# The value of habitats of conservation importance to climate change mitigation in the UK

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***Abstract***

The twin pressures of climate change and biodiversity loss mean that it is imperative to manage land in ways that benefit carbon storage and biodiversity conservation. We focus on a set of UK habitats of recognised conservation value, first quantifying the carbon stored in the vegetation and top 30 cm of soil in these areas. We estimate that these areas store 0.55 gigatonnes of carbon in vegetation and soil to a depth of 30cm, approximately 30% of the UK terrestrial carbon store to a similar depth, on 20% of the land area. Most of these high carbon, high conservation value habitats are in upland areas, with particularly notable extents and mass of carbon in Scotland. In their current condition, we estimate these areas to exert a net sequestration effect of more than 8 million tonnes of $CO_2$ equivalents per year. Furthermore, restoration of these habitats from their current, generally poor condition could result in an extra 6-7 million tonnes of $CO_2$ equivalents per year, in the context of the UK's total emissions of 455.9 million tonnes $CO_{2eq}$ in 2017. Restoration of degraded bogs would

avoid significant annual emissions (currently negating significant sequestration by woodlands and coastal habitats) and should be a particular priority.

## *Introduction*

Current trends in global biodiversity and climate are a source of concern to academic and civil society alike (IPCC, 2018, IPBES, 2019), as is the role of land management in both these challenges (IPBES, 2019, IPCC, 2019). However, whilst there is a general consensus that it is possible to address both issues together as 'twin crises' (Cohen-Shacham et al., 2016, Griscom et al., 2017), the mechanisms and pitfalls of doing so remain the subject of debate (Griscom et al., 2019, Runting et al., 2019, Lewis et al., 2019, Anderson et al., 2019).

While global agreements and conventions set the tone, action is undertaken at the national level. The UK is characteristic of the challenges inherent in addressing these problems. It has a relatively large population and small land area, with many competing land-uses and ecosystem services required from this finite land resource, coupled with complex issues of land tenure and political governance. The government has signalled its commitment to global biodiversity through its being a party to the current set of CBD targets that were to be met by 2020 (Convention on Biological Diversity, 2010), while the UK Biodiversity Action Plan (BAP) sets national targets for species and habitats conservation (UK Government, 1994) demonstrates it commitement to national biodiversity. As with many countries, the UK is an Annex I and II party to the UNFCCC, meaning the UK is required to produce a full annual greenhouse gas (GHG) inventory, and a signatory to the 2015 Paris agreement to restrict global climate change to 1.5° C above pre-industrial levels. The UK government has ambitious targets of its own and has recently committed to 'net zero' greenhouse gas emissions by 2050 under the 2008 Climate Change Act (UK Government, 2019a). In order to reach net zero, all sectors will need to contribute by reducing GHG emissions. However, increased GHG removals from the atmosphere will also be required (Griscom et al., 2017, IPCC, 2018, Anderson et al., 2019).

Progress to net zero could be achieved through multiple combinations of reductions in emissions and removals across sectors (Green Alliance, 2019, National Farmers' Union, 2019, Committee on Climate Change, 2020), with Land Use, Land Use Change and Forestry (LULUCF) being one sector. Figures suggest that in 2017 this sector contributed around -0.016 Gt $CO_2$ equivalents of the UK's net emissions (Department for Business Energy & Industrial Strategy, 2018), i.e. a small net sink, although Evans et al. (2017) report that emissions from degraded peatlands are vastly underreported such that the total LULUCF figure may be a considerable underestimate. This is in the context of GHG emissions from other sectors, with total UK annual emissions at 0.455 Gt $CO_{2eq}$ (Department for Business Energy & Industrial Strategy, 2018). Greenhouse gas removals technologies (GGRT) have been proposed to form part of future climate change mitigation pathways. However, many remain unproven or untested at scale (Smith, 2016, Smith et al., 2016), so the priority must be to cut emissions as much as possible in order to reduce the size of the deficit that GGRT are required to fill.

Photosynthesis, however, does offer a proven and scalable GGRT opportunity. Importantly, natural processes and management activities that benefit biodiversity - so called 'natural climate solutions' (NCS, *sensu* Griscom et al. (2017); i.e. ecosystem-based climate change mitigation, a subset of 'Nature-based Solutions' (NbS) to climate change (Cohen-Shacham et al., 2016)) - have been proven to be an effective GGRT. Sequestration of atmospheric carbon by vegetation occurs through natural ecosystem processes (photosynthesis) as part of the maintenance or creation of natural or semi-natural habitats. In the inescapable context of competition for land use to provide different benefits, NbS are likely to provide not only strong climate change mitigation benefits, but also a range of other services. For instance, more natural forests tend to provide both more biodiversity and a wider range of benefits than commercial forestry (Seddon et al., 2020).

Numerous studies have tested the spatial congruence and synergies between maintenance of biodiversity (in terms of spatial congruence of species distributions) and supply of a range of ecosystem services (e.g. Naidoo et al., 2008). For the UK, Anderson et al. (2009), Moilanen et al. (2011) and (Thomas et al., 2013) used spatial prioritisation software ('Zonation') to assess spatial conflicts and synergies between different services and biodiversity (of UK BAP Priority Species). Complementarity and irreplaceability approaches were used to assess the efficiency of different conservation strategies for delivering the full range of benefits. They found that strategies focussed solely on protecting carbon stores were largely inadequate for protecting biodiversity but when carbon and biodiversity value were given joint priorities, up to 90% of both could be protected. Reside et al. (2017) however, found that this synergy of carbon and biodiversity conservation is not universal, and complementarity approaches may be less valuable in areas of high endemism. They advocated caution in relying on climate change mitigation alone as a tool to simultaneously conserve biodiversity. Thus, the promotion of NbS will likely be beneficial to biodiversity, but should not be seen as a panacea for conservation of all species. Conflicts between carbon storage and biodiversity will likely arise.

Site-based conservation informed by species' distribitions is a widely used approach to efficient nature conservation. It is possible in a data-rich location like the UK. However, many species' distributions are poorly known, so habitat maps can be a further important vehicle for prioritisation, while acknowleding that they do not fully capture all species distributions. For instance, Anderson and Ferree (2010) and Beier et al. (2015) used a combination of geology and climate variables to predict species' distributions, the combination of which could be described as habitat or landform, in the context of climate change adaptation of land use planning for conservation. Habitat conservation does, therefore, form a key component of biodiversity conservation. In the UK for instance, many habitats contribute to the multiple national and international obligations the UK government has to biodiversity conservation, under various conventions and commitments (European Union Habitats and Birds Directives (Council of the European Communities, 1992, European Parliament and of the Council of the European Communities, 2009); UK Government 25-year Environment Plan for England (UK Government, 2018)).
A range of previous studies have estimated the scale or potential of climate change mitigation offered by such natural or semi-natural habitats or land use strategies encompassing them (Anderson et al., 2009, Gilroy et al., 2014, Williams et al., 2018, Alonso et al., 2012). However, the scale of the benefit of these habitats, for either biodiversity conservation or climate change mitigation, can be greatly affected by their condition. The links between ecological condition (and biodiversity conservation value) and the provision of ecosystem services, particularly climate regulation, have been investigated at the site or landscape scale. There is robust evidence that conservation, or restoration, of natural or semi-natural ecosystems supports the provision of greater levels of carbon storage and removal of atmospheric carbon dioxide, both in peatlands (Minayeva et al., 2016, Minayeva and Sirin, 2012, Joosten et al., 2016) and in forests (Chazdon, 2008, Alexander et al., 2011). However, benefits for, and synergies between, biodiversity and carbon storage/sequestration are greater from the conservation of intact habitats than from restoration (e.g. Martin et al., 2013), though still greater than continued degradation.

In this study we estimate the contribution to achieving net zero of the priority habitat element of UK biodiversity conservation strategy. We assess the co-occurrence of habitats of high conservation value and climate change mitigation value of different locations, based on the spatial distribution of habitats (specific land covers) of defined conservation importance (restricted or important assemblages / communities of species). Specifically, we use UK BAP Priority Habitats (hereafter BAP Habitats), as defined by the UK Joint Nature Conservation Committee (UK Government, 1994, Maddock, 2008). These habitats, with their characteristic species assemblages and communities, have been prioritised for their 'principal importance for the conservation of biodiversity' on the basis of their national or international importance, rarity or threat status, or functional importance to ecosystems or threatened species (UK Government, 1994). They form a central plank of UK's biodiversity conservation framework (Joint Nature Conservation Committee and Department for Environment Food & Rural Affairs, 2012). Continued and improved conservation of these areas forms the practical foundation for both conservation of priority communities of species and individual priority species. We assess the current carbon storage within BAP Habitats. We quantify the level of protection given to these habitats and their carbon stores, considering current national and international nature conservation designations. We go on to assess the current climate change mitigation contribution made by these areas and the likely influence of habitat condition on their current and potential future contribution.

***Methods***

To indicate the spatial distribution of natural and semi-natural habitats likely to be of significant biodiversity conservation value (hereafter 'High Conservation Value' HCV) we created a map of surrogate BAP Habitats across the entire UK, based on Land Cover Map 2015 (hereafter LCM, (Rowland et al., 2017)). BAP habitat descriptions were defined by the UK-wide Joint Nature Conservation Committee, in collaboration with the country statutory conservation bodies (English Nature, Scottish Natural Heritage, Countryside Council for Wales) at the end of the 20$^{th}$ century, to include the biogeographic variation across the UK. Political devolution has led to a divergence of conservation practice across the four nations of the UK (the devolved UK's constituent nations of England, Scotland, Wales (together also known as Great Britain) and Northern Ireland), such that whilst conservation priorities remain broadly unchanged and overseen by JNCC, mapping and surveying methods differ significantly. Consequently, to map HCV habitats across the UK requires unification of national land cover datasets that differ in underlying methodology, resolution and habitat discrimination. Therefore, we chose to use LCM as a UK-wide dataset of suitable thematic and spatial resolution to map HCV habitats. LCM is 'a parcel-based land cover map for the UK, created by classifying satellite data into 21 land cover classes'. These classes are based on the UK BAP Broad Habitat definitions, which are broadly aligned with BAP Priority habitats (Jackson, 2000). We refine the LCM data with other data sources to approximate a supra-national map of HCV land covers as a proxy for BAP Habitats, to compensate for the lack of BAP Habitat inventories at high resolution in countries other than England. We use data from national forest inventories (Forestry Commission, 2015, Northern Ireland Forest Service, 2013) ancient woodland surveys (Natural England, 2015, Scottish Natural Heritage, 2014, Natural Resources Wales, 2011, Woodland Trust, 2008), inter-tidal substrate surveys(Department for Environment Food and Rural Affairs, 2004, Natural Resources Wales, 2005) and the CORINE land cover map (Cole et al., 2015) to supplement LCM data and discriminate areas of broad LCM classes likely to be of significant biodiversity conservation value. We then assess the extent to which they contribute to climate change mitigation by;

a) storage of carbon already removed from the atmosphere, in soil and vegetation, using data from the Harmonised World Soils Database (Nachtergaele et al., 2011) and Milne and Brown (1997) respectively; and

b) potential for further ongoing removal of carbon from the atmosphere to vegetation and soil, and reduction of emissions of greenhouse gases from vegetation and soil. For this, we used annual emissions factors (net flux of the three major greenhouse gases ($CO_2$, $CH_4$ and $N_2O$)) from literature sources, based on the annualised climate warming potential of these gasses over 100 years (Forster et al., 2007).

We then assess the degree to which the resultant HCV habitats are protected under national and international legislation. We also assess the effect of ecological condition of HCV land covers on the scale of climate change mitigation.

**Representation of HCV (~BAP) Habitats**

Land cover data were taken from LCM at 25m resolution (Rowland et al., 2017). Data were supplied as a 2 band 25m Raster, with land cover represented by 21 target classes, carried in the first band pixel information.

Broad target classes were matched to BAP Habitats (Maddock, 2008) (Table A1). Land covers containing little, or no (or poorly quantified) vegetation or soil carbon are excluded ('Inland waters' (standing and flowing); 'Inland bare ground'; 'Supralittoral rocks'; 'Supralittoral sediments'; 'Littoral rocks,). Habitats that could not be associated with land cover types covered by the LCM (i.e. those below mean low water neap/sublittoral land cover) but included in the BAP Habitats list ('Sublittoral sediments' and 'Rock') and land cover with no semi-natural component ('Tilled land' ,'Urban' and 'Suburban' areas) were also excluded.

To better define several HCV habitats, we used additional habitat-specific information to discriminate between those land cover areas that approximate BAP Habitats and similar land covers of lower conservation priority. 'Broadleaved/mixed' and 'Coniferous woodlands' were cross-referenced with both the National Forest Inventory for Great Britain (Forestry Commission, 2015), the Northern Ireland Woodland Base Map ((Northern Ireland Forest Service, 2013) and with the four UK Country Ancient Woodland surveys (Natural England, 2015, Scottish Natural Heritage, 2014, Natural Resources Wales, 2011, Woodland Trust, 2008). This was done to exclude woodlands unlikely to be of high conservation value (i.e. plantation and production woodlands) while keeping those of higher biodiversity value (Ancient natural and semi-natural and replanted woodlands). It also enabled remaining woodland stands to be characterised, to allow representative vegetation carbon values to be apportioned.

Grasslands occur in several LCM classes. The 'Semi-natural calcareous', 'Neutral' and 'Acid' grassland categories were cross-referenced with the boundaries of Special Areas of Conservation (SAC,(Council of the European Communities, 1992)), Special Protection Areas (SPA,(European Parliament and of the Council of the European Communities, 2009))and Sites of Special Scientific Interest (SSSI) (Areas of Special Scientific Interest (ASSI) in Northern Ireland, (UK Government, 2019c) designated in each of the four UK countries, to gain a measure of HCV. All grassland polygons within 'Semi-natural calcareous', 'Neutral' and 'Acid' grassland categories that fell within designated areas were retained. Coastal and floodplain grazing marshes were contained within the 'Wet grassland' category, which contains many other lower conservation value management types. We cross-referenced this with the Corine Land Cover data class 411: 'Inland wetlands, inland marshes' (Cole et al., 2015), in order to separate HCV locations from lower value areas.

'Heathlands', 'Bog' and 'Fen, Marsh and Swamp' cover classes were retained in their entirety, as their vegetation is sufficiently distinct to separate them from confusion with other covers of lower conservation value.

'Littoral sediments' were also cross-referenced with other survey data to retain only those littoral sediments of HCV and some carbon storage value. This is because the broad LCM class contains shingles and sand, which may in some circumstances have biodiversity value, but contain little organic carbon. The cover class does not differentiate between these and muddy substrates, which are of high biodiversity value as well as containing large amounts of organic matter. To do this we used the Intertidal Substrate Foreshore (England and Scotland) (Department for Environment Food and Rural Affairs, 2004), and the Intertidal Substrate (Wales)(Natural Resources Wales, 2005) data layers. No such data layer exists for Northern Ireland, so we have excluded all littoral sediments in Northern Ireland from the maps. The only intertidal habitat represented in Northern Ireland is 'Saltmarsh'. This cover class was included in its entirety across the UK.

We tested our derived surrogate map of HCV Habitats for England against the Natural England BAP Habitats inventory map for England (Natural England, 2019) for spatial congruity. This revealed that the LCM-derived map is 74.6% correct in apportioning HCV habitats to BAP Habitats (Table A4). This gave us confidence that our surrogate map of HCV habitats across the UK was a good proxy for specific BAP habitats where mapping was not carried out outside England. However, there were some categorisation mismatches, notably between areas mis-classified as bog, heath and semi-natural grassland in either the source PH or target LCM classes.

We also calculated the vegetation and carbon stores for all UK land not covered by HCV cover as described above.

**Vegetation carbon stock attribution**

Vegetation (above, below ground & litter) carbon density values for each HCV class were taken from Milne and Brown (1997)(Table A2), for all cover classes except 'Broadleaved/mixed' and 'Coniferous woodland'. For these classes, where Forestry Commission/Northern Ireland Forest Service data gave more detail on management and habitat succession, we used a combination of carbon content data from Milne and Brown (1997) and IPCC (2006) (Table A2).

**Soil data and soil carbon stock attribution**

Soil data were taken from the Harmonised World Soil Database (HWSD)(Nachtergaele et al., 2011). Total soil organic carbon in the topsoil (**$C_{30}$**, $tCha^{-1}$) to 30cm depth was calculated as follows from the HWSD bulk density (**BD**, kg $dm^{-3}$) and soil organic carbon content (**SOC**, %):

$$\mathbf{C_{30} = ((SOC \times 0.01) \times (BD \times 10{,}000)) \times 0.3)}$$

Soil data for the top 30cm of soil from the HWSD were used for all except LCM/HCV classes 'Littoral sediments' and 'Saltmarsh', although it should be noted that much peat is likely to be deeper than this, and hence contain more carbon. Nearly all of the Littoral sediment and Saltmarsh habitats lie outside the coverage of the terrestrial soil data of the HWSD and furthermore, the bulk of carbon stored in these habitats is allochthonous, and largely independent of the underlying post-glacial soils or geology. Therefore, for these classes we used an independent estimate of below ground 'soil' carbon from Beaumont et al. (2014).

**Global Warming Potential data attribution**

Emissions factors, in $tCO_{2eq}$ $ha^{-1}$ $y^{-1}$ global warming potential over 100 years ($GWP_{100}$, after Forster et al. (2007)), were extracted from published literature and assigned to each of the broad HCV classes as defined in Table A3 (negative values represent removal from the atmosphere). In apportioning these factors, we also account for the effect of ecological condition on emissions, where it is likely to be important. We used the last condition estimates for features on SSSIs and Natura 2000 sites in the UK (Williams, 2006b), as a guide to the proportion of each HCV habitat class across the whole UK in good or poor condition and applied these proportions to the areas of each class, apportioning appropriate emissions factors to these areas. The greenhouse gas emissions from soil and vegetation of some habitat classes (semi-natural grasslands on mineral soils, woodlands, saltmarsh and intertidal mudflats) were assumed to be largely unaffected by condition, in as much as management sufficient to change emissions is deemed to be akin to land use change (e.g., conversion of forest to grassland, afforestation of heathland) and therefore we used a single $GWP_{100}$ value for each of these classes, regardless of condition. In other cover classes, management is known or likely to have a significant effect on emissions without engendering land cover change. Drainage of deep peat is known to increase $CO_2$ emissions to the atmosphere (Artz et al., 2012, Bain et al., 2011, Couwenberg et al., 2011, Evans et al., 2017,) and dissolved and particulate organic carbon loss in water. Changing vegetation communities (Couwenberg et al., 2011, Crowle and McCormack, 2009) and vegetation burning increases both greenhouse gas emissions and dissolved organic carbon loss *via* drainage (Clutterbuck and Yallop, 2010, Turetsky et al., 2014). These managements are listed as the most frequent activities leading to poor (= 'unfavourable') condition of blanket bogs on SSSIs and Natura 2000 sites in the UK (Williams, 2006b). We also included one instance of apparent land cover change as an indicator of poor condition. We assume that the presence of dwarf shrub heath or acid grassland vegetation on deep peat is the result of management-induced land cover change by drainage on blanket bog (e.g. (Ellenberg, 1988, Bragazza et al., 2006, Gunnarsson et al., 2002, Crowle and McCormack, 2009, Worrall et al., 2007). If these managements were reversed, blanket bog vegetation would re-establish. For these classes, we used an additional dataset to more precisely locate deep peat soils. The HWSD, at a resolution of 1km, gives soil information for the dominant soil type within each pixel and therefore may overestimate the amount of any habitat on peat and skew estimates of emissions. Therefore, we used a peat-specific data set to estimate areas, and therefore emissions, of dwarf shrub heath and acid grassland occurring on deep peat. We used the 'PEAT' polygons of the British Geological Survey's 1: 625,000 Superficial Geology Data (British Geological Survey, 1977). This category refers to ombrotrophic or minerotrophic peats formed under blanket or raised bogs, or fens (McMillan and Powell, 1999). For the absence of doubt, this includes only dwarf shrub heath established on deep peat and excludes heath vegetation established on mineral and shallow podzols. Whilst this distinction is poorly made by LCM (Rowland et al., 2017), use of the BGS superficial geology layer precludes non-peat areas. We apply $GWP_{100}$ values for good and poor condition in proportion to that indicated by Williams (2006b) For those habitats where condition is linked to underlying soil type, we use the poor condition $GWP_{100}$ value for all areas of these classes occurring on peat, and the poor/good condition $GWP_{100}$ values in proportion to the Williams (2006b) values for areas on mineral soil types.

**Legislative Protection**

Inclusion of land areas under national (SSSI/ASSI) or international (SAC, SPA) designations provides the highest degree of legal protection available to habitats in the UK. Whilst this protection is not strict, in the sense that much of the land so-designated is owned and managed privately, commercial or recreational land uses are permitted and enforcement of designated status is patchy, these designations are the basis for legal protection of biodiversity in the UK. Maps of HCV land covers were therefore overlain with the boundaries of these national/international nature protected site designations, to assess the degree to which the level of protection they have been afforded also co-delivers climate change mitigation, in the absence to date of specific legislation for protection of land to mitigate climate change.

***Results***

**Carbon Stocks**

Around 42% of land cover in Scotland is identified as being of high conservation value by our measures, compared to 17% in Wales, 16% in Northern Ireland and 9% in England. HCV habitats in Scotland account for approximately 66% of both area and amount of carbon stored in the UK's high conservation value land whilst England holds a further 24%, Wales 7% and Northern Ireland the remaining 3% (Figure 1, Table 2). We estimate that 0.55 Gt of carbon, the equivalent of 2. 6Gt $CO_{2eq}$ is currently stored in above ground vegetation and the top 30cm of soils within these HCV habitats in the UK as compared with 1.3 Gt of carbon in all non-HCV land (Table 1, Table 2). The largest carbon stocks are held within dwarf shrub heath (0.24 Gt C), bogs (0.10 Gt C), semi-natural grassland (0.09 Gt C) and woodlands (0.08 Gt C). Fifty-three percent of this carbon store is within areas protected under national and/or international designation (Table 1). The largest absolute protected stocks (0.094 Gt) are in protected semi natural grasslands. However, we do not know how much HCV semi-natural grassland exists outside protected areas, since we have no measure of conservation value of these grasslands save the inclusion within designated areas. Second is heaths with 0.08 Gt protected, accounting for just 34.4% of their stock (Table 1). Of the total area of HCV land cover (5.02 million hectares), 1.5 million hectares are estimated to be on peat soils (Table 3). Of this, 0.59 million hectares are covered by non-peat-forming vegetation types, thus likely to be the result of changed land management (e.g. drainage, damaging grazing, burning etc.), and likely to be losing carbon (Bain et al., 2011).

The majority of carbon stocks in HCV habitats in Scotland are in upland areas, especially on the mainland north of the Highland Boundary Fault. There were also notable areas in the Hebrides, Shetland and Southern Uplands. Upland areas also contain the most extensive carbon stocks in HCV habitats in Wales and England, but there are a number of coastal and estuarine areas (e.g. The Wash, Morecambe Bay) also highlighted (Figure 1). This upland distribution of carbon is reflected in the high levels seen in semi-natural grassland, bogs and heaths in Table 1.

**Net greenhouse gas emissions**

Net GHG blance of the three most important greenhouse gases from the total area of HCV habitats, in its current range of condition, is estimated to be around -0.0087 Gt $CO_{2eq}$ per year (Table 3), indicating that overall the land in HCV areas is sequestering marginally more greenhouse gas than it is emitting. Condition monitoring scores vary considerably between habitats, but overall, they indicate that approximately 1,600 kha of the total area of 5,000 kha are in good condition (31.6%). If all land not in good ecological condition were to be restored to good ecological condition (especially by rewetting of peat soils), this net sequestration could be increased by between c. 58% and 68%, to between -0.014 and -0.015 $GtCO_{2eq}$ per year (Table 3). The exact level will depend on the intermediate flux response of restored ecosystems to restoration (Table A3).

***Discussion***

Our analysis highlights that land of nature conservation value is also a significant store of carbon, by virtue of both relatively high carbon densities, but also large areas of land. This carbon is stored chiefly in soils (Milne and Brown, 1997), but also in vegetation, especially trees. This is not distributed equally across habitats or the UK but is concentrated in bogs, heaths and semi-natural grasslands. These are primarily distributed in areas of higher altitude and latitude, although there is also an East-West gradient, with the preponderance of grass and heath in the wetter western parts of the UK. It is likely that much of what is termed 'heath' and 'grassland' in this study is actually degraded blanket bog (as detailed in the Methods) and is therefore losing stored carbon, and not sequestering at all. We report that these HCV habitats are currently cumulatively sequestering carbon, but estimates indicate that restoration of all of these habitats could increase this by over a half. The time taken to realise these full benefits, is however, uncertain, but would probably be decades and is unlikely to be fully realised in practise, given the complexities of land tenure and land-use culture in the UK.

Thomas et al. (2013) explored the degree of synergy between carbon and biodiversity conservation strategies for the UK. They used BAP priority species' distributions as indicators of spatial conservation value, traded-off against vegetation and soil carbon distribution. They found that using species' distributions weighted by range size, to favour areas important to range-restricted and rare species, there was some synergy between strategies to optimally conserve either carbon or biodiversity. However, this was not ubiquitous, and a strategy that aimed to maximise both, would conserve around 90% of each. Their single-benefit-driven strategies showed that areas important for carbon or biodiversity conservation in the UK are quite different, most carbon concentrated in the north, and most rare species in the south. In contrast, using BAP habitat distribution (in which prioritisation for conservation is made by virtue of the co-occurrence of species assemblages, rather than individual species ranges) we find that most carbon conserved through habitat-based conservation is concentrated in the north of the UK. This difference is likely to be due to the forces that have driven conservation and biodiversity trends in the UK over centuries. The south of the UK, heavily populated and cultivated, has long seen pressure on semi-natural habitats and the species they support through land use and agricultural intensification (Wilson et al., 2009, Proctor, 2013). Therefore, species with biogeographical characteristics suited to the southern UK have been reduced in range and distribution, leading to them being more likely to have been prioritised as in need of conservation (or have been lost entirely to land use change). Conversely, the northern parts of the UK, less populated or suited to intensive agriculture, have experienced fewer pressures on biodiversity (viewed either as species' or habitats' distributions). This has favoured larger areas of less altered semi-natural vegetation and the species they support, thus driving the co-occurrence of carbon and biodiversity value, when not based directly on species' rarity.

This provides an alternative view of the considerable contribution of conservation to climate change mitigation to that of Thomas et al (2013), but as conservation in the UK is viewed both through the lenses of species and habitats, future strategies will have to take both into account. For climate change mitigation, our study provides information on the value of semi-natural habitats to carbon sequestration and storage and the co-benefits that increased protection and better management can bring. We caution, though, that land use-based climate change mitigation (through, for example, tree planting) should not damage these HCV habitats if we are to conserve biodiversity and current carbon socks. This does not preclude the need to conserve range-restricted and rare species, as studied by Thomas et al. (2013), but that multi-objective mechanisms may not always be the correct vehicles to achieve this.

The scale of topsoil and vegetation carbon storage under HCV land covers is not insignificant as a proportion of UK total soil and vegetation carbon estimates, either by our method (1.84Gt C) or made by other authors. Milne and Brown (1997) estimated UK total carbon storage at 0.114 Gt in vegetation and 9.84 Gt in soil, but this covered the entire soil profile depth, including deep peats of up to several meters. Bradley et al. (2005) estimated soil organic carbon to 30cm in topsoil at 2.54Gt whilst Reynolds et al. (2013) give a figure of 1.39Gt to a depth of 15cm. Our estimate for HCV land covers of 0.551 Gt C (including soil – to 30cm depth - and vegetation carbon) equates to around 30% of the UK's topsoil and vegetation carbon stock from approximately 20% of the land area.

We have quantified carbon in the top 30cm of soil profiles, partly as this is likely to contain the bulk of the soil organic carbon of mineral soils (Bradley et al., 2005), but also as this is the part of the soil profile most at risk to current and near-future management changes that might jeopardise this store (Kimble et al., 2001, Woomer et al., 2001). As such, our estimates of the contribution of HCV land covers to UK carbon storage do not account for the significant amounts of carbon below 30cm, particularly in deep peat (Milne and Brown, 1997) and hence are likely to be conservative. However, in terms of climate change mitigation, it is the surface soil horizons' carbon stocks that are likely to be most influenced by soil and land management and so it is these we have assessed. Such management-driven changes may be both negative (loss through tillage, burning, grazing etc) or positive, either by reversal of damaging management practices through ecological condition improvement or through land use change. Much work has been done on the responses of peatland soils and vegetation to restoration (usually reversal of drainage by ditch blocking) and the effects of this on GHG fluxes (see e.g. Wilson et al. (2016), Evans et al. (2017)). Generally, restoration of peatlands entrains a reduction in $CO_2$ emissions from microbial degradation of peat, but increased waterlogging leads to an initial increase in methane emissions (Couwenberg et al., 2011). With time and careful management of water levels (to just below the surface) there is a reduction in overall GHG balance, towards that of pristine habitats (Evans et al., 2017, Bain et al., 2011).

Soil carbon stocks dominate the total land cover carbon store in all UK habitats, though the smallest difference between soil and vegetation pools is in woodland habitats (Milne and Brown, 1997). Therefore, the largest functional threat to the UK's ecosystem carbon stores is likely to be management or land-use changes that affect the stability or function of soils. Many UK land covers (e.g. improved grassland, commercial forestry, arable land) are subject to large scale soil and vegetation management practices which have significant impacts on carbon storage (e.g. tillage, grazing, fertiliser application, drainage) (Dawson and Smith, 2007, Ostle et al., 2009, Smith et al., 2007). However, managing these areas to conserve declining species associated with low intensity agriculture (High Nature Value farming (e.g. Finch et al., 2019, Lamb et al., 2019), could make a further synergistic contribution to countering climate change and biodiversity loss. If such action was targeted on soils with high carbon sequestration potential (e.g. organic soils on drained lowland fens), there could be even greater gains. Whilst we have elected to concentrate upon land covers of higher conservation value, many of these semi-natural habitats are still subject to varying degrees of anthropogenic influence (including practices mentioned above), which may affect their ecological or climate regulation value. In reality, many land covers in the UK lie on a continuum of land use intensity, by virtue of which they may, or may not be, classed as HCV. Those for which this intensity of management has been sufficient to reduce their conservation value will lie outside our definition, whilst others may retain some conservation value currently, but be subject enough to these practises that they may be deemed as being in unfavourable ecological condition. With most of these land covers, there is insufficient evidence of the influence of ecological condition (a proxy for management) on their carbon storage or net emissions, unless that condition were akin to land cover change (loss of woodland cover for example). However, and importantly, there are some land covers for which management is known or likely to have a significant effect on emissions without engendering land cover change (e.g. burning or drainage on blanket bog, maintenance of grassland on lowland fen soils). For these, we use emissions factors derived from a large body of evidence linking management practices (reflected in condition measures) to changes in greenhouse gas fluxes(See Methods and Table A3), largely caused by a switch in predominant soil conditions from anaerobic to aerobic metabolism of organic soil microbial flora. This switch, largely brought about by drainage and reflected in altered characteristic vegetation communities on peat soils, entrains a loss of soil carbon through microbial respiration, as well direct physical loss e.g. through erosion. Thus, expansion of draining in these areas could result in substantial loss of soil carbon. Inversely, restoration of these habitats has the potential to prevent further soil carbon loss and sequester considerable volumes of carbon from the atmosphere.

We have excluded semi-natural grasslands that do not lie within designated sites because of the paucity of accurate data on the nature of non-intensive grasslands in the UK and are therefore likely to be underestimating the current and future climate change mitigation (and conservation) potential of this broad land cover class. Indeed, there are significant areas of floodplain grazing marsh and upland in-bye grassland that are not designated but, nonetheless, are (or could be) under favourable management for biodiversity conservation (P. Grice, Personal communication). For a country that is now much less forested that it used to be, there is, however, potential for much further sequestration through additional tree cover, which might logically be located on what is currently low productivity grassland. However, given the current lack of certainty of the location of undesignated species-rich grassland, the pros and cons of additional tree cover in different places will first need to be elucidated to inform such action, to ensure that loss of grasslands to trees does not entail loss of grasslands of conservation importance.

Our results indicate that in current condition, HCV land covers exert an annual net negative GHG balance of approximately 8 million tonnes of $CO_2$ equivalents. This represents around half of all annual negative emissions in the UK due to LULUCF ((Department for Business Energy & Industrial Strategy, 2018), from around 20% of the land area, or equivalent to the entire UK annual agricultural emissions of around 5.5 million tonnes of $CO_2$ equivalents (Department for Business Energy & Industrial Strategy, 2018). We estimate that restoration of degraded bogs to a more favourable ecological condition might increase this figure to around -14 million tonnes of $CO_2$ equivalents. This is the equivalent of approximately 10% of all the UK's annual transport emissions (Department for Business Energy & Industrial Strategy, 2018). In this study, we have only considered the influence of condition and management on peat soils, since there is a considerable body of evidence to link management and land cover condition to greenhouse gas fluxes. However, the evidence of such links is lacking for most other semi-natural habitats at all but the simplest of habitat descriptions and condition estimates are old and incomplete across all these habitats. Thus, we think that our estimates of the scale of emissions and potential improvements through improved management are likely to be very conservative and quantifying contributions from other, non-peat substrates should be a priority for further work.

There are clear benefits to climate change mitigation from the continued protection and improved management of the HCV land covers identified in this study, but beyond this, there are also benefits to wider biodiversity. Currently, whilst a proportion of HCV areas enjoy a degree of statutory protection, around half does not, and even within those sites protected, a high proportion of features were deemed to be in suboptimal condition in 2006 (Williams, 2006a). The fraction of this area that has been assessed more recently continues to be unfavourable (UK Government, 2019b). Improvement of the ecological condition of both the statutorily protected and unprotected areas would not only improve their climate change mitigation potential but significantly improve their conservation value (Lawton et al., 2010). Lawton et al. (2010) concluded that the current state of protected areas in England was insufficient to provide adequate safeguard against further species and habitat losses and that semi-natural habitats outside this network were under-protected and under-managed to achieve the same goals. This varied tenure (from private to state owned), lack of rigorous protection and patchy management also places a substantial proportion of the carbon stored within these habitats at continued risk, and improvements in these would benefit both biodiversity and climate change mitigation.

If renewed effort were made to redress some of these shortcomings, by improving the ecological status of HCV areas, there would be significant benefits to both the status of their conservation value and the climate change benefits they provide. We do note, though, that where conservation is the prime driver of land management, there may be trade-offs with climate change mitigation. For example, lowland heathland and species-rich grassland management for conservation often results in the removal of trees, or at least interruption of vegetation succession, which clearly limits the potential benefits for carbon storage.

Our approach has necessitated several assumptions which have generated uncertainties that must be borne in mind when interpreting the results. The lack of consistent UK-scale mapping of the UK's priority habitats has meant that we have had to create a proxy map of these from other UK land cover data, of varied resolutions, ages and discriminatory power (to determine true ecological character). This will likely mean that our estimates of the area and conservation value of land covers will be less than optimal, but we have no reason to believe that this introduces any systematic bias to our results. Given the lack of quantified uncertainties in the underlying vegetation and soil carbon datasets, the absolute estimates of carbon stocks/removals should be treated with caution, though the relative stocks among different habitats should be more robust. These assumptions and 'work arounds' have been necessary because of the paucity of national coverage and knowledge of the UK's semi-natural priority habitats, and their condition. These limitations are likely to be generic across many countries, and whilst this will limit the applicability of our study in many countries, those with similar knowledge of land covers should be able to replicate this approach. The lack of condition assessment of protected areas in the UK in recent years is a serious issue that urgently needs addressing and, at the very least, has inhibited assessment of whether the UK is meeting its international biodiversity obligations, or whether national policies are working. We believe that better national strategies of mapping and condition assessment of valuable semi-natural habits will enable better stewardship of these, by statutory, NGO and private land managers. If this were to happen, targeted and cost-effective ecologically sensitive management could be implemented across large areas, leading to increased climate change benefits and ecological connectivity, to the benefit of UK nature conservation and society.

Despite these assumptions, we contend our analysis makes two points clear. First, it provides a quantitative indication of the large-scale climate change mitigation co-benefit already afforded by the continued existence of (HCV) land covers that provide significant conservation benefits. Second, it indicates that improvement in the ecological condition of these areas of land could effectively double this climate change mitigation service, whilst providing a necessary boost to the status of several nationally and internationally important priority habitats and the species they support. This scale of conservation action is the least that will be required to achieve, for example, the UK government's aims under the 25 year environment plan for England but, if achieved, will clearly also provide a significant contribution towards the aim of Net Zero under the climate change act (UK Government, 2019a).

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Table 1. Estimated total vegetation and soil carbon stores by HCV cover class, and levels of statutory protection.

| | **Total Area (ha)** | **Protected area (ha)** | **C store (Gt)** | **Protected C store (Gt)** | **% C protected** |
|---|---|---|---|---|---|
| **Bog (blanket & raised)** | 956,853 | 426,877 | 0.10 | 0.05 | 47.0 |
| **Fen** | 17,414 | 8,794 | 0.002 | 0.001 | 50.0 |
| **Heath** | 2,440,951 | 836,930 | 0.24 | 0.085 | 34.4 |
| **Littoral Mud** | 163,872 | 154,341 | 0.02 | 0.02 | 94.7 |
| **Saltmarsh** | 79,589 | 62,302 | 0.01 | 0.01 | 77.8 |
| **Semi natural grass** | 940,686 | 940,686 | 0.09 | 0.09 | 100* |
| **Wet grassland** | 2,348 | 893 | 0.000 | 0.000 | - |
| **Woodland** | 421,943 | 164,742 | 0.08 | 0.01 | 11.0 |
| **Total** | **5,023,656** | **2,595,566** | **0.55** | **0.26** | 47.0 |
| | **$CO_{2eq}$** | | 2.02 | 0.95 | |
| | **% unprotected** | 48 | | 53 | |

Table 2. Estimated total vegetation and soil carbon stores by UK country, and levels of statutory protection.

<table>
<tr><th></th><th>National Area (ha)</th><th>Area under HCV land cover (ha)</th><th>Protected area (ha)</th><th>C store (Gt)</th><th>Protected C store (Gt)</th><th>Non-HCV C Store (Gt)</th></tr>
<tr><td>England</td><td>13,045,920</td><td>1,129,075</td><td>747,119</td><td>0.13</td><td>0.08</td><td rowspan="4">1.3</td></tr>
<tr><td>Northern Ireland</td><td>1,418,191</td><td>220,024</td><td>62,379</td><td>0.02</td><td>0.01</td></tr>
<tr><td>Scotland</td><td>7,881,042</td><td>3,342,794</td><td>1,274,440</td><td>0.34</td><td>0.12</td></tr>
<tr><td>Wales</td><td>2,078,224</td><td>352,433</td><td>260,654</td><td>0.04</td><td>0.03</td></tr>
</table>

Table 3. Net Greenhouse Gas emissions of broad HCV cover classes by soil type and condition. Negative values represent atmospheric removal (i.e. climate cooling). * Represents fluxes accounting for emissions due to transition process after restoration towards target habitat, of non-peat forming vegetation communities to fen or bog. ** Represents fluxes potentially possible once vegetation communities are restored to peat-forming vegetation types after restoration.

| **Broad Habitat Class** | **Soil Type** | **Total Area (ha)** | **% in Good Ecological Condition** | **Flux in current condition ($tCO_{2eq}$)** | ***Flux if all restored ($tCO_{2eq}$)** | ****Flux if all in good condition ($tCO_{2eq}$)** |
|---|---|---|---|---|---|---|
| **Semi natural grass** | Mineral | 652,612 | 0.31 | -1,011,549 | -1,011,549 | -1,011,549 |
| | Organic | 288,074 | | 786,442 | 233,340 | -2,881 |
| **Broad-leaved Woodland** | Mineral | 274,838 | 0.35 | -2,943,515 | -2,943,515 | -2,943,515 |
| **Conifer woodland** | Mineral | 147,105 | 0.46 | -2,575,809 | -2,575,809 | -2,575,809 |
| **Wet grassland** | Mineral | 1,775 | 0.30 | -2,752 | -2,752 | -2,752 |
| | Organic | 573 | | 1,563 | 3,648 | -349 |
| **Bog** | Organic | 956,853 | 0.53 | 1,222,667 | 359,203 | -9,569 |
| **Fen** | Organic | 17,414 | 0.38 | 25,729 | 65,346 | -10,623 |
| **Heath** | Mineral | 2,138,458 | 0.20 | -4,318,730 | -7,377,680 | -7,377,680 |
| | Organic | 302,493 | | 825,806 | 245,019 | -3,025 |
| **Saltmarsh** | | 79,499 | 0.50 | -333,896 | -333,896 | -333,896 |
| **Littoral Mud** | | 163,872 | 0.66 | -383,460 | -383,460 | -383,460 |
| **Totals** | | **5,023,656** | | **-8,707,503** | **-13,722,104** | **-14,655,106** |
| | | **Total ($GtCO_{2eq}$)** | | -0.0087 | -0.0137 | -0.0147 |
| | | **Difference ($GtCO_{2eq}$)** | | | 0.0050 | 0.0059 |
| | | **% change in mitigation** | | | 57.6 | 68.3 |

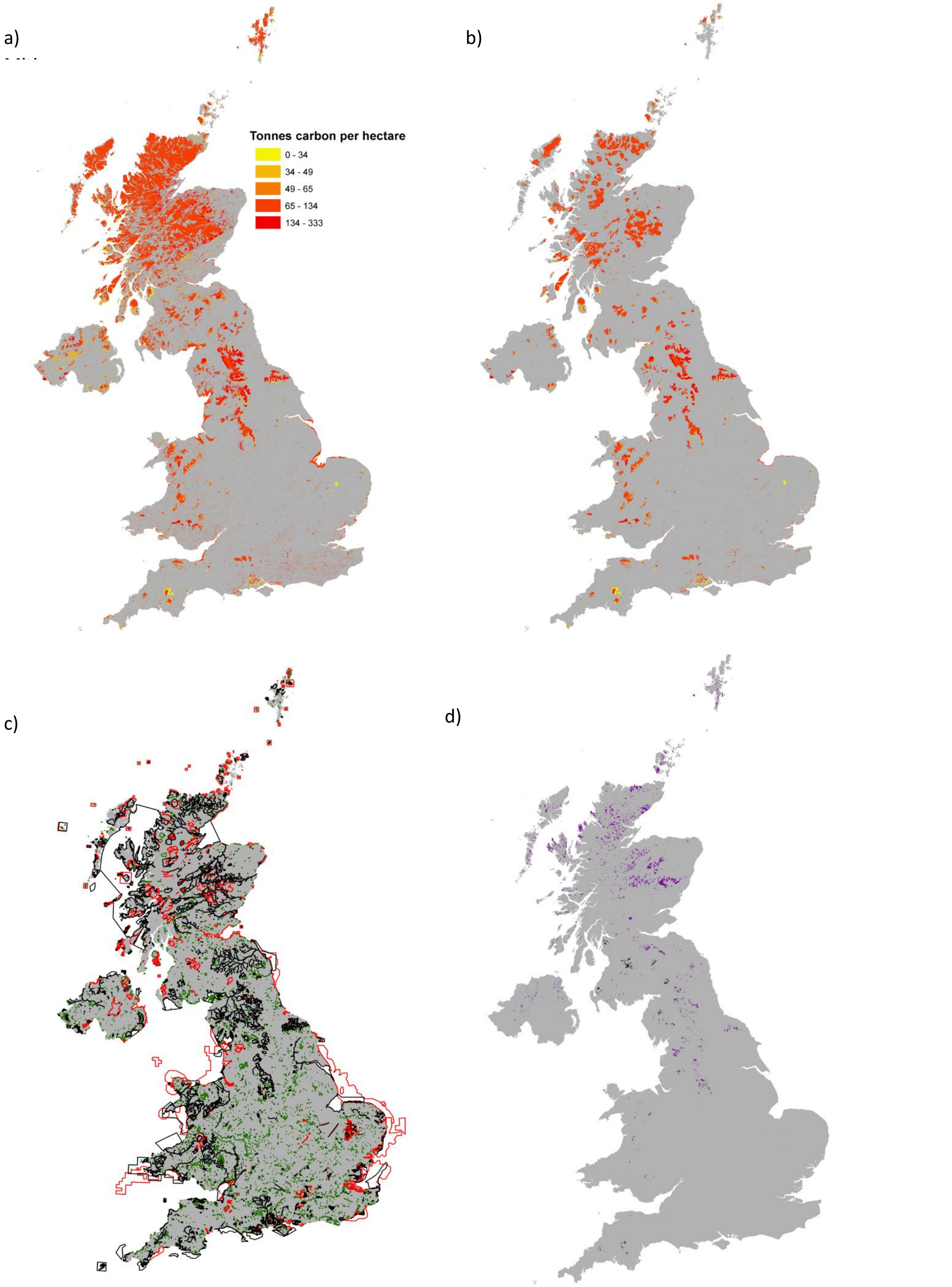


Figure 1. Distribution and land designation status of High Conservation Value habitats in the United Kingdom. a) Soil (to 30cm) and vegetation carbon density in tonnes per hectare in all HCV habitats; b) carbon density in national and internationally designated areas; c) location of nationally and internationally designated areas in the United Kingdom: green = SSSI/ASSI, black = SAC, red = SPA (for details of designations see

text); d) distribution of non-peatland habitats on deep peat (as defined in the BGS superficial geology layer) in the UK: purple = heathland, black = semi-natural grasslands.