Restoration of lowland raised bogs in Scotland: Emissions savings and the implications of a changing climate on lowland raised bog condition

Final report

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Sphagnum hummock and lawn on a regenerating cutover peatland. Rebekka Artz

Drained area of lowland raised bog with high *Calluna* cover. Ian Leith (CEH Bush)

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Damaged raised bog with plantation forestry. Ian Leith (CEH Bush)

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1. Executive Summary

There are various definition issues in describing the lowland raised bog resource in Scotland. Based on the UK Lowland Raised Bog Inventory (LRBI, 1996), **the total extent of raised bog habitat in Scotland with underlying peat that is more than 1 metre in depth is 27,884 ha**. **Using the same definition criteria, the extent of lowland basin peat in the Scottish Soils Knowledge and Information Base (SSKIB) totals 34,903 ha** on underlying peat deeper than 0.5 m. These differences reflect mostly the different definition of minimum peat depth, but also historic land use conversions that resulted in exclusion from the LRBI as well as basin peat deposits that are designated as blanket bogs, intermediate bogs or upland habitat. Due to the small size of many raised bogs, previous national assessments of extent, in particular of relic bogs, can have relatively large errors. Progress could be made in future by integrating remote sensing data.

It was unfortunately not feasible to obtain condition data for the full extent of all lowland basin peats other than the data from the Land Cover of Scotland 1988 (LCS88). There are a number of GIS shape files that include NVC categorisation and condition classes for the LRBI dataset, resulting from a combination of aerial photographic interpretation and field surveys. However, the files obtained during this project contained overlapping data and incomplete coverage of the LRBI and therefore were not useable for the purpose of this review.

Due to these mapping issues above, it is difficult to arrive at consistent figures for the areas of lowland raised bog in different current condition classes. For example, the LRBI figure of **2,515 ha in 1996 for bogs in near-natural condition**, is not verifiable through other means and thus more recent figures cannot be given**. There are substantially larger areas of lowland raised bog (around 10 kha) that still support natural peatland vegetation in good or moderately degraded condition**. Other areas that could potentially be actively peat-forming could include **revegetated or regenerating peat cuttings (estimated between 2-3 kha)**. Many former raised bogs were targeted for **conifer plantations or have become encroached by woodland (ca 9.5 kha)**. Historic drainage for agricultural improvements and/or climatic change may have caused some of the original lowland basin peat areas to become **drier grassland and other pasture land or drier heathland (ca 11 kha)**. A fair proportion was fully converted to **arable land (2-3 kha)**. Even areas that are still raised bog habitat today generally have at least one area within it that has been altered due to drainage, planting, or other factors. It would be beneficial to assess such sites on a case by case basis. It was not possible to extract the area of raised bogs currently under designation as such sites can be designated for a number of features of which lowland raised bog habitat may be only one. However, amongst the sites which carry a designation for raised bog habitat, as a result of substantial management efforts to improve the condition of unfavourable lowland raised bog features, the **proportion of features in favourable condition has risen from 30% in 2005 to 57.7% in March 2010**.

The LRBI identified a total of 27,884 ha of raised bog sites in Scotland, of which 10,071 ha classes as 7110 Active Annex I habitat. Hence, **this would equate more than 15,000 ha of further raised bog remnant that could be suitable for active restoration and may qualify as 7120 Annex I habitats. Using the SSKIB figures, this figure could equate to ca 27,000 ha suitable for restoration**.

On the basis of the definition of lowland peat applied here, the lowland basin peats hold 64 Mt of carbon, which equates to 18 years of the Scottish transport emissions, if based on the total transport emissions in 2009. The deepest and/or largest lowland basin peat deposits, and hence the largest individual carbon stores, appear to coincide with many of the larger LRBI raised bog sites in the North-East of Scotland, the Central Belt and Dumfries and Galloway.

In terms of current carbon accumulation potential, only Auchencorth Moss (ranked fifth in terms of C) has been studied enough to produce a full budget over a number of years. Based on a number of literature values, the likely range of sequestration rates in bogs of varying condition classes based on CO_2 emissions factors or modelled full C balances were produced and the data used to calculate the likely annual sink strength of bogs in good condition and the likely emissions from those in a degraded state. By these calculations, **the current net amount of carbon sequestered by the remaining raised bogs in good condition is between 5,000 and 20,000 t CO_2e yr^{-1}**, most likely nearer the lower end of this figure. Unfortunately, this is more than counteracted by emissions from raised bogs in degraded condition classes. **Our calculations suggest that the total emissions from degraded sites are around 21,000-143,000 t CO_2e yr^{-1}**. These emissions comprise **0.6-4.6% of the total estimated agricultural and other land-use related emissions** of 3.2 Mt of CO_2e yr-1 for 2009. With raised bog only occupying 0.17% of the land area or less, this emphasises that raised bog degradation is making a disproportionately large contribution to land-use related emissions. Therefore, restoration of lowland raised bogs in poor condition has the potential to provide substantial emissions savings.

The likely effect of climate change on raised bogs was examined using bioclimatic envelope and species niche distribution models. These models aid in the identification of areas likely to be under climatic stress. The bioclimatically suitable spaces for raised bog in Scotland suggest a spatial shift in where such habitat may thrive. The climate envelope for raised bogs moves in to upland areas currently dominated by blanket bogs and some of the upland basin peats that did not meet the selection criteria for lowland raised bogs. The areas most likely to be under climatic stress are in the Central Belt and the coastal West, while the Grampian raised bogs were likely to experience less climatic stress. Species niche distribution models for e.g. *Sphagnum* mosses also suggest some spatial changes rather than decline. However, such models were generally trained on a much larger dataset and emphasise change in habitat suitability rather than species cover, which is highly uncertain. The implications for raised bog habitat in Scotland from these models are that many areas may require specifically targeted management that alleviates the stress from altered rainfall patterns or increased summer temperatures.

Measuring full carbon budgets is expensive and time consuming, hence development of proxies has been the focus of recent research efforts. Vegetation cover in particular is often seen as a good indicator of site hydrological condition. These correlations, however, generally do not explain more than 50% of the variation in carbon flux. On the other hand, vegetation may influence GHG fluxes directly (rather than simply providing a proxy for water table) due to the role of some species in transporting methane to the atmosphere, and others (e.g. *Sphagnum*) in sequestering CO_2 into peat. Hence, there is considerable potential in the use of vegetation proxies for GHG flux estimation, but further developmental work is required. Vegetation composition can be directly assessed via remote sensing techniques, using high quality satellite imagery. Extraction of vegetation parameters from multispectral data, such as calculation of the commonly used normalized difference vegetation index (NDVI) is also possible. NDVI can be used as an indicator of relative biomass, greenness, and, if adequate ground truthing data are available, the carbon fixed through primary production. The ability to make such predictions for Scottish peatlands is still some way ahead, due to the need to parameterise vegetation indices for the vegetation types found within these peatlands.

Coordinated flux measurement programmes are needed to help calibrate proxy models and to help refine carbon emission and stock calculations. There are currently a small number of high-quality research programmes ongoing, but as yet no integrated national-level flux measurement programme. Given the complexity and heterogeneity of peatlands in general, and raised bogs in particular, there would be clear benefits to coordinating any new measurement activities within Scotland, and more broadly in aligning these with existing and new research and monitoring elsewhere in the UK. Given the relatively high cost of establishing a full C/GHG measurement programme, this could permit more complete coverage of different peat management and condition types. Long term monitoring of the impact of climate change would also aid testing of the bioclimatic envelope and species niche models, by validation of the suggested shift in vegetation. Such work could help advise whether restored and remaining active raised bogs are more resilient or can adapt to climate change impacts.

Considerable peatland restoration work has already been initiated in Scotland. The largest of these initiatives was an EU-LIFE Project with the Scottish Raised Bog Partnership (a partnership between Forest Enterprise (Forestry Commission Scotland), Scottish Natural Heritage and the Scottish Wildlife Trust), which completed restoration work at eleven sites. In addition, the Grampian Lowland Bog Scheme (2003-2006) as well as the South Scotland Bog Scheme (SSBS, 2006 onwards) still have ongoing management agreements. Its successors, the current SRDP Axis 2 options, have two schemes, the options for 'Management/Restoration of Lowland Raised Bogs', with or without a grazing management, as well as the 'Buffer areas for Fens and Lowland Raised Bogs' scheme. Both include payments for capital costs as well as annual measures for restoration and monitoring. The benefits of this restoration work should be seen within the next decade if those sites continue to follow a trajectory of return to an active raised bog habitat. Hence, there will already be some avoided loss that has been realised through the various restoration programmes carried out to date and the current SRDP targeted programme, although we were not able to quantify this within this report.

An initial review of the costs and benefits of raised bog restoration highlights the economic benefits. **The present value of the entire raised bog carbon stock would equate to £317 million – £8.2 billion**, depending on which carbon pricing indicator is used. **The annual carbon savings that sequestration performed by the sites that are still currently in good condition provide would be valued at between £35k and £2.5 million (depending on whether the LRBI or LCS88 derived condition categories are used, and which carbon price is applied). The annual emissions from the raised bogs in degraded condition run to an estimated average of £350k to £7 million**.

The figures collated for the cost of restoration suggest an initial capital expenditure for one-off costs such as tree or shrub removal and/or drain blocking of ca. £1,280 ha^{-1}. In addition, annual maintenance costs may be between £40 and £500 ha^{-1}, depending on whether only light grazing management may be required or whether scrub may need controlled and dams checked at regular intervals. **Scaled up to the entire raised bog resource, this suggests capital restoration costs in the order of £20-£32 million for the whole of Scotland, and a further £650k annually for maintenance costs**.

Such estimates suggest a good cost:benefit trade-off could be reached relatively quickly. **It may be beneficial to see the capital expenditure as expenditure to safeguard the total carbon stock in the long term as well as reducing emissions in the short term and the annual management costs as the tool to produce carbon savings by reducing annual net emissions**. Hence, the large capital expenditure figures, when set against even the value of the stock appear a good investment, with the potential savings from reducing the net annual emissions forming the return on investment.

A formal economic cost:benefit analysis for carbon savings on raised bogs would necessitate a number of data that are not currently readily available. While these figures illustrate the potential economic benefits of restoration in carbon terms, figures for the actual reduction in carbon emissions through restoration are extremely scarce. To calculate a return on investment, it is necessary to know the trajectory of carbon emissions from a site that has undergone restoration; in other words, the carbon improvement in t C per ha^{-1} per yr^{-1}. As yet, there are few peatland restoration projects that have produced full carbon budgets before, during, and after restoration and hence there is a paucity of values on which to base such calculations. However, the trajectory in emission reductions that has been observed in various European studies suggests that a benefit may be reached within a relatively short timeframe. The precise time post-restoration when a significant carbon benefit can be measured is likely to be dependent on the starting condition of the peatland and the historical types and severity of disturbances. Consequently, cost:benefit ratios will be highly site-specific.

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2. Background

Lowland raised bogs are rain-fed peatland ecosystems which develop primarily in areas with topographic depressions, where drainage may be impeded by a high groundwater table, or by low permeability of the underlying substrata such as estuarine, glacial or lacustrine clays. The resulting constant waterlogging, decreased oxygen availability and hence anaerobic conditions impede the decomposition of plant material, leading to an accumulation of peat. Continued peat accumulation elevates the bog surface above groundwater levels to form a dome from which the term 'raised' bog is derived. The key distinction between fen and raised bog is the source of moisture; raised bogs receive all water inputs from precipitation and not groundwater that supplies the fen peat below. Peat depths can vary considerably but can exceed 12 metres. In Scotland, peat is defined as an organic soil of more than 0.5 meters depth.

The accumulated peat separates the surface vegetation from the nutrient influence of the underlying groundwater and vegetation relies exclusively on precipitation for nutrients to support growth. Hence, all raised bog ecosystems are rain-fed (ombrotrophic) environments, which support a distinctive suite of vegetation types. Although low in overall diversity, raised bogs support specialised plant assemblages dominated by mosses of the genus *Sphagnum,* as well as a number of higher plant species, some of which are scarce, including, for example, bog rosemary *Andromeda polifolia* or great sundew *Drosera anglica.* The raised bog surface microtopography generally consists of a patterned mosaic of pools, hummocks and lawns, created in part by the existing vegetation types. This provides a range of hydrological regimes which support different species assemblages at the microtope level. Lowland raised bogs also support a distinctive range of animals including a variety of breeding waders and wildfowl and invertebrates.

Lowland raised bogs are a significant and highly modified part of the national peatland resource. The Aichi targets have prompted revision of the UK Biodiversity Action Plans and hence the habitat restoration targets for both raised and blanket bogs have been set to 100,000 hectares (Scottish Government, 2012). The present review of the evidence base and potential for management of raised bogs in terms of their current and future carbon sequestration potential thus offers a unique opportunity to offer policy advice on priority areas for restoration and revised management strategies on Scottish raised bogs. This report aims to

- Summarise the implications, particularly in carbon terms, of a changing climate for lowland raised bogs in Scotland, under different scenarios of healthy active, degraded or restored bog
- Identify the available research and research needs in order to provide ongoing assessment and monitoring of climate change impacts, especially on greenhouse gas (GHG) flux, on lowland raised bogs under different climatic scenarios
- Provide an outline cost benefit analysis of the different scenarios in carbon terms

3. Current and historic raised bog locations

3.1. <u>Classification issues</u>

Although it may seem an easy task, defining which areas of Scotland can be classified as raised bogs has presented some difficulties in this project. We had access to a number of historic datasets to achieve this. From a soils perspective, the main source of information builds on the Memoirs of the Soil Survey of Great Britain (Scotland (Bown and Heslop, 1979). In the first relevant series of these Memoirs, raised bogs are described within the category of confined (=basin) mires (see foot note). It was these Memoirs that formed the basis of the current 1:250,000 database of Scottish Soil classes held at the James Hutton Institute (Soil Survey of Scotland, 1981). In the Soil Survey of Scotland (SSS), basin peats are categorised as deposits of peat > 0.5 m, in line with the general Scottish definition of a peat soil. These data now form part of the GIS-referenced database within the Scottish Soils Knowledge and Information Base (SSKIB) at the James Hutton Institute.

However, almost all documents in circulation that report the aerial extent of raised bog habitat appear to have been based on the original UK Lowland Raised Bog Inventory (LRBI) by Lindsay and Immirzi (1996). The LRBI used a different underlying soil dataset for the assessment of the Scottish raised bogs: the British Geological Survey (BGS) 1 inch maps and 1:50,000 Drift Edition Map Series, augmented for gaps in Scotland with the 1:50,000 Soil Survey of Scotland maps. The BGS datasets classified peat as being deposits over 1 m depth rather than the 0.5 m Scottish definition. Therefore, we attempted to re-map lowland raised bogs in Scotland by querying the Scottish Soils Knowledge and Information Base (SSKIB) 1:250 000 digitised data, as the abovementioned 1:50,000 SSS maps unfortunately do not have full national coverage. Of course, the 1:250,000 dataset will also have a degree of inaccuracy, this time due to the mapping resolution. Very small bogs (< 10 ha), or small basin peat polygons within other soil mosaics, will not be represented individually at this mapping level as such small areas will be incorporated into a larger polygon, representative of the surrounding soil types.

<u>*Foot note: Definition of basin peat in the Soil Survey of Scotland:*</u>
Within the area surveyed, three main types of deposit are distinguished - confined (basin) mire, unconfined (blanket) mire and partly-confined (intermediate) mire. The term mire is used to define all peatland types (bog, moss, moor, fen, etc.) irrespective of their topographical, hydrological, or phytosociological relationships. Confined mires form locally under the influence of ground water (soligenous mires) and are typically located within poorly drained hollows or basins. As deposition continues, the mire surface may ultimately rise above the level of inflow. This fundamental change in hydrological conditions is accompanied by an equally important change in nutrient source from ground water (minerotrophic) to rainwater (ombrotrophic). Typically, a fully developed confined raised mire has a convex or dome-shaped configuration and shows, in section, several quite distinct horizons which reflect the changing environmental conditions from the minerotrophic 'low moor' stage, represented by lake mud and grass and sedge peat, to ombrotrophic raised bog or moss in which the main components are Sphagnum, cotton grass and ericaceous plants.

Another issue is the definition of 'lowland'. The authors of the UK Lowland Raised Bog Inventory were clearly aware of the difficulties in defining 'lowland raised bog' habitats as distinct entities from blanket bog. They mention that 'In Dumfries and Galloway, for example, sites above a mere 30 m above sea level begin to display many of the characteristics of blanket mire, whereas in more easterly Berwickshire (etc) it is possible to identify distinct raised bogs which have formed at altitudes of several hundred metres above sea level'. Thus, their definition of raised bog excluded:

1) areas of domed, basin, peat which is wholly within surrounding blanket peat units, and
2) areas of basin peat that lies beyond or outwith the enclosed land, commonly referred to as uplands (Usher and Thompson, 1988).

We therefore applied a similar approach to filtering the basin peat data in the SSKIB to those within 'lowlands'. In the basin peats in Shetland, for example, though at low altitude, the vegetation community of such basin peats is not significantly different from surrounding blanket bog. Hence, such basin peats were excluded. We also filtered out basin peat areas that occur at altitudes above the limits of agricultural enclosure, (above 250-400 m as per the JNCC definition of uplands; http://jncc.defra.gov.uk/page-1436). Each remaining candidate lowland raised polygon was checked on a digitised 1:50,000 OS map for the nearest enclosed land.

Based on these definitions, the extent of lowland basin peat within the SSKIB totals 38,344 ha (equating to 0.17 % of the Scottish land area, Figure 1). Many lowland basin peats are rather small in area and hence, if plotted realistically on a map (Figure 1a), would not be very visible on a map at national scale. In contrast, most documents on raised bog distribution in Scotland cite 28,000 ha as the former extent, referencing the LRBI. The most current estimates are from the UK Biodiversity Action Plan UKBAP (Jackson and MacLeod, 2000 & 2008) that suggest that only 8,900 ha of these areas are remaining and that only 2,500 ha of these are still active raised bog. **The LRBI (1996) suggested a more limited distribution along primarily the areas of the Grampian, Central Scotland and Southwest areas of Scotland (Figure 2), with a total extent of raised bog habitat in Scotland of 27,884 ha.** We therefore overlaid the LRBI data points onto our lowland basin peat maps to show differences in coverage (yellow, Figure 2). It is clear that a number of raised bogs identified in the LRBI do not correspond to lowland basin peat in the 1:250,000 soils map. As mentioned above, this may be due to the mapping resolution in the SSKIB, however, in a minority of cases, the predominant soil units for such locations are not peat related, notably in Fife. On the other hand, a number of quite sizeable lowland basin peats did not appear in the LRBI (Fig. 2). Some of these are designated as containing raised bog habitat or were known as raised bog deposits from surveys done by the Scottish Wildlife Trust (blue in Fig. 2). However, there is also a substantial number of sites with land cover that led to them being designated as intermediate (blanket) bog, blanket bog, or upland assemblages (red in Fig 2; 29 sites, totalling 3,441 ha) that needs to be subtracted. **Thus, depending on the data sources used, the extent of lowland raised bogs may be between 27,884 and 34,903 ha.**

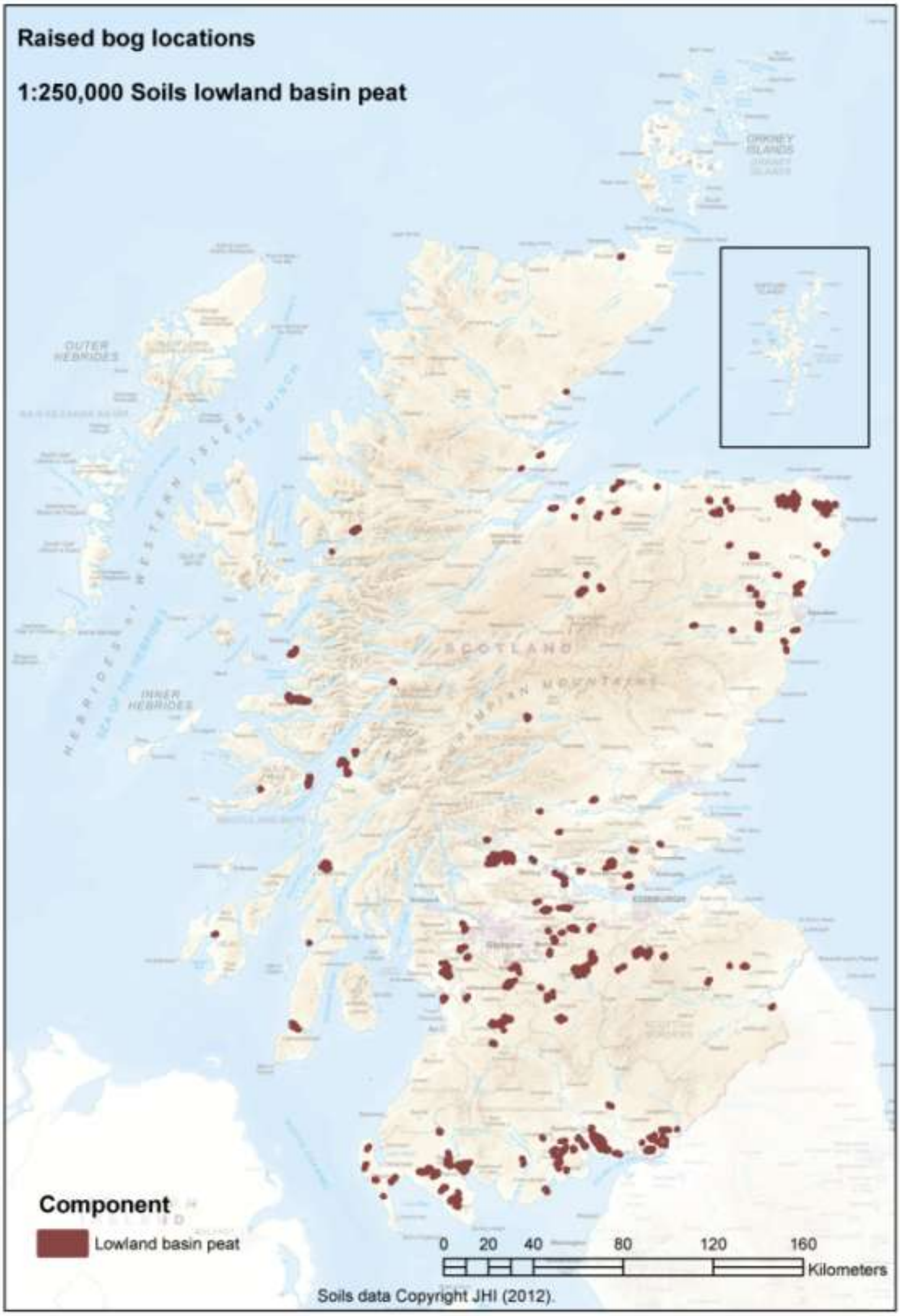


Figure 1. Distribution of lowland basin peat areas in Scotland, as based on the basin peat definition within the SSKIB and further compliance with the definition of lowland raised bog as per Lindsay and Immirzi (1996). Areas have been graphically enhanced with a wide border, for clarity only.

Figure 1a: Realistic depiction of the distribution of lowland basin peat in Scotland, showing the areas depicted in Figure 1 without the wide border. This image shows more clearly the true areal extent.

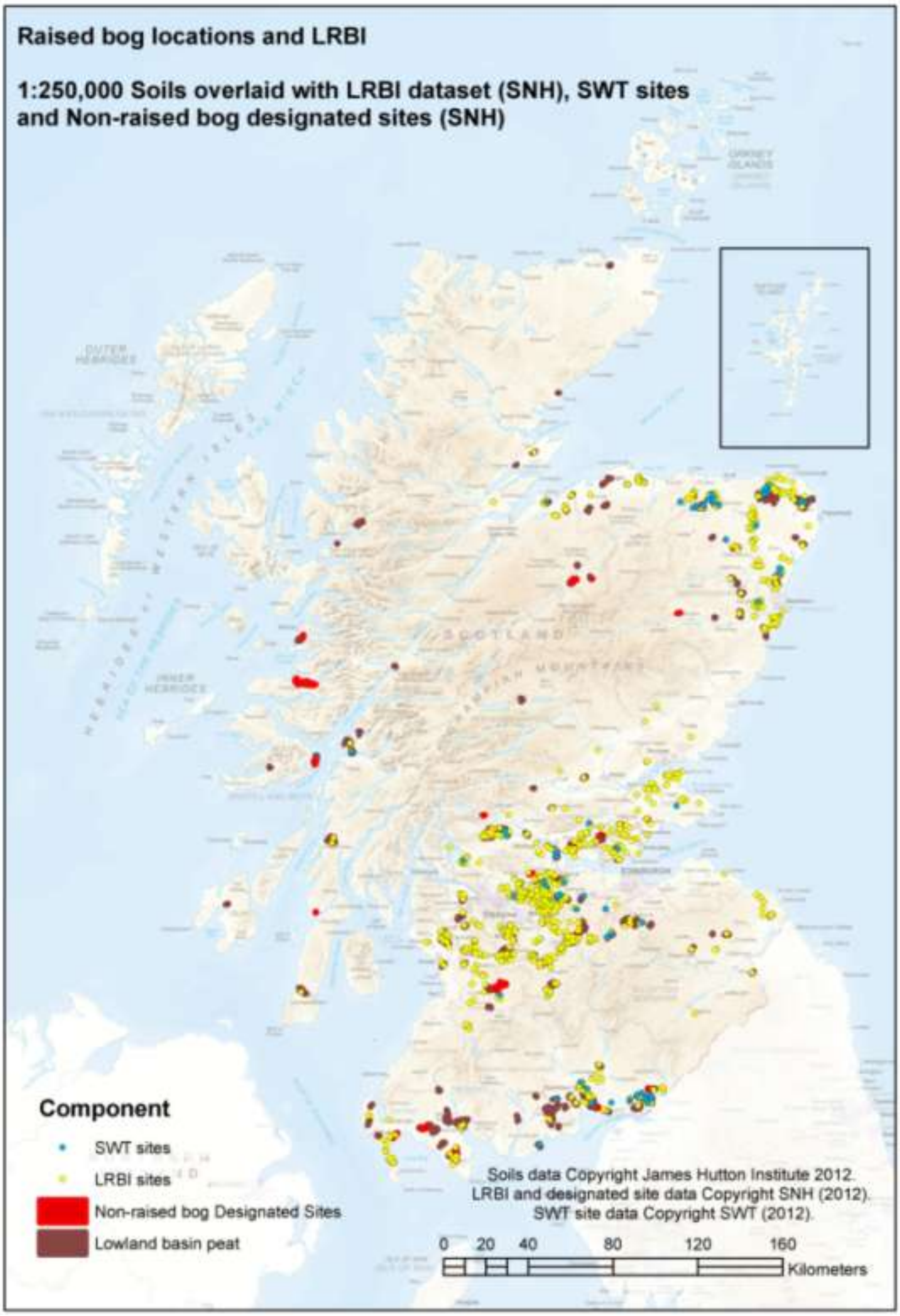


Figure 2: Location of raised bogs in Scotland as identified in the Lowland Raised Bog Inventory (yellow, LRBI, Lindsay & Immirzi, 1996) on lowland basin peat areas as based on the filtered SSKIB. Also shown are lowland raised bog sites surveyed by SWT for condition in 2011-2012 (blue, Matthews, 2012). Finally, the proposed lowland basin peats were filtered for designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities (red, SNH Sitelink data).

3.1.1. Database discrepancies explained through examples

Many of the peatlands in the LRBI are of rather small size, in fact, out of a total 851 records, 593 are less than 20 ha in size. Sites in the LRBI were sometimes being reported multiple times (851 records for 807 lowland raised bogs) due to the peculiar reporting style at the then District level, which caused LRB sites spanning across district boundaries to be reported for each of the relevant districts.

These 593 small site records in total occupy 4056 ha (14% of the LRBI area, Table 1). It is these small sites that cannot not be easily matched to the soils mapping within the SSKIB as the latter has as its smallest basin peat polygon a site of 11 hectares (see foot note).

Table 1. LRBI records split by size categories

Size category	Number of records	Total area	% of total LRBI area
< 20 ha	593	4056 ha	14 %
20-50 ha	118	3590 ha	13 %
50-100 ha	72	4882 ha	17 %
>100 ha	68	15356 ha	55 %

Between the LRBI and a database held by the Scottish Wildlife Trust (SWT) of raised bogs surveyed since 1994, there are a number of discrepancies in terms of the areal extents of those sites that match both databases (Annex Table 1). However, there are also some medium and large SWT sites that are not represented in the LRBI at all. An example is Nether Longford Moss in West Lothian (59 ha) in the SWT database, which does not correspond to an LRBI entry, nor does the location appear to fall into a basin peat category. Others, e.g. the SWT sites Side Moss (53 ha) and Whitley Moss (104 ha), are not found in the LRBI, but fall within the underlying lowland basin peat complex of nearby LRBI sites. Finally, a number of SWT database entries larger than 50 ha are a complex of smaller raised bogs in the LRBI (e.g. Blar nam Fiadh, Branteth Flow and Dunmore Moss).

All of these observations do point out that none of these databases are fully approaching a valid classification of the raised bog resource in Scotland due to limitations in all of the individual data sources. However, it can be said with relative confidence that the larger (>50 ha) raised bog areas can be identified (Figure 3), by selecting the larger LRBI locations (>50 ha, 72% of LRBI area) as well as larger SWT sites and viewing these against the basin peat areas in Figure 1. Table 1 in the Annex lists all of these larger raised bogs. For these larger complexes, the soils data and LRBI entries match up reasonably well, with a few exceptions (Figure 3). For LRBI entries < 50 ha, less than half are located on a basin peat polygon (data not shown).

Foot note: There are only 5 lowland basin peat polygons of < 20 ha in the SSKIB.

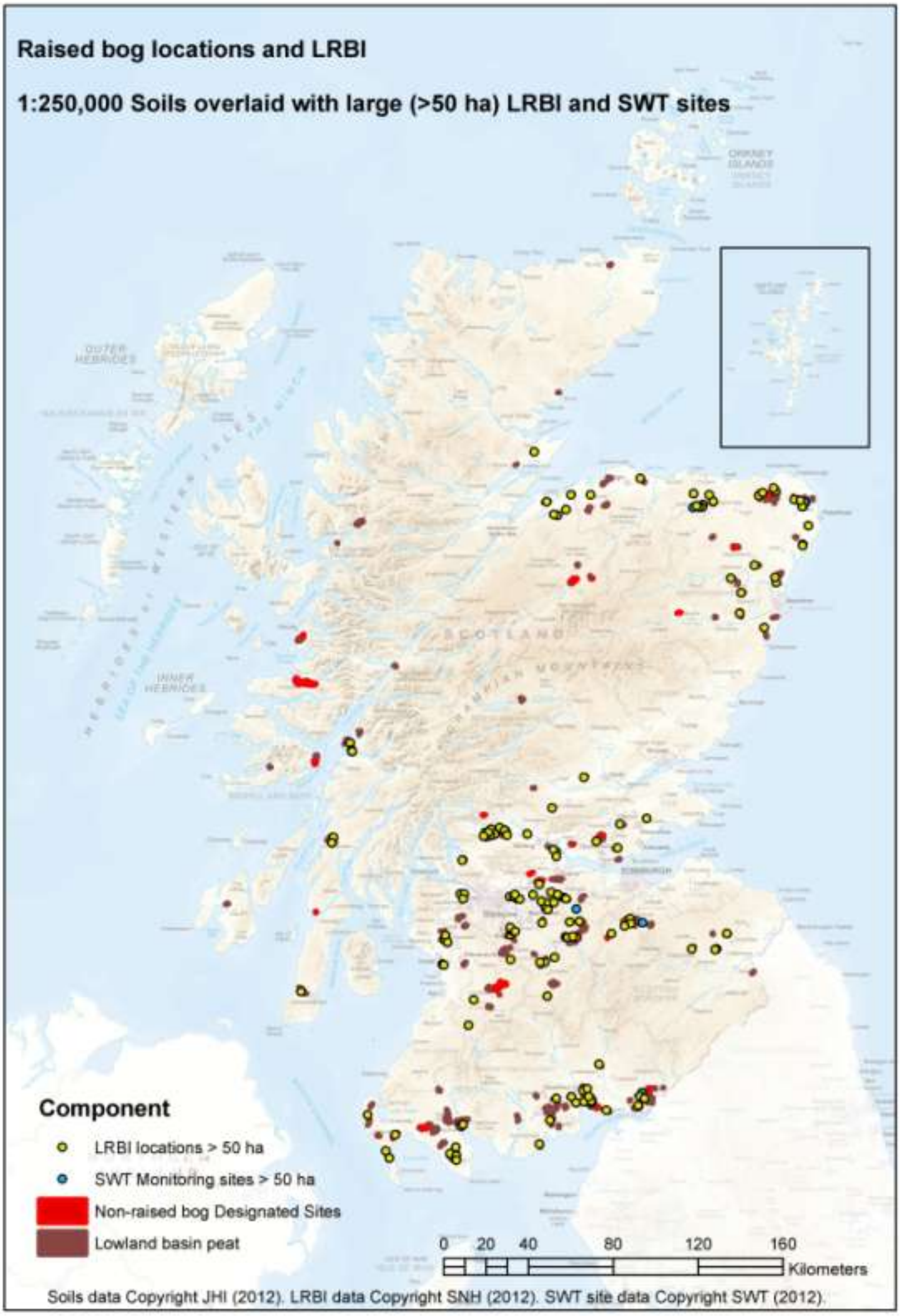


Figure 3. Location of the larger (>50 ha) LRBI raised bog entries and SWT monitoring sites on lowland basin peats as per SSKIB. Red areas are designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities.

3.2. Condition of the raised bog resource

Site condition data that span the whole of the lowland raised bog resource are not available at national scale. The LRBI report (Lindsay and Immirzi, 1996) provides information on the predominant land cover, as well as the areal extent of any part of the site that is still in near-natural or primary degraded raised bog condition. However, the areal extent of other condition classes is not specified. As mentioned earlier, there are gaps in the LRBI with regards to at least a few known lowland raised bogs and hence there it was not possible to achieve full coverage of the lowland basin peat areas. We were able to obtain a number of ArcGIS shapefiles that include NVC categorization and condition classes for the LRBI dataset from SNH. These originated from a combination of aerial photographic interpretation and field surveys collated by The Wildlife Partnership (1999). However, the files obtained from SNH contained overlapping data and incomplete coverage of the LRBI and therefore were not useable for the purpose of this review.

Another source of data would have been the SNH Site Condition Monitoring dataset, but this only applies to sites under designation and hence also underreports at national scale. The only other datasets that provide full coverage across Scotland are the 1988 Land Cover of Scotland mapping and the CEH land Cover 2007. The latter was not yet available at the start of this project. The LCS88 contains polygon data of different land use categories and thus allows for a relatively broad condition classification and was thus used as the predominant data source to map the lowland raised bog resource into condition classes in this report. Data from the LRBI and the SNH Site Condition Monitoring programme were also reported for comparison.

3.2.1. <u>Near-natural or moderately degraded raised bog, supporting active peatland vegetation</u>

The original LRBI defined various subcategories of raised bog, depending on their land cover, using aerial photography. These subcategories (Box 1) described their likely condition and could be considered either ‘active raised bog’ (7110 Annex I habitat) or degraded raised bog still capable of regeneration (7120 Annex I habitat)’, as per the EC Habitats Directive. Within the active raised bog (7110) categories, Lindsay and Immirzi originally included the P1, P2, P3 and S1 categories (Box 1).

The original LRBI suggested that there were only 2,701 ha of P1 near-natural habitat, 3,137 ha of P2 degraded, 1,362 ha of P3 drained raised bog habitat and 2,871 ha of S1 revegetating peat cuttings at the time of the report. This equals a total of 10,071 ha of potentially active raised bog (7110 Annex I habitat). **More recent figures put the P1 category as nearer 2,500 ha (UK BAP, Jackson and MacLeod, 2000 & 2008)**.

Box 1. Subcategories within the LRBI, based on land cover from aerial photography

Primary (P)	Secondary (S)	Archaic (A)
P1 natural or near-natural vegetation	S1 Revegetated or regenerating cutover	A1 Bogs soils in agricultural use
P2 Degraded vegetation (usually burnt or dry)	S2 Commercial or domestic workings	A2 Built development
P3 Drained		
P4 Open-canopy scrub or woodland		
P5 Closed-canopy woodland		

Reproduced in modified form from Lindsay and Immirzi, 1996

The LCS88 (see Annex 2 for the complete listing) allows for 126 land cover types to be identified. Area features are delineated where they are > 10 ha for semi-natural ground vegetation, > 5 ha for built-up land or > 2 ha for woodland. The LCS baseline scale is 1:25,000. Peatland vegetation in the LCS is grouped together for both lowland raised bogs and blanket bog vegetation types. There is no subdistinction for such peatland land cover in specific habitat condition, hence the LCS88 classification system does not allow for a classification into the same categories as in the LRBI. We therefore assumed that, in order to distinguish between near-natural (P1) and other active raised bog categories (P2, P3, and S1), a site still covered with peatland-specific vegetation in the LCS88 that did not have peat cutting features (see below) was likely to be in a relatively active bog state.

Querying the LCS88 database for all categories that should include such presumably active peatland vegetation in good or moderately degraded condition resulted in an area of 7,789 ha (22%) out of the total lowland basin peat resource (Table 2, Figure 4). This figure is similar to the total area considered by Lindsay and Immirzi for the LRBI as active raised peatbogs (Table 2) but excludes categories of revegetated or regenerating peat cuttings as the condition of previously cutover sites cannot be identified from the LCS88 classification. Unfortunately, it is not possible to determine from the LCS88 categories which of these sites are likely within the natural or near-natural category (P1) and which are in categories of low intensity of degradation (P2 or P3). **It is possible that the 'best' category within the LCS88 is that of bog vegetation with no erosion (in bold in Table 2, 5,381 ha).** However, we made the assumption that areas classified within the LCS88 as still covered with peatland or wetland types of vegetation were all at the lower end of any degraded spectrum. In contrast, any site on lowland basin peat that had LCS88 land cover classes that were more aligned with altered states (e.g. wet/dry heath; coarse grassland) were included in a 'degraded vegetation' category that is unlikely to be active, but may be restorable since the underlying soil is still deep lowland basin peat. There were also some minor ordination discrepancies between the SSKIB and LCS88 maps that result in a small degree of skew between the data layers. This causes some mapping errors, such as inclusion of wetlands or water in the case of lochs close by raised bogs, and salt marshes in the case of estuarine bogs. Due to ordination issues, we were also unable to exclude areas that were designated as non-raised bog habitat (see above) from calculations, hence these were highlighted in maps and a best guess made on the basis of their location to subtract such areas from the total.

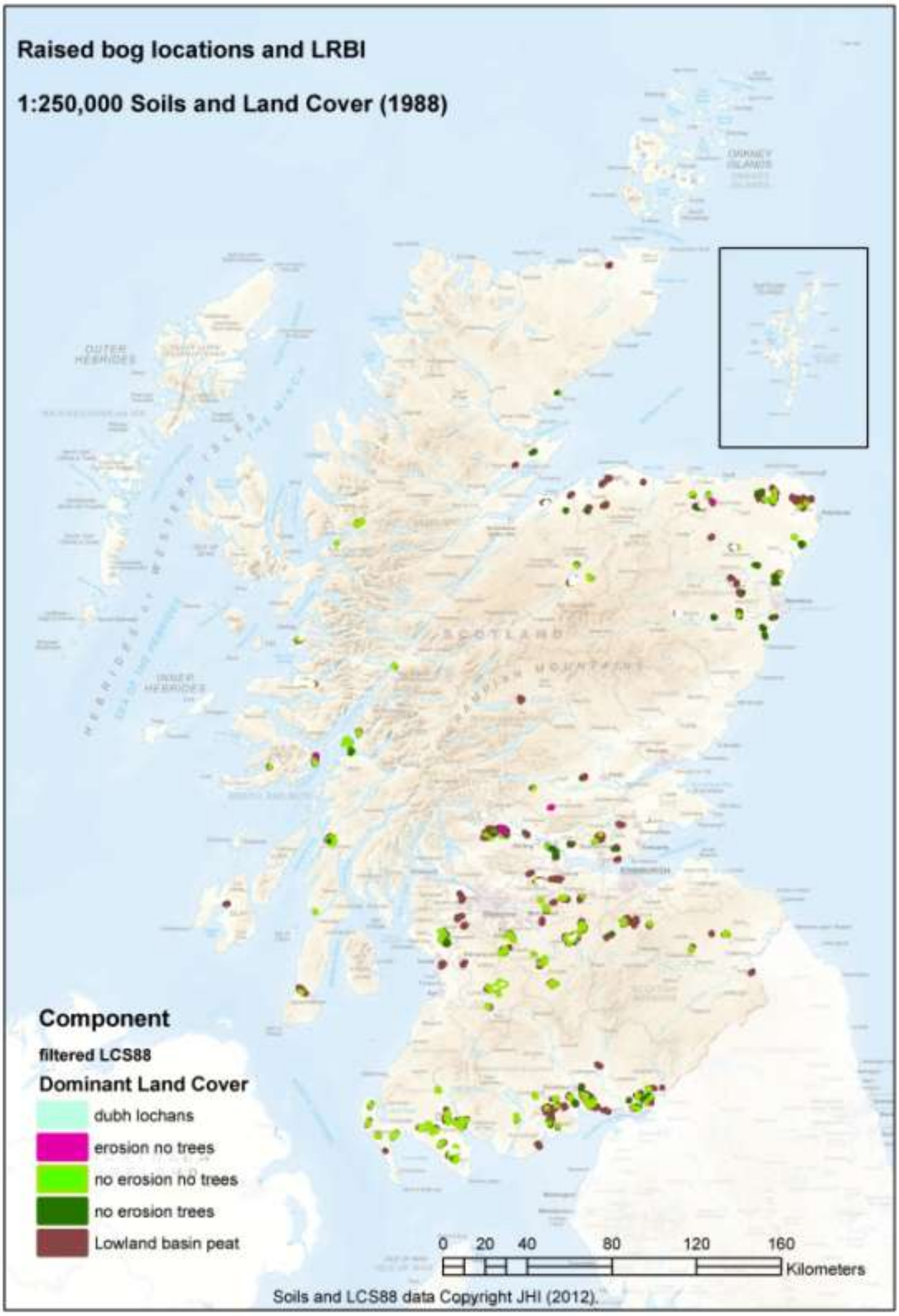


Figure 4. Lowland basin peat areas overlaid with categories of land cover most likely representing functional peatland vegetation (in 1988).Classes within the same category (i.e. all blanket bog vegetation; all water; all wetlands) have been given the same colour coding. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

Table 2. Lowland raised (basin) peatland categorized by land cover as per the LCS 1988. Note some mapping discrepancies due to ordination error (* see main text).

Nearest LCS88 class for raised bog condition	Area (ha)	Total area per category (ha)	Area in LRBI (ha)
Near-natural vegetation			
Not identifiable through LCS88	N/A	N/A	2,701
Active vegetation			
Blanket bog/peatland vegetation - dubh lochans	35	7,789	10,071 (includes 2,701 ha in near-natural category)
Blanket bog and other vegetation – erosion	420		
Blanket bog and other vegetation – no erosion	**5381**		
Blanket bog and other vegetation – no erosion (trees)	1650		
Water*	55		
Wetlands*	215		
Salt marsh*	24		
Dune lands *	5		
Estuary*	4		
Degraded vegetation (no planting)			
Wet heather moor	45	11,911	Partially included in active category above
Dry heather moor	145		
Undifferentiated heather moor	337		
Improved pasture – no trees	6520		
Other improved pasture	433		
Smooth grassland	2138		
Coarse grassland (Nardus/Molinia)	1305		
Undifferentiated smooth grass	263		
Recent ploughing	725		
Scrub or woodland			
Low scrub	27	9,554	9,725
Bracken	26		
Young plantation [1]	1637		
Coniferous plantation	6148		
Recent felling	343		
Undifferentiated broadleaf	626		
Mixed woodland	747		
Arable conversion			
Arable land	2350	2,350	3,081
Cutover			
Industrial peat cuttings	1361	2,855	2,284 active workings
Other peat cuttings	1494		
Lost/archaic land			
Airfields	4	408	950
Bings	84		
Caravan parks	14		
Quarries	109		
Road	3		
Built over	69		
Factory/urban	108		
Golf course	17		
Total	**34,903 incl errors**		**27,884** [2]

[1] Status in 1988. There has been substantial further planting since the LCS88 was completed. [2] Another 2,871 ha of revegetating cuttings are included in active category. [3] Includes 1,771 ha of unknown major land cover.

Within the areas identified as still containing peatland vegetation on basin peat, a number of the larger peat complexes stand out, such as Flanders Moss East and Letham Moss in the Stirling area. Other examples include the Longbridge Muir and Kirkconnell Flow complexes, as well as the Moss of Cree and Ravenstone/Grennan/Auchness Moors in Dumfries and Galloway. While most of these peat bogs are included as raised bogs in the LRBI, there are examples of some of these that aren't, notably in the North East of Scotland as well as some of the West Coast basin peat locations (Annex 3 Fig A1).

We will show a brief example of the complexities of completing these calculations. Longbridge Muir, which is part of the Solway Mosses North SAC complex, is part of a once large lowland raised bog complex (Figure 5). The polygon size for the lowland basin peat in the SSKIB indicates a former extent of 2611 ha (in brown in Fig. 5). In the LRBI, there were three sites that are located on this peat deposit: Longbridge Muir (1,056 ha), Racks Moss (504 ha) and an unnamed area of 41 ha (yellow dots in Fig 5), together these would form an area of only 1,601 ha, leaving 1,010 ha of the original basin peat area unallocated. In the LRBI, the major land cover for all of these three sites was P5, closed canopy woodland, although for Longbridge Muir, an area of 170 ha was given as being in P1 primary natural condition.

In contrast, in the LCS88 data, the area covered by peatland vegetation (green in Figure 5) is 262 ha, a much larger figure than given for the P1 condition in the LRBI. The remaining land cover out of the total 2,611 ha at this lowland basin peat complex is predominantly plantation forestry, with smaller areas of grassland, mixed woodland and arable land. To further complicate matters, due to minor ordination discrepancies, the far eastern end of this basin peat deposit (purple in Figure 5) overlaps slightly with the large Ramsar/SPA/SSSI Upper Solway Flats and Marshes, which is designated for its bird species rather than habitat.

Similar discrepancies apply to almost every lowland basin peat deposit identified in this study.

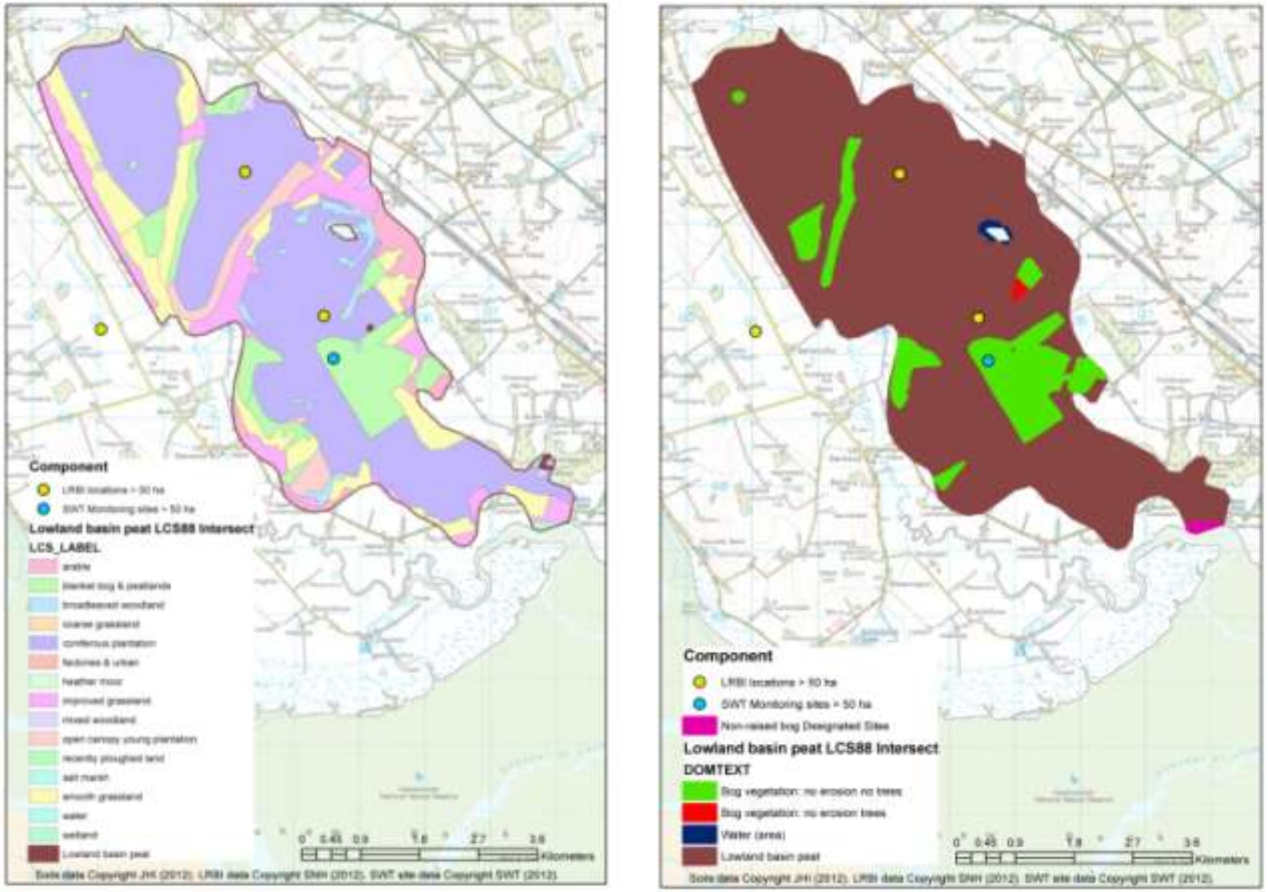


Figure 5. Longbridge Muir site as an example of condition classification using the LCS88. Left image shows the different land cover types found at this site. The right image shows only the blanket bog vegetation types (intact and eroded). Yellow markers indicate the centroid of an LRBI site. The blue point indicates the centroid of the SWT monitoring site Longbridge Muir (Lochar Moss).

3.2.2. Cutover raised bogs

Undoubtedly, many of the original raised bogs have been damaged, in many cases beyond recognition. In the LRBI, cutover sites that showed evidence of revegetation or regeneration were classified as S1, and included as potentially active raised bog habitat (7110, Annex I) under the EC Habitats Directive. Lindsay and Immirzi (1996) estimated a total of 2,871 ha of such raised bog remnants. In addition, another 2,284 ha were identified in the LRBI as being in the S2 (non-active) cutover peatlands category, potentially part of the Annex I 7120 category of degraded raised bog still capable of regeneration.

Peat cutting categories in the Land Cover of Scotland 1988 survey make up a smaller area than in the LRBI. A total of 2,855 ha (8%) of the total lowland basin peat area are cutover, of which just over half are domestic peat cuttings. The latter are likely still sporting relatively high vegetation cover if best practice guidelines have been adhered to by replacing the surface vegetation after cutting (Table 2; Figure 6). The discrepancy between the LRBI and LCS88 derived figures is slightly perplexing as peat cutting is relatively easily identified in aerial photography. Conversely, some of the more highly cutover raised bogs may not be represented in the LRBI as extensive cutting would have resulted in fairly dry vegetation cover or even bare peat (see also grazing land below, Section 3.2.4).

The worst affected areas are in the former Grampian region and in the Central Belt, where practically all LRBI sites show indicators of extensive domestic and/or industrial scale peat extraction. In some cases, the entire site has been cutover, and hence appears not to have warranted inclusion in the LRBI (Annex 3, Fig A2). Such sites may still be interesting to study in detail as they may well warrant inclusion in categories of 'degraded raised bogs still capable of natural regeneration' (7120 Annex I habitat). This habitat category should only include examples "where the hydrology can be repaired and where, with appropriate rehabilitation management, there is a reasonable expectation of re-establishing vegetation with peat-forming capability within 30 years". Such sites can include the following:

- Conifer plantations
- Improved pasture
- Scrub woodland (usually birch Betula spp.)
- Bare peat
- Impoverished vegetation dominated by species including purple moor grass *Molinia caerulea*, hare's-tail cottongrass *Eriophorum vaginatum* and heather *Calluna vulgaris*, and lacking significant cover of bog-mosses *Sphagnum* species.

It is these last two categories that can be representative of harvested sites. Site-specific assessments may need to take place to assess whether such areas could be restored within the definition given in the 7120 Annex I habitat classification. However, planning consents may still be in place for some such areas. In Scotland, there were 72 extraction sites in 2003: 20 were active, 16 had expired, 3 were pending and 33 were awaiting confirmation (Brooks 2003).

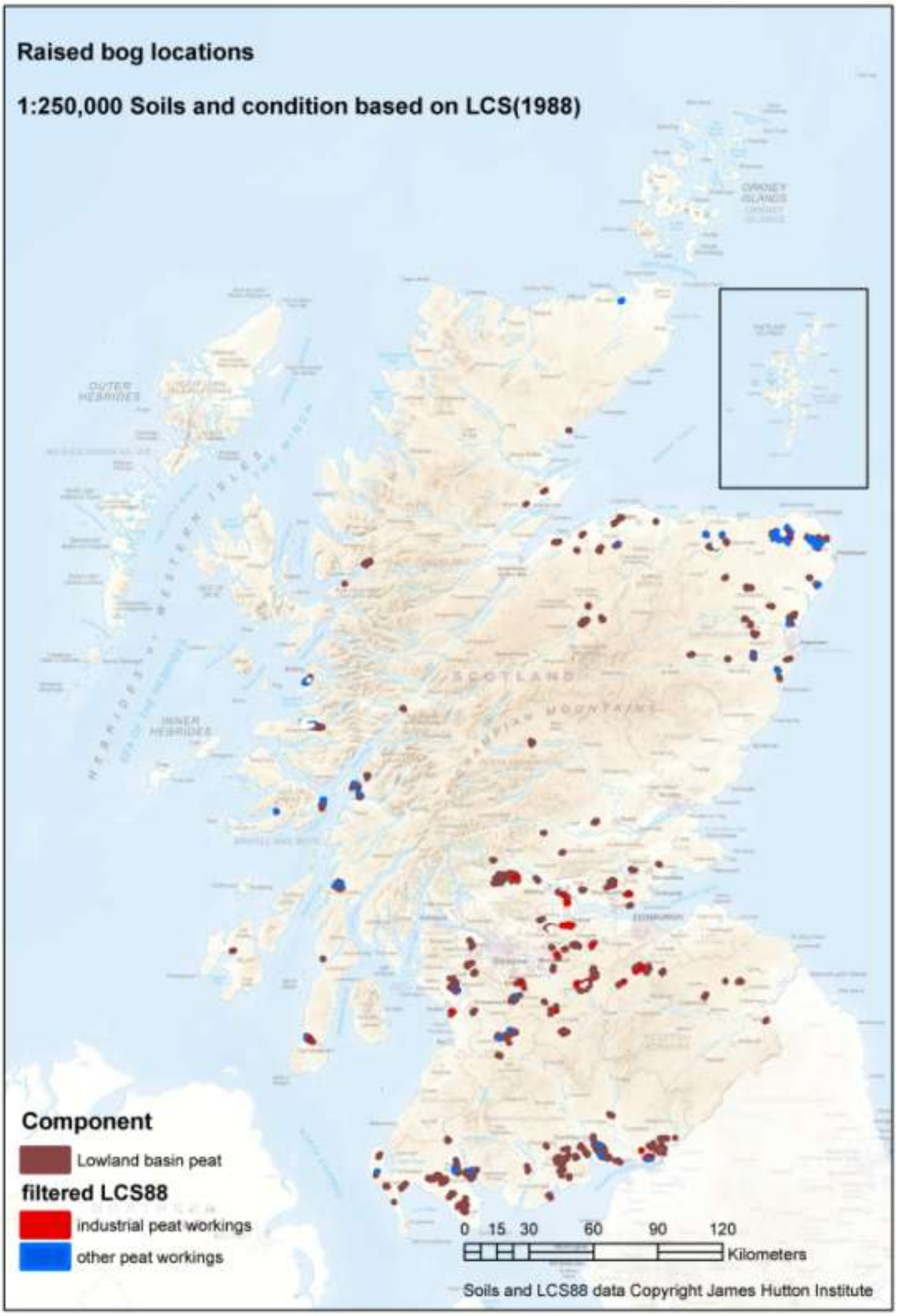


Figure 6. Cutover raised bogs as identified through the LCS88 on lowland basin peat as per SSKIB. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

3.2.3. Raised bogs with vegetation conversion to woodland

Similar to the expansion of forestry plantations on blanket bog, many former raised bogs were targeted for conifer plantations. The LRBI identified 9,549 ha of former raised bog as having 'closed canopy cover' and an additional 176 ha as scrub or open-canopy woodland. In the LCS88, a total of 9,554 ha of former lowland basin peat were identified as having woodland cover (27%, Table 2, Figure 7). Within these areas, many of the larger LRBI sites can be identified, such as West Flanders Moss, the Longbridge Muir complex, as well as many other, smaller raised bogs in the Central Belt and Dumfries and Galloway, however there were a significant number of sites with woodland cover that were not represented in the LRBI (Annex 3, Fig A3). Hence, although the figures for this category of raised bog remnants are similar for both the LRBI and LCS88 based estimates, they are both slightly inaccurate.

Further planting of Forestry Commission land since 1988 may explain the recent ploughing category (725 ha in 1988) in the LCS88, or alternatively establishment of semi-natural mixed or broadleaf woodland may include instances of woodland encroachment (Figure 7a). This can be clearly seen in some of the raised bogs in the Aberdeenshire area, for example, Red Moss of Candyglirach (104 ha; Plate 1), a non-designated peatland of which the LRBI entry (NJ745009) mentions woodland encroachment by seeding from nearby plantations as cause for concern (1994 assessment). It would be beneficial to assess, on a case by case basis, if the woodland or shrub cover should be reduced or removed, if the individual raised bog has high value for carbon or other benefits. There certainly appears to be a lot of scope for further management.

Plate 1. *Red Moss of Candyglirach, Aberdeenshire, a non-designated raised bog remnant with woodland encroachment and historic drainage for domestic peat cutting. Photo: Rebekka Artz.*

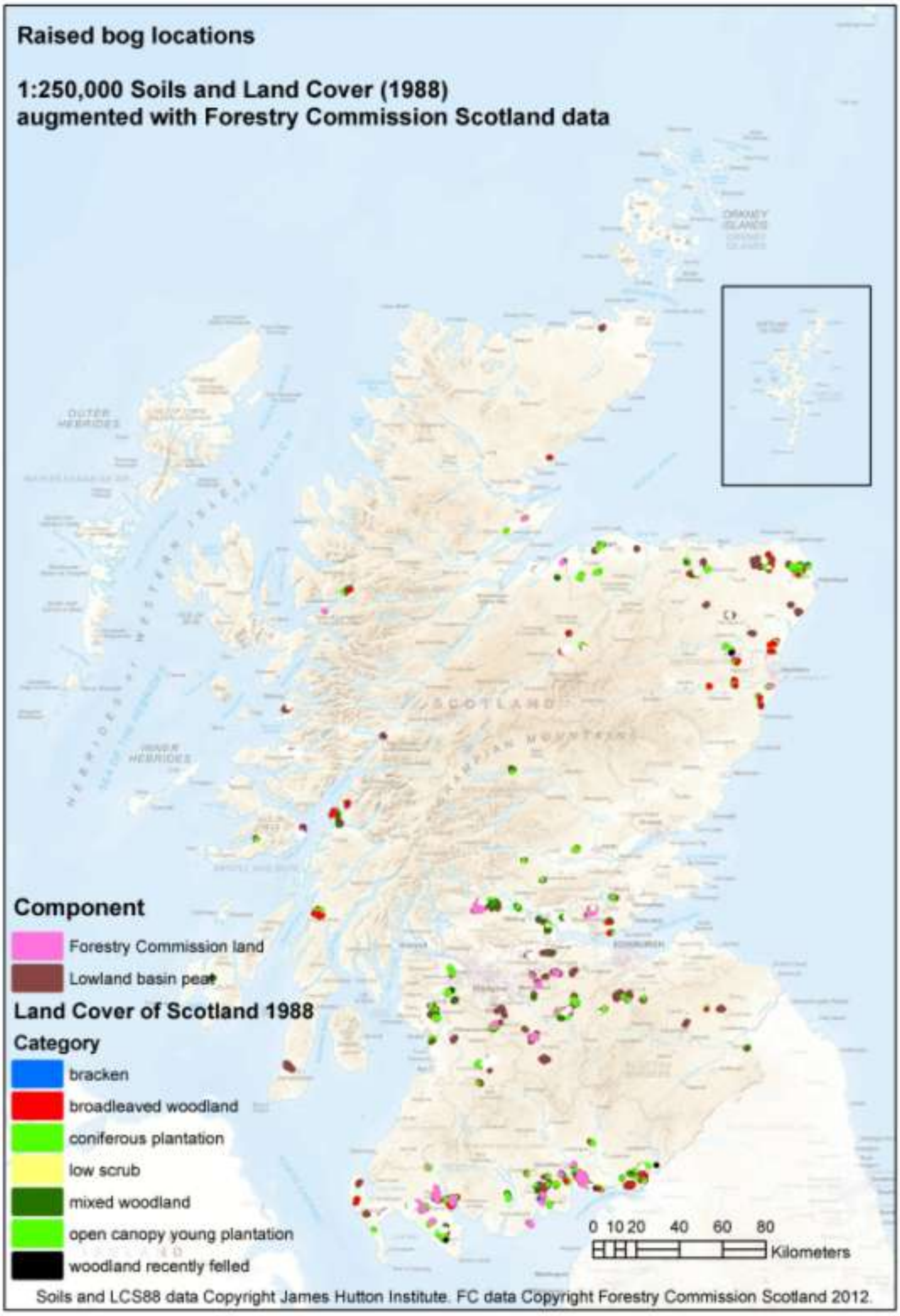


Figure 7. Lowland basin peat areas with woodland expansion or plantation forestry. Forestry Commission land on basin peat in purple (as of November 2011). Coniferous plantations and open canopy young woodland have been given the same colour codings as these were presumably mostly new plantation areas. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

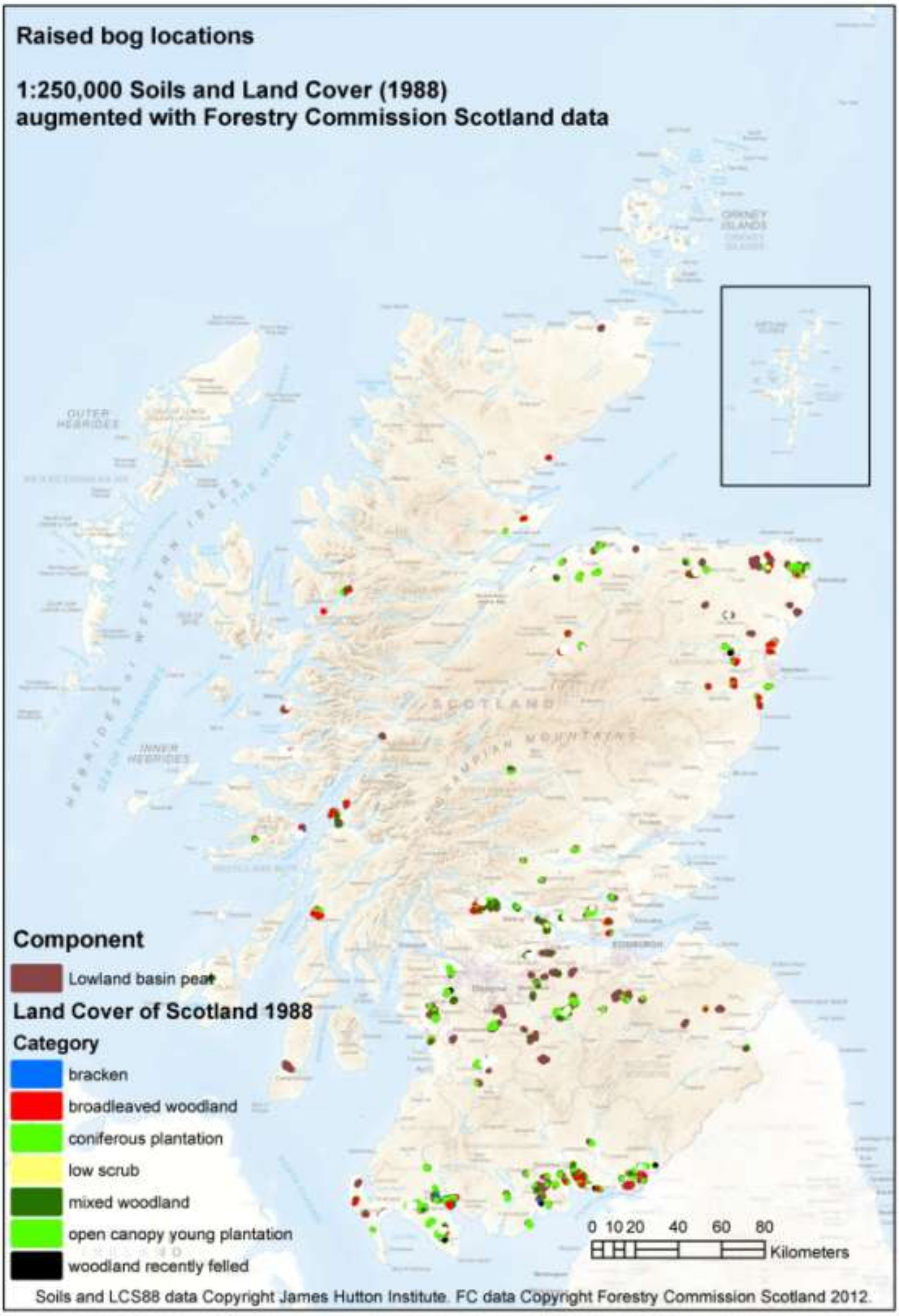


Figure 7a. Land Cover categories connected with woodland without inclusion of Forestry Commission land (Figure 7). This illustrates that most of the FC areas on lowland basin peat in Figure 7 were coniferous plantations in 1988, with smaller areas of broadleaved woodland. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

3.2.4. Raised bogs with vegetation conversion to grazing or arable land

Historic drainage and/or climatic change may have caused some of the original lowland basin peat areas to become drier. Grassland and other pasture categories, as well as drier heathland vegetation, within the LCS88 on lowland basin peat totals 11,911 ha (Table 2, Figure 8). It is possible that, whilst some of this area is probably included in estimates of the P2 and P3 categories within the LRBI, a fair proportion was ignored as the land cover was too different from a raised bog habitat (Annex 3, Fig A4). Similar to observations for other condition classes, many of the areas of lowland basin peat now supporting grassland vegetation are part of larger complexes that still support peatland vegetation of some form in other sections of the former raised bog. Such areas that still support peatland vegetation are also largely affected by grazing and other pressures that have rendered the vegetation in less than a near-natural condition. It may thus be useful to study in detail whether larger scale restoration may be possible to convert grassland areas on former raised bogs back to larger peat forming complexes. Similar to the assessment of extensively cutover bogs, it may be necessary to ascertain whether such areas would fall within the 7120 Annex I category of restorable sites, on a site-by-site basis. Areas that would warrant further interest are in the North-East of Scotland, many of the raised bogs in the Central Belt and in the southern regions of Dumfries and Galloway. Some of the raised bog remnants that are at the upper end of the altitude for consideration as lowland raised bogs show evidence of land cover conversion to heather moor, both as dry and wet heather moor classes. Such examples are found in Figure 8, for example for Fornighty Moss near Nairn, and for the larger peat remnants near Abernethy Forest and Greenlaw Moor/Dogden Moss. It may be beneficial to review any outstanding muirburn practices in such areas.

A fair proportion of former lowland basin peat was converted for agricultural use through drainage practices. The original LRBI categories of 'lost', 'archaic' raised bog comprised such land converted to arable, but also areas with built development. The latter component can be thought to be completely lost, but only comprises a relatively small proportion of the basin peat resource within the LCS88 data (408 ha, <1 %, Table 2). The total area in the LCS88 converted to arable land is 2,350 ha (3.4% of all basin peat, Figure 9, Table 2). Although this figure is lower than the LRBI estimate of 3,081 ha, it is clear that the LRBI did not include some raised bogs converted to arable land (Annex 3, Fig A5). Many arable areas are predominant in the Grampian raised bogs and in the sites near Stirling, and are often contiguous with areas now sporting grassland or woodland vegetation. Many of these areas are adjacent to LRBI sites and in areas of contiguous lowland basin peat and it may thus be beneficial to look at restoration potential of such areas in further detail.

In conclusion, most of the raised bog resource has at least one area within it that has been altered due to drainage, planting, or other factors. It would be beneficial to assess such sites on a case by case basis. For a large number of sites, there will have been further alterations since 1988, for example, at Flanders Moss (Annex 1), there is more extensive forestry planting on East Flanders Moss than the LCS88 indicates.

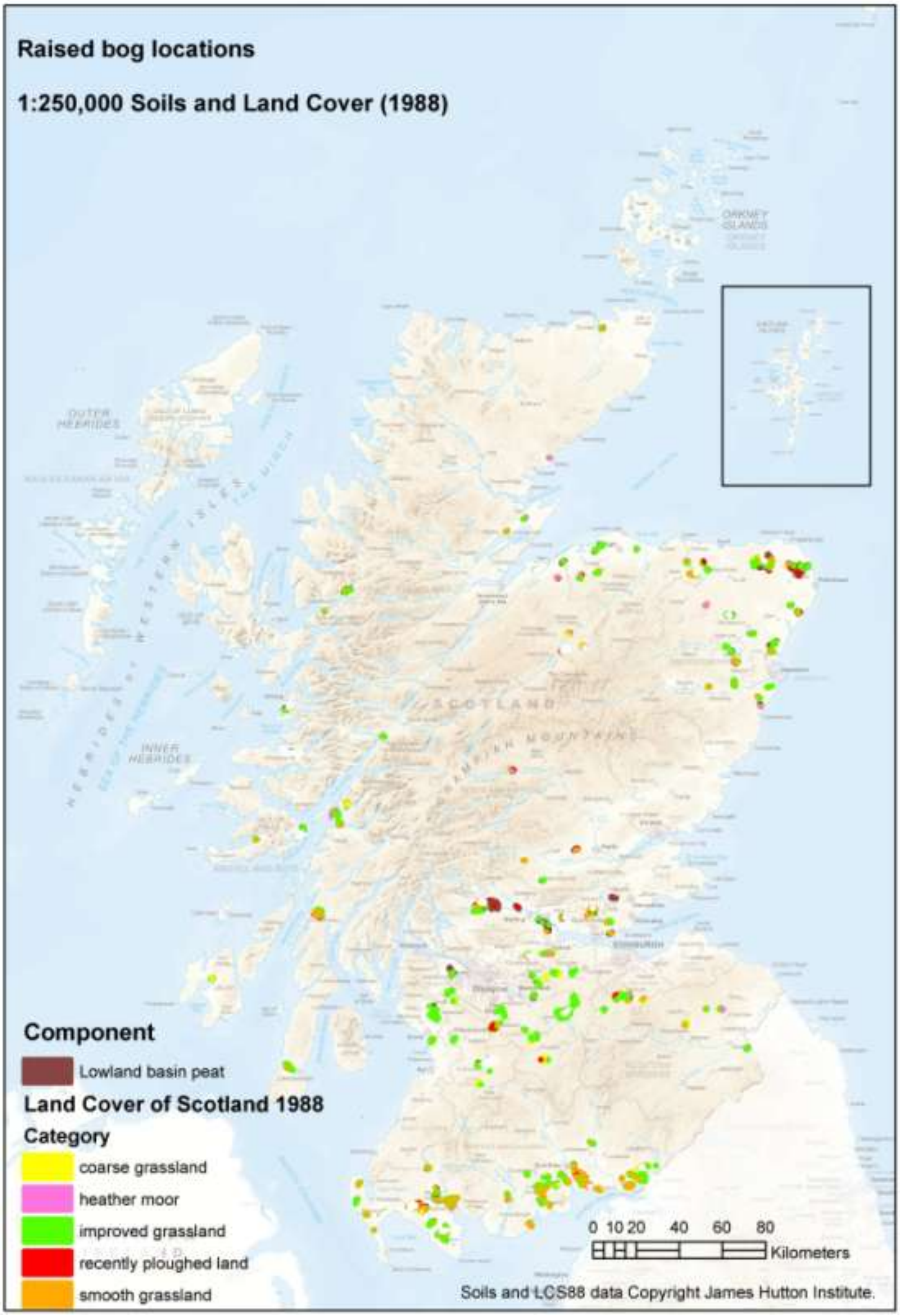


Figure 8. Former lowland basin peat now showing grassland vegetation cover. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

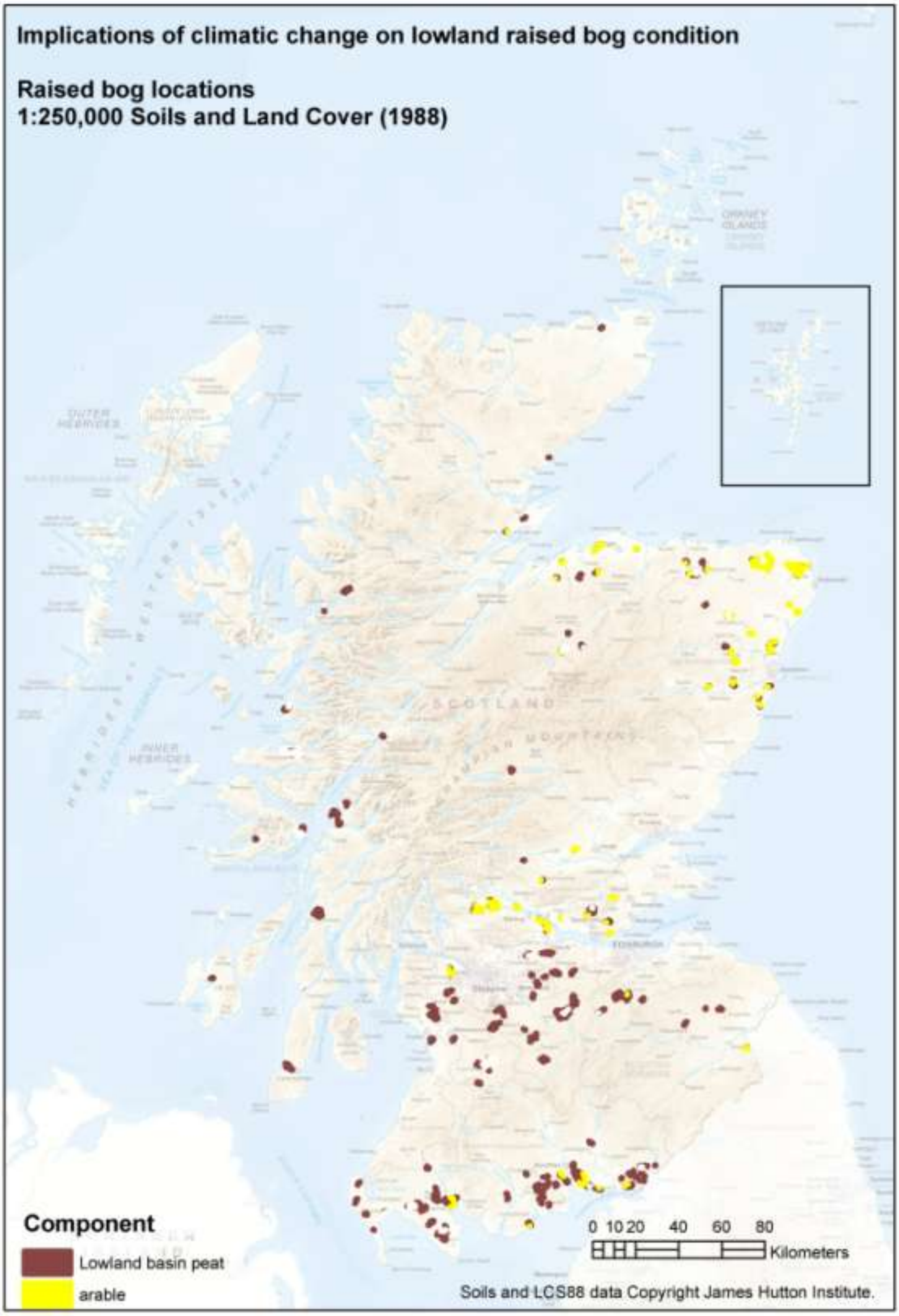


Figure 9. Former lowland basin peat converted to arable land. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

3.2.5. Designations on lowland raised bogs

A substantial number of Scottish raised bogs are under designation (Table 3, Figure 10). The SSSI selection guidelines for raised bogs (JNCC, 1994) stipulate a minimum size of 10 ha (although some existing sites are smaller, Table 3) and a minimum of two criteria from a list including presence of proportion of the original dome and/or lagg fen, low density of peat cuttings, presence of plant species indicative of low disturbance and peat forming capability, an area of appropriate surface patterning and an absence of woodland or low scrub cover (unless a raised bog woodland). In Scotland, there are 41 SACs with raised bog features and 63 SSSIs (some of which component parts of SACs). It is difficult to extract a precise figure for the coverage of designated sites as a percentage of the total areal extent of raised bog, as the data available were only centroid point data for the entire SSSI, which often has designated features for other interests together with raised bog interest. Abernethy Forest SSSI, at 5796 ha (Table 3), is a prime example, as the raised bog feature is only a small part of this large site. Other notable large designated sites are the rather rare estuarine raised bogs, represented in Scotland by the Moine Mhor complex as well as parts of the Kirkconnell Flow.

There are again a number of data set discrepancies with the LRBI. Of the larger LRBI entries (Annex Table 1), 36 are SSSI, however, three of these sites are counted twice due to the original LRBI being structured by district boundaries. Moss of Crombie is mentioned as an intermediate blanket bog in the SNH records rather than a raised bog SSSI as per the LRBI. Similarly, the SSSI record for Gordon Moss is designated for wet woodlands rather than raised bog. Finally, Waukenwae Moss has designated site status as SSSI for raised bog features but this was not mentioned under either of its two entries in the LRBI in 1996. Finally, as in previous sections, there are many smaller raised bog SSSI that are not represented in the LRBI but are on basin peat, for example the Forest of Alyth Mires and the adjacent Dun Moss saddle mire, both north of Blairgowrie.

On the other hand, a number of raised bog SSSI in Figure 10 do not correspond to basin peat, but this may again be due to polygon size of the 1:250,000 soils mapping units. For the smaller SSSI this is an obvious conclusion, but for medium sized SSSI the underlying soil type is often a mosaic of peat patches with shallower organic soils. Glims Moss and Durka Dale SSSI, situated on West Mainland, Orkney, is one of the examples of raised bogs that were not mentioned in the LRBI, and would not be classified as raised bog using our SSKIB-based assessment either, as the underlying soil at this location in the 1:250,000 maps is a peaty gley with peat components. The Dun Moss saddle mire and Forest of Alyth raised bog SSSI are on similar soil mosaics of subalpine soil with peat components or peaty gleys. This observation adds further weight to the likely total area of potentially active raised bogs (P1, P2, P3 and S1) being higher than the suggested 10,071 ha in the LRBI as this misses some designated areas, as well as degraded raised bog on <1 m but >0.5 m depth within already identified raised bog complexes.

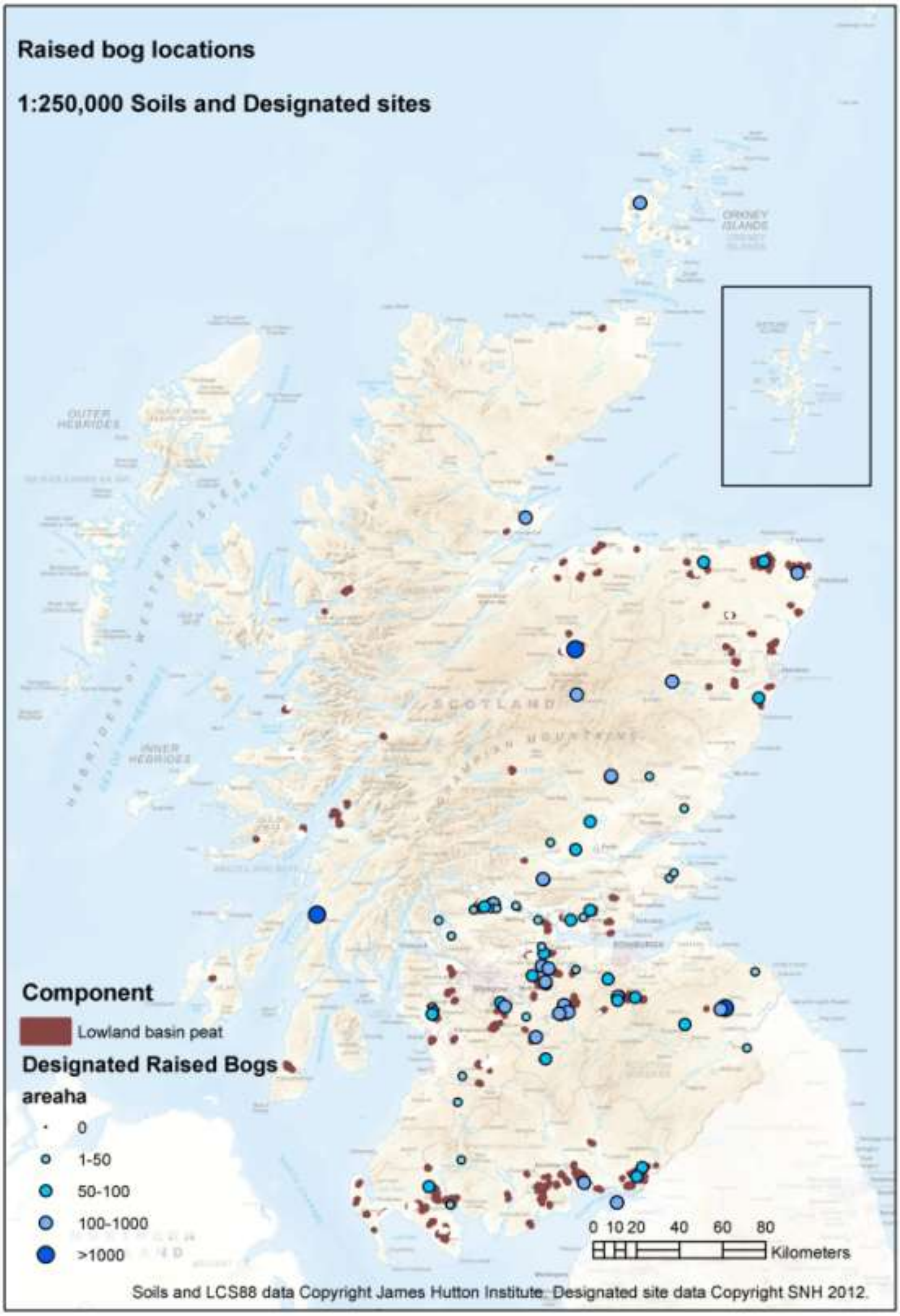


Figure 10. Designated raised bog areas by size. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

Site condition monitoring is carried out by SNH on a rolling 6-year programme under the Common Standards Monitoring guidelines. Results of monitoring for the 2005 Baseline Monitoring revealed that only 30% of the raised bog features were in favourable condition at that time, with another 28% unfavourable recovering, following historical damage such as drainage and afforestation. A large proportion of the features (42%) were in unfavourable condition.

The most widespread negative factor highlighted was tree or shrub encroachment; this was seen on all unfavourable lowland bog sites as well as some of the favourable features. Active drains were also noted as a negative influence. As a direct result of substantial active restoration efforts to improve the condition of unfavourable lowland raised bog features, the proportion of features in favourable condition has risen from 30% in 2005 to 57.7% in March 2010 (SNH, 2010, Figure 11).

While there has clearly been great progress in the last decade through the implementation of large scale or targeted restoration works at a large number of designated sites (Table 3), some further targeted measures will be required. There are still many of the sizeable SSSI that have not yet been included in a larger restoration plan. In some cases, such as for Turclossie Moss, negotiations have already been carried out but have not been completed, or, for Methven Moss, this process has only recently been concluded (see section 3.4).

Unfortunately, 2% of the former raised bog SSSI have recently been classed as destroyed and it may be beneficial to review whether such areas hold value from a carbon perspective. The carbon stocks of lowland basin peat areas, irrespective of designation, are further discussed in section 3.3.

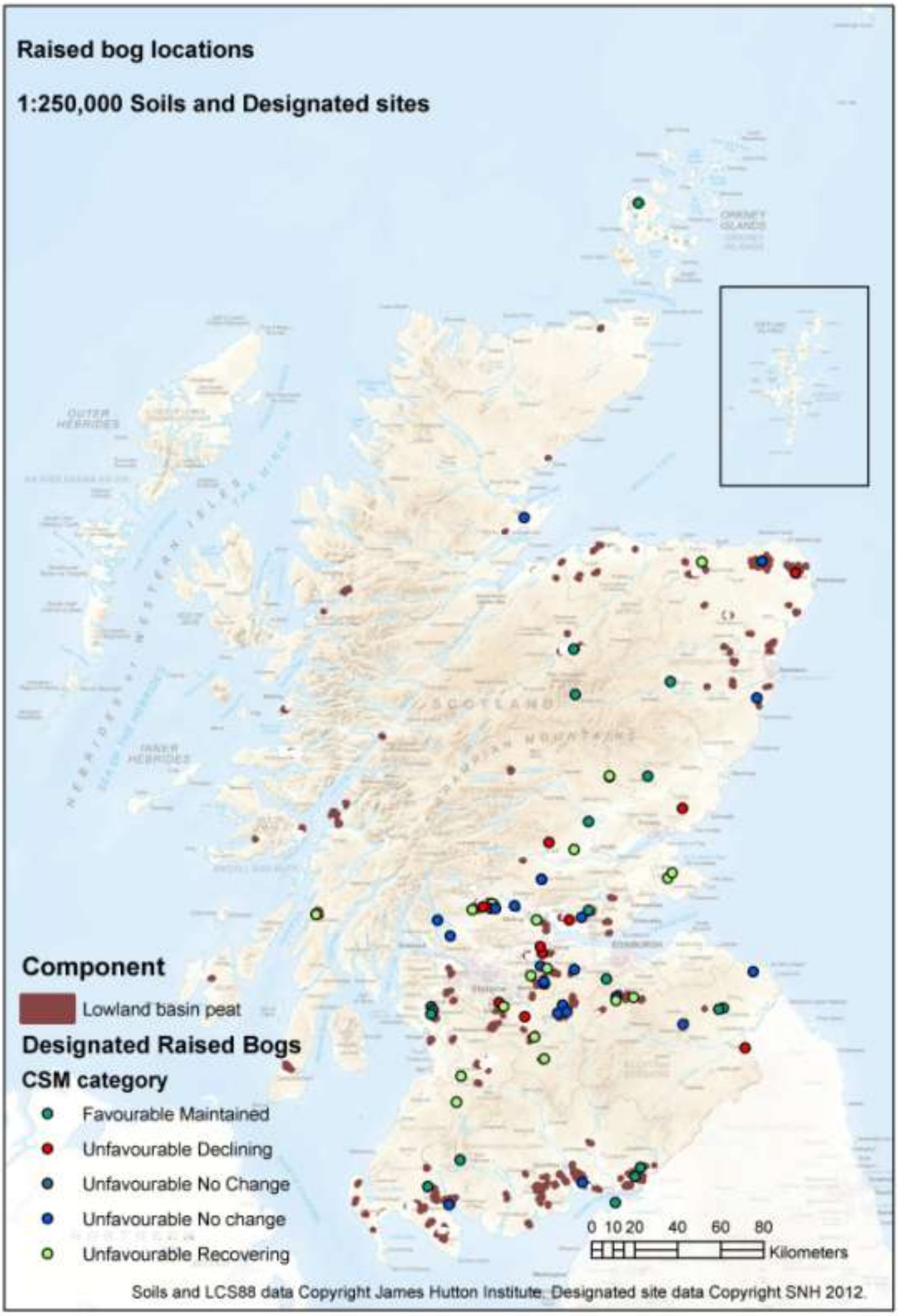


Figure 11. Site condition of areas designated for lowland raised bog features. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

Table 3. Condition of designated areas with raised bog features. *Please note that the area pertains to the whole designated site area, not the raised bog feature within it (see Abernethy Forest SSSI, total 5796 ha, but the raised bog feature area is unknown)*

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other features	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
Abernethy Forest	SSSI	5796 *	Raised bog	Favourable Maintained	NJ010165	Yes				
Auchencorth Moss	SSSI	105	Raised bog	Unfavourable No change	NT208552					
Balerno Common	SSSI	62.7	Raised bog	Favourable Maintained	NT162635	Yes				Raised bog site is called Red Moss in LRBI
Balloch Moss	SSSI	16	Raised bog and Laggs of raised bog	Favourable Maintained	NO353576				Yes	
Bankhead Moss	SSSI	8.4	Raised bog	Unfavourable Recovering	NO447102					
Bankhead Moss, Beith	SAC/SSSI	32.5	Raised bog	Favourable Maintained	NS347506		FM			Lagg fen present
Barlosh Moss	SSSI	36.4	Raised bog	Unfavourable Recovering	NS486185	Yes				Scots pine encroachment, drainage
Black Loch Moss	SAC/SSSI	108	Raised bog	Unfavourable No change	NS855695		UNC	UNC		Drainage, grazing impacts
Blairbeich Bog	SSSI	20.8	Raised bog	Unfavourable No change	NS435835					Drainage, shrub encroachment (rhododendron)
Blantyre Muir	SSSI	51.2	Raised bog	Unfavourable Declining	NS663525					Drainage , shrub encroachment

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other features	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
Blawhorn Moss	SAC/SSSI	109	Raised bog	Unfavourable Recovering	NS886684		UR	UR	Yes	
Bell's Flow	SSSI	71.8	Intermed. bog (raised)	Favourable Maintained	NY320760				Yes	
Braehead Moss	SAC/SSSI	122	Intermed. bog (raised)	Unfavourable No change	NS958515		UR	UNC	Yes	
Cairnleith Moss	SSSI	80.6	Laggs of raised bog	Favourable Maintained	NO079365	Yes				
Cander Moss	SSSI	29.6	Raised bog	Unfavourable Declining	NS782460				Yes	
Carnwath Moss	SSSI	145	Raised bog	Unfavourable No change	NS977482				Yes	
Carsebreck and Rhynd Lochs	SSSI	219	Raised bog	Favourable Maintained	NN861097	Yes				
Carsegowan Moss	SAC/SSSI	49.9	Raised bog	Unfavourable No change	NX429588		UNC	UNC	Yes	
Cassindonald Moss	SSSI	11.5	Raised bog	Unfavourable Recovering	NO467128	Yes			Yes	
Coalburn Moss	SAC/SSSI	224	Raised bog	Unfavourable Recovering	NS827365		FM	UR	Yes	
Cockinhead Moss	SAC/SSSI	48.4	Raised bog	Unfavourable No Change	NS356490		UNC	UD		Drainage, tree and scrub encroachment
Connachan Marsh	SSSI	23.3	Raised bog	Unfavourable Declining	NN895268					
Cranley Moss	SAC/SSSI	101	Raised bog	Unfavourable No change	NS935475		FM	UR	Yes	
Dalmellington Moss	SSSI	27.4	Raised bog	Unfavourable Recovering	NS465064					Drainage, nutrient enrichment
Darnrig Moss	SSSI	77.5	Raised bog	Unfavourable Declining	NS863755				Yes	

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other features	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
Dilty Moss	SSSI	36.5	Raised bog	Unfavourable Declining	NO515427				Yes	
Din Moss and Hoselaw Loch	SSSI	45.7	Raised bog	Unfavourable Declining	NT806315	Yes				Tree encroachment, drainage
Drone Moss	SSSI	23.5	Raised bog	Unfavourable No change	NT844669					Scrub encroachment, potential nutrient enrichment
Greenlaw Moor	SSSI	1175	Raised bog	Favourable Maintained	NT705500	Yes				
Dogden Moss	SAC	156	Raised bog	Favourable Maintained	NT684495		FM	N/A		
Dun Moss and Forest of Alyth Mires, comprising:	SAC	469	Raised bog	Favourable Maintained	NO176573		FM			
1.Dun Moss	SSSI	130	Upland saddle raised mire	Favourable Maintained	NO 169559					
2.Forest of Alyth Mires	SSSI	339	Raised bog	Unfavourable Recovering	NO175577					
Dykeneuk Moss	SAC/SSSI	61.6	Raised bog	Favourable Maintained	NS345472		FM	FR	Yes	
Ellergower Moss	SSSI	34.3	Raised bog	Favourable Maintained	NX482796				Yes	
Flanders Moss, also containing:	SAC/SSSI	859	Raised bog	Unfavourable Recovering	NS630985	Yes	UD	UR	Yes	
2.Killorn Moss	SSSI	34.9	Raised bog	Unfavourable Declining	NS622962					

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other featu res	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
3.Collymoon Moss	SSSI	99	Raised bog	Unfavourable Declining	NS588971					
4.Offerance Moss	SSSI	42.2	Raised bog	Unfavourable Recovering	NS538957				Yes	
5.Shirgarton Moss	SSSI	38.3	Raised bog	Unfavourable No change	NS646962				Yes	
Glims Moss and Durkadale	SSSI	225.3	Raised bog	Favourable Maintained	HY310237	Yes				
North Shotts Moss, incorporating:	SAC	53.3	Raised bog	Favourable Maintained	NS874612		FM	UNC		
Hassockrigg North Shotts Mosses	SSSI	107	Raised bog	Unfavourable No change	NS870622					
Howierig Muir	SSSI	20.5	Raised bog	Unfavourable Declining	NS854786					Tree encroachment
Inchmoan	SSSI	45.7	Raised bog	Unfavourable No change	NS376907				Yes	
Lady Bells Moss	SSSI	59.4	Raised bog	Unfavourable Recovering	NS810651					
Lockshaw Mosses	SSSI	59.2	Raised bog	Unfavourable Declining	NS989909					Drainage, scrub encroachment
Moine Mhor	SAC/SSSI	1150	Estuarine raised bog	Unfavourable Recovering	NR812934	Yes	UR	UR	Yes	
Methven Moss	SAC/SSSI	82.7	Raised bog	Unfavourable Recovering	NO011236		FM	UR		Scrub and tree encroachment
Muir of Dinnet	SAC	415	Raised bog	Favourable Maintained	NJ459015	Yes	N/A	FM		

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other features	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
Ochtertyre Moss	SSSI	35.1	Raised bog	Unfavourable No change	NS735975					Scrub and tree encroachment
Park Hill/Tipperton Mosses	SSSI	90.6	Raised bog	Favourable Maintained	NT078954					Drainage issues
Peeswit Moss	SAC/SSSI	52.9	Raised bog	Unfavourable Recovering	NT288550		UR	UR		Drainage, scrub encroachment
Pitmaduthy Moss	SSSI	119	Raised bog	Unfavourable No change	NH780776	Yes				Tree encroachment, drainage
Raeburn Flow	SSSI	63.9	Raised bog	Favourable Maintained	NY295718		FM	UR	Yes	
Red Moss	SAC/SSSI	75.8	Raised bog	Unfavourable Recovering	NS872265		UR	N/A	Yes	
Red Moss of Netherley	SAC/SSSI	93.8	Raised bog	Unfavourable No change	NO861940		UNC	UNC	Yes	
Reidside Moss	SAC/SSSI	83.8	Raised bog	Unfavourable Recovering	NJ605570	Yes	UR	UR	Yes	
Ring Moss	SSSI	52.4	Raised bog	Favourable Maintained	NX332672					
Rora Moss	SSSI	165	Raised bog	Unfavourable Declining	NK040520					Drainage, scarification
Shelforkie Moss	SAC	111	Raised bog	Unfavourable No change	NN859098		UNC	UNC		
Solway Mosses North, incorporating:	SAC	653	Raised bog	Favourable Maintained	NY203597		FM	UR		
1.Kirkconnell Flow	SAC/SSSI	145	Estuarine raised bog	Unfavourable Recovering	NX 970700		FM	UR	Yes	
2.Longbridge Muir	SSSI	514	Raised bog	Unfavourable No change	NY050690				Yes	

Site Name	Designation	Area (ha)	Feature name	Main Condition Assessment	Central point	Other featu res	SAC Active raised bog condition	SAC Degraded Raised bog Condition	Restoration works?	Comments
Steelend Moss	SSSI	6.94	Raised bog	Unfavourable No change	NT046922	Yes				Drainage, scrub encroachment
Tailend Moss	SSSI	30.2	Raised bog	Unfavourable No change	NT013678					Drainage, fire
Threepwood Moss	SAC/SSSI	53.5	Raised bog	Unfavourable No change	NT518424		UR	UNC	Yes	
Turclossie Moss	SAC/SSSI	62.8	Intermed. bog (raised)	Unfavourable No change	NJ884574		FM	FM		Drainage, peat cutting, tree encroachment
Waukenwae Moss	SAC/SSSI	155	Raised bog	Unfavourable Recovering	NS684507		FR	UR	Yes	
Wester Moss	SSSI	30.4	Raised bog	Unfavourable Recovering	NS837909				Yes	
Whim Bog	SSSI	93.8	Raised bog	Unfavourable Recovering	NT206535					Drainage, active peat cutting, woodland encroachment

FM – Favourable Maintained, FR – Favourable Recovered; UD – Unfavourable Declining; UR – Unfavourable Recovering, UNC – Unfavourable No Change

3.3. Current and likely past carbon stock

On the basis of the definition of lowland peat applied here, the lowland basin peats hold 64 Mt of soil carbon (SSKIB data). This equates to 4 % of the overall carbon stored in peat soil in Scotland, or 2 % of the total soil carbon stock in Scotland (see footnote). In terms of Scottish transport emissions, this equates to 18 years if based on the 2009 emissions (Scottish Government, 2009).

A recent survey by the Scottish Wildlife Trust (Matthews, 2012) extrapolated to a total estimated carbon stock of 59.4 Mt of carbon on the basis of their 62 site depth and carbon content measurements. Both methods of estimating total carbon stock have caveats. The SWT approach assumes that the peat depth across each site is constant. This will lead to over-estimates in some sites, but also underestimates in others, for example where the depth probe did not reach the underlying mineral soil (pers. Comm., Peter Matthews). In addition, the SWT figures did not include archaic peat sites.

Conversely, the SSKIB data suffer from other biases, for example the classification problems associated with the definition of 'lowlands' as described in section 3.1. Secondly, as mentioned previously some smaller lowland raised bogs will have been missed as the lowland basin peat mapping is based on the Scottish Soil mapping exercise which was done at 1:250,000 scale, missing very small bogs (< 10 ha) or small basin peat polygons within other soil mosaics. Hence, neither of the total carbon stock figures that are presented will be entirely accurate, although both figures are, remarkably, within the same order of magnitude.

The original peat carbon calculations presented by Chapman and colleagues (2009) were based on relatively extensive coverage of bulk density, carbon content and depth measurements of the total basin peat resource (Table 4). Many of the peat depth figures originated from the Peat Surveys (Department of Agriculture and Fisheries for Scotland, 1962-68, incorporated into the SSKIB), which primarily aimed to quantify the available peat for extraction. It is likely that many of the visited peat bogs already had an altered, drier, surface vegetation at this point. Lindsay (2010) describes such degraded peatbogs as 'haplotelmic', where the surface bulk density is too high to have originated from a *Sphagnum*-dominated acrotelm. The total coverage of depth measurements across the lowland basin peat resource (67 sites out of all 372 data sources) can be seen in Figure 12. It is striking that the values used for the calculations of the carbon stock content of our study come mostly from different sites than the SWT study of Matthews (2012, Figure 12) and hence there could be scope to further refine carbon stock figures.

Foot note: Chapman et al. (2009) originally calculated that basin peats as a whole (i.e. lowland basin peats as well as upland basin peats) stored 120 Mt of C in total, with the majority of this carbon (77 Mt) stored below 1 metre.

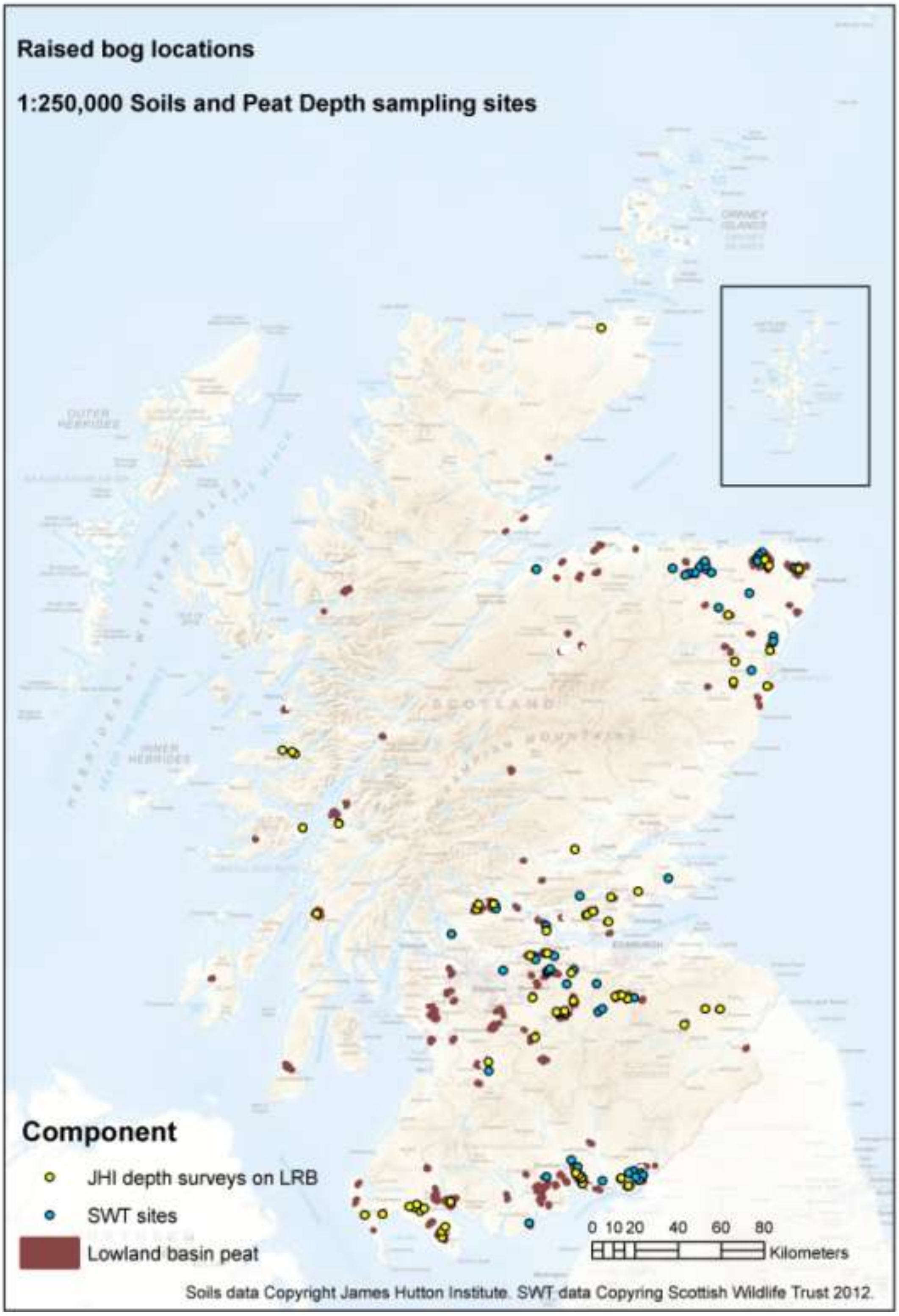


Figure 12. Distribution of lowland basin peat sites with known depth data. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

The bulk density figures used for the calculation of carbon stock originated primarily from the SSKIB, which had comparatively low coverage of the basin peats amongst the total peatland cover (Table 4). Again, as mentioned earlier, it is evident from these figures that the majority of the sampled peatbogs will have no longer supported a functioning acrotelm, as the bulk density for such layers would be expected to be at least an order of magnitude lower (Lindsay, 2010). However, for the purpose of calculating the carbon stock of predominantly haplotelmic peatlands, these figures are entirely adequate.

Finally, the carbon contents (% carbon), which is the final required value for stock calculations, also originate from the SSKIB. The data are more reliable down to 1 m, with 25 and 43 measures of carbon content for the top peat layers at 0-0.3 m and 0.3-1 m, respectively, however, the carbon content of deeper peat > 1m is based on only 2 observations.

Table 4. Mean values and standard errors in the carbon content calculations for raised bogs (from Chapman et al., 2009) Values in brackets are the numbers of observations the value is based on.

Variable	0-0.3 m	0.3 – 1 m	below 1m
Carbon content (%)	51 ± 1.0 (41)	48.6 ± 1.1 (45)	60.8 ± 3.4 (18)
Bulk density (g cm^3)	0.136 ± 0.022 (12)	0.114 ± 0.017 (17)	0.092 ± 0.004 (16)
Depth of basin peat		2.87± 0.09 (360) (> 0.5 m)	
Depth of eroded basin peat		2.72 ± 0.39 (4) (>0.5 m)	
Depth of basin peat in other soil units		2.87 ± 0.34 (8)	

Taking these caveats into account, calculations of the relative distribution of carbon stocks within the basin peat/raised bog resource should be relatively straightforward. From a carbon stock perspective, the highest scoring raised bogs would be those with the highest areal extent and/or greatest depth. Doing this naively, on the basis of the definitions used in this report, results in a distribution map of carbon stock in lowland raised bogs such as Figure 13. Unfortunately, we cannot directly match the lowland raised bogs within the LRBI or SWT databases to our carbon stock data. As mentioned throughout the report, some of the sites in the LRBI inventory are so small they do not appear to correspond to basin peat in the 1:250,000 scale soils map, and conversely, some LRBI sites are close enough together to be on the same basin peat polygon in the 1:250,000 soils maps. Hence, in this report, we will have to consider carbon storage in individual lowland basin peat deposits and match these to lowland raised bog sites in the SWT and LRBI.

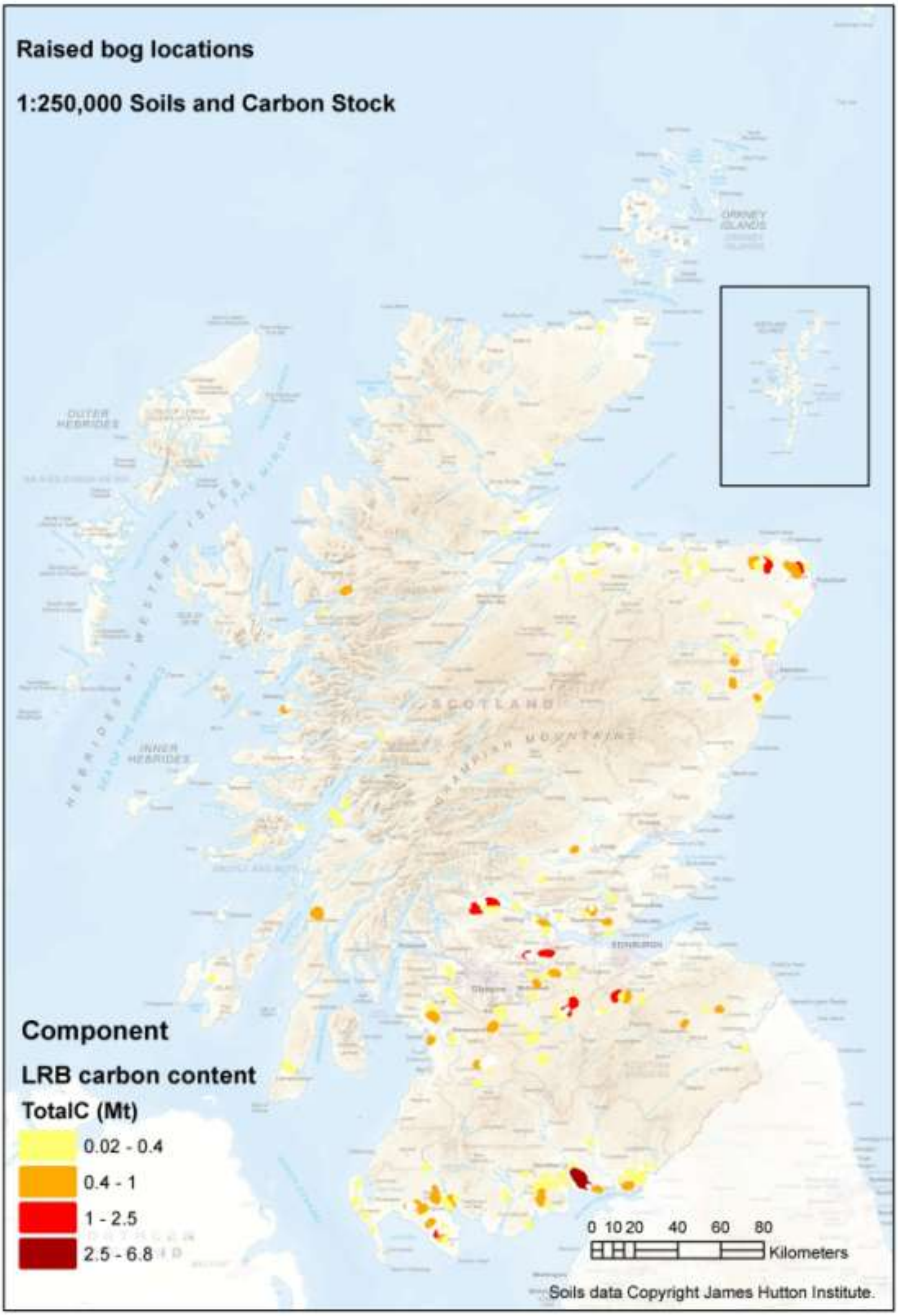


Figure 13. Distribution of carbon stock in the lowland basin peat resource in Scotland. Designated sites that are classified as blanket or intermediate (blanket) bog or for their upland assemblage habitat qualities have been masked.

The deepest and/or largest basin peat deposits, and hence the largest individual carbon stores (Figure 13) appear to coincide with many of the larger LRBI and SWT raised bog sites in the North-East of Scotland, the Central Belt and Dumfries and Galloway (Table 5). The raised bog remnants with the highest carbon stock are the eastern raised bogs of what is now part of the Solway Mosses North complex (rank 1). The western site of Solway Mosses North, Kirkconnell Flow, which appears as a separate basin peat deposit with Drungans Moss, has a much lower carbon stock (not in top 20). The next highest ranking sites from a carbon stock perspective are the two Flanders Moss complexes, followed by Carnwath and Ryeflat Moss. Auchencorth Moss (rank 5) has ongoing measurements of carbon budgets, including measurements of GHG exchange by eddy covariance techniques. Other sites with high total carbon stock value are located predominantly in the central belt and North-East Grampian. Table 5 lists the 20 raised basin peat deposits with the largest carbon stores.

Table 5. Highest ranging peat polygons matching LRBI and SWT sites based on total carbon stock

Rank	Carbon stored in peat deposit (Mt)	Matching LRBI and/or SWT sites (indicated in italics) on the relevant peat soil polygon. Where LRBI/SWT sites do not match well, OS place names are used where possible, in brackets.
1	> 6.8	Racks Moss, Longbridge Muir, unnamed site at NY048727, *Lochar Moss* (part of Solway Mosses North, includes Craigs Moss on OS map, but not Kirkconnell Flow)
2	> 2.5	Flanders Moss West (*also SWT*), Gartrenich Moss, Garchell Moss, Gartur Moss, unnamed site at NS579975, Arnochoile Wood Moss
3	> 2.4	Flanders Moss East (also SWT), unnamed site at NS612998
4	> 2	Carnwath Moss, Ryeflat Moss, site NS94.11
5	> 1.9	Springfield Moss (Auchencorth Moss on OS map)
6	> 1.4	Drumscallan Moss and surrounding remnants (Dowalton Marshes and Drumoddie Moss)
7	> 1.3	St. Fergus Moss
8	> 1.2	No matching site (Fannyside Muir on OS map)
9	> 1.2	No matching site (Stallashaw Moss/ The Kames)
10	> 1.2	Cutover, no matching site (Middlemuir, Greenloch on OS map) Western end of deposit is Turclossie Moss SSSI (Intermediate raised bog)
11	> 1.0	No matching site (Gardrum and Darnrig Moss on OS map)
12	> 1.0	Rora Moss
13	> 0.9	Nutberry Moss and surrounding area
14	> 0.9	Moss of Cree
15	> 0.9	No matching site (Mossmulloch on OS map)
16	> 0.8	No matching site (Mindork and Drumdow Moss on OS map)
17	> 0.8	Mhoine Mhor and Crinan Moss
18	> 0.7	No matching site (NW of St Fergus Moss)
19	> 0.7	No matching site (estuarine bog near Lochcarron)
20	> 0.7	No matching site (Merton Hall Moss, High Moor of Killiemuir, Barvennan Moss on OS map)

3.4. Current likely sequestration rates

Only a very small number of UK peatlands have been studied in enough detail to produce full carbon budgets or long term accumulation rates. Within Scotland, only Auchencorth Moss, near Penicuik (site ranked fifth in terms of C stock in Table 5), has been studied enough to produce a full budget over a number of years. The site is not entirely representative of many raised bogs as it is affected by drainage to some degree but also by historically rather high atmospheric pollution typical of central and southern Scotland. There are further measurements ongoing at Whim Bog nearby (L. Sheppard, CEH Edinburgh, unpublished). Other, less extensive campaigns, have taken place at Newton of Middlemuir (Middlemuir Moss, site ranked twelfth in Table 5), near Strichen, Aberdeenshire, on several areas with varying degrees of revegetation on previously cutover bog (Section 3.7). Finally, there have been at least two years of chamber-based gas exchange measurements from Flanders Moss, (Section 3.7). Hence, a JNCC review of the carbon fluxes and GHG emissions from UK peatlands used modelled fluxes or compound values from a number of European or world-wide studies to produce emissions factors for peatlands in different classes based on the most predominant land use (Worrall et al., 2011). A more recent evaluation of such emissions factors by Artz et al (2012) critically appraised modelled figures and published values to produce best and worst case scenario CO_2 emissions factors for several peatland categories (reproduced in Table 6).

In some cases, the figures that have been collated for Table 6 include all relevant carbon fluxes, i.e. carbon dioxide, methane and nitrous oxide exchange, as well as runoff budgets of fluvial fluxes. However, in many cases, values have been predominantly based on measured carbon dioxide fluxes, with other terms estimated from literature or modelled values. Site-specific variation could well prove to seriously affect the figures published to date. Nitrous oxide emissions are generally negligible from ombrotrophic peatlands, unless nitrogen fertiliser is involved, or possibly in the Central Belt where atmospheric N deposition is still a factor (Drewer et al., 2010). This has not been fully assessed in a UK context (see footnotes of Table 6), but it is worth keeping in mind when assessing the likely net global warming potential (GWP) of a site. In contrast, methane emissions can be relatively high, and this can especially be the case in sites immediately after rewetting. For example, the only currently available Scottish data relate to total C balances for Auchencorth Moss. These net C balances are between 8.3 g C m^2 yr^{-1} (net uptake, Billett et al. 2004) and -72.4 g C m^2 yr^{-1} (net loss, Dinsmore et al. 2010), in climatically different years. These budgets included, for a certain part, estimated methane fluxes or chamber-derived data, converted to GWP using the 100 year time horizon IPCC factor of 25. In their initial estimates of the total C budget, methane fluxes from both terrestrial and aqueous sources made only a small contribution to the total budget (around 10-12 g CO_2e ha^{-1} yr^{-1}, offset against a net CO_2 uptake (NEE) of 343-500 g CO_2e ha^{-1} yr^{-1}) .

In contrast, more recent estimates of methane emissions, using eddy covariance measurements over a year (2010-11), at Auchencorth Moss amounted to a global warming potential of 58 g CO_2e m^{-2} y^{-1}, (0.58 t CO_2e ha^{-1} yr^{-1}), under the 100 year time horizon conversion. (Helfter et al., 2011). Such high emissions may have the capacity to counteract a more significant proportion of the net sequestration of CO_2. It may be possible that such differences are caused by the different measurement techniques as eddy covariance measurements integrate across a section of the landscape, whereas chamber measurements only cover a very small (often <0.2 m^{-2}) area and may thus under-represent methane hotspots. However, another possibility is interannual variability. 2011 in particular was a year with relatively high rainfall and thus methane fluxes may have been higher due to climatic variation in that particular year. However, in most years, it seems that Auchencorth Moss, despite no longer being a natural mire, has a net cooling effect in terms of carbon emissions.

There is still a paucity of methane emissions data from peatlands and hence assessments of the contribution of such emissions to the total, long term, fate of carbon in peatlands are still difficult to make. In addition, the use of the global warming potential conversion factors makes the contribution of methane and nitrous oxide to the total carbon budget dependent on the timescale in question. Over the timescale of bog formation (1000's of years), the effect of such gaseous emissions on the total C budget is even less important, unless it is persistently high enough to override the net fixation through photosynthetic biomass production. A recent collation of methane emissions estimates from drainage ditches in peatlands for the IPPC also suggests that our current emission factor estimates for damaged peatlands may be inaccurate. Scaled to the whole peat surface, fluxes are circa 0.4 g CH_4 m^{-2} yr^{-1} (ca. 0.10 t CO_2e ha^{-1} yr^{-1}) for forestry-drained peats, 2.6 g CH_4 m^{-2} yr^{-1} (ca. 0.65 t CO_2e ha^{-1} yr^{-1}) for peat extraction sites and around 5.4 g CH_4 m^{-2} yr^{-1} (ca. 1.31 t CO_2e ha^{-1} yr^{-1}) for intensive agricultural sites, depending on drainage density (C. Evans, CEH Bangor, unpublished data). In other words, the total emissions from sites affected by drainage for land use conversion may be much higher than current estimates, because CH_4 is generally assumed to be zero in budgets from drained sites, ignoring potential methane production in the sediment within the drains and emissions resulting from lateral transfer of methane from the water table to the drainage network. It is often assumed that the net GWP from near-natural sites includes a large proportion resulting from CH_4 emissions. In restored sites in the first few years, methane fluxes are often expected to temporarily increase above that of natural sites, due to the rewetting. However, if such 'ignored' emissions from methane from drains or N_2O losses before restoration were fully taken into account, the net savings in terms of GWP may be even more substantial. Keeping these caveats in mind, the values in Table 6 nevertheless give a framework of reference for the likely sequestration status of peatlands in various conditions. These data are derived from an attempt to summarise the likely carbon emissions from peatlands in various condition categories, and are based on a compilation of available literature values from modelled and measured carbon balances. The measured carbon balances in Table 6 include both long-term accumulation rates (LARCA, see below) and short-term carbon fluxes (studies similar in nature to the approach for Auchencorth Moss as above).

Table 6. Likely sequestration rates in bogs based on a compilation of literature values of carbon balances from all available sources (long-term measures as well as short term experiments) without taking into account whether complete coverage of all carbon terms was achieved (i.e. no weighting factors were applied to more complete datasets). Data in g C m^{-2} y^{-1} were converted to CO_2 equivalents using the generic multiplier (3.66) to calculate emissions factor equivalents. The 'likely' values column gives the range between the lower and upper quartile (avoiding inclusion of outlier values). Negative values denote net uptake of CO_2.

Land use	Likely current emissions factors (EF) (t CO_2e ha^{-1} yr^{-1})	Literature EF range (t CO_2e ha^{-1} yr^{-1})	References
Near-natural	-2.6 to -0.7	-4.1 to +0.3 (n=14)	(Gorham 1991; Cannell *et al.* 1999; Charman 2002; Turunen 2003; Billett *et al.* 2004; Worrall *et al.* 2009; Dinsmore *et al.* 2010; Evans & Lindsay 2010; Koehler *et al.* 2011; Worrall *et al.* 2011)
Bare peat (eroded or harvested[1])	0.6 to +5.2	0 to +9 (n=12)	(IPCC 2006; Bortoluzzi *et al.* 2006; Waddington *et al.* 2010; Evans & Lindsay 2010; Couwenberg 2011; Couwenberg *et al.* 2011; Worrall *et al.* 2011)
Afforested[2]	+0.1 to +1.7	-2.4 to +4.3 (n=7)	(IPCC 2006; Couwenberg 2011; Lohila *et al.* 2011)
Drained (for forestry or grazing improvements[3])	-0.03 to +6	-0.3 to +20 (n=9)	(Rowson *et al.* 2010; Lohila *et al.* 2011; Couwenberg *et al.* 2011; Couwenberg 2011; Worrall *et al.* 2011)
Cultivated[2]	+8 to +17	+5.5 to +24 (n=5)	(Couwenberg *et al.* 2011; Couwenberg 2011)
Restored[4]	(-6.5 to +0.9) Highly variable, dependent on site history and time since restoration	-8.1 to +2.8 (n=11)	(Byrne *et al.* 2004; Bortoluzzi *et al.* 2006; Yli-Petäys *et al.* 2007; Waddington *et al.* 2010; Samaritani *et al.* 2011)

[1] Recently harvested peatlands or fresh erosion features carry the high-end of the range EF values. [2] Afforested sites may also have N_2O emissions arising from fertilisation at time of planting. No UK values for this in existence. Couwenberg et al (2011) suggest use of boreal or temperate peat soil values which range from 0.1-6.4 kg N_2O-N ha^{-1} yr^{-1}. N_2O emissions from grazed or arable sites may be as high as 6.8 kg N_2O-N ha^{-1} yr^{-1} Negligible on other sites except where high N deposition exists (Drewer *et al.* 2010). [3] Drained sites showing vegetation conversion to grassland vegetation probably at the higher end. [4] Probably insufficient data available at present, all from previously bare sites. Very high net sequestration rates may be observed only temporarily during initially slow decomposition.

It must be stressed, however, that the collated data in Table 6 are still only a first attempt to categorise emissions from peatlands into condition classes and that the approach suffers from a number of potential biases, for example by including both data from long and short term measures of C balances, using a simple conversion of the total budget to CO_2 equivalents irrespective of how large the contribution of methane and/or nitrous oxide to the total budget is, and finally the inclusion of modelled and largely non-UK data.

A way of measuring the carbon sequestration rates of peatlands that avoids the interannual variability that many GHG exchange-based budgets include is via the actual or long term rate of carbon accumulation, ARCA and LORCA, respectively. Unfortunately ARCA is generally derived from peatland growth models and not actively measured. LORCA values are derived from radio isotope measurements (generally ^{14}C) to the basal peat and generated by wiggle matching peat ages through the peat horizons. Lindsay (2009) reviewed a wide range of published data sets. The most relevant of these, from a climatic perspective, have been collated in Table 7. Lindsay pointed out that often reported figures are derivatives of earlier studies and hence not entirely robust. However, a general picture of the ranges of likely C accumulation emerges from these values (Table 7) similar to those derived from CO_2 emissions factors alone. The range of these values (29-70 g C m^{-2} yr^{-1}) would equate to a range of -1.1 to -2.6 t CO_2e ha^{-1} yr-^{1} if the simple conversion factor of 3.66 is applied. In a recent UK wide review, Billett *et al.* (2010) quote historic values of -1.3 to -7.7 t CO_2e ha^{-1} yr^{-1}.

Table 7. Likely sequestration rates in bogs based on long term accumulation rates (negative value means net loss of C). Modified from Lindsay (2009).

Categories	g C m^{-2} yr^{-1}	Reference
semi-natural'		
boreal region bogs	29	Gorham, 1991
southern Finland raised bogs	30-35	Turunen, 2003
UK – reviewed data	30.5	Charman, 2002
UK - estimate	40-70	Cannell, 1999

Based on the emissions factor ranges in Table 6, some cautious calculations are possible. If we assume that the LCS88-based figures for condition categories (Table 2) are within the correct range, calculating current carbon sequestration ranges is quite straightforward. Table 8 gives the estimated range for each category, also taking into account the original figures quoted in the LRBI (Lindsay and Immirzi, 1996) for comparative purposes. We have stated the full range as well as the likely average for each condition class as many of the sites reported for carbon sequestration purposes are often already in a state that shows evidence of degradation (Auchencorth Moss being the prime example). By these calculations, the current net amount of carbon sequestered by the remaining raised bogs in relatively good condition is between 5000 and 20,000 t CO_2e yr^{-1}. It is likely that the lower end of this figure is a better estimate as the majority of the sites that would be classed as belonging into our 'active condition' class in the LCS88 will have been degraded sites to a minor degree. An estimate provided only using the LRBI near-natural condition areas suggests a similar figure (Table 8). Unfortunately, this is more than counteracted by emissions from raised bogs in degraded condition classes. **Our calculations suggest that the total emissions from such degraded sites are around 21,000 to 143,000 t CO_2e yr^{-1} (Table 8). These emissions comprise 0.6-4.6% of the total estimated agricultural and other land-use related emissions of 3.2 Mt of CO_2e yr^{-1} for 2009 (SG, 2009). With raised bog only occupying 0.17% of the land area or less (Section 3.1), this emphasises that raised bog degradation is making a disproportionately large contribution to land-use related emissions.**

Table 8. Likely current carbon emissions in the condition classes occurring for the Scottish raised bog resource. Net cooling categories in green, net global warming categories in red.

Condition class	Area (ha) based on LCS88 (Table 2)	Likely range and average (brackets) of C sequestration (kt $CO_2e\ yr^{-1}$)	Area (ha) based on LRBI (Lindsay and Immirzi, 1996)	Likely range and average (brackets) of C sequestration (kt $CO_2e\ yr^{-1}$)
Natural or near-natural (**or best LCS88 category, which includes areas showing moderate signs of degradation**)	7,789	-5 to -20 (likely at lower end due to categorisation in LCS88)	2,701	-1.9 to -7 (likely at upper end due to categorisation in LRBI)
Degraded vegetation (no planting)	11,911	-0.4 to +71	3,137 (degraded) + 1,362 (drained)+ 2,871 (revegetating harvested)	-0.22 to +44
Afforested (including scrub and woodland encroachment)	9,552	+1 to +16	176 (scrub) + 9,549 (afforested)	+1 to +16.5
Harvested	2,855	+1.7 to +15	2,284 (active workings)	+1 to +11
Cultivated (Arable)	2,350	+19 to +40	3,081	+24 to +52
Total potential for emissions abatement within degraded condition classes (excluding lost, archaic)	**26,668**	**+21 to +143**	**25,161**	**+27 to +125**

Ongoing and planned raised bog restoration programmes have the potential to offset a part of this emission; it is possible that relatively high CO_2 sequestration rates may be achieved during the early years post-restoration, increasing the carbon benefits, although the lack of measurement data makes this conclusion highly uncertain. If full abatement of the total current emissions from degraded peatlands (0.02 to 0.143 Mt of CO_2e yr^{-1}) could be achieved through restoration of the ca. 15-27,000 ha of lowland raised bogs in poor condition, this could be a significant contribution to overall carbon emission savings for Scotland. Unfortunately, it is not possible to easily calculate the likely emissions reduction until 2020 or 2050 if policy measures were introduced to restore such areas of lowland raised bog, as a) the interannual variability in emissions from peatland sites is high (section 3.4.) and b) it is not yet known from a sufficient number of studies how long it takes for emissions to reduce post-restoration. While data from habitat monitoring studies post-restoration suggest that positive impacts can sometimes be seen within a decade, there have as yet been insufficient studies of restoration projects to adequately determine the timeframe of such impacts on the C balance of peatlands. This will be further discussed in Section 3.10. It is however likely that raised bog restoration measures could lead to emissions reductions in the same order of magnitude as enforced 70 mph speed limits and smart metering of energy consumption in small to medium-sized businesses (Audit Scotland, 2011).

3.5. The future of raised bogs in Scotland: Types of, limitations of, and progress on, peatland models in relation to future climate

Future climate could have severe consequences on where peat accumulation may still take place. The latest UKCIP09 predictions for Scotland include a year-round increase in temperature of 2-3 degrees by 2080 (medium emissions scenario), a reduction in summer rainfall by ca. 20%, and an increase in winter precipitation of ca. 15% (http://www.ukcip.org.uk/essentials/uk-impacts/key-findings/). Warmer and drier summers especially could have a severe impact on species composition and hydrological status of a peatland and could also impact on the net carbon emissions from lowland raised bogs as temperature, levels of incident sunlight and water levels are all important drivers of photosynthetic uptake of atmospheric CO_2 as well as decomposition of plant litter and soil carbon. There are currently no published forecasts of carbon balances under climatic change in peatlands that are relevant to the UK situation. Some inferences can be made, however, from models that attempt to model future boundaries for peatland extent using statistical bioclimatic models that describe the relationship between climate and peatland occurrence.

Recent work in the UK has focused on mapping change in climate associated with the extent of blanket peat in relation to precipitation, temperature, potential evapotranspiration and similar derived parameters (Clark *et al.*, 2010; Gallego-Sala *et al.*, 2010). Such models could be used to classify areas of Scotland where peatland regeneration may be favoured by future climate, or indeed, constrained. However, it must be noted that these predictions are exclusively based on data of current peatland distribution, but the models generally do not include an assessment of whether the peatlands in the starting dataset were still actively peat forming, and hence may not adequately represent potential peat forming areas in the future projections. Models present a good starting point to be able to pinpoint those areas where future bog formation (and hence carbon accumulation) may be climate constrained, and such information can be added to other constraints, such as e.g. continued peat harvesting or renewable energy developments. Conversely, areas at highest risk could benefit from targeted management to increase resilience to climate change.

In spite of the popularity of bioclimatic models for mapping species distributions, few studies have examined the extent of actual ecosystems. There are no known bioclimatic models for raised bogs in Great Britain. Models of raised bog distributions in other countries do not provide sufficient information to apply models to the UK (Parviainen & Luto, 2007; Essl et al., 2011). Therefore, for the purpose of this study, three simple bioclimatic models describing the bioclimatic space associated with Lowland Raised Bogs were defined from the same Met Office 5 km gridded climate data sets used for Blanket Bogs by Clark et al. (2010). Due to issues with mapping the extent of lowland raised bogs, the National Lowland Raised Bog Inventory (Lindsay & Immirizi, 1996) was used as this dataset provided a clear definition of confined lowland raised bog extent and indication of condition (i.e. active peat formation). Bogs classified with the best condition on site as either primary or secondary were used. Archaic bogs and other classes were excluded from the analysis as these areas could be ruled out as areas of active peat formation. The whole data set for Great Britain (i.e. Scotland, England and Wales) was used to improve classification of the envelope model. Points were converted to presence or absence across the Ordnance Survey 5km grid. 70% of the grid cells with recorded presence and 70% with recorded absence were selected to calibrate the model, leaving 30% of the data to the check model fits.

A simple threshold model for the three most important climatic indices (based on the same list of 15 variables used by Clark et al., 2010) was defined to cover 90% of raised bog presence. Climatic variables were selected based on those with the lowest number of cells with false prediction of raised bog presence.

Climatic variables chosen were:

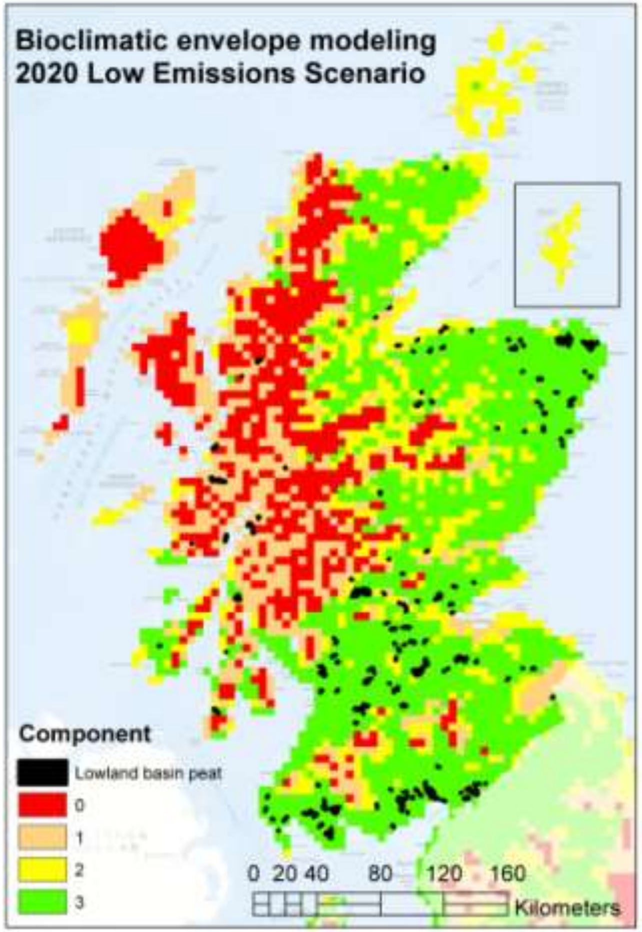
- Mean monthly maximum temperature of the warmest month (Tmax, ^{o}C)
- Thornthwaite-Mather Moisture Index (TMI, -1 to 1), which is an index based on total annual precipitation and total annual potential evaporation
- Annual Accumulated Monthly Water Deficit (AAMWD, mm/yr), which is the accumulated total of monthly deficit between rainfall and potential evaporation calculated using Hargreaves evaporation model.

A full description of these climatic variables and this method can be found in Clark et al. (2010). One key limitation with bioclimatic models is that correlation between presence and climatic index does not necessarily mean the two factors are related. Therefore, future projects of changing space have been made using an ensemble of models rather than selecting one model. The projected bioclimatic spaces for raised bog in Scotland are summarized in Figure 14. Using the same UKCIP02 high and low emission scenarios as in Clark et al. (2010), projections show a shift in bioclimatic space, though not the more significant decline predicted for blanket peat. Raised bogs towards the lower edge of the climate space move outside of the envelope, though many remain within. The climate envelope for raised bogs moves in to upland areas currently dominated by blanket bogs and some of the upland basin peats that did not meet the selection criteria for lowland raised bogs. The likely areas where peatlands are under climatic stress are the Central Belt and coastal West coast bogs (under both low and high emissions scenario's), and the areas forecast to be under the least climatic stress would be the bogs in the Southwest and Grampian raised bogs, at least if a high emissions scenario is avoided (Fig 14). As would be expected, the critical lower thresholds for lowland raised bog presence are either warmer and/or drier than for blanket bogs, however there is some overlap between climatic spaces (see Table 9). More accurate information on the extent of active peat formation in both raised bogs and blanket peat would help to correctly define the bioclimatic envelop between these two habitats. However, it is also noted that overlap between variables is also likely to be due to the simplicity of these single variable threshold models and need to consider multiple factors limiting peat development.

Table 9. Comparison between critical climatic thresholds defining blanket peat and lowland raised bog extent using single climatic indices. Data from this study and Clark et al. (2010: 137).

Climatic Index	95% blanket peat presence (single upper or lower threshold)	90% lowland raised bog presence (upper and lower threshold)
Tmax	<17.9	>16.6 to <20.2 oC
TMI	>0.37	>0.11 to <0.61
AAMWD	>-103.98	>-187.1 to <-26.1 mm/yr

The data in Fig 14 suggest that climatic conditions would be placing a substantial proportion of the lowland raised bog resource under stress. This may manifest itself in a change in the species composition on such areas towards the more stress tolerant species, and/or an alteration in the evapotranspiration balance. Both of these could make a transition to drier sites with higher likelihoods of shrub encroachment and this could negatively affect carbon emissions from such areas. This is a strong incentive to safeguard the carbon stores and reduce emissions where possible in such areas by promoting the best possible habitat condition through policy measures aimed at re-instating or maintaining raised bog hydrology and control of invading shrub or woodland.



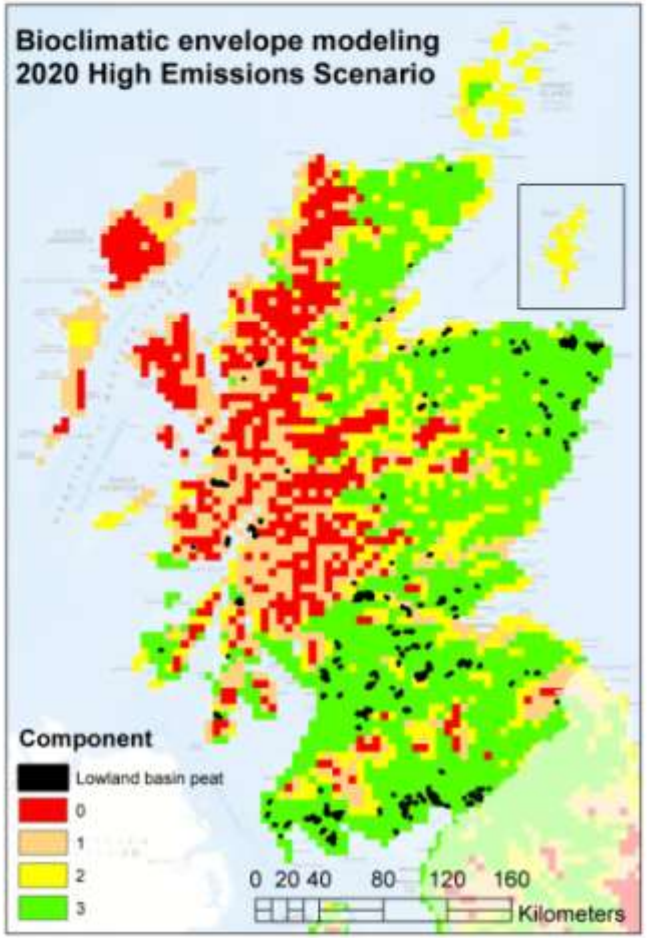


Figure 14. Bioclimatic envelope model (BEM) outputs showing likely areas where raised bogs may be under climatic stress under the high and low emissions UKCIP02 scenarios (Unpublished data).The numbers in the legend refer to the number of BEM predicting climatic stress, i.e. red = high stress, green = agreement on low stress in all three models.

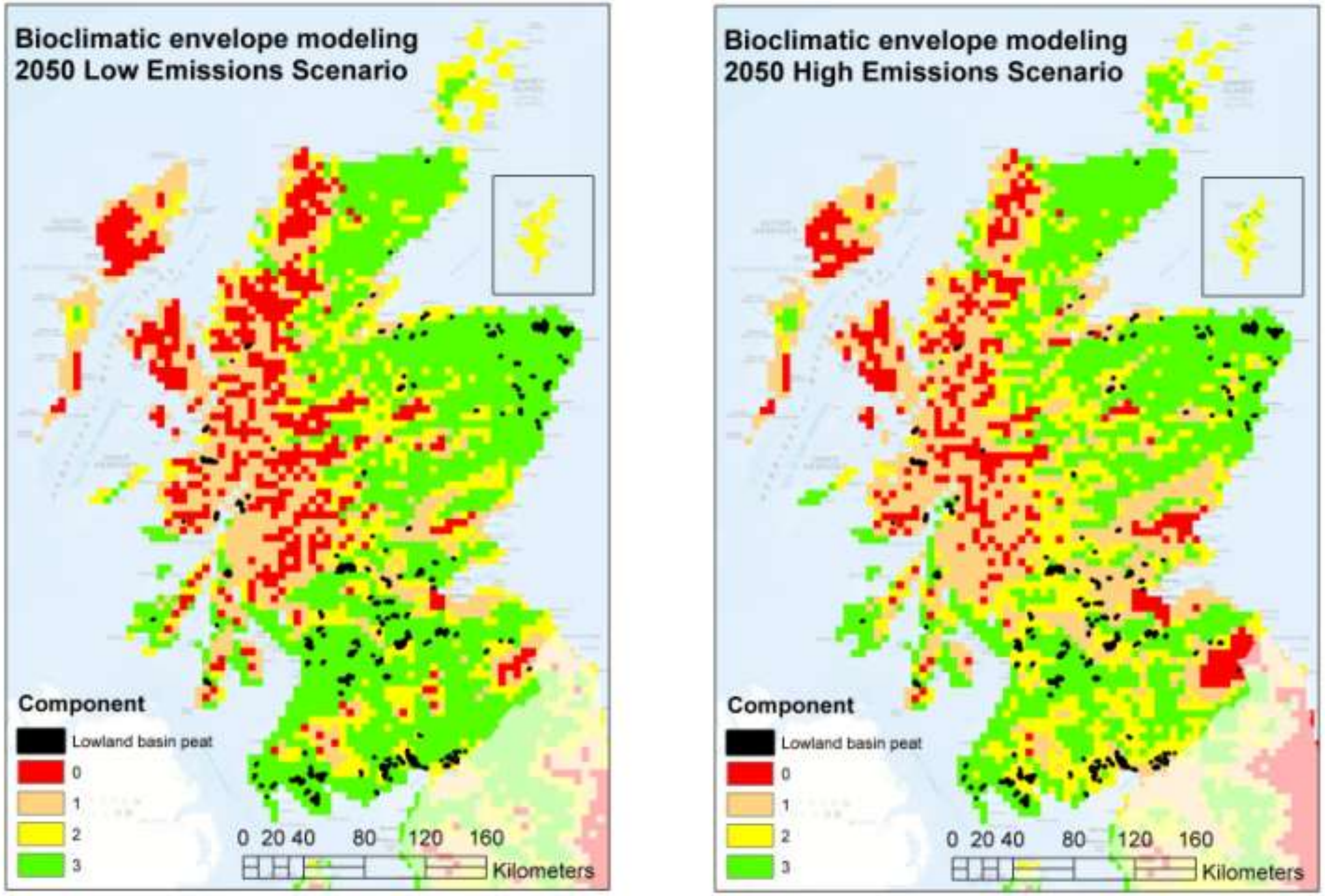


Figure 14 (CONTINUED). Bioclimatic envelope model (BEM) outputs showing likely areas where raised bogs may be under climatic stress under the high and low emissions UKCIP02 scenarios (Unpublished data).The numbers in the legend refer to the number of BEM predicting climatic stress, i.e. red = high stress, green = agreement on low stress in all three models.

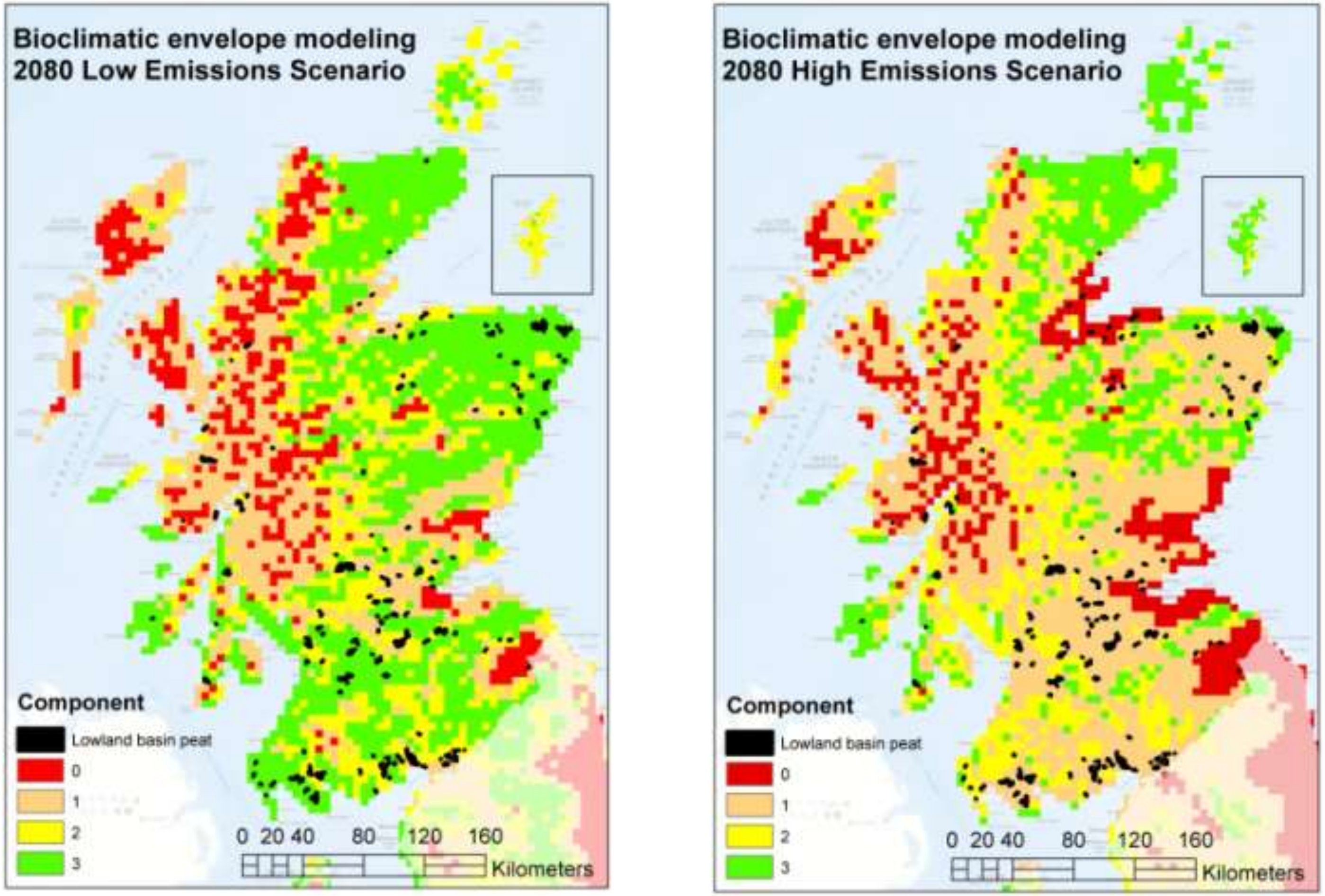


Figure 14. Bioclimatic envelope model (BEM) outputs showing likely areas where raised bogs may be under climatic stress under the high and low emissions UKCIP02 scenarios (Unpublished data).The numbers in the legend refer to the number of BEM predicting climatic stress, i.e. red = high stress, green = agreement on low stress in all three models.

3.5.1. Assessment of *Sphagnum* species niche models and their applicability to lowland ombrotrophic peat in Scotland

A similar approach to bioclimatic envelope models is to model the niche distribution of individual species that may be key indicators of particular habitats. There have been two such studies using species that are common in peatlands, including lowland raised bogs. Smart *et al.* (2010) approached the question of where peatlands may exist under future climatic conditions by building a niche model of ombrotrophic *Sphagnum* spp distribution in response to future climate and pollution scenarios. Their research used cover data for ombrotrophic Sphagna, classed as four morphological classes (red/green + fat/thin), from the Countryside Survey (Smart *et al.*, 2003) and used regression based modelling approaches to determine the niche occupied by these groups. Their findings suggested little change across the UK in the cover of ombrotrophic Sphagna under UKCIP09 scenarios. An earlier attempt at modelling species niche distribution by Berry et al. (2003) using UKCIP98 scenarios also suggested little cover change for Sphagna, but suggested that they species may be moving away from it's most eastern and southern occurrences, whilst expanding into other niches. Other raised bog species, however, such as *Eriophorum vaginatum* and *Andromeda polifolia*, appeared to be at risk of losing climate space.

For this present report, the limitations and strengths of two niche modelling approaches in terms of their ability to assess vulnerability of Scottish lowland ombrotrophic peats to climate change and other drivers were assessed. Both approaches are based on statistical modelling of the correlation between present-day or recent historical patterns of *Sphagnum* species distribution and explanatory variables that are hypothesised to be influential in causing the observed pattern and responsive to drivers such as climate change, pollution and management that can change abiotic conditions to be more or less favourable for continued persistence. Note that there are many caveats with this approach.

1. Niche models are trained on spatial relationships. This means that temporal forecasts of change cannot be directly equated with these spatial patterns. Model forecasts of change in time are interpreted as changes in habitat suitability rather than expectations of actual change in species cover or presence over that time period.
2. The models assume that the observed coupling between species and environment in the data used to build the models represents an equilibrium. Even though there may be considerable noise in the data, the assumption is that our forecasts of habitat suitability are not undermined by the likelihood that present-day *Sphagnum* cover may reflect historical rather than present-day conditions.
3. Both caveats have the potential to be addressed by dynamic models of peat growth in response to various factors. Such models are hard to build, parameterise and apply over large areas but contribute key understanding of peatland dynamics. While static niche models cannot simulate rates of change, feedbacks and lags their strength is that they can be applied with minimal data across many areas. The best approach is therefore to combine

the understanding from both. Therefore niche model forecasts of change in habitat suitability should be interpreted alongside other information gained from ecological knowledge and dynamic modelling. We return to the desirable practice of using multiple models to build confidence and consensus below.

4. Regarding lowland raised bog development, since the niche models are trained on current presence they do not predict potential for ombrogenous nuclei to form from terrestrialisation of water bodies, i.e. raised bog formation *de novo* cannot be predicted.

Modelling change in cover of ombrotrophic *Sphagnum* species

Using two statistical techniques (Generalised Additive Mixed Models and Generalised Linear Mixed Models) Smart et al (2010a) built a niche model of ombrotrophic *Sphagnum* cover across British peatlands. The model was trained on occurrences of ‘red’ *Sphagnum* species in a stratified random sample of 623 200m^2 plots located in 172 1 km squares in Britain. The minimum adequate models for both techniques contained the explanatory variables in Table 10. The GAMM model also included various terms for covarying out spatial autocorrelation effects (Smart et al 2010a). The final models were then used to produce a first approximation of predicted changes in cover under UKCIP02 and UKCP09 scenarios, with and without the interacting influence of projected atmospheric deposition of sulphur and nitrogen.

Table 10. Explanatory variables for the final minimum adequate models that predict Sphagnum cover across British peatlands.

Explanatory variable	GAMM	GLMM
Cover-weighted vascular plant canopy height	Yes	Yes
Substrate C:N ratio	No	Yes
Mean monthly precipitation	Yes	Yes
Soil C content	Yes	No
Mean max July temperature	Yes	No

The applicability of the *Sphagnum* cover model to lowland peats in Scotland should be higher the greater the extent to which the training data represent the target habitat. While the majority of the Countryside Survey plots used to build the models were in Scotland, only 16% (28) of the training quadrats out of a total of 172 containing red *Sphagnum*, were located in 1km squares <135m in maximum altitude (Figure 15). Basin peats and lowland raised bog were therefore covered but this proportion as well as the total dataset of presences remains low. While the training data was based on a random stratified sample of Scotland, sparse coverage of lowland peatlands, because of their scarcity, risks predicting lower favourability of lowland situations and increasing the uncertainty around prediction in this region of ecological and climate space.

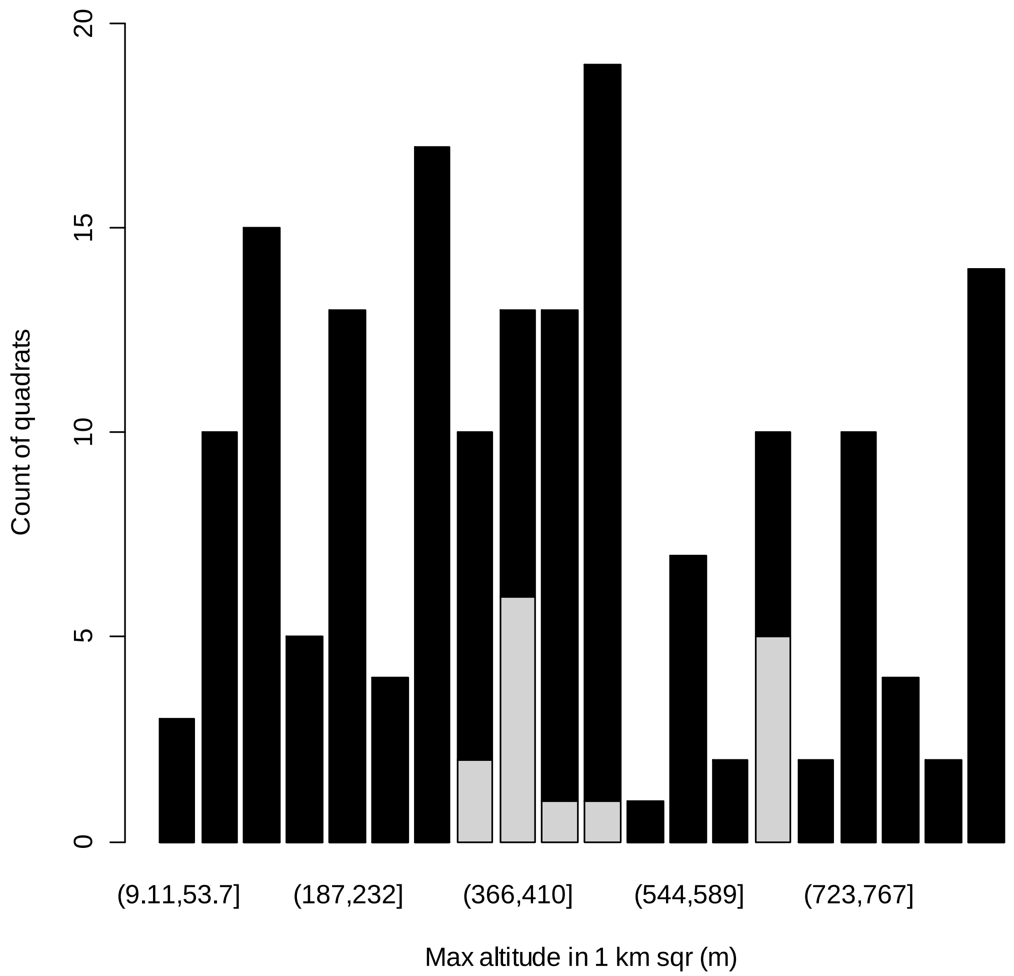


Figure 15: Counts of quadrats by maximum altitude of the 1 km Countryside Survey squares within which the red Sphagnum species group was recorded in 1998. Black indicates quadrats in Scotland; grey is England & Wales.

By overlaying the forecast maps of *Sphagnum* change from Smart *et al* (2010a) with the LRB and basin peat layer we can quantify the distribution of the size and directions of expected changes for these points. **These results should be treated with caution since the models were trained on occurrence of peat bog defined by the satellite LCM2000 (Figure 16) rather than the LCS88 as in this report. When overlaid, 78% of lowland raised bog inventory (LRBI) point locations and 51% of the lowland basin peat (SSKIB) point locations did not coincide with LCM2000 peatland. Of those points that did coincide the majority of expected changes were negative, less than 0.1% in cover and highly uncertain (Smart *et al* 2010).** More comprehensive coverage of both peatland types but especially lowland raised bogs would be needed if to try and produce reliable niche models.

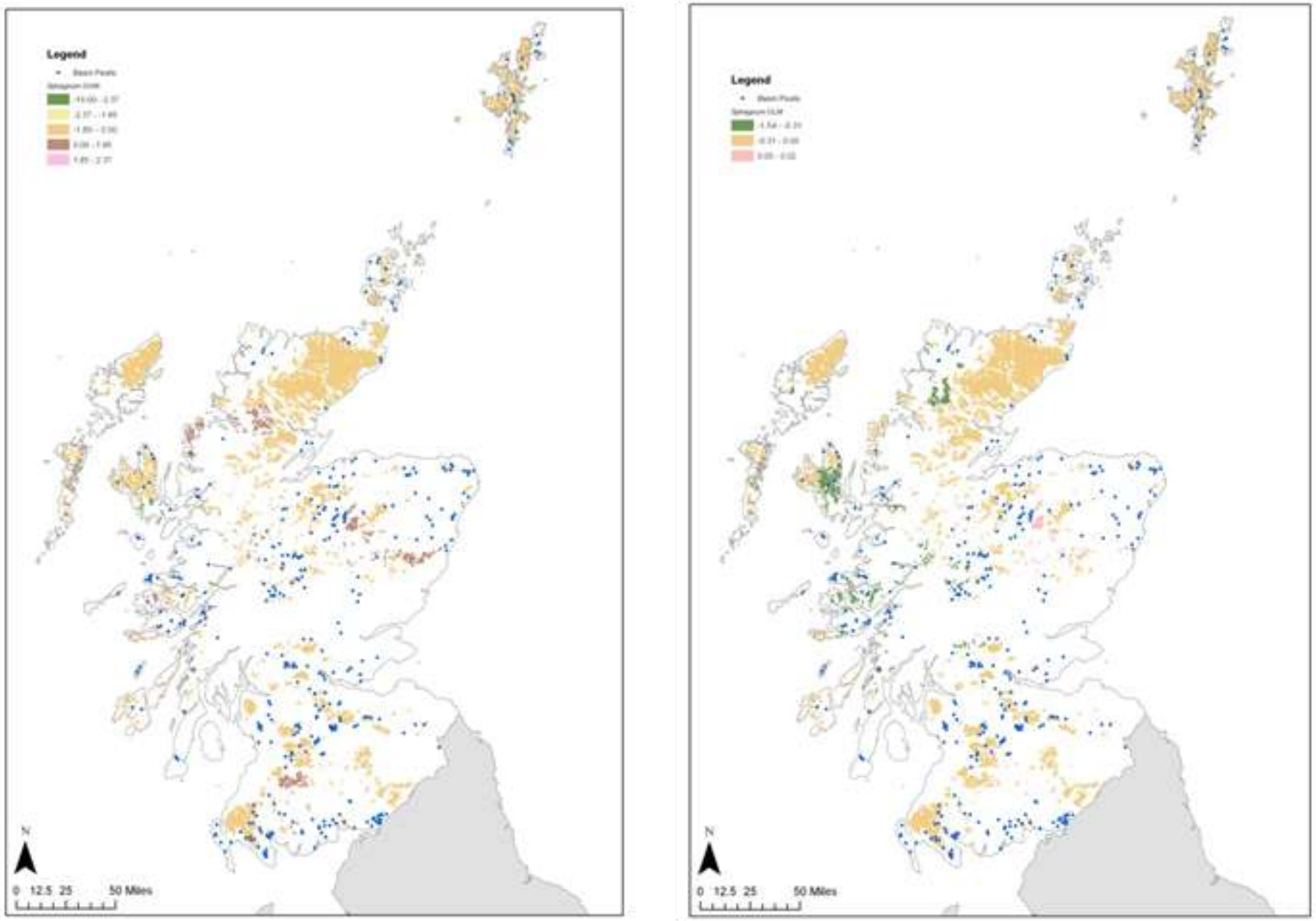


Figure 16: Forecast change in ombrotrophic Sphagnum cover by the 2050s in Scotland based on solving a GLMM based on UKCP09 high emissions climate projections and modelled impacts of nitrogen deposition (Left figure) and a GAMM based just on UKCP09 climate projections. Models were run for 1km squares containing peatland according to Land Cover Map 2000. Blue dots show the locations of basin peats (right figure).

MultiMOVE presence/absence models

A large number of individual species models have been produced using three statistical techniques (GAM, GLM and Multiple Adaptive Regression Splines (MARS)). This small ensemble predicts habitat suitability in terms of abiotic and climate variables that are similar to the *Sphagnum* cover model described above however there are important differences. The models were trained on a much larger presence/absence dataset comprising NVC, GB Woodlands Survey and Key Habitats surveys (described and referenced in Smart *et al* 2010b and their application described in DeVries *et al* 2010). Using these data, models for 1140 higher and lower plants were developed including most *Sphagnum* species (an example for *S. papillosum* is in Figure 17).

Since the NVC dataset was included, the models guarantee coverage of those NVC communities represented in raised bogs across Britain. This is a clear advantage. Another advantage of the MultiMOVE models is that they are based on predictions from three statistical modelling techniques and so a consensus forecast can be generated that builds confidence in the mean prediction and robustly quantifies the uncertainty around the mean predictions. A weakness is, that, to allow all quadrats to be included in the model building process, positions along abiotic gradients were estimated using indices based on mean Ellenberg values (Ellenberg *et al* 1991) rather than on directly measured soil conditions as in the red *Sphagnum* cover models. Thus, mean Ellenberg values for nutrient availability, soil moisture and substrate pH were calculated for each plot based on the plant species present but excluding the values for the particular species being measured. Quantitative links between mean Ellenberg values and soil conditions were based on regression models constructed for a subset of quadrats with measured soil data plus plant species composition (Rowe *et al* 2011; Smart *et al* 2010b). In addition to these indirect measures of abiotic conditions, the MultiMOVE models were also trained on cover-weighted canopy height and three climate variables; long-term averages for annual precipitation, maximum July temperature and minimum January temperature. Since interaction terms were included as well as main effects, the models allow for responses to one driver, for example climate, to be conditional on position on another gradient. For example expected change in response to warming may differ between drier or wetter soil moisture starting conditions. As with the red *Sphagnum* cover models, the caveats noted there also apply here. Consequently we never interpret model forecasts as expectations of species occurrence but as changes in habitat suitability, with species more likely to occur at higher suitability only if other factors such as dispersal and establishment filters have been alleviated. MultiMOVE models could be applied to each LRB and basin peat location. By changing the climate variable values in each model according to UKCP scenarios it would be possible to carry out an initial assessment of vulnerability to climate change. However, such an assessment would need to be cautiously interpreted in light of local knowledge and other model predictions.

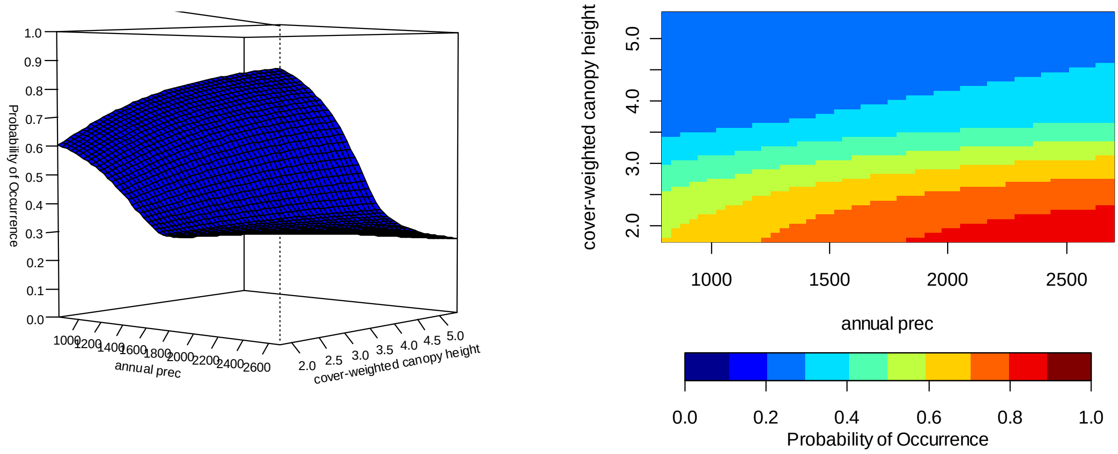


Figure 17: Example output from the MultiMOVE model is shown for Sphagnum papillosum. The model average from the three techniques is fitted to observed data from the NVC database. The modelled response to annual precipitation and canopy height is shown. All other explanatory variables are set to their median values in the observed dataset. S. papillosum is clearly strongly restricted to high rainfall and short vegetation but can coexist better with taller plants as rainfall increases.

In conclusion, an initial assessment of climate change impacts on raised bogs/lowland basin peatlands could be carried out using the MultiMOVE individual *Sphagnum* species models. These were trained on a much larger dataset and emphasise change in habitat suitability rather than species cover, which is highly uncertain. These assessments could be validated against field observations to ensure that predictions of the current situation satisfactorily matched observed presence/absence (see for example Supplementary Material on model testing in Smart *et al* 2010). A better approach would be to build new models focused on basin peats and lowland raised bogs. This would provide an opportunity to build in additional explanatory variables such as finely resolved slope, upslope catchment area, peatland unit area, management status and surrounding land-use as well as extending to more *Sphagnum* species groups and other useful composite variables such as vascular plant:*Sphagnum* ratio.

3.6. Proxies for C sequestration

As the methodology required in order to obtain full carbon budgets is laborious and expensive, many research efforts have been directed towards the development of more easily monitored, field-based, proxies, especially where large scale restoration of peatland habitats is required. Vegetation cover in particular is often seen as a good indicator of both the site hydrological condition, as many species have a good correlation in their distribution with water table (Ellenberg, 1988). Similarly, Couwenberg et al (2011), along with many empirical modelling studies of carbon flux in peatlands (e.g. Dias et al, Ecosystems 2010; Levy et al., GCB 2011), demonstrate a good correlation of the water table dynamics with net carbon dioxide and methane emissions. These correlations, although often linear within reasonable water table limits, however generally do not explain more than 50% of the variation in carbon flux. On the other hand, vegetation may influence GHG fluxes directly (rather than simply providing a proxy for water table) due to the role of some species (e.g. *Eriophorum* and other aerenchymatous species) in transporting methane to the atmosphere, and others (e.g. *Sphagnum*) in sequestering CO_2 into peat. Hence, there is considerable potential in the use of vegetation proxies for GHG flux estimation, but further developmental work is required, and vegetation composition should best be considered to provide a reasonable approximation of the net carbon savings from restoration practices in the current absence of better empirical or modelling approaches.

As an example of the use of a vegetation proxy approach, Couwenberg and colleagues (2011) tested their water table and carbon emission proxies, which were derived from regression models, based on data obtained from predominantly western European locations, on two Belarusian raised bogs. They also included forward projections of the vegetation composition in 2039 with or without rewetting restoration scenarios and used the resultant changes in projected vegetation composition to calculate rough carbon emissions for each scenario. This enabled them to estimate the reductions in carbon emissions under the rewetting as opposed to 'do nothing' scenario.

Such an approach is entirely feasible for UK peatlands. Dynamic vegetation modelling is feasible for models such as LPJ-GUESS, which will, however, require calibration for some of the vegetation types found in UK peatlands as it is currently focused on forest habitats. A version of LPJ more specific to peatlands, LPJ-Why, which is parameterized to include flood-tolerant C3 graminoids (e.g., *Carex* spp., *Eriophorum* spp.) and *Sphagnum* spp. has been developed by Wania et al (2009). The abovementioned empirically-based GBMOVE model (Smart et al., 2010) provides predictions of the probability of occurrence of key species based on measured or modelled soil, hydrologic, climatic and management variables, and is based on UK-specific data from the Countryside Survey. The model would also require further development for specific application in this context, and could be augmented by a dynamic model of peat vegetation competition and growth.

Vegetation composition can be directly assessed via remote sensing techniques, using high quality satellite imagery. The Landsat 7 ETM+ libraries would allow for calculation of the commonly used normalized difference vegetation index (NDVI) or the enhanced vegetation index (EVI), which can, on occasion, be more useful as it does not saturate to the same degree during peak primary production (green-ness, Schubert et al., 2010). However, Landsat 7 data are not available at a high enough resolution for the heterogeneity of small raised bogs. In contrast, the highest level of resolution currently available from satellite data is in data from Ikonos or Quickbird, where panchromatic resolution approaches that of aerial photography. Multi-spectral satellite imagery is still an order of magnitude off large scale aerial photos. Airborne multi-spectral scanning is feasible, but expensive. Extraction of vegetation parameters from such multispectral data at a spatial resolution of 2-4 m is also possible (Harris and Bryant, 2009). NDVI can be used as an indicator of relative biomass and greenness, or, if adequate ground truthing data are available, the NDVI can be used to calculate and predict primary production, dominant species, and, at the highest resolution, impacts of grazing, peat extraction, drainage and burning practices. For peatlands, the current state of research has progressed such that major plant functional types have been classified in various remote sensing projects (Bubier et al., 1997; Schaepman-Strueb et al., 2009). Schubert et al (2010) also recently demonstrated that such measures were also able to aid model predictions of net primary production (NPP) as well as ecosystem respiration (ER) in Sweden and hence allow for landscape scale modelling of carbon exchange as well as vegetation diversity. The ability to make such predictions for Scottish peatlands is still some way ahead, due to the need to parameterise vegetation indices for the vegetation types found within these peatlands. Scottish peatlands may present a more heterogeneous landscape than elsewhere in Northern Europe due to the much more varied hydrological conditions resulting from climatic differences within the landscape, topographic variation, and the impacts of previous drainage and other land-use factors. However, due to the volume of research that has already been carried out, there would be much scope for developing these tools for Scottish peatlands. Hydrological conditions that induce physiological stress in plants (drought) affect photosynthetic rates and hence net primary productivity. As the productivity of *Sphagnum* species in particular can be very sensitive to drought conditions, much research has focused on producing models that allow for adequate identification of the hydrological status of Sphagna in remotely sensed data and subsequent calculation of the net primary production capacity at the landscape scale (e.g. Harris, 2008). At least a measure of vegetation diversity at national scale, through NDVI and/or EVI, would enable a much more detailed assessment of site condition on the basis of vegetation condition to be made. Anderson et al. (2010b) have provided an example of this approach for Wedholme Flow (Cumbria), using a combination of multispectral ICONOS and Lidar data to classify peat condition based on microtopographical and vegetation condition features.

3.7. Ongoing research on GHG fluxes on raised bogs

Current research on GHG fluxes in British peatlands was summarised in JNCC Report 433 (Evans et al., 2011). Most existing research is taking place within upland blanket bogs, with some notable exceptions. In Scotland, the Auchencorth Moss CEH Carbon Catchment monitoring site (Dinsmore et al. 2010; Helfter et al., 2011) and Whim Moss field nitrogen manipulation experimental (Sheppard et al., 2011) are located within the same large raised bog complex near Edinburgh. Auchencorth Moss now has the longest-running and most complete C and GHG budget of any UK peatland site, including eddy covariance CO_2 and CH_4 measurements. The study catchment is considered to represent a raised bog in reasonably good condition, although it was subject to past drainage, and overgrown drainage ditches may still influence water levels. A small area of the catchment (outside the gas flux measurement area) lies within an adjacent peat extraction site, and the dominance of graminoid species (*Deschampsia flexuosa*, *Eriophorum vaginatum* and *Juncus effusus*) suggests some degree of degradation. Whim Bog site has a natural hummock-hollow topography with *Calluna vulgaris*, *Eriophorum vaginatum* and *Sphagnum* dominant, although water table may be affected somewhat be adjacent peat extraction. A substantial amount of CH_4 and CO_2 flux data, as well as porewater DOC measurements have been collected at the site, under both ambient and elevated N conditions. Data extent over approximately a 5 year period, although not all measurements are continuous, and some remain unpublished. Further measurements will be made under a new EU project, ECLAIRE. The nearby Deep Syke research site, also part of the same raised bog complex, investigated the effects of elevated N and S deposition on the biogeochemical cycling of an afforested area of shallow (< 1m) peat. Some CH_4 and N_2O measurements were made as part of the project, but the site has now been decommissioned.

At Flanders Moss, Yamulki and colleagues (2012) recently completed a 2 year chamber-based study of the effects of tree felling and drainage associated with the plantation on fluxes of CO_2, CH_4 and N_2O and concluded that draining pristine peatlands for forestry increases net global warming potential, even accounting for carbon uptake by the plantation, due to increased soil net respiratory losses of CO_2. Restoration of the afforested areas was found to increase CH_4 emissions, leading to net global warming potential above that of an afforested peatland. It will remain to be seen whether such methane losses are a temporary feature during restoration.

Finally, a 2 year monitoring project concluded in 2006 at Middlemuir Moss on a series of four areas of a formerly mechanically cutover raised bog. Both CO_2 and CH_4 exchange were studied using chambers. The site was a machine extracted raised/intermediate lowland bog, using sites ranging from being still bare 10 years post extraction to naturally regenerating vegetation 5-50 years post (manual and mechanical) extraction. The oldest site sported an accumulation of a new acrotelm of 4-10 cm of relatively poorly decomposed vegetation, but was not floristically or hydrologically equivalent to a raised bog in good condition. None of the sites seemed to be obvious C sinks, although the losses were lower in the 50 year old sites. While the data were insufficient to model carbon budgets, the results nevertheless suggested that the revegetated areas (ca. 5-10 and over 50 years since last cutting activity) were emitting less gaseous carbon and were actually actively fixing carbon for at least some of the year (Artz et al., 2008). The data showed similar trends to other cutover sites in Europe. For example, Bortoluzzi et al (2006) saw emissions reductions, to the point of returning to a net C gain, in terms of CO_2 and CH_4 exchange in sites allowed to naturally revegetate for 20 years. These sites were in an upland area of the French Jura mountains, had been hand cut, and had little drainage associated with them. Samaritani et al (2011) used 3 sites in the Swiss Jura mountains at 26, 42, 51 years post manual extraction and observed emissions reductions with age, with sites approximating 50 years returning to an actively C fixing state. The site history for these was similar to the French sites above, but has slightly different climatic and floristic characteristics. Finally, Yli-Petays et al (2007) studied 50 year old Finnish lowland raised bog sites post mechanical harvesting and found these to be modest C sinks compared with natural peatlands close by, again suggesting substantial emissions reduction can be achieved through restoration management. Although these non-UK examples may not be directly comparable, it nevertheless points out that returns to fully carbon sequestering systems for such highly altered sites may be a long term project, that will benefit from targeted interventions. For example, both the studies above, and more directly, Kivimaki et al. (2008) have pointed to the benefits of having *Sphagnum* as part of the re-establishing vegetation cover, as this maximizes the carbon benefits from restoration as the *Sphagnum* carpet reduces methane emissions by facilitating methane oxidation and adds to the carbon sequestration capacity.

Further work on raised bogs in relation to climate impacts is taking place within England and Wales. A substantial body of research exists on Cors Fochno, an estuarine raised bog in mid-Wales, including detailed work on peat hydrology (e.g. Baird et al., 2008) and methane emission (e.g. Baird et al., 2004). Since 2010, Cors Fochno has also been one of two UK field sites in the PEATBOG EU project (http://www.sste.mmu.ac.uk/peatbog/), and has been subject to a factorial field warming and drought experiment to examine the possible climate change effects on peatland carbon cycling. Measurements include CO_2, CH_4 and DOC fluxes under ambient and manipulated conditions. The second UK project site is at Whixhall Moss, on the Welsh-English border, a site with high historic impacts from drainage and peat extraction, as well as atmospheric N deposition. The site has recently been restored via re-wetting and tree removal. Experimental manipulations parallel those at Cors Fochno. Initial results (Rowson et al., 2012) indicate that warming and drought treatments have both reduced the CO_2 sink at Cors Fochno, with the combined warming+drought treatment having a greater negative effect than the sum of the individual treatments. At Whixhall Moss, early observations suggest that the climate manipulations may have turned the site into a net CO_2 source (J. Rowson, pers comm.). While these last two examples have not included restoration treatments, these observations further strengthen the argument that it would be beneficial to mitigate potential climate-induced emissions through ensuring best possible habitat conditions in lowland raised bogs.

Also in England and Wales, a new Defra-funded research project (SP1210) will be measuring full C and GHG budgets for a total of 13 sites across six lowland peat areas. Four of these areas are on fen peat (East Anglian Fens, Somerset Levels, Norfolk Broads and Anglesey Fens), and two on raised bogs, the Manchester Mosses (Chat Moss) and Humberhead Levels (Thorne Moor). The aim of the project is to derive estimates of the C and GHG balance of representative lowland peat sites under contrasting land-use and management, in order to derive emission factors for GHG accounting. Measurements (which will start in 2012) include detailed vegetation and peat characterization, hydrological monitoring, gas flux measurement via a combination of static chamber and eddy covariance methods, and fluvial C flux measurement. The management classes represented for raised bogs are semi-natural vegetation (at Chat Moss and Thorne Moor, both with some historic effects of peat-cutting), active peat extraction and drained arable (both at Chat Moss). The project is also undertaking a systematic review of the effects of land-management on greenhouse gas fluxes specifically for lowland peats, with an initial draft produced during 2012.

3.8. Identifying research needs for future monitoring

A recent JNCC report 443 (Evans et al., 2011) provides a framework for UK peatland C and GHG measurement. This comprises a combination of baseline measurement sites representing different peat and management types, and well-designed (controlled, before-after, replicated and/or catchment scale) experimental studies to examine land-use transitions such as drainage and re-wetting. The study proposes three levels of peatland C/GHG measurements, to provide a balance between intensive measurements at a few sites, and lower-intensity measurements across a broader area. Level I sites would provide comprehensive, detailed and long-term flux measurement data for a core set of sites such as the CEH Carbon Catchments (including Auchencorth Moss, as described above). It would be envisaged that these would cover a few examples of the relevant condition classes for peatlands, so that reliable emissions factors can be calculated. The report suggested 14 candidate sites on peatlands for Level I monitoring across the UK, including Flanders Moss and Auchencorth Moss in Scotland (Forsinard was mentioned as a blanket bog site). Level II sites would include less intensive flux measurements, for example those that would be obtained from the control plots of experimental studies based on monthly sampling. These sites, more numerous in nature, would help to validate the resulting emissions factors from Level I sites. Level III sites would provide larger-scale, infrequent survey-type data on peat condition, vegetation type and growth rates, and carbon stock. Defra Project SP1210 largely adopts the approaches set out in the report, with a focus on the comprehensive (Level I) monitoring of a set of representative sites under stable management. There have been no suggestions thus far on the number of candidate sites for Level II and III sites, as the JNCC report did not attempt a power analysis. However, if such an approach is adopted, the minimum monitoring requirements under Level III have been reproduced in Box 2.

Box 2. Minimum monitoring requirements for Level III monitoring (Evans et al 2011).

Initial and 5-yearly vegetation surveys at permanent quadrats (proportional cover of major plant functional types and key indicator species, to include key plant functional types and indicator species, and recording of bare peat areas)

Initial C stock measurement based on whole-profile coring, and 5-yearly soil C stock change measurements based on shallow core sampling (depth, bulk density and %C) to a dateable horizon or fixed point.

Initial collection of a full peat core for basal age measurement, long term C accumulation rate and contemporary C accumulation rate estimation.

Initial collation of aerial photograph and LIDAR data, if available for the site, and recording of ditches, bare peat or burnt areas, erosion features and microtopography.

Installation and monitoring of a network of dipwells, to provide an indication of average water table. Water table loggers may be more cost-effective than manual recording,

Annual fixed-point photographs to provide a record of vegetation and site condition.

Annual recording of site management, biomass offtake (if relevant), restoration activities, burning etc.

In Scotland, there are currently a small number of high-quality research programmes ongoing (as described above), but as yet no integrated national-level flux measurement programme. Given the complexity and heterogeneity of peatlands in general, and raised bogs in particular, there would be clear benefits to coordinating any new measurement activities within Scotland, and more broadly in aligning these with existing and new research and monitoring elsewhere in the UK. Given the relatively high cost of establishing a full C/GHG measurement programme, this could permit more complete coverage of different peat management and condition types (for example, the Defra project does not include afforested, grassland or degraded raised bogs). In addition, sites in England and Wales may provide 'climate analogues' for Scottish bogs under future climate change, indicating how the C/GHG balance of Scottish raised bogs might be expected to change under a given management regime. The establishment of a cost-effective Scottish raised bog monitoring programme could in part be achieved through the integration and alignment of existing research sites, augmenting these with additional measurements where necessary. Given the limited number of such sites at present, however, it may be necessary to add additional monitoring locations in order to achieve coverage.

At present, there is no large-scale (Level III type) survey-based monitoring of peat carbon stocks in the UK. Data on peat C stocks are largely derived from historic national soil surveys with no current plans for further revisits. The UK Countryside Survey incorporates periodic measurements of vegetation condition and surface organic horizon quality for bog habitats, but these data are insufficient to quantify stock changes. However, the possibility exists to augment future Countryside Survey (or other) monitoring programmes to provide information on stock change (Evans et al., 2010, 2011), or to establish new monitoring programmes specifically for peats, for example linked to agri-environment monitoring. In general, both intensive site-scale and extensive survey-based measurement programmes should aim to generate data useful for the development of emission factors, and data for the calibration of proxy methods (e.g. detailed vegetation data as described above) for mapping emissions at the larger scale.

3.9. Current management schemes and potential areas for restoration

The UK Biodiversity Action Plan is delivered in Scotland through the Scottish Biodiversity Strategy. The current status in relation to raised bog habitats is unclear. The Biodiversity Action Reporting System (BARS) reports on progress in relation to the biodiversity targets. At present, the following UK BAP targets are in place for Scotland:

T1 Maintain the extent of the existing UK resource of BAP habitat (i.e. primary and secondary raised bog resource) with no loss. Target (13,000 ha) not achieved. 2008 & 2010

T2 Rehabilitate degraded bog habitat still capable of natural regeneration (in targeted areas) to bring most of the primary and secondary resource into or approaching favourable condition by 2020 through appropriate management. Targets 4,333 ha (2010) 8,666 ha (2015) 13,000 ha (2020). Some progress (behind schedule). 2008.

T3 Restore Lowland Raised Bog immediately or via succession from fen on chosen areas of archaic peat to ensure a sustainable hydrological regime for adjacent extant habitat and to restore LRB to its former geographical range as part of a national series. Targets: 35 ha (2010) 70 ha (2015) 100 ha (2020). Some progress (behind schedule). 2008

In Scotland, significant early progress in raised bog habitat restoration and public awareness was made through an EU-LIFE Project with the Scottish Raised Bog Partnership (a partnership between Forest Enterprise (Forestry Commission Scotland), Scottish Natural Heritage and the Scottish Wildlife Trust, Brooks, 2007). Restoration work was completed successfully at eleven sites, 10 of which are SACs:

Bankhead Moss,
Braehead Moss,
Carsegowan Moss,
Coalburn Moss,
Cranley Moss,
Dykeneuk Moss,
Flanders Moss,
Moine Mhor,
Longbridge Muir,
Kirkconnell Flow
Threepwood Moss

Work could not be started at a twelfth site in the Grampian area (Turclossie Moss). The project as a whole involved the removal of 430 ha of trees, clearance of 253 ha of encroaching scrub, installation of 2,153 dams into ditches, erection of 12,101 m of fencing and removal of 3.6 ha of rank heather. The objective was to safeguard an active raised bog area of 1,256 ha, to increase the area of active raised bog by 315 ha by clearing trees, scrub and heather and create suitable conditions that will encourage the natural regeneration of additional degraded raised bog. The project also aimed to devise, implement and monitor a range of site management techniques aimed at improving the conservation management of a cSAC area of 3,700 ha of active raised bog in Scotland. The total project value was 2,139,262.00 € (around £1.8 to 1.9 million), which translates into an average restoration cost of £1,177 per hectare.

Through SWT and the North East Scotland Biodiversity Partnership, efforts to clear scrub and restore the water table have been implemented at Red Moss of Netherley. Management plans involving restoration programmes for a further 8 sites in this region are completed but no current effort is taking place. Many of these peatlands were part of the Natural Care Grampian Lowland Bog Scheme (2003 – 2006). It aimed to stimulate restoration management within the following areas:

Black Moss SSSI cSAC (Muir of Dinnet NNR)
Moidach More SSSI cSAC
Moss of Crombie SSSI
Parkins Moss SSSI cSAC (Muir of Dinnet NNR)
Red Moss of Netherley SSSI cSAC
Reidside Moss SSSI cSAC
Rora Moss SSSI
Turclossie Moss SSSI cSAC
The Four Bogs of New Pitsligo

The Grampian Lowland Bog Scheme (2003-2006) as well as the South Scotland Bog Scheme (SSBS, 2006 onwards) still have ongoing management agreements. The South Scotland Bog Scheme was available for all lowland raised bog sites designated as SSSI or SAC within three Scottish Natural Heritage Areas: Forth and Borders, Argyll and Stirling, and Strathclyde and Ayrshire, and hence was potentially applicable to 42 raised bogs. The SSBS paid for management plan costs as well as basic management at £40 ha^{-1} for the first 150 ha, then £20 for additional areas, up to a maximum of £9000 per year. Provision of buffer land was included at payment rates of £248 ha^{-1} yr^{-1}. Drain blocking as well as grazing management options were available as one-off payments at similar rates to the current Axis 2 SRDP options (below). Monitoring and maintenance costs of dams across drained areas attracted an annual flat fee of £450 per management area. Scrub and tree encroachment control was also provided, at similar rates to the current SRDP programme. It is thought that the total cost of the SSBS programme at the end of the 6 years, in October 2014, will be £89,000.

In contrast, the GLB scheme paid for management and maintenance costs of £20 ha^{-1} for sites up to 100 ha and £12 ha^{-1} for larger sites, with a minimum payment of £200 but no more than £3,800 each year. Public access management was funded by an additional £5 ha^{-1}. The scheme did not have very high uptake, presumably due to the rather low funding levels and the stipulated cessation of peat cutting rights.

Its successors, the current SRDP Axis 2 options, have two available schemes. The 5 year options for 'Management/Restoration of Lowland Raised Bogs' includes a payment rate of £40 ha^{-1} yr^{-1} with a supplement of £43 ha^{-1} yr^{-1} for the grazing management option. The 5 year 'Buffer Areas for Fens and Lowland Raised Bogs' scheme includes a flat payment of £267.90 ha^{-1} yr^{-1}. Capital costs for a range of peatland restoration measures are supported (Table 11). New uptake rates for 2009 were encouraging, with 629 hectares under the basic raised bog scheme and a further 519 ha under the scheme with grazing supplement. The buffer areas scheme was in place for 57 ha by 2009. Uptake is geographically varied, with some of the larger raised bog deposits in the North East Grampian region and the Central belt supported by this scheme. There are also a number of peatland areas that are supported by this scheme that do fall on lowland basin peat but do not correspond to LRBI entries, notably in the far North, South-East and South-west of Scotland. As at 2010, there were 21 entrants in the Management/Restoration of Lowland Raised

Bog scheme, for both the basic option and the additional grazing management. The payments for these were £316,993 and £422,981, respectively (The Highland Council, 2010). This translates into a cost of ca £645 ha^{-1}, a figure almost 50% lower than the Scottish Raised Bog Partnership project. A further breakdown of these figures (data not shown), however, shows much more limited actual restoration works taking place under these payment, with much of the cost going towards annual maintenance rather than capital works. In addition, the option for Buffer areas for fens and lowland raised bogs attracted 22 entrants, of which one for a fen area, with payments totalling £ 582,019. Restoration work under the current SRDP scheme for approved at the September 2011 Rural Priorities assessment round included Newmiln Farm, at Tibbermore, Perth, which was awarded £57,200 to continue work undertaken as part of a Management Agreement with Scottish Natural Heritage to manage the Methven Moss SSSI/Special Area of Conservation. Restoration work includes tree felling and removal, scrub control and installing three dams in an adjacent water course to maintain the water table at a high level and help prevent the drying out of the peat.

Table 11. Restoration capital costs supported by the SRDP Raised Bog options

Management	**£/ha**
Light cover, open scrub clearance	600
Light cover, scrub removal	500
Closed cover, intermediate scrub clearance	850
Closed cover, intermediate scrub removal	1050
Closed cover, dense vegetation clearance	1250
Closed cover, dense vegetation removal	1450
Grip blocking costs	60-280*
Bracken treatment	200
Rhododendron removal (manual)	3700
Rhododendron removal (chemical/mechanical)	1750

*depending on grip spacings. Further payments can be made available for livestock bridges or fencing costs.

Matthews (2012) presented the results of a Scottish Wildlife Trust initiative to analyse the costs associated with restoration towards favourable site condition. Their data indicate an average cost of £1,280 ha^{-1} for capital restoration costs such as tree and shrub felling and drain blocking and an additional £40 ha^{-1} towards annual management costs for grazing management or maintenance of dams.

3.10. Review of specific restoration costs and benefits

A lot of restoration work has now been done and the benefits of this should be seen within the next decade if those sites continue to follow a trajectory of return to a raised bog habitat. High carbon benefits can be achieved in the early stages of restoration projects (section 3.4). In cases where site degradation was minimal or where little site disturbance is required for site restoration, it may even be possible to achieve reversion to a net C accumulating system fairly quickly. Hence, there will already be some avoided loss that has been realised through the various restoration programmes carried out to date and the current SRDP targeted programme. Lunt et al (2010), in one of the IUCN reviews, pointed out the multiple benefits of peatland restoration and presented a meta analysis of the effects of peatland restoration on various ecosystem services. Their analysis indicated that grip blocking first benefited the stability and height of the water table, followed by a lagged effect on carbon sequestration potential due to the time taken for paludification of rewetted drains. Improvements in biodiversity targets were likely to take even longer. Conversely, the effects of grazing management or active vegetation restoration may bring biodiversity benefits sooner.

A full economic cost:benefit analysis for carbon savings on raised bogs would necessitate a number of data that are not currently readily available. The following data should be collated for a number of sites ranging in area between the lower end of the raised bog resource (i.e. 10-30 ha) to the upper end (>100 ha):

1) Initial restoration cost (initial management) ha^{-1}
2) Annual management and monitoring cost ha^{-1}
3) The changes in net carbon emissions over time between the different stages before, during, and after restoration to enable calculation of Carbon flow improvement in tonnes C ha^{-1} $year^{-1}$

At present, both the initial and annual costs of restoration can be estimated within some uncertainties. The previous section (3.9) gave examples of average restoration costs of around £1,280 ha^{-1} and annual management costs at around £40 ha^{-1} yr^{-1}. The missing data are the changes in C emissions over time. If such data were available, cost:benefit calculations could be made that take into account the current market price of carbon or, alternative measures that include the social cost of carbon. The market price of carbon has fluctuated rather a lot since this measure was introduced. The last year has seen a 50 per cent decline in the volume-weighted average carbon price compared with the start of 2011 to €6.6 per tonne. Early in 2012, the price collapsed to an all-time low of €5.99 (= £4.95; Table 12). **At this lowest carbon market price, the present value of the entire raised bog carbon stock would equate to £316.8 million. The annual carbon benefits that sequestration performed by the sites that are currently in good condition provide would most likely be valued at between £35,000 and £99,000** (depending on whether we use the most likely ends of the relevant LRBI or LCS88 derived condition categories). **However, the cost of the carbon emitted from sites in degraded state**

would run to an annual financial loss of approximately £0.1-0.7 million (LRBI and LCS88 figures are similar, Table 12).

In addition, as many have pointed out, the market price of carbon is only one potential measure and many have advocated a pricing structure that reflects the full, social, price of carbon. The social cost of carbon (SCC) is an indication of the cost of one incremental unit of carbon (or equivalent for other greenhouse gases) that is emitted in the present, which is calculated by summing the full global cost of damage imposed over the whole of its atmospheric lifetime. The SCC provides an indication of the scale of the externality that needs to be incorporated into policy decisions and investment options (Defra, 2007). The IPPC 2007 figures were based on peer-reviewed estimates of the social cost of carbon for 2005, and used an average value of \$43/tC (=£26.9, Table 12). The Stern review pointed out further required inclusions to conclude that the price should be > \$300 per tC. Anthoff et al (2009) recently adapted figures from the Stern review, correcting these for empirical risk preferences of governments, and equity weightings (such as income differences across the world) and calculated a figure of \$205.5 per tC (= £128.8; Table 12).

Losing the entire carbon stock of the Scottish raised bog resource, at the social carbon price, would translate into a financial loss of £ 1.7-8.3 billion, depending on which social carbon price (IPCC or Stern) is used, or, as pointed out in Section 3.4, equate to 18 years of the net transport emissions. **At these prices, the currently carbon benefits that raised bogs in good condition provide translates into between £51k and £2.5 million annually**, depending on the SCC value used, but also factoring in the uncertainties of categorizing the peatlands into condition classes. **The social price of net emissions from sites in degraded condition, however, runs to an annual cost of between £0.7 million to £18 million** depending on the figures used (Table 12).

Table 12. Value of the total raised bog carbon stock, annual carbon sequestration from sites in good condition and annual carbon losses through emissions from sites in degraded condition, for three carbon pricing scenario's.

Carbon prices		Total raised bog carbon stock	Current annual C sequestered in sites in good condition		Current annual emissions from sites in degraded condition	
		64 Mt	LRBI (1.9-7 kt)	LCS88 (5-20kt)	LRBI (26-125 kt)	LCS88 (21-143 kt)
	GBP					
Lowest Market	4.95	**£316.8 mi**	**£9.4-34.9k**	**£25-99k**	**£0.1-0.6 mi**	**£0.1-0.7 mi**
Social/ IPCC	27	£1.728 bi	£51-189k	£135-540k	£0.7-3 mi	£0.7-4 mi
Revised Stern	129	£8.256 bi	£0.2-0.9 mi	£0.6-2.5 mi	£3-16 mi	£3-18 mi

The restoration costs themselves can also fluctuate wildly. Matthews (2012) collected data on the initial restoration and annual management costs for 58 sites undergoing monitoring by the Scottish Wildlife Trust. Initial restoration costs for these sites ranged from <£10 ha^{-1}, for sites where only light intervention was required, to costs > £1500 ha^{-1}, generally on sites where woodland removal and/or extensive rewetting through dam blocking was required. Similarly, annual costs for monitoring inclusive of dam maintenance were reported by Matthews as a flat £450 yr^{-1} cost per site, irrespective of the size of the raised bog. Evans et al (2011) suggested a number of minimum observations for carbon stock and flow monitoring as part of a terrestrial peatland framework. The figures collated in their report would suggest a figure of around £5000 yr^{-1} per site for monitoring costs. In addition, some sites may require ongoing scrub control and/or grazing management adjustments. Matthews (2012) gives a flat £150 yr^{-1} cost for scrub control, which may only be realistic in the years immediately following initial restoration. Current SRDP funding calculates a rate for low scrub removal and clearance of £525 plus £500 ha^{-1}, respectively, rising to £1250 and £1450 ha^{-1} for dense vegetation removal and clearance, respectively. If the area is under designation, these funds are even higher. Grazing management measures through SRDP under the raised bog schemes is paid under a supplement of £43 ha^{-1} yr^{-1} but is dependent on site requirements. Indeed, Matthews (2012) presents figures for grazing management costs ranging from as little as £3 ha^{-1} for very large sites with little need for grazing management. The figures presented for the overall costs to restore the whole of the Scottish raised bog resource, by extrapolation, was £20.79 million in capital expenditure plus an additional £650,000 annual management cost by Matthews (2012). **A simple calculation of their average capital restoration cost figure (£1,280 ha^{-1}) x the area of degraded raised bog (ca. 25,000 ha) would suggest initial capital costs to be closer to £32 million. Further annual maintenance costs would be suggested to be in the order of £650,000 as per Matthews (2012).**

Even at the lowest market price, and especially when considering the social cost of net emissions, such estimates suggest a good cost:benefit trade-off could be reached relatively quickly, especially if restoration is prioritized for sites where the largest or fastest carbon gains are feasible. It may be beneficial to see the capital expenditure as expenditure to safeguard the total carbon stock in the long term as well as reducing emissions in the short term and the annual management costs as the tool to produce carbon savings by reducing annual net emissions. Hence, the large capital expenditure figures of £20,79-£32 million, when set against even the lowest market value of the stock alone (£316.8 million) look like a good investment, with the potential savings from reducing the net annual emissions forming the return on investment. If all carbon emissions for the stock could be mitigated, at the lowest estimate of the social cost of current carbon emissions, this would be worth £0.7-3 million annually (Table 12), a figure still higher than the estimated annual management costs.

While these figures illustrate the potential economic benefits of restoration in carbon terms, figures for the actual reduction in carbon emissions through restoration are still extremely scarce. To calculate a return on investment, it is necessary to know the trajectory of carbon emissions from a site that has undergone restoration; in other words, the carbon improvement in t C per ha^{-1} per yr^{-1}. As yet, there is no single restoration project in Scotland that has produced full carbon budgets before, during, and after restoration and hence there are no valid literature values on which to base such calculations. The comparative studies that have been performed on restored sites elsewhere in Europe generally did not include a 'before' scenario, nor have all carbon pools been adequately studied to produce a full C balance (sections 3.4. and 3.7). However, general assumptions can be made by utilising the likely emissions figures from peatlands in different current condition, augmented with observations about the timescale required to shift between these post-restoration (Figure 18). However, it is generally observed that improvement in net emissions in restoration sites is not linear through time (Figure 18), and hence more data through a suitable time frame from a number of restoration sits in different starting conditions should be collected.

The likely time post restoration until significant emissions savings can be measured is likely to be dependent on the starting condition of the peatland and the historical types and severity of disturbances. In sites where drain-blocking or large scale tree removal may need to be applied, there is a possibility that the restoration activities themselves cause a temporary increase in emissions during restoration (Figure 18, worst case scenario, red line). There may, for example, be a temporary increase in methane emissions after measures such as the blocking of drainage ditches. The worst case scenario may be the temporary soil disturbance caused by restoration practices such as tree felling for harvest, as the soil surface is severely disrupted in such management efforts. There has been anecdotal evidence of increased DOC in aqueous fluxes as well as increased CO_2 emissions during the breakdown of needle litter and the decaying tree stumps. However, such increased fluxes are only a short term effect of major disruption and appear to often cease after only a short few years. Hence, it will take longer for such projects to achieve net emissions reductions. However, the short term carbon benefits in such projects are more obvious than in site where current losses are lower (e.g. overgrazed or slightly drained areas). The total carbon savings in such scenarios may be lower, but they also carry a much lower risk of failure (Figure 18, blue line, best case scenario). In addition, the likely capital expenditure in such sites is much lower (grazing exclusion or addition management, perhaps minimal scrub treatment). Consequently, cost:benefit ratios will be highly site-specific. It is worth noting that the higher temporary emissions from restoring the more damaged bogs will only apply to a small area of the total available for restoration, for example a maximum of ca. 9,500 ha of forestry or woodland and between 2,300 and 3,000 ha of cultivated raised bogs.

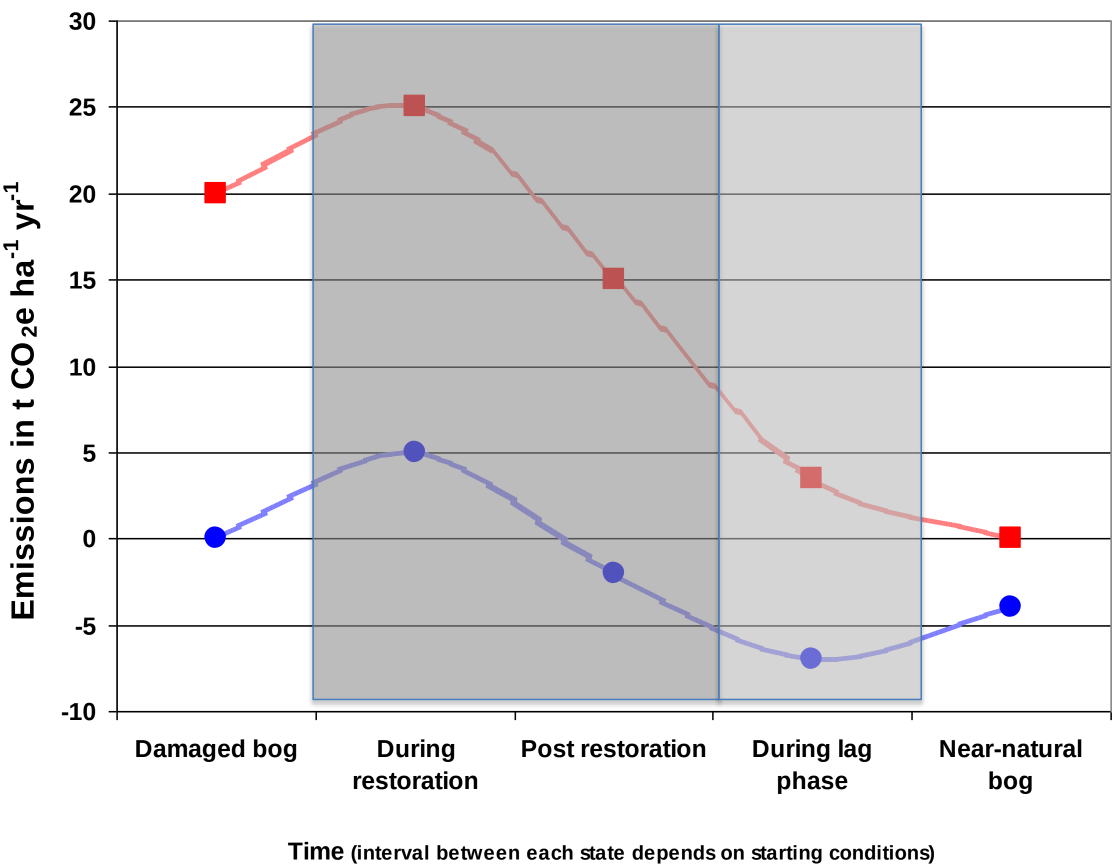


Figure 18. Best and worst case scenarios for carbon abatement potential from restoration in Scottish peatlands. The red line depicts the likely change in carbon emissions resulting from restoration of the damaged peatland sites with highest current emissions (drained or agricultural), while the blue line shows the likely changes resulting from restoring the least damaged sites. The time taken for projects from highly damaged bogs (red line) to stabilize to near-natural bog emissions is likely in excess of 50 years, although very substantial savings will be made in the short term by lowering emissions from the starting scenario to that of post restoration sites. Restoring less damaged areas may show stable C benefits (near-natural bog scenario) within a 10-20 year time frame. The costs of the capital investments to restoring the most damaged bogs are likely substantially higher (felling, drain blocking) than in less damaged areas (grazing management, shrub removal). However, focusing on only either restoring the most damaged bogs may not be the most cost-effective option, as deterioration in the less damaged bogs will continue, neither will the lower investment cost in restoring the less damaged bogs lead to the best carbon reductions. The grey areas indicate our uncertainties in data, the darker the shading the less underpinning data are available.

The focus in policy options for raised bog management from a carbon perspective should be one of encouraging a condition that minimizes further losses in the case of sites with large carbon stocks that are in poor condition, particularly those in areas with poor climatic prognosis. It may be possible to judge, on a case by case basis, whether any particular site would benefit from even more intensive management to try to optimize conditions for renewed net carbon sequestration, where this is in balance with, or not likely to lead to trade-offs with, the other ecosystem services that are provided by that site.

Extrapolating the current payments from the EU Life Project, the SSBS as well as current running SRDP options, the total payments of ca. £2-3 million to date have gone into safeguarding a 64 Mt carbon stock as well as improving the annual sequestration function of bogs in good condition or minimizing losses from those in poorer states. At present, we cannot estimate the carbon benefits that such projects have already provided. In addition, Christie et al (2010; in Reed et al., 2010) estimated that the total use and non-use value of the ecosystem services that is delivered by the UK BAP for lowland raised bog ecosystems is around £1.5 million annually. Moxey (2010) pointed out that the equivalent of the difference between the CO_2 emissions between a degraded and natural site are driving an executive petrol car for 13,000 km, which is not far off the UK average annual mileage per vehicle owned. Moxey estimated the mitigation cost for near natural peatland to be £6 per t CO_2e and for grip blocking activities at £13 per t CO_2e. Both of these figures are within the same range as domestic building insulation costs, biomass boilers, and afforestation programmes and within an order of magnitude or closer of the abatement potential per year. Although the raised bog resource is only a small component of the overall UK peatland resource, with most of the remaining raised bog habitat in Scotland, restoration efforts overall appear to be good value for money.

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