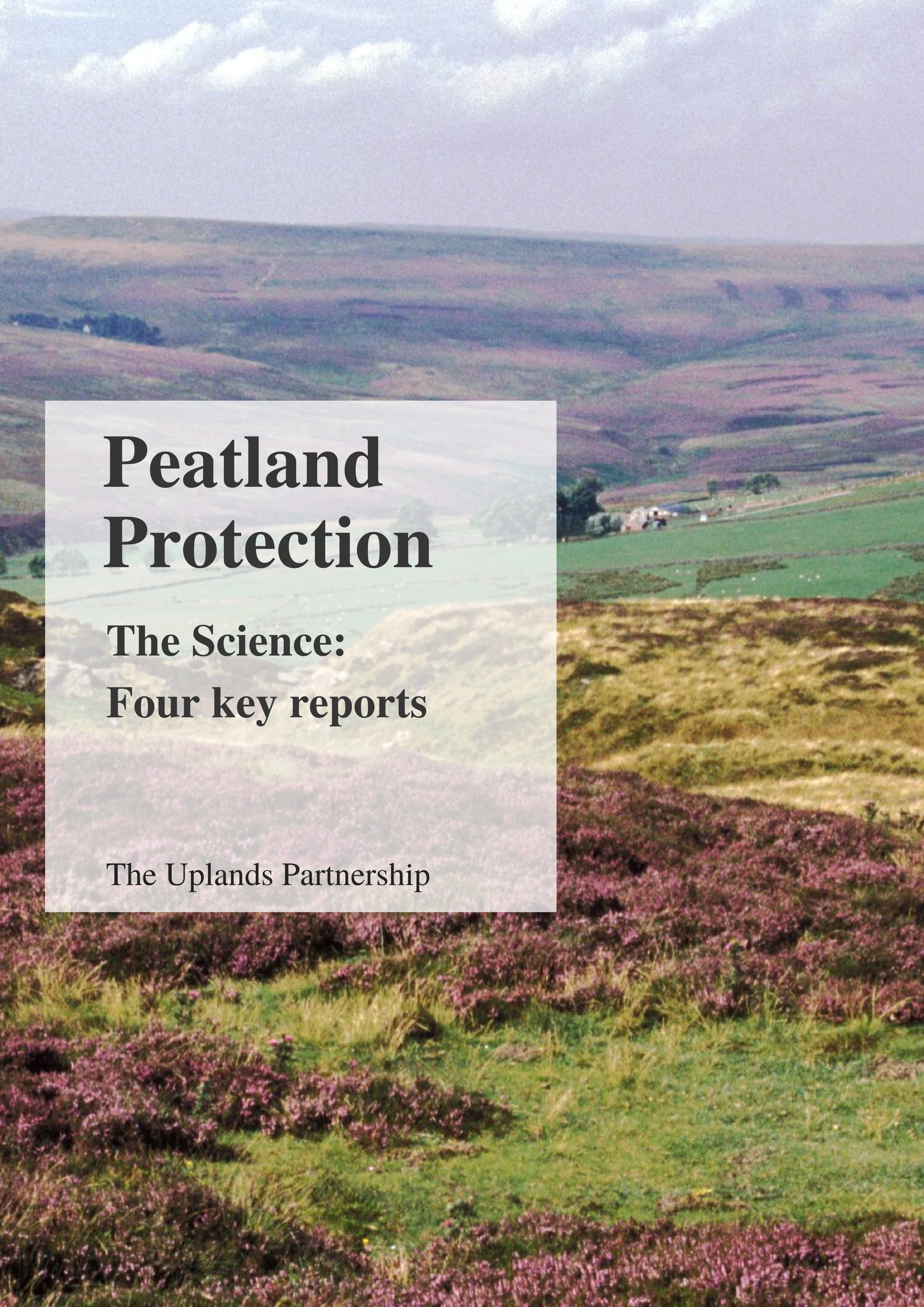

Peatland Protection

The Science: Four key reports

The Uplands Partnership

Foreword

All of us with the countryside at heart are agreed the uplands are among our most treasured landscapes.

We are also united in recognising the importance of ensuring these lands contribute significantly to improving biodiversity and tackling climate change. Opinions are, however, divided on how best we achieve these admirable and shared objectives.

The debate over peatland protection has become particularly polarised and no more so than when the issue of heather burning is being discussed.

This debate should surely rely heavily on the best and most rigorously scrutinised science available.

This document sets out contemporary and authoritative findings by scientists, Dr Mark Ashby, Lancaster University / Whitebeam Ecology, Dr Andreas Heinemeyer, University of York, Dr Gavin Stewart, Newcastle University, and Nick Sotherton, Director of Research, the Game & Wildlife Conservation Trust.

Their conclusions seek to make a meaningful and constructive contribution to assist decision-makers as they plan the way ahead for peatland protection – and the future of our uplands.

BASC The British Association for Shooting & Conservation

The Moorland Association

Countryside Alliance

CLA

The National Gamekeepers' Organisation

The Uplands Partnership

The Game & Wildlife Conservation Trust is an advisory body to The Uplands Partnership

Key findings from four reports

- Conclusions from previous science are now out of date and not safe to be used in policy-making.
- A more coherent policy framework is required which could include integrated adaptive trial designs and monitoring the impacts of different types of management to provide more robust evidence.
- Heather burning can have a positive effect on carbon capture.
- Burning does not cause water discolouration.
- Environmentally-important Sphagnum moss recovers quickly from low severity ‘cool’ burning.
- The loss of controlled burning in the USA led to declines in bird life and wildfires.
- Greenhouse gas emissions from controlled burning are relatively insignificant compared to emissions from wildfire or indeed severely degraded lowland peatlands used for agriculture.

Contents

Report 1:

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

A review of the post-Glaves et al. (2013) evidence

Dr Mark Ashby, Lancaster Environment Centre, Lancaster University, Whitebeam Ecology.

Peer review by Dr Gavin B. Stewart, Newcastle University (Appendix A)

Executive Summary: Key differences between conclusions drawn from evidence up to 2013 review compared to 2013-2020

Summary

Fundamental changes to the evidence base now disagrees with previous science

These changes should inform policy and any proposed regulation change, the England Peatland Strategy and Natural England's position on restoration burning

- Burned areas of blanket bog **ARE** capable of carbon capture.
- Production of charcoal during managed burning has a **POSITIVE** impact on long-term carbon storage.
- Burning **DOES NOT** cause water discolouration
- Controlled burning reduces fuel loads and helps **PREVENT AND LIMIT WILDFIRES**
- Over abundance of heather is **LIMITED** by burning. Environmentally important Sphagnum **MOSS RECOVERS** from 'cool' managed burning within three years.

Introduction

Natural England reviewed the science evidence base on heather burning up to 2013 and found it damaging for water colour, carbon storage and biodiversity. Following a dispute resolution process, NE and MA undertook to review the science from 2013- present following an agreed and consistent method. Peer Review was conducted by an independent scientist of NE's choosing.

Findings

1. Water Quality and storage

Water colouration: Glaves concluded that here was strong evidence that burning increased colour and so too does a heather dominant sward. However, updated evidence concludes that there is neutral effect – burning does not cause an increase or decrease in colouration. This disagrees with Glaves.

Glaves found moderate evidence that burning was associated with an increase in pH and weak evidence that the water table depth becomes shallower post burn. Fresh evidence is inconsistent with both neutral and slightly negative effects found for pH and also inconsistent on water table depth effect with higher and lower WTD found on burnt areas compared to unburnt or not recently burnt controls.

2. Carbon

Glaves concluded that comparing 10-year rotation plots and plots unburnt since 1954, burning reduced peat accumulation and reduced above and below ground carbon storage compared to no burning and carbon loses through burning in conversion to char. The updated evidence base disagrees with these three assertions. New evidence on whether peat is accumulated in burnt areas is now neutral not negative and concludes that burnt areas of blanket bog accumulate rather than lose carbon in the peat profile. The rate of accumulation in flat and wet areas of blanket bog subject to longer burning rotations of circa 20 years appears broadly the same as that recorded in unburnt or not recently burnt areas. Additionally, there is now consistent but very weak evidence that the production of charcoal during managed burning has positive impacts on long-term carbon storage - which therefore requires more study.

Glaves concluded that here was strong evidence that burning increased dissolved organic carbon (DOC)but updated evidence is consistent in that there is neutral effect – burning does not cause an increase or decrease in DOC disagreeing with Glaves.

Glaves concluded that burning resulted in an increase in small scale bare ground but the updated evidence reveals this is a transient effect lasting four to ten years.

3. Biodiversity

Flora – The concern of Glaves was that burning caused heather dominance which may affect the structure and function of Blanket bog. Whether burning was the original cause or not, (it is difficult to unpick other factors such as drainage) the evidence base now concludes that *Calluna vulgaris* becomes more abundant and eventually dominant with increasing time since burn, even in wetter areas and is highest on unmanaged areas whilst abundance is lowest on recently and/or frequently burnt areas. (See consequences of increasing abundance for water colouration and fuel load for wildfire severity).

Both reviews reveal inconsistent evidence on the effects on vegetation diversity, surface topography and Sphagnum moss diversity but the new review concludes that burning has a neutral effect on Sphagnum abundance and after initial damage done by low severity fire, the *Sphagnum capillifolium* almost fully recovers within three years and in high severity fires shows signs of recovery in that time period. Lower 'cool burn' severities cause minimal damage to *S.capillifolium* plants relative to unburnt controls.

Moderate evidence in Glaves concluded that the diversity and composition of aquatic Invertebrates assemblages changed including declines in mayfly and stonefly. The latest evidence is inconsistent in the abundance of pollution intolerant aquatic invertebrates so disagrees with Glaves.

4. Wildfire

Glaves found moderate evidence that fuel load and structure are critical factors in fire behaviour particularly in 'fireline' intensity (heat output per unit length of fire front) and rate of spread, although residence time and depth of penetration of lethal temperatures in to soil are important in determining severity of impact. Yet little evidence on the types of burning practice taking place in the English uplands including the 'extent' to which 'cool burning' is practiced was found. Burning reduces fuel load and may therefore have benefits for fire risk management and recognised the increased need for fire risk management as climate change scenarios become a reality. There was moderate evidence that 'heather moorland' in the Peak District, which was mostly managed by rotational burning, is less prone to the occurrence of wildfires than other moorland habitats.

Even the latest data on burning extent and frequency is ten years out of date and may have now changed with extensive wildfires, some very severe, having occurred in the last three years.

NE position on restoration burning - February 2018

It was always proposed as guidance to how they would consent application for restoration burning and would be updated in the light of new science.

"…..burning on blanket bog is generally considered to be harmful." (our emphasis).

Is the current evidence base supportive of this 'generally considered' position? For carbon storage, water quality and biodiversity the harmful effects concluded by Glaves, do not now appear to be upheld by the up to date evidence base.

"The UK government is responding to infraction proceedings from the EU requiring measures to halt deterioration of blanket bog condition as a result of regular burning." (our emphasis).

Are we still sure that the evidence base consistently and strongly links regular burning with the deterioration of blanket bog given the findings of this review?

"We remain committed to long term restoration plans which focus on a range of outcomes to be achieved from <u>functioning blanket bog</u>." (our emphasis).

If carbon storage, clean water and peat accumulation are key outcomes from functioning blanket bog, the evidence suggests that these outcomes can be delivered on flat and wet blanket bog areas, with a burn cycle of 20 years.

Conclusion

The time is now right to review Natural England's February 2018 position statement which guides its decisions on consenting burning and examine the circumstances in which burning may be a necessary tool to accelerate peatland restoration where restoration is impeded through over dominant heather but also where the structure and function of the site is intact but will deteriorate if burning is removed. Adaptive management through test and trials across multiple sites is the recommended approach to explore the best balance of outcomes under differing conditions.

Due to Climate Change the increased threat and impact from severe wildfires must now also be taken into account in terms of mitigating damage to structure and function of blanket bog.

A REVIEW OF THE POST-GLAVES ET AL. (2013) EVIDENCE:

INVESTIGATING THE EFFECTS OF MANAGED BURNING ON UPLAND PEATLAND BIODIVERSITY, CARBON, GREENHOUSE GAS EMISSIONS AND WATER.

Produced by Dr Mark A. Ashby a* on behalf of the Moorland Association and in consultation with Natural England

Peer reviewed by Dr Gavin B. Stewart b

Burning on peatlands evidence statement by Dr Gavin B. Stewart and Dr Mark A. Ashby

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Burning on peatlands evidence statement by Dr Gavin Stewart, independent peer review

Purpose of the review

To summarise the updated evidence-base regarding peatland burning and ascertain key implications for policy, practice and research.

Summary of updated evidence

The evidence-base underpinning decisions about burning management is highly uncertain despite the plethora of papers published on the topic. The three major causes of uncertainty are i) difficulties synthesising studies measuring different outcomes on different spatial and temporal scales ii) high inconsistency in effects across multiple studies iii) high risk of bias resulting from deficiencies attributing causation and/or high potential for confounding. Interpretation of recent evidence, reviewed by Ashby differs from the interpretation of older evidence reviewed by Glaves, notably with respect to *Sphagnum* abundance and carbon accumulation. The impact of burning on the former is heavily context-dependent and varies in relation to post-burning succession. The latter is subject to a fierce academic debate that remains polarised and unresolved. Uncertainties in the evidence base are exacerbated by changing climatic baselines which may interact with floristic responses, carbon budgets, wildfire frequency and other important components of peatland systems. Further uncertainty is added by habitat heterogeneity, particularly for large scale studies which may incorporate both deep and shallow peats.

Implications for policy and practice

Deficiencies in the evidence-base necessitate decision-making under high uncertainty. One approach is to utilise the precautionary principle to minimise potential deleterious effects. This may be particularly appropriate for high-value sites on deeper peats, particularly where hydrological functioning is intact or easily restored. An alternative approach is to utilise an adaptive management framework whereby different management is undertaken at different sites subject to monitoring outcomes. Such an approach mitigates uncertainty by hedging and avoiding a one size fits all solution. It can also facilitate evidence acquisition, especially if this is built into the policy. This may be a viable option on peatlands with a lower conservation value where ecosystem service provision and sporting interest may be easier to align.

Implications for Research

The current deficiencies in the evidence-base are unlikely to be resolved by the accumulation of more studies alone. A more coherent framework is required based around a consensus regarding the core common outcomes required for studies of peatland management. This requires augmenting with work ascertaining how research-intensive measurements relate to easily measurable surrogates that could be collated at scale by automated sensors, remote sensing, citizen scientists and land managers. Large scale long term studies are required, which may be more cost-effective if surrogate outcomes have been identified, and comparator treatments are implemented by existing land managers. Incorporating elements of randomisation or adaptive trial design would help resolve uncertainties regarding causation.

Further information

Dr Gavin Stewart [Gavin.stewart@newcastle.ac.uk]

Bias Statement

This evidence briefing was prepared jointly by Gavin Stewart and Mark Ashby following peer review (GS) and authorship (MA) of an updated review of evidence funded by the Moorland Association. Peer review was undertaken at the bequest of Natural England and the Moorland Association. Open science and evidence synthesis have important roles in reconciling stakeholders with polarised beliefs and values about peatland management and allowing the development of a robust evidence-base in this domain.

Summary and key findings

Background

In 2013, Glaves et al. (2013) published a systematic review on "*The effects of managed burning on upland peatland biodiversity, carbon and water (NEER004)*". Since then, a substantial amount of evidence has emerged. However, rather than clarifying our understanding, the emerging evidence seems to have intensified the scientific debate about the use of managed burning on peatland ecosystems in the UK. In an attempt to provide clarity for land managers and policymakers, the Moorland Association commissioned a review of the evidence that has emerged since Glaves et al. (2013).

Review question

The overarching review question is: What are the effects of managed burning on the maintenance and restoration of upland peatland biodiversity, carbon, soil and water?

Objectives

This review has four objectives:

1. To produce a coded Excel database of post-Glaves et al. (2013) studies that can be used and expanded upon by researchers and policymakers moving forward as the basis of an up-to-date 'living review'.

2. To critically appraise and summarise the post-Glaves et al. (2013) evidence.

3. To highlight contradictions and similarities between the findings summarised in this review and those reported by Glaves et al. (2013).

4. To determine research gaps and priorities.

Search strategy

Evidence searches we conducted in four stages. First, we used a standardised search term to search the title, abstract and keywords of articles contained within the Web of Science and Scopus online databases. Second, we examined the reference lists of six recent and relevant literature reviews. Third, we searched the title, abstract and subject keywords of PhD and

MSc theses contained within the Ethos British Library online database. Finally, we added any articles known to the authors that were not picked up during stages one to three.

Inclusion criteria

We used the following inclusion criteria to accept or reject studies for review:

1. The study must have been published since 2012 (inclusive).

2. The study must not have been included within Glaves et al. (2013).

3. The study must be an original empirical investigation. Modelling studies, systematic/literature reviews, meta-analyses, commentaries, and descriptive books, book chapters and reports were not included within this review. However, if relevant, they were categorised (by reference type) and put within a table in the appendices.

4. The study must focus on temperate and boreal peatland in the northern hemisphere (especially blanket bog but including other bogs/mire/fen/wet heath), biodiversity (flora and fauna), carbon sequestration, GHG emissions, water (quality and flow), soil (erosion, moisture, temperature and chemistry), and (managed) burning. In general, references that did not specifically relate to burning were excluded. However, to address the potential indirect effects of burning on vegetation composition and structure in relation to sub-questions (b) (fauna), (c) (carbon sequestration and GHG emissions) and (d) (water), references relating to the effects of changes in vegetation composition and structure were accepted.

5. Studies must not solely focus on dry heath, mineral soils, forest/woodland/trees, tropical/arctic/tundra and wildfire (unless related to the effect of managed burning).

Data collection and analysis

Articles accepted for inclusion within this review were separated into individual studies and summarised using a range of coding variables and critical appraisal questions. Critical appraisal data was used to assign each study a quality rating based on their ability to ascribe causation. The quality ratings used were very high quality" (+++), "high quality" (++), "medium quality" (+) and "low quality" (-). Evidence for a range of outcome measures was summarised using a narrative synthesis approach. We also noted whether the evidence for a given outcome agreed with or contradicted the corresponding evidence outlined in Glaves et al. (2013).

Main findings

Sixty-two studies derived from 65 different articles were included in this review. These studies provided evidence for 55 different outcome measures. Most studies adopted a correlative, short-term and plot-scale approach to assessing burning impacts. Consequently, the overall quality of evidence for each outcome measure is low. Furthermore, the majority of outcome measures (64%) are supported by inconsistent evidence.

The strongest and most consistent evidence is for *Sphagnum* (principally *S. capillifolium*) abundance. Specifically, the evidence included in this review suggests that managed burning has a neutral impact on *Sphagnum* abundance within upland peatlands. Finally, the evidence for 23 of the outcome measures assessed is inconsistent with the findings presented in Glaves et al. (2013). Notable contradictions include carbon accumulation, dissolved organic carbon fluxes, water colour and *Sphagnum* abundance.

Conclusions

The contradictory nature and low quality of the evidence mean that it is difficult to draw firm conclusions about the impacts of burning on upland peatland ecosystems. As such, it would also be unwise to make any policy recommendations. However, we do have a series of general research recommendations that are informed by our findings:

1. Future studies must investigate burning impacts on upland peatlands using a robust and real-world approach. A robust approach would be the adoption of an experimental design that can accurately ascribe causality, such as a randomised controlled before-and-after trial. A real-world approach is an approach which examines burning in the same way upland land managers apply it, e.g., every year, multiple patches of varying size (but usually ~2500 m^2) are burnt on rotation across an extensive area of moorland using rotations that are suited to the local environmental (i.e. growing) conditions.

2. Both the pre- and post-Glaves et al., 2013 evidence must be collated and categorised.

3. We need to develop an objective approach for summarising the highly heterogeneous burning evidence base.

4. We also need to develop a series of standardised protocols for measuring peatland ecosystem services. This would enable researchers to assess the impact of different land management options using objective approaches, such as meta-analysis.

Contents

1. Introduction

In 2013, Glaves et al. (2013) published a systematic review on "*The effects of managed burning on upland peatland biodiversity, carbon and water (NEER004)*". Since then, a substantial amount of new evidence has emerged (e.g. Harper et al., 2018). Yet, rather than clarifying our understanding, the emerging evidence seems to have intensified the scientific debate surrounding the impacts of managed burning on peatland ecosystems in the UK (e.g. Brown et al., 2016; Davies et al., 2016b; Davies et al., 2016c; Douglas et al., 2016a; Ashby and Heinemeyer, 2019a; Ashby and Heinemeyer, 2019b; Baird et al., 2019; Brown and Holden, 2019; Evans et al., 2019; Heinemeyer et al., 2019b; Marrs et al., 2019b). Indeed, some peatland researchers argue that the evidence suggests the overall effect of burning on peatlands is unclear due to insufficient, contradictory or unreliable evidence (Davies et al., 2016b; Ashby and Heinemeyer, 2019a; Ashby and Heinemeyer, 2019b; Marrs et al., 2019b). Other peatland researchers challenge this assessment and assert the evidence shows that burning is significantly damaging to UK peatlands and the ecosystem services they provide (Brown et al., 2016; Douglas et al., 2016a; Baird et al., 2019; Brown and Holden, 2019).

Debate further intensified in 2016 when, in response to complaints submitted by the Royal Society for the Protection of Birds (RSPB, 2016), the European Commission threatened legal action (i.e. infraction proceedings) against the UK government if it failed to put a stop to rotational burning on blanket bog habitats within English Special Areas of Conservation (European Commission, 2017). The UK government responded by adopting a voluntary approach to stopping rotational burning on blanket bog habitats within SACs, with burning only being allowed as a one-off for restoration purposes under very strict criteria (Natural England, 2019a; Natural England, 2019d; Natural England, 2019c; Natural England, 2019b).

In an attempt to provide clarity, the author (Mark Ashby) was approached and contracted by the Moorland Association to collate and synthesise the evidence that has emerged since Glaves et al. (2013). However, even though there is some value in reviewing the post-Glaves et al. (2013) evidence, it would be much more valuable to researchers, land managers and policymakers if the entire evidence base were reviewed. Such a review would enable one to ascertain whether the cumulative evidence base changes any of the conclusions outlined in Glaves et al. (2013). Nevertheless, at this stage, Natural England[1] suggested it

[1] Natural England were consulted during every stage of the review process.

would be more appropriate to collate and synthesise the most recent and unreviewed evidence.

1.1. The review topic

1.1.1. What is considered in this topic review?

This review considers the effects of burning on upland peatland habitats, and the effects on carbon, soil, and water (quality and flow) related ecosystem services.

1.1.2. The overarching review question

The overarching review question is: What are the effects of managed burning on the maintenance and restoration of upland peatland biodiversity, carbon, soil and water?

The following sub-questions were the focus of the topic review (all but sub-question h are taken from Glaves et al., 2013):

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

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

c) **Carbon sequestration and greenhouse gas emissions -** What are the effects of managed burning of upland peatlands on carbon sequestration and greenhouse gas (GHG) emissions, either directly or indirectly through changes in vegetation composition and structure?

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

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

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

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

h) **Soils -** What are the effects of managed burning of upland peatlands on peat soils (erosion, temperature and chemistry), either directly or indirectly through changes in vegetation composition and structure?

1.1.3. Review objectives

This review has four objectives:

1. To produce a coded Excel database of post-Glaves et al. (2013) studies that can be used (and expanded upon) by researchers and policymakers moving forward. It is hoped that the evidence used by Glaves et al. (2013) will be added to this database and that both evidence bases form the basis of an up-to-date 'living review' (sensu Elliott et al., 2017).

2. To critically appraise and summarise the post-Glaves et al. (2013) evidence relating to the overarching review question and sub-questions.

3. To highlight contradictions and similarities between the findings summarised in this review and those reported by Glaves et al. (2013).

4. To determine research gaps and priorities.

1.1.4. Study inclusion criteria

To be included in this review, studies had to pass the following inclusion/exclusion criteria:

6. The study must have been published since 2012 (inclusive).

7. The study must not have been included within Glaves et al. (2013).

8. The study must be an original empirical investigation. Modelling studies, systematic/literature reviews, meta-analyses, commentaries, and descriptive books, book chapters and reports were not included within this review. However, if relevant, they were categorised (by reference type) and put within a table in the appendices.

9. The study must focus on temperate and boreal peatland in the northern hemisphere (especially blanket bog but including other bogs/mire/fen/wet heath), biodiversity (flora and fauna), carbon sequestration, GHG emissions, water (quality and flow), soil (erosion, temperature and chemistry), and (managed) burning. In general, references that did not specifically relate to burning were excluded. However, to address the potential indirect effects of burning on vegetation composition and structure in relation to sub-questions (b) (fauna), (c) (carbon sequestration and GHG emissions) and (d) (water), references relating to the effects of changes in vegetation composition and structure were accepted.

10. Studies must not focus on dry heath, mineral soils, forest/woodland/trees, tropical/arctic/tundra and wildfire (unless related to the effect of managed burning).

2. Methods

This review attempted to use a similar methodology to Glaves et al. (2013) but, due to several reasons (e.g. logistics), this could not always be achieved. Significant departures from the Glaves et al. (2013) methodology are highlighted throughout the subsequent sections.

2.1. General principles

During the review process, all the available studies providing evidence for the review sub-questions were systematically identified (a-h, listed in section 1.1.2 above). This involved sifting through a list of articles returned during systematic literature searches to ensure that the only articles included were those that met the pre-defined inclusion criteria.

The following PICO framework was used to focus searches on:

- **Population:** upland peatland habitats in England.
- **Intervention:** managed burning.
- **Comparison:** no burning, at least in recent decades.
- **Outcome:** impact of burning on the maintenance and restoration of upland peatland biodiversity, carbon, soil and water.

2.2. Evidence searches

2.2.1. Search term development and optimisation

Glaves et al. (2013) conducted evidence searches using different combinations of relevant search words and wildcard operators. In contrast, we used a fixed search term that contained a string of relevant search words and wildcard operators. Our search term was developed by testing different combinations of specific words and wildcard operators relating to (i) managed burning; (ii) peatland restoration; (ii) peatland habitats; and, (iv) soil, water, GHG sequestration and biodiversity-related ecosystem services. The search term was refined by cross-referencing the search results of different word and wildcard operator combinations to the reference lists of the six recent literature reviews on managed burning impacts within the British uplands (Brown et al., 2015a; Heinemeyer and Vallack, 2015; Davies et al., 2016b;

Thompson et al., 2016; Sotherton et al., 2017; Harper et al., 2018). This resulted in the final search terms listed in Table 1, which are the same apart from minor formatting differences to account for the syntax requirements of the different online databases used. Database searches were conducted using the advanced search function to restrict results to the English language and the period between 2012 and 2019 (November). A start date of 2012 was selected because the final search phase of Glaves et al. (2013) took place during 2012 (D. Stone pers. comm., June 6, 2019). Field codes were used to limit searches to the title, abstract and keywords of the articles within each database (Table 1).

Table 1. The search term we used during the Web of Science and Scopus database searches. Note how the search words and Boolean operators are identical, but the formatting is different (*e.g.* the use of parentheses, quotation marks and asterisk differs). TS and TITLE-ABS-KEY are field codes used in the separate databases that restrict the search to the title, abstract and keywords of an article.

Web of Science: TS=((burn* OR "fire") AND (peat* OR heath* OR moor* OR "blanket" OR "bog" OR "mire") AND ("habitat management" OR "biodiversity" OR "grouse" OR restor* OR bird* OR plant* OR "vegetation" OR sphagnum* OR invertebrate* OR insect* OR amphibian* OR reptile* OR mammal* OR "water quality" OR "water colour" OR "flow" OR "saturated" OR "dissolved organic carbon" OR "DOC" OR hydrolog* OR infiltrat* OR "soil" OR carbon budget* OR "carbon cycling" OR carbon flux* OR "carbon sequestration" OR carbon stock* OR "carbon storage" OR "wildfire" OR ecosystem* OR environment*))
Scopus: TITLE-ABS-KEY((burn* OR {fire}) AND (peat* OR heath* OR moor* OR {blanket} OR {bog} OR {mire}) AND ({habitat management} OR {biodiversity} OR {grouse} OR restor* OR bird* OR plant* OR {vegetation} OR sphagnum* OR invertebrate* OR insect* OR amphibian* OR reptile* OR mammal* OR {water quality} OR {water colour} OR {flow} OR {saturated} OR {dissolved organic carbon} OR {DOC} OR hydrolog* OR infiltrat* OR {soil} OR carbon budget* OR {carbon cycling} OR carbon flux* OR {carbon sequestration} OR carbon stock* OR {carbon storage} OR {wildfire} OR ecosystem* OR environment*))

2.2.2. Search strategy

The first stage of our search strategy involved using the following online databases to extract relevant peer-reviewed journal articles:

1. Web of Science
2. Scopus

These databases were searched in the order shown using the appropriate search term. Search results were then downloaded from each database into an EndNote file. The second stage of the search strategy involved examining the reference lists of the six literature reviews used during search term development to extract additional articles not picked up during stage one:

1. Harper et al. (2018), "Prescribed fire and its impacts on ecosystem services in the UK".

2. Sotherton et al. (2017), "An alternative view of moorland management for Red Grouse *Lagopus lagopus scotica*".

3. Davies et al. (2016b), "The role of fire in UK peatland and moorland management: the need for informed, unbiased debate".

4. Thompson et al. (2016), "Environmental impacts of high-output driven shooting of Red Grouse *Lagopus lagopus scotica*".

5. Brown et al. (2015a), "Effects of fire on the hydrology, biogeochemistry, and ecology of peatland river systems".

6. Heinemeyer and Vallack (2015), "Potential techniques to address heather dominance and help support 'active' *Sphagnum* supporting peatland vegetation on blanket peatlands and identify practical management options for experimental testing".

These reviews were examined in reverse chronological order for any additional references not been picked up during the literature database search. Any additional references were added to the Endnote database.

The third stage of the search strategy involved extracting relevant PhD and MSc theses using the EThOS e-theses database provided by the British Library website (https://ethos.bl.uk/Home.do). The advanced search function was used to carry out three separate searches that focussed on the title, abstract and subject keywords of the theses within the EThOS database (Figure 1). During each search, the following search string was used: "prescribed fire OR prescribed burning OR rotational burning OR heather burning OR muirburn" (Figure 1). Again, any additional studies retrieved during the EThOS searches were added to the EndNote database.

Finally, a small number of relevant studies known to the author were retrospectively added to the EndNote database because the search strategy failed to capture them, or they were released after the literature searches had concluded.

2.2.3. Removal of duplicates

After all the literature searches were completed, duplicates were removed from the EndNote database using the eight-step de-duplication methodology outlined in Appendix C. This method was taken and modified from Bramer et al. (2016). Due to the importance of page numbers during the de-duplication process, the EndNote display settings were changed so that reference 'Pages' were visible within the library window (Bramer et al., 2016). Then, steps 1-8 were followed until all duplicates were removed (Appendix C). Each step involved searching for duplicates using different combinations of EndNote fields (e.g. title, author, pages), with the final step being a manual scan and removal of duplicate references (Appendix C).

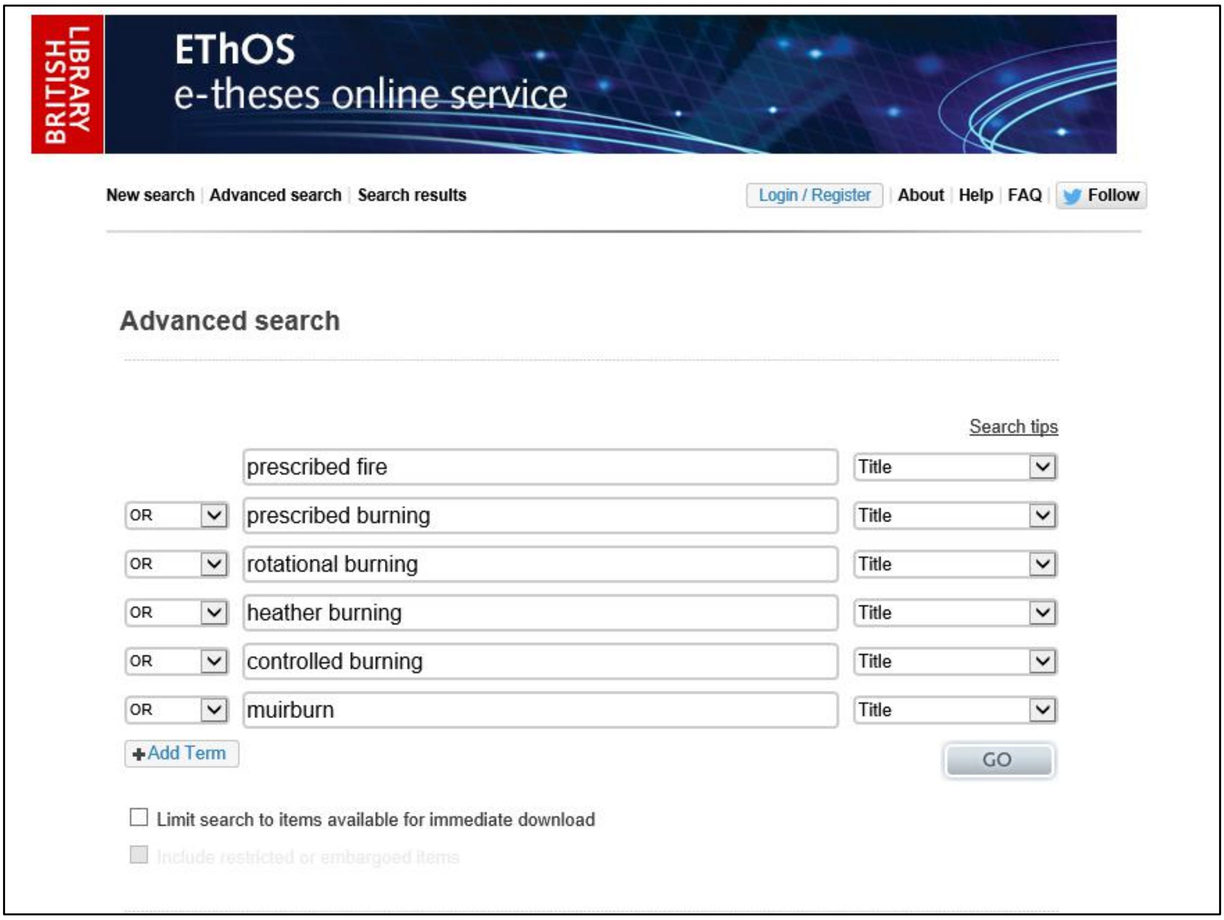


Figure 1. The search string entered into the EThOS e-theses database during the title, abstract and subject keyword advanced searches.

2.2.4. Study screening

Similar to Glaves et al. (2013), studies were screened for inclusion by a single reviewer (M. Ashby). However, unlike Glaves et al. (2013), quality assurance by a second reviewer was not carried out. Articles retrieved during evidence searches were screened for inclusion at two successive levels. First, all unduplicated references were exported from the EndNote database into a Microsoft Excel spreadsheet. Then, the date, title and abstract[2] of each article was examined to see whether it passed or failed each of the four review inclusion criteria (outlined in section 1.1.4. above). In cases of uncertainty (*e.g.* the title and/or abstract were implicitly but not explicitly relevant), the article was included. Second, articles accepted at stage one were read in full to ensure they met all the inclusion/exclusion criteria (see section 1.1.4.). Any articles accepted at this stage were grouped into studies and then entered into a Microsoft Excel spreadsheet using the following coding variables:

Column A: Study ID (unique numeric code given to each study)

Column B: First Author (surname and initial of the first author of the source article).

Column C: Title (Full title of the source article).

Column D: Year (the year in which the source article was published).

Column E: Harvard Reference (full Harvard reference of the source article).

Column F: Reference type (journal, report, book chapter, PhD theses, MSc theses).

Column G: Source (which search method was the source article obtained from).

Column H: Primary sub-question (the primary review sub-question that the study relates to).

Column I: Secondary sub-question(s) (the secondary sub-questions that the study relates to).

Column J: Country (the country or countries in which the study took place)

Column K: Region (the region or regions in which the study took place)

Column L: Study type (e.g. randomised control trial, non-randomised controlled trial, case-controlled trial, cohort study)

Column M: Study length (the amount of time [rounded up to full years] during which data collection started and finished for each study plot)

Column N: Linked study (if applicable, the name of the wider study or experiment which the article relates to, e.g. the Hard Hill experimental plots)

Column O: Surrogate predictor? (Were burning impacts measured directly? Yes/No)

[2] If an article did not have an abstract, then the reviewer read the summary, executive summary or introduction.

Column P: Habitat(s) (the habitat in which the study took place)

Column Q: Predictor variable(s) (the predictor variables used during the study)

Column R: Predictor variable notes (a brief description of the predictor variable)

Column S: Outcome variable(s) (the relevant outcome variables measured by the study)

Column T: Outcome variable measurement units (the units which were used to measure the relevant outcome variables)

Column U: Outcome variable notes (a brief description of the outcome variables)

Column V: Study findings (a brief description of the effect of the predictor variable on each of the outcome measurements investigated)

Column W: Study quality (the quality of the study – determined using the method outlined in section 2.3. below)

Column X: Study quality notes (explanatory notes about the quality designation given to each study)

The spreadsheet containing the data described above will be shared with scientists, land managers and policymakers working on fire impacts within the British uplands.

2.3. Critical appraisal of studies

Each study was critically appraised using 16 yes/no questions that, while considering several aspects of bias (e.g. internal and external validity), were primarily used to rank studies based on their ability to ascribe causality (Table 2). Using this approach, studies were ranked as "very high quality" (+++), "high quality" (++), "medium quality" (+) and "low quality" (-) based on the number of 'yes' responses returned for each of the critical appraisal questions (Table 3). Studies with a low risk of bias are those studies which combine a real-world approach with an experimental design robust enough to attribute causality (Table 3). A "real-world approach" is one which examines burning in the same way it is applied by upland land managers, e.g., every year, multiple patches of varying size (but usually ~2500 m^2) are burnt on rotation across an extensive area of moorland using rotations that are suited to the local environmental (i.e. growing) conditions.

This method of critical appraisal departs from that used in Glaves et al. (2013) in two ways. Firstly, our critical appraisal questions are different (See Appendix 12 in Stone, 2013). Secondly, instead of using a binary yes/no response, the critical appraisal questions used by Glaves et al. (2013) were answered using a graded response that related to whether the reviewer thought the study had low (++), moderate (+) or high (-) levels of bias (Stone, 2013). Each study was then classed as "high quality" (++), "medium quality" (+) and "low quality" (-) based on the general trend of responses across all the critical appraisal questions (i.e. overall, were there more ++, + or – responses) (*ibid*). This critical appraisal system was challenging to replicate when applying it to a small sample of studies included within Glaves et al. (2013). Therefore, a more explicit and repeatable critical appraisal methodology was developed. However, it is worth noting that both approaches are subjective and are, therefore, not definitive assessments of study quality (or bias).

Table 2. The 16 yes/no questions used to critically appraise each study included in this review.

1. Was there a spatial replicate? e.g. Were treatment measurements taken from multiple plots?
2. Was there a temporal replicate? e.g. Were treatment measurements taken across multiple time points?
3. Were significant confounding variables adequately controlled for during data analysis?
4. Was pseudoreplication avoided during data analysis? e.g. Multiple measurements were taken from individual monitoring units (e.g. plots) at a single point in time and/or across several points in time. Individual measurements were then used as replicates (instead of summing or averaging measurements taken from each survey plot) during data analysis without accounting for their lack of independence (e.g. by using appropriate nesting or random effects) (Davies and Gray, 2015).
5. Do the populations studied relate to the target habitats and setting(s) considered by this review (e.g. upland peatlands in the UK and particularly England)? This assessment considered whether the study was conducted in the UK and how representative it was of the English upland peatland resource. This required a comparison with the 'favourable condition' vegetation composition characteristics of upland peatlands in England (JNCC, 2009; JNCC, 2011).
6. Were treatments or study plots randomly allocated?
7. Was there a control? e.g. Was there an unburnt or not recently burnt control plot?
8. Was the study conducted in the field?
9. Was the study experimental?
10. Was the study conducted across multiple peatland sites? e.g. Data collection sites are considered separate if they are >5 km apart
11. Did the study measure burning impacts across more than one burning rotation? e.g. If managed burning was carried out on the burning treatment plots every ten years, then measurements were taken after at least two burns had been applied (once in the first ten years and once in the second ten years).
12. Did the study measure burning impacts across at least three different years within each burning rotation studied? e.g. After managed burning had been carried out, measurements were taken during at least three years before the plot was burnt again.
13. Were baseline measurements taken before burning treatments were applied?
14. Was the effect of burning studied at the catchment or moorland scale?
15. Did the treatments include different burn rotation lengths? e.g. 10-year and 20-year burn rotation treatments.
16. Did the treatments include different fire severities? e.g. low and high fire severity treatments (i.e. low and high fire temperature treatments which usually reflect low and high vegetation/soil moisture contents).

Table 3. The bias ratings ascribed to each study included in this review. Bias ratings were calculated using the critical appraisal questions in Table 2.

Quality rating	Criteria and definition
-	**Low-quality study.** A study that fails to pass questions 1-4.
+	**Medium quality study.** A study that passes questions 1-4 but fails questions 5-9.
++	**High-quality study.** A study that passes questions 1-9 but fails to pass questions 10-16.
+++	**Very high-quality study.** A study that passes questions 1-16.

2.4. Evidence synthesis

We followed the methodology set out in Glaves et al. (2013) and conducted a narrative synthesis of the evidence. We then produced evidence statements that described the quantity, quality, direction and consistency of the evidence for each outcome measure investigated by this review. Evidence consistency and direction were not assessed for outcome measures supported by a single study. Furthermore, evidence was only classed as consistent if ≥75% of the studies for a given outcome measure reported similar results (i.e. the direction of the effect was consistent across studies). Next, we made a series of general and outcome-specific research recommendations. Finally, in addition to providing evidence summary statements and research recommendations for each outcome measure, we also produced an evidence summary table. This table provides a condensed summary of the consistency, direction and strength of evidence for each outcome measure investigated by this review. It also notes whether any of our findings contradicted the findings described in Glaves et al. (2013). Evidence strength was assessed using the following criteria:

- **Strong evidence:** At least three very high-quality studies (+++) or eight high-quality studies (++) reporting consistent results.

- **Moderate evidence:** At least two very high-quality studies (+++) or five high-quality studies (++) reporting consistent results.

- **Weak evidence:** At least three high-quality studies (++) or eight medium-quality studies (+) reporting consistent results.

- **Very weak evidence:** Less than three high-quality studies (++) or less than eight medium-quality studies (+) reporting consistent results.

3. Characteristics of the post-Glaves et al. (2013) evidence base

3.1. Search results

Overall, 65 articles were included in this review, with 54 (83%) of these articles being obtained during the Web of Science search (Table 4). Of the 65 articles included in this review, 59 were from peer-reviewed journals, four were reports, and two were PhD theses. The 65 included articles were condensed into 62 individual studies for further analysis (Table 4).

Table 4. The number of articles retrieved during each search stage. Searches were carried out on the 25/11/2019.

Search method or review stage	Number of articles (number of articles accepted in this review)
1. Web of Science	1341 (54)
2. Scopus	316 (0*)
3. Harper et al. (2018)	25 (1)
4. Sotherton et al. (2017)	12 (3)
5. Davies et al. (2016)	25 (2)
6. Thompson et al. (2016)	12 (0)
7. Brown et al. (2015)	16 (2)
8. Heinemeyer & Vallack (2015)	2 (0)
9. EThOS British Library	5 (1)
10. Added Retrospectively	12 (2)
Total articles retrieved including duplicates	1765
Total articles retrieved minus duplicates	1505
Articles remaining after title, date, and abstract assessment	127
Articles remaining after the full-text assessment	65
Number of studies included within this review[1]	62

[1]The 65 articles included within this review were condensed into 62 studies.
* Most of the duplicates removed were studies retrieved during the Scopus search that had been picked up by We of Science.

3.2. Description of studies included within the review

Most of the studies included in this review were conducted in England (n = 43), followed by Scotland (n = 11), England and Scotland (n = 2), England, Scotland and Wales (n = 2), Norway (n = 2), Wales (n = 1), and Northern Ireland (n = 1) (Table 5). The majority of studies were correlational (n = 17) or case-control studies (n = 13) (Table 6). Sixty-five percent of studies (n = 40) were short-term (i.e. <10 years long), with the majority of short-term studies only collecting data over a single year (n = 20). However, ten paleoecological studies were included, and these studies examined data spanning for >1000 years.

Table 5. The number of accepted studies by country of origin.

Country	Number of studies
England	43
England & Scotland	2
England, Scotland & Wales	2
Scotland	11
Wales	1
Northern Ireland	1
Norway	2

Table 6. The number of accepted studies by type of study. In general, experimental studies (i.e. controlled trials) have the lowest risk of bias (Hurlbert, 1984; Smokorowski and Randall, 2017). The randomisation of treatments and collection of baseline data (i.e. a before-and-after study) further reduces bias (*ibid*).

Type of study	Number of studies
Randomised controlled before-and-after trial	6
Randomised controlled trial	9
Non-randomised controlled before-and-after trial	2
Non-randomised controlled trial	4
Before-and-after study*	1
Case-controlled study	13
Correlational study	17
Cohort study	3
Case report	8

Note: the total is 62 rather than 61 because one study used two approaches
*Differs from a randomised or non-randomised controlled before-and-after trial in that there is no control or treatment randomisation.

Forty of the 62 accepted studies measured burning impacts directly. Whereas, 22 of the accepted studies measured burning impacts indirectly by using vegetation structure (e.g. vegetation height), vegetation composition (e.g. cover of different peatland species), simulated ash deposition, simulated increased bulk density or charcoal macrofossils as proxies. Such variables can be used as proxies for burning management because:

- Upland peatland vegetation composition and structure are both influenced by managed burning (see Glaves et al., 2013 and references therein). For example, burning seems to initially promote the dominance of *Eriophorum*, followed by the long-term dominance of *Calluna vulgaris* (*ibid*). There is also a positive relationship between time since burn and vegetation canopy height within upland peatlands (Whitehead and Baines, 2018).

- Burning leads to the production of ash and charcoal (Allen, 1964; Worrall et al., 2013a; Leifeld et al., 2018) which can be added to the peat profile or removed via overland flow (Johnston and Robson, 2015; Heinemeyer et al., 2018).

- Managed burning can lead to an increase in peat bulk density (Noble et al., 2017; Heinemeyer et al., 2018).

3.3. Quality of studies included within the review

Only 18% of studies had a moderate risk of bias (n = 11), with the remaining 82% of studies having either a high (n = 26) or very high (n = 25) risk of bias (Table 7). More importantly, none of the studies included in this review were classified as having a low risk of bias. Consequently, no study can be said to have accurately or adequately assessed the impacts of burning on upland peatlands (i.e. by using a robust real-world approach).

Table 7. The number of accepted studies by the level of bias.

Study quality	Number of studies
Very high risk of bias	26
High risk of bias	25
Moderate risk of bias	11
Low risk of bias	0

Table 8 lists the number of "Yes" or "No" response to each of the critical appraisal questions used to assess study bias. Overall, most studies had a spatial or temporal replicate (n = 56 and n = 53, respectively), and avoided significant confounding effects (n = 47) or pseudoreplication (n = 53). Also, all but two studies related were not directly relatable to the English upland peatland resource. These two studies were conducted in the coastal wet heaths of Norway and were the only studies conducted on areas of shallow peat (<50cm) (Velle et al., 2014; Velle and Vandvik, 2014).

Table 8. The number of "Yes" or "No" responses to each of the critical appraisal questions used to assess study quality.

Critical appraisal question	Yes	No
1. Was there a spatial replicate? e.g. Were treatment measurements taken from multiple plots?	56	6
2. Was there a temporal replicate? e.g. Were treatment measurements taken across multiple time points?	53	9
3. Were significant confounding variables adequately controlled for during data analysis?	47	15
4. Was pseudoreplication avoided during data analysis?	53	9
5. Do the populations studied relate to the target habitats and setting(s) considered by this review?	60	2
6. Were treatments or study plots randomly allocated?	20	42
7. Was there a control? e.g. Was there an unburnt or not recently burnt control plot.	31	31
8. Was the study conducted in the field?	60	2
9. Was the study experimental?	24	38
10. Was the study conducted across multiple peatland sites?	28	34
11. Did the study measure burning impacts across more than one burning rotation?	2	60
12. Did the study measure burning impacts across at least three different years within each burning rotation studied?	6	56
13. Were baseline measurements taken before burning treatments were applied?	11	51
14. Was the effect of burning studied at the catchment or moorland scale?	20	42
15. Did the treatments include different burn rotation lengths?	14	48
16. Did the treatments include different fire severities?	6	56

Approximately half of the studies were conducted within a single site (n = 34) and did not have an experimental control (n = 32). Conversely, only a minority of studies included in this review:

- Were experimental (n = 24).
- Investigated the impact of different fire severities (n = 6) or burn rotation lengths (n = 14).
- Randomly assigned treatment or study plots (n = 20).

- Took measurements for more than three years within a rotation (*n* = 6) or across several burning rotations (*n* = 2).

- Took baseline measurements before treatments were applied (*n* = 11).

- Measured burning impacts at the catchment of moorland scale (*n* = 20)

4. Narrative review

4.1. Flora

Thirty-four studies investigated the effects of managed burning on upland peatland vegetation composition, structure and function (Fyfe and Woodbridge, 2012; Ward et al., 2012; Chambers et al., 2013; Lee et al., 2013b; Worrall et al., 2013a; Calladine et al., 2014; Velle et al., 2014; Velle and Vandvik, 2014; Alday et al., 2015; Blundell and Holden, 2015; Swindles et al., 2015; Taylor, 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Swindles et al., 2016; Chambers et al., 2017; Douglas et al., 2017; Grau-Andrés et al., 2017; McCarroll et al., 2017; Noble et al., 2017; Robertson et al., 2017; Fyfe et al., 2018; Ludwig et al., 2018; Milligan et al., 2018; Noble et al., 2018a; Noble et al., 2018b; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019a; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al., 2019a; Noble et al., 2019b). Two of these studies were conducted outside the UK within the coastal wet heaths of Norway (Velle et al., 2014; Velle and Vandvik, 2014). Thirteen studies measured burning impacts indirectly by:

- Using paleoecological charcoal analysis (Fyfe and Woodbridge, 2012; Chambers et al., 2013; Blundell and Holden, 2015; Swindles et al., 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Swindles et al., 2016; Chambers et al., 2017; McCarroll et al., 2017; Fyfe et al., 2018), vegetation composition and structure (Calladine et al., 2014), artificial ash additions (Johnston and Robson, 2015; Noble et al., 2017), and changes to bulk density (Noble et al., 2017) as proxies for managed burning.

- Including managed burning as part of a ‘grouse moor’ management variable which also included predator control (Ludwig et al., 2018).

Paleoecology studies are considered separately within this sub-question evidence summary.

It is worth highlighting that seven of the 34 studies investigating the effects of managed burning on upland peatland vegetation used the Hard Hill experimental plots in Moor House National Nature Reserve, Upper Teesdale (Ward et al., 2012; Lee et al., 2013b; Alday et al., 2015; Milligan et al., 2018; Noble et al., 2018a; Marrs et al., 2019a; Noble et al., 2019b). The Hard Hill experiment was established in 1954/55, which makes it the longest-running study investigating the impacts of managed rotational burning and grazing in the UK (Marrs et al., 1986). Located within Moor House National Nature Reserve in Upper Teesdale

(British grid reference: NY 74124 33091), the experimental set-up consists of four 90 x 60 m experimental blocks (A, B, C and D), each of which are divided into six 30 x 30 m sub-plots (Noble et al., 2018a). The experimental blocks are positioned at regular intervals along a gentle hillslope, with block A being the lowest and block D being the highest (Marrs et al., 1986). At the start of the experiment each block was burnt in a single large burn: blocks A, B and D were burned in 1954 and block C was burned in 1955 (Lee et al., 2013a). Thereafter, two grazing treatments (fenced or grazed) and three burning treatments were applied (N = burnt in 1954 only; S = burnt in 1954 and every ten years after; L = burnt in 1954 and every 20 years after) (Rawes and Hobbs, 1979; Marrs et al., 1986). Treatments were assigned within each experimental block by using a randomised split-plot design as follows: four blocks (A-D) × two main treatments (fenced and grazed) × three sub-treatments (N, S, L) (Marrs et al., 1986). In addition to the main plots, unfenced reference plots (R) were established alongside each block outside of the initial 1954 burn areas (Fig 1) (Lee et al., 2013a). It is thought that these plots have not been burnt since 1923 (Rawes and Hobbs, 1979).

4.1.1. Vegetation diversity

Two low-quality studies (-) (Whitehead and Baines, 2018; Grau-Andrés et al., 2019a), two medium quality studies (+) (Velle et al., 2014; Velle and Vandvik, 2014) and three high-quality studies (++) (Milligan et al., 2018; Heinemeyer et al., 2019c; Marrs et al., 2019a) investigated the effect of managed burning on upland peatland vegetation diversity. Two of these studies used vegetation data from the Hard Hill experimental plots (Milligan et al., 2018; Marrs et al., 2019a). The Hard Hill data suggests that, since the start of the experiment (1954), vegetation diversity has marginally increased in the S plots (burnt every ten years) and L plots (burnt every 20 years), but decreased in the N plots (unburnt since 1954) (*ibid*).

A study by Grau-Andrés et al. (2019a) found that species and plant functional type diversity increased after a managed burn relative to unburnt controls. Conversely, Heinemeyer et al. (2019c) found no differences in vegetation diversity between burnt and unburnt peatland plots after four years post-burn. Furthermore, Whitehead and Baines (2018) measured vegetation species richness within unburnt control plots (last burnt >17 years before the start of the study), and plots burnt 1-2, 3-6, 7-10 and 11-17 years before the beginning of the study. Whitehead and Baines (2018) found that vegetation species richness differed across all treatments, but there was no clear pattern (*ibid*). However, when looking at just *Sphagnum* species richness, Whitehead and Baines (2018) found that: i) the unburnt control plots supported the lowest number of *Sphagnum* species; ii) plots burnt 1-2 years ago had the second-lowest numbers of *Sphagnum* species; and, iii) plots burnt 3-6, 7-10 and 11-17 years ago supported the highest number of *Sphagnum* species.

A further two studies examined the effect of managed burning on the vegetation communities within the coastal wet heaths of Norway (Velle et al., 2014; Velle and Vandvik, 2014). Both studies used before-and-after data from the same study plots and found that managed burning leads to an increase in vegetation diversity up to three years post-burn (*ibid*).

4.1.2. Vegetation structure

The impact of managed burning on upland peatland vegetation structure was assessed by three low-quality studies (-) (Robertson et al., 2017; Noble et al., 2018a; Whitehead and Baines, 2018), three medium quality studies (+) (Calladine et al., 2014; Douglas et al., 2017; Noble et al., 2019b) and four high-quality studies (++) (Alday et al., 2015; Heinemeyer et al., 2019a; Heinemeyer et al., 2019c; Noble et al., 2019a).

Four studies measured the impact of managed burning on the structure of the peatland surface (i.e. surface microtopography) (Noble et al., 2018a; Heinemeyer et al., 2019a; Noble et al., 2019a; Noble et al., 2019b). Heinemeyer et al. (2019a) found no differences in peatland surface microtopography between burnt and unburnt plots. Similarly, Noble et al. (2018a) collected data from the Hard Hill experimental plots and found that *Sphagnum* hummock height was similar within S plots (burnt every ten years) and R plots (unburnt since 1923). However, *Sphagnum* hummock height was greater within both S plots and R plots than in L plots (burnt every 20 years) and N plots (unburnt since 1954) (*ibid*). Conversely, Noble et al. (2019a) found that *Sphagnum capillifolium* height increased within unburnt control plots but decreased within burnt plots up to five months post-burn. Finally, Noble et al. (2019b) studied plots burnt one, five and ten years before the start of the study and found that moss depth (cm) generally increased with time since burn.

Seven studies also examined the impacts of managed burning on the structure of the vegetation canopy (usually *Calluna vulgaris* height) within upland peatlands (Calladine et al., 2014; Alday et al., 2015; Douglas et al., 2017; Robertson et al., 2017; Whitehead and Baines, 2018; Heinemeyer et al., 2019c; Noble et al., 2019b). Overall, all but one of these studies (Calladine et al., 2014), indicates that managed burning leads to changes in vegetation canopy height (Alday et al., 2015; Douglas et al., 2017; Robertson et al., 2017; Whitehead and Baines, 2018; Heinemeyer et al., 2019c; Noble et al., 2019b). Obviously, managed burning leads to an initial reduction in the height of the vegetation canopy. However, the height of the vegetation canopy subsequently increases with time since burn (Alday et al., 2015; Douglas et al., 2017; Whitehead and Baines, 2018; Heinemeyer et al., 2019c; Noble et al., 2019b). Interestingly, Robertson et al. (2017) found a positive relationship between variability in *C. vulgaris* canopy height and burning extent across their moorland study sites.

4.1.3. Sphagnum species

Five low-quality studies (-) (Noble et al., 2017; Noble et al., 2018a; Noble et al., 2018b; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a), three medium quality studies (+) (Lee et al., 2013b; Grau-Andrés et al., 2017; Noble et al., 2019b) and five high-quality studies (++) (Taylor, 2015; Milligan et al., 2018; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al., 2019a) investigated the impacts of managed burning on *Sphagnum* species (herein known as "*Sphagnum*"). Five of these studies collected data from the Hard Hill experimental plots (Lee et al., 2013b; Milligan et al., 2018; Noble et al., 2018a; Marrs et al., 2019a; Noble et al., 2019a) and all but one study measured burning impacts on *Sphagnum* directly (Noble et al., 2017).

Most studies took two approaches to measure the effect of managed burning on *Sphagnum*. The first approach involved measuring the abundance of *Sphagnum*. Overall, these studies seem to suggest that burnt areas of upland peatland can support similar amounts of *Sphagnum* than unburnt or not recently burnt areas (Grau-Andrés et al., 2017; Milligan et al., 2018; Noble et al., 2018a; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al., 2019b). However, two studies reported that burning reduces the abundance of *Sphagnum* (Noble et al., 2017; Noble et al., 2018b), with one of these studies using increased peat bulk density and ash deposition as proxies for burning management (Noble et al., 2017).

The second approach involved measuring the heat damage inflicted by managed burning on *Sphagnum* plants (e.g. by measuring cell damage, photosynthetic capacity, net primary productivity, amount of bleaching or the amount of new growth) (Taylor, 2015; Grau-Andrés et al., 2017; Noble et al., 2019a). All of these studies show that managed burning leads to post-fire heat damage of *Sphagnum* plants (*ibid*). However, *Sphagnum* plants show signs of recovery within the space of three years (Taylor, 2015; Grau-Andrés et al., 2017). Thus, given the multiple studies suggesting that *Sphagnum* can be equally abundant on burnt and unburnt areas of upland peatland, the damage to *Sphagnum* plants caused by managed burning seems to be a transient effect. It is also worth considering that the damage inflicted by managed burning on *Sphagnum* plants is dependent on fire temperatures (Taylor, 2015; Grau-Andrés et al., 2017; Noble et al., 2019a), which is itself primarily driven by fuel load and vegetation moisture content (Davies et al., 2010b; Davies et al., 2016a; Grau-Andrés et al., 2018). For example, Noble et al. (2019a) found that, compared to an unburnt control, burning at low temperatures (≤137 ^{o}C) did not cause significant *S. capillifolium* cell damage.

An additional study by Lee et al. (2013b) used a third approach to investigate burning impacts on *Sphagnum*. This study measured the proportion of *Sphagnum* propagules in the top 7 cm of the peat profile within the Hard Hill plots. Lee et al. (2013b) found that the proportion of *Sphagnum* propagules within surface peat increased as burning rotation increased (i.e. *Sphagnum* propagules were lowest in the S plots and highest in the R plots) (*ibid*). This suggests that managed burning reduces the percentage of *Sphagnum* propagules within the peat layers, which contradicts the multiple studies suggesting that *Sphagnum* abundance is not adversely affected by managed burning (Grau-Andrés et al., 2017; Milligan et al., 2018; Noble et al., 2018a; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Marrs et al., 2019a; Noble et al., 2019b).

It should be noted that the evidence on *Sphagnum* impacts included in this review is largely based on data for the most abundant peatland *Sphagnum* species: *S. capillifolium*. Indeed, because the abundance of other *Sphagnum* species is very low, many researchers decide to pool survey data for individual *Sphagnum* species during data analysis. However, the pooled data is often, but not always, dominated by *S. capillifolium*. Whereas, other researchers choose to focus on *S. capillifolium* because it is the most abundant *Sphagnum* species within their study site(s).

4.1.4. Eriophorum species

The impact of managed burning on *Eriophorum*[3] species (henceforth known as "*Eriophorum*") was examined by four low-quality studies (-) (Worrall et al., 2013a; Noble et al., 2018b; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a), two medium quality studies (+) (Grau-Andrés et al., 2019b; Noble et al., 2019b) and five high-quality studies (++) (Ward et al., 2012; Taylor, 2015; Milligan et al., 2018; Heinemeyer et al., 2019c; Marrs et al., 2019a). Three of these studies used data collected from the Hard Hill experimental plots (Ward et al., 2012; Milligan et al., 2018; Marrs et al., 2019a). Across nine of the 11 studies, the abundance (percentage cover or biomass) of *Eriophorum* within burnt plots was greater or

[3] Some of the studies used graminoid abundance (cover or biomass). This was considered a proxy for *Eriophorum* abundance because *Eriophorum* species are usually the most dominant graminoid species in upland peatlands within the UK.

equal to that found in unburnt or not recently burnt plots (Ward et al., 2012; Worrall et al., 2013a; Milligan et al., 2018; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al., 2019b). Conversely, Noble et al. (2018b) found that *Eriophorum vaginatum* cover was greater within unburnt than burnt plots on upland peatland sites. Furthermore, three studies suggest that managed burning leads to an initial increase in the abundance of *Eriophorum* for up to ten years post-burn (Noble et al., 2018b; Whitehead and Baines, 2018; Noble et al., 2019b). However, after ten years have elapsed, *Eriophorum* abundance declines due to the rise in *C. vulagris* cover (*ibid*).

4.1.5. Calluna vulgaris

Four low-quality studies (-) (Worrall et al., 2013a; Noble et al., 2018b; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a), five medium quality studies (+) (Lee et al., 2013b; Velle and Vandvik, 2014; Ludwig et al., 2018; Grau-Andrés et al., 2019b; Noble et al., 2019b) and six high-quality studies (++) (Ward et al., 2012; Alday et al., 2015; Taylor, 2015; Milligan et al., 2018; Heinemeyer et al., 2019c; Marrs et al., 2019a) investigated the effect of managed burning on *C. vulgaris.* Five of these studies collected data from the Hard Hill experimental plots (Ward et al., 2012; Lee et al., 2013b; Alday et al., 2015; Milligan et al., 2018; Marrs et al., 2019a).

Fourteen studies examined the impact of managed burning on *C. vulgaris* abundance (cover or biomass)[4]. Thirteen of these studies found that managed burning leads to a short-term reduction in *C. vulgaris* abundance (Ward et al., 2012; Worrall et al., 2013a; Velle and Vandvik, 2014; Alday et al., 2015; Taylor, 2015; Ludwig et al., 2018; Milligan et al., 2018; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al., 2019b). However, *C. vulgaris* then increases and starts to become dominant within areas that have remained unburnt for more than ten years (e.g. Milligan et al., 2018; Whitehead and Baines, 2018; Marrs et al., 2019a; Noble et al., 2019b). Thus, *C. vulgaris* abundance is lowest on areas of upland peatland that are recently and/or frequently burnt, and highest on unburnt or not recently burnt areas of upland peatland (Ward et al., 2012; Alday et al., 2015; Milligan et al., 2018; Whitehead and Baines, 2018; Heinemeyer et al., 2019c; Marrs et al., 2019a; Noble et al.,

[4] Some of the studies used dwarf shrub abundance (cover or biomass). This was considered a proxy for *C. vulgaris* abundance because *C. vulgaris* is usually the most dominant dwarfshrub species in upland peatlands within the UK.

2019b). Conversely, Noble et al. (2018b) found that burnt plots contained a greater abundance of *C. vulgaris* than unburnt plots (condition monitoring data).

A study by Lee et al. (2013b) used a different approach and investigated the effect of managed burning on *C. vulgaris* propagule banks within i) the litter and peat layers of a burning chronosequence in the peak district; and, ii) the peat layers of the Hard Hill experimental plots. The findings of this study show that i) across the burning chronosequence, *C. vulgaris* propagules were mainly found in the litter layer, which acted as a barrier of transfer to the peat layer; ii) *C vulgaris* propagules within the litter layer increased with time since burn; and, iii) across the Hard Hill experimental plots, *C. vulgaris* propagules within the peat layer increased with burning rotation length (i.e. *C. vulgaris* propagules were lowest in the S plots and highest in the N plots). In short, frequent burning reduces the amount of *C. vulgaris* propagules within the litter and peat layers in upland peatlands (*ibid*). In contrast, Heinemeyer et al. (2019c) found that burnt plots had higher levels of *C. vulgaris* germination than unburnt plots, but only for the first three years post-burn. In fact, Heinemeyer et al. (2019c) found that no *C. vulgaris* plants germinated from seed within unburnt plots throughout the four-year monitoring period.

4.1.6. Bare ground

The impact of managed burning on the creation of bare ground within upland peatlands was investigated by two low-quality studies (-) (Worrall et al., 2013a; Grau-Andrés et al., 2019a), three medium quality studies (+) (Velle and Vandvik, 2014; Grau-Andrés et al., 2019b; Noble et al., 2019b) and one high-quality study (Heinemeyer et al., 2019c). One study found that managed burning did not lead to an increase in the amount of bare ground (Worrall et al., 2013a). Conversely, five studies found that burning leads to an increase in bare ground, at least initially (Velle and Vandvik, 2014; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c; Noble et al., 2019b). However, bare ground percentage cover values recorded within quadrats located in burnt plots are usually <10% (Velle and Vandvik, 2014; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Noble et al., 2019b). Moreover, Heinemeyer et al. (2019c) and Noble et al. (2019b) found that bare ground all but disappears four and ten years post-burn, respectively. Thus, managed burning leads to only a small-scale and transient increase in bare ground.

4.1.7. Paleoecology studies

Ten paleoecology studies were included in this review[5]. Two of these were medium quality studies (+) (Fyfe and Woodbridge, 2012; Fyfe et al., 2018), while the other eight were low-quality studies (-) (Chambers et al., 2013; Blundell and Holden, 2015; Swindles et al., 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Swindles et al., 2016; Chambers et al., 2017; McCarroll et al., 2017). Overall, nine of the ten paleoecology studies found that evidence of fire (wildfire or managed burning) within the peat profile (measured by calculating the number of charcoal macrofossils in the peat layers) was coincident with changes in upland peatland vegetation (measured by calculating the number of different plant macrofossils and pollen species in the peat layers) (Chambers et al., 2013; Blundell and Holden, 2015; Swindles et al., 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Swindles et al., 2016; Chambers et al., 2017; McCarroll et al., 2017; Fyfe et al., 2018). A consistent finding was a decrease in *Sphagnum* macrofossils being coincident with evidence of fire throughout the peat profile (Fyfe and Woodbridge, 2012; Chambers et al., 2013; Blundell and Holden, 2015; Swindles et al., 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Chambers et al., 2017; McCarroll et al., 2017).

4.2. Fauna

Nineteen of the 62 studies included in this review investigated the effect of managed burning on the fauna present within upland peatlands (Dallimer et al., 2012; Johnston, 2012; Turner and Swindles, 2012; Ward et al., 2012; Brown et al., 2013; Ward et al., 2013; Calladine et al., 2014; Douglas et al., 2014; Douglas and Pearce-Higgins, 2014; Johnston and Robson, 2015; Newey et al., 2016; Roos et al., 2016; Buchanan et al., 2017; Douglas et al., 2017; Ludwig et al., 2017; Robertson et al., 2017; Ludwig et al., 2018; Heinemeyer et al., 2019c; Littlewood et al., 2019). Eleven of these studies measured burning impacts directly (Johnston, 2012; Turner and Swindles, 2012; Ward et al., 2012; Brown et al., 2013; Douglas et al., 2014; Newey et al., 2016; Roos et al., 2016; Douglas et al., 2017; Robertson et al., 2017; Heinemeyer et al., 2019c; Littlewood et al., 2019), whereas seven studies measured burning impacts indirectly by using proxies such as different levels of ash deposition (Johnston and Robson, 2015), vegetation structure and composition (Ward et al., 2013; Calladine et al., 2014; Douglas and Pearce-Higgins, 2014; Roos et al., 2016; Buchanan et al., 2017), and general grouse moor management (which including managed burning alongside, vegetation cutting, predator control and reductions in grazing) (Ludwig et al., 2017; Ludwig et al., 2018).

[5] Studies that examine pollen, plant macrofossils and charcoal macrofossils down through the peat profile. This is done to investigate long-term vegetation change and, in some cases, drivers of vegetation change.

4.2.1. Birds

The impact of managed burning on upland peatland bird communities was examined by two low-quality studies (-) (Roos et al., 2016; Robertson et al., 2017) and ten medium quality studies (+) (Dallimer et al., 2012; Calladine et al., 2014; Douglas et al., 2014; Douglas and Pearce-Higgins, 2014; Newey et al., 2016; Buchanan et al., 2017; Douglas et al., 2017; Ludwig et al., 2017; Ludwig et al., 2018; Littlewood et al., 2019). Six of these studies measured burning impacts directly (Dallimer et al., 2012; Douglas et al., 2014; Newey et al., 2016; Douglas et al., 2017; Robertson et al., 2017; Littlewood et al., 2019). In contrast, six studies measured burning impacts indirectly by using proxies for burning management such as vegetation structure and composition (Calladine et al., 2014; Douglas and Pearce-Higgins, 2014; Roos et al., 2016; Buchanan et al., 2017), and general grouse moor management[6] (Ludwig et al., 2017; Ludwig et al., 2018). Furthermore, all twelve bird studies used a correlative study design.

The only consistent result that emerged from these studies is that, by promoting areas with shorter and/or more varied vegetation structure across a moorland, managed burning is likely to have a positive effect on *Pluvialis apricaria* populations within upland peatland habitats (Calladine et al., 2014; Douglas and Pearce-Higgins, 2014; Newey et al., 2016; Buchanan et al., 2017; Douglas et al., 2017; Littlewood et al., 2019). However, even if not recorded, managed burning is usually coincident with predator control in many upland areas, which makes it extremely hard to disentangle the relative effect of managed burning on upland bird species.

4.2.2. Aquatic invertebrates

Three low-quality studies (-) investigated the impact of managed burning on aquatic invertebrate communities within upland peatland streams (Johnston, 2012; Brown et al., 2013; Johnston and Robson, 2015). Two of these studies measured the impact of managed burning directly (Johnston, 2012; Brown et al., 2013). In contrast, Johnston and Robson (2015) used different levels of ash added to within stream mesocosm trays as proxies for managed burning (high, low and no ash additions).

Johnston (2012) and Brown et al. (2013) found that streams draining burnt catchments had slightly higher aquatic invertebrate biodiversity than streams draining

[6] General grouse moor management includes managed burning alongside vegetation cutting, predator control and reductions in grazing.

unburnt catchments. Furthermore, Johnston (2012) and Brown et al. (2013) also found that the abundance of pollution sensitive taxa (e.g. Ephemeroptera) was slightly lower in streams draining burnt catchments than in streams draining unburnt catchments (*ibid*). Conversely, both studies found that the abundance of pollution tolerant taxa (e.g. Chironomidae) was slightly higher in streams draining burnt catchments than in streams draining unburnt catchments (*ibid*). However, the studies of Johnston (2012) and Brown et al. (2013) confounded study site with treatment (managed burning versus no managed burning), and this was not controlled for during statistical analysis. Thus, we cannot be sure whether the results of these studies are due to burning management (managed burning versus no managed burning) or differences between sites. For example, the sites used by Brown et al. (2013) were geographically and environmentally distinct, with burnt catchments receiving less rainfall than unburnt catchments (Ashby and Heinemeyer, 2019a; Ashby and Heinemeyer, 2019b).

A third study by Johnston and Robson (2015) found that different levels of ash additions (high, low and no ash additions added to within stream mesocosm trays) had little effect of aquatic invertebrate communities.

4.2.3. Terrestrial invertebrates

A single high-quality study (++) examined the impact of managed burning on terrestrial invertebrates. This study, by Heinemeyer et al. (2019c), compared cranefly (Tipulidae) emergence between burnt and unburnt control plots (and mown plots) for three years post-management. Cranefly emergence was slightly higher within unburnt plots in the first year post-management (*ibid*). However, in year two and three, cranefly emergence was greater within burnt plots (*ibid*). Importantly, these findings were related to differences in soil surface moisture (top 8 cm) (*ibid*). For example, soil moistures of between 80-95% represent the optimal range for cranefly larval development and emergence (*ibid*). During the first post-management year (a dry year), Heinemeyer et al. (2019c) found that soil surface moisture within unburnt plots was within this optimum range, whereas soil surface moisture within burnt plots was below it (i.e. <80%). Conversely, in the second and third post-management years (both wet years), Heinemeyer et al. (2019c) found that soil surface moisture within burnt plots was within this optimum range, but soil surface moisture within unburnt plots was above it (i.e. >95%). In general, soil surface moisture was lower in burnt plots (*ibid*). Thus, while unburnt plots probably provide better conditions for cranefly emergence in dry and normal years, burnt plots provide better conditions for cranefly emergence in wetter years.

4.2.4. Soil microorganisms

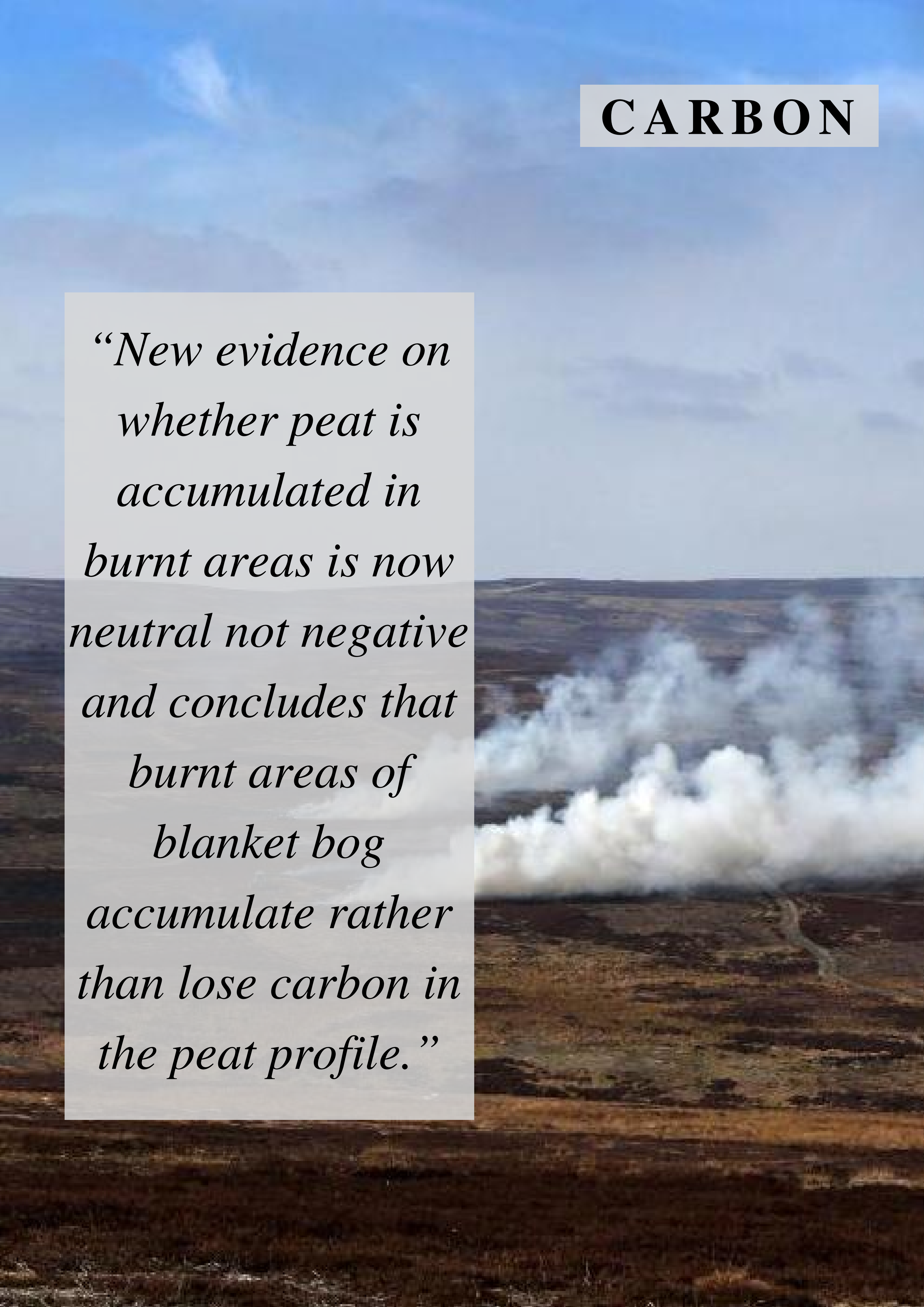
One low-quality study (-) (Turner and Swindles, 2012) and one high-quality study (++) (Ward et al., 2012) investigated the impact of managed burning on soil microorganisms. Ward et al. (2012) used two of the Hard Hill experimental burning treatments and found that S plots (burnt every ten years) had a lower soil fungal biomass than N plots (unburnt since 1954). Conversely, burning did not affect soil bacterial biomass (*ibid*). A second study by Turner and Swindles (2012) found differences in testate amoebae communities between burnt and unburnt areas of upland peatland. For example, while median Shannon diversity values were similar within unburnt and burnt areas of blanket bog, unburnt areas recorded the highest individual Shannon diversity value (*ibid*). Furthermore, the testate amoebae communities within burnt areas of blanket bog were slightly more indicative of drier conditions (i.e. lower water tables) (*ibid*).

4.3. Carbon sequestration and greenhouse gas emissions

A total of 13 studies investigated the effect of managed burning on carbon sequestration and/or GHG emissions (Ward et al., 2012; Ward et al., 2013; Worrall et al., 2013a; Worrall et al., 2013b; Clay et al., 2015; Dixon et al., 2015; Parry et al., 2015; Taylor, 2015; Walker et al., 2016; Heinemeyer et al., 2018; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c; Marrs et al., 2019a). Eight of these studies measured burning impacts directly (Ward et al., 2012; Worrall et al., 2013a; Worrall et al., 2013b; Clay et al., 2015; Taylor, 2015; Heinemeyer et al., 2018; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c; Marrs et al., 2019a), whereas four studies used vegetation composition (Ward et al., 2013; Parry et al., 2015; Walker et al., 2016) or structure (Dixon et al., 2015) as proxies for burning management.

4.3.1. Carbon and peat accumulation within upland peatland soil profiles

One medium quality study (+) (Heinemeyer et al., 2018) and one high-quality study (++) (Marrs et al., 2019a) investigated the effect of managed burning on peat and/or carbon accumulation within upland peatlands. Heinemeyer et al. (2018) used peat core analysis across three upland peatland sites subject to managed burning to investigate carbon accumulation within three time periods: 1700-1850, 1850-1950 and 1950-2015. Heinemeyer et al. (2018) found that there was considerable net carbon accumulation during all three time periods, which suggests that areas of blanket bog subject to managed burning accumulate,



CARBON

“New evidence on whether peat is accumulated in burnt areas is now neutral not negative and concludes that burnt areas of blanket bog accumulate rather than lose carbon in the peat profile.”

rather than lose, carbon. Moreover, Heinemeyer et al. (2018) also found a positive relationship between carbon accumulation rates and charcoal macrofossil concentration throughout the peat profile (number of charcoal pieces per cm^3 of peat), which indicates that burning, via the production of charcoal, may have a positive effect on peatland carbon accumulation. The positive impact of low-severity fires on peatland carbon storage via charcoal production has also been highlighted by studies elsewhere (Leifeld et al., 2018; Flanagan et al., 2020).

A second peat core study by Marrs et al. (2019a) investigated the effect of managed burning on peat and carbon accumulation by using all three of the Hard Hill experimental plots (S, L and N plots) and the R plots outside the main experimental area. Marrs et al. (2019a) found that all the plots showed net carbon and peat accumulation. However, the frequently burnt S plots (burnt every ten years) accumulated significantly less peat and carbon than the R plots (unburnt since at least 1923). It is worth noting that the ten-year rotation of the S plots is not an appropriate burning rotation for many upland peatland sites, which, due to slow *C. vulgaris* growth rates (owing to cold and wet climates), are much more suited to the 20-year rotation of the L plots (Alday et al., 2015). Furthermore, Marrs et al. (2019a) also found no differences in peat height across plots, which means that the differences in peat accumulation between S and R plots are likely due to differences in peat density (i.e. peat density was greater in S plots) and organic carbon content (which was not directly measured) (*ibid*).

4.3.2. Upland peatland carbon fluxes

One low-quality study (-) (Clay et al., 2015), two medium quality studies (+) (Dixon et al., 2015; Grau-Andrés et al., 2019b) and five high-quality studies (++) (Ward et al., 2012; Ward et al., 2013; Taylor, 2015; Walker et al., 2016; Heinemeyer et al., 2019c) investigated the effect of managed burning (either directly or indirectly) on carbon dioxide fluxes. Three of these studies used indirect measurements to examine managed burning impacts on carbon fluxes. Firstly, an indirect study by Dixon et al. (2015) measured carbon fluxes across plots with increasing *C. vulgaris* canopy height. Dixon et al. (2015) found a negative relationship between canopy height and net ecosystem exchange (NEE) [7] and that there was no canopy height at which *C. vulgaris* dominated upland peatland would be a net annual sink of carbon

[7] Net ecosystem exchange is the sum of the carbon dioxide released when plants respire, and the carbon dioxide absorbed when plants photosynthesise. Thus, NEE can be positive or negative, with negative values indicating a carbon dioxide sink.

dioxide. Consequently, the authors suggest that turning upland peatlands into carbon sinks requires a shift away from *C. vulgaris* dominance (*ibid*).

Another indirect study by Walker et al. (2016) measured carbon fluxes across upland peatland plots subject to ambient or warmed climatic conditions and containing different combinations of plant functional types: dwarf shrubs, graminoids, bryophytes, mixed vegetation and bare peat. Walker et al. (2016) found that: i) ecosystem respiration flux (ER) was highest in plots in which dwarf shrubs (i.e. *C. vulgaris*) or graminoids (i.e. *Eriophorum*) were present; ii) artificial climate warming increased ER in the bare peat and dwarf shrub only plots, but had no effect on ER within the bryophyte, graminoid only or fully vegetated treatments (mixture of plant functional types); iii) under ambient conditions, the bryophyte only treatment led to an increase in the respiration of older carbon stocks (the mean age of carbon released was 412 years before present compared to only 40 years before present for the dwarf shrub only treatment); and, iv) under artificial climate warming conditions, the graminoid only treatment, dwarf shrub only treatment and mixed vegetation treatment all led to an increase in the respiration of older carbon stocks (the mean age of carbon released in each treatment was 300, 900 and 2100 before present, respectively).

The third and final indirect study was conducted by Ward et al. (2013), who used the same experimental plots as those used by Walker et al. (2016). Ward et al. (2013) measured NEE of carbon dioxide and found that plots containing dwarf shrubs (i.e. *C. vulgaris*) had the strongest carbon sink function, including dwarf shrub only plots, dwarf shrub and graminoid plots, and dwarf shrub and bryophyte plots.

The remaining five studies measured the impacts of burning on carbon fluxes directly. Clay et al. (2015) examined unburnt and burnt peatland plots that were burnt one, three, five, six, seven, eight, ten and 11 years before the start of the study. The findings suggested that the amount of carbon dioxide absorbed by plants during photosynthesis varied across all treatments, but plots burnt one and 11 years before the start of the study absorbed the highest and lowest amount of carbon dioxide, respectively (*ibid*). ER of carbon dioxide also varied across plots but was highest in the plots burnt one and ten years before the start of the study (*ibid*). Clay et al. (2015) also found that, in general, NEE was negative for young burns but positive for older burns and one out of the two unburnt control sites (i.e. younger burns are carbon sinks, and older burns are net emitters of carbon dioxide).

Taylor (2015) found that ER did not differ between burnt and unburnt plots spread across three upland peatland sites. Conversely, Grau-Andrés et al. (2019b) and Heinemeyer et al. (2019c) found that burnt plots emitted lower levels of carbon dioxide via ER than unburnt plots. However, both studies found that NEE was higher on burnt plots relative to unburnt controls (*ibid*). Heinemeyer et al. (2019c) collected baseline (pre-burn) data and found that burnt plots switch from a net carbon sink to a net carbon source after management, but carbon losses were decreasing over time. Finally, using two of the three Hard Hill treatments, Ward et al. (2012) found no differences in ER, gross primary productivity (GPP)[8] and NEE between the S plots (burnt every ten years) and N plots (unburnt since 1954).

4.3.3. Upland peatland methane fluxes

The effect of managed burning on upland peatland methane fluxes was investigated by one medium quality study (+) (Grau-Andrés et al., 2019b) and four high-quality studies (++) (Ward et al., 2012; Ward et al., 2013; Taylor, 2015; Heinemeyer et al., 2019c). Two of these studies found no differences in methane emissions between burnt and unburnt areas of upland peatland (Ward et al., 2012; Taylor, 2015). However, Grau-Andrés et al. (2019b) found that burnt plots had higher methane emissions than unburnt plots, especially in summer. Conversely, Heinemeyer et al. (2019c) found that burnt plots emitted less methane than unburnt plots in vegetated areas, but in unvegetated areas, burnt and unburnt plots emitted similar amounts of methane. Heinemeyer et al. (2019c) also found a weak positive correlation between the cover of *Eriophorum* species and methane fluxes across all study plots. Furthermore, an indirect study by Ward et al. (2013) found that upland peatland plots containing graminoids (*Eriophorum* species) and no dwarf shrubs (*C. vulgaris*) had the highest methane emissions under both ambient and warmed climatic conditions.

4.3.4. Upland peatland dissolved organic carbon fluxes

The impact of managed burning on upland peatland dissolved organic carbon (DOC) fluxes was investigated by one low-quality study (-) (Worrall et al., 2013b), two medium-quality studies (+) (Parry et al., 2015; Grau-Andrés et al., 2019b) and two high-quality studies (++) (Ward et al., 2013; Heinemeyer et al., 2019c). Three of these studies measured burning impacts directly (Worrall et al., 2013b; Grau-Andrés et al., 2019b; Heinemeyer et al., 2019c), whereas two studies investigated the effect of different plant functional types on upland peatland dissolved organic carbon fluxes (Ward et al., 2013; Parry et al., 2015). Firstly, Grau-Andrés et al. (2019b) and Heinemeyer et al. (2019c) found no differences in soil water DOC concentrations between burnt and unburnt peatland plots. Secondly, Worrall et al. (2013b)

[8] Gross primary productivity is the amount of carbon dioxide uptake by plants during photosynthesis.

investigated the impact of burning, cutting and no vegetation management (i.e. an unmanaged control) on soil water and overland flow DOC concentrations within an upland peatland. This study found that plots subject to burning or cutting treatments had lower soil water DOC concentrations, whereas overland flow DOC concentrations did not differ across treatments (*ibid*). Thirdly, Parry et al. (2015) tested how slope and vegetation composition (plant functional types) influence stream water DOC concentrations within 119 peatland catchments spanning three drainage basins. This study found that different plant functional types[9] had little influence on stream water DOC concentrations (*ibid*). Finally, Ward et al. (2013) found that the removal of dwarf shrubs (i.e. *C.vulgaris*) led to an increase in soil water DOC concentrations.

4.3.5. Charcoal production

The incomplete combustion of vegetation during wild or managed fires leads to the production of a carbon-rich substance called charcoal (Leifeld et al., 2018; Wei et al., 2018; Jones et al., 2019). Charcoal is resistant to oxidation[10], which means it has the potential to lock away large amounts of carbon when it is added to the soil profile on upland peatlands (Worrall et al., 2013a; Heinemeyer et al., 2018; Leifeld et al., 2018). Two studies included in this review examined the relationship between managed burning and charcoal production within upland peatlands (Worrall et al., 2013a; Heinemeyer et al., 2018). Firstly, a low-quality study found that: i) charcoal production "*was approximately 2.6% of the carbon consumed during the fire*"; and, ii) fast burns (<1 minute) at high temperatures (600 ^{o}C) within older stands of *C. vulgaris* (≥15 years old) lead to charcoal additions that increase upland peatland carbon sequestration relative to a no burning policy.

Secondly, a medium quality study (+) by Heinemeyer et al. (2018) carried out peat core analysis using cores taken from three different upland peatland sites managed as grouse moors. This study found that charcoal concentrations (number of charcoal pieces per cm^3 of peat) were positively related to peat bulk density, peat carbon content and thus, carbon accumulation rate (*ibid*). Therefore, the study by Heinemeyer et al. (2018) highlights the potential of managed burning, via charcoal inputs, to increase long-term carbon storage within upland peatland soils. Nevertheless, the results of Heinemeyer et al. (2018) have been debated within the literature (Evans et al., 2019; Heinemeyer et al., 2019b), but, at the same

[9] Ericaceous shrubs, bare peat, mixed vegetation, graminoids or sedges – all assessed using remote sensing.
[10] This is where oxygen is absorbed by carbon molecules and then emitted as carbon dioxide.

time, they are also supported by a study on low-severity fires within a North American peatland (Flanagan et al., 2020).

4.3.6. Upland peatland greenhouse gas budgets

One low-quality study (-) (Clay et al., 2015) and one high-quality study (Heinemeyer et al., 2019c) calculated the effect of managed burning on GHG budgets. Firstly, Clay et al. (2015) estimated[11] GHG budgets across unburnt plots and plots that were burnt one, three, five, six, seven, eight, ten and 11 years before the start of the study. GHG gas budgets were estimated by Clay et al. (2015) using: i) the annual flux of carbon dioxide through photosynthesis; ii) the annual flux of DOC through ecosystem respiration; iii) the annual flux of particulate organic carbon (POC); iv) the annual DOC flux; v) the annual flux of dissolved carbon dioxide; and, vi) the annual methane flux. Clay et al. (2015) found that all the treatment plots were net sources of GHGs, but the most recently burnt plots were smaller sources of carbon than older burns and control plots, which suggests that the "*burning of Calluna-dominated landscapes leads to an 'avoided loss' of carbon*".

Secondly, Heinemeyer et al. (2019c) measured GHG budgets within burnt and unburnt plots over five years (with one-year pre-burn) using: i) NEE; ii) the annual flux of DOC; iii) the annual flux of POC; and, iv) the annual methane flux. The five-year mean suggests that both burnt and unburnt plots were net sources of GHG emissions, but burnt plots showed (expectedly) higher losses over the first five years post-management due to the removal of vegetation biomass (*ibid*).

4.4. Water quality and flow

Six studies investigated the impact of managed burning on water quality and water flow (Johnston, 2012; Brown et al., 2013; Worrall et al., 2013b; Holden et al., 2015; Parry et al., 2015; Heinemeyer et al., 2019c). Five of these studies examined burning impacts directly (Johnston, 2012; Brown et al., 2013; Worrall et al., 2013b; Holden et al., 2015; Heinemeyer et al., 2019c), whereas one study examined burning impacts indirectly by using vegetation composition as a proxy for burning management (Parry et al., 2015).

4.4.1. Water quality

[11] This study estimated (using secondary data) rather than measured some of the elements making up the GHG budget.

Three low-quality studies (-) (Johnston, 2012; Brown et al., 2013; Worrall et al., 2013b), one medium quality study (+) (Parry et al., 2015) and one high-quality study (Heinemeyer et al., 2019c) investigated the impacts of managed burning (either directly or indirectly) on water quality within upland peatlands. Three of these studies used water colour (measured using specific absorbance) as a measure of water quality. Firstly, a plot-scale study by Worrall et al. (2013b) found that managed burning had no effect on water colour within soil pore water or surface run-off within an upland peatland. Secondly, another plot scale study by Heinemeyer et al. (2019c) also found that burning had no effect on soil pore watercolour, but recorded several relationships between water colour and vegetation composition (e.g. increased water colour under increased *Eriophorum* and *Sphagnum* cover, and decreased water colour under increased *C. vulgaris* cover). Thirdly, Parry et al. (2015) investigated the influence of slope and vegetation type (plant functional types) on stream water colour within 119 peatland catchments spanning three drainage basins. Parry et al. (2015) found that different plant functional types[12] had little influence on stream water colour.

Three studies investigated how managed burning influences other aspects of upland peatland water quality. For example, Brown et al. (2013) compared the water quality of five streams draining unburnt peatlands to that of five streams draining burnt peatlands. They found that water within rivers draining burnt peatland had a lower pH and higher concentrations of Si, Mn, Fe, Al, coarse organic matter, and fine organic matter (*ibid*). Similarly, Johnston (2012) compared the water quality of ten streams draining burnt peatlands to that of ten streams draining unburnt peatlands and ten streams draining eroding peatlands. Johnston (2012) found that stream water pH did not differ across catchment types. However, conductivity was higher in streams within burnt and degraded catchments (*ibid*). Finally, Heinemeyer et al. (2019c) found that burning did not affect soil pore water pH.

4.4.2. Water flow

Three low-quality studies (-) (Johnston, 2012; Worrall et al., 2013b; Holden et al., 2015) and one high-quality study (Heinemeyer et al., 2019c) investigated the impacts of managed burning on water flow within upland peatlands. Three of these studies investigated the impact of managed burning on peatland water table depth and/or overland flow (Worrall et al., 2013b; Holden et al., 2015; Heinemeyer et al., 2019c). Worrall et al. (2013b) found that burnt plots had higher water tables than unburnt plots, which they attributed to a reduction in

[12] Ericaceous shrubs, bare peat, mixed vegetation, graminoids or sedges – all assessed using remote sensing.

evapotranspiration that was mediated by the removal of vegetation biomass. Worrall et al. (2013b) also found that burnt plots had a higher frequency of overland flow events than unburnt plots. In contrast to Worrall et al. (2013b), Holden et al. (2015) found that, on average, burnt plots had slightly (~5cm) deeper water tables than unburnt plots. However, when considering the burning age of each burnt plots, this study suggested that water tables recover to a similar level to those found in unburnt plots after >10 years (*ibid*). Holden et al. (2015) also measured overland flow occurrence and, like Worrall et al. (2013b), found that the occurrence of overland flow was greater on burnt than unburnt plots. But the positive effect of burning on overland flow occurrence was not apparent when comparing plots with different burning ages (plots burnt <2 years, 4 years, 7 years and 10+ years since the start of the study) with unburnt plots. In line with Holden et al. (2015), Heinemeyer et al. (2019c) also found that burnt plots had slightly lower water tables than unburnt plots.

Finally, two studies investigated the effect that managed burning has on streamflow by comparing burnt to unburnt upland peatland catchments (Johnston, 2012; Holden et al., 2015). Holden et al. (2015) calculated multiple streamflow metrics for the largest 20% of storm events. Only hydrograph intensity[13] revealed any significant differences in river storm response between burnt and unburnt catchments (it was higher in burnt catchments) (*ibid*). In contrast, Johnston (2012) found no differences in streamflow between burnt, unburnt and eroding peatland catchments.

4.5. Fire ecology

Thirteen studies investigated how differences in burn severity[14] or frequency[15] affect upland peatland ecosystem services (Lee et al., 2013b; Worrall et al., 2013a; Alday et al., 2015; Taylor, 2015; Grau-Andrés et al., 2017; Grau-Andrés et al., 2018; Heinemeyer et al., 2018; Milligan et al., 2018; Noble et al., 2018a; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b; Marrs et al., 2019a; Noble et al., 2019a). All 13 studies measured burning impacts directly (*ibid*). Also, six of the studies collected data from the Hard Hill experimental plots (Lee et al., 2013b; Alday et al., 2015; Milligan et al., 2018; Noble et al., 2018a; Marrs et al., 2019a; Noble et al., 2019a).

4.5.1. Burn severity

[13] This is calculated by dividing peak flow values by total stormflow values.
[14] Burn severity relates to the temperatures experienced during a managed burn – the higher the temperatures, the higher the burn severity.
[15] Burn frequency is the number of times an area of interest (e.g. a vegetation plot) has been burnt.

The impact of burn severity was investigated by two low-quality studies (-) (Worrall et al., 2013a; Grau-Andrés et al., 2019a), three medium quality studies (+) (Grau-Andrés et al., 2017; Grau-Andrés et al., 2018; Grau-Andrés et al., 2019b) and two high-quality studies (++) (Taylor, 2015; Noble et al., 2019a). The only consistent finding to emerge across these studies is a positive relationship between *S. capillifolium* damage and burn severity, with lower burn severities causing only very negligible damage to *S. capillifolium* plants relative to unburnt controls (Taylor, 2015; Grau-Andrés et al., 2017; Noble et al., 2019a). Nevertheless, *S. capillifolium* plants are still able to recover even after experiencing a high severity burn (e.g. Clymo and Duckett, 1986; Taylor, 2015; Grau-Andrés et al., 2017)

4.5.2. Burn frequency

One low-quality study (-) (Noble et al., 2018a), two medium quality studies (+) (Lee et al., 2013b; Heinemeyer et al., 2018) and three high-quality studies (++) (Alday et al., 2015; Milligan et al., 2018; Marrs et al., 2019a) measured the effect of burning frequency on upland peatland ecosystem services. The only consistent results are from the five vegetation studies that all analyse data from the Hard Hill experimental plots (Lee et al., 2013b; Alday et al., 2015; Milligan et al., 2018; Noble et al., 2018a; Marrs et al., 2019a). These studies suggest that frequent burning reduces *C. vulgaris* abundance (adult plants and propagules) and increases *Eriophorum* abundance (Lee et al., 2013b; Alday et al., 2015; Milligan et al., 2018; Marrs et al., 2019a). Furthermore, the frequently burnt S plots (burnt every ten years) support similar amounts of *Sphagnum* (mainly *S. capillifolium*) than are found within the L plots (burnt every 20 years), N plots (unburnt since 1954) and R plots (unburnt since 1923) (Milligan et al., 2018; Noble et al., 2018a; Marrs et al., 2019a).

Two further studies investigated the impact of burn frequency on carbon and/or peat accumulation within upland peatland soil profiles (Heinemeyer et al., 2018; Marrs et al., 2019a)

Firstly, Heinemeyer et al. (2018) analysed peat cores taken from three upland peatlands subject to managed burning. Heinemeyer et al. (2018) found that carbon accumulation rates were greater on the most frequently burnt site during 1950–2015 and 1700–1850, which was linked to increases in peat bulk density and charcoal macrofossil concentrations (*ibid*). Secondly, Marrs et al. (2019a) used the Hard Hill experimental plots and found that, while all treatments were accumulating peat and carbon, there was a negative relationship between the number of managed burns a plot has received (S plots = six burns; L plots = three burns, N plots = one burn; R plots = no burns) with peat and carbon accumulation. However, this relationship was driven by the significantly lower peat and carbon accumulation rates recorded in the S plots relative to the R plots (*ibid*). Furthermore, as previously mentioned, the S plots (burnt every ten years) do not represent a realistic rotation length for the local growing conditions in many upland peatlands, which are much more suited to the 20-year rotation of the L plots (Alday et al., 2015).

4.6. Wildfire

No study directly examined the relationship between managed burning and wildfire, but three studies examined this relationship indirectly (Ward et al., 2012; Alday et al., 2015; Heinemeyer et al., 2019c). Two of these studies were conducted using the Hard Hill experimental plots (Ward et al., 2012; Alday et al., 2015)

4.6.1. Fuel loads

Three high-quality studies (++) found that burning reduces heather fuel loads (i.e. dwarf shrub biomass) (Ward et al., 2012; Alday et al., 2015; Heinemeyer et al., 2019c). Two of these studies collected data from the Hard Hill plots (Ward et al., 2012; Alday et al., 2015). The first of these studies by Ward et al. (2012) only used two of the three Hard Hill burning treatment plots: S plots (burnt every ten years) and N plots (unburnt since 1954). Ward et al. (2012) found that dwarf shrub biomass (g m^{-2}) within N plots was between 1,117 and 3476% higher than in S plots (*ibid*).

A second Hard Hill study conducted by Alday et al. (2015) used all three burning treatment plots: S plots (burnt every ten years), L plots (burnt every 20 years) and N plots (unburnt since 1954). The study by Alday et al. (2015) also investigated vegetation biomass within the R plots (plots outside the experimental area unburnt since at least 1923). Alday et al. (2015) found that *C. vulgaris* biomass decreased with increasing time since burn in the main experimental plots: S plots = 60 ± 16 g m^{-2}; L Plots = 672 ± 39 g m^{-2}; and, N plots = 808 ± 16 g m^{-2}. However, *C. vulgaris* biomass within R plots (705 ± 73 g m^{-2}) was intermediate between N and L plots (*ibid*). Similarly, total vegetation biomass increased with increasing time since burn across all the plots investigated: S plots = 1198 ± 165 g m^{-2}; L Plots = 1593 ± 119 g m^{-2}; N plots = 2079 ± 144 g m^{-2}; and, R plots = 2223 ± 201 g m^{-2} (*ibid*).

A third study by Heinemeyer et al. (2019c), which did not use the Hard Hill plots, also found that burning reduces *C. vulgaris* biomass. For example, mean *C. vulgaris* biomass was 97.0 ± 24.9 g within unburnt plots and 6.0 ± 1.4 g within burnt plots two-years post-management (biomass measurements per 660 cm^2).

4.7. Burning extent

Three studies investigated the extent, frequency, practice and/or type of managed burning on upland peatlands (Thacker et al., 2014; Douglas et al., 2015; Allen et al., 2016).

4.7.1. The current extent of managed burning

Three medium quality studies (+) measured the current extent of managed burning on upland peatlands (Thacker et al., 2014; Douglas et al., 2015; Allen et al., 2016). One study measured the extent of managed burning on a single moorland site using management maps and aerial photography[16] (Allen et al., 2016). This study found that an area of 4.16 km^2 was burned at least once over a 22-year-period (1988-2009 broken down into six discrete time periods), which equated to 20% of the entire moorland area or 29% of the"potentially-burnable" area[17] (*ibid*). In addition, over the 22-year-period, the annual amount of burning ranged between 0.5 and 1.6% (0.10 and 0.33 km^2) of the entire moor area or between 0.7 and 2.4% (0.10 and 0.35 km^2) of the potentially-burnable area (*ibid*). Obviously, because this study collected data from a single site (*ibid*), the results cannot be extrapolated across the wider peatland resource.

A second study measured burning extent on upland peatlands by using aerial imagery from upland areas of the UK (images were from the years 2001 to 2010, inclusive) (Douglas et al., 2015). This study found that 278 km^2 of deep peat is currently subject to managed burning in England (*ibid*). According to the extent data provided by the emission inventory of UK peatlands (Evans et al., 2017), the area of peatland subject to burning recorded by Douglas et al. (2015) equates to 8.6% of the total blanket bog (all bog types) or 4.1% of the total peatland area in England. However, the study by Douglas et al. (2015) also found that across England, Scotland and Wales, the mean area of moorland (all soil types) burned per 1 km^2 was higher inside than outside protected areas, such as Special Areas of Conservation and Special Protection Areas (SACs and SPAs, respectively) (*ibid*). Importantly, Douglas et al. (2015) did not validate their methodology by ground-truthing any of the burnt patches digitised using aerial imagery (Davies et al., 2016d; Douglas et al., 2016b). Consequently, the results presented by Douglas et al. (2015) should be treated with caution.

A third study mapped the current (2010) extent of managed burning using aerial imagery that covered 1612 km^2 (80%) of the dwarf shrub-dominated (i.e. *C. vulgaris*

[16] Aerial images were used to validate, digitise and georeference the burn patches determined using estate management maps.
[17] The potentially burnable area is the total moor area minus areas where burning is restricted or not desired.

dominated) upland peatland in England (Thacker et al., 2014). The results of this study suggest that >33 km^2 of new burns are carried out on *C. vulgaris* dominated deep peat soils within the uplands every year (*ibid*). According to Thacker et al. (2014), >33 km^2 per year equates to 3.76% of the total dwarf shrub-dominated upland peatland in England. Conversely, according to the extent data provided by the emission inventory of UK peatlands (Evans et al., 2017), 33 km^2 equates to 1% of the total blanket bog (all bog types) area or 0.5% of the total peatland area in England. However, Thacker et al. (2014) suggest that 33 km^2 is likely to be an imprecise estimate of the annual area burned because 20% of the *C. vulgaris* dominated peatland in the English uplands was unmapped by their study. Furthermore, as with Douglas et al. (2015), Thacker et al. (2014) did not validate their methodology by ground-truthing, which further calls into question the accuracy of their results.

Thacker et al. (2014) also estimated the current (up to 2014 for some sites) extent of managed burning on deep peat within protected areas, such as SACs, SPAs and Sites of Special Scientific Interest (SSSIs). Burning extent on upland peatland ranged from 0-18.8 km^2 per year across all the protected areas studied (*ibid*). However, burning extent was generally below <5 km^2 per year across most sites (*ibid*). However, these results were also not validated by ground-truthing, which means they should be treated with caution.

4.7.2. Temporal changes to the extent of managed burning

Two medium quality studies (+) measured temporal changes in the extent of managed burning on upland peatlands (Thacker et al., 2014; Allen et al., 2016). Allen et al. (2016) measured temporal changes in the extent of managed burning on a single moorland site using estate management maps and aerial photography spanning six sampling periods: i) 1988-1990; ii) 1991-1995; iii) 1996-1999; iv) 2000-2002; v) 2003-2005; and, vi) 2006-2009. They found that the annual extent of managed burning was smaller in 1988-1990 (0.10 km^2; 0.5% of the total area; 0.7% of the potentially burnable area) than in 2006-2009 (0.34 km^2; 1.6% of the total area; 2.4% of the potentially burnable area) (*ibid*). However, the annual extent of managed burning did not increase linearly across all six time periods (*ibid*). As stated previously, the results of Allen et al. (2016) are from a single site. Therefore, Allen et al. (2016) cannot be used to infer temporal increases in burning extent across the wider upland peatland resource.

A second study by Thacker et al. (2014) used a random sample of aerial images covering 2% of the English uplands and found that managed burning on deep peat has increased from 5.3km^2 yr^{-1} in 1945-1959 to 38.9km^2 yr^{-1} in 2010. Nevertheless, and as previously mentioned, the results of Thacker et al. (2014) should be treated with caution because they did not validate their methodology by ground-truthing digitised burning extent.

4.7.3. Managed burning return intervals

Two medium quality studies (+) measured managed burning return intervals[18] on upland peatlands (Thacker et al., 2014; Allen et al., 2016). Firstly, Allen et al. (2016) measured managed burning return intervals on a single moorland site using estate management maps and aerial photography. They found that the annual amount of burning ranged between 0.5 and 1.6% (0.10 and 0.33 km^2) of the entire moorland area or between 0.7 and 2.4% (0.10 and 0.35 km^2) of the“potentially-burnable” area (*ibid*). These values translate into burning return intervals of 142–42 and 200–63 years, respectively (*ibid*). However, being from a single site, the results of Allen et al. (2016) cannot be used to infer burning return intervals across the wider peatland resource.

Thacker et al. (2014) used aerial imagery to measure managed burning return intervals across England as well as within a range of SACs, SPAs and Sites of Special Scientific Interest (SSSI) (the aerial images used were from 2006-2014 inclusive). Their data suggest that fire return intervals on deep peat are 26.6 years for the whole of England and between 11.4 to >100 years across individual SACs, SPAs and SSSIs (*ibid*). Nevertheless, caution is required when interpreting the results of Thacker et al. (2014) because they did not validate their methodology by ground-truthing digitised burn areas.

4.7.4. The frequency of managed burning

In total, two medium quality studies (+) measured the frequency[19] of managed burning within upland peatlands (Douglas et al., 2015; Allen et al., 2016). Allen et al. (2016) measured temporal changes in the frequency of managed burning on a single moorland site using estate management maps and aerial photography spanning six sampling periods: i) 1988-1990; ii) 1991-1995; iii) 1996-1999; iv) 2000-2002; v) 2003-2005; and, vi) 2006-2009. Overall, 2,561 burns were carried out across the six sampling periods, which equates to a mean of 116 burns per year (*ibid*). The frequency of burns carried out in the most recent sampling period (2006-2009) was higher than during the earliest sampling period (1988-1990), but temporal trends were not analysed using statistical tests (*ibid*). Furthermore, the number of burns carried out during each sampling period fluctuated considerably: 1988-1990 = 61 burns; 1991-1995 =

[18] The length of time, in years, for an entire region of interest to be burnt (Thacker et al., 2014).
[19] The number of burns carried out within a defined time period (e.g. a year).

716 burns; 1996-1999 = 555 burns; 2000-2002 = 498 burns; 2003-2005 = 201 burns; and, 2006-2009 = 530 burns (*ibid*). Importantly, because the results of Allen et al. (2016) are from a single moorland site, they cannot be used to infer burning frequencies across the wider peatland resource.

A second study measured burning frequency on upland peatlands by using aerial imagery from upland areas of the UK (images were from the years 2001 to 2010, inclusive) (Douglas et al., 2015). This study found that "*In England and Scotland, where we had country-wide peat depth data, there was a significant overall increase in annual burn trends*" (*ibid*). However, the raw burn frequency data (for burns on peat) is not provided within the paper or supplementary materials, which means we cannot see the variability of burning frequency across survey years. Also, Douglas et al. (2015) did not validate their methodology by ground-truthing any of the burn patches digitised using aerial imagery (Davies et al., 2016d; Douglas et al., 2016b). Consequently, the results presented by Douglas et al. (2015) should be treated with caution.

4.7.5. The size of management burning patches

One medium quality study (+) measured the size of managed burning patches on upland peatlands (Allen et al., 2016). This study measured the size of managed burning patches on a single moorland site using estate management maps and aerial photography (the latter were used to validate, digitise and georeference the burn patches determined using estate management maps) spanning six sampling periods: i) 1988-1990; ii) 1991-1995; iii) 1996-1999; iv) 2000-2002; v) 2003-2005; and, vi) 2006-2009. Across the entire study period, the mean burn patch size was 2098 ± 67 m^2 and burn patch sizes ranged from 33-110,000 m^2 (*ibid*). However, most of the burn patches throughout the study period were between 501 and 1000 m^2 (*ibid*). It is also worth noting that burn patch size varied considerably: 1988-1990 = 5080 ± 1780 m^2; 1991-1995 = 1800 ± 80 m^2; 1996-1999 = 1530 ± 85 m^2; 2000-2002 = 2060 ± 94 m^2; 2003-2005 = 2640 ± 284 m^2; and, 2006-2009 = 2580 ± 141 m^2 (*ibid*). However, results from this single site tell us very little about the size of burning patches across the wider peatland resource.

4.8. Soils

Ten studies investigated the impact of managed burning on peat soils (Rosenburgh et al., 2013; Vane et al., 2013; Brown et al., 2015b; Clay et al., 2015; Grau-Andrés et al., 2018;

Grau-Andrés et al., 2019b; Heinemeyer et al., 2019a; Heinemeyer et al., 2019c; Morton and Heinemeyer, 2019; Noble et al., 2019a). All ten of these studies measured burning impacts directly (as opposed to using proxies for managed burning, such as vegetation height or composition).

4.8.1. Post-fire soil temperatures

One high-quality (+) study (Heinemeyer et al., 2019c), two medium quality (+) studies (Grau-Andrés et al., 2018; Grau-Andrés et al., 2019b) and one low quality (-) study (Brown et al., 2015b) investigated the impact of managed burning on post-fire soil temperatures. Three of these studies found that, compared to unburnt or not recently burnt plots, post-fire soil temperatures were higher in burnt plots (Brown et al., 2015b; Grau-Andrés et al., 2018; Grau-Andrés et al., 2019b). However, in general, the mean differences in post-fire soil temperatures (at various depths) between burnt/recently burnt and unburnt/not recently burnt plots were generally <1^{o}C (*ibid*). Furthermore, Grau-Andrés et al. (2018) and Grau-Andrés et al. (2019b) found that burning only minimally increased post-fire soil temperatures during the summer months, but not in Spring or Autumn. Grau-Andrés et al. (2018) also found that, relative to unburnt plots, burning did not affect soil accumulated heat, which is "*the daily growing degree hours for each plot, i.e. the sum of ^{o}C above 4^{o}C, the minimum temperature for plant growth, in each hour during a day*". Finally, Heinemeyer et al. (2019c) found that mean post-fire soil temperatures were similar within burnt and unburnt plots. However, burnt plots had larger soil temperature ranges (increased maxima and minima) and slightly higher maximum soil temperatures (*ibid*).

4.8.2. Soil compaction

Three high-quality studies (++) (Heinemeyer et al., 2019a; Morton and Heinemeyer, 2019; Noble et al., 2019a) and one low-quality study (-) (Rosenburgh et al., 2013) examined the impact of burning on soil compaction. Heinemeyer et al. (2019a) found no differences in soil compaction between unburnt and burnt treatments, which was measured using soil bulk density and peat depth pre and post-management. Simalarly, Rosenburgh et al. (2013) found that time since burn had no effect of soil compaction, which was measured using soil bulk density.

Conversely, Morton and Heinemeyer (2019) found that, relative to an unburnt control, burning reduced peat height after two years post-management (Morton and Heinemeyer, 2019). However, the interaction between site and management recorded in the study suggested that the negative effect of burning on peat height was driven by the results from one of the three sites used, most likely in relation to slope position and peat compaction (shrinkage) due to lower water tables (*ibid*). Finally, Noble et al. (2019a) found that burning led to an increase in peat bulk density after five months post-treatment, but only when unburnt plots were compared to "high-temperature" plots[20]. Indeed, there were no differences in post-treatment peat bulk density between "low temperature" plots[21] and unburnt plots. Importantly, however, increased peat bulk density has been linked to increased charcoal inputs (Heinemeyer et al., 2018).

4.8.3. Soil moisture

One medium quality study (+) (Grau-Andrés et al., 2019b) and two high-quality studies (++) (Heinemeyer et al., 2019c; Noble et al., 2019a) examined the impact of burning on post-fire soil moisture. Grau-Andrés et al. (2019b) measured soil moisture in the top 6 cm of the peat surface and found that burning did not affect post-fire soil moisture compared to unburnt controls. Conversely, two additional studies measured soil moisture in the top 6-8 cm of the peat surface and found that, relative to an unburnt control, burning decreased post-fire soil moisture (Heinemeyer et al., 2019c; Noble et al., 2019a).

4.8.4. Soil chemistry

Two low-quality studies (-) investigated the impact of burning on soil chemistry (Rosenburgh et al., 2013; Vane et al., 2013). One study examined the concentrations of polycyclic aromatic hydrocarbons (PAH) added to the peat surface after vegetation burning (Vane et al., 2013). This study found that, compared to unburnt vegetation, burnt surface ash had much higher concentrations of the 18 PAH studied, which suggests vegetation burning on upland peatlands leads to the net addition of PAH to the soil surface (*ibid*). Nevertheless, "*there was no evidence to suggest that the amounts of PAH accumulating from moorland burning are harmful to humans since these are below the generic assessment criteria for soils*" (*ibid*).

The other study investigated how time since burn affects the concentration of multiple chemical properties within peat soils (Rosenburgh et al., 2013). Overall, time since burn did not affect most of the soil chemical properties measured within this study (*ibid*). However, the study did record a negative relationship between time since burning and soil C:N (carbon

[20] Plots where fire temperatures were between 324-538^{o}C
[21] Plots where fire temperatures were between 33-137^{o}C

to nitrogen) ratios, which suggests that peatlands become gradually more saturated with nitrogen as time since burning increases (*ibid*).

4.8.5. Upland peatland soil erosion

One low-quality study (-) investigated the impact of time since burn and soil erosion (Clay et al., 2015). This study used erosion pins[22] and found that more recently burnt plots[23] lost peat, whereas plots burnt seven or more years before the start of the study actively accumulated peat (*ibid*). However, erosion pins inserted into the top 200mm of the peat surface are not a reliable way to measure soil erosion. For example, the peat surface is likely to have moved during the study period due to natural soil contraction and expansion (e.g. wet-dry cycles) (Morton and Heinemeyer, 2019), rather than peat erosion or accumulation.

[22] Pins were 600mm long, 2mm diameter stainless steel rods inserted 200mm into the peat surface.
[23] Plots burnt one, three and six years before the start of the study.

5. Evidence summary statements

The studies included within this review use a diverse range of experimental designs, predictor variables and measurements. Such heterogeneity prevents the use of meta-analysis to objectively summarise the impacts of managed burning on peatland ecosystem services (Haidich, 2010; Shorten and Shorten, 2013). Consequently, the evidence compiled herein has been summarised using a narrative synthesis approach (Grant and Booth, 2009). As such, the following evidence statements are subjective and should, therefore, be considered as highly uncertain. Nevertheless, the methods, rationale and supporting data behind these evidence statements are fully transparent. Thus, even if other researchers disagree with the evidence summaries provided below, they will understand how they were formed.

5.1. Flora

5.1.1. Vegetation diversity

Quantity and quality of the evidence assessed for this topic: Seven studies examined the effect of burning on vegetation species richness or diversity – two low-quality studies (-), two medium quality studies (+) and three high-quality studies (++). One low-quality study (-) investigated the impact of managed burning on *Sphagnum* species richness.

Is the direction of the evidence consistent? No, for vegetation species richness or diversity. NA, for *Sphagnum* species richness – cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? NA.

5.1.2. Vegetation structure

Quantity and quality of the evidence assessed for this topic: Four studies examined burning impacts on the microtopography of the peatland surface – one low-quality study (-), one medium quality study (+) and two high-quality studies (++). Seven studies examined the impacts of managed burning on the structure of the vegetation canopy (usually heather height) – two low-quality studies (-), three medium quality studies (+) and two high-quality studies (++).

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"The new review concludes that burning has a neutral effect on Sphagnum abundance and initial damage done by low severity fire to Sphagnum capillifolium almost fully recovers within three years..."

Is the direction of the evidence consistent? No, for burning impacts on surface microtopography. Yes, for burning impacts on vegetation canopy height.

If so, what is the direction of the evidence? Unsurprisingly, managed burning leads to a short-term reduction in canopy height, but canopy height then increases with time since burn.

5.1.3. Sphagnum species

Quantity and quality of the evidence assessed for this topic: Ten studies examined the impact of burning on *Sphagnum* abundance (primarily, *S. capillifolium* abundance) – five low-quality studies (-), two medium studies (+) and three high-quality studies (++). Three studies examined the temperature-induced *S. capillifolium* damage during managed burning – one medium quality study (+) and two high-quality studies (++). One medium quality study (+) examined the impact of burning on the proportion of *Sphagnum* propagules in the surface peat layers.

Is the direction of the evidence consistent? Yes, for S. *capillifolium* abundance. Yes, temperature-induced *S. capillifolium* damage. NA, for the proportion of *Sphagnum* propagules in the surface peat layers – cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? Managed burning seems to have a neutral impact on *S. capillifolium* abundance. However, managed burning does lead to short-term damage of *S. capillifolium* plants, with affected plants recovering within the space of three years (two out of three studies). Although, damage to *S. capillifolium* plants is likely to be minimal to absent when managed burns do not exceed 137 ^{o}C at the soil or vegetation surface.

5.1.4. Eriophorum species

Quantity and quality of the evidence assessed for this topic: Eleven studies – four low-quality (-), two medium quality studies (+) and five high-quality studies (++).

Is the direction of the evidence consistent? No.

If so, what is the direction of the evidence? NA.

5.1.5. Calluna vulgaris

Quantity and quality of the evidence assessed for this topic: Fourteen studies examined the impact of burning on *C. vulgaris* abundance – four low-quality studies (-), four medium quality studies (+) and six high-quality studies (++). One medium quality study (+) examined the impact of burning on the proportion of *C. vulgaris* propagules in the surface peat and litter layers. One high-quality study (++) examined the impact of burning on *C. vulgaris* germination.

Is the direction of the evidence consistent? Yes, for *C. vulgaris* abundance. NA, for the proportion of *C. vulgaris* propagules – cannot assess evidence consistency using a single study. NA, for *C. vulgaris* germination – cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? Frequent managed burning (i.e. rotational burning) reduces *C. vulgaris* abundance, but *C. vulgaris* increases and eventually becomes dominant within areas left unburnt for long periods (e.g. it has remained dominant for 90+ years within the Hard Hill Experiment unburnt reference plots: Milligan et al., 2018).

5.1.6. Bare ground

Quantity and quality of the evidence assessed for this topic: Six studies – two low-quality studies (-), three medium quality studies (+) and one high-quality study (++).

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Burning leads to the small-scale increase in bare ground, but this seems to be a transient effect (lasting four to ten years).

5.1.7. Paleoecology studies

Quantity and quality of the evidence assessed for this topic: Nine studies explored the relationship between fire (wildfire or managed burning) and *Sphagnum* occurrence down through the peat profile – eight low-quality studies (-) and one medium quality study (+). Nine studies explored the relationship between fire (wildfire or managed burning) and *C. vulgaris* occurrence down through the peat profile – seven low-quality studies (-) and two medium quality studies (+). Six studies explored the relationship between fire (wildfire or managed burning) and *Eriophorum* occurrence down through the peat profile – five low-quality studies (-) and one medium quality study (+).

Is the direction of the evidence consistent? Yes, for *Sphagnum.* No, for *Eriophorum* and *C. vulgaris*.

If so, what is the direction of the evidence? In general, increased evidence of fire within the peat profile (i.e. the abundance of charcoal macrofossils) is coincident with declines in *Sphagnum* abundance. However, it is important to note that none of the paleoecology studies included in this review tested the relationship between fire occurrence and *Sphagnum* abundance. Nor did they consider other drivers of vegetation change, such as grazing (wild or domesticated), drainage, climate, carbon dioxide levels (in terms of photosynthesis) or atmospheric pollution (e.g. sulphur or nitrogen). Thus, the paleoecology studies included in this review should be considered as circumstantial evidence. Another issue with the paleoecology studies included in this review is the lack of spatial replication. Indeed, half of the studies only explored relationships using a single master peat core from within a single site (Blundell and Holden, 2015; McCarroll et al., 2016b; McCarroll et al., 2016a; Swindles et al., 2016; McCarroll et al., 2017). The lack of spatial replication means that the results cannot be generalised across the wider peatland resource. In short, while paleoecology studies provide valuable insights into historical vegetation change within UK peatlands, any results from such studies should be considered as potential hypotheses to be tested with more robust methods that are better able to ascribe causation (i.e. randomised controlled experiments).

5.2. Fauna

5.2.1. Birds

Quantity and quality of the evidence assessed for this topic: Twelve studies overall – two low-quality studies (-) and ten medium quality studies (+). Six medium quality studies (+) investigated burning impacts (either directly or indirectly) on *P. apricaria* populations.

Is the direction of the evidence consistent? Yes, but only for *P. apricaria.*

If so, what is the direction of the evidence? By promoting areas with shorter and/or more varied vegetation structure across a moorland, managed burning seems to have a positive effect on *P. apricaria* populations within upland peatlands. However, managed burning often coincides with predator control in many upland areas, which means we do not know the relative importance of managed burning in promoting *P. apricaria* populations (but see Littlewood et al., 2019).

5.2.2. Aquatic invertebrates

Quantity and quality of the evidence assessed for this topic: Three low-quality studies (-).

Is the direction of the evidence consistent? No.

If so, what is the direction of the evidence? NA.

5.2.3. Terrestrial invertebrates

Quantity and quality of the evidence assessed for this topic: One high-quality study (++).

Is the direction of the evidence consistent? NA – Cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? NA.

5.2.4. Soil microorganisms

Quantity and quality of the evidence assessed for this topic: Two studies – one low-quality study (-) and one high-quality study (++).

Is the direction of the evidence consistent? No. While both studies show that burning leads to changes in microorganism communities found within peatland soils, each study investigates a different taxon.

If so, what is the direction of the evidence? NA.

5.3. Carbon sequestration and greenhouse gas emissions

5.3.1. Carbon and peat accumulation within upland peatland soil profiles

Quantity and quality of the evidence assessed for this topic: Two studies investigated the impact of managed burning on carbon accumulation within upland peatland soil profiles –

one medium quality study (+) and one high-quality study (++). One high-quality study (++) investigated the impact of managed burning on peat accumulation within upland peatland soil profiles.

Is the direction of the evidence consistent? Yes, for carbon accumulation. NA, for peat accumulation – cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? Both studies suggest that upland peatlands subject to managed burning accumulate, rather than lose, carbon within the peat profile (Heinemeyer et al., 2018; Marrs et al., 2019a). However, it is important to note that the findings of Heinemeyer et al. (2018) and Marrs et al. (2019a) have been debated within the scientific literature (Baird et al., 2019; Evans et al., 2019; Heinemeyer et al., 2019b; Marrs et al., 2019b). Nevertheless, the general finding that flat, fully vegetated and wet upland peatlands (like those studied by Marrs et al., 2019a and Heinemeyer et al., 2018) subject to managed burning accumulate (rather than lose) carbon is supported by previous work (Garnett et al., 2000). Furthermore, the work of Marrs et al. (2019a) suggests that flat and wet areas of blanket bog under longer rotations (e.g. 20 years) accumulate peat and carbon at a similar rate to areas that have remained unburnt for between ~60 to 90 years.

There is a potential caveat that should be considered when interpreting the results of the near-surface[24] carbon accumulation assessments of Heinemeyer et al. (2018) and Marrs et al. (2019a). As Heinemeyer et al. (2018) acknowledge, near-surface carbon accumulation assessments often show rapid carbon accumulation due to lower decomposition rates at the peat surface, but the same peat section could be losing carbon from the opposite (bottom) end of the profile (as shown in the modelling study by Young et al., 2019). Therefore, researchers should ideally assess carbon accumulation throughout the entire peat core (*ibid*). Alternatively, when only near-surface peat core sections are used, researchers should consider site conditions when interpreting their findings (*ibid*). For example, sites affected by deep drainage ditches or that have become very dry for other reasons, are likely to be losing carbon from lower down the peat profile (*ibid*). In such scenarios, one should not relate near-surface carbon accumulation rates to the rest of the peat body (*ibid*). However, any such carbon losses should be indicated by a sharp decline in organic carbon content, which neither Heinemeyer et al. (2018) or Marrs et al. (2019a) observed. Furthermore, near-surface carbon

[24] Near-surface means near the top of the peat profile.

accumulation data taken from wet peatland sites (and with no indication for deep C loss) *can* be generalised to the entire peat body because such places are unlikely to be losing carbon from the deeper peat layers. Consequently, the flat, fully vegetated and wet upland peatland areas studied by Heinemeyer et al. (2018) and Marrs et al. (2019a) are unlikely to be losing considerable amounts carbon from the base of the peat profile. However, future work must verify such an assertion.

5.3.2. Upland peatland carbon fluxes

Quantity and quality of the evidence assessed for this topic: Seven studies – one low-quality study (-), two medium quality studies (+) and four high-quality studies (++).

Is the direction of the evidence consistent? No.

If so, what is the direction of the evidence? NA

5.3.3. Upland peatland methane fluxes

Quantity and quality of the evidence assessed for this topic: Five studies – one medium quality study (+) and four high-quality studies (++).

Is the direction of the evidence consistent? No.

If so, what is the direction of the evidence? NA.

5.3.4. Upland peatland dissolved organic carbon fluxes

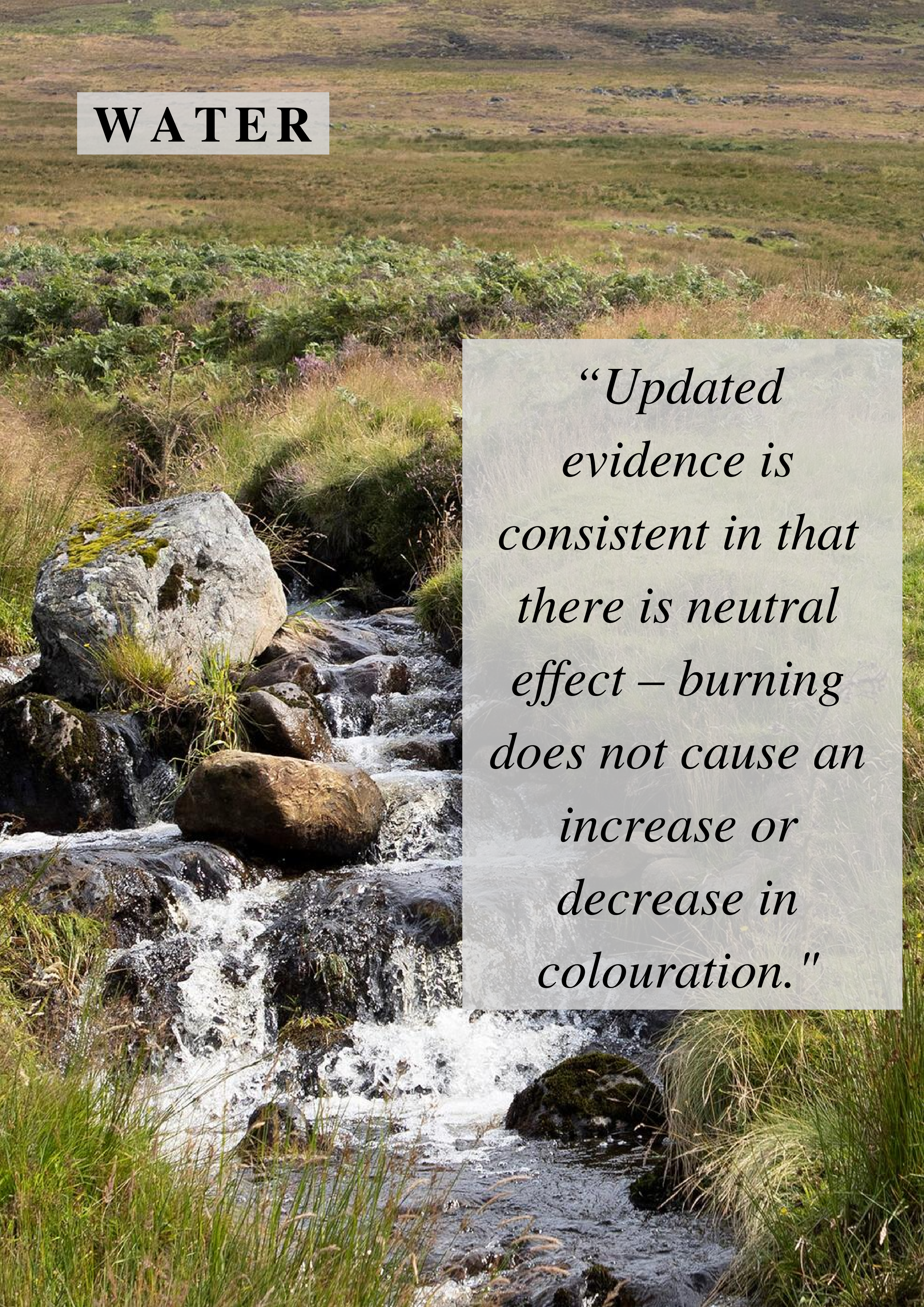
Quantity and quality of the evidence assessed for this topic: Five studies – one low-quality study, two medium-quality studies (+) and two high-quality studies (++).

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Four out of the five studies suggest that managed burning has no impact on dissolved organic carbon fluxes in upland peatlands (either directly or indirectly via changes to vegetation composition).

5.3.5. Charcoal production

WATER



"Updated evidence is consistent in that there is neutral effect – burning does not cause an increase or decrease in colouration."

Quantity and quality of the evidence assessed for this topic: Two studies – one low-quality study (-) and one medium quality study (+).

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Both studies suggest that by adding charcoal to the peat profile, managed burning may lead to long-term carbon storage benefits.

5.3.6. Upland peatland greenhouse gas budgets

Quantity and quality of the evidence assessed for this topic: One low-quality study (-) and one high-quality study (++).

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Both burnt and unburnt plots are net sources (rather than sinks) of GHG emissions. However, one study (++) suggests that burnt plots are greater sources of GHG during the first four post-management years. In contrast, a second study (-) suggests that the more recently burned areas are smaller sources of GHGs than older burns. A major issue with both studies is the limited study length, which is much less than a complete burn rotation (or, even better, several burning rotations).

5.4. Water quality and flow

5.4.1. Water quality

Quantity and quality of the evidence assessed for this topic: Three studies examined the impact of managed burning on water colour (measured at different scales and locations across studies) – one low-quality study (-), one medium quality study (+) and one high-quality study (++). Three studies measured the impact of burning on water pH (within either soil water or stream water) – two low-quality studies (-) and one high-quality study (++).

Is the direction of the evidence consistent? Yes, for water colour. No, for pH.

If so, what is the direction of the evidence? Managed burning has no impact on water colour.

5.4.2. Water flow

Quantity and quality of the evidence assessed for this topic: Two low-quality (-) studies investigated the impact of managed burning on overland flow. Two low-quality (-) studies investigated the impact of managed burning on streamflow Three studies investigated the impact of managed burning on water table depth - two low-quality studies (-) and one high-quality study (++).

Is the direction of the evidence consistent? Yes, for overland flow. No, for streamflow. No, for water table depth.

If so, what is the direction of the evidence? Managed burning leads to an increase in overland flow on upland peatlands. However, these findings are from two low-quality studies with serious methodological flaws (e.g. pseudoreplication and/or significant confounding as shown by Ashby and Heinemeyer, 2019a; Ashby and Heinemeyer, 2019b). Also, it is not evident that the increases in overland flow mediated by managed burning lead to increased flood risk. For example, the impact of managed burning on streamflow is unclear (Johnston, 2012; Holden et al., 2015).

5.5. Fire ecology

5.5.1. Burn severity

Quantity and quality of the evidence assessed for this topic: Seven studies investigated the impact of burn severity on peatland ecosystem services – two low-quality studies (-), three medium quality studies (+) and two high-quality studies (++). Specifically, three studies investigated the impact of burn severity on *S. capillifolium* damage – one medium quality study (+) and two high-quality studies (++).

Is the direction of the evidence consistent? Yes, for the relationship between burn severity and *S. capillifolium* damage.

If so, what is the direction of the evidence? There is a positive relationship between burn severity and *S. capillifolium* damage, with lower burn severities causing minimal damage to *S. capillifolium* plants relative to unburnt controls. Nevertheless, *S. capillifolium* plants are still able to recover after experiencing high severity burns (within the space of three years).

5.5.2. Burn frequency

Quantity and quality of the evidence assessed for this topic: Four studies examined the relationship between burn frequency and *C. vulgaris* abundance – three high-quality studies (++) and one medium quality study (+). Two high-quality studies (++) examined the relationship between burn frequency and *Eriophorum* abundance. Three high-quality studies (++) examined the relationship between burn frequency and *Sphagnum* (mainly *S. capillifolium*) abundance.

Is the direction of the evidence consistent? Yes, for *C. vulgaris, Eriophorum* and *Sphagnum* (mainly *S. capillifolium*) abundance.

If so, what is the direction of the evidence? There is a negative relationship between burning frequency and *C. vulgaris* abundance (adult plants and propagules), and a positive relationship between burning frequency and *Eriophorum* abundance. Also, frequently burnt plots (burnt every ten and 20 years) can support similar amounts of *Sphagnum* (mainly *S. capillifolium*) than plots left unburnt for 60-90 years. It is important to note that these findings come from a single experiment: The Hard Hill experimental plots.

5.6. Wildfire

5.6.1. Fuel loads

Quantity and quality of the evidence assessed for this topic: Three high-quality (++) studies.

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Frequent managed burning significantly reduces fuel loads on upland peatlands. However, two of the three studies measuring fuel loads collected data from a single experimental site in the Northern Pennines: The Hard Hill experimental plots. Nevertheless, several additional studies have shown that the cessation in burning management also leads to significant increases in the percentage cover of dwarf shrubs (mainly *C. vulgaris*) on upland peatlands (Lee et al., 2013a; Milligan et al., 2018; Whitehead and Baines, 2018; Grau-Andrés et al., 2019a; Noble et al., 2019b); and, percentage cover is closely correlated with vegetation biomass (Muukkonen et al., 2006; Axmanová et al., 2012).

5.7. Burning extent

5.7.1. The current extent of managed burning

Quantity and quality of the evidence assessed for this topic: Three medium quality (+) studies.

Is the direction of the evidence consistent? No – burning extent is variable across studies.

If so, what is the direction of the evidence? NA

5.7.2. Temporal changes to the extent of managed burning

Quantity and quality of the evidence assessed for this topic: Two medium quality studies (+).

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Both studies suggest that burning extent has increased in recent decades (at least up to 2009/2010). However, one study was from a single moorland site, and the second study did not validate the method used to calculate burning extent and only assessed 2% of the English uplands. Moreover, both studies are out of date because their last sampling point was over ten years ago (Thacker et al., 2014; Allen et al., 2016) – burning extent may have changed since then.

5.7.3. Managed burning return intervals

Quantity and quality of the evidence assessed for this topic: Two medium quality studies (+).

Is the direction of the evidence consistent? No – burning return intervals are variable across studies and sites. This probably reflects differences in *Calluna vulgaris* growth rates across sites with different environmental conditions (e.g. Santana et al., 2015).

If so, what is the direction of the evidence? NA

5.7.4. The frequency of managed burning

Quantity and quality of the evidence assessed for this topic: Two medium quality studies (+)

Is the direction of the evidence consistent? Yes.

If so, what is the direction of the evidence? Both studies suggest that the number of burns has increased between 1988-2009 (Allen et al., 2016) and 2001-2010 (Douglas et al., 2015). However, one study was from a single moorland site, and both studies are out of date because their last sampling point was over ten years ago (Douglas et al., 2015; Allen et al., 2016) – the trend in burning frequency may have changed since then. Also, the burning frequencies recorded by Allen et al. (2016) were highly variable across sampling periods.

5.7.5. The size of management burning patches

Quantity and quality of the evidence assessed for this topic: One medium quality study (+)

Is the direction of the evidence consistent? NA – Cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? NA

5.8. Soils

5.8.1. Post-fire soil temperatures

Quantity and quality of the evidence assessed for this topic: One high-quality study (++), two medium quality studies (+) and one low-quality study (-).

Is the direction of the evidence consistent across studies? Yes

If so, what is the direction of the evidence? When considering the impact across all seasons examined within each study[25], managed burning seems to have a neutral impact on post-fire soil temperatures.

5.8.2. Soil compaction

Quantity and quality of the evidence assessed for this topic: Three high-quality studies (++) and one low-quality study (-).

Is the direction of the evidence consistent? No.

[25] For example, Grau-Andrés et al. (2019b) found no differences in post-fire soil temperatures during two out of the three seasons investigated.

If so, what is the direction of the evidence? NA.

5.8.3. Soil moisture

Quantity and quality of the evidence assessed for this topic: Two high-quality studies (++) and one medium quality study (+).

Is the direction of the evidence consistent? No.

If so, what is the direction of the evidence? NA.

5.8.4. Soil chemistry

Quantity and quality of the evidence assessed for this topic: Two low-quality studies (-).

Is the direction of the evidence consistent? No. While the two studies show that burning leads to changes in the chemical properties within peatland soils, each study investigates a different range of chemical properties.

If consistent, what is the direction of the evidence? NA.

5.8.5. Upland peatland soil erosion

Quantity and quality of the evidence assessed for this topic: One low-quality study (-).

Is the direction of the evidence consistent? NA – cannot assess evidence consistency using a single study.

If so, what is the direction of the evidence? NA.

6. Research recommendations

Before providing research recommendations for each review sub-question, the list below outlines a series of generic research recommendations that are informed by the critical appraisal of the evidence included in this review. With this in mind, future studies investigating the impact of managed burning on upland peatland ecosystems should consider:

- Randomly allocating treatment or survey plots.
- Including an unburnt or not recently burnt control.
- Using an experimental, rather than correlative, study design.
- Collecting data from multiple peatland sites, with each site containing treatment replicates to avoid the confounding of burning management with study site (and other environmental variables).
- Collecting data from across more than one burning rotation and for at least three different years within a burning rotation.
- Collecting baseline data.
- Examining the effect of managed burning at both the plot and catchment scale.
- Investigating different burn rotation lengths and burn severities.

The above research recommendations provide a framework to investigate burning impacts on upland peatlands using a robust[26] and real-world[27] approach that is largely absent within the current evidence base. Indeed, just one of the studies included in this review has adopted such an approach (Heinemeyer et al., 2019c), but this project has only been running for ten years and does not compare burning to an unburnt control at the catchment scale (catchment-scale comparisons are made between burning and mowing). The remaining studies in this review

[26] An experimental approach that allows you to ascribe causation, e.g., a randomised controlled before-and-after trial.
[27] One which examines burning in the same way it is applied by upland land managers, e.g., every year, multiple patches of varying size (but usually ~2500 m^2) are burnt on rotation across an extensive area of moorland using rotations that are suited to the local environmental (i.e. growing) conditions.

generally measured burning impacts using a plot-scale approach in which burning impacts were measured for only a short period post-burn (<4 years). As a result, the evidence base largely provides data about the short-term impacts of managed burning on upland peatland ecosystem services.

6.1. Flora

6.1.1. Vegetation diversity

The number and geographical distribution of studies investigating burning impacts on peatland vegetation diversity are still limited. For example, studies are mostly conducted in northern England, with 40% of UK studies using data from the Hard Hill experimental plots. Therefore, future studies should be conducted across a wider geographical area.

6.1.2. Vegetation structure

The impact of managed burning on peatland surface microtopography was investigated using a short-term approach whereby measurements were taken for no more than a couple of years at the start of a burning rotation (Noble et al., 2018a; Heinemeyer et al., 2019a; Noble et al., 2019a; Noble et al., 2019b). Therefore, future studies should document how peatland surface microtopography changes across the entire burning rotation relative to unburnt control areas. Alongside this, we also need to know how changes to peatland surface microtopography influence important ecological parameters such as carbon and peat accumulation, flood prevention, water quality improvement and wildfire mitigation.

It is self-evident that burning initially reduces the height of the vegetation canopy and that canopy height recovers as time since burning increases. Nevertheless, there is very little research into the wider implications of this relationship. For example, by removing the shade and competition caused by a dense *Calluna vulgaris* canopy, managed burning may provide more conducive conditions for *Sphagnum* growth (e.g. Gunnarsson et al., 2002; Benscoter and Vitt, 2008). Furthermore, by reducing fuel loads, managed burning may also play a role in wildfire prevention and mitigation (Santana et al., 2015; Santana et al., 2016; Santana and Marrs, 2016). Both aspects require urgent research attention.

6.1.3. Sphagnum species

Most of the studies included here and within Glaves et al. (2013) focus on *S. capillifolium*, which is probably because other *Sphagnum* species are less frequent within many upland peatland sites. Consequently, even if other *Sphagnum* species are recorded, there is usually insufficient data to carry out robust statistical analyses. Nevertheless, we still need to know how managed burning effects the full range of *Sphagnum* species found within upland peatlands across the UK. Therefore, future studies should attempt to address this important research gap.

Another issue is that ~40% of the *Sphagnum* studies use data collected from the Hard Hill experimental plots, which suggests future studies should try to collect data from different sites to reduce the geographical bias of the evidence base. Finally, none of the studies investigating the effect of fire damage on *Sphagnum* plants collected data for more than three years. This represents a significant research gap that will inform us of whether fire-induced heat damage of *Sphagnum* plants leads to long-term ecological consequences (this seems unlikely given that burning does not reduce the abundance of *Sphagnum* spp.).

Before we can consider the wider implications of managed burning impacts on *Sphagnum* spp., we also need robust experimental data on the ecological functions of *Sphagnum* within upland peatlands. For example, experimental evidence about its contribution to peat and carbon accumulation, water storage capacity and flood mitigation, water quality, and wildfire prevention and mitigation. Indeed, it has long been stated within the literature that *Sphagnum* species have a positive effect on peat and carbon accumulation, but this is based on circumstantial (rather than causal) evidence from paleoecology studies (see Shepherd et al., 2013; Gillingham et al., 2016 and references therein).

6.1.4. Eriophorum species

Before we can consider the wider implications of burning impacts on *Eriophorum* species, we need robust experimental data on the ecological functions that *Eriophorum* species provide within upland peatlands. For example, the contribution they make to peat and carbon accumulation, methane emissions, water storage capacity and flood mitigation, water quality improvements, and wildfire prevention and mitigation. Again, as with *Sphagnum*, it has long been stated within the literature that *Eriophorum* species have a positive effect on peat and carbon accumulation, but this is based on circumstantial (rather than causal) evidence from peat record (see Shepherd et al., 2013; Gillingham et al., 2016 and references therein).

6.1.5. Calluna vulgaris

Before we can consider the wider implications of managed burning impacts on *C. vulgaris* abundance, we need robust experimental data on the ecological functions of *C. vulgaris* within upland peatlands. For example, its contribution to peat and carbon accumulation, methane emissions, water storage capacity and flood mitigation, water quality, and wildfire prevention and mitigation. The role of *C. vulgaris* abundance in wildfire prevention and mitigation is a particularly urgent research priority given its flammability (Davies and Legg, 2011; Santana and Marrs, 2016) and the predicted rise in moorland wildfires due to warmer and drier summers (Albertson et al., 2009; Albertson et al., 2010).

6.1.6. Bare ground

Only two studies included in this review examined the temporal changes to bare ground after a managed burn has been applied (Heinemeyer et al., 2019c; Noble et al., 2019b). Future studies should address this research gap. What would be particularly useful would be information on how long it takes for the small post-burn patches of bare ground to revegetate, and how this varies with changes in climate, water table depth, peat depth, fire severity, fire frequency, burn rotation length and vegetation community. Also, and as mentioned for other aspects of peatland vegetation, before we can consider the wider implications of these findings, we need robust experimental data on how the small-scale and transient creation of bare ground affects ecological functions within upland peatlands (e.g. peat and carbon accumulation, methane emissions, water storage capacity and flood mitigation, water quality improvements, wildfire prevention and mitigation, and biodiversity). We should also consider that the creation of small patches of bare ground may provide benefits, such as providing micro-habitats for invertebrates (Cameron and Leather, 2012).

6.1.7. Paleoecology studies

Before any valid conclusions or lessons can be drawn from paleoecology studies on upland peatlands, we really need a greater number of studies which: i) are multi-site and analyse numerous well-distributed peat cores per site; ii) statistically test the effect between fire occurrence (i.e. the presence of charcoal macrofossils) and vegetation change throughout the peat profile; and, iii) examine the effect of other explanatory variables (e.g. climate, drainage, grazing) on historical vegetation change within upland peatlands. Ideally, any findings that emerge from paleoecology studies should also be confirmed using experimental approaches.

6.2. Fauna

6.2.1. Birds

P. apricaria prefers shorter areas of vegetation in which to breed (Whittingham et al., 2000; Whittingham et al., 2002). Thus, managed burning could be used to promote *P. apricaria* breeding habitat on upland peatland (Whitehead and Baines, 2018). But the same result could be achieved by mowing. Therefore, one research priority would be to establish whether managed burning or mowing best promotes *P. apricaria* breeding habitat. Such studies should also consider the wider environmental impacts (e.g. on water quality, GHG emissions, flood mitigation) and practicalities of both vegetation management techniques (e.g. getting equipment to inaccessible areas).

Future studies should also: i) use more experimental approaches in order to better establish if any relationships exist between managed burning and the abundance of certain bird species on upland peatlands; ii) determine how managed burning influences bird populations on upland peatland (e.g. by changes to habitat structure, food resources or predation exposure); iii) examine the wider implications of burning induced changes to bird populations within upland peatlands (e.g. the effect on upland peatland food webs); and, iv) attempted to separate the impact of burning from other aspects of grouse moor management, such as predator control.

6.2.2. Aquatic invertebrates

To accurately establish the effect of managed burning on aquatic invertebrate communities within upland peatlands, we need a greater number of high-quality or very high-quality studies (see Table 2 and 3). A priority should be to use study designs that do not confound site with burning management (burning versus no burning) so that the effect of managed burning can be isolated from other environmental or management variables. Once we understand how managed burning influences aquatic invertebrate communities, we can assess the wider implications of any findings that emerge. For example, how burning induced changes influence within stream invertebrate-mediated ecosystem services or peatland food webs.

6.2.3. Soil microorganisms

Only two studies investigated the impact of managed burning on peatland soil microorganisms. Thus, much more research is required to clarify the effect of burning on the different microorganisms living within upland peatland soils. Once this is established, we can examine the wider implications of any research findings that emerge. For example, whether the taxa promoted or inhibited by managed burning promote or inhibit different peatland ecosystem services (e.g. carbon storage and water quality).

6.3. Carbon sequestration and greenhouse gas emissions

6.3.1. Carbon and peat accumulation within upland peatland soil profiles

We need a greater number of high-quality and multi-site studies that measure the impact of managed burning on upland peatland carbon and peat accumulation. Any future studies should attempt to examine carbon and peat accumulation throughout the entire peat profile (i.e. by using full-length cores) using multiple peat cores that are well distributed across each study site or treatment plot. Furthermore, calculations of carbon and peat accumulation should take account of detailed soil bulk density and carbon content assessments (sensu Heinemeyer et al., 2018).

6.3.2. Upland peatland carbon fluxes

The contradictory results across studies suggest that more work is required to establish the relationship between managed burning and upland peatland carbon fluxes. The work of Walker et al. (2016) and Ward et al. (2013) indicates that vegetation composition may be a key driver of upland peatland carbon fluxes. Thus, future work should investigate how changes to upland peatland vegetation composition mediated by managed burning influences ecosystem carbon fluxes. Moreover, none of the carbon flux studies included in this review took measurements across a complete burning rotation – this research gap clearly needs to be addressed.

6.3.3. Upland peatland methane fluxes

The contradictory results across studies suggest that additional research is required to establish the relationship between managed burning and upland peatland methane fluxes. Given that most of the methane flux studies included in this review took measurements for three years or less (i.e. they are short-term assessments), future studies should attempt to capture methane fluxes over at least an entire burning rotation (see Harper et al., 2018). Furthermore, a recent meta-analysis of 87 studies covering 186 sites suggests that peatland methane emissions are primarily driven by water table depth, vegetation composition, pH and temperature (Abdalla et al., 2016). Consequently, future studies should also measure these covariates to see how they interact with burning management to influence upland peatland methane fluxes.

6.3.4. Upland peatland dissolved organic carbon fluxes

The contradictory results suggest that more research is required to fully understand any causal links between burning and DOC fluxes from upland peatlands. Any future studies should try to move away from plot scale measurements and calculate the impact of burning management on DOC fluxes at the catchment scale.

6.3.5. Charcoal production

Given the low number of studies and the debate surrounding the results of Heinemeyer et al. (2018) (Evans et al., 2019; Heinemeyer et al., 2019b), we clearly need more data on the contribution of charcoal to upland peatland carbon budgets. Also, to address some of the criticisms of Heinemeyer et al. (2018) (see Evans et al., 2019; Heinemeyer et al., 2019b) and improve our knowledge of how charcoal influences upland peatland carbon budgets, future studies must: i) include an unburnt control[28]); ii) use a greater number of peat cores spread across a wider area within each study site or plot (Heinemeyer et al., 2018 used three cores per site that were each within a five-metre radius); and, iii) use complete peat core sections to address the criticisms of Young et al. (2019) outlined above (or provide evidence that no deep carbon losses have occurred, e.g. peat profile data on constant or increasing carbon content).

6.3.6. Upland peatland greenhouse gas budgets

We clearly need a greater number of high-quality studies that assess the impact of managed burning on GHG budgets within upland peatlands. Future studies should attempt to measure (rather than estimate or model) each individual pathway that contributes to upland peatland GHG budgets, including the contribution of charcoal (Worrall et al., 2013a; Heinemeyer et al., 2018; Leifeld et al., 2018). Such assessments should also be carried out over the entire burning rotation.

6.4. Water quality and flow

6.4.1. Water quality

We need a much greater number of higher-quality studies that measure burning impacts on water quality directly. This would enable us to accurately detect any causal links that exist between managed burning and different water quality metrics. To achieve this aim, future studies must examine burning impacts on peatland water quality across multiple sites, at the

[28] Heinemeyer et al. (2018) explored relationships between different peat property variables. Therefore, a control was not required and the work by Heinemeyer et al. (2018) should be considered as the first step in exploring a causal relationship between burning frequency, charcoal concentrations in the peat profile and carbon accumulation. The next step would be to carry out a more robust study that would enable causal links to be established.

catchment scale, and, to allow comparisons between studies, use similar methodologies and measure the same water quality metrics. We also need to know the wider ecological and societal implications of any changes to water quality metrics that are mediated by burning management (i.e. does it matter that burning leads to a small decrease in stream water pH?) and whether any potential damage could be mitigated by improving burning practice (burning away from watercourses) or habitat manipulation (e.g. gill planting).

6.4.2. Water flow

We clearly need a much greater number of high-quality studies that measure burning impacts on overland flow and streamflow within upland peatlands. Once we have established robust causal links between managed burning and peatland hydrology, we can investigate the wider implications and whether any potential damage could be mitigated by improving burning practice (burning away from watercourses) or habitat manipulation downstream (e.g. coarse woody debris)?

6.5. Fire ecology

6.5.1. Burn severity

We need many more studies that investigate the effect of burn severity on a wider range of environmental parameters within upland peatlands. Once we have this information, we will be able to manipulate the temperatures of managed burns (by using local environmental conditions – such as peat and vegetation moisture) so that they do not exceed the threshold temperature over which multiple ecosystem services are adversely affected. Finally, to get an accurate and complete picture, future studies should assess the impact of different burn severities across entire burning rotations.

6.5.2. Burn frequency

We desperately need studies that investigate how burning frequency affects a wider range of environmental parameters on upland peatland (rather than just vegetation composition). Such studies should also try to reduce the geographic bias within the evidence base (e.g. all but one study used the Hard Hill experimental plots).

6.6. Wildfire

6.6.1. Fuel loads

WILDFIRE

"Even the latest data on burning extent and frequency is ten years out of date and may have now changed with extensive wildfires, some very severe, having occurred in the last three years."

We clearly need more studies that measure the impact of burning on fuel loads within upland peatlands. These studies should ideally be from a wide range of peatland sites across the British uplands to reduce the current geographical bias within the evidence base (e.g. fuel load studies are predominantly restricted to data collected from the Hard Hill plots). More importantly, we need to know whether reductions in fuel loads mediated by managed burning reduces wildfire risk and damage. For example, an increase in fuel loads on upland peatlands is likely to increase the severity of any wildfires that take hold (Davies et al., 2010b; Davies et al., 2016a). Crucially, high severity wildfires could potentially be extremely damaging to the moss, litter and soil layers within upland peatlands (Davies et al., 2010b; Grau-Andrés et al., 2017; Grau-Andrés et al., 2018; Taylor et al., 2018; Grau-Andrés et al., 2019a; Noble et al., 2019a).

Alternatively, peatland rewetting (e.g. by gulley and ditch blocking), combined with the cessation of vegetation management (e.g. managed burning or cutting) and the planting of *Sphagnum* spp., has been put forward as a better and less damaging way of reducing wildfire risk on upland peatlands (Baird et al., 2019). Proponents of the rewetting hypothesis state that: "*Naturally wet and rewetted peatlands do not experience deep burning because a suite of ecohydrological processes and bog moss traits maintain a surface with a high moisture content, and thereby increase the energy required to ignite peat and restrict the burn depth if fires do occur*" (*ibid*).

It is certainly possible that wetter peatlands *could* reduce the chances of the moss and peat layers igniting or limit the spread of a fire if the moss and peat did ignite. For example, a group of British studies show that the soil and moss layer within (wet) blanket bog ecosystems are generally buffered from the effects of a managed burn, whereas the soil and moss layer within (drier) heathland ecosystems is not (Grau-Andrés et al., 2017; Grau-Andrés et al., 2018; Grau-Andrés et al., 2019a; Grau-Andrés et al., 2019b). But these studies were testing the effect of a management burn (*ibid*). Such burns are carried out in winter during saturated soil conditions, which means they are likely to be significantly cooler than wildfires (especially at the soil surface) (Davies et al., 2010a; Davies et al., 2010b; Davies et al., 2016b). Furthermore, upland peatlands of the UK are largely heather dominated even across areas with more 'natural' (i.e. high) water tables[29] (Lee et al., 2013a; Alday et al., 2015; Milligan et al., 2018; Marrs et al., 2019b). Consequently, rewetted upland peatlands with

[29] The peatland underlying the Hard Hill plots has water tables like that of 'natural' peatland (see Marrs et al., 2019b and references therein). Note also, that summer water tables within 'natural' peatlands can drop to well below (~34 cm) the surface (*ibid*). Thus, the soil surface and moss layers within rewetted peatlands may still be dry, and thereby easily ignitable, during the summer months.

unmanaged vegetation are likely to have high fuel loads, which would lead to higher fire temperatures if a wildfire does manage to ignite (Hobbs and Gimingham, 1984; Davies et al., 2010b; Davies et al., 2016a; Noble et al., 2019a). Furthermore, ignition is certainly possible in summer when bog vegetation becomes very dry, especially during the prolonged dry spells that are becoming more frequent. For example, heather moisture content only has to drop below 60% for it to become flammable (Davies and Legg, 2011). The key question is: would the temperatures during summer wildfires be high enough to ignite the peat within rewetted areas of upland peatland? In truth, we do not know. Thus, the wildfire mitigation potential of rewetting and managed burning both require urgent research attention

6.7. Burning extent

We need more studies at the moorland, regional and national scale that use validated (i.e. ground-truthed) methodologies which include more recent (2015 up to 2020) measurements of burning extent, frequency, return intervals and patch sizes. Once we have accurate estimates of these parameters, we can assess the wider implications of any findings that emerge. However, this will also require a complete understanding of burning impacts, as well as the relationship between burning extent and peatland ecosystem services at the moorland, regional and national scale.

6.8. Soils

6.8.1. Post-fire soil temperatures

Due to the contradictory results reported between studies, more research is required to fully establish the effect of burning on post-fire peatland soil temperatures. Moreover, we have no idea whether small and seasonal differences in mean post-fire soil temperatures are ecologically relevant, that is, do small post-burn increases in mean soil temperatures ($<1^{o}$C) significantly reduce peatland ecosystem functioning.

6.8.2. Soil compaction, moisture and chemistry

Due to the inconsistent results across studies, more research is required to clarify the effect of burning on peatland soil compaction, moisture and chemistry and how these different soil parameters influence the provision of peatland ecosystem services.

6.8.3. Upland peatland soil erosion

More research is required to clarify the effect of burning on peatland soil erosion. Future studies should avoid using erosion pins and instead try to develop more robust methods of measuring peatland soil erosion (perhaps at the catchment scale so that fluvial export of peat can be measured).

6.9. Notes for policymakers, land managers and peatland researchers

Policymakers and land managers require robust and conclusive evidence to underpin decision-making (Sutherland et al., 2004; Pullin and Knight, 2009; Dicks et al., 2014a; Dicks et al., 2014b). However, it would be unwise to use the results of this review to develop clear policy and land management advice because of the considerable uncertainty within the evidence base. Thus, moving forward, peatland researchers must work together to fill research gaps and develop an objective approach for summarising a highly heterogeneous evidence base. Hopefully, the data collated during this review will provide the foundations for achieving the latter objective. Indeed, collating and categorising the complete managed burning evidence base should be an urgent research priority. Another priority moving forward is to develop a series of standardised protocols for measuring managed burning impacts on peatland ecosystem services. This would enable researchers to assess the impact of managed burning using objective approaches, such as meta-analysis.

7. Evidence summary table

Table 9 below summarises the findings of this review and notes whether they are consistent with Glaves et al. (2013). A detailed evidence summary table key is provided in Table 10.

Table 9. Evidence summaries for each of the outcome measure investigated within this review. A description of the data contained in each column is given in Table 10 below.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Flora	Vegetation diversity	No	NA	NA	Positive: 2++, 2+, 1- Negative: 0 Neutral: 1++, 1-	NA	Yes
	Sphagnum diversity	No	NA	NA	Positive: 1- Negative: 0 Neutral: 0	NA	No
	Surface microtopography	No	NA	NA	Positive: 0 Negative: 1++, 1+ Neutral: 1++, 1-	NA	No
	Canopy height	Yes	Negative	This is a short to medium-term impact which is reversed once the vegetation canopy has regrown after ~10-20 years. Thus, frequent burning would have a negative impact on canopy height, but for longer rotations, the impact would be Neutral.	Positive: 0 Negative: 2++, 2+, 2- Neutral: 1+	Very weak	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Flora	*Sphagnum* abundance (principally *Sphagnum capillifolium*)	Yes	Neutral	When considering different rotation lengths or times since burning, burning seems to have a neutral impact on *Sphagnum* abundance relative to non-intervention.	Positive: 0 Negative: 2- Neutral: 3++, 2+, 3-	Weak	No
	Sphagnum capillifolium damage	Yes	Positive	Burning causes temperature-induced damage to *S. capillifolium.* However, two out of the studies suggest that *S. capillifolium* plants recover within under three years.	Positive: 2++, 1+ Negative: 0 Neutral: 0	Very weak	Not assessed by Glaves et al. (2013)
	Sphagnum propagules in surface peat	No	NA	NA	Positive: 0 Negative: 1+ Neutral: 0	NA	Not assessed by Glaves et al. (2013)
	*Eriophorum a*bundance	No	NA	NA	Positive: 3++, 1- Negative: 1++, 1- Neutral: 1++, 2+, 2-	NA	No
	Calluna vulgaris abundance	Yes	Negative	This is a short-term impact. Heather becomes more abundant and eventually dominant with increasing time since burn. Thus, *C. vulgaris* abundance is lowest on recently and/or frequently burnt areas, and highest on unmanaged areas.	Positive: 1- Negative: 6++, 4+, 3- Neutral: 0	Moderate	Yes
	*Calluna vulgaris g*ermination	No	NA	NA	Positive: 1++ Negative: 0 Neutral: 0	NA	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Flora	*Calluna vulgaris* propagules in litter and surface peat	No	NA	NA	Positive: 0 Negative: 1+ Neutral: 0	NA	Not assessed by Glaves et al. (2013)
	Amount of bare ground	Yes	Positive	Burning leads to the small-scale increase in bare ground, but this seems to be a transient effect (lasting four to ten years).	Positive: 1++, 3+, 1- Negative: 0 Neutral: 1-	Very weak	Yes
	Sphagnum historical abundance	Yes	Negative	Circumstantial evidence from peat cores suggests that episodes of fire (denoted by charcoal macrofossils) are coincident with a decline in *Sphagnum* macrofossils.	Positive: 0 Negative: 1+, 7- Neutral: 1-	Very weak	Not assessed by Glaves et al. (2013)
	Eriophorum historical abundance	No	NA	NA	Positive: 1+, 3- Negative: 1- Neutral: 1-	NA	Not assessed by Glaves et al. (2013)
	Calluna vulgaris historical abundance	No	NA	NA	Positive: 5- Negative: 1- Neutral: 2+, 1-	NA	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Fauna	*Pluvialis apricaria* populations	Yes	Positive	Note: managed burning often coincides with predator control, which means it is hard to determine the relative contribution of managed burning in promoting *P. apricaria* populations	Positive: 6+ Negative: 0 Neutral: 0	Very weak	Yes
	Cranefly emergence	No	NA	NA	Positive: 0 Negative: 0 Neutral: 1++	NA	Not assessed by Glaves et al. (2013)
	Aquatic invertebrate diversity	No	NA	NA	Positive: 0 Negative: 2- Neutral: 1-	NA	No
	Abundance of pollution tolerant aquatic invertebrates	No	NA	NA	Positive: 2- Negative: 0 Neutral: 1-	NA	No
	Abundance of pollution intolerant aquatic invertebrates	No	NA	NA	Positive: 0 Negative: 2- Neutral: 1-	NA	No
	Soil microorganisms	No	NA	NA	Studies: 1++, 1-	NA	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Carbon sequestration and GHG emissions	Carbon accumulation	Yes	Neutral	Both studies suggest that burnt areas of blanket bog accumulate (rather than lose) carbon within the peat profile. One study suggests that carbon accumulation rates on blanket bog subject to longer burning rotations (~20 years) appear broadly the same as those recorded in unburnt or not recently burnt areas.	Positive: 0 Negative: 0 Neutral: 1++, 1+	Very weak	No
	Peat accumulation	No	NA	NA	Positive: 0 Negative: 0 Neutral: 1++	NA	No
	Carbon fluxes	No	NA	NA	Positive: 1++, 1+ Negative: 1+ Neutral: 4++, 1-	NA	No
	Methane fluxes	No	NA	NA	Positive: 1++, 1+ Negative: 0 Neutral: 3++	NA	Not assessed by Glaves et al. (2013)
	Dissolved organic carbon fluxes	Yes	Neutral	NA	Positive: 1++ Negative: 0 Neutral: 1++, 2+, 1-	Low	No
	Influence of charcoal on carbon storage	Yes	Positive	The production of charcoal during managed burning and its subsequent incorporation into the peat profile may have positive impacts on long-term carbon storage.	Positive: 1+, 1- Negative: 0 Neutral: 0	Very weak	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Carbon sequestration and GHG emissions	Greenhouse gas budgets	No	NA	Note: in relation to the direction of evidence, "Positive" means increased GHG emissions and "Negative" means reduced GHG emissions relative to unburnt or not recently burnt controls.	Positive: 1++ Negative: 1- Neutral: 0	NA	Yes
Water quality and flow	Water colour	Yes	Neutral	Note: two of the three studies measured water colour at the plot scale (in soil pore water or overland flow), whereas the third study measured water colour at the catchment scale (within stream water).	Positive: 0 Negative: 0 Neutral: 1++, 1+, 1-	Very weak	No
	pH	No	NA	NA	Positive: 0 Negative: 1- Neutral: 1++, 1-	NA	No
	Water table depth	No	NA	Note: in relation to the direction of evidence, "Positive" means higher water tables and "Negative" means lower water tables relative to unburnt or not recently burnt controls.	Positive: 1- Negative: 1++, 1- Neutral: 0	NA	No
	Overland flow	Yes	Positive	NA	Positive: 2- Negative: 0 Neutral: 0	Very weak	Yes
	Streamflow	No	NA	NA	Positive: 1- Negative: 0 Neutral: 1-	NA	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Fire ecology	Fire severity and *Sphagnum capillifolium* damage	Yes	Positive	There was a positive relationship between burn severity and *S. capillifolium* damage, with lower burn severities causing minimal damage to *S. capillifolium* plants relative to unburnt controls. Nevertheless, *S. capillifolium* plants are still able to recover after experiencing high severity burns.	Positive: 2++, 1+ Negative: 0 Neutral: 0	Very weak	Not assessed by Glaves et al. (2013)
	Burn frequency and *Calluna vulgaris* abundance	Yes	Negative	Note: the evidence is exclusively from the Hard Hill experimental plots.	Positive: 0 Negative: 3+, 1+ Neutral: 0	Very weak	No
	Burn frequency and *Eriophorum* abundance	Yes	Positive	Note: the evidence is exclusively from the Hard Hill experimental plots.	Positive: 2++ Negative: 0 Neutral: 0	Very weak	No
	Burn frequency and *Sphagnum* abundance (mainly *S. capillifolium*)	Yes	Neutral	Note: the evidence is exclusively from the Hard Hill experimental plots.	Positive: 0 Negative: 0 Neutral: 3++	Weak	Not assessed by Glaves et al. (2013)
	Carbon accumulation	NA	NA	NA	Positive: 1+ Negative: 1++ Neutral: 0	NA	Not assessed by Glaves et al. (2013)
	Peat accumulation	NA	NA	NA	Positive: 0 Negative: 1++ Neutral: 0	NA	Not assessed by Glaves et al. (2013)

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Wildfire	Fuel loads	Yes	Negative	NA	Positive: 0 Negative: 3++ Neutral: 0	Weak	Yes
Burning extent and frequency	Current extent	NA	NA	NA	Studies: 3+	NA	No
	Temporal changes in extent	Yes	Positive	Note: one study was from a single moorland site, and the second study did not validate the method used to calculate burning extent and only assessed 2% of the English uplands. Moreover, both studies are out of date because their last sampling point was over ten years ago – burning extent may have changed since then.	Positive: 2+ Negative: 0 Neutral: 0	Very weak	No
	Burn return intervals	No	NA	NA	Studies: 2+	NA	No
	Temporal changes in frequency	Yes	Positive	Note: one study was from a single moorland site, and the second study did not validate the method used to calculate burning extent and only assessed 2% of the English uplands. Moreover, both studies are out of date because their last sampling point was over ten years ago – the trend in burning frequency may have changed since then. Also, in one study, burning frequencies were highly variable between years.	Positive: 2+ Negative: 0 Neutral: 0	Very weak	No

Table 9. Continued.

Sub-question	Outcome measure	Is the evidence consistent?	Direction of evidence	Further info	Evidence profile	Strength of evidence	Consistent with Glaves et al. (2013)?
Burning extent and frequency	Burn patch size	No	NA	NA	Studies: 1+	NA	Not assessed by Glaves et al. (2013)
Soils	Post-fire soil temperatures	Yes	Neutral	NA	Positive: 1- Negative: 0 Neutral: 1++, 2+	Weak	Not assessed by Glaves et al. (2013)
	Soil compaction	No	NA	NA	Positive: 2++ Negative: 0 Neutral: 1+, 1-	NA	No
	Soil moisture	No	NA	NA	Positive: 0 Negative: 2++ Neutral: 1+	NA	Not assessed by Glaves et al. (2013)
	Soil chemistry (various metrics)	No	NA	NA	Studies: 2-	NA	No
	Soil erosion	No	NA	NA	Positive: 1- Negative: 0 Neutral: 0	NA	No

Table 10. A descriptive key to the evidence summary table (Table 9)

Sub-question	The review sub-question to which the evidence applies.
Outcome measure	The outcome measure being assessed.
Is the evidence consistent?	Evidence was only classified as consistent if ≥75% of the studies for a given outcome variable reported similar results (positive, negative, or neutral).
Direction of evidence	If the evidence is consistent, does it indicate burning has a positive, negative, or neutral impact on the selected outcome variable? Note that "positive" and "negative" are not value judgements (i.e. better or worse) – *they relate to the direction of evidence*.
Further info	Clarificatory information about the evidence for the selected outcome measure.
Evidence profile	The number and quality of studies reporting a positive, negative, or neutral effect of managed burning. Note that "positive" and "negative" are not value judgements (i.e. better or worse) – *they relate to the direction of evidence*.
Strength of evidence	Strong, Moderate, Weak or Very weak.
Consistent with Glaves et al. (2013)?	Are the findings for the selected outcome variable consistent with the findings of Glaves et al. (2013)? Yes, No or Not assessed by Glaves et al. (2013).

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Appendix A: Peer review comments

Below are the peer review comments provided by Dr Gavin B. Stewart (Newcastle University) on the original draft of this review (Italicised text). Also shown are the responses of Dr Mark A. Ashby (Blue text).

Overall comments

Ashby and colleagues review burning management post Glaves to provide a summary of the most recent evidence. The review scope is problematic in evidence synthesis terms- with questions based on an extant narrative review where specific questions are not always fully defined and are very broad particularly with respect to outcomes. Nonetheless the author's provide a review which has elements of systematic review including use of inclusion criteria, search strategy, and critical appraisal. A protocol was not utilised but this is not yet universal in the field of environmental review. I would judge that despite this, the acquisition of evidence was demonstrably unbiased, repeatable and of sufficient sensitivity to draw valid conclusions. A repeatable critical appraisal was included which provided a useful dichotomy of evidence based on causation despite some problems with spatio-temporal relationships which are often hard to address in this domain. An evidence synthesis was undertaken addressing a series of a priori questions. Narrative synthesis was used and is justified by review scope and high heterogeneity amongst studies in terms of methods, taxon and outcomes. The narrative synthesis and evidence summaries provide transparent but not repeatable statements regarding evidence alongside components of a very useful database of study characteristics. Extending this database with explicit information about study outcomes, judgements about direction, size and precision of effects, and habitat would allow more nuanced judgements about strength of evidence. Deficiencies in the evidence-base are recognised in implications for research but not fully articulated in the synthesis or evidence summaries. There are therefore considerable uncertainties in the statements about the effects of burning management in this review, as there are in primary studies and other reviews on the topic. The transparency with which the review has been undertaken and provision of data provide important foundational steps for the development of a more robust evidence-base. The potential for open science to provide a collaborative mechanism to develop evidence-informed policy in this contested environmental space should be recognised and embraced by researchers, policy makers and stakeholders alike.

Specific comments on the introduction, methods and results, general comments on the evidence synthesis and implications for research

L12 paragraph preceding: I understand the brief is to review post Glaves literature- But the science objective of interest in a policy context is to understand if the new evidence changes the conclusions one would draw from the cumulative evidence-base. Being more explicit about the controversy and the policy context might be helpful? Reference to any tender briefs would improve transparency.

I have been more explicit about why I was contracted by the Moorland Association, and I have also been more explicit about the controversy and policy context. Furthermore, while I have mentioned that "the science objective of interest in a policy context is to understand if the new evidence changes the conclusions one would draw from the cumulative evidence-base", I have outlined that this can only be done by reviewing the entire evidence base. However, after consultation with Natural England, they suggested we do not go over old ground (i.e. the evidence in Glaves et al., 2013), but instead review the evidence that has emerged since 2012.

L18 inclusion criteria: The inclusion criteria are very broad- with some specific questions defining outcomes more precisely replicating the questions posed by the glaves review. However, more details of both population and intervention are needed to derive repeatable inclusion criteria (are there habitat, taxonomic or geographical restrictions on inclusion, is accidental burning relevant, how is experimental burning treated, is post burn recovery relevant). I note that greater detail is provided later in the review but this would be better consolidated and defined here. Greater definition of outcome measures and defining primary and secondary outcomes would reduce the probability of selective reporting and HARKING. Such definitions would be mandatory in a medical setting, ideally specified a priori.

I have moved the inclusion criteria to section 1.1.

L62 This is a commendable objective. Providing a transparent and accessible data-base of relevant studies is very valuable contribution to moving the debate on the pros and cons of prescribed burning forward; more so as this includes the oft neglected critical appraisal necessary to inform overall strength of evidence assessments. So called "living reviews" are emerging in some domains, allowing for continuous updating of important evidence-bases as knowledge is accrued. The database provided here and the Glaves review could form the precursor of such a living review to inform upland land management decisions across GB.

I have added an additional sentence alluding to the potential of both reviews providing the basis of a ‘living review’ on the impacts of managed burning on upland peatlands in the UK.

L70 Short explicit paragraph on differences and rationale

I have changed this short passage to: “This review attempted to use a similar methodology to Glaves et al. (2013) but, due to several reasons (e.g. logistics), this could not always be achieved. Significant departures from the Glaves et al. (2013) methodology are highlighted throughout the subsequent sections”. Thus, I will describe significant differences in methodology throughout the methods section.

L143 Short explanation of how much or how little redundancy there was between articles from the search and articles from included reviews would help define the sensitivity of the search (put this in the results) The detail is provided in the supplementary material which is gold standard in terms of search inclusion transparency, but the salient details need reporting.

I have inserted this information in Table 4.

Table 2 – superfluous?? move to appendix?

I have moved this Table to the appendices.

L173- these inclusion criteria need to be specified earlier (see comment re L18)

I have moved the inclusion criteria to section 1.1.

L233 [General comment] Notwithstanding specific criticisms regarding outcome definition, the acquisition of evidence for the review appears sound and conforms with bias minimisation strategies employed in contemporaneous meta-analyses and reviews in the environmental field. Information specialists would no doubt advocate use of a broader more sensitive search and multiple reviewers but experience of reviews in this field and emerging evidence from rapid reviews suggest the bias associated with less exhaustive searches and single reviewers is minimal.

L251- Use of subjective domain based assessment for study appraisal (sensu glaves) is a standard approach to considering risk of bias in many evidence synthesis contexts. The score *based system you have utilised is more repeatable than the overall domain based-judgement but no less subjective. Suggest rephrasing to make this more apparent.*

I have rephrased to highlight that both approaches are subjective and, therefore, open to criticism. However, the approach used in this review is clearer and more repeatable.

L253- I really like the idea of identifying gold standard studies and think this is very useful. However- I don't think that there is a single optimal spatio-temporal scale (landscape level management manipulations). Rather, this is linked to outcomes. e.g. changes in sphagnum abundance may be optimally measured at a patch scale especially if considering single species; whereas bird population abundance is likely to be more usefully measured at a larger scale? Identifying the spatio-temporal thresholds of gold standard studies would be a great suggestion for further work- but here represents a study characteristic rather than a quality component per se. Conversely, consideration of causation is a critical and oft-neglected element of assessing the strength of evidence. Ascertaining how (or if) only considering evidence with strong causal claims, changes the evidence-base should be a key focus of this review/and/or future work. There is a general consensus in clinical medicine that policy should be informed by a single (or few) studies with robust inference (despite generalisability concerns) rather than a larger number of studies where causation cannot be attributed (garbage in, garbage out). The trade-off here is that studies with strong causal claims (randomised, replicated studies) are difficult to implement especially at the larger spatio-temporal scales required to capture management effects or ecosystem process.

I have replaced "Gold standard" with "Very high quality". I have also made this category only obtainable by passing all 16 of the critical appraisal questions.

Tables 3&4. The questions posed are unambiguous and helpful in characterising study methodology. The value judgements underpinning assessment of risk of bias in Table 4 are fully transparent and have a strong rationale grounded in ability to ascribe causation. However, studies that are very informative but at high risk of bias will be described as low quality. e.g. well conducted paleoecology based on peatcores. Rephrasing in terms of risk of bias and nuanced interpretation with discussion of how study methods impact conclusions accompanying the bifurcation of data into causality classes might be useful modifications.

I have made it clear that the quality rating is primarily an assessment of a studies ability to ascribe causation. I have also highlighted that, whilst being designated as low quality, paleoecology studies are extremely informative. However, due to the diminished capability of such studies to ascribe causation, they should be considered as the basis for further investigation (e.g. via experimentation).

L307. Consider addition of PRISMA diagram (mandatory in medical domains, increasingly prevalent in environmental fields).

Table 4 now serves the same function as a PRISMA diagram because it has been modified to include the number of articles obtained during each search method, the number of duplicates removed, the number of articles accepted at each stage of screening and the number of accepted articles obtained using each search method.

L318. Succinct and discriminatory study description. It might be worth defining the design terminology and relating to risk of bias? Maybe an additional column or two in table 8?

I have not added extra columns, but I have changed the table title to this: "The number of accepted studies by type of study. In general, experimental studies (i.e. controlled trials) have the lowest risk of bias (Hurlbert, 1984; Smokorowski and Randall, 2017). The randomisation of treatments and collection of baseline data (i.e. a before-and-after study) further reduces bias (ibid)".

Evidence synthesis

L1144 [Evidence synthesis narrative] The evidence synthesis is well structured considering study findings in relation to the original review questions stratified by risk of bias. However, the value judgements, diverse study designs and heterogeneous outcomes measured by the studies make statements about the overall evidence problematic. Arguably, these problems are common to all narrative reviews but the breadth of evidence considered and the contested policy context exacerbate the issue. In my view the five problems with the approach should be elucidated and the uncertainty arising from them acknowledged.

1) *The value judgements about the evidence are transparent and the arguments underpinning them are clear, but there is a multiverse of alternative arguments and value judgements. It is problematic to consider studies too diverse to be formally combined in a meta-analysis as consistent in effect especially where precision and effect magnitude for individual studies aren't ascertained. Such evidence statements should be defined explicitly as highly uncertain or unpacked further in my view.*

   I have discussed the uncertainty of the evidence statements at the beginning of section 5.

2) *There are frequently differences between the author's conclusions and the data presented in primary studies. This can arise because authors selectively report or emphasise some results rather than others, choose particular end-points or methods of analysis. The current review relies too heavily on the author's interpretation of the data rather than utilising studies simply as a means to acquire data for synthesis. This can be illustrated with the hard hill data for Sphagnum response to burning [based on ecn dataset]. Despite representing the best evidence available on sphagnum response to burning it is possible to present and interpret the data from this monitoring in multiple ways. Six species of sphagnum are recorded in the data-set but despite use of an objective outcome measure rather than subjective cover and high intensity monitoring, data is too sparse for five species to draw any conclusions at all regarding changing species composition or diversity. (see figure 1). Note that this point is made in the review research recommendations but not in the evidence synthesis or summary. There is a large volume of data on Sphagnum capillifolium, and it is therefore possible to make many different claims about the effects of burning on this species depending on choice of endpoint and comparison. However, the variance in the data make it abundantly clear, that predicting differences in treatments with certainty is impossible (figure 2). One definition of reliable evidence,- is that a hypothetical future study would almost certainly not change your conclusions. This is clearly not the case here and contrasts with the conclusion in the evidence summary 5.13 that 12 consistent studies allow inference. This example illustrates the problem of relying on authors interpretations of evidence.*

   I have addressed this by reporting study findings rather than author conclusions.

3) *A related problem to relying on the author's interpretation is that there is no consideration of effect magnitude or precision when combining information across studies. One large precise study with a negative effect could outweigh any number of smaller positive studies, despite appearing to be an outlier when summing across studies numerically. This problem of "vote counting" is well known but is very difficult to address unless meta-analysis can be undertaken.*
4) *The fact that ecological studies frequently measure different things even when exploring the same construct has frustrated those involved in synthesis for some time (cf http://www.comet-initiative.org/Studies/Details/1278). Combining surrogate*

outcomes with directly relevant measures across studies adds hugely to uncertainty and this requires more acknowledgement and unpicking.

5) *Habitat type (specifically deep v shallow peats) is a major potential reason for heterogeneity in effect with important policy implications. This is considered for some outcomes but requires standardised reporting and treatment across all the questions addressed by the review. If it is unclear what habitat or habitats were investigated in a specific study this should be specified.*

   All but two of the studies were conducted on deep peat (either exclusively or areas of deep peat areas constituted part of the area from which data were collected). However, I have noted the habitat type of each study within the "Supplementary Database 3.xlxs".

Fully addressing all of these problems is clearly resource intensive, beyond the scope of narrative review and requires changes to the way we fund and undertake primary research as well as our interpretation and synthesis methods. Nonetheless, there are a range of options for mitigating some of the uncertainty engendered. The simplest option would be to include discussion of these problems and add caveats regarding uncertainty to the textual statements in the synthesis and evidence summaries. More usefully, the supplementary material detailing study inclusion, characteristics and methodology could be combined and extended to include study outcomes (as stated by the author) and habitat type. This could be the basis of a relational database, but even simple pivot tables could be used to provide standardised templates to underpin both the textual synthesis and evidence summaries. Including categorical variables defining the information content (study effect, direction, precision) and directness of outcomes would provide a means of attempting to address the vote counting and surrogate outcome problems. However, some analysts would likely dispute the value of the endeavour given the high subjectivity.

Research recommendations

The research recommendations appear thoughtful and sensible, but it is not clear how they relate to uncertainties in the evidence base. Directly and explicitly linking the need for research to uncertainty related to i) poor causation/confounding ii) imprecision iii) inconsistency iv) indirectness and v) potential bias would be useful. These form elements of *the GRADE framework widely utilised in medicine and applied increasingly to environmental contexts. Some form of prioritisation would also be useful given that this review can usefully highlight the deficiencies in the current evidence-base and has a potentially useful role in shaping the future research agenda.*

Implications for policy

There were no implications for policy

Whilst uncertainty remains very high due to deficiencies in the evidence-base, implications for policy should be discussed. These might be more easily discernible, following further development of a database if this is pursued.

I have provided a brief discussion about policy implications within section 6.9.

Supplementary material

Supplementary material regarding the search and study inclusion is extensive and high quality

Improved formatting of supplementary material, provision of .txt and .csv files would be desirable

I have improved supplementary material formatting. However, I have only supplied supplementary material as .xlxs files because, due to the use of multiple tabs, .txt and .csv files are not appropriate.

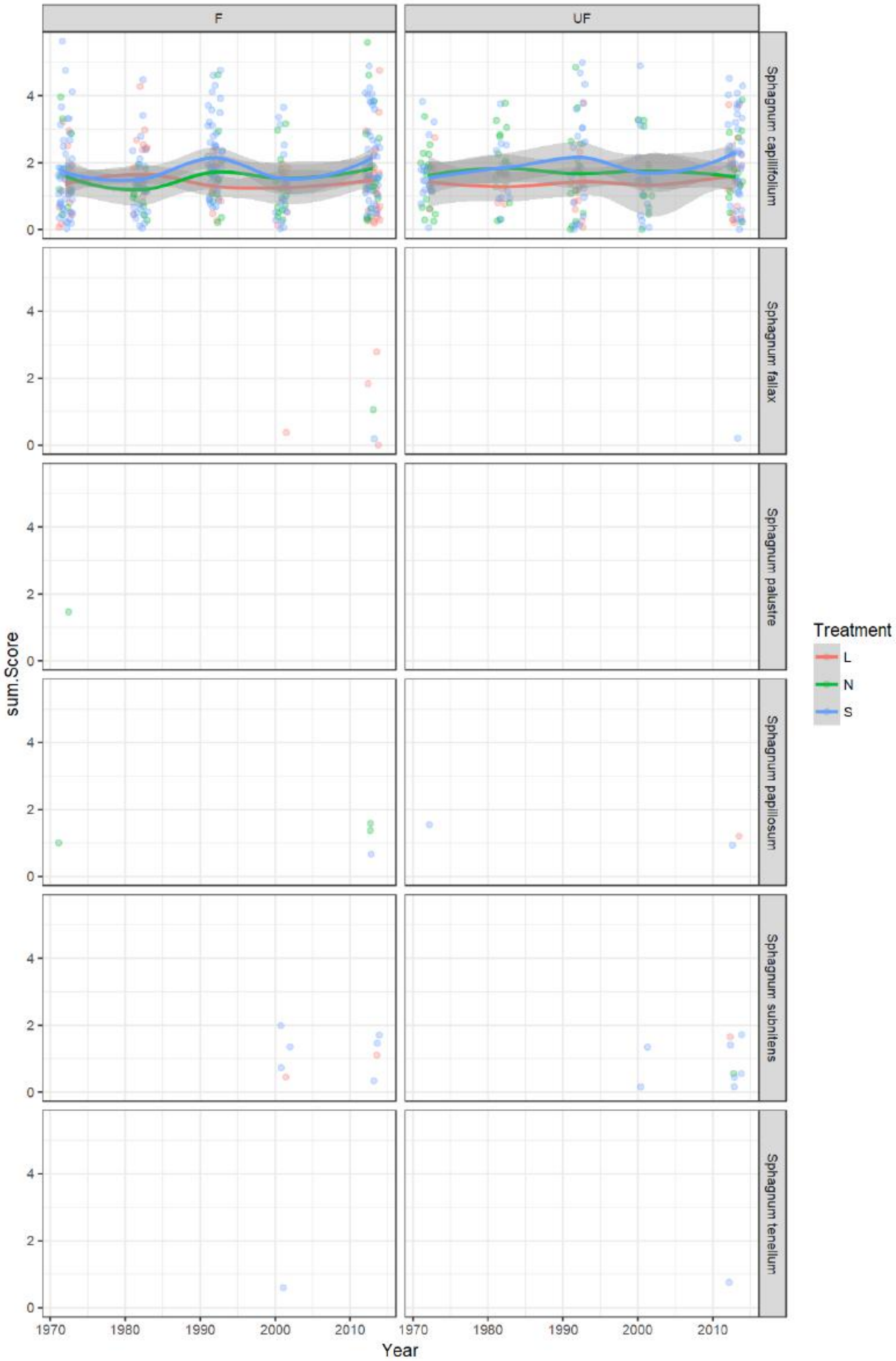
I advocate for development of a database to inform the narrative synthesis and evidence summaries in the review.

I have stated this explicitly within the review objectives in section 1.1.

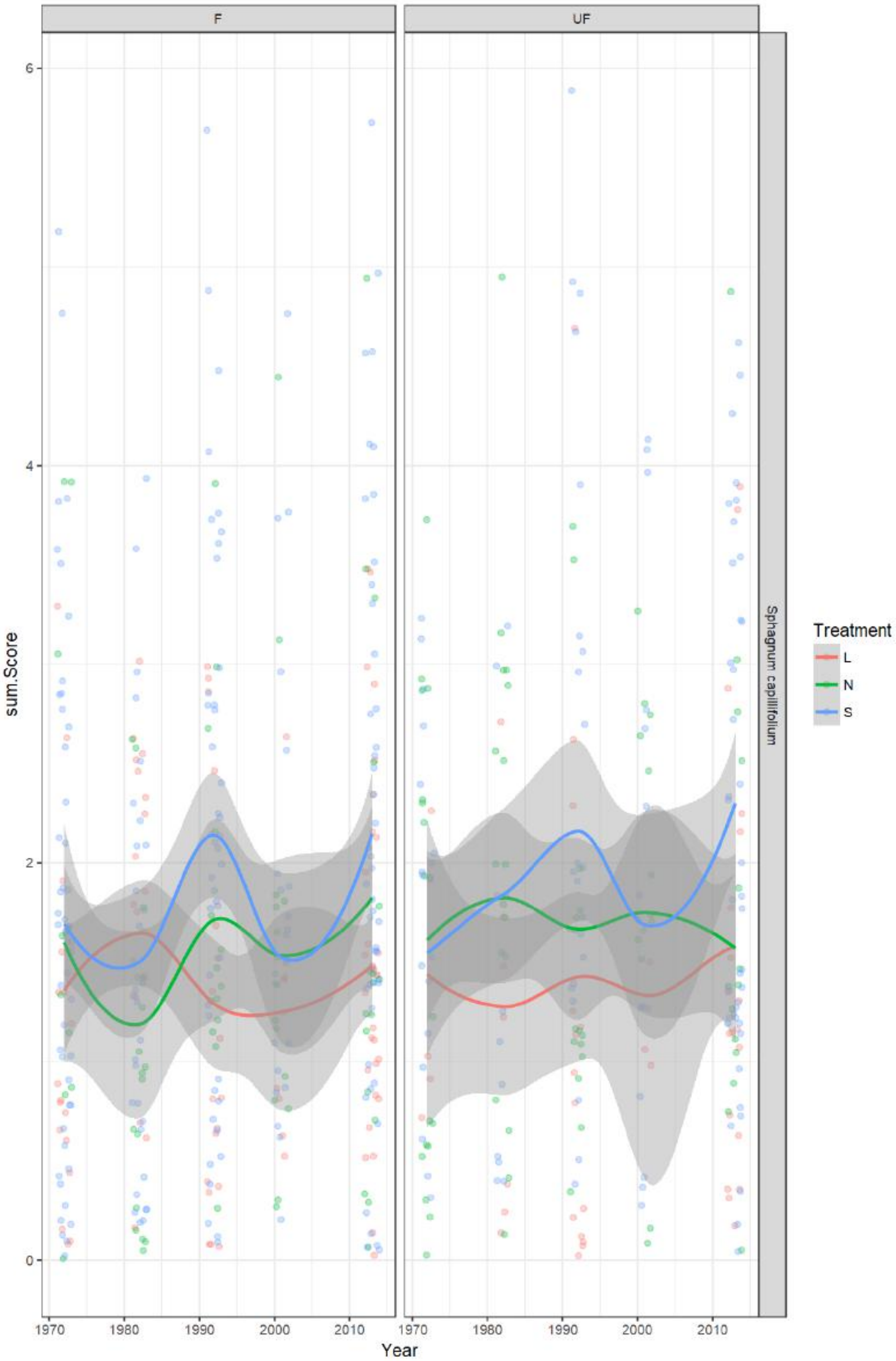
***Bias statement**. I have attempted to provide a full and fair appraisal of the evidence synthesis undertaken by the review team. I have focused on evidence synthesis methodology not the details of ecology, geology or hydrology. I have previously worked (and continue to work) not only on evidence synthesis but also upland management. Funders include NERC, Natural England, RSPB, and the Moorland Association (who have paid for this review). I have personal friends and collaborators who have divergent views on upland policy. I remain committed to the principles of open science and robust evidence synthesis including critical* *appraisal, and believe that a consensus on how to manage British uplands can be found and implemented based upon sound science.*

Gavin B Stewart

April 2020



The figure above graphs *Sphagnum* (various species) pin-frame data from the Hard Hill experimental plots. N = unburnt since 1954, L = burnt every 20 years, S = burnt every 10 years. F = fenced, UF = unfenced.



The figure above graphs *Sphagnum capillifolium* pin-frame data from the Hard Hill experimental plots. N = unburnt since 1954, L = burnt every 20 years, S = burnt every 10 years. F = fenced, UF = unfenced.

Appendix B: Relevant articles not included in this review

The articles in the table below were not included within this review because they are not primary empirical investigations or, if they were, they did not meet all the review inclusion criteria. Nevertheless, they are included here because they are relevant to answering or interpreting the overarching review question and sub-questions.

Reference	**Reference type**
ALONSO, I., WESTON, K., GREGG, R. & MORECROFT, M. 2012. Carbon storage by habitat: Review of the evidence of the impacts of management decisions and condition of carbon stores and sources. Natural England Research Report NERR043. Peterborough, UK: Natural England.	Literature Review
ANDERSEN, R., CHAPMAN, S. J. & ARTZ, R. R. E. 2013. Microbial communities in natural and disturbed peatlands: A review. Soil Biology & Biochemistry, 57, 979-994.	Literature Review
ASHBY, M. A. & HEINEMEYER, A. 2019. Prescribed burning impacts on ecosystem services in the British uplands: A methodological critique of the EMBER project. Journal of Applied Ecology, 00, 1-9.	Comment Paper
ASHBY, M. A. & HEINEMEYER, A. 2019. Whither Scientific Debate? A Rebuttal of “contextualising UK Moorland Burning Studies: Geographical Versus Potential Sponsorship-bias Effects on Research Conclusions” by Brown and Holden (biorxiv 2019; 731117). EcoEvoRxiv, October 31.	Comment Paper
BAIRD, A. J., EVANS, C. D., MILLS, R., MORRIS, P. J., PAGE, S. E., PEACOCK, M., REED, M., ROBROEK, B. J. M., STONEMAN, R., SWINDLES, G. T., THOM, T., WADDINGTON, J. M. & YOUNG, D. M. 2019. Validity of managing peatlands with fire. Nature Geoscience, 12, 884-885.	Comment Paper
BIXBY, R. J., COOPER, S. D., GRESSWELL, R. E., BROWN, L. E., DAHM, C. N. & DWIRE, K. A. 2015. Fire effects on aquatic ecosystems: an assessment of the current state of the science. Freshwater Science, 34, 1340-1350.	Literature Review
BROWN, L. E. & HOLDEN, J. 2019. Contextualising UK moorland burning studies: geographical versus potential sponsorship-bias effects on research conclusions. bioRxiv, 731117.	Comment Paper
BROWN, L. E., HOLDEN, J. & PALMER, S. M. 2016. Moorland vegetation burning debates should avoid contextomy and anachronism: a comment on Davies et al. (2016). Philosophical Transactions of the Royal Society B-Biological Sciences, 371.	Comment Paper

Reference	Reference type
BROWN, L. E., HOLDEN, J., PALMER, S. M., JOHNSTON, K., RAMCHUNDER, S. J. & GRAYSON, R. 2015. Effects of fire on the hydrology, biogeochemistry, and ecology of peatland river systems. Freshwater Science, 34, 1406-1425.	Literature Review
DAVIES, G. M. 2013. Understanding Fire Regimes and the Ecological Effects of Fire. In: BELCHER, C. M. (ed.) Fire phenomena and the Earth system: an interdisciplinary guide to fire science. London, UK: Wiley.	Book Section
DAVIES, G. M., KETTRIDGE, N., STOOF, C. R., GRAY, A., ASCOLI, D., FERNANDES, P. M., MARRS, R., ALLEN, K. A., DOERR, S. H., CLAY, G. D., MCMORROW, J. & VANDVIK, V. 2016. The role of fire in UK peatland and moorland management: the need for informed, unbiased debate. Philosophical Transactions of the Royal Society B-Biological Sciences, 371.	Literature Review
DAVIES, G. M., KETTRIDGE, N., STOOF, C. R., GRAY, A., MARRS, R., ASCOLI, D., FERNANDES, P. M., ALLEN, K. A., DOERR, S. H., CLAY, G. D., MCMORROW, J. & VANDVIK, V. 2016. Informed debate on the use of fire for peatland management means acknowledging the complexity of socio-ecological systems. Nature Conservation-Bulgaria, 59-77.	Comment Paper
DAVIES, G. M., KETTRIDGE, N., STOOF, C. R., GRAY, A., MARRS, R., ASCOLI, D., FERNANDES, P. M., ALLEN, K. A., DOERR, S. H., CLAY, G. D., MCMORROW, J. & VANDVIK, V. 2016. The peatland vegetation burning debate: keep scientific critique in perspective. A response to Brown et al. and Douglas et al. Philosophical Transactions of the Royal Society B-Biological Sciences, 371.	Comment Paper
DAVIES, G. M., STOOF, C. R., KETTRIDGE, N. & GRAY, A. 2016. Comment on: Vegetation burning for game management in the UK uplands is increasing and overlaps spatially with soil carbon and protected areas. Biological Conservation, 195, 293-294.	Comment Paper
DOUGLAS, D. J. T., BUCHANAN, G. M., THOMPSON, P. & WILSON, J. D. 2016. The role of fire in UK upland management: the need for informed challenge to conventional wisdoms: a comment on Davies *et al.* (2016). Philosophical Transactions of the Royal Society B: Biological Sciences, 371, 20160433.	Comment Paper
DOUGLAS, D. J., BUCHANAN, G. M., THOMPSON, P., SMITH, T., COLE, T., AMAR, A., FIELDING, D. A., REDPATH, S. M. & WILSON, J. D. 2016. Reply to comment on: vegetation burning for game management in the UK uplands is increasing and overlaps spatially with soil carbon and protected areas. Biological Conservation, 195, 295-296.	Comment Paper

Reference	Reference type
EVANS, C. D., BAIRD, A. J., GREEN, S. M., PAGE, S. E., PEACOCK, M., REED, M. S., ROSE, N. L., STONEMAN, R., THOM, T. J., YOUNG, D. M. & GARNETT, M. H. 2019. Comment on: "Peatland carbon stocks and burn history: Blanket bog peat core evidence highlights charcoal impacts on peat physical properties and long-term carbon storage," by A. Heinemeyer, Q. Asena, W. L. Burn and A. L. Jones (Geo: Geography and Environment 2018; e00063). Geo-Geography and Environment, 6.	Comment Paper
GILLINGHAM, P., STEWART, J. & BINNEY, H. 2016. The historic peat record: Implications for the restoration of blanket bog, Natural England Evidence Review, Number 011.	Systematic Review
HARPER, A. R., DOERR, S. H., SANTIN, C., FROYD, C. A. & SINNADURAI, P. 2018. Prescribed fire and its impacts on ecosystem services in the UK. Science of the Total Environment, 624, 691-703.	Literature Review
HEINEMEYER, A. & VALLACK, H. W. 2015. Literature review on: potential techniques to address heather dominance and help support 'active' Sphagnum supporting peatland vegetation on blanket peatlands and identify practical management options for experimental testing. York, UK: University of York draft report to Defra and Natural England.	Literature Review
HEINEMEYER, A., BURN, W. L., ASENA, Q., JONES, A. L. & ASHBY, M. A. 2019. Response to: Comment on "Peatland carbon stocks and burn history: Blanket bog peat core evidence highlights charcoal impacts on peat physical properties and long-term carbon storage" by Evans et al. (Geo: Geography and Environment 2019; e00075). Geo-Geography and Environment, 6.	Comment Paper
JONES, L., STEVENS, C., ROWE, E. C., PAYNE, R., CAPORN, S. J. M., EVANS, C. D., FIELD, C. & DALE, S. 2017. Can on-site management mitigate nitrogen deposition impacts in non-wooded habitats? Biological Conservation, 212, 464-475.	Literature Review
MARRS, R. H., MARSLAND, E. L., LINGARD, R., APPLEBY, P. G., PILIPOSYAN, G. T., ROSE, R. J., O'REILLY, J., MILLIGAN, G., ALLEN, K. A., ALDAY, J. G., SANTANA, V., LEE, H., HALSALL, K. & CHIVERRELL, R. C. 2019. Reply to: Validity of managing peatlands with fire. Nature Geoscience, 12, 886-888.	Comment Paper
PARRY, L. E., HOLDEN, J. & CHAPMAN, P. J. 2014. Restoration of blanket peatlands. Journal of Environmental Management, 133, 193-205.	Literature Review
SOTHERTON, N., BAINES, D. & AEBISCHER, N. J. 2017. An alternative view of moorland management for Red Grouse Lagopus lagopus scotica. Ibis, 159, 693-698.	Comment Paper

Reference	Reference type
SWINDLES, G. T., MORRIS, P. J., MULLAN, D. J., PAYNE, R. J., ROLAND, T. P., AMESBURY, M. J., LAMENTOWICZ, M., TURNER, T. E., GALLEGO-SALA, A., SIM, T., BARR, I. D., BLAAUW, M., BLUNDELL, A., CHAMBERS, F. M., CHARMAN, D. J., FEURDEAN, A., GALLOWAY, J. M., GAŁKA, M., GREEN, S. M., KAJUKAŁO, K., KAROFELD, E., KORHOLA, A., LAMENTOWICZ, Ł., LANGDON, P., MARCISZ, K., MAUQUOY, D., MAZEI, Y. A., MCKEOWN, M. M., MITCHELL, E. A. D., NOVENKO, E., PLUNKETT, G., ROE, H. M., SCHONING, K., SILLASOO, Ü., TSYGANOV, A. N., VAN DER LINDEN, M., VÄLIRANTA, M. & WARNER, B. 2019. Widespread drying of European peatlands in recent centuries. Nature Geoscience, 12, 922-928.	Primary Research
THOMPSON, P. S., DOUGLAS, D. J. T., HOCCOM, D. G., KNOTT, J., ROOS, S. & WILSON, J. D. 2016. Environmental impacts of high-output driven shooting of Red Grouse Lagopus lagopus scotica. Ibis, 158, 446-452.	Comment Paper
TURETSKY, M. R., BENSCOTER, B., PAGE, S., REIN, G., VAN DER WERF, G. R. & WATTS, A. 2015. Global vulnerability of peatlands to fire and carbon loss. Nature Geoscience, 8, 11-14.	Literature Review
WERRITTY, A., PAKEMAN, R. J., SHEDDEN, C., SMITH, A. & WILSON, J. D. 2015. A Review of Sustainable Moorland Management. Battleby: Report to the Scientific Advisory Committee of Scottish Natural Heritage.	Literature Review
YALLOP, A. R., CLUTTERBUCK, B. & THACKER, J. I. 2012. Changes in water colour between 1986 and 2006 in the headwaters of the River Nidd, Yorkshire, UK: a critique of methodological approaches and measurement of burning management. Biogeochemistry, 111, 97-103.	Comment Paper
YOUNG, D. M., BAIRD, A. J., CHARMAN, D. J., EVANS, C. D., GALLEGO-SALA, A. V., GILL, P. J., HUGHES, P. D. M., MORRIS, P. J. & SWINDLES, G. T. 2019. Misinterpreting carbon accumulation rates in records from near-surface peat. Scientific Reports, 9, 17939.	Primary Research

Appendix C: Duplicate removal methodology

The eight-step method used to remove duplicate references. This method was taken and modified from Bramer et al. (2016).

Step	EndNote fields	Process of removal
1	Author | Year | Title | Secondary Title (Journal)	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window and press delete to remove all selected duplicates (no manual assessment required).
2	Author | Year | Title | Pages	Same as Step 1.
3	Title | Volume | Pages	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window and: A. Manually assess the top references with a blank title or author fields, using ctrl-left click to deselect false duplicates. B. Click on the column heading ‘‘Pages’’ to sort all duplicate references by descending order of page numbers. C. Review the top references without page numbers and those with page numbers, starting with number 1 for similar author names. If author names of subsequent references differ, deselect the marked false duplicates with ctrl-left click. D. Remove all selected duplicates by pressing delete.
4	Author | Volume | Pages	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window and: A. Repeat stages A-B in Step 3. B. Deselect the top references without page numbers by pressing ctrl-left click on the first highlighted reference and ctrl-shift-left click on the first highlighted reference with a starting page number greater than 1. Remove the remaining selected duplicates by pressing delete.
5	Year | Volume | Issue | Pages	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window and: A. Right-click on ‘My Groups’ > ‘Create Group’ and then press enter. B. In the group ‘Duplicate References’, click on the column heading ‘Pages’ to sort all duplicate references by descending order of page numbers. C. Select all references with page numbers by left-clicking on the top reference while holding shift, and then left-clicking on the last reference with page numbers present. D. Drag the selected references to the just created ‘New Group’ folder. E. Click on ‘New Group’. Then check ‘New Group’ group for references with just one page and page numbers starting with ‘1’ or with a letter. Select false duplicates from those references and then press delete to remove them from the group (They remain in ‘All References’ but are not de-duplicated in this step). F. Select one of the references in the ‘New Group’ folder. Then run the ‘Find Duplicates’ tool, close the ‘Find Duplicates’ window and press delete to remove all selected duplicates (no manual assessment required).
6	Title	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window and: A. Compare page numbers of consecutive references. If page numbers are present and different, examine journal titles and authors. Deselect false duplicates using ctrl-left click. References with blank pages or pages starting with the ‘1’ are usually true duplicates but check journal titles and author names when in doubt, especially when multiple consecutive blank pages are selected. B. After checking the entire list, remove the remaining selected duplicate references by pressing delete.
7	Author | Year	After the ‘Find Duplicates’ tool has been run, close the ‘Find Duplicates’ window. Then, if a true duplicate is found, deselect all references by left-clicking the first true duplicate. Compare subsequent references on page numbers: if two adjacent references have the same page numbers, select the one with the largest record number with ctrl-left click. After checking the complete list, remove the remaining selected references by pressing delete.
8	Not Applicable	Finally, sort all remaining references by title (A-Z) and manually scan for and remove duplicates.

Appendix D: The articles included within this review

Reference	Reference type
ALDAY, J. G., SANTANA, V. M., LEE, H., ALLEN, K. A. & MARRS, R. H. 2015. Above-ground biomass accumulation patterns in moorlands after prescribed burning and low-intensity grazing. Perspectives in Plant Ecology Evolution and Systematics, 17, 388-396.	Journal
ALLEN, K. A., DENELLE, P., RUIZ, F. M. S., SANTANA, V. M. & MARRS, R. H. 2016. Prescribed moorland burning meets good practice guidelines: A monitoring case study using aerial photography in the Peak District, UK. Ecological Indicators, 62, 76-85.	Journal
BLUNDELL, A. & HOLDEN, J. 2015. Using palaeoecology to support blanket peatland management. Ecological Indicators, 49, 110-120.	Journal
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Appendix E: Supplementary materials

Supplementary database 1.xlsx – Evidence search results.

Supplementary database 2.xlsx – Evidence screening results.

Supplementary database 3.xlsx – Coding variable data for each study included in this review.

Supplementary database 4.xlsx – Critical appraisal data for each study included in this review.

APPENDIX S1: HOW WE EXPLORED ENVIRONMENTAL DIFFERENCES BETWEEN EMBER SITES AND TREATMENT PLOTS

The EMBER study design

The EMBER project used five burnt and five unburnt upland river catchments (sites) to investigate the impact of prescribed rotational burning on water quality, hydrology, aquatic biodiversity and soils within blanket bog biotopes (Table S1.1) (Brown, Holden & Palmer 2014). All ten catchments are geographically separate: the mean (± standard error of the mean; SEM) distance between burnt and unburnt catchments equals 76. 7 ± 10.9 km, whereas, the mean (± SEM) distance between all catchments equals 79.1 ± 8.3 km. The five burnt catchments were all managed as grouse moors and contained a mosaic of recent burn patches ranging from <1 to 25 years since burning (*ibid*). The five unburnt catchments had a varied history of prescribed rotational burning: Green Burn, Moss Burn and Trout Beck had not been burnt for more than 60 years; whereas, Crowden Little Beck and Oakner Clough had not been burnt for between 30 and 50 years, respectively (Table S1.1) (*ibid*). The predominant soil type across all catchments was blanket peat (*ibid*).

Table S1.1. The burnt and unburnt catchment sites used during the EMBER project.

Management/site	Location
Burnt catchments:	
Bull Clough	Midhope Moor, Peak District
Rising Clough	Derwent Moors, Peak District
Woo Gill	Nidderdale, Yorkshire Dales
Great Eggleshope beck	Teesdale, North Pennines
Lodgegill Sike	Teesdale, North Pennines
Unburnt catchments:	
Crowden Little Beck	Longendale, South Pennines
Green Burn	Teesdale, North Pennines
Moss Burn	Teesdale, North Pennines
Oakner Clough	Marsden Moor, South Pennines
Trout Beck	Teesdale, North Pennines

Twelve study plots were selected within each catchment (burnt plots n = 60; unburnt plots n = 60). In burnt catchments study plots were equally divided into four burning age classes (three replicates per age class): <2 years since burning (B2), 3-4 years since burning (B4), 5-7 years since burning (B7) and >10 years since burning (B10+) (Brown, Holden & Palmer 2014). One replicate of each burning age class was positioned at the top, middle or bottom of a hillslope (*ibid*). Within the unburnt catchments, the 12 study plots were chosen at random, ensuring that there were four replicates located in top, middle or bottom hillslope positions (*ibid*).

Our comparisons of environmental differences between EMBER study catchments and treatment plots

The EMBER study and its associated peer-reviewed articles use different combinations of study catchments and plots depending on the response variable investigated. These different combinations formed the basis of our comparisons between EMBER study catchments and treatment plots. Specifically, using a range of variables, we compared the environmental conditions between:

1. Streams within burned catchments and streams within unburned catchments (across all 10 EMBER catchments).

2. Burned and unburned plots (across all 10 EMBER catchments).

3. B2, B4, B7, B10+ and unburned plots (across all 10 EMBER catchments).

4. B2, B4 and B15+ plots within the Bull Clough study catchment, unburned plots within the Moss Burn study catchment and wildfire plots within the Oakner Clough study catchment.

5. B2, B4, B7 and B15+ plots within the Bull Clough study catchment and unburned plots within the Oakner Clough study catchment.

The subsequent sections provide additional information about the environmental variables used during all five comparisons. This information includes a brief description of each variable, how each variable was sourced and calculated, and tabular results and descriptions of any statistical analysis we carried out.

Comparing streams within burned catchments and streams within unburned catchments

- This experimental set-up relates to Brown et al. (2013) and Holden et al. (2015)

Table S1.2. The source of each catchment environmental variable and how it was calculated. Data was matched to each catchment by using the location information provided in Table 2.1 in Brown, Holden and Palmer (2014).

Response variable	Data source	Data calculations
Monthly temperature (^{o}C)	UKCP09 Met Office 5 km gridded long-term monthly climate observations from 1981 to 2010	Monthly temperature data for each catchment were extracted using ESRI ArcGIS 10.4 and then averaged across the year. Data available from http://catalogue.ceda.ac.uk/uuid/87f43af9d02e42f483351d79b3d6162a
Monthly rainfall (mm)	UKCP09 Met Office 5 km gridded long-term monthly climate observations from 1981 to 2011	Monthly rainfall data for each catchment were extracted using ESRI ArcGIS 10.4 and then averaged across the year. Data available from http://catalogue.ceda.ac.uk/uuid/87f43af9d02e42f483351d79b3d6162a
Elevation (m)	Table 1 in Holden et al. (2015)	The upper and lower elevation values given in Table 1 for each catchment were averaged
Area (km^2)	Table 1 in Holden et al. (2015)	No calculations required as the area values for each catchment are given in Table 1
NVC community	Table 1 in Holden et al. (2015); Table 1 in Noble et al. (2018)	No calculations required as the NVC values for each catchment are given in both tables
Geology	Table 2.1 in Brown, Holden and Palmer (2014)	No calculations required as the underlying geology for each catchment are given in Table 2.1

Table S1.3. Mean (± SEM) monthly temperature, monthly rainfall, elevation and area values for the five burnt and five unburnt EMBER study catchments. *F* test statistics and p-values for the comparisons of monthly temperature and monthly rainfall between burnt and unburnt catchments are from one-way ANOVA tests. Chi-square test statistics and p-values for the comparisons of elevation and area between burnt and unburnt catchments are from Kruskal-Wallis rank sum tests (as the data failed to meet the parametric assumption of homogeneity of variances). Significant results ($P < 0.05$) are highlighted in bold.

Response variable	Burnt	Unburnt	d.f.	*F*	χ^2	*P*
Monthly temperature (^{o}C)	6.38 ± 0.45	5.96 ± 0.58	1,8	0.33		0.584
Monthly rainfall (mm)	106.87 ± 4.96	132.53 ± 6.42	1,8	10.01		**0.013**
Elevation (m)	505.90 ± 27.68	562.70 ± 67.01	1		0.54	0.465
Area (km^2)	1.26 ± 0.20	1.84 ± 0.47	1		0.54	0.462

Comparing burned and unburned EMBER plots

- This experimental set-up relates to Holden et al. (2015).

- Three additional plots (plot 13, 14 and 15) from Great Eggleshope Beck that were bunt during the EMBER study were omitted from the analysis because the methods section in Holden et al. (2015) suggests that they were not used (*e.g.* "*At all 10 catchments,12 soil plots were selected*").

Table S1.4. The source of each plot environmental variable and how it was calculated. Data was matched to each plot using location information provided in a Microsoft Excel spreadsheet by one of the EMBER authors (J. Holden, pers. comm., September 28, 2018).

Response variable	Source	Data calculations
Elevation (m)	Ordnance Survey Terrain 50 digital elevation model	Elevation data for each plot were extracted using ESRI ArcGIS 10.4. Data available from https://www.ordnancesurvey.co.uk/opendatadownload/products.html
Slope (°)	Ordnance Survey Terrain 50 digital elevation model	Slope data for each plot were extracted using ESRI ArcGIS 10.4. Data available from https://www.ordnancesurvey.co.uk/opendatadownload/products.html
Aspect (°)	Ordnance Survey Terrain 50 digital elevation model	The aspect of each plot was extracted using ESRI ArcGIS 10.4. The aspect of each plot was refined to northerly (N, NE, NW), southerly (S, SE, SW), easterly (E) and westerly (W) aspect categories. Data available from https://www.ordnancesurvey.co.uk/opendatadownload/products.html

Table S1.5. Mean (± SEM) elevation and slope values for the burnt (n = 60) and unburnt (n = 60) EMBER study plots. Chi-square test statistics and p-values are from Kruskal-Wallis rank sum tests (as the data failed to meet the parametric assumption of normality). Significant results ($P < 0.05$) are highlighted in bold. **Data were analysed at the plot rather than site level to match the analysis of Holden et al., 2015 (N.B. This could be considered as pseudoreplication, but we wanted to match the analysis of Holden et al., 2015).**

Response variable	Burnt	Unburnt	d.f.	χ^2	P

Elevation (m)	485.47 ± 8.03	518.37 ± 12.96	1	7.37	**0.007**
Slope (°)	5.93 ± 0.25	6.88 ± 0.42	1	4.72	**0.030**

Comparing the B2, B4, B7, B10+ and unburned EMBER plots

- This experimental set-up relates to Holden et al. (2015).

- The data sources for elevation and slope values are the same as those listed in Table S1.4 above. However, in this analysis, they are averaged across burning age treatments.

- Three additional plots (plot 13, 14 and 15) from Great Eggleshope Beck that were bunt during the EMBER study were omitted from analysis because the methods section in Holden et al. (2015) suggests that they were not used (*e.g.* "*At all 10 catchments,12 soil plots were selected*").

Table S1.6. Mean (± SEM) elevation and slope values for the "B2" = <2 years old (n = 15), "B4" = 3-4 years old (n = 15), "B7" = 5-7 years old (n = 15), "B10+" = >10 years old (n = 15) and "U" = unburnt (n = 60) EMBER study plots. Chi-square test statistics and p-values are from Kruskal-Wallis rank sum tests (the data failed to meet the parametric assumption of normality). **Data were analysed at the plot rather than site level to match the analysis of Holden et al., 2015 (N.B. This could be considered as pseudoreplication, but we wanted to match the analysis of Holden et al., 2015).**

Response variable	B2	B4	B7	B10+	U	d.f.	χ^2	P
Elevation (m)	486.55 ± 16.83	487.04 ± 15.71	483.02 ± 16.33	485.25 ± 17.05	518.37 ± 12.96	4	7.59	0.108
Slope (°)	5.35 ± 0.47	5.69 ± 0.41	6.12 ± 0.57	6.58 ± 0.53	6.88 ± 0.42	4	8.03	0.090

Comparing the B2, B4 and B15+ plots within the Bull Clough study catchment with unburnt plots within the Moss Burn study catchment and wildfire plots within the Oakner Clough study catchment.

- This experimental set-up relates to Holden et al. (2014).

- The data sources for elevation and slope values are the same as those listed in Table S1.4 above.

- The original spreadsheet sent by one of the EMBER authors did not state which three plots were used at the Moss Burn (unburnt) and Oakner Clough (wildfire) catchments (J. Holden, pers. comm., September 28, 2018). Therefore, we included every plot from both sites within our analyses.

Comparing the B2, B4, B7 and B15+ plots within the Bull Clough study catchment with unburned plots within the Oakner Clough study catchment.

- This experimental set-up relates to Brown et al. (2015) and Holden et al. (2015).

- The data sources for elevation and slope values are the same as those listed in Table S1.4 above.

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Report 2:

Peatland Report 2020

A review of the environmental impacts including carbon sequestration, greenhouse gas emissions and wildfire on peatland in England associated with grouse moor management

Game and Wildlife Conservation Trust (GWCT)

Peatland Report 2020

A review of the environmental impacts including carbon sequestration, greenhouse gas emissions and wildfire on peatland in England associated with grouse moor management

Game & Wildlife Conservation Trust

Contents

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Finally we thank the Moorland Association for help with mapping.

Opposite: Red grouse on heather moorland.

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GWCT Peatland Report 2020: A review of the environmental impacts including carbon sequestration, greenhouse gas emissions and wildfire on peatland in England associated with grouse moor management.

Foreword

By Rt Hon Sir James Paice
Chairman of Trustees
Game & Wildlife Conservation Trust

Rt Hon Sir James Paice

A debate is raging over the management of the English uplands. These iconic and beloved landscapes are also ecosystems, food producing farms and wild game shoots. The wild game is red grouse but those grouse moors are just as important homes to some of the highest populations in England of upland waders such as curlew, golden plover, lapwing, snipe, as well as black grouse. These uplands are water catchments for cities such as Birmingham and Manchester and are designated for the quality of their landscape or the abundance of their wildlife. Now they are also part of the climate change debate because of the huge amount of carbon locked up in peat.

Black grouse © David Kjaer.

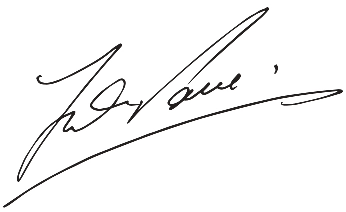
This short description highlights the complexity of management in the uplands, the multifunctionality of its land use; this despite the fact it is some of the least productive land in England.

Both the climate change and biodiversity loss crises highlight afresh the importance of these uplands to the nation, and the responsibility held by policymakers, landowners and land managers to get the management of these special places right.

Land management is not easy – I know, I have been a land manager most of my life. It is, above all, difficult to do well from a distance with blunt policy instruments, I've tried that too. Land management, if it is going to achieve good outcomes, has to be a process of co-creation between the policymakers and the people on the ground. Generalised prescriptions are rarely correct for every circumstance. Recent research is showing that it may not be all as it seems, sometimes the right approach will be counter-intuitive. We need to think very carefully about how we undertake future management in the uplands to ensure we get the best possible outcomes. That means working together to a common purpose.

In the last 10 years we have been rectifying the mistakes of the last big Government-directed land management change in the uplands – draining them to improve livestock productivity. Millions of pounds of taxpayers' money were spent then to achieve that aim, and millions of pounds are being spent now to undo it. Now policy makers and the Climate Change Committee are calling for significant changes to upland management, particularly to vegetation management through burning. Potentially, this represents another huge management change and needs to be handled with great care.

I have seen these issues from both sides – as a farmer/land manager and as a politician. Politicians need to set the direction of travel, then let the land managers work out how to best implement that on their land. The GWCT has a good track record of taking science into practice and finding management solutions that fit with both practical land management and good environmental outcome. This report is intended to help achieve that in the uplands.



Golden plover juvenile on moorland in Yorkshire, England.

Introduction

The aim of this report is to look at carbon management in the English uplands, in particular on areas managed for grouse with an emphasis on vegetation management through burning. We have estimated that grouse moor management covers 423,000ha in total, with 282,000ha above the Defra moorland line. In the English uplands, it is currently essentially one of only three land uses (the others being livestock farming and forestry). The Game & Wildlife Conservation Trust (GWCT) has been researching upland game and wildlife, and the ecology of the uplands since the early 1980s, principally on grouse moors. Historically grouse moor management has acted to conserve heather and other peat-forming plants compared to these alternative land uses and grouse moors are strongholds for upland waders such as curlew, lapwing, and golden plover. All grouse moors are peatland (either dry heath or bog) and the management and restoration of peatlands, which represent a huge carbon store is now attracting considerable policy attention.

Grouse moors have the capacity to contribute significantly to climate change and biodiversity targets in England. In particular, upland wader populations, the restoration of blanket bog and the reduction of carbon emissions. However, the management measures for these outcomes need to be capable of sitting alongside the management measures for the production of grouse which provides the economic and social drivers for the environmental outcomes. Grouse moor management can change to help contribute to climate change targets and we see no reason this should not happen providing the multifunctionality of the land management is acknowledged and the trade-offs between the management outcomes are understood and balanced.

This report has been written because the GWCT is concerned that new Climate Change policies for the management of peatland will need to take more account of the complexities of land management issues, new evidence of how the carbon cycle works on peatland, acknowledge risks such as wildfire, be clear about knowledge gaps and allow individual landowners to develop estate-specific policies. As yet there is insufficient evidence, experience and knowledge to be clear exactly how to create the best possible environmental outcomes for the future alongside the economic and social outcomes of grouse moor management; but we believe there can be a shared desire to achieve that. In this report we attempt to highlight the issues that need to be considered, and some of the pitfalls that need to be avoided, to get to that point.

“Grouse moors have the capacity to contribute significantly to climate change and biodiversity targets in England”

We have liaised closely with experts in peatland ecology working in several UK Universities and experts from the USA and seeks to highlight findings from recently conducted research that hopefully will help Defra as they formulate policies regarding England's peatlands.

Recently, restrictions to manage peatlands by prescribed burning on deep peat have been put in place with the aim of helping to restore deep peat to functioning blanket bog. We support the restoration of blanket bog where this is possible but caution that simple **'no burn policies'** may have unintended negative consequences. This report seeks to set out those concerns and the logic behind them. To do this, we try to clarify the science behind the pros and cons of burning peatland, including carbon budgets and greenhouse gas emissions, risk of wildfire and the potential impacts on biodiversity.

This report anticipates Defra's **'Peatland Strategy'** report which seeks to **'ensure that all peatlands in England meet the needs of wildlife and people'** and **'demonstrates how peatlands can contribute to the UK's target of zero net emissions of greenhouse gases by 2050.'** The GWCT is delighted to contribute to this debate.

AN EXAMPLE OF COMMUNITY CONSERVATION

In the uplands there is a strong element of community conservation. Much of the uplands is isolated and remote, and farming or country sports provide a significant part of employment and economic activity. A policy change that affects someone's ability to manage land for a particular outcome can have serious knock-on consequences for local employment, economic activity and social cohesion. It is a cliché but these are living, working landscapes. Policy solutions need to tick all the sustainability boxes – environment, economic and social – and be practical and appealing to the land manager within his framework of multifunctional management.

25 PEATLAND CATEGORIES – NO ONE-SIZE FITS ALL

There is no formal definition for peatland so estimates of the extent of the peat or its condition will vary depending on the definition used. For example, Natural England uses five types of peatland with 11 types of management. In the most comprehensive inventory of peat yet published, Evans *et al.* (2017) describe 25 peatland conditions categories. The point is that this complexity shows that a **'one size fits all'** approach to managing our peatlands could lead to confusion and be misguided.

PEATLAND TRADE-OFFS

The condition of our peatlands will be strongly related to the land use undertaken on it including arable farming, especially vegetable growing, grassland, growing trees, livestock grazing, extracting peat for commercial reasons and managing for red grouse. Each land use has differing carbon emissions and reducing them will potentially involve trade-offs between carbon storage/emissions, agricultural production, wildlife, conservation and risk of wildfire. We are not aware of research that has identified the relative contributions of all these factors to the condition of our peatlands, but we do know that vegetable growing produces the greatest loss of carbon from our peat. So, do we abandon horticultural production on these grade-1 soils in the Cambridgeshire fenlands? Hence the need to consider trade-offs in the debate.

TYPES OF BURNING

Burning surface vegetation on grouse moors, known as heather burning, is often cited as a contributor to peatland degradation and unwanted carbon emissions. There are two principle types: managed burning also known as prescribed or rotational; and uncontrolled burning or wildfire.

Wildfires, like that on Saddleworth Moor in 2018, are large fires, burning out of control and can cover extensive areas. They result from accidental or deliberate (malicious) ignitions which tend to be in the summer and therefore potentially high risk, or can be a managed burn getting out of control (which will only be in the winter burning season: October to March/April). They can burn at very high temperatures, not only the surface vegetation but also into the underlying peat, possibly down tens of centimetres. Liverpool University (Marrs, pers comm) estimated that Saddleworth wildfire resulted in seven centimetres of peat being lost, and that it will take up to 200 years to restore it (a minimum of 29 years to recreate one cm of surface peat). Wildfires can burn for a long time, smouldering underground and flaring up elsewhere at a later date.

Modern grouse moor managers undertake managed burns on small areas (seldom wider than 30m) of older heather to reduce the heather cover (the surface vegetation) and regenerate the heather to encourage new green shoot growth to feed grouse. These burns are supervised (i.e. a control team very nearby), surrounded by a firebreak, and when operating well move across the surface quickly and so are described as **'cool'** burns. They remove the vegetation canopy but do not burn into the peat or moss layer. The condition of such burns rely on weather, humidity, wind speed, fuel load and other factors. Unfortunately, some managed fires escape this careful control. It is not in a gamekeeper's interest to have a **'hot'** or **'deep'** burn: both severely compromise the heather's ability to regenerate.

Burning patches of heather in different years in this way provides a patchwork of different height heather – a mosaic providing areas for red grouse feeding, breeding and cover – beneficial not only to grouse but other moorland birds.

All managed burning is rotational in the sense that it happens periodically and the burnt vegetation goes through a cycle of recovery and maturity. In policy terms rotational burning has become associated with a prescription to burn on a fixed term of years (say every 15 years) which has been a feature of Natural England's management plans for upland SSSIs which are grouse moors. This rotational burning on deep peat has become highly contentious due to reported negative impacts of burning, especially on peatland ecosystem services. The concept of blanket bog restoration burning has been created for burning associated with restoring blanket bog (reducing heather dominance and restoring peat-forming plants). This is helpful as burning should be for an ecological purpose, not just by rotational rote. In reality what happened on the ground was somewhere between prescribed rote and burning when the heather height dictated a need to manage for grouse. Now, the concept of restoration burning has allowed the development of common middle ground allowing practitioners to assess and manage the land to benefit a much improved blanket bog assemblage of vegetation and health rather than just seeking the quickest heather re-growth of fresh shoots.

To some commentators, burning is burning, and no proper distinction between managed/prescribed/cool burns and wildfires is made, though in our view researchers are clear about this distinction.

Executive summary

1. POLICY CONTEXT.

1.1. This document has been prepared by the Game & Wildlife Conservation Trust (GWCT) working with experts in peatland ecology in several UK Universities. We also draw on expertise from the USA.

1.2. It is in anticipation of Defra's **'Peatland Strategy'** due to be published in 2020. This seeks to **'ensure all peatlands in England meet the needs of wildlife and people'** and show **'how peatlands can contribute to the UK's target of zero net emissions of greenhouse gases by 2050.'**

1.3. We highlight the findings of recently published research of relevance to policy decisions regarding the management of England's peatlands that may not have been considered by Defra and Natural England.

1.4. We give credit to research completed to date and what it tells us, but point out what it cannot tell us. Separate annexes describe both the research limitations and knowledge gaps.

1.5. Recently, restrictions to manage peatlands by rotational burning on deep peat have been put in place with the aim of helping to restore deep peat on functioning blanket bog. We support the restoration of blanket bog where this is possible but caution that simple **'no burn policies'** may have unintended negative consequences. This report sets out these concerns and the science behind them.

2. TYPES OF BURNING.

Not all burning is the same. It is important to distinguish between **'hot'** wildfires (like Saddleworth Moor in 2018) which tend to happen in summer and can burn into the underlying peat, and **'cool',** managed, and prescribed burns designed to burn surface vegetation and only take place within the **'burning season'** (October-April). These are the fires set by gamekeepers managing their moor to create optimum conditions for red grouse. See text box 2 on page 7 for an explanation of different types of burning.

3. **SECTION 1: Carbon Storage in England peatlands – some definitions and terminology.**

3.1. Peatlands cover 11% of England's land area and are estimated to store around 584 million tonnes (mt) of carbon. Peatlands are the UK's largest carbon store. If this carbon store were to be lost to the atmosphere it would be equivalent to 2.14 billion tonnes of CO_2 emissions.

3.2. Carbon fluxes (how carbon comes into and leaves peatland) and carbon stocks are the two key components that need to be measured.

3.3. On grouse moors, carbon is released when heather is burnt, but grouse moors can also capture carbon in the recovering, re-growing vegetation and in the black char left behind (from the burn). This **'flux'** is an immediate release of carbon in the smoke, followed by the slow capture of carbon in the re-growing plant tissue. Carbon is also lost when peatlands dry out. Conversely, carbon can be captured when blanket bogs are restored and start actively laying down peat again.

Dried grass and burnt heather branches.

3.4. As well as CO_2, there are other greenhouse gases (GHG) released by peat. **'Carbon dioxide equivalent'** or CO_2eq. is the term for describing different greenhouse gases in a common unit. In this report both methane (CH_4) and nitrous oxide (N_2O) are included in the carbon dioxide equivalent. A negative number (e.g. -0.61 t CO_2e ha^{-1} yr^{-1} etc) means that 0.61 tonnes of CO_2 equivalent are sequestered or stored per hectare per year.

3.5. The carbon stock is the amount of carbon (and peat) that has accumulated from a certain historical time point or within discrete time periods.

3.6. Data on long-term carbon stocks are still very limited. Data on both carbon fluxes and carbon stocks for peatland are sparse and biased towards a few repeat assessments of the same peatland sites. Data from so few sites need to be interpreted with caution.

4. SECTION 2: What is the current state of knowledge about carbon emissions and capture on upland peat?

4.1. Greenhouse gas emissions from our peatlands represent 4% of the UK's total GHG emissions.

4.2. Peatland not managed by man (near-natural) is regarded as **'close to carbon neutral'** or **'very small net GHG sources'** – a maximum of 0.01 tonnes of carbon dioxide equivalent per hectare per year.

4.3. GHG emissions from modified peatlands (modified by erosion, drainage, cutting, burning or grazing) are higher but are still relatively low (between 2.08 and 4.85 $tCO_2e\ ha^{-1}\ yr^{-1}$) compared to peatlands converted to cropland or grassland, harvested for fuel, or afforested (between 7.91 and 38.98 $tCO_2e\ ha^{-1}\ yr^{-1}$).

4.4. However, the large area of modified peatlands (about 41% of the UK's (not England) peatland resource), means these contribute 15% of all peatland GHG emissions (which includes peatlands converted to agriculture). Unfortunately, we cannot yet separate the figures for each of the modification types.

4.5. However, emissions from modified peatlands, the category including the grouse moors, represents less than 1% of the UK's total annual GHG emissions.

4.6. The crucial comparison is with peatland burned for red grouse compared with unburned or not recently burned areas. Compared to no burning, managed burning leads to short-term losses of above ground carbon when the vegetation is burned. But the carbon released is then stored again as the vegetation vigorously re-grows in subsequent years. Losses of carbon in the smoke can potentially be cancelled out by the vegetation re-growth. However, the science does not yet prove this.

4.7. Studies conducted have been short term i.e. in the year of the burn or the next year, so in the years when the carbon is lost; not over the full cycle of a burning cycle – say 15 years – when we would expect the carbon to have been re-stored. Long-term research to look at the overall net balance of carbon gain/loss over time is desperately needed.

4.8. However, two recent studies contradicted this **'general view'** (initial loss of carbon immediately after burning) and showed recently burnt plots emitted less carbon than older burn or no burned plots. Clearly more work is needed.

4.9. Every carbon stock study conducted thus far has recorded positive carbon and peat accumulation within flat and wet areas of blanket bog whether subject to burning or not. In general, areas of blanket bog burnt on a ten-year rotation accumulate less carbon than unburnt (or not recently burnt) areas. However, a recent study measured similar rates of carbon accumulation between plots burnt on a 20 year rotation, plots left unburnt since 1954 and plots left unburnt since 1923.

4.10. Another recent study explored the issue of pyrogenic charcoal. This is produced when vegetation is burnt and is also called soot, char, black carbon and bio char. It is produced during the incomplete combustion of material. It can store carbon in large quantities and for a very long time. A York University study found a positive relationship between moorland burn frequency and carbon storage through time. Pyrogenic charcoal was the key factor behind this relationship. The more frequently a piece of peatland was burned the more carbon was stored in the charcoal. Most studies ignore the role of pyrogenic charcoal, consequently, the carbon storage potential of burning management may have been underestimated, especially in flat wet areas of blanket bog where peat erosion is limited.

Continued overleaf >

5. SECTION 3: How much peatland is managed for grouse and can we estimate total carbon stored and carbon emissions?

5.1. Working with the Moorland Association (MA), we have mapped land in the UK designated above Defra's moorland line and superimposed over it land owned by grouse moor owners. We use this land owned by members of the MA as a proxy for land managed for red grouse. Our new estimate for the total area occupied by grouse moors is 423,000ha, with 228,000ha within the moorland line and therefore assumed to be on peat.

5.2. This now forms one of three methods we have used to estimate total carbon stored on grouse moors and net carbon emissions from grouse moors. The other two methods rely on different proxies for the area of grouse moor.

5.3. The area of grouse moor on peat in England is estimated using MA data to be 282,000ha, with other estimates being between 27,800 and 170,550ha. Expressed as a % of total peatland area in England, these figures are 41% and between 4% and 25%.

5.4. The total carbon stored on grouse moors using MA data is estimated to be between 66mt and 205mt, or between 11% and 35% of all carbon stored in England peatland.

5.5. Carbon dioxide equivalent emission estimates are necessarily crude as they are based on such varying estimates of area, peat condition and level of emissions.

5.6. An upper limit can be derived from the National Inventory Evans *et al.* (2017) which estimates the total upland peatland emissions at 603,386tCO_2e per year from 324,876ha to peat in varying condition. This would indicate a maximum grouse moor emission of 523,753 tCO_2e per year (based on 282,000ha of grouse moor on peat).

5.7. On that basis we have estimated that English grouse moors emit between 0.98% and 4.82% of total England peatland net carbon dioxide equivalent emissions.

6. SECTION 4: Wildfire.

6.1. Fire is a natural part of the management of many ecosystems around the world.

6.2. Both managed and wildfire are a global phenomenon, most often seen in warmer, dryer regions of the world, but making headlines in 2019 in Australia and California.

6.3. Everyone agrees that wildfires on upland blanket bogs are a problem. Vast areas of surface vegetation can be destroyed and fires can burn into the underlying peat layers destroying them to a considerable depth, even to bedrock. For example, Saddleworth Moor suffered a wildfire in 2018. Researchers at Liverpool University have estimated seven centimetres of peat were lost in addition to all surface vegetation, and that it will take up to 200 years to restore it.

6.4. The evidence surrounding the role of managed burning to manage and mitigate wildfire risk is unclear. Some propose that fires set by gamekeepers reduce fuel loads and burnt plots provide fire breaks that, in the event of a wildfire, help limit its spread, extent and severity. Others propose that these benefits do not exist and that burning dries out the land making it more susceptible to wildfire. Some managed fires escape control leading to wildfire; in the Peak District National Park Ranger Reports from 1976-2004, of those wildfires with a known cause, 25% were from escaped management fires. However, the area burnt by these escaped fires represented 51% of the burnt area of those fires with a known cause. Therefore, we should avoid simple binary statements that **'wildfires are bad and prescribed fire is good'** and instead we should look at the severity of the fire and seek to monitor the long-term environmental responses. Without this long-term view we run the risk of over/under-appreciating the impact of any one fire.

6.5. Managing fuel load through mechanical removal and/or prescribed burning is commonly undertaken around the world to meet wildfire risk reduction objectives. However, in the UK the evidence base is limited on the links (or not) between prescribed burning and wildfires. Consequently we sought the experience of others working in similarly fire-prone ecosystems (see Section 5).

6.6. Peatland restoration has been proposed as a mechanism to reduce wildfire risk in upland blanket peatlands. But wildfire experts state that on restoration sites **'fuel load build-up'** could threaten the success of such schemes if not carefully monitored. In other words, the threat of wildfire remains even on restoration sites. In any transition between vegetation communities (e.g. re-wetting, **'rewilding'**, forestry) wildfire risk should be factored into management plans.

Rewetting of peatlands should improve the resilience to wildfires under typical conditions, but these sites are still potentially flammable, particularly under environmental stress (e.g. persistent drought). Water tables typically drop in the summer especially in dry seasons.

6.7. In summary, rewetting will not prevent wildfire ignition or significant damage – this will require a reduction in fuel loads. Obviously, this is conjecture, but we think it is a valid view given the current evidence.

7. SECTION 5: Lessons from the USA: Managing fire-prone ecosystems via fire exclusion.

7.1. Since inception, the USA has dealt with controversy over how to manage wildland fire in its forests, woodlands, savannas, and grasslands. Evidence of fire history from pre-European settlement suggested frequent fire regimes (large areas with multiple fires per decade) ignited by lightning and Native Americans.

7.2. Late 19th and early 20th century wildfires in northern and western states caused human fatalities and damaged large forested landscapes. National policy focused on rapid fire suppression and bans on prescribed or managed fire by the 1930s.

7.3. As this widespread fire exclusion became the rule, negative ecological consequences were realised, e.g. a severe decline in habitat for the Northern Bobwhite Quail (*Colinus virginianus*), a formerly common upland game bird. When prescribed or managed burns were reintroduced quail numbers recovered. Non-game, rare bird species, in the formerly fire-prone region suffered steep declines without fire. Negative consequences for plants was also observed, namely- substantially reduced floristic richness, replacement of diverse grass-shrub communities and colonization by dense fire-intolerant tree species.

7.4. Late in the 20th century, fire suppression policies led to an increased extent and severity of wildfires, and these continue to the present day. A primary cause of this steep increase in the number of large wildfires and their uncharacteristic severity is the decades of fire exclusion and a **'reduced burn'** policy.

7.5. Fire exclusion led to increased tree density, heavy surface fuel loading, increased prevalence of fire-intolerant tree species, and landscape continuity that all acted to promote high intensity fire with often high severity.

7.6. The consequences of these fires for wildlife, and many rare plants has been severe, and the legacy of fire exclusion has been the large cost of containment and losses of ecosystem services.

7.7. Notable exceptions have been in regions where intentional prescribed fire has continued. High frequency, low intensity prescribed or managed fires maintain substantial local and regional plant and animal biodiversity and complement timber management and other land uses. The effects of prescribed fire on reducing wildfires, results have been overwhelmingly in favour of drastic reductions in wildfire where prescribed fires are common.

7.8. An insidious long-term problem resulting from policies to suppress prescribed burning is the loss of a **'fire culture'** in rural communities. Industries, policy, and public opinion fail to understand the value of prescribed or managed fire.

7.9. The USA experience with fire suppression is one potential path for managing fire-prone ecosystems. Changes in climate, particularly warming and its effects on wildfires is a complicating facet that will likely exacerbate the simplistic policy of reduced burning. Predicting a future without fire in UK's moorlands is complicated, but lessons learned in the USA and in other fire-prone regions of the globe suggest that finding ways to manage fire for biodiversity, wildfire hazard reduction, and carbon storage is an important strategy for long-term sustainability.

8. SECTION 6: Biodiversity and grouse moor management.

8.1. Birds.
Fire management of heather to increase red grouse in the UK may also provide suitable habitats for other upland birds, especially waders (dunlin, golden plover and curlew). The UK holds an estimated 27% of the global population of curlew, which is in steep decline. Numbers of curlew and golden plover were lowest on moors which received no burning.

8.2. Curlew were more numerous on overall shorter vegetation provided by cotton-grass, moss and recently burned heather, but where taller rushes were also present. Golden plover avoided tall heather and, together with red grouse, also preferred shorter vegetation of cotton grass and moss created by heather burning. Our own work on birds on managed heather that is the basis of these conclusions is ongoing and has not yet been peer-reviewed, but the abundance of waders (main species combined) was on average six-fold higher on moors with either high levels of managed burning or higher levels of sheep grazing than on two large moors with no burning and where sheep were virtually absent.

8.3. Cessation of managed burning on peatlands, when combined with the reduced sheep grazing that has occurred over the last two decades, is predicted to have negative repercussions for already declining upland waders.

8.4. Plants.
Heather dominated moorland supports communities of plants that are only found in the UK or are found more abundantly here than elsewhere in the world. Until the early 2000s, heather cover was falling sharply in the UK but a GWCT study found that between the 1940s and 1980s, moors that stopped grouse shooting lost 41% of the heather cover while moors that continued shooting lost 24%. The commitment to grouse management dissuaded moor owners from converting moors to forestry or areas dedicated to sheep.

8.5. *Sphagnum* mosses are particularly valuable for their peat-forming capacity. They contain **'hyaline cells'** which have a high water-holding capacity and form 80% of the plants' volume. This helps create a permanently wet environment in which decomposition of the *Sphagnum* material is inhibited by the water-logged, anaerobic (low oxygen) conditions, and by tannins that are released by the *Sphagnum* moss. This supports a build-up of plant material creating peat.

8.6. Much debate surrounds the role of grouse moor management, particularly burning, on sustaining blanket bog vegetation. A 2013 Natural England report examined burning on peatlands. Most studies indicated an overall increase in species richness or diversity when burning was considered at a whole moor level. Several studies have presented evidence that prescribed burning changes the species composition of blanket bog, promoting heather monocultures and reduced abundance of sedges and mosses. In contrast, other studies have demonstrated that a shorter (less than ten year) interval may be associated with greater cover of peat-building species such as *Sphagnum* mosses and cotton grass.

8.7. Cutting is increasingly being promoted as a less-damaging alternative to burning. Evidence for the effects of this cutting is currently very limited, with very little known about the long-term effects on vegetation structure and composition.

8.8. What happens to blanket bog if no management is undertaken will depend on many factors, including peat depth, altitude, rainfall, exposure, and levels of grazing. In some instances, natural layering of the heather may occur, allowing other plant species to grow up through the opened heather canopy. If sufficiently wet and exposed vegetation succession may be arrested resulting in a **'steady state'** where the blanket bog effectively maintains itself.

8.9. However, in many instances, climate, aspect, altitude and peat depth can all contribute to growing conditions which will require some form of management intervention (be it grazing, burning, cutting or a combination of those) if open blanket bog vegetation is to be maintained. The habitat management that is undertaken on grouse moors, including cutting and burning heather, can therefore help to maintain the conditions that are needed to sustain our blanket bogs, and the associated flora. Although these management interventions may have a carbon **'cost'** associated with them, these costs have to be offset against the outcome of maintaining active blanket bog.

8.10. Invertebrates.
Data to show the effect of burning on many invertebrates associated with heather, moorland vegetation or its management are limited. According to Natural England **'relatively few scarce species are restricted to moorland'** and **'the highest proportion of moorland species (of invertebrates) are among the moths, ground and rove beetles, money spiders and craneflies.'** And **'For invertebrate conservation on moorland, the main management objective is to maintain or increase the habitat diversity and the structural diversity of the vegetation, which will assist in increasing the diversity of invertebrate species.'** But they also add **'Catastrophic management, such as sudden periods of very intensive grazing, burning or cutting causes breaks in the continuity and the condition of habitats… may lead to the loss of invertebrate species.'**

8.11. The small size of these prescribed burns is not likely to create a problem for most invertebrates.

8.12. As with carbon, the timing of the assessment of the impact of burning on invertebrates is key. Burning will remove most invertebrates in the short-term, especially those in the litter layer (such as the moths pupating on the ground) but as long as there are nearby sources of tall vegetation re-colonisation will be first, especially among winged species.

9. CONCLUSION.

9.1. England's peatlands are an enormous carbon store and protecting that is extremely important.

9.2. Grouse moors only occur on upland peat. They are important strongholds for upland waders and most are designated in recognition of the special nature of these habitats and associated species.

9.3. Both Government and grouse moor managers have a vested interest in sustainable environmental and biodiversity outcomes: protecting both peat and the flora and fauna associated with it.

9.4. Grouse moor management is a key economic and social driver which underpins the human effort needed to create the environmental and biodiversity outcomes we all seek. Without that there will be no estate level staff to help fight wildfires, to implement peat bog restoration over large areas of England's uplands, and no predation control protecting vulnerable ground nesting birds such as curlew, dunlin, lapwing, golden plover and black grouse.

9.5. Peatland will emit GHG whether vegetation burning occurs or not; the aim should be to use burning as a vegetation management tool to best effect – to help balance outcomes and manage trade-offs. Burning is one of only three vegetation management tools available to the upland manager (burning, cutting and grazing).

9.6. Peat on grouse moors needs to be protected from wildfire, drying out and erosion. Upland waders need to be protected from predation and provided with a mixture of habitat types including the short vegetation created by managed burning. Cessation of managed burning on peatlands (possibly combined with the reduced sheep grazing since 2005) is predicted to negatively impact on these already declining upland waders. Reduced or no burning may help prevent peat drying out, but it will also allow the build up of fuel load which will make a wildfire potentially harder to control and more likely to burn into the underlying peat.

9.7. The concept of restoration burning on blanket bog has been created to help reduce heather dominance and restore peat-forming plants. It seems clear from the trade-offs that we will need more than this: we will need wildfire prevention and mitigation burning, upland wader habitat creation burning as well as burning for grouse.

9.8. Cutting is increasingly being promoted as a less-damaging alternative to burning but very little is known about the long-term effects on vegetation structure and composition, or associated carbon fluxes.

9.9. In the US well-intentioned policies which stopped managed burning of ground vegetation from the 1930s onwards have directly led to severe declines in some bird species and the incredibly damaging forest wildfires of today. Heather uplands are also fire-prone ecosystems.

9.10. The problem of insufficient evidence, experience and knowledge about how to create the best possible environmental outcomes, amidst complicated trade-offs between carbon storage, emissions, and biodiversity, with potential impacts on the economic, social and cultural aspects that underpin the environmental management means we must focus on the broader picture.

9.11. The only way that we can envisage achieving the complex management needed to balance these trade-offs is for landowners to formulate estate-scale policies that allow for learning through adaptive management. Policy direction will be needed, but these are living, working landscapes and to achieve results we need the harness the knowledge and experience of those who live and work there.

9.12. We believe there is a shared desire to protect peat, enhance biodiversity and maintain living, working landscapes. We also believe grouse moor managers should help achieve that by setting out their **'environmental offer'** for the future and work together to make a difference at scale.

9.13. This approach is endorsed by England's 25 Year Environment Plan (Defra 2018) which sets **'restoring and protecting our peatlands'** as a key target, and recommends using the new concept of **'Nature Recovery Network(s)... (to help achieve) landscape-scale recovery for peatland'**.

Twilight over heather, Rothbury Terraces, Northumberland, with views over the Coquet Valley to the Simonside and Cheviot Hills.

Carbon storage in English peatlands – some definitions and terminology

Peatlands cover 11% of England's land area and are estimated to have around 584 million tonnes of carbon stored there. Peatlands are the UK's largest carbon store. If this carbon store were to be lost to the atmosphere it would be equivalent to 2.14 billion tonnes of CO_2 emissions. (Natural England, 2010).

Peat is an organic material derived from vegetation that has built up in waterlogged conditions with low soil oxygen contents after the plants have died. These oxygen-poor conditions prevent dead plant material from decomposing. It is where carbon captured from the atmosphere is stored. Hence, they are called carbon stores or sinks. In contrast, places where carbon is lost are called carbon sources.

Carbon in peatlands does not just simply sit there. There are a whole number of dynamic processes that constantly release and capture carbon into and from the atmosphere. These dynamics are called the carbon flux.

Carbon fluxes and carbon stocks are the two key components that need to be measured and understood before we ask questions about peatlands.

Carbon flux

The carbon flux consists of ways in which carbon comes into and leaves the peatlands (inputs and outputs).

Inputs include:

- CO_2 take-up from the atmosphere by growing plants.
- Dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) coming in as rainfall.
- Inorganic carbon coming from the weathering of underlying bedrock (many moors sit on carboniferous limestone, some do not).

Outputs include:

- CO_2 and methane (CH_4) gasses escaping to the atmosphere as dead plants are damaged decompose.
- Carbon gases and compounds dissolved in water (DIC and DOC again) but also as particulate organic carbon (POC) and via other pathways.

Carbon stocks

Peat accumulates vertically over time within distinct stratified layers (Rydin *et al.*, 2013). During a carbon stock assessment, vertical peat cores are extracted from a peatland site. Various dating techniques are then used to determine chronological markers and age-depth profiles within each peat core. This enables researchers to calculate the amount of carbon (and peat) that has accumulated from a certain historical time point or within discrete time periods.

So, on grouse moors, carbon is released when heather is burnt, but grouse moors can also capture carbon in the recovering re-growing vegetation and in the black char left behind. This changes over time with the immediate release of carbon in the smoke and the slow capture of carbon in the growing plant tissue. How you assess carbon capture/release on a burnt grouse moor depends on when you measure it.

We discuss this in more detail in the report. But carbon loss is not just from burning. Carbon is also lost when peatlands dry out and carbon can be captured when blanket bogs are restored and start actively laying down peat again.

As well as CO_2 there are other greenhouse gases. Methane is another gas that comes from decomposing vegetation. The scientific jargon surrounding this topic can be confusing. A good source of helpfully clear definitions can be found at: *https://ecometrica.com/assets/GHGs-CO2-CO2e-and-Carbon-What-Do-These-Mean-v2.1.pdf* Authored by Matthew Brander in 2012.

So to simplify things, one term frequently used is **'carbon dioxide equivalent'** or CO_{2eq}. **'It is a term for describing different greenhouse gases in a common unit'. 'It allows bundles of greenhouse gases to be expressed in a single number and it allows different bundles of greenhouse gases to be easily compared in terms of their total global warming impact.'** See above web link.

You will often see GHG emissions data expressed as follows, for example, 0.01 $tCO_2\ ha^{-1}\ yr^{-1}$. This means that 0.01 tonnes of CO_2 equivalents are released per hectare per year. A negative number (e.g. -0.61 tCO_2 etc) means that 0.61 tonnes of CO_2 equivalent are sequestered or stored per hectare per year.

While the number of carbon flux studies from upland peatlands is increasing, data on long-term carbon stocks are still very limited. Furthermore, data on both carbon fluxes and carbon stocks within different types of upland peatland subject to different management are generally sparse and biased towards a few repeat assessments of the same peatland sites. Therefore, again, a cautious approach needs to be taken when interpreting data from so few sites.

It is important to note that studies of both approaches (carbon flux and carbon stock) have limitations as mentioned above. Details of these criticisms are laid out in the Appendix 1.

What is the current state of knowledge about carbon emissions and capture on upland peat?

There are two things that are measured to answer such questions:

Carbon fluxes

The recently published UK peatland GHG emissions inventory (Evans *et al.*, 2017) provides the best and most up-to-date information on the current state of knowledge about carbon fluxes on UK peatlands. This extensive assessment calculated GHG emissions for 13 peat condition categories (TABLE 1) using 1207 individual observations from 110 sites located across the UK and North western Europe. We do not know how many of these sites were from the UK or England, nor do authors distinguish between peatlands managed for grouse and those that do not. So we used their **'heather dominated modified bog'** category as a proxy for peatlands subject to management for red grouse. Even so, the calculations made in The Inventory by Evans *et al.* (2017) indicate that:

- Total peatland GHG emissions represent around 4% of the UK's total annual GHG emissions.
- Near-natural peatlands (peatlands relatively untouched by human management) are **'close to carbon neutral'**, and only **'very small net GHG sources'** (TABLE 1). Near-natural peatlands have emission factors between -0.61 and 0.01 tCO_2e ha^{-1} yr^{-1} (remember, negative numbers indicate GHG sequestration, whereas positive numbers indicate GHG release).
- The GHG emissions from modified peatlands (modified by erosion, drainage, cutting, burning or grazing) are higher than those recorded on near-natural peatlands, but they are still relatively low when compared to peatlands converted to cropland or grassland, harvested for fuel, or afforested (TABLE 1) modified peatlands have emission factors between 2.08 and 4.85 tCO_2e ha^{-1} yr^{-1}.
- Despite producing relatively low GHG emissions, the extent of modified peatlands (41% of the UK peatland resource) means that they contribute around 15% of all peatland GHG emissions (which include emissions from peatlands converted to agriculture). As such, emissions from modified peatlands (this category includes the grouse moors) represent less than 1% of the UK's total annual GHG emissions.
- England's peatlands converted to cropland, grassland and forestry are significant sources of GHG emissions and contribute 27%, 11% and 10% of all peatland GHG emissions respectively.

Crucially, however, due to low data availability, The Inventory published by Evans *et al.* (2017) did not calculate separate emission factors for upland peatlands. Nevertheless, if we remove emissions from lowland peatlands converted to cropland, the contribution of upland peatlands to the UK's total annual GHG emissions will certainly be less than 3%.

The 1% figure refers to emissions from peatlands subject to grouse moor management (using the '**Heather dominated modified bog'** category of Evans *et al.* (2017) as a proxy for grouse moor management). Whereas, the 3% refers to emissions from all upland peatlands regardless of grouse moor management.

In the wider peer-reviewed literature, the only land management option that has received any serious research attention in relation to carbon fluxes on upland peatland is prescribed managed burning, and this is usually compared to unburnt or not recently burnt areas. Carbon flux studies generally show that, compared to no burning, managed burning on upland peatlands leads to (following Harper *et al.*, 2018):

- Short-term losses of above-ground carbon stores due to the combustion of vegetation – the carbon released is usually then re-sequestered (stored again) as the vegetation re-grows in later years.
- Higher atmospheric CO_2 fluxes via plant and soil respiration in years immediately following a burn.

This is because no study has measured the carbon uptake of the vegetation growth post-burn for an entire burning rotation. However, it follows that the biomass emissions lost from a burn can be cancelled out by the vegetation

Female red grouse hiding in thick heather. © Tarquin Millington-Drake.

TABLE 1

Emission factors for peat condition types taken directly from Evans *et al.* (2017). Emission factors are shown in $tCO_2e\ ha^{-1}\ yr^{-1}$. A positive emission factor indicates net GHG emission, and a negative emission factor indicates net GHG removal.

Peatland type	Direct CO_2	CO_2 from DOC	CO_2 from POC	Direct CH_4	CH_4 from ditches	Direct N_2O	Indirect N_2O	Total
Forest	7.39	1.14	0.3	0.12	0.14	0.65	0.17	9.91
Cropland	26.57	1.14	0.3	0.02	1.46	8.97	0.54	38.98
Eroded modified bog drained	0.85	1.14	0.89	1.19	0.66	0.06	0.06	4.85
Eroded modified bog undrained	0.85	0.69	0.71	1.19	0	0.06	0.05	3.55
Heather dominated modified bog drained	-0.14	1.14	0.3	1.36	0.66	0.05	0.03	3.4
Heather dominated modified bog undrained	-0.14	0.69	0.1	1.36	0	0.05	0.02	2.08
Grass dominated modified bog drained	-0.14	1.14	0.3	1.36	0.66	0.05	0.03	3.4
Grass dominated modified bog undrained	-0.14	0.69	0.1	1.36	0	0.05	0.02	2.08
Extensive grassland	13.33	1.14	0.3	1.82	0.66	1.5	0.29	19.02
Intensive grassland	23.37	1.14	0.3	0.37	1.46	2.8	0.48	29.89
Rewetted bog	-2.23	0.88	0.1	2.02	0	0.04	0	0.81
Rewetted fen	0.86	0.69	0.1	4.24	0	0.24	0.04	6.37
Near-natural bog	-3.54	0.69	0	2.83	0	0.03	0	0.01
Near-natural fen	-5.44	0.69	0	3.88	0	0.24	0	-0.61
Extracted domestic	4.73	1.14	0.89	0.2	0.68	0.14	0.13	7.91
Extracted industrial	6.44	1.14	5	0.2	0.68	0.14	0.24	13.84

regrowth once it has achieved a similar biomass to that found pre-burn. This assumes the regrowth resembles the vegetation removed by burning but new growth is much better at taking up C than old growth. So it is likely that the initial loss of C can be made quickly. However, we lack the certainty to be more definitive than **'is usual'**.

Furthermore, there are several additional studies that have investigated burning impacts on carbon loss in water (on dissolved organic carbon DOC or particulate organic carbon POC) from upland peatlands, but the findings between studies are contradictory (dissolved organic carbon) or derived from unreliable field measurements (POC) (Harper *et al.*, 2018; Ashby & Heinemeyer, 2019).

But there have been recent studies that contradict this **'general'** view. Two studies (Clay *et al.*, 2010 and Clay *et al.*, 2015) showed that more recently burnt plots emitted less carbon than older burn or no burn plots. Clearly more work is needed here.

The impact of grazing on upland peatland GHG emissions has also received some research attention, but this has been largely investigated alongside burning using the Hard Hill experimental plots within Moor House NNR, Upper Teesdale (Ward *et al.*, 2007; Clay *et al.*, 2010; Ward *et al.*, 2012). The results of such studies report mixed responses of grazing on different elements of the carbon budget relative to unmanaged and burnt areas.

Carbon stocks

Generally, studies calculating carbon stocks within upland peatlands in the UK have made comparisons between burnt and unburnt (or not recently burnt) areas of blanket bog (Garnett *et al.*, 2000; Marrs *et al.*, 2019a). In summary, every carbon stock study conducted thus far has recorded positive carbon and peat accumulation within flat and wet areas of blanket bog whether subject to burning or not (Garnett *et al.*, 2000; Heinemeyer *et al.*, 2018; Marrs *et al.*, 2019a). It is worth noting that each of these studies examined carbon accumulation near the top of the peat profile (the near-surface) (Garnett *et al.*, 2000; Heinemeyer *et al.*, 2018; Marrs *et al.*, 2019a). However, on dry sites care must be taken not to relate near-surface carbon accumulation rates to the rest of the peat body (Appendix 1 Young *et al.*, 2019). But given that each of these studies examined near-surface peat cores from wet blanket bog sites, it is highly likely that the near-surface carbon accumulation rates can be related to the rest of the peat body.

In general, areas of blanket bog burnt on a ten-year rotation accumulate less carbon than unburnt (or not recently burnt) areas (Garnett *et al.*, 2000; Marrs *et al.*, 2019a). However, a recent study measured similar rates of carbon accumulation between plots burnt on a 20-year rotation, plots left unburnt since 1954 and plots left unburnt since 1923 (Marrs *et al.*, 2019a). Furthermore, another recent study explored the impact of pyrogenic charcoal (produced when vegetation is burnt) on carbon accumulation within peatlands managed for red grouse (e.g. by using managed burning) (Heinemeyer *et al.*, 2018). Pyrogenic charcoal, also called soot, char, black carbon and biochar is produced by the incomplete combustion of organic matter. It is resilient to oxidation so can store carbon for very long periods. This study, which was the first of its kind in the UK, found a positive relationship between moorland burn frequency and carbon accumulation through time, with charcoal being identified as the key factor behind the relationship (Heinemeyer *et al.*, 2018). While more work is required to corroborate this finding, the finding itself is unsurprising, given that pyrogenic charcoal is carbon-rich and resistant to decomposition (Leifeld *et al.*, 2018). Thus, as more charcoal is incorporated into the peat profile via burning, greater amounts of carbon will be locked away (assuming that the peat continues to accumulate) (see, for example, Wei *et al.*, 2018; Jones *et al.*, 2019).

We sought the opinion of some US researchers and they wrote the following comment:

'Burning any living or dead vegetation (fuel) emits stored carbon in smoke. The carbon consequences of wildfires are of global significance whereas the effect of prescribed or managed burning is more nuanced. While burning emits substantial CO_2 it produces considerable black carbon that is deposited in underlying soil as recalcitrant charcoal and dispersed widely in the generated plume as finer black carbon. Both of these solid forms are resistant to decomposition over long (centuries) periods (DeLuca and Aplet 2008). Over successive prescribed burns, the changes to the residual fuels and vegetation enable the remaining ecosystem to uptake atmospheric C more readily and make the ecosystem more resilient to future fires and store more C over time (Wiedinmyer and Hurteau 2010). Frequent prescribed burns are low in intensity and allow for rapid uptake and storage of C because the soil is not sterilized from excessive heat. Wildlands not burned frequently are vulnerable to rapid loss of stored above- and below-ground C when wildfires occur, typically when fuels are dry.'

How accurate are these estimates?

Carbon fluxes

The GHG emission factors produced in the recent UK peatland GHG emissions inventory and comparisons of the rates of loss between peat types (Evans *et al.*, 2017) are the ones being used to formulate peatland management policy but are likely to be inaccurate because:

- They did not distinguish between peatlands in the UK and Europe.
- They did not attempt to split the 'modified bog' categories by land management intervention such as burning, mowing, grazing or non-intervention.
- When calculating GHG emissions from near-natural and re-wetted peatlands, the authors left out data from sites subject to seasonal or continuous inundation.
- Emission calculations did not take into account key factors such as slope and rainfall.
- The study did not publish locations or environmental data (rainfall, peat depth, type of vegetation).
- The study provides only subjective estimates of the error around these estimates and so their accuracy cannot be better scrutinised.

More details regarding these six criticisms are in Appendix 2.

Upland peatland carbon flux data produced in the wider peer-reviewed literature (mainly on burning impacts) are also likely to be inaccurate because:

- It comes from a small number of studies that are often repeat assessments of a single experimental site at the Hard Hill plots (Glaves *et al.*, 2013; Harper *et al.*, 2018), which may not be representative of the wider upland peatland resource (very high and wet) (Baird *et al.*, 2019 but see Marrs *et al.*, 2019b for a contrasting opinion).
- There are few complete assessments for upland peatlands, with most studies focussing on one or several elements of the carbon budget (Glaves *et al.*, 2013; Harper *et al.*, 2018).
- Most carbon flux studies on upland peatlands are short-term (only one or two years) and are conducted within small experimental plots (Glaves *et al.*, 2013; Harper *et al.*, 2018), which means they are greatly influenced by short-term climatic and environmental fluctuations or extreme events. Thus, such studies provide a very limited insight into the long-term carbon fluxes at the moorland or catchment scale.

One important factor that has limited the accuracy of carbon flux studies is the failure to incorporate pyrogenic charcoal inputs into the calculation of emissions for areas of upland peat subject to prescribed burning (Harper *et al.*, 2018). Consequently, the carbon storage potential of burning management may have been underestimated, especially in flat wet areas of blanket bog where peat erosion is limited (e.g. Heinemeyer *et al.*, 2018).

Carbon stocks

Current carbon stock data are also likely to be inaccurate for the following reasons:

- It comes from only three studies and two of these are repeat assessments from one site (see above).
- Most studies do not measure pyrogenic carbon and its impact on carbon content.
- Most studies only take a small number of surface peat cores from small experimental plots and so do not make estimates at the moorland scale, thus they do not take account of carbon fluxes at depth or take into account key factors such as slope, vegetation type etc.

More detail regarding these criticisms are in Appendix 2.

What are the knowledge gaps?

It is very easy just to be critical but if we are to do a better job defining evidence-based policy, we will need better quality research. To get a more accurate picture of peatland GHG emissions and storage, we require more knowledge. This is set out in Appendix 4.

Footpath through the Pentland hills, Scotland, covered by heather flowers.

How much peatland is managed for grouse

and can we estimate total carbon stored and carbon emissions?

The exact area of peatland managed for grouse is unknown due to the lack of national survey data and inaccurate data on the extent of peatland managed for grouse.

We have looked at three ways of estimating the peatland managed for grouse and deriving estimates for carbon stored and emitted.

METHOD 1 – using Glaves *et al.* (2013) and Douglas *et al.* (2015) to estimate of area managed for grouse and data from UK Peatland GHG emissions inventory (Evan *et al.*, 2017).

If we assume that prescribed burning is synonymous with grouse moor management, then according to Glaves *et al.* (2013) grouse moor management occurs on about 25% of **'the total moorland deep peat resource in England'**. Extent data from the UK peatland GHG emissions inventory (Evans *et al.*, 2017) suggests that 25% of English deep peat equates to an area of 170,550 ha. However, a study by Douglas *et al.* (2015) derived from aerial images taken between 2001 and 2010 suggests that grouse moor management (i.e. burning management) on deep peat (peat >0.5 m deep) occurs across 27,800 ha within England. Again, using the peatland extent data from the UK peatland GHG emissions inventory (Evans *et al.*, 2017), this equates to 4.1% of the deep peat resource in England. The wild disparity between the two estimates provided above indicates that this is an area in which more accurate data are urgently required.

METHOD 2 – using previously unpublished maps from the Moorland Association overlaid on Natural England's 2010 carbon storage map.

For the first time in this report we attempt to improve the estimate of how much of England's peatlands are managed as grouse moors by plotting land owned by members of the Moorland Association onto carbon storage maps of peat published by Natural England (2010).

FIGURE 1 shows the map of the English northern uplands with estimates of the amount of carbon stored (in tonnes per hectare) within peaty soils. It also shows how much of this land is managed by members of the Moorland Association (henceforth MA) (423,000 ha) (see TABLE 1A), and how much of that is above Defra's moorland line and therefore assumed to be on peat (282,000ha) (see TABLE 2A).

The MA's membership could be another proxy for the area managed for red grouse but it is still not completely accurate (reasons why are discussed in Appendix 5).

From this map we have calculated the % of land owned by the MA in each of these five soil carbon content categories and also the total estimated carbon stored in them. These data are also compared to the land managed above the moorland line.

Close-up of Sphagnum moss. © Laurie Campbell

The other data source we have analysed quantifies the amounts of GHG emissions (estimated as CO_2 equivalents). The same configuration of emissions on land above the moorland line and land managed by the MA is shown in FIGURE 2 Here there are six categories of GHG emissions expressed as tonnes per hectare per year. Here the % of land on grouse moors emitting different levels of GHGs is very similar to emissions

FIGURE 1

Estimated carbon storage within deep and shallow peaty soils in upland England.

Taken from Natural England. **'England's peatlands: carbon storage and greenhouse gases.'** Natural England Report NE257 (2010).

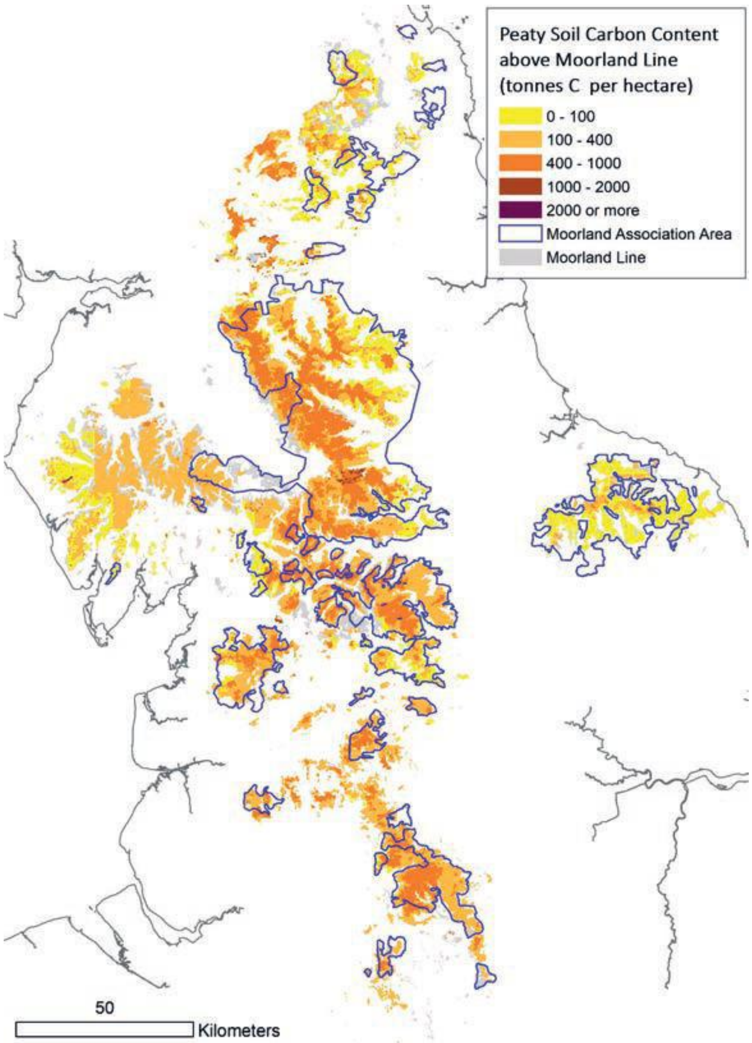


TABLE 1A

Hectarage of carbon storage within the Moorland Line, and the Moorland Association land. Hectares rounded to nearest thousand except†.

CARBON CONTENT RANGE	MOORLAND LINE		MOORLAND ASSOCIATION	
0 - 100	333,000	(22.9%)	118,000	(27.8%)
100 - 400	893,000	(61.5%)	192,000	(45.4%)
400 - 1000	209,000	(14.4%)	111,000	(26.3%)
1000 - 2000	18,000	(1.2%)	2,000	(0.5%)
2000 - 3500*	39†	(0.003%)	85†	(0.02%)
Total hectares	1,453,000		423,000	

* Original data did not specify an upper limit. To provide an upper value 3,500 tonnes C per hectare was used as it is a proportional increase from other ranges.

TABLE 1B

Tonnes of carbon stored within the Moorland Line, and the Moorland Association land. Tonnes rounded to nearest thousand.

CARBON CONTENT RANGE	MOORLAND LINE		MOORLAND ASSOCIATION	
	MIN	MAX	MIN	MAX
0 - 100	0	33,304,000	0	11,793,000
100 - 400	89,343,000	357,370,000	19,244,000	76,975,000
400 - 1000	83,403,000	208,508,000	44,558,000	111,395,000
1000 - 2000	17,509,000	35,019,000	2,255,000	4,510,000
2000 - 3500*	78,000	137,000	169,000	296,000
Total tonnes	190,333,000	634,338,000	66,226,000	204,969,000

* Original data did not specify an upper limit. To provide an upper value 3,500 tonnes C per hectare was used as it is a proportional increase from other ranges.

on land above the moorland line except in the lowest category of emissions (between zero and a net carbon sink) where a greater proportion of land in this category is not managed by MA members (TABLE 2A). This same trend is reflected in the tonnage figures given in TABLE 2B. Using this method:

- 282,000ha of peatland above the moorland line is managed by MA members (a proxy for English grouse moors).

- 29% of the carbon within peat soils above the moorland line is stored on land owned by the MA (TABLE 1A previous page). In terms of tonnes of carbon stored, it is between 66 million tonnes (mt) and 205mt, or between 35% of the minimum amount and 32% of the maximum amount found above the moorland line is stored on land owned by the MA (TABLE 1B previous page).

- This area of peatland has net emissions of between 106,000 and 948,000 tCO_2e per year, or between 0.95% and 8.5% of total England peatland emissions (assuming those to be 10,867,550 tCO_2e per year – see TABLE 4 – Evans *et al.* (2017)).

METHOD 3 – using Heather dominated modified bog as a proxy for grouse moor area and data from UK Peatland GHG emissions inventory (Evan *et al.* 2017).

If we assume that peatland grouse moors are, in general, likely to be heather dominated (this is a reasonable assumption given the relationship between burning and heather dominance, e.g. Glaves *et al.*, 2013), then we can derive some information about grouse moor carbon dynamics by using the drained and undrained **'Heather dominated modified bog'** categories within the UK peatland GHG emissions inventory (Evans *et al.*, 2017). This area totals 106,429 ha (see TABLE 4). For example, heather dominated modified bogs (i.e. grouse moors) take up some CO_2 directly (0.14 tCO_2 ha^{-1} yr^{-1}) but lose more via fluvial DOC (0.69-1.14 tCO_2 ha^{-1} yr^{-1}) and POC (0.10-0.30 tCO_2 ha^{-1} yr^{-1}) exports. Thus, in total, the UK peatland GHG emissions inventory suggests that undrained and drained grouse moors are net sources (rather than sinks) of GHG emissions as they emit between 0.65 and 1.30 tCO_2 ha^{-1} yr^{-1}, respectively. If this is scaled up using the full extent of **'Heather dominated modified bog'** across England, then grouse moors emit approximately 81,664 tCO_2 yr^{-1}. This equates to 1.07% of the peatland carbon emissions (CO_2 only) produced in England. Data from Evans *et al.* (2017) suggests that total England CO_2 emissions are 7,654,052 t yr^{-1}.

However as we have seen, the **'Heather dominated modified bog'** category is only a proxy for grouse moor management and there are limitations to the accuracy of these data. Indeed, the direct uptake figures in the **'Heather dominated modified bog'** category reported in the inventory seems far too low and contradicts other carbon flux studies, Heinemeyer *et al.* (2019) as well as peat core evidence that shows considerable net carbon uptake on UK grouse moors (Heinemeyer *et al.*, 2018; 2019; Marrs *et al.*, 2019a). Finally, the actual fate of carbon losses in water (DOC and POC) remains highly uncertain (is the carbon emitted or is it stored in habitats further downstream?) (Davies *et al.*, 2016).

Butterfly on heather under the evening sun.

FIGURE 2

Estimated greenhouse gas emissions within deep and shallow peaty soils in upland England.

Taken from Natural England. **'England's peatlands: carbon storage and greenhouse gases.'** Natural England Report NE257 (2010).

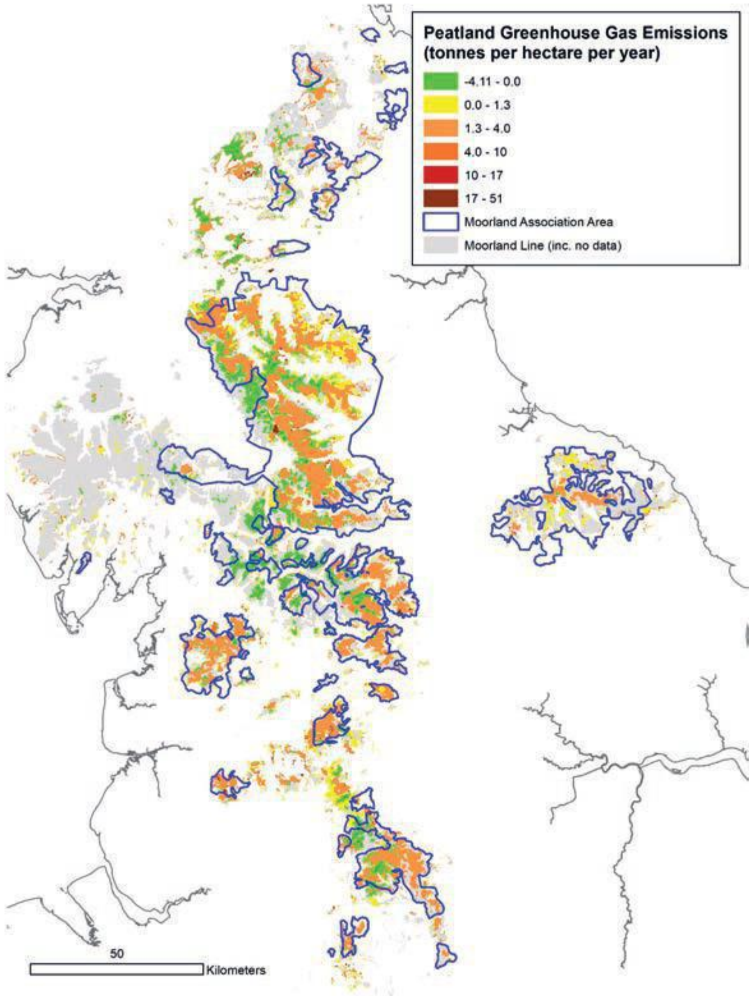


TABLE 2A

Hectarage emitting estimated greenhouse gases (CO_2 equivalents) within the Moorland Line, and the Moorland Association land. Hectares rounded to nearest thousand except.

GREENHOUSE GAS EMISSION RANGE	MOORLAND LINE		MOORLAND ASSOCIATION	
-4.1 - 0	80,000	(24.1%)	45,000	(15.9%)
0 - 1.3	70,000	(21.0%)	71,000	(25.3%)
1.3 - 4	172,000	(51.8%)	156,000	(55.4%)
4 - 10	6,000	(1.7%)	5,000	(1.7%)
10 - 17	2,000	(0.6%)	2,000	(0.7%)
17 - 51	2,000	(0.7%)	3,000	(1.0%)
Total hectares:	332,000		282,000	

TABLE 2B

Tonnes per year of greenhouse gas emissions (CO_2 equivalents) within the Moorland Line, and the Moorland Association land. Tonnes rounded to nearest thousand.

GREENHOUSE GAS EMISSION RANGE	MOORLAND LINE		MOORLAND ASSOCIATION	
	MIN	MAX	MIN	MAX
-4.1 - 0	-329,000	0	-185,000	0
0 - 1.3	0	91,000	0	93,000
1.3 - 4	224,000	689,000	203,000	625,000
4 - 10	23,000	57,000	20,000	49,000
10 - 17	21,000	35,000	19,000	32,000
17 - 51	40,000	120,000	50,000	149,000
Total tonnes:	-22,000	992,000	106,000	948,000

Is there a reasonable approximation for the amount of carbon dioxide equivalent emissions from peat managed for grouse (taking account of methane and nitrous oxide)?

Yes, assuming the **'Heather dominated modified bog'** categories are good proxies for peatland managed for grouse. Then, using this approach, undrained and drained heather dominated modified bogs (i.e. grouse moors) are estimated to produce between 2.08 and 3.40tCO_2e ha^{-1} yr^{-1}, respectively (**TABLE 1B**). Again, if we scale this up using the full extent of **'Heather dominated modified bog'** across England then English grouse moors emit approximately 246,727tCO_2e yr^{-1}. This equates to 2.3% of the total peatland GHG emissions produced in England.

Using data from the UK peatland GHG emissions inventory (Evans *et al.*, 2017), **TABLE 2B** displays the total and proportional contribution of different peatland types to the annual peatland GHG emissions in England. Figures for upland peat can be estimated by combining the **'Eroded'**, **'Heather dominated'**, **'Grass dominated'**, **'near-natural'** and **'rewetted'** bog categories within this table. Consequently, the total upland peat area of 324,876ha emits 603,386tCO_2e per year, or 5.6% of the total peatland GHG emissions produced in England. Thus, 94% of total GHG emissions in England come from lowland peatlands.

If grouse moors emit 2.3% of the peatland emissions produced in England and Scotland, this makes grouse moors the fourth-largest emitters of peatland GHG emissions in England, behind peatlands converted to cropland (66%), intensive grassland (20%) and forestry (6%) respectively (**TABLE 4**). Grouse moors produce relatively low peatland GHG emissions per hectare (**TABLE 4**) but they take up 16% of the total peatland area. However, the figures quoted may be inaccurate and over-estimated because:

- They assume that grouse moor extent and GHG emissions are broadly similar to the **'Heather dominated modified bog'** categories reported in UK peatland GHG emissions inventory (Evans *et al.*, 2017). Given the uncertainties around grouse moor extent and the limitations of UK peatland GHG emissions inventory, we have no idea whether such assumptions are accurate (even if they seem reasonable).

- They ignore the contribution of pyrogenic charcoal to GHG capture and storage within grouse moors (e.g. Harper *et al.*, 2018; Heinemeyer *et al.*, 2018; Leifeld *et al.*, 2018).

Freshly cut peat stacked to dry.

Summary

- The area of grouse moor on peat in England is estimated using MA data to be 282,000ha, with other estimates based on proxies being between 27,800 and 170,550ha. Expressed as a % of total peatland area in England, these figures are 41% and between 4% and 25%.
- The total carbon stored on grouse moors using MA data are estimated to be between 66mt and 205mt, or between 11% and 35% of all carbon stored in England's peatland.
- Carbon dioxide equivalent emission estimates are necessarily crude as they are based on such varying estimates of area, peat condition and level of emissions.
- An upper limit can be derived from Evans *et al.* (2017) which estimates the total upland peatland emissions at 603,386tCO_2e per year from 324,876ha to peat in varying condition. This would indicate a maximum grouse moor emission of 523,753tCO_2e per year (based on 282,000ha of grouse moor on peat), rather than the upper limit of 948,000 derived from the older (and presumably less accurate) 2010 Natural England report (see Method 2 in the summary table above).
- On that basis we have estimated that English grouse moors emit between 0.98% and 4.82% of total England peatland net carbon dioxide equivalent emissions.

TABLE 3

Summarising the results of the three methods.

	UNIT	METHOD		
		1	2	3
Gross area of grouse moor in England	ha		423,000	
Area of grouse moor in England on peatland / above the moorland line	ha	27,800 -170,550	282,000	106,429
Total peatland area in England	ha	682,201	682,201	682,201
Grouse moor as % total peatland in England		4% - 25%	41%	16%
Carbon stored in peat on grouse moors	mt	N/A	66-205	N/A
Total carbon stored in peat in England	mt	584	584	584
Carbon stored in peat on grouse moors as % of England total			11% - 35%	
CO_2 equivalent emissions on grouse moors	tCO_2e per year		106,000 -523,753	246,727
Average CO_2 equivalent emissions per ha of grouse moor	tCO_2e per year per ha		0.37 - 1.86	2.3
CO_2 equivalent emissions on total peat in England	tCO_2e per year	10,867,550	10,867,550	10,867,550
CO_2 equivalent emissions on total upland peat in England	tCO_2e per year	603,386	603,386	603,386
Grouse moors emissions as % total peatland emissions in England			0.98% - 4.82%	2.3%

TABLE 4

The area, GHG emission factors, total GHG emissions (CO_2 + CH_4 + N_2O) and percentage GHG emissions for different peat condition types within England. The data presented are calculated from the data presented in Evans *et al.* (2017). Emission factors are shown in $tCO_2e\ ha^{-1}\ yr^{-1}$ and total emissions are shown in $tCO_2e\ yr^{-1}$. A positive emission factor indicates net GHG emission, and a negative emission factor indicates net GHG removal.

Peatland type	Area (ha)	$tCO_2e\ ha^{-1}\ yr^{-1}$	$tCO_2e\ yr^{-1}$	% emissions
Forest	65,492	9.91	649,026	6.0
Cropland	182,701	38.98	7,121,685	66
Eroded modified bog drained	5,653	4.85	27,417	0.3
Eroded modified bog undrained	43,568	3.55	154,666	1.4
Heather dominated modified bog drained	19,208	3.4	65,307	0.6
Heather dominated modified bog undrained	87,221	2.08	181,420	1.7
Grass dominated modified bog drained	24,053	3.4	81,708	0.8
Grass dominated modified bog undrained	34,825	2.08	72,436	0.7
Extensive grassland	1,895	19.02	36,043	0.3
Intensive grassland	73,681	29.89	2,202,325	20
Rewetted bog	24,070	0.81	19,497	0.2
Rewetted fen	24,537	6.37	156,301	1.4
Near-natural bog	86,278	0.01	863	0.0
Near-natural fen	-	-0.61	-	0.0
Extracted domestic	4,391	7.91	34,733	0.3
Extracted industrial	4,628	13.84	64,052	0.6
TOTAL	682,201	145.49	10,867,550	100

Little owl, north Yorkshire, England.

Wildfire

Fire is a natural part of, and driving force behind, many ecosystems around the world. Several factors influence the occurrence and behaviour of wildfire (e.g. ignition sources, fuel characteristics) which can be described as the fire regime. The fire regime of a given area is effectively the when, where, what and how of fires in that location: when (e.g. seasonality), where (e.g. size and shape), what (e.g. type of fire), and how (e.g. fire intensity, flame length, fuel consumption). Fire regimes may change naturally through time (e.g. changes in vegetation composition) or be altered by human activities (e.g. agricultural activities). Human activities may alter fuel structure, change ignition sources, or the timing of fire activity.

Wildfires are a global phenomena though we commonly observe them, in particular regions such as the Mediterranean, Australia and USA. Indeed, recent major conflagrations in Australia and the Amazon basin have captured headlines around the world. Climate change will impact fire regimes around the world and along with changing land use practices (e.g. building houses in the rural-urban interface) and rural demographics, we need to better understand the global wildfire threat.

UK wildfire

In England alone between financial years 2009/10 and 2016/17 the Fire and Rescue Services (FRS) attended over 258,000 outdoor vegetation fires, an average of over 32,000 each year. Many of these were small (<1 ha), though bigger, **'landscape scale'** fires do occur. Most incidents occurred in built-up areas and gardens. The majority of the area burnt was on arable, improved grassland, semi-natural grassland or **'mountain, heath and bog (open habitats)'**. In 2011/12, 95% of the area burnt that year was classified under one of these four categories, and the greatest area burnt in 2011/12 was on mountain, heath and bog.

Battling the Marsden Moor fire, West Yorkshire © Craig Hannah.

Fire at Dovestones, Saddleworth. © Craig Hannah

Wildfire on upland blanket bogs

Everyone agrees that wildfires on our upland blanket bogs are a problem. Vast areas of heather, grass, and moss can be destroyed and fires can burn into the underlying peat layers destroying them to a considerable depth or even to bedrock, not just removing surface vegetation e.g. Saddleworth Moor where it has been estimated by researchers at Liverpool University that seven centimetres of peat were lost in addition to all surface vegetation.

For some time, there was no separation between wildfires and prescribed burns. That separation is now better acknowledged and understood, but the links between wildfire and prescribed burning are not clearly understood.

Some propose that prescribed burning reduces fuel loads and burnt plots provide fire breaks that help limit the spread, extent and/or the severity of wildfires. Others propose that these benefits do not exist and that burning dries out the land making it more susceptible to wildfire. Some evidence suggests that over 50% of wildfire incidents with known causes may themselves be caused by the loss of control of prescribed or managed burns (source: National Trust Scotland). However, when reviewed by the Scottish Fire and Rescue Service, this figure reduced to 9%. Some managed fires escape leading to a wildfire; in the Peak District National Park Ranger Reports from 1976 – 2004, of those wildfires with a known cause, 25% were from escaped prescribed or managed fires. Also the area burnt by these escaped fires represented 51% of the burnt area of those fires with a known cause (IUCN UK Committee Peatland Programme).

Ignitions

In the UK, most ignitions are man-made in origin, whether that is accidental (e.g. discarded BBQ, escaped prescribed or managed burns) or deliberate (i.e. arson). There are very few cases of wildfires ignited by lightning strikes (there was a notable recent exception in the Cheviot Hills in 2018). In some areas of the UK there is evidence to suggest that there is a connection between public access and wildfire occurrence. In the Peak District, fires were more frequent near to roads and footpaths (e.g. the Pennine way) and at certain times of the year (e.g. Bank Holidays), though more recent modelling suggests these associations may have changed since 2009 (Albertson *et al*;, 2010 and McMorrow *et al*., 2009).

Attributing a definite ignition source for any wildfire is not simple. The Fire and Rescue Service Incident Recording System (IRS) includes a section on the source of ignition, but this remains unconfirmed unless a fire investigation is done, and this is very rare for vegetation fires. Local knowledge from land managers, gamekeepers and rangers can sometimes shed light on suspected causes.

A hiking path cuts through a landscape scene which was once heather and is now ash after fires spread across the land..

Fuel management and impact on wildfires

Managing fuel load through mechanical removal and/or prescribed burning is commonly undertaken around the world to meet wildfire risk reduction objectives. However, in the UK the evidence base is limited on the links (or not) between prescribed burning and wildfires. The 2015 report to Scottish Natural Heritage entitled **'A Review of Sustainable Moorland Management'** written by Werritty *et al.* (2015) concludes that **'overall, the relationship between the use of prescribed fire and the frequency and extent of wildfires as moorland remains contested and this is an area where the evidence-base needs to be developed'**.

A particular challenge for the UK uplands is the need to balance different ecosystem services provided by peatlands in particular (e.g. carbon, water quality, biodiversity). This might not be the case in other areas of the world where vegetation management by fire is better understood (see Section 5).

Environmental impact

Understanding the environmental impact of wildfires requires an assessment of the severity of the fire immediately after a fire, as well as monitoring the long-term environmental response. The challenge for assessing severity is the fact that it not always possible to know the pre-burn vegetation and environmental characteristics. Indeed, most wildfire studies cannot know these. Instead nearby unburnt vegetation is used as the **'control'** site to allow assessments of fire severity.

Studies of fire severity and environmental impacts in UK uplands (e.g. Davies et al, 2016; Clay and Worrall, 2011; Maltby *et al*., 1990) have shown a range of impacts with some wildfire events consuming similar amounts of biomass to a prescribed burn and not impacting the underlying peat, through to catastrophic events leaving long-term damage to a landscape. Equally, poorly conducted prescribed or managed fires can lead to damaging impacts. Therefore, we should avoid simple binary statements that **'wildfires are bad and prescribed fire is good'** and instead we should look at the severity of the fire and seek to monitor the long-term environmental responses. Without this long-term view we run the risk of over/under-appreciating the impact of any one fire.

Restoration

Peatland restoration has been proposed as a mechanism to reduce wildfire risk in upland blanket peatlands. We agree with this, especially if restoration involves re-vegetating bare peat and raising water tables by removing or blocking drains (re-wetting). Grouse moor managers have indeed blocked drainage channels on their moors to re-wet the peat and this has led to positive outcomes for estates (e.g. grouse chicks feed on the insects emerging

Wildfire damage, having burned down into the peat layer.

from these waterlogged areas). Indeed, this is the thinking behind a cool burn undertaken by gamekeepers for red grouse which is restricted to the wetter and colder winter months when the moss and peat are saturated – this results in the moss and peat layers remaining relatively undisturbed during the burn.

However, in the process of restoring these sites, careful monitoring of fuel will be needed to avoid a build-up of fuel load during the transition between vegetation communities. Rewetting of peatlands should improve the resilience to wildfires under typical conditions, but these sites are still potentially flammable, particularly under environmental stress (e.g. persistent drought). Water tables typically drop in the summer especially in dry seasons.

But the wildfire experts also state that on restoration sites **'fuel load build-up'** could threaten the success of such schemes if not carefully monitored. In other words, the threat of wildfire remains even on restoration sites (McMorrow *et al.*, 2009 p427). In any transition between vegetation communities (e.g. re-wetting, **'rewilding'**, forestry) wildfire risk should be factored into management plans.

In summary, rewetting will not prevent wildfire ignition or significant damage – this will require a reduction in fuel loads. Obviously, this is conjecture, but I think it is a valid view given the current evidence we have.

Golden plover, on its nesting site in the heather moorlands of northern England.

Lessons from the USA: Managing fire-prone ecosystems via fire exclusion

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Since its inception, the USA has dealt with controversy over how to manage wildland fire in its forests, woodlands, savannas, and grasslands. Evidence of fire history from pre-European settlement suggested frequent fire regimes (large areas with multiple fires per decade) were evident across the bulk of the continent, ignited by lightning and Native Americans (Guyette *et al.*, 2012). Early European settlers used fire, albeit to a lesser extent and in contrasting ways to the tribes they displaced. Late 19th and early 20th century wildfires in the northern and western states caused human fatalities and damaged large forested landscapes. The resulting national policy was focused on rapid fire suppression and bans on prescribed or managed fire (Stephens and Ruth 2005). These were in place across much of the USA by the 1930s.

As widespread fire exclusion became the rule in the USA, negative ecological consequences were realized. In the south eastern region, Stoddard (1931) discovered that the lack of fire had led to a severe decline in habitat for the Northern Bobwhite Quail (*Colinus virginianus*), a formerly common upland game bird. When fires were reintroduced as prescribed or managed burns, quail numbers recovered. Non-game wildlife in the formerly fire-prone region suffered steep declines without fire. Rare bird species listed as priority species in State wildlife action plans respond positively to managed fire (Gaines *et al.*, 2019). The negative consequences for plants was also observed, namely- substantially reduced floristic richness, replacement of diverse grass-shrub communities and colonization by dense fire-intolerant tree species (Glitzenstein *et al.*, 2012).

Late in the 20th century, another negative consequence of fire suppression policies was revealed, namely the increased extent and severity of wildfires. Areas of the Pacific and Rocky Mountain west suffered large, high severity wildfires beginning in the late 1970s. and continuing to the present day. A primary cause of this steep increase in the number of large wildfires and their uncharacteristic severity is the decades of fire exclusion and a **'reduced burn'** policy. Fire exclusion led to increased tree density, heavy surface fuel loading, increased prevalence of fire-intolerant tree species, and landscape continuity that all acted to promote high intensity fire with often high severity (Agee and Skinner 2005). Small trees in a forest act as ladders for fire to reach the dense canopy and spread as crown fires across areas formerly dominated by frequent low intensity surface fires. The consequences of these fires for wildlife, and many rare plants has been severe (Brennan *et al.*, 1998). Beyond the biodiversity consequences, the legacy of fire exclusion has been the large cost of containment and losses of ecosystem services. Single large wildfire events in the western USA now typically cost ca. $500 million to $1 billion to suppress, not counting the losses in biodiversity, natural resources, timber, tourism, and diminished provision of clean water, air, and rehabilitation required to restore these habitats.

Notable exceptions to these negative patterns have been in regions where intentional prescribed fire has continued. On lands managed for game like the quail lands in the eastern US where prescribed burning occurs, rare birds (federally endangered red-cockaded woodpecker, *Picoides borealis*, and other rare upland non-game birds), and rare plants have not shown the same declines associated with a **'reduced burn'** policy (Ryan *et al.*, 2013, Stephens *et al.*, 2019). High frequency, low intensity prescribed or managed fires maintain substantial local and regional plant and animal biodiversity and complement timber management and other land uses. Prescribed fire in these landscapes have consumed surface fuels, maintained low tree densities, and created horizontal and vertical discontinuities at the patch, stand, and landscape scales resulting in far fewer and less damaging wildfires

Male Northern bobwhite quail, southern Texas, USA.

(Ryan *et al.*, 2013). In the reviews of the effects of prescribed fire on reducing wildfires, results have been overwhelmingly in favor of drastic reductions in wildfire where prescribed fires are common (Kalies and Yocum-Kent 2016).

An insidious long-term problem resulting from policies to suppress prescribed burning is the loss of a **'fire culture'** in rural communities. Centuries of fire knowledge bolstered by science and technology allow for relatively easy application of prescribed fire at a landscape scale in the South eastern USA because fire is part of the culture. Misguided fire suppression policies in other parts of the USA have resulted in generations without a knowledge of fire application, ecological benefits, and wildfire reduction. As such, industries, policy, and public opinion fail to understand the value of prescribed fire.

The USA experience with fire suppression is one potential path for managing fire-prone ecosystems. Changes in climate, particularly warming and its effects on wildfires is a complicating facet that will likely exacerbate the simplistic policy of reduced burning. Predicting a future without fire in UK's moorlands is complicated, but lessons learned in the USA and in other fire-prone regions of the globe suggest that finding ways to manage fire for biodiversity, wildfire hazard reduction, and carbon storage is an important strategy for long-term sustainability.

Red-cockaded woodpecker in Florida, USA. © Robert Emond

Controlled burning of forest understorey, Florida, USA.

Biodiversity and grouse moor management

Heather burning and birds

Managed strip burning of heather-dominated moorland as an integral component of grouse moor management in the UK uplands has recently become highly contentious due to reported negative impacts of burning, especially on peatland ecosystem services. However, fire management of heather for the purpose of increasing red grouse densities and their breeding success may also provide suitable breeding habitat for other upland birds and especially waders (Tharme *et al.*, 2001). Moorland waders include dunlin, golden plover and curlew, the latter being a species in severe decline in the UK, which holds an estimated 27% of the global population (Brown *et al.*, 2015).

Preliminary findings from two on-going analysis of bird data collected by GWCT describe bird-habitat associations on managed grouse moors (D. Baines unpublished data). The first, from 110 1-km plots on 35 moors in northern England suggest that heather burning is beneficial for golden plover, being associated with higher breeding densities, impacts upon skylark, which is associated more strongly with grassland, and is neutral for curlew, lapwing and meadow pipit.

Interpretation of bird-habitat relationships within such multi-site analyses can be difficult due to between-site differences in natural factors such as geology, peat depth and weather as well as anthropogenic factors such as management of predators, sheep grazing intensity and landscape scale mosaic and fragmentation. For this reason, a second study was conducted that considered the same suite of moorland birds on one large, high altitude peatland landscape in the Upper Tees / Tyne catchment. In this second study, the abundance of waders (main species combined) was on average six-fold higher on moors with either high levels of managed burning or higher levels of sheep grazing (i.e. short vegetation) than on two large moors with no burning and where sheep were virtually absent. The remaining moors, with intermediate values of grazing and burning, had intermediate wader densities. The most frequently encountered species of wader were curlew and golden plover, which formed 49% and 35% respectively of waders present summed across all sites.

Curlew and golden plover abundances were lowest on moors which received no burning, but red grouse were at similar densities (1.6-3.3 bird km^{-1}). Pipit densities also varied little across moors, ranging from 3.9 – 7.9 birds km^{-1}, but skylark densities were higher on grassier sites, which had higher levels of sheep grazing. Curlew were more numerous on overall shorter vegetation provided by cotton-grass, moss and recently burned heather, but where taller rushes were also present. Golden plover avoided tall heather and, together with red grouse, also preferred shorter vegetation of cotton grass and moss created by heather burning. Meadow pipits preferred taller cotton grass on shallower peat soils associated with a greater frequency of burning and less heather, but more grass cover. Skylark preferred short vegetation and avoided heather, including that with a higher frequency of burning.

We predict that cessation of managed burning on peatlands, especially when combined with the reduced sheep grazing that has occurred over the last two decades, may have negative repercussions for already declining upland waders. Dunlin, which tend to use the shortest, most eroded bare peat communities (Brown 1938, Lavers & Haines-Young 1997) often towards fell summits, is already in steep, but not fully quantified, decline (Balmer *et al.*, 2013). Necessity for short vegetation for nesting and chick rearing amongst both golden plover and curlew (Whittingham *et al.*, 2001, 2002), which to-date has often been provided by heather burning (Robson 1998), may restrict their future distribution and abundance in the uplands. Provision of consents for cutting of heather on designated sites may help mitigate against imposed burning restrictions, especially if they are done on similar scales.

Reductions in burn-cut management interventions on heathland may similarly impact waders through increasing vegetation height (Stroud *et al.*, 1987). Taller heather swards, especially if interspersed with invasive scrub, may be more attractive to black grouse (Baines 1996) and would certainly benefit passerine communities, particularly stonechat and whinchat (Tharme *et al.,* 2001), together with some species of warbler, for example willow warbler, whitethroat and grasshopper warbler. More passerines would in turn benefit merlin, whose principal prey is small passerines (Newton *et al.*, 1984), and even hen harrier, but only if sufficient grassland areas persisted to retain formerly abundant meadow pipits, skylark and field voles (Smith *et al.,* 2001). Succession to woodland could be fast unless management intervention was instigated, with rapid loss of moorland bird species.

Adult curlew on moorland in Yorkshire during the breeding season.

Common cottongrass or bog cotton.

Higher and lower plants

Heather-dominated moorland supports communities of plants that are only found in the UK or are found more abundantly here than elsewhere in the world. These communities are different to those found under other land uses such as commercial forestry or agriculture. They include species of berry, grass, sedge and moss, including *Sphagnum* moss, which together define habitats that are listed under the EU's Conservation of Natural Habitats and of Wild Flora and Fauna Directive (European Council Directive 92/43/EEC). Many UK upland sites are designated under this Directive as Special Areas of Conservation (SAC) (JNCC 2020), with underpinning UK notification as Sites of Special Scientific Interest (SSSI), in recognition of the special nature of these habitats, and associated plant species, that they support.

Over the last 200 years, heather cover has fallen sharply in the UK, generally as a result of overgrazing and commercial forestry plantations (Stevenson & Thompson 1993). However, a GWCT study found that between the 1940s and 1980s, moors that stopped grouse shooting lost 41% of their heather cover, while moors retaining shooting lost only 24% (Robertson *et al.*, 2001). Historically, a landowner's commitment to grouse management may have dissuaded them from converting moors to other land uses such as forestry or sheep grazing. Both of these activities can destroy the valuable conservation habitats associated with moorland heather or peat bog, though excessive sheep grazing diminished significantly once sheep headage payments were stopped in 2005.

Some of these areas of heather moorland sit on blanket bog, a globally restricted habitat that is confined to cool, wet climates and relies on rainfall to maintain its wetness. The dominant species on bogs in Western Europe are specialised and distinctive and although they can form nine different UK-defined vegetation communities (JNCC 2008), many include the typical blanket mire species of heather *Calluna vulgaris*, cross-leaved heath, *Erica tetralix*, deer grass *Trichophorum germanicum*, cotton grass *Eriophorum* spp. and several of the bog moss *Sphagnum* species.

Sphagnum mosses are particularly valuable for their peat-forming capacity, largely due to their structure and their ability to thrive in nutrient-poor soils. They contain **'hyaline cells'** which have a high water-holding capacity and form 80% of the plants' volume. This helps create a permanently wet environment in which decomposition of the *Sphagnum* material is inhibited by the water-logged, anaerobic (low oxygen) conditions, and by tannins that are released by the *Sphagnum* moss. This supports a build-up of plant material, creating peat which grows approximately 1mm per year in depth.

While some species of *Sphagnum* may be associated with poor-fen or dry heath conditions, others are notable peat-formers. Species such as *Sphagnum capillifolium, S. magellanicum* and *S. papillosum* are all hummock-forming species with a greater water-holding capacity and are more resistant to low water and pH levels than some other species of *Sphagnum* and their presence may be considered indicative of blanket bog in good condition.

The role that grouse moor management can play in sustaining blanket bog vegetation is the focus of much debate, particularly regarding the traditional practice of heather burning. A 2013 report by Natural England (Glaves *et al.*, 2013) examined much of the scientific literature available at that time examining burning on peatlands. Most studies considered in that report indicated an overall increase in species richness or diversity when burning was considered at a whole moor level. Because burning takes place in small areas leaving the majority unburnt in any given year, a mixture of habitats is produced which can support a wider variety of species. Several studies have presented evidence that prescribed burning changes the species composition of blanket bog, promoting heather monocultures (Littlewood *et al.*, 2010) and reduced abundance of sedges and mosses (Harris *et al.*, 2011). In contrast, other studies have demonstrated that a shorter (less than ten year) interval may be associated with greater cover of peat-building species such as *Sphagnum* mosses and cotton grass (Milligan *et al.*, 2018; Whitehead *et al.*, 2018). Cutting is increasingly being promoted as a less-damaging alternative to burning, for maintaining the shorter, more open heather canopy that favours persistence of other blanket bog plant species. Evidence for the effects of this cutting is currently very limited, with very little known about the long-term effects on vegetation structure and composition (Heinemeyer *et al.*, 2019).

What happens to blanket bog if no management is undertaken will depend on many factors, including peat depth, altitude, rainfall, exposure and grazing. In some instances, natural layering of the heather may occur, allowing other plant species to grow up through the opened heather canopy. If levels of wetness and exposure are sufficient to arrest vegetation succession, it may be possible to achieve a **'steady state'** where the blanket bog effectively maintains itself. However, in many instances, climate, aspect, altitude and peat depth can all contribute to growing conditions which will require some form of management intervention (be it grazing, burning, cutting or a combination of those) if open blanket bog vegetation is to be maintained. For example, on areas of blanket bog that are adjacent to forest plantations, there can be a significant problem from reseeding and encroachment of spruce, particularly where grazing levels have been reduced or removed.

The habitat management that is undertaken on grouse moors, including cutting and burning heather, can therefore help to maintain the conditions that are needed to sustain our blanket bogs, and the associated flora. Although these management interventions may have a carbon **'cost'** associated with them, these costs have to be offset against the outcome of maintaining active blanket bog.

Close-up detail of colourful Sphagnum moss in autumn.

Invertebrates

The effect of burning on many invertebrates associated with heather, moorland vegetation or its management are limited. The best research studies seemed to have been conducted in the late 1990s. According to Natural England (2001) **'relatively few scarce species are restricted to moorland'** and **'the highest proportion of moorland species (of invertebrates) are among the moths, ground and rove beetles, money spiders and craneflies.'**

They go on to say **'for invertebrate conservation on moorland, the main management objective is to maintain or increase the habitat diversity and the structural diversity of the vegetation, which will assist in increasing the diversity of invertebrate species.'**

This can be achieved by prescribed burning. But they also add **'catastrophic management, such as sudden periods of very intensive grazing, burning or cutting causes breaks in the continuity and the condition of habitats. This may lead to the loss of invertebrate species, although the scale is obviously important – how catastrophic an event may be depends on the amount of ground covered in relation to the dispersal distance of the invertebrate species.'**

But the small size of these prescribed burns is not likely to create a problem for most invertebrates (Haysom & Coulson 1998). In other studies some authors (Gimingham 1975) found that prescribed burning reduced invertebrate biodiversity by Usher & Jefferson (1991) found conflicting results, concluding that burning maximised the diversity of spiders and beetles.

As with the debate over carbon, the timing of the assessment of the impact of burning on invertebrates is key. Burning will remove most invertebrates in the short-term, especially those in the litter layer (such as the moths pupating on the ground) but as long as there are nearby sources of tall vegetation re-colonisation will be first, especially among winged species.

Clockwise: Cranefly. True lover's knot moth. Green tiger beetle. Rove beetle. © Will George.

As well as environmental benefits, shooting can make an important contribution to the local economy. © Matt Limb

Red grouse in heather, Yorkshire Dales, UK.

Conclusion

England's peatlands are an enormous carbon store and protecting that is extremely important. This report focuses on the current and future environmental and biodiversity contribution of grouse moor management in that context, and how heather burning can be used as a vegetation management tool alongside cutting and burning. It estimates for the first time the amount of carbon stored on grouse moors and estimates GHG net emissions.

Grouse moors only occur on upland peat and its heather and peat-forming plants sustain red grouse. They are important strongholds for upland waders and most are 'designated' in recognition of the special nature of the habitats, and associated plant and bird species. Historically, commitment to grouse management is associated with less forestry or sheep grazing, both which can destroy the valuable conservation habitats associated with moorland heather or peat bog. Both Government and grouse moor managers have a vested interest in sustainable environmental and biodiversity outcomes: protecting both peat and the flora and fauna associated with it.

However, this environmental sustainability is intrinsically linked to economic and social sustainability. Grouse moor management is a key economic and social driver which underpins the human effort needed to create the environmental and biodiversity outcomes we all seek. Without such management there will be no estate level staff to help fight wildfires, to implement peat bog restoration over large areas of England's uplands, and no predation control protecting vulnerable ground nesting birds such as curlew, dunlin, lapwing, golden plover and black grouse.

Creating these balanced outcomes is complex and there will be trade-offs.

All England's peatland types are net emitters of GHG, even near-natural bog emits some (see TABLE 4). The estimated annual total tonnes of CO_2 equivalent emitted is 11 million tonnes. Arable cropping and intensive grass on lowland peat/fen emit the most (86% of the total), upland peat only 5.6%. It is difficult to calculate how much grouse moors contribute total emissions, but our estimate is between less than 1% (0.98%) and 4.8%. Peatland will emit GHG whether vegetation burning occurs or not; the aim should be to use burning as a vegetation management tool to best effect – to help balance outcomes and manage trade-offs. Burning is one of only three vegetation management tools available to the upland manager (burning, cutting and grazing).

Peat on grouse moors needs to be protected from wildfire, drying out and erosion. Upland waders such as golden plover, dunlin and curlew need to be protected from predation and provided with a mixture of habitat types including the short vegetation created by managed burning. Cessation of managed burning on peatlands (combined with the reduced sheep grazing since 2005) is predicted to negatively impact on these already declining upland waders. Reduced or no burning may help prevent peat drying out, but it will also allow the build-up of fuel load which will make a wildfire potentially harder to control and more likely to burn into the underlying peat not just the surface vegetation. Modern grouse moor managers use 'cool' burns to regenerate the heather to encourage new green shoot growth to feed grouse, but this also serves to provide preferred habitat for waders and support a greater diversity of moorland plants.

The concept of restoration burning on blanket bog has been created to help reduce heather dominance and restore peat-forming plants. The difficulty is there is no common view between scientists as to how burning should be best utilised to help restore blanket bog, and there are knowledge gaps around the long-term carbon cycle associated with heather burning. Furthermore, it seems clear from the trade-offs identified above that we will need more than this: we will need wildfire prevention and mitigation burning, upland wader habitat creation burning as well as burning for grouse.

Then there are potential trade-offs between types of vegetation management. Golden plover seem happy to accept short vegetation produced by either burning or sheep grazing. However, sheep numbers have dropped dramatically since 2005 and seem likely to drop further post-Brexit. Cutting is increasingly being promoted as a less-damaging alternative to burning but very little is known about the long-term effects on vegetation structure and composition, or associated carbon fluxes.

These are just some of the trade-offs that need to be managed to achieve long term sustainability (we have not looked at water quality for example). Identifying these trade-offs is one thing. Contextualising and quantifying them is difficult, especially given the variability that exists both between sites and within sites at very small spatial scales. Basing management decisions or restrictions on large scale designations which are historically inadequately monitored is unlikely to succeed.

This gives policymakers a difficult and deeply unenvious role, with huge risk of unintended consequences, such as we are currently living with from the previous policy to drain moorland to improve livestock productivity. Other countries have suffered acutely from historic **'no burn'** policies. Section 5 details how in the US well-intentioned policies which stopped managed burning of ground vegetation from the 1930s onwards have directly led to severe declines in some bird species and the incredibly damaging forest wildfires of today. Heather uplands are also fire-prone ecosystems.

The problem of insufficient evidence, experience and knowledge about how to create the best possible environmental outcomes, amidst complicated trade-offs between carbon storage, emissions, and biodiversity, with potential impacts on the economic, social and cultural aspects that underpin the environmental management means we must focus on the broader picture. Carbon storage should not necessarily trump biodiversity; and economic social and cultural issues should not be forgotten.

The only way that we can envisage achieving the complex management needed to balance these trade-offs is for landowners to formulate estate-scale policies that allow for learning through adaptive management. Policy direction will be needed, but these are living, working landscapes and to achieve results we need the harness the knowledge and experience of those who live and work there.

We believe there is a shared desire to protect peat, enhance biodiversity and maintain living, working landscapes. We also believe grouse moor managers should seek to help achieve that by setting out their 'environmental offer' for the future, and that by working together they can make a difference at scale.

This approach is endorsed by England's 25 Year Environment Plan (Defra 2018) which sets **'restoring and protecting our peatlands'** as a key target, and recommends using the new concept of **'Nature Recovery Network(s)... (to help achieve) landscape-scale recovery for peatland'**.

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Appendices

Appendix 1

More detailed criticisms of studies used to estimate carbon stocks.

The carbon flux approach usually measures GHG emission over short periods (<5 years) and fails to quantify the effect of any longer-term shifts in environmental conditions (e.g. long-term climate and water table dynamics) and vegetation communities on carbon up-take or storage. The carbon stock approach does not account for C export in water. Furthermore, near-surface carbon stock assessments require careful interpretation because they often show rapid carbon accumulation due to lower decomposition rates at the peat surface. However, the same peat section could be losing carbon from the opposite (bottom) end of the profile – this usually happens in very dry or peat pipe eroding peatlands. Therefore, when using the carbon stock approach, researchers should ideally assess carbon accumulation throughout the entire peat core. Alternatively, when near-surface peat core sections are used, researchers should consider site conditions when interpreting their findings. For example, sites affected by deep drainage ditches or that have become very dry for other reasons, are likely to be losing carbon from lower down the peat profile (Young *et al.*, 2019). In such scenarios, one should not relate near-surface carbon accumulation rates to the rest of the peat body. Conversely, near-surface carbon accumulation data taken from wet sites can be and have been (i.e. Garnett *et al.*, 2000; Marrs *et al.*, 2019a; Heinemeyer *et al.*, 2018) generalised (with the knowledge caveats) to the entire peat body because such sites are unlikely to be losing carbon from deeper peat. In some studies of carbon storage/loss of peatlands and grouse moors, these cautions have not been accounted for.

Appendix 2

More detailed criticisms of studies used to estimate carbon fluxes from peatlands published in Evans *et al.* (2017).

- They did not distinguish between peatlands in the UK and Europe. For each peatland category studied, emissions from the UK and European peatlands were assessed together. Indeed, many of the data points used to produce GHG emission factors for UK peatlands were taken from outside the UK in Northwestern Europe. However, UK peatlands, especially in the uplands, are very different from European peatlands, which tend to be lowland fens or raised bogs. UK peatlands also have higher N deposition rates and different site histories than their European counterparts (e.g. less historical cultivation and contemporary grouse moor management). Unfortunately, the report does not state the number of non-UK sites used to calculate the GHG emissions for each peatland category studied. Such information would provide a valuable insight into the accuracy of the emission factor calculations.

Mountain hare in winter.

Freshly cut peat stacked to dry.

- Evans *et al.* (2017) split modified blanket bog into three categories **'Eroded modified bog'**, **'Heather dominated bog'** and **'Grass dominated bog'**. Each of these three categories was then further divided in terms of drainage (drained or undrained). However, due to data availability, Evans *et al.* (2017) did not attempt to split the modified bog categories by land management interventions, such as burning, mowing, grazing or non-intervention. The authors state themselves that land management factors are likely to have a strong influence on peatland emission factors. Thus, their analysis potentially hides large differences between near-natural peatlands and modified peatlands subject to different land management.

- During the calculation of GHG emissions arising from near-natural and rewetted peatlands (previously drained peatlands where a high water table has been restored), the authors omitted data from sites subject to seasonal or continuous water inundation (i.e. some of the wettest peatlands). This omission seems unjustified given near-natural and rewetted peatlands are likely to experience such inundation conditions for prolonged periods. More importantly, according to the authors, CH_4 emissions were extremely high at very wet sites. Consequently, by omitting data from sites subject to seasonal or continuous water inundation, the inventory is likely to have greatly underestimated GHG emissions from near-natural and rewetted peatlands (**TABLE 1**). In fact, the wider literature suggests that wetter peatland sites are likely to have positive emission factors due to high CH_4 emissions, particularly under warmer conditions (Abdalla *et al.*, 2016).

- The emission calculations did not take into account the influence of key factors such as topography (slope) and climate (rainfall). These factors have a strong influence on water table depth and thereby, carbon fluxes (e.g. Tiemeyer *et al.*, 2020).

- No location or environmental data (e.g. temperature, rainfall, peat depth, water-table-depth, type of vegetation) are provided for each of the observations used in the assessment. Therefore, it is difficult to ascertain how representative the GHG emission factors are of UK peatland resource, either overall or for each peat condition category they assessed (i.e. climatic and site conditions could have biased the observations, such as wetter/drier years causing higher/ lower methane emissions only for certain categories).

- Crucially, the report provides only subjective estimates of error for the emission factor calculations. Moreover, the data underpinning the emission factor calculations has not been published. Therefore, their accuracy cannot be properly scrutinised, e.g., by examining the number of studies and observations used to calculate each emission factor and calculating confidence intervals and standard errors for these estimates.

Managed burning on blanket bog vegetation, Hard Hill, Moor House, Upper Teesdale, UK. © www.ecologicalcontinuitytrust.org

Appendix 3

More detailed criticisms of studies used to estimate carbon stocks from published literature.

- It comes from only three studies with two of these being repeat assessments of the Hard Hill plots at Moor House (Garnett *et al.*, 2000; Heinemeyer *et al.*, 2018; Marrs *et al.*, 2019a). As previously mentioned, the Hard Hill plots may not be representative of the wider upland peatland resource (but they are representative of very high and wet blanket bogs); thus a burn frequency of 10 years, which showed the only significant reduction in C accumulation compared to the unburnt plots, is unsuitable due to plants being too small for a realistic rotation).

- Most studies measuring carbon accumulation rates for areas of upland peat subject to prescribed burning do not measure pyrogenic charcoal inputs and their detailed impact on peat bulk density and organic carbon content (Garnett *et al.*, 2000; Marrs *et al.*, 2019a).

- Every carbon stock study on upland peatland has been conducted by taking a low number of surface peat cores from within small experimental plots (Garnett *et al.*, 2000; Heinemeyer *et al.*, 2018; Marrs *et al.*, 2019a). Such an approach provides little information about how carbon stocks vary at the moorland scale due to factors such as water table depth, topography and vegetation type. Also, by only sampling the surface peat layers, this approach can fail to quantify potential carbon losses or gains towards the bottom of the peat profile (Young *et al.*, 2019).

Appendix 4

What are the knowledge gaps?

1. Dissolved organic carbon (DOC) and particulate organic carbon (POC) dynamics. In particular, we have little information about the impacts of vegetation and topography on DOC and POC export from upland peatlands. Furthermore, we do not understand what happens to DOC and POC once it leaves upland peatlands. Most carbon flux studies assume that DOC and POC are mostly oxidised after being exported, which would lead to the release of CO_2 into the atmosphere. However, the DOC and POC exported from peatlands could be transported and deposited in other habitats further downstream, which would lead to off-site carbon storage. Knowledge about the long-term fate of DOC and POC exports would help us to develop a more accurate picture of upland peatland GHG dynamics (Evans *et al.*, 2013; Palmer *et al.*, 2016)

2. Carbon stock and flux data (especially for CH_4) from a wider range of UK peatland types, especially from modified peatlands under different management regimes and near-natural peatlands. Studies collecting such data should also collect data on topography, climate and water table depth so that the influences of these factors on GHG emissions and storage can be properly investigated.

3. The contribution of different plant species to carbon stocks and fluxes within UK upland peatlands. The concept of peat-forming species is used frequently within the literature (see Gillingham *et al.*, 2016 and references therein), with *Sphagnum* and *Eriophorum* spp. purported to be the most important peat-formers. However, the science behind the 'peat-forming species' label is based on correlative evidence, such as higher amounts of *Sphagnum* fragments being found within peat cores during periods of rapid peat growth (Shepherd *et al.*, 2013; Gillingham *et al.*, 2016 and references therein). Therefore, we require experimental data on the contribution of different peatland species to GHG capture and storage. Such knowledge would provide clear targets for land managers concerned with reducing peatland GHG emissions.

4. To promote peatland species with the greatest GHG capture and storage potential, we need to understand the effect of different land management interventions on peatland plant species. We also need to determine whether the efficacy of land management interventions are consistent across different peatlands with different management histories, climates, water tables and baseline vegetation communities (i.e. to promote certain plant species, do we have to tailor management to the site?).

5. We need to determine whether upland areas of shallow peat overlying mineral soils were once areas of deep peat and, if so, whether these areas can be restored. If restoration is viable, such areas have huge GHG capture and storage potential and, due to the high carbon accumulation rates for initial peat formation, the GHG sink potential is much greater than for rewetting deeper peat on modified heather-dominated bogs (with the latter potentially resulting in high CH_4 emissions, e.g., Abdalla *et al.*, 2016).

6. Finally, there are many uncertainties about the synergies and trade-offs between management to promote GHG storage on peatlands and management for other equally important ecosystem services, such as flood alleviation, wildfire mitigation and upland biodiversity. For example, what are the effects of rewetting on peat water storage potential and downstream flood risk? A very high water table will likely limit water storage capacity and most likely lead to increased runoff. Also, what is the wildfire prevention and damage mitigation potential of different land management strategies, such as rewetting, cessation of vegetation management, burning and mowing? Alongside benefits to GHG capture and storage, it is claimed that a cessation of vegetation management and rewetting will prevent wildfire or mitigate the damage if one does ignite (with damage usually including large GHG emissions) (Baird *et al.*, 2019). However, these assumptions have not been tested within a UK upland context, which would consist of a scenario in which ignition potential and wildfire burn severity are measured on a rewetted bog with a high fuel load (the cessation of vegetation management will result in a build-up of burnable biomass, e.g., Alday *et al.*, 2015). Finally, impacts of thick brash layers left after mowing or removal of nutrients with the brash could have fundamental impacts on water quality and plant growth.

Appendix 5

Carbon storage/GHG peatland area digitising

Method

Original maps of the outputs from the 2010 Natural England report (NE257) *England's Peatlands: carbon storage and greenhouse gases* were unavailable for our use. The maps of Carbon Storage and Greenhouse Gas Emission (Map 8 and 9, pages 22 and 28 respectively) were image captured from the PDF at a high zoom level using the Foxit Reader 9.5 SnapShot tool to obtain an image of sufficient resolution. These image-captured maps were georeferenced to the UK Ordnance Survey base map in QGIS 3.6 using the Georeferencer Plugin.

This resulted in some positional anomalies when comparing the georeferencing against the UK coastline and government region boundaries. Further alignment was necessary using a Thin Plate Spline (TPS) algorithm. Identifiable areas on the Carbon storage and Greenhouse gas emission maps were matched to topographic forms (moors, meres, valleys etc.) identified on the Ordnance Survey base map and through visual comparison to the British Geological Survey UK Soils map - using the online map viewer (mapapps2.bgs.ac.uk/ukso/home.html).

Once these maps were successfully georeferenced in ArcMAP 10.6 they were overlaid with the boundary outline of the land ownership of members of the Moorland Association (dated 2013) and the Rural Payments Agency's Moorland Line of England (magic.defra.gov.uk/Datasets/Dataset_Download_MoorlandLine.htm).

Each feature of the data ranges from the maps was digitised to recreate a digital vector version that approximated the same areas illustrated in the report's maps. Only the ranges, or parts thereof, that where within or overlapped the Moorland Association boundary, the Moorland Line of England or were features considered upland areas or grouse moors in Northern England were included.

The area for each digitised Carbon storage and Greenhouse gas emission data range was calculated in order to arrive at a figure of carbon storage and greenhouse gas emission associated with moorland management.

Known issues

The accuracy of the digitised features was limited due to the simplified outlines on the maps in the original report. In addition, the maps of Carbon storage and Greenhouse gas emission areas appear to include a noticeable boundary of unknown thickness. Therefore, the area digitised, and the figures calculated from them will be larger than the original data from the NE257 report.

The original ranges for Map 9 *Estimated carbon storage* did not specify an upper limit (**'2000 or more tonnes C per hectare'**). We set an upper limit of 3500 tonnes C per hectare for the purposes of this work being the approximate value when using the proportional increase in other range values.

The 2017 Department for Business, Energy & Industrial Strategy report *Implementation of an Emissions Inventory for UK Peatlands* was not used. This was due to the lack of mapped data available in this report. As the authors highlight in the text, this is a known shortcoming in how useful their latest (2017) findings will be:

'Finally, it is important to note that the peat mapping datasets used in the project came from multiple sources, and most are subject to licencing restrictions. This is likely to significantly limit wider use of the 'unified' peat layer created during the project. If the final peat map could be made accessible as 'open data' to other organisations and projects this would greatly enhance its future value for policy, land-management and research.'

Red grouse on moorland in UK.

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Report 3:

Constructive criticism of the IUCN "Burning and Peatlands" position statement

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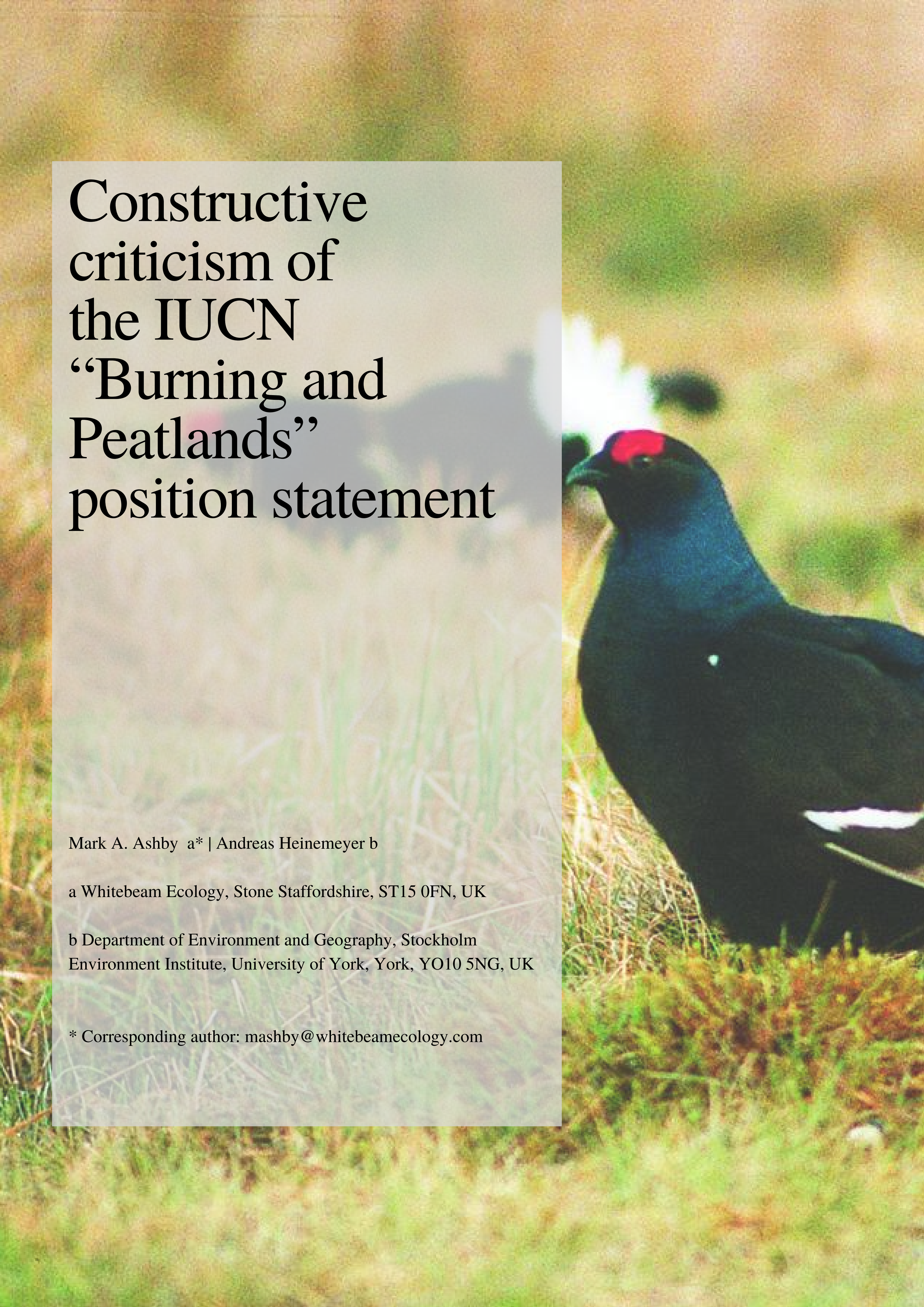

Constructive criticism of the IUCN “Burning and Peatlands” position statement

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Constructive criticism of the IUCN "Burning and Peatlands" position statement

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On the 31st of March, the International Union for Conservation of Nature (IUCN) UK Peatland Programme published an updated position statement about "*Burning and Peatlands*" (IUCN 2020). We strongly agree with several of the statements made within this document, especially those outlined within the "*Areas for further consideration and research*" section. However, the document is littered with unverified assertions and scientific inaccuracies, which may reflect the fact that it has not been peer-reviewed by the peatland research community. Given that this document is published by one of the UK's most prominent peatland conservation organisations, it may be used to inform upland land use policy. Therefore, for the benefit of policymakers, the erroneous statements made within this document must be challenged and corrected. To this end, within Box 1, we have provided a point-by-point critical commentary of the IUCN (2020) "*Burning and Peatlands*" document. For balance, we also highlight points of agreement. Our aim in producing this commentary is to move towards an unbiased and evidence-based position statement about burning on UK peatlands.

In addition to our specific criticisms outlined in Box 1, we would also like to highlight three broader but crucially important points for policymakers to consider when evaluating the impact of prescribed burning on UK peatlands.

<u>The first point for policymakers to consider is</u> that, to date, no study has assessed burning impacts using a **real-world approach**[1]. In short, this means that the impacts of prescribed burning on UK peatlands have not been accurately assessed. For example, in the real world,

[1] The Peatland-ES-UK study will qualify as using a real-world approach once it has measured impacts over a full management cycle of 20+ years (Heinemeyer *et al*. 2019).

gamekeepers burn areas of mature heather to create a mosaic of differently aged heather patches at the moorland scale (Tharme *et al.* 2001). Specifically, gamekeepers want young stands of heather with fresh, nutritious shoots for adults, older stands of heather for cover, and short open stands of heather containing a greater abundance of insect prey for chicks (Miller, Jenkins & Watson 1966; Palmer & Bacon 2001; Buchanan *et al.* 2006). The aim is to burn multiple patches of heather across a moorland during each burning season. However, due to the vagaries of weather and logistics, the number of burning patches per season is highly variable (Allen *et al.* 2016). The size and shape of individual burns are also highly variable, but they are usually no more than 30 x 100 metres (*ibid*). Patches are re-burnt as soon as they become dominated by tall and 'leggy' heather, which can take between 10 and >25 years depending on climate (Glaves *et al.* 2013; Thacker, Yallop & Clutterbuck 2014; Alday *et al.* 2015). Prescribed burning is also applied within a wide range of environmental contexts because each peatland differs in terms of climate, peat depth, water table depth, slope, vegetation composition, the level of drainage, the amount of grazing, levels of atmospheric pollution and management history (Noble *et al.* 2018b; Heinemeyer *et al.* 2019b).

So how are prescribed burning impacts investigated within the scientific literature? Well, studies generally measure burning impacts at the plot (rather than moorland) scale, with experimental plots being uniform in size and much smaller than the prescribed burns created by gamekeepers. Experimental plots are also burnt on strict rotations, rather than when the heather is tall and 'leggy'. Furthermore, pre-burn measurements are rarely taken, and post-burn measurements are usually only taken for a couple of years at the start of a burning rotation or during a single year across multiple burning rotations. Finally, studies often ignore how environmental conditions (e.g. water table depth, slope, peat depth, vegetation composition) vary within or between study sites, and how such variation influences fire behaviour.

Besides being unrealistic, the short-term approach used to study prescribed burning is also biased towards finding adverse effects. For example, prescribed burning is a form of habitat disturbance and all forms of habitat disturbance (natural or anthropogenic) cause short-term ecological damage irrespective of the long-term ecological impacts. Thus, if a similar approach were used to assess other uncontroversial disturbance-based land management techniques (e.g. hedge laying, coppicing, mowing, grazing, rewetting), the results of such studies would undoubtedly be negative (see, for example, the difference between rewetting impacts after one year versus after four years in Holden *et al.* 2017).

The second point for policymakers to consider is that the results of many burning studies are currently unreliable because they use **experimental designs that are unable to detect causal relationships and/or make significant statistical errors** (for a discussion of this issue, see Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b). For example, several studies confound treatment (e.g. burnt versus unburnt) with study site and fail to control for this during data analysis (*ibid*). The results of such studies are unreliable because any observed impacts (i.e. differences) cannot be solely attributed to burning management. Several burning studies also commit pseudoreplication because they fail to account for data structure during analysis (*ibid*). By doing this, such studies artificially inflate treatment-level sample sizes, which means the significance values reported are likely to be much too low and the results cannot be generalised (Davies & Gray 2015). Given these issues, we recommend that any future assessment of the prescribed burning evidence should weight conclusions according to the methodological strength (experimental design and data analysis) of each study, with studies being rejected from consideration if they report unreliable results.

The third and final point for policymakers to consider relates to how the **precautionary principle** is applied to different forms of peatland management. It is suggested within the IUCN (2020) "*Burning and Peatlands*" document that: "*Where there is uncertainty around the benefits of burning for peatland restoration, the precautionary principle should be applied and burning avoided*". We do not object to the IUCN applying the precautionary principle to prescribed burning. However, we *do* object to the fact that it is not applied equally to other forms of peatland management that have not undergone a full environmental cost-benefit-analysis (e.g. rewetting or cutting). For example, compared to burning, we know even less about the impact of rewetting on peatland ecosystem services (e.g. greenhouse gas emissions; water quality; flood mitigation). Even so, the small amount of evidence we do have suggests that, by raising water tables, rewetting *could* lead to increased methane emissions, increased saturated overland flow and reduced water quality (e.g. Holden & Burt 2003; Abdalla *et al.* 2016; Peacock *et al.* 2018). Yet, surprisingly, nowhere in the IUCN (2020) "*Burning and Peatlands*" document is it suggested that the precautionary principle should be applied to rewetting. Instead, the document repeatedly advocates the use of rewetting as a way of reducing wildfire risk (*ibid*), which, incidentally, there is no evidence for (see Box 1).

Our personal view is that the application of the precautionary principle to burning on peatlands is the wrong approach for two reasons. Firstly, the lack of a transparent and objective decision-making process means the precautionary principle is difficult to apply in practice (Peterson 2007; Vlek 2010). Secondly, and more importantly, there is a growing body of evidence which suggests that, in specific contexts (e.g. flat areas with water tables at or near the soil surface), burning only causes minimal short-term impacts to UK peatlands (Lee *et al.* 2013; Taylor 2015; Grau-Andrés, Gray & Davies 2017; Grau-Andrés *et al.* 2018; Milligan *et al.* 2018; Noble *et al.* 2018a; Grau-Andrés *et al.* 2019a; Grau-Andrés *et al.* 2019b; Heinemeyer *et al.* 2019a; Marrs *et al.* 2019a). Furthermore, when negative impacts are reported, they are often for short-term effects or differences that are too small to be ecologically significant (Noble *et al.* 2018b; Grau-Andrés *et al.* 2019b; Noble *et al.* 2019a; Noble *et al.* 2019b).

Instead, given that the impacts of burning are likely to be site-specific (Heinemeyer *et al.* 2019b), upland land managers should adopt an adaptive management approach to prescribed burning. The fundamental tenet of adaptive management is to monitor management interventions and use the results to inform future actions (e.g. by halting any interventions that are found to be damaging) (Holling 1978). We endorse this ‘learn by doing’ approach because it (i) allows management to continue as long as landowners monitor the environmental impacts of their interventions; (ii) encourages landowners to adopt a more cautious approach to management; (iii) ensures more environmentally sensitive management techniques are adopted; and, (iv) contributes to the evidence base. We also recommend that any future research adopts a joint-up and catchment-scale approach in which management interventions are compared across several actively managed sites covering a broad range of environmental conditions (the Peatland-ES-UK study is such an example: Heinemeyer *et al.* 2019b).

Box 1. The statements made within the IUCN (2020) "*Burning and Peatlands*" position statement (black text) and our responses to those statements (blue text).	
Statement 1	There is a consensus within the literature that burning is, or has the potential to be, damaging to peatlands. It is well-established that burning can degrade bog habitats, leading to reductions or loss of key bog species (plants and animals), development of micro-erosion networks, increased tussock formation and increased dominance of non-peat forming vegetation such as heathland species (e.g. heather *Calluna vulgaris* and the moss *Hypnum jutlandicum*). There is no such consensus within the literature. Several recent reviews and commentary papers demonstrate that the overall effect of burning on peatlands is unclear due to insufficient, contradictory or unreliable evidence (Davies *et al.* 2016b; Harper *et al.* 2018; Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b). However, we do agree that, like any other disturbance-based land management intervention (e.g. grazing, mowing, hedge laying, coppicing), burning has "*the potential to be*" damaging, but only when applied in the wrong spatial, temporal or environmental context. For example, prescribed burning causes only very minimal short-term damage to flat areas of peatland with water tables at or near the soil surface (Davies *et al.* 2010b; Lee *et al.* 2013; Kettridge *et al.* 2015; Taylor 2015; Grau-Andrés, Gray & Davies 2017; Grau-Andrés *et al.* 2018; Milligan *et al.* 2018; Noble *et al.* 2018a; Grau-Andrés *et al.* 2019a; Grau-Andrés *et al.* 2019b; Heinemeyer *et al.* 2019b; Marrs *et al.* 2019a). We assert that the negative view of prescribed burning largely comes from the use of short-term results (up to two years post-burn) to infer long-term impacts over the full 15-25+ year burning rotation. We also disagree that it is "*well established*" that burning leads to "*reductions or loss of key bog species (plants and animals)*" and the "*increased dominance of non-peat forming vegetation*". All the available evidence suggests that burnt areas of blanket bog support similar levels of *Sphagnum* and *Eriophorum* spp. (key plant species) to comparable unburnt or not recently burnt areas (Lee *et al.* 2013; Milligan *et al.* 2018; Noble *et al.* 2018a; Noble *et al.* 2018b; Whitehead & Baines 2018; Grau-Andrés *et al.* 2019a). Admittedly, the *Sphagnum* data is largely an assessment of burning impacts on the most abundant species: *Sphagnum capillifolium* (this research gap requires urgent attention) (*ibid*). Nevertheless, our point still stands. As for animal species, most of the work done thus far has been on upland birds and invertebrates (terrestrial and aquatic), with burning seeming to benefit some species and harm others (Harper *et al.* 2018). Importantly, we have no data about the impacts of prescribed burning on mammals, reptiles, or amphibians that inhabit peatland ecosystems in the UK (Harper *et al.* 2018). Furthermore, to claim that burning increases the abundance of non-peat-forming species, such as *C. vulgaris* and *Hypnum jutlandicum*, contradicts most of the evidence base (Lee *et al.* 2013; Alday *et al.* 2015; Milligan *et al.* 2018; Whitehead & Baines 2018; Grau-Andrés *et al.* 2019a; Heinemeyer *et al.* 2019a,b). Of greater concern is that the 'peat-forming' label is not supported by robust experimental evidence. Rather, it is based on circumstantial evidence, such as greater quantities of *Sphagnum* fragments being found within peat cores during periods of rapid peat growth (Shepherd *et al.* 2013; Gillingham, Stewart & Binney 2016 and references therein). But is this cause or effect? In other words, were these periods of rapid peat growth due to a greater abundance of *Sphagnum* or did periods of rapid peat accumulation coincide with conditions favourable to *Sphagnum* growth (e.g. very wet and acidic)? Contemporary evidence suggests that the relationship between *Sphagnum* abundance and peat (carbon) accumulation is unclear (Garnett, Ineson & Stevenson 2000; Marrs *et al.* 2019a; Piilo *et al.* 2019), and there are multiple paleoecology studies in which peat core sections are dominated by non-*Sphagnum* plant fragments (Fyfe, Brown & Rippon 2003; Fyfe & Woodbridge 2012; Shepherd *et al.* 2013; Gillingham, Stewart & Binney 2016; Fyfe *et al.* 2018). A review published by Natural England clearly states that any plant species (including *C. vulgaris*) can form peat in the right conditions (Shepherd *et al.* 2013; Gillingham, Stewart & Binney 2016). Indeed, from a scientific perspective, it is the hydrological (high water table) and environmental (low pH) conditions that determine whether peat forms, regardless of species composition (Gillingham, Stewart & Binney 2016). However, by retaining water and reducing soil pH, *Sphagnum* spp. may facilitate such peat-forming conditions where they are otherwise limiting (Gorham 1957; Clymo *et al.* 1984; van Breemen 1995; University of Leeds Peat Club: *et al.* 2017). Finally, as far as we are aware, there is no published evidence documenting the relationship between prescribed burning and the "*development of micro-erosion networks*" and "*increased tussock formation*" within peatlands. This seems to be pure speculation by the IUCN (2020) unless it is based on yet to be published evidence.

Statement 2	The impacts of fire on bog habitat, and particularly the main peat forming *Sphagnum* species' ability to recover, depends on the frequency and intensity of the burn along with other factors such as prevailing soil water levels, intensity of livestock trampling, climate, altitude and the starting condition of the peatland. As our comments on the previous statement indicate, there is no evidence on the peat-forming capabilities of different peatland plant species. Thus, we have no idea whether *Sphagnum* is the main peat-forming species. What we do know is that peat formation is context and site-specific, and is primarily driven by the environmental and edaphic conditions (Shepherd *et al.* 2013; Gillingham, Stewart & Binney 2016; University of Leeds Peat Club: *et al.* 2017). However, we do agree that burning impacts on *Sphagnum* spp. are dependent on the "*frequency and intensity of the burn*" and the "*prevailing soil water levels*". For example, burns on wetter bogs cause only minimal short-term damage (Taylor 2015; Grau-Andrés, Gray & Davies 2017; Grau-Andrés *et al.* 2018; Grau-Andrés *et al.* 2019a). Conversely, there is currently little evidence for the "*intensity of livestock trampling"* and "*the starting condition of the peatland*" modifying the impacts of prescribed burning on *Sphagnum* spp.
Statement 3	Rotational burning on peatland leads to drier vegetation communities (wet heath and dry heath communities) or a shift towards their dominance (e.g. of *Molinia*) (Bruneau & Johnson, 2014). This is associated with changes to the ecosystem (e.g. increased erosion rates and reduced availability of soil moisture) that can result in significant adverse impact on peatland biodiversity, carbon emissions, drinking water quality and flood management (Brown et al., 2014). We disagree. Current evidence suggests that, compared to unburnt or not recently burnt areas, burnt areas of blanket bog can support: a similar abundance of wetland plants, such as *Sphagnum* and *Eriophorum* spp; and, a lower abundance of plants indicative of drier conditions (e.g. *C. vulgaris*), at least in the short-term (Lee *et al.* 2013; Milligan *et al.* 2018; Noble *et al.* 2018a; Noble *et al.* 2018b; Whitehead & Baines 2018; Grau-Andrés *et al.* 2019a; Heinemeyer *et al.* 2019b). We are also unaware of any study which shows that prescribed burning on blanket bog leads directly to the dominance of *Molinia caerulea*. (note: often there is no distinction made between this and much hotter and more sever wildfires burning into the peat and *Molinia* increase on wet heath and acid grasslands, often on non-peat soils, which has been generalised in summary tables; see Tucker, 2003). Furthermore, the impacts of burning on peatland ecosystem services remain unclear due to insufficient, contradictory or unreliable evidence (Davies *et al.* 2016b; Harper *et al.* 2018; Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b). We would also like to point out that the EMBER report and associated peer-reviewed studies are unreliable in their current form (Brown *et al.* 2013; Brown, Holden & Palmer 2014; Holden *et al.* 2014; Brown *et al.* 2015; Holden *et al.* 2015); since, it has been shown to be methodologically and scientifically flawed (Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b; Brown & Holden 2019). Therefore, these studies should not be cited to support the claim that burning has a "*significant adverse impact on peatland biodiversity, carbon emissions, drinking water quality and flood management*".
Statement 4	The majority of UK peatlands are in a degraded state as a result of various factors including drainage, burning, atmospheric pollution and high livestock numbers (JNCC, 2011; Artz et al., 2019). This statement lacks nuance because it combines peatlands that are 'unfavourable-recovering' with those that are just 'unfavourable'. Using blanket bog within Special Areas of Conservation (SAC) in the UK as an example, 45%, 14% and 39% are in 'Favourable', Unfavourable-recovering' and 'Unfavourable' condition, respectively (2% of this habitat has been destroyed) (JNCC 2006). Thus, the majority of blanket bogs in SACs (59%) are on a positive ecological trajectory (i.e. bogs that are 'Favourable' or 'Unfavourable-recovering') (*ibid*). In fact, the majority of all peatland types (e.g. lowland fens and marshes, upland fends and marshes, lowland raised bogs) are on a positive ecological trajectory (*ibid*). Perhaps more importantly, the criteria used to assess peatland condition is made up of arbitrary pass-fail criteria that do not measure important ecosystem functions, such as water table depth and peat accumulation (JNCC 2009). Thus, in truth, we have no idea about how much of the UK peatland resource is degraded. It could be that some bogs currently classed as unfavourable (i.e. degraded) are actually in good ecological condition (and vice versa), that is, they have water tables at or near the bog surface and are actively accumulating peat.

	Compared to intact peatlands, degraded peatlands generally show: Apart from water table depth, the points below do not describe direct measurements of peatland degradation. To determine whether a peatland is degraded, we need robust empirical measurements of peat accumulation, water table depth and other important ecosystem services, such as water quality. If such measurements indicate that a peatland has a low water table, is losing (rather than accumulating) peat and has low levels of relevant ecosystem service provision, then it should be classed as degraded. However, conflicting outcomes should be expected for different ecosystem services under different land management scenarios (e.g. Bennett, Peterson & Gordon 2009; Power 2010). - a higher proportion of dwarf shrub and graminoid (grasses and sedges) abundance - reduced *Sphagnum* bog moss abundance and diversity of typical bog species - vegetation structural changes such as loss of bog moss hummocks and pools - greater development of tussock and micro-erosion microtopography - denser, more degraded surface peat - a lower water table.
Statement 5	One of the sources of confusion around the impact of management activity on peatland is the misunderstanding as to what constitutes degraded and favourable condition, and failure to assess management trajectories. This is also reflected in some academic studies, which have inconsistent approaches to describing peatland vegetation, the state of peatland or the management objectives for the peatland. Indeed, many published journal papers do not adequately describe, or take account of, the type or current condition of the peatland under investigation. It is our view that the confusion surrounding degraded and favourable condition is due to the fact they are not objective and evidence-based criteria (JNCC 2009). Indeed, they are arbitrary criteria centred around 'typical' vegetation communities (see, for example, the criticisms in Davies *et al.* 2016b). If the most important aspect of a peatland is the peat itself, then a simple and objective definition of favourable condition could be whether a peatland is accumulating (rather than losing) peat. However, such a definition requires that accurate peat accumulation data be collected from across the peatland site being designated, which may be cost and time prohibitive. An alternative approach would be to conduct robust experimental research into proxies of positive peat accumulation that can be rapidly assessed in the field. Such an assessment becomes much more complicated when other key ecosystem services factors are included (e.g. net GHG emissions; water quality; biodiversity). To our knowledge, such a detailed assessment is yet to be done. Given the subjective and unscientific nature of the current peatland condition criteria, we also question why it is relevant if a study adequately describes or takes account of the "*current condition of the peatland under investigation*". What we need to know is how burning effects the functioning of the peatland(s) under investigation relative to baseline conditions. Such studies must also adequately control for environmental and ecological differences between treatment plots and study sites (Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b).
Statement 6	The majority of peatland restoration projects across the UK are able to achieve relatively rapid development vegetation communities typical of blanket bog (within c.5-10 years) through hydrological restoration. Re-wetting a peatland tends to be sufficient that any undesirable vegetation, such as dominant heather cover, dies back naturally to be replaced by Sphagnum-dominated conditions associated with healthy peatbog habitat (Cris, 2011). Effective restoration of peatlands has been widely achieved across Scotland without the need for burning; for example, there are over 200 Peatland Action restoration sites in Scotland that are delivering good practice restoration and have not required burning as part of this process. The IUCN document (IUCN, 2020) assumes that peatland rewetting is a net good, which is exemplified by 'Statement 6'. We question this assumption. For example, rewetting (usually by ditch blocking) aims to saturate peatland soils by raising the water table so that it is at or near the soil surface. However, when rain falls on a saturated peatland, the rainwater will either pond on flat areas or, on slopes, flow to lower ground under the force of gravity across the peat surface (Holden & Burt 2003; Acreman & Holden 2013). The latter process is called

	saturated overland flow and it if it occurs it will increase the volume and speed of water running downhill into river catchments (e.g. Holden & Burt 2003). However, by increasing surface roughness, surface vegetation (e.g. *Sphagnum* spp.) may help to reduce the speed of saturated overland flow (Holden *et al.* 2008), but the extent to which it does is likely to decline as the vegetation itself becomes saturated. Thus, a rewetted and saturated peatland *could* exacerbate (rather than mitigate) downstream flooding. If so, then peatland rewetting may not be the best land management strategy to employ within flood-prone catchments, especially given the projected increases rainfall intensity across the UK (Kendon *et al.* 2019; Met Office 2019). Peatland rewetting may also have a negative impact on climate change mitigation because peatlands with high water tables also emit large amounts of methane, particularly if combined with increasing temperatures (Abdalla *et al.* 2016), with methane having a greater warming potential than carbon dioxide. Thus, if the carbon captured and stored by rewetted peatlands is less than the methane emitted, then such a peatland will be a net emitter of greenhouse gases (GHGs). It is important to note that the potential for rewetting to lead to negative impacts on peatland ecosystem services does not mean traditional peatland drainage (e.g. gripping) is the solution (e.g. Holden *et al.* 2006)[2]. However, it does mean that, based on the current evidence, we should not assume that rewetting only has positive effects on peatland ecosystem services. We also need to consider how the impacts of rewetting may vary according to future climate scenarios (e.g. warmer summers and wetter and warmer winters). We also question two further assumptions outlined in this statement. Firstly, the assumption that a peatland dominated by *C. vulgaris* is undesirable and that "*Sphagnum-dominated conditions*" are "*associated with healthy peatbog habitat*". As we have previously mentioned, there is no robust evidence for an association between type of peatland vegetation and peat accumulation (i.e. ecological function): it is the hydrological (high water table), environmental (low pH) and climatic (lower temperatures) conditions that determine peat accumulation rates, and not vegetation composition (Gillingham, Stewart & Binney 2016). Therefore, peatlands should not be classed as 'undesirable' because they fail to pass the arbitrary vegetation composition criteria outlined within the peatland condition assessment (JNCC 2009). Secondly, we question the assumption that rewetting reduces *C. vulgaris* dominance. As far as we are aware, there is no empirical evidence to support this claim. In fact, wet and undrained sites can remain dominated by *C. vulgaris* even after 90+ years post-management (Lee *et al.* 2013; Alday *et al.* 2015). Given the lack of evidence, we find it strange that the IUCN document provides a supportive citation for this claim (e.g. Cris *et al.* 2011). However, on closer inspection, the supporting citation contains only an unreferenced statement about rewetting reducing *C. vulgaris* dominance (*ibid*).
Statement 7	Burning has been advocated by some land managers as a tool in peatland restoration to remove rank, leggy heather (*Calluna vulgaris*) (Uplands Management Group, 2017)). Burning carries a risk of causing more serious damage, further degradation and compromising the onset of peatland recovery. The substantial plant biomass load and the often dry nature of the underlying peat beneath the heather, are susceptible to uncontrolled or "hot burns" that can damage peat forming Sphagnum species, peatland seedbanks, underlying peat soil and lower the water table for a period of several years. The role of "cool burns" as a means of reducing risks has not been assessed in the peer reviewed scientific literature and in view of the large number of successful peatland restoration schemes that do not use any form of burning, the need for a "cool burn" on peatlands is untested. So called "hot" and "cool burns" are an untested management tool with no certainty as to whether differences can be controlled and no robust studies on the relative impacts. Successful restoration of blanket bog on numerous upland sites around the UK, without the use of muirburn or any other form of burning, demonstrates that burning is not a necessary tool for peatland restoration. It is undeniable that burning removes rank, leggy *C. vulgaris*, at least in the short-term (Alday *et al.* 2015; Whitehead & Baines 2018). However, due to the lack of evidence, it is unclear whether it promotes *Sphagnum* development in the long-term. Nevertheless, data from the Hard Hill experiment indicates that, over a 60-year period, repeatedly burnt and unburnt plots support similar levels of *Sphagnum* (Milligan *et al.* 2018; Noble *et al.* 2018a). At the very least, this suggests that prescribed burning does not have a negative impact on the long-term survival of *Sphagnum* populations (note – this mainly applies to *Sphagnum capillifolium*).
Statement 8	A number of recent studies have presented misleading conclusions resulting in the mistaken interpretation that burning is beneficial for peatland conservation and restoration (e.g. Marrs

[2] It could be that intermediate water tables are the key to increasing flood and climate change mitigation potential within peatlands.

et al., 2019; Heinemeyer et al., 2018; Milligan et al., 2018). Common factors presented in academic literature that can lead to confusion include:

This is untrue. Whether the cited studies present misleading results is currently unresolved (Heinemeyer *et al.* 2018; Milligan *et al.* 2018; Baird *et al.* 2019; Evans *et al.* 2019; Heinemeyer *et al.* 2019a; Marrs *et al.* 2019a; Marrs *et al.* 2019b). Failing to mention this key point is a glaring omission. Furthermore, Heinemeyer et al. (2018) did not assess the impact of burning on either peatland conservation or restoration. Also, we are assuming[3] that the criticism of Heinemeyer *et al.* (2018) and Marrs *et al.* (2019a) is primarily based on the model produced by Young *et al.* (2019). If so, it is important to note that the model used by Young *et al.* (2019) is unvalidated, unspecified and only relates to the impact of deep drainage, and does include any burning management or C-cycle processes (e.g. the impact of charcoal). Consequently, it cannot be used to criticise studies looking at C accumulation on rotationally burnt areas of blanket bog with minimal drainage impacts. In fact, a recent study by Flanagan *et al.* (2020) supports the findings of Heinemeyer *et al.* (2018) and Marrs *et al.* (2019a) that low-severity fires (i.e. prescribed burns) can have result in high C accumulation rates on peatlands. Specifically, Flanagan *et al.* (2020) found that the positive impact of low-severity fires on carbon accumulation was mediated by charcoal production and addition to the peat profile and reduced decomposition, with both processes being hypothesised in Heinemeyer *et al.* (2018) and Heinemeyer *et al.* (2019b).

a) Inconsistent approaches to the definition of peatland vegetation and its condition; of particular concern are studies that do not consider whether the vegetation recorded is typical of bog habitat or representative of more dry habitats. (It is overly simplistic to report only on the abundance of moss species or generic Sphagnum species, as these can also be associated with poor-fen or dry heath conditions rather than bog formation).

   This is not a valid criticism. Excluding species of conservation concern, peatland vegetation is, in a sense, irrelevant – ecosystem functioning is what should be important. In the case of peatland, this means peat building, which is primarily driven by hydrological (high water table), environmental (low pH) and climatic (lower temperatures) conditions (Gillingham, Stewart & Binney 2016). However, land managers may also want to enhance other ecosystem services (e.g. flood, climate change and wildfire mitigation, water quality, or avian biodiversity).

b) Inadequate methodologies to make a full assessment of baseline conditions or summary of any potential confounding effects. Existing environmental and management factors such as drainage, subsidence, grazing pressure, historic burning regime, surrounding land use pressures such as forestry plantations and atmospheric pollution can all impact on study sites. To fully consider the effects of fire on peatland carbon balance a full net balance needs to be conducted to allow for comparison between burned and unburned sites.

   Yes, but, barring a few studies (Heinemeyer *et al.* 2019b), this applies to the entire evidence base, especially the four peer-reviewed studies published as part of the EMBER report (Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b; Brown & Holden 2019). So why does this document only mention Heinemeyer *et al.* (2018), Milligan *et al.* (2018) and Marrs *et al.* (2019a) in this respect? An unbiased assessment would rightly highlight, as we do above, that no study to date has examined burning as it is applied in the real world using a study design robust enough to detect causal relationships (one exception is the Peatland-ES-UK, which was set up specifically to address this issue: Heinemeyer *et al.* 2019b).

c) Failure to consider the impact of land management regimes in relation to trajectory for a habitat. Simply comparing burned areas with unburned areas is unhelpful if the aims of the site are to restore functioning peatland habitat. Burning of a heavily degraded heather dominated peatland may simply produce a constrained, degraded peatland state, retaining vegetation associated with drier conditions, such as Calluna that could limit further recovery towards the near natural state.

   The first sentence of this specific criticism is unclear. We disagree that comparing burnt to unburnt areas is unhelpful. Such comparisons would be extremely helpful if they were made using a randomised before-after-control-impact (BACI) design in

[3] A supporting citation would clarify this.

	which important ecological functions (e.g. peat accumulation and water table depth) were measured over several management cycles. Also, note the word "*may*" in the last sentence of this statement. It is key, because, due to the lack of empirical evidence, we actually have no idea whether the burning of "*a heavily degraded heather dominated peatland may simply produce a constrained, degraded peatland state, retaining vegetation associated with drier conditions, such as Calluna that could limit further recovery towards the near natural state*". d) Comparing the burned to unburned state can produce data that shows a change in vegetation including an increase in Sphagnum species. However, in burned plots, consideration should be given to the type of Sphagnum species and whether these are typical of bogs, as well as the likelihood of reversion of the degraded peatland back towards abundant heather. Agreed. But again, this applies to the entire evidence base. *Sphagnum* species are often grouped by researchers because the abundance of most species is low, which inhibits statistical analysis. Also, individual *Sphagnum* species may not be the most sensitive habitat indicators because i) of their wide environmental tolerances (c.f. Plates 1i - 1viii in Daniels & Eddy 1985); and, ii) we lack data on their contribution to important peatland functions (e.g. peat and carbon accumulation). e) A distinction also needs to be made between studies of a single burn, compared with frequent managed burns on a cycle of 30 years or less. The latter can give rise to substantial cumulative impact due to long recovery times of particular blanket bog Sphagnum species from damage through burning (Noble et al., 2019). Agreed. Every study should consider aspects of fire ecology to provide relevant and useful guidance to land managers and enhance scientific understanding (Davies *et al.* 2010b; Davies *et al.* 2016b; Grau-Andrés *et al.* 2018; Grau-Andrés *et al.* 2019a). It is also worth noting that Noble *et al.* (2019b) used a correlative space-for-time approach in which baseline vegetation data was not collected (see 'Statement 8b'). Furthermore, Noble *et al.* (2019b) provide no data on the environmental and management differences between treatment plots (see 'Statement 8b').
Statement 9	In addition to the failings to accurately describe peatland vegetation and condition described above, studies can also lead to the mistaken view that burning is inconsequential or even beneficial for both the ecology and the carbon store of a bog if they do not fully account for: - the negative long-term carbon trends associated with atypical plant species abundance There is no robust causal evidence for the impact of atypical plant species abundance on long-term carbon accumulation or storage within UK peatlands. - damaged state of the acrotelm (thin living surface layer of peat-forming vegetation) Again, no evidence is provided for the state of the acrotelm (definition and impacts). A complete assessment of the acrotelm would be extremely complex because it would have to consider its physical (e.g. bulk density), chemical (e.g. organic carbon content) and biological (e.g. microbial communities) properties. Currently, we only consistently record the physical and chemical properties of the acrotelm. However, as specific physical and chemical properties relate to multiple factors (e.g. management, climate and vegetation) (Morton & Heinemeyer 2019), it can be difficult to determine the state of the acrotelm. Conversely, soil infiltration measurements can provide useful information on the hydrological state of the acrotelm for a specific moment in time. Furthermore, we would argue that short-term management impacts to the acrotelm, such as exposed peat surfaces after a prescribed burn, should not be used to infer long-term impacts. Finally, there is no evidence to support the 'peat-forming' label (peat formation is driven by the environmental conditions - see our comment on Statement 1) and the definition of the acrotelm used by the IUCN (2020) differs from the standard definition first described by Ingram (1978): • "*The surface layer of a mire soil, differing from the subjacent layer in the nature, greater range or more abrupt variation of its physical properties and biological attributes and in function the principal site of matter and energy exchange in the mire ecosystem*". • "*We consider the lower boundary to be the level above which the water conditions*

	and degree of decomposition vary rapidly, while below this level they either remain constant or vary slightly". - consequent impacts on the catotelm (permanently waterlogged peat store under the acrotelm). Past changes to deep C stores can also give rise to misleading conclusions about the previous rates of C accumulation. Negative impacts on the catotelm are only really achieved by deep drainage ditches or gulleys. Standard moorland drains (i.e. grips) usually only lead to small increases in water table depth (generally only a few centimetres) that only extend a couple of metres either side of the ditch (e.g. Luscomb et al., 2016; Wilson et al., 2010; Holden et al., 2004). We are assuming[4] the second sentence in this statement is based on the model published by Young *et al.* (2019). As previously noted, this is an unvalidated and unspecified drainage-based model (*ibid*). Therefore, the results of this model cannot be applied to burning management. A far better modelling study (with model validation) to consult on C storage impacts has previously been published by Heinemeyer and Swindles (2018). - loss of microtopography and overall reduction in environmental resilience. The first part of this statement is untrue. For example, several studies show that, relative to unburnt controls, prescribed burning has no effect on blanket bog microtopography (Noble *et al.* 2018a; Heinemeyer, Sloan & Berry, 2019). The second part of the statement is too ambiguous to comment on – the term "*environmental resilience*" needs to be clearly and objectively defined.
Statement 10	Bogathon and Sphagathon (Moorland Association & Heather Trust, 2015) have demonstrated that there is support for maintaining and restoring peatlands to a healthy condition. It has also demonstrated recognition among land managers that healthy peatlands can support driven grouse shooting and stock grazing. There is indeed support for peatland restoration. However, we need clearly defined and objective restoration goals that are based on ecological function. Once such criteria have been developed, we suggest that scientists and government agencies then need to work together with land managers and grouse shooting estates to carefully manage an evidence-based and site-specific transition to alternative management. We advocate using a series of 'champion estates' distributed across the UK (to capture different site conditions) that implement alternative and traditional management using a moorland-scale BACI approach. "Landowners and grouse moor managers appreciate that raising the water table builds resilience into their land to provide protection from the impacts of climate change and the increasing risk of damage from wildfire – 'wetter is better'." (BASC & Moorlands Association, 2016) As previously noted, we cannot assume that wetter is always better. Blanket bogs with water tables at or above the soil surface can emit large amounts of methane (Abdalla *et al.* 2016; Evans *et al.* 2017)[5], which could counteract the carbon accumulated within the peat body (Heinemeyer *et al.* 2019b). Saturated peatlands may also contribute to flooding downstream (via increased saturated overland flow) and have a negative impact on important invertebrate taxa, such as Tipulidae (an important food source for rare upland birds) (Holden & Burt 2003; Holden *et al.* 2008; Carroll *et al.* 2015; Holden *et al.* 2017; Heinemeyer *et al.* 2019b). We also cannot assume that rewetting will be enough to mitigate wildfires because the wildfire mitigation potential of rewetting has never been tested within a UK context. There are two aspects to wildfire mitigation: ignition prevention and damage limitation. Firstly, it seems intuitive that wetter bogs would be less likely to ignite (Davies & Legg 2011). However, in summer, bog vegetation becomes very dry, especially during the prolonged dry spells that are becoming increasingly common due to climate change. As the vegetation becomes drier, it becomes more flammable. For example, *C. vulgaris* becomes flammable when moisture content drops below 60% (Davies & Legg 2011). Thus, in theory, ignition of heather dominated peatlands is possible any time the moisture content of the *C. vulgaris* canopy drops below 60%.

[4] Again, we have to assume because no supporting citation is given.
[5] Especially under the warmer and wetter conditions we are expecting due to future climate change (Heinemeyer *et al*. 2019b).

	Now, even if a canopy fire took hold, it is highly likely that a wetter bog *would* reduce the chances of the underlying moss and peat layers igniting, or limit the spread of a peat fire if the peat body did ignite. Indeed, a group of British studies show that the moss and peat layer within (wet) blanket bog ecosystems are generally buffered from the effects of a prescribed burn (i.e. minimal damage and no peat ignition) (Grau-Andrés, Gray & Davies 2017; Grau-Andrés *et al.* 2018; Grau-Andrés *et al.* 2019a; Grau-Andrés *et al.* 2019b). But these studies were testing the effect of a prescribed management burn. Such burns are carried out between late autumn and early spring (1st October – 15th April) when peatland water tables are higher (DEFRA 2007; Heinemeyer *et al.* 2019b). Consequently, prescribed burns are likely to be significantly cooler than wildfires (especially at the soil surface) (Davies *et al.* 2010a; Davies *et al.* 2010b; Davies *et al.* 2016b), with the latter generally occurring in the summer months (Albertson *et al.* 2009). Another consideration is that, even on hydrologically intact peatlands (i.e. those that undisturbed by human management), the water table draws down by as much as 20-30 cm during the summer months (Labadz, Hart & Butcher 2007; Holden *et al.* 2011), which, combined with dry vegetation, is likely to significantly increase the flammability of the peat. Furthermore, rewetting is often associated with a cessation of vegetation management, which, as the Hard Hill Experiment indicates, leads to a significant increase in above-ground biomass (Alday *et al.* 2015; Marrs *et al.* 2019a). Consequently, rewetted bogs are likely to have a higher fuel load, which will lead to higher fire temperatures if a wildfire does manage to ignite (Hobbs & Gimingham 1984; Davies *et al.* 2016a; Noble *et al.* 2019a). Given all the points raised above, rewetting may not be as effective at mitigating wildfire as its proponents claim. However, due to the lack of data, the true role of rewetting in wildfire mitigation remains unknown and, thus, requires urgent research attention[6].
Statement 11	When examining the evidence on wildfire impacts, it is important to distinguish between studies based on dry heath/grasslands on shallow soils, as opposed to deep peat sites. Concerns over wildfire risk do not generally apply to wet blanket bog habitat where there is naturally minimal dry biomass load and high water tables prevent burning of the peat mass. This statement is, at best, an unverified assertion. Firstly, we lack data on the impact of rewetting on vegetation biomass. However, we do have data from a long-term (60 years) experiment situated in an area of undrained, wet and high-altitude blanket bog: the Hard Hill Experiment. In contrast to what the IUCN document asserts, the unmanaged plots (plots unmanaged since 1923 and 1954) within the Hard Hill Experiment support the greatest amount of biomass (Alday *et al.* 2015). Secondly, rewetting may raise water tables during the wetter months (October to April), but water tables will drop significantly during the summer months (Labadz, Hart & Butcher 2007). This, combined with dry conditions, is likely to significantly increase the flammability of the peat. But again, we lack data on this key issue.
Statement 12	However, a large proportion (c. 80%) of our peatlands are considered to be in a degraded condition. Degraded peatlands with abundant heather have been described by some managers as a fire risk when naturally high water tables are absent. The larger fuel load on a damaged peatland can mean that if a fire occurs that it is more damaging; greater fuel load ≈ greater heat intensity ≈ prolonged fire ≈ potential for greater damage to vegetation and ignition of the underlying peat soil. There are numerous scientific studies which demonstrate that wet peatlands are less prone to wildfire (e.g. Turetsky et al., 2015, Swindles et al., 2019; Grau-Andres et al., 2017;) or that rewetting is a better strategy than burning to achieve peatlands that are resilient to wildfire (Baird et al., 2019) . Re-wetting peatlands is therefore viewed as crucial in mitigating wildfire risk. For a rebuttal of the first sentence, see our comments underneath 'Statement 4'. We agree with this: "*greater fuel load ≈ greater heat intensity ≈ prolonged fire ≈ potential for greater damage to vegetation and ignition of the underlying peat soil*". Indeed, it is well established (Davies *et al.* 2010b; Davies *et al.* 2016a; Davies *et al.* 2016b; Noble *et al.* 2019a), as is the fact that prescribed burns reduce fuel load on UK peatlands, at least in the short-term (Lee *et al.* 2013; Alday *et al.* 2015; Milligan *et al.* 2018; Whitehead & Baines 2018; Grau-Andrés *et al.* 2019a; Heinemeyer *et al.* 2019b; Marrs *et al.* 2019a). However, we disagree with the unverified assertion that rewetting peatlands is crucial for mitigating wildfire risk. This has not been tested and, as our comments on 'Statement 10' suggest, it may also be a flawed assumption. Furthermore, we question some of the references used to support the

[6] Several large wildfire projects are currently live in the UK.

	statement "*that wet peatlands are less prone to wildfire*". Swindles *et al.* (2019) do not test this and Grau-Andrés *et al.* (2018) examine prescribed burning, not wildfire.
Statement 13	On UK peatlands, high fuel loads of heather and grasses and dry exposed peat are consequences of lower water tables from drainage, compounded by over-grazing and repeated burning. A healthy peatland with high, stable, water tables and Sphagnum growth, naturally suppresses excess heather and other dry understory ground vegetation. For many sites rewetting (raising the water table) is a rapid process following restoration works and there will be no need for additional vegetation management. However, some severely degraded sites or sites with complex topography (e.g. sites with severe peat hags) may still have significant areas of drier peat and excess heather and other dry vegetation following rewetting activity. For these sites there may be the need to consider measures to control fire risk during the transition period, such as cutting fire breaks in certain areas and restricting burning on adjacent areas. The mixing of different management aspects is unhelpful for an informed debate. As outlined previously, we agree that deep drainage is a serious issue (e.g. Young *et al*., 2019), but it should be judged independently from burning as deep drainage can be implemented in the absence of burning (and vice versa). We also think that prescribed burns should not be carried out on steeply sloping and dry areas or areas in which a significant amount of the peat surface is exposed (the latter is extremely unlikely to be burned because it will not be dominated by a dense *C. vulgaris* canopy). However, prescribed burns within flat and wet areas of blanket bog are likely to cause only minimal short-term damage to the moss layer (Davies *et al.* 2010b; Lee *et al.* 2013; Kettridge *et al.* 2015; Taylor 2015; Grau-Andrés, Gray & Davies 2017; Grau-Andrés *et al.* 2018; Milligan *et al.* 2018; Noble *et al.* 2018a; Grau-Andrés *et al.* 2019a; Grau-Andrés *et al.* 2019b; Heinemeyer *et al.* 2019b; Marrs *et al.* 2019a). In addition, we take issue with the following passage: "*A healthy peatland with high, stable, water tables and Sphagnum growth, naturally suppresses excess heather and other dry understory ground vegetation. For many sites rewetting (raising the water table) is a rapid process following restoration works and there will be no need for additional vegetation management*". If by 'healthy', the IUCN mean a peatland that is relatively undisturbed by human management, then such peatlands can also experience a significant drop in the water table during the summer months (Labadz, Hart & Butcher 2007; Holden et al., 2011). Also, very wet peatland sites can still be dominated by *C. vulgaris* well after any management has ceased (Lee *et al.* 2013; Alday *et al.* 2015; Milligan *et al.* 2018).
Statement 14	There are a range of approaches to reducing fire risk in habitats. For peatlands, the approach used must not lead to increased deterioration of the peatland sites as this will exacerbate fire risk. In many peatland restoration projects, managers will seek to rewet and diversify the vegetation composition to naturally reduce biomass. This may involve vegetation cutting in strategic locations, seeking to influence visitor behaviour, responding directly to visitor behaviour at high risk times and participating in local fire response groups. We recognise that there is a need to investigate the most effective mechanisms for wildfire risk mitigation to support the development of management plans for restoration projects during transition periods. As we highlight in our comments underneath 'Statement 6', rewetting may have an adverse impact on certain ecosystem services. There are also many potential issues with alternatives to burning, such as cutting (e.g. sedge dominance, methane emissions, water quality impacts; see Heinemeyer *et al*., 2019 – BD5104). Furthermore, it might not always be possible to restrict access and it only takes one ignition incident to set off a devastating wildfire across a heather dominated blanket bog. We need to consider these risks, measure them, and try to predict them accurately in relation to different land management interventions, including prescribed burning.
Statement 15	Wildfires on peatland are rare outside of situations where people have been involved in the origin of the fire, whether as a result of an out of control managed burn, arson or carelessness. We agree that the greatest wildfire threat to blanket bog comes from people, particularly on blanket bogs near densely populated urban areas (as the number of visits to these bogs would be greater) (Albertson *et al.* 2009). However, even though wildfires on blanket bog are currently rare, they may increase in frequency due to climate change (Albertson *et al.* 2010).
Areas for further consideration and	- An agreed methodology for defining different peatland states should be developed for use in academic studies along with protocols for describing peatland vegetation which include

research	vegetation type and structure. Yes, we agree. However, such a methodology must be based on direct measurements of ecosystem functioning (e.g. net peat and carbon accumulation, net GHG emissions, water storage and quality; biodiversity) – we need to move away from using unverified vegetation metrics as proxies for peatland ecosystem functioning. We would also need to produce a set of thresholds (based on actual ecological data). - Agree how the impact of burning on C storage and C accumulation should be measured. We think that, overall, there is strong agreement on this issue amongst peatland researchers, but it would be extremely useful if a standardised measurement protocol were developed. We recommend that any such protocol should consider the following: • The measurement of C storage from fluxes should include all major C-flux components (i.e., Net Ecosystem Carbon Balance, NECB), certainly both main gaseous C flux components, carbon dioxide and methane, using eddy covariance towers or ground-level chambers. • Any management assessment needs to consider the entire rotation period (i.e. regrowth of vegetation to maturity). For example, a robust flux monitoring approach for managed heather burning can be expected to require at least 25 years (Heinemeyer et al., 2019b) • Any peat core assessment needs to consider detailed peat physical assessments and ideally assess full-length peat cores (i.e. bulk density in relation to peat moisture (Morton & Heinemeyer, 2019) and peat C storage impacts from deep drainage (Young et al., 2019). • The measurement of charcoal impacts on carbon accumulation (i.e. recalcitrant, long-term C storage) and carbon fluxes (i.e. its influence on peatland microbial activity and, thus, decomposition). • Measurements of dissolved and particulate organic carbon exports using soil and stream water analysis. To properly understand the impact of burning on carbon storage and accumulation, the above measurements must be collected using the following scientifically robust and real-world approach: A multisite BACI design where treatments (burnt versus an unburnt control) are randomised within each site and data is collected during at least one complete burning rotation (but, ideally several). - Instigate a number of long-term monitoring and survey plots for peatlands under different management conditions to determine the impact of burning on the trajectory towards peatland restoration. Yes, we could not agree more. However, government funding is needed to achieve this. For example, the BD5104 (Peatland-ES-UK) project was intended to be one such long-term monitoring study (Heinemeyer *et al.* 2019b). However, long-term monitoring sites require a long-term commitment to funding. Furthermore, such studies must utilise a randomised and multisite BACI design (see, for example, Heinemeyer *et al.* 2019b). This is because, compared to other experimental designs, randomised and multisite BACI designs can control for confounding variables and are significantly more accurate in detecting management impacts (França *et al.* 2016; Smokorowski & Randall 2017; Ashby & Heinemeyer 2019a; Ashby & Heinemeyer 2019b) - A systematic review of the response of peatlands following restoration under different management treatments. Agreed. We urgently need a holistic and clinical systematic review of the impact of different management impacts on peatland ecosystems. Such a review must consider the strength of the experimental designs used and the reliability of the data used. Reviewers should not be afraid to reject studies with unreliable results. Consideration should also be given to selective reporting and titles or conclusions which contradict the study findings. Furthermore, the review should be conducted by an independent scientific group to prevent any bias. - Further research to support the development of accessible good practice guidance in

	managing wildfire risk for peatlands which are under restoration and are in transition to a wet and naturally fire resilient state. Agreed. This is a research priority because, due to the lack of data, we have no idea whether rewetting will mitigate wildfire risk. In the UK, there are several live projects investigating this topic, which is to be welcomed, especially given that the UK has very different conditions/challenges compared to most of the available literature from North America, Scandinavia and the Tropics.

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Report 4:

Prescribed burning impacts on ecosystem services in the British uplands: a methodological critique of the EMBER project

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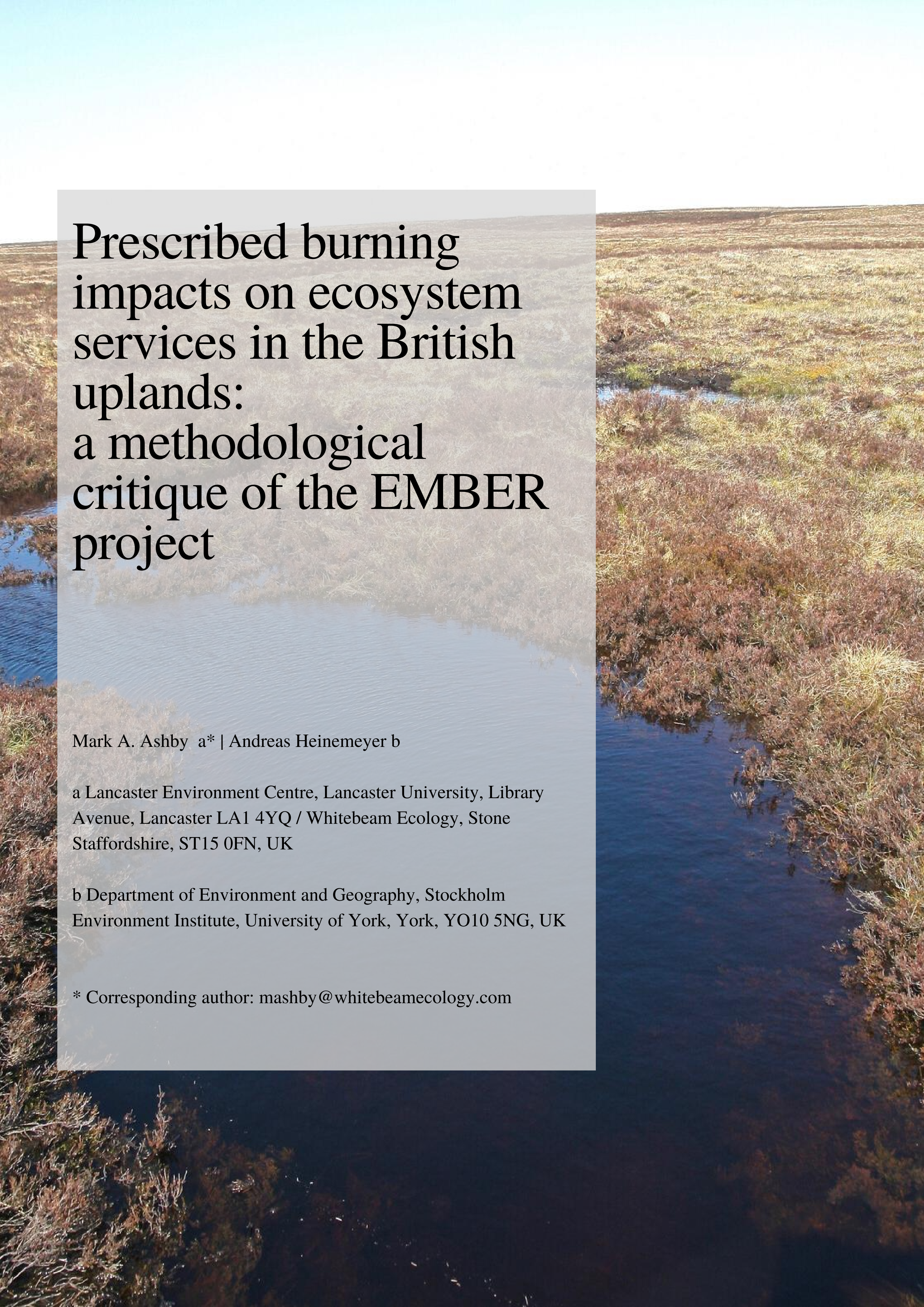
Data sources and analysis (Appendix, S1)



Prescribed burning impacts on ecosystem services in the British uplands: a methodological critique of the EMBER project

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Prescribed burning impacts on ecosystem services in the British uplands: a methodological critique of the EMBER project

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Abstract

1. Due to its novelty and scale, the EMBER project is a key study within the prescribed burning evidence base. However, it has several significant but overlooked methodological flaws.

2. In this paper, we outline and discuss these flaws. In doing so, we aim to highlight the current paucity of evidence relating to prescribed burning impacts on ecosystem services within the British uplands.

3. We show that the results of the EMBER project are currently unreliable because: it used a correlative space-for-time approach; treatments were located within geographically-separate and environmentally-distinct sites; environmental differences

between sites and treatments were not accounted for during statistical analysis; and, peat surface temperature results are suggestive of measurement error.

4. *Policy Implications*. Given the importance of the EMBER project, our findings suggest that (i) policymakers need to re-examine the strengths and limitations of the prescribed burning evidence base; and, (ii) future work needs to control for site-specific differences so that prescribed burning impacts on ecosystem services can be reliably identified.

Keywords

Ecosystem services, Evidence-based policy, Experimental design, Prescribed rotational burning, the EMBER project, Upland habitats, Upland land management

1. INTRODUCTION

In recent years, researchers have begun to highlight the limited evidence surrounding prescribed burning impacts on ecosystem services within the British uplands (Glaves et al. 2013; Davies et al. 2016; Harper et al. 2018). The EMBER project (Effects of Moorland Burning on the Ecohydrology of River basins) aimed to address part of this knowledge gap by conducting the most extensive study thus far on the ecosystem effects of prescribed rotational burning (Brown, Holden & Palmer 2014). However, we believe that this study suffers from a series of important but overlooked methodological flaws. Our objective in this paper is to describe and discuss these flaws. In doing so, we aim to stimulate a broader debate about the current evidence linking prescribed burning with the degradation of upland ecosystems. We fully acknowledge that every scientific study (including ours) is limited by practical considerations such as time and cost. Nevertheless, such practical considerations do not preclude a study from being critically assessed in order to provide a more nuanced view of the evidence base and encourage improvements to study design and data analysis. Furthermore, a thorough examination of the evidence is particularly important in applied ecology where the implementation of the results will have practical, policy and economic consequences.

2. THE EMBER CRITIQUE

2.1. Background

The EMBER project aimed to improve our current understanding about the effects of prescribed rotational burning on water quality, hydrology, aquatic biodiversity and soils within upland peat-dominated river catchments (Brown, Holden & Palmer 2014). It did this over five years using five burnt and five unburnt river catchments and 120 soil plots located within the English Pennines (*ibid*). Both its novelty and its scale make it an important study within the prescribed burning evidence base.

Overall, results from the EMBER project suggest that prescribed burning on blanket bog has clear negative effects on aquatic invertebrates, river water quality, peat hydrology, peat chemistry, peat structure and peat surface temperatures (Brown, Holden & Palmer 2014). Unsurprisingly, these findings meant that the project received a lot of positive media attention upon its release in 2014 (see, for example, Amos 2014; Avery 2014; Bawden 2014; Webster 2014). However, we assert that the findings of the EMBER study are currently unreliable because: it used a correlative space-for-time (SfT) approach; treatments were located within geographically separate and environmentally distinct sites; environmental differences between sites and treatments were not accounted for during statistical analysis; and, peat surface temperature results are suggestive of additional methodological inaccuracies.

Our critique uses the methodological information provided by four peer-reviewed research studies relating to parts 3-6 of the main EMBER report (Table 1). It is worth noting that, depending on the response variable investigated, the EMBER study used different combinations of study catchments and soil plots (Table 1). Additional information about the EMBER experimental design is given within Appendix S1, which also contains a detailed description of data sources, collection methods and statistical analysis for the data presented and discussed in the following sections.

2.2. Correlative space-for-time approach

The EMBER project used a correlative SfT approach whereby comparisons were made between unburnt controls and burnt treatments (and between a chronosequence of different burn ages) well after burning had taken place (Brown, Holden & Palmer 2014). This approach is a cheaper and quicker alternative to conducting controlled field experiments. However, SfT studies assume that control and treatment plots had similar pre-disturbance conditions, which is unlikely to be true due to the environmental heterogeneity of most ecosystems (Pickett 1989; Johnson & Miyanishi 2008). Consequently, the results of SfT studies are not as reliable or accurate as those produced through experimentation (França et al. 2016).

2.3. Geographical separation of treatments

The authors of the EMBER project chose to locate treatments (unburnt and burnt catchments + soil plots) within geographically-separate sites (Fig. 1). This study design assumes that sites are similar in every respect except for burning management (*c.f.* Schwarz 2014a, b). We believe that this assumption is flawed because each study site differed in one or more of the following environmental variables: mean monthly temperature (^{o}C), mean monthly rainfall (mm), elevation (m), underlying geology and vegetation communities (Table 2 and 3 and Appendix S1). Many of these variables are known to affect the ecohydrology of upland river basins (*e.g.* Simmons 2003; Durance & Ormerod 2007; Yallop, Clutterbuck & Thacker 2010; Armstrong et al. 2012; Ritson et al. 2014; Armstrong et al. 2015; Parry et al. 2015; Bell et al. 2018). For example, Armstrong et al. (2012) found that peatland vegetation type had a significant effect on dissolved organic carbon (DOC) levels within soil and drain water samples. Moreover, elevation exerts a strong influence on precipitation which, in turn, effects peatland water tables and overland flow (Heinemeyer et al. 2010).

Plot and catchment specific data also indicate that there were environmental differences between treatments (Fig. 2-6). These data are highlighted below and grouped by study focus using the different catchment and soil plot combinations adopted by Brown et al. (2013), Holden et al. (2014), Brown et al. (2015) and Holden et al. (2015).

2.2.1. Streams within all five burnt catchments vs streams within all five unburnt catchments

This approach was used to investigate burning impacts on aquatic invertebrate communities, stream ecosystem functioning, water quality (Brown et al. 2013) and streamflow (Holden et al. 2015) (Table 1). The five burnt catchments are significantly drier than the five unburnt catchments (Fig. 2a). Burnt catchments were also smaller, at lower elevations and warmer, although these differences were not statistically significant (Fig. 2b).

2.2.2. Burnt vs unburnt plots across all ten catchments

Holden et al. (2015) used this experimental set up to investigate burning impacts on water table depth. Unburnt plots were at significantly greater elevations and on significantly steeper slopes (Fig. 3a & b). Also, a higher proportion of unburnt plots had a northerly aspect (Fig. 4a).

2.2.3. Burnt plots of different burn ages vs unburnt plots across all ten catchments

Plots of different burn ages (ranging from <2 years to >10 years) were compared with each other and with unburnt plots by Holden et al. (2015) while investigating burning impacts on water table depth. Plots of different burn ages were at similar elevations to each other but at lower elevations than unburnt plots; however, this pattern was not significant (Fig. 3c). Conversely, slope angle varied between plots of different burn ages and unburnt plots, but again this pattern was not significant (Fig. 3d). The proportion of plots with a northerly aspect also varied between plots of different burn ages and unburnt plots (Fig. 4b). This pattern is most pronounced when comparing unburnt plots with burnt plots that were <2 years old (Fig. 4b).

2.2.4. Burnt plots of different burn ages within Bull Clough vs unburnt plots within Moss Burn vs wildfire plots within Oakner Clough

This approach was used by Holden et al. (2014) to examine the impact of burning on peat near-surface infiltration and macropore flow. Plots of different burn ages were positioned at a similar elevation to each other but a lower elevation than unburnt plots and a higher elevation than wildfire plots (Fig. 5a). In terms of between treatment differences in slope angle: wildfire plots were located on steeper slopes than all treatments except for burnt plots that were >15 years old; burnt plots that were >15 years old were located on steeper slopes than burnt plots that were <2 years and 3-4 years old; plots that were 3-4 years old were located on steeper slopes than burnt plots that were <2 years old; and, the slope angle of unburnt plots varied considerably but was shallower than wildfire plots (Fig. 5b).

2.2.5. Burnt plots of different burn ages within Bull Clough vs unburnt plots within Oakner Clough

This approach was used to investigate prescribed burning impacts on peat temperature (Brown et al. 2015), water table depth and overland flow (Holden et al. 2015). Plots of different burn ages were positioned at a similar elevation to each other but a higher elevation than unburnt plots (Fig. 6a). Unburnt plots and burnt plots that were >15 years old had similar slope angles, but both these treatments were located on steeper slopes than burnt plots that were <2, 3-4 and 5-7 years old (Fig. 6b). Burnt plots that were <2 and 5-7 years old also had similar slope angles but were located on shallower slopes than burnt plots that were 3-4 years old.

2.4. Statistical inaccuracies

When analysing ecological field data, there are usually multiple covariates acting upon a response variable in addition to the predictor variable of interest (Zuur, Ieno & Smith 2007; Schwarz 2014a). If covariates are known and measured, they can be dealt with to some extent by including them as variables during data analysis: this partitions the variation in the dataset accounted for by the covariate(s) so that the effect of the predictor variable can be examined in isolation (Zuur et al. 2009; Pourhoseingholi, Baghestani & Vahedi 2012). Failure to include a covariate can produce misleading results (Gail, Wieand & Piantadosi 1984). Furthermore, researchers conducting multi-site ecological field studies where treatment replicates are located within each site should include ‘site’ and/or any known environmental factors as covariates during data analysis; since, even though sites may appear similar, they are highly likely to be different in some unknown way, and these unknown differences may influence the results (Schwarz 2014a). Such site level effects are likely to exert a greater influence on the results of ecological field studies where treatments are within separate sites.

In such cases, it is even more important to control for known site differences during data analysis.

As discussed above, the burnt and unburnt treatments within the EMBER study were located within separate sites, and both sites and treatment plots differed in a range of key environmental variables that are likely to have influenced the results (*e.g.* elevation, rainfall, temperature, slope, aspect and vegetation composition). However, to our surprise, none of the peer-reviewed articles part of the main EMBER report attempted to control for any of these site/treatment differences during data analysis (Table 1). Interestingly, Brown et al. (2013) state that "*Differences between individual rivers* (*i.e.* sites) *were not assessed with MANOVA as the main focus of the study was on management effects*." We believe that this statistical approach is flawed and, combined with the choice to locate treatments in separate and environmentally-distinct sites, means that the results reported by the EMBER project cannot solely be attributed to burning management. Perhaps the EMBER authors did not control for site effects because they found it had no bearing on the results. If so, then they should have stated this and ideally provided some supporting analyses.

In contrast, while not associated with the main EMBER project, Noble et al. (2018) did control for site when they examined the effect of several environmental variables (including burning management) on the cover of different plant species within the EMBER study plots. They state that "*Site was included in all models* (generalised linear mixed models) *as a random effect to account for grouping of plots within sites*" (*ibid*).

2.5. Peat temperature measurements

Brown et al. (2015) used Gemini PB-5001 thermistors to measure how vegetation removal through burning influences peat temperature. This type of thermistor has a long metal external sensor that will artificially heat up if any part (but mostly the tip) is exposed to the sun. Exposure to the sun can result in large short-term temperature spikes (see graphs in Appendix of Heinemeyer et al., forthcoming[1]). Brown et al. (2015) report extremely high maximum peat surface (0-1 cm) temperatures (up to 52.8 ^{o}C) within burnt plots of different ages. The relatively low occurrence of these maxima events (*c.f.* Fig. 2 in Brown et al., 2015) suggests that the thermistor sensor was periodically-exposed to the sun and that the temperatures recorded were, therefore, artificially high.

3. CONCLUSIONS

The EMBER project is currently the only published multi-site study to examine the effects of burning on multiple ecosystem processes at both the plot and catchment level (but see, Heinemeyer et al., forthcoming[1]). Consequently, it is likely to have had a strong influence on environmental policy and land management decisions. However, we have demonstrated that the results of the EMBER project should be treated with caution due to a series of statistical inadequacies and what appear to be several important methodological flaws. These findings suggest that: (i) policymakers need to re-examine the strengths and limitations of the prescribed burning evidence base; and, (ii) researchers need to fully account for potential site-specific differences in any future work so that prescribed burning impacts on ecosystem services can be reliably identified.

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AUTHOR’S CONTRIBUTIONS

M.A. and A.H. conceived the paper; M.A. collected and analysed the data and wrote the first draft of the manuscript. Both M.A. and A.H. interpreted the results, revised the manuscript and gave final approval for publication.

DATA ACCESSIBILITY

All data sources are listed in Appendix S1. Grid references and treatment information for each EMBER study plot can be obtained by contacting the authors of the report directly.

ENDNOTES

1. This is the previously Defra-funded (BD5104) and now extended Peatland-ES-UK project. A report presenting the results from the first five years is currently pending final approval by Defra.

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Table 1. Summary of the peer-reviewed articles associated with the EMBER project (Brown et al., 2014).

Authors	Related chapter	Response variables	Experimental set-up and analysis
Brown et al. (2013)	Chp 6	Aquatic biodiversity, stream ecosystem functioning and water quality	Compared streams in five burnt catchments to streams in five unburnt catchments. The fact that unburnt and burnt streams were in separate sites **was not accounted for during statistical analysis**.
Holden et al. (2014)	Chp 3 & 4	Peat near-surface infiltration and macropore flow	Compared plots of different burn ages within the Bull Clough catchment (burnt 2 years, 4 years and >15 years prior to the study) to unburnt plots within the Moss Burn catchment, as well as plots affected by a recent wildfire (<1-year-old) in the Oakner Clough catchment. Three 400 m^2 plots were used for each burning treatment. These were positioned in top, middle and bottom hillslope positions. The fact that unburnt, burnt and recent wildfire plots were in separate sites **was not accounted for during statistical analysis**.
Brown et al. (2015)	Chp 5	Peat temperature	Compared plots of different burn ages within the Bull Clough catchment (burnt <2 years, 3–4 years, 5–7 years and 15-25 years prior to the study) to unburnt plots within the Oakner Clough catchment. Three 400 m^2 plots were used for each burning treatment, positioned in top, middle and bottom hillslope positions. The fact that unburnt and burnt plots were in separate sites **was not accounted for during statistical analysis.**
Holden et al. (2015)	Chp 4	Water table depth, overland flow and streamflow	This study compared (response variables in parentheses): (i) Streams in five burnt catchments to streams in five unburnt catchments (streamflow). (ii) 60 burnt and 60 unburnt 400 m^2 plots across all ten catchments. Within each catchment, three plots were positioned in low, mid and high slope positions (water table depth). (iii) Plots of different burn ages in the five burnt catchments to plots within the five unburnt catchments. The burn age treatments were <2 years, 3–4 years, 5–7 years and >10 since burning. Within each burnt catchment there were three 400 m^2 plots per burn age with one of these corresponding to low, mid or high slope positions. Within each unburnt catchment, three 400 m^2 plots were positioned in low, mid and high slope positions (water table depth). (iv) Plots of different burn ages within the Bull Clough catchment (burnt <2 years, 3–4 years, 5–7 years and 15-25 years prior to the study) to unburnt plots within the Oakner Clough catchment. Three 400 m^2 plots were used for each burning treatment. These were positioned in top, middle and bottom hillslope positions (overland flow and water table depth). The fact that unburnt and burnt streams and plots were within separate sites **was not accounted for during statistical analysis**.

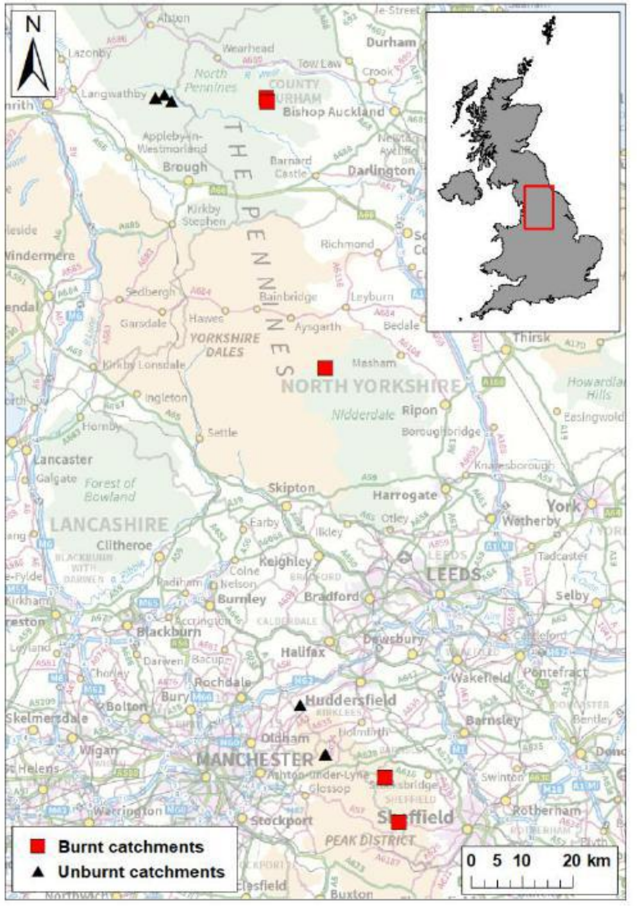

Figure 1. Map showing the locations of the five burnt (red squares) and five unburnt (black triangles) EMBER catchments. The Ordnance Survey MiniScale basemap TIFF (version 01/2018) was downloaded on the 30th October 2018 from: https://www.ordnancesurvey.co.uk/opendatadownload/products.html

Table 2. Locations and environmental conditions of the five burnt and five unburnt EMBER catchments. Location information was taken from Brown et al. (2014). Catchment environmental data was obtained from a variety of sources which are described in Table S1.2 within Appendix S1.

Management/ catchment	Location	British grid reference	Monthly temperature (°C)	Monthly rainfall (mm)	Elevation (m)	Area (km^2)	Geology
Burnt catchments:							
Bull Clough	Midhope Moor, Peak District	SK1915897463	6.33	123.56	498	0.7	Carboniferous and Jurassic sandstone
Rising Clough	Derwent Moors, Peak District	SK2180288624	7.63	97.93	415.5	1.8	Carboniferous gritstone and sandstone
Woo Gill	Nidderdale, Yorkshire Dales	SE0723278444	7.14	112.94	488	1	Carboniferous and Jurassic mudstone
Great Eggleshope beck	Teesdale, North Pennines	NY9558732021	5.39	99.95	566.5	1.6	Carboniferous mudstone, sandstone and limestone
Lodgegill Sike	Teesdale, North Pennines	NY9572631276	5.39	99.95	561.5	1.2	Carboniferous mudstone, sandstone and limestone
Unburnt catchments:							
Crowden Little Beck	Longendale, South Pennines	SE0728701970	6.77	130.6	468.5	3.1	Carboniferous gritstone and sandstone
Green Burn	Teesdale, North Pennines	NY7674331473	5.46	147.29	641	0.7	Carboniferous sandstone, limestone and shale
Moss Burn	Teesdale, North Pennines	NY7535632708	5.46	147.29	664	1.4	Carboniferous sandstone, limestone and shale
Oakner Clough	Marsden Moor, South Pennines	SE0224111836	7.71	117.11	345.5	1.2	Carboniferous gritstone and sandstone
Trout Beck	Teesdale, North Pennines	NY7348532097	4.38	120.37	694.5	2.8	Carboniferous sandstone, limestone and shale

Table 3. The dominant national vegetation classification (NVC) types and plant species found within burnt and unburnt EMBER study catchments. Information taken from Hedley (2013), Holden et al. (2015) and Noble et al. (2018).

Management/site	NVC type	Dominant plant species
Burnt catchments:		
Bull Clough	H9b	*Calluna vulgaris, Eriophorum* spp., *Rubus chamaemorus, Vaccinium myrtillus*
Rising Clough	H9b	*Calluna vulgaris, Eriophorum* spp., *Campylopus introflexus*
Woo Gill	M19a	*Calluna vulgaris, Eriophorum* spp., *Campylopus, Hypnum jutlandicum, Vaccinium myrtillus*
Great Eggleshope beck	M19a	*Calluna vulgaris, Eriophorum* spp., *Campylopus, Hypnum jutlandicum, Vaccinium myrtillus, Sphagnum capillifolium*
Lodgegill Sike	M19a	*Calluna vulgaris, Hypnum jutlandicum, Polytrichum commune*
Unburnt catchments:		
Crowden Little Beck	M20b	*Vaccinium myrtillus, Empetrum nigrum, Eriophorum* spp., *Deschampsia flexuosa*
Green Burn	M19b	*Empetrum nigrum, Eriophorum vaginatum, Hypnum jutlandicum, Plagiothecium undulatum, Pleurozium schreberi, Rhytidiadelphus loreus, Sphagnum capilifolium*
Moss Burn	M19b	*Calluna vulgaris, Empetrum nigrum, Eriophorum vaginatum, Hypnum jutlandicum, leurozium schreberi, Sphagnum capillifolium*
Oakner Clough	M20b	*Eriophorum* spp., *Molinia*
Trout Beck	M19b	*Calluna vulgaris, Eriophorum vaginatum, Hypnum jutlandicum, Plagiothecium undulatum, Pleurozium schreberi, Rhytidiadelphus loreus*

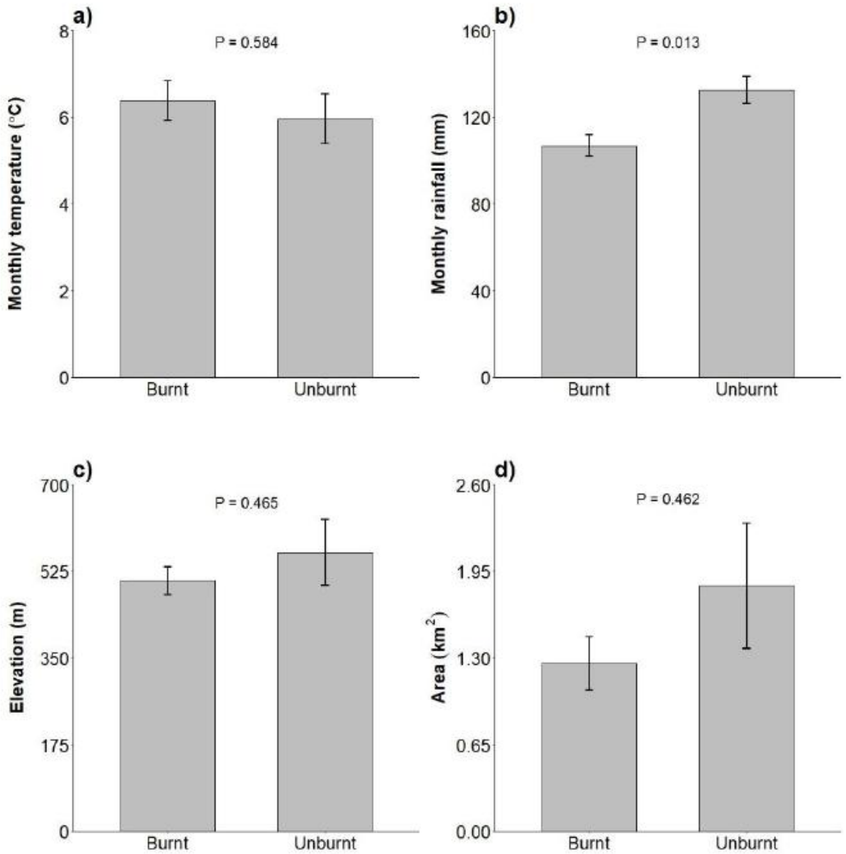

Figure 2. The environmental and physical differences of the five burnt and five unburnt EMBER study catchments. Showing the mean (± SEM) differences in (a) monthly temperature, (b) monthly rainfall, (c) elevation and (d) area. P-values are from one-way ANOVA (a & b) or Kruskal-Wallis rank sum tests (c & d).

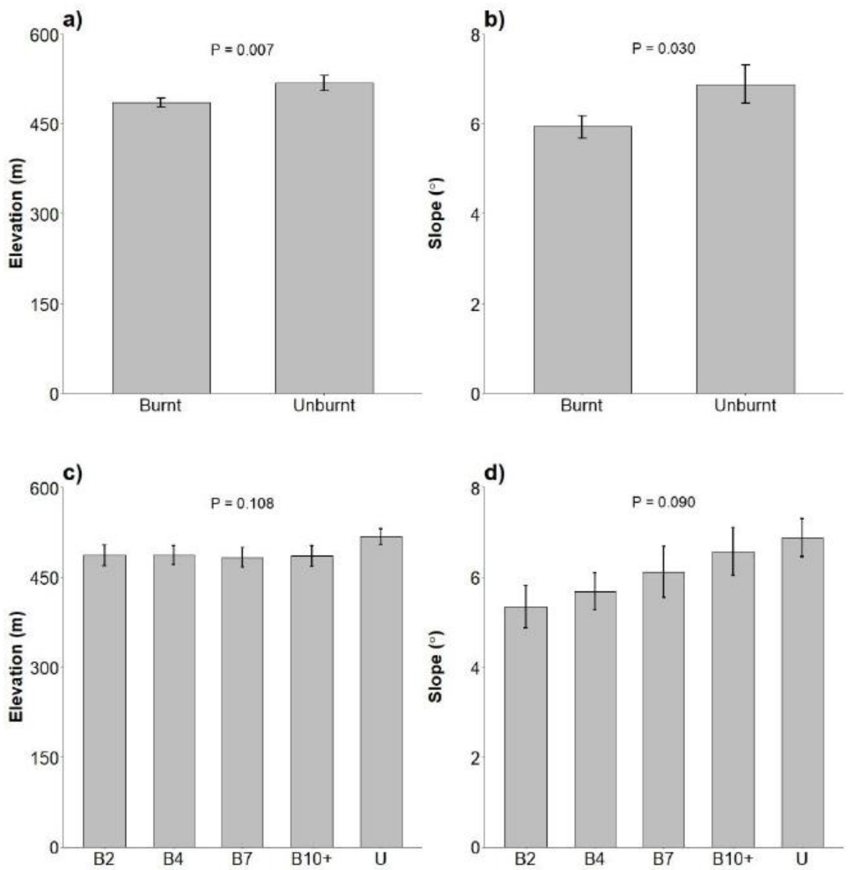

Figure 3. The topographical differences of the EMBER treatment plots. Showing the mean (± SEM) differences in (a) elevation and (b) slope values of the burnt (n = 60) and unburnt (n = 60) EMBER study plots. Also shown are the mean (± SEM) differences in (c) elevation and (d)

slope values for the same plots when they are grouped by burn age treatment: "B2" = <2 years old (n = 15), "B4" = 3–4 years old (n = 15), "B7" = 5–7 years old (n = 15), "B10+" = >10 years old (n = 15) and "U" = unburnt (n = 60). P-values are from Kruskal-Wallis rank sum tests.

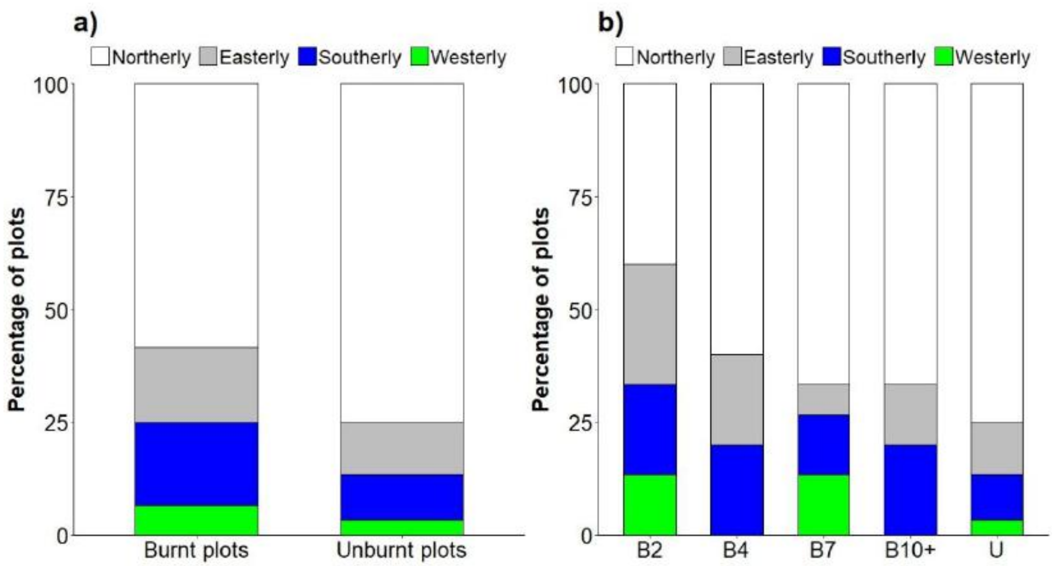

Figure 4. The different aspects of the EMBER treatment plots. a) Showing the percentage of burnt (*n* = 60) and unburnt (*n* = 60) EMBER plots with a northerly (N, NE, NW), easterly (E), southerly (S, SE, SW) or westerly (W) aspect. b) Showing the percentage of plots of different burn ages with a northerly, easterly, southerly or westerly aspect: “B2” = <2 years old (*n* = 15), “B4” = 3–4 years old (*n* = 15), “B7” = 5–7 years old (*n* = 15), “B10+” = >10 years old (*n* = 15) and “U” = unburnt (*n* = 60).

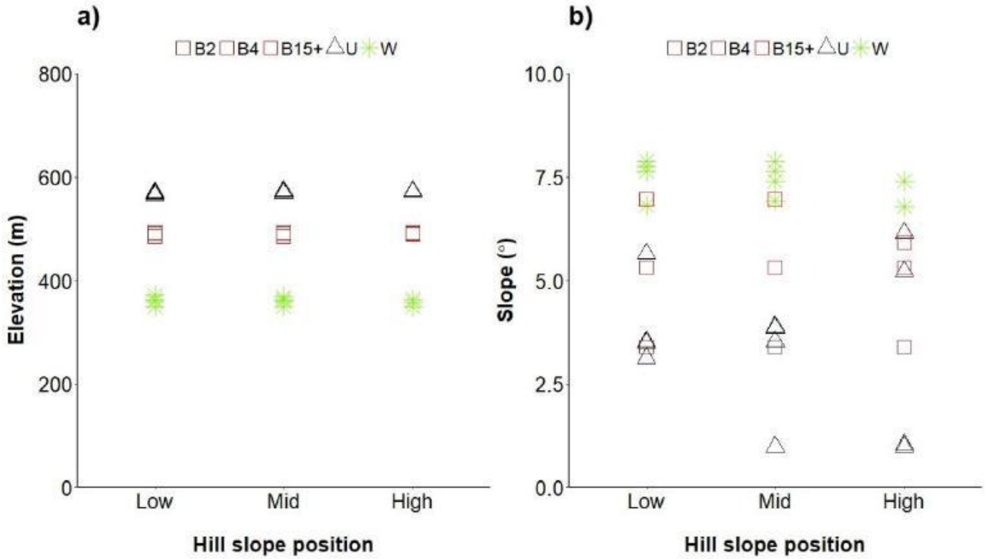

Figure 5. The topographical differences of the sub-set of EMBER treatment plots used by Holden et al. (2014). a) Showing the elevation values for: “B2” (<2 years old; *n* =3), “B4” (3–4 years old; *n* =3) and “B15+” (>15 years old; *n* =3) plots within the Bull Clough study catchment; “U” (unburnt; *n* =12) plots within the Moss Burn study catchment; and, “W” (recent wildfire; *n* =12) plots within the Oakner Clough study catchment; values are grouped along the x axis by plot slope position: low, medium and high. b) Showing the slope values for: B2, B4 and B15+

plots within the Bull Clough study catchment; unburnt (U) plots within the Moss Burn study catchment; and, recent wildfire (W) plots within the Oakner Clough study catchment; values are grouped along the x axis by plot slope position: low, medium and high.

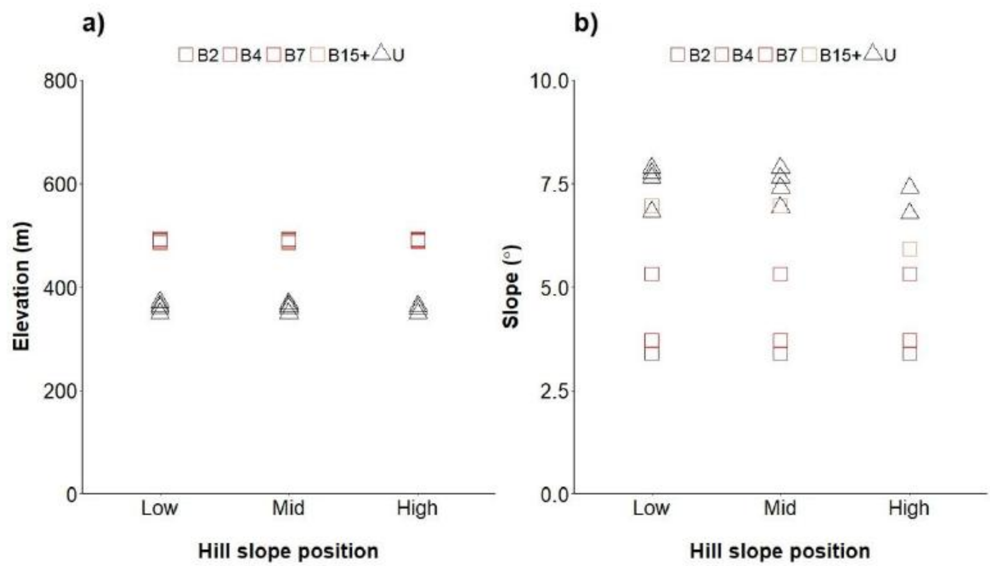

Figure 6. The topographical differences of the sub-set of EMBER treatment plots used by Brown et al. (2015) and Holden et al. (2015). a) Showing the elevation values for: “B2” (<2 years old; *n* = 3), “B4” (3–4 years old; *n* = 3), “B7” (5-7 years old; *n* = 3) and “B15+” (>15 years old; *n* = 3) plots within the Bull Clough study catchment; and, “U” (unburnt; *n* = 12) plots within the Oakner Clough study catchment; values are grouped along the x axis by plot slope position: low, medium and high. b) Showing the slope values for: B2, B4, B7 and B15+ plots within the

Bull Clough study catchment; and, unburnt (U) plots within the Oakner Clough study catchment; values are grouped along the x axis by plot slope position: low, medium and high.

Note: Following publication of this report, there was a non-peer reviewed response from the authors of the EMBER report, Brown and Holden. https://www.biorxiv.org/content/10.1101/731117v1) In turn, we (Ashby & Heinemeyer), responded stating the view that none of the major criticisms had been addressed. https://ecoevorxiv.org/68h3w/)

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