or species groups, though these varied between faunal groups. For the largest group, birds (16 studies), most studies assessed breeding numbers, pairs and/or density (14, 88% of studies), followed by breeding success (8, 50%), species richness/ diversity and presence (both 2), and survival and change over time (both 1), with all but two studies including multiple outcome measures. The bird studies reported on either selected moorland bird species or groups (though not necessarily all species in groups and a few studies included a wider range of species, with one including all species): waders (10, 63% of bird studies), grouse (9, 56%, all but one red grouse), passerines (8, 50%) and raptors (5, 31%). Four studies involved single bird species on single sites or wider areas/regions: Douglas and others (2014 [2+, EV+]) on curlew *Numenius arquata*; Douglas & Pearce-Higgins (2014 [2+, EV-]) on golden plover *Pluvialis apricaria*; Roos and others (2016 [2+, EV+]) on black grouse *Tetrao tetrix*; and Ludwig and others (2018 [2+, EV-]) and Robertson and others (2017 [2+, EV++]) on red grouse. In addition, Wilson and others (2021 [2++, EV++]), analysed national data sets on egg-laying dates for moorland and 'moorland edge' bird species from the British Trust for Ornithology (BTO) Nest Record Scheme (NRS) or nestling ('pullus') ringing records collected over a 43-year period between 1976 and 2019 in relation to burning dates. This in part is an update to a previous analysis by Moss and others (2005 [2++, EV++], NEER004, para. 5.26, pp. 29–30, Appendix 4, p. 109), though the approach and analyses, as well as the longer time-period, differ, hence the current report was treated as a separate study in this review. Another study assessed the timing of breeding activity of three passerine species at a single Dartmoor moorland site in relation to burning and other management (Zonneveld and others (2024 [2+, EV-]).

5.11 Recent bird studies tended to also record a wide range of potential explanatory variables, particularly in relation to management (burning and/or cutting), habitat/vegetation type and vegetation structure (Figure 5), though there was overlap between some of them. All studies assessed the effect or potential effect of burning, though in four cases this was only indirectly.

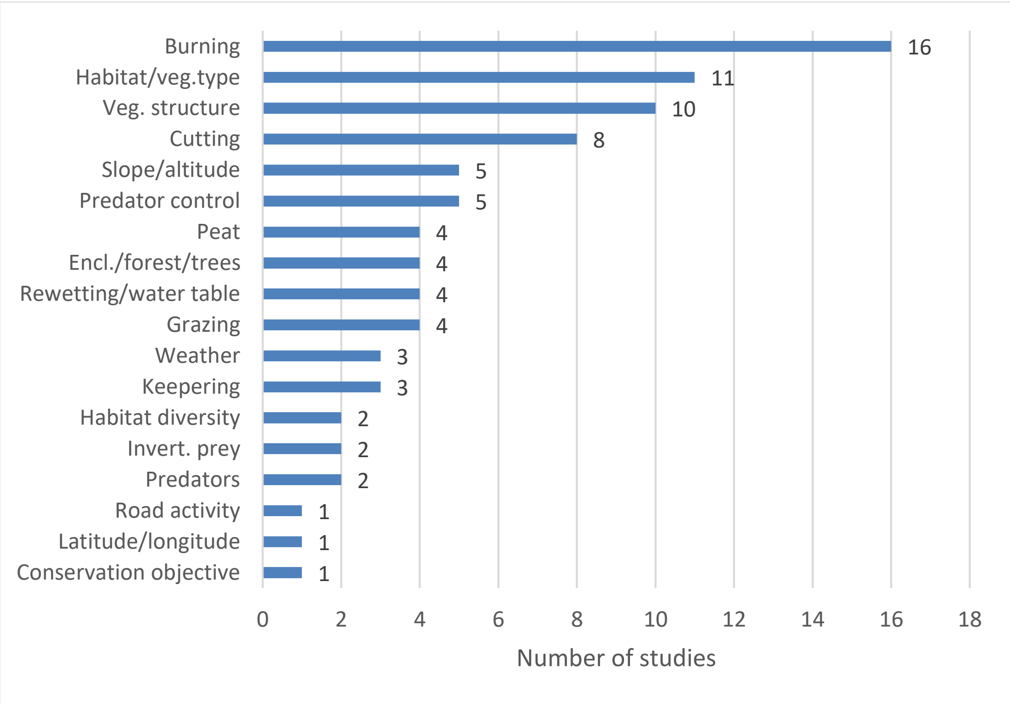

Figure 5. Frequency of explanatory variables recorded in recent upland bird studies *(n = 16 studies).*

5.12 For other, non-bird faunal groups (9 primary studies), all studies recorded outcome measures related to abundance: counts of individual species or individuals in species groups (8 studies); total abundance/density or assemblage(s) (5); species or taxonomic group richness/diversity (4); species traits or functional groups (6); rare species (2); and presence of species in 10-km or 1-km grid squares, i.e., mapped distribution (1).

Applicability of recent evidence on the effects of managed burning on fauna

5.13 This section reviews the applicability of the evidence from the mostly primary, evaluated studies on the effects of burning on fauna on UK, and in particular, English upland peatlands. It draws on some of the characteristics of studies discussed in paras. 5.5 – 5.12 and the assessment of external validity carried out in assessing the individual studies and across the studies as a whole, including such factors as geographical location and how representative the habitat/vegetation type(s) and intervention(s) were of the characteristics and state of upland peatlands in the study area(s) and especially nationally.

Countries, areas and number of sites

5.14 All but two of the recent fauna studies were from the UK (22, 92%), with the others from Norway (on carabid beetles in coastal dry heaths, Bargmann and others, 2015/2016 [2+, EV-]) and Denmark (five different insect taxa on dry, wet and differently managed heaths, Byriel and others, 2023 [2+, EV-]). Within the UK, the majority of primary studies (21) were from or included sites in England (14, 67% of UK studies, including five studies which also included Scottish sites), followed by Scotland (12, 57%) and two studies that included sites in Wales (10%), one a GB mainly BTO moorland breeding bird survey data set (Wilson and others, 2021 [2++, EV++]), and the other including breeding bird survey sites in four regions across England, Scotland and Wales (Buchanan and others, 2017 [2+, EV++], involving 159 plots). Two linked evidence reviews on heather beetle ecology and burning management by Gillingham and others (2016a/b [4+, EV+]) also included studies from across GB and elsewhere, especially Europe.

5.15 While the recent fauna studies are geographically and ecologically applicable to upland peatlands in the UK and especially England, they were concentrated in northern England, especially the Pennines, and Scotland (Figure 6). Although the range of variation in the upland peatland resource as a whole is not fully represented, two studies included Welsh sites and two were in SW England (Figure 6Figure 2), and the national data set on moorland breeding bird egg laying dates was from across the GB uplands (Wilson and others, 2021 [2++, EV++]).

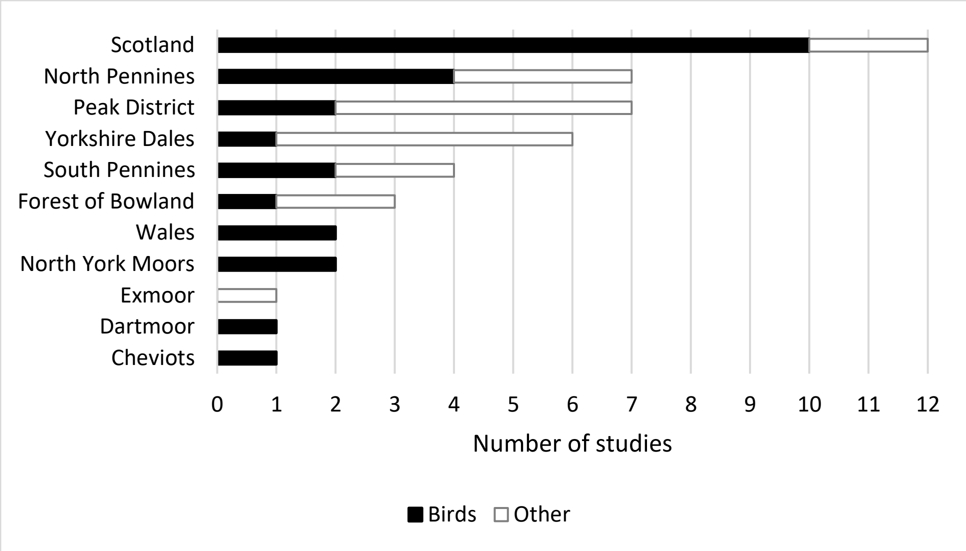

Figure 6. Number of recent peatland fauna primary studies including English upland areas, Scotland and Wales (n = 21 primary studies, several covering multiple countries and areas).

5.16 Fauna surveys, especially of birds, tended to cover larger areas (than, for example, vegetation surveys), often recording within 1-km squares or other compartments and often covering multiple areas and/or sites. In addition to the two large GB surveys (para. 5.14), bird studies within England included one covering 36 grouse moors across northern England (Robertson and others, 2017 [2+, EV++], sampled within 1-km^2 per site); two covering single, large upland areas (the North York Moors (Drewitt, 2015 [2+, EV+], sampled for merlin *Falco columbarius* in 59 1-km squares); and the Peak District (Dallimer and others, 2012 [2+, EV+], sampled in 37 paired sites each comprising an area of moorland and an area of farmland within *c*.2 km); and one a single, large site in the North Pennines (Douglas and others, 2017 [2+, EV-], Geltsdale). In addition, Heinemeyer and others (2019c [1+, EV-]) modelled the effects on cranefly larval emergence on golden plover in response to management and water moisture on three Pennine/Bowland blanket bog sites.

5.17 Within Scotland, in addition to the two large GB-wide bird surveys (para. 5.14), two surveys included moorland sites in both Scotland and England: a curlew study covering two large areas in south Scotland and the South Pennines (Douglas and others, 2014 [2+, EV++], sampling 41 and 36 1-km squares, respectively); and a survey of three southern Scottish and 15 northern English sites (Littlewood and others, 2019 [2++, EV+], sampling a total of 104 1-km squares). A further five studies included Scottish bird surveys: Newey and others (2016 [2+, EV+]) surveyed 26 *Calluna*-dominated moorland sites in the Highlands sampled by up to four 1-km squares; Douglas & Pearce-Higgins (2014 [2+, EV-]) monitored golden plover density and breeding success in relation to cranefly prey and vegetation structure in northern Scotland; Roos and others (2016 [2+, EV+]) studied associations between young black grouse broods and habitat characteristics in four areas of moorland-forest mosaic in the Scottish Highlands; and single sites were surveyed by Calladine and others (2014 [2+, EV-]) and Ludwig and others (2018 [2+, EV-], for red grouse) in relation to vegetation composition and structure, both in SW Scotland.

5.18 Studies of other faunal groups were less numerous (para. 5.8) and generally covered smaller geographic areas, although aquatic invertebrates were included in three linked studies covering a range of upland watercourses from up to around 30 Pennine sites between the Peak District and North Pennines (Johnston, 2012 [2-, EV+]; Johnston & Robson, 2015 [1+, EV+]; Brown and others (2013 [1,2+, EV+]). In addition to a Norwegian study (para. 5.14), terrestrial invertebrates were surveyed in two other primary studies, both on craneflies as breeding wader prey, from three Pennine/Bowland sites (Heinemeyer and others, 2019c [1+, EV-]) and one northern Scotland area (Douglas & Pearce-Higgins, 2014 [2+, EV-], in 33 1-km squares), and two linked evidence reviews on heather beetle ecology and management included studies from across GB and elsewhere (Gillingham and others, 2016a/b [4+, EV+]). Bedson and others (2022b [2+, EV+]) studied mountain hare *Lepus timidus* at three main sites in the Peak District, each in 13 1-km squares, across 26 random 1-km squares across their range. Finally, Newey and others (2020 [2+, EV++]) reported on the distribution of a range of faunal (and fewer floral) species across a near census of the distribution of burned grouse moors in Scotland in relation to area of managed burning. The species were selected on the basis that the association between species distribution and grouse moor management was considered less well understood or unknown. The fauna species comprised six birds: golden plover, curlew, merlin, kestrel *Falco tinnunculus*, whinchat *Saxicola rubetra* and lesser redpoll *Acanthis cabaret*; and single reptile (adder *Vipera berus*) and invertebrate (green hairstreak *Callophrys rubi*) species. Soil microbes (Burn, 2021 [1/2+, EV+]) are also included in this fauna section, though they are also relevant to carbon and soil.

Habitat and vegetation type

5.19 Some fauna studies, especially of birds, did not differentiate peatland from wider moorland habitats, in part reflecting the fact that surveys they tend to cover large areas and hence cross habitat boundaries, transitions and mosaics. To some extent this limits their relevance and hence value for this review update, though many recent studies do make this distinction, including in some cases by recording peat or deep peat as an explanatory variable (in four out of 16 bird studies, Figure 5) than in NEER004. In terms of the main broad habitats reported, most studies (15) were described as moorland, followed by blanket bog (9), heath (3), moorland fringe (2) and moorland/forest mosaic (1), with limited overlap (four studies reporting two main habitat types). Given other evidence included in study publications, it seems likely that most classed as moorland included at least some areas of blanket bog and/or wet heath.

Types of burn and fire

5.20 As far as can be determined from published information, all of the burns included in recent fauna studies were managed burns undertaken by land managers.

External validity

5.21 Reflecting the higher number of sites included in most fauna studies (paras. 5.16–5.17) and relatively wider geographic spread of them in relation to the UK and English upland peatland resource as a whole (than in vegetation studies, Section 4, paras. 4.19–4.22), albeit still with most in the northern England and Scotland (paras. 5.145.14–5.15, Figure 6), most recent studies were classed as [EV+] for external validity (10, 42% of studies), with eight (33%) as [EV-] and six (25%) as [EV++] (Table 13). All those classed as [EV++] were type 2 studies involving relatively large numbers of areas, sites and/or samples.

Table 13. Categorisation of external validity (EV) of recent fauna studies by the number and type of studies. EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	1	1	2
1/2: both type 1 and 2	0	2	0	2
2: quantitative observational or correlative	6	6	7	19
3: qualitative	0	0	0	0
4: review	0	1	0	1
Total	6	10	8	24

Summary of recent evidence on the effects of burning on fauna

5.22 This section presents a summary, synthesis and brief interpretation of the findings of 24 recent studies on the effects of burning on fauna. This includes the range of outcome measures reported in relation to burning and other management interventions, and other explanatory variables reported (paras.5.11–5.12). It is given as narrative descriptions by individual fauna groups and in some cases as summary tables indicating direction and magnitude of responses in relation to biodiversity (fauna) objectives.

Breeding birds

5.23 The importance of the UK uplands, including peatlands, for specific bird populations, breeding bird assemblages, UK BAP (Biodiversity Action Plan) and other species of conservation concern, including those listed on Annex I of the Wild Birds Directive and as features as designated sites, is described in NEER004 (paras. 5.3–5.4, 5.6, pp. 25–26).

5.24 Fifteen recent studies provide relevant information on the direct and indirect effects of burning, other habitat management, and a range of other explanatory variables, particularly physical characteristics related to climate and topography, on upland peatland breeding birds. Many of the studies only provide information on potential indirect effects through burning or other management-induced 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 negative effects from burning through loss of bird's nests, eggs and young and potentially adult birds, reptiles, and invertebrates in various developmental stages. Some studies assessed a wide range of explanatory variables which enable the relative importance of burning to be assessed compared to other factors.

Burning effects

5.25 Ten recent studies provide information relevant to the direct effects of burning on breeding birds. Most of these report on the effects on breeding numbers and/or density, though some also provide information on a range of other related outcome measures: breeding success, survival, species richness/diversity and community composition, presence and change over time. Relatively few recent studies showed significant direct effects of burning on breeding birds. Burning effects can be hard to separate from other related management activities, especially predator control, though more recent studies do separate out the effects of burning and predator control than in NEER004.

5.26 A study of the effect of burning on red grouse across 36 grouse moors in northern England (Robertson and others (2017 [2+, EV++]) showed no effect on pre-breeding density, but positive effects on breeding success and post-breeding density. Densities and breeding success were similar on heath and blanket bog. In a survey of three southern Scottish and 15 northern English sites by Littlewood and others (2019 [2+, EV+]), the only evidence for an effect of area burnt was a very weak positive relationship for golden plover numbers, although burning was not included in the best model for this species (a stronger association was shown for predator control including for a wider range of species, para. 5.38) and a weak negative effect of burning on wren abundance. Monitoring over a 14-year period on a single site in the North Pennines (Douglas and others, 2017 [2+, EV-]), where a greater percentage area of a plot was burned between surveys, showed relatively weak increases in golden plover and declines in red grouse (and no change for curlew).

5.27 Two studies compared breeding bird abundance and distribution data with mapped burning extent and different management objectives. In the first, Wilson and others (2021 [2++, EV++]) estimated the proportion of the breeding population of each of 40 species (14 moorland species, 11 moor edge species and 15 additional upland species) found in areas where rotational burning is practised in GB (taken from the moorland burning data set described by Douglas and others, 2015 [2++, EV++]). This was primarily to assess the impacts of burning on birds nesting in these areas in the context of impacts on their wider populations. Relative abundance of birds in each tetrad was estimated from Bird Atlas Timed Tetrad Visit breeding season count data (Balmer and others, 2013). Many moorland species were more abundant in tetrads with burned heather moorland ('burn') than in tetrads with unburned heather moorland ('moor') or in the wider countryside ('other'). Species with notably higher densities included golden plover, meadow pipit *Anthus pratensis*, red grouse and ring ouzel *Turdus torquatus*. The only two moorland species with lower densities in 'burn' tetrads were dunlin *Calidris alpina* and greenshank *Tringa nebularia*, which are both species that tend to reach their highest densities in wetter peatland habitats. The pattern was more mixed for grassland/moorland edge species, with waders in particular showing strongly contrasting patterns of association. Densities of curlew, lapwing *Vanellus vanellus* and snipe *Gallinago gallinago* were all much higher in 'burn' tetrads than on 'moor' or 'other' tetrads. Oystercatcher *Haematopus ostralegu*s, redshank *Tringa totanus* and ringed plover *Charadrius hiaticula* were, however, less abundant on 'burn' and 'moor' than on 'other' tetrads, as was twite *Carduelis flavirostris*. Most of the widespread species were less abundant in 'burn' tetrads than in either of the other two categories, but cuckoo *Cuculus canorus* and dipper *Cinclus cinclus* were both exceptions to this rule. Some of these associations may reflect other differences between burned and other tetrads than just burning and associated management. In England, tetrads with burned moorland accounted for 61% to 78% of the populations of black grouse, golden plover, red grouse and ring ouzel, though the percentages were less in Wales and Scotland. This indicates that any negative effects of burning on these species could have larger consequences for their populations, particularly in England.

5.28 In the second study, Newey and others (2020 [2+, EV++]) compared species distribution data with an index of mapped area burnt (the percentage area of grouse moor burnt per hectad in five burn classes) on Scottish grouse moors for selected flora and fauna species, including six bird species (in 10-km squares or hectads rather than 1-km squares as used for non-bird species). Curlew occurred in 120 assessed grouse moor hectads (35% of all occupied hectads in Scotland), with the majority of curlew records from squares with little or moderate burning (<40%), though the percentage of occupied hectads gradually increased with increasing percentage of area burnt. However, as the greatest burn area category (81–100%) only occurred in one square at the hectad scale (used for all bird species), an artefact of this was that the percentage of hectads occupied by a species in that burn area category was either 100% (2 species) if present or 0% (four species) if absent in that one hectad. Thus, the results for the 81–100% burn area category are excluded from the data for bird species summarised here. This could affect other burn area classes which occur in a low percentage of hectads (especially 61–80%).

5.29 Golden plover occurred in 68 grouse moor hectads (38% of all occupied hectads in Scotland), with most records from hectads with low to moderate burning with the peak hectads having 21–40% burn area. The percentage of occupied hectads gradually increased with increasing burn area up to 41-60%. Merlin occurred in 102 grouse moor hectads (41% of all occupied hectads in Scotland, the highest percentage association with grouse moors of the six bird species), with most records from hectads with low to moderate burning with the peak hectads having 21–40% burn area. The percentage of occupied hectads gradually increased with increasing burn area up to 41–60%. Kestrel occurred in 102 grouse moor hectads (29% of all occupied hectads in Scotland), with most records from hectads with low to moderate burning, peaking in hectads with 21–40% burn area. The percentage of occupied hectads gradually increased with increasing burn area up to 41–60%. Lesser redpoll occurred in 85 grouse moor hectads (23% of all occupied hectads in Scotland), with most records from hectads with low to moderate burning, peaking in hectads with 21–

40% burn area. The percentage of occupied hectads gradually increased with increasing burn area up to 61–80%, which seems surprising for a species associated with trees. Whinchat occurred in 80 grouse moor hectads (26% of all occupied hectads in Scotland), with most records from hectads with low to moderate burning, peaking in hectads with 21–40% burn area.

5.30 The percentage of occupied hectads gradually increased with increasing burn area. The authors note that it is difficult to draw any firm conclusions from these associations and that for all species, especially birds which were assessed in fewer, larger squares (hectads), care is needed in interpreting species occurrence in the high percentage area burn classes as the sample size of both the number of assessed squares and the number of species records are low. Species may be responding to aspects of moorland management other than burning and occurrence was likely influenced by the wider landscape.

5.31 Newey and others (2016 [2+, EV+]) reported on aspects of breeding bird community composition from 26 *Calluna*-dominated moorland sites in the Scottish Highlands. There was no significant effect of any management practice, including percentage of burnt ground, or dominant management objective (grouse shooting, deer staking, sheep grazing and conservation) on bird species richness or diversity. Ordinations revealed that the percentage of burnt ground (and management for grouse shooting along with latitude) had significant effects on absolute and relative abundance of species and community composition. Common sandpiper *Actitis hypoleucos,* dunlin, golden plover and red grouse were strongly influenced by burning (and latitude) and were more likely to be prevalent on estates in the east where there is a higher percentage of moorland burnt. While grouse shooting as a dominant management objective appears to have a strong influence on the occurrence and absolute abundance of only a few species, these estates were associated with a distinctive avian assemblage characterised by waders such as common sandpiper, curlew and golden plover, as well as black-headed gull *Chroicocephalus ridibundus*, buzzard *Buteo buteo*, meadow pipit, red grouse and short-eared owl *Asio flammeus.* However, these estates were also negatively associated with corvids, merlin and some passerine species. Conservation as the dominant management objective had significant effects on the relative abundance of species, though the effect on absolute abundance and occurrence was less clear. These estates tended to be positively associated with passerine species, corvids and merlin, and negatively associated with red and black grouse, some wading birds, buzzard, common gull *Larus canus*, ring ouzel and short-eared owl.

5.32 Following an apparent decline in the numbers of breeding merlin in the North York Moors between two national surveys in 1993/94 and 2008 (Rebecca & Bainbridge, 1998; Ewing and others, 2011), Drewitt (2015 [2+, EV+]) investigated the possible effects of changes in burning practices in a sample of 59 1-km squares, including those occupied by breeding merlin in one or both survey years and in random unoccupied squares. *Calluna* extent and recent burning were identified using aerial photographic interpretation by Yallop & Thacker (2015) for 1995 and 2009, specifically for this project. There were no significant differences in the total area of mature or burnt heather between years. However, there were significant differences in the size and number of burns, both across all squares and those squares occupied by nesting merlin in 1993/94 but not in 2008. In both cases the number of burns in 2008 was greater than in 1995, with 120% more burns in the breeding squares formerly occupied by merlin. Individual burn size reduced over time by 31% for all squares, and by 44% in squares that supported merlin in 1993/94. The squares occupied by breeding merlin in 2008 showed a similar trend, although the magnitude of change over the time period was smaller. These changes in burning practice may have had the effect of reducing the extent of formerly unbroken patches of mature heather which could have affected their suitability for merlin nest sites.

5.33 Modelling from a sample survey of 37 paired moorland and nearby inbye farmland sites in the Peak District by Dallimer and others (2012 [2+, EV+]) indicated that reducing the intensity of burning management on moorland can result in a two-fold increase in curlew density on nearby farmland (presumably foraging birds) without altering current farmland habitat. Similarly, reducing burning intensity on moorland has the potential to lead to a five-fold increase in nearby farmland snipe density.

5.34 Unlike Robertson and others (2017 [2+, EV++], para. 5.26), Ludwig and others (2018 [2+, EV-]) found no significant relationships between grouse productivity, post-breeding density, and heather management intensity by burning and cutting at a single Scottish site (Langholm). However, the average intensity of heather management was five-fold lower than that described by Robertson and others (2017). It was suggested that either the degree of heather management was too low to deliver the benefits to grouse or other factors, such as predation, may have been more important in determining grouse numbers.

5.35 An intensive study of curlew across 18 sites in northern England and southern Scotland (Douglas and others, 2014 [2+, EV++]) showed no effect of the area burned on population change or breeding success.

Predator control

5.36 Five recent studies provide information on the possible direct effects of legal predator control, or surrogates for it (mainly gamekeeper density), and predator abundance on breeding birds (along with a wide range of other potential explanatory variables, Figure 5. Frequency of explanatory variables recorded in recent upland bird studies *(n = 16 studies).*Figure 5). Illegal control of protected predators, especially raptors, is an issue that has an effect on the populations of some species (Murgatroyd and others, 2019; Newton, 2020; RSPB, 2023) but is beyond the scope of this review.

5.37 In a large-scale survey across 159 plots in four regions in England, Scotland and Wales (Buchanan and others, 2017 [2+, EV++]), predator control variables (density of gamekeepers and counts of carrion and hooded crow *Corvus corone* and *C. cornix*, respectively) explained 7% of absolute, or 20% of relative, variance among the waders and grouse. The average amount of absolute variance explained by predator control was greater than the absolute variance explained by any of the other variable groups. The relative contribution of predator control was lower among the passerines, with the only significant effects being positive associations with crow groups, which were therefore unlikely to be causal. Gamekeeper density was positively correlated with the abundance of curlew, golden plover and red grouse.

5.38 In a survey of three southern Scottish and 15 northern English sites by Littlewood and others (2019 [2+, EV+]), an index of predator control intensity was derived based on estimates of the number of full-time equivalent staff exclusively carrying out predator control per km^2. There were positive associations between predator control and the abundance of the three most widespread species of ground-nesting wader, with strong effects for curlew and golden plover, and less strongly for red grouse and snipe. These effects saturated at low levels of predator control for the three waders, with marked increases in numbers associated with increasing predator control effort up to a point, after which further intensifying of predator control had little effect. Coefficients calculated across all models in which predator control and burning occurred individually indicate much stronger effects of predator control than burning on the abundances of golden plover, curlew, red grouse and snipe.

5.39 In a curlew study covering two large areas in south Scotland and the South Pennines (Douglas and others, 2014 [2+, EV++]), population changes over an 8- to 10-year period were positively related to gamekeeper density (as a surrogate of predator control intensity) and inversely to the area of woodland surrounding sites, as a likely source of predators to adjacent open ground. Model predictions suggested that to achieve curlew population stability, increasing woodland cover from 0% to 10% of the land area within 1 km of populated sites requires an increase in human predator control effort of about 48%, a level associated with high-intensity grouse moor management. For a subset of sites (31) with no gamekeeping in the surveys, the area of woodland surrounding study sites showed a highly significant positive correlation with the fox abundance index but no clear correlation with crow abundance.

5.40 Predator control showed no significant effect on bird community composition across 26 *Calluna*-dominated moorland sites in the Scottish Highlands (Newey and others, 2016 [2+, EV+]).

5.41 Calladine and others (2014 [2+, EV-]) monitored carrion crow abundance annually over ten years at a single moorland restoration site in SW Scotland, compared to concurrent UK moorland-specific background trends and changes in the number of crows killed annually as part of predator control. The numbers of crows killed each year varied, with no significant trend. Crows increased markedly relative to background trends despite being actively removed by predator control measures. The authors suggested that ineffective control measures may have resulted in a recorded decline in red grouse breeding success and contributed to some species declines.

Vegetation structure and composition

5.42 Ten recent studies provide information on the possible indirect effects of burning and other management on breeding birds through changes in vegetation structure and composition. The wider effects of burning on vegetation composition and structure are described in Section 4.

5.43 In a large-scale survey across 159 plots in four regions in England, Scotland and Wales by Buchanan and others (2017 [2+, EV++]), vegetation composition and structure variables contributed approximately equally to models of abundance for waders, grouse and passerines. Combined, the amount of absolute variance explained was similar at about 7%, although the relative variance was about 50% larger for passerines. Curlew, red grouse and snipe benefited from increased compositional heterogeneity. Curlew and lapwing abundance increased with structural complexity and snipe increased with increasing vegetation height. The opposite was true for golden plover, which was more abundant in areas of shorter vegetation. Stonechat *Saxicola rubicola* and whinchat benefited from vegetation that was more complex in terms of species composition and height, respectively, whereas skylark *Alauda arvensis* appeared to benefit from vegetation that was less complex in terms of species composition. Wheatear *Oenanthe oenanthe* was weakly associated with vegetation that was more variable in height.

5.44 Douglas & Pearce-Higgins (2014 [2+, EV-]) collected data on large-scale spatial variation in vegetation structure (and prey abundance, para. 5.59) to understand drivers of golden plover breeding abundance and success at a single site in northern Scotland. Breeding densities were highest where vegetation was shortest, probably reflecting greater prey accessibility. In contrast, breeding success was not strongly related to vegetation height but positively correlated with both cranefly abundance and daily minimum temperatures. When combined to model the number of likely successful pairs in any one year, the magnitude of the vegetation height effect far exceeded that of cranefly abundance.

5.45 Dallimer and others (2012 [2+, EV+], para. 5.16) quantified how the habitat associations of bird species and assemblages occurring within two distinct, but adjacent habitat types ('moorland' and nearby inbye farmland) in the Peak District, determine a suite of bird density and richness indicators. There was a clear association between onsite avian density and richness and offsite habitat structure (for example, vegetation height, percent cover of dominant plant species, land management practices) in combined models. Although such effects were not universal across all species and assemblages, where present (for five farmland and three moorland indicators), the increase in model explanatory power offered by including offsite habitat structure can be large.

5.46 Roos and others (2016 [2+, EV+]) located brood-rearing black grouse females in four areas of moorland-forest mosaic in the Scottish Highlands. Each brood was paired with a random reference location 100 m away. Cover of different habitats and fine-scale vegetation and structure within squares of 1 and 0.25 ha, respectively, and associations between brood occurrence and habitat and vegetation variables were examined at these two scales. Black grouse broods were positively associated with wet flushes (1 ha), and with *Calluna*, sedges, grasses, *Sphagnum* bog-mosses and taller vegetation with intermediate levels of height variability (0.25 ha).

5.47 Monitoring over a 14-year period at a single, large site in the North Pennines showed that five bird species showed significant changes in abundance in relation to changes in vegetation structure and composition (Douglas and others, 2017 [2+, EV-]). Golden plover increased where *Sphagnum* cover increased. Skylark declined where graminoid height increased. Whinchat increased where vegetation density increased, and wheatear declined where graminoid height increased and where vegetation height became more variable. Meadow pipits increased where graminoid cover increased.

5.48 Four recent studies reported little or no effect of vegetation structure (mostly height) on breeding birds (Douglas and others, 2014 [2+, EV+], curlew; Calladine and others, 2014 [2+, EV-], waders and some passerines; Robertson and others, 2017 [2+, EV++] and Ludwig and others, 2018 [2+, EV-], both red grouse, both of which showed small increases with increasing heather cover).

Timing of breeding of upland birds

5.49 Wilson and others (2021 [2++, EV++]) carried out a re-analysis of data on the timing of breeding of upland birds in England, Scotland and Wales, to assess whether rotational burning poses a threat to populations of these species and whether any such threat varies in space and time (updating a similar review by Moss and others (2005 [2++, EV++], NEER004, para. 5.26 and Appendix 4, pp.109–110). In England and Scotland, the burning period runs from 1 October to 15 April in the uplands in the former, but in Scotland permission can be granted to extend the burning season to 30 April. In Wales, since 2008 the period runs from 1 October to 31 March. First-egg dates were estimated from the BTO Nest Record Scheme (NRS) records for 17 species of moorland and moorland edge habitats. Data from early years of the NRS were relatively sparse, so the data set used for global analyses of laying dates for all species was restricted to records collected during the 44-year period from 1976 to 2019. General Linear Models related laying date to species, year, latitude, longitude, elevation, habitat, rainfall, temperature, proximity of roads and occurrence of moorland burning. Laying date models for individual species were also constructed, based on first-egg dates derived from NRS and Ringing Scheme records, and including year, latitude and longitude as explanatory variables. Independent datasets for species poorly represented in the NRS dataset were obtained from existing studies of these species in upland areas. These were used to validate the results of models based on NRS and Ringing Scheme records. Timing of breeding information from studies of red grouse in Strathspey (1992–2016) and Langholm (2008–2016) in Scotland was analysed in relation to overlap with burning seasons, relationship with elevation, and (for Strathspey only) trends over time.

5.50 Mean laying date varied considerably between species. Standardised values predicted for different species by the all-species model for the year 2019 spanned a two-month period between late March and late May. Over the four-decade period from which the modelled data were drawn, mean laying date across all species in the model advanced by about one day every eight years. Other variables significantly related to laying date in the model were easting, northing, cover of conifer woodland and semi-natural grassland, rainfall and temperature. Across the available time series, 3% of red grouse clutches in Strathspey and 4% of red grouse clutches at Langholm were initiated before 15 April. The proportions of clutches started before the end of April were considerably higher, being 72% in Strathspey and 93% at Langholm.

5.51 Many moorland species were more abundant in tetrads with burned moorland than in tetrads with unburned moorland or in the wider countryside. In England, tetrads with burned moorland accounted for 61% to 78% of the populations of black grouse, golden plover, red grouse and ring ouzel. Negative effects of burning on these species could have larger consequences for their populations, particularly in England.

5.52 Overlap for most species between burning season and laying dates remains relatively small. Even among early breeding birds in moorland areas, the risk presented by burning is low for many species. Early breeders include species such as golden plover and lapwing that typically breed in short vegetation that is unlikely to be targeted for burning, and species such as golden eagle *Aquila chrysaetos* and peregrine *Falco peregrinus* that breed on crags, most of which are unlikely to be included in rotational burns. The overall risk to populations posed by burning depends on several factors. Some of these, such as timing of breeding, nesting ecology and the proportion of each population that nests in areas of rotationally burned moorland, are species-specific. Others, like the area of moorland burned each year and the proportion of burning done in the spring, are dependent on management practices and will affect risk in a similar way for many species.

5.53 Taking all of these factors into account, the species for which burning poses the greatest population level risk are not necessarily the earliest breeding species. Stonechat breeding attempts are probably among the most frequently destroyed by burning. However, this species is a habitat generalist which also breeds in lowland areas (though some lowland habitats are burnt), so no more than 0.3–0.5% of all stonechat nests are likely to be destroyed by burning. Conversely, populations of species that often nest in deep heather, such as ring ouzel and merlin, are concentrated in areas where moorland burning takes place, especially in England. For these species, the proportion of breeding attempts directly impacted by heather burning each year is likely to be less than 1% if burning is restricted to the legal burning season. If burning on all moorlands continued until 30 April, this proportion could rise to 4–5% for merlin and 6–7% for ring ouzel. Although national population level risk may be low, it is possible that burning and other interventions might have greater effects more locally and, perhaps reflecting this, various other legislation, regulations and guidance specify different spring dates for cessation of other regulated activities or access (Glaves and others, 2005 [4+, EV+], NEER004, p. 72).

5.54 Spatial variation in breeding phenology across most of England, Scotland and Wales, according to both the empirical information and model outputs is modest. Recent changes made to the upland burning season in Wales, which now ends two weeks before the standard upland season in England and Scotland, mean that the risk posed by moorland burning in Wales is now likely to be substantially lower than elsewhere in Britain.

5.55 Zonneveld and others (2024 [2+, EV-]) studied the timing of breeding of three moorland passerines, meadow pipit, stonechat and whinchat at a Dartmoor site over six years. The sites comprised a mix of wet and dry heath with areas of blanket bog and grassland in SW England where breeding is likely to be earlier than further north. Nests were found by intensive searching. Each nest was visited between one and eight times whilst active, when breeding stage and nest contents were recorded following the BTO Nest Record Scheme protocol. The findings were compared with dates of cessation of moorland burning specified in regulations (15 April, week 15 of the year), best practice recommendations/guidance in the SW (Defra, 2013, 31 March, week 13) and a potential earlier date (15 March, week 11). The degree of overlap of breeding with burning was quantified for different breeding stages (nest building, egg laying, egg incubation and nestling).

5.56 For all years combined, the median earliest week of onset of breeding was week 12 for stonechat, week 14 for meadow pipit and week 18 for whinchat. Stonechat showed the greatest overlap in all three burning cessation scenarios, whereas whinchat was not affected by burning management under any scenario. Under the week 15 burning cessation which matches current regulations, stonechat breeding activity overlapped with the permitted burning season by six weeks. In week 15, 6% of nests were at the nest building stage, 13% at laying stage, 41% at incubation stage and 1% at the nestling stage, with 61% of all nests being active in this burning cessation scenario. Meadow pipit breeding activity overlapped with week 15 burning cessation by three weeks, overlapping with 29% of nests, 14% nest building, 11% laying and 4% incubating. Both stonechat and meadow pipit still showed overlap under the earlier week 13 burning cessation scenario: four weeks or 41% of nests for stonechat (28% building, 10% laying, and 3% incubating), and one week or 4% of nests (all at the building stage) for meadow pipit.

Invertebrates

5.57 The invertebrate fauna of upland peatlands is described in NEER004 (paras. 5.3–5.5, p.25), including UK BAP (Biodiversity Action Plan) and other species of conservation concern and invertebrate species assemblages.

5.58 Five recent primary studies and one review provide information on the effects of burning and other management interventions (affecting vegetation structure and composition) on terrestrial invertebrates, with a further three studies on burning effects on aquatic invertebrates. The latter were reported under water quality and flow in NEER004 (Section 7, para. 7.14, p.36 and Appendix 7, pp.139–140), but here are reported under fauna and cross referenced as an aspect of water quality in Section 7.

Terrestrial invertebrates

5.59 Douglas & Pearce-Higgins (2014 [2+, EV-], at a single, less-modified blanket bog site in northern Scotland) and Heinemeyer and others (2019c/2023 [1+, EV-], at three modified, *Calluna*-dominated blanket bog sites in the Pennines/Bowland) investigated potential management effects, including burning, through changes to vegetation structure and moisture content on cranefly (tipulid) abundance in relation to their importance as prey for breeding golden plover. Douglas & Pearce-Higgins (2014 [2+, EV-]) showed that modelled cranefly abundance varied widely, was positively related to altitude and the proportion of deep peat (used as a proxy for soil moisture) and varied between years. But there was no significant relationship between vegetation height and cranefly abundance. Spatial variation in golden plover density was negatively related to mean vegetation height, but there was little evidence that cranefly abundance, the proportion of deep peat or between-year effects strongly influenced spatial variation in breeding density. The probability of a pair fledging young was positively related to cranefly abundance. The strongest driver of variation in the number of successful pairs was vegetation height, but spatial variation in cranefly abundance was positively related to altitude (with the highest numbers at cool, high-altitude locations) and an index of peat depth (with the highest numbers at wet, deep peat areas).

5.60 Heinemeyer and others (2019c/2023 [1+, EV-]) showed significantly higher cranefly numbers in mown compared to burnt sub-catchments between 2014 and 2016 which was highest in the driest year, but subsequently no difference from 2017 to 2022. Modelling by from transect counts predicted that mean cranefly abundance over the then three years of the study was 64% higher in the mown compared to the burnt sub-catchments. This translated into a predicted probability of a pair of golden plover successfully fledging young to be 33% higher under the mown treatment. Similarly, modelling from cranefly emergence traps in treatment plots translated into similar patterns for predicted cranefly and bird numbers (in this case dunlin, golden plover and red grouse). It was suggested that mowing may provide some resilience to the predicted increase in future summer droughts for a key ecological food chain on blanket bog, whereas continued burning practice is likely to aggravate the impacts. However, it was noted that hydrological conditions during winter, specifically considering predicted increases in precipitation, warrants further investigation considering an observed possible upper soil moisture limitation for cranefly emergence.

5.61 Bargmann and others (2015/2016 [2+, EV-]) investigated the species diversity and composition of carabid ground beetles over a 22-year chronosequence of time since burning in two adjacent coastal heaths in western Norway. Burning increased alpha species richness and was considered particularly important for the richness of typical open habitat species. There were clear compositional differences between assemblages along the chronosequence and species richness was also increased by higher species turnover between consecutive years in recently burnt patches compared to longer-unburnt patches. Bargmann and others (2016) identified characteristic carabid species of post-burn successional stages and identified traits that were characteristic of species in burnt areas and areas dominated by older stands. Ten species were identified as indicator species for the pioneer stage (0–5 years old), and single species for the building (6–14 years) and mature stages (15–25 years). Moisture preference and diet were identified as traits that determine species response to prescribed fire. Specialist springtail (Collembola) predators and species with no moisture preference were most abundant in burnt patches, whereas generalist predators and species with a high moisture preference are less tolerant of fire. Carabids are probably less characteristic of peatlands than drier habitats.

5.62 Byriel and others (2023 [2+, EV-]) investigated how ‘old-growth’ heath (>30 years unmanaged) affected species richness and composition of bees (Anthophila), craneflies (Tipulidae), ground beetles (Carabidae), hoverflies (Syrphidae) and rove beetles (Staphylinidae) in relation to their hygropreference (relative humidity range) compared to wet heath and managed dry heath. Species composition differed between intensively managed (burned, grazed, cut), old-growth and wet heathland for all taxa. Indicator species and richness analyses showed a predominance of xerophilic (affinity with low water availability) bee species in managed heathland. Old-growth heathland showed a predominance of mesophilic (affinity with moderate environment) indicator species, and higher richness of mesophilic craneflies and of hygrophilic (affinity for high water availability) ground and rove beetles compared to managed heathland. As would be expected, wet heathland was generally dominated by hygrophilic species. Soil moisture, bare soil and vegetation height density were important drivers explaining the contrasting responses in richness and composition between heathland types. But perhaps surprisingly, there was no difference in red-listed species between managed, old-growth and wet heathland, and xerophilic species seemed to persist to some extent in old-growth heathland sites. It was concluded that the results demonstrate that heathland management, such as burning, focusing on early successional vegetation stages may homogenize insect communities. Rather, management practices should focus on improving structural vegetation heterogeneity. This can be achieved by allowing patches of old-growth vegetation stages to develop and by conserving existing ones.

5.63 Newey and others (2020 [2+, EV++]) compared species distribution data with an index of mapped burning intensity on Scottish grouse moors for selected flora and fauna species including one invertebrate (in 1-km squares), the green hairstreak butterfly. It was recorded from 1,475 1-km squares in Scotland of which only 83 (5.6%) overlapped with assessed grouse moor squares. Most grouse moor records occurred within squares with less than 40% burning, though there was no clear pattern in the percentage of occupied squares with increasing burn area.

5.64 In linked reviews for Natural England, Gillingham and others (2016a/b [4+, EV+]) reviewed the ecology and effect of burning and other management on heather beetle and post beetle-damage recovery. They concluded that there is currently no evidence that burning serves to reduce the likelihood of heather beetle outbreaks or reduce existing heather beetle numbers. Evidence on whether burning encourages ‘regeneration’ of damaged or killed *Calluna*-dominated vegetation was considered generally of low quality, with some suggestion that management techniques other than burning might be more effective at encouraging regrowth. In addition, some sites have been observed to regenerate naturally in the absence of management, so there remains a question over whether management is necessary.

Aquatic invertebrates

5.65 Brown and others (2013/2019 [2+, EV+]) investigated the effects of burning and sedimentation on aquatic invertebrates in ten Pennine peatland headwaters, sampled six times each over a period of 20 months as part of the EMBER project, along with a riverside mesocosm experiment comprising 24 channels alongside one watercourse (Brown and others, 2019). Significant effects of burning, season and their interaction were found on upland peatland river macroinvertebrate communities, with watercourses draining burned catchments having significantly lower taxonomic richness and diversity. There was also a significant effect of burning on macroinvertebrate community composition, typically with reduced mayfly (Ephemeroptera) abundance and diversity and greater abundance of chironomids (Chironomidae) and brown stoneflies (Nemouridae). Grazer and collector-gatherer feeding groups were also significantly less abundant in watercourses draining burned catchments. These biotic changes were associated with lower pH and higher Si, Mn, Fe and Al in burned systems. It was concluded that vegetation burning on peatlands has effects beyond the terrestrial part of the system. Brown and others (2019 [1,2+, EV+]) show an adverse impact of peat deposition on invertebrate community biodiversity in both sediment deposition gradients in experimental mesocosms and headwater surveys of the same ten watercourses. This was evident at the community level, including decreases in density and richness, and increased beta diversity. Traits analysis of mesocosm assemblages suggested biodiversity loss was driven by decreasing abundance of invertebrates with trait combinations sensitive to sedimentation (longer life cycles, active aquatic dispersal of larvae, fixed aquatic eggs, shredding feeding habit).

5.66 Johnston & Robson (2015 [1+, EV+]) investigated the effects of ash input (as a proxy for burning) in four Pennine upland peatland headwater streams (also EMBER sites) on macroinvertebrate assemblages by depositing ash onto trays filled with natural stream substrata. Before the experiment, streambed samples were taken to describe ambient macroinvertebrate assemblages. Macroinvertebrate response after 21 days was compared among low, high and top-up (dosed twice) ash-addition treatments and control trays (no ash addition). Additions increased tray ash-free dry mass and by the end of the experiment some trays retained more ash than others. Macroinvertebrate assemblages differed among streams and treatments. Streambed samples contained fewer shredders than did other treatments. A significant relationship was found between assemblages and environmental conditions. Stream depth and ash-free dry mass showed strongest correlations with assemblages.

5.67 Johnston (2012 [2-, EV+]) compared catchment characteristics, physicochemical variables and macroinvertebrate community assemblages in 30 peatland watercourses within ‘intact’ (not burnt, drained, eroded or afforested), burnt and eroded (not burnt) catchments (10 in each, some EMBER sites). Intact sites had greater macroinvertebrate diversity than burnt or eroded sites and higher proportions of pollution sensitive mayflies (Ephemeroptera), caddisflies (Trichoptera) and stoneflies (Plecoptera), it was suggested as a result of being less disturbed and having higher habitat and water quality than other site types. Macroinvertebrate diversity of burnt and eroded sites was similar, whilst that for eroded sites was lowest. It was suggested that this may reflect the impact of the release of various chemicals, including heavy metals, from eroding peat (reviewed, for example, by Abraham and others, 2017), and also increased organic loads and sedimentation in burnt catchments. Across site types, those with larger catchment areas had a greater diversity of macroinvertebrates.

Reptiles

5.68 Only one study reported on the effect of burning on reptiles. Newey and others (2020 [2+, EV++]) compared species distribution data with an index of mapped burning area on Scottish grouse moors for selected flora and fauna species including adder. It was recorded from 810 1-km squares in Scotland of which only 77 (9.5%) overlapped with assessed grouse moor squares. Although most grouse moor records occurred within squares with less than 20% burning, there was no clear pattern in the percentage of occupied squares with increasing burn intensity.

Mammals

5.69 Only one study reported on the effect of burning on mammals. Bedson and others (2022b [2+, EV+]) systematically surveyed mountain hare on transects at three main sites in the Peak District, each in 13 1-km squares, and across 26 random 1-km squares across their range. Following bog restoration, hare densities (32.6 individuals km^{-2}) were notably higher than on neighbouring ‘degraded (unrestored) bog’ (24.4 km^{-2}). Hare density on ‘restored peatland’ was 2.7 times higher than on bogs managed for grouse shooting (12.2 km^{-2}) and 3.3 times higher than on ‘heather moorland’ managed for grouse shooting (10.0 km^{-2}). Acid grassland used for sheep farming had a similar density to grouse moorland (11.8 hares km^{-2}). Unmanaged dwarf shrub heath had the lowest density (4.8 km^{-2}). On both ‘grouse moor bog’ and ‘heather moorland’, *Calluna* existed in such large deep expanses that movement for hares through them was considered to be difficult. Yearly estimates varied most on habitats managed for grouse, perhaps indicative of the impact of habitat management, for example, heather burning and/or possible hare culling to control potential tick-borne louping ill virus in gamebirds. During an earlier survey in 2002, total abundance throughout the Peak District National Park was estimated at 3,361

(95% CI: 2,431–4,612) hares. The present study estimated 3,562 (2,291–5,624) hares suggesting a stable population over the last two decades despite fluctuations likely influenced by weather and anthropogenic factors.

5.70 Hesford & Macleod (2022) report on a subsequent a non-random, night transect (using lamps) survey of 12 sites from nine Peak District grouse moor estates surveyed by gamekeepers between December 2021 and January 2022, which suggested higher densities reaching 52–125 km^{-2}. In response, Bedson and others (2022a) note that the method was based on a weak, non-significant relationship between hare encounter rates using spotlight surveys of walked transects at night and estimated densities derived from spatial capture-recapture methods. These were tested in a review and the authors, Newey and others (2018), recommended they should not be used to estimate hare densities, and that the reportedly high mountain densities were biased and based on a model with little predictive power.

Soil Microbes

5.71 Burn (2021 [1,2+, EV+]) used samples from the three Peatland-ES-UK experiment sites at Moor House NNR and three other upland peatland areas, to investigate soil microbial community taxonomy and fungal community function (para. 5.6), including in relation to burning and cutting management. The microbial community varied significantly with management across fungi, bacteria and archaea, with the principal differences being between the Peatland-ES-UK sites managed as grouse moors (burnt, mown and uncut plots), land previously (pre-1950s) managed as grouse moor (Moor House NNR) and three other 'national' areas (Exmoor, Peak District and Forsinard Flows, Scotland) with multiple sites reflecting differences in degree of habitat modification and time since restoration treatments. Sites in the three 'national' areas showed very little difference. Fungal trophic groups varied between management types, but only between burnt plots and everything else in the case of the symbiotrophs, and burnt and previously burned but recently unmanaged plots and everything else in the case of the saprotroph-symbiotrophs (although notably the uncut plots were burnt *c*.25 years prior to sampling as part of previous grouse moor rotational burning management before the experiment treatments started). Shannon diversity did not differ between management type in the fungi or the archaea. In the bacteria, only Moor House was different, having much higher Shannon diversity, but notably these samples were collected a few months later in summer, possibly explaining this difference. The hypothesis that 'intact' sites would show higher diversity was not confirmed. Microbial communities varied significantly with 'national' site location. Climate variables played a significant part in explaining this difference, with rainfall and soil temperature also significant.

5.72 Microbial community measures were significant in relation to soil pore water quality variables including DOC (dissolved organic carbon), SUVA (Specific ultraviolet absorbance) and Hazen (colour). Saprotroph-symbiotroph fungi were significantly different between burnt plots and mown plots, as well as between burnt plots and all 'national' sites, although not different to uncut plots. Burnt plots had the highest relative abundance of saprotroph-symbiotrophs of all management treatments. In a piecewise structural equation model examining the relationship between microbial and pore water quality variables, saprotroph-symbiotrophs were found to be significant in their effect on DOC and Hazen. As their abundance increased, so did the water quality variables. It was noted that these results only infer an effect of burning on water quality via saprotroph-symbiotrophs, rather than proving a causal link, though they suggest that burning has a modifying effect on peat microbial communities. The main variables associated with an increase in saprotroph-symbiotrophs were pH, soil moisture and the amount of bare ground, all of which might be affected by burning, especially immediately post-burn.

6. Recent evidence on the effects of managed burning of upland peatlands on carbon balance

6.1 The full text of this sub-question is:

What are the effects of managed burning of upland peatlands on carbon balance?

Introduction

6.2 The carbon storage capacity and balance of upland peatlands is described in NEER004 (para. 6.2). NEER004 also assessed relevant studies on the effects of burning on carbon sequestration up to 2012 (see Appendix 6, pp. 121–129), with a summary and brief interpretation across studies as evidence statements (main text paras. 6.5–6.12) and research recommendations (para. 6.12 and paras. 12.35–12.36). The evidence statements also feature in summary form in the Conclusions (paras. 12.3–12.6) and Summary (pp. v-vi). NEER004 also included studies of other aspects of soil in the carbon section as does this update.

6.3 There is some crossover between this sub-question and those concerning vegetation (Section 2) and water (Section 7). Vegetation is directly linked to aboveground carbon stock, whilst the fluvial export of DOC and POC (particulate organic carbon) have implications for water quality as well as carbon budgets. There is also some overlap in relation to soil microbes which is included in Section 5 (fauna, paras. 5.71–5.72).

Recent studies on the effects of burning on carbon balance

6.4 Twenty recent studies (reported in 29 references) since NEER004 provide evidence on the effects of managed burning on carbon storage and fluxes.

6.5 Five studies were at least partly based on data from Moor House NNR in the North Pennines, including four which used data from the Hard Hill burning and grazing experiment. Although these were not direct continuations of existing studies, nine of the studies relating to carbon reviewed in NEER004 also used data from Moor House NNR. Additionally, Worrall and others (2013a [1,2+, EV-]) collected plant material from Moor House NNR for use in a laboratory experiment, though the field-based elements of this study were conducted elsewhere.

6.6 Eighteen studies related specifically to carbon stocks and fluxes. Two other studies included the collection of data relevant to carbon outcomes (Rosenburgh and others, 2013 [2+, EV-]; Chapman and others, 2017 [2,4+, EV-]).

6.7 Fifteen studies on carbon outcomes related directly to managed burning, while four examined the effect of vegetation composition, which may itself be affected by burning (Ward and others, 2013 [1,2+, EV-]; Parry and others, 2015 [2+, EV+]; Ritson and others, 2016 [1+, EV+]; Dunn and others, 2016 [2-, EV-]). One study investigated the effect of heather canopy height, which was considered a proxy for time since burning on burned sites (Dixon and others, 2015 [2+, EV-]), and one prescribed burning study also included data from a wildfire which occurred during the study period (Brown and others, 2014 [2+, EV+], wildfire element reported in Blundell and others, 2013).

Study type and quality

6.8 A summary of the type and quality of the 20 recent carbon studies is given in Table 14.

Table 14. Categorisation of recent carbon evidence studies by study type and quality. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	0	5	0	3
1/2: both type 1 and 2	1	5	0	8
2: quantitative observational or correlative	0	8	1	9
3: qualitative or 4 review	0	0	0	0
Total	1	18	1	20

6.9 Ten studies involved field experiments. Of these, three studies used the existing Hard Hill burning and grazing experiment (Ward and others, 2012 [1,2+, EV-]); Alday and others, 2015 [1,2+, EV-]; Marrs and others, 2019 [1,2+, EV-]). A further experimental study was conducted at Moor House NNR (Ward and others, 2013 [1,2+, EV-]), which involved warming treatments and manipulations of vegetation composition rather than burning. Of the remaining field experiment studies, two involved experimental burns at sites in Scotland (Grau-Andrés and others, 2017a [1+, EV-]; Grau-Andrés and others, 2019b [1+, EV-]), two were based on burning and cutting experiments in Northern England (Qassim, 2015 [1+, EV-]; Heinemeyer and others, 2019c [1,2+, EV-]), one investigated decomposition of different plant species from sites in South West England (Ritson and others, 2016 [1+, EV+]), and one exposed plant matter burned in a laboratory to terrestrial and aquatic field conditions in the Peak District (Kennedy-Blundell, 2020 [1+, EV-]).

6.10 Seven studies were primarily based on observation and/or monitoring in the field. Of these, four used a chronosequence approach (see Section 4, para. 4.14) to investigate carbon-related outcomes during the post-burning succession. Of the remainder, Worrall and others (2013a [1,2+, EV-]) used pre- and post-burn measurements to investigate biomass losses and char production during fire, and Parry and others (2015 [2+, EV+]) investigated impacts of slope and vegetation type. Heinemeyer and others (2019c [1,2+, EV-]) included observational work which focussed on the management history of the study sites, in addition to the experimental part of the study.

6.11 The observation and monitoring studies employing a chronosequence approach used burn patches ranging from newly burned up to 10-20 years old, and all included comparison plots which had been unburned for 13+ to 40+ years but not unmanaged or unmodified sites. The work of Alday and others (2015 [1,2+, EV-]) and Ward and others (2012 [1,2+, EV-]), which investigated the Hard Hill experimental plots (which do include unburned treatments on a previously burned site), can also be considered chronosequence studies as both surveyed plots at a single point in time when they were at various burn ages.

6.12 Five studies had significant ex-situ components. This includes mesocosm work by Burn (2021 [1,2+, EV+]) using samples from the experimental sites of Heinemeyer and others (2019c [1+, EV-]) and additional sites, and laboratory-based experimental burns conducted by Worrall and others (2013a [1,2+, EV-]) and Kennedy-Blundell (2020 [1+, EV-]), as well as laboratory analysis of plant material by Ritson and others (2015 [1+, EV+]) and soil samples by Dunn and others (2016 [2-, EV-]).

6.13 Modelling was the main component of three studies. Chapman and others (2017 [2,4+, EV-]) and Li and others (2017 [2+, EV+]) used mostly remotely-sensed and pre-existing data as model inputs. Santana and others (2016 [2+, EV+]) used data from four existing studies (Chapman and others, 1975; Miller and others, 1979; Allen and others, 2013; Alday and others, 2015 [1,2+, EV-]).

Outcome measures

6.14 The 20 carbon studies assessed 11 main carbon outcome measures given in Figure 7. Respiration and DOC were the most common measures, with ten studies providing evidence. Several studies reported on multiple carbon outcome measures, but only two studies attempted to calculate an overall carbon budget or Net Ecosystem Carbon Balance (NECB) (Clay and others, 2015 [2+, EV-] and Heinemeyer and others, 2019c [1+, EV-]).

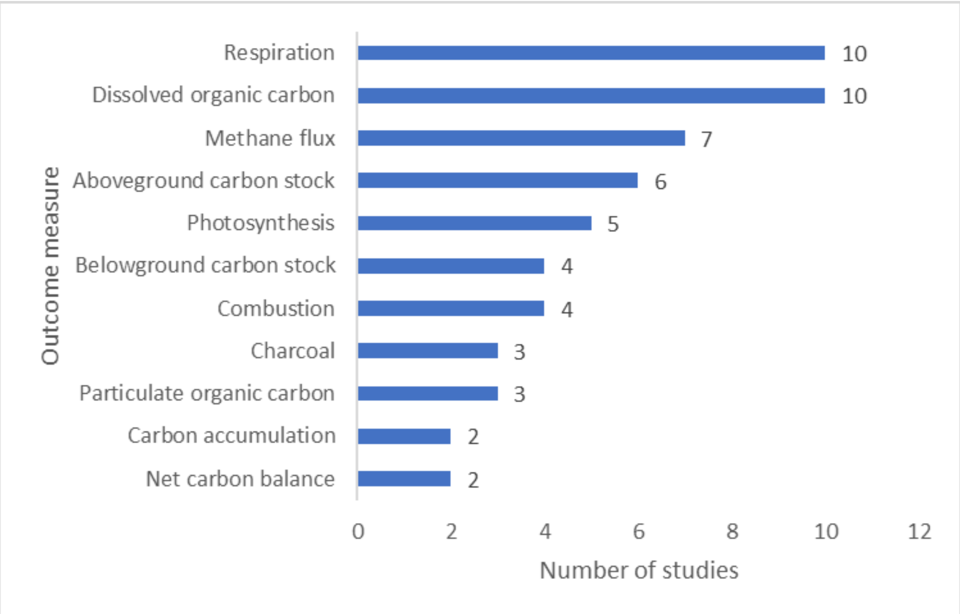

Figure 7. Outcome measures recorded or derived from recent carbon studies (n = 20 studies) - some studies covered more than one outcome.

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

Countries, areas and number of sites

6.15 All 20 of the recent carbon studies were from the UK. Most (17) were from or included sites in England, with three from or including sites in Scotland and a single study from Wales. Within England, most studies primarily used data from northern regions (Figure 8), with most of the research concentrated in the Pennines. The only exceptions to this were experimental work in Dartmoor and Exmoor by Ritson and others (2015 [1+, EV+]) and data taken from lowland sites in Dorset by Santana and others (2016 [2+, EV+]). This reflects the fact that a large proportion of English blanket bog occurs in the Pennines, but nevertheless the full range of variation in the upland peatland resource is not represented by the studies available.

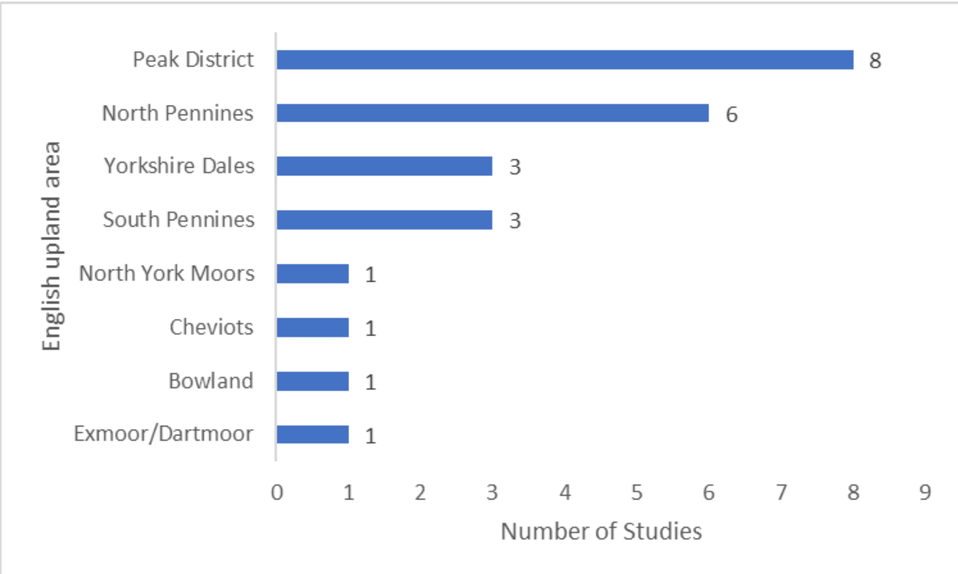

Figure 8. English upland areas covered by recent peatland carbon studies (n = 18 studies) – some of the studies covered more than one area.

6.16 Most of the studies involved a small number of sites with seven single site studies and 12 involving two to five sites. Only one study (Brown and others, 2014 [2+, EV+]) involved more, with ten sites (five burned and five unburned) variously located in the North and South Pennines, Yorkshire Dales and Peak District.

Habitats and vegetation types

6.17 Most studies (17) were carried out primarily on blanket bog habitats. Exceptions to this were studies focusing on heath (Grau-Andrés and others, 2017a [1+, EV-]), both heath and raised bog (Grau-Andrés and others, 2019b [1+, EV-]), and undifferentiated 'moorland' which likely included wet heath (Chapman and others, 2017 [2,4+, EV-]). Some studies included other habitats in addition to blanket bog, for example lowland heath in Santana and others (2016 [2+, EV+]).

6.18 Where NVC type was given or could be derived (13 studies), M19 and M20 community types were most common, with all 13 studies including one or both types. Additional NVC types represented included M6, H9 and H12 in Brown and others (2014 [2+, EV+]), as well as H2 and H10/12 in Santana and others (2016 [2+, EV+]), with at least some of these occurring on deep peat.

Types of burn and fire

6.19 The burns investigated by the 15 studies which focussed directly on burning were mainly ‘routine’ managed burns (6) or experimental burns (7). The remaining two studies modelled managed burning based on existing data. Worrall and others (2013a [1,2+, EV-]) included laboratory-based burning in addition to managed burns, whilst Heinemeyer and others (2019c [1+, EV-]) included cutting alongside burning treatments.

External validity

6.20 As with the recent vegetation studies, most of the recent carbon studies included a relatively small number of sites with limited geographic spread and bias towards the Pennines in relation to the English upland peatland resource. Therefore, most carbon studies were classed as [EV-] for external validity (Table 15). The remainder of studies were classed as EV+ as the spread of sites or coverage of modelling provided evidence at a larger (regional or national) scale.

Table 15. Categorisation of recent carbon studies by study type and EV (external validity). EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	1	4	5
1/2: both type 1 and 2	0	1	5	6
2: quantitative observational or correlative	0	4	5	9
3: qualitative or 4 review	0	0	0	0
Total	0	6	14	20

Summary of recent evidence on the effects of burning on carbon balance

6.21 This section presents a summary and synthesis of the findings of the 20 recent studies on the effects of burning on carbon balance. It is organised by themes which encompass each of the 11 outcome measures listed in Figure 7.

Aboveground carbon stock

6.22 Six studies reported on aspects of aboveground carbon stock in relation to burning. Some of their findings are relevant to vegetation as well as carbon and these are discussed in Section 4.

6.23 Four studies reported total aboveground biomass and all found that this increased with time since burning (Table 16). Two of these studies also calculated aboveground carbon stock, with carbon making up approximately half the weight of total aboveground biomass (Worrall and others, 2013a [1,2+, EV-]; Clay and others, 2015 [2+, EV-]).

Table 16. Values for total aboveground biomass and C (carbon) stock according to time since burn from recent carbon studies. Arrows indicate increase or decrease with time.

Main study reference	Years post-burn	Aboveground biomass g m^{-2} for years post-burn	C stock g m^{-2}
Alday and others 2015 [1,2+, EV-]	5, 16, 56, 87+	1198 ↑ 1593 ↑ 2079 ↑ 2223	Not measured
Clay and others 2015 [2+, EV-]	1–13, 13+	409 ↑ 860	201 ↑ 437
Ward and others 2012 [1,2+, EV-]	1, 53	248 ↑ 862	Not measured
Worrall and others 2013a [1,2+, EV-]	<1, pre-burn (age unknown)	197 ↑ 880	97 ↑ 445

6.24 Four studies investigated various fractions of aboveground biomass. This included one study which reported increases in both *Calluna* and litter biomass with time since burning (Santana and others, 2016 [2+, EV+]). Heinemeyer and others (2019c [1+, EV-]) measured *Calluna* biomass and reported large reductions on burned and cut plots compared to pre-treatment and unmanaged comparisons. Data from the Hard Hill experiment suggests that the proportional contribution of different fractions to total aboveground biomass varies with burn age, with bryophytes representing a larger proportion of the total in the immediate post-burn period (Ward and others, 2012 [1,2+, EV-]; Alday and others, 2015 [1,2+, EV-]).

6.25 Santana and others (2016 [2+, EV+]) found that biomass accumulation rate varied between sites, though there was no clear relationship with site climate. At three of the four sites included in their study, plateaux in both *Calluna* and litter biomass were observed 20-30 years after burning, though the time taken and maximum biomass reached varied between sites. At a fourth site, biomass had a linear relationship with time over the period the data covered (0–50 years).

Belowground carbon stock

6.26 One study (Heinemeyer and others, 2018 [2-, EV-]) quantified soil carbon stock for three burned sites. The results showed that the site with the greatest average historical burn rotation length (28 years) had a lower total belowground carbon stock (65 kg m^2) than more frequently burned sites (25- and 23-year average historical rotation lengths with 81 and 95 kg m^2 carbon stock respectively). However, this finding may be confounded by other differences between sites, including drainage history and climate (Evans and others, 2019).

6.27 Two studies investigated soil carbon or organic matter concentration. Rosenburgh and others (2013 [2+, EV-]) reported no change in carbon concentration with time since burning, but Brown and others (2014 [2+, EV+]) found that burning led to a lower proportion of organic matter in soil. This was hypothesised to be a result of dilution by ash and therefore does not necessarily indicate a change in overall carbon stock.

Combustion

6.28 Five studies investigated aspects of combustion and associated carbon loss. All of the modelled and measured carbon losses in these studies were based on combustion of aboveground biomass.

6.29 One study examined the impact of burning rotation length on annual carbon loss associated with burning and concluded that the loss was lower at longer rotations in the absence of wildfire (Santana and others, 2016 [2+, EV+]).

6.30 By expressing aboveground carbon stocks 1–13 years after burning as a proportion of control plot stocks and extrapolating the regression line to the time of burning, Clay and others (2015 [2+, EV-]) estimated that around 360 g m^2, or 80% of total stocks, may be consumed by a burn. Worrall and others (2013a [1,2+, EV-]) calculated a similar figure for consumption (76%) by measuring pre- and post-burn biomass in the field, with lab experiments in the same study showing that carbon loss was greater at higher temperatures and longer durations of exposure. Heinemeyer and others (2019c [1+, EV-]) estimated combustion carbon losses of 543–593 g m^2 using manual harvesting methods.

Charcoal

6.31 Worrall and others (2013a [1,2+, EV-]) found that higher burning temperatures were associated with greater production of charcoal (which is a recalcitrant material in which carbon is stored over very long timescales). In the field, around 2% of pre-burn biomass survived as charcoal.

6.32 Heinemeyer and others (2018 [2-, EV-]) used charcoal presence as an indicator of past fire and found that it was associated with increased bulk density and carbon accumulation in the peat profile, though carbon flux and peat depth measurements from the same site showed different effects of burning (see paras. 6.51 and 6.53).

6.33 Kennedy-Blundell (2020 [1+, EV-]) studied burned plant material (referred to as pyrogenic carbon or PyC) ranging from lightly charred biomass to more completely burned material including charcoal. Burn severity was found to impact the characteristics of PyC, with higher severities resulting in greater PyC surface area, greater aromaticity and lower proportional hydrogen and oxygen content. However, higher severity burns are also likely to result in higher CO_2 losses on combustion which may affect the amount of PyC produced. Changes in PyC composition after field exposure were also observed, with a loss of labile carbon over short timescales (1 month) followed by stabilisation in the longer term (12 months). PyC from lower severity burning was associated with greater loss of carbon during the first weeks of terrestrial and aquatic exposure than that from higher severity burning.

Gas exchange

6.34 Nine studies investigated aspects of gaseous fluxes of carbon, including gross primary productivity (GPP), ecosystem and soil respiration, net ecosystem exchange of CO_2 (NEE), methane flux and net greenhouse gas (GHG) flux.

6.35 Of four studies reporting GPP, two found negative impacts of burning and two found no difference. Grau-Andrés and others (2019b [1+, EV-]) observed smaller CO_2 uptake from photosynthesis between 123 and 744 days after burning compared to before burning. Heinemeyer and others (2019c [1+, EV-]) also observed smaller GPP on recently (2-3 years) burned plots compared to longer-unburned plots. However, a comparison between plots one and 53 years after burning at the Hard Hill experiment found no difference in GPP (Ward and others, 2012 [1,2+, EV-]). Clay and others (2015 [2+, EV-]) found that although GPP varied according to burn year in plots burned 1-11 years previously, there was no trend over time or consistent difference between burn and control plot values.

6.36 Four studies measured ecosystem respiration, three in relation to burning and one in relation to vegetation composition. Of these, two found smaller respiration fluxes on burned plots than pre-burning (Grau-Andrés and others, 2019b [1+, EV-]) or longer-unburned comparisons (Heinemeyer and others, 2019c/2023 [1+, EV-]). Cutting treatments in the experiment of Heinemeyer and others (2019c [1++, EV+]) also had greater ecosystem respiration than comparison plots. In contrast with field measurements, soil mesocosms from the sites of Heinemeyer and others (2019c [1+, EV+]) showed no difference in respiration according to burning treatment. As with GPP, Clay and others (2015 [2+, EV-]) found that ecosystem respiration varied between plots burned in different years, but with no trend over time or consistent difference to control plots. Ward and others (2013 [1,2+, EV-]) found that plant species manipulations impacted ecosystem respiration, with shrubs and grasses contributing more than bryophytes.

6.37 Three studies (Brown and others, 2014 [2+, EV+]; Grau-Andrés and others, 2017a [1+, EV-]; Heinemeyer and others, 2019c [1+, EV-]) found changes to soil thermal regime after burning. Brown and others (2014 [2+, EV+]) found that mean and maximum soil temperatures were higher and minima lower on recently burned plots compared to plots burned ≥15 years previously. Grau-Andrés and others (2017a [1+, EV-]) measured temperatures for a year following experimental burning and found greater daily range, higher summer mean and lower winter mean temperatures in burned compared to control plots. The same study then estimated soil respiration from temperature-driven models and reported higher summer values for burned plots. Heinemeyer and others (2019c [1+, EV-]) found that maximum temperatures were higher in the two years after burning than before but hypothesised that greater differences observed in other studies may be inflated by temperature sensors heating up (to a greater extent than the soil surface) when exposed to direct radiation.

6.38 Heinemeyer and others (2019c/2023 [1+, EV-]) found that burned plots had a lower rate of soil respiration than longer-unburned comparisons over an 8-year period, whereas in cut plots where the cut material (brash) was removed, a lower rate was only apparent in the short term (3 years). Dunn and others (2016 [2-, EV-]) found that peat from beneath *Sphagnum* emitted less CO_2 than peat from beneath *Calluna* or *Juncus effusus*, indicating lower rates of soil respiration.

6.39 Five studies reported on NEE (net ecosystem exchange of CO_2 considering both GPP and ecosystem respiration). Of three studies looking directly at burning impacts, two found that recently burned plots had positive NEE (i.e. net CO_2 losses) compared to pre-burning (Grau-Andrés and others, 2019b [1+, EV-]) or longer-unburned comparisons (Heinemeyer and others, 2019c/2023 [1+, EV-]) which had negative NEE values. In the latter study, recently cut plots also had positive NEE. As with respiration, work by Burn (2021 [1,2+, EV+]) on soil mesocosms from the sites of Heinemeyer and others (2019c [1+, EV-]) showed no difference in NEE according to burning treatment. Clay and others (2015 [2+, EV-]) found that plots burned one year ago had the most negative NEE, whilst plots burned 8-10 years ago had the most positive NEE (greatest CO_2 losses). Of two longer-unburned control sites used by Clay and others (2015 [2+, EV-]), one had on average positive NEE and one negative NEE. One study which used heather height as a proxy for burn age found that all heights had positive NEE, and that this became more positive with increasing height as photosynthesis per unit respiration declined (Dixon and others, 2015 [2+, EV-]). Finally, Ward and others (2013 [1,2+, EV-]) found that plant species manipulations impacted NEE, with GPP by shrubs and graminoids contributing to negative NEE values, whilst plots containing only bryophytes had a positive NEE.

6.40 Methane fluxes in relation to burning were reported in three studies, with mixed impacts observed. Heinemeyer and others (2019c /2023 [1+, EV-]) found that recently burned plots had lower median methane fluxes than longer-unburned comparisons, with recently cut plots intermediate. Leaving brash after cutting also led to greater methane fluxes than removing it. As with other fluxes, this impact was not evident in work by Burn (2021 [1,2+, EV+]) on mesocosms from the same sites. In contrast, Grau-Andrés and others (2019b [1+, EV-]) observed a large increase in methane fluxes after burning at a raised bog site, with summer values of 1.16 $\mu mol\ m^{-2}\ s^{-1}$ before burning and 25.3 $\mu mol\ m^{-2}\ s^{-1}$ one year after. This study also measured methane fluxes at a heathland site, but these were found to be mostly negligible. Methane values reported by Clay and others (2015 [2+, EV-]) were derived from water table data and showed no impact of burning.

6.41 Two studies investigated the impacts of vegetation composition on methane, with both finding that graminoids were associated with greater methane emissions than dwarf shrubs or bryophytes. Dunn and others (2016 [2-, EV-]) found that net methane fluxes from peat dominated by *Juncus effusus* were always positive, while fluxes from *Calluna* and *Sphagnum* dominated peat were negative in samples taken at 10 and 30 cm depth in summer and at 10 cm in winter, but positive at 30 cm in winter. Meanwhile in a vegetation removal experiment, Ward and others (2013 [1,2+, EV-]) found that plots with *Eriophorum vaginatum* alone had greater positive methane fluxes than plots with dwarf shrubs or bryophytes alone, mixed vegetation communities, or no vegetation. These results suggest that methane fluxes may be influenced by vegetation change in the post-burning period, particularly by periods of graminoid dominance.

6.42 One study calculated net GHG fluxes over ten years and found that recently burned and recently cut plots were a significant source of GHG (CO_2 equivalent of 232 and 338 t km^{-2} yr^{-1} respectively) compared to longer-unburned comparison plots, which were a GHG sink (CO_2 equivalent of -104 t km^{-2} yr^{-1}) (Heinemeyer and others, 2019c [1+, EV-] reported in Heinemeyer and others, 2023).

Fluvial carbon export

6.43 Ten studies reported on DOC, including five examining the effect of burning (two of these also included cutting treatments), one the effect of wildfire and four the effect of vegetation. Nine of these studies reported on soil water DOC concentrations, with four additionally reporting on run-off DOC and two on watercourse DOC. One study (Kennedy-Blundell, 2020 [1+, EV-]) reported on DOC fluxes from burned plant material in a laboratory setting.

6.44 Clay and others (2015 [2+, EV-]) found no trend in the flux of soil DOC in plots burned 1–13 years ago but measured higher soil DOC fluxes at longer-unburned control sites. For run-off DOC no trend with time since burn was observed and there was no difference between burned and longer-unburned sites. Grau-Andrés and others (2019b [1+, EV-]) found no difference in soil DOC concentration prior to and within the first two years after burning. Similarly, Heinemeyer and others (2019c [1+, EV-]) found no effect of management on either soil or watercourse DOC in the three years following burning and cutting, though the sites had been rotationally burnt prior to the experiment. Qassim (2015 [1+, EV-]) found that vegetation management had complex relationships with soil and runoff DOC which were influenced by factors including hydrological status. At dry localities, soil and runoff DOC concentrations increased with time up to 15+ years after burning, whilst at wet localities soil DOC increased but runoff DOC decreased with time since burning. Brown and others (2014 [2+, EV+] reported in Blundell and others, 2013) found that on plots affected by wildfire on an EMBER site, DOC increased in soil water but decreased in overland flow. However, this study found no impact on watercourse DOC in the weeks following wildfire or two years after.

6.45 Kennedy-Blundell (2020 [1+, EV-]) found that burned plant material from low severity burns resulted in higher DOC fluxes than that from high severity burns, and that these peaked after one week of terrestrial exposure. Similarly, material from low severity burns lost more carbon (DOC) after a month of submersion in water.

6.46 Four studies reported on water colour which is often correlated with increased DOC export (Wallage & Holden, 2010). Chapman and others (2017 [2,4+, EV-]) found a spatial association between burning and increased water colour, whilst Heinemeyer and others (2019c [1+, EV-]) found no difference in water colour between burned, cut or less recently managed plots, nor between watercourses draining burned and cut catchments. Brown and others (2014 [2+, EV+] reported in Blundell and others, 2013) found no influence on stream water colour in the weeks following wildfire or two years after. Kennedy-Blundell (2020 [1+, EV-]) found that burned plant material can have a significant effect on water colour during the early weeks of field exposure.

6.47 Four studies reported evidence of vegetation effects on soil DOC. Dunn and others (2016 [2-, EV-]) found that peat from beneath *Sphagnum* had the lowest DOC overall, and that DOC from *Juncus effusus* and *Calluna* dominated areas varied seasonally and with depth. Ward and others (2013 [1,2+, EV-]) found that the removal of dwarf shrubs increased DOC concentrations measured one year later. Ritson and others (2016 [1+, EV+]) compared litter from different peatland species and found that *Molinia* produced the most DOC followed by *Calluna* and *J. effusus*, then *Sphagnum*, then peat. This study also found that vascular plants produced more litter with pronounced seasonal peaks compared to *Sphagnum*. Parry and others (2015 [2+, EV+]) concluded that the impact of broad plant functional type (PFT) on DOC and colour at the catchment scale was weak compared to other drivers.

6.48 Four studies considered POC fluxes. Clay and others (2015 [2+, EV-]) found a tendency for greater POC export in the early years after burning, while Brown and others (2014 [2+, EV+]) found up to four times higher POC in streams draining burned catchments than those draining unburned catchments. Heinemeyer and others (2019c [1+, EV-]) found no overall impact of management when comparing POC export from burned and cut catchments at three sites, though export was significantly higher from the burned catchment at one site. Finally, in a modelling study where future land management scenarios were considered, Li and others (2017 [2+, EV+]) found that an intensive management scenario (including grazing, artificial drainage and regular burning) was generally associated with increased peat erosion rates and sediment yields, while the opposite was true of a ‘carbon management’ scenario where burning, grazing and drainage stopped.

Carbon balance

6.49 Two studies used peat cores to determine carbon accumulation rates under different burning regimes (Marrs and others, 2019 [1,2+, EV-]; Heinemeyer and others, 2018 [2-, EV-]). The validity of this approach has been questioned by Young and others

(2019, 2021) as continued decomposition over time means that apparent carbon accumulation rates (aCAR) are not comparable between more and less recently formed peat, and the observed rate for a particular period cannot account for concurrent carbon losses from deeper peat layers.

6.50 In a study on the Hard Hill experiment, Marrs and others (2019 [1,2+, EV-]) found that burning significantly reduced carbon accumulation in the short rotation treatment (6 burns between 1954 and 2016) compared to the reference treatment (no burns since 1923). The intermediate treatments followed a linear trend whereby each additional burn reduced the apparent carbon accumulation rate by 1.9 g cm^{-2} yr^{-1}.

6.51 A study comparing three sites with a history of burning found that increasing burn frequency was associated with a greater rate of carbon accumulation (Heinemeyer and others, 2018 [2-, EV-]). The methods and conclusions of this part of the study have been questioned by Evans and others (2019) who raised several issues, including the small difference in historical burning frequencies, similar recent burning regimes and potential for confounding differences between sites, as well as potential inaccuracies in the dating of peat cores. The same study also measured peat depth using ground penetrating radar in 2012 and 2016 and concluded that peat accumulation over this period did not differ significantly between burned and unburned treatments.

6.52 Two studies calculated NECB (net ecosystem carbon balance, also referred to as carbon budget) based on measured and/or modelled fluxes of gaseous CO_2, methane, DOC and POC. One (Clay and others, 2015 [2+, EV-]) also included dissolved CO_2 flux, though noted that this makes a proportionally low contribution to peatland carbon balance and used the same modelled value in all calculations. All sites measured in this study were net carbon sources, and two longer-unburned comparison sites were greater sources (249 and 338 g C m^{-2} yr^{-1}) than plots burned 1–11 years previously (4–269 g C m^{-2} yr^{-1}). Within the recently burned site, the two most recent burn categories (one and three years since burning) were the smallest net carbon sources. The calculations do not include the loss of CO_2 on combustion (para. 6.30). It should be noted that where repeated burning takes place, a large proportion of the carbon accumulated by plants during a burning cycle is likely to be lost in combustion during the next burn, rather than being sequestered in peat.

6.53 In contrast, on sites monitored for ten years by Heinemeyer and others (2019c [1+, EV-] reported in Heinemeyer and others, 2023), plots with ongoing burning management were on average net sources of carbon (71 g C m^{-2} yr^{-1} including combustion losses, 60 g C m^{-2} yr^{-1} without), as were plots with cutting management (103 g C m^{-2} yr^{-1}), whilst longer-unburned comparisons were net sinks (-17 g C m^{-2} yr^{-1}).

7. Recent evidence on the effects of managed burning of upland peatlands on water quality and hydrology

7.1 The slightly revised, full text of this sub-question 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 distribution and flow (including downstream flood risk)

Introduction

7.2 Issues relating to water quality and hydrology in upland peatlands are described in NEER004 (para. 7.2). NEER004 also assessed relevant studies and on the effects of burning on water quality and hydrology (see Appendix 7, pp. 130–141), with a summary and brief interpretation across studies provided as evidence statements (main text paras. 7.2–7.17) and research recommendations (p. 37 and paras. 12.35–12.36). The evidence statements also feature in summary form in the Conclusions (Section 12, paras. 12.16–12.21) and Summary (pp. vii-viii).

7.3 The scope of this question includes all aspects of hydrology in upland peatland catchments relating to water quality, distribution and flow, including soil water, watercourses, overland flow and the water table.

7.4 There is some crossover between this sub-question and those concerning carbon (Section 6) and fauna (Section 5). In particular, the fluvial export of DOC and POC has implications for water quality as well as carbon budgets (see Section 6, paras. 6.43–6.48) and aquatic invertebrates can be indicators of water quality (reported in Section 5 on fauna, paras. 5.65–5.67).

Recent studies on the effects of burning on water quality and hydrology

7.5 Twenty-two recent studies (reported in 31 references) since NEER004 provide evidence on the effects of managed burning on water quality and flow.

7.6 Fourteen related specifically to water quality and hydrology. Of the remainder, five focussed primarily on carbon fluxes including DOC and/or POC. A further three studies included the collection of data relevant to water-related outcomes.

7.7 Fourteen studies primarily investigated the effects of managed burning, while one focussed on wildfire, one palaeoecological study included both managed burning and wildfire, and a further study investigated carbon fluxes from laboratory-burned material. Five studies investigated the impact of vegetation composition on water outcomes.

Study type and quality

7.8 A summary of the type and quality of the 22 recent water studies is given in Table 17.

Table 17. Categorisation of recent water evidence studies by study type and quality. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	0	8	0	8
1/2: both type 1 and 2	1	1	0	2
2: quantitative observational or correlative	0	10	2	12
3: qualitative or 4 review	0	0	0	0
Total	1	19	2	22

7.9 Seven studies involved field experiments and eight included field observation and/or monitoring. Heinemeyer and others (2019c [1+, EV-]) included both aspects and are thus included in both. Most of the experiments included were recently established rather than long-term, though Qassim (2015 [1+, EV-]) used the same experimental sites as Worral and others (2012), whose work was evaluated in NEER004.

7.10 Of the field observation/monitoring studies, two used a chronosequence approach to examine change in water outcomes over time after burning. The other observation/ monitoring studies generally made comparisons between areas or time periods with different burning or vegetation characteristics: Heinemeyer and others (2018 [2-, EV-]) used peat cores to examine site histories; Johnston & Robson (2012 [2-, EV+]) made comparisons between catchments; Brown and others (2014 [2+, EV+] reported in Blundell and others, 2013) Parry and others (2015 [2+, EV+]) investigated the influence of slope and vegetation; Turner & Swindles (2012 [2+, EV-]) recorded testate amoebae communities; and Vane and others (2013 [2+, EV-]) traced the transport and fate of water pollutants.

7.11 The studies using a chronosequence approach were those of Brown and others (2014 [2+, EV+]) who compared sites with patches burned 2–10+ years ago with sites where there was no recent history of burning management, and Clay and others (2015 [2+, EV-]) who compared sites with patches burned 1–13 years ago with sites unburned for 13+ years.

7.12 Four studies included laboratory experiments which investigated vegetation impacts on water quality (Dunn and others, 2016 [2-, EV-]; Ritson and others, 2016 [1+, EV+]); the effects of ash deposition (Noble and others, 2017 [1+, EV-]) and heating-induced soil hydrophobicity (Wu and others, 2020 [1+, EV-]).

7.13 Modelling was a main component of six studies (Chapman and others, 2017 [2,4+, EV-]; Gao and others, 2016/2017, both [1+, EV-]; Heinemeyer & Swindles, 2018 [2+, EV-]; Li and others, 2017 [2+, EV+]; Odoni, 2016 [2+, EV-]). These studies mainly used pre-existing and/or remotely sensed data with modelled scenario parameters informed by past empirical work. Heinemeyer & Swindles (2018 [2+, EV-]) additionally used the subfossil archive of testate amoebae to reconstruct past moisture conditions.

Outcome measures

7.14 The 22 recent water studies assessed 11 main outcome measures given in Figure 9. Five of these relate to water quality, with DOC the most frequently reported. The remaining six relate to water distribution and flow within peatlands and/or the watercourses they supply. Most studies reported on multiple outcomes.

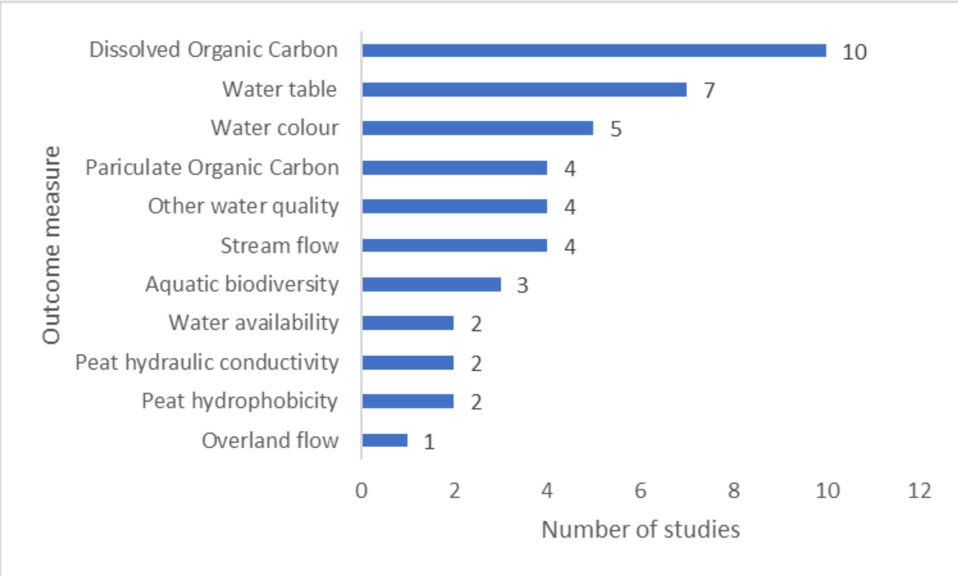

Figure 9 Number of recent studies recording or deriving data on individual water outcome measures.

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

Countries, areas and number of sites

7.15 Twenty-one of the 22 recent water studies were from the UK, with a single laboratory study from Canada. Within the UK most of the studies (19) were based in England, with one of these also including a site in Wales. One study was based entirely in Wales and one in Scotland. As with other sub-questions, research was concentrated in the Pennine region of England (Figure 10).

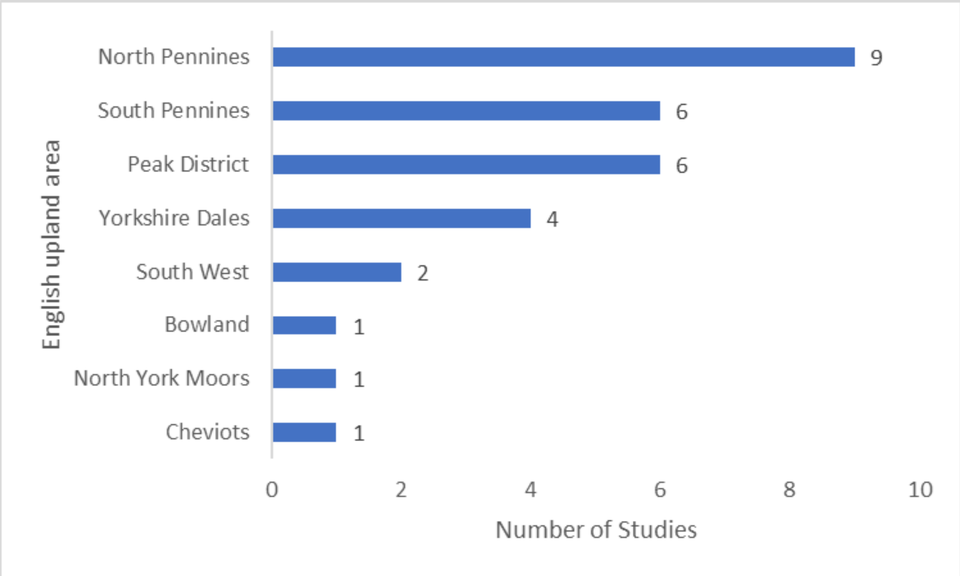

Figure 10. English upland areas covered by recent peatland water studies (n = 18 studies).

7.16 Most of the water studies involved a small number of sites with eleven single site studies and ten studies involving two to four sites. Two studies involved a greater number of sites, with ten used by Brown and others (2014 [2+, EV+]) and 30 used by Johnston & Robson (2012 [2-, EV+]). The way in which sites were used ranged from taking samples or monitoring at plot scale to modelling at whole catchment scale.

Habitats and vegetation types

7.17 Over half of the studies (15, 65%) were carried out specifically on blanket bog habitats. A further five (22%) focussed on ‘moorland’ or ‘upland’ habitat, though all of these were at least partly peatland. One study included both heath and raised bog (Grau-Andrés and others, 2019b [1+, EV-]) and another involved boreal peatlands (Wu and others, 2020 [1+, EV-]).

7.18 Of seven studies where NVC types were described, M19 was represented in six and M20 in four. The study of Brown and others (2014 [2+, EV+]) additionally included M6, H9 and H12 NVC vegetation communities on deep peat.

Types of burn and fire

7.19 Of the 19 studies primarily focussed on burning and/or wildfire, eight (44%) studied ‘routine’ managed burning, two conducted experimental burns, one studied wildfire and a further palaeoecological study considered historical fire including both managed burns and wildfire. Two studies modelled managed burning based on existing data and three simulated specific impacts of fire in laboratory experiments. Two of the studies included experimental cutting treatments in addition to burning.

External validity

7.20 As with other sub-questions, most of the water studies included a relatively small number of sites with limited geographic spread and bias towards the Pennines.

Therefore, most water studies were classed as [EV-] for external validity (Table 18). The remainder were classed as [EV+] as the spread of sites or coverage of modelling provided evidence at a larger (regional or national) scale.

Table 18. Categorisation of recent water studies by study type and EV (external validity). EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	1	8	9
1/2: both type 1 and 2	0	1	1	2
2: quantitative observational or correlative	0	3	8	11
3: qualitative or 4 review	0	0	0	0
Total	0	5	17	22

Summary of recent evidence on the effects of burning on water quality and hydrology

7.21 This section presents a summary and synthesis of the findings of the 17 recent studies on of burning on water qualify and hydrology. It is organised by themes which encompass each of the 11 outcome measures listed in Figure 10.

Water quality - organic carbon

7.22 The findings from ten studies investigating DOC, five investigating water colour and four investigating POC are described in section 6 as they are relevant to carbon balance as well as water quality.

7.23 Of additional relevance for water quality, Ritson and others (2016 [1+, EV+]) investigated water treatment of DOC derived from peat and different peatland vegetation species, the distribution and abundance of which may be affected by managed burning. The results showed that DOC from *Sphagnum* is more labile than that from vascular plants. The proportion of DOC microbially mineralised without treatment was greatest in *Sphagnum*-derived DOC followed by that derived from *Calluna*, then *Juncus effusus*, then *Molinia*, then peat. The ease of DOC removal by chemical treatment was in the order peat > *Sphagnum* and *J. effusus* > *Molinia* > *Calluna*, while the potential of DOC to form chloroform during treatment was in the order *Calluna* > *Sphagnum*, peat and *J. effusus* > *Molinia*.

Water quality - other

7.24 Two studies measured pH in relation to burning. Brown and others (2014 [2+, EV+]) found no difference in soil pH, but observed that stream water pH was generally lower (more acidic) in streams draining burned catchments. Heinemeyer and others (2019c [1+, EV-] reported in Heinemeyer and others, 2023) found no difference in soil pH between recently burned plots and cut and recently unmanaged comparisons, and no difference in the pH of streams draining burned and cut sub-catchments.

7.25 The same two studies measured various water quality attributes in stream water. Brown and others (2014 [2+, EV+]) found that streams draining burned catchments had lower concentrations of calcium and greater silica, manganese, iron and aluminium compared to unburned catchments. Heinemeyer and others (2019c/2023 [1+, EV-]) found three times greater phosphorus and significantly lower lead concentrations in stream water draining cut catchments compared to burned catchments, but no difference in the concentration of other elements measured or in conductivity of stream water.

7.26 Noble and others (2017 [1+, EV-]) found that the addition of heather ash to peat resulted in increased concentrations of Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}, but that these cations were easily leached with rainwater. However, the study did not investigate the fate of the leached cations or implications for stream water quality.

7.27 One study investigated the transport and fate of polycyclic aromatic hydrocarbons from a burned peatland catchment (Vane and others, 2013 [2+, EV-]). Although there was no unburned comparison, the results suggested that managed burning was a significant source of the polycyclic aromatic hydrocarbons found in reservoir sediment.

Peat hydrology

7.28 In total, nine studies included outcomes relating to peat hydrology including water inputs, movement, storage and outputs.

7.29 Two studies reported on hydrophobicity of the peat surface. Wu and others (2020 [1+, EV-]) found that heating temperatures and durations comparable to those of managed burning (250 °C and 300 °C for <5 minutes) induced peat hydrophobicity due to evaporative water loss. Turner & Swindles (2012 [2+, EV-]) did not measure hydrophobicity directly, but inferred increased hydrophobicity after fire at Ilkley Moor in the South Pennines based on the presence of testate amoeba taxa such as Hyalosphenia subflava, which was associated with moss species colonising hydrophobic peat surfaces, along with the presence of standing water suggesting surface pooling after rainfall.

7.30 Two studies reported on hydrophobicity of the peat surface. Wu and others (2020 [1+, EV-]) found that heating temperatures and durations comparable to those of managed burning (250 °C and 300 °C for <5 minutes) induced peat hydrophobicity due to evaporative water loss. Turner & Swindles (2012 [2+, EV-]) did not measure hydrophobicity directly, but inferred increased hydrophobicity after fire at Ilkley Moor in the South Pennines based on the presence of testate amoeba taxa such as Hyalosphenia subflava, which was associated with moss species colonising hydrophobic peat surfaces, along with the presence of standing water suggesting surface pooling after rainfall.

7.31 Seven studies investigated water table depth. Brown and others (2014 [2+, EV+]) found that water tables were deeper in burned catchments and showed evidence of gradual recovery from 2–10+ years after burning. Heinemeyer and others

(2019c/2023 [1+, EV-]) compared cutting and burning treatments up to four years post-treatment and found that cut plots with brash left in-situ had the shallowest water tables (mean -8.9 cm), followed by cut plots with brash removed (-10.3 cm) and plots that were less recently managed (-10.5 cm), then burned plots (-13.2 cm or -10.8 cm when a plot with a peat pipe was excluded), although these differences are small. However, after a further five years of monitoring, water tables on less recently managed plots were on average the deepest when considering the entire post-management period for all sites. This apparent gradual post-burn recovery in water table in burnt plots is consistent with Brown and others (2014 [2+, EV+]). Two other studies observed shallower water tables on plots more recently managed by burning (Qassim, 2015 [1+, EV-]; Grau-Andrés and others, 2019b [1+, EV-]). Two studies showed deeper water tables associated with wildfire (Brown and others, 2014 [2+, EV+] on a burnt EMBER site reported in Blundell and others, 2013) and over historical periods of grouse moor management at Moor House NNR which included burning (Heinemeyer & Swindles, 2018 [2+, EV-]).

7.32 Two studies measured peat bulk density, which can affect water storage and retention (Boelter, 1968), and hence water availability to plants. Brown and others (2014 [2+, EV+]) found that bulk density (at four depth increments, 0–5, 5–10, 10–15 and 15–20 cm) was much greater on plots burned two years ago (mean 0.249 g cm-3) than plots from unburned catchments (0.124 g cm-3), while plots burned four and 15+ years ago had intermediate bulk densities (0.166 and 0.136 g cm-3 respectively) indicating recovery over time. Heinemeyer and others (2019c [1+, EV-]) found no difference in bulk density (measured in depth increments to 200+ cm) between burned, cut and less recently managed treatment plots. However, peat cores from the same study showed a positive correlation between bulk density and charcoal which was considered a proxy for historical fire incidence.

7.33 Brown and others (2014 [2+, EV+]) reported on some additional aspects of peat hydrology including infiltration, hydraulic conductivity and macropore flow. The study found that steady state infiltration rates and hydraulic conductivity were both lower on plots recently subjected to managed burning (two or four years prior) or wildfire, compared to less recently burned (15+ years) and unburned plots. Blundell and others (2013) also reported reduced hydraulic conductivity after wildfire on one of the same EMBER study sites as part of the same study. The proportion of flow moving through macropores was also lower for recently burned plots compared to less recently burned and not recently burned plots. Suggested mechanisms for these observations included the collapse of peat structure due to drying and/or blocking of macropores by ash and fine sediment mobilised by burning.

7.34 One study (Brown and others, 2014 [2+, EV+]) found that overland flow was more frequent at burned sites. This study also observed a relationship between rainfall and overland flow incidence on both unburned plots and plots burned over 15 years previously. In contrast, plots burned less than 15 years previously did not exhibit this relationship, suggesting a modification of hydrological functioning after burning.

Stream hydrology

7.35 Four studies provided evidence that stream flow is affected by burning and vegetation composition. Heinemeyer and others (2019c/2023 [1+, EV-]) found that flow volumes were greater from burned than cut catchments at two of three study sites. This study also modelled downstream river levels and found that they were significantly higher when a burning scenario was compared to a cutting scenario for grouse moors in the catchment of the River Ouse (Yorkshire). Other modelling work has also suggested that burning has the potential to increase flow peaks (Gao and others, 2017 [1+, EV-]). Gao and others (2016 [1+, EV-]) found that denser vegetation results in a lower and more delayed flow peak with buffers around headwater streams the most effective location for dense vegetation. Finally, Brown and others (2014 [2+, EV+]) observed an overall greater lag time from peak-rainfall to peak-discharge, and significantly longer recession time for storm hydrographs, for burned catchments compared to unburned catchments. Specifically, burning was associated with longer lag times in response to smaller rainfall events and more intense hydrographs in response to larger storm events.

8. Recent evidence on the effects of burn severity, frequency, scale, location and other characteristics on upland peatlands

8.1 The slightly revised full text of this sub-question is:

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

Introduction

8.2 An introduction and explanation of terminology relating to aspects of the fire regime is given in NEER004 (para. 8.2). Fire severity describes the immediate effects of fire on vegetation, litter and/or soils. These effects may be quantified based on post-fire appearance, biomass loss, depth of burn or other characteristics. As such, severity is closely linked to impacts on vegetation (Section 4) and carbon loss via combustion (Section 6), and also overlaps with the extent, frequency and especially the type of burning (Section 11).

Recent studies on the effects of burn characteristics on biodiversity, carbon and water

8.3 Eight recent studies (reported in 12 references) since NEER004 provide evidence on the effects of burn characteristics on biodiversity, carbon and water. All have been evaluated in relation to sub-questions 1 to 4 (Sections 4 to 7), so this section aims to draw out effects relating specifically to the characteristics of burns.

8.4 Six studies related primarily to aspects of fire severity or factors affecting fire severity, with two of these also investigating the effect of location (habitat or position within site). The remaining two studies related primarily to the frequency of burning, though this may also impact fire severity due to differences in the aboveground biomass accumulated in the time between burns.

Study type and quality

8.5 A summary of the type and quality of the eight recent burn characteristics studies is given in Table 19.

Table 19. Categorisation of recent burn characteristic studies by study type and quality. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	1	3	1	5
1/2: both type 1 and 2	1	0	0	1
2: quantitative observational or correlative	0	2	0	2
3: qualitative or 4 review	0	0	0	0
Total	2	6	0	8

8.6 Five studies involved field experiments. Of these, three used the existing Hard Hill burning and grazing experiment: Milligan and others (2018 [1++, EV-]) presented the latest of a series of approximately ten-year routine vegetation surveys (para. 4.38); while Noble and others (2019a [1,2++, EV+]) observed fire temperatures during burning of the experimental plots and subsequent effects on *Sphagnum*, with additional laboratory work; and Marrs and others (2019 [1,2+, EV-]) measured the apparent rate of carbon accumulation in peat cores from the experimental plots. The two other related field experiment studies featured experimental burns on sites in Scotland with moss/litter layer removal (Grau-Andrés and others, 2017a [1+, EV-]) and drought simulation treatments (Grau-Andrés and others, 2019b [1+, EV-]).

8.7 One experimental study burned plant matter in a laboratory at different severities, before exposing it to terrestrial and aquatic field conditions in the Peak District (Kennedy-Blundell, 2020 [1+, EV-]). Another study (Davies and others, 2016a [2+, EV+] +]) took field measurements after wildfire and made comparisons with managed burns.

Outcome measures

8.8 The burn characteristic studies mainly reported outcomes relating to vegetation (five studies), carbon (four studies) and water (one study). Additionally, two studies considered fire severity and two fire temperatures as outcomes potentially influenced by pre-fire conditions.

Applicability of the evidence on the effects of effects of burn characteristics on biodiversity, carbon and water

Countries, areas and number of sites

8.9 Seven of the eight recent burn characteristics studies were from the UK and one was from Canada. Within the UK, there were four studies based in England, two in Scotland, and one which used sites in both.

8.10 Of the four studies based in England, three were conducted at the Hard Hill experiment in the North Pennines and one in the Peak District. Davies and others (2016a [2+, EV+]) used wildfire sites in the South and West Pennines as well as wildfire and managed burn sites in Scotland.

8.11 Six of the studies used single sites with only Davies and others (2016a [2+, EV+]) and Grau-Andrés and others (2017a [1+, EV-]) using more (seven and two sites respectively).

Habitats and vegetation types

8.12 Most studies were carried out on blanket bog habitat with the exceptions of heathland at some sites in Davies and others (2016a [2+, EV+]), heathland with wet flushes on peaty podzols in Grau-Andrés and others (2017a [1+, EV-]) and heathland on peaty podzols and raised bog in Grau-Andrés and others (2019b [1+, EV-]).

8.13 All four studies where NVC community type was given included sites with M19 vegetation, with the addition of H12 and M25a in Davies and others (2016a [2+, EV+]).

Types of burn and fire

8.14 Five studies used experimental burns in the field, two conducted burning or heating in the laboratory and one took measurements following wildfire.

External validity

8.15 Small numbers of sites resulted in most of the studies being categorised as [EV-] for external validity (Table 20).

Table 20. Categorisation of recent burn characteristics studies by study type and EV (external validity). EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	0	4	4
1/2: both type 1 and 2	0	1	1	2
2: quantitative observational or correlative	0	1	1	2
3: qualitative or 4 review	0	0	0	0
Total	0	2	6	8

Summary of recent evidence on the effects of burn characteristics on biodiversity, carbon and water

8.16 This section presents a summary and synthesis of the findings of the eight recent studies of burn characteristics on vegetation, carbon and water.

Vegetation

8.17 Four studies investigated the impact of burn characteristics on outcomes relating to vegetation. This included two linked experiments where burn severity was manipulated (Grau-Andrés and others, 2017a, 2019b [both 1+, EV-]). The remaining two studies used the Hard Hill experiment and focussed on the effects of burn frequency (Milligan and others, 2018 [1++, EV-]) and burn temperature (Noble and others, 2019a [1,2++, EV+]).

8.18 Grau-Andrés and others (2017a [1+, EV-]) simulated higher fire severity by removing the moss/litter layer in some plots before burning. When compared to burned plots with an intact moss/litter layer, these plots differed in community composition and had a greater frequency of *Calluna* resprouting 2.5 years (three growing seasons) after burning. Additional treatments where the moss/litter layer was removed either after burning or alongside cutting showed similarities in vegetation community to where it was removed before burning, suggesting that moss removal and exposure of peat substrate are important mechanisms for impacts on vegetation.

8.19 A second linked experiment by Grau-Andrés and others (2019b [1+, EV-]) used a drought treatment to increase fire severity. Plant community composition (both species frequency and plant functional type) measured in the first two years after burning differed between high and low severity burns, though these were more similar to each other than to unburned plots. Generally, dwarf shrubs, graminoids and acrocarpous mosses, including *C. introflexus,* were more associated with higher fire severity, whilst pleurocarpous mosses were associated with lower fire severity. The two study sites showed some differences in vegetation response including that of bell heather *Erica cinerea*, which was associated with lower fire severity at the raised bog site and higher severity at the heathland site. Beta diversity was greater after high severity burning at the heathland site but not at the raised bog.

8.20 Grau-Andrés and others (2019b [1+, EV-]) also measured *Sphagnum* abundance and photosynthetic capacity in experimental plots at the raised bog site. Low and high severity burns had similar *Sphagnum* cover to unburned plots 10–24 months after fire. High severity burns lowered *Sphagnum* photosynthetic capacity 1–8 months after fire to a greater extent than low severity burns, but this had recovered in both treatments within two years.

8.21 Milligan and others (2018 [1++, EV-]) reported the most recent vegetation survey results from the Hard Hill experiment. These are summarised in the vegetation section (Section 4). Differences between the short- and long-rotation plots (burned at 10- and 20-year intervals respectively) suggest impacts of burn frequency on vegetation. These differences included increased species diversity and richness of vascular plants in short-rotation plots along with increased abundance of *Eriophorum vaginatum*, *Campylopus paradoxus*, liverworts and *Sphagnum* (a comprehensive survey of *Sphagnum* in the experiment including reference plots was also reported in Noble and others, 2018a [1,2++, EV-] – see Section 4). The height profile of short-rotation burn plots also differed from long-rotation burn plots, with more vegetation in lower height strata and less over 20 cm.

8.22 One study investigated how the temperature at the moss surface during fire affected *Sphagnum* by measuring cell damage (Noble and others, 2019a [1,2++, EV+]). Higher fire temperatures in the field tended to lead to greater *S. capillifolium* cell damage (measured ten and 21 weeks after fire), and the same pattern was observed immediately after heating in the laboratory for five *Sphagnum* species (*S. capillifolium*, *S. papillosum*, *S. medium*, *S. austinii* and *S. angustifolium*).

Carbon

8.23 Four studies investigated the influence of burn characteristics on aspects of the carbon cycle. Of these, three studies used experimental manipulation of fire severity. The moss/litter layer removal treatment used by Grau-Andrés and others (2017a [1+, EV-]) led to increased soil heating during burning, as well as an altered post-fire thermal regime with greater diurnal and annual temperature ranges, including higher summer temperatures at the soil surface, compared to plots where the moss/litter layer was left intact. Post-fire temperatures were used to estimate relative respiration, and estimated rates were highest in plots where the moss/litter layer had been removed, particularly in the warm summer months.

8.24 The drought treatment used by Grau-Andrés and others (2019b [1+, EV-]) led to greater soil heating and moss/litter layer consumption during burning at both a dry heath and a raised bog site. For outcomes relating to carbon (CO_2, CH_4 and DOC fluxes), there was little difference between drought and non-drought treatments in the two years following fire. The only significant difference was a greater rate of ecosystem respiration in drought plots during autumn at the dry heath site. However, this did not lead to a difference in net ecosystem exchange of CO_2 between treatments over the study period.

8.25 Kennedy-Blundell (2020 [1+, EV-]) found that the severity of laboratory burning affected the characteristics of the resulting burned plant material (referred to as pyrogenic carbon or PyC), including lability. This subsequently had impacts on carbon fluxes when the material was exposed to field terrestrial or aquatic conditions (see carbon Section 6), with less combusted material from low-severity burns releasing more carbon in the early weeks of exposure.

8.26 At the Hard Hill burning and grazing experiment, Marrs and others (2019 [1,2+, EV-]) found that burning more frequently led to a reduced apparent rate of carbon accumulation over a 53-year period (para 6.50).

Water

8.27 Two studies measured peat moisture content but found no significant differences between severity treatments (Grau-Andrés and others, 2019b [1+, EV-]) or correlation with fire temperature (Noble and others 2019a [1,2++, EV+]). Similarly, Noble and others (2019a [1,2++, EV+]) found no correlation between fire temperature and peat bulk density after burning.

Vegetation effects on fire severity and temperature

8.28 Three studies provided evidence that vegetation characteristics influence fire severity and temperature. Davies and others (2016a [2+, EV+]) found that variation in severity both within and between wildfires was influenced by vegetation type, structure and moisture, and concluded that blanket bog may be less at risk of severe wildfire than heath habitats due to differences in vegetation moisture content. Grau-Andrés and others (2019b [1+, EV-]) found that an experimental drought treatment decreased moss/litter layer moisture content at both heathland and raised bog sites but only affected soil moisture content at the heathland site. These changes led to greater soil heating and fire severity (measured by moss/litter consumption) at both sites, with a greater increase at the heathland site. Finally, Noble and others (2019a [1,2++, EV+]) found that dwarf shrub cover was a better predictor of temperature at the moss surface during fire than graminoid cover or dwarf shrub height.

9. Recent evidence on the interaction between managed burning and grazing on upland peatlands

9.1 The full text of this sub-question is:

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

Introduction

9.2 The context of research on the extent, frequency, practice and type of burning is described in NEER004 (paras. 9.2–9.3, p.41). NEER004 also assessed relevant studies (in Appendix 9 pp. 146–149), with a summary and brief interpretation across studies provided as evidence statements (paras. 9.4–9.18) and research recommendations in Section 9 (para. 9.19) and across sub-questions in Section 12 (paras. 12.35–12.36). The evidence statements also feature in summary form in the Conclusions (Section 12, para. 12.27) and Summary (p. ix).

Recent studies on the interaction of managed burning and grazing

9.3 Two recent studies (reported in two references) since NEER004 provide evidence on the interaction of managed burning and grazing.

Study type and quality

9.4 A summary of the type and quality of the two recent studies relating to the interaction of managed burning and grazing is given in Table 21. Both studies collected data from the same long-term burning and grazing experiment at Hard Hill, Moor House NNR.

Table 21. Categorisation of recent burning and grazing interaction studies by study type and quality. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	1	0	0	1
1/2: both type 1 and 2	1	0	0	1
2: quantitative observational or correlative	0	0	0	0
3: qualitative or 4 review	0	0	0	0
Total	2	0	0	2

Outcome measures

9.5 Both studies investigated the effect of interaction between burning and grazing on vegetation related outcomes. Milligan and others (2018 [1++, EV-]) reported on vegetation community composition, properties and plant species abundance from a long-running series of vegetation surveys, whilst Noble and others (2018a [1,2++, EV-]) reported on *Sphagnum* patch size, frequency and species measured in a one-off survey.

9.6 No studies investigated the effect of the interaction between grazing and burning on outcomes specifically relating to carbon or water, although some vegetation outcomes may subsequently influence these.

Applicability of the evidence on the interaction of managed burning and grazing

Countries, areas, habitats and vegetation types

9.7 Both studies were from the same experiment at a site in the North Pennines, England. The habitat at the experimental site is blanket bog dominated by *Calluna*, *Eriophorum vaginatum* and *Sphagnum*, which conforms to NVC blanket mire type M19. Grazing at this site is mainly by sheep and is currently at a very low intensity, having declined through time and thus is unlikely to be representative of the range of grazing intensities and stock types found across UK upland peatland sites.

External validity

9.8 Both studies reporting on the interaction of managed burning and grazing were based at a single site and therefore classed as [EV-].

Summary of recent evidence on the interaction of managed burning and grazing

9.9 Milligan and others (2018 [1++, EV-]) reported some interaction between burning and grazing treatments at the Hard Hill experiment, at which vegetation was monitored over time from 1972 (18 years after the start of the experiment) to 2013. For vegetation community composition, 10- and 20-year rotation burning treatments had a slightly different trajectory of change over time according to grazing status. Specifically, ungrazed plots had a slightly enhanced trajectory towards a greater abundance of *Eriophorum* compared to grazed plots. This effect of grazing was not apparent in the no-burn since 1954 plots.

9.10 This study also reported that burning and grazing interacted to affect vegetation community properties, with an increase in vascular plant abundance in ungrazed 10-year burning rotation plots compared to the grazed equivalent, the opposite effect of grazing in no-burn since 1954 plots, and no effect of grazing in 20-year rotation plots.

9.11 Grazing and burning also had interactive effects on the abundance of some plant species, including vascular plants (*Calluna*, *Empetrum nigrum*), mosses (*Sphagnum* spp., *Hynum jutlandicum*, *Campylopus paradoxus, Pohlia nutans*), liverworts (*Calypogeia muelleriana*, *C. bicuspidata, Lophozia ventricosa)* and lichens, as detailed below.

9.12 At the start of the monitoring period, *Calluna* abundance was lowest in the 10-year rotation, grazed plots and there was a greater difference between grazing treatments in this burning treatment, though abundance also increased at a greater rate in these plots over time. *Empetrum nigrum* was initially more abundant in ungrazed plots for 10- and 20- year burn rotations, though over time abundance decreased in both the grazed and ungrazed 20-year burn rotation plots and the ungrazed 10-year rotation plots but increased slightly in the grazed 10-year rotation plots. In the no-burn since 1954 plots, *E. nigrum* decreased with grazing and increased under no grazing.

9.13 *Sphagnum* had greater abundance in ungrazed plots burned on 10- and 20-year rotations but not in the no-burn treatment, though it increased more in the grazed 10- and 20-year rotation plots over time. *Hypnum jutlandicum* increased greatly over time in the no-burn since 1954 plots and this increase was enhanced by lack of grazing. *Calypogeia paradoxus* increased over time in the grazed 10- and 20-year burn plots, with the greatest disturbance (grazing and 10-year burning rotation) resulting in the highest abundance, but it declined over time in all other treatments. *Pohlia nutans* had a high initial abundance in ungrazed 10-year rotation plots but subsequently declined rapidly to reach a similar level to other treatments.

9.14 For all three liverwort species, removal of grazing tended to induce or enhance a decline over time for 10- and 20-year burn rotations, whereas declines were similar for both grazing treatments in no-burn since 1954 plots. *Calypogeia bicuspidata* and *Lophozia ventricosa* declined over time in all treatments, but *C. muelleriana* increased over time in the grazed burned plots. Finally, grazing removal slowed decline of lichens to a greater extent with more frequent burning rotations.

9.15 These changes in species abundance translated into some changes in abundance-weighted Ellenberg values (para. 4.42). Whereas there was no effect of grazing treatment in the no-burn since 1954 plots, in the 10- and 20-year rotation plots grazing treatment modified the rate of change for some Ellenberg values, particularly for soil fertility in the 10-year rotation plots where grazing resulted in a decrease over time compared with an increase where grazing was excluded.

9.16 Noble and others (2018a [1,2++, EV-]) reported on a detailed survey of *Sphagnum* on the Hard Hill experimental plots but found no effect of the grazing treatments on any outcome measures, nor any interaction between grazing and burning.

10. Recent evidence on the relationship between managed burning of upland peatlands and wildfire

10.1 The full, slightly revised text of this sub-question is:

Is there a relationship between managed burning of upland peatlands and wildfire risk, hazard, occurrence, severity, extent and habitat resilience?

Introduction

10.2 The context to research on the relationship between managed burning and wildfire is described in NEER004 (paras. 10.2–10.5).

10.3 NEER004 also describes relevant studies and their findings up to 2012, which provide evidence on the relationship between managed burning of upland peatlands and wildfire in Appendix 10 (pp. 150–151). In NEER004 a summary, synthesis and brief interpretation is given across wildfire-related studies, including as evidence statements in Section 10 (paras. 10.6–10.13) and research recommendations (p. 47) including across sub-questions in Section 12 (paras. 12.35–12.36). The evidence statements were also given in summary form in the Conclusions (para. 12.28) and Summary (p. ix).

10.4 Evidence on the relationship between managed burning of upland peatlands and wildfire was also included as part of a more recent, wider Natural England evidence review on the causes and prevention of wildfire on heathlands and peatlands in England (NEER014), which included and updated the findings from NEER004. The sections of NEER014 that relate to the relationship between managed burning of upland peatlands and wildfire (rather than wider wildfire issues) are especially Sections 4 (Wildfire risk and occurrence, pp.14–28), 5 (Wildfire ignition sources, pp.29–38), 6 (Wildfire behaviour and severity, paras. 6.7, 6.9, 6.12), 8 (Reducing the impact of wildfires, paras. 8.4–8.5, 8.9–8.11) and 9 (Summary, paras. 9.10, 9.20, 9.24–25, 9.35–39, 9.50, 9.63–64, 9.66–68, 9.78). It should be noted that NEER014 covers all evidence up to 2020 and relates not just to upland peatlands, but also to other upland and lowland semi-natural habitats and, to a more limited extent, other land uses.

Recent studies on the relationship between managed burning and wildfire

10.5 Eight recent studies since NEER014 provide additional evidence on the relationship between managed burning and wildfire.

Study type and quality

10.6 A summary of the type and quality of the eight recent studies relating to the relationship between managed burning and wildfire is given in Table 22. Most studies were type 2

Table 22. Categorisation of recent studies on the relationship between manged burning and wildfire by type and quality of evidence. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	0	0	0	0
2: quantitative observational or correlative	0	7	0	7
3: qualitative or 4 review	0	0	1	1
Total	0	7	1	8

10.7 Three studies were based on analysis of satellite-derived wildfire burn data at large scale (Perry and others, 2022 [2+, EV++]; Cardíl and others, 2023, [2+, EV+]; Kirkland and others, 2023 [2+, EV-]). One study analysed large, empirical wildfire site data sets (Wilkinson and others, 2023 [2+, EV+]) which included seasonality. Other single studies reported on ignition cause (Cosgrove, 2004 [3-, EV-], which was missed in NEERs 004 and 014 but was reported in Holland and others, 2022), analysis of (Scottish) Fire and Rescue Service wildfire incident data (Gagkas and others, 2022 [2+, EV++]), coincidence with weather conditions in Norway (Log and others, 2017 [2+, EV-]), and related bare ground cover resulting from a wildfire to time since last managed burning (Swindell, 2017 [2+, EV-]).

Applicability of the evidence on the relationship between managed burning and wildfire

Countries, areas and number of sites

10.8 Five of the studies covered large geographic areas or regions, in two cases inter-continental zones, and the remaining three studies covered single case study sites/areas.

10.9 Four studies were from the UK. One (Perry and others, 2022 [2+, EV++]) covered the whole UK, two were in Scotland (Gagkas and others, 2022 [2+, EV++] and Cosgrove, 2004 [2-, EV-]), and one in England (Swindell, 2017 [2+, EV-]). Another study covering north-west Europe included the UK (Cardíl and others, 2023, [2+, EV+]).

10.10 The three remaining non-UK studies covered northern (non-permafrost boreal and temperate) peatlands (Wilkinson and others, 2023 [2+, EV+]), the Polesia region of northern Ukraine and southern Belarus (Kirkland and others, 2023 [2+, EV-]) and a single site in Norway (Log and others, 2017 [2+, EV]).

Habitats and vegetation types

10.11 Though most studies covered wide geographic areas, most related specifically to peatlands. However, three that included the UK included wider habitats and land uses affected by wildfires (Gagkas and others, 2022 [2+, EV++]; Perry and others, 2022 [2+, EV++]; Cardíl and others, 2023, [2+, EV+]), as did NEER014.

External validity

10.12 There was a range of categorisations of external validity (Table 23. Categorisation of recent studies on the relationship between manged burning and wildfire by study type and EV (external validity).) reflecting the fact that, although many of the studies covered large geographical ranges, some covered single sites and three were from outside the UK. All but one of the studies were classed as type 2, with quality spread across the three categories.

Table 23. Categorisation of recent studies on the relationship between manged burning and wildfire by study type and EV (external validity). EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	0	0	0
2: quantitative observational or correlative	3	2	2	7
3: qualitative or 4 review	0	0	1	1
Total	3	2	3	8

Summary of recent evidence on the relationship between managed burning and wildfire

Escaped managed burns as a cause of wildfire occurrence

10.13 Cosgrove (2004 [2-, EV-]) reported on an assessment of the source of wildfire ignition by Badenoch and Strathspey Fire Protection Group, which found that “the supposed cause of 13 out of 14 (93%) wildfires tackled in Badenoch and Strathspey in the Cairngorms, Scotland, in 2003, was human action. [Of these,] 29% were started by ‘muirburns’ that got out of control, affecting 1,135 ha of habitat.”

Relationship between time since managed burning and subsequent wildfire severity

10.14 Swindell (2017 [2+, EV-]) reported significantly higher cover of bare ground following a wildfire at a Peak District blanket bog site in younger *Calluna* stands (i.e., more recently burned prior to the wildfire), with a mean of 78% post-wildfire bare ground cover across 0–6, 7–15 and 16–29 years post-burn classes compared with 35% in 30–40 years and 26% in >40 years post-managed-burning. This may be related to vegetation composition and/or wetness, as *Sphagnum* cover was highest in longer unburned stands (para. 4.49).

Seasonality of wildfires in relation to the burning season

10.15 Perry and others (2022 [2+, EV++]) and Gagkas and others (2022 [2+, EV++]) covering the UK and Scotland, respectively, and Cardíl and others (2023 [2++, EV++]) covering north-west Europe including the UK, all identified a peak in wildfire occurrence in spring, especially in March and April and into May in Scotland. This spring period overlaps in part with the ‘burning season’ (up to 15 April in England) within which management of vegetation by burning is permitted. Neither Perry and others (2022 [2+, EV++]) or Gagkas and others (2022 [2+, EV++]) found a trend in wildfire occurrence over the period of data collection.

10.16 Perry and others (2022 [2+, EV++]) and Cardíl and others (2023 [2+, EV++]) also identified a secondary peak of fire activity in summer. Cardíl and others (2023 [2+, EV++]) also reported that the spring fire season identified in northwest Europe (including the UK) is clearly different to the Mediterranean fire season, which experiences a minor peak in spring but a stronger peak towards the months of July to September. This summer peak in the Mediterranean fire season coincides with the period of lowest fuel moisture. Typically, the temperate climate of northwest Europe is wetter and more humid in the summer and the period of lowest fuel moisture tends to fall before the phenological ‘green-up’ period in late spring (Cardíl and others, 2023 [2+, EV++]), which may also coincide with cold, easterly drying winds (Log and others, 2017 [2+, EV-]; also see NEER014, paras. 4.18e, 9.10, 9.15). Thus, it is likely that local phenology and weather play a role in the timing of peak fire activity in northwest Europe, particularly in spring. The overlap of part of this period with the burning season, and other potential accidental and deliberate causes of fires in spring, potentially pose a wildfire risk when already low moisture content coincides with drying weather conditions. This is addressed in part by potential closure of open access land when extreme fire weather conditions are predicted by the Met Office Fire Severity Index (MOFSI, Met Office, 2003; also see NEER014, paras. 7.10–7.13). There is, however, no evidence of the effectiveness of such closures which are relatively infrequent (NEER014, para. 9.10).

Habitat resilience in relation to water table and habitat restoration

10.17 Kirkland and others (2023 [2+, EV-]) investigating fires in eastern Europe, where open peatlands covered 4,241 km^2 and fen and transition mires covered 2,598 km^2, found evidence that landscape scale fires disproportionately affected high conservation value peatlands. The lower moisture levels, indicative of drainage, disturbance and/or degradation, meant that open peatlands, meadows and deciduous forests were far more likely to burn than pristine peat habitats. The low occurrence of large fires in healthy, wet peatlands indicates that peatland restoration is crucial for protecting these threatened, carbon-storing ecosystems and promoting landscape resilience in relation to fire events.

10.18 Wilkinson and others (2023 [2+, EV+]), who undertook a synthesis of empirical datasets from a range of natural, degraded and restored northern peatlands (in non-permafrost boreal and temperate regions), found that wildfire reduced carbon uptake in pristine peatlands by 35% and further enhanced emissions from degraded peatlands by 10%. They also found that restoration of peatlands before a fire event mitigated extensive carbon release, but that restored peatlands remained a small source of carbon. Modelling the scenario of all degraded peatlands being restored resulted in an estimated increase in the carbon sink by almost 90% by 2100 compared to the current scenario based upon the existing condition of the peatlands.

10.19 There is evidence from Kirkland and others (2023 [2+, EV-]), Wilkinson and others (2023 [2+, EV+]) and Granath and others (2016 [2+, EV+]) that drained and degraded peatlands are more susceptible to damage and carbon loss following wildfire than pristine or restored peatlands.

11. Recent evidence on the extent, frequency and type of managed burning on upland peatlands

11.1 The full, slightly revised, text of this sub-question is:

What is the extent, frequency, practice and type of managed burning on upland peatlands (including in relation to designated sites and peatland)

Introduction

11.2 The context of research on the extent, frequency and type of burning is described in NEER004 (paras. 11.2–11.4).

11.3 NEER004 also describes relevant studies and their findings up to 2012, which provide evidence on the extent, frequency and type of burning in Appendix 11 (pp. 152–155). In NEER004 a summary and brief interpretation across studies, including the evidence statements, in Section 6 (paras. 11.5–11.14) and research recommendations in Section 6 (p. 50) and across sub-questions in Section 12 (paras. 12.35–12.36). The evidence statements also feature in summary form in the Conclusions (Section 12, paras. 12.29–12.34) and Summary (pp. ix–x) of NEER004.

Recent studies on the extent, frequency and type of burning

11.4 Twelve recent studies (reported in 14 references) since NEER004 provide evidence on the extent, frequency and type of burning.

11.5 Eleven of these had objectives related directly to the extent, frequency and type of upland burning used aerial photographs (3) and/or satellite (8) imagery to collect data on burning. The other (Critchley and others, 2016 [2++, EV++]) reported data on recent burning collected as part of a wider moorland habitat monitoring exercise.

11.6 Some studies, especially those using satellite imagery, were generally not able to separate burning from cutting and hence reported on the combined area of management, though burning was generally thought to predominate. One study (Shewring and others, 2024 [2+, EV++]) did separate out burning and cutting, although the method was only partially successful, with 28% of a sample of known cuts incorrectly mapped as burns. This suggests that while the reduction in photosynthesising vegetation following burning or cutting is mostly distinguishable, the spectral signature in some cuts and burns is similar. The extent to which studies were able to differentiate between managed burning and wildfire also varied depending on methods and was not always reported.

Study type and quality

11.7 A summary of the type and quality of the 12 recent studies relating to extent, frequency or type of burning is given in Table 24. All studies were observational, with the majority employing earth observation techniques over extensive areas.

Table 24. Categorisation of recent studies on the extent, frequency and type of burning by study type and quality of evidence. Quality categories denote low (++), moderate (+) or high (-) risk of bias.

Study type	Quality ++	Quality +	Quality -	Total
1: quantitative experimental	0	0	0	0
2: quantitative observational or correlative	4	8	0	12
3: qualitative or 4 review	0	0	0	0
Total	4	8	0	12

Outcome measures

11.8 The 12 studies on burning extent, frequency or type studies assessed six main outcome measures shown in Figure 11. Most related to extent and change in this over time.

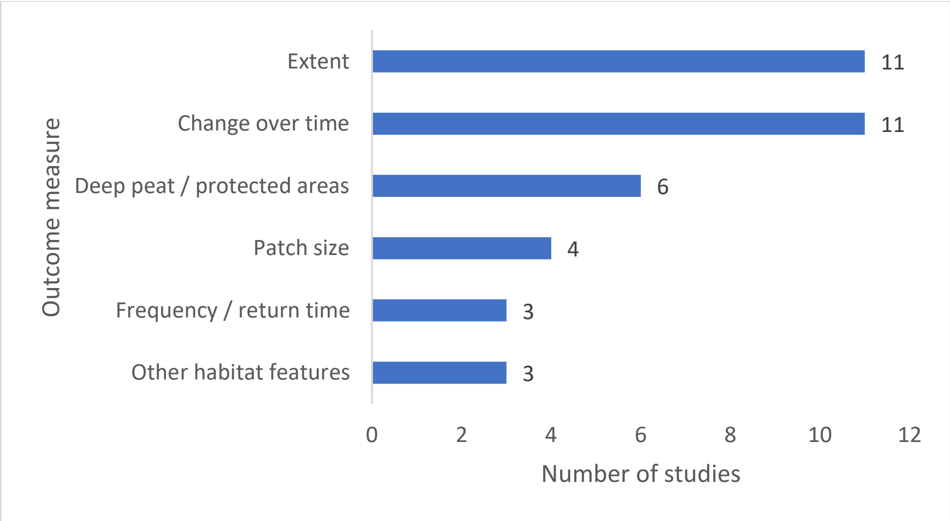

Figure 11. Number of recent studies recording or deriving data on individual outcome measures relating to extent, frequency and type of burning.

Applicability of the evidence on the extent, frequency and type of burning

Countries, areas, habitats and vegetation types

11.9 All 12 of the recent studies on burning extent, frequency or type were from the UK. Of these, eight were England-based, two were based in Scotland, and two covered England, Scotland and Wales.

11.10 Eight studies (seven in or including England) were based in multiple regions/areas chosen variously to represent upland areas, heather-dominated or 'moorland' habitats, or areas with a history of managed burning. The moorland landscapes in these studies generally include both deep and shallow peat deposits, with habitat types including blanket bog and upland heath. One study was based on a sample of 59 1-km squares in the North York Moors and the remaining two studies were based on single sites in the Peak District, both of which supported blanket bog habitat.

External validity

11.11 Many of the recent studies on the extent, frequency and type of burning covered multiple upland regions and were considered representative and generalisable nationally. Therefore, eight studies (67%) were classed as [EV++] for external validity (Table 25).

Table 25. Categorisation of recent studies on the extent, frequency and type of burning by study type and EV (external validity). EV categories denote studies nationally representative (++), regionally representative (+) or less representative (-) of UK upland peatlands.

Study type	EV ++	EV +	EV -	Total
1: quantitative experimental	0	0	0	0
2: quantitative observational or correlative	8	2	2	12
3: qualitative or 4 review	0	0	0	0
Total	8	2	2	12

Summary of recent evidence on the extent, frequency and type of burning

11.12 This section presents a summary and synthesis of the findings of the 12 recent studies on the extent, frequency and type of burning. It is organised by themes which encompass each of the six outcome measures listed in Figure 11.

Burning extent

11.13 Eleven studies reported on the spatial extent of managed burning. The results of these studies were generally expressed as a proportion of the study area burnt, with values ranging from 0.1 to 29% burned annually, though most studies also recorded the total area burnt (Table 26). There was considerable variation in the area burnt per year probably reflecting both variations in suitability of weather conditions and temporal changes in intensity of burning. The latter potentially reflected a response to the new Heather and Grass etc. Burning (England) Regulations 2021 which came into force on 1 May 2021, i.e., from the 2021/22 burning season (Appendix 5). However, due to differences in the methods used to define study areas and identify and classify burning, together with differences in the actual study regions/areas included, proportions and burn areas are not necessarily directly comparable between studies.

Table 26. Summary of results for burning extent.

Study	Study area (period)	Burning extent (area)
Allen and others, 2016 [2+, EV-]	Howden Moor, Peak District (1988–2009)	20% of total area, 29% of 'potentially burnable (PB) area' over 22 years excepting repeat-burned area (para. 11.23) (0.5–1.6% of total area or 0.7–2.4% of PB area year^{-1}).
Blundell and others, 2021 [2+, EV++]	N York Moors, Bowland, N Pennines, Yorks Dales/S Pennines/Peak District (2016/17–2019/20)	0.1–5.2% managed (burned/cut) year^{-1}.
Douglas and others, 2015 [2++, EV++]	Various GB upland regions/ areas and SAC/SPAs (2001–2010)	SAC/SPAs had 0.1–11.5% cover of detectable burns (< c.25 years old).
Drewitt, 2015/ Yallop & Thacker, 2015 [2+, EV+]	59 1-km squares within North York Moors SAC (1995–2009)	Mean 25 ha burns and 32 ha mature heather in each 1-km square (excluding 200 m edge buffer).
Lees and others, 2021 [2+, EV++]	North York Moors, Peak District, Yorkshire Dales and North Pennines (2015/16–2019/20)	0.2–6.9% managed (burned/cut) year^{-1} depending on region and year.
Matthews and others, 2020 [2+, EV++]	Rough grazing within grouse moor holdings in Scotland (2005–2018)	19% of grouse moor holdings and 38% of land within 500 m of grouse butts had detectable burns. Total area with evidence of burning of rough grazing for these holdings was c.163,000 ha (c.20%).
Natural England, 2021 [2+, EV++]	Heather-dominated area for England Moorland Change Map c.2,000 km^2, across 10 upland areas (2017/18–2021/22)	2.6–6.6% managed (burned/cut) year^{-1}. Total area burned over deep peat (>40 cm) range 2,244 (2021/22) to 7,261 (2018/19) ha year^{-1}.
Shewring and others, 2024 [2++, EV++]	GB uplands, including protected areas (PAs), over 5 burn seasons (2017/18–2021/22)	Burning (>1 ha) recorded in 14% of PAs and, within these, the percentage area of moorland burned varied from 2 to 31%. Total GB area burnt ranged from 8,333–20,974 ha (mean 15,250 ha year^{-1}); on deep peat (>40 cm) in England 5,022–766 ha (mean 3,814 ha year^{-1}).
Spracklen & Spracklen, 2023 [2+, EV+]	Eastern Scotland, Grampian Mountains and Southern Uplands over 38 years (1985–2022)	Mean 0.7% (4,670 ha) of the Grampians area burnt year^{-1} and 1.5% (1,410 ha) of the Southern Uplands year^{-1}, with large interannual variability.
Swindell, 2017 [2+, EV-]	Stainery Moor, Peak District (1976–2010)	Between 10 and 29% of study area burned in various years.
Thacker and others, 2015 [2++, EV++]	13 English upland areas (2005–2014); also 106 1-km square (2%) sample across English uplands (1945–2011)	3.8% on deep peat, 4.0% on other soils year^{-1}. Total area burnt year^{-1} 3,310 ha on deep peat and 5,069 ha on other soils (on c.80% of upland dwarf-shrub-dominated area).

Burn patch size

11.14 Five studies reported on burn patch size. These highlighting a large range from the smallest to largest burns, with the majority being small, as well as variation in average sizes according to region and year/time (Table 27). The burn patch sizes compare with the maximum recommendation of 2 ha in the Heather and Grass

Burning Code (Defra, 2007) guidance and the prohibition of burns over 10 ha in the Heather and Grass Burning Regulations 2007 except under licence. The data suggest that sometimes these guidance and regulatory burn size maxima may be exceeded.

Table 27. Summary of results for burn patch size

Study	Burn patch size (ha)
Allen and others, 2016 [2+, EV-]	0.21 (mean), 10.91 (max)
Blundell and others, 2021 [2+, EV++]	0.02–0.05 (median, depending on region/area)
Drewitt, 2015 [2+, EV+]	1.34 (mean for 1995), 0.09 (mean for 2009)
Lees and others, 2021 [2+, EV++]	0.02–2.78 (range); 0.38, 0.19 and 0.24 (mean, 3 sample areas).
Spracklen & Spracklen, 2023 [2+, EV+]	Grampians: burns >1 ha comprised 17% of total burn area, range 5–36%, and >5 ha comprised 8% of burn area, range 1–31%. Southern Uplands: burns >1 ha comprised 21% of burn area, range 5–38%, and >5 ha comprised 8% of burn area.

Burning over deep peat and in protected areas

11.15 Six recent studies reported on burning in relation to peat depth and areas protected by biodiversity conservation designations comprising SSSI (Sites of Special Scientific Interest), SAC (Special Areas of Conservation) and SPA (Special Protection Areas).

11.16 All six studies found evidence of widespread burning over 'deep peat' (although different peat depths were used to define it). Douglas and others (2015 [2++, EV++]) reported that 60% of the burned area in England and 40% in Scotland was on areas they classified as deep peat (>50 cm), whilst Thacker and others (2015 [2++, EV++]) reported that that 40% of the total area burnt within their English areas was on deep peat (>40 cm). Blundell and others (2021 [2+, EV++]) found that burning on deep peat (>40 cm) appeared to follow overall trends in their data from English areas, i.e. deep peat areas were not burned less than areas with shallow or no peat. From more recent data, Natural England (2021 [2+, EV++]) reported that 39–40% of the total area burnt was on deep peat (>40 cm) consistently from 2016–2020, falling to 32% in 2020/2021 and 34% in 2021/2022. In Eastern Scotland, Spracklen & Spracklen (2023 [2+, EV+]) reported that in the Grampian Mountains, 34% (1,590 ha year^{-1}) of burning occurred on deep peat (>50 cm) compared to 23% (330 ha year^{-1}) of burning in the Southern Uplands and there was no significant change in burn area on deep peat over the long study period (1985–2022). Across the GB uplands, Shewring and others (2024 [2++, EV++]) reported an average of 34% of the area recently burnt occurred over deep peat (>40 cm in England and Wales, >50 cm in Scotland), varying from 28% (5,246 ha) in 2020/21 to 41% (8,552 ha) in 2018/19. Within GB, 52% by area of burning over deep peat occurred in England (30–60% of the annual GB area between seasons), 47% in Scotland (40–70% between years) and 1% in Wales (1–4% between years).

11.17 Five recent studies reported on burning in relation to protected areas (PAs, including SSSIs, SACs and SPAs) all of which showed that it was widespread in such sites, though unlike the others described below, one study (Thacker and others (2015 [2++, EV++]) did not directly compare burning inside and outside PAs. There are issues with making such a comparison as PAs are likely to differ in underlying environmental characteristics from non-PA land. This was addressed by Douglas and others (2015 [2++, EV++]) and Shewring and others (2024 [2++, EV++]) by comparing similar matched areas inside and outside of PAs.

11.18 Douglas and others (2015 [2++, EV++]) reported that burning in GB was significantly greater per 1-km square within areas designated as SAC and/or SPA than in matched areas that were not. Shewring and others (2024 [2++, EV++]) reported that the relative proportion of available moorland burned inside and matched areas outside PAs (SAC/SPAs) differed between years for both England and Scotland. In England, a significantly greater proportion of available moorland was burned inside PAs than outside in two of five years; whereas in Scotland, the proportion of moorland burned inside PAs was significantly lower than that outside PAs in all years. As with deep peat, Blundell and others (2021 [2+, EV++]) found that burning on SSSIs tended to follow overall trends across habitats in four English regions/areas. In Eastern Scotland, Spracklen & Spracklen (2023 [2+, EV+]) reported that, on average, 0.7% was burnt annually in the Grampian Mountains compared with a mean of 0.5% of the SAC, SPA and SSSI area, though this increased to 0.7% when the less intensively burned Cairngorms were excluded. This compared with a mean of 1.5% of the Southern Uplands burnt annually, compared with means of 1%, 5% and 1.4% within SACs, SPAs and SSSIs, respectively.

Burning in relation to other habitat features

11.19 Three studies also reported on burning in relation to other habitat features including on montane[19] habitats and by elevation and slope. Shewring and others (2024 [2++, EV++]) recorded 158 ha of burning (0.2% of total burned area) on montane habitats mostly in Scotland (plus a small extent in NW England) over five years which varied little between years (range 26–37 ha, mean 32 ha). Perhaps not surprisingly, the greatest area of burning in these habitats occurred in the 650–750 m altitude band (though the habitat occurred in between the 250–350 m to 950–1,050 m bands).

11.20 Spracklen & Spracklen (2023 [2+, EV+]) reported mean elevation of burns in the Grampians as 443 m and similar (338 m) in the Southern Uplands, and mean slope of burns as 10° and 9.2°, respectively. Three percent of the burnt area was on steep slopes greater than 27° (1 in 2) in the Grampians (which the Muirburn Code states should be avoided, as do the other GB burning codes) and 2% in the Southern Uplands. Similarly, across GB only a small area of burning (range 87–319 ha annually) occurred on slopes greater than 27° (Shewring and others, 2024 [2++, EV++]).

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[19] A range of high-altitude habitats, generally above c.600 m, which are particularly sensitive to perturbations and are also referred to as mountain or alpine habitats.

11.21 Douglas and others (2015 [2++, EV++]) reported that the percentage area of moorland burned peaked at 400–600 m elevation and in 1-km squares with a mean gradient of about 10°, with very little on slopes greater than 25°.

Burn frequency

11.22 For studies that measured burning extent as an annual proportion of the study area, frequency can be calculated by working out how many years it would take to burn 100%. Using this method, Lees and others (2021 [2+, EV++]) reported a mean frequency of 20–66 years depending on English region/area. Thacker and others (2015 [2++, EV++]) reported frequencies or 'return periods' for dwarf shrub dominated areas in England of 27 years on deep peat and 25 years on other soils including shallow peat. They also found regional/area variation, with return times as frequent as 11 years on deep peat in the North York Moors.

11.23 One study at a single site (Allen and others, 2016 [2+, EV-]) reported the actual frequency of burning during a 22-year period and found that 23% of burned area was burned twice, 7% burned thrice and smaller proportions four or five times. Of the area burned in the first three years, 59% had been burned at least twice by the end of the study period.

Change over time

11.24 Eleven studies reported on trends over time regarding the extent, frequency or type of managed burning.

11.25 Three found that the extent of burning had increased over recent decades at national scales (Douglas and others, 2015 [2++, EV++]; Thacker and others, 2015 [2++, EV++]; Matthews and others, 2020 [2+, EV++]), with Douglas and others (2015 [2++, EV++]) additionally reporting an accelerating increasing trend between 2001 and 2011. The trends were generally similar between deep peat and other soils.

11.26 Four other studies reported either fluctuating (Allen and others, 2016 [2+, EV-]; Swindell, 2017 [2+, EV-]) or relatively consistent (Drewitt, 2015 [2+, EV+]; Critchley and others, 2016 [2++, EV++]) burning extent over time, though the first two showed fluctuations in burning extent covered longer periods at single sites (1988–2009 and 1976–2010, respectively), but didn't include more recent, post-2010, trends.

11.27 A national-scale Moorland Change Map (MCM) study (Natural England, 2021 [2+, EV++]) found a gradual small decline in burning extent in England in more recent years (2016–22). An exception to this trend was the 2018/19 burning season when a greater burned and/or cut area was recorded, possibly reflecting the occurrence of some large wildfires which may have been included in the data. A decline in the proportion of burning over deep peat compared with other soils was observed in the MCM data over the same period (para. 11.16). Similarly, another more recent GB-scale study (Shewring and others, 2024 [2++, EV++]) showed a marked reduction in the total burned area in 2021/22, driven mainly by a reduction in area in England. The area burned in England in 2021/ 22 (1,859 ha) was 73% lower than the mean of the four previous seasons (6,913 ha). This decline was greater than that detected in the Natural England (2021 [2+, EV++]) study, perhaps reflecting a switch to cutting that was not separated from burning in the MCM data but was attempted by Shewring and others (2024 [2++, EV++], para. 11.6). It may also reflect other differences in methods and the exact regions/areas covered.

11.28 Drewitt (2015 [2+, EV+]) found that there were a greater number of smaller burns in 2009 than 1995 in sample 1 km-squares in the North York Moors. Allen and others (2016 [2+, EV-]) also reported a decrease in maximum burn patch size between 1988 and 2009 at a Peak District site.

11.29 Two other studies reported on trends in burning extent over longer time periods. Thacker and others (2015 [2++, EV++]) summarised long-term trends in estimated burning extent on peat and other soils from 74 representative sample 1-km squares across the English uplands between 1945 and 2011. In the sample as a whole, the extent of burning increased approximately five-fold from the 1940s to 2011 and the increase was greater over deep peat than other soils (Figure 12).

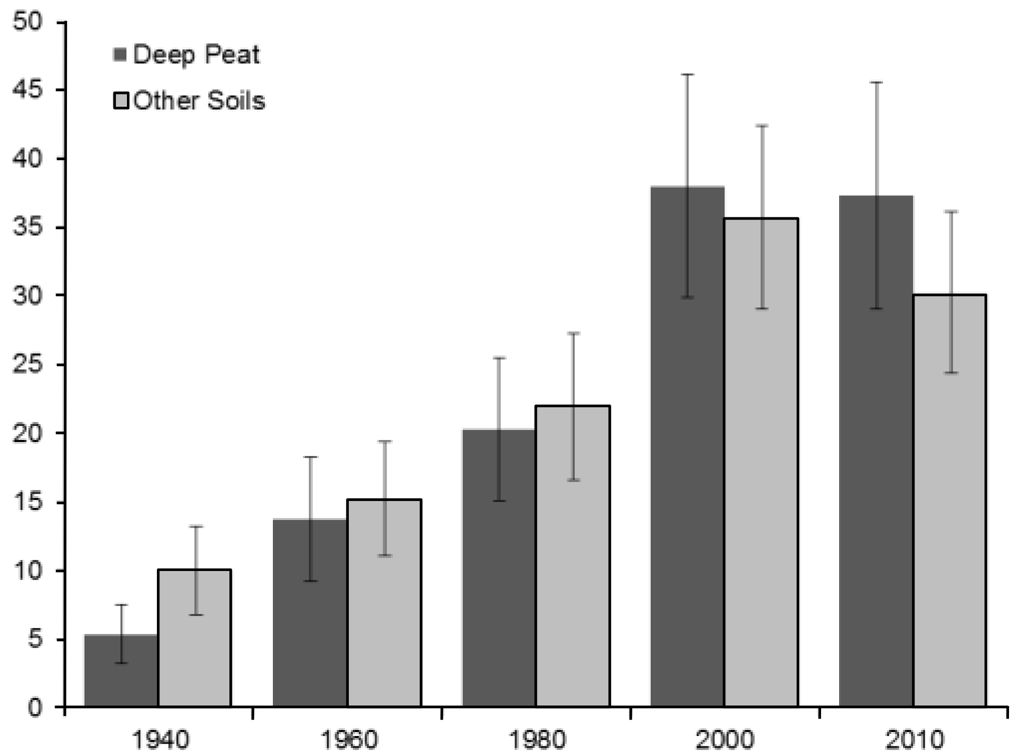

Figure 12. Estimated annual burning extent (km^2) on deep peat and other soils in the English uplands 1945–2010 based on 74 sample 1-km squares for which imagery was available for all decades. Bars show standard errors. (Taken from Thacker and others, 2015, Figure 3).

11.30 Conversely, Spracklen & Spracklen (2023 [2+, EV+]) found no significant trend in estimated area burnt annually over the period 1985 to 2022 in the Grampian

Mountains and Southern Uplands in Eastern Scotland, though there was considerable year-to-year variability. They note, though, that an earlier study showed a decline in the area burnt in the Grampians from the mid-1940s to the 1980s (Hester & Sydes, 1992). The recent lack of consistent trends in these areas in Eastern Scotland is in contrast to reported increasing burn extent across Scotland from 2001 to 2011 (para. 11.25), though this may reflect differences between areas and methods.

11.31 Based on an annual muirburn area of 6,100 ha $year^{-1}$ and only combustion of aboveground vegetation and various other assumptions, Spracklen & Spracklen (2023 [2+, EV+]) also made a preliminary estimate of particulate air pollution emissions from recent burning in Scotland of 1,000 tonnes annually.

12. Comparison of findings from NEER004 with findings from this update

The effects of managed burning on the vegetation of upland peatland habitats

Summary of similarities and differences

12.1 A comparison of the evidence on the effects of managed burning on upland peatland vegetation outcomes in NEER004 and this review update is given in Table 28. A similar number of studies were considered in the update as in NEER004, and the geographic bias of these studies was also similar.

12.2 The recent studies were broadly consistent with the findings of NEER004. For example, evidence from recent vegetation studies supported many of the findings of NEER004, including the conclusion that overall burning results in a change in species composition, at least for a period. This is characterised by an initial period of graminoid dominance especially of *Eriophorum vaginatum*. Dwarf shrubs initially decline, though this is typically followed by a gradual increase, sometimes to dominance or alternatively to a steady state growing through *Sphagnum* and other bryophyte hummocks and mats. Recent evidence also supported the findings that burning is associated with the creation of bare ground, that other dwarf shrubs, especially *Empetrum nigrum*, may decline and not necessarily recover following burning, and that bryophytes decline in cover and biomass following burning but some early colonising, especially acrocarpous moss species may rapidly increase while others, typically pleurocarpous species, only gradually increase with increasing time post-burn.

12.3 Both NEER004 and the update found inconsistent evidence on the effects of burning on *Sphagnum* species and *Rubus chamaemorus*, though for the latter, a long-term analysis showed a decline with short-rotation burning. New evidence included that burning reduces *Sphagnum* propagule frequency in peat, and moss mat depth which gradually increases post-burn.

Table 28. Comparison of evidence assessed for the vegetation sub-question of NEER004 and this update. Individual study findings are described in Section 4.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
Number and location of studies	39 studies (25 primary). All 25 (primary) studies UK-based (most in England [16], especially Pennines [13], or Scotland [6], with 2 N Ireland/ Ireland and 1 Wales).	45 studies (all primary). 37 UK-based (most [32] including England, especially Pennines [26], Scotland [7], and 1 each Wales and N Ireland).	Similar number (though more recent primary studies) and geographic bias.	The available evidence is likely to be applicable to northern England and Scotland, and some to the wider UK uplands.
Overall conclusion on species composition	Strong evidence that burning results, at least for a period, in a change in species composition (18 studies, see below). This includes some evidence from ordination plots indicating a general tendency of separation of burnt and not-recently-burnt treatments. However, there is limited information on longer-term temporal trends with the only experimental study covering multiple rotations, the Hard Hill experiment ([1++]).	Similar evidence that burning affects post-burn vegetation species composition typically followed by a gradual transition back towards pre-burn composition (31 studies). This also includes evidence from ordination plots which show separation of burnt and not-recently-burnt treatments.	Recent evidence consistent with NEER004 findings.	Evidence that burning results, at least for a period, in changes in vegetation species composition.
Overall species richness (SR) (Also see SR outcomes for species groups below.)	Evidence (from Hard Hill expt., [1++]) showed marginally highest total SR in unburned since 1954 plots, and lowest in the 20-year burn, treatments, though the differences were relatively	Evidence of variable, relatively small effects on total plant SR in 3 studies: an initial post-burn decline followed by an increase back to above pre-burn levels after	Recent evidence consistent with NEER004. Similar inconsistent post-burn trends in NEER004 and recent studies perhaps reflecting differences in post-burn	Evidence of inconsistent, often relatively small effect of burning on overall SR. Species contributing to post-burn change in SR and diversity may not be

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	small. In the longer-unburned 'reference' plots, SR did not change significantly (though moss SR increased significantly, whilst lichen SR declined). Contrary to this, SR declined (slightly from low starting level) post-burn and was lowest in long-unburned (≥35 years) stands (on 5 severely modified sites) (Harris *et al*., 2011b [2+]), though the additional species post-burn were mostly acid grassland or heath species rather than those characteristic of blanket bog.	just 2 years (Velle & Vandvik, 2014 [1+, EV-]); a post-burn increase lasting up to at least 9 years (Heinemeyer and others, 2019c/2023 [1+, EV-]); no change on resurvey of 10 wet heath sites after 28 years (Muñoz *et al*., 2014 [2-, EV-]).	timescales monitored and in degree of modification of peatland vegetation pre-burn.	typical mire species and may include species more characteristic of heath, grassland or other habitats, indicative of poor or unfavourable condition, or be colonisers of early successional stages. Thus, species richness and diversity derived from all species present may have limited usefulness for interpreting change in peatland vegetation in terms of quality and condition. An increase may even reflect a move away from characteristic vegetation of the habitat depending on the species contributing.
Vascular plant species richness (SR)	No evidence.	Vascular plant SR relatively low across treatments (between 6 and 9 spp.) at Hard Hill, though showed little temporal change in not burned since 1954 plots compared with increases under the burn treatments, greatest in shorter, 10-year plots (Milligan *et al*., 2018	New evidence.	Evidence of low vascular plant species richness and relatively little temporal change, though a slight increase post-burn, though not all species were characteristic of upland peatland habitats.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		[1++, EV-]). It was similarly lowest in the oldest post-burn age class (>17 years) in another single site study (Whitehead & Baines, 2018 [2+, EV-]).		
Overall species diversity	No evidence.	Burning had little effect on various diversity indices in 2 studies (Muñoz *et al*., 2014 [2-, EV-]; Whitehead *et al*., 2021 [2+, EV-]), though in 2 others, diversity (Shannon-Weiner index) increased following burning but declined over time in recently unmanaged treatments (Milligan *et al*., 2018 [1++, EV-]; Heinemeyer *et al*. 2019c/2023 [1+, EV-]).	New evidence.	Inconsistent evidence on the effect of burning on species diversity with no change and increases reported.
Ecological indicators	No evidence.	2 studies reported on the effects of burning on Ellenberg ecological indicator scores (Milligan *et al*., 2018 [1++, EV-], Hard Hill; Velle & Vandvik, 2014/Velle *et al*., 2014 [1+, EV-]). In the former, there were increases in fertility (only 20-year burn), acidity, light and moisture scores under burning treatments compared with a slight decline for acidity and moisture under no burn since 1954. In the latter study, there	New evidence.	Limited evidence of post-burn increases in Ellenberg ecological indicator scores for fertility, acidity, light and moisture mostly from a single study and for acidity in another study, compared with a slight decline for acidity and moisture under a no burn since 1954 treatment.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		was a similar increase in acidity score following burning.		
Species composition: graminoids (as a group) initial response	Strong evidence that burning leads to an initial period of graminoid dominance, typically of 10–20 years and at least an initial decline in dwarf shrub cover and in some cases diversity (11 studies: 2 [1++]: Hard Hill experiment; Stewart *et al*., 2004; 4 [1+]: Miles, 1971; Ross *et al*., 2003; Marrs *et al*., 2004; Ward *et al*., 2007 [1,2+]; 2 [2+]: Currall, 1981; Harris 2011b; and 3 [2-]: Elliott, 1953; Forrest & Smith, 1975; McFerran *et al*., 1995).	Similar evidence of a relatively rapid post-burn increase in graminoid cover/frequency varying in magnitude (6 studies: 4 related to *Eriophorum* spp., see studies below; and 2 to *Carex* sedge species or grasses: Velle *et al*., 2012; Velle & Vandvik, 2014, both [1+]).	Recent evidence consistent with NEER004 findings.	Evidence that burning frequently results, for at least a period of 10-20 years, in graminoid dominance.
Species composition: graminoid species initial response	Strong evidence that: *Eriophorum vaginatum* attains initial dominance, particularly in the Pennines (3 studies: Hard Hill experiment [1++]; Ward *et al*., 2007 [1+] at same site; Harris *et al*., 2011b [2+]); or alternatively that other graminoids especially *Molinia* and *Trichophorum* tend to initial dominance in the wetter, oceanic west (5 studies: 3 [1+]: Miles, 1971; Ross *et al*., 2003; Marrs *et al*.,	Similar recent evidence of a post-burn increase in *Eriophorum vaginatum*/*E*. spp. varying in magnitude (range at peak 16–55% cover perhaps reflecting degree of modification and/or wetness) (4 studies: Milligan *et al*., 2018/Hard Hill experiment [1++]; Heinemeyer *et al*., 2019c /2023 [1+]; 2 [2+], Whitehead & Baines, 2018; Whitehead *et al*., 2021).	Recent evidence of initial dominance of *Eriophorum vaginatum,* particularly in the Pennines consistent with NEER004. No recent evidence confirming initial dominance of *Molinia* or *Trichophorum* in the west.	Evidence that especially in the Pennines, burning results in initial dominance of *Eriophorum vaginatum* or alternatively, other graminoids especially *Molinia* and *Trichophorum* in the wetter, oceanic west.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	2004; and Currall, 1981 [2+]; Forrest & Smith, 1975 [2-]; Anderson *et al*., 2006 [4+]).	No recent studies showed effects on *Molinia* or *Trichophorum* probably reflecting the scarcity of more recent study sites in the west.		
Species composition: graminoid species longer-term response	The period of graminoid dominance tends to be longer than typically occurs on dry heath or more severely modified, drier bog (Harris *et al*., 2011b [2+]), probably reflecting the greater competitive advantage of wetland graminoids.	Especially on *Calluna* dominated blanket bog sites, included in the above studies, *Eriophorum* species and other graminoids tended to then decline over time as *Calluna* increased but, in some cases, *Eriophorum* still maintained moderate, or in one case relatively high, cover/ frequency (same 4 studies as above). Related to this, *Eriophorum* species may sometimes occur at moderate or high cover in relatively long-unburned sites (1 study, 2 data sets: Noble and others, 2018b [2+, EV++]), perhaps reflecting an interaction with grazing.	Recent evidence generally consistent with NEER004, although limited additional evidence that *Eriophorum* may maintain moderate or high cover even in relatively long unburned sites perhaps reflecting an interaction with grazing.	Evidence that graminoid dominance tends to be longer than typically occurs on dry heath or more severely modified, drier bog and rather than then declining may maintain moderate or high cover even in relatively long unburned sites perhaps reflecting an interaction with grazing.
Graminoid biomass	Evidence that burning increased graminoid dry weight biomass 10 years post-burn by 88% compared to the no-burn since 1954 (50 years) treatment at Hard Hill (1 study: Ward *et al*., 2007 [1,2+]).	Similar recent evidence of an increase the proportion of total biomass represented by graminoids (and bryophytes/ lichens) soon after burning from the same site (1 study: Ward *et al*., 2012 [1,2+, EV-]). [check actual biomass]	Recent evidence consistent with NEER004 findings.	Limited evidence of a post-burn increase in graminoid biomass followed by a decline.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
Species composition: *Calluna* initial response and recovery	Strong evidence that: *Calluna* tends to decline during the initial post-burning graminoid-dominant phase; but typically then increases (7 studies: 2 [1++]: Hard Hill experiment [1++]; Stewart *et al*., 2004; 3 [1+]: Ross *et al*., 2004; Marrs *et al*., 2004; Ward *et al*., 2007[1,2+]; and Currall, 1981 [2+]; McFerran *et al*., 1995 [2-]), especially on drier sites, though it may take 15–20 years to regain dominance on less modified, wetter blanket bog (Hard Hill experiment [1++]; Ward *et al*., 2007 [1+], same site) and this may not occur, for example, with too frequent or severe burning and/or heavy grazing (Currall, 1981 [2+]).	Similar evidence of an immediate or initial post-burn decline in *Calluna* (or in 2 cases ericoid dwarf shrubs collectively) cover, often down to a few % live cover, usually from dominance pre-burn suggesting modified habitat state (10 studies: Heinemeyer *et al*., 2019c/2023 [1++, EV+]; Milligan *et al*., 2018 [1++, EV-]/Hard Hill (in NEER004); Velle *et al*., 2012 [1+, EV-]; Velle & Vandvik, 2014 [1+, EV-]; Grau-Andrés, 2018 [1+, EV-]; Worrall *et al*., 2013a [1/2+, EV-]; Whitehead & Baines, 2018 [2+, EV-]; Noble *et al*., 2019b [2+, EV+]; Whitehead *et al*., 2021 [2+, EV-] Garnett *et al*., 2019 [2-, EV-]). In the 9 of these studies that included subsequent monitoring, the decline was followed by a relatively rapid recovery back towards, and in some cases reaching, pre-burn cover within around 10–12 years. Evidence of slower increase in *Calluna* cover on the wettest of three Peatland-	Recent evidence consistent with NEER004 findings.	Conclusion unchanged.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		ES-UK sites (1 study: Heinemeyer *et al*., 2019c/ 2023 [1++, EV+]).		
Species composition: *Calluna* longer-term response	*Calluna* may continue to increase or maintain high cover for a considerable period. 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 its stems tend to be constantly reburied by growth of *Sphagnum* and through the rejuvenation of stems, an uneven aged stand of *Calluna* is produced in a 'steady state' where other mire species are well represented (Rawes & Hobbs, 1979, and Hobbs, 1984, under the Hard Hill experiment [1++]; and 4 [4+]: Mowforth & Sydes, 1989; Coulson *et al*., 1992; Tucker 2003; Lindsay, 2010).	No evidence.	No new evidence	Conclusion unchanged.
***Calluna* seed density**	No evidence.	A single study in 2 areas (Lee *et al*., 2013b [1,2+, EV-]) showed: *Calluna* seed density in peat at Hard Hill expt. increased with increasing time since	New evidence.	Limited evidence of much greater *Calluna* seed density at a severely modified site than a less modified site and that at the former

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		burning, being highest in the longest unburned ‘reference’ plots (with an interaction with grazing, with generally greater seed densities in ungrazed treatments); *Calluna* seed density was much higher than at Hard Hill and greater in peat than in litter, at 3 Peak District sites. Elapsed time since burning had no effect on peat seed density, but in the litter layer it increased with time since burning suggesting that the aboveground litter fraction acted as a barrier to seed transfer to the underlying peat.		that aboveground litter acted as a barrier to seed transfer to the underlying peat.
Species composition: other dwarf shrubs	Weak evidence that other dwarf shrubs, especially *Empetrum nigrum*, may decline following burning (3 studies: Cotton & Hale, 1994 [1-]; and Elliott, 1953; Forrest & Smith, 1975, both [2-]) and sometimes not recover after more severe fires (1 study: Stewart *et al*., 2004, Hard Hill [1++]).	Single recent studies reported: a temporal decline in *Empetrum nigrum* frequency associated with rotational burning (and generally lower cover with grazing) (Milligan *et al*., 2018, Hard Hill [1++, EV-]); and reduced cover of *Vaccinium myrtillus* with increasing intensity of burning (Newey *et al*., 2020 [2+, EV++]).	Recent evidence consistent with NEER004 findings.	Evidence that some other dwarf shrubs, especially *Empetrum nigrum*, may decline following burning and sometimes not recover if it is severe.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
Dwarf shrub biomass	Evidence that burning reduced ericoid (mostly *Calluna*) dwarf shrub dry weight biomass 9 years post-burn by 51% compared to the no-burn since 1954 (then 50 years post-burn) treatment at Hard Hill (1 study: Ward *et al*., 2007 [1+]). Grazing had a similar, but smaller magnitude effect.	Similar evidence of increasing *Calluna* biomass with increasing time since burning at the same site across all treatments (1 study: Alday *et al*., 2015 [1,2+, EV-]). Only the shortest (10-year) treatment (at 5 years post-burn) had significantly reduced *Calluna* biomass, though modelled growth curves suggested an asymptote for biomass after 20 years and 15 years for vegetation height, though actual biomass levelled-off, with a decline in mean *Calluna* biomass in the longest unburned 'reference' plots compared with the no-burn since 1954 treatment.	Recent evidence consistent with NEER004 finding (but extended to all treatments and the longer-unburned 'reference' plots at Hard Hill and to other sites which showed a levelling off of *Calluna* biomass accumulation in longest-unburned plots.	Evidence that burning results in an initial major reduction *Calluna* biomass followed by a relatively rapid recovery (with an absolute growth rate [AGR] peak after eight years) which then tends to slow (asymptote after 20 years)
Species composition: *Rubus chamaemorus*	Moderate but inconsistent evidence, with 2 related Moor House studies indicating that burning leads to an increase in *Rubus chamaemorus* (Hard Hill experiment, 1961–2001 [1++]; Taylor & Marks, 1971 [1+]), although another study on the same site reported a decline (Forrest & Smith, 1975 [2-]) and re-analysis of data from the main	Most recent resurvey from Hard Hill expt. indicated a decline over time in *R. chamaemorus*, but only in the 10-year burn treatment where abundance was generally greatest at the start (Milligan *et al*., 2018 [1++, EV-]). There were no effects of sheep grazing.	Inconsistency in findings between some earlier analyses from Hard Hill included in NEER004 (initially indicating an increase, then no change) and most recent analysis of temporal change which shows a decline only under the short rotation burn treatment (and no grazing effect). It is	Inconsistent evidence on the effects of burning on *R. chamaemorus* from one site (Moor House NNR, especially the Hard Hill experiment) with the latest long-term analysis indicating a decline but only under the short (10-year) burn rotation treatment.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	experiment between 1972 and 2001 failed to detect a difference between burn treatments (but showed a decline in the grazed, 'unburnt' treatment) (Lee *et al*, 2013, as part of the Hard Hill study [1++]). This suggests that grazing may be an important factor that may interact with burning (Taylor & Marks, 1971 [1+]).		possible that this might reflect initial differences between blocks as well as burn treatments.	
Species composition: bryophytes as a group	Strong evidence that overall, bryophytes tend to decline in cover/frequency initially after burning, although some early-colonising species may quickly become frequent or even abundant (6 studies: Hard Hill experiment [1++]; 2 [2+], Currall, 1981; Harris, 2011b; and 3 [2-], Forrest & Smith, 1975; Burch 2008 [2-]; McFerran *et al*., 1995).	Other bryophytes as a group showed initial post-burn declines in cover/frequency, mostly from high cover pre-burn (7 studies: Velle *et al*., 2011 [1+, EV-]; Velle & Vandvik, 2014 [1+, EV-]; Grau-Andrés *et al*., 2019b/a [1+, EV-]; Milligan *et al*., 2018 [1++, EV-], Hard Hill; Whitehead & Baines, 2018 [2+, EV-]; Heinemeyer *et al*., 2019c [1+, EV-]); Whitehead *et al*., 2021 [2+, EV-]). In all cases but one (where there was no extended post-burn monitoring), the decline was followed by a relatively rapid increase in most cases back up to or towards pre-burn levels.	Recent evidence consistent with NEER004 finding.	Evidence that overall bryophytes tend to initially decline in cover/frequency after burning which is followed by a relatively rapid increase in most cases back up to or towards pre-burn levels.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
Species composition: acrocarps	Some early-colonising, typically acrocarpous moss species, may relatively quickly become frequent or even abundant after burning (2 studies: Hard Hill experiment [1++]; Burch, 2008 [2-]).	Acrocarps as a group increased rapidly post-burn (2 studies: Grau-Andrés *et al*., 2019b, 2018 [both 1+, EV-]) and another increasing up to c.30% within 3 years (Noble *et al*., 2019b [2+, EV+]). An individual acrocarpous species, *Campylopus introflexus* increased in frequency/cover post-burn (2 studies: Milligan *et al*., 2018 [1++, EV-], Hard Hill; Noble *et al*., 2018b [2+, EV++]).	Recent evidence consistent with NEER004.	Conclusion unchanged.
Species composition pleurocarps	No evidence.	An individual pleurocarpous moss species, *Hypnum jutlandicum*, showed no increase in 10-year burn plots, a delayed increase in 20-year burn plots and a large increase in unburned since 1954 plots (especially ungrazed) (Milligan *et al*., 2018 [1++, EV-], Hard Hill) and a large immediate decline from 60% pre- to 19% post-burn (Grau-Andrés *et al*., 2019b/a [1+, EV-]).	New evidence.	Evidence that an individual pleurocarpous moss species, *Hypnum jutlandicum*, showed a large immediate post burn decline, no initial increase, a delayed increase in 20-year burn treatment and a large increase in unburned since 1954 plots.
Bryophyte biomass (including *Sphagnum* species)	Evidence that burning reduced bryophyte dry weight shoot biomass 9 years post-burn by 92% compared to the no-burn since 1954 (then 50	Similar evidence of increasing bryophyte biomass with increasing time since burning at the same site across all treatments including the	Recent evidence consistent with NEER004 finding.	Conclusion unchanged.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	years) treatment at Hard Hill (1 study: Ward *et al*., 2007 [1+]). (Grazing had a similar but much smaller magnitude effect.)	longer-unburned 'reference' plots (1 study at same site: Alday *et al*., 2015 [1,2+, EV-]).		
Moss depth	No evidence.	2 recent chronosequence studies showed lowest moss depth in the youngest post-burn age classes (12–18 months and up to 4 years) (Noble *et al*., 2019b [2+, EV+]; Whitehead *et al*., 2021 [2+, EV-]). Another study reported moss depth indirectly as immediate moss/litter consumption: depth declined by 0.1 cm in 'no-drought' and 1.4 cm in simulated 'drought' (increased severity) burn plots (Grau-Andrés *et al*., 2019b/a [1+, EV-]).	New evidence available.	Evidence that moss depth is lowest post-burn, after which it increases, and that it declines more with more severe burning.
Species composition: *Sphagnum* species	*Sphagnum* spp. as a group showed mixed responses, in some cases increasing in the early post-burn stages (Lee *et al*., 2013 [1++], as part of the Hard Hill expt.; Forrest & Smith, 1975 and Hamilton, 2002 both [2-]), sometimes declining or being killed (Lindsay, 1977 [2++]; Hamilton, 2000 [2-]) and sometimes later increasing or	*Sphagnum* spp. as a group were reported as showing a range of responses. Some studies suggest small initial declines immediately after burning, often from relatively low pre-burn cover, followed by some initial recovery (Heinemeyer *et al*., 2019c /2023 [1+, EV-]; Whitehead & Baines, 2018 [2+, EV-]; Garnett, 2023 [2+, EV-]) while	Recent evidence generally consistent with NEER004 findings.	There is moderate evidence that *Sphagnum* (generally reported as a group) may show a range of responses to burning. Some studies suggest (i) initial declines immediately post-burn, sometimes being killed; (ii) little or no change or recovery; or (iii) some recovery or

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	recolonising after varying periods (Miles, 1971 [1+]; Lindsay & Ross, 1994 [2++]; Burch, 2008 [2-]).	others indicated little or no change, though again generally over relatively short periods (Grau-Andrés *et al*., 2019b, 2017b [both 1+, EV-]); Heinemeyer *et al*., 2019c/2023 [1+, EV-]) or similarly, no difference between burnt and not-recently-burnt stands (Noble *et al*., 2018b [2+, EV++]). Conversely, 1 national study showed lower *Sphagnum* spp. cover in recently burnt than not-recently-burnt sites, though the difference was relatively small (Noble *et al*., 2018b [2+, EV++]) and another study showed highest *Sphagnum* spp. cover in longest-unburned areas (>40 years) (Swindell, 2017 [2+, EV-]). [Hard Hill] Covering a much longer period, re-analysis of 1965 data from the (grazed) unburnt since 1954 and longer-unburnt but otherwise comparable 'reference' plots at Hard Hill expt. showed that the latter had significantly greater *Sphagnum* cover		recolonisation after varying periods.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		(Noble *et al*., 2018a [12+, EV-]). This suggests that the (relatively large, whole block) 1954 burns had a negative effect of on *Sphagnum* that was apparent 11 years after burning and which was still observed in the 2015/16 survey over 60 years later		
***Sphagnum* species richness (SR)**	The total number of *Sphagnum* spp. was highest in the short (10-year) burn treatment, and lowest under the 20-year burn, with no burn since 1954 intermediate (Hard Hill experiment [1++]). However, 4 spp. occurred in both the expt. blocks and 'reference' plots with 5 additional spp. in the latter and 4 in the former despite the former covering a much larger area (making comparison problematic as frequency is scale-dependant).	1 study showed a decline in *Sphagnum* SR following burning from 2.2 spp. pre burn to absence for 2 years followed by only a gradual increase to approaching the pre-burn number after 13 years (c.f. no decline from 2.3 spp. following cutting and a gradual increase to 4.2 spp. over 13 years) (Garnett, 2023 [2+, EV-]).	Recent evidence not consistent with NEER004 finding.	Limited, inconsistent evidence on the effect of burning on *Sphagnum* species richness.
***Sphagnum* propagule frequency**	No evidence.	*Sphagnum* propagule frequency increased with time since the last burn, with highest frequency in the longest unburned 'reference' plots and very low frequency under both burn treatments	New evidence available.	Burning reduces *Sphagnum* propagule frequency in peat.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
		and was absent from 3 additional burnt sites in the Peak District (1 study: Lee *et al*., 2013a, Hard Hill expt. [1,2+, EV-]). This suggests that frequent burning reduces the *Sphagnum* propagule bank and hence its regeneration potential from peat.		
Overall propagule frequency	No evidence.	A single study in 2 areas (Lee *et al*., 2013a [1,2+, EV-], Hard Hill and Peak District) showed that: (i) propagule banks in both areas were species-poor and comprised mostly common species; (ii) species-richness, frequency of *Sphagnum* and *Polytrichum* spp. and *Calluna* seed density all increased with time since burning in peat at Hard Hill; (iii) *Calluna*, seed density was much higher and greater in peat than litter at the Peak District sites. While elapsed time since burning had no effect on peat seed density, it increased in the litter layer over time. *Juncus effusus* increased in litter with time from burning at 1 site.	New evidence.	

Outcome	NEER004	2024 update	Comparison	Updated conclusion
Long-term changes in overall species composition	Moderate evidence (from one long-term study as part of the Hard Hill experiment, Lee *et al*., 2013a [1++]) that the composition of blanket bog vegetation can continue to show change more than 80 years since 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 individual moss species.	No evidence.	No new evidence.	Conclusion unchanged.
Bare ground	Strong evidence that burning is associated with the creation of bare ground, at least at a fine scale (5 studies: Hard Hill expt. 2061–2001 [1++]; Cotton & Hale, 1994 [1-]; Currall, 1981 [2+]; Elliott, 1953 and McFerran *et al*., 1995, both [2-]), including in both less modified and severely modified blanket bog, and wet heath (Currall, 1981 [2+]). The evidence indicated that cover of bare ground was greatest initially after burning and variable, ranging up to c.50%, before gradually declining, though it	5 recent studies reported on changes in cover of bare ground, all of which showed an initial post-burn increase, though in only three cases did this specifically relate just to bare ground (Velle & Vandvik, 2014 [1+, EV-]; Worrall *et al*., 2013a [1,2+, EV-]; Noble *et al*., 2019b [2+, EV+]) with the 2 others being combined with ‘duff’ and ‘brash’/’burnt’ (Grau-Andrés *et al*., 209b/2018 and Heinemeyer *et al*., 2019c [both 1+, EV-]), respectively. Cover of bare ground tended to be greatest initially after burning and ranged up to	Recent evidence consistent with NEER004 findings.	Evidence statement unchanged, though recent evidence that bare ground cover may be relatively high for a period.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	was still apparent after up to 9 years in one case (with longevity potentially affected by other factors such as trampling by livestock, Currall, 1981 [2+], and erosion, especially on slopes, Elliott, 1953 [2-]).	c.60%, mostly lower, before gradually declining, though it was still apparent after up to 4+ years. 1 study showed increased cover of bare ground with increased severity of burning (Grau-Andrés *et al*., 2019b/2018 [1+, EV-]).		
Vegetation structure and microtopography	Few 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 (2 studies: Lindsay & Ross, 1994 [2++]; Hamilton, 2000 [2-]).	2 recent studies reported on microtopographic variation in the Hard Hill experiment plots (Clutterbuck and others, 2020 [1,2+, EV-], full survey of all blocks yet to be reported; Noble and others (2018a [1,2+, EV-]/O'Reilly, 2016) which showed some differences between treatments and blocks.		
Longer-term effects of differences in burning rotations	Relatively little evidence of the longer-term effects of differences in burning rotations on peatland vegetation with the only long-term experimental study that has covered multiple rotations of differing lengths (10- and 20-years) being the Hard Hill experiment [1++]. This	The Hard Hill expt. remains the only long-term burning study involving multiple, different rotations, with the latest resurvey in 2013 representing 6 burn cycles for the 10-year burn and 3 for the 20-year burn treatments, respectively, and 60 years of recovery in the unburned	Recent evidence consistent with NEER004 findings (from same extended study).	Conclusion unchanged.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	provides moderate evidence that differences in frequency of burning affect the vegetation composition and structure of blanket bog in the medium to long term. At this site, more frequent burning has promoted dominance of *Eriophorum vaginatum*, with *Calluna* achieving higher cover but not necessarily overwhelming dominance under the long rotation and longer-unburned 'reference' plots.	since 1954 treatment (Milligan *et al*., 2018 [1++, EV-], as part of the Hard Hill study).		
Palaeoecological studies	2 studies: Chambers *et al*. (2007 [2+]) documented a 20th Century rise to dominance of *Molinia* and *Calluna* on 2 different sites in South Wales which was considered unprecedented. Previously *Sphagnum austinii* had shown millennial-scale dominance. Though there was evidence of increased burning, it was suggested that a range of factors including increased atmospheric input and changes in grazing pressure may have been responsible. Ellis (2008, [2+]) suggested environment-	Evidence from 4 studies that vegetation burning post-Industrial Revolution likely contributed to subsequent change of vegetation community in the peat archive to (i) graminoid dominance (3 studies: Rowney *et al*., 2023 [2+, EV-]; Fyfe *et al*., 2018, [2+, EV-]); McCarroll *et al*., 2017, [2+, EV-]); and (ii) *Calluna* dominance (1 study: Blundell & Holden, 2015 [2+, EV+/-]), though atmospheric pollution may have contributed to these changes.	Recent evidence consistent with NEER004 findings.	

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	vegetation interactions over millennial-scale peatland development and landscape change in NW Scotland. With contrasting plant groups responding to changes in surface hydrology: “Drier conditions, along with more burning are associated with more Ericaceae and *Racomitrium.* Wetter conditions with less burning, are associated with monocotyledons and *Sphagnum*.”			
Summary	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	Recent evidence supports the NEER004 vegetation summary.	Recent evidence consistent with NEER004.	Conclusion unchanged.

Outcome	NEER004	2024 update	Comparison	Updated conclusion
	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*.			

The effects of managed burning on the fauna of upland peatlands

Summary of similarities and differences

12.4 A comparison of the evidence on the effects of managed burning on upland peatland fauna outcomes in NEER004 and the current review update is given in Table 29. A similar number of studies were considered in the update as in NEER004, and the geographic bias of these studies was also similar.

12.5 The recent studies were broadly consistent with the findings of NEER004. For example, evidence from recent breeding bird studies supported the findings of associations between habitat type and especially vegetation structure and composition, and burning and/or predator control, and breeding bird numbers of some species. There was also consistent evidence on overlap of bird egg laying with the burning season and stronger evidence of gradual advancement to earlier egg-laying dates across most species. There was also consistent evidence that burning influences terrestrial invertebrate community composition and is associated with changes in the diversity and composition of aquatic invertebrate assemblages in upland peatland watercourses.

12.6 There was increased evidence of stronger positive associations between predator control and some birds than with burning, especially for some waders and red grouse, and negative associations between burning and/or predator control and a few bird species, especially passerines. New evidence suggested that burning, cutting and restoration interventions may affect cranefly abundance and availability as avian prey and that in Scotland adders mostly occur in 1-km squares where grouse moor management does not take place.

Table 29. Comparison of evidence assessed on the effects of managed burning on upland peatland fauna in NEER004 and this review update. Recent study findings are described in Section 5.

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
Number and location of studies	23 studies (20 primary). 21 UK-based (mostly N England or Scotland). Most on birds (12,10 primary); also invertebrates (4 terrestrial primary, plus 3 aquatic primary studies reported in water section), and mammals (1 primary).	24 studies (23 primary). 22 GB-based (mostly N England or Scotland). Most on birds (16 primary); also invertebrates (9, 8 primary including 3 aquatic); and single primary studies on reptiles, mammals and soil microbes.	Similar number and geographic bias.	Evidence is likely to be applicable to northern England and Scotland, and some to the wider UK uplands.
Breeding birds: habitat type and vegetation structure & composition	Associations between moorland **habitat and especially vegetation structure/composition** and density of some birds, especially waders (6 studies: 1 [2++]: Amar *et al*., 2009; 4 [2+]: Haworth & Thompson, 1990; Smith *et al*., 2001; Tharme *et al.*, 2001; Pearce-Higgins & Grant 2006; 1 [2-]: Daplyn & Ewald, 2006).	Associations between moorland **habitat and especially vegetation structure/composition** and numbers and density of some birds (7 studies: Buchanan *et al*., 2017 [1+, EV++]; Dallimer *et al*., 2012 [2+, EV+]; Douglas & Pearce-Higgins, 2014 [2+, EV-]; Douglas *et al*., 2017 [2+, EV-]; Ludwig *et al*., 2018 [2+, EV-]; Robertson *et al*., 2017 [2+, EV++]; Roos *et al*., 2016 [2+, EV+]).	Recent evidence consistent with NEER004 finding.	Increased evidence of associations between habitat type and especially vegetation structure and composition, and breeding bird numbers, density and other bird variables.
Breeding birds: burning and/or predator control intensity	Mixed effects including positive associations between **burning and/or predator control** and	Fewer associations between **burning** and numbers and **densities** of birds apart from golden plover (2 studies: Littlewood *et al*., 2019 [2+, EV+];	New evidence suggests that predator control has a stronger effect	Increased evidence that indicates positive associations between predator control and some

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	densities of some moorland birds, particularly waders (6 studies: 5 [2+]: Picozzi, 1968; Haworth & Thompson, 1990; Smith *et al.*, 2001; Tharme *et al.*, 2001; and Pearce-Higgins & Grant, 2006; and 1 [2-]: Daplyn & Ewald, 2006). Of these, 2 studies showed higher densities of curlew (1 [2+]: Haworth & Thompson, 1990 [2+]; and 1 [2-]: Daplyn & Ewald, 2006), golden plover and red grouse (both [2+]: Picozzi, 1968; Tharme *et al.*, 2001) and curlew (1 [2+]: Haworth & Thompson, 1990 [2+]; and 1 [2-]: Daplyn & Ewald, 2006); and single studies higher densities in lapwing (Daplyn & Ewald, 2006 [2-]), redshank (Haworth & Thompson, 1990 [2+]) and ring ouzel (Daplyn & Ewald, 2006 [2-]). Weak evidence of a positive association between **burning and/or predator control** intensity and overall **diversity** of moorland breeding birds,	Douglas *et al.*, 2017 [2+, EV-]); and on **breeding success**/**post-breeding density** of red grouse (1 study: Robertson *et al.*, 2017 [2+, EV++]). No effect of **burning** on bird **species richness/diversity** (1 study: Newey *et al.*, 2016 [2+, EV++]). Mixed effects reflecting **type/management objectives of holdings**, especially grouse moors on absolute and relative abundance of species, and community composition/assemblage with waders and red grouse tending to be associated with grouse moors and corvids, merlin and passerines with other moorland (2 studies: Newey *et al.*, 2020 [2+, EV++]; Wilson *et al.*, 2021 [2++, EV++]). Stronger positive associations between **predator control** and densities of birds than burning (3 studies: Buchanan *et al*. 2017 [2+, EV++]; Douglas *et al.*, 2014 [2+, EV++]; Littlewood *et al*. 2019 [2+, EV+]).	than burning (in most cases they were not separated out in NEER004).	birds, especially some waders and red grouse, and weaker positive associations with burning compared to predator control for some species.

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	although the same study showed no relationship with **species richness** (1 study: Smith *et al*., 2001 [2+]). Mixed evidence on changes in numbers of waders associated with **burning** (1 study: Amar *et al.*, 2009 [2++]) with greater declines in golden plover under more intensive (rather than less intensive) burning management and with curlew and lapwing declining more on '*Calluna*-dominated' plots than on 'bog' plots. Moderate evidence of an increase in breeding success and numbers of curlew, golden plover, lapwing and red grouse and breeding success of meadow pipit in response to **predator control** (1 study: Fletcher *et al*., 2010 [1+]).			
Breeding birds: burning and/or predator control	A few negative associations between **burning and/or predator control** on passerine bird species (2 studies: both [2+]: Smith *et*	A few negative associations between **burning** and two bird species (2 studies: Littlewood *et al*., (2019 [2+, EV+]; Douglas *et al*., 2017 [2+, EV-]), showing	Similar findings of a few negative associations, in review update	Increased evidence of some negative associations between burning and/or

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	al., 2001; Tharme *et al*., 2001 on meadow pipit; and 1 study: [2-], Daplyn & Ewald, 2006 on skylark, wheatear and twite). However, in these studies most other species did not show significant correlations with burning/predator control.	weak negative associations with wren and red grouse, respectively. Also mixed effects reflecting type/management objectives of holdings with waders and red grouse tending to be associated with grouse moors, and corvids, merlin and passerines with other moorland (2 studies: Newey *et al*., 2020 [2+, EV++]; Wilson *et al*., 2021 [2++, EV++]).	specifically concerning burning.	predator control, mostly on passerine species.
Breeding birds: timing of egg-laying in relation to burning season	**Burning season** overlaps with **timing of first egg-laying** in some bird species and moderate evidence of earlier nesting for 8 species (1 study: Moss *et al*., 2005 [2++] based on two separate large national BTO data sets, and two studies/ reviews (Tucker, 2003 [2+]; Ratcliffe, 1990 [4+]) of smaller, preliminary national data sets). The impact on populations depends on the proportion of the population nesting on moorland likely to be subject to burning, i.e., species nesting on ground or in relatively short vegetation (Moss *et al*.,	Similar overlap of egg-laying with **burning season**, with evidence of an advancement of mean laying date across all species by about one day every eight years over a 44-year period from 1976 to 2019 (1 study: Wilson *et al*., 2021 [2++, EV++], based on re-analysis of same, updated BTO data sets (as in NEER004; and for 3 passerines on a SW site, 2 overlapped with the burning season: stonechat by six weeks: 6% of nests at the nest building stage, 13% at laying stage, 41% at incubation stage, and 1% at the nestling stage; meadow pipit breeding activity overlapped by three weeks, with 29% of nests; 14% nest building, 11% laying, and 4% incubating (1 study: Zonneveld *et al*., 2024 [2+, EV-]).	Consistent evidence on overlap of egg laying with burning season and stronger evidence of gradual advancement of egg-laying dates across most species.	Increased evidence of some overlaps of egg-laying with burning season (up to 15 April in England) with some risk of impact for a few vulnerable, early-laying, ground-nesting bird species, especially those that nest in relatively short vegetation (likely to be burnt) and have large populations on regularly burnt moorland. Revised stronger evidence of a gradual, small advancement of mean egg-laying date of moorland birds in response to climate change.

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	2005 [2++]; Glaves *et al*., 2005 [4+]; Grant *et al*., 2012 [4+]).			
Terrestrial invertebrate community	Burning indirectly influences the **invertebrate community composition** of upland peatland habitats, typically benefiting open-ground species such as ground beetles and surface-active spiders (8 studies: one [2++]: Hochkirch & Adorf, 2007; two [2+]: Eyre *et al*., 2003; McFerran *et al*., 1995; and 5 [2-]: Coulson, 1988; Curtis & Corrigan, 1990; Usher, 1992; Holmes *et al*., 1993; Stone, 2006). Not all are necessarily characteristic species of (less modified) peatlands.	Different **carabid (ground beetle) species** characteristic of different post-burn successional stages, including some species showing traits characteristic of burnt areas or areas dominated by older-aged vegetation stands (1 study: Bargmann *et al*., 2015/2016 [2+, EV-]). Similarly, **species composition of five invertebrate families** differed between burnt/cut, old-growth and wet heathland for all taxa (5 families). Soil moisture, bare soil and vegetation height density were important drivers explaining contrasting responses in richness and composition between heathland types (1 study: Byriel *et al*., 2023 [2+, EV-]).	Generally consistent with NEER004.	Increased evidence that burning influences invertebrate community composition, with different species and species-groups associated with different successional stages and vegetation types/habitats particularly in relation to soil moisture, open- and bare-ground and vegetation height/density.
Large heath butterfly	Weak evidence that too **frequent burning** is likely to render peatland sites less suitable or unsuitable for the large heath butterfly, but that occasional burning may be beneficial perhaps favouring the larval foodplant, *Eriophorum*	No new evidence.	Only evidence from NEER004.	Same evidence that too frequent burning is likely to make sites less suitable or unsuitable for large heath butterfly, though occasional burning may be beneficial.

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	vaginatum, and in reversing succession on at least some drier sites (1 study: Dennis & Eales, 1997/1999 [2+]).			
Cranefly emergence/abundance	No evidence.	New evidence that cranefly emergence/abundance is related to moisture content and hence may be affected by different vegetation management and restoration interventions (2 studies: Douglas & Pearce-Higgins, 2014 [2+, EV-]; Heinemeyer *et al*., 2019c/2023 [1+, EV-]), and evidence that it is not related to vegetation height, though taller vegetation may reduce the availability of prey for waders (1 study: Douglas & Pearce-Higgins, 2014 [2+, EV-]).	New evidence available.	New evidence that burning, cutting and restoration interventions may affect cranefly abundance and availability as avian prey.
Aquatic invertebrates (also relevant to water quality)	Moderate evidence that burning is associated 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 (3 studies:	Significant effects of burning, season and their interaction on peatland watercourse aquatic invertebrate **communities**, with significantly lower taxonomic **richness** and **diversity**. There was also a significant effect of burning on macroinvertebrate community composition, typically with reduced mayfly, and grazer and collector-gatherer feeding groups, abundance and diversity, and greater abundance of non-biting midges (3 studies: Brown *et al*., 2013/2019 [2+,	Evidence consistent.	Increased evidence that burning is associated with changes in the diversity and composition of aquatic invertebrate assemblages in upland peatland watercourses.

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
	2 [2++]: Aspray, 2012; Ramchunder *et al*., 2013/Ramchunder 2010/ Brown *et al*. 2009; and Ramchunder *et al*., 2009 [2+]).	EV+]; Johnston, 2012 [2-, EV+]; Johnston & Robson, 2015 [1+, EV+]).		
Mammal abundance	Limited evidence, no evidence statements.	New evidence that following bog restoration in the Peak District, mountain hare densities were notably higher than in neighbouring 'degraded bog', bogs managed for grouse shooting and on 'heather moorland' (1 study: Bedson *et al*., 2022b [2+, EV+]).	New evidence.	New evidence that mountain hare densities in the Peak District are higher on restored than degraded bog managed for grouse shooting and 'heather moorland'.
Reptile abundance	Limited evidence only from a review (Glaves *et al*., 2005 [4+]), no evidence statements.	New evidence for adders, which were recorded from 810 1-km squares in Scotland of which only 77 (10%) overlapped with assessed grouse moor squares. Although most grouse moor records occurred within squares with less than 20% burning, there was otherwise no clear pattern in the percentage of occupied squares with increasing burn intensity (1 study: Newey *et al.,* 2020 [2+, EV++]).	New evidence.	New evidence that in Scotland adders mostly occur in 1-km squares where grouse moor management does not take place.
Soil microbes	Limited evidence, no evidence statements.	New evidence for the microbial community varied significantly with management across fungi, bacteria and archaea, with the principal differences being between the Peatland-ES-UK managed sites (burnt, mown and uncut plots), land	New evidence.	New evidence that fungi, bacteria and archaea communities differed principally between managed sites and

Outcome	NEER004 findings	Review update findings	Comparison	Updated conclusion
		previously (pre-1950s) managed as grouse moor (Moor House NNR), and three other ‘national’ areas (Exmoor, with multiple sites reflecting differences in degree of habitat modification and time since restoration treatments. Sites in the three ‘national’ areas showed very little difference (1 study, Burn [1,2+, EV+]).		modified sites undergoing restoration.

The effects of managed burning of upland peatlands on carbon balance

Comparison of findings

12.7 A comparison of the evidence on the effects of managed burning on upland peatland carbon balance in NEER004 and this update is given in Table 30. A similar number of studies were considered in the update as in NEER004, and although the update had more UK-based studies, the geographic bias was also similar.

12.8 For several outcomes, evidence from the recent studies supported the findings of NEER004, including the overall conclusion that there is strong evidence that burning affects various aspects of the carbon cycle on upland peatlands. Recent evidence also supported the findings that a significant amount of carbon is lost in combustion, leading to reduced aboveground carbon stocks, and that near-surface soil temperatures are higher after burning. Both NEER004 and the update found inconsistent evidence of burning effects on methane, soil DOC and overall carbon budget.

12.9 Findings from recent studies were partly consistent with NEER004, including evidence that vegetation can influence DOC production and export. However, whilst NEER004 identified *Calluna* as a key driver of DOC production, it was not always associated with the greatest DOC concentration in soil or stream water in recent studies. One study in NEER004 showed greater POC flux after burning, and while three of four studies in the update had similar findings, one did not find an overall difference between cutting and burning at the catchment scale. Meanwhile, one study in NEER004 found that burning reduced peat accumulation, and whilst one new study supported this finding, another reported a positive relationship between peat accumulation and burning (though there has been published discussion of the limitations of both recent studies). The latter study reported greater belowground carbon stocks on sites which were slightly more frequently burned historically (on average 23- or 25- compared to 28-year rotations), in contrast to the finding from one study in NEER004 that burning reduces belowground carbon stocks.

12.10 One study reviewed for NEER004 provided evidence that burning initially increased photosynthesis, respiration and net CO_2 uptake associated with post-burn regrowth, whereas inconsistent effects of burning on these fluxes were found in recent studies at different times post-burn. For watercourse DOC, evidence from two recent studies found no effect of fire compared to the strong evidence of increase identified in five studies reviewed for NEER004. Finally, recent studies provided evidence for additional outcomes relating to the significance of charcoal and pyrogenic carbon in carbon cycling, and the effects of burning on soil respiration and net GHG exchange.

Table 30. Comparison of evidence assessed on the effects of managed burning on upland peatland carbon balance in NEER004 and this review update. Individual study findings are described in Section 6.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Number and location of studies	18 studies 15 UK-based (mostly Pennines, 9 Moor House, 4 Hard Hill experiment).	20 studies 20 UK-based (mostly Pennines, 4 Moor House, 3 Hard Hill experiment).	Similar Number and geographic bias.	The available evidence is likely to be applicable to the Pennines and possibly wider UK.
Overall conclusion	Strong evidence that burning affects various aspects of carbon cycling (11 studies, see below).	Evidence that burning affects various aspects of carbon cycling (see below for studies).	Recent evidence consistent with NEER004 finding.	Evidence that burning affects various aspects of carbon cycling.
Aboveground carbon (C) stocks	Burning reduces aboveground C stock (1 study: Ward *et al*., 2007 [1++]).	Burning reduces aboveground C stock, which can then increase for at least several decades after burning (6 studies: Alday *et al*., 2015 [1/2+, EV-]; Clay *et al*., 2015 [2+, EV-]; Heinemeyer *et al*., 2019c [1,2+, EV-]; Santana *et al*., 2016 [2+, EV+]; Ward *et al*., 2012 [1,2+, EV-]; Worrall *et al*., 2013a [1,2+, EV-]).	Recent evidence consistent with NEER004 finding.	Burning reduces aboveground C stock.
Belowground C stocks	Burning reduces belowground C stock (1 study: Ward *et al*., 2007 [1+]).	Greater C stocks associated with more frequent burn history (1 study: Heinemeyer *et al*., 2018 [2+, EV-]).	Recent evidence not consistent with NEER004 finding.	Inconsistent evidence on belowground C stock.
Combustion	C is lost in combustion (3 studies: Allen *et al*., 2013 [2+]; Clay & Worrall, 2011 [2+]; Farage *et al*., 2009 [2-]).	C (76–80%) is lost in combustion (4 studies: Clay *et al*., 2015 [2+, EV-]; Heinemeyer *et al*., 2019c [1,2+, EV-]; Santana *et al*., 2016 [2+, EV+]; Worrall *et al*., 2013a [1,2+, EV-]).	Recent evidence consistent with NEER004 finding.	Evidence that [aboveground] carbon is lost in combustion.
Pyrogenic carbon / charcoal	Char production contributed 4.3% of biomass lost on combustion and, though it may degrade over time, it has	Burn severity impacts char characteristics including subsequent degradation (2 studies: Kennedy-	New evidence available.	Charred material has a role in post-burn carbon cycling

Outcome	NEER004	2023 update	Comparison	Updated conclusion
	a long residence time (1 study: Clay & Worrall, 2011).	Blundell, 2020 [1+, EV-]; Worrall *et al*., 2013a [1,2+, EV-]).		which can be influenced by burn severity.
NEE (Net Ecosystem Exchange) of CO_2	Burning created a greater net sink for CO_2 by increasing photosynthesis to a greater extent than respiration (1 study: Ward *et al*., 2007 [1++]).	5 recent studies. 2 report that burning increased NEE CO_2 resulting in a net carbon source after burning (Grau-Andrés *et al*., 2019b [1+, EV-]; Heinemeyer *et al*., 2019c [1+, EV-]). 1 found no effect of burning (Clay *et al*., 2015 [2+, EV-]). 2 found effects of vegetation (Dixon *et al*., 2015 [2+, EV-]; Ward *et al*., 2013 [1,2+, EV-]).	Recent evidence not consistent with NEER004 finding.	Inconsistent evidence of how burning affects net CO_2 flux
GPP (Gross Primary Productivity)	Burning increases rate of gross photosynthesis (1 study: Ward *et al*., 2007 [1++], c.9 years after burning).	4 recent studies considered GPP. 2 found a reduction (less negative flux) (Grau-Andrés *et al*., 2019b [1+, EV-] 4 months–2 years post burn; Heinemeyer *et al*., 2019c [1+, EV-] 2–3 years post-burn) and 2 found no difference following burning (Clay *et al*., 2015 [2+, EV-] 1–13 years post-burn; Ward *et al*., 2012 [1,2+, EV-] 18 months post-burn).	Recent evidence not consistent with NEER004 finding.	Inconsistent evidence of how burning affects photosynthesis
Ecosystem respiration	Burning increases CO_2 rate of respiration (1 study: Ward *et al*., 2007 [1++], c.9 years after burning).	Burning can reduce ecosystem respiration (2 studies: Grau-Andrés *et al*., 2019b [1+, EV-] 4 months–2 years post burn; Heinemeyer *et al*., 2019c [1+, EV-] 2–3 years post-burn), though 1 study found no consistent effect (Clay *et al*., 2015 [2+, EV-] 1–13 years post-burn). 1 study found plant species effects on respiration (Ward *et al*., 2013 [1,2+, EV-]).	Recent evidence not consistent with NEER004 finding.	Inconsistent evidence of how burning effects respiration.
Soil temperature	Burning increases soil temperature (2 studies: Orwin & Ostle, 2012 [1+]; Yallop *et al*., 2008)/White *et al*., 2004 [2+]).	Burning increases soil temperature in early (1–4) years following burning (3 studies: Brown *et al*., 2014 [2+, EV+];	Recent evidence consistent with	Evidence that burning increases soil temperature.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
		Grau-Andrés *et al*., 2017a [1+, EV-]; Heinemeyer *et al*., 2019c [1+, EV-]).	NEER004 finding.	
Soil respiration	No studies.	Evidence of lower rates of soil respiration after burning (1 study: Heinemeyer *et al*., 2019c [1+, EV-] 1–8 years post burn) and that vegetation can affect soil respiration (1 study: Dunn *et al*., 2016 [2-, EV-]).	New evidence available.	Evidence that burning can decrease soil respiration.
Methane	Mixed impacts of burning on methane/ methanotroph activity (3 studies: Ward *et al*., 2007 [1++]; Chen *et al*., 2008 [2++]; Gray & Levy, 2009 [4+]).	3 studies provided inconsistent evidence of burning effects on methane fluxes, with findings of a decrease (Heinemeyer *et al*., 2019c [1+, EV-]), increase (Grau-Andrés *et al*., 2019b [1+, EV-]) and no effects (Clay *et al*., 2015 [2+, EV-]) following burning. 2 studies found effects of vegetation composition (Dunn *et al*., 2016 [2-, EV-]; Ward *et al*., 2013 [1,2+, EV-]).	Recent evidence consistent with NEER004 finding.	Inconsistent evidence of burning impacts on methane fluxes
Net GHG (CO_2, CH_4 and N_2O fluxes)	No studies	1 study (Heinemeyer *et al*., 2019c [1+, EV-]) found that burning increased net GHG emissions and that burned and cut plots were GHG sources compared to longer unburned comparisons which were GHG sinks.	New evidence available.	Evidence that burning can impact net GHG emissions, potentially switching peatlands from sink to source.
Soil DOC	Inconsistent evidence that burning affects DOC and water colour at the plot scale (4 studies: Ward *et al*. 2007 [1+]; Worrall *et al*., 2007/Clay *et al*., 2009b [1+]; Clay *et al*., 2012 [2+]; Worrall *et al*., 2010 [2-]). The differing responses are likely to reflect differences in time since burning at the time of sampling, especially as most of these studies were from the Hard Hill	Inconsistent evidence that burning effects DOC and water colour at the plot scale. No difference (2 studies: Grau-Andrés *et al*., 2019b [1+, EV-]; Heinemeyer *et al*., 2019c [1+, EV-]) or more DOC in longer-unburned plots (2 studies: Clay *et al*., 2015 [2+, EV-]; Qassim, 2015 [1+, EV-]). Some evidence of a short-term increase after wildfire (1 study: Brown *et al*., 2014	Recent evidence consistent with NEER004 finding.	Inconsistent evidence that burning effects DOC and water colour at the plot scale

Outcome	NEER004	2023 update	Comparison	Updated conclusion
	expt. at different times post-burn. Clay *et al*. (2009b) showed a peak in DOC and water colour in the weeks after a new burn c.f. no effect of burn treatments ten years after the last burn.	[2+, EV+] reported in Blundell *et al*., 2013).		
Watercourse DOC	Strong evidence that burning leads to increased DOC and water colouration at the watercourse or catchment scale (4 multiple catchment studies: Yallop & Clutterbuck, 2009 [2++]; Clutterbuck & Yallop, 2010 [2++]; Yallop *et al*., 2010 [2++]; Yallop *et al*., 2008 [2+]; a modelling study: Grayson et al., 2012 [2+]; a lab study: Mitchell & McDonald 1995/McDonald *et al*., 1991 [2-]) and a critical synthesis review: Holden *et al*., 2011/2012 [2++]). 2 other single catchment studies did not show such effects (O'Brien *et al*., 2005 [2-]; Chapman *et al*., 2010 [2-]), though these studies did not map the extent of recent burning or deep peat (Yallop *et al*., 2011).	No difference in watercourse DOC after recent burning (in burnt EMBER catchments) or a wildfire at an EMBER burn study site (1 study: Brown *et al*., 2014 [2+, EV+] reported in Blundell et al., 2013) or in sub-catchments with continued burning and cutting treatments on previously burned sites (1 study: Heinemeyer *et al*., 2019c [1+, EV-]).	Recent evidence not consistent with NEER004 finding.	Overall, still strong evidence that burning leads to increased DOC/water colouration in watercourses, though such effects were not found in 4 studies of single or in 1 case, 3 sub-catchments.
Vegetation effects on DOC	Moderate evidence (2 studies: Beharry-Borg, 2009 [2+]; 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.	Evidence of vegetation effects on DOC, though *Calluna* was not always the most important driver (4 studies: Dunn *et al*., 2016 [2-, EV-]; Ritson *et al*., 2016 [1+, EV+]; Parry *et al*., 2015 [2+, EV+]; Ward *et al*., 2013 [1,2+, EV-]).	Recent evidence partly consistent with NEER004 finding.	Evidence that plant species composition can affect DOC flux.
Erosion / POC	Moderate evidence that managed burning can result in erosion and reduction in the level of the soil surface (1 study: Kinako & Gimingham, 1980 [2+]).	2 studies observed higher POC fluxes from more recently burned plots (Brown *et al*., 2014 [2+, EV+]; Clay *et al*., 2015 [2+, EV-]); 1 study modelled higher erosion rates from 'intensively managed' peatlands (Li *et al*., 2017	Recent evidence partly consistent with NEER004 finding.	Moderate evidence that burning increases erosion and watercourse POC, with 4/5 studies suggesting burning leads to an increase.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
		[2+, EV+]) and 1 study found no overall difference in POC export between burned and cut catchments (Heinemeyer *et al*., 2019c [1+, EV-]), though not compared with no management.		
Peat accumulation	Burning reduces peat accumulation (1 study: Garnett *et al*., 2000 [1+]).	Inconsistent evidence of burning impacts on peat/carbon accumulation with reports of both a reduction (1 study: Marrs *et al*., 2019 [1,2+, EV-]) and an increase (1 study: Heinemeyer *et al*., 2019c [1+, EV-]).	Recent evidence partly consistent with NEER004 finding.	Inconsistent evidence of how burning impacts peat accumulation
Carbon balance	Strong evidence that burning affects the processes controlling C budgets, but inconsistent evidence with predictions of both positive and negative overall effects (5 studies: Garnett *et al*., 2001 [2++]; Clay *et al*., 2010b [2++]; Worrall *et al*., 2010 [2+]; Clay & Worrall, 2011 [2+]; Farage *et al*., 2009 [2-]).	Estimates of both positive (1 study: Clay *et al*., 2015 [2+, EV-]) and negative (1 study: Heinemeyer *et al*., 2019c [1+, EV-]) effects of burning on C balance.	Recent evidence consistent with NEER004 finding.	Inconsistent evidence of how burning affects overall upland peatland carbon balance.

The effects of managed burning of upland peatlands on water quality and hydrology

Summary of similarities and differences

12.11 A comparison of the evidence effects of managed burning on water quality and hydrology outcomes between NEER004 and the current update is given in Table 31. A similar number of studies were considered for this sub-question in the update as in NEER004, and the geographic bias of these studies was also similar. The recent studies were broadly consistent with the findings of NEER004, with mixed evidence of burning effects on soil water DOC and water pH levels, evidence of changes to various aspects of water chemistry, and increased frequency of runoff after burning. However, for water table, seven recent studies found various effects of burning compared to the evidence of shallower water tables after burning identified in two studies reviewed for NEER004. Similarly, for watercourse DOC, evidence from three recent studies found no effect or mixed effects of burning compared to the increase identified in five studies reviewed for NEER004 (see carbon section, para 6.44 - 6.45, Table 30).

12.12 The new studies also provided evidence for additional outcomes including burning effects on bulk density and hydrophobicity as well as plant species effects on DOC treatability. Furthermore, whereas NEER004 found no evidence relating to downstream flow in watercourses draining burned catchments, three recent studies suggested that burning can affect flow, with a further study suggesting that vegetation density may be an important driver.

Table 31. Comparison of evidence assessed for the water sub-question of NEER004 and this update. Individual study findings are described in section 7. Evidence on soil and watercourse DOC and colouration is summarised in Table 30 on carbon balance and storage.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Number and location of studies	22 studies. All UK (all including England). Most Pennines.	22 studies . 21 UK (19 including England). Most Pennines, some other regions.	Similar number and geographic bias.	The available evidence is likely to be applicable to the Pennines and possibly wider UK.
DOC treatability	No studies.	1 study (Ritson *et al*., 2016 [1+, EV+]) showed that the plant species origin of DOC has implications for water treatment, with *Calluna*-derived DOC most difficult to remove and most likely to form chloroform during treatment.	New evidence available.	Evidence that changes to peatland vegetation may influence DOC treatability.
Water chemistry	Weak evidence of changes in soil water/runoff chemistry (1 study: Worrall & Adamson, 2008/Clay *et al*., 2010a [1+]).	Evidence of changes to water chemistry including stream water (2 studies: Brown *et al*., 2014 [2+, EV-]; Heinemeyer *et al*., 2019c [1+, EV+]); leaching of cations from ash (1 study: Noble *et al*., 2017 [1+, EV-]); and introduction of polycyclic aromatic hydrocarbons (PAHs) from burning (1 study: Vane and others, 2013 [2+, EV-]).	Recent evidence is consistent with NEER004 finding.	Evidence that burning influences various aspects of soil, runoff and stream water chemistry.
Soil water, runoff and stream water pH	Weak evidence from 3 laboratory studies (McDonald *et al*., 1991 [2++]; Miller, 2008 [2++]; Allen, 1964 [2+]) that burning increases pH. Weak evidence from 1 study (Worrall & Adamson, 2008/Clay *et al*., 2010a [1+]) of lower soil water pH following burning.	Inconsistent evidence of changes to pH after burning with reports of no difference in soil water (2 studies: Brown *et al*., 2014 [2+, EV+]; Heinemeyer *et al*., 2019c [1+, EV-]); and no difference (Heinemeyer *et al*., 2019c [1+, EV-]) or a decrease (Brown *et al*., 2014 [2+, EV+]) in stream water.	Recent evidence is consistent with NEER004 finding (of inconsistent evidence).	Inconsistent evidence of burning effects on water pH.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Water table	Weak evidence of initially shallower water table following burning (2 studies: Worrall *et al*., 2007/Clay *et al*., 2009a [1+]; Worrall *et al*., 2010 [2-]) and contra, weak evidence of oscillating, deeper water table following burning (1 study: Yallop *et al*., 2008/White *et al*., 2004 [2+]).	Inconsistent recent evidence of effects on water table: 1 study showed deeper water tables after burning (Heinemeyer & Swindles, 2018 [2+, EV-]; 2 large studies using different methods showed lower water tables following burning, followed by a gradual recovery over time (Brown *et al*., 2014 [2+, EV+] including Blundell *et al*., 2013 in relation to a wildfire; Heinemeyer *et al*., 2019c [1+, EV-]); and 2 studies showed shallower water tables following burning (Qassim, 2015 [1+, EV-]; Grau-Andrés *et al*., 2019b [1+, EV-]). These differences may reflect different timescales of study post burn and change in relation to weather and season and how this is accounted for, for example, by using controls.	Recent evidence is not fully consistent with NEER004 finding.	Inconsistent evidence of burning effects on water table depth.
Runoff	Moderate evidence (2 studies: Clay *et al*., 2009a [1+]; Clay *et al*., 2012 [2+]) of more frequent runoff after burning.	1 study showed more frequent runoff after burning (Brown *et al*., 2014 [2+, EV+]).	Recent evidence is consistent with NEER004 finding.	Evidence of more frequent runoff after burning.
Bulk density	No studies.	1 study showed an increase after burning (Brown *et al*., 2014 [2+, EV+]). 1 study showed no difference in recently burned plots, though higher bulk density was associated with charcoal in peat cores from the same sites (Heinemeyer *et al*., 2019c [1+, EV-]).	New evidence available.	Inconsistent evidence of burning impacts on bulk density.
Saturated hydraulic conductivity and macropore flow	Evidence of lower saturated hydraulic conductivity and lower macropore flow (as contribution to overall infiltration) in recently burnt plots (with no difference with wildfire) and indication of recovery	Similar evidence from same study of lower steady state infiltration rates, proportion of flow moving through macropores and hydraulic conductivity in recently burned plots (Brown *et al*., 2014 [2+, EV+]).	Recent evidence is consistent with NEER004 finding.	Evidence of reduced steady state infiltration rates, proportion of flow moving through macropores and hydraulic conductivity after recent burning.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
	within 2 decades (1 study: Holden *et al*., 2013 [1+], 3 EMBER sites).			
Hydrophobicity	No studies.	2 studies found that fire increased hydrophobicity at the peat surface (Wu *et al*., 2020 [1+, EV-]; Turner & Swindles, 2012 [2+, EV-]).	New evidence available.	Evidence that burning can increase peat surface hydrophobicity.
Watercourse flow	No studies.	3 studies suggest that burning can affect downstream flow. 1 showed increased flow volume from burned catchments at 2 of 3 study sites and modelled higher downstream river levels as a result of burning (Heinemeyer *et al*., 2019c [1+, EV-]). 1 observed greater hydrograph lag times and a flashier response to large storm events (Brown *et al*., 2014 [2+, EV+]). 1 modelling study suggested that burning can increase flow peaks (Gao *et al*., 2017 [1+, EV-]). A further modelling study showed an impact of vegetation density on flow peaks (Gao *et al*., 2016 [1+, EV-]).	New evidence available.	Evidence that burning can affect flow in watercourses draining the catchment.

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

Comparison of findings

12.13 A comparison of the evidence for outcomes relating to the effects of differences in the severity, frequency, scale, location and other characteristics of burns between NEER004 and the current update is given in Table 32. A greater number of studies were considered for this sub-question in the update than in NEER004, with slightly wider geographic spread. Findings from NEER004 were generally consistent with the newer evidence. Both reviews considered studies from the Hard Hill experiment, with both finding greater abundance of *E. vaginatum* with increasing burn frequency, and the newer study also reporting effects on other species (Table 32). Evidence of the effects of pre-burn vegetation on fire severity was also consistent, with evidence that fuel load and structure are important characteristics, and additional evidence in the review update that fuel moisture also plays a role.

12.14 The review update found new evidence for several outcomes relating to this sub-question including evidence that burn severity affects vegetation species abundance and can also temporarily affect *Sphagnum* cell viability and photosynthetic capacity. Burn severity may also affect aspects of carbon cycling, though there was no evidence of impacts on net carbon fluxes. There was, however, evidence that burn frequency can affect carbon accumulation and storage. Finally, evidence from one wildfire study suggested that fire severity interacts with moss species and microtopography to influence water availability at the peat surface following fire.

Table 32. Comparison of evidence assessed for the burn severity sub-question of NEER004 and this update. Individual study findings are described in section 8110.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Number and location of studies	5 studies, 4 UK, 3 Scotland, 1 England (Peak District).	8 studies, 7 UK, 4 England (3 Hard Hill and 1 Peak District), 2 Scotland, 1 England and Scotland.	More studies, slightly wider geographic coverage.	Findings likely to be applicable to the Pennines and potentially wider England and Scotland.
Effects of burn severity on vegetation	No evidence.	Evidence that fire severity affects vegetation, with higher severity benefiting shrubs including *Calluna* (2 studies: Grau-Andrés *et al*., 2017a [1+, EV-], 2019b [1+, EV-]), acrocarpous mosses (2 studies: Grau-Andrés *et al*., 2019b [1+, EV-]; and graminoids (1 study: Grau-Andrés *et al*., 2019b [1+, EV-]), but leading to lower abundance of pleurocarpous mosses (1 study: Grau-Andrés *et al*., 2019b [1+, EV-]) and damage to *Sphagnum* cells (2 studies: Grau-Andrés *et al*., 2019b [1+, EV-]; Noble *et al*., 2019a [1,2++, EV+]).	New evidence available.	Evidence that fire severity affects vegetation composition and function.
Effects of burn frequency on vegetation	Moderate evidence that frequency of burning affects vegetation composition and structure (more frequent = increase in *Eriophorum vaginatum*, less frequent = increase in *Calluna vulgaris*) (1 study: Hard Hill experiment [1++]).	Evidence from 1 study (Milligan *et al*., 2018 [1++, EV-] updating the Hard Hill experiment findings in NEER004) that frequency of burning affects vegetation composition and structure (more frequent = greater abundance of *E. vaginatum*, *Campylopus paradoxus*, liverworts and *Sphagnum* spp. and shorter vegetation).	Recent evidence consistent with NEER004 (both used 1 study from Hard Hill experiment, Moor House NNR).	Evidence that frequency of burning affects vegetation composition and structure.
Effects of burn severity on carbon	No evidence.	Evidence that burn severity can affect soil thermal regime (1 study: Grau-Andrés *et al*., 2017a [1+, EV-]) and lability of pyrogenic carbon (1 study: Kennedy-Blundell, 2020 [1+, EV-]) but no evidence of effects on GHG	New evidence available.	Some evidence that burn severity may affect aspects of carbon cycling, but no evidence

Outcome	NEER004	2023 update	Comparison	Updated conclusion
		or DOC fluxes (1 study: Grau-Andrés *et al*., 2019b [1+, EV-]).		of net change to gaseous or fluvial carbon fluxes.
Effects of burn frequency on carbon	Evidence from modelling that more frequent burning can increase carbon loss (though patterns were modified when wildfire frequency was included in models) (1 study: Allen *et al*., 2013 [2+, EV-]).	Evidence that apparent carbon accumulation rate can decrease with more frequent burning (1 study: Marrs *et al*., 2019 [1,2+, EV-]).	New evidence available.	Evidence that frequency of burning can affect carbon accumulation and storage.
Effects of fire severity on water	No evidence.	No evidence of differences in peat moisture content after different burns of different severity or temperature (2 studies: Grau-Andrés *et al*., 2019b [1+, EV-]; Noble *et al*., 2019a [1,2++, EV+]).		
Effects of vegetation on fire severity	Moderate evidence that fuel load and structure influence fire behaviour and severity (3 linked studies: Davies, 2005; Davies *et al*., 2010a; Davies & Legg, 2011 [all 2+]; and one review: Legg & Davies, 2009 [4+]).	Evidence that fuel load and structure influence fire severity (2 studies: Davies *et al*., 2016a [2+, EV+]; Noble *et al*., 2019a [1,2++, EV+]) and that fuel moisture also has an impact (2 studies: Davies *et al*., 2016a [2+, EV+]; Grau-Andrés *et al*. 2019b [1+, EV-]).	Recent evidence consistent with NEER004.	Evidence that fuel load, structure and moisture influence fire severity.

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

Comparison of findings

12.15 A comparison of the evidence for outcomes relating to burning and grazing between NEER004 and the current update is given in Table 33. Fewer studies were considered for this sub-question in the update than in NEER004, and both recent studies were based on the same burning and grazing experiment. While one new study did not report any interactions between burning and grazing, the other (which monitored a wider range of vegetation outcomes over a much longer time frame) found interactive effects on various vegetation community characteristics and the abundance of several plant species. This is consistent with the findings of NEER004 that interactions between grazing and burning occur, although the individual species affected were different.

Table 33. Comparison of evidence assessed for the grazing interaction sub-question of NEER004 and this update. Individual study findings are described in section 9.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Number and location of studies	11 studies, 9 primary. All UK, 8 including England (most Pennines), 4 Moor House NNR; 2 Scotland.	2 studies, both Hard Hill experiment, Moor House NNR	Fewer studies and less geographic spread in update	The available evidence is likely to be applicable to specific sites and possibly the Pennines.
Vegetation	Some interactions between burning and grazing including increased bare ground and increased grazing of *Rubus chamaemorus* after burning (1 study: Lee *et al*., 2013b/ Hard Hill main experiment [1++]; Taylor & Marks, 1971 [1+]).). In another grazing and burning experiment at Moor House NNR, heavy sheep grazing resulted in a loss of *Calluna* and rapid increase in *Eriophorum vaginatum* irrespective of whether burning occurred or not (1 study: Rawes & Williams, 1973/ Rawes & Hobbs, 1979 [2+]). Moderate evidence that burning results in increased grazing of *Molinia caerulea* (3 studies: Miles, 1971; Ross *et al*., 2003; and Marrs *et al*., 2004, all [1+]) but that inadequate grazing can lead to *Molinia* dominance (Anderson *et al*., 2006 [4+]).	Evidence of interactions between burning and grazing affecting the trajectory of vegetation community change as well as abundance of several species including vascular plants (*Calluna*, *Empetrum nigrum*), mosses (*Sphagnum* spp., *Hypnum jutlandicum*, *Campylopus paradoxus, Pohlia nutans*), liverworts (*Calypogeia muelleriana*, *Cephalozia. bicuspidata, Lophozia ventricosa)* and lichen spp. (1 study: Milligan *et al*., 2018 [1++, EV-] updating the Hard Hill experiment findings). At the same site, no interaction with grazing was identified in a separate, one-off survey of *Sphagnum* spp. (1 study: Noble and others (2018a [1,2++, EV-]).	Recent evidence is consistent with NEER004 findings that a burning and grazing interaction can affect vegetation species abundance and change over time.	Evidence that burning and (mostly low intensity sheep) grazing can interact to affect several vegetation-related variables.

	Weak evidence that grazing can lead to a prolonged phase of graminoid dominance after burning (2 studies: Currall, 1981 [2+]; Ward *et al*., 2007 [1+]).			
Burning and grazing intensity	Evidence that the effects of and interactions between grazing and burning depend on burning rotation length, extent and location as well as stocking intensity and the seasonal timing of grazing (Tucker, 2003 [4+]).	Evidence that burning rotation length can affect vegetation response to grazing (1 study: Milligan *et al*., 2018 [1++, EV-]).	Recent evidence is at least partly consistent with NEER004 findings.	The characteristics of both grazing and burning regimes can influence how they interact to affect vegetation outcomes.

The relationship between managed burning of upland peatlands and wildfire risk, hazard, occurrence, severity, extent and habitat resilience

Comparison of findings

12.16 A comparison of the evidence on the relationship between managed burning of upland peatlands and wildfire risk, hazard, occurrence, severity, extent and habitat resilience between NEER004 and NEER014, and the current update is given in Table 34. Fewer studies were included in this update reflecting the relatively short time since NEER014 was published (2020). Though many were from the UK more were from elsewhere reflecting the higher incidence of wildfires in some other parts of the world, and hence associated research and prevention/mitigation measures.

12.17 Where there was more recent evidence it tended to be consistent with the larger body of evidence included in NEER004 and especially NEER014. This included consistent findings of a spring peak in wildfire in the UK and NW Europe, a proportion of wildfires resulted from managed burns escaping control and that pristine, less modified, restored and wetter peatlands were less susceptible and more resilient to wildfire, though still at risk in dry conditions.

Table 34. Comparison of evidence assessed in the burning and wildfire sub-question of NEER004 and the NEER014 wildfire review with this update. Individual study findings are described in Section 10. Findings relate to UK studies unless stated otherwise and to NEER014 unless NEER004 mentioned.

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
Number and location of studies	7 evaluated studies in NEER004 (4 primary). All from UK (all in N England or Scotland). Total of 174 evaluated studies in NEER014, though its scope was wider, so many were not relevant to the narrower wildfire question in this update with 30 studies included, just 8 from the UK.	8 more recent evaluated studies (all primary) reflecting the relatively short time since NEER014 was published (2020). 4 studies from and another included the UK.	Fewer studies from the UK (16) than outside the UK (29) reflecting the higher incidence of wildfire, and research and prevention/mitigation measures.	Evidence likely to be applicable to upland peatlands in England.
Seasonal timing compared with the 'burning season'	Strong evidence of wildfire peaks in **summer** and especially **spring** in the UK, the latter including part of the 'burning season' (up to 15 April in English uplands) (8 studies: 1 [2++]: de Jong *et al*., 2016; 6 [2+]: Alberston *et al*. 2009; McMorrow *et al*., 2009; Jollands *et al*., 2011; Krivstov & Legg, 2011; Davies & Legg, 2016; NEER014, Appendix 2 and Figure 3; 1 [3+]: Martin, 2018).	Similar evidence of wildfire peak in **spring** with a lower secondary peak in **summer** (Perry *et al.* [2+, EV++]; Gagkas *et al.* [3+, EV+] Cardil *et al.* [2+, EV+]). Evidence that this pattern differs from the Mediterranean fire season, which experiences a minor peak in the spring but a stronger peak towards July-September (Cardil *et al.* ([2+, EV+]).	Recent evidence consistent with NEER004/014 findings.	Evidence of wildfire peaks in summer and especially in spring the latter of which overlaps in part with the burning season in England.
Seasonal timing in uplands compared with. lowlands	Strong evidence of a difference in the **seasonal pattern** of wildfires between **lowland** and upland **areas** in England with a higher percentage in the **uplands** in spring than in the lowlands where there is a more even spread	No evidence.	No recent evidence.	Evidence of a difference in the seasonal pattern of wildfires between lowland and upland areas in England with

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	between spring and summer (2 national studies: Forestry Commission, 2019b [2++]; NEER014, Appendix 2 and Figure 4 [2+]).			a higher percentage in the uplands in spring than in the lowlands
Monthly timing	Strong evidence of peaks in the uplands in **March and April** compared to summer, and especially autumn and winter, months (2 national data sets: 1 [2++]: Forestry Commission, 2019b; 1 [2+]: NEER014 Appendix 2 and Figure 4 [2+]; and 1 case study [3+]: Martin, 2018).	Evidence of peaks in March and April in the UK, Perry *et al.* [2+, EV++], Gagkas *et al.* [3+, EV+] Cardil *et al.* [2+, EV+]. Gagkas *et al.* [3+, EV+], also found a later peak in May in Scotland.	Recent evidence consistent with NEER004/014	Evidence of wildfire peaks in English uplands in March and April.
Specific ignition sources	Strong evidence from the same recent English wildfire data set (NEER014 Appendix 2 [2+, EV++] and Figure 6) that, in the minority of cases when a more specific cause was assigned (382, only 12% of all fires), the main causes were '**campfires**' (49%), **management burns** (15%), **barbeques** (10%), and 'reignited' fires and military training (both 5%) with no other causes greater than 3%.	No evidence	No new evidence.	Evidence that that, in the minority of cases when a more specific cause was assigned, the main causes were campfires (49%), management burns (15%), barbeques (10%), and 'reignited' fires and military training (both 5%) with no other causes greater than 3%.
Specific causes in uplands compared with lowlands	Moderate evidence from the same recent English wildfire data set (NEER014 Appendix 2 [2+, EV++] and Figure 6) of a difference in the main causes in the **uplands** where the most were assigned to **managed burns** escaping control (68%), followed by	No evidence.	No new evidence.	Evidence of a difference in the main causes in the uplands where the most were assigned to managed burns escaping control (68%),

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	campfires (9%) and barbeques (8%), and the **lowlands** where most were due to camp fires (56%), barbeques (11%), 'reignited' fires (8%) and **managed burns** (8%), though more of the fires with specific causes assigned (84%) were in the lowlands.			followed by campfires (9%) and barbeques (8%), and the lowlands where most were due to campfires (56%), barbeques (11%), 'reignited' fires (8%) and managed burns (8%).
Managed burns escaping control	Strong evidence that **managed burns** escaping control cause a proportion of wildfires, particularly in the uplands (7 studies from 4 national and 3 regional/ local data sets: de Jong *et al*., 2016 [2++, EV++]; 3 [2+]: Luxmoore, 2018 [EV++]; Moors for the Future, 2009 [EV+]; NEER014 Appendix 2 [EV++]; 3 [3+]: Legg *et al*., 2006 [EV+]; Worrall *et al*., 2011 [3+]; Martin, 2018 [EV-]). These give a range for the proportion of wildfires resulting from escaped managed burns (where a specific cause assigned) was between 15% and 60%, or 24–68% if data from the lowlands (where managed burning is much less common) are excluded, though the studies cover different UK geographical areas and periods.	Cosgrove [3-, EV-] in Cairngorm, Scotland found that whilst 93% of wildfires were supposed to be caused by human activity, 29% of these were caused by muirburn getting out of control.	Recent evidence consistent with NEER004/014 findings.	Evidence that a proportion of wildfires, especially in the uplands, are caused by managed burns getting out of control.
Fire behaviour and severity: fuel load and structure	NEER004 and NEER014 combined. Moderate evidence that fuel load and vegetation structure, and hence vegetation and habitat type (though	No evidence.	No new evidence.	Evidence that fuel load and vegetation structure, and hence vegetation and habitat

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	most evidence relates to *Calluna*-dominated vegetation), are critical factors in fire behaviour in UK peatlands and heathlands, 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, but are less well understood (3 primary studies: 4 [2+]: Davies, 2005; Albertson *et al*., 2010; Davies *et al*., 2010; Davies & Legg 2011; and 4 reviews: Davies *et al*., 2008 [2+]; Legg & Davies 2009 [3+]; McMorrow *et al*., [2+]).			type are critical factors in fire behaviour in UK peatlands and heathlands, particularly in fireline intensity and rate of spread, (although residence time and depth of penetration of lethal temperatures into the soil are also important in determining severity but are less well understood).
Fire behaviour and severity: vegetation and habitat types	NEER014. Moderate evidence that fire severity (including ground fuel consumption, ground heating and changes in post-fire soil thermal dynamics) vary by habitat/vegetation type in the UK (3 primary studies: Hudspith *et al*., 2014; Grau-Andrés *et al*., 2018/2019b [all 2++]) and elsewhere, e.g., Canada (Camill *et al*., 2009 [2++]). This includes moderate evidence that in the UK, *Calluna* dry heath and tree-dominated sites suffer more severe burning than bog, flushes/fens and bog woodland (Hudspith *et al*., 2014 [2++]; Grau-	No evidence.	No new evidence.	Evidence that fire severity varies by habitat/vegetation type in the UK. This includes evidence that in the UK, *Calluna* dry heath and tree-dominated sites suffer more severe burning than bog, flushes/fens and bog woodland.

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	Andrés, 2016 [1,2++]; and Grau-Andrés *et al*., 2018/2019b [1,2++]).			
Fire severity: relationship with time since managed burning	No evidence.	Evidence of significantly higher cover of bare ground following a wildfire at a Peak District blanket bog site in younger *Calluna* stands (i.e., more recently burned prior to the wildfire), with a mean of 78% post-wildfire bare ground cover across 0–6, 7–15 and 16–29 years post-burn classes compared with 35% in 30–40 years and 26% in >40 years post-managed-burning (1 study: Swindell, 2017 [2+, EV-]).	New evidence.	Evidence of higher cover of bare ground following a wildfire at a Peak District blanket bog site in younger *Calluna* stands (i.e., more recently burned prior to the wildfire).
Effect of degree of modification and water table on habitat vulnerability and resilience	Moderate evidence from continental Europe and North America that 'pristine' and 'less modified' peatlands, especially where the water table is high, are less vulnerable to severe, smouldering fires (2 studies; Granath *et al*., 2016 [2++]; Turetsky *et al*., 2014 [2+]).	Evidence that that drained and degraded peatlands are more susceptible to damage and carbon loss following fire than pristine or restored peatlands (2 studies: Kirkland *et al*., 2023 (2+, EV+], Wilkinson *et al*. [2+, EV+]). Evidence that intact, wet, open peatland habitats are more resilient to fires and could act as barriers to fire spread, but under dry conditions, peatlands are at much greater risk of burning (1 study: Kirkland *et al*., 2023 [2+, EV-]).	Consistent with NEER004/014.	Evidence that pristine and restored peatlands are more resilient to wildfire.
Wildfire prevention: habitat restoration/ resilience	Restoration of upland peatlands, including through rewetting and treatments to reduce cover of 'over-dominant' species, has been recommended to reduce risk of, and increase resilience to, wildfire in the UK (e.g., McMorrow & Lindley, 2006 [2+]; Aylen *et al*., 2007 [3+]), and for wider benefits and there is moderate	Evidence that the lower moisture levels, indicative of drainage, disturbance, and /or degradation, mean that open peatlands, meadows and deciduous forests are far more likely to burn than pristine peat habitats (1 study: Kirkland *et al*., 2023 [2+, EV-]).	New evidence.	Evidence that the severity and perhaps incidence of wildfires may be reduced when wetter conditions, in particular high water tables, are maintained or restored.

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	evidence that the severity and perhaps incidence of wildfires may be reduced when wetter conditions, in particular high water tables, are maintained or restored (Grau-Andrés, 2016/2019b [2++]; Aylen *et al*., 2007 [3+].			
Managing biomass - UK	Although monitoring and managing biomass by burning or mechanical treatment is often advocated by some in the UK, (e.g., McMorrow & Lindley, 2006 [2+]; Albertson *et al*., 2010 [2+]; Marrs *et al*., 2018 [2+]), there is limited evidence of its direct effect on wildfire ignition, behaviour, severity and extent, or in reducing wider negative impacts.	No evidence.	No new evidence.	Although monitoring and managing biomass by burning or mechanical treatment is often advocated by some in the UK, there is limited evidence of its direct effect on wildfire ignition, behaviour, severity and extent, or in reducing wider negative impacts.
Managing biomass outside the UK: general, especially from modelling and theoretical studies	Managing biomass by **mechanical treatments** and/or '**prescribed'** (and sometimes 'traditional' managed) **burning** is widely practiced elsewhere in the world, particularly in shrub and forest habitats in southern Europe, North America and Australia, and there is strong, but in some cases contradictory, evidence particularly from **modelling** and **theoretical** investigations, and in some cases empirical studies, that this can be beneficial in reducing hazard and	No evidence.	No new evidence.	Strong but in some cases contradictory, evidence from outside UK, particularly from modelling and theoretical investigations, and in some cases empirical studies, that managing biomass can be beneficial in reducing hazard and hence the incidence,

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	hence the incidence, intensity, severity and extent of wildfires, and in facilitating fire suppression efforts, (4 [2++]: Hering *et al*., 2009; Marino *et al*., 2012, 2014; Stevens-Rumann *et al*., 2013 [2+]; Brose & Wade, 2002; Nunez-Regueira *et al*., 2002; Shang *et al*., 2004; King *et al*., 2006; Cary *et al*., 2009; Mitchell *et al*., 2009; Cassagne *et al*., 2011; Arkle *et al*., 2012; Shive *et al*., 2013; Wu *et al*., 2013; Volkova *et al*., 2014; Waltz *et al*., 2014; Penman *et al*., 2015; Oliveira *et al*., 2016; Fernandes & Botelho, 2003 [3+]; McCarthy & Tolhurst, 2004a [3-]). However, the magnitude and length of the effect, and the cost/benefit ratio, trade-offs and difficulty of implementation vary between sites, habitats, and wider landscapes. In addition, operational, social, ecological and wider environmental issues and objectives may constrain fuel load management.			intensity, severity and extent of wildfires, and in facilitating fire suppression efforts. The magnitude and length of the effect, and the cost/benefit ratio, trade-offs and difficulty of implementation vary between sites, habitats, and wider landscapes. In addition, operational, social, ecological and wider environmental issues and objectives may constrain fuel load management.
Managing biomass outside the UK: from empirical studies	There is less extensive evidence on the effects of biomass management from **empirical** (rather than modelling and theoretical) **studies**, mostly from case studies and analysis of fire regimes in the presence of fuel management, especially of	No evidence.	No new evidence.	Less extensive evidence on the effects of biomass management from empirical studies. More generally, there is moderate evidence

Outcome/effect	NEER004 and NEER014	2023 update	Comparison	Updated conclusion
	'**prescribed burning**'. More generally, there is moderate evidence that there remain considerable apparently **unresolved questions** over the **effects of fuel load management**, in particular in relation to the spatial arrangement, size, extent and type of fuel treatments, and severity of fire weather conditions (3 [2+]: Keeley *et al*.,1999; Keeley & Fotheringham, 2001; Cary *et al*., 2009; Price, 2012 [2+]; Fernandes & Botelho, 2003 [3+]).			that there remain considerable apparently unresolved questions over the effects of fuel load management, in particular in relation to the spatial arrangement, size, extent and type of fuel treatments, and severity of fire weather conditions.

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

Comparison of findings

12.18 A comparison of the evidence between NEER004 and the current update for outcomes relating to the extent, frequency, practice and type of burning is presented in Table 35. A similar number of studies were considered for this sub-question in the update as in NEER004, and both reviews included several studies with national or multi-region/area samples. The recent evidence considered in the review update was consistent with the evidence in NEER004 for several outcomes, with studies showing regional variation in burning extent and frequency, a similar rate of burning on deep peat compared to shallow peat and non-peatland habitats, and on protected areas compared to non-protected areas.

12.19 Regarding change over time, the recent evidence is consistent with the finding in NEER004 of an increase in burning at a national scale in recent decades, but with an indication that this trend may have started to reverse since around 2016 and especially more recently. Some new evidence in the update also suggests more burning on SAC/SPA than non-SAC/SPA areas, as well as a trend towards smaller burn patch size over time. For both NEER004 and the update, there was little evidence regarding the practice and type of burning, and its location in relation to watercourses. New evidence suggested that burning on montane habitats and steep slopes only covered a small area, though such burning is potentially damaging.

Table 35. Comparison of evidence assessed for the extent sub-question of NEER004 and this update. Individual study findings are described in section 11.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Number and location of studies	9 studies. All England. Most national samples.	12 studies. 10 including England, 4 including Scotland, 2 including Wales. 7 multiple region or national samples.	Similar number and geographic coverage.	Evidence is likely to be applicable to upland peatlands in England.
Burning extent	Strong evidence that burning extent varies by region/area and year with the proportion of study area burned between (5 studies: ADAS, 1997a [2++]; Penny Anderson Associates, 2012 [2++]; Yallop *et al*., 2012 [2++], 2006a [2+]; Anderson *et al*., 2009 [2-]).	Evidence that burning extent varies depending on region/area and year with the proportion of study area burned between 0.1 and 29% year^{-1} (11 studies: Allen *et al*., 2016 [2+, EV-]; Douglas *et al*., 2015 [2++, EV++]; Thacker *et al*., 2015 [2++, EV++]; Shewring and others, 2024 [2++, EV++]; 2016 [2+, EV-]; Blundell *et al*., 2021 [2+, EV++]; Drewitt, 2015 [2+, EV+]; Lees *et al*., 2021 [2+, EV++]; Matthews *et al*., 2020 [2+, EV++]; Natural England, 2021 [2+, EV++]; Spracklen & Spracklen, 2023 [2+, EV+]; Swindell, 2017 [2+, EV-]).	Recent evidence consistent with NEER004 finding.	Burning extent varies by region/area and year.
Burning frequency	Moderate evidence that burning frequency varies by region (3 studies: Yallop *et al*., 2012 [2++]; ADAS, 1997a [2++]; Yallop *et al*., 2006a [2+]).	Evidence that burning frequency varies by region (2 studies: Lees *et al*., 2021 [2+, EV++]; Thacker *et al*., 2015 [2++, EV++]) and can range between 11 and 66 years. Evidence of actual frequencies of 1–5 burns of individual patches (e.g., 23% of burned area burned twice) over a 22-year study period (1 study: Allen *et al*., 2016 [2+, EV-]).	Recent evidence consistent with NEER004 finding.	Burning frequency varies by region/area and year.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
Location – designated sites	Frequency of burning is similar between designated and non-designated areas (2 studies: Yallop *et al.*, 2006a [2+], 2012 [2++]).	Overall burning is widespread in protected areas (PA) and generally occurs on a relatively similar or greater proportion of the available area in and outside PAs (5 studies: Douglas *et al*., 2015 [2++, EV++]; Thacker *et al*., 2015 [2++, EV++]; Blundell *et al*., 2021 [2+, EV++]; Shewring *et al*., 2024 [2++, EV++]; Spracklen & Spracklen, 2023 [2+, EV+]), though some variation between areas and years. At GB scale, mean area of burning per 1-km square greater in SPA/SACs than outside them (1 study: Douglas *et al*., 2015 [2++, EV++]) and similar, less strong evidence, in matched areas at England-scale but more burned outside SPA/SACs in Scotland (1 study: Shewring *et al*., 2024 [2+, EV++]).	Recent evidence consistent with NEER004 finding (SSSI), with new evidence available (SAC/SPA).	Designated sites are burned at similar or greater rates than wider upland regions.
Location – peat and other soils	Frequency of burning is similar between peatland and dry heath habitats (3 studies: ADAS, 1993 [2++]; Yallop *et al*., 2006a [2+], 2012 [2++]).	Around 30–60% of burning occurs over deep peat (3 studies: Douglas *et al*., 2015 [2++, EV++]; Thacker *et al*., 2015 [2++, EV++]; Natural England, 2021 [2+, EV++]). Frequency of burning is similar between areas with and without deep peat (1 study: Blundell *et al*., 2021 [2+, EV++]).	Recent evidence consistent with NEER004 finding.	Habitats with different peat depths and mineral soils are burned at similar rates.
Location – steep slopes and montane habitats	No evidence.	Evidence that burning on steep slopes and montane habitats only covers small areas (3 studies: Shewring and others, 2024 [2++, EV++]; Spracklen & Spracklen, 2023 [2+, EV+]; Douglas and others, 2015	New evidence.	Evidence that burning on steep slopes and montane habitats covers small areas, though it is potentially damaging in these situations.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
		[2++, EV++]), though it is potentially damaging in these situations.		
Location – watercourses	No evidence.	No evidence.	No evidence.	No evidence.
Practice and type	Little evidence on burning type other than records of burning into the bryophyte or lichen layer (2 studies: Critchley *et al*., 2011a [2++], 2011b [2++]) in national samples of blanket bog and wet heath, which may also relate to burn severity.	No evidence.	No recent evidence.	Little evidence of burning type other than records of burns into the bryophyte or lichen layer.
Patch size	Evidence that median burn patch size was 0.25–0.28 ha in a national upland sample in 2000 (1 study: Yallop *et al*., 2005/2006b) [2+]).	Evidence that burn patch size varies according to region and time with a recent trend towards smaller patches (2 studies: Drewitt, 2015 [2+, EV+]; Allen *et al*., 2016 [2+, EV-]). More/ size.	New evidence available.	Evidence that burning patch size varies according to region and time with a trend towards smaller patches.
Change over time	Moderate evidence of an increase in extent and frequency of burning over time (2 studies: ADAS, 1997 [2++], greater on AES agreement land; Yallop *et al*., 2006a [2+]). This was supported by moderate evidence of a then 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 (Natural England, 2009 [2+]), though this related to all heather-dominated moorland rather than specifically peatland.	Mixed evidence of change over time with reports of increased burning in recent decades (3 national studies: Douglas *et al*., 2015 [2++, EV++]; Matthews *et al*., 2020 [2+, EV++]; Thacker *et al*., 2015 [2++, EV++]) as well as consistent (2 studies: Drewitt, 2015 [2+, EV+]; Critchley *et al*., 2016 [2++, EV++]) or fluctuating (2 studies: Allen *et al*., 2016 [2+, EV-]; Swindell, 2017 [2+, EV-]) burning extent in specific regions. The most recent evidence (1 national study: Natural England, 2021 [2+, EV++]) suggests a decline between 2016–2022. Similarly, another more recent GB-scale study (Shewring and others, 2024 [2++, EV++]) showed a	Recent evidence partly consistent with NEER004 finding.	Evidence that burning extent has changed over time at a national scale, with a long-term increase followed by an indication of a recent decrease since 2016 following increases in previous decades especially in 2021/2022.

Outcome	NEER004	2023 update	Comparison	Updated conclusion
		marked reduction in the total burned area in 2021/22, driven mainly by a reduction in area in England.		

13. Summary and conclusions

Introduction

13.1 This section provides a high-level summary of the evidence from across NEER004/014 and this update for each of the eight sub-questions. It draws on the summaries of evidence and comparisons made between the findings of the original review and recent evidence produced since then, which was presented in the tables for each of the eight sub-questions in Section 12. Where there is clear evidence of specific effects of burning, this is summarised as an evidence statement. Thus, not all the evidence considered in NEER004/014 and this update appears in an evidence statement. For example, this may be the case where evidence of an effect is lacking, minor, weak or inconsistent, or not clearly related directly or indirectly to burning.

13.2 The statements are given in **bold** as retained, revised (from NEER004/014) or new evidence statements, including their strength (paras. 2.32–2.3421) and the number of studies from NEER004/014 and this update that contribute to them. More detailed information, including lists of the supporting studies (including type and quality), is given in the comparison tables in Section 12. The statements are ordered from broad to more specific effects. Some limited interpretation and context are also included with some statements, sometimes drawing on non-evaluated references including other reviews.

13.3 A total of 93 evidence statements were developed from the evidence derived from 255 evaluated studies (listed in Appendix 1) across the combined evidence base (Table 36). There were some differences in the quantity and quality of evidence, and hence strength of evidence statements in relation to the eight sub-questions. There were a greater number of statements developed for the fauna, carbon, vegetation and wildfire sub-questions (between 14 and 19) than for water, burning extent, grazing and severity (nine or less). This in part reflected a greater number of supporting studies. Overall, most of the statements were classed as either moderate (42%) or strong (33%), with fewer classed as weak (13%) or inconsistent (12%).

Table 36. The number of studies (across the combined NEER004 and update evidence base) and strength of evidence statements by sub-question. Many studies contributed to multiple sub-questions.

Sub-question	No. studies supporting	Strong evidence	Moderate evidence	Weak evidence	Inconsistent evidence	Total no. evidence statements
Vegetation	84	6	8	0	2	16
Fauna	41	9	7	3	0	19
Carbon	38	4	6	2	6	18
Water	45	0	4	2	3	9
Severity etc.	13	0	3	1	0	4
Grazing	12	1	3	1	0	5
Wildfire	45	6	5	3	0	14
Extent etc.	21	5	3	0	0	8
Total	-	31	39	12	11	93

The effects of burning on the vegetation of upland peatland habitats

Introduction

13.4 Across the combined evidence base, a total of 84 studies reported on the direct or indirect effects of burning on upland peatland vegetation. There was a relatively similar number included in NEER004 (39) and this update (45). Most were from the UK, especially northern England and Scotland. More detailed information, including lists of the supporting studies, is given in the comparison table for vegetation in Section 12 (Table 28).

Effects on vegetation

13.5 **There is strong evidence from the majority of studies of vegetation response to burning that overall, managed, rotational burning results in change in the species composition of blanket bog and upland wet heath vegetation, at least for a period (i.e., in the short to medium term - years to decades)** (18 NEER004 studies, 31 recent studies). This results from species (a) changing relative abundance or (b) disappearing and recolonising or potentially colonising (Harris and others, 2011b [2+]). The actual species involved vary 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 and others, 1992 [4+]; Harris and others, 2011b [2+]).

13.6 **There is inconsistent evidence on the effect of burning on overall vegetation species richness and diversity including increases, no change and declines compared to no recent burning** (two NEER004 studies, five recent studies). Upland peatlands are typically not species-rich (especially in vascular plants, though may be more so in bryophytes) and unmodified and less-modified states are characterised by wetland species. Thus, interpretation of changes in species-richness and diversity need to consider the ecological characteristics of species involved.

13.7 **There is strong evidence that burning of blanket bog and wet heath typically leads to an initial period of graminoid dominance, typically of around 10–20 years, and at least an initial corresponding decline in dwarf shrub cover and diversity** (11 NEER004 studies, five recent studies).

13.8 **There is strong evidence that *Eriophorum vaginatum* tends to initial dominance after burning, particularly on blanket bog in the Pennines, or other graminoids, particularly *Molinia* and *Trichophorum germanicum*, especially in the oceanic, wetter west of the country** (eight NEER004 studies, three recent studies). This period of graminoid dominance tends to be longer than typically occurs on dry heath or more severely modified, drier bog (Harris and others, 2011b [2+], NEER004), probably reflecting a greater competitive advantage of wetland graminoids in peatlands. Especially on *Calluna* dominated blanket bog sites, *Eriophorum* species and other graminoids tend to decline over time as *Calluna* cover increases but, in some cases, *Eriophorum* still maintains moderate, or in one case relatively high, cover/ frequency. Related to this, *Eriophorum* species may sometimes occur at moderate or high cover in relatively long-unburned sites (Noble and others, 2018b [2+, EV++]), perhaps reflecting an interaction with grazing.

13.9 **There is strong evidence that *Calluna* tends to decline during the initial post-burning graminoid-dominant phase, but typically then increases** (seven NEER004 studies, ten recent studies), **especially on drier sites, though this may take 15–20 years or longer on less modified, wetter blanket bog** (two NEER004 studies, one recent study) **and may not occur, for example, with too frequent or severe burning and/or heavy grazing** (one NEER004 study, no recent studies). *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 its stems tend to be constantly reburied by growth of *Sphagnum* and sometimes other bryophytes and through the rejuvenation of the 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 experiment papers [1++]; Mowforth & Sydes, 1989; Coulson and others, 1992; Tucker, 2003; Lindsay, 2010, all [4+]). This reflects the fact that *Calluna* is “not typically a wetland species, but [is] often found growing on tussocks [and hummocks] of other species on ombrogenous mire and at the edges of soligenous mire" (Grime and others, 1988). It shows signs of stress after 40 days waterlogging (Bannister, 1964) and where soil aeration is thus reduced (Specht, 1979).

13.10 The responses in *Calluna* abundance are not surprisingly, also reflected in biomass changes, with an initial major reduction in biomass followed by a relatively rapid recovery which then tends to slow. At the Hard Hill experiment this was reflected in an absolute growth rate (AGR) peak in biomass after eight years, an asymptote after 20 years, followed by a decline in the longer-unburned 'reference' plots (Alday and others (2015 [1,2+, EV-]). This decline may be linked to increased occurrence of *Calluna* layering in *Sphagnum* moss within the 'reference' plots.

13.11 **There is moderate evidence that other dwarf shrubs, especially *Empetrum nigrum*, may decline following burning and sometimes not recover if it is more severe** (four NEER004 studies, two recent studies).

13.12 **There is inconsistent evidence on the effect of burning on *Rubus chamaemorus* with increases, no change and declines reported over time from the Hard Hill experiment and another Moor House NNR study** (three NEER004 studies, no more recent studies other than continuation of the Hard Hill study). These inconsistent findings are reflected in differences between some earlier analyses from Hard Hill included in NEER004 (initially indicating an increase, then no change) following burning and the most recent analysis of temporal change which showed an overall decline but only under the short (10-year) burn rotation treatment (and no grazing effect compared with previously reported negative effects) (Milligan and others, 2018 [1++, EV-]). It is possible that this might reflect interactions with other impacts and/or initial differences between the experiment blocks, as well as the burn treatments.

13.13 **There is strong evidence that overall, bryophytes tend to decline initially after burning followed by a more general increase, in most cases back up to or towards pre-burn levels** (six NEER004 studies, seven recent studies). The responses in bryophyte abundance are not surprisingly also reflected in biomass increases. In the Hard Hill experiment, bryophyte biomass was greatest in the no-burn and 'reference' plots and lowest in the longer (20-year) burn-treatment (Ward and others, 2007 [1+], NEER004; Alday and others, 2015 [1,2+, EV-]). This probably reflects an initial post-burn increase and subsequent decline in acrocarps, followed by a gradual increase of pleurocarps over time, peaking in the longest unburned plots. Linked to this, there is evidence that moss depth is lowest post-burn, after which it increases, and that it declines more with more severe burning (Grau-Andrés and others, 2019b/a [1+, EV-]; Noble and others, 2019b [2+, EV+]; Whitehead and others, 2021 [2+, EV-]).

13.14 **There is moderate evidence that *Sphagnum* (generally reported as a group rather than individual species) show a range of responses to burning. Some studies suggest (i) initial declines immediately post-burn, sometimes being apparently killed; (ii) little or no change or recovery; or (iii) some recovery or recolonisation after varying periods** (nine NEER04 studies, nine recent studies).

Covering a much longer period, re-analysis of 1965 data from the (grazed) unburnt since 1954 and longer-unburnt but otherwise comparable ‘reference’ plots at Hard Hill experiment showed that the latter had significantly greater *Sphagnum* cover (Noble and others, 2018a [12+, EV-]). This suggests that the (relatively large, whole block) 1954 burns had a negative effect on *Sphagnum* that was apparent 11 years after burning and which was still observed in the 2015/16 survey over 60 years later. These differences in response probably reflect a variety of factors including the actual species and abundance of *Sphagnum* pre-burn (some being colonisers and other typical bog species slower at colonisation) and over time since burning, together with different resurvey intervals (some studies lacking the immediate post-burn response and many others only covering relatively short post-burn periods) and the severity of the burn. Nevertheless, high *Sphagnum* cover and diversity is characteristic of unmodified or less-modified peatland habitats, with different species associated with different microtopes and degrees of habitat modification (for example, Lindsay, 1977/Lindsay & Ross, 1994 [2++], NEER004; Turetsky and others, 2012; Godfrey & Rogers, 2021; Smith, 2022).

13.15 **There is moderate evidence that some early-colonising, typically acrocarpous moss species, may relatively quickly become frequent or even abundant after burning** (two NEER004 studies, three recent studies).

13.16 **There is moderate evidence that an individual pleurocarpous moss species, *Hypnum jutlandicum*, showed a large immediate post burn decline, no initial increase, a delayed increase (in 20-year burn treatment plots) followed by a large increase in unburned since 1954 plots** (no NEER004 studies, two recent studies). The evidence comes from two experiments (Milligan and others., 2018 [1++, EV-], Hard Hill; Grau-Andrés and others., 2019b/a [1+, EV-]), mostly the former.

13.17 **There is moderate evidence that burning reduces *Sphagnum* propagule frequency in peat, with frequency increasing with time since the last burn and highest frequency in the longest unburned ‘reference’ plots and very low frequency under both burn treatments** (one recent study). The study was at the Hard Hill experiment (Lee and others, 2013a [1,2+, EV-]). It suggests that frequent burning reduces the *Sphagnum* propagule bank and hence its regeneration potential from peat.

13.18 **There is moderate evidence that the composition of blanket bog vegetation can continue to show change more than 80 years since the last burn** (one NEER004 study, no recent studies) From the Hard Hill experiment study (Lee and others, 2013b [1++]). The changes included increase in *Calluna*, though not at the expense of other species, and an increase in *Sphagnum* diversity and in some other moss species.

13.19 **There is strong evidence that burning is associated with the creation of bare ground, which is greatest initially after burning and variable, ranging up to c.50–60% in two cases, before gradually declining, though it was still apparent after up to nine years in one case** (six NEER004 studies, five recent studies). One study showed increased cover with increased severity of burning (Grau-Andrés and others, 2019b/2018 [1+, EV-]) and it may also be affected by the cover of the bryophyte, lichen and litter ground layer remaining (Lindsay 2010 [4+], NEER004). The longevity of bare ground may potentially be affected by other factors such as trampling by livestock (Currall, 1981 [2+]; NEER004), and erosion, especially on slopes (Elliott, 1953 [2-], NEER004). Areas of bare peat tend to be flatter and lack microtopographic variation (Lindsay 2010 [4+], NEER004).

13.20 **There is moderate evidence that differences in the frequency of burning affect the vegetation composition and structure of blanket bog in the longer-term** (one NEER004 study, no more recent studies other than continuation of the Hard Hill study). At the Hard Hill experiment ([1++, EV-]/Milligan and others, 2017), there is evidence of differing long-term trends in some cases across all treatments. But there are also differences between no-recent-burning and the burning treatments, and between the two the different burning frequencies. More frequent burning has generally promoted dominance of *Eriophorum vaginatum*, with *Calluna* achieving higher cover under the longer rotation. Otherwise there appears to be relatively little evidence of the long-term effects of burning and differing rotation lengths on peatland vegetation.

13.21 **There is moderate evidence from palaeoecological studies that vegetation burning post-Industrial Revolution likely contributed to subsequent change of vegetation community in the peat archive to graminoid or *Calluna* dominance, though other factors, especially atmospheric pollution and changes in grazing regimes may have contributed to these changes** (two NEER004 studies, four recent studies). It is likely that these and other evaluated upland palaeoecological studies (Table 11) represent only a proportion of such recently published studies, though an included review assessed 28 studies (Gillingham and others, 2016c [4+, EV++]). However, they include those that were identified in searches as specifically relating to fire/burning and hence are potentially relevant to burning management and in helping identify potential restoration trajectories and objectives.

13.22 Overall, 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, 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*.

The effects of burning on the fauna of upland peatlands

Introduction

13.23 Across the combined evidence base, a total of 41 studies reported on the direct or indirect effects of burning on upland peatland fauna. There was a similar number included in NEER004 (26) and this update (24). Most were from the UK (38 92% of studies across the two review periods), especially northern England and Scotland. Most related to breeding birds (28), followed by terrestrial invertebrates (10), aquatic invertebrates (six reported under the water section in NEER004, para. 7.14), and single studies including a mammal, reptile and soil microbes, which were reported in this update under Section 5 (fauna) (and in part under carbon, paras. 6.39–6.40). More detailed information, including lists of the supporting studies, is given in the comparison table for fauna in Section 12 (Table 29).

Breeding birds

13.24 Relatively few breeding bird studies have specifically related to upland peatlands rather than moorland in general, though some recent studies compare blanket bog with other moorland habitats. Similarly, relatively few studies specifically relate to the effects just of burning which can be difficult to separate from wider gamekeeper and other land manager activities, especially predator control (paras. 12.5-12.6, but see para.13.29). Nevertheless, burning on upland peatlands is an important factor in influencing changes in vegetation composition and especially structure (see vegetation section, paras. 4.91–4.96) which may, for example, affect suitability for bird nesting and food availability. As an example, Pearce-Higgins & Grant (2006 [2+, EV++]) and Grant and others (2012 [4+, EV+]) showed that many species tend to be associated with particular moorland habitat/vegetation characteristics that may be influenced by burning (NEER004, para. 5.22).

Habitat types, composition and structure

13.25 **There is strong evidence of associations between moorland habitat types, particularly their vegetation structure and composition (which may be influenced by burning and other management) and numbers and densities of some moorland breeding birds** (six NEER004 studies, seven recent studies).

Burning and/or predator control

13.26 **There is strong evidence of mixed effects of burning and/or predator control on numbers and densities of a range of moorland breeding birds, with some species showing increases, some declines and some no effect** (12 NEER004 studies, 16 recent studies). **This includes**:

- **Strong evidence of positive associations between burning and/or predator control intensity and numbers, densities, assemblages (and some other bird-related variables) of some moorland breeding birds, particularly waders and red grouse** (five NEER004 studies, two recent studies). Species showing such effects in more than one study were curlew, golden plover and red grouse.

- **Moderate evidence of negative associations between burning and/or predator control intensity and numbers, densities, assemblages (and some other bird-related variables) of some moorland breeding birds, particularly some passerines** (2 NEER004 studies, 4 recent studies).

13.27 **There is weak (and inconsistent) evidence of associations between burning and/or predator control intensity and overall moorland bird species diversity, and no effect reported on species richness of moorland birds** (one NEER004 study, one update study).

13.28 **There is moderate evidence of greater declines in golden plover under more intensive (rather than less intensive) burning management and greater declines in curlew and lapwing on '*Calluna*-dominated' plots than on 'bog' plots** (one NEER004 study, no recent studies).

Predator control

13.29 **For species that have shown evidence of positive associations between burning and/or predator control, there is strong evidence that predator control has a greater effect than burning** (one NEEER004 study, three recent studies). Species showing this effect in more than one study were curlew, golden plover and red grouse.

Timing of breeding

13.30 **There is strong evidence that the timing of first egg-laying of some moorland bird species overlaps with the burning season in spring in the English uplands** (five NEER004 studies including two large national BTO data sets; two recent studies, one based on the same, now extended, national data sets). As well as egg-laying, burning may may also coincide with the pre-nuptial period on site and other breeding activities, including nest building and, to a lesser extent, the incubation and nestling stages (Moss and others, 2005 [2++, EV++], NEER004; Wilson and others, 2021 [2++, EV++]; Zonneveld and others, 2024 [2+, EV-]).

13.31 The risk and potential effect on bird populations depends on a range of factors, including the degree of overlap of the dates, the proportion of the population nesting on upland habitats likely to be burnt (i.e., species that nest in relatively short, burnable vegetation), the frequency and extent of moorland burning and proportion of burning in spring, and the effect on breeding success including re-nesting (Moss and others, 2005 [2++, EV++], NEER004; Wilson and others, 2021 [2++, EV++]; Zonneveld and others, 2024 [2+, EV-]; Glaves and others, 2005 [4+, EV+], NEER004). Early nesting moorland species that may be vulnerable to losing some first nests and eggs towards the end of the burning season in the English uplands include: golden plover, lapwing, snipe (though these three waders tend to prefer shorter swards that are less likely to be burnt); short-eared owl, hen harrier, and stonechat (Wilson and others, 2021 [2++, EV++]; Zonneveld and others, 2024 [2+, EV-]); and in the south-west, Dartford warbler (Bibby, 1979; Wotton and others, 2009). The same 'burning season' end date in England (15 April in the uplands[20]) also applies in Scotland (where it can be extended to 30 April with landowner permission), but in Wales it was moved back to 31 March in the uplands in 2008 in large part to reduce the risk to early nesting birds (Newson and others, 2007; WAG, 2008). The same date of voluntary earlier cessation of burning is also recommended on the south-west moors in England (Defra, 2013).

13.32 **There is strong evidence of gradually earlier egg-laying in many moorland bird species** (one NEER004 study, one recent study). This mainly comes from two long-term, large national BTO data sets. The initial analysis of these data showed evidence of earlier nesting for eight species (Moss and others, 2005 [2++, EV++], NEER004), with the most recent re-analysis showing evidence of an advancement of mean laying date across all species by about one day every eight years over the 44-year period from 1976 to 2019 (Wilson and others, 2021 [2++, EV++]).

Invertebrates

13.33 Few studies specifically addressed the impacts of managed burning on the invertebrates of upland peatlands. Most considered burning amongst other management and environmental variables to interpret differences in invertebrate community composition, often in site comparison studies, some of which related to wider moorland habitats. Many studies related to specific insect groups, though some considered invertebrate assemblages or the community as a whole. Some of the species reported are not necessarily characteristic of less-modified peatland/wetland or watercourse habitats (for example, see Webb and others, 2010).

Terrestrial invertebrates

13.34 **There is strong evidence mostly from studies involving multiple sites, and in one case a national sample (from Wales), that burning and other management influences the invertebrate community composition of upland peatland habitats** (eight NEER studies, two recent studies). **This includes:**

- **Strong evidence of differences in species, species-groups and assemblages associated with different post-burn successional stages and vegetation types/habitats.**
- **Strong evidence that these differences are related to a range of factors including soil moisture and nutrient status, presence of open- and bare-ground, vegetation height/density and altitude, most of which are directly or indirectly influenced by burning and other management.**

13.35 **There is moderate evidence that cranefly emergence and abundance is related to soil moisture content and hence may be affected by different**

[20] In the Severely Disadvantaged Areas (SDA) of the Less Favoured Areas (LFA).

vegetation management and restoration interventions (no NEER004 studies, two recent studies). **This includes:**

- **Weak evidence that cranefly emergence and abundance is not related to vegetation height (which is affected by management), though taller vegetation may reduce the availability of prey for waders** (no NEER004 studies, one recent study).

13.36 **There is weak evidence across most known and potential sites in the large heath butterfly's main range in England in Northumberland, that too frequent burning is likely to make peatland sites less suitable or unsuitable, but that occasional burning may be beneficial, perhaps in favouring it's larval foodplant, *Eriophorum vaginatum*, and in reversing succession on at least some drier sites** (one NEER004 study, no recent studies). Despite some declines reported in the 1990s, since that time there has been a very large (407%) increase in the large heath's abundance at monitored sites overall in the UK and little change in distribution (-2%). Nevertheless, at some sites in northern England, a reduction has been reported in the abundance of *Eriophorum* species., the larval foodplants, which could reflect management, although it has been suggested that climate change could be a factor (Fox and others, 2023).

Aquatic invertebrates

13.37 **There is moderate evidence that burning, season and their interaction is associated with changes in peatland watercourse aquatic invertebrate communities, including reduced taxonomic richness and diversity. These changes reflect declines in certain species groups, especially mayflies and stoneflies, and grazer and collector-gatherer feeding groups, and increases in others including non-biting midges and flies** (three NEER004 studies reported in water section [Section 7, p. 36, and Appendix 7, pp. 139–140]; and three recent studies, though many of these studies are linked through the inclusion of EMBER sites as well as other additional sites). These biotic changes were associated with lower pH and higher Silicon, Manganese, Iron and Aluminium in burned catchments and increased sedimentation (Brown and others, 2014/2019 [1,2+, EV+]) likely also related to water quality.

Mammals

13.38 **There is moderate evidence that mountain hare densities following bog restoration are higher than on neighbouring 'degraded bog', 'bogs managed for grouse shooting' and on other 'heather moorland' in the Peak District** (no NEER004 studies, one recent study).

Reptiles

13.39 **There is moderate evidence that in Scotland adders mostly occur in 1-km squares where grouse moor management does not occur** (no NEER004 primary studies, one review; one recent study). Adders were recorded from 810 1-km squares in Scotland of which only 77 (10%) overlapped with assessed grouse moor squares (Newey and others, 2020 [2+, EV++]). Though no primary studies in NEER004 identified evidence of effects on reptiles, an evaluated review (Glaves, 2005 [4+, EV+]) summarised similar, earlier evidence that suggests that adders may be less frequent or absent in frequently burned upland areas and that they are potentially at risk from fires in late winter and spring (Wild & Entwistle, 1997; Whiteley, 1997, 2003; Frazer, 1983; Offer and others*,* 2003; Baker and others*,* 2004). Adders and common lizards *Zootoca vivipara* may be capable of recolonising burnt areas over time (Simms, 1972), presumably from any neighbouring, recently unburnt areas of suitable habitat.

Soil microbes

13.40 **There is moderate evidence that fungi, bacteria and archaea communities differ between managed (burned or cut) sites and modified sites undergoing restoration. There was, however, little difference between the restoration sites across three geographic areas.** (no NEER004 studies, one recent study). The study sampled eight widely distributed sites/areas between Exmoor and Forsinard Flows, Scotland, (Burn, 2021 [1,2+, EV+]). The effects of microbial community composition on carbon balance and water quality are reported under carbon and water in sections 6 and 7.

The effects of burning of upland peatlands on carbon balance

Introduction

13.41 Across the combined evidence base, a total of 38 studies reported on the effects of burning on upland peatland carbon balance. There was a similar number included in NEER004 (18) and this update (20). All but three were from the UK (92%), mostly from or including northern England, with three others from Scotland and one from Wales. More detailed information, including lists of the supporting studies, is given in the comparison table for carbon in Section 12 (Table 30).

Effects on carbon balance

13.42 **There is strong evidence that managed burning affects various aspects of the carbon balance of upland peatlands** (18 NEER004 studies, 20 recent studies). **This includes**:

- **Strong evidence that burning reduces aboveground carbon stock, which can then increase for at least several decades after burning** (one NEER004 study, six recent studies).
- **Weak, inconsistent evidence of burning reducing and increasing belowground carbon stock** (one NEER004 study, one recent study).

- **Strong evidence that above-ground carbon is lost in combustion** (three NEER004 studies, six recent studies).
- **Moderate evidence that charred material has a role in post-burn carbon cycling, which can be influenced by burn severity** (one NEER004 study, two recent studies).
- **Inconsistent evidence on the effects of burning on Net Ecosystem Exchange (NEE) of CO_2** (one NEER004 study, five recent studies). This includes two studies that found plant species effects on NEE of CO_2.
- **Inconsistent evidence on the effects of burning on Gross Primary Productivity** (**GPP)** (one NEER004 study, four recent studies).
- **Inconsistent evidence on the effects of burning on ecosystem respiration** (one NEER004 study, four recent studies). This includes one study that found plant species effects on respiration.
- **Moderate evidence that burning increases soil temperature for an initial period following burning** (two NEER004 studies, three recent studies).
- **Weak evidence of lower rates of soil respiration after burning** (one recent study) **and that plant species can affect soil respiration** (one recent study).
- **Inconsistent evidence on the effects of burning on methane (CH_4) fluxes** (three NEER004 studies, three recent studies) **and vegetation composition effects on methane fluxes** (three recent studies).
- **Moderate evidence that burning increases net greenhouse gas (GHG) (CO_2, CH_4 and N_2O) emissions, with burned and cut plots GHG sources compared to longer-unburned comparisons which were GHG sinks** (one recent study). This was from the Peatland-ES-UK experiment on three modified blanket bog sites up to nine years post-burn (Heinemeyer and others, 2019c/2023 [1+, EV-]).
- **Overall, moderate evidence of an increase in DOC/water colouration at the plot scale initially post burn with little or no longer-term effect** (four NEER004 studies, five recent studies). Some of the apparent differences in response are likely to relate to different timescales of sampling after usually one-off burn treatments in experimental plots (with effects occurring soon after burning). For example, Clay and others (2009b [1,2+], NEER004) showed peaks in DOC/water colouration at Hard Hill between three and seven weeks after burning. Thus, these findings are not necessarily inconsistent between plot studies or with those from laboratory- and catchment-scale watercourse studies as the effect appears to occur for an initial period after burning (rather than over the longer post-burn recovery period that may be studied in individual 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 (NEER004, para. 7.11).
- **Overall, strong evidence that burning is associated with increases in DOC/water colour in peatland watercourses** (from nine primary NEER004 studies and a review), although there was no evidence of an effect in two recent

studies. The NEER004 studies comprised five multiple catchment studies, three laboratory studies, a model and a critical synthesis (last, Holden and others, 2011/2012 [2++]), whereas the two recent studies sampled single catchments. These apparent differences may in part relate to accuracy in mapping the extent of deep peat and/or recent burning over time, and the number of catchments included (see, for example, Yallop and others, 2011) and perhaps other factors such as burn severity and burn locations in relation to slope and distance to watercourses. Unlike in plot studies used to monitor post-burn response/recovery over time, typically after single burns, with rotational burning of different patches across a site/catchment, there is a post-burn effect each year with the effect subject to variations in burn extent and severity between years.

- **Moderate evidence of plant species or vegetation composition effects on DOC or water colouration in watercourses** (two NEER004 studies, four recent studies). Two studies showed an association between the *Calluna*-dominated area and DOC/water colouration, but four others indicated associations with other plant groups.
- **Moderate evidence that burning is associated with increased erosion and POC in watercourses** (one NEER004 study, three recent studies), though one recent study showed no difference between burned and cut catchments (but did not include an unmanaged treatment).
- **Inconsistent evidence on the effects of burning on peat accumulation** (one NEER004 study, two recent studies).
- **Inconsistent evidence of how burning affects the overall upland peatland carbon balance** (five NEER004 studies, two recent studies). This probably at least in part reflects variations and gaps in the extent of evidence on the effects of burning on different aspects of the carbon balance and over medium- to long-term, rather than short-term, timescales.

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

Introduction

13.43 Across the combined evidence base, a total of 44 studies reported on the effects of burning on peatland water quality, distribution and flow. These studies were evenly split between NEER004 (22) and this update (22). All but two were from the UK, all but two of these from or including England, mostly the Pennines. More detailed information, including lists of the supporting studies is given in the comparison table for water in Section 12 (Table 31).

Water colouration/DOC

13.44 **There is moderate evidence that changes to peatland vegetation composition may influence DOC treatability with *Calluna*-derived DOC most difficult to remove and most likely to form chloroform during treatment** (one recent study).

13.45 Evidence statements on the wider effects of burning on DOC, POC and water colouration, which are also related to water quality, are given under carbon in this section (para. 13.42).

Soil and water chemistry

13.46 **There is moderate evidence that burning influences various aspects of soil, runoff and stream water chemistry** (one NEER004 study, four recent studies).

13.47 **There is inconsistent evidence on burning effects on water pH with increases, declines and no change in soil and stream water reported** (four NEER004 studies, two recent studies).

Watercourse aquatic invertebrates

13.48 An evidence statement on the effects of burning, season and their interaction on watercourse aquatic invertebrate in peatland watercourse aquatic invertebrate communities is given under fauna in this section (para. 13.37).

Hydrology and water flow

13.49 **There is inconsistent evidence of burning effects on water table depth with four studies reporting lowering of the water table following burning, in two cases followed by gradual recovery, and four studies reporting shallower water tables following burning** (three NEER004 studies, five recent studies). These differences may reflect differences in timescales of study post-burn and changes in relation to weather and season and how this is accounted for, for example, by using controls. The reported gradual recovery of water tables following burning is consistent with a similar response to the introduction of restoration management treatments (including cessation of burning) (ref).

13.50 **There is moderate evidence of increased frequency of surface runoff after recent burning** (two NEER004 studies, one recent study).

13.51 **There is weak evidence of reduced steady state infiltration rates, proportion of flow moving through macropores and hydraulic conductivity after recent burning** (one NEER004 study, one recent study).

13.52 **There is weak evidence that burning can increase peat surface hydrophobicity** (two recent studies).

13.53 **There is moderate evidence that burning can increase flow in watercourses draining upland catchments** (four recent studies). One study showed increased flow volume from burned catchments at two of three study sites and modelled higher downstream river levels (Heinemeyer and others, 2019c [1+, EV-]). Another showed greater hydrograph lag times and a flashier response to large storm events (Brown and others, 2014 [2+, EV+]). A modelling study suggested that burning can increase flow peaks (Gao and others, 2017 [1+, EV-]) and another showed a possible mechanism for this, with bare ground increasing flow peaks and denser *Sphagnum* ground cover density reducing them (Gao and others, 2016 [1+, EV-]).

13.54 **There is inconsistent evidence of burning impacts on peat bulk density,** which may affect water retention and availability to plants (two recent studies).

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

Introduction

13.55 Across the combined evidence base, a total of 13 studies (one in both NEER004 and the update counted as a single study) reported on the effects of differences in the severity, frequency and other characteristics of managed burns on upland peatland vegetation, fauna, carbon and water. There was a similar number included in NEER004 (six) and this update (seven, excluding one that was also included in NEER004). All but two were from the UK (83%) and either northern England and/or Scotland. Most studies related to severity and frequency and none to scale and location, although five studies (one a review) related vegetation fuel load and structure to fire behaviour and severity, and two to fuel moisture. More detailed information, including lists of the supporting studies is given in the fire characteristics comparison table in Section 12 (Table 32).

13.56 Little evidence was identified on the types of burning practice taking place in the English uplands in general and specifically on deep peat, including the extent to which ‘cool burning’ is practiced.

Burn severity

13.57 **There is moderate evidence that the severity of burns affects vegetation composition, with higher severity benefiting dwarf shrubs including *Calluna*, acrocarpous mosses and graminoids, and leading to lower abundance of pleurocarpous mosses and damage to *Sphagnum* cells** (no NEER004 studies, four recent studies).

13.58 **There is weak evidence that higher severity burns can affect the soil thermal regime and lability of pyrogenic carbon** (no NEER004 studies, two recent studies). No effects were detected on GHG or DOC fluxes in one of the studies.

13.59 **There is moderate evidence that burns into the bryophyte and lichen layer occur in a proportion of cases on blanket bog and wet heath** (two NEER004 studies, no recent studies) The two studies comprised national sample condition surveys of blanket bog and wet heath (Critchley and others, 2011a,b [2++], NEER004) which were reviewed under extent, frequency and type of burning (Section 11, Table 35. Comparison of evidence assessed for the extent sub-question of NEER004 and this update. Individual study findings are described in section 11.).

Burn frequency

13.60 **There is moderate evidence that increasing the frequency of burning reduces carbon accumulation and storage** (no NEER004 studies, two recent studies [one modelling and one on carbon stocks]). For carbon stock studies, it should be noted that comparisons of the apparent rate of peat carbon accumulation (aCAR) between treatments and sites has been criticised by Young and others (2019, 2021) who suggest that aCAR should not be used to compare amounts of carbon stored in surface peat (para. 6.49).

13.61 Burn frequency is a key issue at a site or larger scale. At any one-time, rotational burning creates a patchwork of burns at different scales and ages since burning. Subject to (often weather-related) variations between years and trends in extent/ intensity over time, increased frequency of burning results in shorter rotations and hence a greater proportion of a site being burned annually. As a result, any effects of burning are increased and recovery timescales shortened. It also results in higher proportions of a site being at shorter or moderate times since burning, and hence any initial and medium-term effects of burning applying over a greater area. Such effects may include, for example, carbon losses on combustion (paras. 4.89–4.90), increased DOC/water colouration in watercourses (Table 30), and initial dominance of graminoids, especially *Eriophorum vaginatum* (paras. 4.44, 4.63–4.65). Rotational burning, especially with short return intervals, prevents full restoration of habitat composition, structure and function, including characteristic associated species, recovery of natural processes and ecosystem services, and resilience to climate change and other impacts.

The interaction between burning and grazing

Introduction

13.62 Across the combined evidence base, a total of 12 studies (one in both NEER004 and the update counted as a single study) reported on the interaction between burning and grazing. Eleven were included in NEER004 and just one in this update (excluding one that was also included in NEER004). All were from the UK, mostly northern England, including four at Moor House NNR, with two from Scotland. All ten primary studies involved experimental elements with grazing (in all but one case by sheep) and burning as treatments, and reported on the effects of the interaction on vegetation. More detailed information, including lists of the supporting NEER004 and recent studies (including type and quality), is given in the comparison table for burning and grazing interaction in Section 12 (Table 33).

13.63 The relatively small number of evaluated studies that included grazing treatments reported 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 Martin and others, 2013, for evidence on grazing effects). It is however possible that interactions may occur at a relatively large scale (for example, moorland grazing unit) and are less easy to pick up in smaller plots. For example, new growth, particularly of graminoids, following burning generally attracts stock and burning is specifically used for stock management to produce 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 and hence the impact of a given stocking rate (for example, Phillips, 2012).

Effects on vegetation

13.64 **There is strong evidence that burning and (mostly sheep) grazing may interact to affect species composition, dominance and the abundance of individual species and species groups** (11 NEER004 studies, no recent studies). **This includes:**

- **Moderate evidence, especially from the Hard Hill experiment but also other studies, that the interaction between burning and grazing may affect the trajectory of vegetation community change, including prolonging the initial post-burn graminoid phase and resulting in changes in abundance of individual species and groups** (described in NEER004, para. 4.20) (five NEER004 studies, no recent studies). This is consistent with NEER004, which suggested that over time the graminoid phase tends to transition to increasing *Calluna* cover especially on drier sites but "... may not occur, for example, with too frequent or severe burning and/or heavy grazing …" (para. 4.22). Changes in abundance have been reported across many groups and a number of species including vascular plants (declines in *Calluna*, *Empetrum nigrum* and increases in graminoids especially *Eriophorum vaginatum*), mosses (*Calypogeia muelleriana, Campylopus paradoxus, Hypnum jutlandicum, Pohlia nutans, Sphagnum* spp.), liverworts (*Calypogeia muelleriana*, *Cephalozia. bicuspidata, Lophozia ventricosa)* and lichens.
- **Moderate evidence that burning results in increased grazing of *Molinia* by sheep and deer, and increased grazing of *Rubus chamaemorus* by sheep, but these effects may be relatively short-lived** (two NEER004 studies, no recent studies).
- **Weak evidence that grazing following burning results in increased cover of bare ground** (1 NEER004 study, no recent studies).
- **Moderate evidence that the characteristics of both grazing and burning regimes can influence how they interact to affect vegetation outcomes** (one NEER004 study, one recent study). This includes stocking levels and regimes, including seasonal timing, and burn rotation length, extent and location (Tucker,

2003 [4+, EV+], NEER004; Milligan and others, 2018 [1++, EV-], Hard Hill experiment).

The relationship between managed burning and wildfire

13.65 Across the combined evidence base, a total of 45 studies reported on the relationship between managed burning and wildfire. Thirty-eight studies were included in NEER004/014 and eight in this update reflecting the relatively short time since NEER014 was published (2020). Though many were from or included the UK (17), more were from outside the UK (29) reflecting the higher incidence of wildfires in some other parts of the world, and hence associated research and prevention/mitigation measures. More detailed information, including lists of the supporting NEER004/014 and recent studies (including type and quality), is given in the comparison table for burning and wildfire in Section 12 (Table 34).

Escaped managed burns as a cause of wildfire occurrence

13.66 Most wildfire ignitions in the UK are anthropogenic in origin, bring classed either as 'accidental', associated with public access, recent/current wildfire or managed burning activity, or deliberate ('arson'), with very few documented instances of 'natural' wildfires due to lightning strikes. Evidence from a recent English wildfire data set maintained by, and including data submitted to, Natural England (NEER014, Appendix 2 [2+, EV++]) showed that, where a broad cause of fire was assigned (2,726 fires), the majority (77%) were classed as deliberate and the minority (23%) accidental.

13.67 **There is strong evidence that managed burns escaping control cause a proportion of wildfires in the UK, particularly in the uplands** (seven NEER014/014 studies, one more recent study). These give a range for the proportion of wildfires resulting from escaped managed burns (where a specific cause was assigned which is the minority of cases) of between 15–60% or (if data from the lowlands, where managed burning is less widely used, are excluded) 24–68%. It should be noted that the studies cover different UK geographical areas and periods. **This includes:**

- **Strong evidence for England that, the main cause of wildfire ignition is 'campfires' (49% of fires), followed by management burns escaping control (15%), barbeques (10%), and 'reignited' fires and military training (both 5%) with no other causes greater than 3%.** (one NEER014 study, no more recent studies). The data for this come from the minority of wildfires where a more specific wildfire ignition cause was assigned (12% of all fires) in a recent English wildfire data compiled by and submitted to Natural England (NEER014, Appendix 2 [2+, EV++] and Figure 6).

- **Strong evidence of differences in the causes of wildfire ignition in: (i) the English uplands with the main cause being management burns escaping control (68% of cases), followed by campfires and barbeques; and (ii) in**

the English lowlands with the main cause being campfires (56%), followed by barbeques (11%), 'reignited' fires (8%) and managed burns (8%) (one NEER014 study, no more recent studies). This is based on studies where a specific ignition source was assigned in the same recent Natural England English wildfire data set (NEER014, Appendix 2 [2+, EV++]).

Seasonality of wildfires in relation to the burning season

13.68 **There is strong evidence of a wildfire peak in spring and a lower secondary peak in summer in the UK, the former overlapping in part with the 'burning season' (up to 15 April in the English uplands)** (eight NEER014 studies, three recent studies) **This includes:**

- **Strong evidence of a difference in the seasonal pattern of wildfires between lowland and upland areas in England, with a higher percentage in the uplands in spring than in the lowlands where there is a more even spread between spring and summer** (two NEER014 studies, no more recent studies).

- **Strong evidence of a wildfire peak in the English uplands in the months of March and April compared to lower incidence in the summer and especially autumn and winter months** (three NEER004/014 studies, no more recent studies).

Fire severity

13.69 **There is moderate evidence that vegetation biomass and structure, and hence vegetation and habitat type, are critical factors in upland peatland fire behaviour, particularly fireline intensity and rate of spread – although residence time and the depth of penetration of high temperatures into the soil are also important in determining severity, these are less well understood** (seven NEER004/014 studies, no more recent studies). Most evidence relates to *Calluna*-dominated vegetation. **This includes:**

- **Moderate evidence that fire severity (including ground fuel consumption, ground heating and changes in post-fire soil thermal dynamics) vary by habitat/vegetation type in the UK** (three NEER014 studies).

- **Moderate evidence that *Calluna* dry heath and tree-dominated sites in the UK suffer more severe burning than bog, fens and bog woodland** (three NEER014 studies, no more recent studies).

- **Moderate evidence that the severity and possibly incidence of wildfires may be reduced when wetter site conditions are maintained or restored, in particular through a high water table** (three NEER014 studies, two more recent studies). Restoration of upland peatlands, including through rewetting and treatments to reduce cover of 'over-dominant' species, has been recommended to reduce risk of, and increase resilience to, wildfire and deliver wider benefits (for example, McMorrow & Lindley, 2006 [2+]; Aylen and others, 2007 [3+], both NEER014).

- **Weak evidence of higher cover of bare ground following a wildfire in areas subject to previous, relatively recent managed burning than in areas not burned for longer periods** (no NEER004/014 studies, one recent study). The study involved a single Peak District blanket bog site. Mean post-wildfire bare ground cover was 78% across 0–6, 7–15 and 16–29 years post-managed-burn classes compared with 35% in 30–40 years and 26% in >40 years post-managed-burn (Swindell, 2017 [2+, EV-]). This may be related to vegetation composition and/or wetness, as *Sphagnum* cover was highest in longer unburned stands.

Biomass management

13.70 **There is weak evidence of any direct effect of managing biomass by burning or mechanical treatment on wildfire ignition, behaviour, severity or extent in the UK** (NEER014, paras. 8.4, 9.63–9.68). This is even though it represents a potential mechanism for reducing wildfire 'hazard' (Holland and others, 2022), the use of which has been advocated by some (for example, McMorrow & Lindley, 2006 [2+], NEER014; Albertson and others, 2010 [2+], NEER004/014; Marrs and others, 2018 [2+, EV-]).

13.71 **There is strong, but in some cases contradictory, evidence particularly from modelling and theoretical investigations and in some cases empirical studies, that managing biomass by mechanical treatments, 'prescribed' burning, and (sometimes) 'traditional' managed burning (as is widely practiced elsewhere in the world, particularly in forest and shrub habitats in southern Europe, North America and Australia) is beneficial in reducing hazard and hence the incidence, intensity, severity and extent of wildfires, and in facilitating fire suppression efforts** (20 NEER014 studies, no more recent studies). However, the magnitude and length of the effect (the latter generally short), and the cost/benefit ratio, trade-offs and difficulty of implementation vary between sites, habitats, and wider landscapes. In addition, operational, social, ecological and wider environmental issues and objectives may constrain biomass management. It should also be noted that the majority of these studies relate to: forests rather than open habitats, particularly peatlands; often hotter, more fire-prone environments where natural, lightning-induced, fires are much more frequent; and larger-scale, frequent use of 'prescribed burning' often by professionals, rather than more traditional managed patch burning by land managers on moderate rotations. Thus, the findings may not necessarily be applicable to UK upland peatlands.

13.72 **There is moderate evidence from outside the UK of considerable, apparently unresolved, questions over the effects of biomass management, in particular in relation to the spatial arrangement, size, extent and type of fuel treatments, and severity of fire weather conditions** (five NEER014 studies, no more recent studies). There is less extensive evidence on the effects of fuel management from empirical (rather than modelling and theoretical) studies outside the UK, mostly from case studies and analysis of fire regimes in the presence of fuel management, especially of 'prescribed burning' specifically to reduce biomass (also see NEER014, paras. 8.6–8.7, 9.66–9.67).

The extent, frequency and practice of managed burning

Introduction

13.73 Across the combined evidence base, a total of 21 studies reported on the extent, frequency and practice of managed burning. Nine were included in NEER004 and 13 in this update. All were from the UK, all but two were from or included England, four were from or included Scotland, and two included Wales. More detailed information, including lists of the supporting studies, is given in the comparison table for managed burning extent, frequency and practice in Section 12 (Table 35).

Extent

13.74 **There is strong evidence that burning extent varies by GB region/area and year** (five NEER004 studies, 11 recent studies).

Frequency

13.75 **There is strong evidence that burning frequency varies by GB region/area and year** (three NEER004 studies, three recent studies).

Types of land

Designated sites

13.76 **There is strong evidence that burning in SSSIs, SACs and SPAs occurs at a similar or greater frequency as non-designated areas in the same regions/areas and nationally** (two NEER004 studies, five recent studies).

Peatlands

13.77 **There is strong evidence that burning over deep peat occurs at a similar frequency as on other soil types in the same regions/areas and nationally** (three NEER004 studies, four recent studies).

Other sensitive areas

13.78 **There is moderate evidence that burning on steep slopes and montane habitats covers small areas, the latter mostly in Scotland, though it is potentially damaging in these situations** (three recent studies). No evidence was found on the extent or frequency of burning over other 'sensitive areas' as listed in the upland Common Standards Monitoring guidance (JNCC, 2009) and the Heather and grass burning code (Defra, 2007), including adjacent to watercourses.

Practice and type of burning

13.79 **There is moderate evidence 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%)** (two NEER004 studies, no recent studies). Although there is little evidence on the types of burning practice taking place in the English uplands, including on the extent to which 'cool burning' is practiced, these data from two national sample surveys (Critchley and others, 2011a,b [2++], NEER004) suggest that that severity of burning is variable, both within and between sites.

Burn patch size

13.80 **There is moderate evidence that burn patch size varies by region and over time, with a recent trend towards smaller patch size** (one NEER004 study, two recent studies).

Change over time

13.81 **There is strong evidence that burning extent has changed over time at a UK national scale, with a long-term increase followed by an indication of a recent decrease since 2016, especially in 2021/22 in England** (one NEER004 study, three recent studies).

Conclusion

13.82 The combined evidence from NEER004 and the update suggests that burning can affect peatlands, and the ecosystem services they provide relating to biodiversity, carbon and water, with numerous potential pathways for influence. Key changes such as altered vegetation composition and structure recover on varying timescales, ranging from months to decades. Repeated burning risks interrupting the trajectory of recovery, resulting in a sustained departure from characteristic peatland structure and function. Furthermore, UK peatlands where burning occurs are often degraded due to past and current stressors including fire, suboptimal grazing and atmospheric pollution, and continued burning may inhibit recovery or restoration.

Limitations

13.83 This review has several limitations, including the possibility that there may be evidence which was not identified in the literature searches. This is more likely to apply to unpublished data and reports or grey literature. Other limitations of the review include limitations of the studies evaluated. These include geographic biases due to research being concentrated in certain study regions (notably the Pennines) and sites, though this in part reflects the geographic distribution of recent and current managed burning. The mismatch in time periods between study length (often influenced by funding), burning rotation length and peatland recovery timescales can also be considered a limitation, though there were some longer studies and use of space-for-time substitution. A further limitation for many studies was the availability and selection of less modified or unmodified comparison sites, as many upland peatlands in regions where managed burning occurs have been burned at some point in history or degraded by other influences.

13.84 When studying nature, it is often problematic to disentangle the effects of myriad variables. Even when multiple studies give consensus on the direction and magnitude of an impact it is often difficult to generalise a precise quantification, due to variations in the history, geography, and management of study sites. However, the availability of a range of study types from controlled experiments to national scale observations formed a robust evidence base on which to draw conclusions for many outcomes.

Research recommendations

13.85 Recommendations for future research and other evidence gathering to address gaps were made in NEER004 for each sub-question and were summarised across sub-questions at the end of the Conclusions (Section 12, paras. 12.35 and 12.36, pp. 57–58). An assessment of the extent to which those recommendations have been or are being addressed is given in Appendix 4.

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Glossary

The following list of technical terms and acronyms used in the report draws on a range of sources including FAO (1986), Davies and others (2008) and Keeley (2009).

Term	Definition
API	Aerial photographic interpretation, for example, of habitats or burns.
BAP	Biodiversity Action Plan.
Bog	An 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.
Duff	Decaying and decayed organic matter usually below but sometimes including the litter level.
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.
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.

Term	Definition
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.
Meso-scale	An intermediate scale.
Microtopography	Small-scale surface features.
Mire	A habitat that forms peat.
Moorland Line	Definition of semi-natural moorland vegetation in the uplands (Severely Disadvantaged Areas (SDA) in the Less Favoured Areas (LFA) produced for MAFF (now Defra) by aerial photographic interpretation (API) with ‘ground truthing’.
Muirburn	Scottish term for managed burning of vegetation on moorland.
Nanotope	Individual small-scale bog structures such as a hummock, low ridge, or *Sphagnum* hollow.
N_2O	Nitrous oxide.
NECB	Net ecosystem carbon balance, also referred to as carbon budget.
NEE	Net Ecosystem Exchange (of CO_2).
Ombrogenous	Formed due to the influence of precipitation.
Ombrotrophic (habitat or ecosystem)	Receiving all its nutrient supply from precipitation or atmospheric deposition.
PAR	Photosynthetically active radiation.
Palaeoecological	Relating to the ecology of fossil and subfossil animals and plants.
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).
Peat pipe	Underground channel through peat that water flows through.
PFT	Plant functional type.
pH	A measure of the acidity or alkalinity of a solution or material.
POC	Particulate Organic Carbon.

Term	Definition
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.
SUVA	Specific ultraviolet absorbance.
Synusia/synusial	A distinct vegetation layer that is composed of plants of a similar life-form

Appendices

Appendix 1. Evidence tables of evaluated studies.

Appendix 2. Evidence search strings.

Appendix 3. Copies of quality assessment checklists.

Appendix 4. Research recommendations from previous reviews.

Appendix 5. Burning regulation and guidance.

Appendix 6. Upland peatland habitats and vegetation communities.

Appendix 1. Evidence tables of evaluated studies

Table A1.1. Evidence table of recent evaluated studies included in this review update in alphabetic order by lead author and date (oldest first). The study type, quality and external validity are shown in the first column (see Tables 1–3 for details of the scoring system). Where applicable, supplementary studies and relevant comments (C) and responses (R) are included in columns 2 and 3. The other columns show which country/nations/areas each study was located in (a key to abbreviations used is given after the table) and which sub-questions it applies to. Studies included in the update from the wildfire review (NEER014) are listed separately in Table A1.2.

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Alday *et al*., 2015 [1,2+, EV-]*	Santana *et al*., 2016; Hard Hill vegetation study (NEER004 and Hard Hill studies in this update)	(C): Clutterbuck *et al*., 2020	UK	E	NP	1	3, 7
Allen *et al*., 2016 [2+, EV-]*	-	-	UK	E	PD	8	-
Bargmann *et al*., 2016 [2+, EV-]	Bargmann *et al*., 2015	-	Norway	-	-	2	-
Bedson *et al*., 2022b [2+, EV+]	-	(C): Hesford & MacLeod, 2022; (R): Bedson *et al*. 2022a	UK	E	PD	2	-
Blundell & Holden, 2015 [2+, EV-]*	Blundell *et al*. 2016	-	UK	E	SP	1	-
Blundell *et al*., 2021 [2+, EV++]	-	-	UK	E	FB, NP, NYM, PD, SP	8	-
Brown *et al*., 2014 [2+, EV+]*	Blundell *et al*., 2013; Brown *et al*., 2013*, 2015a,b*, 2019; Hedley, 2013; Holden *et al*., 2013, 2015*; Aspray *et al*., 2017; Nobel *et al*., 2018b	(C): Davies *et al*., 2016b; Ashby & Heinemeyer, 2019a,b; (R): Brown *et al*., 2016; Brown & Holden, 2019, 2020	UK	E	NP, PD, SP, YD	4, 3	2, 5

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Buchanan *et al*., 2017 [2+, EV++]*	-	-	UK	E, W, S	NP, SP	-	-
Burn, 2021 [1,2+, EV+]	Heinemeyer *et al*., 2019c, 2023; Burn *et al*., 2021		UK	E, S	EX, FB, PD, YD	2	1, 3, 4
Byriel *et al*., 2023 [2+, EV-]	-	-	Denmark	-	-	2	1
Calladine *et al*., 2014 [2+, EV-]*	-	-	UK	S	-	2	1
Cardíl *et al*., 2023 [2+, EV++]	-	-	NW Europe	UK	-	7	-
Chambers *et al*., 2013 [2+, EV+]*	-	-	UK	W	-	1	-
Chambers *et al*., 2017 [2+, EV++]*	-	-	UK	E	NP, NYM SP	1	-
Chapman *et al*., 2017 [2,4+, EV-]	-	-	UK	E	NYM	4	3
Clay *et al*., 2015 [2+, EV-]*	Clay *et al*., 2012	-	UK	E	NU	3	1, 4, 7
Clutterbuck *et al*., 2020 [2,1+, EV-]	Ecus, 2013; Clutterbuck & Midgley, 2015; Milligan *et al*., 2018; Clutterbuck & Lindsay, 2021; Lindsay & Clutterbuck, in prep; Hard Hill vegetation study (NEER004)	-	UK	E	NP	1	4
Cosgrove, 2004 [2-, EV-]	Holland *et al*., 2022	-	UK	S	-	7	8
Critchley *et al*., 2016 [2++, EV++]	Nisbet, 2004a,b; JNCC, 2009; Defra, 2016; ADAS *et al*., 2017	-	UK	E	National sample	1	8
Dallimer *et al*., 2012 [2+, EV+]*	Dallimer *et al*., 2010a,b	-	UK	E	PD	2	-
Davies *et al*., 2016a [2+, EV+]	Legg *et al*., 2007	-	UK	S	-	5	7
Davies *et al*., 2019 [1,2+, EV-]	-	-	UK	S	-	7	5

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Dixon *et al*., 2015 [2+, EV-]*	Dixon, 2012; Worrall *et al*. 2012; Qassim, 2015 [1+, EV-]	-	UK	E	PD, SP	-	-
Douglas & Pearce-Higgins, 2014 [2+, EV-]*	-	-	UK	S	-	2	-
Douglas *et al*., 2014 [2+. EV++]*	Pearce-Higgins & Grant, 2006 [in NEER004]; Buchanan *et al*., 2007	-	UK	E, S	SP	2	1
Douglas *et al*., 2015 [2++, EV++]*	Anderson *et al*., 2009 (in NEER004)	(C): Davies *et al*., 2016b,e; (R): Douglas *et al*., 2016a,b	UK	E, W, S	Across uplands	8	
Douglas *et al*., 2017 [2+, EV-]*	Garnett *et al*., 2019; Garnett, 2023	-	UK	E	NP	2	1
Drewitt, 2015 [2+, EV+]	Yallop & Thacker, 2015	-	UK	E	NYM	2	8
Dunn *et al*., 2016 [2-, EV-]	-	-	UK	W	-	3	1, 4
Fyfe & Woodbridge, 2012 [2+, EV-]*	-	-	UK	E	DM	1	-
Fyfe *et al*., 2018 [2+, EV+]*	-	-	UK	E	EX	1	-
Gagkas *et al*., 2022 [2+, EV++]	-	-	UK	S	-	7	-
Gao *et al*., 2016 [1+, EV-]	Gao *et al*., 2015, 2017	-	UK	E, W	DM, NP	4	-
Gao *et al*., 2017 [1+, EV-]	Gao *et al*., 2015, 2016	-	UK	E	YD	4	-
Garnett, 2023 [2+, EV-]	Garnett, 2013; Garnett & Thompson, 2016; Douglas *et al*., 2017 (esp. Appendix 2); Garnett *et al*., 2019; Thompson & Wilson, 2020	-	UK	E	NP	1	2
Gillingham *et al*., 2016b [4+, EV++]	Gillingham *et al*., 2016a	-	UK+	UK	National	2	1
Gillingham *et al*. 2016c [4+, EV++]	-	-	UK+	UK	National	1	-

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Granath *et al.*, 2016 [2,4+, EV+]	-	-	Canada & N Europe	UK	-	7	3
Grau-Andrés *et al.*, 2017a [1+, EV-]	Grau-Andrés, 2016	-	UK	S	-	-	-
Grau-Andrés *et al.*, 2017b [1+, EV-]*	Grau-Andrés, 2016	-	UK	S	-	-	-
Grau-Andrés *et al.*, 2019b [1+, EV-]	Grau-Andrés, 2016; Grau-Andrés *et al.*, 2018*, 2019a*	-	UK	S	-	5	7
Hedley, 2013 [2+, EV+]	Brown *et al.*, 2014; Noble *et al.*, 2018b	-	UK	E	NP, PD, SP, YD,	1	-
Heinemeyer & Swindles, 2018 [2+, EV-]	Heinemeyer *et al.*, 2019c	-	UK	E	NP	3, 4	-
Heinemeyer *et al.*, 2018 [2-, EV-]*	Heinemeyer *et al.*, 2019c	(C): Evans *et al.*, 2019; Young *et al.*, 2019, 2021; (R): Heinemeyer *et al.*, 2019b; (C & R): Defra, 2020; Heinemeyer, 2020	UK	E	FB, YD	3	-
Heinemeyer *et al.*, 2019a [1,2+, EV-]*	Heinemeyer *et al.*, 2019c	(C & R): Heinemeyer *et al.*, 2019c	UK	E	FB, YD	1	-
Heinemeyer *et al.*, 2019c [1+, EV-]*	Defra, 2011; Carroll *et al.*, 2015; Morton, 2016; Heinemeyer & Swindles, 2018; Heinemeyer *et al.*, 2018, 2019a, 2023a,b,c; Morton & Heinemeyer, 2018, 2019; Lindsay, 2020; Burn, 2021; Burn *et al.*, 2021; Heinemeyer, 2021, 2023a,b; Heinemeyer *et al.*, 2023	(C): Evans *et al.*, 2019; Young *et al.*, 2019, 2021; (R): Heinemeyer *et al.*, 2019b; (C & R): Defra, 2019; Heinemeyer, 2019c	UK	E	FB, YD	1	2, 3, 4

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Johnston, 2012 [2-, EV+]*	-	-	UK	E	NP, PD, SP, YD	2	4
Johnston & Robson, 2015 [2+, EV+]*	-	-	UK	E	NP, SP	2	4
Kennedy-Blundell, 2020 [1+, EV-]	Kennedy-Blundell *et al*., 2023	-	UK	E	PD	3	-
Kirkland *et al*., 2023 [2+, EV-]	-	-	Belarus, Ukraine	-	-	7	-
Lee *et al*., 2013a [1,2+, EV-]*	Lee & Marrs, 2020	-	UK	E	NP, PD	-	1
Lees *et al*., 2021 [2+, EV++]	-	-	UK	E	NP, NYM, PD, YD	8	1
Li *et al*., 2017 [2+, EV+]	-	-	UK	E	NP	4	3
Littlewood *et al*., 2019 [2+, EV+]	-	-	UK	E	CH, NP	4	3
Log *et al*., 2017 [2+, EV-]	-	-	Norway	-	-	5	7
Ludwig *et al*., 2018 [2+, EV-]*	Ludwig *et al*., 2017*, 2020	-	UK	S	-	2	1
Marrs *et al*., 2019 [1,2+, EV-]*	Garnett *et al*., 2000, 2001; Milligan *et al*., 2018; Hard Hill vegetation and carbon studies (NEER004) [more]	(C): Baird *et al*., 2019; Young *et al*., 2019, 2021; (R): Heinemeyer *et al*., 2019b; Heinemeyer, 2020 [more].	UK	E	NP	3	1
Matthews *et al*., 2020 [2+, EV++]	Matthews *et al*., 2018	-	UK	S	-	8	-
McCarroll *et al*., 2016a [2+, EV-]*	-	-	UK	E	SP	1	-
McCarroll *et al*., 2016b [2+, EV-]*	c.f. McCarroll *et al*., 2016a, 2017	-	UK	E	YD	1	-
McCarroll *et al*., 2017 [2+, EV-]*	c.f. Heinemeyer *et al*., 2018	-	UK	E	YD	1	-

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Milligan *et al*., 2018 [1++, EV-]*	Hard Hill vegetation study (in NEER004, pp. 87–90); Bailey 2019; Marrs *et al*., 2019*	(C): Gray and Levy, 2009; Lindsay, 2010; Ecus, 2013; Clutterbuck *et al*., 2020; Clutterbuck & Lindsay, 2021	UK	E	NP	1	6
Muñoz *et al.,* 2014 [2-, EV-]	Basanta *et al*., 1989	-	Spain	-	-	1	6
Natural England, 2021 [2+, EV++]	Natural England Moorland Map data	-	UK	E	National	8	-
Newey *et al*., 2016 [2+, EV+]*	-	-	UK	S	-	2	1
Newey *et al*., 2020 [2+, EV++]	Matthews *et al*., 2020	-	UK	S	-	1, 2	8
Noble *et al*., 2017 [1+, EV-]	Noble, 2018	-	UK	E	PD, NP	1	4
Noble *et al*., 2018a [1,2+, EV-]	O'Reilly, 2016; Milligan *et al*., 2018; Noble *et al*., 2019a; Noble, 2018, Hard Hill vegetation study (NEER004)	-	UK	E	NP	1	4
Noble *et al*., 2018b [2+, EV++]	Critchley *et al*., 2011a; Hedley, 2013; Brown *et al*., 2014; Noble, 2018; Noble *et al*., 2018c	-	UK	E	PD, SP, YD, NP + national sample	1	6
Noble *et al*., 2019a [1,2++, EV-]	Noble, 2018; Noble *et al*., 2018a; Hard Hill vegetation study (NEER004) [more]	-	UK	E	NP	1	5, 6
Noble *et al*., 2019b [2+, EV-]	Noble, 2018	-	UK	E	CH, NP, PD	1	-
Odoni, 2016 [2+, EV-]	-	-	UK	E	SP	4	-
Parry *et al*., 2015 [2+, EV+]*	-	-	UK	E	NP, SP	4	3
Perry *et al*., 2022 [2+, EV++]	-	-	UK	UK	National	7	-

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Qassim, 2015 [1+, EV-]	Worrall *et al*., 2013 [1,2+, EV-]; Qassim *et al*., 2013, 2014,	-	UK	E	PD	3, 4	-
Ritson *et al*., 2016 [1+, EV+]	-	-	UK	E	EX, DM	4	3
Roberston *et al*., 2017 [2+, EV++]*	-	-	UK	E	NP, FB, NYM, PD	2	1, 3
Roos *et al*., 2016 [2+, EV+]*	-	-	UK	S	-	2	-
Rosenburgh *et al*., 2013 [2+, EV-]*	-	-	UK	E	PD	4	3
Rowney *et al*., 2023 [2+, EV+]	Ombashi, 2019; Rowney *et al*., 2022	-	UK	E	EX	-	-
Santana *et al*., 2016 [2+, EV+]	Chapman *et al*., 1975; Miller, 1979; Alday *et al*., 2015; Santana *et al*., 2015; Santana & Marrs, 2016 [in NEER014]; Hard Hill vegetation study (NEER004)	-	UK	E, S	PD, NP	3	1
Shewring *et al*., 2024 [2++, EV++]	-	-	UK (GB)	E, S, W	National	8	-
Spracklen & Spracklen, 2023 [2+, EV+]	Scottish Natural Heritage, 2016; Scottish Government, 2020	-	UK	S	-	8	-
Swindell, 2017 [2+, EV-]	-	-	UK	E	PD	1	7, 8
Swindells *et al*., 2015 [2+, EV-]*	-	-	UK	NI	-	1	-
Swindells *et al*., 2016 [2+, EV-]*	-	-	UK	E	YD	1	-
Taylor *et al*., 2017 [1+, EV-]*	Taylor, 2015	-	UK	S	-	1	-
Thacker *et al*., 2015 [2++, EV++]*	Yallop *et al*., 2005; 2006a,b [all in NEER004]	-	UK	E	National	8	-
Turner & Swindells, 2012 [2+, EV-]*	-	-	UK	E	SP	4	2, 3

Main study reference [type, quality, external validity]	Supplementary study references	Relevant Comments and Responses	Country	UK nations	English area	Main sub-question(s)	Other sub-questions
Vane *et al*., 2013 [2+, EV-]*	-	-	UK	E	PD	3, 4	-
Velle & Vandvik, 2014 [1+, EV-]*	Velle *et al*., 2014	-	Norway	-	-	1	-
Velle *et al*., 2012 [1+, EV-]	-	-	Norway	-	-	1	-
Ward *et al*., 2013 [1,2+, EV-]	Walker *et al*., 2015, 2016*	-	UK	E	NP	3	1, 4
Ward *et al*., 2012 [1,2+, EV-]	Ward *et al*., 2007	-	UK	E	NP	3	1
Whitehead & Baines, 2018 [2+, EV-]	-	-	UK	E	NP	1	-
Whitehead *et al*., 2021 [2+, EV-]	-	-	UK	S	-	1	-
Wilkinson *et al*., 2023 [2+, EV+]	-	-	Northern peatlands (boreal, temperate)	UK	-	7	5
Wilson *et al*., 2021 [2++, EV++]	Moss *et al*., 2005; Newson *et al*., 2007; Fletcher *et al*., 2013	-	UK	E, S, W	National	2	-
Worrall *et al*., 2013a [1,2+, EV-]*	-	-	UK	E	PD	3	1
Wu *et al*., 2020 [1+, EV-]	-	-	Canada	-	-	4	3
Yusup *et al*., 2022 [1+, EV-]	Yusup *et al*., 2023	-	China	-	-	1	-
Zonneveld *et al*., 2024 [2+, EV-]	Zonneveld, 2019	-	UK	E	DM	2	-

UK nations: E = England, S = Scotland, W = Wales.

England areas: CH = Cheviot Hills, DM = Dartmoor, EX = Exmoor, FB = Forest of Bowland, NP = North Pennines, NU = Northumberland, NYM = North York Moors, PD = Peak District, SP = South Pennines, YD = Yorkshire Dales.

* Studies also included in post-NEER004 review by Ashby (2020).

Table A1.2. Evidence table of additional evaluated studies included in NEER014 reported in this review update in relation to the wildfire sub-question, in alphabetic order by lead author. The study type, quality and external validity are shown in the first column (see Tables 1-3 for details of the scoring system). Where applicable, supplementary studies and relevant comments (C) and responses (R) are included in columns 2 and 3. The other columns show which country/nations/areas each study was located in (a key to abbreviations used is given after the table) and which sub-questions it applies to. Studies included in NEER014 in relation to the wildfire sub-question but also included in NEER004 and this update in relation to other main sub-questions are not included in this table but area in Tables A1.1 and A1.3.

Main study reference [type, quality, EV]	Supplementary study references	Country	UK nation	English area	Main sub-question(s)	Other sub-questions
Arkle *et al.*, 2012 [2+, EV-]	-	USA	-	-	7	-
Brose & Wade, 2002 [2+, EV-]	-	USA	-	-	7	-
Camill *et al.*, 2009 [2++, EV-]	-	Canada	-	-	7	-
Cary *et al.*, 2009 [2+, EV-]	-	Australia, Canada, USA	-	-	7	-
Cassagne *et al.*, 2011 [2+, EV+]	-	France	-	-	7	-
Fernandes & Botelho, 2003 [3+, EV-]	-	Australia, USA	-	-	7	-
Hering *et al.*, 2009 [2++, EV-]	-	USA	-	-	7	-
Hudspith *et al.*, 2014 [2++, EV+]	-	Ireland	-	-	7	-
Keeley & Fotheringham, 2001 [2+, EV-]	-	USA	-	-	7	-
Keeley *et al.*, 1999 [2+, EV-]	-	USA	-	-	7	-
Legg *et al.*, 2006 [3+, EV+]	-	UK	UK	-	7	5
Luxmoore, 2018 [2,4+, EV+]	-	UK	S	-	7	-
Marino *et al.*, 2012 [2++, EV-]	Marino *et al.*, 2010; Marino *et al.*, 2014	Spain	-	-	7	-
Marino *et al.*, 2014 [2++, EV-]	Marino *et al.*, 2010; Marino *et al.*, 2012	Spain	-	-	7	-

Main study reference [type, quality, EV]	Supplementary study references	Country	UK nation	English area	Main sub-question(s)	Other sub-questions
Martin, 2018 [3+, EV-]	-	UK	E	WP	7	5
McCarthy & Tolhurst, 2004a [3-, EV-]	McCarthy & Tolhurst, 2004b	Australia	-	-	7	-
Mitchell *et al*., 2009 [2+, EV-]	-	USA	-	-	7	-
Moors for the Future, 2009 [2+, EV+]	-	UK	E	PD	7	-
NEER014, Appendix 2 [2+, EV++]	NE English wildfire data set	UK	E	National	7	-
Nuñez-Regueira *et al*., 2002 [2+, EV-]	-	Spain	-	-	7	-
Oliveira *et al*., 2016 [2+, EV-]	-	Spain	-	-	7	-
Penman *et al*., 2015 [2+, EV-]	Penman *et al*., 2014	Australia	-	-	7	-
Price, 2012 [2+, EV-]	Price & Bradstock, 2010, 2011; Price *et al*., 2012	Australia	-	-	7	-
Shang *et al*., 2004 [3+, EV-]	-	USA	-	-	7	-
Shive *et al*., 2013 [2+]	-	USA	-	-		-
Stevens-Rumann *et al*., 2013; [2++, EV-]	-	USA	-	-	7	-
Turetsky *et al*., 2014 [2+]	-	Worldwide	-	-	7	-
Volkova *et al*., 2014 [2+, EV-]	-	Australia	-	-	7	-
Waltz *et al*., 2014 [2+, EV-]	-	USA	-	-	7	-
Wu *et al*., 2013 [2+, EV-]	-	China	-	-	7	-

UK nations: E = England, S = Scotland. England area codes as in Table A1.1 apart from WP = West Pennines.

Table A1.3. Evidence table of evaluated studies in NEER004 in alphabetic order by lead author (oldest first). The study type, quality and external validity are shown in the first column (see Tables 1–3 for details of the scoring system). Where applicable, supplementary studies and relevant comments (C) and responses (R) are included in columns 2 and 3. The other columns show which country/nations/areas each study was located in (a key to abbreviations used is given after the table) and which sub-questions it applies to.

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
ADAS, 1997a [2++, EV+]	MAFF, 1993; ADAS, 1997b	-	UK	E	PD	8	-
Albertson *et al*., 2010 [2+, EV+]	Albertson *et al*., 2009; McMorrow *et al.,* 2009	-	UK	E	PD, YD	7	-
Allen, 1964 [1,2++, EV-]	-	-	UK	E	NP, LD (lab)	4	-
Allen *et al*., 2013 [2+]	-	-	UK	E	PD	3	-
Amar *et al*., 2011 [2++, EV++]	Sim *et al*., 2005	-	UK	-	-	2	-
Anderson *et al*., 2006 [4+]	-	-	UK	W	-	1	6
Anderson *et al*., 2009 [2-, EV+]	-	-	UK	-	-	8	-
Armstrong *et al.,* 2009 [3-]	-	-	UK	E		4	-
Armstrong *et al.,* 2012 [3+]	-	-	UK	E	PD	4	-
Aspray, 2012 [2++, EV+]	-	-	UK	E	PD	4	2
Aylan *et al.,* 2007 [4-]	-	-	UK	E	PD	7	-
Beharry-Borg *et al.,* 2011 [2+, EV-]	-	-	UK	E	YD	4	-
Benscoter *et al.,* 2011 [1+, EV-]	-	-	Boreal, (lab)	-	-	5	-
Brown & Bainbridge, 1990 [4-, EV+]	-	-	UK	-	-	2	-
Burch, 2008 [2-, EV-]	Burch, 2009	-	UK	E	NYM	1	-
Chambers *et al.,* 2007 [2+, EV-]	Chambers *et al*., 2000	-	UK	W	-	1	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
Chapman *et al.,* 2010 [2-, EV-]	-	(C): Yallop *et al*., 2011; (R): Chapman *et al.,* 2011	UK	E	YD	4	-
Chapman *et al.,* 2009 [2-, EV+]	-	-	UK	E	PD	1	-
Chen *et al.,* 2008 [1,2++, EV-]	-	-	UK	E	PD, (+ lab)	3	-
Clay & Worrall, 2011 [2+, EV-]	Clay, 2009	-	UK	E	PD	3	-
Clay *et al.,* 2010b [2++, EV-]	Clay, 2009	-	UK	E	NU	3	-
Clay *et al.,* 2012 [2+, EV-]	-						
Clutterbuck & Yallop, 2010 [2++, EV++]	Clutterbuck, 2009; Yallop & Clutterbuck, 2009	-	UK	E	SP, YD	4	-
Cotton & Hale, 1994 [1-, EV-]	Hale & Cotton, 1988, 1993	-	UK	E	SP	1	-
Coulson, 1988 [2-, EV+]	Butterfield & Coulson, 1983; Coulson & Butterfield, 1986	-	UK	E	North	2	-
Coulson *et al.,* 1992 [4+, EV+]	-	-	UK	-	-	1	-
Couwenberg *et al.,*2011 [2-, EV-]	-	-	Belarus	-	-	3	-
Critchley *et al.,* 2011a [2++, EV++]	JNCC, 2009; Glaves, 2017; Noble *et al*., 2018b	-	UK	E	-	8	-
Critchley *et al.,* 2011b [2++, EV++]	JNCC, 2009	-	UK	E	-	8	-
Currall, 1981 [1,2+, EV+]	Currall, 1989; Tucker, 2003	-	UK	S	-	1	6
Curtis & Corrigan, 1990 [2-, EV-]	-	-	UK	S	-	2	-
Daplyn & Ewald, 2006 [2-, EV+]	-	-	UK	E	PD	2	-
Davies, 2005 [2+, EV-]	-	-	UK	S	-	5	-
Davies & Legg, 2008 [2+, EV-]	Davies, 2001	-	UK	S	-	1	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
Davies & Legg, 2011 [1,2+, EV+]	-	-	UK	S	-	5	-
Davies *et al.,* 2008 [4+, EV+]	-	-	UK	-	-	7	-
Davies *et al.,* 2010a [2+, EV+]	-	-	UK	S	-	5	-
Dennis & Eales, 1997 [2+, EV+]	Dennis & Eales, 1999	-	UK	E	NU	2	-
Elliott, 1953 [2-, EV+]	-	-	UK	E	PD	1	-
Ellis, 2008 [3+, EV-]	-	-	UK	S	-	1	-
Eyre *et al,* 2003 [2+, EV-]	-	-	UK	S	-	2	-
Farage *et al.,* 2009 [2-, EV-]	-	(C): Legg *et al*., 2010; (R): Farage *et al*., 2010	UK	E	YD	3	-
Fletcher *et al.,* 2010 [1+, EV+]	GWCT, 2010	-	UK	E	NU	2	-
Forrest & Smith, 1975 [2-, EV-]	Forrest, 1971	-	UK	E	NP	1	-
Fullen, 1983 [3+, EV-]	-	-	UK	E	NYM	3	-
Garnett *et al.,* 2000 [1+, EV-]	Garnett & Stevenson, 2004	(C): Gray and Levy 2009; Lindsay, 2010; Clutterbuck *et al*. 2020; Young *et al*., 2019, 2021	UK	E	NP	3	-
Garnett *et al.,* 2001 [2++, EV-]	-	-	UK	E	NP	3	-
Glaves *et al*., 2005 [4+, EV+]	-	-	UK	-	-	2	-
Grand-Clement, 2008 [1,2-, EV-]	-	(C): Gray and Levy 2009; Lindsay, 2010; Clutterbuck *et al*. 2020; Young *et al*., 2019, 2021	UK	E	NP, PD	3	-
Grant *et al.,* 2012 [4+, EV+]	-	-	UK	-	-	2	-
Gray & Levy, 2009 [4+, EV+]	-	-	UK	-	-	3	-
Grayson *et al.*, 2012 [2+, EV+]	Grayson *et al.*, 2008	-	UK	E	SP, PD	4	-
Hamilton, 2000 [1,2-, EV-]	-	-	UK	S	-	1	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
Hard Hill vegetation study, 1965–2001 [1++, EV-] (NEER004, Appendix 3, pp. 87–90)	Forrest, 1961; Rawes & Williams, 1973; Rawes & Hobbs, 1979; Hobbs & Gimingham, 1980; Hobbs, 1981, 1984; Marrs *et al*., 1986; Adamson & Kahl, 2003; Stuart *et al*., 2004; Lee *et al*., 2013b; Milligan *et al*., 2018*; Bailey, 2019; Marrs *et al*., 2019b*	(C): Gray and Levy, 2009; Lindsay, 2010; Ecus, 2013; Clutterbuck *et al*., 2020; Clutterbuck & Lindsay, 2021	UK	E	NP	1	6
Hard Hill hydrology studies, 2005–08 [1+, EV-] (NEER004, Appendix 7, pp. 130–131)	Worrall *et al*., 2007; Worrall & Adamson, 2008; Clay, 2009; Clay *et al*., 2009a,b, 2010a	(C): Gray and Levy, 2009; Lindsay, 2010; Clutterbuck *et al*., 2020; Clutterbuck & Lindsay, 2021	UK	E	NP	4	6
Harris *et al.,* 2006 [2+, EV-]	-	-	UK	E	PD	1	-
Harris *et al.,* 2011a [2+, EV-]	Harris, 2011	-	UK	E	PD	5	3
Harris *et al.,* 2011b [2+, EV+]	Harris, 2011	-	UK	E	PD	1	-
Haworth & Thompson, 1990 [2+, EV+]	-	-	UK	E	SP	2	-
Hochkirch & Adorf, 2007 [2++, EV-]	-	-	Germany	-	-	2	-
Holden, 2005a [1,2+, EV++]	Holden, 2005b	-	UK	-	-	3	-
Holden *et al.,* 2012 [4++, EV+]	Holden *et al.,* 2011	-	UK	-	-	4	-
Holden *et al.,* 2013 [2+, EV+]	EMBER (see Table A1.1), especially Brown *et al*., 2014 [2+, EV-].	(C): Davies *et al*., 2016b; Ashby & Heinemeyer, 2019a,b; (R): Brown *et al*., 2016; Brown & Holden, 2019b, 2020	UK	E	PD, NP	4	-
Holmes *et al.,* 1993 [2+, EV+]	-	-	UK	W	-	2	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
IUCN, 2011 [4+, EV+]	Littlewood *et al.,* 2011; Lunt *et al.,* 2011; Worrall *et al*., 2011a	-	UK	-	-	1	-
JNCC, 2009 [4+, EV+]	Previous versions: JNCC, 2004, 2005, 2006, 2008; Jerram *et al*., 2001 and previous versions: Jerram & Drewitt, 1997, 1998; MacDonald *et al*., 1998	(C): Davies *et al*., 2016b	UK	-	-	1	-
Jones, 2005 [4+, EV+]	Sherry, 2005	-	UK	W	-	1	-
Kinako & Gimingham, 1980 [2+, EV-]	-	-	UK	S	-	3	-
Legg & Davies, 2009 [4+, EV+]	-	-	UK	S	-	5	-
Lindsay, 2010 [2,4+, EV+]	-	-	UK	-	-	3	-
Lindsay & Ross, 1994 [2++, EV-]	Lindsay, 1977	-	UK	E	CU	1	-
Littlewood *et al.,* 2011 [4+, EV+]	-	-	UK	-	-	1	-
Loftus, 1994 [2+, EV-]	-	-	Ireland	-	-	1	-
Lunt *et al.,* 2011 [4+, EV+]	-	-	UK	-	-	1	-
MacDonald, 2008 [2,4+, EV-]	-	-	UK	S	-	1	-
Marrs *et al.,* 2004 [1+, EV+]	-	-	UK	E	PD, YD	1	6
McDonald *et al.,* 1991 [2+, EV+]	Mitchell & McDonald, 1995	-	UK	E	-	4	-
McFerran, *et al.,* 1995 [2-, EV-]	-	-	UK	E	-	4	-
McMorrow *et al.,* 2009 [2,4+, EV+]	Albertson *et al*., 2009, 2010	-	UK	E	PD	7	-
Miles, 1971 [1+, EV-]	-	-	UK	S	-	1	6
Miller, 2008 [2++, EV]	-	-			-	4	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
Mitchell & McDonald, 1995, [2+, EV+]	McDonald *et al*., 1991	-	UK	E	YD	4	-
Moss *et al.,* 2005 [2++, EV++]	Ratcliffe, 1990; Tucker, 2003; Joys & Crick, 2004; Glaves *et al*., 2005; Crick *et al*., 2006; Newson *et al*., 2007; Wilson *et al*., 2021	-	UK	E	-	2	-
Mowforth & Sydes, 1989 [4+, EV+]	-	-	UK	-	-	1	-
Natural England, 2009 [2+, EV++]	-	-	UK	E	-	8	-
O'Brien *et al.,* 2005 [2-, EV-]	O'Brien *et al.,* 2009	-	UK	E	PD	4	-
O'Reilly, 2008 [2+, EV+]	-	-	UK	E	NU	1	-
Orwin & Ostle, 2012 [1,2+, EV-]	-	-	UK	E	-	3	1
Penny Anderson Associates, 2012 [2++, EV++]	Natural England, 2010	-	UK	E	-	8	3
Pattison & Lane, 2011 [4+]	-	-	USA	-	-	4	-
Pearce-Higgins & Grant, 2006 [2+, EV++]	-	-	UK	-	-	2	-
Pearce-Higgins *et al.,* 2009 [4+, EV+]	-	-	UK	-	-	2	
Picozzi, 1968 [2+, EV+]	-	-	UK	S	-	2+	-
Pietikäinen *et al.,* 1999 [2+, EV-]	-	-	Finland	-	-	3	-
Ramchunder *et al.,* 2009 [2+, EV+]	-	-	UK	E	NP	4	2
Ramchunder *et al.*, 2013 [2++, EV+]	Ramchunder *et al.*, 2010; Brown *et al*. 2009	-	UK	E	NP	4	2
Ratcliffe, 1990 [2,4+, EV+]	-	-	UK	-	-	2	-
Rawes & Williams, 1973 [2+, EV-]	Rawes & Hobbs, 1979	-	UK	E	NP	1	6

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
Ross *et al.,* 2003 [1+, EV-]	-	-	UK	E	NU	1	6
Rowell, 1980 [4+, EV+]	-	-	UK	-	-	1	-
Shaw *et al*., 1996 [4+, EV+]	Shaw *et al*., 1997	-	UK	UK	-	1	6
Shelter *et al*., 2008 [2+, EV-]	-	-	USA	-	-	3	-
Smith *et al.,* 2001 [2+, EV++]	-	-	UK	-	-	2	-
Sotherton *et al.,* 2009 [4+, EV+]	-	-	UK	-	-	2	-
Stewart *et al.,* 2004 [1++, EV+]	Stewart *et al*., 2005	-	UK/Ireland	-	-	1	-
Stone, 2006 [2-, EV-]	-	-	UK	E	PD	1	2
Taylor & Marks, 1971 [1+, EV-]	Marks & Taylor, 1972	(C): Clutterbuck *et al*. 2020	UK	E	NP	1	6
Tharme *et al.,* 2001 [2+, EV++]	-	-	UK	S, E	-	2	-
Thompson *et al.,* 1995 [4+, EV+]	-	-	UK	-	-	1	-
Tucker, 2003 [2,4+, EV+]	Tucker, 2004a,b	-	UK	-	-	2	2, 6
Usher, 1992 [2-, EV-]	-	-	UK	E	NYM	2	-
Ward *et al.,* 2007 [1,2+, EV-]	Ward *et al*., 2012	(C): Gray and Levy 2009; Lindsay, 2010; Clutterbuck *et al*. 2020	UK	E	NP	1	3, 4, 6
Worrall & Warburton, 2009 [2+, EV+]	-	-	UK	E	NP	4	-
Worrall *et al.,* 2011 [4+]	IUCN, 2011	-	UK	-	-	1–4	-
Worrall *et al.,* 2010a [2,4+, EV+]	-	-	UK	-	-	3	-
Worrall *et al.,* 2012 [2-, EV-]	Same as Worrall *et al.,* 2013b*	-	UK	E	PD	4	-
Yallop & Clutterbuck, 2009 [2++, EV+]	Clutterbuck & Yallop, 2010	-	UK	E	NYM, PD, SW,	4	-
Yallop *et al.,* 2005 [2+, EV++]	Thomas *et al*., 2004; Yallop *et al*., 2006b;	(C): Davies *et al*., 2016b	UK	E	-	8	-

Main study reference [type, quality, EV]	Supplementary study references	Relevant Comments and Responses	Country	UK nation	English area	Main sub-question	Other sub-questions
	Yallop *et al*., 2009; Thacker *et al*., 2015*						
Yallop *et al.* 2008 [2+, EV+]	White *et al*., 2004	-	UK	E	YD	4	-
Yallop *et al.,* 2010 [2++, EV+]	-	-	UK	E	SP	4	-
Yallop *et al.,* 2012 [2++, EV++]	-	-	UK	E	NP, NYM, PD,	8	-
Yallop *et al.,* 2006a [2+, EV+]	-	-	UK	E	NP	8	-

UK nations: E = England, S = Scotland, W = Wales.

England areas: CH = Cheviot Hills, CU = Cumbria, DM = Dartmoor, EX = Exmoor, FB = Forest of Bowland, NU = Northumberland, NP = North Pennines, NYM = North York Moors, PD = Peak District, SP = South Pennines, YD = Yorkshire Dales.

* Studies also included in post-NEER004 review by Ashby (2020).

In most cases the internal and external validity scores from NEER004 were retained (but the latter were not published in the report), though a few tweaks were made, especially to EV, to improve consistency across the wider evidence base spanning NEER004 and this update.

Appendix 2. Evidence search strings

The following search strings were used in searches run in the Scopus database in February 2021. The same strings were repeated in September 2023 with unnecessary, duplicate terms removed, which are the versions given below. A general, across sub-question, search string was also used in 2023.

Vegetation:

> (TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (biodiversity OR ecosystem* OR habitat* OR vegetation OR flor* OR plant* OR communit* OR sphagn* OR heather OR calluna OR molinia OR eriophor* OR composition OR structure OR function OR condition OR microtopograph* OR acrotelm OR restor* OR paleoecol* OR (peat AND record))) AND LANGUAGE (english) AND PUBYEAR > 2011

Fauna:

> (TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (fauna OR animal* OR bird* OR mammal* OR insect* OR invertebrate* OR reptile* OR population OR breeding* OR grouse*)) AND LANGUAGE (english) AND PUBYEAR > 2011

Carbon:

> (TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (carbon* OR sequestration OR storage OR “carbon budget*" OR "carbon stock*" OR "carbon cycle*" OR flux*)) AND LANGUAGE (english) AND PUBYEAR > 2011

Water:

> (TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (water* OR flow OR flood* OR hydrolog* OR chemistry OR doc OR dom OR poc OR pollut* OR metal* OR infiltrat* OR overland*)) AND LANGUAGE (english) AND PUBYEAR > 2011

Grazing interaction:

> (TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (

graz* OR livestock OR stock*)) AND LANGUAGE (english) AND PUBYEAR > 2011

Severity, wildfire and extent combined:

(TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland) AND TITLE-ABS-KEY (intens* OR severity OR rotation* OR (return AND period*) OR frequency OR extent OR area OR map* OR aerial* OR areal* OR satellite* OR (remote AND sensing) OR (land AND cover) OR (designated AND site*) OR sssi OR wildfire* OR hazard* OR risk* OR prevent* OR resilien* OR (fuel AND load))) AND LANGUAGE (english) AND PUBYEAR > 2011

General search:

(TITLE-ABS-KEY (fire* OR burn*) AND TITLE-ABS-KEY (peat* OR blanket OR bog OR mire OR fen OR moor* OR heath* OR upland)) AND LANGUAGE (english) AND PUBYEAR > 2020

Appendix 3. Copies of quality assessment checklists

Table A3.1. Quantitative experimental quality checklist.

Review sub-topic question(s)	*Vegetation, fauna, carbon, water, fire behaviour, grazing interaction, wildfire interaction, extent/frequency.*	
Study citation		
Study type categories	**Broad type**: *1: quantitative experimental OR 1-4: review [only use this form if meta-analysis]*	**Specific type**: *RCT, meta-analysis, (controlled) before & after, laboratory, gaseous/fluvial fluxes, soil/water chemistry, hydrology, water quality/flow, peat/carbon accumulation, other (describe)*
Assessed by & date(s)		

Section 1: Areas/population(s)		
1.1 Were the wider source area(s) identified and well described? What area(s) did the study occur in and cover? Was a national sample done or, if not, which wider regions/areas/ groups of sites (e.g. GO Region(s), NCAs etc.) were the study site(s) in and were there multiple areas? What were the target habitat(s)/ vegetation types (peatland or other(s)) and other biodiversity (species/groups) of the area(s) and were they well described?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
1.2 Were the eligible area(s) (the 'sample frame') to be sampled representative of the source area(s)?	☐++	**Comments**:

What habitats and vegetation types occurred and was the floristic diversity (or spp. including fauna) and condition/state representative of the source area(s)? [In a national sample, the eligible area could be the same as the source area.] How was the eligible area selected? Were important vegetation types/species/groups under-represented?	☐+ ☐- ☐NR ☐NA	
1.3 Were the sampled area(s) (e.g. plots, transects, quadrats etc.) representative of the eligible area(s) and was their sampling unbiased? What was the method of selection and was it well described (was it random)? Did it include replication (spatial/temporal)? Were there any sources of bias? Were any inclusion/exclusion criteria explicit and appropriate? Were the habitat(s) typical, un-/little-modified or modified/degraded (e.g., *Calluna*- or other single spp.-dominated, spp.-poor etc.)?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
Section 2: Method of allocation to intervention (or comparison/control)		
2.1 How were samples allocated to management intervention(s) and/or any comparisons and controls, and how was selection bias minimised? Was allocation randomised (++)? If not randomised, was significant confounding likely/not likely?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.2 What were the management intervention(s) (usually burning) and any comparisons (e.g., cutting and	☐++	**Comments**:

controls (e.g., 'unburned') and were they well described and appropriate? In sufficient detail to replicate? Were comparisons/ controls appropriate? Were treatments repeated at different intervals? Was there an 'unburnt'/not recently burnt control? Were 'baseline' measurements taken prior to interventions and for how long? Was pre-intervention (burning/other) management described?	☐+ ☐- ☐NR ☐NA	
2.3 Was exposure to the management intervention(s) and/or any comparisons and controls adequate? Was any lack of exposure (e.g., burning incomplete/very low severity) sufficient to cause important bias? Consider consistency of implementation (e.g., was there any unplanned variation in timing/frequency of treatment(s)?).	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.4 Was any contamination acceptably low? Did any of the comparison/control treatments receive the management intervention(s), e.g., burning, or vice versa? If so, was it sufficient to cause important bias?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.5 Were any other intervention(s) received and, if so, were they similar across treatments, any comparisons and controls, and blocks/plots etc?	☐++ ☐+	**Comments**:

Did any treatments/plots etc. receive additional 'interventions' (i.e. unplanned management, e.g., cutting, 'wildfire') and/or were any treatments done at different time intervals or missed?	☐- ☐NR ☐NA	
2.6 Are the source/eligible/sampled area(s) representative and hence applicable to the UK/England upland peatland resource [EV++]? If not, might they be representative of smaller geographical areas, e.g., regions, NCAs, site(s) [EV+/-] etc?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.7 Did the intervention(s) and any comparisons and controls reflect normal UK 'real-world' practice(s)? e.g., shape, size/scale, patterning, overall extent and frequency, and method/type, of burning or other treatments? At what scale was sampling done (e.g., plot, transect, (sub-)catchment, site, area etc.).	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
Section 3: Outcomes		
3.1 What were the outcome measures (response variables and any explanatory variables), and were they and procedures reliable? Were they subjective or objective? How reliable were the outcome measures (e.g., inter- or intra-reliability scores, observer bias assessments)? Was there any indication that measures had been validated/subject to other QA?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:

3.2 Were all outcome measurements complete? Were they completed across all/most of the study area(s) (that met the defined study outcome definitions)?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.3 Were all important outcomes assessed? Were all important positive and negative effects recorded and assessed by the variables/measurements used? Were any important outcomes not recorded or assessed? Was it possible to determine the overall balance of benefits and/or harms versus comparisons?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.4 Were the outcome measures relevant to the study objectives and especially to the review questions? And if surrogate/proxy outcome measures/variables were used, what were they and did they provide a reliable indication of the scale and direction of important effect(s)?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.5 Were there similar post-treatment time intervals across treatments, and in any comparison and control treatments? Were variables measured at multiple time intervals post-treatment?	☐++ ☐+ ☐- ☐NR	**Comments**:

	☐NA	
3.6 Was the post-treatment time interval meaningful? What were the time intervals and were they long enough to assess medium-/long-term effects?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
Section 4: Analyses		
4.1 Were exposure and any comparison and control groups similar at baseline? If not, was this taken into account and adjusted for in the analyses and interpretation? Were there any differences between groups in important confounding factors at baseline?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
4.2 Was the study sufficiently powered to detect an intervention effect (if one existed)? Was a power calculation given (a power of 0.8 is the conventionally accepted standard)? If not, was there a statement on expected effect size (from comparison/control/other data)? Does the sample size seem adequate to allow an effect to be detected?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
4.3 Were estimates of effect direction, size and ecological/environmental importance (and whether meaningful) given or calculable?	☐++	**Comments**:

	☐+ ☐- ☐NR ☐NA	
4.4 What analytical methods were used and were they appropriate? Were any important differences in post-treatment time and likely confounding factors controlled/adjusted for and pseudoreplication avoided? Were any sub-group analyses pre-specified (or explanatory)? (Explanatory sub-group analyses can provide valuable information but can be underpowered and should not be over emphasised.)	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
4.5 Was the precision of the intervention effects given or calculable? Were they meaningful? Were confidence intervals and or p-values for the effect estimates given or calculable? (If they are wide, it may suggest that the study was underpowered.)	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:

Section 5: Summary		
5.1 Are the results of the study internally valid (i.e. unbiased)? How well did the study minimise sources of bias (i.e. adjusting for potential confounding factors)?	☐++ ☐+	**Comments**: *++ 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 methodological 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).*

Were there any significant flaws in the study design and analysis?	☐-	- *Few or no methodological criteria have been fulfilled. The conclusions of the study are thought likely or very likely to alter (high risk of bias).*
5.2 Are the findings generalisable to the wider source area(s) and nationally (i.e. externally valid/applicable)? Are sufficient details given to determine whether the findings can be generalised nationally or across area(s)/population(s) (i.e. habitat, species etc.)? This should draw particularly on 1.1-1.3 (which may also relate to internal validity/quality, especially re. any potential sampling bias [1.3]) and 2.6 and 2.7 (last may also relate to internal validity).	☐++ ☐+ ☐-	**Comments**:

Results

A concise summary of key results relevant to the topic and sub-topic questions (for Evidence Table) This should concentrate more on the results than the authors' interpretation, though the latter can be referred to and commented on.	**Results:**

Notes:

Any limitations identified by author(s):

Any limitations identified by others' formal comments/critiques:

Any limitations identified by the review team:

Any evidence gaps and/or recommendations for further research:

Source(s) of funding and any conflicts of interest given:

Any other notes:

Table A3.2. Quantitative observational/correlative studies quality checklist.

Review 'sub-topic' question(s)'	*Vegetation, fauna, carbon, water, fire behaviour, grazing interaction, wildfire interaction, extent/frequency.*	
Study citation		
Study type categories	**Broad type**: *2: quantitative observational/correlative* *OR: 1-4: review [could use expt. form for meta-analysis of RCT, CBA etc or qualitative form if more appropriate, esp. for traditional lit. review]*	**Specific type**: *meta-analysis, systematic review, critical synthesis, traditional review, before & after, site comparisons, space-for-time (chronosequence), survey/monitoring, Earth Observation (remote sensing), laboratory, modelling, correlational, case study, gaseous/fluvial fluxes, soil/water chemistry, hydrology, water quality/flow, peat/carbon accumulation, palaeoecological, other (describe)*
Assessed by & date(s)	D Glaves, 02/12/21	

Section 1: Areas/populations		
1.1 Were the wider source area(s) identified and well described? What area(s) did the study occur in and cover? Was a national sample done or, if not, which wider regions/areas/groups of sites (e.g. GO Region(s), NCAs etc.) were the study site(s) in and were there multiple areas? What are the target habitat(s)/ vegetation types (peatland or other(s)) and other biodiversity (species/groups) of the area(s) and were they well described?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
1.2 Were the eligible area(s) (the 'sample frame') to be sampled representative of the source area(s)? What habitats and vegetation types occurred and was the floristic diversity (or spp. including fauna) and condition/state representative of the source area(s)? [In a	☐++ ☐+ ☐-	**Comments**:

national sample, the eligible area could be the same as the source area.] How was the eligible area selected? Were important vegetation types/species/groups under-represented?	☐NR ☐NA	
1.3 Were the sampled area(s) (e.g. plots, transects, quadrats etc.) and habitats/flora /fauna representative of the eligible area(s) and was their sampling unbiased? What was the method of selection and was it well described (was it random)? Did it include replication (spatial/temporal)? Were there any sources of bias? Were any inclusion/exclusion criteria explicit and appropriate? Were the habitat(s) typical, un/little modified or modified/degraded (e.g., *Calluna*- or other spp.-dominated, spp.-poor etc.) and representative of the wider eligible area sampled?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
Section 2: Method of selection/allocation to intervention (or comparison/control)		
2.1 What were the management intervention(s) (usually burning) and any comparisons (e.g., cutting) and controls (e.g., 'unburned')? How were the intervention, comparison and control areas selected and how was selection bias minimised? Were comparisons/controls appropriate? Were 'baseline' measurements taken prior to interventions and pre-treatment (burning/other) management described?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.2 What explanatory variables were recorded (that may explain changes in the response variables) and was their selection based on a sound theoretical basis?	☐++	**Comments**:

	☐+ ☐- ☐NR ☐NA	
2.3 Was any contamination acceptably low? Did any of the comparison/control areas receive the exposure or vice versa? If so, was it sufficient to cause important bias?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.4 How well were likely confounding factors identified and controlled? Were there likely to be other confounding factors not considered or appropriately adjusted for? Was this sufficient to cause bias?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
2.5 Is/are the setting(s) representative and hence applicable to the UK/England and did the intervention(s) reflect normal UK practice [EV++]? If not, might they be representative of smaller geographical areas, e.g., regions, NCAs, site(s) etc. [EV+/-]?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:

Section 3: Outcomes		
3.1 What were the outcome measures (response variables) and were they and procedures reliable? Were they subjective or objective? How reliable were the outcome measures (e.g., inter- or intra-reliability scores, observer bias assessments)? Was there any indication that measures had been validated/subject to other QA?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.2 Were all outcome measurements complete? Were they completed across all/most of the study area(s) (that met the defined study outcome definitions)?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.3 Were all important outcomes assessed? Were all important positive and negative effects recorded and assessed by the variables/measurements used? Were some important outcomes not recorded or assessed?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.4 Were the outcome measures relevant to the study objectives and especially to the review questions? And if any surrogate/proxy outcome measures were used, what were they and did they measure what they set out to?	☐++ ☐+ ☐-	**Comments**:

	☐NR ☐NA	
3.5 Were there similar follow up times in exposure across treatments, and in any comparison and control groups? Were variables measured at multiple time intervals?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
3.6 Was the follow up time meaningful? What were the time intervals and were they long enough to assess medium-/long-term effects?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
Section 4: Analyses		
4.1 Was the study sufficiently powered to detect an intervention effect (if one existed)? Was a power calculation given (a power of 0.8 is the conventionally accepted standard)? If not, was there a statement on expected effect size (from comparison/control/other data)? Does the sample size seem adequate to allow an effect to be detected?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
4.2 Were sufficient, multiple explanatory variables considered in the analysis?	☐++	**Comments**:

	☐+ ☐- ☐NR ☐NA	
4.3 Were the analytical methods appropriate? What methods were used? Were important differences in follow-up time and likely confounding factors controlled and adjusted for? Were any sub-group analyses pre-specified (or explanatory)?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:
4.4 Was the precision of the intervention effects given or calculable? Was any association meaningful? Were confidence intervals and or p-values for the effect estimates given or calculable? Was the direction and size of the effects given and are they ecologically/environmentally meaningful and important?	☐++ ☐+ ☐- ☐NR ☐NA	**Comments**:

Section 5: Summary		
5.1 Are the results of the study internally valid (i.e. unbiased)? How well did the study minimise sources of bias (i.e. adjusting for potential confounding factors)?	☐++ ☐+ ☐-?	**Comments**: *++ 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 methodological 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).*

Were there significant flaws in the study design and analysis?		- *Few or no methodological criteria have been fulfilled. The conclusions of the study are thought likely or very likely to alter (high risk of bias).*
5.2 Are the findings generalisable to the wider source area(s) and especially nationally (i.e. externally valid/applicable)? Are there sufficient details given to determine if the findings of can be generalised across the source population (i.e. habitat, species) and nationally? This should draw particularly on 1.1-1.3 (which may also relate to internal validity, especially re. any potential sampling bias [1.3]), and 2.5.	☐++ ☐+ ☐-	**Comments**:

Results

A concise summary of key results/findings relevant to the topic and sub-topic questions (for Evidence Table) This should concentrate more on the results than the authors' interpretation, though the latter can be referred to and commented on**.**	**Summary:**

Notes:

Any limitations identified by author(s):

Any limitations identified by others' formal comments/critiques:

Any limitations identified by the review team:

Any evidence gaps and/or recommendations for further research:

Source(s) of funding and any conflicts of interest given:

Any other notes:

Table A3.3. Qualitative studies quality checklist.

Review 'sub-question(s)'		
Study citation		
Study type categories	Broad type: *1: quantitative experimental, 2: quantitative observational/correlative, 3: qualitative, 1-4: review, [4: opinion/consensus]*	Specific type: *RCT, meta-analysis, systematic review, critical synthesis, traditional review, CBA, site comparisons/TFS, survey/monitoring, EO, laboratory, modelling, correlational, case study, gaseous/fluvial fluxes, soil/water chemistry, hydrology, water quality/flow, peat/carbon accumulation, palaeoecological*
Assessed by & date		

Section 1: Theoretical approach		
1.1 Is a qualitative approach appropriate? For example: - Does the research question seek to understand processes or structures, or illuminate subjective experiences or meanings? - Could a quantitative approach have better addressed the research question?	☐ Appropriate ☐ Inappropriate ☐ Not sure	**Comments**:
1.2 **Is the study clear in what it seeks to do?** For example: - Is the purpose of the study discussed – aims/objectives/ research questions? - Is there adequate/appropriate reference to literature? - Are underpinning values/ assumptions discussed?	☐ Clear ☐ Unclear ☐ Mixed	**Comments**:
1.3 How defensible/rigorous is the research design/methodology? For example:	☐ Defensible ☐ Indefensible	**Comments**:

-Is the design appropriate to the research question? -Is a rationale given for using a qualitative approach? - Are there clear accounts of the rationale for sampling, data collection and data analysis techniques used? - Is the selection of cases / sampling strategy theoretically justified?	☐ Not Sure	
Section 2: Study design		
2.1 How defensible/rigorous is the research design/methodology? For example: -Is the design appropriate to the research question? -Is a rationale given for using a qualitative approach? - Are there clear accounts of the rationale for sampling, data collection and data analysis techniques used? - Is the selection of cases/sampling strategy theoretically justified?	☐ Defensible ☐ Indefensible ☐ Not Sure	**Comments**:
Section 3: Data collection		
3.1 How well was the data collection carried out? For example: -Are data collection methods clearly described? -Were the appropriate data collected to address the research question? - Was the data collection and record keeping systematic?	☐ Appropriately ☐ Inappropriately ☐ Not sure/ inadequately reported	**Comments**:
Section 4: Trustworthiness		
4.1 Is the role of researcher clearly described? For example: -Has the relationship between the researchers and intervention group been adequately considered?	☐Clearly described ☐Unclear ☐ Not described	**Comments**:

4.2 Is the context clearly described? For example - were observations made in a sufficient variety of circumstances? - was context bias considered?	☐Clear ☐ Unclear ☐ Not sure	**Comments**:
4.3 Were the methods reliable? For example: -Was data collected by more than one method? -Is there justification for triangulation or for not triangulating? - Do the methods investigate what they claim to?	☐ Reliable ☐ Unreliable ☐ Not sure	**Comments**:
Section 5: Analyses		
5.1 Is the data analysis sufficiently rigorous? For example: -Is the procedure explicit? -How systematic is the analysis, is the procedure reliable? -Is it clear how the themes and concepts were derived from the data?	☐ Rigorous ☐ Not rigorous ☐ Not sure/not reported	**Comments**:
5.2 Are the data 'rich'? For example: -how well are the contexts of the data described? -Has the diversity of perspective and content been explored? -Are responses compared and contrasted?	☐ Rich ☐ Poor ☐ Not sure/not reported	**Comments**:

5.3 Is the analysis reliable? For example: -Did more than one researcher theme and code data? -If so, how were differences resolved? -Were negative/discrepant results addressed?	☐ Reliable ☐ Unreliable ☐ Not sure/not reported	**Comments**:
5.4 Are findings convincing? For example, are: - Findings clearly presented? - Findings internally coherent? - Extracts from original data included? - Data appropriately referenced? - Is reporting clear and coherent?	☐ Convincing ☐ Unconvincing ☐ Not sure	**Comments**:
5.5 Are the findings relevant to the aims of the study?	☐ Relevant ☐ Irrelevant ☐ Partially relevant	**Comments**:
5.6 Are the conclusions clear and justified? For example: - How clear are the links between data interpretation and conclusions? - Are the conclusions plausible and coherent? - Have alternative explanations been explored and discounted? - Do they enhance understanding of the research topic? - Are the implications of the research clearly defined?	☐ Adequate ☐Inadequate ☐ Not sure	**Comments**:

- Is there adequate discussion of the limitations encountered?		
Section 6: Ethics		
6.1 How clear and coherent is the reporting of ethics? For example: - Have ethical issues been taken into consideration? - Are they adequately considered? - Have the consequences of the research been considered? - Was the study approved by an ethics committee?	☐ Appropriately ☐ Inappropriately ☐ Not sure/not reported	**Comments**:

Section 7: Overall assessment		
7.1 As far as can be ascertained from the publication/report, how well was the study conducted? For example: - Are data collection methods clearly described? - Were the appropriate data collected to address the research question? - Was the data collection and record keeping systematic?	☐++ ☐+ ☐-	**Comments**:

Notes:

Any limitations identified by author(s)

Any limitations identified by the review team

Any evidence gaps and/or recommendations for further research

Source(s) of funding and any conflicts of interest given

Any other notes

Appendix 4. Assessment of research recommendations identified in the NEER004 and NEER014 reviews

Table A4.1. Research recommendations from NEER004 and the extent to which they have or are being addressed.

NEER004 research recommendations	Evidence in review update and from other relevant ongoing or published studies that Natural England is aware of
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 years) which are currently restricted to the Hard Hill experiment at Moor House [NNR, North Pennines]. Ideally these should consider type of burning (for example, 'cool' burns). Such studies should also [incorporate] 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 [then] nearly completed NERC-funded EMBER (Effects of Moorland Burning on the Ecohydrology of River basins) project [(Brown *et al*., 2014 [2+, EV+]): https://water.leeds.ac.uk/our-missions/mission-1/ember/.]	Overall, studies in the Pennines still dominate the literature, with the M19 NVC vegetation community being studied most. Most recent studies were still in the Pennines and Scotland. Little recent research has been carried out on wet heath and even less on flushes, fens (including valley mires) and swamps. Major recent studies by Brown *et al*. (2014 [2+, EV+]) and Heinemeyer *et al.* (2019c/2023 [1+, EV-]) are focused upon the Pennines and adjacent Bowland Fells. Grau-Andres' experimental work (classed as three studies in this update: 2017a/b, 2019b [all 1+, EV-]) was based in Scotland with one study site being dry heath and the other a raised bog. This said, there were 13 recent experimental studies (including laboratory studies) and a further 13 that included experimental aspects, though like Grau-Andres' studies all covered relatively short time periods, and the only one covering multiple burn rotations was at the Hard Hill experiment which remains the most investigated and long-running site for burning (and grazing interaction) studies.
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.	Brown *et al.* (2014 [2+, EV+]) and Heinemeyer *et al*. (2019c/2020 [1+, EV-]) found that recovery of water table depths to pre-burning levels could take up to a decade following burning. Four palaeoecological studies recorded changes in the vegetation community within the peat archive likely at least in part as a result of burning: Rowney *et al*. [2+, EV+], Fyfe *et al*. [2+, EV+], McCarroll *et al*. [2+, EV-], Blundell and Holden [2+, EV-].

NEER004 research recommendations	**Evidence in review update and from other relevant ongoing or published studies that Natural England is aware of**
	Though *Sphagnum* spp. were recorded as a group in 11 studies in most cases they were not recorded or reported to individual *Sphagnum* species level (in some cases because some species weren't sufficiently frequent to include in statistical analyses), though in some cases they were: Milligan *et al*. (2018 [1++, EV-]/Lee and others 2013b) and Noble *et al*. (2018a [1,2+, EV-]) at the Hard Hill experiment (but using different methods), Noble *et al*. (2018b [2+, EV++]/Hedley, 2013, condition dataset), Grau-Andrés et al. (2019b/a, 2017b [both 1+, EV-]); Heinemeyer *et al*. (2019c/2023 [1+, EV-]).
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 [micro]topography and condition, not just in vegetation but in wider studies[/subjects], 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.	The Hard Hill experiment plots have been further investigated and described more completely by Clutterbuck *et al.* (2020 [1,2+, EV-]) including past burns prior to the experiment, gross morphology, microtopographic variation, peat depth, and for Block D nanotope features and synusial vegetation of nanotope features, now extended to all blocks (Lindsay & Clutterbuck in prep.). Ecus (2013) also reported on some characteristics of the Hard Hill site. The EMBER project sites had a separate vegetation assessment carried out, including NVC community/sub-community and CSM habitat condition (Hedley, 2013 [2+, EV+]. However, some study sites continue to not be very well described in terms of vegetation and habitat types, degree of modification, condition and microtopography.
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 microtopography) and function.	See above regarding study site characteristics.
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,	Heinemeyer *et al.* 2019c [1+, EV-] BD51047/Peatland-ES, though the study objectives and focus have to some extent evolved since the project inception when the focus was testing methods of reducing *Calluna* over-dominance to promote habitat restoration (Defra, 2011). The focus now seems to be more on comparing the effects of burning with cutting (the methods tested) particularly on carbon, but also on

NEER004 research recommendations	Evidence in review update and from other relevant ongoing or published studies that Natural England is aware of
in a [then] current Defra Environmental Stewardship research project, BD5104.	water and biodiversity. This may in part reflect the priorities of the co-funders of the second, Peatland-ES-UK, phase after the first five years of Defra funding.
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).	BD5104/Peatland-ES-UK (above) has monitored cranefly abundance as avian prey as did Douglas & Pearce-Higgins (2014 [2+, EV-]). A further three recent studies the effects of burning on terrestrial invertebrates: Bargmann *et al*. (2015/2016 [2+, EV-]), Byriel *et al*. (2023 [2+, EV-]) and Newey *et al*. (2020 [2+, EV++]) compared species distribution data with an index of mapped burning intensity on Scottish grouse moors for selected flora and fauna species (in 1-km squares) including one invertebrate, the green hairstreak butterfly and one reptile, the adder. Fifteen recent studies provide information on the direct and indirect effects of burning, other habitat management, and a range of other explanatory variables, particularly physical characteristics related to climate and topography, on upland peatland breeding birds (Section 5, paras. 5.24–5.56).
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).	Wilson *et al*. (2021 [2++, EV++]) carried out a re-analysis of data on the timing of breeding of upland birds in England, Scotland and Wales, to assess whether rotational burning poses a threat to populations of these species and whether any such threat varies in space and time (updating a similar review by Moss and others (2005 [2++, EV++], NEER004, para. 5.26 and Appendix 4, pp.109–110). In addition, Zonneveld *et al*. (2024 [2+, EV-]) studied the timing of breeding of three moorland passerines, stonechat, whinchat and meadow pipit, at a Dartmoor in relation to the timing of burning and bracken management.
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.	Recent studies have provided more evidence on this (see carbon comparison of NEER004 and recent studies, Table 30) including;

NEER004 research recommendations	Evidence in review update and from other relevant ongoing or published studies that Natural England is aware of
	• 5 recent studies on NEE CO_2. (Grau-Andrés *et al*., 2019b [1+, EV-]; Heinemeyer *et al*., 2019c /2023 [1+, EV-]; Clay *et al*., 2015 [2+, EV-]; Dixon *et al*., 2015 [2+, EV-]; Ward *et al*., 2013 [1,2+, EV-]). • 3 recent studies on methane (Heinemeyer *et al*., 2019c /2023 [1+, EV-]; Grau-Andrés *et al*., 2019b [1+, EV-]; Clay *et al*., 2015 [2+, EV-]). • 1 recent study on net GHG flux (Heinemeyer *et al*., 2023 [1,2+, EV-]). • 2 recent studies on char production (Kennedy-Blundell, 2020 [1+, EV-]; Worrall *et al*., 2013a [1,2+, EV-]).
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].	3 recent studies reported on burning effects on aquatic invertebrates: Brown *et al*. (2013/2019 [2+, EV+]); Johnston & Robson (2015 [1+, EV+]); Johnston (2012 [2-, EV+]).
Studies of the effects of differences in the intensity/severity of fires and characteristics of burn patches such as size, shape, location [for example, in relation to slope, watercourses etc], distribution etc.	New studies have provided some evidence on this [see severity comparison table for detail]: 4 studies reporting severity effects on vegetation (Grau-Andrés *et al.* 2017a, 2019b, both [1+, EV-]; Noble *et al.*, 2019a [1,2++, EV+]. 3 studies reporting severity effects on carbon: Grau-Andrés *et al.*, 2017a, 2019b, both [1+, EV-]; Kennedy-Blundell, 2020 [1+, EV-]. 2 studies reporting severity effects on water (Grau-Andrés *et al.*, 2019b [1+, EV-]; Noble *et al.,* 2019a [1,2++, EV+]).
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.	National data still collected and collated by FRS and some by other bodies, e.g., Natural England, MoD, some National Parks etc. But no collation between these different sources. Some data collected on characteristics, e.g., severity of wildfires, but little on the relationship with managed burning. Also see Table A4.2 below. Recent evidence that managed burns escaping control cause a proportion of wildfires, particularly in the uplands in spring: de Jong *et al*. [year] [2++, EV++]; 3 [2+]: Luxmoore, 2018 [2+, EV++]; Moors for the Future, 2009 [2+, EV+; NEER014, Appendix 2 [2+, EV++];

NEER004 research recommendations	Evidence in review update and from other relevant ongoing or published studies that Natural England is aware of
	Cosgrove, 2004 [2-, EV-]; 3 [3+]: Legg *et al*., 2006 [EV+]; Worrall *et al*., 2011a [EV+]; Martin, 2018 [EV-].
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.	Recent studies have provided considerable new evidence on this (see extent comparison table for detail): 9 studies reporting burning extent: Allen *et al*., 2016 [2+, EV-]; Blundell *et al*., 2021 [2+, EV++]; Douglas *et al*., 2015 [2++, EV++]; Drewitt, 2015 [2+, EV+]; Lees *et al*., 2021 [2+, EV++]; Matthews *et al*., 2020 [2+, EV++]; Natural England, 2021 [2+, EV++]; Swindell, 2017 [2+, EV-]; Thacker *et al*., 2015 [2++, EV++]). 8 studies reporting mixed evidence of change over time: Douglas *et al.,* [2++, EV++]; Matthews *et al*., [2+, EV++]; Thacker *et al*., 2015 [2++, EV++]; Drewitt, 2015 [2+, EV+]; Critchley *et al*., [2++, EV++]; Allen *et al*., 2016 [2+, EV-]; Swindell, 2017 [2+, EV-]; Natural England, 2021 [2+, EV++].
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.	Still not available for grouse moors, though available for designated sites and peat soils as reported in several recent studies (see above).
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.	Little progress in relation to this recommendation.

Table A4.2. Research recommendations from the wildfire evidence review (NEER014) relevant to the relationship between managed burning and wildfire and the extent to which they have or are being addressed.

NEER014 research recommendation	Relevant ongoing or published studies that Natural England is aware of
Where possible, standardization of the range of variables recorded and definitions used, particularly cause of ignition, between the Home Office's national Incident Recording Scheme (IRS) and other wildfire recording schemes to enable compatibility of data nationally.	Review of wildfire Incident Recording Scheme in progress by Home Office and FRS. Ongoing NERC UKFDRS project investigating data required, and environmental variables of UK vegetation, to develop a Fire Danger Rating System for UK (information gathering and not producing an actual FDR system) led by University of Manchester: https://ukfdrs.com/partners/ and https://gtr.ukri.org/projects?ref=NE%2FT003553%2F1. Also, the Scottish Fire Danger Rating System project: https://www.scottishfiredangerratingsystem.co.uk/ and Taylor and others (2021): https://www.scottishfiredangerratingsystem.co.uk/sites/www.scottishfiredangerratingsystem.co.uk/files/SFDRS-Research-Report-Final-15-2-2022.pdf.
Investigation of the relationship between routine managed burning and prescribed burning (and cutting/mowing and other management with a fuel management objective) and wildfire occurrence, extent, and ideally severity and impact. This should consider the potentially beneficial effect of fuel management and how factors such as the scale, pattern frequency and targeting (in relation to risk factors) affect this, and the effect of burns escaping control resulting in wildfires and the factors that contribute to and cause loss of control. The latter may include consideration of indirect effects such as effects of managed/ prescribed burns on vegetation composition (e.g., dominance of more flammable species/vegetation types) and water table (lowering).	PhD at the University of Exeter due to commence in autumn 2024 investigating the role of managed burning in preventing or reducing wildfire occurrence and spread.

NEER014 research recommendation	Relevant ongoing or published studies that Natural England is aware of
Investigation, potentially involving modelling, of the most effective burn configuration (patch size, shape, pattern, scale, frequency) and targeting of managed/prescribed burning to manage fuel load to reduce wildfire occurrence, severity, extent and impact. This would need to consider habitat/vegetation type and composition, including types other than just *Calluna*-dominated vegetation.	PhD at University of Exeter (see above) investigating burn patterns in relation to wildfire.
Extension of recording/mapping of managed/prescribed burning in England potentially using Earth Observation, particularly in the uplands, in part to contribute towards investigation of the relationship with wildfire occurrence.	The process of mapping burning and wildfire is now well explored and, in addition to the recent papers evaluated in this review update (Table 28), the following have been recently published: Greenpeace. 2022. *Satellites reveal widespread burning on England's protected peatlands, despite government ban:* https://unearthed.greenpeace.org/2022/05/30/satellites-fires-burning-england-peatland-grouse-shooting/. Roteta and others. 2021. A preliminary global automatic burned-area algorithm at medium resolution in Google Earth Engine: https://doi.org/10.3390/rs13214298. Roteta and others. 2019. Development of a Sentinel-2 burned area algorithm: Generation of a small fire database for sub-Saharan Africa: https://doi.org/10.1016/j.rse.2018.12.011. Roy and others. (2019. Landsat-8 and Sentinel-2 burned area mapping - a combined sensor multi-temporal change detection approach: https://doi.org/10.1016/j.rse.2019.111254. RSPB (2023) *Upland burning*. https://upland-burning-rspb.hub.arcgis.com/. Spracklen & Spracklen. 2023. Assessment of peatland burning in Scotland during 1985–2022 using Landsat imagery: https://besjournals.onlinelibrary.wiley.com/doi/10.1002/2688-8319.12296.

NEER014 research recommendation	Relevant ongoing or published studies that Natural England is aware of
	Tanase and others. 2020. Burned area detection and mapping: Intercomparison of Sentinel-1 and Sentinel-2 based algorithms over tropical Africa: https://doi.org/10.3390/rs12020334.
Carry out a broader investigation of the effects of wider management interventions, e.g., grazing, scrub and bracken management, and drainage, on wildfire occurrence, severity, extent and impact.	-
Extension of research into fire behaviour, fuel moisture dynamics, severity, extent and impact, especially in non-*Calluna*-dominated vegetation, and across habitat transitions, potentially including to forestry/woodland and the urban-fringe, in part to input to future development of a full FDRS.	Under investigation through NERC UKFDRS Project led by Manchester University (see above)
In reviewing factors associated with wildfire impact, potential impact should also be considered. This could include assessments of, and inputs to, risk registers, and tools developed for wildfire management planning including risk assessment, scoring and mapping, and fuel mapping.	-
Further research on the design and effectiveness of fire and fuel breaks, and fire suppression in open habitats (and forestry).	-
Research into the influence of sward composition and structure on the occurrence, severity, extent and impact of wildfire.	Andersen and others. 2024: https://doi.org/10.1186/s42408-024-00256-0 as part of NERC FIREBLANKET project: https://nora.nerc.ac.uk/id/eprint/530346/.
Research and monitoring of the effect of peatland and other habitat restoration on wildfire risk/hazard, occurrence, severity, extent and impact, and its effect on habitat resilience.	Andersen and others (2024) (see above).
Investigation into the natural (and historic) fire regime in the UK (probably involving palaeoecological and perhaps restoration/reconstruction ecology studies), its impact upon	University of Exeter evidence review on UK plant species fire adaptation traits *in prep*. for Natural England.

NEER014 research recommendation	Relevant ongoing or published studies that Natural England is aware of
vegetation communities, including an assessment of the extent to which they are fire- adapted, and hence the implications for the use of fire in managing UK vegetation.	

Appendix 5. Burning regulation and guidance

Appendix 2 of NEER004 (pp. 64–86) provides a summary of the Heather and Grass Burning Regulations (England) 2007[22], the Heather and Grass Burning Code (Defra, 2007), other good/best practice guidance on burning, burning under agri-environment scheme agreements and designated site consents, and previous reviews on burning effects. This appendix provides a brief update on regulation and guidance.

Burning Regulations

The Heather and Grass etc. Burning (England) Regulations 2021[23] came into force on 1 May 2021 and supersede parts of The Heather and Grass Burning Regulations (England) 2007 for burning on Sites of Special Scientific Interest (SSSI) and European sites (SAC/SPA) in England. The 2021 Regulations prohibit a person from burning any specified vegetation on areas of peat over 40 cm deep in a SSSI that is also a European site, unless an exception applies, or the burning is carried out under, and in accordance with, a licence issued by the Secretary of State (see below). Exemptions include areas of vegetation which have a slope of more than 35 degrees or where more than half of that area is covered by exposed rock or scree.

“The Secretary of State may grant a licence where it is expedient or necessary:

(a) for the conservation, enhancement or management of the natural environment for the benefit of present and future generations;

(b) for the safety of any person;

(c) to reduce the risk of wildfire; or

(d) because the specified vegetation is inaccessible to mechanical cutting equipment and any other method of management is impracticable.”

Natural England Position Statement 2020

Burning as a tool for the restoration of upland blanket bog: Position Statement from Natural England (2020)[24]. The position statement clarified the position Natural England will take where a request is made to carry out a burn for restoration purposes on blanket bog.

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22. http://www.legislation.gov.uk/uksi/2007/2003/contents/made.

23 www.legislation.gov.uk/uksi/2021/158/contents/made.

24 https://publications.naturalengland.org.uk/.

Natural England set out that the restoration of blanket bog habitats is necessary for the delivery of climate change mitigation, water quality improvement, flood risk mitigation and biodiversity recovery for the benefit of the economy and society.

The Position Statement was responding to evidence that burning on blanket bog is damaging to peatland and that while burning on blanket bog is generally considered to be harmful, in exceptional circumstances it may be appropriate to carry out a one-off burn for the purposes of restoration.

The Position Statement set out that Natural England as the statutory body is responsible for consenting certain activities on protected sites, assessing any likely effects on the notified features, and the position it will take where a request is made to burn blanket bog for restoration purposes.

The Position Statement gave the example that burning on peat over 40 cm in depth will only be consented where there is evidence that, having considered all other alternative management interventions, it is directly connected with or necessary for the management of the habitat for which the site has been designated.

The Position Statement was accompanied by three annexes including a decision-making framework, a monitoring protocol and a question-and-answer document.

Favourable Conservation Status

The Definition of Favourable Conservation Status for blanket bog (Crowle and others, in press) sets out Natural England’s view on favourable conservation status for blanket bog in England and that “Favourable conservation status is the situation when the habitat can be regarded as thriving in England and is expected to continue to thrive sustainably in the future.” The Definition states that “fully functioning blanket bog is a climax habitat that does not require management intervention. Until [it] is fully functioning, management interventions may be needed to fix outstanding drainage issues, inappropriate grazing levels or burning” and that for the function attributes to have been met requires that “management is in place to achieve or maintain target condition in the long term” and “inappropriate management such as rotational burning, drainage or unsustainable levels of grazing have been removed.”

Appendix 6. Upland peatland habitats and vegetation communities

Upland peatland habitats are described in paragraphs 3.3 to 3.6 of NEER004. They comprise the UK BAP priority habitats (and corresponding habitats of principle importance for the conservation of biodiversity in England under the NERC Act[25]) blanket bog, upland flushes, fens and swamps (except where these occur on mineral or at least humic soils) and the wet heath component of upland heathland (some of which also occurs on mineral or at least humic soils). All of these occur on moorland in the uplands. Intermediate bog occurs in places in the uplands as does transitional bog, and raised bog very locally. The following text and table are based on paras. 4.2–4.7 in NEER004.

Trichophorum germanicum 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, higher, drier eastern bogs typically support a higher proportion of *Eriophorum vaginatum* and *Vaccinium myrtilus*. Similarly, *Molinia* and bog-myrtle *Myrica gale* are much more widespread and typical on western bogs (Rodwell, 1991; Averis and others, 2004). *Calluna* also occurs widely, especially in drier situations, including on hummocks where it often regenerates through layering (MacDonald and others, 1995) and may achieve a 'steady state' through the continuous layering and rejuvenation of stems among the *Sphagnum* carpet (Forrest, 1971; Jones and others, 1971; Rawes & Hobbs, 1979; 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; Crowle and others, in press).

Upland flushes, fens and swamps receive water and nutrients from surface and/or groundwater sources as well as rainfall (Rodwell, 1991 *et seq*.; Averis and others, 2004; BRIG, 2008). The soil is generally waterlogged with the water table close to or above the surface for most of the year. Whilst they sometimes occur on peat, they are also found on mineral-based soils, including liquid/silty muds and gleyed podzolic, stagnogley and stagnohumic soils. 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. But species-poor *Molinia* swards and species-poor or 'weedy' soft-rush *Juncus effusus* swards are excluded from the BAP priority habitat. Swamps are generally included except where they fringe standing waters. Though often too wet to be burned, burning does occur in places particularly in more extensive stands such as some valley mires.

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[25] Natural Environment and Rural Communities Act 2006.

Wet heath is widespread in the wetter west and north of the UK. Less-modified forms are dominated by mixtures of *Erica tetralix*, *Trichophorum germanicum*, *Calluna* and *Molinia*, over an understorey of mosses often including carpets of *Sphagnum* bog-mosses (BRIG, 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.

The National Vegetation Classification (NVC) provides a systematic and comprehensive catalogue and description of plant communities in Britain (Rodwell, 1991 *et seq*.). A list of relevant NVC community types occurring in upland peatlands is given in Table A8.1 which draws on Rodwell (1991), Elkington and others (2002), and Averis and others (2004). These include bog pool communities (M1-3) that occur on the surface of the blanket peat and several 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, M17 and M18) represent modified types of blanket bog at least to a degree. More severely modified blanket bog includes dry heath vegetation types, particularly H9 and H12, and some grassland types, particularly U6.

Table A6.1. Upland bog (active and non-active) NVC community types on deep 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 *Trichophorum germanicum* [*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	Mire community replacing M17 and M19 blanket mire communities associated with heavy grazing, burning, and drying of peat.
M17 *Tricophorum germanicum–Eriophorum* blanket mire	Western/oceanic blanket mire community characterised by *Trichophorum*, *Molinia*, *Eriophorum*, *Calluna, Erica tetralix*, and *Sphagnum capillifolium* and *S. papillosum.*
M18 *Erica tetralix–Sphagnum papillosum* raised and blanket mire	Similar to M17, with typical species *Calluna*, *Erica tetralix* and *Eriophorum* on waterlogged peat typically at lower altitudes. *Trichophorum* and *Molinia* generally less common.
M19 *Calluna vulgaris–Eriophorum vaginatum* blanket mire	Extensive blanket mire type, dominated by *Eriophorum* and *Calluna*. Less species-rich compared to M17 and M18. One of the drier bogs which, with drainage and regular burning can lead to change to a dry heath community.

NVC community	Characteristics
M20 *Eriophorum vaginatum* blanket and raised mire	Species poor *Eriophorum vaginatum*-dominated ombrogenous bog community derived from M19 through intensive management. Can also occur in a more ‘natural’ form on drier bog edges.
M21 *Narthecium ossifragum–Sphagnum papillosum* valley mire	Valley mire community particularly in SW and southern England.
M25 *Molinia caerulea–Potentilla erecta* mire	Mire community with overwhelming dominance of *Molinia*, often associated with areas of peat aeration.

Heath and grassland like communities on blanket bog and wet heath.

Wet heath comprises M15 and M16 generally on shallow peat, and where in degraded forms M25 mire (see Table A6.1 for brief descriptions) and U6 *Juncus squarrosus–Festuca ovina* grassland.

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.

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