Prescribed heather burning on peatlands: A review of ten key claims made about heather management impacts and implications for future UK policy

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SUMMARY

In a previous *Mires and Peat* article, Bacon *et al*. (2017) questioned ten common assumptions frequently made about peatlands "*in the academic literature, practitioner reports and the popular media which are either ambiguous or in some cases incorrect*". In a similar vein, here, we critically examine ten claims frequently made by the UK governmental, non-governmental organisations, popular media and scientists in relation to the effects of prescribed burning of heather on peatlands. The ten claims are:

1. Prescribed heather burning causes a net peat carbon loss and contributes to the climate crisis;
2. Fire and heather dominance are a result of recent management changes;
3. Prescribed heather burning reduces *Sphagnum* moss abundance and peat formation;
4. Rewetting reduces heather dominance and thus protects peatlands against wildfire;
5. Cessation of heather burning results in wetter peat, less heather cover and no need to burn;
6. Seventy-five percent of global heather moorland is found in the UK;
7. Prescribed heather burning causes water colour and quality issues;
8. Prescribed heather burning causes flooding;
9. Peatlands offer huge carbon sequestration potential and are climate change 'saviours'; and
10. Prescribed heather burning causes loss of biodiversity.

We critically examine the evidence surrounding each of these claims and use our findings to make policy and research recommendations for those interested in the future management of UK peatlands and to facilitate an informed and unbiased debate. The key findings of our assessment are that: (a) government agencies and policymakers need to **re-examine the strengths and limitations of the evidence base and be wary of generalisations** around management needs and options on heather-dominated peatlands, especially for prescribed burning; (b) **researchers need to fully account for potential site-specific and pre-management differences and limitations in temporal and spatial scales**, especially in urgently needed systematic reviews; (c) in any future work, all **major alternative management scenarios should be compared adequately and robustly to burning and assessed** for short-term (disturbance) and long-term (trajectory) impacts across appropriate landscape scales, so that management effects (benefits and risks) on ecosystems, their functions and services can be reliably identified to inform policy.

KEY WORDS: *Calluna vulgaris,* fire, peatland management, *Sphagnum*, wildfire

INTRODUCTION

Peatlands in the UK represent a significant natural carbon store, support a unique and diverse range of flora and fauna, and provide multiple ecosystem services to wider society (Bonn *et al*. 2014). Their importance and the fact that many are classified as being degraded (Lindsay 2010), has led to the creation of national and regional peatland action plans (e.g., the English, Welsh, Scottish and Northern Irish plans, but also regional plans in the Cairngorms or Argyll). For example, the *England Peat Action Plan* (DEFRA 2021) outlines management and restoration plans for approximately 1.42 million ha of peatlands. The document references a £50 million Nature for Climate Peatland Grant Scheme to support best practices for protecting, managing and restoring peatlands across England and Wales (*ibid*). A common theme across UK peatland action plans is the call for evidence-based peatland restoration policies and initiatives *(ibid*). However, producing sound peatland management policies is extremely challenging in the UK as one must rely on a limited and often conflicted evidence base (e.g., Ashby & Heinemeyer 2021) to balance the needs of various stakeholders whose main focus is to protect peatlands and the numerous ecosystem services they provide (e.g. Uplands Management Group 2017). Such difficulties are highlighted by ongoing debates within the scientific and practitioner communities about the validity and efficacy of certain peatland management approaches such as prescribed heather (*Calluna vulgaris*) burning (e.g., Davies *et al*. 2016a,b,c, Ashby & Heinemeyer 2021).

Indeed, the use of prescribed heather burning as a peatland vegetation management tool is a much-debated and often controversial topic which has polarised peatland researchers and management practitioners due to the contested implications for peatland functioning and ecosystem service provision compared to alternative land use and management strategies (Davies *et al.* 2016a,b, Harper *et al.* 2018, Crowle *et al.* 2022). Whilst we welcome such debate, we assert that key claims or statements about prescribed heather burning and peatlands must be well-defined and based on robust and applicable evidence but, unfortunately, this is frequently not the case (e.g., Davies *et al.* 2016b,c, Ashby & Heinemeyer 2021). Therefore, we believe there is an urgent need to refocus the debate on the actual evidence, making sure that its limitations and knowledge gaps are recognised and understood. In addition, prescribed heather burning is widely used in other heathland habitats in Europe (Davies *et al*. 2016a,b,c) with several studies discussing their disadvantages and benefits (e.g., Vandvik *et al*. 2005, Ascoli *et al*. 2009, Gjedrem & Log 2020). However, here we focus on upland peatlands in the UK, where prescribed fire is often seen within a special historic context of grouse moor management (Simmons 2003).

This article builds on two recent critical assessments by Ashby & Heinemeyer (2021) and Heinemeyer & Ashby (2023) that challenged several unverified assertions and misleading arguments made about prescribed heather burning impacts on peatland habitats by a statutory conservation agency and peat conservation NGO, respectively. Based on these evaluations and two previous reviews (Davies *et al.* 2016b, Harper *et al.* 2018), we have identified ten claims frequently made by UK governmental and non-governmental organisations (NGOs) about peatland vegetation management and restoration, especially in relation to prescribed heather burning. Here, we review the validity of these claims against the quantity and quality of available evidence. We then summarise our findings (see Table 1 at the end), identify key commonalities between the ten statements, and provide recommendations for moving the debate on peatland vegetation management forward.

REVIEWING VALIDITY OF THE TEN CLAIMS

1. Prescribed heather burning causes a net peat carbon loss and contributes to the climate crisis

It is often claimed that prescribed heather burning on peatlands contributes to climate change (e.g., Gregg *et al*. 2021) through causing a carbon loss via the emissions given off during combustion. Even if this claim were true, the contribution of controlled heather burning to climate change would likely be insignificant. For example, only ~ 30 % of the UK's blanket bog is managed by burning (Evans *et al.* 2014), emissions data from UK peatlands (and at appropriate time scales) are limited and estimates are very uncertain (Evans *et al*. 2022, Williamson *et al.* 2023). This must also be considered in the context of comparing emission estimates from all three 'modified bog' categories (heather, grass, eroding) of less than 10 % (note: CO_2 only emissions are much lower) compared to that of lowland agricultural peatlands which are estimated to account for ~ 60 % of total UK peatland greenhouse gas (GHG) emissions or > 80 % for England (Evans *et al.* 2017). The challenge with this statement is the paucity of empirical evidence to demonstrate either a positive or a negative effect of prescribed burning on carbon storage and emissions (Harper *et al.* 2018, Ashby & Heinemeyer 2021, Heinemeyer *et al.* 2023). A recent example is a review on prescribed burning (i.e., muirburn) for Nature Scot (Holland *et al.* 2022) where the key statement on linking muirburn to peat loss had to be revised as there was no evidence to support it. However, we agree that even small losses of carbon are important and need to be considered, but it is important to see this in the wider context of long-term net carbon sequestration.

Supporting data for this statement is often cited from single study and model sites across the UK, with the position that the burning of heather directly contributes to long-term carbon loss (e.g., Garnett *et al.* 2000, Worrall *et al.* 2009), with Garnett *et al.* being the only study cited in a review by Evans *et al.* (2014). Studies that support this statement are often divided between either single study references (e.g., Garnett *et al.* 2000) or examine the role of wildfire with direct translation into the impact of burning as a heather management tool (e.g., Wilkinson *et al.* 2023). However, Garnett *et al.* (2000) has some methodological limitations (Davies *et al*. 2016b, Heinemeyer *et al.* 2019a). Moreover, another challenge with the evidence from peat core studies, as obtained from sites subjected to rotational burning (Heinemeyer *et al.* 2018, Marrs *et al.* 2019a), is their limitation in interpreting recent carbon accumulation rates, as they represent not fully decomposed organic matter and do not represent a carbon budget or balance for the peatland (Heinemeyer *et al.* 2018). Crucially, this limitation does not invalidate the value or use of such studies and net carbon accumulation rates as implied by the IUCN UK Peatland Programme (IUCN 2020, 2023), based on unjustified and misleading criticisms made by Young *et al*. (2019, 2021), but requires consideration of such limitations and context (Heinemeyer *et al*. 2018, Young *et al*. 2019). Notably, the study of Young *et al*. (2019) is based on a model of unrealistic constant deep (50 cm) drainage and, as in Young *et al*. (2021), does not include any representation of controlled burning (charcoal). Both studies are, therefore, not directly applicable as a generic criticism to invalidate the peat core studies by Marrs *et al*. (2019a) or Heinemeyer *et al*. (2018), which explicitly assess the impacts of heather burning and charcoal, respectively (as discussed in Ashby & Heinemeyer 2021). Moreover, neither study claims that their carbon accumulation rates represent a net carbon balance, which is specifically highlighted by Heinemeyer *et al*. (2018) together with a clarification that carbon accumulation rates near the peat surface are higher as they reflect largely undecomposed peat, which aligns with the findings presented within both of the Young *et al*. publications and thus invalidates their direct criticisms. An added issue is that carbon flux studies provide contradictory evidence, most likely because so far, no such studies have conducted a comprehensive assessment of the net ecosystem carbon balance (NECB) or budget (i.e., an assessment of all NECB elements), and studies fail to capture the appropriate timescale of the entire burning management cycle (e.g., ~20 years) (Evans *et al*. 2022, Heinemeyer & Ashby 2023). It has been suggested that ageing heather continues to increase above-ground carbon stocks for up to 18 years (although this clearly depends on site growth conditions) with a subsequent stagnation and small "net biomass loss" phase over time with maturity (Kopittke *et al.* 2013). In peatlands this has also been linked to a declining water table depth (Worrall *et al.* 2013, Brown *et al.* 2014, Heinemeyer *et al.* 2023) and likely increased net carbon losses (Heinemeyer *et al.* 2023). The latter study also identified a threshold for NECB of about 12 cm mean annual water table depth with wetter conditions likely a carbon sink, and indicated that rejuvenating vegetation can stimulate nutrient levels in leaves (Heinemeyer *et al*. 2025) and net carbon uptake over time, especially after burning (Heinemeyer *et al.* 2023), confirming other studies on heather growth and carbon increments over time (Kopittke *et al.* 2013, Santana *et al.* 2016).

The statement also ignores the evidence that demonstrates the potential benefit of prescribed burning on long-term carbon storage from likely suppressed peat decomposition (Heinemeyer *et al.* 2019a) and from partly bypassing decomposition by converting biomass into charcoal (Leifeld *et al.* 2018, Gao *et al.* 2022, Ashby & Heinemeyer 2021, Heinemeyer *et al.* 2023), which agrees with previous model predictions by Clay & Worrall (2011) and Worrall *et al.* (2013), and other potential charcoal benefits (e.g., reduced methane emissions). Moreover, root carbon input is often overlooked but has been recognised as an important input from heather as a 'peat-forming' species by the IUCN (2014), although the net input after burning remains unclear as dead root biomass continues to decompose over time. This statement also fails to consider site and time-dependent variables ascribed to long-term monitoring of burning impacts (Heinemeyer *et al.* 2023). For example, recent evidence suggests low-severity fires (associated with pyrogenic charcoal/carbon production) may suppress peatland methane emissions via chemical-microbial interactions in various northern peatland ecosystems (Davidson *et al.* 2019, Flanagan *et al.* 2020, Ashby & Heinemeyer, 2021, Gray *et al.* 2021). This methane impact is also corroborated by recent laboratory experimental charcoal work undertaken by Sun *et al.* (2021). However, the results are unclear in the context of application to peatland management practice and further research on charcoal aspects is required (Evans & Gauci 2023). For example, the results of Heinemeyer *et al.* (2023) and Sun *et al.* (2021) are unclear on the quantity of pyrogenic carbon created during these experimental burning events and the effect this has on methane production/oxidation and thus net emissions. It is also unclear how these results relate to the habitat state (e.g., nutrient status) and peatland type (rain versus groundwater fed) and how this influences the interaction between functional microbial groups and pyrogenic carbon, which has been shown to reduce microbial processes and suppress methane production (Sun *et al.* 2021).

The relationship between heather burning and carbon loss is complex and challenging. According to Heinemeyer *et al.* (2023), while there is immediate biomass combustion, there are uncertain charcoal gains (depending on many factors such as fuel load, moisture and combustion temperature), short-term losses are likely outweighed by long-term net carbon gains from rejuvenating biomass (increasing efficiency of photosynthesis versus losses from respiration) and the net ecosystem carbon budget (NECB, including all key carbon storage aspects) is rarely measured. Notably, the NECB has never been compared to alternatives at the appropriate, replicated scales (plot to catchment) and over a complete management cycle (regrowth to ageing heather). For any comparable statement about the impacts of management practices, there are key gaps in the literature that include the role of spatial complexity and multi-site evidence synthesis. An important gap in our knowledge is how the net carbon balance of combustion losses and charcoal gains compares to losses of decomposing plant litter as partly explored in a model scenario by Clay & Worrall (2011) and Worrall *et al*. (2013). Consideration should also be given to the influence of climate change in driving carbon losses from peatlands and the challenge of balancing existing management plans with the potential effects of future climate change on the state of upland peatlands and which management (if any) is required to enhance biodiversity and prevent wildfires (Marrs *et al.* 2019a,b). Recent examples of such wildfires on UK upland peatlands include fires in the Peak District (Marsden Moor) and Scotland (Cannich), with an estimation of carbon losses provided in a subsequent report for the former (Titterton *et al*. 2021).

One solution for providing robust and applicable data is to deploy better experimental and long-term monitoring approaches, especially considering longer-term experimental monitoring and multiple sites (Harper *et al*. 2018), with much work on carbon accumulation in UK upland peatlands having focused on a single site, Moor House (e.g., Garnett *et al*. 2000, Worrall *et al.* 2003, Ward *et al*. 2007, Worrall *et al.* 2007, Marrs *et al.* 2019a). For example, the Peatland-ES-UK project (Heinemeyer *et al.* 2019a, 2023) adopted a randomised, multi-site, catchment-scale, Before-After-Control-Impact (BACI) approach at three upland blanket bog sites across northern England. For the first time, this project collected one year of pre-management change and 10+ years of post-management plot- and catchment-scale (burning and mowing) data with the aim of collecting data for 20+ years (i.e., the full management cycle for those sites), to ensure realistic and accurate assessment of how prescribed burning influences blanket bog NECB (Heinemeyer & Ashby 2023). This study also gives us a better understanding of the plot-scale temporal and spatial variation and a comparison of managed to unmanaged heather, and the long-term effect over a complete management cycle (Heinemeyer & Ashby 2023).

As outlined here and previously by Heinemeyer & Ashby (2023) there is no clear evidence to support this statement and, until there is a shift in the approaches used to assess the effect of prescribed burning on carbon budgets (see their critique of Gregg *et al.* 2021), the evidence remains limited by incomplete assessments as long as only part of the management and recovery cycle and not all major NECB elements are being considered (e.g., Clay *et al.* 2010, Clay *et al.* 2015, Grau-Andrés *et al.* 2019a, Gray *et al.* 2021).

2. Fire and heather dominance are a result of recent management changes

This statement implies that prescribed heather burning is a relatively recent management phenomenon and that its use has led to heather dominance on many peatlands across the UK. One challenge with this claim is defining ‘recent management’ and the timescale in which the evidence on management history is contextualised. Prescribed burning intensified as a management practice (on grouse moors) during the last 200 years (Simmons 2003). However, peat core evidence suggests that heather-dominated vegetation occurred throughout the previous ~6,000 years at several upland peatland sites in the UK as shown in high pollen counts (e.g., McCarroll *et al.* 2017, Webb *et al.* 2022) and plant remains (e.g., Webb *et al.* 2022). Similarly, various peat core studies have also found frequent charcoal layers throughout deep peat profiles across the UK (Fyfe *et al.* 2003, Ellis 2008, Fyfe & Woodbridge 2012, McCarroll *et al.* 2017, Fyfe *et al.* 2018, Webb *et al.* 2022). Such records support the view that both heather dominance (high cover either confirmed as plant remains or based on pollen counts) and fires (evidenced as charcoal layers from likely intentional burning or wildfires) have been a feature of many UK upland peatlands throughout the Holocene (Heinemeyer & Ashby 2023). Some intensification of fire management during the past 200 years is supported, but not for all sites, by increases in surface peat charcoal records (e.g., McCarroll *et al.* 2017, Webb *et al*. 2022).

At sites across Exmoor, South Wales and the Brecon Beacons, Chambers (2022) used palaeoecological records to demonstrate that shifts in *Sphagnum*, grass and heather cover have often been cyclical during the past millennium. At all sites it is unclear what factors were key drivers (i.e., grazing, drainage, pollution, climate or burning) of particular vegetation changes. Furthermore, at sites across northern England, Chambers *et al.* (2017) provide evidence for high heather cover and frequent local fires well before the height of the industrial revolution and intensification of grouse moor management (notably, although the authors repeatedly suggest otherwise, the data clearly demonstrate this for three of the four sites studied).

It is worth noting that there are limitations in using peat core studies. For instance, when a charcoal layer is found within a peat core, we mostly cannot determine whether it resulted from a wildfire or a prescribed burning (i.e., anthropogenic) episode (e.g., Webb *et al.* 2022), although some interpretation is possible (Crawford & Belcher 2022). Moreover, temporal negative relationships between *Sphagnum* and heather cover (or charcoal abundance) in peat profiles (e.g., Chambers *et al.* 2017) do not necessarily imply a cause-and-effect relationship because other factors, such as drainage or climate, are likely to play a key role in explaining this change (Ashby & Heinemeyer 2021). Similarly, the palaeoecological relationship between peatland burning episodes and drier edaphic conditions is complex, and causes often remain unclear (Sim *et al.* 2023); is it the presence of heather that drives burning or is burning the result of increased heather cover (a point originally made by Davies *et al.* 2016b)? However, the two possibilities are not mutually exclusive.

In summary, the palaeoecological record tells us that both fire (either wildfire or prescribed burning) and heather dominance are unlikely to be recent phenomena within UK peatlands and that the key drivers of historical and more recent changes to peatland vegetation remain largely unknown (e.g., is it climate, drainage, grazing, burning or atmospheric pollution?). We need to better understand these drivers, as shown for industrial atmospheric pollution (i.e., lead, nitrogen and sulfuric acid deposition) severely reducing *Sphagnum* cover in the UK's Peak District (e.g., Lee *et al*. 1990, Tallis *et al*. 1997) and including the potential historical influence of fire in shaping UK peatlands.

3. Prescribed heather burning reduces *Sphagnum* moss abundance and peat formation

The greatest challenge surrounding this statement, as already outlined by Bacon *et al.* (2017), is the definition of peat-forming species and the frequent claims made or implied regarding *Sphagnum* mosses in this process (i.e., Ashby & Heinemeyer 2021, Heinemeyer & Ashby 2023). In Gregg *et al.* (2021), Natural England states that, in the UK, *Sphagnum* species "*are the main contributors to peat formation in bogs*". However, it seems that the assumption that *Sphagnum* species are the primary 'peat-formers' within UK blanket bogs is not supported by evidence (Shepherd *et al.* 2013, Gillingham *et al.* 2016, Ashby & Heinemeyer 2021). As stated by Heinemeyer & Ashby (2023), the challenge of assigning *Sphagnum* as the primary driver of peat formation is that "*we lack mechanistic data on their contribution to important peatland functions*". It has been suggested instead that any plant species can also form peat when the conditions allow (e.g., Shepherd *et al.* 2013, Gillingham *et al.* 2016). However, *Sphagnum* moss is clearly an important peat-forming species for several reasons, for example, as outlined in a review by van Breemen (1995).

Again, we can turn to the palaeoecological record to assess this statement. As outlined in Ashby & Heinemeyer (2021), there is evidence across several UK based palaeoecological studies (Fyfe *et al.* 2003, Fyfe & Woodbridge 2012, Shepherd *et al.* 2013, Gillingham *et al.* 2016, McCarroll *et al.* 2017, Fyfe *et al.* 2018, Webb *et al.* 2022) of sustained and frequently high peat accumulation dominated by non-*Sphagnum* plant remains across the Holocene period (often with high heather/Ericales and especially high sedge contributions; although likely, in many studies it is often not clear if Ericales refers to common heather). Confusingly, whilst the IUCN (2014) has included heather as a peat-forming species in their peatland "*Key Facts*" Briefing Note 2 "*Peat Bog Ecosystems: Structure, Form, State and Condition*", it has recently claimed that heather is not a peat-forming species in their position statement 'Burning and Peatlands' (IUCN 2023). Ashby & Heinemeyer (2021) also highlight previous observations that peatlands in Indonesia, the Amazon Basin and the Everglades do not contain *Sphagnum* moss species (Bacon *et al.* 2017, Hodgkins *et al.* 2018). A key technical report by Lindsay (2010) cites only one reference to support the claim that > 99 % of peat mass is formed by *Sphagnum* remains, citing Wallén (1992). On closer inspection, this report does not provide any actual data to support such a generic claim, as calculations were based on unverified/assumed model decomposition values. Crucially, peatland model predictions by Frolking *et al.* (2010) already showed that proportions of remaining plant matter in a Canadian bog are likely to be very different, with vascular plants (mostly sedges and shrubs) accounting for about 36 % versus 64 % non-vascular (mostly *Sphagnum* moss). It is clear from these case studies that *Sphagnum* as a species required for peat formation is a false/misleading terminology (and equally that heather is a non-peat-forming species) as claims of nearly all peat matrix to consist of *Sphagnum* matter are unsubstantiated and heather and other species are found often substantially throughout many peat cores. There is also a possible circularity in the argument about what drives what regarding peatlands' peat-forming conditions and the presence of *Sphagnum* moss.

Regarding the effect of prescribed burning on *Sphagnum* there is further evidence to contradict this statement at sites across the UK where prescribed burning is used as a management tool. For example, at sites in the North Pennines, Whitehead *et al.* (2021) observed at ten years post-burn an overall increase in heather and non-*Sphagnum* moss cover, and *Sphagnum* cover averaged five times higher in plots burnt 8–10 years earlier than in the no-burn control plots. This confirmed a similar finding of increased *Sphagnum* cover after prescribed burning at a long-term experimental burn site by Lee *et al*. (2013), Milligan *et al*. (2018) and Heinemeyer *et al*. (2023). Indeed, several studies support the view that older and dense heather cover suppresses *Sphagnum* moss cover, which can be addressed by burning management (e.g., Milligan *et al*. 2018, Whitehead & Baines 2018, Grau-Andrés *et al*. 2019b).

This statement is further undermined by the imprecise way that peatland condition is defined, which remains ill-defined, vegetation-focused and ignores peatland functions (Ashby & Heinemeyer 2021) at the point of assessment (from intact to modified to degraded) and how this may change depending on whether the assessment is based on *Sphagnum* (or other key plant) species presence or cover versus ecological function. Moreover, as already alluded to above, there can be a circularity in translating evidence of *Sphagnum* in peat cores into evidence of its role as a peat-forming species. Heinemeyer & Ashby (2023) highlight this point: "*it is difficult to determine if such periods of rapid peat growth are due to greater Sphagnum abundance or the presence of conditions favourable to peat formation and Sphagnum growth (e.g., high water tables and low pH)*". However, *Sphagnum* moss is clearly an important peatland species; perhaps a more appropriate term for *Sphagnum* moss and other 'peat-forming' species is therefore *'peat-formation enhancing'* or *'supporting'* species (Heinemeyer & Ashby 2023). We previously recommended the consideration of such more nuanced terminology to prevent misconceptions around peat formation as evident in key policy-informing publications (Ashby & Heinemeyer 2021).

4. Rewetting reduces heather dominance and thus protects peatlands against wildfire

This statement implies a causal relationship between rewetting, a decline in heather cover, and, thus, increased peatland resilience to wildfires. Frequently, in discussions on the topic, it is asserted that restored peatlands reduce the risk of wildfires (Baird *et al.* 2019, Grau-Andrés *et al.* 2019a, Swindles *et al.* 2019). Raising water tables to encourage more 'typical' peatland vegetation is commonly proposed (Granath *et al.* 2016, Glaves *et al.* 2020) because of claims that this limits heather cover (Baird *et al*. 2019, Glaves *et al*. 2020) and, therefore, lacks the requirement for managing fuel loads (IUCN 2020, 2023). Part of this discussion also relies on claims around potential negative impacts from prescribed burning (Davies *et al.* 2016b,c), such as altering microtopography (peat and surface vegetation), affecting runoff and decreasing peat wetness (surface and lower water tables). Indeed, it is often suggested that the cessation of prescribed burning management will support other restoration approaches, such as blocking drainage ditches, to improve wildfire resilience (e.g., Baird *et al.* 2019, IUCN 2020). However, there seems to be no directly applicable evidence to support such generic assumptions, especially in relation to reducing existing heather-domination, and previous claims remain unevidenced (see sections in Ashby & Heinemeyer 2021, Heinemeyer & Ashby 2023). Moreover, the reported negative impacts of prescribed burning on blanket bog moisture are likely short-lived (Heinemeyer *et al.* 2023). In contrast, unmanaged, ageing heather has repeatedly been shown to dry out surface peat over time (Worrall *et al.* 2007, 2013; Brown *et al.* 2014, Heinemeyer *et al.* 2023), which, in conjunction with a higher fuel load, poses a greater risk of more frequent and damaging (i.e., igniting peat) fires (Heinemeyer & Ashby 2023).

There is clearly ecological value in rewetting and revegetating bare and eroding peat as outlined previously (Heinemeyer & Ashby 2023). However, we need to consider that using rewetting to limit heather cover and mitigate against wildfire will have limitations and has never been tested within UK upland peatlands (Ashby & Heinemeyer 2021). Indeed, heather shows physiological adaptations to, and can thrive in conjunction with, a wide range of water table depths, including wet conditions (Bannister 1964), which is confirmed for UK blanket and raised bog sites (Grau-Andrés *et al*. 2018, Heinemeyer *et al*. 2019a). The key variables that need to be considered for wildfire prevention and mitigation include fuel load, moisture content and weather conditions. Although it is frequently assumed that wetter bogs are less likely to ignite (e.g., Davies & Legg 2011, Baird *et al*. 2019), with hotter and drier summers, peatland vegetation and typically wet ground is also likely to become very dry during extreme conditions (apart from the wettest sites) and is, therefore, at a higher combustion/ignition risk in connection to high vegetation fuel loads (Ashby & Heinemeyer 2021, Heinemeyer & Ashby 2023). Whilst rewetting should generally reduce fire risk and limit the risk of fires burning into the peat, this does not imply that wet peatlands are generically 'fireproof'. Long-term assessments are needed as vegetation composition changes over time, with higher heather cover likely during succession, especially under drier conditions. Moreover, heather becomes flammable when moisture content drops below 60 % (Davies & Legg 2011) and is, therefore, likely to be at risk because of climate change rather than management measures (Barber-Lomax *et al.* 2022).

As Ashby & Heinemeyer (2021) outlined, rewetting is unlikely to end the need for wildfire-related vegetation management at UK peatland sites. The cessation of burning management in the experimental burn trials at Hard Hill (Moor House) caused a significant increase in above-ground biomass (Alday *et al.* 2015, Marrs *et al.* 2019a). This increase in above-ground biomass may provide a higher fuel load for potential wildfires. In turn, this could cause higher fire temperatures if a wildfire does occur (Hobbs & Gimingham 1984, Davies *et al.* 2016b, Noble *et al.* 2019). It is also essential to recognise that there are natural (climate, topography) limitations to rewetting a particular site and that not all sites are drained, thus offering limited or no potential for effective rewetting. This is in addition to the likelihood that taller heather vegetation will reduce peat moisture (Worrall *et al.* 2013, Heinemeyer *et al.* 2023), thus increasing the risk of a burn into the peat itself when a hot wildfire occurs (Heinemeyer & Ashby 2023).

Further undermining this generic statement is a lack of evidence on the future resilience of rewetted and restored 'healthy' peatlands, the limitations of a site's potential for wetness and fire risk/resilience, and the potential need for management approaches that reduce fuel loads to limit wildfire risk (with peat loss rather than only vegetation combustion). As stated by Ashby & Heinemeyer (2021), this balance between a view of an unmanaged 'healthy' resilient peatland and current management approaches is poorly understood because peatland restoration has a very small amount of robust (BACI) and long-term impact monitoring studies associated with it (and seemingly none in relation to wildfire mitigation) to test such assumptions or hypotheses.

Overall, there is no direct evidence for the relationship between the rewetting of peatlands, cessation of prescribed burning and subsequent resilience of peatlands to wildfire. This statement needs measurements, trials and models to test if, where and to what degree rewetting provides resilience to wildfire. This will be particularly important considering the increasingly frequent warmer and drier spring/summer 'blocking' weather conditions expected due to climate change (Barber-Lomax *et al.* 2022) and the effects this may have on peatland resilience to future wildfires.

5. Cessation of heather burning results in wetter peat, less heather cover and no need to burn

This statement directly challenges the use of prescribed burning as a management tool and draws upon similar evidence-based resolutions as in Statement 4. Similarly, the evidence base for Statement 5 draws directly on the narrative of 'wetter is better' combined with the assumption that a restored 'healthy' peatland requires little to no management. This approach is often linked to a call for an ill-defined 'precautionary' policy approach to stop burning (e.g., IUCN 2020), which should not be used as a basis for decision-making in this evidence-limited context as outlined by Ashby & Heinemeyer (2021).

As outlined in Statement 4, the relationship between wetter conditions and declining heather cover, and the direct implications for improving management and peatland resilience (Glaves *et al.* 2020, IUCN 2020) seems purely based on opinions (Baird *et al.* 2019) or studies not applicable to the context (e.g., Granath *et al.* (2016) who studied the unrelated restoration of industrially drained and mined bare peat; see Ashby & Heinemeyer (2021)). Notably, in the long-term study by Heinemeyer *et al*. (2023), mature heather (~ 35+ years) was increasingly tallest (~ 45 cm) on the wettest site, and the overall cover was equally high (~ 75 %) on the driest and the wettest blanket bog sites, similar to a comparison of a dry heath to a wet raised bog by Grau-Andrés *et al*. (2018). Moreover, we know that management of upland peatland sites requires an understanding of site history and the conditions that may affect potential wetness and long-term trajectories in terms of response to climate, topography, management and restoration (if required). Other key factors to consider include the fuel load and the potential for increasing fuel loads with rewetting (Arkle *et al*. 2012) and, more importantly, the implications for long-term wildfire risk management scenario building versus no management approaches (Barber-Lomax *et al.* 2022).

A common assumption is that prescribed burning is the only vegetation management approach that negatively affects peatland condition, health and processes (Ashby & Heinemeyer 2021). However, alternative vegetation management approaches, such as cutting, are even less well studied and do not necessarily provide an alternative 'safe', 'beneficial' or 'only positive' solution (Heinemeyer *et al.* 2023). Prescribed burning as a tool for effective management of upland peatland and heathland sites has been linked to maintaining key aspects of biodiversity and management objectives (Vandvik *et al.* 2005, Worrall *et al.* 2009, Davies *et al.* 2016b, Whitehead *et al.* 2021) and, as outlined in Statement 4, unmanaged heather can result in drier peat (Worrall *et al.* 2007, 2013; Heinemeyer *et al.* 2023). It seems important to recognise that not every site will be as wet as the 'ideal' site elsewhere, especially when considering ecohydrological differences in peatland types (Glatzel *et al*. 2023); and that permanently inundated rewetting sites where heather will not thrive are likely to become high methane emitting sites (Gatis *et al*. 2023), especially under a warmer climate (Abdalla *et al*. 2016). There are clear limitations in this approach and what is needed is a site-by-site and a BACI trial approach based on growing evidence, comparing management methods in a robust and applicable way (Harper *et al.* 2018).

6. Seventy-five percent of global heather moorland is found in the UK

Since 1998, several organisations and publications have stated that the UK has the highest proportion of heather moorland (including lowland and upland areas) in the world, often followed by the assertion that it is a habitat rarer than global rainforests (see online post by Carver 2019). The 75 % figure appears to originate from two publications (Holden *et al.* 2007, Aebischer *et al.* 2010) citing previous literature (Gimingham *et al*. 1979, Tallis *et al.* 1997). Carver (2019) observes, however, that nobody has provided actual data to support this claim and has instead cited a calculation by Diemont *et al.* (1996).

This 75 % figure has been used by different stakeholders to promote their own narratives around the use and value of upland moorlands. It has been used both to support and to deter the use of prescribed burning on upland moorlands. This narrative tends to be split between the value of conserving current moorlands (e.g., The Moorland Association, The Countryside Alliance) and the value of these environments for grouse breeding (e.g., British Association for Shooting and Conservation, The Game and Wildlife Conservation Trust (GWCT), the National Gamekeepers Association). This claim could, therefore, be misused without consideration of its accuracy or its limitations in relation to justifying management options. However, many other organisations, including conservation groups and government departments, have also used this figure (Carver 2019).

As outlined by Carver (2019), this figure is an unsubstantiated claim based on two major issues:

1) The 75 % estimate is very uncertain as the data from Diemont *et al.* (1996) were based on only seven European countries and fail to account for the heather moorland cover figures from elsewhere in Europe (e.g., Ireland, Estonia, Finland) or across the globe (e.g., New Zealand, Australia). However, we note that this includes countries where heather is an invasive species.
2) The 75 % estimate was made prior to the development of the European Nature Information System (EUNIS) and is based too broadly on the term 'upland moorland vegetation', rather than considering the up to 62 different land cover classes that are likely to contain heather and other dwarf shrub species.

The online response by the GWCT (2020) outlined a rebuttal focusing on heather moorland definition by Thompson *et al*. (1995) on upland areas, which, therefore, considers only 12 different plant communities in which heather is dominant, as defined by the EU Habitats Directive. They also point out that many of the other heather habitats across the globe reflect an invasive species issue and should not be included as they are not comparable to natural heather moorland habitats.

Whatever the final conclusion or percentage, it is clear that this statement is currently based on poor definitions of heather moorland and lacks robust calculations of percentage coverage in the UK and globally. It has been estimated by Carver (2019) that, using the EUNIS classifications, the actual proportion of global heather moorland coverage in the UK could be as low as 13 %. We do not have the data to support an accurate estimate, and on this basis we would caution against the future use of percentages for heather moorland areas without specific calculations and acknowledgement of the large uncertainties around such estimates, calculations and definitions. However, we believe the heather moorland area in the UK is substantive and globally important.

7. Prescribed heather burning causes water colour and quality issues

We lack clear evidence on direct impacts of heather burning on water colour and quality within the UK (see sections in Davies *et al.* 2016b and in a review by Harper *et al.* 2018), and further research is required (Ramchunder *et al.* 2009, Worrall *et al.* 2009, Harper *et al*. 2018, Holland *et al*. 2022). Associating any impact is often compounded by the difficulty of directly associating one single factor, such as burning, among several confounding aspects (e.g., acidification, drainage, grazing, vegetation) to impacts on water colour and quality (Davies *et al.* 2016b, Ashby & Heinemeyer 2021). This is especially true for studies lacking an appropriate experimental design and controls (e.g., Brown *et al.* 2014 as highlighted by Allott *et al.* 2019).

Where the evidence does exist, there are some key considerations regarding robustness and the generality of findings. Holden *et al.* (2012) and Williamson *et al.* (2023), for instance, present contradictory evidence and raise uncertainties around confounding factors in observations and assessments of prescribed burning impacts, not only between laboratory and site evaluation but also between plot and catchment-level evaluation. For example, laboratory studies seem to indicate that burning increases colour production and has direct implications on water quality (Holden *et al.* 2012). However, the same study demonstrates that the laboratory evidence does not translate into the field, particularly when dealing with plot and catchment studies and considering temporal scales, challenges in measurement equipment and the complexity of site hydrology. Coincidentally, a recent BACI peatland restoration study about grip blocking effects on water quality showed no clear positive (Peacock *et al*. 2018) and even some lasting negative water quality implications (Gatis *et al.* 2023). Williamson *et al.* (2023) raise uncertainty issues regarding the use of regression analysis results to infer impacts on DOC concentration from managed catchments, including burning, with little evidence to support negative or positive effects of management apart from afforestation and catchment or site-specific aspects (peat cover, rainfall and altitude). The mixed outcomes of previous studies, and likely causes such as limitations in time and space and no adequate controls, are presented and discussed in a review by Harper *et al.* (2018), which suggests that drainage is likely to be the major cause of increased DOC concentration (Williamson *et al.* 2023). Moreover, long-term climatic effects have explained deteriorating water quality (Klante *et al*. 2021), thus confounding tests of management, and the lack of any clear and robust evidence linking prescribed burning to water quality issues is confirmed by the Holland *et al.* (2022) review, which also cites a supporting discussion by Davies *et al.* (2016c).

8. Prescribed heather burning causes flooding

This statement relates directly to the narrative that prescribed rotational burning of heather on peatlands is potentially linked to flooding (e.g., review by Brown *et al.* 2015) and should, therefore, cease as a method of peatland management. Although the review indicates little evidence to support a clear link between burning and flooding, this claim has fed directly into an IUCN position statement on 'Burning and Peatlands' (IUCN 2020, 2023), which forms part of a series of their key recommendations on the management of upland peatlands. Moreover, one of the key studies in Brown *et al.* (2014) (specifically, Holden *et al*. 2015; part of the EMBER project), as cited by the IUCN (2020, 2023), is experimentally constrained as it lacked an adequate control (Allott *et al.* 2019) for generic site differences and confounding factors (Ashby & Heinemeyer 2019). Moreover, Holden *et al*. (2015) did not investigate any flooding-related aspects.

The challenge with this statement also seems to be that there is a perception that peatlands act as natural water storage and flood alleviation systems, holding large amounts of water like 'sponges' (see Bacon *et al.* 2017), which is an analogy often used when referring to the negative influence of prescribed burning. As outlined by IUCN (2020), the key areas that are often attributed to increased flooding from prescribed burning focus around:

1) Loss of vegetation cover: prescribed burning removes the vegetation, which plays a crucial role in regulating water flow by intercepting rainfall, promoting infiltration, and reducing and slowing runoff. Without vegetation, rainwater runs off the burned surface more quickly, leading to increased surface runoff and the potential for flooding (e.g., Grayson *et al.* 2010, Holden *et al.* 2015).
2) Alteration or damage to peatland properties: it has been stated that burning can change the physical properties of the peatland's acrotelm (top layers), and the impact on micro-erosion (Clutterbuck *et al.* 2020) reduces the ability of peatlands to absorb water, increasing surface runoff (Holden *et al*. 2015) and thus potentially contributing to flooding (IUCN 2020).

According to Bacon *et al.* 2017, "*the perception created by the sponge analogy, that peatlands can soak up most rainwater and thereby reduce downstream flood risk, is not the reality in most cases*". Furthermore, the 'sponge' analogy undermines the need for a site-by-site assessment, as it fails to recognise crucial variables that can affect susceptibility or resilience to flooding such as topography, vegetation and management (Bacon *et* *al.* 2017). The evidence required to substantiate this statement would be results from BACI studies to develop and compare burnt versus unburnt datasets over suitable timescales (Ashby & Heinemeyer 2021), and this needs to be conducted site-by-site, ideally at catchment scales (Allott *et al.* 2019), to better understand flood risk and resilience and how a site's current state and management may affect this.

Specifically, the first point regarding water flow on bare or burnt peat is related to some reported localised effects (Brown *et al.* 2014). Allott *et al*. (2019) note that there is no concrete evidence linking this to flooding, especially considering how unlikely it is for small patches of managed heather with total cover only a small proportion of a landscape to influence the entire catchment flow and thus downstream flooding (Heinemeyer & Ashby 2023). The second point regarding the effects of vegetation burning on peat physical properties and micro-erosion and their ecological importance again seems to lack any evidence to link it to flooding (Ashby & Heinemeyer 2021, Heinemeyer & Ashby 2023). Notably, Holden *et al*. (2015) collected data from burnt and unburnt catchments that were in geographically separate and environmentally distinct locations (rather than comparing adjacent catchments), which means their study has a high risk of bias. The only other catchment-scale assessment of burning impacts on flow to date uses a robust BACI approach to compare burning versus cutting management (i.e., both catchments are managed with unmanaged heather areas only compared at the plot scale), and the impacts are highly site dependent, with only two out of three sites indicating reduced flow from cut versus burnt (but notably not versus unmanaged) heather catchments (Heinemeyer *et al.* 2023). However, microtopography was greatly affected by heather cutting (Heinemeyer *et al.* 2019b).

9. Peatlands offer huge carbon sequestration potential and are climate change 'saviours'

This statement relates to the current perception of peatlands as a potential 'climate saviour' with significant carbon sequestration potential and a vital role in mitigating climate change, both globally (Leifeld & Menichetti 2018) and in the UK (Gregg *et al.* 2021). The source of this perception, which is mentioned in almost every written piece on peatlands, is often centred around the estimated global peatland cover of only around 3 % of the Earth's land surface, containing about 30 % of the global soil organic carbon stocks (e.g., Gorham 1991, GPA 2022). This vast storage reflects slow carbon accumulation in peatlands over millennia (during much of the Holocene), mostly under waterlogged and acidic conditions that slow down the decomposition of organic matter (Gorham 1991). This long-term sequestration certainly makes them valuable long-term carbon sinks (Harris *et al.* 2021). However, this carbon stock should not be confused with current interests in near-future carbon sequestration potential. Moreover, peatlands not only sequester carbon, but also release methane, a very potent GHG (e.g., Abdalla *et al*. 2016), especially at wet sites (Heinemeyer *et al*. 2023), and after rewetting (Vanselow-Algan *et al*. 2015, Gatis *et al*. 2023), especially in ditches (Cooper *et al*. 2014). Furthermore, peatlands' carbon sequestration and GHG mitigation potential are site-dependent, often limited and affected by environmental change.

When intact, peatlands act as net carbon and GHG sinks as they normally offset their methane loss in the long-term (Harris *et al.* 2021). Many peatlands, especially those that have been drained for agriculture, are now contributing 5–10 % of the global anthropogenic carbon dioxide emissions annually (Loisel & Gallego-Sala 2022). Clearly, the obvious solution is to invest in peatland restoration, reverting drainage and revegetating bare peat to stop this emission and restart peat formation and carbon sequestration (Strack *et al.* 2022). Considering the immense peatland carbon stock and substantial sequestration potential, their role in the global carbon cycle and the multiple benefits they provide, we fully support these aspects of the statement. However, it is the claim of their value as a 'saviour' in the fight against climate change that is too simplistic.

The future potential of peatlands to sequester carbon and contribute to climate-change mitigation needs to be assessed on a regional or site-by-site basis. For example, the current state of a peatland in the UK depends largely (besides climate) on historic management or use (i.e., peat cutting, agricultural cultivation, drainage), which likely translates into potential future carbon accumulation and peat growth (Heinemeyer *et al.* 2019a). Moreover, peat accumulation rates generally show an asymptotic curve over time, as initial peat accumulation rates are much higher than those of old peatlands (Heinemeyer & Ashby 2023). This is especially true for blanket bogs, which are much more limited in their carbon accumulation potential (due to slope related fluvial export and higher water table fluctuations) than wetter peatlands such as raised bogs (Glatzel *et al.* 2023). Therefore, whereas restoration and rewetting of sites with huge historic losses will offer potentially enormous accumulation rates and thus carbon gains, sites with less historic disturbance or site-specific limitations are likely to offer only minimal gains. In a single-site context it is, therefore, important to know what management or restoration is likely to achieve in terms of additional carbon sequestration. Although approximately 39 % of UK peatlands sites are classified as 'unfavourable' (JNCC 2006, 2009), such terminology is not based on functions (Ashby & Heinemeyer 2021) and implicitly assumes that prescribed fire is damaging (Davies *et al.* 2016b), and, therefore, should not be used to make general predictions on generic potential carbon gains from 'restoration'.

Finally, all assumptions on future carbon gains are uncertain in the face of future environmental change, especially in terms of how increasing temperatures and precipitation changes (i.e., droughts) will affect peatland ecosystems (Gallego-Sala *et al.* 2010). A changing climate requires us to also consider potential risks to carbon gains from changing management and restoration in light of potential future losses due to extreme events like wildfire (see Statement 4). Climate change could potentially undermine all of the long-term benefits of peatland restoration unless we consider consequences like increased vegetation growth with increased fuel loads and likely drying out of peat alongside it. Notably, we lack data on peatland NECBs, net GHG emissions and any applicable evidence on how rewetting/restoration protects peatlands against wildfire (Ashby & Heinemeyer 2021), especially if vegetation/fuel load management is not carried out. Therefore, although substantial carbon gains and climate benefits from peatlands are important, they are site dependent, and it is most important to safeguard existing stocks against climate change impacts such as wildfire (Belcher *et al.* 2021). It seems more important to safeguard peatlands' existing carbon stocks and prevent further losses than to promote their anticipated future carbon gains as part of a climate 'saviours' storyline, which might literally 'go up in smoke' if we continue to ignore key evidence.

10. Prescribed heather burning causes loss of biodiversity

This statement relates directly to the reported loss of biodiversity in peatlands as a consequence of prescribed burning (Middleton *et al.* 2006, Ramchunder *et al.* 2009, Brown *et al.* 2013). The primary concern in this statement revolves around the unique biodiversity and habitat conservation opportunity that many peatlands present for plant and animal species. It is often assumed that prescribed burning negatively impacts peatland ecological functioning. Yet, the evidence indicates mixed results, depending on experimental and site context and timescales (Davies *et al.* 2016b,c, Harper *et al.* 2018, Ashby & Heinemeyer 2021), and the shift in functions due to prescribed burning may have cascading effects on the entire ecosystem and the species that rely on it (Middleton *et al.* 2006, Ramchunder *et al.* 2009, Littlewood *et al.* 2010, Brown *et al.* 2013).

The challenge with this statement is that (i) it generalises; (ii) biodiversity impact does not equal a loss; and (iii) there is very little evidence to support claims of "*significant adverse impacts*" of prescribed burning as stated in the IUCN (2020, 2023) position statements (Ashby & Heinemeyer 2021). In fact, the overall evidence base indicates varied impacts and findings on nearly all ecosystem aspects including the limited evidence on biodiversity impacts (i.e., Harper *et al.* 2018), highlighting complexity, inconsistency, site condition, scale and many other factors (Holland *et al.* 2022). There is certainly no evidence to allow a generalisation of overall negative impacts, even on *Sphagnum* moss (Holland *et al.* 2022); the same review also points out that often reported negative impacts are short-lived and need to be seen as part of a patchwork across larger scales (catchment/landscape).

There is a clear need to consider the state of the peatland, specific management practices, their temporal and spatial scale and the ecological context. Any management choice results in 'winners and losers' and biodiversity impacts need to be contextualised against conservation status of many species, including many bird species (i.e., curlew, lapwing and golden plover), not just plants. When prescribed burning is carefully planned, taking into account the specific characteristics of the peatland ecosystem and the conservation objectives, it is possible to minimise the negative impacts of burning whilst promoting positive aspects of biodiversity (Holland *et al.* 2022) and many other aspects (Harper *et al.* 2018), especially when compared to alternative cutting or no management (Heinemeyer *et al.* 2023). Proper management strategies, including monitoring and adaptive management, are essential to mitigate the risks and ensure the long-term conservation of biodiversity in peatlands. It is also important to consider management trade-offs (i.e., some species will benefit whilst others will lose out). Benefits to beta diversity at the ecosystem scale (e.g., Davies & Bodard 2015, Grau-Andrés *et al*. 2019b) likely increase gamma diversity at the landscape scale, but these often remain unconsidered in short-term patch/plot-level studies (i.e., considering long-term regrowth and patchworks of vegetation ages/communities). As Harper *et al.* (2018) conclude in their review "*Prescribed burning, under a changing climate, could either be a useful land* *management tool or a highly damaging process if implemented without sufficient impact research. Based on the current knowledge it is still unclear which category prescribed burning falls into in the UK*."

CONCLUSIONS AND RECOMMENDATIONS

We have provided an evidence-based assessment of ten key claims or statements commonly made in relation to the use of prescribed burning in the management of heather-dominated peatland. As stated by Ashby & Heinemeyer (2021), "*the complete range of impacts caused by burning to peatland ecosystem services remains unclear due to insufficient, contradictory or unreliable evidence*" (Davies *et al.* 2016b, Harper *et al.* 2018, Ashby & Heinemeyer 2019). Key evidence and recent literature reviews clearly indicate that certain claims and statements or their generalisations cannot be upheld. We have identified several reasons for rejecting or questioning the ten statements, but there are clear commonalities between several of them, which translates into similar recommendations for how to address issues, evidence and knowledge gaps.

In Table 1 we summarise our responses to, and reasons to reject, each of the ten statements reviewed. We also outline recommendations for better assessing the validity of the ten statements.

We have identified four key aspects shared by the ten statements and how to address them within the evidence base:

1) **Definitions:** Studies need to improve on agreed and universal definitions, notably for 'heather moorland', 'peat-forming' species, habitat conditions like 'degraded', 'modified', 'intact'. Also, national/global agencies should start to link condition assessments to functions rather than to arbitrary criteria such as key species or management presence/absence. Ideally this would be achieved by inclusive, multi-stakeholder (policy, academia, industry, non-governmental organisations, end users) focus groups (e.g., Reed *et al.* 2022) to capture different views, avoid bias and agree on outcomes.
2) **Methodology:** Experimental studies must improve on and/or evaluate their methodologies, including pseudo-replication (limiting generalisation), multi-site analysis (capturing a range of conditions), space-for-time (unknown confounding factors), BACI approach (accounting for possible confounding factors) and examining alternative management regimes (allowing direct comparisons). Study limitations and their implications for interpreting results should be acknowledged and clarified. This should not necessarily prohibit the use of a study but will greatly help when evaluating evidence in systematic reviews of a topic, which needs to independently assess such factors.
3) **Scale:** Evidence reviews need to carefully consider limitations in temporal and spatial scales when interpreting study findings. It does not help to report (i) short-term disturbance outcomes when they are interpreted as 'impacts' - often long-term outcomes differ substantially as there is a recovery and regrowth phase and likely a new trajectory; or (ii) small plot-scale assessments when they are interpreted as generic, landscape-scale impacts - often this does not capture catchment-scale processes important for hydrology and biodiversity as soils and landscapes are connected. Similarly, studies of only a few years do not capture enough climatic variability to allow credible upscaling beyond a decade. Ideally, 'gold standard' plot-to-catchment scale studies should be performed as part of long-term monitoring projects within a BACI design, allowing various aspects and processes to be captured at the necessary scales. While no study is perfect, this does not justify ignoring obvious limitations and constraints in the interpretation of findings/results. Moreover, Government agencies and research councils should initiate and support such 'gold standard' research platforms as they are likely to deliver the most ecologically meaningful and relevant outcomes in relation to both policy evidence (Lindenmayer *et al.* 2012) and scientific impact (Hughes *et al.* 2017).
4) **Site:** Frequently, findings are site-specific because they reflect (historic) management, environment and climatic conditions and cannot be generalised without testing or monitoring several key variables. As shown for the first claim around carbon studies, much of the UK literature on upland peatlands focuses on one site (i.e., Hard Hill plots on Moor House; see Ashby 2020). Ideally, multi-site studies or critical meta-analyses of existing studies, also considering site and historical information, should be conducted to obtain or identify robust and general findings or outcomes (considering all the enumerated study limitations).

In conclusion, legislating against or banning prescribed burning as a management tool for heather-dominated peatlands, even on precautionary grounds, remains neither a well-informed nor an evidence-based solution (Davies *et al.* 2016b,c, Harper *et al.*

Table 1. A summary of the ten common statements made in relation to heather-dominated peatlands and their vegetation management in the UK that have been reviewed in this article, with our responses. The final column outlines recommendations for better assessing the validity of the ten statements.

Statement summary	Response	Recommendations
1. Prescribed heather burning causes a net peat carbon loss and contributes to the climate crisis.	Unsubstantiated, too generic, site and time dependent, confounding factors, not enough robust data to support/negate and several datasets to reject statement.	Methodological approach: randomised, multi-site, catchment-scale, BACI approach to test long-term impacts, NECB components, complete management cycle.
2. Fire and heather dominance is a result of recent management changes.	Unsubstantiated, too generic, confounding factors, ill-defined and several data sets to reject statement.	Definition: better define and contextualise the evidence for 'recent' management in the context of ecological history on a site-to-site basis, assess/consider confounding factors.
3. Prescribed heather burning reduced *Sphagnum* abundance and peat formation.	Unsubstantiated, too generic, ill-defined terminology and no robust data to support but several to reject statement.	Definition: better define peat-forming (enhancing/supporting). Methodological approach: more measurements on the role and amount of *Sphagnum* and other plant species in peat formation.
4. Rewetting reduces heather dominance and thus protects peatlands against wildfire.	Unsubstantiated as no adequate data or models to support statement, site dependent.	Methodological approach: measurements, trials and models to test if, where and to what degree rewetting provides resilience to wildfire.
5. Cessation of heather burning results in wetter peat, less heather cover and no need to burn.	Unsubstantiated as too generic, site and time dependent, no data to support statement and confounding factors.	Methodological approach: BACI approach, measurements, trials and models to test different long-term scenarios/conditions.
6. Seventy-five percent of global heather moorland is found in the UK.	Almost certainly wrong but likely large proportion, ill-defined habitat, too vague and lack of consistent data.	Definition: better define heather moorland for UK calculations. Methodological approach: include adequate and consistent data for specific definitions/habitats.
7. Prescribed heather burning causes water colour and quality issues.	Unclear, lacking robust data to support or negate statement, confounding factors.	Methodological approach: BACI approach, multi-factor trials/analyses to identify causation.
8. Prescribed heather burning causes flooding.	Unsubstantiated, lacking robust data to support or negate statement, confounding factors.	Methodological approach: BACI approach, multi-factor trials/analyses to identify causation.
9. Peatlands offer huge carbon sequestration potential and are climate change 'saviours'.	Ill-defined parameters, site and time dependent, too generic and likely misleading hyperbole statement.	Definitions: site and time dependence of peat accumulation rates (considering historic impacts, current conditions, future climate). Methodological approach: more long-term NECB and net GHG data for peatland categories/conditions.
10. Prescribed heather burning causes biodiversity loss.	Ill-defined, confounding factors, unclear and/or not enough data to support such generic statement, scale (time and space) dependent, several studies showing positive and/or negative depending on species and/or group.	Definitions: site and time dependence of biodiversity changes and too broad (e.g., which species/group?). Methodological approach: BACI approach, multi-factor assessments to identify causation, catchment/landscape-scale trials/analyses to identify overall impacts.

2018, Heinemeyer & Ashby 2023). As stated recently by Smith *et al.* (2023) for UK lowland heathlands, this approach could put habitat conditions at risk and have negative implications related to several statements discussed here. Davies *et al.* (2016b) captures the dilemma around inadequate evidence, misinformation, misleading and false claims shaping management policy: "*Despite the complex, long-term role of fire in peatland management, there is a growing trend of simplifying the narrative around burning in the uplands of the UK. This can present managed burning as an ecological practice that is only ever damaging and responsible for the poor ecological condition of many heathland and peatland ecosystems*".

Based on all this information, we ask that all parties in this debate acknowledge (i) the outlined limitations, misleading and sometimes even false claims, and (ii) that instead of an outright ban and shift to other management options, a more robust and applicable evidence base is required for all management options based on replicated long-term plot-to-catchment scale studies with methodological approaches that allow separation of key issues, avoid pseudo-replication, include multiple sites and use the BACI approach to capture both short-term and long-term impacts alongside confounding factors. This procedure would ideally be done collaboratively with all major stakeholders to underpin an adaptive management approach based on UK wide management trials to capture risks and benefits for the ecosystems, their functions and services. Only this will provide a range of site-specific and ecologically meaningful data on long-term trajectories suitable for scenario upscaling to aid policymakers in reaching informed decisions on the best, and likely combinations of, heather management options within a site-specific and climate change context. A precautionary principle should not be applied to burning, especially when even less is known about consequences of the alternatives (Ashby & Heinemeyer 2021). Finally, in light of our findings and several previous assessments (e.g., Davies *et al.* 2016b,c, Harper *et al.* 2018, Ashby & Heinemeyer 2021, Heinemeyer & Ashby 2023), we suggest that reviews commissioned by statutory conservation agencies and peat conservation NGOs (i.e., IUCN 2020, 2023; Glaves *et al*. 2013, Werritty *et al*. 2019, Gregg *et al.* 2021, Holland *et al.* 2022, Noble *et al*. 2025) need to be reviewed to ensure that any online content and publicity material is critically assessed and rewritten to correct or acknowledge any false, misleading or unevidenced claims to ensure unbiased, applicable and sound evidence is presented to policymakers and land-user communities.

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AUTHOR CONTRIBUTIONS

AH conceived the assessment and prepared the first complete draft of the assessment, AH and MA identified the key claims based on interactions with many stakeholders over many years, and all other authors contributed to further manuscript edits.

REFERENCES

Abdalla, M., Hastings, A., Truu, J., Espenberg, M., Mander, Ü., Smith, P. (2016) Emissions of methane from northern peatlands: a review of management impacts and implications for future management options. *Ecology and Evolution*, 6, 7080–7102.

Aebischer, N., Ewald, J., Tapper, S. (2010) Driven grouse shooting in Britain: A form of upland management with wider conservation benefits. In: Allen, P. (ed.) *Ecologic and Economic Benefits of Hunting*, Proceedings of the World Symposium on Hunting Activities, World Forum on the Future of Sport Shooting Activities (WFSA), Rome, 186–201.

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 and Evolution Systems*, 17, 388–396.

Allott, T., Auñón, J., Dunn, C., Evans, M., Labadz, J., Lunt, P., MacDonald, M., Nisbet, T., Owen, R., Pilkington, M., Proctor, S., Shuttleworth, E., Walker, J. (2019) *Peatland Catchments and Natural Flood Management*. Report to the IUCN UK Peatland Programme's Commission of Inquiry on Peatlands Update, IUCN UK Peatland Programme, Edinburgh UK. Online at: https://research.bangor.ac.uk/en/publications/peatland-catchments-and-natural-flood-management, accessed 09 Aug 2025.

Arkle, R.S., Pilliod, D.S., Welty, J.L. (2012) Pattern and process of prescribed fires influence effectiveness at reducing wildfire severity in dry coniferous forests. *Forest Ecology and Management*, 276, 174–184.

Ascoli, D., Beghin, R., Ceccato, R., Gorlier, A., Lombardi, G., Lonati, M., Marzano, R., Bovio, G., Cavallero, A. (2009) Developing an Adaptive Management approach to prescribed burning: a long-term heathland conservation experiment in north-west Italy. *International Journal of Wildland Fire*, 18, 727–735.

Ashby, M.A. (2020) *The Effects of Managed Burning on Upland Peatland Biodiversity, Carbon and Water. A Review of the Post-Glaves et al. (2013) Evidence.* In: Peatland Protection The Science: Four Key Reports, The Uplands Partnership. Online at: https://keele-repository.worktribe.com/output/426269, accessed 09 Aug 2025.

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*, 57, 2112–2120.

Ashby, M., Heinemeyer, A. (2021) A critical review of the IUCN UK Peatland Programme's "Burning and Peatlands" Position Statement. *Wetlands*, 41, 56.

Bacon, K.L., Baird, A.J., Blundell, A., Bourgault, M.A., Chapman, P.J., Dargie, G., Dooling, G.P., Gee, C., Holden, J., Kelly, T.J., McKendrick-Smith, K.A., Morris, P.J., Noble, A., Palmer, S.M., Quillet, A., Swindles, G.T., Watson, E.J., Young, D.M. (2017) Questioning ten common assumptions about peatlands. *Mires and Peat*, 19, 12.

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.

Bannister, P. (1964) The water relations of certain heath plants with reference to their ecological amplitude. III. Experimental studies: general conclusions. *Journal of Ecology*, 52, 499–509.

Barber-Lomax, A., Battye, R., Gibson, S., Castellnou, M., Bachfischer, M. (2022) *Peak District National Park Wildfire Risk Assessment*. Peak District National Park Authority, Bakewell UK, 142 pp. Online at: https://www.peakdistrictwildfire.co.uk/, accessed 09 Aug 2025.

Belcher, C., Brown, I., Clay, G., Doerr, S., Elliott, A., Gazzard, R., Kettridge, N., Morison, J., Perry, M., Santin, C., Smith, T. (2021) *UK Wildfires and their Climate Challenges*. Expert led report prepared for the Third Climate Change Risk Assessment, Global Systems Institute, University of Exeter, 79 pp. Online at: https://www.ukclimaterisk.org/wp-content/uploads/2021/06/UK-Wildfires-and-their-Climate-Challenges.pdf, accessed 09 Aug 2025.

Bonn, A., Reed, M.S., Evans, C.D., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M.A., Birnie D. (2014) Investing in nature: developing ecosystem service markets for peatland restoration. *Ecosystem Services*, 9, 54–65.

Brown, L.E., Johnston, K., Palmer, S.M., Aspray, K.L., Holden, J. (2013) River ecosystem response to prescribed vegetation burning on blanket peatland. *PLoS One*, 8(11), e81023.

Brown, L.E., Holden, J., Palmer, S.M. (2014) *Effects of Moorland Burning on the Ecohydrology of River Basins: Key Findings from the EMBER Project*. University of Leeds, 33 pp. Online at: https://water.leeds.ac.uk/our-missions/mission-1/ember/, accessed 09 Aug 2025.

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(4), 1406–1425.

Carver, S. (2019) *No Moor Myths?* Landethicsblog. Online at: https://landethicsblog.wordpress.com/2019/09/17/no-moor-myths/, accessed 09 Aug 2025.

Chambers, F. (2022) The use of paleoecological data in mire and moorland conservation. *PAGES*, 30(1), 16–17.

Chambers, F., Crowle, A., Daniell, J., Mauquoy, D., McCarroll, J., Sanderson, N., Thom, T., Toms, P., Webb, J. (2017) Ascertaining the nature and timing of mire degradation: using palaeoecology to assist future conservation management in northern England. *AIMS Environmental Science*, 4, 54–82.

Clay, G.D., Worrall, F. (2011) Charcoal production in a UK moorland wildfire – How important is it? *Journal of Environmental Management*, 9(32), 676–682.

Clay, G.D., Worrall, F., Rose, R. (2010) Carbon budgets of an upland blanket bog managed by prescribed fire. *Journal of Geophysical Research: Biogeosciences*, 115, G04037.

Clay, G.D., Worrall, F., Aebischer, N.J. (2015) Carbon stocks and carbon fluxes from a 10-year prescribed burning chronosequence on a UK blanket peat. *Soil Use and Management*, 31(1), 39–51.

Clutterbuck, B., Lindsay, R., Chico, G., Clough, J. (2020) *Hard Hill Experimental Plots on Moor House - Upper Teesdale National Nature Reserve: A Review of the Experimental Set Up*. Natural England Commissioned Report NECR321, 59 pp. Online at: https://publications.naturalengland.org.uk/file/4799418618609664, accessed 09 Aug 2025.

Cooper, M.D.A., Evans, C.D., Zielinski, P., Levy, P.E., Gray, A., Peacock, M., Norris, D., Fenner, N., Freeman, C. (2014) Infilled ditches are hotspots of landscape methane flux following peatland re-wetting. *Ecosystems*, 17, 1227–1241.

Crawford, A.J., Belcher C.M. (2022) A morphological analysis of Holocene charcoal particles from a peatland in southwest England. *Mires and Peat*, 28, 33.

Crowle, A.J.W., Glaves, D.J., Oakley, N., Drewitt, A.L., Denmark-Melvin, M.E. (2022) Alternative future land use options in the British uplands. *Ibis,* 164, 825–834.

Davidson, S.J., Van Beest, C., Petrone, R., Strack, M. (2019) Wildfire overrides hydrological controls on boreal peatland methane emissions. *Biogeosciences,* 16, 2651–2660.

Davies, G.M., Bodart, J. (2015) Changes in vegetation diversity and composition following livestock removal along an upland elevational gradient. *iForest - Biogeosciences and Forestry*, 8(5), 582–589.

Davies, G.M., Legg, C.J. (2011) Fuel moisture thresholds in the flammability of *Calluna vulgaris*. *Fire Technology*, 47, 421–436.

Davies G.M., Domènech, R., Gray, A., Johnson, P.C.D. (2016a) Vegetation structure and fire weather influence variation in burn severity and fuel consumption during peatland wildfires. *Biogeosciences*, 13, 389–398.

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. (2016b) The role of fire in UK peatland and moorland management: the need for informed, unbiased debate. *Philosophical Transactions of the Royal Society B*, 371, 20150342.

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. (2016c) Informed debate on the use of fire for peatland management means acknowledging the complexity of socio-ecological systems. *Nature Conservation*, 16, 59–77.

DEFRA (2021) *England Peat Action Plan*. Department for Environment Food & Rural Affairs (DEFRA), UK Government, 37 pp. Online at: https://www.gov.uk/government/publications/england-peat-action-plan, accessed 09 Aug 2025.

Diemont, W.H., Webb, N., Degn, H.J. (1996) A pan European view on heathland conservation. In: *Proceedings National Heathland Conference,* 18–20 September 1996, New Forest, Hampshire, 21–32.

Ellis, C.J. (2008) Interactions between hydrology, burning and contrasting plant groups during the millennial-scale development of sub-montane wet heath. *Journal of Vegetation Science*, 19(5), 693–704.

Evans, C., Gauci, V. (2023) *Wetlands and Methane Technical Paper* (Wetlands International). Online at: https://www.wetlands.org/publications/wetlands-and-methane/, accessed 09 Aug 2025.

Evans, C.D., Bonn, A., Holden, J., Reed, M.S., Evans, M.G., Worrall, F., Couwenberg, J., Parnell, M. (2014) Relationships between anthropogenic pressures and ecosystem functions in UK blanket bogs: Linking process understanding to ecosystem service valuation. *Ecosystem Services*, 9, 5–19.

Evans, C., Artz, R., Moxley, J., Smyth, M., Taylor, E., Archer, N., Burden, A., Williamson, J., Donnelly, D., Thomson, A., Buys, G., Malcolm, H., Wilson, D., Renou-Wilson, F., Potts, J. (2017) *Implementation of an Emissions Inventory for UK Peatlands.* Report to the Department for Business, Energy and Industrial Strategy, Centre for Ecology and Hydrology, Bangor UK, 88 pp. Online at: https://uk-air.defra.gov.uk/assets/documents/reports/cat07/1904111135_UK_peatland_GHG_emissions.pdf, accessed 08 Aug 2025.

Evans, C., Artz, R., Burden, A., Clilverd, H., Freeman, B., Heinemeyer, A., Lindsay, R., Morrison, R., Potts, J., Reed, R., Williamson, J. (2022) *Aligning the Peatland Code with the UK Peatland Inventory*. Final SP0822 Report to Defra and the IUCN Peatland Programme, March 2022 (updated January 2023). Online at: https://nora.nerc.ac.uk/id/eprint/534668/1/N534668CR.pdf, accessed 09 Aug 2025.

Flanagan, N.E., Wang, H., Winton, S., Richardson, C.J. (2020) Low-severity fire as a mechanism of organic matter protection in global peatlands: thermal alteration slows decomposition. *Global Change Biology*, 26, 3930–3946.

Frolking, S., Roulet, N.T., Tuittila, E., Bubier, J.L., Quillet, A., Talbot, J., Richard, P.J.H. (2010) A new model of Holocene peatland net primary production, decomposition, water balance, and peat accumulation. *Earth Systems Dynamics*, 1, 1–21.

Fyfe, R.M, Woodbridge, J. (2012) Differences in time and space in vegetation patterning: Analysis of pollen data from Dartmoor, UK. *Landscape Ecology*, 27, 745–760.

Fyfe, R., Brown, A., Rippon, S. (2003) Mid-to late-Holocene vegetation history of greater Exmoor, UK: estimating the spatial extent of human-induced vegetation change. *Vegetation History and Archaeobotany*, 12, 215–232.

Fyfe, R.M., Ombashi, H., Davies, H.J., Head, K. (2018) Quantified moorland vegetation and assessment of the role of burning over the past five millennia. *Journal of Vegetation Science*, 29, 393–403.

Gallego-Sala, A.V., Clark, J.M., House, J.I., Orr, H.G., Prentice, I.C., Smith, P., Farewell, T., Chapman, S.J. (2010) Bioclimatic envelope model of climate change impacts on blanket peatland distribution in Great Britain. *Climate Research*, 45, 151–162.

Gao, C., Cong, J., Sun, Y., Han, D., Wang, G. (2022) Variability in pyrogenic carbon properties generated by different burning temperatures and peatland plant litters: Implication for identifying fire intensity and fuel types. *International Journal of Wildland Fire*, 31, 395–408.

Garnett, M., Ineson, P., Stevenson, A. (2000) Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK. *The Holocene*, 10, 729–736.

Gatis, N., Benaud, P., Anderson, K., Ashe, J., Grand-Clement, E., Luscombe, D.J., Puttock, A., Brazier, R.E. (2023) Peatland restoration increases water storage and attenuates downstream stormflow but does not guarantee an immediate reversal of long-term ecohydrological degradation. *Scientific Reports*, 13, 15865.

Gillingham, P., Stewart, J., Binney, H. (2016) *The Historic Peat Record: Implications for the Restoration of Blanket Bog.* Evidence Review NEER011, Natural England, Peterborough, 28 pp. Online at: https://publications.naturalengland.org.uk/publication/5155418650181632, accessed 09 Aug 2025.

Gimingham, C.H., Chapman, S.B., Webb, N.R. (1979) European heathlands. In: Specht, R.L. (ed.) *Heathlands and Related Dwarf Shrublands*, Ecosystems of the World 9A, Elsevier, Amsterdam, 365–413.

Gjedrem, A.M., Log, T. (2020) Study of heathland succession, prescribed burning, and future perspectives at Kringsjå, Norway. *Land*, 9(12), 485.

Glatzel, S., Worrall, F., Boothroyd, I.M., Heckman, K. (2023) Comparison of the transformation of organic matter flux through a raised bog and a blanket bog. *Biogeochemistry*, 167(4), 443–459.

Glaves, D., Morecroft, M., Fitzgibbon, C., Owen, M., Phillips, S.K., Leppitt, P. (2013) *The Effects of Managed Burning on Upland Peatland Biodiversity, Carbon and Water*. Evidence Review NEER004, Natural England, Peterborough, 155 pp. Online at: https://publications.naturalengland.org.uk/publication/5978072, accessed 09 Aug 2025.

Glaves, D., Crowle, A.J.W., Bruemmer, C., Lenaghan, S.A. (2020) *The Causes and Prevention of Wildfire on Heathlands and Peatlands in England*. Evidence Review NEER014, Natural England, Peterborough, 93 pp. Online at: https://publications.naturalengland.org.uk/publication/4741162353295360, accessed 09 Aug 2025.

Gorham, E. (1991) Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, 1(2), 182–195.

GPA (2022) *UNEP Global Peatlands Assessment: The State of the World's Peatlands: Evidence for Action Toward the Conservation, Restoration, and Sustainable Management of Peatlands*. Online at: https://www.unep.org/resources/global-peatlands-assessment-2022, accessed 09 Aug 2025.

Granath, G., Moore, P.A., Lukenbach, M.C., Waddington, J.M. (2016) Mitigating wildfire carbon loss in managed northern peatlands through restoration. *Science Reports*, 6, 28498.

Grau-Andrés, R., Davies, G.M., Gray, A., Scott, E.M., Waldron, S. (2018) Fire severity is more sensitive to low fuel moisture content on *Calluna* heathlands than on peat bogs. *Science of the Total Environment*, 616, 1261–1269.

Grau-Andrés, R., Gray, A., Davies, G.M., Scott, E.M., Waldron, S. (2019a) Burning increases post-fire carbon emissions in a heathland and a raised bog, but experimental manipulation of fire severity has no effect. *Journal of Environmental Management*, 233, 321–328.

Grau-Andrés, R., Davies, G.M., Waldron, S.R., Scott, E.M., Gray, A. (2019b) Increased fire severity alters initial vegetation regeneration across *Calluna*-dominated ecosystems. *Journal of Environmental Management*, 231, 1004–1011.

Gray, A., Davies, G.M., Domènech, R., Taylor, E., Levy, P.E. (2021) Peatland wildfire severity and post-fire gaseous carbon fluxes. *Ecosystems*, 24, 713–725.

Grayson, R., Holden, J., Rose, R. (2010) Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology*, 389, 336–343.

Gregg, R., Elias, J.L., Alonso, I., Crosher, I.E., Muto, P., Morecroft, M.D. (2021) *Carbon Storage and Sequestration by Habitat: A Review of the Evidence (Second Edition)*. Research Report NERR094, Natural England, York, UK. Online at: https://publications.naturalengland.org.uk/publication/5419124441481216, accessed 09 Aug 2025.

GWCT (2020) *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 (GWCT), Fordingbridge, Hampshire, 55 pp. Online at: https://www.gwct.org.uk/media/1157594/GWCT-Peatland-Report-2020-lr.pdf, accessed 09 Aug 2025.

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.

Harris, L.I., Richardson, K., Bona, K.A., Davidson, S.J., Finkelstein, S.A., Garneau, M., McLaughlin, J., Nwaishi, F., Olefeldt, D., Packalen, M., Roulet, N.T. (2021) The essential carbon service provided by northern peatlands. *Frontiers in Ecology and the Environment*, 20(4), 222–230.

Heinemeyer, A., Ashby, M. (2023) Prescribed fire in UK heather-dominated blanket bog peatlands: A critical review of “carbon storage and sequestration by habitat: A review of the evidence (Second Edition)” by Gregg *et al.* 2021. *Fire,* 6(5), 204.

Heinemeyer, A., Asena, Q., Burn, W.L., Jones, A.L. (2018) Peatland carbon stocks and burn history: blanket bog peat core evidence highlights charcoal impacts on peat physical properties and long-term carbon storage. *Geo*: *Geography and Environment*, 5, e00063.

Heinemeyer, A., Vallack, H.W., Morton, P.A., Pateman, R., Dytham, C., Ineson, P., McClean, C., Bristow, C., Pearce-Higgins, J.W. (2019a) *Restoration of Heather-Dominated Blanket Bog Vegetation on Grouse Moors for Biodiversity, Carbon Storage, Greenhouse Gas Emissions and Water Regulation*: *Comparing Burning to Alternative Mowing and Uncut Management.* Final Report, Defra Project BD5104, Stockholm Environment Institute University of York, UK, 252 pp. (plus Appendix by Richard A. Lindsay). Online at: https://sciencesearch.defra.gov.uk/ProjectDetails?ProjectId=17733, accessed 09 Aug 2025.

Heinemeyer, A., Berry, R., Sloan, T.J. (2019b) Assessing soil compaction and micro-topography impacts of alternative heather cutting as compared to burning as part of grouse moor management on blanket bog. *PeerJ*, 7, e7298.

Heinemeyer, A., David, T., Pateman, R. (2023) *Restoration of Heather-Dominated Blanket Bog Vegetation for Biodiversity, Carbon Storage, Greenhouse Gas Emissions and Water Regulation: Comparing Burning to Alternative Mowing and Uncut Management*. Final 10-year Report to the Project Advisory Group of Peatland-ES-UK, University of York, UK, 188 pp. Online at: https://doi.org/10.15124/yao-2wtg-kb53, accessed 09 Aug 2025.

Heinemeyer, A., Morton, P.A., David, T., Holmes, T., Jones, A.L., Liu, B. (2025) Ecological implications of changes in vegetation elemental composition under different heather (*Calluna vulgaris*) managements on British blanket bog. *Journal of Environmental Management*, 392, 126720. https://doi.org/10.1016/j.jenvman.2025.126720

Hobbs, R.J., Gimingham, C.H. (1984) Studies on fire in Scottish heathland communities: I. Fire characteristics. *Journal of Ecology*, 72, 223–240.

Hodgkins, S.B., Richardson, C.J., Dommain, R., Wang, H., Glaser, P.H., Verbeke, B., Winkler, B.R., Cobb, A.R., Rich, V.I., Missilmani, M., Flanagan, N. (2018) Tropical peatland carbon storage linked to global latitudinal trends in peat recalcitrance. *Nature Communications*, 9(1), 3640, 13 pp.

Holden, J., Gascoign, M., Bosanko, N.R. (2007) Erosion and natural revegetation associated with surface land drains in upland peatlands. *Earth Surface Processes and Landforms*, 32(10), 1547–1557.

Holden, J., Chapman, P.J., Palmer, S.M., Kay, P., Grayson, R. (2012) The impacts of prescribed moorland burning on water colour and dissolved organic carbon: a critical synthesis. *Journal of Environmental Management*, 101, 92–103.

Holden J., Palmer S.M., Johnston K., Wearing C., Irvine B., Brown L.E. (2015) Impact of prescribed burning on blanket peat hydrology. *Water Resources Research*, 51, 6472–6484.

Holland, J.P., Pollock, M.L., Buckingham, S., Glendinning, J.P.G., McCracken, D.I. (2022) *Reviewing, Assessing and Critiquing the Evidence Base on the Impacts of Muirburn on Wildfire Prevention, Carbon Storage and Biodiversity*.

Research Report 1302, NatureScot, Inverness UK. Online at: https://www.nature.scot/doc/naturescot-research-report-1302-reviewing-assessing-and-critiquing-evidence-base-impacts-muirburn, accessed 09 Aug 2025.

Hughes, B.B., Beas-Luna, R., Barner, A.K., Brewitt, K., Brumbaugh, D.R., Cerny-Chipman, E.B., Close, S.L., Coblentz, K.E., De Nesnera, K.L., Drobnitch, S.T., Figurski, J.D. (2017) Long-term studies contribute disproportionately to ecology and policy. *BioScience*, 67(3), 271–281.

IUCN (2014) *International Union for Conservation of Nature UK Peatland Programme: Peat Bog Ecosystems: Structure, Form, State and Condition*. Briefing Note 2, IUCN, Edinburgh UK. Online at: https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2024-03/Briefing%202%20-%20Peat%20Bog%20Ecosystems%20Structure%2C%20Form%2C%20State%20and%20Condition.pdf, accessed 09 Aug 2025.

IUCN (2020) *International Union for Conservation of Nature UK Peatland Programme: Burning & Peatlands.* Position Statement Version 2, IUCN, Edinburgh UK. Online at: https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2020-03/IUCN%20UK%20PP%20Burning%20and%20Peatlands%20Position%20Paper%202020%20Update.pdf, accessed 30 Oct 2024.

IUCN (2023) *International Union for Conservation of Nature UK Peatland Programme: Burning & Peatlands.* Position Statement, Version 4, IUCN, Edinburgh UK. Online at: https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2023-04/Position%20Statement%20-%20Burning%20and%20Peatlands%20V4%20-%20FINAL_1.pdf, accessed 30 Oct 2024.

JNCC (2006) *Common Standards Monitoring for Designated Sites: 2006 Report.* Joint Nature Conservation Committee, Peterborough, UK. Online at: https://hub.jncc.gov.uk/assets/15967de5-9da9-4d1f-b067-a8e76549bdca, accessed 09 Aug 2025.

JNCC (2009) *Common Standards Monitoring Guidance for Upland Habitat: 2009 Report.* Joint Nature Conservation Committee, Peterborough, UK. Online at: https://data.jncc.gov.uk/data/78aaef0b-00ef-461d-ba71-cf81a8c28fe3/CSM-UplandHabitats-2009.pdf, accessed 09 Aug 2025.

Klante, C., Larson, M., Persson, K.M. (2021) Brownification in Lake Bolmen, Sweden, and its relationship to natural and human-induced changes. *Journal of Hydrology*, 36, 100863.

Kopittke, G.R., Tietema, A., van Loom, E.E., Kalbitz, K. (2013) The age of managed heathland communities: implications for carbon storage? *Plant and Soil*, 369, 219–230.

Lee, J.A., Baxter, R., Emes, M.J. (1990) Responses of *Sphagnum* species to atmospheric nitrogen and sulphur deposition. *Botanical Journal of the Linnean Society*, 104(1–3), 255–265.

Lee, H., Alday, J.G., Rose, R.J., O'Reilly, J., Marrs, R.H. (2013) Long-term effects of rotational prescribed burning and low-intensity sheep grazing on blanket-bog plant communities. *Journal of Applied Ecology*, 50, 625–635.

Leifeld, J., Menichetti, L. (2018) The underappreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, 9(1), 1–7.

Leifeld, J., Alewell, C., Bader, C., Krüger, J.P., Mueller, C.W., Sommer, M., Steffens, M., Szidat, S. (2018) Pyrogenic carbon contributes substantially to carbon storage in intact and degraded northern peatlands. *Land Degradation & Development*, 29, 2082–2091.

Lindenmayer, D.B., Likens, G.E., Andersen, A., Bowman, D., Bull, C.M., Burns, E., Dickman, C.R., Hoffmann, A.A., Keith, D.A., Liddell, M.J., Lowe, A.J. (2012) Value of long-term ecological studies. *Austral Ecology*, 37, 745–757.

Lindsay, R. (2010) *Peatbogs and Carbon: a Critical Synthesis to Inform Policy Development in Oceanic Peat Bog Conservation and Restoration in the Context of Climate Change*. Environmental Research Group, University of East London, 315 pp. Online at: https://repository.uel.ac.uk/item/862y6, accessed 09 Aug 2025.

Littlewood, N., Anderson, P., Artz, R., Bragg, O., Lunt, P., Marrs, R. (2010) *Peatland Biodiversity.* IUCN UK Peatland Programme, Edinburgh UK. Online at: https://www.iucn-uk-peatlandprogramme.org/sites/www.iucn-uk-peatlandprogramme.org/files/Review%20Peatland%20Biodiversity,%20June%202011%20Final.pdf, accessed 30 Oct 2024.

Loisel, J., Gallego-Sala, A. (2022) Ecological resilience of restored peatlands to climate change. *Communications Earth & Environment*, 3(1), 208.

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. (2019a) Experimental evidence for sustained carbon sequestration in fire-managed, peat moorlands. *Nature Geoscience*, 12, 108–112.

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. (2019b) Reply to: Validity of managing peatlands with fire. *Nature Geoscience*, 12(11), 886–888.

McCarroll, J., Chambers, F.M., Webb, J.C., Thom, T. (2017) Application of palaeoecology for peatland conservation at Mossdale moor, UK. *Quaternary International*, 432, 39–47.

Middleton, B.A., Holsten, B., Van Diggelen, R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Milligan, G., Rose, R.J., O'Reilly, J., Marrs, R.H. (2018) Effects of rotational prescribed burning and sheep grazing on moorland plant communities: results from a 60-year intervention experiment. *Land Degradation and Development*, 29, 1397–1412.

Noble, A., Crowle, A., Glaves, D.J., Palmer, S.M., Holden, J. (2019) Fire temperatures and *Sphagnum* damage during prescribed burning on peatlands. *Ecological Indicators,* 103, 471–478.

Noble, A., Glaves, D.J., Leppitt, P., Crowle, A., Key, D., Rodgers, A. (2025) *An Evidence Review Update on the Effects of Managed Burning on Upland Peatland Biodiversity, Carbon and Water*. Natural England Evidence Review NEER155, Natural England, York UK, 322 pp. Online at: https://publications.naturalengland.org.uk/file/5554225990598656, accessed 09 Aug 2025.

Peacock, M., Jones, T.G., Futter, M.N., Freeman, C., Gough, R., Baird, A.J., Green, S.M., Chapman, P.J., Holden, J., Evans, C.D. (2018) Peatland ditch blocking has no effect on dissolved organic matter (DOM) quality. *Hydrological Processes*, 32, 3891–3906.

Ramchunder, S.J., Brown, L.E., Holden, J. (2009) Environmental effects of drainage, drain-blocking and prescribed vegetation burning in UK upland peatlands. *Progress in Physical Geography*, 33(1), 49–79.

Reed, M.S., Young, D.M., Taylor, N.G., Andersen, R., Bell, N.G.A., Cadillo-Quiroz, H., Grainger, M., Heinemeyer, A., Hergoualc'h, K., Gerrand, A.M., Kieft, J., Krisnawati, H., Lilleskov, E.A., Lopez-Gonzalez, G., Melling, L., Rudman, H., Sjogersten, S., Walker, J.S., Stewart, G. (2022) Peatland core domain sets: building consensus on what should be measured in research and monitoring. *Mires and Peat*, 28, 26.

Santana, V.M., Alday, J.G., Lee, H., Allen, K.A., Marrs, R.H. (2016) Modelling carbon emissions in *Calluna vulgaris*-dominated ecosystems when prescribed burning and wildfires interact. *PLoS ONE*, 11, e0167137.

Shepherd, M., Labadz, J., Caporn, S., Crowle, A., Goodison, R., Rebane, M., Waters, R. (2013) *Restoration of Degraded Blanket Bog*. Natural England Evidence Review NEER003, Peterborough UK, 124 pp. Online at: https://publications.naturalengland.org.uk/publication/5724822, accessed 08 Aug 2025.

Sim, T.G., Swindles, G.T., Morris, P.J., Baird, A.J. and 83 others (2023) Regional variability in peatland burning at Mid-to High-Latitudes during the Holocene. *Quaternary Science Reviews*, 305, 108020.

Simmons, I. (2003) *The Moorlands of England and Wales: An Environmental History, 8000 BC – AD 2000*. Edinburgh University Press, Edinburgh UK, 288 pp.

Smith, B.A., Carpenter, D., Holland, J., Andruszko, F., Gathorne-Hardy, A., Eggerton, P. (2023) Resolving a heated debate: The utility of prescribed burning as a management tool for biodiversity on lowland heath. *Journal of Applied Ecology*, 60, 2040–2051.

Strack, M., Davidson, S.J., Hirano, T., Dunn, C. (2022) The potential of peatlands as nature-based climate solutions. *Current Climate Change Reports*, 8, 71–82.

Sun, T., Guzman, J.J., Seward, J.D., Enders, A., Yavitt, J.B., Lehmann, J., Angenent, L.T. (2021) Suppressing peatland methane production by electron snorkeling through pyrogenic carbon in controlled laboratory incubations. *Nature Communications,* 12, 4119.

Swindles, G.T., Morris, P.J., Mullan, D.J., Payne, R.J. and 34 others (2019) Widespread drying of European peatlands in recent centuries. *Nature Geoscience*, 12, 922–928.

Tallis, J.H., Meade, R., Hulme, P.D. (eds.) (1997) *Blanket Mire Degradation*: *Causes, Consequences and Challenges*. Macaulay Land Use Research Institute, Aberdeen UK, 222 pp.

Thompson, D.B.A., MacDonald, A.J., Marsden, J.H., Galbraith, C.A. (1995) Upland heather moorland in Great Britain: a review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation*, 71(2), 163–178.

Titterton, P., Hunt, M., Crouch, T., Pilkington, M. (2021) *A Case Study into the Estimated Amount of Carbon Released as a Result of the Wildfire that Occurred on Marsden in April 2019*. Report to Moors for the Future Partnership, Edale. Online at: https://www.moorsforthefuture.org.uk/our-work/our-projects/moorlife2020/moorlife2020-research-and-monitoring/carbon-audit, accessed 09 Aug 2025.

Uplands Management Group (2017) *MoorLIFE*

2020: Blanket Bog Land Management Guidance. Online at: https://www.moorsforthefuture.org.uk/our-work/our-projects/moorlife2020/conservation-works/blanket-bog-land-management-guidance, accessed 09 Aug 2025.
van Breemen, N. (1995) How *Sphagnum* bogs down other plants. *Trends in Ecology & Evolution*, 10(7), 270–275.
Vandvik, V., Heegaard, E., Måren, I.E., Aarrestad, P.A. (2005) Managing heterogeneity: the importance of grazing and environmental variation on post-fire succession in heathlands. *Journal of Applied Ecology*, 42, 139–149.
Vanselow-Algan, M., Schmidt, S.R., Greven, M., Fiencke, C., Kutzbach, L., Pfeiffer, E.M. (2015) High methane emissions dominated annual greenhouse gas balances 30 years after bog rewetting. *Biogeosciences*, 12, 4361–4371.
Wallén, B. (1992) Methods for studying below-ground production in mire ecosystems. *Suo*, 43, 155–162.
Ward, S.E., Bardgett, R.D., McNamara, N.P., Adamson, J.K., Ostle, N.J. (2007) Long-term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems*, 10, 1069–1083.
Webb, J.C., McCarroll, J., Chamber, F.M., Thom, T. (2022) Evidence for the Little Ice Age in upland northwestern Europe: Multiproxy climate data from three blanket mires in northern England. *The Holocene*, 32(5), 451–467.
Werritty, A., Hester, A., Jameson, A., Newton, I., Oddy, M., Reid, C., Davies, S., MacDonald, C., Smith, A., Thompson, D. (2019) *Grouse Moor Management Review Group: Report to the Scottish Government.* Scottish Government, Edinburgh UK, 93 pp. ISBN: 978-1-83960-434-8 (web only). Online at: https://www.gov.scot/publications/grouse-moor-management-group-report-scottish-government/, accessed 09 Aug 2025.
Whitehead, S.C., Baines, D. (2018) Moorland vegetation responses following prescribed burning on blanket peat. *International Journal of Wildland Fire*, 27, 658–664.
Whitehead, S., Weald, H., Baines, D. (2021) Post-burning responses by vegetation on blanket bog peatland sites on a Scottish grouse moor. *Ecological Indicators*, 123, 107366.
Wilkinson, S.L., Andersen, R., Moore, P.A., Davidson, S.J., Granath, G., Waddington, J.M. (2023) Wildfire and degradation accelerate northern peatland carbon release. *Nature Climate Change*, 13, 456–461.
Williamson, J.L., Tye, A., Lapworth, D.J., Monteith, D. and 29 others (2023) Landscape controls on riverine export of dissolved organic carbon from Great Britain. *Biogeochemistry*, 164, 163–184.
Worrall, F., Reed, M., Warburton, J., Burt, T.P. (2003) Carbon budget for British upland peat catchment. *Science of the Total Environment*, 312, 133–146.
Worrall, F., Armstrong, A., Adamson, J.K. (2007) The effects of burning and sheep-grazing on water table depth and soil water quality in a upland peat. *Journal of Hydrology*, 339, 1–14.
Worrall, F., Evans, M.G., Bonn, A., Reed, M.S., Chapman, D., Holden, J. (2009) Can carbon offsetting pay for upland ecological restoration? *Science of the Total Environment*, 408, 26–36.
Worrall, F., Clay, G.D., May, R. (2013) Controls upon biomass losses and char production from prescribed burning on UK moorland. *Journal of Environmental Management*, 120, 27–36.
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.
Young, D.M., Baird, A.J., Gallego-Sala, A.V., Loisel, J. (2021) A cautionary tale about using the apparent carbon accumulation rate (aCAR) obtained from peat cores. *Scientific Reports*, 11, 9547.

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