Vegetation burning for game management in the UK uplands is increasing and overlaps spatially with soil carbon and protected areas

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Burning for habitat management is globally widespread. Burning over carbon-rich soils is a global environmental concern due to the potential contribution to climate change. In the UK, upland heath and blanket bog, so-called ‘moorland’, often overlies carbon-rich soils, and has internationally important conservation value, but is burned as management for gamebird shooting and to a lesser extent for livestock grazing. There is little detailed information on the spatial extent or temporal trends in burning across the UK. This hinders formulation of policies for sustainable management, given that the practice is potentially detrimental for soil carbon storage, water quality and habitat condition. Using remotely sensed data, we mapped burning for gamebird management across 450 000 km² of the UK. Burning occurred across 8551 1-km squares, a third of the burned squares in Scotland and England were on peat ≥0.5 m in depth, and the proportion of moorland burned within squares peaked at peat depths of 1–2 m. Burning was detected within 55% of Special Areas of Conservation and 63% of Special Protection Areas that were assessed, and the proportion of moorland burned was significantly higher inside sites than on comparable squares outside protected areas. The annual numbers of burns increased from 2001 to 2011 irrespective of peat depth. The spatial overlap of burning with peat and protected areas and the increasing number of burns require urgent attention, for the development of policies for sustainable management and reversal of damage to ecosystem services in the UK uplands.

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1. Introduction

Prescribed burning of natural and semi-natural vegetation is an established management across the world. It can reduce wildfire risk (Boer et al., 2009), manage forest succession (Sah et al., 2006), enhance grazing pasture (Augustine and Milchunas, 2009), manage game (Kilburg et al., 2015) and deliver nature conservation by mimicking natural fire events to maintain desired habitat conditions (Williams et al., 2012). However, across a range of systems, there is increasing evidence of negative environmental impacts of burning, including soil erosion (Cawson et al., 2012), alteration of soil processes including nutrient cycling and soil hydrology (Neary et al., 1999), water quality (Battle and Gollday, 2003) air pollution (Tian et al., 2008) and habitat condition and biodiversity impacts (Suárez and Medina, 2001). These impacts must in some cases be evaluated against purported benefits of burning, for example wildfire risk reduction, with the benefits and disbenefits debated (Altangerel and Kulla, 2013).

Burning in the UK uplands for game management is a case where benefits and disbenefits need to be better understood (Grant et al., 2012; Sotherton et al., 2009). This burning is widely practised above the altitudinal limits of enclosed agriculture on upland heath and blanket bog dominated by ling heather Calluna vulgaris, for the endemic red grouse Lagopus lagopus scotica. On these ‘moorlands’, burning is undertaken in small strips recommended up to 2 ha and in rotations of 10–25 years (Defra, 2007), to maintain a fine–scale mosaic of different ages which provide nesting cover and food resources for red grouse. Together with intensive predator control and use of medication to reduce grouse parasites and disease, this management supports a globally unique form of recreational shooting which depends on achieving high post-breeding densities of red grouse (≥ 200 birds km−2, Moorland Association, 2006). The area of heather moorland managed principally for grouse shooting has previously been imprecisely estimated at
6600–17 000 km² (Bunce and Barr, 1988; Grant et al., 2012; Hudson, 1992), representing 5–15% of the UK uplands, and about 20–60% of heather dominated moorland, although not all of this may be subject to rotational burning (Ball et al., 1983; Brown and Bainbridge, 1995).

Grouse management is therefore a major land use in the UK, but concerns have been raised about the environmental impacts of burning practiced for grouse shooting (Bain et al., 2011; Glaves et al., 2013). Many moorlands are underlain by blanket peat soils, of which the UK holds 10–15% of the global resource (Dunn and Freeman, 2011; Lindsay et al., 1988; Milne and Brown, 1997), storing 3.2 billion tonnes of carbon (Bain et al., 2011) — the largest terrestrial carbon reserve in the UK (Grayson et al., 2012). There is particular concern about the effects of burning over deep peat (usually defined as ≥0.5 m depth; Bain et al., 2011) on peat hydrology, chemistry and physical properties. Where blanket bog is damaged by burning, impacts include a lowered water table, breakdown of the active peat-forming structure and resulting long-term loss of the carbon store (Lindsay, 2010; Brown et al., 2014).

Moorlands are also of high international conservation value for their vegetation, invertebrate and bird communities (Pearce-Higgins et al., 2009; Thompson et al., 1995) with large areas given legal protections for nature conservation under the European Commission Habitats Directive (Special Areas of Conservation (SACs), European Commission, 2015a) and Birds Directive (Special Protection Areas (SPAs), European Commission, 2015b). Burning alters vegetation composition and structure and, where fire temperatures are high and rotation lengths are short, it may be damaging to fire-sensitive, peat-forming species such as Sphagnum spp. mosses (Brown et al., 2014). These problems are cited as a primary reason for poor condition of upland sites designated for their conservation value, contributing to the reasons for “unfavourable” condition in 53% of the total area in England (Natural England, 2008) and on 87% of “unfavourable” upland bog features in Scotland (SNH, 2010).

Finally, upland areas provide >70% of the UK’s water supply (Watts et al., 2001), however water colouration and the occurrence of particulate and dissolved organic carbon (POC and DOC) from peat soils in upland areas are major factors affecting water quality, incurring high financial costs for treatment (Grayson et al., 2012). Growing evidence links burning over peat to a range of impacts on water quality including discolouration (White et al., 2007), lower pH and higher DOC content (Brown et al., 2014; Clay et al., 2012). Furthermore, riverine invertebrate diversity may be lower within burned catchments (Brown et al., 2013).

Before policies for sustainable resource management can be developed, there is a need for a detailed knowledge of the extent of burning in the UK, and how this overlaps with areas of high concern (e.g. deep peat and areas designated under EU conservation directives). Despite the widespread use of burning for gamebird management in the UK, and increasing evidence of its wider environmental impacts, objective reporting of the spatial extent of moorland burning, and variation in spatial and temporal patterns, is currently lacking except at coarse (10-km square, Anderson et al., 2009) resolutions or in localised studies (Amar et al., 2012; Hester and Sydes, 1992; Yallop et al., 2006). However, the characteristic pattern of strip burning is readily visible on high resolution aerial and satellite images (Fig. A1), allowing fine-scale repeatable mapping over large areas, whilst the heat generated by active fires is detectable by fire monitoring satellites. We use these data to map spatial and temporal patterns in strip burning across three countries (Scotland, England and Wales) of the UK at a 1-km resolution. We use high resolution aerial and satellite imagery (henceforth images or imagery) to assess the extent to which burning takes place over deep peat soils and in protected areas designated for their conservation value. From satellite-derived thermal anomaly data we measure trends in the annual number of burns from 2001 to 2011 nationally, and in relation to peat depth.

For the conservation reasons above, we focus on strip burning for red grouse in the UK, however we demonstrate the utility of combining remotely sensed burning data with wide scale maps of environmental correlates, which is widely applicable where land use is associated with environmental impacts (Willis, 2015).

2. Methods

2.1. Mapping burning extent

Areas assessed were based on Anderson et al. (2009), who visually estimated the extent of strip burning for red grouse management within a single image of each of 2707 10-km British National Grid squares (Ordnance Survey, 2014) covering the mainland UK (Scotland, England and Wales). We selected all 10-km squares with burning present and 60 additional 10-km squares that Anderson et al. (2009) did not assess due to cloud-obscured imagery. From the resulting 488 10-km squares, we used a single image of each constituent 1-km British National Grid from Getmapping (2011) (n = 46 239 squares, aerial photography) or Google Earth (2011) (n = 863 images, satellite images); the latter was obtained for a separate study (Newey et al., 2012). We excluded squares with less than 5% land or inadequate imagery (cloud cover or blurred), and assessed a total of 47102 images. For all Getmapping images, and most Google Earth images, the image year was known. For some Google Earth images we had not recorded the exact year but could assign images to one of two years; 2001, <0.1%, 2002, 0.1%, 2003, <0.1%, 2004, 7.3%; 2005, 27.5%; 2005 or 2006, 0.1%, 2005 or 2008, 0.1%, 2006, 11.0%; 2007, 47.0%; 2007 or 2008, <0.1; 2008, 6.7%; 2009, 0.2%; and 2010, <0.1%.

Getmapping images were viewed in ArcGIS 9.3 (Esri, 2009). Shapefiles of 1-km squares to be viewed in Google Earth were first opened in MapWindowGIS (2011), converted to KML files using the plug-in “Shape2Earth” before being opened in Google Earth. The percentage of each square comprising moorland and the percentage of the moorland with strip burning was visually assessed (Fig. A1). Between-observer variation in image scoring was minimal (Appendix A). Both measures were estimated to the nearest 5%, including an additional category of “1%” for the area of moorland burned, to identify squares with very limited burning where rounding to zero would have incorrectly classified the square as “no burning”, but rounding to 5% would have over-represented burning extent. The percentage of moorland multiplied by the percentage of moorland burned gave an estimate of the percentage of the square burned. Using the colour tone of vegetation types on images, moorland was defined as an unenclosed habitat dominated by ericaceous dwarf shrubs (typically ling heather but also bell heather Erica cinerea, cross-leaved heath Erica tetralix and crowberry Empetrum nigrum). Our assessment of moorland area will also include some areas of grassland and bracken Pteridium aquilinum cover where these occur in mixes with dwarf shrub heath, but only where dwarf shrub heath dominates. Burning of grass-dominated moorland also occurs for grazing management, but generally comprises larger burn patches, is less tightly rotational and is a much less significant feature of upland landscapes nationally than strip burning for red grouse (Yallop et al., 2006). Therefore, whilst some of this burning may be included within the areas assessed, in general we ascribe burning within our study to that undertaken by gamekeepers for grouse management. Mechanised cutting of moorland may occur as an alternative to burning, and may also be detectable on imagery. Whilst we were unable to quantify the relative extent of cutting and burning, cutting can only take place where machinery can be deployed (not on wet, steep or rocky ground (Defra, 2007)), and we expect any cutting to comprise only a very limited area relative to that burned.

Post-burning re-growth of ling heather typically occurs in four stages over subsequent years, which are distinguishable on aerial images (Yallop et al., 2006) from newly burned (stage 1) to degenerate with no remaining visible burn scar (stage 4). Observers scoring images independently confirmed that stages 1–3 were distinguishable from surrounding unburned vegetation (consistent with Yallop et al., 2006).
and together constituted the estimate of burning extent in a 1-km square. We therefore assume that all burning detectable in these images was conducted within a maximum of ≤25 years of the image date (Yallop et al., 2006), although post-burning regeneration rates vary according to environmental factors including soil wetness (Yallop et al., 2009).

2.2. Trends in burning frequency

Temporal burning patterns were extracted from MODIS Thermal Anomalies data (MODIS, 2013). Sensors on the Aqua and Terra satellites detect ground surface temperatures twice daily, identifying events when the fire strength is sufficient to be detected relative to its background (to account for variability of the surface temperature and reflection by sunlight). Daily MODIS data for Scotland, England and Wales were downloaded in November 2013 for 2001–2011. After re-projection to British National Grid (Ordnance Survey, 2014), thermal anomalies with probability of ≥8 (Giglio, 2013) were defined as fires, probably excluding some fires, but reducing false detection rates of non-fire thermal anomalies. We included burns within the fully permissible annual period for burning (1 October–15 April in England and Scotland (Defra, 2007; Scottish Government, 2011)) and 1 October–31 March in Wales (Welsh Assembly Government, 2008) and these data were summarised as the annual number of burns per 1-km square. Thermal anomalies will only be detected if they are sufficiently hot when the satellite passes overhead. Similarly, fires will not be detected on cloudy days as the ground will be obscured from the sensor. Consequently, an unknown proportion of fires remain undetected, and our data are considered a minimum value.

2.3. Environmental data

Mapped peat depth data, produced using a combination of measured and extrapolated peat depths, were available separately for the whole of England and Scotland, but not Wales. We derived a value of mean peat depth per 1-km square. The dataset for England (Natural England, 2012) comprised 25 m raster data with a peat depth value per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations per pixel, derived from two data layers describing peat types above and below 150 m elevation, explaining 28.4% and 37.8% of variations.

We classified squares into three categories of average peat depth; no peat = 0 m (squar es overlying non-peat soils), shallow peat of < 0.5 m or deep peat of ≥ 0.5 m (Bain et al., 2011) (Fig. A2). Mean altitude and gradient per 1-km grid square were calculated using a digital terrain model with a 50-m grid of points.

2.4. Burning in protected areas

Special Areas of Conservation (SACs) comprise high-quality conservation sites aiming to provide increased protection to a variety of wild animals, plants and habitats (JNCC, 2014a), whilst Special Protection Areas (SPAs) are considered to be of international importance for a range of rare and vulnerable bird species and regularly occurring migratory species (JNCC, 2014b). Using GIS we identified all 1-km squares assessed for burning lying wholly within one of these protected area types; squares overlapping site boundaries were excluded as it was not possible to easily define how much of the moorland or burning lay within the site boundary. Thus, of the sites within the area assessed for burning, we achieved coverage of 40.1% (range 10.0–75.3% per site) of the area of these SACs and 43.3% (range 10.1–65.4% per site) of the area of these SPAs (Tables A2 and A3). The areas burned per site should therefore be considered minima. Where SACs and SPAs overlapped these were treated as separate sites for clarity.

2.5. Analysis

All analyses were conducted in R 3.0.2 (R Core Team, 2013) unless otherwise stated.

2.5.1. Spatial correlation between burning and environmental correlates

Using the burning dataset from imagery, we examined the relationship between the percentage of moorland burned per 1-km square and peat depth, across England and Scotland (with peat data for all squares), removing squares with no moorland (yielding 32905 squares). The response variable was the percentage burning measure, rescaled to a proportion and logit-transformed (thereby normalising it) after adding 0.01 to all values (Warton and Hui, 2011). The key continuous explanatory variable was peat depth, with additional covariates of altitude (m), gradient (degrees), image year (averaged where an image was from one of two possible years), easting and northing, all fitted as linear and quadratic terms to test for potential non-linear relationships.

The proportion of moorland burned per square might show spatial autocorrelation, especially within individual estates subject to similar management practices. Estate boundaries were unavailable so we examined whether such autocorrelation occurs, and over what scale, by first fitting a non-spatial GLM with response and explanatory variables as described above, normal errors and identity link. We examined spatial autocorrelation in the residuals using Moran’s I in SAS 9.4 (SAS, 2012), at increasing increments of 1-km lags between squares, to identify at what scale Moran’s declined to 0.1, when spatial autocorrelation effects in model residuals are unlikely to compromise inference (Ryan et al., 2004; Kraan et al., 2009). This occurred at a 20-km lag and we created a factor grouping squares within 20 × 20-km blocks along OS grid lines. We fitted this as a random factor, thereby assuming spatially non-independent errors at a 20-km scale, in a Generalised Linear Mixed Model (GLMM) with normal errors, identity link, and response variable and explanatory variables as above.

2.5.2. Spatial overlap between burning and protected areas

Globally, protected areas are not randomly distributed with respect to key environmental characteristics that might relate to their effectiveness (Joppa and Pfaff, 2009). Therefore, before we tested whether the percentage of moorland burned per 1-km square differed inside and outside protected sites, we controlled for potentially confounding effects. Covariates potentially associated with the percentage area burned (peat depth, area of moorland, altitude, gradient, image year, easting and northing) differed markedly inside and outside protected areas (Table A1). We therefore selected squares inside and outside protected areas in England and Scotland, with moorland present and with similar distributions in the covariates, following an established matching method (Andam et al., 2008). We used the ‘matchit’ procedure in the R package Matchit, selecting squares for both SPAs and SACs, using nearest-neighbour covariate matching and a caliper of 0.5 standard deviations of each covariate (Andam et al., 2008). We compared the percentage of moorland burned per square within and outside SPAs and SACs, respectively, using two-sample unpaired t-tests applied to the matched samples of squares, where the response variable was logit-transformed to normalise (Warton and Hui, 2011).

2.5.3. National trends in burning frequency and its environmental correlates

From the MODIS data, we extracted all 1-km squares assessed for burning that contained moorland. We summed the total annual number of burns across all squares from 2001 to 2011. These data showed extremely high counts in 2003 and 2007; a generalised Extreme Studentized Deviate test, for detecting one or more outliers (Rosner, 1983) revealed that these years were statistically significant outliers. MODIS cannot distinguish between wildfires or prescribed burns, and
with 2003 in particular having a high incidence of UK wildfires, including some spring fires within the permissible burning period (McMorrow, 2011), 2003 and 2007 were removed. Whilst some wildfires may be included in the remaining years, by restricting our study to known areas of red grouse management at the 10-km square scale, this risk is minimised. We tested for a trend in the annual number of burns, fitting a Generalised Linear Model (GLM) with quasi-Poisson errors (accounting for overdispersion where residual deviance/residual d.f. > 2 (Lindsey, 1999)) and log link, with total annual burns as the response variable and year as a continuous explanatory variable. We conducted two further tests by modifying this model, first testing whether trends differed between England, Scotland and Wales, using quasi-Poisson errors, fitting country as a three-level factor and testing a year * country interaction. Secondly, we tested whether trends in annual burns differed between squares in the three peat depth categories, in a model using quasi-Poisson errors, using all squares in England and Scotland, fitting peat depth as a three-level factor and testing a year * peat depth interaction. Trends inside and outside of protected areas were not analysed due to very small numbers of burns recorded by MODIS in the matched samples of squares (<7 burn events annually, inside or outside, across all squares per site type).

As the number of burns detected by MODIS could be influenced by cloud cover, we tested for a trend in the annual number of sunshine hours, using publicly available data from four weather stations situated ≤10 km from areas assessed for burning, with complete sunshine data from 2001 to 2011 (Met Office, 2014). We included data from months overlapping with the permissible burning period (October to April inclusive), summing total annual sunshine hours across stations and fitting this as the response variable in a GLM with quasi-Poisson errors, log link and year as a continuous explanatory variable.

3. Results

3.1. Mapping burning extent

Strip burning was detected in 8551 1-km squares, with 5245 in Scotland, 3139 in England and 167 in Wales (Fig. 1a). Within squares, the percentage area of moorland burned varied from 1 to 95% (mean across squares with burning 16.7 ± 0.2%; England 19.0 ± 0.3; Scotland 15.7 ± 0.2%; Wales 5.9 ± 0.6%).

3.2. Burning and peat depth

A third (33.5%) of burned 1-km squares (43.6% in England and 27.6% in Scotland) were classified as overlying deep peat (Fig. 1b). Of the total area burned, approximately 466 km² (39%) occurred in squares classified as deep peat, c278 km² (60% of burned area) in England and c188 km² (40% of burned area) in Scotland.

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3.3. Burning in protected areas

Burning was significantly greater inside protected areas than in matched areas that were not protected (Fig. 3), and burning was widespread across protected areas (Fig. 4). Burning was detected in 26 of 47 (55%) SACs assessed (Table A2), with the percentage of total site area burned ranging from <0.1 to 11.5% per site. The North Pennine Moors

Fig. 1. (a) Extent of moorland burning within Scotland, England and Wales. Map shows the percentage area of moorland burned within 1-km grid squares, assessed from a single image per square obtained from 2001 to 2010; (b) extent of moorland burning within 1-km squares classified as deep peat (averaging ≥0.5 m) in England and Scotland. Shown are only squares overlying deep peat and with burning present.
SAC in the north of England was notable in having high relative and absolute levels of burning, with the 118.2 km² burned representing 51.7% of all assessed SAC burning (despite accounting for only 16.5% of assessed SAC area).

Burning was detected in 20 of 32 (63%) SPAs assessed (Table A3), with <0.1–10.8% of site areas burned. The North Pennine Moors SPA (overlapping partly with the North Pennine Moors SAC) had the largest area burned, both as a percentage of site area and also total area burned, with 158.4 km² burned representing 59.5% of the total area of burning detected within all SPAs assessed, but only 23.6% by total area.

### 3.4. National trends in burning frequency

The annual number of burns from 2001 to 2011 increased significantly overall across Scotland, England and Wales at a rate of 11% per year (0.106 ± 0.029, $\chi^2_1 = 64.77, P = 0.011$), with an accelerating trend in more recent years (Fig. 5). This was unlikely to be related to declining cloud cover and improved burn detection, as there was no significant trend in sunshine hours over this period ($-0.012 \pm 0.011$, $\chi^2_1 = 31.91, P = 0.285$). There was no significant difference in annual burn trends between the three countries ($\chi^2_2 = 0.77, P = 0.950$). In England and Scotland, where we had country-wide peat depth data, there was a significant overall increase in annual burn trends ($0.105 \pm 0.029$, $\chi^2_1 = 62.64, P < 0.001$) but the rate of increase did not differ between squares overlying zero peat, shallow peat or deep peat ($\chi^2_1 = 0.75, P = 0.926$).

### 4. Discussion

This study provides the first fine-scale assessment of the extent, trends and environmental correlates of burning for recreational shooting of red grouse in the UK uplands. Burning occurred across 8551 1-km squares and based on typical vegetation regeneration rates we assume that this burning took place within the last c25 years (Yallop et al., 2006). This estimate of the extent of moorland burning is within the range of previous estimates of the area under management for red grouse of 6600–17 000 km² (Bunce and Barr, 1988; Grant et al., 2012; Hudson, 1992), albeit towards the lower end; this may be because other activities such as predator control may extend beyond the area burned and may be included in estimates of the total area under grouse management.

The annual number of burning events has increased significantly from 2001 to 2011 at a rate of c11% per annum. These national trends in burning are consistent with smaller scale studies such as in northern England where Yallop et al. (2006) reported markedly increasing burn events, with the percentage of the area newly burned (~4 years) doubling from 15 to 30% between the 1970s and 2000, and declining repeat burning times to squares from 20 to 16 years. Our results suggest that increases in the annual number of burns are widespread across the uplands of the mainland UK. These increases may be due to general intensification of management for red grouse; for example, the numbers of gamekeepers in employment, and the potential number of ‘shooting days’ that could be sold, increased in northern England from 2001 to 2009 (Natural England, 2009).

Set against annual increases in burning, aerial and satellite imagery used to estimate burning may rapidly require updating, or the true extent of burning may be underestimated. In fact, our study is likely to be a conservative estimate since >90% of data used to map burning are derived from images pre-dating 2008, and therefore before much of the recent large increase in the number of burns was detected.

This increase occurred irrespective of peat depth, so that burning is now widespread over deep (≥0.5 m) peat in England and Scotland, especially in England, where 44% of all squares with burning overlies deep peat, and where 60% of the measured area of burning across these two countries over deep peat occurred. This result is set in the context of increasing global concern surrounding burning on carbon-rich soils, and its impacts on carbon storage and emissions, where burning occurs across temperate-boreal and tropical peatlands (Goldammer, 2013; Hoscilo et al., 2011; Tansey et al., 2008).

A growing evidence base highlights negative impacts of burning on deep peat for a range of ecosystem services including drinking water quality (Glaves et al., 2013; White et al., 2007), for example through elevated DOC contents (Clay et al., 2012; Holden et al., 2012). There is also evidence of both negative (Garnett et al., 2000; Worrall et al., 2010) and positive (Clay et al., 2010) impacts of burning on carbon storage and further research is needed on the impacts on overall carbon...
budgets. Much of this evidence has been obtained from studies at a limited number of sites, but our study provides a broader picture of where burning is practised over deep peat, notably northern England and the central Highlands and far north of Scotland (Fig. 1b). The impacts of burning on carbon budgets may not have previously been accounted for in national emission statistics (UK Government, 2015).

The significantly greater percentage area of burning inside protected areas than matched areas outside these sites is set in the context of increasing concern surrounding the condition of upland protected areas. Special Areas of Conservation and Special Protection Areas are designated for the conservation value of their habitats and birds, however in the uplands many of these sites are failing to reach standards of condition required as part of their designation (Adaptation Sub-Committee, 2013). Inappropriate burning is a major contributor to the poor (“unfavourable”) condition of blanket bog in upland protected areas in England (Natural England, 2008) and Scotland (SNH, 2010) and our results provide strong evidence that a greater percentage of moorland is burned within these sites than outside sites.

The UK government has agreed to effectively protect more than 17% of its land by 2020 (Convention on Biological Diversity, 2010). The impact of burning within SPAs and SACs, together with the evidence of an increase in burning, suggests that current moorland management is not making a sufficient contribution to this target; indeed it may be detracting from it. Our results highlight the high overlap of this intensive burning management with environmental concerns such as soil carbon and protected areas. The semi-natural moorlands of the UK uplands are of high international biodiversity conservation value (Pearce-Higgins et al., 2009; Thompson et al., 1995) and store important carbon resources (Bain et al., 2011). Whilst burning may in some circumstances enhance plant species diversity (Harris et al., 2011) or support carbon storage (Clay et al., 2010), a wide range of negative environmental impacts of burning are now increasingly well documented (Brown et al., 2014). Policies to reverse these damaging effects, for example over deep peat, must be implemented as a matter of urgency. These should include management agreements between land managers and statutory agencies to halt damaging burning practices, and tighter enforcement of these agreements.

Detailed remote mapping of the extent of widespread land uses such as burning has other uses. It could be used to improve our understanding of the impacts of burning on the distribution of species associated with these upland areas; game management, including burning, is...
known to be associated with the distribution and abundance of upland birds (e.g. Tharme et al., 2001; Douglas et al., 2014). National mapping will allow for a greater study of these effects, updating previous coarse resolution assessments of not just burning (Anderson et al., 2009) but also the interaction between grouse management and bird distributions (Gibbons et al., 1995).

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Appendix A. Supplementary data

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References


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