Peatland Biodiversity

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Summary

Land covered in accumulated peat is known as peatland. It is active peatland if peat is being formed now. Estimates of the extent of peatland in the UK vary widely but most are between 1.5 and 2.5 million ha. The UK may host between 8.8 and 14.8% of Europe's peatland area and about 13% of the world resource of one peatland type, namely blanket bog. Indeed blanket bog forms the largest expanse of semi-natural habitat in the UK. The review focuses primarily on blanket bogs and on lowland raised bogs.

Biodiversity Features

Active bog is characterised by an abundance of bryophytes, especially the bog moss, *Sphagnum*. Different *Sphagnum* species, with different preferences for degree of ground wetness, form the characteristic hummock and pool systems and thus create topographical variation. The plant assemblage also includes a range of sedges and dwarf-shrubs and grades into associated habitats such as wet and dry heathland. The peatland vegetation assemblage, alongside high water levels, provides the key ecosystem service of laying down new peat accumulations and maintaining the peat store.

Peatland biodiversity includes a range of rare, threatened or declining habitats, plants and animals. Some plant assemblages are better represented in the UK than anywhere else in the world. A significant number of peatland plant communities are considered to be of European importance.

UK peatlands, especially blanket bogs, have a rich and unique breeding bird assemblage. It is a species-poor assemblage, although it contains an exceptionally high proportion of species with legal protection under UK and European conservation law. Iconic species for which UK peatlands are particularly important include golden plover, greenshank, red-throated diver, dunlin and common scoter. Some species, such as red grouse and hen harrier, will utilise habitat mosaics of peatlands along with wet and dry upland heathland and indeed many such areas are managed primarily to maximise grouse numbers for commercial shooting.

Invertebrate assemblages on peatlands can be relatively species rich, especially for families that respond to small scale structural variation in vegetation and topography. Invertebrates on blanket bog play a key role in fragmenting plant litter as part of the peat accumulation process. Below-ground biodiversity is much less studied and the role that it plays in influencing vegetation change is little understood.

Challenges

There have been significant challenges to peatlands over the last 300 years in particular. A number of drivers cause peatland degradation. Peatlands close to industrial centres have previously suffered from SO_2 pollution whilst N-deposition has increased over the last 50 years. Both of these adversely impact on *Sphagnum* in particular, whilst water-borne nutrient-enrichment has modified conditions for lowland raised bog and fen vegetation. Peatlands have historically been drained for agriculture and more recently been used for commercial forestry. Drain construction was carried out extensively through much of the 20th century and though it has now largely ceased, its legacies of peat shrinkage and erosion remain.

Over-grazing and burning are currently the most significant ongoing activities that pose threats to blanket bog. Peatlands have a low carrying capacity for livestock and high grazing levels can suppress typical peatland vegetation. Burning impacts are poorly understood but may include adverse affects on *Sphagnum*. In lowland raised bogs, hydrological change is

the most significant threat with drainage of the bog or adjacent land lowering the water table and causing loss of vegetation and other biodiversity that depends on waterlogged conditions.

Recent construction, especially of wind farms and their associated infrastructure, have the potential to affect peatland habitats, particularly blanket bog, significantly.

Climate change may exacerbate some of the negative drivers. Wildfire will become a greater threat in a drier landscape and increased storminess may cause greater erosion. Additionally there is already evidence of mismatches that could occur in the timing of seasonal activity between predator and prey populations.

Impacts

Only 18% of the most extensive peatland type in the British Isles, blanket bog, is currently in a natural or near-natural condition. Of the remainder, 16% is eroded, 16% is afforested, 11% is affected by peat cutting and 40% is modified. The impacts are not uniform. Blanket bog in Scotland is in better condition than further south. Lowland raised bog also tends to be in a better condition further north, although the picture is more mixed. Available evidence suggests that habitats on SSSI-designated peatland sites are in better condition that on non-designated sites. Peatland species show mixed trends but a majority of those designated as part of the UK Biodiversity Action Plan have declining populations.

Peatland management

Effective peatland management for biodiversity requires a good understanding of existing environmental and hydrological conditions. Under ideal circumstances peatland hydrology and grazing livestock can be controlled. However this is often difficult. Burning is generally discouraged. Peatland restoration is a realistic option in most situations and the best results for returning peatland biodiversity will occur where the hydrology can be controlled over a wide area in order to restore a peatland to an active state. However restoration may not always achieve a natural peatland and benefits may only be seen in the long-term.

Co-ordination and dissemination of management information is important for maximizing the biodiversity potential of peatland management. Management for other benefits (e.g. carbon sequestration), if undertaken correctly, could promote typical peatland species and bring assemblage-level benefits at least in the long-term. The conservation of some species, though, may require further actions within and beyond peatland sites.

Peatland management requires long-term commitment and can be costly to the practitioner. However society must recognise that it is good value compared to the overall costs of continued peat loss. Stakeholders should input to development of funding schemes to ensure that they can be implemented to the maximum benefit of peatland habitats.

Key Points

1. Blanket bog forms the largest area in large blocks of terrestrial semi-natural habitat in the UK. It often occurs in a matrix with related habitats.

2. Peatland biodiversity is characterised by specialized species adapted to thriving in a waterlogged, mostly acidic, nutrient-poor environment.

3. The value of peatland habitats is recognised through UK and European legal obligations for their protection and restoration.

4. The peatland bird assemblages is recognised as internationally important. Many species breeding on peatlands have UK or European conservation designations and legal protection.

5. Peatlands have been subject to significant multiple negative drivers including burning, pollution, over-grazing and draining.

6. Only 18% of UK's blanket bog is now in a natural or near-natural state. The remainder is eroded, modified or has undergone land-use change (e.g. to forestry or peat extraction).

7. Biodiversity has been lost through peatland degradation. Evidence suggests that populations of many key species are in decline.

8. Restoration management has the potential to return peatland to an active state and to restore biodiversity on some sites. However reversion to a natural state with the full compliment of peatland species may be an unrealistic aim in the most degraded situations.

9. Restoration needs realistic aims and a long term approach. It should be accompanied by well planned and resourced monitoring.

10. Peatland management needs to take a flexible adaptive approach to address different drivers. Management advice should be disseminated widely.

1. Introduction: aims, scope and objectives of review

Land covered by accumulated peat is known as peatland. It is active peatland if peat is being formed now or, under certain definitions, if it still supports vegetation capable of peat formation (see glossary in JNCC report 445, JNCC 2011). Estimates of the extent of peatland in the UK vary widely depending on how peat is defined and measured but most are between 1.5 and 2.5 million ha (Lindsay 2010, JNCC 2011). Blanket bog and lowland raised bog are globally rare habitats. The UK has between 8.8 and 14.8% of Europe's peatland area (Montanarella *et al*. 2006) and about 13% of the world resource of blanket bog (Lindsay *et al*. 1988).

Peatland habitats are recognised as being a conservation priority under UK and EU law with many sites classified under the EU Habitats and Species Directive. Peatland biodiversity is typically species-poor with a large proportion of highly adapted species. These species include a range of rare, threatened or declining plants and animals. The bird assemblage is highly valued in an European context, leading to protection of large areas as Special Protection Areas. Some plant assemblages are better represented in the UK than anywhere else in the world (the best are designated as Special Areas of Conservation). However, peatland biodiversity is sensitive to changes in land management and a range of other external drivers.

The aims of this review are to highlight the importance of peatlands for biodiversity, and specifically to:

- identify biodiversity features and characteristics that are specific to peatlands;
- review progress in species and habitat conservation;
- identify where peatland management for a range of services may be beneficial to typical and valued peatland species;
- identify threats to biodiversity that arise from both external drivers and peatland management; and
- make recommendations for maximising future benefits for peatland biodiversity.

1.1 Scope and Definitions

For clarity the following definitions are adopted for use throughout (Bragg & Lindsay 2003).

- **Peat** is partly decomposed plant material that has accumulated *in situ* (rather than being deposited as a sediment) as a result of waterlogging.
- A **peatland** is an area where peat has accumulated *in situ*.
- A **mire** is an area that supports at least some vegetation known to form peat, and usually includes a peat deposit.
- A **bog** is a type of mire that is fed exclusively by precipitation (which is normally a poor source of plant nutrients)
- A **fen** is a type of mire that receives not only precipitation but also water that has been in contact with soil or rock, and so has higher nutrient status

Throughout this review a standard terminology has also been adopted (Lindsay 2010) for defining peatland topography and scale (Figure 1.1). This terminology is useful for defining the structure and function of peatlands, and is effectively a description of diversity at different scales (landscape to plant community). For example, bird assemblages are typically measured at the macrotope scale, many invertebrates respond to differences at the microtope scale and micro-organism populations will vary at the vegetation scale. All these contribute to the distinctiveness and value of peatland biodiversity

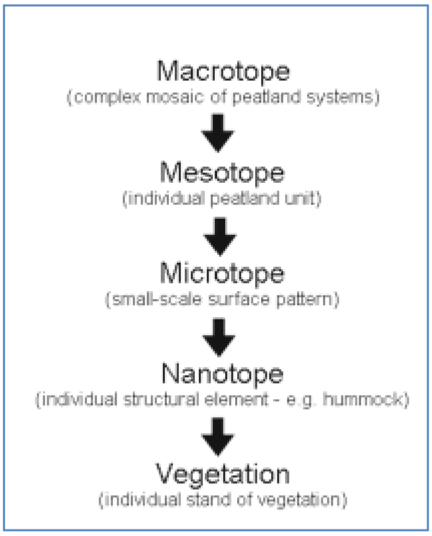


***Figure 1.1** The peatland hierarchy of elements (after Lindsay 2010). These components make up the hierarchy of building blocks and functional units within any peatland system. This is most readily seen in blanket bog landscapes (modified after Whittaker 1960). Reproduced courtesy of Richard Lindsay.*

Two or three types of mire habitat may be distinguished on the basis of water source, nutrient and mineral status and vegetation characteristics. Bog (ombrotrophic mire) obtains mineral nutrients exclusively from precipitation which, at least in unpolluted areas, is a poor source of plant nutrients. Raised bogs are discrete domed peatlands, whereas blanket bog covers entire, usually upland, landscapes though is often fragmented by features such as roads or afforestation. Fen (minerotrophic mire) receives mineral nutrients from both precipitation and water that has been in contact with soil or rock, and so has higher mineral concentrations. Fens may be acidic or calcareous depending on the water sources. Intermediate types, termed transitional mire, may also be recognised (Bragg & Lindsay 2003). These habitats do not always occur in isolation. Lowland raised bogs are a specific mesotope occurring in a variety of landscape situations. They are often part of a mire macrotope, set within a wetland landscape with other components which may include ‘lagg’ fen (a surrounding wetland area fed by surface ground water unlike the rain-fed bog), as well as blanket bog and also archaic (heavily modified) peatlands in the surrounding landscape. Many areas of upland blanket bog comprise an extensive, complex macrotope, with adjacent areas at the mesotope scale of blanket bog, wet heath, pools, flushes, springs, rock exposures and acid grassland, which may interact hydrologically.

This review focuses primarily on UK peatlands. In particular, the primary focus is on lowland raised bog and blanket bog, though it will draw on some evidence from fens and other peatland types elsewhere. Peatland is frequently defined arbitrarily as having deep peat soils with an organic layer deeper than 40 cm in England and Wales or 50 cm in Scotland (see JNCC 2011). However this review recognises that there are functional bogs, on which peatland biodiversity is represented, that fall outside this definition and these too are included where appropriate.

2. Biodiversity Values

Raised bogs, mires and fens are recognised in the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and wild fauna and flora) as of high wildlife value in a European context. There are nine different habitat types recognised as of community importance (Table 2.1), all of which occur in the UK (see JNCC 2011 for maps of the extent of peatlands in the UK).

Table 2.1 *The habitats regarded under the Habitats Directive as of community importance*

Active raised bog
Degraded raised bogs still capable of regeneration
Blanket bogs
Transition mires and quaking bogs
Depressions on peat substrate of the *Rhynchsporion*
Calcareous fen with *Cladium mariscus* and the species of the *Caricion davallianae*
Petrifying springs with tufa formation
Alkaline fens
Alpine pioneer formations of the *Caricion bicoloris-atrofuscae*

In comparison with some other habitats, peatlands tend to have relatively species-poor wildlife assemblages and simple measures of species diversity, such as species-richness, are not helpful for measuring the intrinsic value of peatland biodiversity. However, many of the typical species are highly specialised and occur in unique species assemblages within peatland habitats.

One of the most important attributes of blanket bog in particular is its extensive area. It occurs in large contiguous blocks, more so than any other habitat of value within the British Isles (for comparison, woodland cover is about 12% (Carey *et al*. 2008) but only a small proportion of this is of high wildlife value and the habitat is highly fragmented for the most part).

Only relatively few of the species occurring on peatlands are listed as UK Biodiversity Action Plan Priority Species (see Table 2.2) and most of these are not restricted to peatlands. However, peatlands are valued for biodiversity. They are included as priority habitats in the UK Biodiversity Action Plans because of the contribution they make to maintaining species diversity at the national and international level and because of the nature of the assemblages that they host. In particular peatland species are highly specialised and adapted to thriving in waterlogged, often acidic and nutrient-poor conditions.

In the UK there are significant gradients in altitude and in wetness from north and west to south and east, each of which influences the resultant species and assemblages of peatland habitats. For example, southern fens may have species that are typical of continental European wetlands but which in the UK are at the northern edge of their range. Conversely, as blanket bog is most abundant in northern upland Britain, it may host species more typical of boreal and arctic environments at the southern edge of their ranges. There are, though, features common to peatlands throughout this range.

Although the biodiversity importance of peatlands has high intrinsic value, it, together with the concomitant ecosystem functioning of peatlands, also underpins a number of ecosystem services. Crucially the vegetation is instrumental in fixing carbon within accumulating peat (which is itself a major carbon store). The whole ecosystem adds value to recreational enjoyment whether this is for bird watching or the 'wilderness' experience. In this way it is a supporting service amongst the other ecosystem services that it might provide. Add to this the importance of peatlands in water gathering catchments and in flood control, then the value of the biodiversity (a healthy, good condition ecosystem will be better able to contribute to ecosystem services) can be seen to increase significantly. In the light of the recognition of such ecosystem services (Van der Wal *et al*. in press), society needs to re-evaluate radically the value that it places on these habitats. Within such re-evaluation peatland biodiversity should be recognised for its specialisation and naturalness and its role in shaping the whole system.

2.1 Plants

As far as blanket and lowland raised bog are concerned, unmodified bog is characterised by an abundance of bryophytes, especially those of the bog moss genus (*Sphagnum*). Different *Sphagnum* species have different preferences for degree of ground wetness and hence form the characteristic hummock and pool systems creating habitat diversity at the nanotope level (see Figure 2.1). *Sphagnum* is crucial to the functioning of active peatlands and has a major role in carbon sequestration (Kivimäki *et al*. 2008). It is more resistant to decay than is vascular plant tissue and thus is the primary constituent of peat (Lindsay 2010). Furthermore, *Sphagnum*-dominated vegetation can suppress methane release far better than can vegetation dominated by vascular plans (Lindsay 2010). *Sphagnum* also plays a significant role in moderating water flow and thus reducing downstream impacts of heavy rain (e.g. Whitfield *et al*. 2009).

Sedges, such as common cottongrass *Eriophorum angustifolium*, hare's-tail cottongrass *E. vaginatum* and deergrass *Trichophorum cespitosum*, are typical of active peat. Cranberry *Vaccinium oxycoccos* and bog rosemary *Andromeda polifolia* are less common associates, whilst cloudberry *Rubus chamaemorus* forms dense patches in some places. Nutrient-poor bog conditions are the main environment utilised in the UK by a number of carnivorous plant species, such as sundews *Drosera* spp. and butterworts *Pinguicula* spp. Occurring in low density in the most waterlogged areas but more abundantly elsewhere in peatland habitat mosaics are dwarf shrubs, especially heather *Calluna vulgaris* but also cross-leaved heath *Erica tetralix*, bilberry *Vaccinium myrtillus*, crowberry *Empetrum nigrum* and others. Ombrotrophic peatlands host just a small a number of UK Priority Species of both higher and lower plants (Table 2.2) though knowledge of the range trends of these species is notably lacking.

Sixteen plant communities described in the National Vegetation Classification (NVC) (Rodwell *et al*. 1991) may be associated with blanket bog and lowland raised bogs (Table 2.3). High quality blanket bog supports distinctive plant communities with well-defined microtopographical variation on the bog surface and a two-layered peat profile. Active blanket bog may be characterised by expanses of vegetation with affinities to the NVC communities M17–19 where cottongrasses predominate along with deergrass and a constant presence of bog asphodel *Narthecium ossifragum* and tormentil *Potentilla erecta* (M17) or hare's-tail cottongrass and heather (M19), which become more widespread eastwards in Britain. The *Sphagnum*-rich M18 community provides a broad overlap between these two main communities while the bog pool communities M1–M3 occur in localised wetter areas. On shallower peat M15 or M16 communities may be present. Fens encompass a far wider range of communities from those that represent a gradation from bog communities, especially those that are sedge-dominated, through to distinctive habitats such as reedbeds dominated by common reed *Phragmites australis*.

Peatland habitats feature prominently within UK and international habitat designations. The UK Biodiversity Action Plan lists Lowland Raised Bogs, Blanket Bog and Lowland Fens as Priority Habitats. At the European level, the best examples of certain habitats are eligible for designation as Special Areas of Conservation which confers protection under the EC Habitats Directive (92/43/EEC).

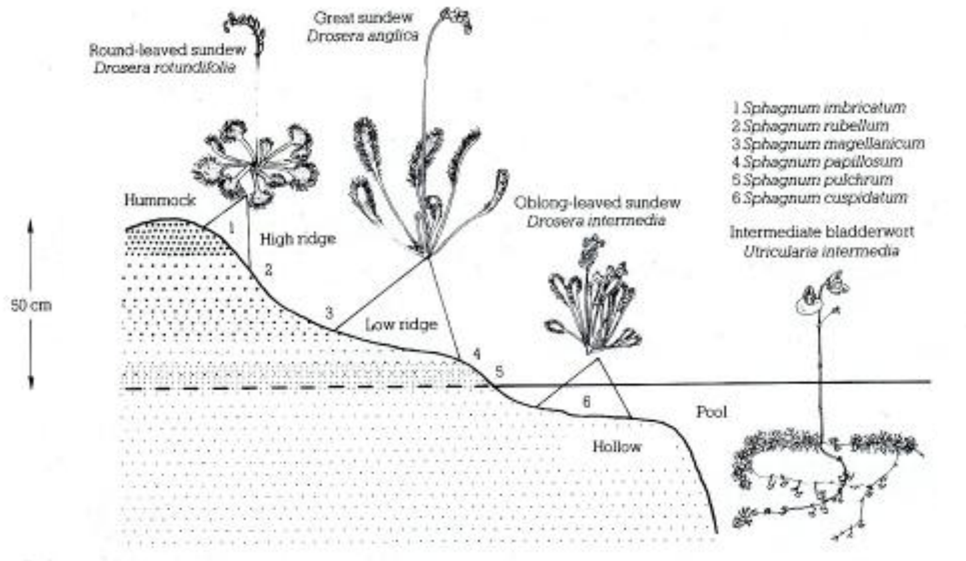


Figure 2.1 *The zonation of vegetation types within the microtopography of a bog. It can be seen here that different Sphagnum species occupy distinct vertical ranges within the typical microtopographic structures found on a bog, as indeed do the various carnivorous plants found in such habitats. Figure reproduced courtesy of Richard Lindsay (2010).*

Table 2.2 *UK BAP priority species occurring on ombrotrophic bogs. ■: occurs; ?: possible habitat. [1]Trends given are for those from the 2008 BAP reporting round (or, where stated, the 2005 reporting round). No trend information yet exists within the BAP system for species that were added to the list of UK Priority Species in 2007. Additional sources used are: [2] breeding population trends for 1995-2008 (Risely et al. 2010); [3]population trend for 1978–2004 and distribution trend for 1970–82 vs. 1995–2004 (Fox et al. 2006a); [4]population trend for 1978–2002 (Fox et al. 2006b); [5]population trend for 1995–2008 (Risely et al. 2010).*

Group	**English name**	**Scientific name**	**Blanket Bog**	**Lowland Raised Bog**	**Trend**
Amphibians	Common Toad	*Bufo bufo*	■	■	
Ants	Black Bog Ant	*Formica candida*		■	
Beetles	Blue Ground Beetle	*Carabus intricatus*		■	Stable[1]
	10 Spotted Pot Beetle	*Cryptocephalus decemmaculatus*		■	
	Mire Pill Beetle	*Curimopsis nigrita*		■	Declining slowly[1]
	a ground beetle	*Pterostichus aterrimus*		■	No clear trend[1]
Birds	Sky Lark	*Alauda arvensis*	■	■	Declining (slowing)[1]; 11% decline (significant)[2]
	Greenland White-fronted Goose	*Anser albifrons* subsp. *flavirostris*	■		
	Nightjar	*Caprimulgus europaeus*		■	Increasing[1]
	Twite	*Carduelis flavirostris*	■		
	Black-throated Diver	*Gavia arctica*	■		
	Red Grouse	*Lagopus lagopus*	■		9% decline (non-significant)[2]
	Grasshopper Warbler	*Locustella naevia*	■	■	24% increase (non-significant)[2]
	Woodlark	*Lullula arborea*		■	Increasing[1]
	Common Scoter	*Melanitta nigra*	■		Declining (continuing/accelerating)[1]
	Curlew	*Numenius arquata*	■		42% decline (significant at p<0.05)[2]
	Red-necked Phalarope	*Phalaropus lobatus*	■		
	Arctic Skua	*Stercorarius parasiticus*	■		
	Black Grouse	*Tatrao tetrix*	■	■	
Butterflies	Large Heath	*Coenonympha tullia*	■	■	26% population decline, 43% distribution decline[3]
Craneflies	a cranefly	*Tipula serrulifera*	■		

Flowering plants	Juniper	*Juniperus communis*	■		Declining (continuing/accelerating)[1]
Fungi	Agaric	*Armillaria ectypa*	■		No clear trend[1]
	Lousewort Rust	*Puccinia clintonii*		■	
Grasshoppers & Crickets	Large Marsh Grasshopper	*Stethophyma grossum*		■	Stable (2005 reporting round)[1]
Hoverflies	Bog Hoverfly	*Eristalis cryptarum*	■		Fluctuating – probably stable
Liverworts	Marsh Earwort	*Jamesoniella undulifolia*		■	Status unknown[1]
Mammals	Water Vole	*Arvicola terrestris*	■		Fluctuating - probably declining[1]
	Mountain Hare	*Lepus timidus*	■		29% decline (not significant)[5]
	Otter	*Lutra lutra*	■		Increasing[1]
	Soprano Pipistrelle	*Pipistrellus pygmaeus*		?	Fluctuating - probably stable[1]
Mosses	Carrion-moss	*Aplodon wormskjoldii*	?		
	Waved Fork-moss	*Dicranum bergeri*	■		
	Baltic Bog-Moss	*Sphagnum balticum*	■	■	Stable[1]
Moths	Haworth's Minor	*Celaena haworthii*	■		80% decline[4]
	Argent and sable	*Rheumaptera hastata*	■	■	Declining (slowing)[1]
Reptiles	Adder	*Vipera berus*	■		
Spiders	A money-spider	*Erigone welchi*		?	
	A money-spider	*Notioscopus sarcinatus*	■	■	
	A money-spider	*Saaristoa firma*	■	■	
	A money-spider	*Semljicola caliginosus*	■		

Table 2.3 *Plant communities of ombrotrophic peatlands identified in the British National Vegetation Classification (Rodwell et al. 1991). State indicates if the community is usually associated with semi-natural or modified peatlands or if it is a wet heath community that can also occur on peatlands.*

	NVC class	**Description**	**State**
Mires & Wet Heath	M1	*Sphagnum auriculatum* pool	Semi-natural
	M2	*Sphagnum auriculatum/recurvum* bog pool	Semi-natural
	M3	*Eriophorum angustifolium* bog pool	Semi-natural
	M15	*Scirpus cespitosus - Erica tetralix* wet heath	Semi-natural
	M16	*Erica tetralix - Sphagnum compactum* wet heath	Semi-natural
	M17	*Scirpus cespitosus - Eriophorum vaginatum* blanket mire	Semi-natural
	M18	*Erica tetralix - Sphagnum papillosum* raised and blanket mire	Semi-natural
	M19	*Calluna vulgaris -Eriophorum vaginatum* blanket mire	Semi-natural
	M20	*Eriophorum vaginatum* blanket and raised mire	Degraded
	M21	*Narthecium ossifragum - Sphagnum papillosum* valley mire	Semi-natural
Heath	H9	*Calluna vulgaris - Deschampsia flexuosa* heath	Degraded
	H10	*Calluna vulgaris - Erica cinerea* heath	Wet heath
	H12	*Calluna vulgaris - Vaccinium myrtillus* heath	Degraded
	H16	*Calluna vulgaris - Arctostphylos uva-ursi* heath	Wet heath
	H21	*Calluna vulgaris - Vaccinium myrtillus -Sphagnum capillifolium* heath	Wet heath
Upland	U6	*Juncus squarrosus - Festuca ovina* grassland	Degraded

2.2 Birds

Blanket bogs, lowland raised bogs and fens all hold distinct bird assemblages. However it is the breeding bird assemblage of UK blanket bogs that is really outstanding in an European and global context. None of the constituent birds are obligate peatland species although breeding greenshank *Tringa nebularia* and dunlin *Calidris alpina* are largely confined to peats and wet heather-dominated moors. Most species will utilize a range of upland habitats (see, for example, Pearce-Higgins & Grant, 2006), but within blanket bogs, the wettest habitats (open pools and un-drained bog) are utilised by wildfowl and dunlin, areas of tall vegetation in bogs (and in mosaics with upland heathland) provide cover and nesting sites for black grouse *Lyrurus tetrix*, curlew *Numenius arquata*, greenshank, hen harrier *Circus cyaneus* and merlin *Falco columbarius*, whilst golden plover and redshank *Tringa totanus* and skylark *Alauda arvensis* preferentially select shorter vegetation (e.g. Pearce-Higgins *et al*. 2009a).

Breeding birds on peatlands include a very high proportion of species that are covered by conservation designations. These are listed in Table 2.4 and include fourteen EC Annex 1 species (Stroud *et al*. 1987), thirteen species on the Red List of Birds of Conservation Concern for the UK and fifteen on the Amber List (Eaton *et al*. 2009), thirteen UK Biodiversity Action Plan Species and fourteen Schedule 1 species of the UK Wildlife and Countryside Act.

Table 2.4 *Breeding bird species on UK peatlands and their conservation status. Annex 1: European Commission Birds Directive; BoCC: Birds of Conservation Concern Red and Amber lists; W&C: Wildlife and Countryside Act 1981, Schedule 1. This list is not exhaustive and further breeding species may make some use of peatlands. See also Table 2.2.*

		Annex 1	**BoCC Red**	**BoCC Amber**	**UK BAP**	**W&C**
Common Scoter	*Melanitta nigra*		■		■	■
Red Grouse	*Lagopus lagopus*			■	■	
Black Grouse	*Tetrao tetrix*	■	■		■	
Red-throated Diver	*Gavia stellata*	■		■		■
Black-throated Diver	*Gavia arctica*	■		■	■	■
Hen Harrier	*Circus cyaneus*	■	■			■
Golden Eagle	*Aquila chrysaetos*	■		■		■
Merlin	*Falco columbarius*	■		■		■
Peregrine	*Falco peregrinus*	■				■
Golden Plover	*Pluvialis apricaria*	■		■		
Temminck's Stint	*Calidris temminckii*		■			■
Dunlin	*Calidris alpina*		■			
Ruff	*Philomachus pugnax*	■	■			■
Whimbrel	*Numenius phaeopus*		■			■
Curlew	*Numenius arquata*			■	■	
Common Sandpiper	*Actitis hypoleucos*			■		
Greenshank	*Tringa nebularia*					■
Redshank	*Tringa totanus*			■		
Wood Sandpiper	*Tringa glareola*	■		■		■
Red-necked Phalarope	*Phalaropus lobatus*	■	■		■	■
Arctic Skua	*Stercorarius parasiticus*		■		■	
Great Skua	*Stercorarius skua*			■		
Common Gull	*Larus canus*			■		
Short-eared owl	*Asio flammeus*	■		■		
Nightjar	*Caprimulgus europaeus*	■	■		■	
Woodlark	*Lullula arborea*	■		■	■	■
Skylark	*Alauda arvensis*		■		■	
Grasshopper Warbler	*Locustella*		■		■	
Twite	*Carduelis flavirostris*		■		■	
Reed Bunting	*Emberiza schoeniclus*			■	■	

Of cultural importance and also of conservation and economic significance are populations of red grouse of the sub-species endemic to UK and Ireland, *Lagopus lagopus scoticus*. These largely occupy upland heathland rather than peatlands but will make some use of blanket bog (and to a lesser extent lowland raised bogs). Shooting of these birds for sport provides a commercial return in northern upland regions and substantial areas of dwarf shrub heath are managed to maximise their densities. Management of bog areas for this species by regular, repeated burning may lead to habitat degradation through an increase in heather dominance at the expense of more typical bog plants. The most successful populations of grouse are those which utilise wet areas such as bog flushes, especially in summer months, when chicks feed especially on soft-bodied invertebrates, such as craneflies *Tipulidae* (Park *et al*. 2001). Management of estates for red grouse can help to maintain at least some aspects of biodiversity (e.g. Thompson *et al*. 1997). However there has been a suggestion that illegal persecution of raptors on some estates, as a result of perceived conflicts between breeding hen harriers and driven grouse shooting, may be associated with regional declines in some raptors (e.g. Sim *et al*. 2007).

2.3 Other Vertebrates

The more important mammals of peatlands are populations of wild deer (red deer *Cervus elaphus* in the uplands and on some lowland raised bogs and fallow deer *Dama dama* on lowland fens within a landscape where there are woodland refuges). Some introduced species are becoming well established in lowland habitats, in particular sika deer *Cervus nippon* in the New Forest, and their activities can be damaging to peatlands. Wild deer have few natural predators and populations can increase if not managed adequately. This can result in damage to peat. For example, in many areas in Scotland heavy trampling causes bare patches which can further erode in exposed locations, such as happens in the Monadhliath Mountains (Anon 2008c).

Mountain hare *Lepus timidus* is the other distinctive upland mammal that occurs on blanket bog in Scotland and (re-introduced) in the Peak District. The Irish hare *Lepus timidus hibernicus* is a distinct subspecies and is endemic, making it of high conservation interest (Corbet & Harris 1991). Mountain hares are listed in Annex V of the Habitats Directive as a species 'of community interest whose taking in the wild and exploitation may be subject to management measures'. There is a further pressure to limit numbers where hares carry ticks, which carry viruses that affect grouse and sheep (e.g. Harrison *et al*. 2010).

Peatlands, particularly ombrotrophic bogs provide good habitats for adders *Vipera berus* whilst common lizards *Zootoca vivipara* and grass snakes *Natrix natrix* are common in lowland sites, particularly where there are amphibians present. Of the latter, smooth *Lissotriton vulgaris* and palmate newts *Lissotriton helveticus* and common frog *Rana temporaria* are the most readily encountered and the habitat is important for these species where they have declined elsewhere.

2.4 Invertebrates

Far less is known about peatland invertebrates than about larger animals or plants. However they form a very significant element of peatland biodiversity. For example the numbers of invertebrate species on peatlands may be up to 30 times higher than vertebrates and their biomass an order of magnitude larger (Coulson *et al*. 1995). In particular, upland blanket bog supports high numbers of enchytraeid worms, mites, spiders and bugs with a high biomass of flies, bugs, mites and springtails (summarised by Tallis 1998).

The true interest of peatlands for invertebrate conservation *per se* is very variable. Lowland fens, particularly those with good water quality and hence a diverse vegetation structure, can be very rich in diversity and support a high number of specialist species. This is particularly true of sites with relatively stable water levels. Small pools with emergent vegetation and muddy edges of wetlands can also be very important. Fens composed entirely of *Phragmites*, which are generally of poor water quality, tend to have a poor insect fauna, whilst sites with varying water tables have an invertebrate fauna more similar to a floodplain pond. Lowland bogs are richer in invertebrate species than blanket bogs, but in some cases this is because of more diverse edge habitats and species associated with encroaching trees and scrub. However many lowland peatland sites have dried out and often, in the case of valley mires, the invertebrate fauna is regularly considered as part of the surrounding heathland system. As a result, the invertebrate faunas of lowland peatlands, in particular, tend to be overlooked and underestimated. Upland bogs are often poor in overall species richness with a high invertebrate biomass. However, they do also often support specialist species of high biodiversity interest, particularly within *Sphagnum* ponds and around the edges of flushes.

Invertebrate groups that are species-rich on peatlands tend to be those that are more dependent on vegetation structure than species composition. These include spiders (Scott *et* *al*. 2006), especially money spiders (Lyniphidae) (Coulson & Butterfield 1986) and ground beetles (Usher 1992). Indeed in addition to the UK Priority Species (Table 2.2) three spiders with Red Data Book (RDB) status, *Carorita limnaea*, *Maro lepidus* and *Sitticus floricola*, are confined solely to blanket bog (Anon 2010). Variation in hydrology is also a driver of diversity in invertebrate assemblages. This may be through directly altering ground saturation (Coulson *et al*. 1990) or less direct effects through influences to foodplant availability for phytophagous species (Fowles *et al*. 2004). Bog pools on both upland and, especially, lowland sites are particularly important for dragonflies and damselflies. For two thirds of Britain's 38 species, bogs are among the habitats utilised. Eleven of these species are virtually restricted to peatland habitats in this country of which seven are regarded as rare or local in Britain (Brooks 1997).

Coulson & Butterfield (1978) consider that blanket bog invertebrates play an essential role in the initial fragmentation of plant litter (termed 'comminution', Swift et al., 1979), prior to fungal and bacterial attack, which plays a part in the peat accumulation process. Furthermore peatland invertebrates have an important role as prey for other peatland fauna (e.g. Buchanan *et al*. 2006). Many species are in their adult stage in early summer and in large numbers, thus representing important prey items for breeding birds. For example the peak emergence of craneflies in late May/early June is significant for wading birds and their chicks as well as for red grouse and other birds (e.g. Pearce-Higgins *et al*. 2005).

The effect of peatland management on invertebrates is generally little understood. There are some studies that have focussed on key species (e.g. Fowles *et al*. 2004) but in most cases invertebrates are not the driving force behind peatland management decisions. There is, though, concern that management for other services, such as carbon sequestration, might be detrimental to some invertebrates. In particular the practise of re-wetting, through installation of dams, may be detrimental to rare and priority species. This may be because changes in the water table may make conditions unsuitable for species that are present whilst the relative speed of such changes may preclude their moving elsewhere, if indeed suitable conditions remain at all in the vicinity (Verbeck *et al*. 2006). Furthermore restored habitat from rewetting management may lack the surface heterogeneity of unmodified bogs (Verbeck *et al*. 2010) whilst artificially created habitat such as bog pools can experience very slow colonisation of invertebrates. This can lead to an impoverished fauna compared to what was present before the onset of restoration management, especially where such peatlands are isolated (Mazerolle *et al*. 2006; Mazerolle & Poulin 2007; van Duinen *et al*. 2007). Most research of rewetting effects on invertebrates has been carried out overseas and their remains a need for investigation of impacts of management on rare or priority invertebrate species within the UK.

Few peatland invertebrates are afforded legal conservation status although at least 15 species that are regularly associated with ombrotrophic peatlands are UK Biodiversity Action Plan Priority Species (see Table 2.2) and many others are likely to make at least some use of such sites. Lowland raised bogs host more UK Priority Species (11) than blanket bog (8). A large range Red Data Book species have also been identified from peatlands with at least 22 known from lowland raised bog and 17 from blanket bog. Fens are especially rich in this context and host at least 81 Red Data Book invertebrate species (Anon 2010).

2.5 Microbial Biodiversity

Below-ground peatland biodiversity is a much neglected topic despite evidence that the microbial community plays an important role in the functioning of peatlands. It seems straightforward to assume a linkage between the microbial community composition, the activity of the relevant microbiota within it, and net ecosystem carbon exchange, as the microbiota are the predominant processors of decaying organic matter and thus represent the bottleneck in the extent to which ecosystem respiration returns C to the atmosphere in peatlands. However, methodological constraints as well as limitations in our understanding of the functions of soil biota have made finding evidence for this hypothesis rather elusive. A review of the literature suggests that the taxonomic microbial community composition of peatlands is closely linked to vegetation assemblages and will change in parallel with the vegetation during succession (Artz, 2009 *and therein*). This evidence is strongest for a link between peatland vegetation composition and the methanogenic archaeal community composition (Galand *et al.* 2005; Rooney-Varga *et al.* 2007). However, significant relationships have been also found to be in evidence for the wider, bacterial (Morales *et al*. 2006; Opelt *et al*. 2007) and fungal communities (Thormann *et al.* 2006; Artz *et* al. 2007). Finding direct functional relationships of microbial biodiversity within ecosystem functioning has been more elusive. Although a relationship between the succession stage of vegetation in peatlands and the physiological capacity to decompose a variety of simple carbon compounds has been established (Artz *et al.* 2008), ascribing key roles within the carbon cycle in peatlands to particular members of the soil microbiota has been one of the crucial research questions in soil microbiology for decades. For methanogenic communities and net methane fluxes, some direct relationships in peatlands have indeed been found (e.g. Hines *et al.* 2008), however our understanding of the functions of many other microbiota is still rather limited as scientific technology has predominantly focused on taxonomic rather than functional understanding of soil microbiota. Similarly, many of the techniques for cataloguing taxonomic microbial diversity are currently limited and much remains to be learnt about the role of microbiota in influencing, for example, success of peatland vegetation restoration.

3. Drivers of Change

3.1 Drainage

Drainage or 'gripping' of peatlands has been carried out particularly on blanket peats (Wallage *et al.* 2006) in the past primarily for purposes of grazing or game management but also to direct water flows into reservoirs, to drain major pool systems and to prepare ground for tree planting. About 20,000 ha/year of largely blanket bog was drained in the 1960s and 1970s (Stewart & Lance 1983) funded by Ministry of Agriculture grants (70% of cost), although grant aid ceased in 1985. Past drainage has degraded peatland mesotopes by causing localised drying and disrupting overland flows. This causes, in particular, reductions in plants that are highly dependent on waterlogged conditions such as cottongrasses and *Sphagnum capillifolium* (Stewart & Lance 1991) although long-term impacts of draining are not well understood. Even relatively small changes in the water table can have a significant effect on the species composition and particularly on the nature of the primary peat forming *Sphagnum* species. Drainage has also led to peatland erosion, especially where drains are on slopes, as well as to an increase in suspended sediment and dissolved organic carbon in water flows (Holden *et al.* 2007b). Draining of peatlands has now been demonstrated to be of limited actual value for grazing or game management (Holden *et al*. 2007a; Stewart & Lance 1991) with the potential to encourage spread of unpalatable grasses (Coulson *et al*. 1990) and must be regarded as inappropriate on peatlands.

Lowland raised bogs have also been extensively drained, but this was generally related more to peat extraction or afforestation. It was less systematic but equally damaging to drainage on blanket bogs. Lowland raised bogs have also been extensively affected by drainage of surrounding land for agriculture, which may cause interfere with bog hydrology and cause drying of the peat. In some cases direct bog drainage has been carried out to improve conditions for livestock rearing (Anon 1999).

Reversal of former drainage management, by drain-blocking, is frequently successful at stalling erosion, reducing sediment run-off, moderating flood peaks and facilitating re-vegetation (e.g. Grayson *et al*. 2010). However research primarily carried out in the Peak District has shown that success tends to be higher within smaller drains on shallow slopes (Evans *et al*. 2005). Severely eroded sites and drains on steep slopes can prove resistant to restorative management (Armstrong *et al*. 2009) and this may especially inhibit formation of peat-forming mosses such as *Sphagnum capillifolium* and other plants that similarly rely on waterlogged conditions. Drain-blocking is a relatively recent management practice with a significant rise in its application in recent years on blanket bogs (Armstrong *et al*. 2009) and also in lowland raised bogs where re-wetting is a primary objective (for example Hatfield and Thorne Moors, Solway Mosses and Fenns and Whixhall Mosses). Whilst drain-blocking necessarily concentrates on restoring peatland processes, the potential of the activity to reinstate peatland biodiversity has not generally been investigated. For further information on peat draining see JNCC (2011), Lunt *et al.* (2010) and Natural England (2010).

3.2 Forestry

Tallis (1998) gives an estimated 3,500 km^2 of blanket bog in Britain and Ireland that is afforested, or 16% of the total area. Afforestation on peatlands requires deep-ploughing and draining. This leads to long-term erosion, shrinkage, deep-cracking and oxidation both within and beyond the plantation area. The planted region loses peatland vegetation when the forest closes to thicket after 10-15 years and the trees dry out the bog (e.g. Stroud *et al*. 1987). The impact on peatland fauna may also extend some distance beyond the planted area with, for example, Hancock *et al*. (2009) demonstrating negative affects on either population density or trends close to forest edge for dunlin and red grouse. Hence afforestation destroys peatland habitats and impacts on peatland species at the mesotope or macrotope scale.

Whilst the most extensive areas of peatlands that have been damaged and destroyed by afforestation have been in the uplands, lowland raised bogs have also been impacted with 17% of the area of this habitat in England having been planted (Natural England 2010). In recent years, restoration management has entailed felling of plantation woodland. Anderson (2010) showed that, on blanket bogs, this achieves most success, in terms of raising the water table, when trees are felled (either removed or left to waste) and plough furrows are dammed. In the same work, on lowland raised bogs, whole tree removal produced the best water table results though with little difference between treatments where furrows were left open or dammed. As well as promoting reinstatement of peatland vegetation, such felling of woodlands on lowland raised bogs has benefited priority bird species such as nightjars *Caprimulgus europaeus* and woodlarks *Lullula arborea* (Conway *et al*. 2007; Conway *et al*. 2009; Langston *et al*. 2007).

Government policy currently favours significant expansions in UK woodland cover over future decades. The Scottish Forestry Strategy, for example, calls for an increase of woodland cover from 17% to 25% by the second half of this century (Anon 2006). It is important that this expansion is made away from deep peat sites and indeed there is now acknowledgment within the forestry sector of the need to conserve blanket and raised bogs and to remove trees where they have previously been planted on such ground (Anon 2006).

3.3 Cutting/Extraction

Peat cutting can take the form of small scale operations for domestic use or operations at the more industrial-scale. The use of peat for domestic fuel occurs principally in far northern and north-western parts of the UK (see Figure 4.2d) and although impacts vary between regions it has been estimated that around 7% of Scotland's blanket bogs show some sings of cutting (Coupar *et al*. 1997). Industrial peat extraction, especially for horticultural use, has much greater effects, impacting at the mesotope or macrotope scale. In 2009 some 0.94

million m^3 of peat was supplied to the UK horticultural industry from UK sources with around 2 million m^3 supplied from elsewhere, mostly from Ireland and the Baltic states (Defra 2010).

Peat extraction results in drying and loss of the peat mass, loss of surface vegetation and trampling/compaction of access routes. The long-term impact on peatland biodiversity will differ between sites but in many cases there will be changes away from assemblages typical of active peatlands (e.g. Tallis 1998). However some formerly cut-over peatlands may still be rich in biodiversity (e.g. Cooper *et al*. 2001; Hulme, 2006). For further information, see JNCC (2011).

3.4 Grazing

There is a polarisation between many lowland raised bogs where there has been insufficient grazing, which, coupled with drainage, has resulted in significant scrub and woodland invasion, for example Holme Fen near Peterborough (Marrs, 1984), and blanket bogs where grazing has tended to be too high for many years. High grazing levels by domestic livestock have long been recognised as a driver of upland vegetation change, for example from dominance of dwarf-shrubs to grasses or cottongrasses, at the macrotope scale. It can be difficult, however, to separate the effects solely of high levels of grazing from those of regular burning plus grazing.

Blanket bogs are usually set within a wider landscape of moorland habitats grazed as a single unit. There is little to attract grazing animals to blanket bogs. Some bog species, such as the ‘mosscrop’ of cottongrass flower buds, are a significant attractant in early spring but most other species are not particularly palatable or attractive as forage (Yalden 1981). The amount of grazing on the blanket bog is therefore more a function of the degree of availability of more palatable vegetation elsewhere within the foraging area and the number of animals (domestic livestock or wild deer).

The impact of livestock grazing on blanket bog species tends to be negative when grazing occurs all year and at a high level. Biomass is reduced, dwarf shrubs are reduced in cover and structure (Grant *et al*. 1985), some species such as bog asphodel and cloudberry can become scarce and various mosses disappear. Blanket bogs have a lower carrying capacity than most other upland plant assemblages in livestock terms. For example on blanket bogs in the Tees Valley, northern Pennines, a grazing density of just one sheep per 4 acres (= 0.62 sheep ha^{-1}) suppressed dwarf-shrubs and facilitated increased proportions of graminoid species such as purple moor grass *Molinia caerulea* and heath rush (Welch & Rawes 1966). Furthermore high grazing levels, especially if coupled with burning, can exacerbate problems of erosion (Yeloff *et al*. 2006, Worrall *et al*. 2010b). Increases in sheep numbers, the emergence of new hardy varieties, the use of winter supplementary feeding, changes in sheep management (such as removal of wethers from the uplands) and the removal of cattle from many peatland and other moorland sites have all been factors in stock management that have resulted in reduced plant diversity (e.g. Tallis 1998).

Since the late 1990s there has been a reversal of the trend of over-grazing in at least some regions. Although detailed figures for grazing on peatlands are not available, there has been a steady decline in UK sheep numbers since 2000. For example, in Scotland, there has been a reduction in sheep numbers from almost 10 million in 1998 to less than 7 million in 2009 (RERAD 2010) and some upland areas, where peatland habitats are best represented, are being abandoned altogether for grazing of domestic livestock. Whilst there is some research on the impacts of livestock removal from upland heathland, less is known about the processes acting following the cessation of domestic grazing on peatlands. However Rawes (1983) showed an increase in heather and crowberry as well as cloudberry and bog asphodel within fenced blanket bog plots at Moor House National Nature Reserve when grazing was removed for 15 years. At the same site Marrs *et al*. (1988) demonstrated an accelerated trend from heath rush-dominance towards heather and hare's-tail cottongrass with release of grazing pressure. Similar results were reported from blanket bogs in SW Scotland under a reduction in grazing levels (Grant *et al*. 1985). Contrasting results were found from experimental enclosures at Butterburn Flow in the Border Mires where cessation of sheep grazing lead to a loss of plants typical of wet ombrotrophic conditions, especially towards the bog edge, and an increase in species generally associated with drier moorland vegetation (Smith *et al*. 2003).

The impact of wild herbivores, such as red deer, on peatlands are very little studied. There have been some studies on upland heathland (e.g. Grant *et al*. 1981) that have shown that high stocking levels (>2 hind equivalents ha^{-1}) have led to a diminution of heather and replacement by graminoids and similar effects may be expected at least on those peatlands that have a reasonable cover of dwarf shrubs. It is likely that deer, sheep and other herbivores interact, possibly with compensatory processes especially when numbers of sheep are reduced. However the characteristics of such processes on peatlands are little understood.

3.5 Burning/Muirburn

Prescribed burning is controlled by legislation and it is generally not allowed in the late-spring and summer months (Anon 2007; Anon 2008a; Anon 2008b). It is carried out primarily to remove the surface vegetation and litter, after which the vegetation regenerates. Although it is a practice especially associated with upland heathland (on grouse moors) it has long also been a part of peatland management. Farey (1815) for example, describes shepherds taking their tinder boxes out for the day on horseback, and some of the resultant fires burning for weeks, resulting in peat collapse in the blanket bogs of Derbyshire. This would largely have been to provide a flush of new growth for sheep grazing, but the signature in peat could also be wildfire which has affected many peatlands severely. On many peatlands there is evidence of some burning throughout their existence though burning activity has increased considerably since the Industrial Revolution (Yeloff *et al*. 2006; Chambers *et al*. 2007).

Prescribed burning can significantly alter vegetation assemblages and reduce the amplitude of surface patterning features (Hamilton *et al*. 1997). There is conflicting evidence on the form that such changes to the vegetation take. For example *Sphagnum austinii* was formerly a major part of some peatland systems at the mesotope or macrotope scale (Chambers *et al*. 2007) but its demise in the peat record coincided with an increase in burning activity. Its revival in recent decades at Cors Fochno, Wales, was linked to the control of burning activity on that reserve (Bailey 2003). However Shaw *et al*. (1997) found no firm evidence that managed burning results in long term damage to *Sphagnum* cover of blanket bogs. It has also been argued that prescribed burning leads to a dominance of heather and a reduction in other species (McVean & Ratcliffe 1962) although a systematic review of the impacts of prescribed burning on blanket bogs found that there was a tendency for burning to cause an increase in bryophytes and bare ground and a switch from ericoids to graminoids (Stewart *et al*. 2005).

Burning is currently discouraged on blanket bogs, particularly where these are still rich in mosses (largely *Sphagna*) (Anon 2007). On those sites that are heather-dominated a longer burning cycle may be agreed. However heather-dominance on blanket bogs itself is a sign of a degraded and modified bog (Chambers *et al*, 2007) often as a product of repeated burning (Yallop *et al*. 2006) for grouse management. Holden *et al.* (2007c) also suggests that the dominance of heather is correlated with elevated Dissolved Organic Carbon production levels from the peat and thus a degeneration of the peat carbon store and reduced water quality. Thus continued burning may perpetuate this modified state and greater scientific scrutiny on how to reduce heather dominance on peat is required.

Unburned, heather-dominated, peatlands may be at greater risk from wildfire. Wildfire is either unplanned, often unintentional fire, usually outside the legal burning season, which can be much more extensive than any managed fire. It usually occurs after periods of persistently dry and often hot weather (Anderson 1986). It may also occur within the legal burning season and result from prescribed fires getting out of control. Wildfire has a greater risk of damaging the root mat after which the peat can become exposed. Overland flow increases on bare peat, rills and then gullies can form and in the worse cases, extensive bare gully systems can develop as in many areas in the Peak District and South Pennines (Phillips *et al*. 1981). Blanket bogs are more vulnerable than any other moorland habitat to severe and long lasting damage from wildfire (McMorrow *et al*. 2005). Moreover, recovery from such damaged states can take an extremely long time (Maltby *et al*. 1990; Anderson *et al*. 1997). For more information on the effects of burning on peatlands, see Worrall *et al.* (2010b).

3.6 Pollution

Emissions of sulphur dioxide (SO_2) linked with heavy use of fossil fuels have impacted peatlands at the mesotope scale since the onset of the industrial revolution. There is a disproportionately affect on peatlands downwind of areas of heavy industry. *Sphagnum* especially is vulnerable to SO_2 pollution (Baxter *et al*. 1991) and the Peak District and South Wales, in particular, suffered its disappearance in the mid 19th century linked to emissions of SO_2 from nearby centres of industry (Yeloff *et al*, 2006). Pollution may act in an even more substantial way than simply altering the vegetation composition. The onset of erosion in Pennine peatlands has been linked to this loss of the *Sphagnum* layer (e.g. Tallis 1998; Yeloff *et al*. 2006) and its absence may inhibit colonisation by other plants of bare ground, such as after wildfire. The long term effect of SO_2 pollution on the peats near industrial centres has been a reduction in pH to as low as 2.8 (Anderson *et al*. 1997). It is impossible to restore vegetation onto bare peat with pH this low without adding lime and fertiliser to raise levels to within the more normal range.

Global production and emission of reactive nitrogen has increased substantially over the last 200 years (Galloway & Cowling 2002). Peatland vegetation is generally oligotrophic (adapted to low nutrient conditions). Elevated nitrogen levels will impact differently on different plant species leading to a change in vegetation composition. The growth of *Sphagnum* spp., crucial to peat accumulation, may be inhibited by nitrogen- (N) deposition as increased nitrogen levels in plant material can lead to tissue breakdown and death (Limpens & Berendse 2003; Phuyal *et al*. 2008). Although atmospheric nitrogen levels have reduced in recent years, the sensitivity of species, such as the peat-forming *Sphagnum capillifolium*, to even low levels of deposition (e.g. Gunnarsson & Rydin 2000) means that N-deposition will continue to pose an ongoing threat to bogs (Sheppard *et al*. 2008).

N-deposition may further impact on peatland vegetation by increasing insect herbivory. In some cases this can lead to “outbreaking” of insect populations and the associated decimation of foodplants. For example heather may respond to enhanced nitrogen availability with increased foliar nitrogen levels and increased growth (although with reduced tolerance to stressors such as frost) (Carroll *et al*. 1999). Recent years have seen significant rises in ‘outbreaking’ populations of heather beetle *Lochmaea suturalis* with resultant defoliation and vegetation change (e.g. Rosenburgh & Marrs 2009).

Due to their ombotrophic nature, well-managed bogs are not prone to water-borne pollution or enrichment. Fens, on the other hand, can be vulnerable to run-off and other sources of diffuse pollution. Fen vegetation is strongly nitrogen-limited and is therefore at risk from even low levels of enrichment, a problem particularly associated with agricultural intensification (McBride *et al*. 2010).

3.7 Construction and Development

The most frequent construction works on peatlands are windfarms and communications masts. Associated infrastructure, such as access tracks and foundations, can interfere with peatland hydrology, thus altering vegetation at the microtope and possibly mesotope scale. Construction can lead to significant areas of peat disturbance and bare ground and, due to hydrological interference, typically affects an area considerably larger than the footprint of the development itself. Furthermore some bird species actively avoid wind turbines. For example golden plover show an avoidance effect at up to 200 m from turbines (Pearce-Higgins *et al*. 2008) and breeding densities of this and other key species present on peatland, including hen harrier and curlew, may be depressed in the proximity of wind farms (Pearce-Higgins *et al*. 2009b). This is especially an issue on blanket bogs in northern Britain due to the disproportionate number of wind farms located there relative to other environments. In Scotland for example, 55% of wind farms installed up to 2007 were on peatlands (Bright *et al*. 2008b) and, together with planned installations, they have the potential for a significant cumulative detrimental impact on peatland birds, peatland habitat, water quality and runoff characteristics (e.g. Bright *et al*. 2008a).

3.8 Restoration Management

Peatland restoration typically involves managing areas to reinstate peatland function and biodiversity. This may involve reversal of several of the above mentioned drivers of change, such as through grazing reduction, drain-blocking to raise water levels or cessation of burning regimes to allow recovery of bog vegetation such as *Sphagnum* mosses. Restoration may be a high-intensity activity in some cases where the damage is greatest. For example, areas of eroded pare peat, especially in the Peak District, have been restored by laying down an artificial substrate (geojute) to stabilise bare peat prior to re-seeding. A 'nurse crop' of grasses and heather brash with seed attached is sometimes then used to rapidly establish a stable peat surface into which peatland vegetation can then become established (see, for example, Anderson & Radford 1988, Anderson *et al*. 1997). *Sphagnum* re-introduction has been applied to a number of lowland raised bog restoration sites (e.g. Robroeck *et al*. 2009), whilst Moors for the Future (www.moorsforthefuture.org.uk) and the National Trust (http://www.nationaltrust.org.uk/main/w-chl/w-countryside_environment/w-nature/w-nature-peatlands.htm) are planting large numbers of cotton grass and other plugs into bare upland peat.

Restoration management of afforested peats has also increased significantly in magnitude in recent years. For example, around 2500 ha was deforested and restored in Great Britain between 2001 and 2005 compared to less than 1000 ha over the preceding 15 years (Anderson, 2010). This is still small in proportion to the total forested peatland area (<2%) but with an increasing understanding of restoration techniques, there is likely to be an impetus for such restoration to become more widespread (e.g. Wilkie & Mayhew 2003).

Most restoration management is geared towards reinstating peatland function, whilst subsequent recovery of peatland biodiversity can be a particularly slow process. Restoration management can be expensive though the recent focus on carbon-storage within peatlands has raised the profile of the benefits of such management and shown that it is likely to be good value compared to the cost to society placed on carbon loss from non-restored peats (Anon 2009). Furthermore significant funds are now available for well planned projects with clear outcomes from, for example, EU LIFE funding. For further information on peatland restoration, see the Lunt *et al.* (2010).

4. Status and Trends

4.1 Historical Vegetation Trends

Our perception of what constitutes a desirable or optimum peatland vegetation can be biased by living-memory recollections or recent monitoring data although old accounts (e.g. Moss 1913) provide some historical context. However palaeoecological evidence has recently shown that peatland vegetation has been more dynamic than is often appreciated. In some Exmoor locations, for example, restoration and management work on peatlands may be aimed at reversing a recent dominance of purple moor-grass and reinstating a mixed *Sphagnum* and dwarf-shrub assemblage. However, palaeoecological evidence has shown that whilst purple moor-grass may have risen to dominance in recent years there were also previous periods when the species formed at least a substantial part of the vegetation (Chambers *et al*. 1999).

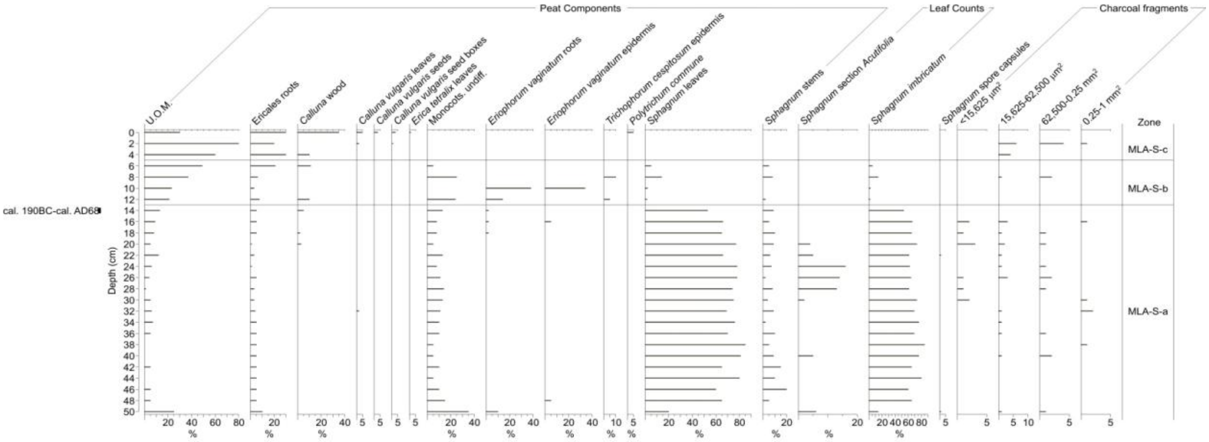


Figure 4.1 *Selected plant macrofossil data for Mynydd Llangatwg. This site is presently a Calluna vulgaris-Eriophorum vaginatum mire, Erica tetralix sub community (M19a: (Rodwell et al. 1991). The profile shows the former presence of Sphagnum imbricatum. Its decline around 2000 years BP may be linked to climatic deterioration (Mauquoy & Barber 1999). Charcoal records at 8 to 10 cm depth indicate prevalence of fire and mark a change to a more 'xeric' mire community with increase in Ericales roots (Calluna and Erica spp.) and virtual disappearance of remaining Sphagnum spp. Such features in profiles at this and other sites mark major vegetation changes post-dating the start of the industrial revolution whilst increased atmospheric input along with changes in grazing are also likely to have been influential. Figure reproduced with permission of Elsevier from Chambers et al. (2007).*

What palaeoecological analysis of many UK peatlands shows is that *Sphagnum* sp. have generally been far more dominant formerly than is currently the case. There is strong evidence that Callunetum (heather-dominated vegetation) at many sites may have little historical precedence. Instead sites may have been dominated by other species with, for example, millennial-scale dominance of *Sphagnum imbricatum* recorded from Welsh and English blanket bogs (see Figure 4.1; Chambers *et al*. 1999; Chambers *et al*. 2007; McClymont *et al*. 2008). Similarly, at a series of West Pennine bogs, *Sphagnum* spp., especially *Sphagnum papillosum*, were dominant for around 2000 years up to the start of the twentieth century when, as a result of erosion, they were replaced by hare's-tail cottongrass, bilberry, crowberry, *Hypnum cupressiforme*, *Dicranum scoparium* and wavy hair-grass *Deschampsia flexuosa* (Mackay & Tallis 1996). Thus there is a real danger in considering solely recent vegetation or vegetation adjacent to degraded sites as representing a target condition for vegetation management. In particular vegetation targets for designated sites may need to have a greater focus on *Sphagnum* and less on dwarf shrubs. For more information on vegetation trends, see JNCC (2011).

4.2 Condition Trends

Only around 400,000 ha (18%) of the blanket bog resource in the British Isles is in a natural or near-natural condition whilst 16% is eroded, 16% is afforested, 11% is affected by peat cutting and 40% is otherwise modified such as by management for sheep or deer (Tallis 1998) (for further information see JNCC (2011)). Section 2.1 describes the plant communities associated with such peatlands. Modification of a blanket bog is usually reflected in the vegetation which tends to have lower plant diversity and significantly reduced *Sphagnum* and sometimes other bryophytes. There is often a predominance of hare's-tail cottongrass (NVC M20) or purple moor-grass and communities representing drier bog surfaces (some of the heathland H9 or H12 vegetation types) (see Table 2.3). Figure 4.2 depicts some of these different states of peatland condition.

Lowland raised bogs have fared even worse than blanket bog. The extent of such sites that are relatively undisturbed has declined by 94% over the last 100 years from 95,000 ha to approximately 6,000 ha. The most significant declines have been due to afforestation, peat extraction and agricultural intensification, including drainage. Ongoing declines in extent and condition are expected to occur through gradual drying as a result of these activities (Anon 1999).

Site condition or extent of modification can be assessed in several ways but there is a trade-off between broad scale surveys, with general findings, and more detailed studies with narrower remits. Some of the data that are potentially useful are collected as part of monitoring of agri-environment schemes. However deriving trends may be hampered by the use of site-specific metrics such as the "Indicators of Success" used to measure scheme compliance on designated sites in England. Higher Level Stewardship, the primary financial means of delivering management on priority sites in England, does not provide for adequate long-term monitoring to assess outcomes for biodiversity. Whilst rapid and easily applied assessment methods might be desirable for use within agri-environment schemes, peatland management and restoration schemes require long-term detail on hydrological function, habitat change and key species trends for their efficacy to be reliably assessed.

Figures 4.2a-f *Blanket bog states and impact on biodiversity*

Figure 4.2a. *Severely degraded blanket bog in the Western Pennines. This site is heavily grazed by cattle and sheep and has been spread with manure. As a result the surface is rather bare. Little of the typical blanket bog vegetation remains though the site has a thin cover of hare's-tail cottongrass. (Photo Penny Anderson)*

Figure 4.2b. *This Western Pennines site has been subject to drains dug into blanket peat along with severe subsequent overgrazing. The vegetation comprises a number of common pasture herbs colonising the vestiges of a cottongrass blanket bog vegetation. (Photo: Penny Anderson)*

Figure 4.2c. *This image, from the South Pennines, shows the effect of wildfire in producing 'peat pans' which fill with water in wet weather, then dry out with caked algal mat in the bottom. They do not fill with Sphagnum and whilst some common cottongrass may colonise, they mostly stay bare under high grazing pressure. It is possible that these were once the hummocks comprising species such as crowberry. These burn hotly and become hollows once eroded out. Gully formation may follow if the peat pans are connected with others. (Photo: Penny Anderson)*

Figure 4.2d. *Peat cutting at Graven in north Mainland, Shetland. Small-scale shallow peat cutting may permit recovery of some elements of the peatland vegetation, such as the cottongrasses in the lower part of this view that have re-colonised the previously cut blanket bog. (Photo: Nick Littlewood)*

Figure 4.2e. *Blanket bog in good condition near Vidlin in north Mainland, Shetland. This site has a varied topography with cottongrasses and Sphagnum dominating the lower parts and ericoids in the more elevated sections. (Photo: Nick Littlewood)*

Figure 4.2f. *Close-up of a blanket bog in Bowland, north Lancashire in good condition with cranberry growing through Sphagnum. (Photo: Penny Anderson Associates)*

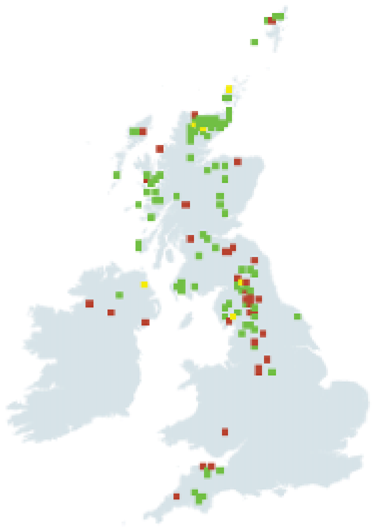
None of the monitoring schemes discussed below directly measures physical factors, for example the integrity of macroptopes and hydrological function. Instead they use biodiversity measures or the presence/absence of particular features such as drains as indicators of peat condition. There remains, therefore, a clear need for comprehensive monitoring and research to show how peatlands are changing, why such changes are happening and how such changes affect peatland biodiversity.

The only statistically reliable monitoring system for wide-scale habitat trends is the **Countryside Survey** (Carey *et al*. 2008), which covers all ecosystems within the UK. This provided an estimate for the UK extent of blanket bog at 2,393,000 ha (9.7% of the land area) in 2007 and this figure is little changed from the previous survey in 1998. Whilst condition trends cannot be directly inferred from Countryside Survey results, some aspects of data collection allow assessment of change in the characteristics of plant assemblages and of soils of blanket bogs. In particular, between 1998 and 2007 plant species richness declined. Grasses and other competitive plants increased, whereas ruderal plants, associated with disturbance, decreased. Other vegetation changes reflected a decreasing nutrient status and increasing acidity. Whilst some of these trends may be indicative of deterioration in bog condition, the Countryside Survey report cautions that further investigation is required. Insufficient lowland raised bog samples were included in the Countryside Survey to draw conclusions for this habitat.

Very coarse summary information is collated on Priority Habitats within the **UK Biodiversity Action Plan** and published through the Biodiversity Action Reporting System website (http://ukbap-reporting.org.uk). For the 2008 reporting round the extent of blanket bog is estimated at 2,208,533 ha and this is assessed as “declining (slowly)”, however with acknowledgement that the trend is difficult to assess due to lack of useful data. For lowland raised bogs, the extent is estimated at 53,537 ha and this is assessed as “fluctuating – probably declining”. Gains have been made by restoration measures while other sites are deteriorating due to such causes as failure to achieve suitable hydrological regimes and failure to follow up on scrub clearance.

On designated sites monitoring of SSSI features has been carried out as part of the **Common Standards Monitoring** programme (Williams 2006). These permit some comparison of trends in peatland condition by country within the UK. Whilst 58% of condition assessments of blanket bogs at designated sites came out as favourable, there was a distinct northerly bias in the results. At most Scottish sites the overwhelming majority of SSSI features at sites containing blanket bogs were assessed as favourable whereas English sites were shown to be in much poorer condition (see Figure 4.3a). Over-grazing and burning were the most frequent adverse activities recorded that contributed to unfavourable status. The situation for lowland raised bogs was considerably worse with just 22% of the 79 assessments returning a ‘favourable’ outcome. Again there was a tendency for a higher proportion of assessed features to be favourable on more northerly sites. A considerable proportion of bogs in Northern Ireland were assessed as being ‘unfavourable-recovering’ reflecting the ongoing impacts of positive conservation management (see Figure 4.3b). The most frequent adverse activities on lowland raised bogs were poor water management, lack of remedial management and invasive species. Information for fens and other habitats are given in Williams (2006) whilst further information about condition monitoring on SSSIs is given in JNCC (2011).

Current condition of SSSI/ASSI features

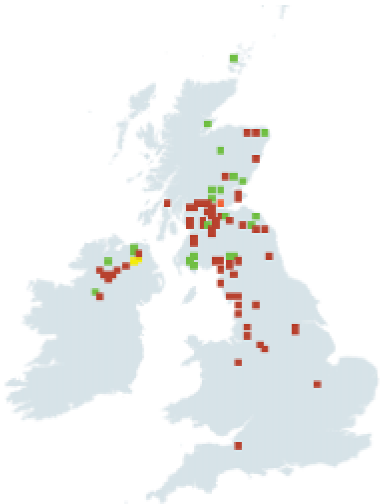


Condition of SSSI/ASSI features, with those currently reported as unfavourable-recovering shown as 'favourable'

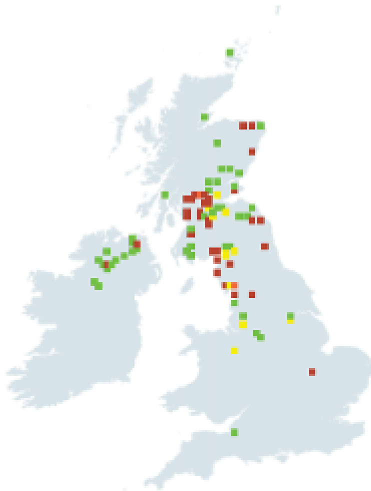
Key: Proportion of assessed features on 10 km squares that are favourable:

80-100% 50-80% 20-50% 0-20%

Figure 4.3a *The condition of features on designated blanket bogs assessed by Common Standards Monitoring. Reproduced from Williams (2006) with permission of JNCC.*



Current condition of SSSI/ASSI features



Condition of SSSI/ASSI features, with those currently reported as unfavourable-recovering shown as 'favourable'

Figure 4.3b *The condition of features on designated lowland bogs assessed by Common Standards Monitoring. Reproduced from Williams (2006) with permission of JNCC. Key as for Figure 4.2a.*

The condition of non-designated peatland is not assessed or collated nationally. However local monitoring can provide at least some insight into trends on non-designated areas. For example in the Yorkshire Dales National Park over half of SSSI-designated blanket bog was in a 'favourable' or 'unfavourable-recovering' state. Although not directly comparable, 79% of blanket bog on non-designated land was assessed as requiring some form of restoration or enhancement (see Table 4.1). It is unclear whether this trend is because only bogs in better condition have been designated or because enhanced incentives or effort exists for appropriate management and restoration on SSSI-designated sites. Greater effort to monitor non-designated peatland sites on a national scale would be needed to address this question fully and to assess the contribution of non-designated sites to the UK peatland biodiversity resource.

For more information on habitat condition trends see JNCC (2011).

Table 4.1 *Condition of designated and non-designated blanket bog in the Yorkshire Dales National Park surveyed between 2002 and 2008. For non-designated sites, condition assessment is based on the Higher Level Scheme FEP (Farm Environment Plan) condition assessment. It is not comparable with condition monitoring on SSSIs. However it does allow us to say that around 79% of non-designated blanket bog (categories B and C) is in need of some form of enhancement or restoration. Data courtesy of Tim Thom, Yorkshire Dales National Park Authority (see www.yppartnership.org.uk).*

	SSSI			non-SSSI		GRAND TOTAL AREA (Ha)
	Measured Area (Ha)	Calculated Area (Ha)	Condition (%)	Measured Area (Ha)	Condition (%)	
Favourable	2292	2292	**18**			
Unfavourable -no trend	731					
Unfavourable-recovering	4254	4577	**36**			
Unfavourable-no change	5377	5785	**46**			
Unfavourable-declining	8	9	**0**			
Unknown condition	2219	2219				
A				1472	**21**	
B				1264	**18**	
C				4330	**61**	
Unknown condition				10252		
TOTAL AREA (Ha)	**14881**			**17318**		**32199**
AREA IN MOSAIC (Ha)	**420**			**984**		**1404**
GRAND TOTAL (Ha)	**15301**			**18302**		**33603**

4.3 Trends in Key Species

Priority species, designated under the UK Biodiversity Action Plan process, are subject to published targets but monitoring and reporting of progress towards these targets is patchy. For species that occur across other habitats as well as peatlands, progress is not collated in a way that allows a break-down by habitat type. However, for ombrotrophic peatland species where UK trend information from a variety of sources is available, ten were found to be declining and only four increasing (see Table 2.2). Greater emphasis on establishing baseline data for these species is essential if we are to understand future trends in populations or ranges.

Trends in breeding bird populations on peatlands are difficult to define as few data are published that separate peatlands from other habitats. In particular most trend analyses (e.g. Sim *et al.* 2005) appear to lump together bogs with heather moorland. Local trends must be interpreted with care as factors affecting birds may vary geographically. However as described in Section 2, the Flow Country of Caithness and Sutherland is an internationally important peatland area for breeding birds and has been subject to repeat surveys. Stroud *et al*. (1987) showed how numbers of some key species declined in the 1970s and 1980s as a result of habitat loss to forestry. In addition to the direct loss of habitat, subsequent surveys in 1993 and 1994 showed continued losses of these species (Whitfield 1997) (see Table 4.2). As some plantations were adjacent to some of the best bogs for these birds, this may have been due to 'edge-effects' of altered hydrology or cover for predators (Avery 1989; Hancock *et al*. 2009).

Finally, priority species listing does not always guarantee conservation action and the policy frameworks of key organisations may be influenced by other factors. For example the UK government conservation agencies concentrate work on designated sites (SSSIs etc.). This may be to the benefit of species on lowland raised bogs in England (most of which by area is designated as SSSI) but less advantageous to those inhabiting blanket bogs in Scotland (of which only about 11% is designated).

Table 4.2 *Trends in the number of breeding pairs of three wading bird species on the Flow Country of Caithness and Sutherland. The primary period of forest expansion was late 1970s and early to mid 1980s.*

	Pre-afforestation (early 1970s) (Stroud *et al*. 1987)	**1987 (Stroud *et al*. 1987)**	**1993/94 (Whitfield 1997)**
Golden Plover	4900	3980	3767
Dunlin	4620	3830	3095
Greenshank	760	630	464

5. Good Management Practice

The success of imposed management depends on the starting point and the objective to be achieved. Good management requires an assessment of the initial floristic composition and of physical factors, including hydrology, at all scales from macrotope to vegetation. A consideration of the needs of key animals for the site is also good practice. Site targets and objectives will depend on the nature of the site and degree of degradation at the outset. At severely degraded sites, for example, lowland raised bogs that have been damaged by cutting, creation of a poor fen rather than raised bog might be regarded as success, at least in the short term, especially if the long-term trajectory is towards eventual bog formation. Monitoring of management schemes is crucial to assessing their efficacy (and the importance of monitoring, including collection of baseline data, should be fully recognised in funding schemes). However aims need to be appropriate for the timescales involved with intermediate benchmarks set where progress towards active peat formation is a very long-term goal.

5.1 Grazing

Some peatlands appear not to require grazing to maintain bog plant communities (e.g. Smith 1995). However Tallis (1998) concluded that natural blanket bog vegetation may be the result of long-continued low-intensity grazing. He cited the example of areas in northern Britain where grazing on bogs below the tree line prevents colonization by birch *Betula* spp., pine *Pinus* spp. and willow *Salix* spp. (Welch 1997). Different peatland habitats respond differently to grazing depending on the species they contain, their palatability and digestibility, their coverage and the mixture on offer across a site (which usually extends well beyond the peatland area). Grazing exclusion might produce benefits when trying to restore peatland vegetation from a highly degraded state (e.g. Rawes 1983, Anderson *et al.* 1997). However in many situations light grazing is likely to be beneficial, especially summer grazing, in situations where grasses might otherwise rise to dominance (as demonstrated by Hulme *et al*. (2002) in wet upland heathland). Grazing impacts will, though, vary from site to site and an adaptive management approach is required to produce the most favourable habitat outcomes.

Wild herbivores, particularly red deer, have also long been natural grazers on peatlands. In areas with fewer sheep, especially northern Scotland, they may continue to be the main grazer. Although the differing impact of sheep and deer have been studied on upland heathland (e.g. Milne *et al*. 1992; Hester & Baille 1998), there is little currently known about how they interact on peatlands or about how deer should be managed for biodiversity gains.

Some breeding birds may benefit from grazing, especially, for example, golden plover and curlew where grazing can help to break up the vegetation at sites with tall uniform heather (Grant 2002; Pearce-Higgins & Grant 2006). Furthermore the structural diversity of vegetation associated with light grazing may promote increased invertebrate diversity in some upland habitats. For example Gardner *et al*. (1997) found this effect on upland heathlands and the same may apply on bogs with related vegetation types.

5.2 Hydrology

Hydrological integrity is crucial for good management for peatland biodiversity, both in upland blanket bogs and lowland raised bog systems and in fens. Where there is good hydrological integrity this must be maintained; where hydrological processes are sub-optimal they should be improved. The hydrology of blanket peats is complex. In upland blanket bogs water flow may occur at multiple depths (Holden & Burt 2003) and good hydrological management may, therefore, be difficult to achieve. Ideally hydrological management should be carried out on the mesotope although localised management, such as drain blocking, may produce some benefits (e.g. Armstrong 2009). The extent of modification to hydrology may not be immediately apparent and, for example, the extent of peat pipes resulting from wildfire or other damage may be considerable (e.g. Holden 2005). Restoration of the hydrology may be impossible where the site is in the later stages of complete erosion and loss of much of the peat mass (Anderson *et al.*1988). Sites where part of the mesotope has been previously modified by agricultural reclamation will present particular challenges to prevent bog desiccation. Whilst peat-bunding can produce some success at buffering active peatlands from surrounding drained land (e.g. Bailey 2003) taking control of hydrology on adjacent land is a better option where this is possible.

5.3 Scrub Management

On sites where the water table has been lowered there is a risk from scrub and tree encroachment, especially on lowland raised bogs. Species involved may include natives, such as willows and birch (e.g. Hulme 2006) or invasive alien species such as *Rhododendron ponticum*. In some cases, native scrub may be viewed as a natural part of bog vegetation and may help form very specific habitat requirements for rare species, such as the 10-spotted pot beetle *Cryptocephalus decemmaculatus*, which has only been found on small willows and birches within bog habitats (Anon 2010). At other sites, scrub-encroachment may be by non-native species such as regenerating lodgepole pine *Pinus contorta* from adjacent forestry (Anderson, 2010). At sites subject to succession where the nutrient status has been elevated, such as from surrounding agriculture, and there is no prospect of reversing this, allowing succession to fen carr rather than trying to re-create a raised bog may be appropriate. Indeed woodland on lag fen surrounding lowland raised bogs may be considered a natural and biodiversity-rich part of the system (Bowler 2002). However an expansion of scrub will usually be viewed as a threat to typical peatland biodiversity and scrub control forms a major part of management at some key sites.

5.4 Burning

Burning is a common management tool to maintain heather dominated vegetation for red grouse, to protect larger areas from wildfire and to provide fresh vegetation for sheep grazing. It is carried out in particular on upland heathland but has also been carried out extensively on peatlands. However, burning on blanket bogs is now discouraged (Anon 2007; Anon 2008a; Anon 2008b) and there is little evidence to recommend otherwise. A longer cycle of burning is permitted on some drier heather-dominated peatland sites in England under the relevant code (Anon 2007). However most such sites are likely to represent a modified vegetation that is actually perpetuated by the continuation of burning and the burning cycles needs to be broken for peatland vegetation to recover. For further details, see Worrall *et al.* (2010b).

5.5 Restoration

There are probably more peatland restoration projects being undertaken or planned currently and over the last five years than at any time in the past. These schemes are using special project funding (such as LIFE), investment permitted by the water utility companies through the Drinking Water Inspectorate and OFWAT or agri-environment schemes. Much has been driven by the Government target for favourable condition for SSSIs and should thus also enhance biodiversity, but water quality is also a key driver and flood risk reduction is a main player on some schemes (DEFRA/EA funded). Indeed restoration may be carried out for multiple purposes including carbon sequestration, recreation and biodiversity. In many cases good practice management for one of these factors will benefit the others but this may not always be the case, especially where one of the other factors is being managed to its maximum. For example over-intensive recreation can lead to species disturbance and erosion from unmanaged access tracks.

Restoration projects must have clear aims and be planned over timescales that are realistic for these aims. Restoration should also be at an appropriate scale and consideration should be given to re-establishing integrated habitat networks. Some projects, such as those purely involving withdrawal or control of grazing livestock, may take many years before recovery of the desired vegetation becomes apparent (e.g. Yalden 2004). Funding schemes need to recognise these long-term challenges.

Monitoring should be a common element in all restoration schemes where biodiversity conservation or enhancement is a stated goal of management. This needs to include good baseline data as well as ongoing monitoring to be able to assess the effectiveness, or otherwise, of management. Management should be adaptive to take account of such monitoring and the results of successful schemes should be disseminated through a variety of media to increase the all round effectiveness of restoration initiatives. In particular there needs to be ready access to case studies and demonstration sites to disseminate knowledge of both successful and unsuccessful management actions.

5.6 Barriers to Good Practice

Information for land managers is often scattered or even conflicting. Much is hidden in the scientific literature or in unpublished reports. There are collations of information in a Management Handbook for bogs (Brooks & Stoneman 1997) and, recently, one for fens (McBride *et al*. 2010) and a new handbook on good practice upland restoration is in preparation. These should be promoted among peatland management practitioners. Further initiatives to make information on management for biodiversity more readily available include the publication of 'user-friendly' habitat management information on BAP priority Habitats (Anon 2010) and advice sheets from DEFRA relating to sustainable grazing on heather-dominated sites. Peatland practitioners should be encouraged where possible to report back on the results of management initiatives. There is a free journal dedicated to mires and peat (www.mires-and-peat.net) and the dissemination of results from management interventions is facilitated through initiatives such as the online journal, Conservation Evidence (www.conservationevidence.com).

Skills shortages for peatland management may take a number of forms. For example, species identification skills are required for accurate monitoring of ecosystems but there is increased consensus that these are generally lacking in university training. This issue is especially acute for peatlands as much monitoring relies on knowledge of more 'difficult' groups such as bryophytes. There are initiatives to address this shortfall, such as identification courses run by the Field Studies Council, but there needs to be a much more extensive development of field skills training than at present. Further skills shortages may result from a reduction in rural work forces. This may make it more difficult to carry out land management actions, such as livestock shepherding.

Peatland management and, especially, restoration is expensive. Work carried out under Higher Level Stewardship in England requires payment up front by the landowner, making uptake impossible for some. Even when funding is more readily available, peatland managers may have priorities that differ from those of promoting biodiversity. Furthermore management hierarchies in upland areas are often complicated by separation of ownership, shooting rights and grazing rights, especially on commons, which may serve to make coherent management difficult or impossible. Restoration or management for retaining peatlands in favourable condition has thus been largely carried out in a piecemeal approach. There is a real need now to create a co-ordinated policy framework for such management. Given the huge losses that have occurred, lowland raised bogs in particular need a co-ordinated plan for restoration and management of the remaining resource.

Despite the above limitations, successful peatland restoration and management projects have been carried out for the benefit of biodiversity and other peatland services in many parts of the UK. These range in scale from multi-site projects, such as the public-private Moors for the Future partnership (www.moorsforthefuture.org.uk) which is working in particular to tackle erosion across the Peak District, the Peatscapes project (North Pennines AONB http://www.northpennines.org.uk/index.cfm?articleid=12218), the Yorkshire Peat Partnership (http://www.yppartnership.org.uk/) and the Mires on the Moors project (www.exmoor-nationalpark.gov.uk/mire) which is lead by South West Water to focus primarily on re-wetting catchment areas, to single site projects such as restoration by tree-removal of Foulshaw Moss by the Cumbria Wildlife Trust. Such sites act as exemplars of what it is possible to achieve from which the benefits of management should be promoted to encourage and inform long-term good practice across all peatland sites.

6. Possible Future Climate Change Impacts

Current predictions indicate that peatlands will be subject to pressures of drying (with possible increased peat cracking, which will act as a positive feedback effect by increasing erosion), increased storminess (also with associated potential for erosion) and increased temperatures with a concomitant effect on plant decomposition and on drying. Bioclimate envelope modelling suggest that, under a high emission scenario for 2071 to 2100, the bioclimatic space for peatlands reduces by 84% with only parts of western Scotland remaining within this space. Increased summer temperatures are the primary driver of change in this analysis (Gallego-Sala *et al*. in press).

The extent to which peatlands are resistant to damaging perturbations and how resilient peatlands are after any damage has been imposed by the perturbation remains to be seen. Resistance is the ability of the ecosystem to resist a damaging force and resilience is the ability of the ecosystem to bounce back after damage has occurred (see Mitchell *et al*. 2000 for discussion). The relative balance between these two ecosystem properties will have crucial importance for the protection of peatland biodiversity and of the carbon store it contains. Significant changes to peatland vegetation assemblages are likely to occur and indeed have done so periodically in the past (see Section 4). However the fact that each former vegetation type laid down deposits in accumulating peat layers indicates a degree of resilience to climate change, insofar as peat formation has continued through previous climate changes. Part of the mechanism giving rise to this resilience involves changes in microtope characteristics, such as a reduction in pools and increase in hummocks and ridges which in turn will alter the quantity and quality of habitat for other life forms. Degradation may reduce the ability of a peatland to further adapt to climate change. Hence identifying boundaries to such resilience would be of considerable use for long term peatland conservation planning.

Species using peatlands may undergo climate-induced range shifts and changes to the timing of their seasonal activity. There is already general evidence of northward and uphill movement in the distributions of a wide range of species in Britain (e.g. Hickling *et al*. 2006). The impact of this may be particularly severe on northern blanket bogs. As discussed in section 2, these host a suite of species that are at or close to the southern edge of their distribution and are likely to be particularly at risk from a warming climate (Hampe & Petit 2005). The rates of change of seasonality of activity may vary (e.g. Thackerey et al. 2010) and this may lead to mismatches between phonology of predator and prey species. Some bird species, for example, only use peatlands during the spring and early summer breeding season. Climate change-related mismatches between Golden Plover breeding and the emergence of a primary food source, adult craneflies, have already been identified as potentially occurring in the Peak District (Pearce-Higgins *et al*. 2005). It is likely that the impact of such asynchronous shifts in activity as well as general climatic impacts on prey availability will be greater on species that are at or close to the edge of their range, as is the case for many of the important breeding birds on UK peatlands and, thus, the resilience of such populations may be seriously threatened under future modelled climate scenarios (Pearce-Higgins *et al*. 2010).

For further details on climate change impacts see Worrall *et al.* (2010a).

7. Conclusions and Key Messages

- Simple diversity indicators, such as species richness, are generally inappropriate for assessing the value of peatland biodiversity. Instead the naturalness of the system should be recognised and assessment should encompass habitat condition, specialist species, microtope patterns and key species trends.
- There has been significant modification of peatlands over time, but particularly in the last 300 years from aerial deposition, high grazing levels, regular burning (managed and wildfire), nutrient input and scrub colonisation (lowland raised bogs) and drainage together with other losses of systems to forestry, peat extraction and other developments. Many of these modifications are ongoing processes.
- The extent of functioning peatland habitats has suffered large declines over the last 100 years in particular. Declines in extent and habitat condition are most acute for lowland raised bog but also significant for blanket bog, especially towards the south of its extent within the UK.
- Modification of peatlands has negatively impacted key wildlife species. More UK Priority Species of peatlands are declining than are increasing though information is incomplete. Breeding bird populations on peatlands are under pressure from multiple factors including habitat degradation and climate change.
- Maintaining hydrological function of the macrotope, where it has not been modified, is of paramount importance for safeguarding peatland biodiversity. On raised and blanket bogs this typically means maintaining a high water table.
- Restoration of modified sites may not produce pristine peatland vegetation, at least in the short term, but has the potential to set sites on a trajectory of change in the direction of becoming functioning, more resilient semi-natural systems. Restoration prioritisation should be given to sites where the hydrology of the macrotope can be controlled and where there is a good remnant population of bog species. Restoration of bare and eroding peat should not be neglected where biodiversity and other ecosystem benefits can be identified.
- Target-setting for peatland management and restoration projects should, where possible, take account of available palaeoecological evidence. Whilst long-term change in peatland vegetation may be a normal process, recent and/or adjacent vegetation may have been promoted by moderately recent human activity and might not necessarily be considered an optimum target.
- Progress on safeguarding peatland biodiversity is restrained by a lack of data on populations, trends and ranges of species and habitats. In particular we know very little about peatland invertebrates and the functional role that they play.
- Monitoring is a crucial component of any peatland management scheme. Knowledge of trends in key species is very poor, even for UK Priority Species. There is very little co-ordinated monitoring of habitat condition of non-designated peatland sites and this needs to be addressed.
- Peatland management needs to take a flexible approach to address different drivers influencing each site. Management advice should be disseminated widely.

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