Peatland Report 2020

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

Game & Wildlife
CONSERVATION TRUST

Contents

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Opposite: Red grouse on heather moorland.

Foreword

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

Rt Hon Sir James Paice

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

Black grouse © David Kjaer.

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

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

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

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

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

Golden plover juvenile on moorland in Yorkshire, England.

Introduction

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

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

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

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

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

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

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

AN EXAMPLE OF COMMUNITY CONSERVATION

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

25 PEATLAND CATEGORIES – NO ONE-SIZE FITS ALL

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

PEATLAND TRADE-OFFS

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

TYPES OF BURNING

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

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

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

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

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

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

Dried grass and burnt heather branches.

Executive summary

1. POLICY CONTEXT.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Continued overleaf >

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

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

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

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

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

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

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

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

6. SECTION 4: Wildfire.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

8. SECTION 6: Biodiversity and grouse moor management.

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

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

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

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

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

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

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

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

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

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

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

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

9. CONCLUSION.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Carbon storage in English peatlands – some definitions and terminology

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

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

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

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

Carbon flux

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

Inputs include:

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

Outputs include:

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

Carbon stocks

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

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

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

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

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

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

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

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

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

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

Carbon fluxes

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

- Total peatland GHG emissions represent around 4% of the UK's total annual GHG emissions.

- Near-natural peatlands (peatlands relatively untouched by human management) are **'close to carbon neutral'**, and only **'very small net GHG sources'** (TABLE 1). Near-natural peatlands have emission factors between -0.61 and 0.01 $tCO_2e\ ha^{-1}\ yr^{-1}$ (remember, negative numbers indicate GHG sequestration, whereas positive numbers indicate GHG release).

- The GHG emissions from modified peatlands (modified by erosion, drainage, cutting, burning or grazing) are higher than those recorded on near-natural peatlands, but they are still relatively low when compared to peatlands converted to cropland or grassland, harvested for fuel, or afforested (TABLE 1) modified peatlands have emission factors between 2.08 and 4.85 $tCO_2e\ ha^{-1}\ yr^{-1}$.

- Despite producing relatively low GHG emissions, the extent of modified peatlands (41% of the UK peatland resource) means that they contribute around 15% of all peatland GHG emissions (which include emissions from peatlands converted to agriculture). As such, emissions from modified peatlands (this category includes the grouse moors) represent less than 1% of the UK's total annual GHG emissions.

- England's peatlands converted to cropland, grassland and forestry are significant sources of GHG emissions and contribute 27%, 11% and 10% of all peatland GHG emissions respectively.

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

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

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

- Short-term losses of above-ground carbon stores due to the combustion of vegetation – the carbon released is usually then re-sequestered (stored again) as the vegetation re-grows in later years.

- Higher atmospheric CO_2 fluxes via plant and soil respiration in years immediately following a burn.

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

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

TABLE 1

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

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

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

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

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

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

Carbon stocks

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

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

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

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

How accurate are these estimates?

Carbon fluxes

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

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

More details regarding these six criticisms are in Appendix 2.

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

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

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

Carbon stocks

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

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

More detail regarding these criticisms are in Appendix 2.

What are the knowledge gaps?

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

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

How much peatland is managed for grouse

and can we estimate total carbon stored and carbon emissions?

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

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

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

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

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

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

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

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

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

Close-up of Sphagnum moss. © Laurie Campbell

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

FIGURE 1

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

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

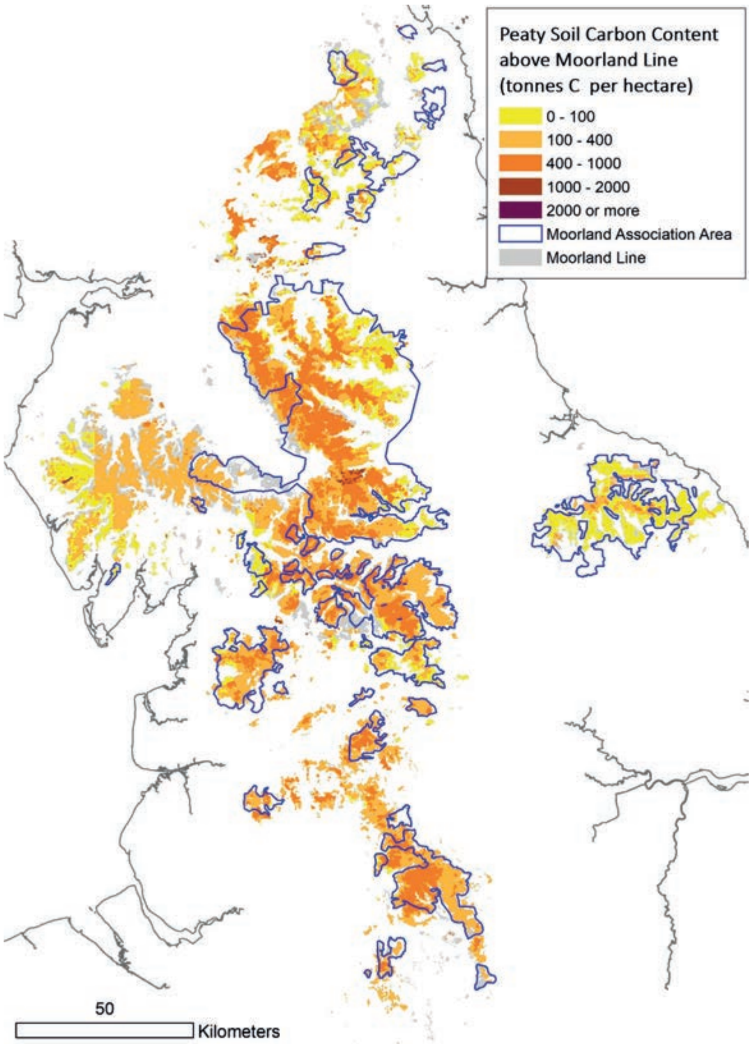

TABLE 1A

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

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

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

TABLE 1B

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

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

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

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

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

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

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

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

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

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

Butterfly on heather under the evening sun.

FIGURE 2

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

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

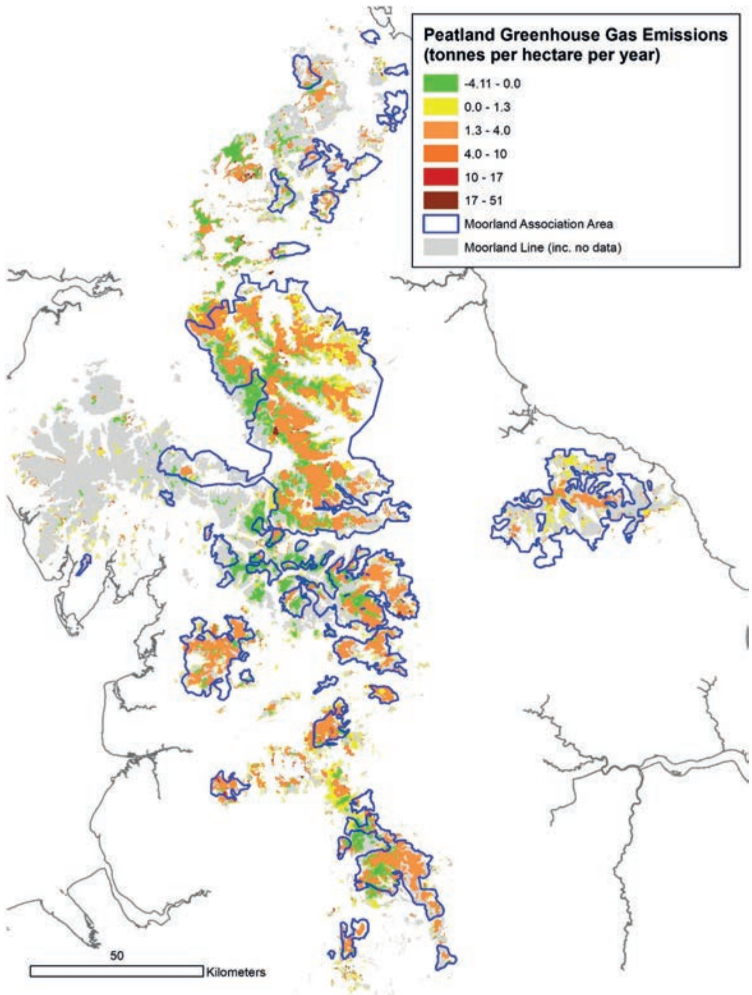

TABLE 2A

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

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

TABLE 2B

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

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

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

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

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

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

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

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

Freshly cut peat stacked to dry.

Summary

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

TABLE 3

Summarising the results of the three methods.

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

TABLE 4

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

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

Little owl, north Yorkshire, England.

Wildfire

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

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

UK wildfire

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

Fire at Dovestones, Saddleworth. © Craig Hannah

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

Wildfire on upland blanket bogs

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

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

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

Ignitions

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

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

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

Fuel management and impact on wildfires

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

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

Environmental impact

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

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

Restoration

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

Wildfire damage, having burned down into the peat layer.

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

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

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

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

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

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

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

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

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

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

Male Northern bobwhite quail, southern Texas, USA.

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

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

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

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

Controlled burning of forest understorey, Florida, USA.

Biodiversity and grouse moor management

Heather burning and birds

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

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

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

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

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

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

Adult curlew on moorland in Yorkshire during the breeding season.

Common cottongrass or bog cotton.

Higher and lower plants

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

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

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

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

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

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

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

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

Close-up detail of colourful Sphagnum moss in autumn.

Invertebrates

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

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

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

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

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

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

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

Red grouse in heather, Yorkshire Dales, UK.

Conclusion

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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Appendices

Appendix 1

More detailed criticisms of studies used to estimate carbon stocks.

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

Appendix 2

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

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

Mountain hare in winter.

Freshly cut peat stacked to dry.

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

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

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

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

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

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

Appendix 3

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

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

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

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

Appendix 4

What are the knowledge gaps?

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

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

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

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

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

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

Appendix 5

Carbon storage/GHG peatland area digitising

Method

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

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

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

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

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

Known issues

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

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

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

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

Red grouse on moorland in UK.

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