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A Critical Review of the IUCN UK Peatland Programme's "Burning and Peatlands" Position Statement

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Abstract
Despite substantial contrary evidence, there has been a growing tendency to present prescribed vegetation burning as a management practice that is always damaging to peatland ecosystems in the UK. This is exemplified by the "Burning and Peatlands" position statement published by the International Union for Conservation of Nature UK Peatland Programme. Indeed, while we strongly agree with several of the statements made within this position statement, it also contains a series of unverified assertions and misleading arguments that seemingly serve to simplify the narrative and paint prescribed burning as a wholly damaging peatland management tool. Given that this position statement is published by one of the UK's most prominent peatland conservation organisations, it is likely to be consulted when debating upland land use policy. Therefore, for the benefit of policymakers, we provide a point-by-point critical review of the "Burning and Peatlands" position statement. We also discuss several further points for researchers and policymakers to consider that are consistently ignored by those attempting to simplify the narrative about prescribed burning. Our aim in producing this discussion paper is to encourage the research and policy community to move towards an evidence-based position about prescribed burning impacts on UK peatlands.

Keywords Blanket bog · Evidence-based policy · Fire ecology · Peatlands · Prescribed burning · Upland land management

Introduction

In 2016, Davies et al. (2016b) published a seminal paper calling for an informed and unbiased debate about the impacts of prescribed vegetation burning on upland ecosystems in the UK (i.e. on heather *Calluna vulgaris* L. dominated areas). They made this plea because, despite a substantial amount of contrary evidence, there was a growing tendency to present prescribed burning as "*an ecological practice that is only ever damaging and responsible for the poor ecological condition of many heathland and peatland ecosystems*" (Davies et al. 2016b). Since 2016, the number of studies investigating prescribed burning impacts on UK peatlands has increased. And, as with the evidence presented by Davies et al. (2016b), these new studies suggest that prescribed burning has a variety of impacts (positive, negative or neutral) across a range of peatland ecosystem services (McCarroll et al. 2016a; McCarroll et al. 2016b; Buchanan et al. 2017; Chambers et al. 2017; Douglas et al. 2017; Grau-Andrés et al. 2017; Ludwig et al. 2017; McCarroll et al. 2017; Noble et al. 2017; Robertson et al. 2017; Grau-Andrés et al. 2018; Ludwig et al. 2018; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2018b; Whitehead and Baines 2018; Grau-Andrés et al. 2019a; Grau-Andrés et al. 2019b; Heinemeyer et al. 2019c; Littlewood et al. 2019; Marrs et al. 2019a; Noble et al. 2019a; Noble et al. 2019b). A conclusion that was also drawn by a more recent review of burning impacts by Harper et al. (2018). At the same time, a debate has ignited about the quantity and reliability of certain studies within the prescribed burning evidence base (Ashby and Heinemeyer 2019; Brown and Holden 2020).

Given these recent developments, one would expect that peatland researchers and conservation organisations would have begun to heed Davies et al.'s cogent plea (Davies et al. 2016b). But, alas, this is not the case: there have been a series of recent publications implying that prescribed burning only ever leads to negative impacts on peatland ecosystems in the UK (Thompson et al. 2016; Baird et al. 2019; Natural England

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2019; Wild Justice 2019; IUCN 2020). One such publication is the "*Burning and Peatlands*" position statement by the International Union for Conservation of Nature UK Peatland Programme (IUCN UK PP) (IUCN 2020). We strongly agree with several of the statements made within this document, especially those within the "*Areas for further consideration and research*" section (IUCN 2020). Nevertheless, the document contains a series of unverified assertions, scientific inaccuracies and misleading arguments that seemingly serve to simplify the narrative and paint prescribed burning as a peatland management tool that is damaging in *every* context (IUCN 2020). In contrast, alternative peatland management tools such as rewetting (e.g. by ditch or gulley blocking), are assumed to only ever have positive impacts (IUCN 2020). The document is also ill-defined in that it uses an overly broad title (e.g. "*Burning and Peatlands*") which belies the narrow focus of the document (e.g. prescribed vegetation burning within UK blanket bog habitat).

The lack of balance and scientific accuracy within the "*Burning and Peatlands*" position statement may reflect the fact that it has not been subject to peer-review by the wider research community. However, given that this document is published by one of the UK's most prominent peatland conservation organisations, it is very likely to be used to inform upland land use policy. Therefore, the document must stand up to academic scrutiny. To this end, we herein present a point-by-point critical review of the IUCN UK PP "*Burning and Peatlands*" document (IUCN 2020). We also highlight several additional points for researchers and policymakers to consider that are consistently ignored by those attempting to simplify the narrative about burning impacts (e.g. NGOs, environmentalists, journalists). We are neither pro nor anti burning; our aim in producing this discussion paper is to reiterate the plea of Davies et al. (2016b) and encourage the research and policy community to move towards an unbiased and evidence-based position about prescribed burning within UK peatlands. While this paper focuses on the burning and peatlands debate in the UK, the debate is of international significance because the themes and arguments within will be familiar to wetland ecologists and managers in other parts of the world.

A Point-by-Point Critical Review of the IUCN "Burning and Peatlands" Document

Below, we have separated each statement within the IUCN UK PP "*Burning and Peatlands*" document into individual boxes (IUCN 2020). Underneath each box, we describe where we agree and where we disagree with what the IUCN UK PP have written. It is worth noting that prescribed burning on UK peatlands is predominantly, if not exclusively, carried out within blanket bog habitats as part of grouse moor management. As international readers may be unfamiliar with blanket bog habitat and grouse moor management, these are described in the 'Definitions' box below.

Position Statement 1: "There is a consensus within the literature that burning is, or has the potential to be, damaging to peatlands. It is well-established that burning can degrade bog habitats, leading to reductions or loss of key bog species (plants and animals), development of micro-erosion networks, increased tussock formation and increased dominance of non-peat forming vegetation such as heathland species (e.g. heather Calluna vulgaris and the moss Hypnum jutlandicum)."

There is no such consensus within the literature that prescribed burning "*is*" damaging to peatlands. Several recent reviews and commentary papers demonstrate that the overall effect of burning on peatlands is unclear due to insufficient, contradictory or unreliable evidence (Davies et al. 2016b; Harper et al. 2018; Ashby and Heinemeyer 2019). Nevertheless, we do agree that, like any other disturbance-based land management interventions (e.g. grazing, mowing, hedge laying, coppicing), burning has "*the potential to be*" damaging, but only when applied in the wrong spatial, temporal or environmental context. For example, when used on the correct scale, in the right location (i.e. flat areas away from bare peat and watercourses) and during the right environmental conditions (i.e. 'cool burns' that are carried out on flat areas between October 1st and April the 15th, when the peat and moss layers have a high moisture content), prescribed burning leads to, at worst, only very minimal, small-scale and short-term damage to the moss and litter layers within UK peatlands (Davies et al. 2010b; Lee et al. 2013a; Kettridge et al. 2015; Taylor 2015; Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Milligan et al. 2018; Noble et al. 2018a; Grau-Andrés et al. 2019a; Grau-Andrés et al. 2019b; Heinemeyer et al. 2019c; Marrs et al. 2019a). Such negligible and transient damage should be judged against the longer-term impacts (positive, negative or neutral) of prescribed burning. Yet, this is not the case, and we assert that the negative view of prescribed burning largely comes from using short-term results (≤ 3 years) to infer long-term impacts over the full 15 to >25-year burning rotation (e.g. Turner and Swindles 2012; Ramchunder et al. 2013; Brown et al. 2015; Holden et al. 2014; Holden et al. 2015; Johnston and Robson 2015; Grau-Andrés et al. 2019b; Noble et al. 2019a; Noble et al. 2019b).

We also disagree that it is "*well established*" that burning leads to "*reductions or loss of key bog species (plants and animals)*" and the "*increased dominance of non-peat forming vegetation*". Notwithstanding the obvious short-term 'disturbance' impacts, much of the available evidence suggests that, over longer timescales, burnt areas of blanket bog can support similar levels of *Sphagnum* and *Eriophorum* spp. (purported 'peat-forming' plant species) to comparable unburnt or not recently burnt areas (e.g. Lee et al. 2013a; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2018b; Whitehead and Baines 2018; Grau-Andrés et al. 2019a). Admittedly, the *Sphagnum* data we have is largely for the most abundant species, *S. capillifolium* (e.g. Lee et al. 2013a; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2018b; Whitehead and Baines 2018; Grau-Andrés et al. 2019a). Nevertheless, if *Sphagnum* species as a group are indeed key 'peat-forming' species, then, from a peatland functioning perspective (with peat formation being a key peatland function), does it matter which individual *Sphagnum* species are present? If so, then the evidence for this claim should be cited within the document. Although, we doubt this evidence currently exists as we are unaware of any robust experimental study investigating the peat-forming capabilities of individual *Sphagnum* species.

Most of the studies investigating burning impacts on peatland fauna have focussed on birds and invertebrates (terrestrial and aquatic) (Glaves et al. 2013; Davies et al. 2016b; Thompson et al. 2016; Sotherton et al. 2017; Harper et al. 2018). Unsurprisingly, prescribed burning seems to benefit some taxa and harm others (Glaves et al. 2013; Davies et al. 2016b; Harper et al. 2018). Thus, the decision to use prescribed burning will depend on the taxa the land manager is trying to promote. More importantly, we lack data about the impacts of prescribed burning on mammals, reptiles, or amphibians that inhabit peatland ecosystems in the UK (Harper et al. 2018).

To claim that burning management (relative to no management) increases the abundance of 'non-peat-forming' species, such as *C. vulgaris* and *Hypnum jutlandicum,* contradicts most of the evidence base (Lee et al. 2013a; Alday et al. 2015; Milligan et al. 2018; Whitehead and Baines 2018; Grau-Andrés et al. 2019a; Heinemeyer et al. 2019c). However, we do accept that there are likely to be anecdotal cases or sites which have a long history of inappropriate burning (frequent high-severity burns), notably often alongside deep drainage, which have led to conditions favouring a gradual shift towards *Calluna* and pleurocarp mosses. Any general vegetation assessment needs to take account of site-specific conditions and different management combinations over a sufficiently long timescale (e.g. at least one burning rotation).

Our greatest concern with this statement is that the 'peat-forming' label is not supported by robust experimental evidence (i.e. evidence that enables us to ascribe causality). Instead, the 'peat-forming' label seems to be based on circumstantial evidence, such as greater quantities of *Sphagnum* fragments being found within peat cores during periods of rapid peat growth (Shepherd et al. 2013; Gillingham et al. 2016). But is this cause or effect? In other words, were these periods of rapid peat growth due to a greater abundance of *Sphagnum* or did periods of rapid peat accumulation coincide with conditions favourable to *Sphagnum* growth (e.g. very wet and acidic)? Contemporary evidence suggests that the relationship between *Sphagnum* abundance and peat (carbon) accumulation is unclear (Garnett et al. 2000; Marrs et al. 2019a; Piilo et al. 2019), and there are multiple UK based paleoecology studies in which core sections encompassing periods of rapid peat growth are dominated by non-*Sphagnum* plant fragments (Fyfe et al. 2003; Fyfe and Woodbridge 2012; Shepherd et al. 2013; Gillingham et al. 2016; McCarroll et al. 2017; Fyfe et al. 2018). Outside the UK, peatlands within Indonesia, the Amazon Basin and the Everglades (about a third of the world's peat stores) do not contain *Sphagnum* species (Bacon et al. 2017; Hodgkins et al. 2018). And, the suggestion that shrubs are 'non-peat-forming' is, for example, not supported by North American observations (e.g. Rollins et al. 1993).

Thus, it seems that any plant species (including *C. vulgaris*) can form peat in the right conditions (Shepherd et al. 2013; Gillingham et al. 2016). Indeed, from a scientific perspective, it is the hydrological (high water table) and environmental (e.g. low pH) conditions that determine whether peat forms, regardless of species composition (Gillingham et al. 2016). Nevertheless, by retaining water, reducing soil pH and/or altering peat chemistry, *Sphagnum* spp. may enhance or facilitate such peat-forming conditions where they are otherwise limiting (Clymo et al. 1984; Gorham 1957; van Breemen 1995; Bacon et al. 2017). Furthermore, by leaching phenolic compounds, woody species may inhibit peat decomposition as shown during dry periods (Fenner and Freeman 2020).

Lastly, as far as we are aware, the evidence documenting the relationship between prescribed burning and the "*development of micro-erosion networks*" and "*increased tussock formation*" within peatlands is scant. This process has only been investigated within one 0.54 ha area that is part of a larger experiment located within a wet and high-altitude site in the North Pennines, UK (e.g. the Hard Hill experiment) (Clutterbuck et al. 2020). Therefore, the claim that burning leads to the development of micro-erosion networks and tussock formation is highly speculative.

Position Statement 2: "The impacts of fire on bog habitat, and particularly the main peat forming Sphagnum species' ability to recover, depends on the frequency and intensity of the burn along with other factors such as prevailing soil water levels, intensity of livestock trampling, climate, altitude and the starting condition of the peatland."

Again, there is no robust causal evidence about the specific and relative peat-forming capabilities of different peatland plant species. Thus, we have no idea whether *Sphagnum* is the main peat-forming species, nor do we know the different peat-forming capacities of individual *Sphagnum* species. What we do know is that peat formation is context and site-specific, and is primarily driven by the environmental and edaphic conditions, which can be supported or enhanced by *Sphagnum* species (via moisture retention, pH reduction; litter quality) (Shepherd et al. 2013; Gillingham et al. 2016; Bacon et al. 2017).

Conversely, we agree that burning impacts on *Sphagnum* are dependent on the "*frequency and intensity of the burn*" and the "*prevailing soil water levels*". For example, there is a negative relationship between site wetness (soil and vegetation moisture) and damage to the moss, litter and peat layers within blanket bog habitat (Taylor 2015; Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Grau-Andrés et al. 2019a).

Position Statement 3: "Rotational burning on peatland leads to drier vegetation communities (wet heath and dry heath communities) or a shift towards their dominance (e.g. of Molinia) (Bruneau & Johnson, 2014). This is associated with changes to the ecosystem (e.g. increased erosion rates and reduced availability of soil moisture) that can result in significant adverse impact on peatland biodiversity, carbon emissions, drinking water quality and flood management (Brown et al. 2014)."

Position Statement 3 fails to consider the wider evidence base. The current evidence suggests that, compared to unburnt or not recently burnt areas, burnt areas of upland peatland can support a similar abundance of wetland plants, such as *Sphagnum* and *Eriophorum* spp.; and a lower abundance of plants indicative of drier conditions (e.g. *C. vulgaris*), at least in the short-term (Lee et al. 2013a; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2018b; Whitehead and Baines 2018; Grau-Andrés et al. 2019a; Heinemeyer et al. 2019c).

We are also unaware of any study which shows that prescribed burning leads to a peatland dominated by *Molinia caerulea*. Much hotter and more severe wildfires that burn into the peat *can* increase *M. caerulea* on wet heath and acid grassland, but this should not be confused with the impacts of prescribed burning on deep peat (as it has been in some documents, e.g. Tucker 2003). In reality, 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 and Heinemeyer 2019). We would also argue that the EMBER report and associated peer-reviewed studies are unreliable in their current form (Brown et al. 2013; Brown et al. 2014; Brown et al. 2015; Holden et al. 2014; Holden et al. 2015); since they have serious methodological flaws that have yet to be addressed (e.g. the failure to control for pseudoreplication or the confounding of study site and corresponding differences in environmental conditions with burnt and unburnt treatments) (Ashby and Heinemeyer 2019; Brown and Holden 2020). Therefore, until this issue is resolved, the EMBER report studies should not be cited to support the claim that burning has a "*significant adverse impact on peatland biodiversity, carbon emissions, drinking water quality and flood management*" and a much wider evidence base needs to be considered.

Finally, any assessment of burning impacts on carbon emissions must also consider methane fluxes, especially given the recent evidence that suggests low-severity fires may suppress peatland methane emissions (Davidson et al. 2019; Gray et al. 2020).

Position Statement 4: "The majority of UK peatlands are in a degraded state as a result of various factors including drainage, burning, atmospheric pollution and high livestock numbers (JNCC 2011; Artz et al. 2019)."

Position Statement 4 on habitat status lacks nuance because it combines peatlands that are 'unfavourable-recovering' with those that are just 'unfavourable'. Using blanket bogs within UK Special Areas of Conservation (SAC) as an example, 45%, 14% and 39% are in 'Favourable', 'Unfavourable-recovering' and 'Unfavourable' condition, respectively (2% of this habitat has been destroyed) (JNCC 2006). Therefore, the majority of blanket bogs in SACs (59%) are on a positive ecological trajectory (i.e. bogs that are 'Favourable' or 'Unfavourable-recovering') (JNCC 2006). In fact, the majority of all peatland types (e.g. lowland fens and marshes, upland fens and marshes, lowland raised bogs) are on a positive ecological trajectory (JNCC 2006).

Perhaps more importantly, the criteria used to assess peatland condition is made up of arbitrary pass-fail criteria that do not measure important ecosystem parameters and functions, such as water table depth (WTD) and peat accumulation (JNCC 2009). Thus, in reality, we have no idea about how much of the UK peatland resource is "degraded". It could be that some bogs currently classed as unfavourable (i.e. degraded) are actually in good ecological condition (and vice versa), that is, they have water tables at or near the bog surface and are actively accumulating peat. Finally, this Position Statement 4 includes drainage alongside burning, which are often confused (e.g. Young et al. 2019) or overlooked within the evidence base (i.e. natural and artificial drainage is often an overlooked confounding factor in peatland burning studies). It is very clear that deep drainage has negative impacts on key peatland functions such as C storage (e.g. Young et al. 2019). But when commenting specifically on burning, we must consider this management intervention in isolation, especially as many remaining drainage ditches are actively blocked or have naturally infilled over time.

Position Statement 4a: "Compared to intact peatlands, degraded peatlands generally show: a higher proportion of dwarf shrub and graminoid (grasses and sedges) abundance; reduced Sphagnum bog moss abundance and diversity of typical bog species; vegetation structural changes such as loss of bog moss hummocks and pools; greater development of tussock and micro-erosion microtopography; denser, more degraded surface peat; a lower water table"

Apart from WTD, the points provided as indicative of a degraded peatland are not direct measurements of peatland degradation. Nevertheless, they may serve as proxies of peatland state, but we need experimental evidence to confirm and define this. To determine whether a peatland is degraded, we need robust empirical measurements of peat accumulation, WTD and other important ecosystem services, such as water quality and in relation to favourable or "intact" status. If such measurements (or their evidence-based proxies) indicate that a peatland has a low water table, is losing (rather than accumulating) peat and has low levels of relevant ecosystem service provision, then it should be classed as functionally degraded. However, conflicting outcomes should be expected for different ecosystem services under different land management scenarios (e.g. Bennett et al. 2009; Power 2010). For example, should one expect a permanently saturated peatland (i.e. a peatland with a water table at or above the peat surface) to have a high catchment flood mitigation potential or to be a net greenhouse gas (GHG) sink (e.g. wetter peatland sites have higher methane emissions, see Abdalla et al. 2016)?

Position Statement 5: "One of the sources of confusion around the impact of management activity on peatland is the misunderstanding as to what constitutes degraded and favourable condition, and failure to assess management trajectories. This is also reflected in some academic studies, which have inconsistent approaches to describing peatland vegetation, the state of peatland or the management objectives for the peatland. Indeed, many published journal papers do not adequately describe, or take account of, the type or current condition of the peatland under investigation."

It is our view that the confusion surrounding degraded and favourable condition is due to the fact they are not objective and evidence-based criteria (JNCC 2009). Rather, they are arbitrary criteria centred around 'typical' vegetation communities (see, for example, the criticisms in Davies et al. 2016b). If the most important aspect of a peatland is the peat itself, then a simple and objective definition of favourable condition could be whether a peatland is accumulating (rather than losing) peat. However, such a definition requires accurate peat accumulation data to be collected from across the peatland site being designated, which may be cost and time prohibitive.

An alternative approach would be to conduct robust experimental research at a broad range of representative sites as part of a national assessment and/or to use the same sites to validate proxies or tests of positive peat accumulation that can be rapidly assessed in the field. Such an assessment becomes increasingly complicated when other confounding management (e.g. drainage or grazing), site conditions (e.g. topography; climate) and additional ecosystem services are included (e.g. net GHG emissions; water quality; biodiversity). To our knowledge, such a detailed assessment is yet to be done.

Given the subjective and unscientific nature of the current peatland condition criteria and the lack of criteria in relation to burning, we also question why it is relevant if a study adequately describes or takes account of the "*current condition of the peatland under investigation*". What we need to know is how burning effects peatland functions relative to site-specific baseline conditions and independently of other impacts such as (deep) drainage, (over) grazing and atmospheric pollution. Such studies must also adequately control for environmental and ecological differences between treatment plots and study sites (Ashby and Heinemeyer 2019).

Position Statement 6: "The majority of peatland restoration projects across the UK are able to achieve relatively rapid development of vegetation communities typical of blanket bog (within c.5–10 years) through hydrological restoration. Re-wetting a peatland tends to be sufficient that any undesirable vegetation, such as dominant heather cover, dies back naturally to be replaced by Sphagnum-dominated conditions associated with healthy peatbog habitat (Cris et al. 2011). Effective restoration of peatlands has been widely achieved across Scotland without the need for burning; for example, there are over 200 Peatland Action restoration sites in Scotland that are delivering good practice restoration and have not required burning as part of this process."

The IUCN UK PP "Burning and Peatlands" position statement (IUCN 2020) assumes that peatland rewetting leads to a net positive impact, which is exemplified by Position Statement 6. We question this assumption. For example, rewetting, usually by ditch blocking, aims to saturate peatland soils by raising the water table so that it is at or very near the soil surface. We are not disputing that this process enhances carbon storage and peat accumulation (Leifeld and Menichetti 2018). However, when rain falls on a saturated peatland, the rainwater will either pond and partially drain on flat areas or, on slopes, flow to lower ground under the force of gravity across the peat surface (Holden and Burt 2003; Acreman and Holden 2013). The latter process is called saturated overland flow, and it can increase the volume and possibly the speed of surface water running downhill into river catchments (e.g. Holden and Burt 2003). By increasing surface roughness, surface vegetation (e.g. *Sphagnum* spp.) may help to reduce the speed of saturated overland flow (Holden et al. 2008). But the extent to which it does is likely to decline as the vegetation itself becomes saturated. Thus, a rewetted and saturated upland peatland *could* potentially exacerbate (rather than mitigate) downstream flooding (Acreman and Holden 2013) and lead to negative impacts on local communities. If so, then peatland rewetting that leads to complete soil saturation may not be the best land management strategy to employ within flood-prone and generally very wet catchments, especially given the projected increases in rainfall intensity across the UK (Kendon et al. 2019; Met Office 2019). We urgently need robust catchment-scale experiments to ascertain the flood mitigation potential of peatlands in different hydrological and vegetative states.

Peatland rewetting may also have a negative impact on climate change mitigation because peatlands with high water tables also emit large amounts of methane, particularly if combined with increasing temperatures (Abdalla et al. 2016), with methane having a much greater warming potential (GWP) than carbon dioxide. Thus, if the GHG benefit of carbon captured and stored by rewetted peatlands is less than the net GHG emission contribution of the methane emitted, then such a peatland will be a net source of GHGs. In fact, this issue has been highlighted by both short-term and long-term assessments of peatland methane emissions (Cooper et al. 2014; Vanselow-Algan et al. 2015), but we desperately need more data about the GWP and net GHG budget impacts of peatland rewetting. Moreover, the assessment of GWPs is highly dependent on the time frame and the methodology used; notably, the commonly used GWP over 100 years fails to represent continuous ecosystem emissions and adequate time frames for methane, with actual GWPs for sustained emission being higher (e.g. Neubauer and Megonigal 2015; Balcombe et al. 2018). We should stress that the potential for rewetting to lead to negative impacts on peatland ecosystem services does not mean traditional peatland drainage (e.g. gripping) is the solution (e.g. Holden et al. 2006). Nor are we against the restoration of functionally degraded peatlands. However, we should not assume that rewetting has only positive effects on peatland ecosystem services – this is rarely the case with any land management intervention. Alongside deciding where, when and why to rewet, we should ideally also consider how the impacts of rewetting may vary according to future climate scenarios, such as warmer summers, and warmer and wetter winters (Kendon et al. 2019; Met Office 2019). To prevent excessive methane emissions, it could be that we have to facilitate lower water tables by a few (but for the net GHG balance potentially crucial) centimeters to maintain a thin aerobic layer favourable to methanotrophs (Roslev and King 1996).

We question two further assumptions outlined in Statement 6. Firstly, the assumption that a peatland dominated by *C. vulgaris* is undesirable and that "*Sphagnum-dominated conditions*" are "*associated with healthy peatbog habitat*". As we have pointed out, there is no robust causal evidence for an association between peatland vegetation type and peat accumulation (i.e. ecological function): it is the hydrological (high water table), environmental (low pH) and climatic (lower temperatures) conditions that determine peat accumulation rates, and not vegetation composition (Gillingham et al. 2016). Therefore, in terms of key peatland functions (e.g. peat accumulation), peatlands should not be classed as 'undesirable' because they fail to pass the arbitrary vegetation composition criteria outlined within the peatland condition assessment (JNCC 2009). Secondly, we question the assumption that rewetting reduces *C. vulgaris* dominance. As far as we are aware, there is no empirical evidence to support this claim. In fact, wet and undrained sites can remain dominated by *C. vulgaris* even after 90+ years post-management (Lee et al. 2013a; Alday et al. 2015). Peatlands dominated by *C. vulgaris* may also support considerable amounts of *Sphagnum* in the below-canopy layer (e.g. Heinemeyer et al. 2019c). Given the lack of evidence, we find it strange that the IUCN document provides a supportive citation for this claim (e.g. Cris et al. 2011). Yet, on closer inspection, the supporting citation (Cris et al. 2011) seems only to contain an unreferenced statement about rewetting reducing *C. vulgaris* dominance.

Position Statement 7: "Burning has been advocated by some land managers as a tool in peatland restoration to remove rank, leggy heather (Calluna vulgaris) (Uplands Management Group 2017). Burning carries a risk of causing more serious damage, further degradation and compromising the onset of peatland recovery. The substantial plant biomass load and the often dry nature of the underlying peat beneath the heather, are susceptible to uncontrolled or "hot burns" that can damage peat forming Sphagnum species, peatland seedbanks, underlying peat soil and lower the water table for a period of several years. The role of "cool burns" as a means of reducing risks has not been assessed in the peer reviewed scientific literature and in view of the large number of successful peatland restoration schemes that do not use any form of burning, the need for a "cool burn" on peatlands is untested. So called "hot" and "cool burns" are an untested management tool with no certainty as to whether differences can be controlled and no robust studies on the relative impacts. Successful restoration of blanket bog on numerous upland sites around the UK, without the use of muirburn or any other form of burning, demonstrates that burning is not a necessary tool for peatland restoration."

It is undeniable that burning removes dense or rank and often leggy *C. vulgaris,* at least in the short-term (Alday et al. 2015; Whitehead and Baines 2018). However, due to the lack of evidence, it is unclear whether it hinders or even promotes *Sphagnum* development in the long-term. Nevertheless, data from the UK's Hard Hill experiment (which are cool burns) indicates that over 60 years, repeatedly burnt and unburnt plots support similar levels of *Sphagnum* (Milligan et al. 2018; Noble et al. 2018a). At the very least, this suggests that prescribed "cool burning" does not inhibit the long-term survival of *Sphagnum* populations (note – this mainly applies to *S. capillifolium*). We concede that this evidence comes from a single and imperfect experiment. Still, it is the only long-term experimental data we have, and longer-term space-for-time chronosequences are not robust enough to ascribe causality or to be generalised (e.g. they have too many assumptions that are rarely met and often fail to or cannot account for confounding variables) (Davies et al. 2016b; França et al. 2016; Ashby and Heinemeyer 2019).

Position Statement 8: "A number of recent studies have presented misleading conclusions resulting in the mistaken interpretation that burning is beneficial for peatland conservation and restoration (e.g. Marrs et al. 2019a, 2019b; Heinemeyer et al. 2018; Milligan et al. 2018)."

Position Statement 8 on misleading recent studies is incorrect. Whether the cited studies present "*misleading conclusions*" is currently unresolved (Heinemeyer et al. 2018; Milligan et al. 2018; Baird et al. 2019; Evans et al. 2019; Heinemeyer et al. 2019b; Marrs et al. 2019a; Marrs et al. 2019b). Failing to mention this key point is a glaring omission. Furthermore, Heinemeyer et al. (2018) did not assess the impact of burning on either peatland conservation or restoration. Also, due to the lack of a supportive citation, we are assuming that the criticism of Heinemeyer et al. (2018) and Marrs et al. (2019a) is primarily based on the model produced by Young et al. (2019). If so, it is important to note that the model used by Young et al. (2019) is unvalidated, unspecified, only relates to the impact of deep drainage, and omits key burning management impacts on peat properties and related C-cycle processes (e.g. the impact of charcoal on bulk density and carbon content as well as on microbial decomposition). Consequently, the model produced by Young et al. (2019) cannot be used to criticise studies looking at C accumulation on rotationally burnt areas of blanket bog with minimal drainage impacts (such as the studies by Heinemeyer et al. 2018 and Marrs et al. 2019a).

In fact, a recent study by Flanagan et al. (2020) supports the findings of Heinemeyer et al. (2018) and Marrs et al. (2019a) that low-severity fires (i.e. prescribed burns) can reduce long-term rates of heterotrophic respiration, potentially increasing net C accumulation rates in peatlands. Specifically, Flanagan et al. (2020) found that the positive impact of low-severity fires on carbon accumulation was mediated by charring and thermal-alteration of the peat aggregate surface, and through charcoal production, with both processes being hypothesised in Heinemeyer et al. (2018) and Heinemeyer et al. (2019c). Moreover, during the last 300 years, the mean annual C accumulation rates for the three grouse moor sites sampled by Heinemeyer et al. (2018) are around 40 gC m^{-2} (1700–1950) to 90 gC m^{-2} (1950–2015). These rates are similar to, and at depth even slightly higher than, those for the three non-grouse moor temperate bogs used in the Young et al. (2019) paper. Crucially, if there were any C losses, this should have been also evident in the peat samples analysed by Heinemeyer et al. (2019a, 2019b, 2019c) as reduced organic carbon content (%Corg), but this was not the case (e.g. %Corg over the first 30 cm depth was around 52%, which showed an overall increase with depth to about 57% at 100 cm).

Position Statement 8a: "Common factors presented in academic literature that can lead to confusion include: a) Inconsistent approaches to the definition of peatland vegetation and its condition; of particular concern are studies that do not consider whether the vegetation recorded is typical of bog habitat or representative of more dry habitats. (It is overly simplistic to report only on the abundance of moss species or generic Sphagnum species, as these can also be associated with poor-fen or dry heath conditions rather than bog formation)."

This is not a valid criticism. Excluding species of conservation concern, peatland vegetation is, in one sense, irrelevant – ecosystem functioning is what should be important. In the case of peatland, this primarily means peat building, which is primarily driven by hydrological (high water table), environmental (low pH) and climatic (lower temperatures) conditions (Gillingham et al. 2016). However, land managers may also want to enhance other ecosystem services, such as flood, climate change and wildfire mitigation, water quality, or biodiversity. Again, the evidence for specific peat-forming species is not clear.

Position Statement 8b: "Common factors presented in academic literature that can lead to confusion include: b) Inadequate methodologies to make a full assessment of baseline conditions or summary of any potential confounding effects. Existing environmental and management factors such as drainage, subsidence, grazing pressure, historic burning regime, surrounding land use pressures such as forestry plantations and atmospheric pollution can all impact on study sites. To fully consider the effects of fire on peatland carbon balance a full net balance needs to be conducted to allow for comparison between burned and unburned sites."

Yes, but, barring a few studies (e.g. Heinemeyer et al. 2019c), this applies to the entire evidence base, especially the four peer-reviewed studies published as part of the EMBER report (Ashby and Heinemeyer 2019). So why do IUCN UK PP only mention Heinemeyer et al. (2018), Milligan et al. (2018) and Marrs et al. (2019a) in this respect? An unbiased assessment would rightly highlight, as we do below, that **no** study to date has examined burning as it is applied in the real world using a study design robust enough to detect causal relationships. One exception is the Peatland-ES-UK study (https://peatland-es-uk.york.ac.uk/), which was specifically set up to address this issue (Heinemeyer et al. 2019c).

Interestingly, this statement refers to the impact of the "*historic burning regime*". Historically, fire or controlled burning seems to have played a role in peatland development within the UK uplands (Simmons 2003). Several studies have found charcoal throughout peat profiles taken from wet heath and blanket bog sites across the UK (Fyfe et al. 2003; Ellis 2008; Fyfe and Woodbridge 2012; McCarroll et al. 2017; Fyfe et al. 2018). This suggests that fire occurred throughout the Holocene (~8000 years before present) and that the burning of vegetation is a long-term feature of wet heath and blanket bog development.

Position Statement 8c: "Common factors presented in academic literature that can lead to confusion include: c) Failure to consider the impact of land management regimes in relation to trajectory for a habitat. Simply comparing burned areas with unburned areas is unhelpful if the aims of the site are to restore functioning peatland habitat. Burning of a heavily degraded heather dominated peatland may simply produce a constrained, degraded peatland state, retaining vegetation associated with drier conditions, such as Calluna that could limit further recovery towards the near natural state."

The first sentence of this specific criticism is unclear, and definitions of trajectories seem, at present, ill-defined. We disagree that comparing burnt to unburnt areas is unhelpful. Such comparisons would be extremely helpful if they were made using a randomised Before-After-Control-Impact

(BACI) design in which important ecological functions (e.g. peat accumulation and water table depth) were measured over several management cycles (e.g. Peatland-ES-UK). Also, note the word "*may*" in the last sentence of this position statement. It is key, because, due to the lack of empirical evidence, we have no idea whether the burning of "*a heavily degraded heather dominated peatland may simply produce a constrained, degraded peatland state, retaining vegetation associated with drier conditions, such as Calluna that could limit further recovery towards the near natural state*". Any burn assessment must also consider site condition as well as elements of fire ecology (burn rotation length, severity and intensity). Clearly, there will be instances where site conditions (e.g. too dry and/or too steep) would have benefited from alternative management (e.g. alternative cutting/mowing or no management) or where burning was done inappropriately (e.g. burns were too severe or too frequent).

Position Statement 8d: "Common factors presented in academic literature that can lead to confusion include: d) Comparing the burned to unburned state can produce data that shows a change in vegetation including an increase in Sphagnum species. However, in burned plots, consideration should be given to the type of Sphagnum species and whether these are typical of bogs, as well as the likelihood of reversion of the degraded peatland back towards abundant heather."

Agreed. But again, this applies to the entire evidence base. *Sphagnum* species are often grouped by researchers because the abundance of most species is low, which inhibits adequate statistical analysis. Also, individual *Sphagnum* species may not be the most sensitive habitat indicators because i) of their wide environmental tolerances (c.f. Plates 1i - 1viii in Daniels and Eddy 1985); and, ii) we lack mechanistic data on their contribution to important peatland functions (e.g. peat and carbon accumulation).

Position Statement 8e: "Common factors presented in academic literature that can lead to confusion include: e) A distinction also needs to be made between studies of a single burn, compared with frequent prescribed burns on a cycle of 30 years or less. The latter can give rise to substantial cumulative impact due to long recovery times of particular blanket bog Sphagnum species from damage through burning (Noble et al. 2019a, 2019b)"

Agreed. Ideally, every study should consider aspects of fire ecology (e.g. burn rotation length, severity and intensity) to provide relevant and useful guidance to land managers and enhance scientific understanding (Davies et al. 2016b; Davies et al. 2010b; Grau-Andrés et al. 2018; Grau-Andrés et al. 2019a).

Position Statement 9a: "studies can also lead to the mistaken view that burning is inconsequential or even beneficial for both the ecology and the carbon store of a bog if they do not fully account for:
- the negative long-term carbon trends associated with atypical plant species abundance"

Again, there is no robust causal evidence for the impact of "*atypical*" plant species abundance on long-term carbon accumulation or storage within UK peatlands.

Position Statement 9b: "studies can also lead to the mistaken view that burning is inconsequential or even beneficial for both the ecology and the carbon store of a bog if they do not fully account for:
- damaged state of the acrotelm (thin living surface layer of peat-forming vegetation)"

A complete assessment of the acrotelm "*state*" would be extremely complex to carry out because it would have to consider its physical (e.g. bulk density), chemical (e.g. organic carbon content) and biological (e.g. microbial communities) properties. Currently, peatland researchers mainly record the physical and chemical properties of the acrotelm. However, as specific physical and chemical properties relate to multiple factors (e.g. management, climate and vegetation) (Morton and Heinemeyer 2019), it can be difficult to determine the current state of the acrotelm. Conversely, soil infiltration measurements can provide useful information on the hydrological state of the acrotelm for a specific moment in time. Furthermore, we would argue that short-term management impacts on the acrotelm, such as exposed peat surfaces after a prescribed burn, should not be used to infer long-term impacts. Finally, the definition of the acrotelm used by the IUCN (2020) (e.g. the "*thin living surface layer of peat-forming vegetation*") differs from the standard definition first described by Ingram (1978):

- "*The surface layer of a mire soil, differing from the subjacent layer in the nature, greater range or more abrupt variation of its physical properties and biological attributes and in function the principal site of matter and energy exchange in the mire ecosystem*".
- "*We consider the lower boundary to be the level above which the water conditions and degree of decomposition vary rapidly, while below this level they either remain constant or vary slightly".*

Position Statement 9c: "studies can also lead to the mistaken view that burning is inconsequential or even beneficial for both the ecology and the carbon store of a bog if they do not fully account for:
- consequent impacts on the catotelm (permanently waterlogged peat store under the acrotelm). Past changes to deep C stores can also give rise to misleading conclusions about the previous rates of C accumulation."

Negative impacts on the catotelm are only really achieved by deep drainage ditches or gullies. Standard moorland drains (i.e. grips) usually lead to small increases in WTD (generally only a few centimetres) that only extend a couple of metres either side of the ditch (e.g. Wilson et al. 2010; Holden et al. 2004; Luscombe et al. 2016). We are assuming the second sentence in this position statement is based on the model published by Young et al. (2019) (again, a supporting citation would confirm this). As previously noted, this is an unvalidated and unspecified drainage-based model without any representation of rotational burning processes (Young et al. 2019). Therefore, the results of this model do not apply to peatlands subject to burning management. A more applicable modelling study in relation to C storage impacts on burning and drainage (with some hydrological model validation) is the one published by Heinemeyer and Swindles (2018).

Position Statement 9d: "studies can also lead to the mistaken view that burning is inconsequential or even beneficial for both the ecology and the carbon store of a bog if they do not fully account for:
- loss of microtopography and overall reduction in environmental resilience."

The first part of this position statement on microtopography impacts is incorrect. For example, several studies show that, relative to unburnt or not recently burnt controls, prescribed burning has no effect on blanket bog microtopography (Heinemeyer et al. 2019c; Noble et al. 2018a). However, alternative cutting might cause damage to the peat/vegetation surface but remains understudied, as shown by Heinemeyer et al. (2019a). The second part of Position Statement 9 (d) is too ambiguous to comment on – the term "*environmental resilience*" needs to be clearly and objectively defined (e.g. resilience to what and what aspect of a peatland needs to be resilient?).

Position Statement 10: "Bogathon and Sphagathon (Moorlands Association & Heather Trust 2015) have demonstrated that there is support for maintaining and restoring peatlands to a healthy condition. It has also demonstrated recognition among land managers that healthy peatlands can support driven grouse shooting and stock grazing."

There is indeed support for peatland restoration toward a "*healthy condition*". However, we need clearly defined and objective restoration goals that are based on ecological function. Once such criteria have been developed, we suggest that scientists and government agencies then work together with land managers and, in the UK context, grouse shooting estates (but also other land owners/managers) to facilitate an evidence-based and site-specific transition to alternative management. We advocate using a series of 'champion estates' distributed across the UK (to capture different site conditions) that implement alternative and traditional management using a moorland-scale BACI experimental/monitoring approach.

Position Statement 10a: "Landowners and grouse moor managers appreciate that raising the water table builds resilience into their land to provide protection from the impacts of climate change and the increasing risk of damage from wildfire – 'wetter is better' (BASC & Moorlands Association 2016)."

As previously noted, we cannot assume that wetter is *always* better. Blanket bogs with water tables at or above the soil surface are likely to emit large amounts of methane (Abdalla et al. 2016; Evans et al. 2017), especially under the warmer and wetter conditions we are expecting due to climate change (Heinemeyer et al. 2019c). The key question is whether such increases in methane emissions will counteract the carbon accumulated within the peat body by rewetting. Saturated peatlands may also contribute to flooding downstream (via increased saturated overland flow) and have a negative impact on important invertebrate taxa, such as Tipulidae (an important food source for rare upland birds) (Holden and Burt 2003; Holden et al. 2008; Carroll et al. 2015; Holden et al. 2017; Heinemeyer et al. 2019c;).

We also cannot assume that rewetting will be enough to mitigate wildfires because the wildfire mitigation potential of rewetting has never been tested within a UK context. There are two aspects to wildfire mitigation: ignition prevention and damage limitation. Firstly, it seems intuitive that wetter bogs would be less likely to ignite (Davies and Legg 2011). However, in summer, bog vegetation becomes very dry, especially during prolonged dry spells. As the vegetation becomes drier, it becomes more flammable. For example, *C. vulgaris* becomes flammable when moisture content drops below 60% (Davies and Legg 2011). Thus, in theory, ignition of *C. vulgaris*-dominated peatlands is possible any time the moisture content of the *C. vulgaris* canopy drops below the 60% threshold.

If a canopy fire did take hold, it seems intuitive that a wetter bog *should* reduce the chances of the underlying moss and peat layers igniting, or limit the spread of a peat fire if the peat body did ignite. Indeed, a group of British studies show that the moss and peat layer within (wet) blanket bog ecosystems are generally buffered from the effects of a prescribed burn (i.e. minimal damage and no peat ignition) (Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Grau-Andrés et al. 2019a; Grau-Andrés et al. 2019b). But these studies were testing the effect of a prescribed management burn rather than a wildfire. Such managed burns are carried out between late autumn and early spring (October 1st to April 15th) when peatland water tables are higher and ground temperatures are cold, which limits any temperature damage (DEFRA 2007; Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Grau-Andrés et al. 2019a; Heinemeyer et al. 2019c). In contrast, wildfires generally occur in the summer months (Albertson et al. 2009). Consequently, they are likely to be significantly hotter than prescribed burns, especially at the soil surface (Davies et al. 2010a; Davies et al. 2010b; Davies et al. 2016b). And, crucially, the heat generated during a wildfire creates a front that may be sufficient to dry out the surrounding peat and facilitate ignition (Prat-Guitart et al. 2016; Huang and Rein 2017).

Another consideration is that, even on near-natural and hydrologically intact peatlands (i.e. peatlands largely undisturbed by human impacts), the water table can lower by as much as 20–30 cm during the summer months (Labadz et al.

2007; Holden et al. 2011), which, combined with dry vegetation, is likely to increase the flammability of the peat significantly. Peat on sloping terrain or hilltops will be particularly prone drying during periods of drought because such areas a more freely draining (Heinemeyer et al. 2010; Carroll et al. 2015). Furthermore, rewetting is likely to be implemented alongside a cessation of vegetation management, which, as the Hard Hill Experiment indicates, leads to a significant increase in above-ground biomass (Alday et al. 2015; Marrs et al. 2019a). Consequently, rewetted bogs may have a higher fuel load over extensive areas (i.e. not broken up into a mosaic of heather ages and such biomass load), which will lead to higher fire temperatures if a wildfire does manage to ignite (Hobbs and Gimingham 1984; Davies et al. 2016a; Noble et al. 2019a). Finally, contrary to the claims made in the literature (Baird et al. 2019), there are indications that higher water tables may not inhibit the horizontal and downward spread of smouldering peatland wildfires (Huang and Rein 2017). Given all the points we raise above, rewetting may not be as effective at mitigating wildfire as its proponents claim. Yet, due to the lack of data, the true role of rewetting in peatland wildfire mitigation remains unknown and, thus, requires urgent research attention.

Position Statement 11: "When examining the evidence on wildfire impacts, it is important to distinguish between studies based on dry heath/grasslands on shallow soils, as opposed to deep peat sites. Concerns over wildfire risk do not generally apply to wet blanket bog habitat where there is naturally minimal dry biomass load and high water tables prevent burning of the peat mass."

Part of this position statement on wildfire impacts is, at best, an unverified assertion. Firstly, productivity and, hence, fire risk will generally be higher on lowland dry heath and grasslands, and so it is important to distinguish them from deep peat sites. Secondly, we lack detailed data on the impact of rewetting on blanket bog vegetation biomass. However, we do have data from a long-term (60 years) experiment situated in an area of undrained, wet and high-altitude blanket bog: the Hard Hill experiment at Moor House National Nature Reserve in the North Pennines, UK (Marrs et al. 1986). In contrast to what the IUCN UK PP document asserts, the unmanaged plots (plots unmanaged since 1923 and 1954) within the Hard Hill Experiment support the greatest amount of biomass (Alday et al. 2015). Thirdly, rewetting may raise water tables during the wetter months (October to April), but water tables will likely still drop significantly during the summer months (Labadz et al. 2007). This, combined with dry conditions, and higher porosity, is likely to significantly increase the flammability of the peat (Huang and Rein 2017). But again, we lack data on this key issue.

Position Statement 12: "However, a large proportion (c. 80%) of our peatlands are considered to be in a degraded condition. Degraded peatlands with abundant heather have been described by some managers as a fire risk when naturally high water tables are absent. The larger fuel load on a damaged peatland can mean that if a fire occurs that it is more damaging; greater fuel load ≈ greater heat intensity ≈ prolonged fire ≈ potential for greater damage to vegetation and ignition of the underlying peat soil. There are numerous scientific studies which demonstrate that wet peatlands are less prone to wildfire (e.g. Turetsky et al. 2015, Swindles et al. 2019; Grau-Andrés et al. 2017;) or that rewetting is a better strategy than burning to achieve peatlands that are resilient to wildfire (Baird et al. 2019). Re-wetting peatlands is therefore viewed as crucial in mitigating wildfire risk."

For a rebuttal of the first sentence, see our comments about Position Statement 4. Moving on, we agree that: "*greater fuel load ≈ greater heat intensity ≈ prolonged fire ≈ potential for greater damage to vegetation and ignition of the underlying peat soil*". Indeed, this concept is well established (Davies et al. 2016a; Davies et al. 2016b; Davies et al. 2010b; Noble et al. 2019a), as is the fact that prescribed burns can be used to reduce fuel loads on UK peatlands (Lee et al. 2013a; Alday et al. 2015; Milligan et al. 2018; Whitehead and Baines 2018; Grau-Andrés et al. 2019a; Heinemeyer et al. 2019c; Marrs et al. 2019a). However, we disagree with the unverified assertion that rewetting peatlands is crucial for mitigating wildfire risk. As we have discussed in previous sections, this has not been tested and, as our comments on Position Statement 10 suggest, it may also be a flawed assumption. Furthermore, we question some of the references used to support the statement "*that wet peatlands are less prone to wildfire*". Swindles et al. (2019) do not test this, Grau-Andrés et al. (2018) examine prescribed burning, not wildfire and the issue of smouldering fires in relation to peat porosity and moisture content must be considered (e.g. Huang and Rein 2017), and Baird et al. (2019) is not an experimental study but a 'Matters Arising' opinion/response to another study.

Position Statement 13: "On UK peatlands, high fuel loads of heather and grasses and dry exposed peat are consequences of lower water tables from drainage, compounded by over-grazing and repeated burning. A healthy peatland with high, stable, water tables and Sphagnum growth, naturally suppresses excess heather and other dry understory ground vegetation. For many sites rewetting (raising the water table) is a rapid process following restoration works and there will be no need for additional vegetation management. However, some severely degraded sites or sites with complex topography (e.g. sites with severe peat hags) may still have significant areas of drier peat and excess heather and other dry vegetation following rewetting activity. For these sites there may be the need to consider measures to control fire risk during the transition period, such as cutting fire breaks in certain areas and restricting burning on adjacent areas."

The conflation of different management aspects is unhelpful for an informed, constructive and precise debate about the use of prescribed burning on peatlands. We agree that deep drainage is a serious issue (e.g. Young et al. 2019), but it should be judged independently from burning because vegetation burning can be implemented in the absence of deep drainage (and vice versa). We also think that prescribed burns should not be carried out on areas of steeply sloping or dry peatland or areas in which a significant amount of the peat surface is exposed (the latter is extremely unlikely to be burned because it will not be dominated by a dense *C. vulgaris* canopy – the pre-requisite for applying a prescribed burn). However, prescribed burns within flat and wet areas of blanket bog are likely to cause only minimal short-term damage to the moss layer (Davies et al. 2010b; Lee et al. 2013a; Kettridge et al. 2015; Taylor 2015; Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Milligan et al. 2018; Noble et al. 2018a; Grau-Andrés et al. 2019a; Grau-Andrés et al. 2019b; Heinemeyer et al. 2019c; Marrs et al. 2019a).

In addition, the following passage seems to be unsupported by empirical data: "*A healthy peatland with high, stable, water tables and Sphagnum growth, naturally suppresses excess heather and other dry understory ground vegetation. For many sites rewetting (raising the water table) is a rapid process following restoration works and there will be no need for additional vegetation management*". If by 'healthy', the IUCN UK PP mean a peatland that is relatively undisturbed by human management, then, as previously noted, such peatlands can also experience a significant drop in the water table during the summer months (Labadz et al. 2007). Also, wet peatland sites can still be dominated by *C. vulgaris* well after any management has ceased (Lee et al. 2013a; Alday et al. 2015; Milligan et al. 2018; Heinemeyer et al. 2019c). It seems a shame that the large investment into peatland restoration has a disproportionally small amount of robust (BACI) and long-term impact monitoring associated with it to test such assumptions or hypotheses.

Position Statement 14: "There are a range of approaches to reducing fire risk in habitats. For peatlands, the approach used must not lead to increased deterioration of the peatland sites as this may exacerbate fire risk. In many peatland restoration projects, managers will seek to rewet and diversify the vegetation composition to naturally reduce biomass. This may involve vegetation cutting in strategic locations, seeking to influence visitor behaviour, responding directly to visitor behaviour at high risk times and participating in local fire response groups. We recognise that there is a need to investigate the most effective mechanisms for wildfire risk mitigation to support the development of management plans for restoration projects during transition periods."

As we highlight in our comments underneath 'Statement 6', rewetting may, in specific contexts, have adverse impacts on certain ecosystem services. There are also many potential issues with alternatives to burning, such as cutting (e.g. sedge dominance, methane emissions, water quality and microtopography impacts; see Heinemeyer et al. 2019c). Furthermore, it might not always be possible to restrict access, and it only takes one ignition incident to set off a devastating wildfire across a *C. vulgaris*-dominated blanket bog. We need to consider these risks, measure them, and try to predict them accurately in relation to different site conditions and **all** available land management options/interventions, including prescribed burning.

Position Statement 15: "Wildfires on peatland are rare outside of situations where people have been involved in the origin of the fire, whether as a result of an out of control prescribed burn, arson or carelessness."

We agree that the greatest wildfire threat to blanket bog comes from people, particularly on blanket bogs near densely populated urban areas (as the number of visits to these bogs would be greater) (Albertson et al. 2009). However, even though wildfires on blanket bog are currently rare, they may increase in frequency due to climate change (Albertson et al. 2010).

Areas for Further Research (a): "*An agreed methodology for defining different peatland states should be developed for use in academic studies along with protocols for describing peatland vegetation which include vegetation type and structure.*"

We broadly agree with this suggestion. However, such a methodology must be based on or at least be partly validated or supported by direct measurements of ecosystem functioning (e.g. net peat and carbon accumulation, net GHG emissions, water storage and quality; biodiversity) – we need to move away from using unverified vegetation metrics as proxies for peatland ecosystem functioning. We would also need to produce a set of thresholds (based on actual ecological data) and an agreed and evidenced set of definitions (concerning habitat states) within an applied context (considering statutory bodies) and a way of defining and assessing a 'trajectory of recovery'.

Areas for Further Research (b): "*Agree how the impact of burning on C storage and C accumulation should be measured.*"

We think that, overall, there is strong agreement amongst peatland researchers about this issue. Still, it would indeed be advantageous if a standardised measurement protocol were developed. We recommend that any such protocol should consider the following:

- The measurement of C storage from fluxes should include all major C flux components (e.g. Net Ecosystem Carbon Balance, NECB), certainly both main gaseous C flux components, carbon dioxide and methane, using eddy covariance towers and/or ground-level chambers.
- Any management assessment needs to consider the entire burn rotation to capture the regrowth of vegetation to maturity. Thus, a robust flux monitoring approach for assessing prescribed burning in UK blanket bog should last between about 15 to 25 years depending on the local climate (Heinemeyer et al. 2019b, 2019c).
- Any peat core assessment must include a detailed physical evaluation of the complete peat profile (i.e. bulk density

changes in relation to peat moisture dynamics; Morton and Heinemeyer 2019) and consider potential C storage impacts due to deep drainage (Young et al. 2019).

- The measurement of charcoal (and partial charring of woody biomass) impacts on carbon accumulation (i.e. recalcitrant, long-term C storage) and carbon fluxes (i.e. its influence on peatland microbial activity and, thus, decomposition and methane emissions).
- Measurements of dissolved and particulate organic carbon exports (and ideally also its long-term fate) using soil and stream water analysis.

To properly understand the impact of prescribed burning on blanket bog carbon storage and accumulation, the above measurements should ideally be collected using the following scientifically robust and real-world approach: a multisite paired site/catchment BACI design where treatments (burnt versus an unburnt control) are randomised within each paired site/catchment and data is collected over at least one complete (but, ideally several) burn rotations.

Areas for Further Research (c): "*Instigate a number of long-term monitoring and survey plots for peatlands under different management conditions to determine the impact of burning on the trajectory towards peatland restoration.*"

Yes, we could not agree more. However, government funding is needed to achieve this and to allow already established experimental work to continue despite any changes to upland land use policy that may occur. For example, the BD5104 (Peatland-ES-UK) project was intended to be one such long-term monitoring study (Heinemeyer et al. 2019c). However, long-term monitoring sites require a long-term commitment to funding. Furthermore, such studies must utilise a randomised and multisite BACI design (see, for example, Heinemeyer et al. 2019c). The use of randomised and multisite BACI designs is crucial because, compared to other experimental designs, they minimise confounding variables (meaning results can be generalised) and are, therefore, significantly more accurate in detecting management impacts (França et al. 2016; Smokorowski and Randall 2017; Ashby and Heinemeyer 2019).

Areas for Further Research (d): "*A systematic review of the response of peatlands following restoration under different management treatments.*"

Agreed. We urgently need a holistic and clinical systematic review of the impact of different management impacts on peatland ecosystems. Such a review must consider the strength of the experimental designs used and the reliability of the data presented by each study. Reviewers should not be afraid to reject studies with unreliable results. Consideration should also be given to selective reporting and titles or conclusions which are not backed up by the study's findings. Given how polarised and political the burning debate has become (Davies et al. 2016b), we urge the review should be conducted by an independent scientific group to prevent any bias. By independent, we mean researchers not invested in the debate about prescribed burning impacts on UK peatlands, which probably means that reviewers from outside the UK should be selected. Alternatively, the review team should include researchers with varying opinions about prescribed burning impacts.

Areas for Further Research (e): "*Further research to support the development of accessible good practice guidance in managing wildfire risk for peatlands which are under restoration and are in transition to a wet and naturally fire resilient state.*"

Agreed. This is a research priority because, due to the lack of data, we have no idea whether rewetting will mitigate wildfire risk. In the UK, there are several live projects investigating this topic, which is to be welcomed, especially given that the UK has very different conditions/challenges compared to most of the available literature from North America, Scandinavia and the Tropics.

Additional Concerns about the Simplified Narrative

Further to our specific comments on the IUCN (2020) "*Burning and Peatlands*" document, we would also like to highlight three additional but crucially important points for researchers and policymakers to consider when evaluating the impact of prescribed burning on UK peatlands. These considerations are intentionally or unintentionally ignored within publications that attempt to simplify the narrative about prescribed burning impacts on UK peatlands.

Spatiotemporal Resolution

To date, no study has fully assessed prescribed burning impacts using a real-world approach, with measurements taken across active grouse moors and extending over a complete management cycle (and covering the entire catchment), but, ideally, several management cycles (the Peatland-ES-UK will have once measurements have been taken over a complete management cycle of about 20 years). In short, this means that the impacts of prescribed burning on UK peatlands have yet to be assessed using the correct spatiotemporal context. For example, in the real world, gamekeepers burn areas of mature *C. vulgaris* to create a mosaic of differently aged *C. vulgaris* patches at the moorland scale (Tharme et al. 2001). Specifically, gamekeepers want young stands of *C. vulgaris* with fresh, nutritious shoots for adult *L. lagopus scotica*, older stands of *C. vulgaris* for cover, and short open stands of *C. vulgaris* containing a greater abundance of insect prey for *L. lagopus scotica* chicks (Miller et al. 1966; Palmer and Bacon 2001; Buchanan et al. 2006). The aim is to burn multiple patches of *C. vulgaris* across a moorland during each burning season. However, due to the vagaries of weather and logistics, the number of burning patches per season is highly variable (Allen et al. 2016). The size and shape of individual burns are also highly variable, but they are usually no more than 30 × 100 m (Allen et al. 2016). Patches are re-burnt as soon as they become dominated by tall and 'leggy' *C. vulgaris*, which takes about 15 to >25 years depending on climate (Glaves et al. 2013; Thacker et al. 2014; Alday et al. 2015). Prescribed burning is also applied within a wide range of environmental contexts because each peatland differs in terms of climate, peat depth, water table depth, slope, vegetation composition, the level of drainage, the amount of grazing, levels of atmospheric pollution and management history (Noble et al. 2018b; Heinemeyer et al. 2019c).

So how are prescribed burning impacts investigated within the scientific literature? Well, the principal UK burning study at Moor House National Nature Reserve measures burning impacts at the plot (rather than moorland) scale, with experimental plots being uniform in size (30 × 30 m or 0.09 ha) and much smaller than the prescribed burns created by gamekeepers (ca. 30 × 100 m or 0.3 ha); experimental plots are also burnt on strict 10 and 20-year rotations, rather than when the *C. vulgaris* becomes dominant, old and leggy (Marrs et al. 1986; Lee et al. 2013a; Lee et al. 2013b; Milligan et al. 2018; Noble et al. 2018a; Marrs et al. 2019a; Noble et al. 2019a). Furthermore, many studies fail to make pre-burn measurements (e.g. Lee et al. 2013a; Lee et al. 2013b; Alday et al. 2015; Noble et al. 2017; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2018b; Marrs et al. 2019a; Noble et al. 2019b), and crucial post-burn measurements (such as carbon fluxes) are usually only taken for ≤3 years at the start of a burning rotation (e.g. Grau-Andrés et al. 2018; Grau-Andrés et al. 2017; Grau-Andrés et al. 2019b; Noble et al. 2019a) or during a single year across multiple burning rotations (e.g. Lee et al. 2013b; Milligan et al. 2018; Noble et al. 2018a; Noble et al. 2019b). Some studies also ignore how environmental conditions (e.g. climate, water table depth, slope, peat depth, vegetation composition) vary within or between study sites, and how such variation influences fire behaviour (e.g. Brown et al. 2013; Ramchunder et al. 2013; Holden et al. 2014; Holden et al. 2015).

Besides being unrepresentative, the short-term approach used to study prescribed burning impacts is biased towards finding adverse effects. This is because prescribed burning is a form of habitat disturbance and all forms of habitat disturbance (natural or anthropogenic) cause *immediate* ecological damage (e.g. via soil disturbance, removal of vegetation and/or physiological damage to plants) irrespective of whether they are beneficial over longer timescales. Thus, if one were to measure the *immediate* impacts of other uncontroversial disturbance-based land management interventions, such as hedge laying, coppicing or mowing, they would, like the short-term investigations of burning, appear to have negative ecological impacts. However, if data collection were extended over the correct timescale for each management intervention, then the ecological impacts would increasingly become positive. We clearly need to measure prescribed burning impacts over complete management rotations (about 15 to >25 years depending on the climate). In the UK, this will require a move away from the short-term research council funding model that is undoubtedly one of the major causes of this issue.

Notwithstanding our comments above, we should also question whether burning rotations of 15 to 25 years are the optimal length for promoting peatland function and ecosystem services. These rotation lengths have a grouse moor management focus. However, it could be that longer rotations (possibly interspersed with alternative cutting) still enhance red grouse populations, but better enhance peatland function and ecosystem services. Future studies should, therefore, investigate burning (and alternative management) impacts over longer timescales.

Evidence Reliability

A second point for policymakers to consider is that the results of many burning studies are currently unreliable or cannot be generalised because they use experimental designs that are unable to detect causal relationships and/or make significant statistical errors (for a discussion of this issue, see Ashby and Heinemeyer 2019). For example, several studies confound burnt and unburnt treatments with site and fail to control for this during data analysis (Ashby and Heinemeyer 2019). The results of such studies are unreliable because any observed impacts (i.e. differences) cannot be solely attributed to burning management. Several burning studies also commit pseudoreplication because they fail to account for data structure during analysis (Ashby and Heinemeyer 2019). By doing this, such studies artificially inflate treatment-level sample sizes, which means the significance values reported are likely to be much too low and the results cannot be generalised (Davies and Gray 2015). Given these issues, we recommend that any future assessment of the prescribed burning evidence should weight conclusions according to the methodological strength (experimental design and data analysis) of each study, with studies being rejected from consideration if they report unreliable results.

The Application of the Precautionary Principle

A third and final point for policymakers to consider relates to how the precautionary principle is applied to different forms of peatland management. It is suggested within the IUCN (2020) "*Burning and Peatlands*" document that:

"*Where there is uncertainty around the benefits of burning for peatland restoration, the precautionary principle should be applied and burning avoided*". If the IUCN UK PP are going to apply the precautionary principle to burning then, for balance, we suggest that they should also apply it to other forms of peatland management that have not undergone a full environmental cost-benefit-analysis, such as cutting, rewetting or the cessation of vegetation management. For example, compared to burning, we know even less about the impact of cutting (an alternative to burning) on peatland ecosystem services (e.g. greenhouse gas emissions; carbon storage; water quality, nutrient cycling). Moreover, the small amount of evidence we do have suggests that, by raising water tables, rewetting *could* lead to increased methane emissions, increased saturated overland flow and reduced water quality (e.g. Holden and Burt 2003; Cooper et al. 2014; Vanselow-Algan et al. 2015; Abdalla et al. 2016; Peacock et al. 2018). Yet, surprisingly, nowhere in the IUCN UK PP "*Burning and Peatlands*" document is it suggested that the precautionary principle should be applied to alternative peatland management interventions, such as cutting or rewetting (IUCN 2020). Nor does the document consider the risks of ceasing vegetation management on UK peatlands (e.g. increased wildfire risk due to increased vegetation biomass) (IUCN 2020). Instead, the document repeatedly advocates the use of rewetting as a way of reducing wildfire risk (IUCN 2020), for which there seems to be no direct evidence absenteeism.

We also assert that the application of the precautionary principle to burning on peatlands might be the wrong approach for two reasons. Firstly, the lack of a transparent and objective decision-making process means the precautionary principle is difficult to apply in practice (Peterson 2007; Vlek 2010). Secondly, and more importantly, there is a growing body of evidence which suggests that in specific contexts (e.g. flat areas with water tables at or near the soil surface), burning causes minimal short-term impacts to UK peatlands (Lee et al. 2013a; Taylor 2015; Grau-Andrés et al. 2017; Grau-Andrés et al. 2018; Milligan et al. 2018; Noble et al. 2018a; Grau-Andrés et al. 2019a; Grau-Andrés et al. 2019b; Heinemeyer et al. 2019c; Marrs et al. 2019a). Furthermore, when negative impacts are reported, they are often for short-term effects or for effects that are so small they may not be ecologically significant (Brown et al. 2015; Noble et al. 2018b; Grau-Andrés et al. 2019b; Noble et al. 2019a; Noble et al. 2019b). Negative impacts are also part of a natural process of fire events, which might be neither positive nor negative in the long-term. It would, therefore, be more appropriate to describe any impacts in a neutral and scientific manner (e.g. impacts are often context, parameter and time-specific).

Instead, given that the impacts of burning are likely to be site-specific (Heinemeyer et al. 2019c), upland land managers should be able to adopt an adaptive management approach to prescribed burning. The fundamental tenet of adaptive management is to monitor management interventions and use the results to inform future actions (e.g. by halting any interventions that are found to be damaging) (Holling 1978). We propose that this 'learn by doing' approach should be endorsed within heather-dominated (including grouse moor) peatland management policy because it (i) allows management to continue as long as landowners monitor the environmental impacts of their interventions (ideally supported by scientific input at representative high-intensity monitoring sites); (ii) encourages landowners to adopt a more cautious approach to management (by realising the benefits and challenges of different options); (iii) potentially ensures more environmentally sensitive management techniques are trailed and tested (before being adopted in general); and, (iv) contributes to evidence base (in a real-world context). We also recommend that any future research on policy-related management (on burning or any alternatives) adopts a joint-up and catchment-scale approach in which management interventions (and potentially also their combinations in relation to site conditions) are compared across several actively prescribed sites covering a broad range of environmental conditions (the Peatland-ES-UK study is such an example: Heinemeyer et al. 2019c).

Conclusions

By critically reviewing the IUCN UK PP "Burning and Peatlands" position statement, we have highlighted how prescribed burning is often wrongly presented as always having negative impacts on peatland ecosystems. We hope that policymakers read and take heed of the points we have made before making decisions about the future of prescribed burning (and possible alternatives) on UK peatlands. We also invite the IUCN UK PP and other peatland researchers with contrasting views to respond to our critical review. Indeed, an open, honest and evidence-based debate is crucial for "*the continuing development of scientific knowledge*" (Egilman 2013). In general, we feel that excluding a management tool categorically, particularly when an ecosystem seems to have evolved with it, is potentially damaging over longer timescales. The risk of uncontrolled wildfires is of particular concern in peatlands. Whilst we do not promote burning, based on the evidence and known risks, we would advocate its continued consideration as part of the management 'toolbox'.

Appendix

Definitions

Blanket bog habitat[1]

Blanket bogs are largely ombrotrophic peatland habitats that form in cool, wet and, in the UK, upland environments. The combination of inhibited surface drainage, high levels of rainfall and low levels of evapotranspiration facilitate peat development in damp hollows and over large areas of undulating ground. Blanket bogs are found throughout the UK, ranging from Devon in southern England to Shetland off the north coast of Scotland.

Peat depth varies but usually lies between 0.5–3 metres, although mean maximum peat depth is estimated to be about 6 metres (Lindsay 2010). One of the characteristics that distinguish blanket bog from other peatland habitats is the vegetation communities they support, which include species such as *Calluna vulgaris*, *Erica tetralix*, *Trichophorum cespitosum*, *Eriophorum* species and several *Sphagnum* species. However, there is no agreed-upon minimum peat depth that can support typical blanket bog vegetation communities.

Blanket bog is also characterised by having water tables at or near the soil surface and the presence of variable surface patterning, ranging from a relatively smooth surface, with the occasional *Sphagnum* hummock and *Eriophorum vaginatum* tussock, to a suite of bog pools and ridges.

Grouse moor management in the UK[2]

Grouse moor management utilises a suite of tools aimed at producing a shootable surplus of red grouse *Lagopus lagopus scotica*. These tools include the legal control of generalist predators (e.g. red foxes, stoats, and carrion crows), disease control (e.g. the application of medicated grit) and vegetation control. As red grouse is an upland species, grouse moors are restricted to the British uplands, mainly in England and Scotland. Consequently, they overlap considerably with blanket bog habitat.

Grouse moor managers control blanket bog vegetation using prescribed burning (also known as rotational burning), but, more recently, they have started to use cutting/mowing as well. Managers burn or mow small areas of mature *C. vulgaris* to create a mosaic of differently aged patches. Usually, multiple patches of *C. vulgaris* are manged across a moorland every year.

Prescribed burning management in the uplands is restricted to the period between October 1st and April 15th. In contrast, cutting is permitted any time outside the bird breeding season, which runs between March 1st and July 31st. The size and shape of individual burns or cuts are highly variable. However, they are usually no greater than 30 x 100 metres. Areas are re-burnt or re-mown as soon as they become dominated by tall and 'leggy' *C. vulgaris*, which takes approximately 15 to >25 years depending on the local climate.

Historically, drainage ditches (~50 cm deep) were added to most blanket bogs currently under grouse moor management, sometimes at a high spatial frequency. The UK government subsidised this activity as a means of increasing post-war upland productivity.

The wider environmental costs and benefits of grouse moor management are fiercely debated. For two contrasting views on the wider impacts of grouse moor management, see Thompson et al. (2016) and Sotherton et al. (2017).

Definitions continued

Photos of burning and mowing management within a UK grouse moor[3]

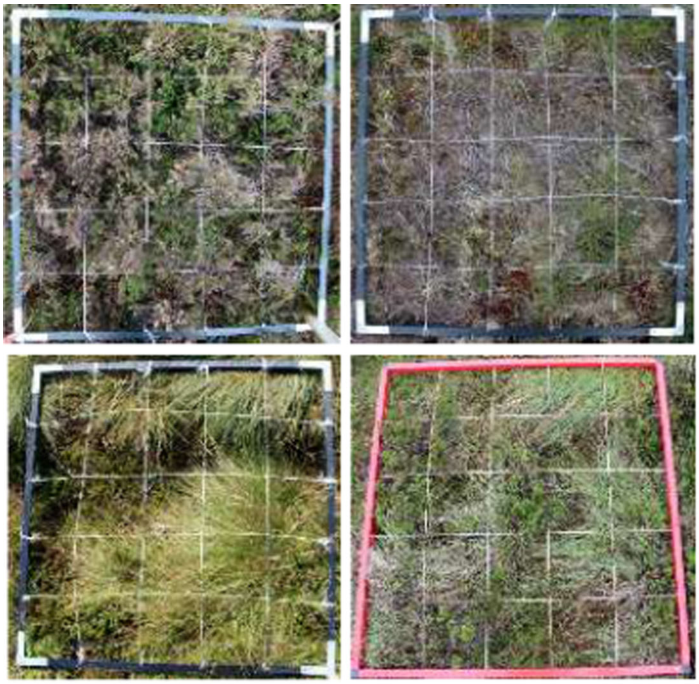
Blanket bog vegetation management using a prescribed burn. Edges of the burn are put out by beating the fire with a flexible shovel.

Blanket bog vegetation management using mowing. Tractors with a set of double wheels to reduce ground pressure are used to prevent peat compaction. The *C. vulgaris* is cut at the rear using blades and returns the brash as a mulch.

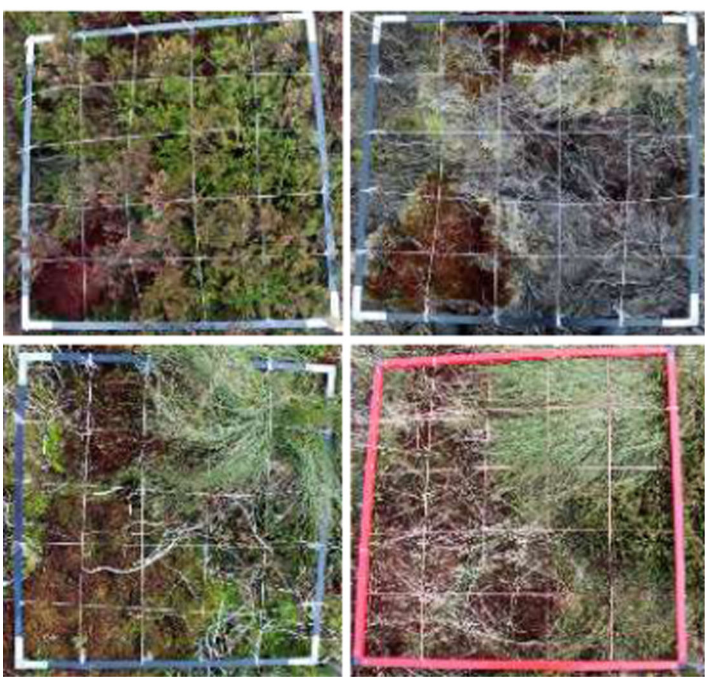
C. vulgaris dominated blanket bog within a UK grouse moor (*C. vulgaris* cover was around 75%) before any burning or mowing management has taken place. Additional plant species included *Eriophorum* spp. (10%), *Sphagnum* spp. (5%), and other mosses (10%) (mainly *Hypnum jutlandicum* and *Campylopus flexuosus*).

Definitions continued

***Photos of burning and mowing management within a UK grouse moor*[3]**



A series of pictures taken during vegetation surveys using 1x1 m quadrats within a mown area of blanket bog (from top left to bottom right: pre-management; < 1-year post-management; 4 years post-management; 8 years post-management). Note the brash layer in the top photo taken shortly after mowing and the quick re-growth of sedges (*E. vaginatum*) at 4 and 8 years post-management.



A series of pictures taken during vegetation surveys using a 1x1 m quadrats within a burnt area of blanket bog (from top left to bottom right: pre-management; < 1-year post-management; 4 years post-management; 8 years post-management). Note the exposed *Sphagnum* layer (mostly *S. capillifolium*) in the top right photo taken shortly after burning showing some fire damage and the quick re-growth of a *Sphagnum* carpet with less sedge (*E. vaginatum*) dominance at 4 and 8 years post-management.

[1] Blanket bog definition taken from Miller et al. (1966), Palmer and Bacon (2001), Tharme et al. (2001), Buchanan et al. (2006), Allen et al. (2016) and Sotherton et al. (2017).

[2] Grouse moor management definition taken from Lindsay (2010) and JNCC (2011).

[3] The photos of grouse moor management were taken between 2012 and 2020 by A. Heinemeyer at the Mossdale site (North Yorkshire, England) used within the Peatland-ES-UK experiment (Heinemeyer et al. 2019c).

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Declarations

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Consent for Publication MA, AH and their respective institutions fully consent to the publication of this manuscript.

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