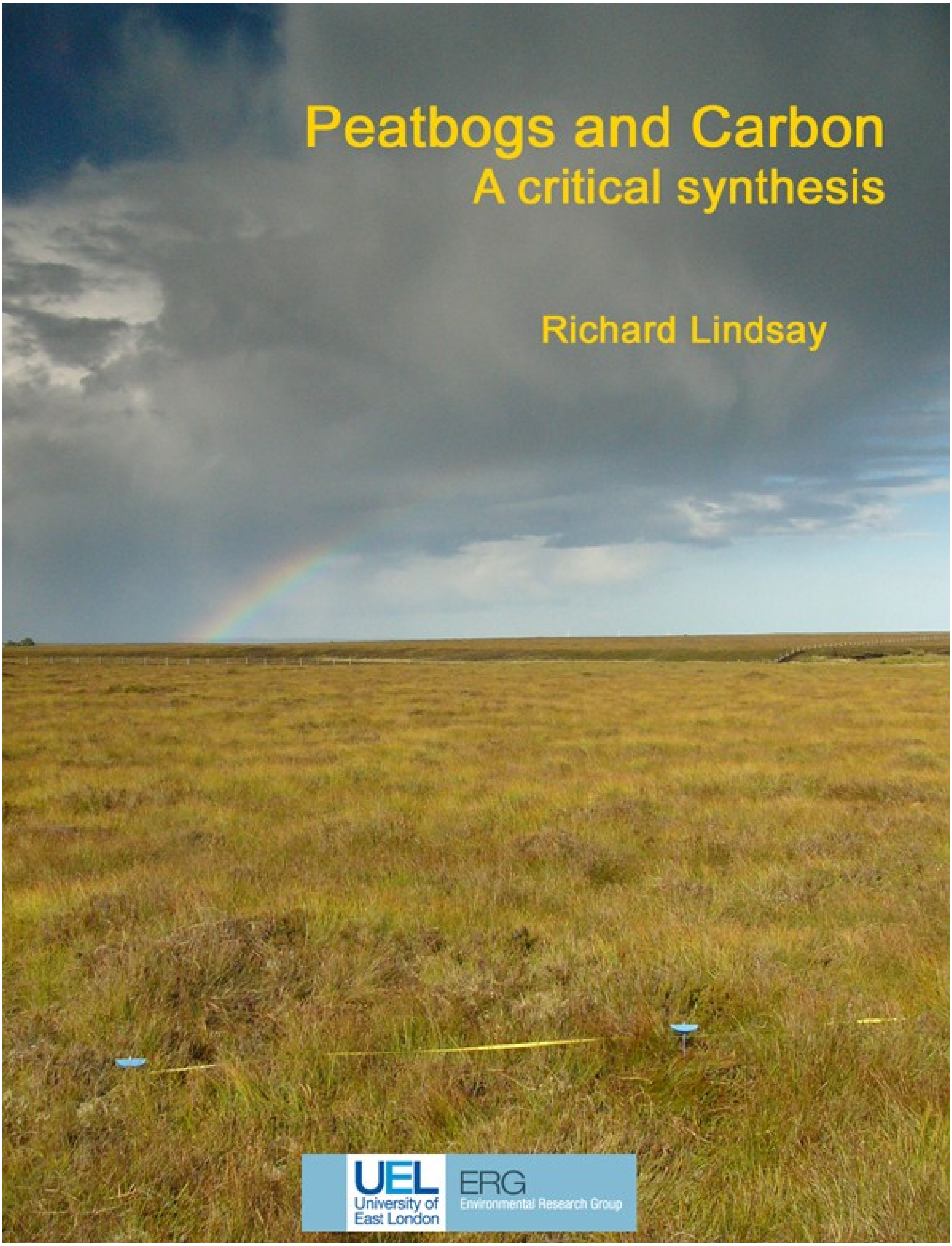
Peatbogs and Carbon
A critical synthesis

Richard Lindsay



UEL University of East London

ERG Environmental Research Group

Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change.

Richard Lindsay
Environmental Research Group
University of East London

r.lindsay@uel.ac.uk

School of Health & Bioscience
University of East London
Stratford Campus
Water Lane
London
E15 4LZ

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Any further queries to:
Clifton Bain
RSPB Scotland
clifton.bain@rspb.org.uk

The report was produced by Richard Lindsay
Environmental Research Group
University of East London
r.lindsay@uel.ac.uk

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PART 1

EXECUTIVE SUMMARY
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KEY FINDINGS

Executive Summary

1) Reviews of peat carbon science have presented often contradictory lines of evidence, in part due to the complexity of the subject but largely as a result of differences in methodology and definition of the peatland under study.

2) An understanding of peatland environments including their different functional, structural and vegetation characteristics is essential in interpreting the findings of peat carbon studies. The main peatland types and forms are described, including the various hydromorphological forms.

3) Estimates of the extent of UK peat vary considerably, from 1.5 million ha to an upper level of 5 million ha if shallower peats <1m are included. From the perspective of an interest in peatlands capable of supporting active peat-forming habitat, the larger estimates may be the most relevant.

4) Peat thickness in blanket bog is usually between 0.3 m and 6 m and can vary considerably even over short distances. Published average national figures, ranging between 1.5 m and 2.4 m for peat thickness are very generalised and have generally not been determined from any systematic measurement across Britain, although significant progress has been made in Scotland.

5) In order to determine the quantity of carbon stored in peat resources of the UK, it is necessary to know how much carbon is stored per unit volume of peat. This largely depends on how densely packed the material is within the peat matrix (bulk density). The bulk density of peat varies according to the management history of the peat habitat and the microtopography but also varies considerably between the two structural layers of the peat (acrotelm and catotelm). One 'standard model' suggests bulk densities of 0.03 g cm^{-3} in the uppermost parts of the acrotelm increasing to 0.12 g cm^{-3} in the lower catotelm. Bulk density may also vary unpredictably down the column of the catotelm where there are weak layers in the peat.

6) In contrast with this standard model, bulk density studies in Britain often show high bulk-density levels in the topmost layer of peat. It is suggested that extensive human influence on peatlands has disrupted or destroyed the acrotelm, creating a dense layer of damaged peat at the surface through compaction and oxidative wastage. The concept of a 'haplotelmic' bog is used to describe sites where these conditions prevail. Such sites, where the peatland has been modified by management, no longer support a functioning *Sphagnum*-dominated acrotelm layer that lays down peat and protects the catotelm peat store. Theoretical values are provided for the carbon content of a "standard cubic metre of peat" under both natural and haplotelmic conditions.

7) When considering the carbon contained within peatlands it is important to recognise the carbon content of the biomass layer as well as the peat soil. It is suggested that a 15 cm *Sphagnum* layer can be considered as living biomass with a carbon content of up to 50 t C per hectare (equivalent, for example, to the total above- and below-ground biomass of some 50-year conifer plantations). On damaged bogs lacking a *Sphagnum* layer and dominated by vascular plants, the total biomass of the bog is lower, at around 10 t C per hectare (including aerial parts and below-ground roots). At present, the UK national greenhouse gas inventory assumes biomass values for bogs that are substantially lower than either of these figures.

8) Information about the carbon content of peatlands at a national level must be regarded with caution in view of the limited measurements of peat thickness and bulk density which have so far been obtained. An estimate of 3,121 Mt C is regarded as the likely minimum for peat soils in the UK.

9) Peatlands are dynamic stores of carbon with interchange between the peat and the atmosphere for most of the year in the predominantly oceanic UK, whereas in boreal regions (*e.g.* Scandinavia, Canada) these exchanges may be frozen during the winter months. Caution must therefore be exercised when using values for peatland carbon-exchange obtained from boreal peatlands.

10) In an active peat bog, the acrotelm takes up carbon dioxide from the atmosphere and converts it into plant material. Much of this is then subsequently lost again through decay as carbon dioxide, methane or as carbon in solution. A small amount of undecayed material is passed to the anaerobic conditions of the catotelm where the rate of decomposition is so slow that material accumulates as peat. The preponderance of *Sphagnum* in such remains reflects the greater resistance of *Sphagnum* to decay compared to vascular-plant tissues. Any decomposition which occurs in the catotelm will tend to produce methane, carbon dioxide, or carbon in solution. It is

the overall balance of accumulation and loss, over time, that determines the carbon dynamics for any particular peatland. Figures for the different components of this system are presented and discussed.

11) Evidence suggests that methane arising from the action of bacterial decomposition in the catotelm may remain largely locked up in the peat under natural conditions, contributing to the low conductivity of this lower peat. In the acrotelm, methane is both produced and oxidised by different microbial populations. Different microtopographical surface features (*e.g.* hummocks, pools) and different vegetation types all show differing methane emissions.

12) *Sphagnum*-dominated swards can suppress methane through oxidation. In contrast, some vascular plants growing in bog pools and hollows can act as routes for direct methane release, transferring methane from lower levels to the atmosphere directly, thus by-passing the oxidising layer in the acrotelm[1]. There have been relatively few long term methane studies on undamaged natural UK peatlands. There have been even fewer studies which take explicit account of the relationship between emission levels, vegetation, and associated hydromorphological structures (*e.g.* pools, carpets, hummocks).

13) Despite the higher global warming potential of methane (25x that of CO_2), *Sphagnum*-rich natural peatlands are on balance likely to be climate-change beneficial, apparently sometimes strongly so. Peatlands currently in a less-than natural state may, depending on their current condition, either be contributing to climate warming or to climate cooling on a 100-year time frame.

14) Following cessation of active damage, most bogs tend naturally towards a state of recovery and this may (or may not) stimulate increased greenhouse gas (GHG) emissions compared to the drained state This is because they may still be releasing CO_2 through oxidation of dry peat and at the same time releasing CH_4 from re-wetting areas (although the CH_4 emissions associated with recovery will not generally exceed emission-levels typical of the original natural state). Depending on their natural or assisted rate of recovery, any climate-warming sites are likely to become climate-cooling on a 100-year timeframe because they will tend towards the natural condition of net CO_2 sequestration and relatively low CH_4 emissions.

15) It must be emphasised that the marked increases in CH_4 emissions from recovering bogs appear still to be generally lower than background emissions from natural bog systems, although blanket bogs in England and Wales may experience a period of CH_4 emissions higher than levels in the natural state because large numbers of blocked drains create more open water than would naturally occur in such regions. However, colonisation of this open water by *Sphagnum* will tend to reduce CH_4 emissions to natural background levels. Furthermore, the emissions from a recovering site should be balanced against the carbon losses that would have continued to occur if the site had not been brought into a recovering state so rapidly – usually through active conservation management.

16) It is proposed in this report that many of the dynamic processes associated with carbon cycling in the peat bog system seem to be most closely linked to vascular-plant tissues. In contrast, the *Sphagnum* sward is largely involved with carbon sequestration and transfer of this into long-term storage within the catotelm. As such, it is perhaps helpful to think of two distinct, though linked, cycles within the surface layer of a bog. On the one hand there is the vascular-plant cycle, which processes the majority of CO_2, CH_4 and DOC but ultimately contributes relatively little to the carbon store, and there is the *Sphagnum* cycle, which processes CO_2 and creates the bulk of the long-term peat store. Although this is undoubtedly a simplistic model, it is presented as a means of clarifying the key components of the bog system-processes and their possible relationship with each other.

Several topics of particular concern in relation to peat bog systems are then considered in more detail. These are summarised below.

[1] This transfer process is often referred to as a 'methane shunt', although a range a different mechanisms may be involved. Prof. Andy Baird (pers. comm.) has suggested that the term 'methane shunt' should be confined to cases where methane is transported through hollow stems such as in common reed (*Phragmites australis*), whereas diffusion through the gas-transport system of plants such as bog bean (*Menyanthes trifoliata*) could more accurately be referred to as 'methane transport'. This suggested approach has been adopted in the present report, but readers may find 'methane shunt' used in the more general sense in other literature.

Discussion Topic 1a: Drainage on peat

17) Drainage removes water from a bog system more rapidly than would occur naturally by infiltration through, and over, the peat surface. In many soils, water removal is readily achieved throughout the soil profile. This is not the case in peat bogs. Rapid water removal is restricted almost entirely to the thin uppermost layer of a bog – the acrotelm. Thus the acrotelm may be readily emptied of its water while the lower catotelm peat remains largely saturated. This has led to the general belief that drainage is ineffective on bogs – which indeed it is from the perspective of a soil-drainage engineer.

18) However, complete water removal from the acrotelm is of major ecological significance. Though it may involve a fall in the water table of only 10-20 cm (or even less) , the lowering produced by drainage can have a substantial impact on the living vegetation, particularly the balance between *Sphagnum* species and vascular plants, because a natural bog water table rarely varies by more than 4-5 cm. Furthermore, because water transmission in this narrow acrotelm layer is so relatively rapid, drainage effects can be felt over considerable distances.

19) Peatland vegetation has the capacity to respond to and alter hydrological conditions within the peat under changing circumstances such as drainage or climate change. Unlike other soils, peatlands can respond to drainage in a number of ways, some of which are physical while other responses are under biological control.

20) Physical responses are most obvious in the form of subsidence due to consolidation as water escapes from the peat, and by compression as the drained material presses down on the peat below. The peat is also oxidised and thus decomposes, leading to further shrinkage of the peat mass. The combined effect is that subsidence can extend many metres beyond a drain in a peat bog depending on the slope, water content of the peat, bulk density of the peat, depth of the drain, and climate.

21) Studies looking at water table levels only in relation to the peat surface rather than to a fixed datum point will tend to under-estimate the water table draw-down effect because they do not take account of surface subsidence, which means that the peat surface follows the water table down to the new lower water table. The picture generally presented is that the water table is drawn down within 2-5 m of a drain. This is often incorrectly considered to be, and generally described as, the prime hydrological impact of drainage.

22) Such a view does not recognise the impacts on the acrotelm, neither does it take into account the existence, and impacts, of subsidence. Subsidence can mean, for example, that the effective width of a drain may be increased substantially over time. However, the main ecological effects of drainage are generally found in the acrotelm.

23) Drainage impacts on the acrotelm are much less frequently measured than the more obvious, if limited, water-table draw-down effects in the catotelm. In the acrotelm, although drainage can lead to extended and even permanent dewatering over substantial distances, in many drainage studies this is described merely as 'removal of surface water'.

24) Drainage causes the acrotelm vegetation to experience drought conditions more often and for longer periods. Topography can also have a major influence on the impact of drainage in blanket bog areas, particularly downslope from drains. Some studies indicate a potential effect on acrotelm hydrology as far downslope as 400m from a drain. The ecological response to even small changes (4-5 cm) in average water levels in the acrotelm can be sufficient to bring about changes in vegetation, although communities exhibit a degree of adaptive resilience to varying water regime, changing species composition according to prevailing conditions.

25) The key stages of topographical change and biological response are presented, from *Sphagnum*-dominated vegetation to dwarf shrubs and grasses, with increasing loss of the acrotelm. Natural conditions with a mixture of drier and wetter *Sphagnum* communities are replaced by *Sphagnum* communities typical only of drier conditions. With continued drainage the vegetation of the acrotelm steadily loses its *Sphagnum* cover and thus ceases to be peat forming. The normally-functional acrotelm is therefore lost and the surface becomes a haplotelmic surface (unprotected catotelm surface) dominated by vascular plants such as dwarf shrubs and grasses, rather than *Sphagnum*.

26) Many drainage studies have either looked only at draw-down in the catotelm and not measured acrotelm effects, or have looked at sites with existing drains and attempted to make assessments of drainage impacts based on what is now visible. Uniquely in peatlands it is possible, when undertaking studies of already-drained ground, to look at the nature of the bog vegetation prior to

drainage (provided the approximate date of drainage is known). This is because the peat archive will hold a record of the pre-drainage condition. Such an approach to assessment of drainage impacts is only now beginning to attract research interest.

27) The carbon impact of drainage essentially consists of increased oxidation of the peat matrix leading to carbon dioxide release to the atmosphere and to dissolved organic carbon in water outflows. Peat is only formed in the acrotelm, largely by *Sphagnum*. As drainage tends to result in loss of *Sphagnum* and a functioning acrotelm, the system increasingly becomes dominated by vascular plants and steadily loses its ability to sequester carbon as peat. There are reductions in methane emissions due to oxidation as air penetrates more deeply into the peat, but cracking can release methane from deeper in the peat even under conditions of drainage. The net effect of drainage is thus a variety of carbon losses and few gains. However, the extent to which different degrees of drainage may halt carbon sequestration and lead instead to carbon losses is not well understood, particularly in relation to UK blanket bogs.

28) Drainage-induced changes to the carbon budget may be significant and can occur even when the water table shows only a limited draw-down relative to the ground surface. Vascular plants are more easily decomposed and therefore provide less net long-term carbon sequestration. Vascular-plant tissues may dominate the carbon store of the surface layer but little of this survives to be passed down into the long-term store of the catotelm because vascular-plant tissues are not as resistant as *Sphagnum* to decay.

29) Vascular plants may, however, therefore contribute significantly to increased dissolved organic carbon (DOC). This increased DOC appears to be derived from elevated quantities of breakdown products generated by the increased biomass of vascular plants resulting from drainage. Furthermore, in the absence of *Sphagnum*, the root systems of vascular plants may increase the loss of carbon from the deeper peat by stimulating decomposer microbial populations with supplies of fresh organic carbon in the form of root tissues, root exudates and oxygen.

30) Drainage also leads to cracking and piping within the peat catotelm. Piping and drainage have been shown to be interlinked through mechanisms which are not yet understood. Such cracking and piping results in greater oxidation and decomposition of the peat, and may make the bog system more susceptible to slope-failure and consequent peatslides. Carbon losses from peatslides can be substantial.

31) Methane emissions on drained peatlands fluctuate considerably over a year, with annual totals often much smaller than indicated by peak rates. Results from short-term studies must therefore be treated with caution. There are few long-term studies of methane and drains in UK peatbogs.

Discussion Topic 1b - Restoration of drained peat bog systems: the carbon balance

32) Restoration of peat bog systems aims to restore high, stable water-table conditions *within* the peat. Large areas of open water are not conducive to habitat recovery.

33) Restoration of drained peatlands through ditch blocking has been shown to raise water levels and stabilise the water table. The vegetation response is different in the ditch and on the bog surface. Growth of aquatic *Sphagnum* species in the open water of the ditches leads to 'terrestrialisation' of the ditch water, and although peat accumulation in such aquatic systems is slow, the presence of *Sphagnum* carpets helps to stabilise the water table in the ditch and reduce CH_4 emissions.

34) Raised water levels in blocked drainage ditches also result in elevated water levels within adjacent areas of the bog surface. This rewetting of the dry bog surface is termed 'paludification', and across the bog surface can result in a switch from vascular-plant dominance to a *Sphagnum*-dominated vegetation characterised by terrestrial *Sphagnum* species. These terrestrial species are more vigorous formers of peat than the aquatic *Sphagnum* species colonising the standing water in the ditch.

35) A decrease in the rate of carbon dioxide released from the peat can be expected as vascular plants are replaced by *Sphagnum*. This is because there are fewer vascular-plant root systems to oxidise the underlying peat. Indeed there may be increased CO_2 uptake and sequestration as

vigorous terrestrial *Sphagnum* species re-establish the peat-forming process. Examples are given of sites which are recovering naturally and where the current rate of carbon sequestration is very high.

36) Methane emissions are known to increase with rising water tables on peatlands but there have been few direct studies of restored UK blanket bogs. Indeed a naturally-recovering site has been found to release extremely low levels of methane. The age of the carbon in methane emissions associated with rewetting suggests that such emissions arise from the decomposition of newly-flooded vascular-plant vegetation in ditches rather than decomposition of catotelm carbon store. If this is the case, then once the vascular-plant material has decomposed, methane emissions can be expected to fall.

37) Methane emissions associated with restoration may thus be relatively short lived (perhaps less than 5-10 years) and of course they are localised within the ditches. There is evidence to suggest that in both ditches and the wider bog surface, re-establishment of *Sphagnum* can suppress methane emissions arising from the peat. *Sphagnum* dominance in the ditches also reduces the vigour and abundance of vascular plants responsible for methane transport.

38) The amounts of methane released from re-wetting are likely to be low in relation to the overall carbon and global warming benefits of restoration. Indeed the limited amount of existing evidence from re-flooded peat bog systems suggests that methane emissions, though showing an increase compared to the drained state, are still lower than those found on natural peatbog systems. In addition, such emissions must be balanced against the long-term losses of carbon dioxide which would have occurred had the site not been brought into restoration management.

39) Dissolved organic carbon has been shown to increase in blocked drains for some years after restoration. A delay in the restarting of certain enzyme reactions following flooding has been suggested as a possible cause. However, this observed increase in DOC might be more simply explained by the death and decomposition, due to rewetting, of the substantial quantities of vascular-plant biomass previously encouraged by the original drainage. The young age of the carbon in DOC suggests that the latter may be the more important factor. After longer periods of restoration, studies show that DOC levels fall substantially in blocked drains. This may be explained by increases in *Sphagnum* and consequent reduced vigour in vascular-plant cover, thereby reducing the source of DOC production.

Discussion Topic 2 - Windfarms on peat

40) The main physical impacts of windfarms on blanket mire arise from the construction and ongoing presence of service roadways running between the turbine towers. Where these roads must cross deep peat (typically 'deep' is defined as 1 m), the approach presented in many recent EIA proposals for windfarm developments is to 'float' the road on the peat using a geotextile. This is a relatively new technology and there have so far been few, if any, published studies on which to draw conclusions about the likely long-term ecological impacts of these roads.

41) Floating roads cause compression of the peat and therefore subside where the peat is softest and wettest, causing hydrological disruption through changes to both surface-water movement and sub-surface seepage. Disruption to surface flows may give rise to hydrological impacts extending 400 m downslope or more depending on the landform, as described in Discussion Topic 1.

42) Drainage proposals for such roads generally involve the installation of culverts. These can cause further erosion by focusing what was originally a diffuse surface seepage into channelled flow, at least over limited distances downslope. An alternative increasingly being proposed in EIA proposals is that the road be engineered to permit diffuse flow through the road to match the seepage of water through peat. This has not yet been demonstrated or tested in practice.

43) Excavated roads in effect operate as ditches or barriers, depending on whether they are infilled to the bog surface or not, and depending on the material used for the infilling. In either case, excavated roads cut down through the peat to the subsoil. This has significant hydrological implications for the bog not merely at the small-scale local level of the immediately-adjacent peat, but also potentially for the hydrological pattern of the bog system as a whole (*i.e.* the mire unit, or 'mesotope'). The hydrological implications include oxidation, subsidence and vegetation responses as covered in Discussion Topic 1.

44) A recently-produced model for estimating the carbon savings of windfarms on peat is examined and considered to be a useful tool. Much of the model provides a valuable quantification of potential carbon-effects, but certain in-built parameter values may merit further review, and there is evident scope for exploring a range of example input-values to reflect the varying conditions generally found in blanket mire environments.

Discussion Topic 3a - Forestry on peat: the carbon balance

45) Forestry plantations on peat require drainage for successful growth. Some tree species themselves further dry out the peat. Forest drains and shallow furrows result in lowering of the water table to an extent that causes subsidence of the peat and, ultimately, oxidative wastage. Determining the relative proportions of oxidative loss and compression responsible for the subsidence below a plantation requires more study for UK blanket bogs.

46) The amount of subsidence depends on, amongst other things, the bulk density of the peatland. For relatively natural blanket bog, bulk density can be relatively low, resulting in up to 100 cm subsidence in 35 years. Trends indicate that this slumping may extend horizontally beyond the forest edge by distances of 50-60 m over time. Ecological drainage effects as described in Discussion Topic 1 could then significantly extend this zone of impact.

47) Based on the limited field research and modelling work available, existing figures suggest that, in the carbon-budget for a full forest rotation (which also takes onto account the trees, timber, other forest products, forest litter and contributions to soil carbon), the overall loss of carbon from an afforested peatbog may exceed the carbon gains made by the plantation forest within a 100-year time-frame and the tipping-point may be as little as 30 years. Recent efforts to model the relative losses and gains of carbon when conifers are planted on peat have used figures which may not be truly representative. The latest data from one of the sites involved are very different from those used in earlier studies.

Discussion Topic 3b - Restoration of afforested peat: the carbon balance

48) In principle, the carbon contained in all conifer plantations on deep peat in Britain could be matched by restoring a *Sphagnum* carpet 3 cm deep across 700,000 ha of peatland. This area is less than the restoration target set out in the UK Habitat Acton Plan for blanket bog and may therefore represent an achievable goal.

49) The full carbon balance of peatlands restored from formerly-afforested areas in the UK has been the subject of little published research. One model, based on conifer plantations on deep peat in the Flow Country, and taking account of both methane and carbon dioxide losses from the decomposition of trees felled to waste, suggests that under the most rapid rates of peat growth likely under the prevailing conditions, the system would be in carbon credit immediately restoration begins. Lower rates of peat growth could put the system in carbon credit within 50 years. If methane outputs decline as the flooded ditches become covered with *Sphagnum*, then even lower rates of carbon sequestration could be in carbon-credit quite quickly. Furthermore, for a forest felled at 15 years it would take the establishment of a *Sphagnum* carpet of only 1.5 cm thick to replace the lost biomass carbon from the trees.

50) In addition, it seems reasonable to expect that a substantial proportion of the below-ground carbon provided by the former forest would remain in place once the above-ground tree biomass had been removed and the ground re-wetted. This preserved, below-ground forest carbon may therefore contribute additionally and positively to the net carbon balance of the peatbog restoration budget.

51) Furthermore, the overall carbon budget should take into account the carbon losses which would have occurred had the trees not been felled when they were. A further 30-40 years' worth of forest growth could have seen carbon emissions from the system of 4-5 t C ha^{-1} yr^{-1} which will not now happen. This would mean that all but the slowest peat accumulation rate would exceed the total GWP emissions within 10 years or so.

Discussion Topic 4 - Hydro-electric schemes in peat-covered catchments

52) When reservoirs are constructed on, or expanded across, peatland habitat or remnant peat soils for the purposes of hydro-electric developments, this can lead to carbon losses from peatlands through flooding of the peat soil and vegetation. The most immediate loss arises from decomposition of the drowned vegetation. Studies show that carbon dioxide and methane emissions rise significantly in the flooded peatland as the inundated bog vegetation decomposes. The loss of carbon-sequestering capability from this vegetation is also relevant. In addition, the waters of such reservoirs have been found to remain super-saturated with carbon dioxide long afterwards.

53) Within peat-covered catchments where the peat is eroding, the supply of peat soil washed down into reservoirs from the catchment is likely to provide an even more significant long-term source of carbon dioxide and methane emissions. The constant input of organic material from the catchment can lead to additional chronic levels of emission from such reservoirs post-construction, particularly of methane, because the bottom sediments of reservoirs generally experience anaerobic conditions.

Discussion Topic 5 – Peatbogs and climate change

54) Natural peatbogs have demonstrated an ability to grow at fairly constant rates over several millennia despite dramatic climate shifts. This can be explained in terms of the capacity of peat bogs to respond biologically to such shifts. Vegetation composition changes in response to changing climate, thereby maintaining waterlogged conditions across the bog surface and enabling peat formation to continue.

55) Damaged peatbogs have less biological capacity to respond to such changes because the primary architects of the biological response are *Sphagnum* mosses. Restoration of peat bog systems to an active condition should re-establish a more robust and resilient habitat capable of responding more successfully to climate change.

56) The most recent climate-change predictions suggest that conditions may become somewhat drier for at least part of the blanket bog resource, but factors such as cloud cover and increased oceanicity may compensate for such changes. In addition, warmer conditions offer the potential for greater dew fall and mists, as well as continued low-lying cloud cover in the hill districts, all of which offer the potential for uptake by *Sphagnum* of ‘occult’ precipitation – *i.e.* precipitation which does not appear in rain gauges. Lacking a cuticle, *Sphagnum* is much more able than vascular plants to take up such moisture.

57) Construction of ‘climate envelopes’ for species, particularly the major peat-forming species *Sphagnum papillosum*, suggest that even under worst-case climate scenarios of 2050, conditions would still allow *S. papillosum* to grow throughout its current range. As one of the main architects of the bog environment, it would appear that *S. papillosum* will continue to act as a keystone species for the blanket bog habitat even under predicted conditions.

58) Moreover the evidence from the peat archive for the past 7-8,000 years suggests that terrestrial *Sphagnum* species such as *S. papillosum* may grow rather more vigorously and lay down peat at a faster rate than the species which predominate during wetter climate phases. It is therefore possible that the scenarios set out for 2020, 2050 and 2080 may even lead to more rapid peat accumulation than at present. This scenario is, however, predicated on the widespread presence of an active, *Sphagnum*-rich bog surface across blanket mire areas. Unfortunately this is not currently the case because the majority of blanket mires in Britain are haplotelmic, lacking a *Sphagnum*-rich acrotelm.

Discussion Topic 6a – Commercial peat extraction and the carbon balance

59) Commercial peat extraction involves considerable loss of the carbon store and carbon-fixing potential because peat is physically removed for use in horticulture and energy production. The extraction process also results in a range of indirect gaseous and particulate carbon losses from the cut-over surface, as well as from adjacent uncut areas of the bog through drainage processes described in Discussion Topic 1. The gaseous emissions can be up to ten times greater than in a natural bog.

60) Peat milling and extraction, together with processing methods, can result in significant loss of windblown peat. As much as 2.9 t C may be lost from a 100ha site in a single harvesting season.

61) There is considerable loss of particulate and dissolved carbon in the runoff water from extraction sites.

62) Commercially exhausted peatbogs which are then afforested are likely to result in further loss of carbon, at rates which may not be compensated for by the trees.

Discussion Topic 6b – Carbon balance and the restoration of commercial peat extraction sites

63) Rewetting of former peat workings has taken place in a number of areas. The carbon implications of these restoration attempts depends on the type of extraction methods previously used and the nature of the restoration methods employed. Techniques which encourage terrestrial *Sphagnum* species rather than extensive development of aquatic *Sphagnum* species may provide lower methane emissions and re-establish peat-forming conditions more rapidly. Re-shaping and flooding of large areas as part of a restoration programme may thus be less beneficial than methods which encourage raised water tables within, and consequent *Sphagnum* colonisation across, the surviving peat surfaces.

64) Methane emissions from restoration sites do increase following restoration, but are reported to remain lower than in natural, undisturbed sites. This may be due to the loss of methane producing-bacteria from the catotelm when the catotelm is exposed to oxidative conditions during extraction. Such methanogenic microbial populations take time to become re-established following cessation of extraction. If conservation management can produce extensive *Sphagnum* swards before these microbial populations can become established, a significant proportion of the methane pulse associated with re-wetting may be avoided.

65) Restoration of old-style cut over-bogs characterised by abundant trenches and hollows can result in a net carbon sink, but the extensive flooded trenches may also be a significant source of methane. Such systems may require longer periods after restoration for the carbon balance to return to that of a natural mire, although management for rapid *Sphagnum* establishment may significantly reduce this period.

66) On former milled areas, there is clearer evidence that restoration can result in the development of a large carbon sink. This is because there are no deep trenches to flood and in-fill, only relatively shallow slit-drains. Recovery on milled surfaces occurs largely through terrestrial *Sphagnum* species rather than those typical of aquatic areas, and restoration techniques are becoming steadily more successful at restoring such *Sphagnum* swards.

Discussion Topic 7 – Burning and peat bog systems

67) Human-induced burning of peatlands as a form of management for game and livestock appears to have been most intensive in the last 200 years, although there is evidence for burning throughout the peat archive going back to Mesolithic times. In Britain the majority of blanket mire areas show evidence of having been burned, often many times over several millennia.

68) Deep peat dominated by vigorous heather is usually a sign that land management is causing the peat to dry out. Burning of tall heather makes growth more vigorous, leading to further drying out and oxidation of the catotelm peat by healthy vascular-plant root systems. Tall heather can

additionally encourage hot fires which further damage or destroy the ground layer of *Sphagnum* mosses.

69) In contrast, heather which grows within a vigorous *Sphagnum* carpet does not enter the classic 'growth and collapse' cycle which drives the demand for a cycle of burning management. Heather shoots growing in a *Sphagnum* carpet are continually engulfed by the vigorously-growing *Sphagnum*, causing the heather to send out fresh shoots and roots within the *Sphagnum* layer (heather 'layering') and thereby heather growth remains relatively young and vigorous.

70) The carbon balance of burned peatlands generally involves a loss of carbon from the peat store as well as loss of the peat-forming capability. Natural fire frequencies on *Sphagnum* peat bogs in boreal Canada have been found to average 1,150 years. It has been shown that if the fire frequency is 5x to 7x greater than this average (*i.e.* between 164-230 years), then the resulting long-term losses from the carbon store mean that the bog achieves zero carbon sequestration. If the fire frequency is more than this, then the bog goes into long-term carbon deficit. The average fire frequency in the past 100 years or so on British blanket mires has been approximately 30 years.

71) Several studies undertaken using the long-term burning-and-grazing monitoring plots established at Moor House, north Pennines, in 1954, indicate that burning cycles tend to reduce or remove the bryophyte (moss) layer, including *Sphagnum*. Overall photosynthesis may increase because vascular-plant cover is increased by burning, but vascular-plant material is a poor former of peat because it decomposes relatively readily within the acrotelm. Loss of *Sphagnum* by repeated burning (on cycles shorter than 200 years or so) means that the long-term carbon inputs do not match losses, as indicated by the study of Canadian bog systems.

72) Studies indicating that there may be short-term carbon gains from certain types of burning on peatlands in Britain need careful interpretation, particularly in terms of the specific nature of the peatland under study. Such work has not fully addressed the complete carbon balance nor the long-term implications of burning.

73) It is proposed that if burning affects the vascular-plant layer but does not touch the moss layer, then processes such as methane release, CO_2 exchange and DOC release will be affected but long-term carbon sequestration would largely remain intact. If, on the other hand, burning affects the moss layer, then the major contributor to carbon sequestration would also be affected. The precise relationships between methane, DOC, carbon store, vegetation and burning have yet to be clarified.

Discussion Topic 8 – Blanket bog erosion

74) When peat covers the landscape as it does in blanket mire, it protects the underlying sub-soil from erosion. However, blanket mires themselves are currently subject to extensive and often severe erosion in many parts of Britain and Ireland. This is a phenomenon largely unique to British and Irish blanket mire systems.

75) It has been estimated that 16% of the blanket mire resource is eroded, but this figure hides a wide range of regional variation and intensity. The most extensively eroded parts of Britain are probably the Shetlands, the Monadhliaths, the Brecon Beacons, and the Peak District, all of which are estimated to be eroded across 70-80% of the blanket mire area. The majority of severe erosion occurs at high altitudes across broad watershed plateaux.

76) Rates of erosion and carbon loss also vary, with rates of surface lowering ranging from 0.6 mm to as much as 4 or 5 cm per year. Typical values of around 1 cm per year equate to approximately 575 g carbon m^{-2} per year in the form of eroded material. Single storm events can cause losses of 40x the rate at which carbon is normally sequestered.

77) Remarkably, it is still not known whether blanket mire erosion is a natural process, perhaps part of a natural cycle of denudation and regeneration. This leads to problems of both terminology and policy, because eroded bogs are generally described as 'damaged', but if the process is natural this is not a logical term to use. On the other hand, if erosion is indeed a product of human impact, then policy needs to encourage as much restoration as possible.

78) Evidence is accumulating to indicate that erosion is often associated with human activity. In particular, evidence points to early phases of erosion being triggered by Neolithic, Bronze Age, or

even later deforestation around the blanket mire margins. Examples are presented of modern blanket mire landscapes in other parts of the world where deforestation has not occurred and there is little sign of erosion. It is suggested that consideration might therefore be given to the re-establishment of a forest cover on the marginal slopes of blanket bog systems by encouraging natural regeneration, though the current value of this ground as open land would also need to be considered.

79) Internal drainage systems (peat pipes) which cause collapse have also been implicated as triggers of erosion. Examples of these are considered, but so, too, are examples where such peat pipes are constructive features of the bog landscape. The presence of peat pipes is therefore not invariably linked to erosion.

80) A commonly-cited natural trigger of erosion is climate change, either in the past, or in response to the changing climate of the last 100 or so years. In particular the Little Ice Age of 1500 AD to 1850 AD is often cited as a key driver of erosion at this time.

81) Possible anthropogenic triggers of erosion are considered. It is noted that only Britain, Ireland and the Falklands have such eroded blanket mires. Other blanket mires elsewhere in the world (*e.g.* Spain, Tierra del Fuego, Canada) do not possess such dramatic erosion landscapes. It is possibly significant that Britain, Ireland and the Falklands all employ the same land-management systems of hill-sheep grazing linked to burning to improve the vegetation for grazing. In Britain and, to a lesser extent in Ireland, blanket mires are also burnt as part of the management regime for deer and grouse. Grazing has been shown to increase the amount of bare peat, sometimes significantly, but it has not been shown conclusively to *trigger* erosion.

82) Burning is widely acknowledged to have demonstrably initiated erosion, and is generally recognised as being capable of causing very severe damage to blanket bog systems. It is not widely accepted as the prime trigger of erosion because individual fire patches are regarded as local features whereas erosion is widespread and almost ubiquitous on high watershed plateaux. However, evidence of fire in blanket bogs *is* almost universal. Any peat core from any blanket bog is likely to show at least some charcoal bands in the peat archive, and most show an increasing frequency towards the modern era.

83) What is often under-estimated in relation to erosion and the potential link to burning are the necessary recovery times from particular fire events and the time taken for erosion to take hold. In the uplands, recovery times will inevitably be slow. If other factors then intervene, or further fires occur, the nascent erosion complex may develop rapidly. Otherwise, such nascent erosion systems may take many decades or even centuries to develop fully. The link between burning and erosion may not therefore be immediately evident.

84) An example is provided of a study which was originally described as showing that erosion was linked closely to, and driven by, climate change. Closer inspection of the information for the site reveals an alternative explanation whereby burning provides the key trigger for erosion, climate change then merely amplifying certain species responses to the erosion and re-vegetation process.

85) Given the acknowledged evidence that burning can trigger erosion and the absence of any proven alternative cause, it would seem reasonable to regard burning as the default trigger for blanket mire erosion until such time as other causes might be demonstrated as alternative triggers. It would also seem desirable to consider the implications of this in terms of land-use policy in the uplands.

Key Findings

Description of study sites

1. Ecological descriptions of locations used in peat-carbon research need to be *substantially* improved if the results obtained are to be understood and interpreted meaningfully.

Estimates of UK carbon stocks

2. Estimates for the amount of carbon stored in UK peat bogs are *highly reliant* on the definitions used for ‘peat’ and ‘peatbog soils’, and on the generalised parameters used for both peat thickness and bulk density of peat. *UK national carbon estimates for peatbog soils are based on an extremely small number of actual field measurements*.

3. Given these sources of variability, estimates of peatbog extent in the UK range from 1.47 to 5.24 Mha, with UK carbon stocks amounting to *a minimum of 3,121 Mt*.

Condition of study sites

4. Natural bogs consist of two layers – a thin surface layer of soft conductive peat (the acrotelm) covering the somewhat denser peat beneath (the catotelm). Natural bogs are therefore ‘diplotelmic’ (two-layered), *but this fact, and the significance of this diplotelmic structure, is often not recognised in the relevant scientific literature.*

5. Where bogs have lost their protective acrotelm (typically through human action) *they are termed ‘haplotelmic’ (single-layered)*. Haplotelmic bogs represent areas where the long-term carbon store is steadily being lost. *A large proportion of the UK peatbog carbon store is haplotelmic and is thus probably experiencing steady carbon loss as a result of human action*.

6. Many peatbog carbon flux studies have been carried out on damaged, haplotelmic peat bog *but this is not recognised or acknowledged in subsequent published accounts*.

Carbon balance of natural peatbog systems

7. Natural bogs show a clear trend of increasingly abundant hollows and pools towards the north and west of the UK. This *may* mean that these so-called 'patterned bogs' of the north and west release more methane than the bogs of northern England, Wales and southern Scotland, where hollows and pools are much less common. *Nonetheless all natural bogs appear to have an overall carbon balance which, to a greater or lesser degree, is greenhouse cooling.*

Carbon-rich living biomass

8. The carbon stored per unit area in the living biomass of natural *Sphagnum*-rich peat bogs is *very much greater than has previously been realised*. This biomass carbon may be equivalent, for example, to the total carbon stored per unit area in the roots, litter and aerial parts of some 50-year conifer forests.

Carbon and drainage

9. Drainage of peat bogs causes the catotelm layer of a bog to undergo *physical* changes (specifically, subsidence and oxidation of peat carbon) which may extend for significant distances. *The major change to water-table behaviour occurs in the surface acrotelm layer only*, which may become permanently emptied and result in significant ecological changes.

10. *Drained bogs are a substantial carbon source*, losing carbon in the form of gaseous emissions and aquatic carbon losses. Drained bogs subject to restoration management show substantially-reduced aquatic losses. Although methane emissions may temporarily increase at the start of

the restoration process, *these do not exceed the range of methane emission levels observed from natural bogs*.

11. Restoration of drained bog also reduces or halts ongoing losses associated with drainage. *Such 'avoided losses' constitute an important part of the overall carbon budget for bog restoration*. Taking into account all carbon fluxes including avoided losses and increased methane emissions, *restored bogs are substantially more global cooling than drained bogs*.

Plantation forests on peat

12. Studies undertaken to date suggest that, in the short term, conifer forests grown on peat may result in a net carbon gain to the system. In the longer term, however, *plantation forests are acknowledged to result in net carbon losses from the system because eventually the carbon-gains of the forest are outweighed by oxidative losses of carbon from the bog.*

13. Projection of data from studies so far presented suggests that the tipping point, beyond which such conifer forests *appear to cause net carbon losses exceeding the maximum possible long-term carbon gains for the forest and its products, could be as little as 30 years*. This projected estimate *does not include oxidative and particulate carbon losses resulting from site preparation, drainage and felling/re-planting*.

Climate change

14. Current climate-change scenarios indicate that the environmental requirements of the main peat-forming species in the UK *will continue to exist even under worst-case conditions*. Furthermore, peatbogs have shown a robust biological response to climate change in the past, but for this response to be possible under future climate change, *peatbog systems will need to be in as 'active' a state as possible*.

Burning

15. Burning, which is widespread on UK blanket bogs, *occurs 5-10x more frequently in the UK than the threshold at which fire has been shown to result in net long-term losses of carbon from peatbog ecosystems in Canada*. Furthermore, apart from large-scale slope failures, *burning is the only activity which has been observed to initiate erosion in recent times*.

Erosion

16. Widespread erosion of blanket bogs in the UK and Ireland remains an unexplained phenomenon but is a major source of carbon loss from such systems. Some evidence exists to suggest that deforestation of marginal slopes in pre-historic times may have initiated some examples of blanket bog erosion. *Re-establishment of forest cover by natural regeneration on the marginal slopes of blanket bog landscapes may thus be a desirable aim.*

17. Until proven otherwise, it would seem reasonable to assume *that the majority of peatbog erosion in the UK results from human action either now or in the past and thus warrants restoration intervention*.

PART 2

CARBON AND PEAT BOGS

1 Introduction

Purpose of report – not a review but a critical examination of key papers; weaknesses in methodology or interpretation identified; attempt to explain/resolve contradictory evidence; highlighting research needs; confusion over units; inadequate description of habitats leads to confusion over results.

This report is not a review, at least not in the commonly-used sense of this word. It is thus most emphatically not a document which summarises the entire range of existing published information and provides an overall synopsis. While the present report does indeed embrace a large body and wide range of published information, its objective has been to provide a critical examination of various key papers which have increasingly come to inform the debate about peat and carbon in recent years. The report attempts to clarify the nature and origins of the evidence presented in these papers, and tries to explain why sometimes apparently-conflicting evidence emerges when comparing aspects of these and associated papers.

As this is a discussion document, it also gives me, as complier and author, some latitude to speculate on possibilities suggested by available evidence. In many cases this points to the need for further research.

The topic of peat and carbon has been attracting ever more attention in recent years with what feels like an almost exponential growth in papers about the subject in the last 5-10 years. Consequently as fast as a critical synthesis such as the present report is written, it is overtaken by large numbers of papers with new findings. This is a problem for all reviews attempting to present the most up-to-date picture. However, it is apparent that many of even the most recent papers contain certain fundamental weaknesses in methodology or interpretation, and in highlighting these, a report such as the present one perhaps still has a valuable role to play in shaping present and future work in the area.

This work seeks to look at the current state of peat carbon science with a view to identifying key messages relevant to policy. In the process of undertaking this study, the sources of many apparently contradictory lines of evidence have been identified. It is hoped that this work can draw together the science of what is undoubtedly a highly complex subject and render the various technical, confusing and sometimes plainly contradictory elements of the subject into a form which clarifies the issues and makes them more accessible. The present study has also highlighted a number of issues where further research is needed in order to provide the necessary answers to questions which, at the moment, cannot be answered with any degree of confidence.

In the course of undertaking the present work, it has become very clear why those who are not actively involved in peat-carbon work find so much that is confusing and seemingly contradictory in the available literature about the topic. Firstly, difficulties are caused by the use of differing units of measurement for the same feature. Sometimes these differences in units occur between different papers, sometimes they occur within a single paper, and sometimes they even occur within the same sentence (see Appendix 1 for further details). Unless the reader is fully conversant with the various forms of measurement unit employed, this can be quite bewildering. Even specialists have been known to confuse themselves in converting from one set of units to another.

The second factor giving rise to confusion is when insufficient detail is given, or insufficient care is taken, in describing the nature of sites being used for peat-carbon research. A site covered with drains cannot be described as ‘undisturbed’, while a value given for carbon-accumulation rates is seen in a different light if it is understood that the values were obtained from a fen in Finland rather than a bog in Britain.

The use of detailed aerial photographs has been a great help during the present review because then it is possible to see fairly precisely the nature and key features of the research-site used for any given study. Extensive use of detailed colour aerial photography supplied by Getmapping.com has thus formed an important part of the present work.

This is not to decry or dismiss the extremely useful work that has already been undertaken in the field of peat-carbon research. The data obtained to the present time from British peatland systems are undoubtedly of very considerable value. Two things are now needed, however:

Firstly, it is necessary for existing data to be explicitly linked to the ground from which these data have been obtained in such a way that the nature of the ground, and the sampling within that ground, is sufficiently detailed to permit the maximum degree of interpretation from such data.

Secondly, there would appear to be a clear and urgent need for a wider range of studies to be undertaken across a wider range of sites, and across a wider range of conditions within those sites, if the true picture of the carbon balance for British (and Irish) peatbogs is to be revealed.

The various issues raised above thus provide the context of, and possibly some of the explanation for, the present review. Its purpose is to examine critically the key works informing the present debate about the carbon dynamics of peatlands in Britain, to present what is reasonably certain as succinctly as possible, to explore and highlight areas and causes of uncertainty, and to identify work still required.

2 Introduction to Peatlands – particularly bogs

Methodological weaknesses in published research result from failure to consider habitat and ecosystem detail; peat bogs are rain-fed wetlands; blanket mire is an internationally important complex of interlinked bog and fen which develops at the landscape scale; bogs are the predominant component of blanket mires, and consist of two structural and hydrological layers – acrotelm and catotelm; mires (peat-forming systems) can be described using a hierarchy of descriptive levels based on hydrology, morphology and vegetation; elements of the hierarchy are capable of responding to prevailing conditions; mires are thus highly responsive ecosystems with homoeostatic capabilities.

2.1 Why this section (and Appendices 2 and 3) are important

There are two ways of looking at this section. It could form the briefest section in the present report because all those involved in peat-carbon research, and those reading the present account of such research, might reasonably be expected to be fully conversant with the habitat on which all this research and policy focus has been based. Thus, by analogy, an architectural review of new building design to resolve housing deprivation in urban centres would not be expected to contain a substantial section about the fundamental building properties of brick, steel and carbon-fibre. Indeed it might be considered inappropriate to find a section about these topics in such a review because it would generally be assumed that architects are already trained in the basic mechanics and building materials of their profession.

The other way of looking at this present section, however, is to say that it is the most important section of all. If an architect has a poorly-developed or incorrect knowledge of building materials, then anything designed by that architect is likely to be a danger to the inhabitants instead of providing them with a solution. Similarly, without a clear knowledge and understanding of the fundamental building blocks and the structural and functional characteristics of a given habitat, it is equally impossible to construct a model or interpretation of that habitat which is reasonable, realistic and meaningful.

However, the consequences of not understanding these essential ecosystem building-blocks are generally much less obvious for a habitat than for a building, and thus it is far more difficult to be sure whether proposed models and interpretations of a habitat are appropriate or not. Other than rare catastrophies such as the 2003 Derrybrien bogslide, poor habitat models are more likely to result in slow deterioration of the habitat over a number of years rather than causing rapid and evident collapse, and thus the consequences of poor modelling tend to be detectable only by careful and extended monitoring.

This digression into architectural expertise and building failure is relevant to the present report because all the evidence suggests that a section about the building blocks and functional architecture of peatlands is needed - indeed is the most important section - in the present review. It is only through an understanding of these habitat elements that it becomes possible to tease out the origins of the confusion and apparent contradictions which currently beset the issue of peat and the carbon balance.

A failure to take into account even quite basic aspects of peatland ecology is widespread, at least in recent scientific literature. This is particularly so in the field of greenhouse gas studies because researchers and those who peer-review their work are generally not ecologists. A few simple examples will suffice to illustrate the problem:

- Papers are full of minute technical detail about the gas-measurement systems used to measure greenhouse-gas fluxes but the precise nature of the habitat which is under such intense investigation is given simply as ‘blanket bog’.
- Gas-flux chambers are placed in stands of vegetation which may be described in terms of the (very broad) National Vegetation Classification (NVC) vegetation classes but often no indication is given of whether the chamber encloses a T1 low ridge, a T3 hummock, or even an A1 hollow, although each has different gas-flux characteristics.

- Changes in water table tend to result in changes to the species composition and biomass of both vascular-plant and bryophyte (moss) vegetation, with consequent implications for gas-exchange characteristics, but measurements of species composition or biomass are rarely given.
- Measurements of bog water-table are often obtained using dipwells and a measuring rule, although more than 30 years ago Goode (1970) described how water levels in dipwells on blanket bog could change by as much as 20 cm simply from the weight of the observer. Consequently he used elevated boardwalk and devised a remote-arm measuring device to measure water tables in his dipwells. Most published studies involving dipwells today make no mention of boardwalk or remote measuring arms.

Perhaps the biggest failing of all, in terms of recent peat-carbon work, has been the failure to understand that bogs are *responsive* ecosystems with homoeostatic mechanisms which are not far removed from those found in living organisms. Unlike mineral soils which are essentially a product of weathering and erosion (albeit mixed with a relatively small proportion of organic matter and a large number of micro-organisms), peat bog soils are a direct product of vegetation growth.

Through changes to the vegetation, peat bogs and the peat soils they generate have the potential to respond to environmental change in much the same way as a living organism might respond. The varying width of tree rings is a readily- and widely-understood response to shifting weather patters and climate change, but what is less widely-understood is that features having many similarities to tree rings can be found in the equally-thin layers of peat which are successively deposited in a bog over millennia. These narrow bands of peat tell the same tale as tree rings, but often over a much longer time-scale.

The peat archive thus shows that for the last 8-9,000 years, bogs have accumulated peat at a remarkably constant rate, despite the very substantial shifts in climate which have occurred over this same period (*e.g.* Belyea and Malmer, 2004). This is made possible because the vegetation, which creates the peat, changes composition in response to changing conditions of climatic wetness, thereby altering the hydrological characteristics of the bog in such a way that peat accumulation remains relatively unchanged despite the altered climatic regime.

This organism-like responsiveness is, however, rarely incorporated into models of climate-change impacts on peat bog systems (*e.g.* Hossell, Briggs and Hepburn, 2000; Berry *et al.*, 2003), or into drainage models of peatland systems – drainage to some extent being a human-induced analogue of climate change.

Key characteristics of the peat bog environment are given in Appendix 2 and 3 and the reader is urged to explore the information given there. Within the text, therefore, reference will be made to Appendix 3 from time to time. For completeness, however, a brief summary of the bog environment is given below, prior to venturing into the peat-carbon story itself, which begins in Section 3.

2.2 Peat – a soil of biotic origin and responsiveness

2.2.1 Peat soils – wetland systems

Peat is a soil distinguished from other soil types by its relatively high content of organic matter, which may range from 30% to virtually 100% organic matter depending on the definitions and conventions used. The organic-matter content results from a combination of plant growth and waterlogging, the latter reducing oxygen diffusion to levels which are so slow that decomposition of the dead plant-matter uses up this oxygen faster than it can be supplied. Consequently conditions in the peat rapidly become anaerobic.

Anaerobic conditions reduce decomposition rates to such an extent that not all the *in-situ* dead plant matter produced each year can be recycled. A proportion of this plant matter is therefore retained each year as semi-waterlogged plant litter. Over several decades this litter becomes overlain by newer litter and ultimately the preserved plant material becomes incorporated into the permanently-waterlogged mass of similarly-preserved plant material accumulating beneath the thin living layer of the bog. This waterlogged mass of semi-decomposed plant material is the thick black or brown soil which is commonly referred to as ‘peat’ in Britain or ‘turf’ in Ireland.

The unusual feature about this peat is that it is a direct product of the vegetation which created it. Consequently the characteristics of this peat soil reflect the nature of the vegetation which created it, while the vegetation itself reflects prevailing hydrological and nutrient conditions. Thus the nature of the peat itself varies in response to these conditions, and in the case of bogs these conditions are controlled almost exclusively by climate and landform.

2.2.2 Bogs – wetlands dependent upon atmospheric moisture

Areas of the landscape in which water tends to collect give rise to peat accumulation caused by waterlogging from groundwater. The water chemistry and flow patterns of such systems reflect the nature of the surrounding catchment in terms of catchment chemistry and flow patterns. Such groundwater-fed peat systems are commonly termed 'fens', though the technically-correct term is 'minerotrophic mire' ('mire' being any peat-forming system).

In regions where precipitation is sufficiently regular throughout the non-frozen period of the year, peat can accumulate to the point where it rises above the surrounding mineral ground-water table and continues to accumulate thanks to sufficient, direct, precipitation inputs to make up for the small quantities of water lost through very slow downward seepage through the peat (it is worth recalling that water is, of course, quite heavy). Such mounds of precipitation-fed peat are termed 'raised bogs' – bog being the term given to peat-forming systems which rely exclusively on atmospheric precipitation, though the technically-correct term for any exclusively rain-fed peat-forming system is 'ombrotrophic mire' (see *e.g.* Lindsay, 1995).

2.2.3 Blanket mire: a landscape of peat-forming systems

In areas where precipitation is sufficiently regular throughout the year, the ground surface itself may remain so wet that peat-forming species are capable of becoming established directly on the ground surface over whole landscapes. The vegetation gives rise to an increasingly thick layer peat which is capable of overwhelming smaller features in the mineral-soil landscape to create a gentler, smoother landscape blanketed with varying thicknesses of peat in what is thereby termed 'blanket mire'.

The term 'mire' is used here in preference to the exclusively rain-fed 'bog' because a blanket mire landscape is a complex peat-forming system which is significantly influenced by the landform over which it develops. Parts of the blanket mire system may be ombrotrophic bog but other parts, where surface- or groundwater-flows tend to collect, will certainly be minerotrophic fen, but these various peat-forming systems tend to be interlinked within the overall blanket of peat. Thus the landscape as a whole is more correctly termed blanket *mire* rather than the more restricted term 'blanket bog' – the latter being reserved more accurately to describe certain rain-fed components of the blanket mire landscape.

Blanket mire is one of the most extensive examples of near-natural (or at least semi-natural) habitat remaining in Britain, and is a globally-rare peatland type for which Britain is internationally recognised as being a 'type location' (Lindsay *et al*., 1988). It is also perhaps the peatland type in which all the various building blocks and functional units characteristic of a peatland system can be most clearly seen.

2.2.4 Bog : a system with two layers

A natural peat bog generally possess two distinct layers. The upper layer is thin – anywhere from a few centimetres to 75 cm – and represents both the layer of living plant material and the zone of water-table fluctuation (and thus of direct oxygen penetration). This layer is known as the 'acrotelm' (Ivanov, 1981), and represents the narrow zone of relatively rapid oxidative decomposition in the bog. The acrotelm typically consists of a *Sphagnum* moss carpet. It therefore comprises a structure with loosely-packed vertical plant stems and small but quite stiff side-branches, which together create a relatively open lattice structure in the upper parts near the bog surface. Bryophytes differ from higher plants in dying upwards from the base, their apical regions continuing to grow upwards as their dead remains are left behind and below them. Thus while the uppermost parts of the living *Sphagnum* carpet form an open, geotextile-like structure, only 10-20 cm below the surface this structure and order begins to break down as the dead branches collapse and stems begin to weaken and fail (Clymo, 1992).

Below the base of this increasingly disordered acrotelm layer is the whole thickness of accumulated peat, which may be 10 metres or even more. This generally much thicker layer of peat is known as the 'catotelm' (Ivanov, 1981). It consists of plant remains which are now largely broken into small, non-oriented fragments, creating a remarkably amorphous structure. This catotelm peat always lies beneath the water table and is thus the zone of the oxygen-free Archaean world (see Appendix 2). The catotelm thickness gives the lowland raised mire system its domed shape, and is what smothers or smoothes the shape of the underlying mineral landform with thick layers of peat in blanket mire landscapes.

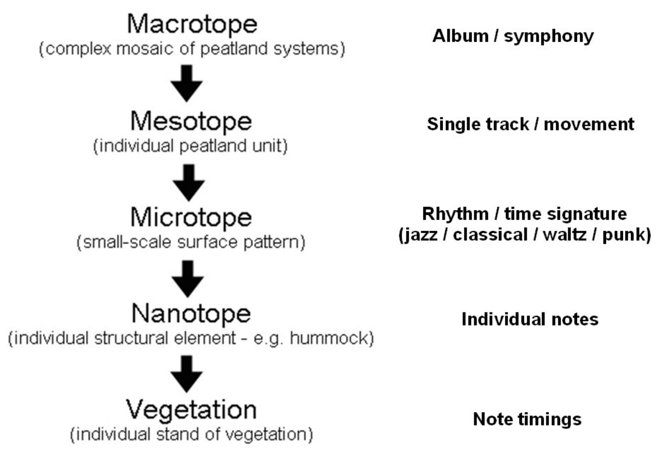
2.3 The building blocks and functional units of blanket mire systems

A blanket mire system can be dismantled into a number of distinctive components which relate to each other in a hierarchical way. This hierarchy of components ranges from major landscape units to individual patches of vegetation, and all are based on, or linked to, hydrological characteristics. The hierarchy, from largest to smallest, comprises:

- macrotope
- mesotope
- microtope
- nanotope
- vegetation

Using these various terms it is possible to assemble or disassemble a blanket mire into its functional components, in the same way that a piece of music consists of many elements which come together to make a coherent sound, song or symphony. The equivalence between these two systems can be seen in Figure 1. A musician must understand all the components of the music in order to perform it well, and so it is with a blanket mire system – all components must be understood and used appropriately if the system is to perform well under any form of management or is to be studied successfully. A brief synopsis of these various components will be given here, and the reader is referred to Appendix 3 for a more detailed account.

Figure 1. The peatland hierarchy of elements and their musical equivalents.
The various components which make up the building blocks and functional units within any peatland system relate to each other in a hierarchy. This is most readily seen in blanket mire landscapes, but is perhaps most easily envisaged conceptually in terms of the various elements which make up a piece of music. Any errors at any point will mean that the musician cannot perform well, but it is worth observing that the lowest parts in the hierarchy – the notes and note timings – are probably the most influential in terms of evident performance failure.



2.3.1 Macrotope

The macrotope consists of all peatland systems which are adjacent to, and hydrologically linked with, each other in a landscape. Thus a range of bog and fen units may share common boundaries with water flowing directly from one to the other. The outer boundary of the macrotope is bounded by landscape elements which are not peat-forming, such as rock outcrops, rivers, and steeply-sloping ground where water is shed too rapidly for peat to form.

2.3.2 Mesotope

The mesotope represents the individual, identifiable peatland units. In blanket mires these are largely described in terms of their landscape position, such as saddle mire, or watershed mire, but the generally-smaller fen mesotopes are described in terms of their hydrology – *e.g.* spring fen, basin fen.

2.3.3 Microtope

Most peatland systems which are not permanently inundated tend to display some form of distinct surface structure consisting of raised elements and lower elements or depressions. The classic description given to this for bogs is 'hummock-hollow', although the pattern is often far more complex than this term suggests. The microtopes of a peatland unit (mesotope) are the various forms of pattern which occur across the unit. Some patterns may be linear, some smooth, some without orientation, but all parts of a mesotope will display some form of microtope pattern.

2.3.4 Nanotope

The small-scale structural elements which create microtope patterns are called nanotopes (in the past they have also been termed 'microforms'). These are the smallest structural components of a bog within the formal hierarchy, and consist of features such as 'hummock' or 'low ridge'.

2.3.5 Vegetation

The final element within the hierarchy is the vegetation itself. Vegetation can also be described in hierarchical terms from broad habitat types to small-scale vegetation patches, but within the peatland hierarchy it is the small-scale patch which is generally of most interest. The UK National Vegetation Classification (Rodwell, 1991) describes bog vegetation only at the scale of mesotope or even macrotope. In contrast the European Union CORINE and EUNIS vegetation systems provide vegetation units which fit more easily as the smallest descriptive element within the peatland hierarchical structure (EU EUNIS bogs web-page) because continental phytosociology has a long tradition of recognising vegetation stands on bogs at the nanotope scale.

2.3.5.1 Bog vegetation – three competing components

What is perhaps not widely appreciated is that bog vegetation consists of three competing elements, but an understanding of this dynamic relationship does much to clarify the nature of any given vegetation stand. Rodwell (1991) alludes to these three elements in assigning distinct vegetation classes to bog hollows and to the terrestrial parts of a bog pattern, and then makes clear that the driest forms of bog vegetation are closely related to heathlands. The distinction is made more explicit within continental phytosociology where bog hollows are assigned to fen systems, dry hummock-tops are assigned to heath, and the remainder of the vegetation is identified as the true bog vegetation.

Synusial phytosociologists of western Europe (*i.e.* plant ecologists studying small-scale patterns of vegetation) are more explicit still and regard the dwarf-shrub communities on the hummock-tops of bogs as outliers of heathland vegetation, while the sedge-dominated communities of hollows are seen as fen-swamp outliers. The *Sphagnum*-communities which range from hummock-top to *Sphagnum* hollow are regarded as the main assemblage of distinctively bog vegetation communities. The synusial approach explicitly recognises that if conditions become drier, the heathland vascular plants gain a certain ascendancy whereas if conditions become wetter then the swamp communities expand. Meanwhile the core *Sphagnum* bog communities remain as the backbone of the system (Philippe Julve, pers. comm.).

2.3.5.2 Sphagnum vegetation – distinct zonations

Exploring the concepts of these small-scale vegetation patterns further, within the (normally) *Sphagnum*-dominated vegetation which makes up the majority of a natural bog surface, the undulations of the nanotopes and microtopes provide a narrow vertical range within which competing species are distributed. This results in the relatively sharp delineation of vertical zones, often no more than a few centimetres high, which provide the niche for a given assemblage of bogland species – particularly the various *Sphagnum* species (see Figure 2).

The fierce competition for living space within these vertical zones means that species are distributed according to their competitive ability rather than their physiological optimum. Thus, for example, *Sphagnum compactum* has a wide physiological and ecological tolerance but is out-competed for the central part of this ecological range by *Sphagnum papillosum*. Consequently in the field, *S. compactum* is found in two distinct habitats – very dry bog surfaces and very wet surfaces – because these are the only parts of its range in which it is able to out-compete the more robust *S. papillosum*. Changing the prevailing conditions to a drier state, however, means that *S. compactum* may be capable of displacing *S. papillosum* across parts or all of its range across a site.

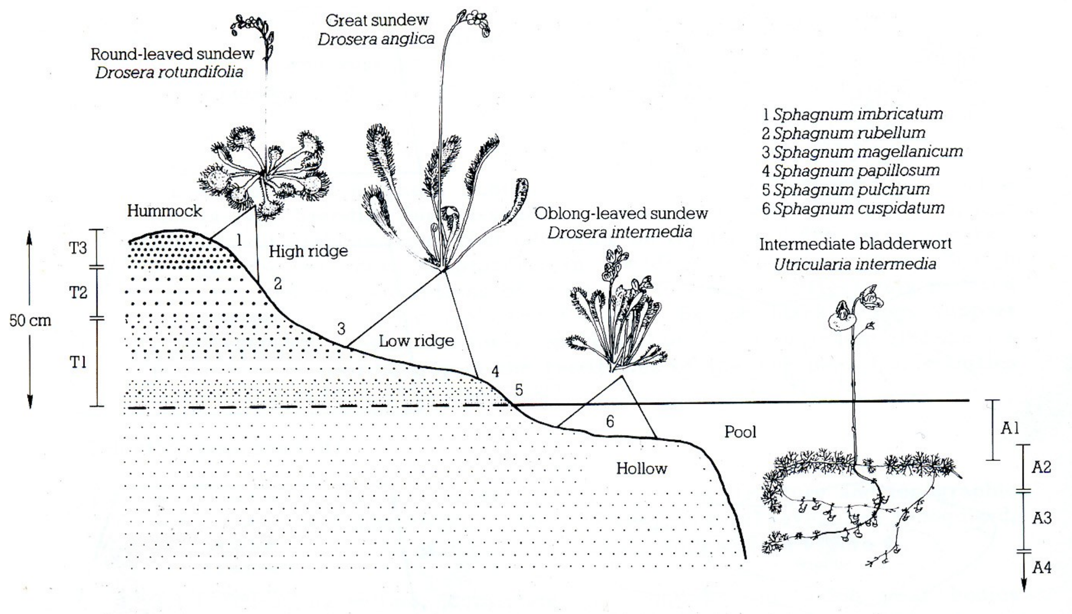


Figure 2. The zonation of vegetation types within the microtopography of a bog.
It can be seen here that different *Sphagnum* species occupy distinct vertical ranges within the typical micro-topographic structures found on a bog, as indeed do the various carnivorous plants found in such habitats.

2.3.5.3 Sphagnum zonation – a responsive system

The capacity of one *Sphagnum* species to replace another when conditions change is of absolutely fundamental importance to the long-term stability of a bog system, because it is by such shifts in species composition that overall hydrological stability can be maintained across the site in the face of substantial shifts in climate, or because of other factors influencing the hydrological regime (Goode, 1973; Barber, 1981).

Thus under drier conditions when a mineral soil may tend to shrink and crack, a *Sphagnum*-rich bog surface is more likely to display a shift to *Sphagnum* species more typical of the hummock-tops because these species are more tolerant of dry conditions and can bind water more effectively than species of wetter conditions. The overall effect is therefore to maintain waterlogged conditions even though inputs have diminished. Interestingly, these hummock-forming species tend to be more effective peat-formers than species found lower in the microtopography. Thus drier conditions may actually stimulate *increased* peat formation, an idea based on discussions in Sections 7.1.3 and 9.1.5 of the present report. This is not a feature generally recognised in discussions about peat and climate change, but is a logical and obvious possible scenario which emerges from an understanding of how the ecology of a bog system interacts with the hydrology to create rather powerful homoeostatic mechanisms.

3 Carbon stores and how to measure them

Sheer scale of the carbon store distinguishes peatlands from other habitats; calculation of the carbon store requires values for extent, thickness and carbon density; there are many estimates for extent of UK peat but we are no nearer an "accurate" figure because totals depend on the mapping objectives; so far, no survey has explicitly mapped peat in a consistent way across Britain; current estimates of peatland extent vary between approximately 1.5 million hectares and 5.2 million hectares; measurements of peat thickness are relatively scarce; thickness can vary substantially over distances of less than 100 m; minimum thickness depends on definition of peat but is generally 30 cm; maximum thickness may exceed 10 m.

The carbon store could be described as the bottom line for peatlands and the carbon debate. True, there are important questions about current accumulation rates and carbon emissions resulting from decomposition, but these are common to all habitats – and will be explored later in the present document. What separates peatlands from practically all other habitats and makes them so unusual is the enormous quantity of carbon they store for periods approaching geological time. The present coal-fields of the world were once peat swamps. Without these fossil peatlands there would probably have been no Industrial Revolution, no anthropogenic CO_2 release, no anthropogenic global warming and therefore no interest in peatlands as carbon stores.

'Modern' peatland stores in Britain have been slowly accumulating carbon for anything between 2,000 and 10,000 years. A few sites may even date back as far as the height of the last period of glaciation 21,000 years ago (Rinns of Islay JNCC website). Not all peat bog sites are currently adding fresh peat to the store. Some peat deposits have lost their peat-forming vegetation and now, for example, support agricultural grass leys, forestry plantations, commercial peat mining, or even sugar beet crops. These altered sites nevertheless retain a peat store as long as at least some of the original catotelm peat remains.

The important question today, of course, is just how much carbon continues to be held in the UK's peatlands whatever their current vegetation cover? Answering this question requires three things:

- accurate figures for the extent of peatlands;
- accurate figures for the thickness of peatlands;
- accurate figures for the carbon density contained within these peatlands.

Given the long history of resource mapping in the UK, and particularly the advent of technologies such as aerial photography, satellite remote-sensing, ground-penetrating radar, and a host of sophisticated test-systems for analysing the mechanical properties of soils, it would seem reasonable to expect that these three sets of figures would be readily available. As we shall see, this is very far from being the case.

3.1 Estimating the extent of UK peatlands and peat bog habitats

Despite the long history of resource mapping and advances in technology referred to above, it is surprisingly difficult to come to an agreed figure for the extent of peat in Britain – even approximately. Although interest in peat-based carbon stores has in recent years shown a marked upsurge associated with concerns about climate change, most estimates of this store are partial, or country-based rather than for GB or the UK as a whole. Table 1 is by no means comprehensive, but presents a substantial number of differing estimates for the peat resources of, variously, England, Wales, Scotland, Britain, Northern Ireland and the UK. Various trends to emerge from such data are then displayed in Figure 3 and discussed below.

Robertson and Jowsey (1968) were the first to provide some sort of estimate for the peatland resource of the UK, proposing what can only have been a well-informed estimate amounting to 1,581,841 ha for the UK. The precision of the number suggests, perhaps, a degree of precision and detail in the original measurement which is not in fact the case. While detailed field survey was carried out, it was by no means comprehensive across the UK, nor sufficiently fine-scale, to provide true total estimates accurate to 1 hectare. Nonetheless, the general scale of the figures proposed has shown a certain degree of robustness over the decades, offering a fairly consistent estimate for at least the minimum amount of (relatively deep) peat for the UK as a whole.

In the 40 years since Robertson and Jowsey's (1968) original estimates, considerable numbers of differing figures for the extent of peat in the UK, or parts of the UK, have been proposed. It is perhaps not surprising that these estimates show such variability because that is precisely what they are – estimates, rather than detailed, accurate measurements based entirely on ground survey. Where the estimates are based on soil surveys, there is uncertainty associated with definition and mapping of peat-based soil classes. There is also the question of mapping resolution, because a survey which samples at 1 km intervals (effectively a scale of 1:50,000) will certainly produce different results from field survey measurements based on a 1:10,000 scale map. Where peat depth is not physically measured but estimated on the basis of vegetation cover, there are questions over whether a particular piece of ground is genuinely peatland or is merely supporting a vegetation which resembles that found on true peat soils.

These estimates are, of course, coupled to an ever-expanding level of understanding about peat soils. As the knowledge-base improves, so we might expect recent estimates of deep peat to be more 'correct', and for the differences between successive estimates to diminish. However, this is not the case. Indeed it is not even necessarily so, because the objectives behind the creation of any area-estimate are equally important and these objectives may differ from survey to survey. A map of peat based on aerial photo-interpretation and remotely-sensed vegetation cover is likely to include substantial areas of shallow peat, and if the objective is to produce a map of 'potentially peat-forming vegetation', this is acceptable. In contrast, a survey concerned with 'exploitable peat resources' will tend to focus on areas of demonstrably deep peat but will seek to exclude areas of peatland vegetation on thin peat and will also tend to exclude small, scattered deep deposits in remote areas because they are not commercially viable even though they may contain substantial depths of peat.

Soil surveys traditionally use the 'soil series' as a mapping unit, and the various soil types thus identified often consist of mixtures made up from differing soil conditions in defined, standardised proportions. Accurate mapping of the peatland component within these mixtures then becomes a matter of judgement for each individual attempting to produce an estimate. Acknowledging that a very wide range of estimates now exists for the extent of peat in the UK, certain other aspects of the data contained in Table 1 and Figure 3 also merit further examination and reflection.

It would seem, for example, that estimates for the extent of peat in England have been shrinking in recent years. The general trend-line in Figure 3 (top left) indicates an overall reduction from a little more than 350,000 ha in 1968 to around 250,000 ha today. Does this reflect a growing understanding of how to define peat soils and a more accurate mapping programme, or does it simply reflect differing approaches to mapping peat?

In this case, the explanation seems to come at least in part from the fact that Lindsay and Immirzi (1996) mapped deep peat only (>1 m thickness) and Tallis and Meade (1997) then took only the blanket mire component of that figure. There is thus a very explicit exclusion of thinner peat from these figures, whereas there is a suggestion that earlier figures contained at least an element of shallower peat soils.

The extent of peat in Wales has followed two clear and distinct paths through the last four decades (see Figure 3 : top centre). One set of estimates at around 75,000 ha appears to represent the extent of relatively deep peat, while the other set of figures remains fairly consistent at around 160,000 ha. This higher figure is explicitly stated by authors such as Taylor (1983) as including peat less than 90 cm thick. The reader therefore has two choices for Wales. Is it the extent of the deep (mainly blanket) peat resource across Wales, or instead the whole spectrum of peat soils from 30 cm to several metres thickness, which is of interest?

Some authors provide data only for the combined peatland resources of England and Wales. When these estimates are mapped (together with the individual data for the two countries combined), a dramatic decline in the scale of estimated extent can be seen to have occurred over the last four decades (Figure 3 : top right). This is not simply a repeat of the diminishing estimates noted above for England alone. These diminishing estimates of extent are largely values in their own right, and none is as dramatic a downsizing as that given by the Countryside Survey 2000 (Haines-Young *et al*. : 2000). The 180,000 ha given by Haines-Young *et al*. (2000) for England and Wales combined is substantially smaller than the lowest estimate given by anybody for peat in England alone. The explanation for this particular value undoubtedly lies in the sampling strategy adopted for the Countryside Survey, but unravelling the precise correlation between this and other estimates lies beyond the scope of the present report.

Table 1. Estimates of total peatland extent (hectares) for England, Scotland, Wales, Northern Ireland, and combinations of these, given by a range of sources (and often including some fen-peat deposits). Orange shading indicates estimates which incorporate some original field-based mapping data. Non-shaded estimates are based on previously-published data, or on re-interpretation of existing soil-mapping units, but add no new original field data. Note that some values represent 'extent of peat soil', rather than 'living mire' or even 'degraded mire habitat'; the current status of such peat soils may thus be found in any one (or more) of the various 'condition classes' described and illustrated by [Section 11] and [Plates 15 – 39] of Lindsay (1995).

Author	Date	England	Wales	England and Wales	Scotland	Britain	N. Ireland	Total (minimum) UK [1]	Sampling effort	Basis of estimate
Robertson and Jowsey [2]	1968	361,690	158,770	520,460	821,381	1,341,841	240,000	1,581,841	Soil survey field mapping	Soil Survey and explicit (exploitable?) peat-resource survey; (possibly including peat>30 cm depth).
Taylor and Tucker	1968		84,200						Some field mapping to supplement BGS and Soil Survey categories	Based on British Geological Survey 1:50,000 Drift Maps, showing mapped areas of peat having 1 m thickness or more; plus Soil Survey data, and additional field survey. Overall, figures are for peat >0.91 m deep.
Hammond	1979	-	-		-	-	166,860		Soil Survey field mapping	Soil-survey categories used to identify extent of 'peat', rather than explicit mapping of peat; extent of peat thus depends on interpretation of categories.
Taylor [3]	1983	361,960	158,770		821,381		166,860	1,508,701		Based on Robertson & Jowsey (1968) and Hammond (1979)
Macaulay Institute for Soil Research	1984	-	-	-	765,000	-	-	-	Sampling only at 1 km grid intersections	Soil Survey of Scotland sampling strategy cannot take into account expanses of soil outside 1 km grid intersections. According to Chapman *et al* (2001) excludes complexes containing significant areas of peat.

Author	Date	England	Wales	England and Wales	Scotland	Britain	N. Ireland	Total (minimum) UK [1]	Sampling effort	Basis of estimate
Bather and Miller	1991	-	-	520,000	820,000	1,340,000	240,000	1,580,000	-	Peat industry figures, largely based on Robertson & Jowsey (1968)
Birnie *et al*.	1991	-	-	-	789,000	-	-	-	Actual mapping	Aerial photo interpretation and field mapping to produce mapped soil units. Area given for 'peat' depends on decisions about mapping categories assigned to 'peat'.
Immirzi *et al*. [4]	1992	-	-	466,700	766,000	1,232,700	240,000	1,472,700	-	Cited values are from Soil Surveys of England and Wales, and of Scotland, thus based on soil-survey categories identified as 'peat'; plus Robertson & Jowsey (1968) for NI
Barr *et al*.	1993	-	-		-	2,237,000	-	[2,368,000]	509 x 1 km squares	Countryside Survey 1990; Sampling effort represents 0.2% of total land area; broad habitat category = 'wet heath & saturated bog' and thus probably >45 cm deep, but missing some peat with 'open heath', peat beneath forestry, and peat under agricultural use (grassland, crops)
Cannell *et al*. [5]	1993	300,000	70,000	370,000	1,742,000	2,112,000 5,112,000	-	[2,243,000] [5,243,000]	Some additional field data	Soil Surveys of England & Wales, and of Scotland; additional detailed mapping data for peatland >45 cm provided by Soil Survey of Scotland; total including estimated peaty gleys and peaty stagnopodzols (<45 cm) also provided by Cannell *et al.* (small font).
Pfaden-hauer *et al*.	1993	-	-	-	-	-	-	1,508,700	?	?

Author	Date	England	Wales	England and Wales	Scotland	Britain	N. Ireland	Total (minimum) UK [1]	Sampling effort	Basis of estimate
Lindsay and Immirzi [6]	1996	252,750	163,000	415,750	1,094,750	1,510,500	-	[1,641,500]	Mapped and measured polygons-	BGS maps, showing mapped areas of peat having 1 m thickness or more. Figures for Wales largely based on Taylor (1983).
Burton; Shier [7]	1996	402,700	157,600	560,300	1,137,500	1,751,000 4,871,400	171,300	1,922,300 5,042,700	?	Based on mapped data from Soil Surveys of England & Wales, and of Scotland, and of NI; thus minimum peat depth ranges from 30-50 cm. Small-font data include 'peaty soils'
Milne and Brown [8]	1997	-	-	356,800 453,700	2,625,300 4,339,000	2,982,100 4,792,700	-	[3,113,100] [4,923,700]	-	Calculated from figures given for Soil Surveys of England & Wales, and of Scotland, using 'blanket and basin peat' for Scotland, and 'raw peat soils' for E&W; also limited Countryside Survey 1990 data incorporated; <u>may</u> be based on peat depth >50 cm; small-font data explicitly <u>include</u> 'peaty soil' types listed by Milne and Brown.
Tallis and Meade [9]	1997	214,000	78,000	292,000	1,056,000	1,348,000	131,000	1,479,000	-	Extent of deep (1 m+) <u>blanket bog</u> only; based on selected data from Lindsay & Immirzi (1996), Yeo (1997) and Hammond (1981)
Haines-Young *et al.* [10]	2000	-	-	180,000	2,038,000	2,218,000	148,000	2,367,000	569 x 1 km squares (366 in E&W, 203 in Scotland); 621 x 1 km squares in NI	Countryside Survey 2000; Sampling effort represents 0.2% of total land area; broad habitat category = 'Bogs' and thus probably >50 cm deep, but missing some peat with 'dwarf shrubs', peat beneath forestry, and peat under agricultural use (grassland, crops)

Author	Date	England	Wales	England and Wales	Scotland	Britain	N. Ireland	Total (minimum) UK [1]	Sampling effort	Basis of estimate
Joosten and Clarke [11]	2002	-	-	-	-	-	-	1,750,000	?	?
Monta-narella *et al*. [12]	2006	-	-	-	-	-	-	4,441,100	Tested with 12,000 sample points	Derived from European Soils Database, CORINE vegetation database and climate data; tested using Soil Survey of E&W and Italian Soil Survey field data; 'peat' defined as containing more than 30% organic matter (*i.e*. includes 'peaty soils' and deep peat); peat depth probably approx. 25+cm.
ECOSSE [13]	2007	-	70,600	-	881,770	-	-	-	-	Based on data from Soil Survey of Wales (Rudeforth *et al*., 1984) and Soil Survey of Scotland; thus based on interpretation of soil-survey classes. In Wales, 'peat' has minimum depth of 40 cm; not clear what minimum depth used for Scotland.
Holden *et al*.	2007a	-	-	357,500 458,900	-	-	-	-	-	Based on Soil Survey of England and Wales, 'raw peat' category. Peat depth at least 30-50 cm. Small font gives total extent including 'earthy peat'; peat depth probably at least 25 cm.
DEFRA Soil Strategy Consul-tation	2008	252,000	-	-	-	-	-	-	-	no information given

[1] [UK minimum total] based on figures for Britain plus smallest figure given in the present table for Northern Ireland (*i.e.* plus 131,000 ha) if no total UK figure given.]

[2] Chapman *et al*. (2001) state that Robertson's (1971) figure for Scottish peat [identical to Robertson & Jowsey's figure] was obtained before completion of the Scottish peat survey, but may have included peat as shallow as 30 cm.

[3] Taylor and Tucker (1986) state that the figure for extent of peat in Wales includes much thin hill peat (less than 90 cm deep).

[4] Immirzi *et al*. (1992) give various figures obtained to that date shown in the present table. The particular values shown here against Immirzi *et al*. (1992) are the cited data for the Soil Survey of Scotland (1:250,000) and the Soil Survey of England and Wales (1:250,000).

[5] Upper large-font values based on peat more than 45 cm deep. Lower, smaller font values represent total area including estimated extent of shallow peat (less than 45 cm) consisting of 'peaty gleys and peaty stagnopodzols'. Cannell *et al*. (1993) also exclude fen peat from their figures.

[6] Based on British Geological Survey Drift Series, showing peat only when it is at least 1 m deep in England and Scotland. BGS data no longer exist for much of Wales. Total figures for Wales were therefore supplemented by (and so dominated by) Taylor (1983) who mapped peat >0.5 m deep. Fen peat soils were also excluded from these figures.

[7] Extent of peat given by Burton (1996) for Britain, based on composite data from Soil Survey of England & Wales, and Soil Survey of Scotland, and by Shier (1996) for Northern Ireland. Definition of 'peat' therefore differs between England & Wales, and Scotland, and NI. Data are also given (in small font) for total extent including 'peaty' soils in the columns for Britain and the UK. [see IPS website and Montanarella *et al*. 2006]

[8] Data give extent of 'raw peat' soils for England and Wales, and extent of 'blanket peat' and 'basin peat' soils for Scotland. Milne & Brown (1997) state that Scottish figures have an estimated 20% margin of error. Data for England and Wales have 'negligible error'. Data also given (in small font) for total extent including 'peaty' soils. For England and Wales, this means including extent of 'earthy peat soils' but not 'humic' soils, while for Scotland the figure includes 'peaty gley, peaty gley (GW), peaty gley (SW), peaty podzol and peaty ranker'.

[9] Estimates for 'blanket bog' only.

[10] Based on the Broad Habitat class 'Bog', which may thus exclude some areas of deep peat supporting 'Open dwarf-shrub' vegetation.

[11] Estimates based on peat >30 cm deep and organic matter >30%, and for all soils whether bog or fen, and whatever their current 'condition class' (*sensu* Lindsay, 1995).

[12] Montanarella *et al*. (2006) use the European Soil Database, together with a number of refining steps, including comparison with more than 12,000 field-sampling points in the UK and Italy, to produce a map of peat with an organic content greater than 25%. The map thus embraces both deep peat and 'peaty' soils.

[13] The ECOSSE Report (2007) uses the same data for Scotland as used by Immirzi *et al*. (1992) within this table, but arrives at a different (somewhat larger) figure. It is not clear why this should be.

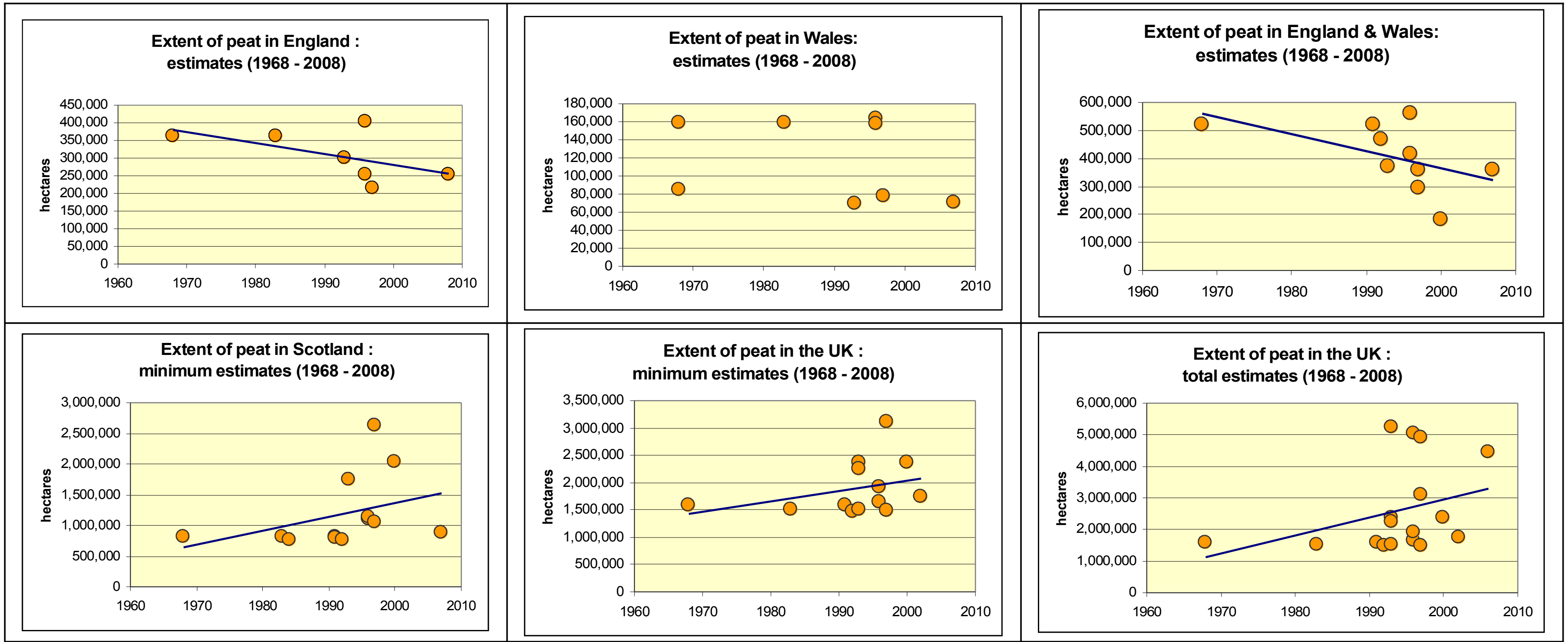


Figure 3. Graphs indicating the estimated extent of peat across England, Wales, Scotland and the UK (estimates produced 1968 – 2008 : See Table 1.).
It can be seen (top left) that estimates of peat in England have shown a distinct downward tendency over the years, but have never fallen below 200,000 ha. Estimates for Wales (top centre) show two distinct groups of values – one around 75,000 ha associated with deep peats, while values of around 160,000 ha include peat soils less than 90 cm thick. The combined data for England and Wales (top right) also show a distinct downward trend, with the lowest estimate below 200,00 ha, while the highest estimate (or deep peat soils) is somewhat less than 600,000 ha. Estimates for Scotland (bottom left) show a clear upward trend, with no estimate less than 760,000 ha while the highest (deep peat?) estimate exceeds 2.5 million ha. Estimates for the UK also show a distinct upward trend, with values of around 1.5 million ha as the minimum extent while the maximum (deep peat?) estimate exceeds 3 million ha. If estimates for all peat soils, including those less than 90 cm are included, the range of values for the UK extends from 1.5 million ha up to more than 5 million ha.

Nonetheless a number of other estimates for the combined extent of peat in England and Wales fall substantially below Robertson & Jowsey's (1968) estimate of more than 500,000 ha. More recent estimates by Milne and Brown (1997) and Holden *et al*. (2007) lie closer to a figure of some 360,000 ha, which is almost twice that given by the Countryside Survey 2000, but clearly much less than Robertson and Jowsey's early (1968) estimate.

It is worth noting that both Milne and Brown (1997) and Holden *et al*. (2007) also provide alternative estimates based on a wider definition of peat soils to include shallower peat, and these estimates again lie fairly close to each other, around 457,000 ha. It seems perhaps, then, that figures for England and Wales which lie somewhere between 350,000 ha and 400,000 ha may be associated with deeper peat deposits while figures in excess of 400,000 ha also include significant amounts of shallower peat.

The values and trends which emerge from Table 1 and Figure 3 for Scotland, and indeed the UK as a whole, are strikingly different in their overall pattern from those see in England and Wales. It is very clear from Figure 3 (bottom left) that there is an underlying core of estimated peatland extent which has never fallen below some 750,000 ha. The general trend over the decades, however, has been inexorably upwards. Taking even just the lowest values across the years, estimates for the extent of peat in Scotland have risen from 821,000 ha in 1968 to 882,000 ha in 2007, but the upper estimates have been getting larger and larger. The gap between an essentially constant lower limit and the expanding upper limit has thus been steadily increasing, leading to an apparent upward trend in the extent of peat for Scotland. The increase is not real, in the sense of there being more peat in Scotland now than in 1968. The apparent increase is purely a reflection of the differing approaches adopted to the mapping of peat soils.

From the fairly consistent minimum of 750,000 ha for Scotland, the trend-line suggests that recent estimates may have raised this figure to something closer to 1.5 million ha, while some of the highest values take this to more than 2.5 million ha. It is not clear whether these high values include significant areas of shallow peat or not, but what is clear is that Milne and Brown (1997) also give a value for Scotland which explicitly includes such shallow peat, and this estimate is twice the size again of the highest value shown in Figure 3 (bottom right).

Peat is almost ubiquitous in Scotland. In addition, much of the Scottish scenery consists of landscapes in which a hard bedrock, even if smoothed by glacial action, presents a surface which is essentially rugged at the micro- or macro-scale (or both) when compared with the smooth landforms of, for example, the English chalk or the London clay. Consequently the range of hollows, basins, pockets, valleys, rounded knolls and rolling landscapes across which peat may form in a continuously varying depth is almost numberless. This means that even if the entire country were mapped carefully at 100 m intervals, a very different result would be obtained from the same mapping exercise carried out at 50 m or at 25 m intervals.

Indeed the most useful way of considering the Scottish peat resource is perhaps to think of it as a fractal measure – a measure which depends entirely on the scale at which measurement is undertaken. Using one scale of measurement produces one result, zooming in to another, finer scale of measurement produces another result, and zooming in yet again produces yet a third result. All are correct, but all are different because each is tied to a particular scale of resolution. There is no 'correct' scale of resolution. This is an important concept to grasp, firstly, because it is clear that different scales are required for different purposes, and, secondly, each increase in resolution increases the survey time and effort required exponentially, raising important issues of resource- and time-constraints.

This digression into the difficulties of measuring the extent of the Scottish peatland resource helps to set the scene for the final group of estimates – those associated with the UK as a whole. Given that the Scottish resource dominates the UK total to such a degree (by somewhere between 55% and 85%), it is hardly surprising that the graphs of estimates for the extent of peat in the UK over the last four decades should show a similar rising trend-line to that already seen for the Scottish data (Figure 3 : bottom centre and bottom right).

However, of particular relevance to the earlier discussion about the fractal nature of the peat resource is the graph in Figure 3 (bottom right). This graph combines the range of 'minimum' estimates for the extent of peat in the UK (essentially those estimates which appear to focus on 'deeper' peat) seen in Figure 3 (bottom centre) with estimates of peat and 'peaty soils' more than 30 cm in thickness. It can be seen that there is a consistent minimum baseline of around 1.5 million ha for the UK. Some figures rise to 2 million or even 3 million ha, though it is not clear to what extent these figures incorporate thinner peat soils. Finally, there is a set of estimates which are clustered around 5 million ha, explicitly incorporating thinner peat soils to a thickness of 30 cm or so (a thickness which corresponds to the widely accepted threshold for the definition of 'peat'). The maximum estimate of somewhat more than

5 million ha thus represents more than a 3-fold increase over the minimum estimates. Perhaps this represents the ratio of deep peat to thin peat in the UK, but it is impossible to say from the figures available. All that can be said on the basis of existing data is that the peatland resources of the UK probably lies somewhere between 1.5 million ha and 5 million ha.

Does 5 million ha represent the absolute maximum possible for the UK peat resource? It is impossible to say. What is clear is that, in recent years, resource estimates for peat in the UK have shown a fairly steady expansion in the range of estimates. More detailed surveys at increasingly fine-scale degrees of resolution would almost certainly produce yet more figures to add to the existing plethora of estimates, in part because of the fractal issues discussed earlier. However, such surveys are unlikely to be undertaken in the foreseeable future simply because of resource constraints. It is more likely that any future surveys will continue to be based on sampling strategies rather than total-resource mapping, and thus the uncertainties inherent in sampling and scaling-up across such a physically complex resource will remain.

This problem is not unique to peatlands. Many natural ecosystem types are equally difficult to map 'accurately'. The most commonly cited example, and the one most frequently used to demonstrate the concept of fractal dimensions, is the coastline. The measured length of the coastline depends entirely on the scale at which the coastline is measured. A coastline mapped using a minimum unit of 1 km will produce a very different result from a coastline mapped using a minimum unit of 10 m, and both will be different from a survey using a minimum resolution of 1 mm. At each different scale a different degree of 'roughness' (or complexity) is being measured. This is an inherent feature of the coastline and no degree of refinement in survey methodology will overcome this. Instead of seeking an increasingly 'accurate' measure of the coastline, a more rational approach would be to decide on the scales appropriate for particular purposes and then accurately calculate the length of coastline at those defined scales. Thus coastline measurements related to navigation of shipping are unlikely to require a resolution of better than 100 m, whereas measurements of erosion rates may require a resolution as fine as 1 cm. The important principle is that the scale of resolution is pre-determined for a particular purpose, and the subsequent mapping is carried out consistently and comprehensively at that scale of resolution.

This has not generally been the approach adopted so far in relation to the peatland resource, particularly in relation to blanket mire. The majority of peatland mapping has been undertaken as part of wider resource-mapping exercises, such as the Soil Survey, the BGS mapping programmes or the Countryside Survey. Peat has obviously featured within the mapping process, but the fundamental principles of fractal mapping have not been applied. Thus important issues arise from the way in which surveys have been undertaken, how peat has been recorded within the survey, and how well this has been documented.

It is essential that there is clarity and consistency in the methodologies employed by those undertaking such survey work. For a national survey, this requires a potentially large number of people to apply rules consistently and record in great clarity their deliberations throughout the survey programme. This is not an easy task. Mosaics, for example, are notoriously difficult to survey quantitatively, especially when the mapping unit is itself not a regular shape. Consequently the assignment of particular localities to particular soil series may owe much to the individual judgement of the surveyor. Similar mixtures in other parts of the country may be assigned by other surveyors to different (albeit closely related) soil series types.

The degree of resolution employed by different surveyors is also an important issue. For example, the British Geological Survey maps at 1:50,000 scale display 'peat' as a drift deposit when it has a thickness of 1 m or more. Detailed examination of maps produced from different parts of the country suggests that the scale of resolution employed by some surveyors was very much greater than that used by others. Some sections of the British drift map display detailed fine-scale mapping, while other sections display the peat only as large, very generalised units. The documented methodology, however, implies that a single consistent approach has been applied across Britain. If the BGS dataset is used as a key source of information about the extent of peat, nothing can be done *post-hoc* to rectify this variation in resolution - it must simply be recognised as a limitation of the dataset. It should be emphasised that the BGS data have not been singled out as a poor example of mapping – the same kinds of issues apply to every dataset underpinning current peatland resource estimates for the UK.

When the field data underpinning resource estimates are not consistent, or are not sufficiently well documented in terms of their inconsistency, subsequent secondary analyses based on such data can generally do little to rectify this problem. However, unless those undertaking secondary analysis are fully aware of the methods used, and the consequent limitations of the data obtained, in these primary surveys, there is a danger that such secondary analyses will fail to recognise these limitations. Problems are then compounded further if secondary analyses themselves involve the re-classification of certain types – for example assigning particular soil types (or even particular parts of mixed-soil polygons) to 'peat soils' while others are assigned to 'heaths' – without providing adequate documentation for this decision-making process. The uncertainties of the original survey are compounded by uncertainties in the secondary survey.

This might seem to be a great deal of fuss over trifles. However, if these differences are compounded across the tens of thousands of mapped polygons embracing the blanket mire resource of the UK, the cumulative effect can ultimately be quite considerable, and perhaps explains why the existing range of estimates for the UK peat resource vary so much. The simple fact is that there has never been a survey designed specifically to map the peat resources of the UK at any defined and consistently-applied scale of resolution. It is as well to be aware of this whenever resource estimates are being considered.

To summarise, therefore:

- estimates for the extent of peat in England have decreased over time from a little more than 350,000 ha to around 250,000 ha, though this may be in part because recent estimates have restricted themselves to deeper peats;
- estimates for the extent of peat in Wales have remained steady, at around 75,000 ha for deeper peat deposits, or 160,000 ha for the combined area of deep and shallower peats;
- estimates for the combined area of peat in England and Wales have also decreased over time, from a little more than 500,000 ha to something closer to 375,000 ha, but if shallower peats are included then the extent may amount to almost 460,000 ha;
- variation between estimates for peat in Scotland has risen steadily, with a consistent minimum base of more than 750,000 ha but some higher estimates pointing to something closer to 1.5 million ha, with a high of more than 2.5 million ha;
- the combined estimates for the area of peat for the UK also show a steadily rising disparity between lowest and highest estimates, with a minimum of around 1.5 million ha, whereas larger estimates today suggest an area of anything between 2 million and 3 million ha for deep peats, while the inclusion of shallower peats results in estimates of around 5 million ha;
- existing estimates of the peatland resource vary so widely precisely because there has never been a survey designed specifically to map UK peatlands at a consistent scale of discrimination and resolution; figures have instead necessarily been obtained from surveys carried out for other purposes, and in which the scale of discrimination and resolution for peatlands has not necessarily been consistent or appropriate;
- the highest estimate for peat soils in the UK is currently more than three times larger than the smallest estimate;
- a single definitive total for the area of peat in the UK is unlikely to be obtained any time soon because this would require the use of a tightly-framed definition of peat, together with extensive fieldwork in which the definition of peat was consistently applied across the whole of the UK.

3.2 Estimating the thickness of UK peat deposits

Having come to some picture, however variable, for the amount of ground covered by peat throughout the UK, the next stage in determining total carbon stores requires that we supplement the data for areal extent (potentially up to 5 million ha) with information about the thickness of this peat layer.

We are immediately confronted with a problem because peat thickness is an invisible property. Unlike maritime regions of the UK where detailed charts render the invisible both clearly and dependably visible – namely the depth of water at low tide - there are no equivalent maps showing the thickness of peat across any but a few scattered localities in the UK. Particular peatland sites have been surveyed in detail in order to assess their exploitable peat resources, or in order to understand their hydro-ecological or palaeo-ecological story but even most of these sites have been surveyed only in terms of a few cross-sectional profiles. Remarkably few sites have been subjected to comprehensive 3-D assessment of peat thickness throughout the site. Indeed the total area of peat in the UK for which any measured depths have been obtained probably represents a vanishingly small proportion of the entire UK peat resource, though there may be localised areas with detailed measurements.

Although in recent years some valuable advances have been made using ground-penetrating radar (GPR) to investigate the nature of peat profiles (*e.g.* Warner, Nobes and Theimer, 1990; Holden, Burt and Vilas, 2002), the almost universally applied method for assessing peat thickness involves probing the peat using connected metal rods.[2] The limitation with probing is that it only gives information about peat thickness for that particular location. The peat thickness may differ substantially only a few metres from the measured point. If enough information of this kind is assembled for a given area, however, it becomes possible at least to discern the general shape of any given deposit.

Unfortunately such probing has also revealed the highly variable nature of the blanket mire mantle at a variety of scales. Thus at the small scale of the bog surface, the topography may vary in height (and thus the peat in thickness) by up to 50 cm or more depending on whether the particularly locality is a hummock or a hollow. Meanwhile below ground, the topography of the bottom sediments underlying the peat mantle may vary by 20-30 cm (or more) across horizontal distances of only 1 m or so.

Over distances of 50 m or more, the peat mantle may vary in thickness by more than a metre. Mounds of mineral ground two or three metres high can become completely overwhelmed by a smothering blanket of peat which may be four or five metres in thickness. What appears to be a relatively uniform expanse of peat may thus, on probing, prove to be highly variable in depth (*e.g.* Charman 1992; Lindsay and Freeman 2008). Other examples of this unpredictability include the many small valleys lying between rock outcrops in the ice-moulded landscape of the Scottish west coast. These are generally infilled by peat, but the precise sub-surface profiles of these valleys can be highly variable. Consequently the ancient Lewisian peneplain on which the peaks of Suilven and Canisp sit so dramatically may be broadly level at the landscape scale, but at the scale of 100 m x 100 m it is a highly complex and irregular terrain. The peat in such ice-moulded landscapes may vary from a thin covering of only a few centimetres to thicknesses of 15 m or more (see, for example, Goode and Lindsay, 1979).

At a larger scale still, broad upland plateaux may support a blanket of peat which varies between 2 m and 7 m in thickness. Thinner parts are generally found towards the margins of the plateaux while the deeper parts dominate the plateau summits, but the thickness may vary substantially wherever drainage-lines between broad peatland units converge. If water seepage in these convergence zones is slow, then very deep deposits can develop, whereas more rapid flow tends to result in much thinner peat formation or even a break in the peat cover if a stream develops along the drainage line.

It is possible to put broad limits on the thickness of this peat mantle. The thinnest deposits are in theory the simplest to define because they correspond to whatever limiting thickness is used for the definition of peat. Consequently the shallowest layers may be anywhere between 30 cm or 1 m thick depending on the definition adopted. In practice, however, these thin deposits can be very difficult to map accurately without exhaustive probing because the vegetation formed on 20 cm thickness of peat may be little different from vegetation growing on 40 cm of peat.

The maximum likely thickness, on the other hand, can be broadly identified by the type of peatland involved. Raised bog peat in the lowlands is capable of attaining thicknesses of up to 10 m, whereas the general expanses of blanket mire deposits rarely exceed 6 m deep. Nonetheless it would be a mistake to assume that blanket mire deposits cannot achieve thicknesses of 10 m or more. In zones of drainage convergence, across basins, and filling steep-sided valleys as described above, it is possible to find peat depths which exceed even the deepest lowland raised bog deposits.

This is a particularly intractable issue for those attempting to produce summary figures for the amount of carbon stored in peat. Published figures for global carbon stores in peatlands generally assume an overall average for peat thickness, giving either a global figure or average thicknesses for particular regions of the globe. Many such studies are usefully summarised in Immirzi *et al*. (1992), from which it emerges that Gorham (1991) assumed an average peat thickness of 2.2 m for Canada, 2.3 m for boreal and sub-arctic regions, and 2.5 m for the Soviet Union. Immirzi *et al*. (1992) themselves assume a global average of 1.5 m thickness for peat soils.

Cannell *et al*. (1993) explore the results of an analysis carried out for 302 samples of peat taken from Forestry Commission holdings across Britain (Pyatt *et al*., 1979). They calculate that the average peat thickness of their 'deep peat' samples was 243 cm. It will be noted that this average value is close to Gorham's (1991) assumed depth for boreal and sub-arctic mires.

As part of the work required for the official Environmental Impact Statement accompanying an application to build Europe's largest onshore windfarm on Lewis in the Outer Hebrides, Lewis Wind Power assembled an extensive dataset of peat depths from parts of the blanket mire expanse

[2] For all but the densest peats, the present author has found that the most cost-effective and practical probing rods comprise 0.5 m lengths of 'M6 threaded studding' linked together using M6 studding connectors available from Wickes DIY stores.

dominating the northern plateau of Lewis (Lindsay and Freeman, 2008). This dataset indicated that the average thickness of peat across the development was somewhere between 2.1 m and 2.3 m (depending on how the data were interpreted). This is close to the estimate given above of 2.43 m obtained by Cannell *et al*. (1993) for their analysis of deep British blanket bog.

While the LWP Lewis dataset is undoubtedly one of the most extensive datasets for peat thickness assembled for a large expanse of blanket mire in the UK, in fact the data were not obtained for the central part of this peat-dominated plateau, nor did the sampling necessarily reflect the full range of peat depths to occur in the Lewis peatlands. This is because the proposed development lay around the fringes of the central plateau, outside the Special Area for Conservation (SAC) designated as the main Lewis Peatlands. Furthermore, the depth measurements were taken along the line of the proposed service road, and it can be assumed that this road had been laid out with a eye to avoiding the very deepest, most difficult areas of peatland. Indeed where possible the route followed old existing tracks originally laid out on shallower areas of peat.

Consequently the average depth of peat obtained from the LWP dataset probably does not reflect the true range of peat depths found across the Lewis Peatlands. Indeed it would be reasonable to assume that because the LWP dataset is somewhat biased towards shallower peats, the true figure for the central Lewis Peatlands may well be greater than this.

The same might be said about the data used by Cannell *et al*. (1993) because, as Chapman *et al*. (2001) observe, the dataset was taken from ground held by the Forestry Commission, presumably with the intention of afforesting this ground at some point. Consequently it is probable that the range of samples described by Cannell *et al*. (1993) are, like the LWP dataset, somewhat biased away from the very deepest peat deposits.

Looking back over the history of peat-depth measurements in Britain, although volume had been an important consideration for the Scottish Peat Surveys begun after the Second World War and continued until the 1970s, this interest lay particularly in exploitable reserves, the term ‘exploitable’ embracing volume, access and local climate (for drying of the harvested peat). On the basis of these available data, Robertson (1971) assumed an average of 2.5 m for the thickness of peat in Scotland. Subsequent accounts of the British peat resource (*e.g.* Moore and Bellamy, 1974; Taylor, 1983) tended to confine themselves to the surface area occupied by peat rather than speculate on the volume of peat involved.

It is only with an upsurge of interest in carbon storage following the adoption of the UNFCCC in 1992 that more detailed consideration has been given to the question of carbon sequestration and storage in a variety of terrestrial ecosystems, including peatlands. Thus it is difficult to find published estimates for the *average* depth of British blanket mires until Cannell *et al*. (1993), and their consideration of the carbon balance associated with afforestation of drained peatlands.

Cannell and Milne (1995) look instead at the carbon stored within vegetation rather than soil. They focus particularly on the balance between carbon sequestration by forests on peat compared with the carbon lost through oxidation of that peat. This approach perhaps reflects the UNFCCC focus on carbon sequestration rather than existing stores.

Howard *et al*. (1995) drew extensively on data held by the Soil Survey of England and Wales and the National Soils Database for Scotland in assembling their estimate of the soil carbon pool. In practice, however, both these major data sources possess only a relatively limited number of actually-measured data for the bulk density and even depth of peat soils. The same two soil surveys are used by Milne and Brown (1997) in drawing together their assessment of carbon stocks in British vegetation and soils. Consequently the same limitations apply, but there is no mention of the uncertainties arising from such data in relation to depth, although the potential errors arising from uncertainty about bulk density are explored in some detail.

Dawson and Smith (2006) acknowledge, as do others, that efforts to estimate the carbon store of British or UK soils are hampered by uncertainty surrounding the depth of (particularly Scottish) peat soils and the bulk density of peat (an issue addressed in the next section of the present report). They are not alone in then devoting little discussion to the question of peat depth. Indeed until 2006 the majority of authors presenting estimated figures for regional or national soil-carbon stores in the UK have done so without exploring in any detail the way in which peat thickness and its various uncertainties can be, or have been, addressed in generating estimated figures for carbon (*e.g.* Howard *et al*., 1995; Milne and Brown, 1997; Cruickshank *et al*., 1998; Bradley *et al*., 2005).

The ECOSSE Report (2007), in contrast, does explicitly address the issue of peat depth in estimating the carbon pool of organic soils in Scotland and Wales. It applies weighted averages to different organic soil units across Wales and Scotland. The weighted averages are derived from a variety of sources but draw heavily on transects and individual records obtained from particular localities of interest rather than from a generally distributed survey across the resource as a whole. In total, the ECOSSE Report (2007) assembles depth data for 278 sites across Scotland. Given that the blanket mire resource in Scotland potentially exceeds 4 million ha (Milne and Brown, 1997) it is probably reasonable to say that this level of sampling provides only a very generalised picture of peat depth in Scotland, but for the mapping resolution of 1:250,000 used in the ECOSSE Report (2007) this generalised picture is appropriate.

Looking elsewhere in Europe, it is possible to find the same degree of generalisation. In summarising the carbon stores of Finland, Turunen (2008) uses mean depths of peat calculated for the range of differing Finnish mire types, based on 1,302 peat cores taken from the various mire regions of Finland. In this case the scale of depth sampling is clear, as is the sampling strategy. Even here, though, such a sampling rate cannot give anything more than the very broadest picture for carbon storage in a country which possesses almost 9 million ha of peat (Lappalainen, 1996).

To summarise the position thus far, it would appear that actual measurements of peat thickness, both for the UK peat resource and elsewhere, are comparatively rare compared to the overall size of the resource. Estimates of peat thickness at regional or national level have been obtained by assigning average thickness values to relevant soil types or peatland types and applying these values across the estimated extent of each type.

This approach can only give a rather low-resolution picture of peat depth across the resource, and thus only a correspondingly low-resolution estimate of the total carbon store. The concern, in terms of existing literature, is that it is rare to find the difficulties and uncertainties associated with estimating peat depth being addressed in any way which actually reflects the problems of limited data and the implications for estimates of carbon storage.

The issue is not just relevant to the mesotope (mire unit) level either, although this is the level at which it is most commonly addressed. Microtope pattern can have important implications for measurements of average peat depth because it is a moot point what an ‘average’ depth of peat would be for a blanket mire covered with microtopes consisting of deep pools. The photograph on the front cover on the NVC book for mire vegetation (Rodwell, 1991) shows Kentra Moss, Argyll, but the foreground is obviously dominated more by water than peat. What exactly would be a meaningful ‘average’ peat-depth measure here?

On shallow peat, the nanotope takes on more significance because on a peat soil which is only 1 m thick, a 40 cm T3 hummock (*sensu* Lindsay, Riggall and Burd, 1985) clearly has the potential to increase a peat-thickness measurement by 40%. Hummocks tend to be rather scattered across a bog surface, but not so hare’s-tail cotton grass tussocks – these can form dense stands. A measurement from the top of a 40 cm tussock will also add 40% to a 1 m thickness of peat, whereas a measurement from the gap between the tussocks will give a measurement of only 1 m. Which measurement is appropriate? Given the much larger extent of shallow peat compared to deep peat, this question is probably quite important to resolve when attempting to estimate carbon stores in organic soils.

There are thus issues to do with peat thickness at the scale of macrotope, mesotope, microtope and nanotope (landscape, mire unit, mire pattern and small-scale surface structures), but there is comparatively little in the literature which addresses the questions raised at each of these scales. Until these questions are addressed, peat thickness will remain a factor which continues to cloud attempts to produce reliable figures for the carbon pool in peat.

Before moving on to the next piece in the puzzle – namely the actual composition of the peat material – it is worth noting that Chapman *et al*. (2001) additionally highlight a hitherto little-reported feature associated with the soil profile as a whole. An increasing body of evidence suggests that a substantial but largely unmeasured amount of carbon passes downwards from the peat deposit to become stored in the mineral sub-soil beneath (Turunen *et al*., 1999; Jobbagy and Jackson, 2000). In Finnish boreal mires, Turunen *et al*. (1999) found that the amount of carbon recorded within the uppermost 70 cm of the underlying mineral sub-soil was equivalent to an additional peat thickness of 18 cm, and they estimated that the total held within this store for Finland may amount to 300 Tg of carbon, representing 5% of the total carbon store associated with Finnish peatlands.

Consequently it may be that estimates of the peat thickness for carbon-storage purposes will in future require that such estimates also include the carbon stored *beneath* the body of peat. Chapman *et al*. (2001) observe that no measurements of this phenomenon have yet been obtained for UK peatlands, so it is not currently possible to judge how significant this might be for carbon auditing, either in terms of depths to which the sub-soil must be assessed, nor in terms of the total contribution to national carbon stores.

In summary:

- it is not yet feasible to create a continuous map of peat thickness; virtually all current assessments rely on probing, which gives a value for the specific location of the probe but no wider information for the peat thickness;
- by combining many probe measurements in the form of a transect or a grid, it is possible to construct a generalised picture of the peat thickness across a site, but the proportion of the UK peat resource so mapped is very small;
- estimates of average peat depth/thickness have been used to make more general assessments of the total peat volume, or carbon store, across countries, regions or even globally;
- a number of regional/global estimates have assumed average peat thicknesses of around 2.4 m, although others have used a value of only 1.5 m;
- average depths calculated from limited probing of the blanket mires of Lewis, in the Outer Hebrides, and of British blanket mires more generally, result in values of between 2.1 m and 2.4 m, although in both UK examples the sampling was probably somewhat biased towards shallower peat;
- probing data highlight the fact that the thickness of the peat mantle can vary substantially over distances of only a few metres although there may be no evidence for this variation in the surface features of the peat;
- in general, the thickness of blanket mire peat tends to range between 30 cm and 6 m but there are many occasions where, locally, the peat may exceed 6 m;
- relatively small-scale variations in peat thickness resulting from hummock growth, or from small-scale roughness in the underlying mineral sediments, can have very large proportional impacts on overall peat thickness in areas of relatively thin peat, and thin peat may be more extensive than deeper peat by a ratio of 3:1; such small-scale variation in thickness may thus have important implications for the total carbon store in peat;
- there is general recognition that uncertainty about peat thickness is an important constraining factor in determining the total amount of carbon stored in UK peatlands, but most accounts in the literature are devoted to methods for estimating variation in bulk density;
- a few accounts, such as the ECOSSE Report (2007), attempt to incorporate an explicit and clear consideration of peat thickness within carbon storage assessments, but the key papers involved in national carbon audits do not address the uncertainties associated with variation in peat thickness and include no discussion of the limitations to such data, although they do provide considerable detail about approaches adopted to cater for variations in bulk density;
- evidence is emerging that peat soils also contribute significantly to a carbon store which accumulates in the sub-soil beneath the peat, but the scale of this additional carbon store has yet to be assessed for the peatlands of the UK.

4 Carbon and the components of bog peat

This is an extensive section largely devoted to the question of bulk density; carbon density of peat depends on mineral content, % carbon in organic matter, and the physical density of the peat (bulk density); mineral content depends on the definition used for peat as a whole, but for bogs it is generally between 1-10%; carbon density of organic matter is generally around 52%; bulk density is a more complex topic; a 'standard model' is presented where the uppermost part of the acrotelm has a bulk density of 0.03 g cm^{-3} while the catotelm is very much denser, with a bulk density of 0.12 g cm^{-3}; bulk density measurements of UK blanket bog are relatively sparse, and many consist of single values rather than distinguishing between acrotelm and catotelm; measurements of bulk density at peat depths of more than 1 m are even scarcer; consequently assumptions are often made about bulk density at depth, but these assumptions may not be correct; measured values of bulk density often contradict the 'standard model', with higher bulk density at the surface of the bog; high bulk-density values at the surface appear to indicate a damaged bog which has lost its acrotelm and has a layer of damaged catotelm ('haplotelm') at the surface instead; microtopography also influences bulk density, with hollows and hummocks giving different bulk-density values to the peat beneath.

Having explored the simple dimensions of extent and depth, and having found that neither parameter is really so simple, the final step in estimating the carbon store of the UK peatland resource requires that the carbon *density* within the peat be measured (*i.e.* how much carbon is stored within a given volume of peat). Several factors influence this parameter:

- the proportion of mineral matter which has become incorporated into the peat matrix from dust, marine-spray inputs, roosting or nesting bird colonies, surface-water flushing or, in the case of mires having a minerotrophic influence, inputs from groundwater sources.;
- the proportion of carbon atoms within the organic compounds which form the chemical structure of the peat fibres;
- the physical density of the peat structure (i.e. how closely the peat fibres are packed together in a given volume, and how dense the material is making up these fibres); this is generally referred to as 'bulk density'.

The first two factors can be addressed relatively easily and will thus be tackled first before going on to consider the rather more challenging question of bulk density. This is not to say that mineral content and carbon content are simple and well-defined values; as we shall see, there are significant uncertainties even here, and some of these uncertainties have the potential to equal those associated with the assessment of bulk density. There is, however, a great deal more published about bulk density and so there is more to say on the subject.

Any reader looking for a remarkably comprehensive yet highly accessible review of the properties of peat from a largely ecological perspective is strongly recommended to obtain a copy of Clymo (1983). The present review has a much narrower focus that that provided by Clymo (1983) but draws on much provided by that earlier review.

4.1 Peat and mineral matter

As is described in Appendix 2, a number of external influences may affect the amount of mineral matter which accumulates within any given peatland system, or indeed within any given part of a peatland system. It is important to understand that the definition of 'peat' plays a fundamental role here, because of course a complete continuum of mineral (ash) content can be found in the soils of the UK. It is only by agreed convention that soils containing a high proportion of organic matter are defined as 'peat', but the actual proportion of mineral matter to organic content used as the basis of this definition varies from author to author. The term 'organic soils' embraces such soil classification types as 'peaty organic soils', 'peaty podzols', or 'peaty rankers', as well as 'raw peat', and different authors choose to set the boundary for peat at different locations along this continuous gradient.

Dachnowski (1912), in one of the earliest definitions of peat, stated that the amount of mineral matter in dry peat must not exceed 25% by weight. More recently, Landva *et al*. (1983) has used a maximum limit of 20% for mineral matter content, while Immirzi *et al.* (1992) give a range of 0.5% - 56% for recorded mineral content, whereas Joosten and Clarke (2002) allow mineral contents as high as 70%

within their definition. Rydin and Jeglum (2006) observe, however, that what the majority of ecologists think of as ‘peat’ mostly has a mineral content of 10% - 20% at most.

The mineral content of bog peat depends entirely on the extent to which the peat at any given point is truly ombrotrophic – *i.e.* fed *only* by direct precipitation – and also whether there has been significant atmospheric inputs of minerals in the form of, for example, salt spray (Moore and Bellamy, 1974) or volcanic ash, which is a major factor in the bogs of Japan, Kamchatka, Iceland and Chile but can be found in bogs world-wide (Blackford, 1997; Hotes, 2004).

There appears to be general agreement that the mineral (ash) content of ombrotrophic bogs is very low indeed. Weber (1902) noted that a peat sample taken from the central plateau of the Augstumal raised bog contained 97.13% of “combustible substances”, leaving 2.87% “mineral substances”. Such figures are more usually expressed in the literature today as ‘loss on ignition’ (LOI) or ‘ash content’. The amount *lost* through burning is taken to be the amount of organic matter originally contained in the sample while the quantity *remaining* after ignition is taken to be the mineral/ash content of the peat.

Immirzi *et al*. (1992) suggest that bog peat typically has a mineral content of 2.27% dry weight, though Evans and Warburton (2007) put this slightly lower, at 2% of dry weight. Rydin and Jeglum (2006) cite values given by the National Wetlands Working Group of Canada for an ombrotrophic raised bog in Canada, giving ash contents of 1.1% to 1.8% for the upper 2 m of the bog. Doyle (1997) describes Irish blanket peat as having ash contents varying between 1.2% - 2.4% of dry weight. Coggins *et al*. (2006) give values ranging from 0.25% to 1.4% for ash content in the uppermost 50 cm of two Irish blanket bogs. Charman (1992) on the other hand, gives LOI values of 96% for ombrotrophic blanket mire in Sutherland (*i.e.* 4% ash content).

Shotyk (1995) analysed 1 m cores from blanket peat at Loch Laxford in north-west Sutherland, and Fleck Lochs on Foula in the Shetlands. The ash content at Loch Laxford varied between 2 – 3%. The ash content of the blanket peat from Foula was rather higher, at a fairly consistent 5%. However, at a depth of 15 cm it also showed a very sharp peak where the ash content rose dramatically to 20% before falling away even more abruptly. At the bottom of the sample the ash content again rose, this time to 15%. Shotyk (1995) observes that this latter rise may represent another sharp peak or may indicate a transition to more generally minerotrophic peat. No explanation is provided for the dramatic first peak. Such data reveal the potential for considerable variability even within this component of the peat-carbon story.

If there is any suggestion of groundwater influence or merely significant surface seepage, the mineral content of the peat will indicate this by rising significantly above the background levels of solutes found in the rain supplying the bog. This is illustrated clearly by the values for a Canadian peatland complex provided by Rydin and Jeglum (2006) already mentioned above. While the area of bog had a mineral content of 1.1% to 1.8%, the fen around the margin of the bog had a mineral content of between 7.2% and 7.9%. Nearby areas of basin fen had a mineral content ranging from 3.3% to 9%, demonstrating the significantly higher mineral content of minerotrophic peat. This puts into context the comparatively high 20% ash content found by Shotyk (1995) in certain layers within Shetland blanket bog.

Where a degree of groundwater or even substantial surface-water influence is identified, a whole range of mineral influences typical of fenland systems opens up and offers a multitude of possibilities in terms of mineral inputs. In the present report, however, we are largely concerned with bog systems because these undoubtedly represent the most extensive peat deposits in the UK, even if agriculturally drained peatlands are included. Consequently in this case the mineral inputs to the peat are generally assumed to reflect the very low solute levels typical of rainwater in the UK – though this may not be an entirely reliable assumption, given Shotyk’s (1995) figures from Shetland. The vitally important point to understand from this, however, is that if a research plot is established in an area with even a small amount of water seepage, this seepage is likely to result in significant differences in the nature of the peat compared with peat nearby which is more truly ombrotrophic.

One further point worth repeating at this juncture is that solute inputs, even if only from precipitation, are significantly higher in Britain and Ireland than they are in, for example, Finland or central Canada. This is for two reasons. Firstly, both Britain and Ireland are sufficiently influenced by winds laden with marine salt spray to mean that all parts of both islands receive at least some input from marine aerosols. This tends to result in precipitation loads which are higher in sodium and chloride ions than is typical for bogs in Finland or much of Canada (Moore and Bellamy, 1974; Proctor, 1995). Secondly, the greater frequency of precipitation in oceanic areas such as Britain and Ireland means that, even if the ionic concentrations of precipitation were the same in Britain and Finland, the volume of solute *throughput* per unit time is greater in the more oceanic area. As a result of both factors, species assemblages typically found in ombrotrophic bog conditions in Britain or Ireland more closely resemble those found in minerotrophic fen conditions in Finland (Lindsay, Riggall and Burd, 1985). The bog moss *Sphagnum* *papillosum*, for example, is a characteristic bog former in Britain but is generally regarded as more of a solute-poor fen species in Finland.

One of the consequences of this biogeographical difference between oceanic and continental peatlands is that the blanket mires of Britain and Ireland tend to have higher mineral contents than is typical for bogs in Finland or central Canada . It is important to bear this in mind when comparing published data from oceanic and continental regions.

Based on the general pattern of values obtained above for the ash content of British and Irish blanket mires, for the present report a value of 3% of dry matter will be taken as the proportion taken up by mineral solids within blanket peat. This leaves 97% of dry matter for carbon-based, organic solids.

4.2 Carbon atoms in peat

While the long-chain organic molecules which make up the bulk of carbon-based solids in peat do have carbon as their backbone, many such molecules also contain a substantial mixture of hydrogen and oxygen atoms as well as occasional sulphur, nitrogen or phosphorus atoms. As a result, the actual weight of *carbon* contained within the organic-matter fraction of peat is substantially less than the total weight of the organic-matter fraction.

Immirzi *et al*. (1992) cite several values for the carbon content of peat, but it is not always entirely clear whether the values quoted are % of organic matter only, or % of *all* dry matter (obviously these are two quite different things). For the element carbon (C) they cite a figure given by Lucas (1982) of 52% as being a 'typical average for oligotrophic peat'. It seems likely that this figure represents a % of the total dry-matter weight. Immirzi *et al*. (1992) also give an 'average carbon percentage' for peat of 55% on two occasions, again apparently on the basis of total dry weight, but also cite values of 58% (described as 'above average'), 49%, 50% and 51.7%. Meanwhile Rydin and Jeglum (2006) cite Gorham's (1991) value of 52% for the carbon content of dry matter when calculating carbon stores based on recent peatland surveys. Turunen (2008) uses a value of 50.3% calculated from 3,670 samples collected by the Geological Survey of Finland, but notes that certain peatland types were calculated to have elevated carbon concentrations of 54%, though this increase appears unrelated either to nutrient status or vegetation cover.

Cannell *et al*. (1993) , when describing their formula for calculating the carbon content of British blanket peats, explicitly give what is described as an assumed value for carbon as 50% of *organic* matter, rather than as % of total dry weight. Dawson and Smith (2006) do not mention the role of % carbon in the estimation of UK carbon stocks. The ECOSSE Report (2007), in testing models of carbon stocks at Plynlimon, Wales, and Glensaugh, Scotland, explicitly recognises the potential for %C to influence estimates of carbon stocks, but then tantalisingly provides no actual figures, merely observing that "since [%C] changed only slightly down the profile it was not considered here".

The ECOSSE Report (2007) talks of % carbon values which are held in the National Soil Database for Scotland at the Macaulay Institute, and describes the use of these in assigning % carbon values to soil horizons. Unfortunately no actual figures are given for Scotland or Wales in this part of the review. Consequently it is not possible to draw any specific conclusions from the ECOSSE Report (2007) about % carbon values contained within the National Soil Database and their possible impact on the estimation of carbon stocks.

More tangible data are, however, provided by the ECOSSE Report (2007) when reviewing the UK carbon inventory. Although actual % carbon values are not given, a graph is displayed showing variation in % organic carbon recorded for topsoils associated with different soil series and their assigned carbon contents. The data clearly show that an increase in % carbon content for the soil series is associated with increased variability in actual values of carbon in the topsoil. For those soil series which have been assigned values of 35% to 40% carbon, the associated topsoils display such a degree of variability that the standard deviation exceeds 20. This high degree of variability suggests that the % carbon of many topsoils might fall as low as 15%, or rise as high as 60%, in soils which are decidedly 'peaty'.

As % carbon values exceed 40%, however, the picture changes significantly. The variance data indicate a substantial fall in variability. The standard deviation in this part of the graph declines to a maximum of 8, which would indicate a potential % carbon range of 42% to 58% for topsoils in soil series with an assigned 50% carbon content. This possible range is not so different from the values cited by Immirzi *et al*. (1992) in their global review, but whether this apparent decline is simply an artefact of the data as displayed, or whether it shows a real tightening of values in the most carbon-rich soils, cannot be judged from the ECOSSE Report (2007). It simply observes of its own data that “the variance [for % carbon] becomes greater as the carbon content increases”. However, S. Chapman (pers. comm.) has since confirmed that this tightening of values was real and was observed in both Scottish and Welsh data.

It is instructive finally to look at three % carbon values recorded by the ECOSSE Report (2007) for three blanket mire study sites in Wales and Scotland. A value of 50.77% carbon is given for an area of blanket mire near Ullapool. The same value is recorded for a blanket mire site (Hafren) in Wales, whereas a slightly higher value of 53.2% is given for an area of blanket mire north of Glensaugh, in the Grampians of Scotland. This set of figures might indeed support the idea that soils with the very highest % carbon show relatively little variability between sites. Given the range of values for % carbon content obtained from these many different sources, for the purposes of the present report it would seem reasonable to take a figure of around 52% dry-matter weight as a typical value for the carbon content of blanket mire peat in the UK.

Assuming that this carbon content is almost entirely restricted to the organic fraction of the dry weight, the solid matter in a block of peat can therefore be separated into 3% mineral matter, 52% organic carbon, and 45% organic elements which are not carbon. In fact the story is not quite as clear-cut as this because carbon also occurs in the mineral fraction as a component of carbonate salts, although the amount is likely to very small indeed in these soils. The release of this as dissolved inorganic carbon (DIC) from the peat is increasingly being recognised as a factor to be included in carbon budgets, but values for the % carbon which this represents in a given block of peat (as opposed to the quantities released into watercourses) are not easy to find.

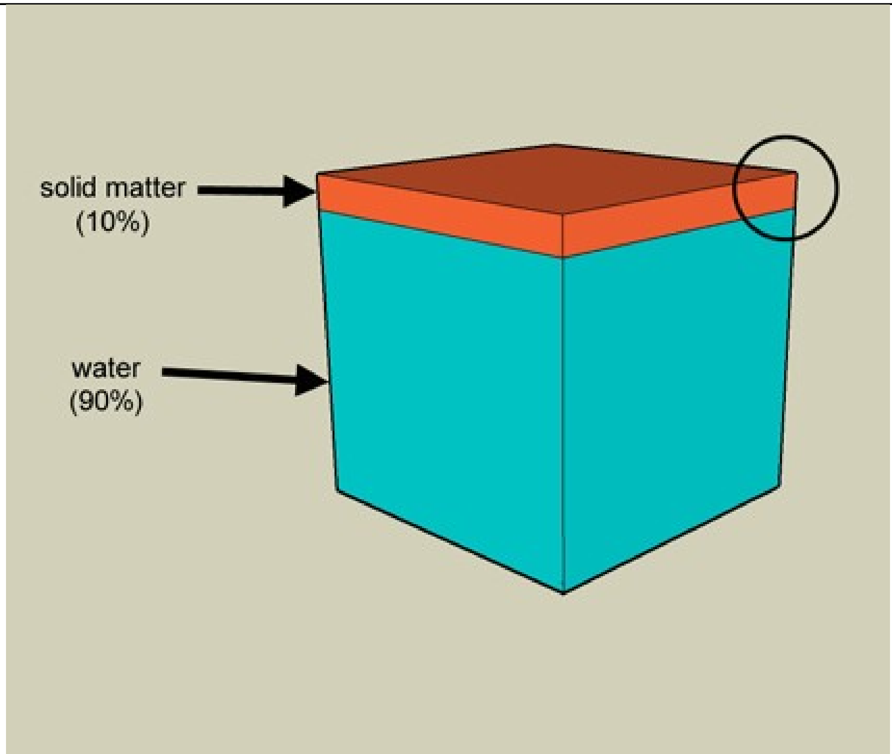
All the foregoing must then be set within the context of two other major features of peat soils which have a significant influence on the size of the potential carbon store for a given peatland area. Firstly, although it is helpful to know the relative *proportions* of the various components which constitute the peat matrix, this information is not sufficient alone to calculate actual carbon stores for a given area. The relative proportions of peatland dry matter will remain the same whether peat is loosely packed or tightly packed. In contrast, the amount of carbon *actually* stored in a given peat thickness will be very much affected by how tightly the peat is compacted. Bulk density is a measure of this compaction, and thus it is a measure of *carbon density*. The whole of Section 4.3 (below) of the present report is devoted to the question of bulk density and its influence on carbon stores.

Secondly, it is important not to forget that by far the most abundant constituent of a peat soil is not peat at all; it is water. As discussed in the opening sections of the present report, a peatland is first and foremost a wetland because complete saturation of the soil is what allows a matrix of peat to accumulate despite the normally ubiquitous forces of decomposition. The peat matrix is extremely light compared to the weight of the enveloping body of water, partly because the peat contains so little mineral matter, and partly because its open geotextile-like structure consists mainly of pore spaces. Thus by weight, water is the by far the dominant component of undisturbed peat soils.

The water content of peat typically ranges from 200% to more than 2,000% of the dry weight (*i.e.* the dry matter may contribute up to 50%, or as little as 5% or less, by weight) (Evans and Warburton, 2007). Blanket peat tends to occupy the wetter end of this spectrum. Typical values generally lie somewhere between 85% and 98% of water (thus giving 15% to 2% of solid matter) by total weight of fresh sample. This means that all the foregoing discussion about the relative proportions of ash to carbon, and carbon to organic matter, may in practice concern a component amounting to no more than 2% of the total weight of material, with water making up the remainder.

To illustrate and summarise, it is perhaps helpful to consider a regular block which represents 100% of all the components in a peat soil considered so far. Assuming a modest 90% for the proportion of water in that soil, we can see rather starkly in Figure 4 the small proportional contribution made by the peat material itself to the overall weight of the peat soil. Focusing on just this peat material, Figure 5 then illustrates the relative proportions of the various components which go to make up that solid matter.

Clearly peatlands contain a great deal of water and not much solid matter. How does this compare with other living entities? The human body consists of between 25% and 55% solid matter, depending on the proportion of body fat (Wikipedia body water website), while barked Longleaf pine (*Pinus palustris*) greenwood, for example, has a solid-matter content of some 53% and water content of around 47% (Husch, Beers and Kershaw, 2003). A jellyfish, on the other hand, is 95% water and only 5% solids, by weight (NSF jellyfish website), while whole milk is generally 87% water and 13% solids (What is milk? website). On this basis, peat bogs would appear to have less tree and more of the jellyfish and milk about them - a sobering thought when sitting on a trembling mass of peat more than 6 m deep and 3 km across.



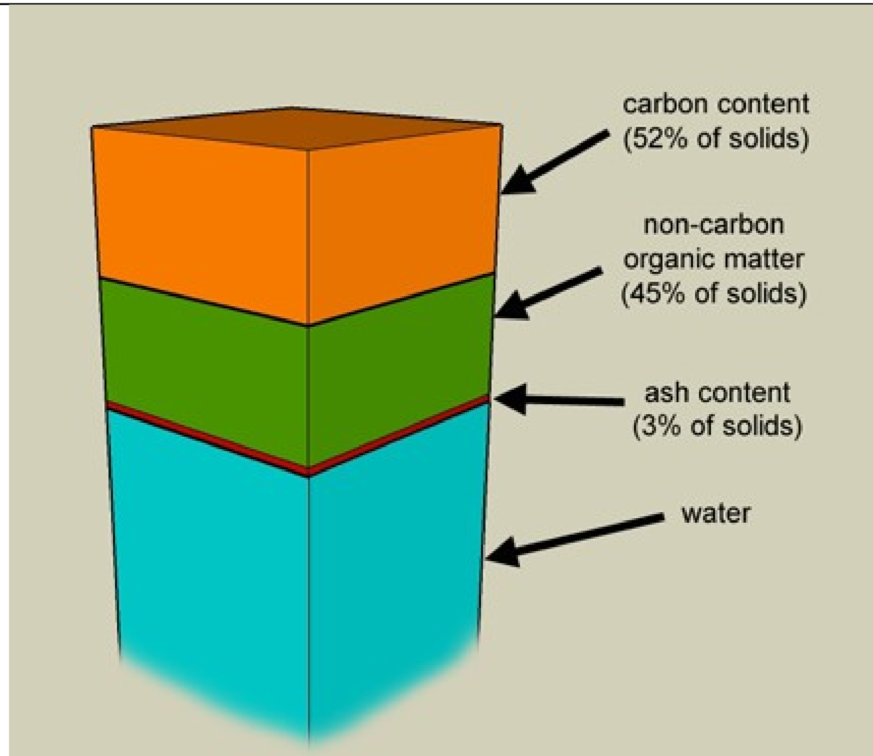


Figure 4. Proportion of solid matter and water in peat.
Representation of the relative proportions, by weight, occupied by water and by dry solid matter within a block of peat. The proportion occupied by water is shaded blue while the proportional weight of dried solid matter is shaded brick red. The area circled at top right is shown in more detail in Figure 5.

Figure 5. Proportions by weight of solid-matter components in peat.
Representation of the relative proportions, by weight, of the various dry-matter components making up the solid-matter proportion of the block of peat shown in Figure 4. Figures based on a synthesis of published literature, as described in the text.

It must be emphasised that Figure 4 and Figure 5 do not represent actual blocks of peat soil. They are merely blocks showing relative proportions of components. For the block to be a true representation of a cubic metre of peat soil, the blocks of solid matter would need to be adjusted to reflect the extent to which this material was tightly or loosely packed. The measure which converts all these estimates into actual quantities of carbon is bulk density.

4.3 Bulk density of peat (and the acrotelm-catotelm transition)

The degree of compaction within the various layers of peat – acrotelm, transition layer, catotelm, and layers within the catotelm - are of considerable significance in the way that they influence the amount of carbon stored in peat. Measures of compaction can be given in various forms, but the most commonly-used is that of dry bulk density, which is calculated by taking a standard volume of wet peat and then determining the amount of dry material contained within that volume. Units are thus typically given as grammes of dry weight per cubic centimetre of original wet volume (g cm^{-3}).

The role played by bulk density in converting relative proportions of materials into actual blocks of peat on the ground does not necessarily mean that it is the most influential factor in the accurate estimation of carbon stores, though the reader of available relevant literature could be forgiven for thinking this to be the case. It is, nevertheless, undoubtedly one of the key factors in making such a determination. As such, accurate assessment of bulk density is important. It is therefore unfortunate that, despite the widespread recognition of the important part played by this parameter, the available data should be so comparatively limited in extent.

Clymo (1992) is an often-cited source for 'standard' figures of bulk density in peatlands, giving figures that are characteristic of bog (rather than fen) peat. He gives a dry bulk density of 0.03 g cm^{-3} for the acrotelm, and then contrasts this with the denser catotelm layer beneath, citing a dry bulk density value of 0.12 g cm^{-3} for this lower layer. In other words, Clymo (1992) indicates a 4-fold increase in density across a vertical distance which may be only a few centimetres thick (see Figure 6).

It can be seen from Figure 6 that in his model of 'structural layers' Clymo (1992) gives four distinct layers. There is a thin uppermost layer of photosynthetic plant material. Then there is a relatively broad zone in which a significant proportion of the material is dead, termed 'litter peat', between 4 cm and 11 cm depth. Underlying this is a narrow 'collapse zone' where all material is dead except the root systems of living vascular plants. This zone spans no more than one or two centimetres. Below this is the zone of true peat, or 'peat proper', which dead plant material enters by crossing the boundary between the acrotelm to catotelm at around 12 cm depth. The uppermost 20 cm or so of this catotelm peat is also penetrated by living plant roots. Living material deeper than this consists entirely of microbial assemblages.

In contrast there are four clear components to Clymo's (1992) bulk-density graph. These consist of an uppermost 1 cm representing the densely-packed capitula (heads) of *Sphagnum*. Below this is a 5 cm section having fairly steady values of 0.03 g cm^{-3}, then a broad transition zone between 6 cm and around 12 cm in which the bulk density increases steadily, then a zone from around 12 cm downwards where the bulk density becomes a fairly constant 0.12 g cm^{-3} (see Figure 6). It seems that a good part of the 'litter peat' zone and the whole of the 'collapse' zone together constitute the broad transition region of increasing bulk density values between 6 cm and 12 cm depth. Integrating the changing values of bulk density for the whole thickness of the acrotelm produces an overall bulk density of the acrotelm of around 0.06 g cm^{-3}.

The reality of this zonation can perhaps be best understood with reference to Appendix 3, Section 23.4, Figure 63, where it can be seen that a proportion of *Sphagnum* immediately beneath the photosynthetic zone may now be dead plant litter but it has yet to undergo any significant decay. Only after a little while, as it is gradually buried deeper within the litter zone, does this plant material begin to show signs of breakdown and collapse. The final phase of collapse is indeed often rather abrupt, as indicated in Clymo's (1992) model of structural layers. A hand forced down into a *Sphagnum* carpet will pass relatively easily through the litter-peat layer but will generally meet distinct and fairly abrupt resistance on reaching this collapse layer.

This overall picture of the surface layer will thus be referred to in the present report as the 'standard Clymo model' (*i.e.* a model which is frequently-cited), with 'standard' dry bulk-density values of 0.03 g cm^{-3} for the upper levels of the acrotelm and 0.12 g cm^{-3} for the catotelm of a bog.

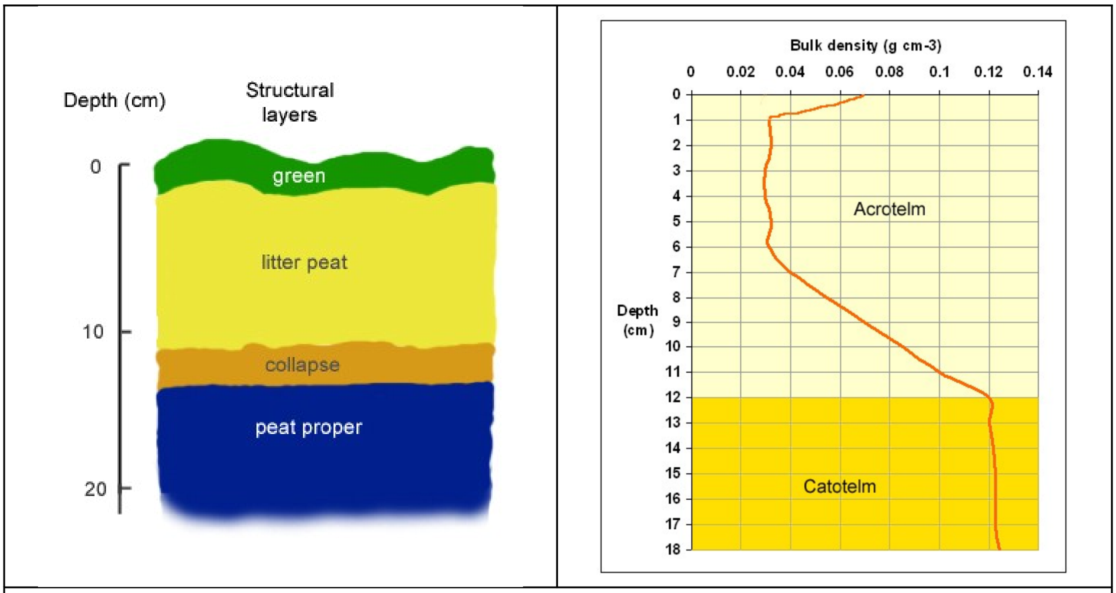


Figure 6. Layering and bulk density in the surface layers of a bog (after Clymo, 1992).
Clymo's (1992) 'standard' model of bulk density and acrotelm/catotelm transition for an ombrotrophic peatland. (**Left**) Clymo's four 'structural layers', with green, litter peat and collapse occupying the acrotelm, while peat proper represents the start of the catotelm. (**Right**) Clymo's values for dry bulk density from the surface to a depth of almost 20 cm. Note that although the vertical scales in the two diagrams are not identical, the start of the catotelm at 12 cm is positioned to correspond with the beginning of the 'peat proper', the lowest of Clymo's structural layers. In both scales, a depth of 0 cm represents the bog surface. Note also the extended zone of transition between 6 cm and 12 cm depth, spanning a wider range of depths than Clymo's relatively narrow 'collapse' zone.

The standard Clymo model would appear to offer an alternative and perhaps convenient means of determining the boundary between the acrotelm and catotelm based on values for bulk density. The boundary between the two layers is instead traditionally given as the 'maximum depth to which the bog water-table falls'. However, Clymo (1965) highlights the fact that this boundary can often be rather diffuse and difficult to determine without undertaking long-term measurements of water-table behaviour.

Bragg (1982), Clymo (1992) and Evans *et al.* (1999) demonstrate the way in which regular water-table readings over an extended period can be used to measure the 'residence time' of the water table at different depths – *i.e.* how much time the water table spends at any given depth. Construction of a 'residence curve' from such data generally reveals that the water table of a bog spends most of its time within 10-20 cm of the bog surface. Below this depth, the curve tends to show a sharp decrease indicating that the water table rarely falls more than 10-20 cm into the peat. Beyond this, there is often a small tail of readings representing the few occasions of drought where the water table may fall as much as 30-40 cm into the peat. The curve does not extend to greater depths because the water table never falls deeper even during the severest droughts (see Figure 7).

Different authors have different opinions about whether the lower boundary of the acrotelm is defined by the depth to which the water table generally falls *apart* from extreme drought conditions (Position A in Figure 7) or by the lowest position to which the water table ever falls (Position B in Figure 7). Whether Position A or B is the more significant, and which represents the 'true' position of the acrotelm/catotelm boundary, rather depends on what is of interest.

Drainage impacts, for example, tend to draw down Position A deeper into the peat but also tend to take Position B to an absolute greater depth. This is because a greater thickness of peat becomes subject to rapid fluctuations in water table, but some parts of the peat column also become subject to aeration for the very first time (Ingram, 1983). There is thus likely to be more rapid decomposition of peat in the upper layers of the profile, but there will also now be a greater overall thickness of peat subject to aeration.

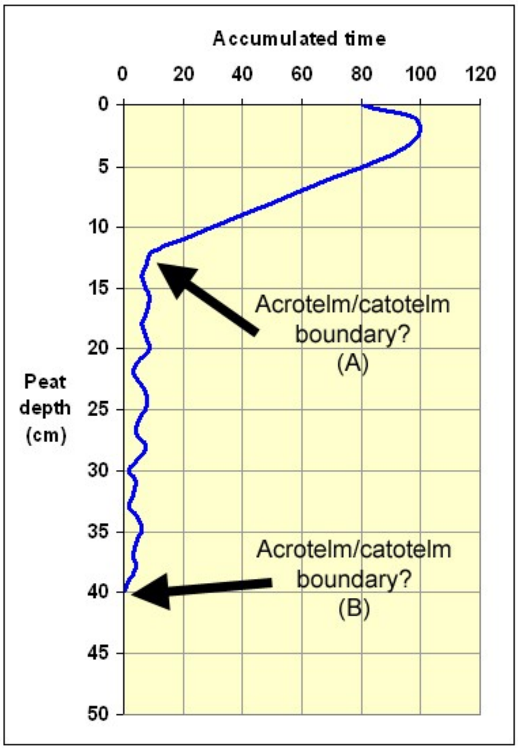


Figure 7. 'Residence curve' for a bog water table. The blue curve represents the accumulated number of hours spent at any given depth by the water table of a bog. The ground surface is at 0 cm. Two possible positions for the boundary between acrotelm and catotelm are shown, depending on the definition used for this boundary.
Adapted from residence curves by Bragg (1980), Evans *et al.* (1999)

If the catotelm is defined as the zone where no aerobic processes occur, then Position B is of more relevance than Position A, and its changed position indicates that peat formerly in the catotelm is now for the first time subject to aerobic intrusion.

If the main feature of interest is the day-to-day behaviour of the bog water table, then Position A perhaps more closely reflects the acrotelm/catotelm boundary. If Position A moves deeper into the peat, this will mean that the zone of frequent and rapid water-table fluctuation has expanded. Some parts of the peat column which formerly only underwent aerobic attack on infrequent occasions now become part of the zone subject to regular aerobic oxidation – even though the point marking the deepest penetration of air (Position B) may not have changed.

What is clear is that both positions can only be determined after extended periods of water-table measurement and analysis. Bragg *et al*. (1994) provide a less labour-intensive means of measuring Position B in the form of a maximum-minimum water-level recorder (Walrag) for bog systems. This can at least record the lowest position to which the water table falls (Position B), but is less help in identifying Position A unless readings are taken on a very regular basis – which rather defeats the purpose of the Walrag.

As yet another alternative, Clymo (1965) demonstrates that, compared to the rather diffuse picture associated with the position of the bog water table, there is often a much sharper division observable between an upper layer of peat containing oxidised sulphur compounds, such as sulphates, and the lower layer dominated by anaerobic (reduced) sulphur compounds – particularly sulphides, which are often highly toxic to both plant and animal life. This boundary clearly relates to the behaviour of the water table but from a functional point of view has added advantages; it is both relatively well-defined and it marks an ecologically-significant threshold produced by waterlogging. The water table itself may oscillate around an 'average', or even a minimum, position, but if the duration at any given depth is insufficient to change the prevailing anaerobic conditions, then the sulphide layer will remain in place and ecological changes are unlikely. Conversely, if the water table sinks for sufficient duration to produce aerobic conditions, the sulphide layer will move deeper into the peat and ecological changes in the upper part of the peat column are likely.

Measurement of this sulphide boundary can be achieved by inserting silver wires into the peat for an extended period and then measuring the position at which the wire turns black due to the reducing conditions prevailing in the sulphide zone. More recently, Belyea (1999) and Booth *et al*. (2005) have used discoloration of PVC tape to measure the same thing with varying degrees of success.

The several methods described above for determining the acrotelm/catotelm boundary all have one thing in common – they all require some form of measurement to be taken over an extended period of time. A means of determining the position of the boundary between acrotelm and catotelm by taking single instantaneous reading would clearly be of considerable value. Clymo's (1992) standard model suggests that theoretically, at least, bulk density could be used as an 'instantaneous' measure of this boundary. Upper parts of the acrotelm are characterised by low bulk density. A steady but marked increase in values indicates the transition zone which forms the basal parts of the acrotelm. The catotelm itself begins where the bulk density finally stabilises at a higher (denser) value.

Such an instantaneous measure of the acrotelm/catotelm boundary would undoubtedly be of considerable value in relation to several key aspects of the peat-carbon story. The critical question, therefore, is whether Clymo's (1992) standard model does indeed provide an instantaneous means of identifying this boundary. How do the 'standard model' values for bulk density compare with actual measured values for bulk density given in the literature, and to what extent do these cited values reflect Clymo's (1992) recognition of at least two, and arguably three, distinct zones of differing bulk density within the peat column?

4.3.1 Bulk density of peat: field measurements

Sampling for bulk density tends to be carried out by extracting a peat column or finding a peat face, then sampling once, or intermittently, or continuously down the peat thickness. Continuous records are much rarer than measurements at two or three depths in the peat.

4.3.1.1 Single cores, intermittent sampling

A great variety of published values for bulk density can be found in the literature, partly because this parameter has become a fairly standard item to measure when investigating the properties, stratigraphy, or even the pollen record of peat. On this basis, it would seem reasonable to conclude that a good picture of bulk density can be built up fairly readily for the bogs of Britain and Ireland (and elsewhere) and thus the standard Clymo (1992) model can be confirmed using this range of data. Unfortunately this is not the case, and there are several important reasons for this.

Boelter (1972) gives bulk-density values of 0.07 and 0.17 g cm^{-3} respectively for acrotelm and catotelm of a densely afforested bog in Minnesota. For a second thinly-wooded bog he gives values of 0.07 and 0.08 g cm^{-3} for the acrotelm and catotelm. For this latter site, which is clearly wetter, the catotelm density barely differs from that of the acrotelm, and has half the density recorded for the catotelm of the drier wooded bog. Clearly in the wetter bog, the difference in bulk density between the 'surface horizon' and 'subsurface horizon' is at least in the right direction (catotelm denser than the acrotelm) but the difference is hardly significant. Most importantly, Boelter (1972) only gives a single value for each layer. No indication is given of the variability encountered (assuming that more than one sample was taken from each layer).

Cannell *et al*. (1993), in analysing the data obtained from the 302 peat samples taken by Pyatt *et al*. (1979) across Britain, do not give separate values for the acrotelm and catotelm, although samples were taken at 5-15 cm and 30-40 cm depths. They simply state that the bulk density of deep peat in British blanket bog ranges from 0.07 to 0.15 g cm^{-3}. They also observe that shallow peats in the British uplands (less than 45 cm thickness) were found to have higher bulk densities of around 0.2 g cm^{-3}.

The depth ranges of their measurements suggest that perhaps the shallower samples included peat from both acrotelm and catotelm in terms of the standard Clymo (1992) model, possibly having been taken from either side of Position A in Figure 7. The deeper samples would all appear to have been taken from the catotelm (or were perhaps taken from just above Position B in Figure 7). It is evident that the sampling strategy described by Cannell *et al*. (1993) gives rise to uncertainty about what exactly has been sampled – is it acrotelm peat, catotelm peat, or a mixture of both?

The ECOSSE Report (2007) examines bulk density in several ways. It begins by estimating total carbon stocks within two particular study areas. As part of this work, the ECOSSE Report (2007) notes that for the two study sites (Plynlimon, Wales and Glensaugh, Scotland), bulk density actually *decreased* with depth – which is in direct contradiction with Clymo's (1992) model shown in Figure 6. The ECOSSE Report (2007) recorded figures of 0.2 g cm^{-3} at depths of 0 – 15 cm depth, and 0.12 g cm^{-3} at depths between 50 – 65 cm. Interestingly both values are clearly more typical of the catotelm than the acrotelm. This does not, however, explain why the very large number of samples taken from these two widely-separated sites should show such a clear *reduction* in bulk density with depth within what appears to be catotelm peat (given the values cited).

The second phase of the work [ECOSSE Sect. 1.5] involved collating a large quantity of soils information held in the National Soils Database. Bulk density values for peat deeper than 1 m were predicted on the basis of regression analysis using existing soil-profile data, reinforced by additional field data gathered specifically for the purpose. This entire dataset, however, still consists of only 18 soil profiles scattered across Scotland and Northern Ireland. The use of only 18 profiles to extrapolate across the whole of Scotland is clearly not entirely satisfactory. This is particularly so given the considerable spatial variability in bulk density recorded on a single site by Shotbolt *et al*. (1998) and Laiho *et al*.'s (2004) conclusion that significant variation in bulk density within a site occurs over distances of 5 m at most.

An appendix to the ECOSSE Report (2007) describes a regression equation to predict dry bulk density values. The regression model is based on 15 cores taken from organic soils in Galloway, the Loch Bradan catchment, the Mourne Mountains in Northern Ireland, the Mharcaidh catchment, and from Shetland. In all, 39 soil horizons were identified and analysed from these cores. The results include the following observations:

> *"Plots of dry bulk density against depth showed a sharp decline up to approximately 15 cm depth but then levelled out until depths of around 100 cm where there was a slight increase in dry bulk density ... As this increase in bulk density with depth is only evident in a few samples, it is perhaps best that further data be collected to substantiate this..."*
>
> *ECOSSE Report, Appendix 1 (2007)*

Finally, the ECOSSE Report (2007) considers data obtained for one site in Wales (Hafren, Plynlimon), one in NE Scotland (Glensaugh) and one in NW Scotland (Ullapool). Bulk-density figures for each site are based on samples taken from 0 – 20 cm depth. These bulk density values range from 0.08 g cm^{-3} at Hafren to 0.14 g cm^{-3} at Glensaugh, with Ullapool between these at 0.12 g cm^{-3}. The latter two values appear to be more typical for catotelm peat while that for Hafren tends towards values found in the acrotelm.

Nothing can be said about whether there is any evidence of the reduced bulk density with depth, as observed elsewhere in the ECOSSE Report, because only a single value for bulk density is given for each of these three sample sites. The depth of sampling to obtain these values extends across the whole acrotelm and well into the catotelm of the standard Clymo (1992) model and includes peat from above and below Position A in Figure 7. Once again, there is therefore uncertainty about what exactly the bulk-density values describe – is it the acrotelm, catotelm, or a mixture of the two? The clear evidence from the ECOSSE Report (2007), however, is that where bulk density can be related to depth, the highest bulk-density values are found in the uppermost layers and there is generally a *decrease* in bulk density beneath these surface layers. The ECOSSE Report (2007) ascribes this greater bulk density in surface layers to the effects of land management.

4.3.1.2 Single cores, continuous sampling

Some measurements of bulk density are based on peat cores which are then analysed at close and regular intervals down their length, thereby in effect giving a continuous record of bulk density down the core. Shotyk (1995), for example, gives a detailed breakdown of bulk density measured at intervals of 5.5 cm down 1 m cores taken from Laxford Bog in north-west Sutherland and from Foula in the Shetlands. Bulk density values are very low (less than 0.05 g cm^{-3}) in both sites for the first 7 cm. This fits well with the standard Clymo (1992) model. Bulk density at the Shetland site then rises to almost 0.14 g cm^{-3} at 20 cm depth before settling down to around 0.12 g cm^{-3}. There is another rise in bulk density near the base of the sample, where bulk density exceeds 0.15 g cm^{-3}. Laxford Bog also rises to around 0.14 g cm^{-3} at 20 cm depth, but then values immediately decline again, falling back to around 0.08 g cm^{-3}. Shotyk's (1995) pattern does, therefore, seem to reflect the standard Clymo (1992) model, although catotelm values appear to be quite variable, and remain so even throughout a single peat column.

Coggins *et al*. (2006) provide detailed peat-soil data for a raised bog and a blanket bog in western Ireland. Two cores to 45 cm depth from the raised bog (Knockroe) and three cores to a similar depth on the blanket bog (Letterfrack) were analysed at 1 cm intervals for a variety of features, including dry bulk density. The results can be seen in Figure 8 (a) and (b) for Knockroe and Figure 8 (c) for Letterfrack.

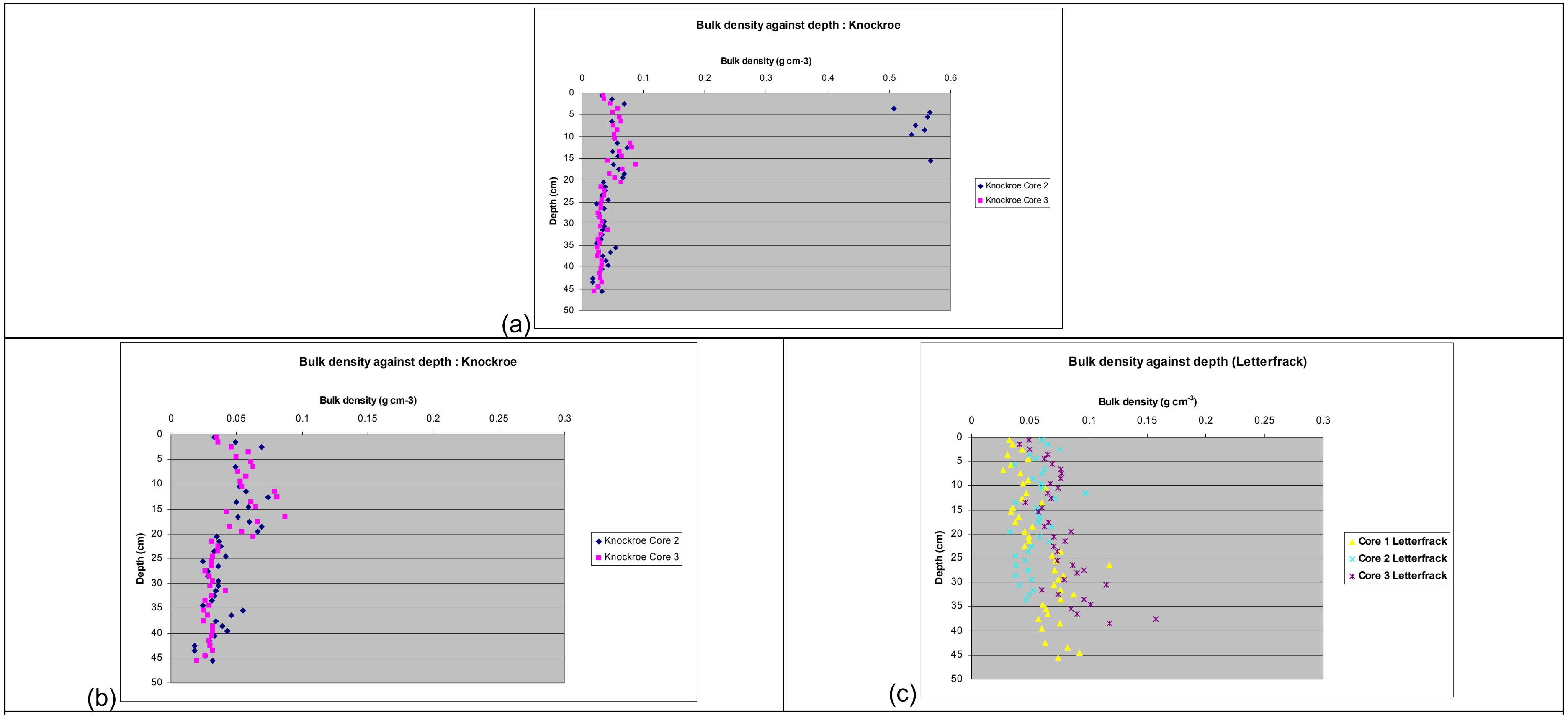


Figure 8. Bulk density profiles for Knockroe raised mire and Letterfrack blanket mire, western Ireland.

Data taken from Coggins *et al*. (2006). Profile (a) shows all data for Cores 2 and 3 of Knockroe raised mire, including a series of extremely high bulk-density values within the uppermost 15 cm of the core. Profile (b) shows the same data but excluding the very high values identified in Profile (a) in order to reveal the pattern displayed by the remaining data. Profile (c) shows data for 3 cores taken from Letterfrack blanket mire.

It can be seen from Figure 8 (a) that the raised bog site at Knockroe has a number of extremely high bulk density values (>0.5 g cm^{-3}) within the first 15 cm of the core. These are intermixed with values more typically associated with acrotelm densities. These high values may be simple errors, or more probably they reflect the influence of past land management. Alternatively, the high values may have been obtained because of dense woody fragments from the dominant heather (*Calluna vulgaris*) cover on what is evidently a somewhat dry site. These are such high values that it is perhaps justifiable to set them aside for the moment as an artefact, and focus instead on the other values in the datasets for the cores obtained for Knockroe.

From Figure 8 (b), it can be seen that the two cores from the Knockroe raised bog initially show a slight trend towards increased bulk density with depth. This continues to a depth of around 20 cm, with values approaching 0.1 g cm^{-3}. There is an abrupt shift to a fairly constant and much *lower* value of around 0.03 g cm^{-3} in the peat below 20 cm depth. This mirrors the trend of reduced bulk density with depth observed in the bulk density profiles described by the ECOSSE Report (2007) but again shows the precise opposite of that predicted by Clymo's (1992) standard model. This further suggestion of a reverse in bulk density with depth compared to the standard Clymo (1992) model is both curious and potentially highly significant.

Perhaps the transition at 20 cm depth in Knockroe does indeed reflect the transition between the acrotelm and the catotelm, but some explanation is then needed as to why the catotelm peat should be *less* dense than the peat of the acrotelm. An alternative explanation is that the shift at 20 cm does not reflect the acrotelm-catotelm boundary, but is instead a shift in bulk density within the catotelm, reflecting a climate phase during which the bog accumulated peat rapidly and with much-reduced levels of decomposition. Without information about the preserved macrofossil remains in this core (and no such evidence is provided), it is impossible to say whether this is the case or not.

The picture from the blanket bog site of Letterfrack (Figure 8 c) is initially similar to that observed at Knockroe. Bulk density values for Cores 1 and 3 begin at around 0.04 g cm^{-3} and then steadily increase to just less than 0.1 g cm^{-3} at a depth of 20 cm. However, there is no abrupt decline in density at this point, as there was at Knockroe. The values continue to increase steadily until by around 40 cm depth they have reached 0.12 g cm^{-3}. In contrast, Core 2 at Letterfrack begins with bulk-density values of around 0.065 g cm^{-3} and these values remain fairly typical for the uppermost 20 cm or so of the core. At around 22 cm, although much less dramatic than the shift in values observed at Knockroe, there is nonetheless a clear shift in values, with bulk density *falling* to around 0.049 g cm^{-3} in the peat below about 22 cm depth.

The data from Letterfrack raise a number of questions. Does the steady, gradual rise in bulk density noted in Cores 1 and 3, and the absence of any abrupt increase in bulk density in these cores, suggest that the acrotelm/catotelm boundary has not been reached by these 0.5 m cores? Alternatively, does the absence of an abrupt change in bulk density suggest that the bog at Cores 1 and 3 has no acrotelm, and the values simply reflect a gradual compaction of catotelm peat? And what of the small though distinct shift in Core 2? If this shift at 22 cm represents the acrotelm/catotelm transition, why are the catotelm values *lower* than those of the acrotelm? A longer, 1 m corer might answer some of these questions because it should reach any acrotelm/catotelm transitions lying at greater depths; the acrotelm rarely, if ever, extends more than 1 m into the peat.

At this point it is worth observing that in purely practical terms it is not easy to obtain bulk density values for peat at depths of more than 50-100 cm. To measure bulk density it is necessary to obtain a fixed volume of undisturbed peat, and obtaining such a sample from depths of more than 50-100 cm into a bog profile is technically difficult. This is especially the case where the peat has a very low bulk density and a very high water content. Such peat is often extremely sloppy and therefore tends to collapse within, or spill from, the chamber used to obtain the peat sample. Indeed the very act of pushing the sampler into the peat may affect the bulk density of the peat by both compressing material further down the peat column when the cutting edge of the sampler is forced through denser layers (such as the root mat) and by lateral squeezing if the sampler itself occupies any significant volume.

Examples of cores taken in the southern Pennines by the UEL Peatland Research Unit (Figure 9) illustrate the difficulties of obtaining undisturbed, or even measurable, cores where the peat is very soft. If it is not possible to sample such peat in any way which permits meaningful bulk-density figures to be obtained, the inevitable result is that such peats will be under-represented within the range of published values.

The more liquid the peat, the less likely it is to have been measured for bulk density. It seems likely, therefore, that there is some bias in the range of published figures for the bulk density of blanket mire peat towards the denser peats and for there to be a threshold of fluidity below which relatively few, if any, measurements have been obtained. In some cases this fluidity will be because the peat is highly decomposed yet water-saturated, while in other cases the soft, liquid nature of the peat reflects the absence of damage to a vigorously-growing, natural mire system.

The lack of bulk-density measurements from such fluid peats is a potentially significant issue for two reasons:

- the least-disturbed, most natural blanket mires, and therefore the mires of highest conservation value, are likely to have some of the softest peats but may thus also have relatively few associated measurements of bulk density;
- the bias towards denser peats will affect overall calculations of carbon stored within peatland soils, tending to make the carbon store larger than it actually is.

Figure 9. Peat cores in soft, wet peat.
Peat cores taken using a Russian peat corer from a blanket mire site in the southern Pennines. The image shows two sections of the same core which together make up a total core length of 1 m, taken from Todmorden Moor, near Rochdale. The uppermost section represents ground surface (on the left) to 50 cm depth (far right),while the lower image represents the section from 50 cm (on the left) to 100 cm depth (far right). It is evident that the peat in the uppermost 20 cm of the core is so wet that no sample was obtained, while the liquid nature of the lower section of core makes it impossible to obtain any meaningful 'undisturbed standard volume' of the peat from this part of the peat column. Consequently it would be very difficult to calculate bulk density values for the peat in this area.

Photo © Richard Lindsay

This is one of the concerns about bulk-density figures obtained using a traditional peat corer, which has a significant bulk which must first be forced down through the peat before the sample is taken. The standard 'Russian' corer is designed to minimise this, but cannot remove the effect altogether. Thin-metal box samplers have been used to depths of 1 m (see Figure 10) but the practicalities of lifting an undisturbed monolith of peat longer than this out of the bog, are considerable.

Certainly there are relatively few bulk-density data available from depths of more than 50 cm from British or Irish peat bogs. Eaton *et al*. (2007) highlight this very problem for peatlands in the Republic of

Ireland. They emphasise the urgent need for a better understanding of bulk density across a wider range of sites than is currently the case. They also highlight the fact that bulk-density values are almost entirely restricted to shallow surface layers of the bog, and there is also thus an urgent need for surveys of bulk density at depth if accurate estimates of the carbon stored in Irish bogs is to be obtained.

Tomlinson and Davidson (2000) found bulk densities of 0.069 g cm^{-3} in the raised bogs of Northern Ireland, and did not observe any increase in bulk density with depth. This contrasts with data from Holden, Burt and Cox (2001), who give bulk-density values for the uppermost 50 cm of the blanket bog mantle at Moor House National Nature Reserve, north Pennines. They recorded densities of 0.15 g cm^{-3} at the surface and 0.18 g cm^{-3} at a depth of 20 cm for vegetated areas. This increased to 0.27 g cm^{-3} at a depth of 50 cm. Areas of bare peat were associated with substantially higher values at the surface, with values of 0.22 g cm^{-3}. This rose slightly to 0.25 g cm^{-3} at 10 cm depth, then showed a marked increase such that by 20 cm the bulk density was 0.35 g cm^{-3}, but it stabilised at this density to 50 cm depth. There is thus no suggestion of a decrease in bulk density with depth in this case, but equally there is no clear evidence for Clymo's (1992) low-density acrotelm either. Nor is there any evidence for what happens to the bulk density at greater depths.

Figure 10. Square box sampler.
Dr Bill Shotyk, University of Heidelberg, taking a square 1 m monolith of peat for lead (Pb) pollution analysis in Shetland. The box sampler can be seen bottom right, while the square-section core obtained from the sampler is about to be sub-sampled by Dr Shotyk.

Photo © Richard Lindsay

Eaton *et al*. (2007) present a range of bulk-density values for the raised and blanket bogs of Ireland. Combining their own measurements with a number of cited values, they highlight the fact that assumptions of a constant value for bulk density are almost certainly not appropriate. They present mean values for blanket bog which vary from 0.14 g cm^{-3} in the uppermost 5 cm to 0.53 g cm^{-3} at 50 cm depth. It has to be said that this is an extraordinarily high mean value (based on only 4 samples) at such a shallow depth. Perhaps the sampled blanket bogs were very shallow and thus the values at 50 cm reflect increasing density close to the mineral sub-soil. The figures they give for raised bog shows the opposite response, with values of 0.12 g cm^{-3} in the surface layers, decreasing to 0.09 g cm^{-3} at 50 cm depth.

4.3.1.3 Sampling of exposed peat faces

One set of values extending to greater depth in a British blanket mire is provided by Tallis (1985), who gives bulk density profiles for six locations and eight peat types to depths of 270 cm from Featherbed Moss, in the southern Pennines. He solves the problem of sampling wet peat by using frozen cores, thereby largely preserving the proportional volumetric relationships within the peat. This method is not, however, one that is, or even can be, applied in the majority of surveys. Specific values thus obtained range from around 0.05–0.21 g cm^{-3}, with *Sphagnum* peat giving the lowest average value of 0.094 g cm^{-3}. Interestingly, it is examples of highly humified (*i.e.* decomposed) *surface* peat which give the highest average bulk-density value of 0.157 g cm^{-3}. Indeed it is quite evident that four of the six profiles examined by Tallis (1985) have their highest bulk densities in their uppermost 10–12 cm, with distinctly lower bulk densities below this. Clymo's (1992) standard model predicts quite the reverse.

Two things are worth noting about the bulk-density data presented by Tallis (1985). Firstly, many of his samples were taken from stream-sides, erosion gullies or peat haggs which offered pre-existing peat faces. These faces were 'cleaned up' (*i.e.* the dried, exposed surface cut away to reveal fresh material) prior to sample chambers being pressed into the peat face.

The use of such exposed faces for bulk-density determinations poses some problems. This is because a well-established peat face acts like a one-sided drain cut into the peat. Drainage from the peat causes the peat matrix to collapse into a more dense state and some of this peat will then also be lost by oxygen-driven decomposition (*e.g*. Hobbs, 1986; Lindsay *et al*., 1988; Edil, 2003; Holden *et al*., 2004; Holden *et al*. 2007a,b). The details of drainage processes will be explored in more detail later in the present document. For the moment it is sufficient to observe that bulk-density estimates obtained from such established faces may be significantly higher than bulk densities typical for the main body of a peat bog.

The second observation to make about Tallis's (1985) figures for bulk density concerns the fact that it appears to reverse Clymo's (1992) bulk-density model. The highest values for bulk density are obtained from the uppermost parts of the peat profile. Why might this be? Featherbed Moss is a blanket mire which is both significantly eroded and which has endured substantial surface impacts in the form of air pollution, burning, and heavy sheep grazing. It may be that the greater density of the peat close to the surface reflects both the drying effect of erosion and the inhibiting effect on vigorous peat growth of these various surface stresses.

As an example of values reflecting an extremely dry set of exposed peat faces, Yeloff, Labadz and Hunt (2006) present bulk-density values for one peat face spanning a depth of 13.5 cm. At the surface, dry bulk density is given as 0.39 g cm^{-3}, rising to a remarkable 0.62 g cm^{-3} at 2.5 cm depth. Values remain above 0.3 g cm^{-3} until a depth of 8 cm, at which point they fall abruptly to 0.18 g cm^{-3} before rising to 0.27 g cm^{-3} at 13.5 cm depth. All of these values are high compared to those generally cited for British blanket mire, some of them quite extraordinarily so. This may in part be explained by the use of long-established peat faces which have dried out significantly some way into the peat. However, examination of the vegetation described for this blanket mire also reveals that it is a system which, at least today, is dominated by a "grass and dwarf shrub" community almost certainly resulting from human land management. As such, it may not be a peat-forming system at present. Once again, however, we see a bulk density which is lower at depth than it is in the near-surface layers.

Giving a picture of bulk density at greater depths, Yang and Dykes (2007) and Dykes and Warburton (in press) have taken samples from exposed peat faces shortly after peat-slides. Their focus has been the failure zone of these peat slides, and thus they have tended to examine conditions towards the bottom of the catotelm close to the transition zone between peat and mineral at the base of the catotelm. Their measurements have typically been taken from depths of around 2 m. At these depths, they have obtained values of between 0.12 and 0.17 g cm^{-3} dry bulk density, which is markedly lower than the values noted by Holden *et al*. (2001) for the upper layers of Moor House peats.

4.3.1.4 Bulk density and humification

Clymo (1983) highlights the fact that a strong and linear relationship has been demonstrated between bulk-density and degree of humification (decomposition-level) in peat. This is interesting and potentially valuable because degree of humification for peat soils has been standardised for more than 70 years using a well-established scale originally developed by von Post (1924) for use in the field.

Clymo (1983) provides a clear and easily-used table for the von Post scale derived from the descriptions provided by von Post and Granlund (1926), and also gives the quantitative relationship between bulk density and von Post value as summarised by Päiväinen (1969):

$$\rho \text{ (bulk density in g cm}^{-3}\text{)} = 0.1 \text{ H (von Post scale)} + 0.04$$

In theory, therefore, a peat sample taken by a corer at any depth could be measured in the field according to the von Post scale and the approximate value of bulk density derived using this formula. However, the approach does not appear to have been widely used in Britain, although von Post values themselves are often included in site descriptions given in the literature.

One problem with this approach, of course, is that softer, more liquid peats may still collapse to some extent within the sampling chamber while being drawn up to the surface. Although the von Post degree of humification may still sometimes be determined in such cases, it is more difficult to be sure of where precisely within the peat column the derived bulk-density value should apply.

Nonetheless, there would seem to be some value in the more extensive use of von Post field measurements at a range of depths within (particularly) blanket mire areas with a view to obtaining at least some rather more accurate picture of bulk density for the entire thickness of peat within the blanket mire resource as a whole.

4.3.1.5 Bulk density and hydraulic conductivity

A further feature of interest to note in relation to bulk density is the fact that as bulk density increases the hydraulic conductivity of the peat decreases; *i.e.* the denser the peat the slower the water-seepage rate through the acrotelm or catotelm. Surprisingly, although Dooge (1975) and Ingram (1983), for example, discuss hydraulic conductivity in relation to degree of decomposition, and despite the close relationship shown between humification and bulk density discussed by Clymo (1983), there appears to be little evidence of von Post values being used to link bulk density with hydraulic conductivity in peat, or even of measured bulk-density values being related directly to hydraulic conductivity.

The apparent lack of such a potentially useful connection is curious, given the somewhat coarse grain of the von Post scale. A well-defined relationship between two such directly-measurable parameters as bulk density and hydraulic conductivity would appear to be a useful tool, but it is all-but impossible to find any such relationship demonstrated within the available literature. Indeed within the literature it is difficult to find examples where both bulk density and hydraulic conductivity have been measured for the same sample of peat.

This mystery may be explained to some extent by Ingram (1983), who notes that although permeability of peat varies inversely with bulk density, Päiväinen (1973) found that the relationship was not as straightforward as might be expected. A clearer relationship could instead be demonstrated between conductivity and the von Post humification scale.

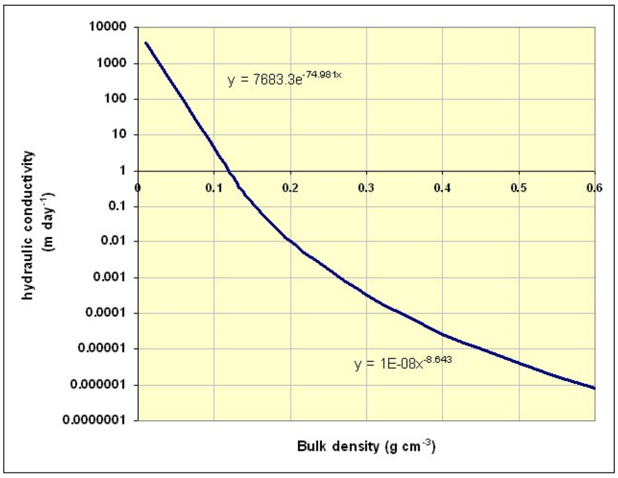


Figure 11. Indicative relationship between bulk density and hydraulic conductivity. Based on measured figures for bulk density and hydraulic conductivity provided by Boelter (1972), Braekke (1983) and Holden, Burt and Cox (2001), this graph gives at least an indication of the possible relationship between bulk density and hydraulic conductivity in bog peat. The curve between bulk densities of 0 and 0.13 g cm^{-3} is described by the upper equation, whereas the relationship between hydraulic conductivity and bulk densities greater than 0.13 g cm^{-3} is described by the lower equation.

Selecting a small number of published values for bulk density and hydraulic conductivity, and using only those combinations which are derived from the same peat samples, produces the graph shown in Figure 11, which does suggest that the relationship may not be a simple linear one. Low bulk densities may give a relatively log-linear response, but as the peat becomes denser than about 0.13 g cm^{-3} the conductivity becomes a log curve which tends towards a limit somewhere around 1×10^{-9} m day^{-1}.

It must be emphasised that the indicative relationship shown in Figure 11 does not take into account possible macropore formation, peat piping, cracking, or any other form of physical disruption of the peat which is often associated with drying of the peat matrix. Such phenomena can be expected to give rise to an increase in bulk density and unpredictability in hydrological pathways.

4.3.1.6 Bulk density and lowland peat soils

In calculating total carbon stocks for Britain, Howard *et al*. (1994, 1995) used a bulk density of 0.35 g cm^{-3} for surface peat. This comparatively high value of bulk density is obtained from Burton and Hodgson (1987) for English and Welsh lowland peats. The choice is perhaps unfortunate because the review by Burton and Hodgson (1987) focuses entirely on lowland peat deposits, which are generally either fen peats of heavily drained raised bog peats. The data presented by Burton and Hodgson (1987) therefore almost explicitly exclude any values of bulk density from English and Welsh blanket mires.

Indeed Burton and Hodgson (1987), and Lindsay and Immirzi (1996), highlight the fact that a very large proportion of the lowland peat resource is now subject to intensive land uses, of which drainage is the major element and in many cases has been so for more than a century. Consequently it is hardly surprising that the 99,645 ha of lowland peat identified by Burton and Hodgson (1987) should be associated with rather high values for bulk density. The important question is whether it is appropriate to apply such values to blanket peat. The measured figures for blanket mire bulk density given so far in the preceding sections would suggest that the bulk density value used by Howard *et al*. (1994, 1995) is at least towards the high end of the typical range, and probably excessively so.

Howard *et al*. (1994, 1995) then further confuse the picture by steadily increasing their bulk-density estimates down through the peat profile to a value of 0.5 g cm^{-3} by 5 m depth because they assumed that the peat at depth became steadily more compressed by the weight of the peat above it. This assumption is probably wrong because of simple hydrophysics. As long as the catotelm peat is submerged beneath the bog water table, the material will be supported within the water, much as Archimedes was in his bath. Consequently any downward compression of material is likely to be somewhat limited. Certainly the various figures for bulk density at depth reviewed so far do not point convincingly to any substantial increase in bulk density with depth.

It is undoubtedly true that bulk density does generally increase markedly right at the base of a peat bog, but this is because the basal peat contains an increasing quantity of mineral matter derived from the mineral sub-soil. In these circumstances the increasing value for bulk density does not represent an increase in carbon content. Indeed it may be quite the reverse because particulate matter from the mineral sub-soil increasingly takes the place of carbon-rich organic matter within the soil matrix.

4.3.1.7 Bad á Cheo: a window into the bulk density of blanket peat

One of the most detailed and illuminating analyses of bulk density for a British blanket mire to date has been provided by Shotbolt, Anderson and Townend (1998). Their study involved an area of blanket mire area known as Bad á Cheo, situated in Caithness, northern Scotland. Part of the site had been planted with conifers in 1968, and there was additional planting in 1987. Prior to planting, a detailed topographic survey of the site was carried out. In 1996, Shotbolt *et al*. (1998) re-surveyed the ground and examined the peat to determine the effects of this afforestation programme on the surface topography and the properties of the peat.

This is such a valuable piece of work that the results are worth considering in some detail. Shotbolt *et al*. (1998) examined two transects. Transect 1 passes through a number of forest blocks before extending some way out into open blanket mire, while Transect 2 is a short transect across open ground lying within the plantation area. As such, both transects have the potential to provide information about blanket peat which has been directly affected by afforestation, but can also give a valuable insight into the pattern of bulk density displayed by blanket peat which may have been subject only to indirect impacts of various intensities.

Shotbolt *et al*. (1998) obtained bulk density values to a depth of 90 cm along their transects. Such measurements are themselves deeper than for many studies, but these measurements were also taken at sufficiently regular intervals along the transect lines to permit a 2-D profile of bulk-density values to be built up along these transects. The resulting picture of the bulk density along their Transects 1 and 2 is shown in Figure 12 and Figure 13, along with aerial photographs to indicate the position of the transects in relation to the forest and open ground.

The first very evident feature of Figure 12 and Figure 13 is the way in which the forest blocks have had an impact on the bulk density of the peat beneath. This is an aspect to be discussed in more detail under Discussion Topics 2 and 3a so will not be explored further here, other than to highlight the fact that land management is clearly able to influence both carbon density and peat thickness. Thus an area which was formerly forested but which is now open land, may still retain a legacy of carbon density and of reduced peat thickness from that period of afforestation.

The second thing to note is that Shotbolt *et al*. (1998) indicate one area as having been planted in 1989 ("7-year forest") on Transect 1, but this would appear not to have been planted. Adjacent parts of the sub-compartment are forested, but the particular section crossed by Transect 1 is treeless and drained only around its margin.

Despite this, it is clear that the bulk density in the uppermost 10 cm surface layer of this unplanted sub-compartment is consistently some 0.02 g cm^{-3} denser than the peat below it – perhaps as a result of the surrounding drainage pressures. Similarly, the comparatively close proximity of forest blocks to the whole length of Transect 2 may explain, at least to some extent, the evidently higher bulk-density values (and thus denser peat) in the surface layers of this transect.

The most significant thing to take from Figure 12 and Figure 13, however, is the fact that the peat beneath the surface layers is clearly highly variable in its bulk density. Lenses at some depth have densities as low as 0.04 – 0.06 g cm^{-3}, although the layers above may attain densities of anything up to 0.1 g cm^{-3}. Such low-density layers at depth are not unique to Bad á Cheo, as can be seen from the peat core shown earlier for the southern Pennines (Figure 9). It is clear from the photographs of the composite core that certain sections of the Pennine core are significantly less dense than other sections.

Indeed one of the surprising things about Transect 1 in Figure 12 is the fact that much of the peat within the top 90 cm of the open blanket mire displays bulk densities of 0.08 g cm^{-3} or less, with no sign of the uppermost peat giving way to a high-density zone (catotelm) beneath. Bulk-density figures from across much of the open mire, even deep into the peat, show values more generally associated with the acrotelm of lowland raised bogs.

Meanwhile the bulk-density profile of Transect 2 shows quite the reverse of the 'standard' Clymo (1992) model, with a higher-density layer near the surface, then grading into much lower densities (again, closer to acrotelm values for raised bogs) in the peat at depths of 40 cm or more. Chapman (2001) confirms these low bulk densities at Bad á Cheo, obtaining values of around 0.05 g cm^{-3} for the uppermost 1 m of the open bog area, with values rising to 0.1 g cm^{-3} in the top 25 cm of the peat within the plantation area, but even here falling to 0.06 g cm^{-3} by 40 cm depth.

What does this mean? Does the acrotelm/catotelm boundary in both Transects 1 and 2 lie at such a depth that that the 90 cm probing used by Shotbolt *et al*. (1998) is insufficiently deep to pick up this transition zone? Such a possibility seems highly unlikely. The only recorded examples of acrotelm layers penetrating to such a depth in temperate bog systems are those found in 'percolation bogs'. These have so far only been identified from the Caucasus Mountains in Georgia (Kaffke *et al*. 2000; Joosten and Clarke, 2002), although perhaps the so-called 'quaking bogs' identified by Lindsay *et al.* (1988) in the Flow Country also fall into this category.

An alternative explanation for the surprisingly low bulk-density values noted at depth by Shotbolt *et al*. (1998) might simply be that our general perception of catotelm peat, with bulk densities of 0.1 g cm^{-3} or more, fails to encompass the full range of densities found naturally in this lower layer. Even if this is the case, however, it does not resolve the puzzle of the missing acrotelm layer of the standard Clymo (1992) model.

If the catotelm has a low bulk density, it would be reasonable to expect that the acrotelm above it would have an even lower bulk density. This is not what we see in the transects at Bad á Cheo – surface layers have a *higher* bulk density. Whatever the explanation for the missing 'Clymo acrotelm', the possibility that the catotelm can display highly variable and sometimes remarkably low values of bulk density has major implications for regional estimates of carbon density and the quantity of carbon stored in British peatlands.

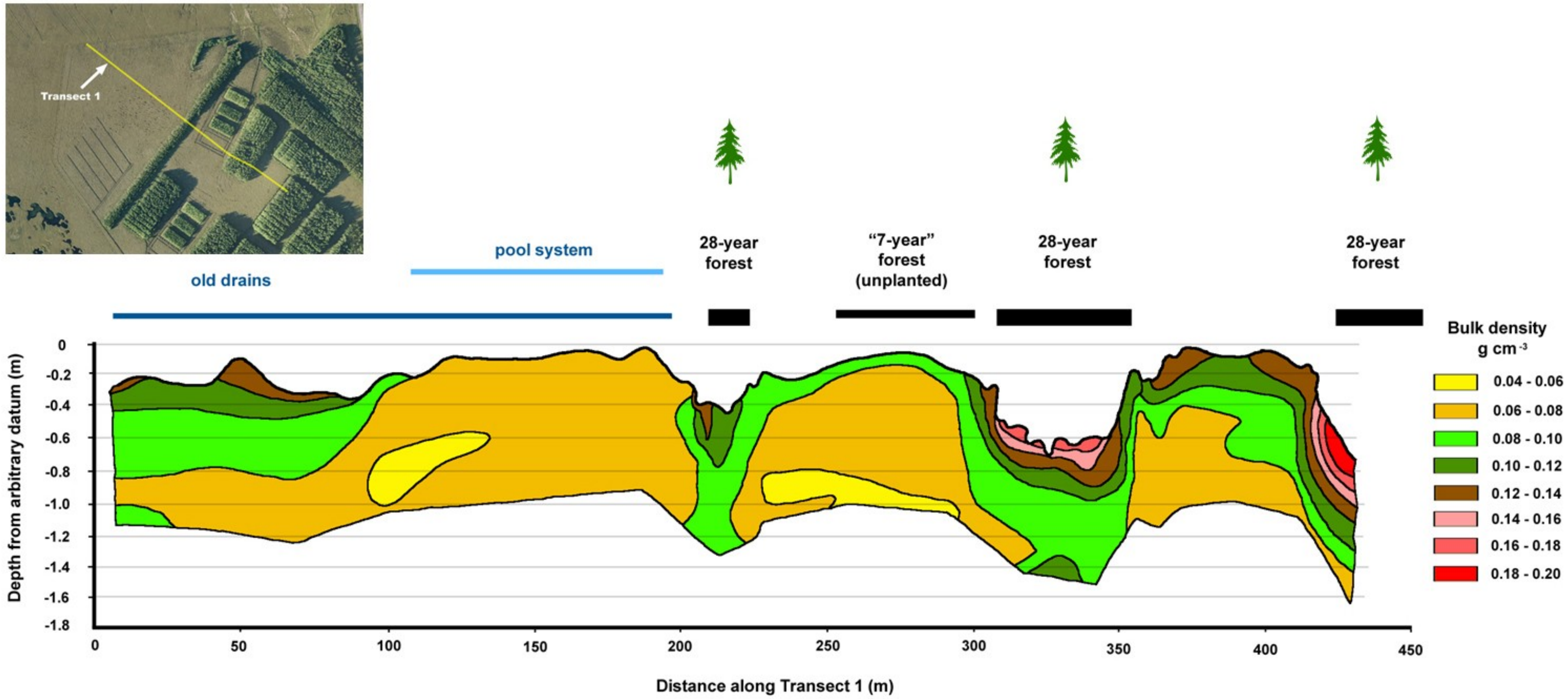


Figure 12. Bulk density profile along Transect 1, Bad á Cheo, Caithness.
An isopleth map of bulk density, obtained by Shotbolt *et al*. (1998) for the uppermost 1 m thickness of peat along their Transect 1 at Bad á Cheo. Bulk density values in g cm^{-3} are colour-coded according to the key, far right. The location of Transect 1 in relation to forestry plantings is shown on the inset aerial photograph, and planting blocks are indicated above the isopleth profile. Note the indicated presence of old drains dominating the whole of the open ground in the first 200 m, and a pool system occupying part of this ground. A rather more obvious set of drains is visible directly to the south of the 'Transect 1' label on the inset aerial photograph.

Diagram adapted from Shotbolt *et al*. (1998); aerial photograph © Getmapping.com

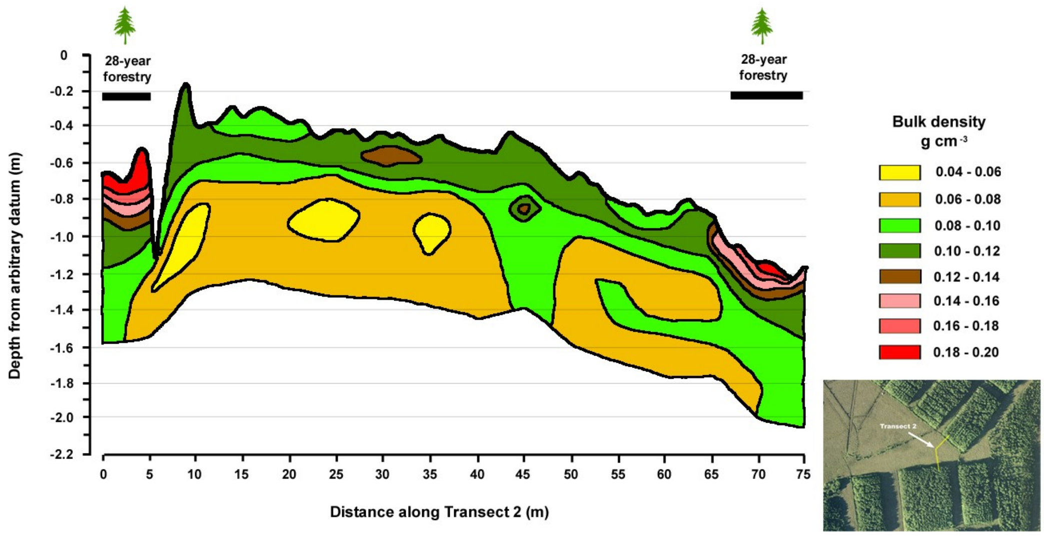


Figure 13. Bulk density profile along Transect 2, Bad á Cheo, Caithness.
An isopleth map of bulk density, obtained by Shotbolt *et al*. (1998) for the uppermost 1 m thickness of peat along their Transect 2 at Bad á Cheo. Bulk density values in g cm^{-3} are colour-coded according to the key, far right. The location of Transect 2 in relation to forestry plantings is shown on the inset aerial photograph, and planting blocks are indicated above the isopleth profile.

Diagram adapted from Shotbolt *et al*. (1998); aerial photo © Getmapping.com

4.3.2 Bulk density values from other mire-forming countries

4.3.2.1 Bulk density in the Boreal Region

It is worth considering an example of bulk-density data from peatlands in the Boreal region of Fennoscandia. Silvola *et al*. (1996) give bulk-density values for a range of peatland sites in Finland, but most of these are types of wooded bogs, with bulk densities of around 0.6 g cm^{-3}, or fen systems which reach values of 1.4 g cm^{-3} because these fens include much mineral matter within the peat. Silvola *et al*. (1996) do, however, include one undisturbed open *Sphagnum* bog, for which they give a value of 0.19 g cm^{-3} for the uppermost 20 cm (a hummock), a value of 0.26 g cm^{-3} for a dwarf-shrub pine bog, and a value of 0.35 g cm^{-3} for a 'low-sedge bog'.

These figures are the lowest cited by Silvola *et al*. (1996) for any of the Finnish mires listed, but are all generally higher than many of the values cited above for British blanket peats. These values, though more recent, are also generally higher than those given by Päiväinen (1969) for Finnish bog systems, and so should be viewed in this light.

Much of the current literature concerning peatlands and carbon storage draws heavily on studies from the Boreal region – either Fennoscandia or Canada. Thus more generally, given the importance of bulk density when calculating carbon stocks in peat, this bias in the literature (and difference in ecology) should be borne in mind.

4.3.2.2 Bulk density in northern Spain

It is worth comparing all the figures discussed so far with values given by Martinez-Cortizas *et al*. (undated poster presentation) and Fraga and Garcia-Rodeja (2008 – see University of Santiago de Compostella Windfarm Symposium 2008 website) for the small ombrotrophic mires (including blanket mires) which occur in Galicia, NW Spain. Of the ombrotrophic sites given in their listings (*i.e.* bog sites with low ash content, more than 1 m thickness of peat and a distinct organic-rich horizon), bulk density values range from 0.07 to 0.32 g cm^{-3}. Fraga and Garcia-Rodeja (2008) present a bulk-density profile which begins at around 0.2 g cm^{-3} at the surface, then falls abruptly to around 0.1 g cm^{-3} by 24 cm depth. The peat remains this dense or even *decreases* slightly for more than 2 m thickness.

It is worth noting that these blanket mires are neither eroded nor burnt, but they are currently subject to significant trampling pressure from cattle and ponies. The surface layer may thus be physically compacted more than has been the case in the past, but for whatever reason, the blanket mires of north-eastern Spain show a pattern of bulk density which, as with so many other examples, appears to be the reverse of that predicted by the standard Clymo (1992) model.

4.3.2.3 Bulk density – Southern Hemisphere

Looking briefly at the wider geographical picture, Clarkson *et al*. (2004a,b) present bulk density values for three bog sites in New Zealand. Their values are given as 0.10, 0.06 and 0.07 g cm^{-3}, yet these bogs have been generated by the growth and litter accumulation of two species of *Sporadanthus* and *Empodisma minus* respectively. *Sporadanthus* is a rush-like species, while *Empodisma* resembles a thread-like horsetail, but both species are southern hemisphere endemics (found only in the southern hemisphere). What is not clear from these figures, however, is whether these southern hemisphere bogs have a distinctive acrotelm and catotelm, and whether figures of 0.06 g cm^{-3} relate to an 'acrotelm-like' layer, or whether this value comes from deeper in the peat.

Nonetheless, what is remarkable about these figures, derived from species so utterly different from *Sphagnum*, is that they appear to create bogs with very much the same conditions of peat structure and bulk density typically found in northern hemisphere *Sphagnum* bogs. In other words, bulk density values of somewhere between 0.05 and 0.15 appear to be something of a universal feature for open bogs in the temperate (or Nemoral) region of both hemispheres, whatever their plant-species composition and whatever their geographical location.

4.3.3 Bulk density, land use and the standard Clymo model

In terms of carbon storage, it is reasonable to ask whether the diplotelmic nature of a bog is important. The critical factor in assessing carbon stores would appear to be the bulk density of the *main* store – *i.e.* the catotelm. However, whereas in most other habitats and soils the nature of the present vegetation has only limited implications for the existing soil-carbon store, this is not true for peat bogs.

A mineral soil is produced by weathering and erosion of bedrock, generally combined with a certain degree of accumulated organic matter which may have developed *in situ* or which may have been transported from elsewhere. In the case of a peat bog soil, the entire thickness of soil has 'grown' exclusively from successive layers of living vegetation whose remains create an (arguably) ever-thickening deposit. The nature of the deposit directly reflects the successive layers of vegetation which created it. The absence of such a peat-forming vegetation means not only that new material cannot be added to the carbon store, but also the *existing* store may undergo significant loss through oxidation processes such as those identified by Holden (2005a), as is discussed more extensively in Section 9.1 of the present report.

It is not sufficient, therefore, simply to note with some curiosity that the standard Clymo (1992) model appears not to apply to many blanket mire systems. The presence or absence of an acrotelm is an issue of prime significance for the carbon store of a peat bog. It would therefore be extremely valuable to have a means of identifying the presence, condition, and thickness of any acrotelm.

For a variety of practical reasons it would also be of considerable value if it were possible to identify the presence and thickness of an acrotelm, and the boundary between that and the catotelm, by taking a relatively simple instantaneous measurement. As we have already seen, long-term monitoring of the water table can provide such information, but, as its name suggests, this is a long-term approach and cannot give an instant result.

It has already been noted that the standard Clymo (1992) model of the acrotelm and catotelm, as displayed earlier in Figure 6, suggests that there is at least the possibility of using bulk density to provide a simple, instantaneous method of identifying the boundary between these two layers. In practice, however, it unfortunately appears that bulk density measurements do not seem to give a reliable guide to the acrotelm/catotelm boundary. Indeed as we have seen in the previous section, bulk density values in British blanket mires rarely seem to follow the 'standard' model.

The pattern seen so far in the information reviewed above has instead been a tendency for a high bulk densities in the uppermost 10 – 15 cm of the bog, displaying values of anywhere between 0.05 and 0.6 g cm^{-3}. Beneath this there lies a thickness of peat in which the bulk density may fall markedly to values of 0.03 - 0.1 g cm^{-3}. This lower density is often maintained for a considerable depth. At the base of the bog, bulk density rises again over a vertical distance of less than half a metre as the basal peat blends into the mineral sub-soil, with values rapidly exceeding values of 0.6 g cm^{-3}. It is worth emphasizing that many of the studies cited above have involved fine-scale sampling at intervals of 1 cm or so. At this level of resolution, they should therefore have been capable of identifying the 'standard' model of acrotelm and catotelm had it been present.

What, then, of this 'standard' (and, as is evident from Appendix 3, Figure 63 and Figure 6, the logically persuasive) Clymo (1992) model of an open acrotelm and a denser catotelm? Why does so much evidence appear to turn the picture quite literally on its head?

4.3.3.1 Land use and the haplotelmic bog

The answer may firstly be that many bogs have experienced significant damage through human action during recent times. The majority of land uses associated with human activity have their greatest impact on the surface layers of a bog and are thus capable of modifying or even destroying the acrotelm to a much greater extent than they can the catotelm. The surface structure of such bogs is not as it would have been had they retained a more natural character. They may be, as Ingram and Bragg (1984) have termed them, 'haplotelmic' bogs – *i.e.* bogs with only one layer.

In the absence of an acrotelm, the surface layer of a haplotelmic bog consists of the now-aerated upper layer of the catotelm. Many of the characteristics of an acrotelm may be observed, but not the pattern of low bulk-density close to the surface nor a gradient of increasing bulk density towards the acrotelm/catotelm boundary. Nor, crucially, is there a layer of vegetation capable of both protecting the peat mass from aerobic penetration and of adding fresh carbon-rich material to the long-term store. In fact the vegetation of haplotelmic bogs generally *aids* aeration of the peat and may contribute little organic matter to the carbon store. Such vegetation may thus be unable to compensate for any carbon lost through increased aeration of the peat.

The properties of the surface layers in a haplotelmic bog are in some ways quite the opposite of those associated with a natural acrotelm. Table 2 attempts to summarise the contrasting properties of these two structures. In an earlier draft of the present document, it was proposed that the modified surface layer of a haplotelmic bog might be called a 'haplotelm', but Olivia Bragg has argued persuasively (O.M. Bragg, pers. comm.) that a haplotelmic bog by definition consists of only a single layer. Introducing a second 'haplotelm' layer thus confuses and negates the whole concept of a haplotelmic bog.

The key point in considering the surface layer of a haplotelmic bog is that is that the surface layer is catotelm peat which differs from the peat below it only because it is undergoing aerobic attack. Any part of the catotelm peat can be transformed in the same way, whereas the functional character of an acrotelm differs fundamentally from that of the catotelm, although Olivia Bragg (pers. comm.) raises the interesting question of the extent to which such an aerobic catotelm layer can in fact mirror the properties of a true acrotelm.

Nonetheless, accepting the concept of a single-layered bog, from a practical point of view there is still a need to distinguish between the current zone of aerated catotelm and the (largely) anaerobic catotelm beneath. Repeated use of the term 'surface or aerobic layer of a haplotelmic bog' is cumbersome and in some cases can be ambiguous.

The present report therefore uses the term 'haplotelm' to mean the main aerated zone of a haplotelmic bog, but does so with the important caveat that the haplotelm should be seen merely as a modified part of the catotelm, rather than as a distinct and fully-functional layer in its own right.

The concept of the haplotelmic mire is a critical issue both in terms of carbon balance and the natural functions of bog systems. There is a possibility that a great many studies undertaken on British and Irish blanket mires have in fact focused on mires which are somewhat more haplotelmic than diplotelmic.

The carbon balance of such sites may thus differ significantly from the natural condition, but there is a danger that if such sites are not recognised for what they are, assumptions will be made and models will be constructed based on a false understanding of how the natural system functions.

Table 2. Contrasting characteristics of acrotelm and haplotelm properties

Property	Acrotelm	Haplotelm
Surface topography	Generally soft and undulating underfoot	Tussock growth-forms, or hard, 'lumpy' surface underfoot, or firm bare peat
Vegetation	Characterised by typical bog-forming species and generally abundant peat-forming *Sphagnum*	Vegetation dominated by tussock-forming bog species, or wet-heath or dry-heath elements of bog vegetation, or largely lacking vegetation
Air penetration	High at surface, decreasing steadily down through the acrotelm, ends at catotelm junction	Moderate through haplotelm profile, though any cracks increase air penetration significantly, and may extend beneath haplotelm
Water conductivity	Easiest at the surface, declines with depth through the acrotelm	May be most difficult at the surface (sometimes almost waterproof), but slightly easier below this
Water fluctuations	Relatively low because pore spaces are generally large	High, because pore spaces are very narrow.
Bulk density	Generally increases down the profile of the acrotelm	May decrease down the haplotelm, with particularly high values close to the surface
Mineral content	Generally very low	Somewhat elevated because decomposition removes % of the organic matter as gas to the atmosphere
Contribution to long-term storage	Provides a relatively steady, if slow, supply of fresh material to the catotelm, and prevents catotelm losses by preventing air penetration.	May provide little if any fresh material to the catotelm; may even result in losses from the catotelm store.

4.3.3.2 Bulk-density sampling, land use and the haplotelmic bog

The fact that comparatively few measurements of bulk density have been obtained from depths of more than 50 cm for European blanket mire systems further complicates the observed relationship between bulk density and the functional layers of a bog. Peat sampled at depths greater than 50 cm was probably laid down when the bog was more natural than it is now, and thus perhaps provides bulk-density values more typical of the natural catotelmic, rather than the current haplotelmic, state. The tendency to take bulk-density measurements from only the uppermost 50 cm of the peat column, however, means that what is being sampled is, very approximately, peat from the last 500 years. The zone being sampled is thus one which has probably experienced more intensive land-use impacts than at any time during the accumulation of the entire peat deposit. Unfortunately this is rarely acknowledged when bulk-density samples are taken and the values then subsequently presented.

The comparative scarcity of bulk-density values from deeper, less disturbed, layers diminishes our capacity to judge the role played by more natural peatlands in the story of carbon balance. Indeed the failure of sampling to reflect the spatial variability of bulk density, and the relative paucity of bulk density values from depths of greater than 0.5 m to 1 m, together represent a very significant constraint on efforts to estimate carbon budgets.

Given these various concerns, it would be of the very greatest value to determine the extent of UK peat bog systems which are truly 'diplotelmic' (*i.e.* have clear examples of both acrotelm and catotelm), and the extent of systems which are now haplotelmic. It would give added confidence and hydro-ecological clarity when reading such statements as:

> *"...Table 1 provides data from an undisturbed blanket peatland hillslope in Upper Wharfedale, UK..."*
>
> *Holden (2005a, p.2897)*

The extent to which any blanket bog in Upper Wharfedale can be described as "undisturbed" is currently a moot point, and the same could be said for a large but as-yet unknown proportion of the British blanket mire resource as a whole. Until a reasonable map of truly 'diplotelmic' bog has been produced, or until authors routinely assess whether their study site is diplotelmic or haplotelmic, it will be difficult to know whether the results of particular blanket mire investigations apply to haplotelmic conditions only, diplotelmic conditions only, or to all blanket mire conditions.

4.3.3.3 The need for more high-resolution studies of bulk density

It is also worth emphasising that the true pattern of bulk density within a peat body only really becomes apparent when detailed, high-resolution studies are undertaken in more than one dimension (*i.e.* using more than single cores), as demonstrated by Shotbolt *et al*. (1998).

Unfortunately such high-resolution studies are very rare. This is a major cause for concern in itself, given the very considerable influence bulk density has on total carbon store, but the question of resolution also raises a further issue which has a substantial influence on the bulk density of bog peat – namely the small-scale micro-topographic patterns which dominate every bog system.

4.3.4 Bulk density and small-scale microtopography (microtopes and nanotopes)

A typical scene from the broad blanket mire plateaux of northern and western Britain perhaps provides the most convincing proof of the need to incorporate microtopes and nanotopes into the process of measuring bulk density, and thereby of estimating carbon stocks.

Figure 14 emphasises the fact that, across parts of the blanket mire landscape, the concept of bulk density has no meaning because the peat no longer exists, having been replaced by an erosion gully. However, erosion gullies are only one form of small-scale pattern found across a bog. The photograph of Kentra Moss, Argyll, displayed on the front cover of the National Vegetation Classification volume for mire vegetation (Rodwell, 1991) shows similar gaps in bulk density, but in this case the gaps are in the form of deep bog pools.

Laiho *et al*. (2004) highlight the fact that, in a study of eleven peatland sites in Finland, variations in bulk-density values represented the second most important source of variance within the peat-soil parameters measured. This variation occurred at a statistically-significant spacing of 5 metres or less.

This result is entirely consistent with the evidence presented by Barber (1981) and, more recently, Karofeld (1998) that hollows and ridges/hummocks are persistent features which each give rise to characteristic types of peat. Karofeld (1998) follows individual hollows on Nigula Bog and Laukasoo Bog (Estonia) back from the present day through almost 2,000 years of growth, identifying the presence and shape of individual hollows by the contrasting nature of the peats formed beneath hummocks and hollows. It is highly likely that these contrasting peat types give rise to contrasting values for bulk density. Unfortunately Karofeld (1998) does not give bulk density figures for his profiles.

Such work highlights the fact that estimates of bulk density based on only a single core, or on a limited range of cores, will almost certainly be influenced by the nature of the nanotope being sampled. Given the range of nanotopes encountered on British mires, this offers considerable scope for variation in bulk density over small horizontal distances. In addition, sampling to any depth within the peat profile opens up the possibility that beneath the evident surface nanotope there is peat which may have undergone one of Barber's (1981) phasic changes to a different nanotope as a result of climate change, or may have suffered damage from fire from which it has recovered to a greater or lesser degree. The sharp peak in bulk density and ash content noted by Shotyk (1995) at around 90 cm depth in the blanket mire of Foula, Shetland, may indicate just such a disjunction.

It should not be particularly difficult to link bulk density values to nanotope if the appropriate features are noted in the field. The current surface can be described according to Lindsay, Riggall and Burd's (1985) nanotope zones, while the past nanotopes found at depth can be identified at least approximately by examining the macrofossil remains as described by Barber (1981) and by a great many other authors. Thus, for example, peat consisting of *Sphagnum capillifolium* litter is likely to have been T2/T3 high ridge/hummock, while abundant *S. cuspidatum* fragments would indicate an A1 *Sphagnum* hollow.

Figure 14. Gaps in the peat - deep gully erosion. An example of a deep erosion gully at Carn nan Tri-tighearnan, Highland Region. A single bulk-density figure for the site will not take into account the fact that in some parts the bulk density is not measurable because the peat has been lost.

Photo © R A Lindsay

Unfortunately, the majority of bulk density values given for British and Irish bogs provide little or no associated information about the precise nature of the nanotope sampled, either in relation to the current surface nanotope or in terms of the macrofossil remains and likely nanotopes associated with bulk densities for peat at different depths. This represents a substantial information-gap. Consequently it would be of the greatest benefit in future for bulk density values obtained from peatland sites to have an associated set of data:

- mesotope type (watershed, saddle, surface-water seepage, percolation fen, etc.);
- position within the mesotope (central, towards the margin, marginal);
- nanotope sampled;
- depth at which bulk density was measured (if sampling is not to be fine-scale and continuous throughout a core, then bulk-density measurements should be taken wherever the core shows a distinct change in the peat);
- generalised macrofossil analysis of each measured sample (*e.g.* *Sphagnum* remains – ideally at least to the broad *Sphagnum* groups or Sections, then other mosses, roots and leaf bases of sedges such as cotton grass, dwarf shrubs such as heather, unidentified organic matter).

While such additional data might seem, on the face of it, somewhat excessive and even overly academic, the simple fact remains that only with such information is it possible to make any really informed judgements about the nature of any bulk-density values obtained for a site. Perhaps most importantly, it helps to emphasise the fact that bulk density on a site can vary substantially for a variety of reasons, often over very short distances both vertically and horizontally.

A single bulk-density value for a site may shed remarkably little light on bulk density for the site as a whole. Indeed a single measurement may be positively misleading. Conversely, some estimate of how bulk density varies between different nanotope elements of any bog surface can be obtained. This can then be combined with an estimate of relative abundance for each of these nanotopes within the different microtope zones across the bog. This rather simple information can then be used to generate a relatively sophisticated model of bulk density across the site – a model which can be fine-tuned in the light of further bulk-density samples for different nanotope types, or as a result of improved microtope mapping.

4.4 Carbon content – a summary

Summarising the picture so far:

- there is no single agreed figure for the minimum proportion of organic matter to mineral matter necessary in order to describe an organic soil as 'peat', with some authors accepting a mineral proportion as high as 70% of dry weight, but many commonly-used definitions do not permit a mineral content greater than 20%;
- ombrotrophic bog peat is generally acknowledged to have a very low mineral content ranging between 1% and 4% of dry weight;
- water movement, surface seepage or groundwater influence can raise this mineral content to 8% or more, which is a significant factor within blanket mire landscapes, where bog and fen units link to form interconnected mire complexes;
- the greater rainfall inputs found in Atlantic regions such as Britain and Ireland mean that the mineral content of the blanket mire peat typical of these areas is higher than in ombrotrophic peats found in Scandinavia;
- the proportion of carbon atoms within the organic fraction of the solid matter in peat is subject to a significant degree of variability, but in highly ombrotrophic blanket mire peats the variation diminishes markedly, typically ranging between 50% and 53% of the dry weight of peat;
- by far the largest component of a peat soil by weight is water, with solid matter sometimes representing as little as 3% of total weight of peat and rarely exceeding 15% in the natural state (which can be compared with 5% solids for jellyfish, and 13% solids for milk);
- a 'typical' example of ombrotrophic peat may thus consist of 90% water, 10% solid matter, while of the dry weight of this solid matter, 3% may be mineral matter, 52% consists of carbon atoms, and 45% is provided by other non-carbon elements which form various side-chains and linkages within the long-chain organic compounds;
- proportionally, therefore, the amount of carbon contained by weight within a block of peat may be no more than 52% of 10% in a 'typical' peat (*i.e.* 5.2%), and may be only 52% of 2% (*i.e.* barely 1%) in very wet peat;
- in order to calculate how much carbon is contained within a specific volume of peat, proportional values for dry-matter components are insufficient because it is necessary to know how tightly packed the solid matter is within the specified volume;
- this compaction is measured as dry bulk density (g m^{-3}) by measuring a defined volume of wet peat and then determining the amount of dry matter within that volume;
- Clymo (1992) provides a 'standard model' of a bog system, in which a thin surface layer has a low bulk density (0.03 g m^{-3}) and combines with a transition zone of increasing bulk density to form the surface 'acrotelm', but then bulk-density values stabilise at a depth of perhaps 12 cm and a bulk density of 0.12 g m^{-3}, representing the main thickness of peat known as the 'catotelm';
- while logically persuasive, this standard Clymo (1992) model is often not found in practice, and in many cases the picture is reversed, with bulk-density values highest at the surface and decreasing with depth;
- one possible explanation for this reverse is that the bog surface has been damaged by human action and the acrotelm has thus been lost, thereby creating a 'haplotelmic' bog consisting only of catotelm whose surface layer is now oxidised, forming a 'haplotelm' rather than an acrotelm;
- the absence of an acrotelm from such sites has important implications for the carbon store, because the acrotelm both protects, and adds new material to, the catotelm whereas haplotelmic vegetation encourages aeration of the catotelm peat;
- a serious limitation with existing bulk-density measurements is the fact that the majority of such measurements are taken from only the uppermost 50 cm or 1 m of the peat profile, while the much smaller number of values obtained from deeper in the peat are often taken from exposed peat faces which will have suffered compaction;

- other than a surprisingly limited and at times somewhat opportunistic set of published measurements, the dry bulk density of fairly natural blanket peat at depths of more than 50 - 90 cm remains something of a mystery for much of the blanket mire resource throughout Britain and Ireland;
- one of the few detailed studies of bulk density in Scottish blanket mire has demonstrated that bulk density in peat varies significantly over vertical distances of 20 cm or less, reflecting historical changes in climate and consequent local shifts in microtopography over time, but most bulk-density sampling rarely reflects or even acknowledges this;
- bulk density in peat varies significantly across horizontal distances of 5 m or less, reflecting the present-day small-scale surface pattern of microtopography, but sampling rarely reflects or even acknowledges this;
- bulk density is one of the key factors in calculating the amount of carbon currently stored in peat, but the number of sites from which measurements have been taken is limited, and often there is only a single measurement for a bog site, taken from the uppermost layers only - consequently even when obtained for a site, bulk-density measurements may be unrepresentative of the peat store as a whole;
- sampling bulk density within the individual nanotopes of a bog, then estimating the relative abundance of these nanotopes within the various microtope patterns found on a site, offers the potential to generate a more refined picture of the bulk density across a site – a picture which can be readily fine-tuned.

5 Carbon storage per unit volume

This section seeks to present a reasonable set of values for a standard cubic metre of peat and its carbon storage; values of carbon storage are most sensitive to changes in bulk density; it is therefore important to select a value for bulk density which is 'reasonable' – but what is 'reasonable'? Examples of 'standard cubic metres of peat' for natural and damaged bogs are presented; methane is included as a product of decay in both acrotelm and catotelm, but it is suggested that most catotelm methane is not released and instead helps to explain aspects of catotelm hydrology.

So far, the present review has considered various components and characteristics of peat soils, and has summarised a range of attempts to determine the total extent and thickness of such peat soils in the UK. In order to estimate the amount of carbon stored within these soils, it is necessary to establish a basic unit of carbon storage, a unit by which any estimate of extent or depth can be multiplied to produce a figure for total carbon storage. This basic unit is the amount of carbon stored in one cubic metre of peat, derived from values for the amount of mineral matter, organic matter, water, the proportion of carbon atoms, and the quantity of this solid matter packed into the defined volume (bulk density).

It is important that this basic unit represents an accurate reflection of carbon storage because if it does not, then any subsequent multiplication by estimates of extent and depth to produce figures for total carbon for the UK will merely compound the error. It is therefore of very considerable importance either to be clear about what might be reasonable as a 'standard' quantity of carbon contained within one cubic metre of peat, or to be clear about the way in which this quantity may vary under differing conditions.

5.1 Calculation of carbon per unit volume

The soil survey undertaken by the Forestry Commission, involving 302 peat samples from FC holdings across upland Britain, and described by Cannell *et al*. (1993), has already been discussed in relation to the issues associated with estimation of bulk density. Cannell *et al.* (1993) use the data obtained from this survey to calculate a carbon content per unit volume. To do so, they use the following formula:

$$\text{carbon content} = 10 \times d \times \rho \times f_{om} \times OM_c$$

where:

10	= simply a conversion factor to turn g cm^{-2} into kg m^{-2}
d	= thickness of peat being considered, in cm
ρ	= dry bulk density in g cm^{-3}
f_{om}	= fraction of the dry matter which is organic matter
OM_c	= the proportion of organic matter which is carbon

The conversion factor of 10 means that the units of g cm^{-3} generally employed in the present report and often used for values of bulk density are converted into units where the spatial dimension becomes metres rather than centimetres, but this is convenient if we are ultimately to consider a 'standard cubic metre' of peat.

Cannell *et al*. (1993) then use the following values for the other parameters of this equation:

ρ	(dry bulk density in g m^{-3})	=	0.1
f_{om}	(organic fraction of dry mass)	=	0.94*
OM_c	(carbon fraction of organic matter)	=	0.5

(*the value for f_{om} has here been calculated using the other values provided by Cannell *et al*., 1993)

Note that the value for bulk density used by Cannell *et al*. (1993) is more characteristic of catotelm peat than acrotelm peat (according to the standard Clymo model), but is somewhat low compared with many values quoted for the catotelm.

The proportion of organic matter within the dry-matter mass as a whole suggests a mineral content for the peat of 6%, which is really rather high for ombrotrophic blanket mire. This may be an indication that much of the peat in Forestry Commission holdings lies towards the margins of very deep ombrotrophic peats, with perhaps significant zones of somewhat flushed ground.

The proportion of carbon given by Cannell *et al*. (1993) is described as the fraction within the organic matter component only, whereas most published figures give carbon as a proportion of the whole dry mass. Such published values generally put the carbon content of blanket peat at around 52% of dry mass. Removing the mineral mass from the picture and considering only the organic fraction would normally *increase* the proportion of carbon to non-carbon atoms and thus a proportion nearer to 0.54 rather than 0.5 might have been expected. However, Cannell *et al.* (1993) simply state that the value was assumed to be 0.5, with no further explanation given.

Using these various values and assuming an area covering 1 m^2 but only 1 cm thick, the calculation given by Cannell *et al.* (1993) is a follows:

carbon content = 10	x	1 cm	x	0.1	x	0.94	x	0.5
		thickness		bulk density		organic matter		carbon proportion

On this basis, Cannell *et al*. (1993) conclude that the average carbon density of the peat soils investigated was 0.47 kg C m^{-2} per centimetre depth. This equates to 47 kg C as the carbon content of our 'standard cubic metre of peat'.

The question is, how representative is this figure? To what extent can the parameters used by Cannell *et al.* (1993) be accepted as 'typical' for peat bog systems throughout Britain? Several earlier sections of the present report have explored the documented variation displayed by these parameters and it is clear that the values used by Cannell *et al*. (1993) are not the only values which might be considered typical.

Employing the same formula as Cannell *et al*. (1993) but using different values for the various parameters, specifically using values which reflect the observed range for each parameter, it is possible to explore the effect of this on the calculated stores of carbon per unit volume.

5.1.1 Mineral content : variability and influence

In one sense, the mineral content of the peat is the most variable factor of all because the definition of 'peat' depends entirely where the line is drawn along a continuum ranging from 70-80% mineral content down to 1% or less. However, as we have seen in Section 5.1, ombrotrophic blanket mire is widely quoted as containing somewhere between 1% and 4% mineral matter.

The calculated value of 6% from the figures provided by Cannell *et al*. (1993), as already observed, is somewhat high compared to this range. What influence does such a range (extending as far as 6%) have on the calculation of carbon content? In other words, how sensitive is the formula to changes in this parameter?

Calculating the resulting carbon content for mineral contents ranging from 6% to 1% reveals that while the high value of 6% used by Cannell *et al*. (1993) gives the carbon content for a cubic metre of peat as 47 kg C, reducing the mineral content to 1% increases the carbon content to 49.5 kg C – an increase of 5%. In other words, the percentage change in carbon content exactly matches the percentage change in mineral content (see Figure 15).

The limited range of mineral content within truly ombrotrophic blanket mire means that the carbon content within such areas is not likely to vary by more than a few percent. Thus in a site such as a lowland raised bog or across the clearly ombrotrophic crown of a blanket mire unit where marked zones of surface seepage tend to be limited or absent, it might be expected that mineral content would be fairly constant for much of the peat thickness and across much of the horizontal extent.

In more complex areas of blanket mire, the surface topography and the underlying landform give rise to distinct zones of seepage, surface flushing, and lines of water movement, and there are also potentially greater grazing and burning pressures. The mineral content of the peat in such areas may differ quite significantly from one location to the next.

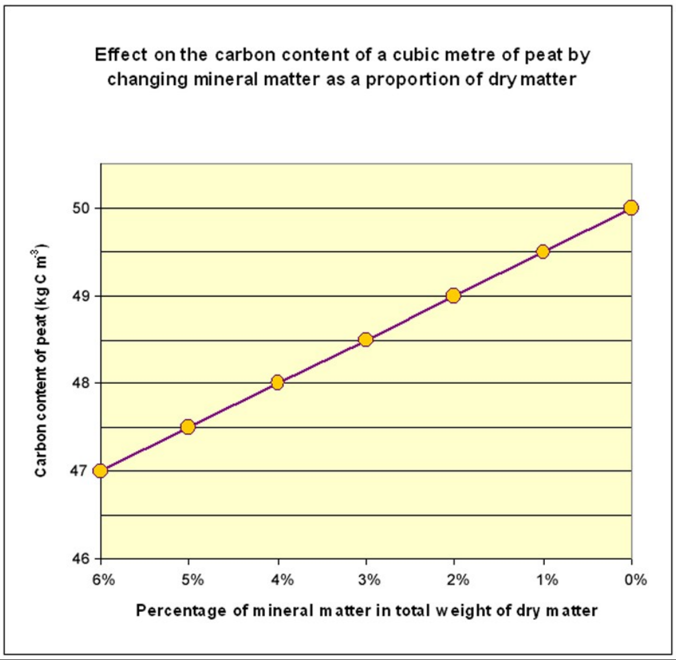


Figure 15. Changes to the carbon content of peat as the % mineral matter content is reduced.
The increased content of carbon within a 1 m^3 block of peat as the percentage mineral content diminishes. Calculations based on the formula presented by Cannell *et al*. (1993), with a dry bulk density of 0.1 g cm^{-3} and a value of 0.5 for the fraction of organic matter which is carbon.

Thus where there is an increased mineral content because the ground is more subject to water seepage, or where, as Shotyk (1995) noted in Shetland blanket peats, there are layers of markedly higher mineral content perhaps as a result of burning or heavy grazing, the mineral content may rise to 20% or more.

The difference between 1% mineral matter and 6% mineral matter in terms of total carbon stored within a 1 m^3 block of peat amounts to 2.5 kg C less in the peat with 6% mineral matter. If on the other hand, the mineral matter were to increase from 1% to 20% throughout the peat block, the carbon content in a 1 m^3 block of peat would be reduced by 9.5 kg C.

5.1.2 % carbon content of dry/organic matter : variability and influence

Given that the present report has settled on 3% of dry-matter weight as the 'standard' mineral content of ombrotrophic blanket peat, the remainder of the dry weight consists of organic material which can be divided into carbon atoms and the remainder (*e.g.* oxygen, hydrogen, sulphur, nitrogen, even iron, calcium and aluminium) all of which may be bound in complex ways to the carbon backbone. Where there are higher levels of sulphur, nitrogen or iron, it would be reasonable to expect a greater proportion of this to become bound into the organic fraction.

With an atomic weight of 56 for iron compared to 12 for carbon, 16 for oxygen and 1 for hydrogen, it is obvious that a relatively small addition of iron to the organic complex has the potential to produce significant changes in the relative contribution of carbon to the dry weight even if the volume occupied by the peat material does not change. If there is significant volume change when these extra atoms take up space within the organic fraction, the bulk density of the peat may increase but the % carbon may remain unchanged or even decrease.

The number of iron atoms incorporated into an organic complex is likely to be relatively small compared to the number or carbon, oxygen and hydrogen atoms making up the bulk of the complex, which probably explains why the range of % carbon in the organic fraction does not vary to any great extent. Nevertheless it is undoubtedly a factor influencing the amount of carbon stored in peat.

Using the Cannell *et al*. (1993) formula to investigate the effect of changing the carbon content from 46% to 54%, it emerges that a 1% change in the carbon content of the organic matter results in a change in the carbon store of 10 g per 1 m^2 per cm thickness. Calculated for a 1 m^3 block of peat, this range of values for carbon content is equivalent to 7.8 kg C.

5.1.3 Bulk density : variability and influence

One possible reason for observed changes in bulk density has been discussed in the sub-section above, but undoubtedly the major factor influencing bulk density is the extent to which the peat is compressed within the peat column. Using the formula employed by Cannell *et al*. (1993) and using values considered reasonable for ombrotrophic blanket bog, Figure 16 illustrates how changes in bulk density influence the amount of carbon stored per unit volume

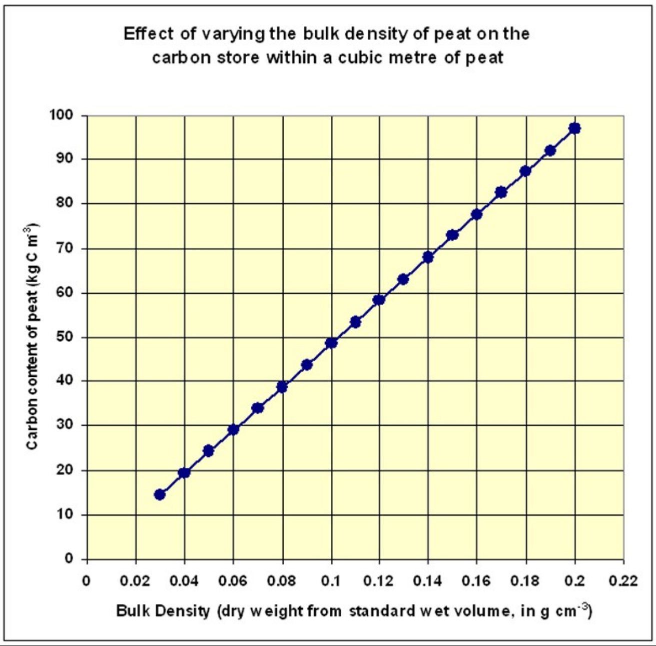


Figure 16. Changes to the carbon content of peat as the bulk density is altered.
The increased content of carbon within a 1 m^3 block of peat as the bulk density of the peat increases. Calculations based on the formula presented by Cannell *et al*. (1993), with a mineral content of 3% of the dry weight and a value of 0.5 for the fraction of organic matter which is carbon.

It can be seen from Figure 16 that bulk densities typically given by the standard Clymo (1992) model for the *acrotelm* (0.03 – 0.09 g cm^{-3}) produce carbon stores of between 14 and 54 kg C within a cubic metre of peat.

Bulk density values for the *catotelm* (0.1 – 0.2 g cm^{-3}) give rise to carbon stores of between 49 and 97 kg C within a cubic-metre block of peat. For every 0.01 g cm^{-3} increase in bulk density, this 1 m^3 block of peat adds 4.85 kg C to the carbon store.

Thus bulk-density values of 0.4 g cm^{-3}, or even 0.6 g cm^{-3}, have the potential to double or triple the amount of carbon displayed in Figure 16 though peats with such extreme values as 0.6 g cm^{-3} probably attain such values by virtue of their high mineral contents.

A cubic metre of peat with a bulk density of 0.1 g cm^{-3} contains 48 kg less carbon than a cubic metre of peat having a bulk density of 0.2 g cm^{-3}, while a cubic metre of peat with a bulk density of 0.3 g cm^{-3} contains 131 kg more carbon than a cubic metre of peat having a bulk density of only 0.03 g cm^{-3}.

Overall, therefore, if reasonable literature-based values for raised and blanket mire systems are used for the various parameters required for the Cannel *et al.* (1993) formula, it is clear that changes in bulk density give rise to the largest changes in carbon store. In other words the formula show most sensitivity to realistic changes in bulk density. The ranking is thus:

bulk density >> **mineral content** > **% carbon content**

5.2 A 'standard' value for carbon per cubic metre?

If a 'standard cubic metre of peat' is to be created as a means of calculating the carbon content of the whole peat resource, the previous sub-section clearly indicates that the values used for mineral content and % carbon will only have a relatively minor effect compared with the value(s) chosen for bulk density.

The ECOSSE Report (2007) describes a practical attempt to judge, amongst other things, the role of bulk density in the estimation of carbon storage. The first stage of this work [ECOSSE Sect. 1.4] selected two sites (Glensaugh in Scotland, and Plynlimon in Wales) for detailed modelling and field investigation. Both areas were subject to intensive field sampling within three 1 km^2 blocks, and from this information a map of carbon content across each square was constructed. This map was then compared with a modelled quantity of carbon for each square derived from DEFRA Project SP0511 (Greenhouse Gas Inventory project). What emerged was that the modelled and measured quantities of carbon differed by substantial amounts. Amongst the individual squares these differences varied between -51.8% to +192.7% (*i.e.* from around half to nearly double the measured amount of stored carbon).

Two major sources of error are highlighted as the source of this significant mis-match in figures. Firstly, it was identified that the soil-mapping series used to create the modelled values was not sufficiently detailed and it was thus not capable of reflecting real variations in soil type and peat depth. Secondly, it was recognised that small changes in bulk-density values produced substantial changes in total carbon for a given area. To quote:

> *"Selecting an unrepresentative* [bulk density] *value gave large errors."*
>
> *ECOSSE Report (2007)*

The question that this observation invites, of course is: "What is a representative value?" Immirzi *et al.* (1992) use a value of 0.1 g cm^{-3} for their global overview of peatlands, while Cannell *et al.* (1993) also use a value of 0.1 g cm^{-3}, based on their analysis of Forestry Commission holdings across Britain. The very high bulk-density values of 0.35 g cm^{-3} to 0.5 g cm^{-3} used by Howard *et al*. (1995) can probably be set aside because they are not based on a type and condition of peat appropriate to the extensive areas of blanket mire in Britain, and the increasing values of bulk density with depth appear to be based on a false assumption.

The picture obtained from Shotbolt *et al*. (1998) suggests that the main body of peat underlying the obvious drainage effects of the forestry at Bad á Cheo has a bulk density of between 0.06 and 0.08 g cm^{-3}. It might thus be argued that somewhere around 0.07 g cm^{-3} might thus be a reasonable bulk-density value for the catotelm. However, against this must be balanced the values of 0.1, 0.12, 0.2, and even 0.25 obtained from other areas of blanket mire by Cannell *et al*. (1993), Holden *et al*. (2001) , the ECOSSE Report (2007), and a number of others.

Milne and Brown (1997), while adjusting the starting point for bulk density used by Howard *et al*. (1995) down to 0.11 g cm^{-3}, nevertheless continue to use the (probably incorrect) assumption made by Howard *et al*. (1995) that bulk density steadily increases with depth. Meanwhile the ECOSSE Report (2007) explicitly highlights the need to recognise that different bulk densities occur at different depths. Consequently in its field testing of carbon models in Wales and Scotland, it uses a value of 0.2 g cm^{-3} for depths of 0–15 cm, and a lower value of 0.12 g cm^{-3} for depths of 50-65 cm. Both these values are clearly more typical of the catotelm rather than of the acrotelm. Clymo's 'standard model' of course has 0.03 g cm^{-3} for the acrotelm layer and then 0.12 g cm^{-3} for the catotelm (Clymo, 1992) – though it should be noted that integrating the bulk-density values given across the entire acrotelm layer of the standard Clymo model, including the zone of collapse, gives an integrated value of 0.06 g cm^{-3} for the acrotelm layer as a whole.

On theoretical grounds and from pragmatic observation there would seem to be a good argument for having at least three 'standard cubic metres of peat':

- an example of the natural condition, and including an acrotelm layer;
- an example of a haplotelmic mire, with a damaged surface layer; and
- an example of a cubic metre of peat entirely within the catotelm.

5.2.1 'Natural cubic metre of peat'

Given that this present report is concerned firstly to set the scene by reviewing what happens in the natural state, it would seem sensible to assume that our 'standard natural cubic metre of peat' does consist of two layers, that the uppermost layer should be a natural acrotelm of around 15 cm thickness with a bulk density of around 0.06 g cm^{-3}, while the lower 85 cm of the peat block should be a natural catotelm with a bulk density of 0.12 g cm^{-3}. The values for the various components are given in Table 3, and are shown shaded in the same colours in the form of a 'standard cubic metre of peat' in Figure 17 below.

Table 3. Values of components for natural cubic metre of peat

	Component	Value	Unit
Acrotelm	Thickness	15.0	cm
	Water (50% saturated acrotelm)	75.0	kg
	Air (approx. 40% acrotelm volume)	40	%
	Dry bulk density	0.06	g cm^{-3}
	Dry weight	7.5	kg
	Mineral content (3% of dry weight)	0.23	kg
	Non-carbon organic matter	3.35	kg
	Carbon (54% of total organic matter)	**3.93**	kg
	Carbon per centimetre thickness	**0.26**	kg cm^{-1} thickness
Catotelm	Thickness	85.0	cm
	Water (100% saturated acrotelm)	850	kg
	Air (no air penetration)	0	%
	Dry bulk density	0.12	g cm^{-3}
	Dry weight	102	kg
	Mineral content (3% of dry weight)	3.06	kg
	Non-carbon organic matter	45.51	kg
	Carbon (54% of total organic matter)	**53.43**	kg
	Carbon per centimetre thickness	**0.63**	kg cm^{-1} thickness

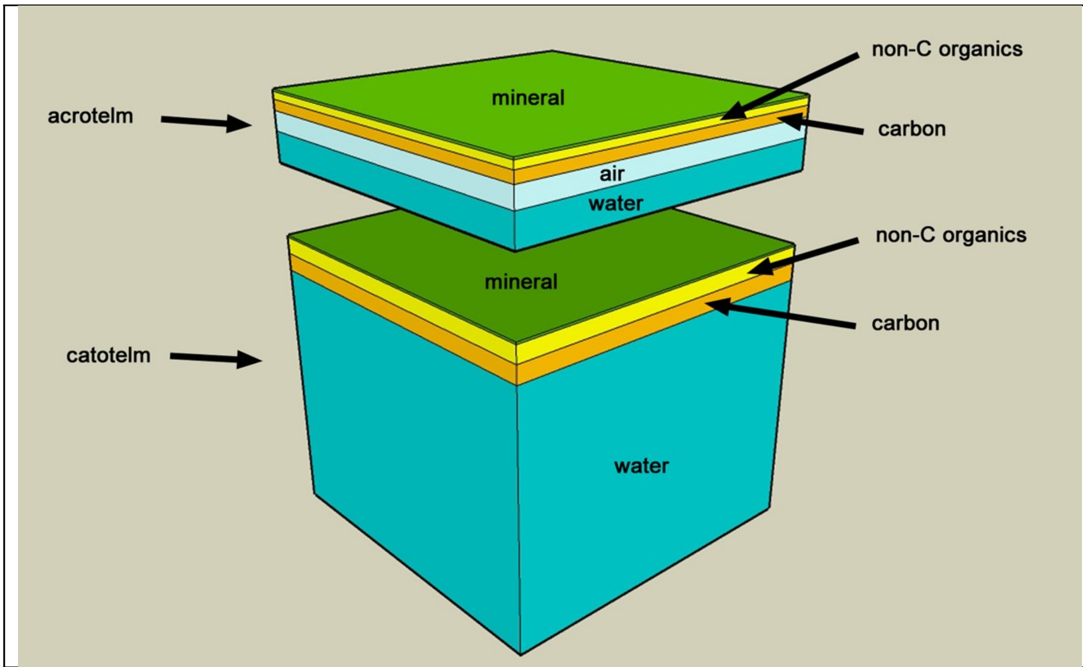


Figure 17. Quantities of material contained within the acrotelm and catotelm of a natural 'standard cubic metre of peat'
Quantities of the different materials which go to make up a 'standard cubic metre of peat' in a natural blanket bog. Materials have been divided appropriately between the acrotelm and the catotelm. Indicated relative volumes are approximately equivalent to the quantities in kg of each component, apart from water which would otherwise dominate the diagram to a much greater extent than it already does. Note the presence of air in the acrotelm but not in the catotelm; this air occupies volume but contributes little weight to the acrotelm.

Having identified the key parameters of bulk density and thickness for these two layers, it is then possible to use the Cannell *et al.* (1993) formula to calculate actual quantities for the various components of the 'natural cubic metre of peat'. In addition to the parameters used in the formula of Cannell *et al*. (1993) for the calculation of carbon, it is important to incorporate values for the quantity of water and of air.

It has been assumed that the water table is currently positioned some 7 cm below the mire surface. In other words, there has been no very recent rain, but neither has the bog been experiencing drought conditions in recent weeks.

An estimate of air volume has been made on the basis of the standard Clymo (1992) model, which indicates that gas is the dominant phase at the acrotelm surface but then steadily declines in importance down through the acrotelm profile. The total quantity of carbon held in the acrotelm is of interest, but so too is the quantity of carbon stored per unit thickness of the different layers.

Although to some extent merely reflecting bulk density values, the amount of carbon per layer thickness is nonetheless useful when comparing different layers and different 'standard cubic metres'.

5.2.2 Haplotelmic cubic metre of peat

As discussed in Section 4.3.3.1 of the present report, there is good reason to suspect that a significant proportion of the blanket mire resource in Britain consists of a modified surface layer which overlies the somewhat less-disturbed peat deposit beneath. This modified surface layer is characteristically more dense than the underlying layers of peat. It may sometimes be a disturbed form of the original acrotelm, but more commonly the original acrotelm has been lost through burning, trampling or drainage, in which case the surface layer is formed by the uppermost surviving layer of the catotelm.

Our 'haplotelmic cubic metre of peat' could be said to consist of two layers, as before, but in this case the uppermost layer is a 25 cm thickness of modified catotelm, with a bulk density of 0.15 g cm^{-3}, while the lower 75 cm of the peat block is catotelm peat with a bulk density of 0.11 g cm^{-3}. Both bulk-density figures are possibly somewhat conservative (*i.e.* the bog is not as damaged as many observed examples of blanket mire), but are based on values from the obviously-affected parts of Bad á Cheo (Shotbolt *et al.*, 1998), which was formerly a typical example of a wet patterned bog system of the Flow Country. Perhaps equivalent figures for a Pennine haplotelm would show significantly higher values for thickness and bulk density.

It should be noted that the % mineral content in the surface layer has increased compared to the 'natural' mire because organic matter in the haplotelmic mire is oxidised and lost to the atmosphere as gas. The % water has *increased* compared to a natural acrotelm because there are no large air spaces, but air penetrates both the surface and lower layer of the catotelm because the surface layer is unable to provide completely-effective protection for the remainder of the catotelm, particularly if there are cracks in the surface layer. The values for the various components are given in Table 4, and are shown in the form of a 'standard cubic metre of peat' in Figure 18 below.

It can be seen from Table 4 that the surface layer contains a significant quantity of carbon, proportionally more than in the catotelm peat beneath. The reason for this greater carbon density is not, however, greater sequestration of CO_2 from the atmosphere. It is instead caused by the partial breakdown of the surface peat after undergoing some form of (generally human-induced) impact , with consequent collapse and consolidation of originally much looser material. This denser carbon store in the surface layer is thus a sign of a system which may be currently releasing carbon rather than sequestering it.

Interestingly, Schneebeli (1991) has proposed that this 'densification' of the surface layer following damage may be of some benefit to the system as a whole. He argues that the denser, more decomposed layer at the surface provides some hydrological protection for the peat beneath, and provides a surface layer through which rainwater is lost more slowly and thus provides the possibility of re-wetting (and re-vegetating) the surface layer more rapidly.

Certainly any bitumens or waxes released during burning can form a waterproof capping (albeit discontinuously) which helps to retain moisture in the peat beneath (Clymo, 1983). The same layer can also form water-filled microsites for species such as *Sphagnum tenellum* to re-colonise the surface and begin the process of re-building a functioning acrotelm (Lindsay and Ross, 1994).

Table 4. Values of components for haplotelmic cubic metre of peat

	Component	Value	Unit
Haplotelm	Thickness	25.0	cm
	Water (75% saturated acrotelm)	212	kg
	Air (approx. 10% acrotelm volume)	10	%
	Dry bulk density	0.15	g cm^{-3}
	Dry weight	37.5	kg
	Mineral content (8% of dry weight)	3.0	kg
	Non-carbon organic matter	15.87	kg
	Carbon (54% of total organic matter)	**18.63**	kg
	Carbon per centimetre thickness	**0.75**	kg cm^{-1} thickness
Catotelm	Thickness	75.0	cm
	Water (99% saturated acrotelm)	713	kg
	Air (approx. 1% penetration)	1	%
	Dry bulk density	0.11	g cm^{-3}
	Dry weight	82.5	kg
	Mineral content (6% of dry weight)	4.95	kg
	Non-carbon organic matter	35.67	kg
	Carbon (54% of total organic matter)	**41.88**	kg
	Carbon per centimetre thickness	**0.56**	kg cm^{-1} thickness

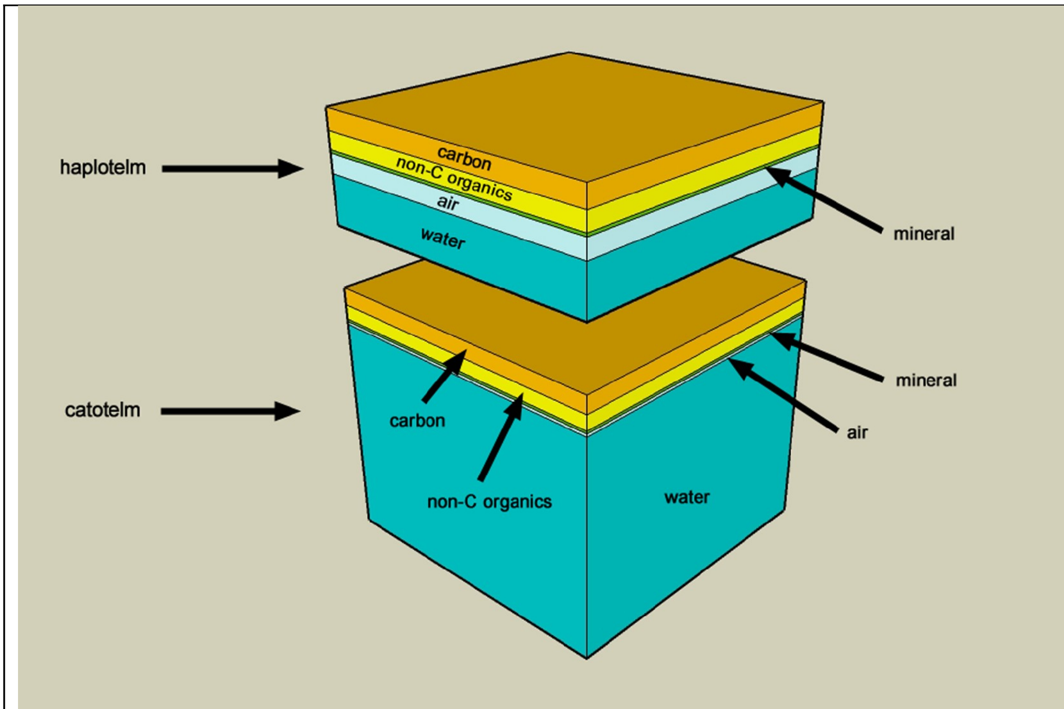


Figure 18. Quantities of material contained within the haplotelm and catotelm of a haplotelmic 'standard cubic metre of peat'
Quantities of the different materials which go to make up a 'standard cubic metre of peat' in a haplotelmic blanket bog. Materials have been divided appropriately between the acrotelm and the catotelm. Indicated relative volumes are approximately equivalent to the quantities in kg of each component, apart from water which would otherwise dominate the diagram to a much greater extent than it already does. Note the presence of air in the haplotelm *and* in the catotelm; this air enables aerobic decomposition to occur in both layers.

5.2.3 Cubic metre of catotelm peat

The nature of catotelm peat varies for a number of reasons, and many of these reasons have been explored at points in the present report. When considering the peat resource at the national level, it is sensible to ask whether there are any broad patterns of variation in bulk density shown by catotelm peat? Are there, for example, regional variations in catotelm peat? Are the pool-dominated blanket mires of the Flow Country characterised by a softer, less-dense peat than the mantle of what seems to be the drier blanket peat which dominates the broad plateaux of the southern Pennines?

5.2.3.1 Regional variation in catotelm bulk density

There is no doubt that the blanket mires east of Rochdale lie in a drier climate than those near Fort William, but does this mean that their peat is also drier? Interestingly there does not seem to be a geographical pattern to support this line of thinking when the range of cited bulk-density values for British blanket bogs is examined. The ECOSSE Report (2007), for example, notes something of a mixed message, with bulk density at Hafren, Wales, recorded as 0.08 g cm^{-3}, while the wetter Ullapool site has a bulk density of 0.12 g cm^{-3}. The site at Glensaugh, near Aberdeen, lies in a markedly dry, cold climate region more typical of boreal (Scandinavian) conditions, and the bulk density of 0.14 g cm^{-3} for the site may reflect this drier climate.

It is possible to construct an argument proposing that the site at Ullapool has a higher bulk density than the site in Wales because the high rainfall leads to almost continuous surface-water flow. This flow provides a little more oxygen and therefore encourages a little more decomposition at Ullapool, in turn leading to increased bulk densities at the wetter site. The Glensaugh site, on the other hand, may have the highest value for bulk density for quite the opposite reason – it is simply the driest site.

Whether this argument is valid or not would require further investigation, but it is clear that there is not a simple north-south divide between the un-patterned mires of the English and Welsh blanket mires and the highly patterned mires of the far northwest of Britain. It is interesting to note that the bulk density recorded from Hafren, in Wales, is very similar to the general bulk-density value obtained by Shotbolt *et al*. (1998) for the catotelm peat beneath, or at distance from, the main layers of disturbed peat affected by the forestry at Bad á Cheo in Caithness.

For bulk densities to be so similar in sites which are so geographically separated perhaps suggests that there is little power in the argument for a bio-geographical pattern in bulk density. The similarity of these bulk densities to values cited for Southern Hemisphere mires formed by completely different plant species lends weight to the argument that geography is not a great determinant of bulk density.

5.2.3.2 Spatial variation in catotelm bulk density

That there is substantial variation in the bulk density of peat is obvious when reviewing the range of figures. This variation appears to be far more closely linked to the condition of a site than to its geographical location. The relationship between bulk density and land-use impacts has already been highlighted in Section 4.3.3, earlier, while the direct link between bulk density and forestry drainage can be seen in the work by Shotbolt *et al*. (1993), as displayed earlier in Figure 12 and Figure 13.

Thus impacts such as burning, trampling, drainage and afforestation might be expected to produce marked increases in bulk density, as would conditions such as intensive erosion (which represents, in effect, a network of drainage gullies). In contrast, areas which are largely free from significant human impact and which are not subject to intensive erosion might be expected to display significantly lower bulk-density values.

Such contrasting areas may be found within close geographical proximity to each other in both the north of Scotland and the southern Pennines, further emphasising the earlier message from Section 4.3.4 of the present report that bulk density varies at the nanotope, microtope, mesotope and macrotope scale. It is thus proposed that two 'standard cubic metres of catotelm peat' are summarised here, representing reasonable extremes of catotelm condition. These two extremes consist, firstly, of a catotelm which is largely free from significant impact and thus has a low bulk density, while the second example is represented by a block of peat from an area of blanket mire which has undergone significant impacts or which lies in the midst of an intense erosion complex.

The values for these two sets of catotelm conditions are set out in Table 5 and the two catotelm blocks are illustrated in Figure 19.

It is very clear that in the case of the eroding/damaged peatland, the resulting increase in bulk density means that the cubic metre of peat contains substantially more carbon than the natural condition. In terms of carbon storage, it is therefore tempting to assume that the damaged or eroded condition is more advantageous than the natural state.

Such an assumption might be tempting, but it would be wrong. One clue as to why this is so can be found in the presence of air within the damaged/eroded block of peat. Not all such peat will be penetrated by air, but the relative abundance of macropores and 'pipes' within damaged, drained peatland as identified by Holden (2005a,b) provides a significant number of ways by which air can penetrate deeply into the catotelm on such damaged sites.

The effect of such things on the carbon store can only be gauged by looking at a number of further factors, including extent of natural versus damaged sites, peat thickness, and the processes of accumulation and decomposition. The next section of the present report begins this assessment by considering the relationship between the various cubic metre blocks of peat described above and the estimated extent and thickness of the peat resource.

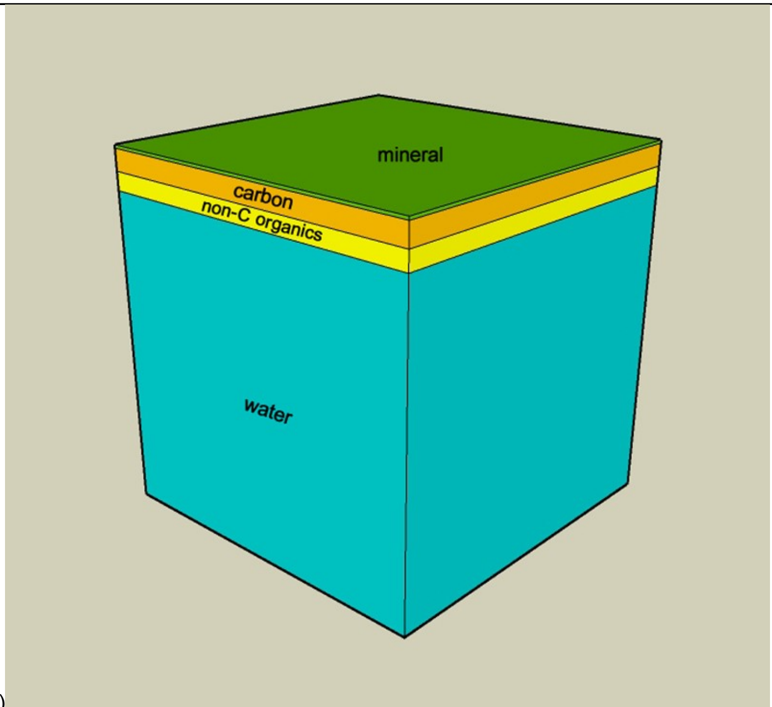
Before concluding this section, however, it is worth looking at the values obtained from one of the few published studies to have explicitly calculated such a figure – namely Cannell *et al.* (1993). Considering only deep peat from their 302 samples, they calculated that such peats contained 0.47 kg C m^{-2} per centimetre thickness, based on bulk-density values obtained for both 5-15 cm and 30-40 cm depth. This would mean that a 'Cannell cubic metre of peat' would contain 47 kg C.

Direct comparison between the 'Cannell cubic metre of peat' and the various peat blocks described above requires that those blocks divided into two layers be combined into a single block in order to produce a directly-comparable cubic metre block. Table 6 illustrates the effects of doing so, in comparison to the figures obtained from Cannell *et al*. (1993).

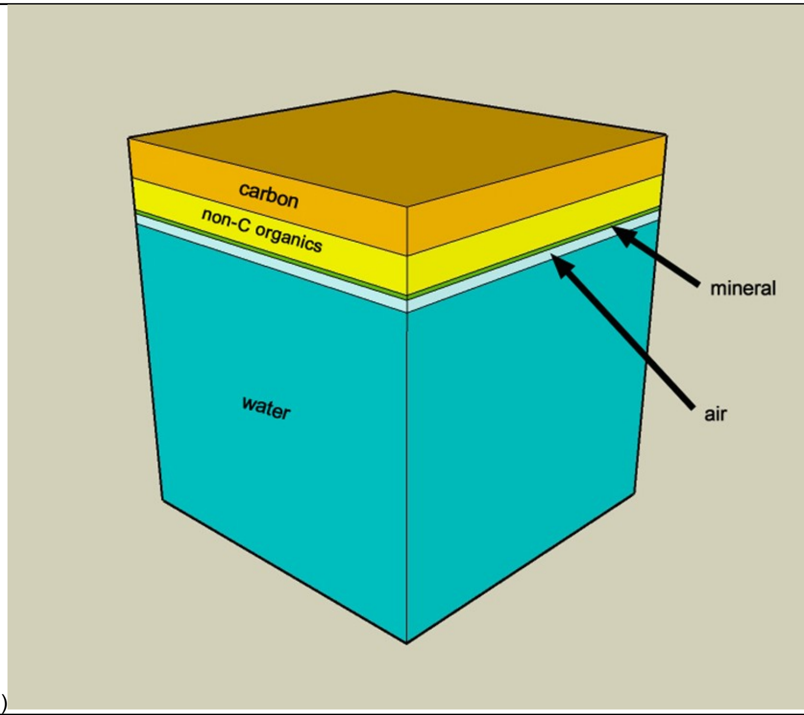
Table 5. Values of components for catotelm cubic metre of peat under two sets of conditions – natural, and eroding (or subject to human impact).

	Component	Value	Unit
Natural	Thickness	100	cm
	Water (100% saturated acrotelm)	212	kg
	No air penetration	0	%
	Dry bulk density	0.1	g cm^{-3}
	Dry weight	100	kg
	Mineral content (3% of dry weight)	3.0	kg
	Non-carbon organic matter	44.62	kg
	Carbon (54% of total organic matter)	**52.38**	kg
	Carbon per centimetre thickness	**0.52**	kg cm^{-1} thickness

Eroding	Thickness	100	cm
	Water (95% saturated acrotelm)	950	kg
	Air (approx. 3% penetration)	3	%
	Dry bulk density	0.2	g cm^{-3}
	Dry weight	200	kg
	Mineral content (4% of dry weight)	8	kg
	Non-carbon organic matter	88.32	kg
	Carbon (54% of total organic matter)	**103.68**	kg
	Carbon per centimetre thickness	**1.04**	kg cm^{-1} thickness



(a)



(b)

Figure 19. Quantities of material contained within the haplotelm and catotelm of a haplotelmic 'standard cubic metre of peat'
Quantities of the different materials which go to make up a 'standard cubic metre of catotelm peat' in two conditions of blanket bog. Condition (a) is a natural site with no erosion and no significant effects of human impact. Condition (b) represents a block of catotelm peat from an area of intense erosion, or an area subject to significant human impact. Indicated relative volumes are approximately equivalent to the quantities in kg of each component. Note the higher quantity of carbon in block (b), but also the presence of air; this air enables some aerobic decomposition to occur.

Table 6. Quantities of carbon contained within various cubic metres of peat
The calculated totals for carbon are based on the formula used by Cannell *et al.* (1993), but the values of the various parameters vary according to the figures provided in Table 3, Table 4 and Table 5 above. Totals for those tables which have two layers are generated by adding carbon totals for each layer together.

	Cannell *et al.* (1993)	**natural cubic metre**	**haplotelmic cubic metre**	**natural wet catotelm**	**damaged catotelm**
kg Carbon	47	57	61	52	104

The consistently higher values obtained in comparison to the 47 kg obtained for the 'Cannell cubic metre of peat' result from the use of a higher proportion of carbon to non-carbon (0.54 rather than 0.5) and generally a lower mineral content.

5.2.4 Gas in peat

There remains one component of the peat which has yet to be discussed. It is, if not controversial, then at least still subject to some debate, but the weight of evidence appears to be increasingly pointing to the presence of such material within the general mass of peat. Perhaps 'mass' is not the appropriate word to use here because the material under discussion is in fact a gas – namely methane.

As already indicated in the construction of the various cubic metres of peat, the standard Clymo (1992) model recognises a gas phase which penetrates, in effect, to the base of the acrotelm. This gas phase of course represents air penetrating into the catotelm during periods of lowered water table. By definition (depending on one's definition) no air penetrates the catotelm and thus the standard Clymo (1992) model is redrawn by Rydin and Jeglum (2006) to emphasise amongst other things that there is no gas phase below the water table.

At the same time, Clymo (1992) develops the ideas of catotelm decomposition, which must by definition involve at least some anaerobic decomposition to methane, while Rydin and Jeglum (2006) devote much attention to the question of methane evolution in the catotelm. Rather less is said about what form this methane might take when produced – is it in the form of large bubbles, micro-bubbles, or perhaps in some way dissolved in solution (though methane is only sparingly soluble)?

Brown (1989, 1995, 1997), Brown and Overend (1993), Brown *et al*. (1989) and Reynolds *et al*. (1992) have explored in some detail the process of methane production within the catotelm of peat bogs. Their findings suggest that while methane is indeed produced in some quantity within the catotelm, this forms part of a methanogen-methanotroph coupling in which methane is produced by one group of micro-organisms but this methane is then assimilated by another group of micro-organisms into the microbial biomass. The net result is that little or no methane is released from the catotelm. The gas instead remains within the peat as micro-bubbles which are digested by the methanotrophs.

Evidence for such a model is put forward by Brown and Overend (1993). They show that a column of peat which is maintained under constant flow in the laboratory displays a steady decline in hydraulic conductivity as the concentration of methane rises over a number of weeks. Performing the same experiment with a sterilised column of peat produces no such decline in conductivity.

Baird *et al*. (1997) also explore the influence of gas bubbles on the hydraulic conductivity of peat, and point out that because gas bubbles are compressible, their presence can help to explain some of the slightly odd features of peat hydrology. Baird and Gafney (1995) and Baird *et al*. (1997) observe that such micro-bubbles are increasingly thought to occur in significant quantity within both the acrotelm and catotelm. Although Clymo and Pearce (1995) present some evidence to suggest that methane bubbles are not present in British blanket mires, Baird *et al*. (1997) question whether the method used is capable of demonstrating this.

On the basis that micro-bubbles of methane are indeed a feature of catotelm peat, it is possible to make one final adjustment to the picture of the natural catotelm. Brown (1997) presents evidence for the degree to which water can be replaced by methane within a catotelm column, indicating that water saturations levels may fall as low as 65% from an initial state of 80%. This would suggest that methane bubbles might occupy 19% of the water volume. Although Baird *et al*. (1997) suggest that the temperatures used at least by Reynolds *et al*. (1992) in this work are not wholly realistic, they nevertheless agree that this kind of picture does appear to reflect what is observed in terms of low hydraulic conductivity and non-Darcian behaviour of the peat.

It is important to recognise that the addition of methane bubbles to the mixture does not affect the proportions of dry-weight materials because the methane is only displacing water within the matrix, not solid matter. Thus the final model of a natural cubic metre of peat can be seen in Figure 20, with a gas phase in both the acrotelm and the catotelm, but in the case of the former this gas is air, while in the latter the gas is methane.

The important thing to understand about this store of methane is that, according to Brown and Overend (1993) and Brown (1995), under natural conditions this methane store remains held in a stable matrix within the catotelm. While the catotelm may thus be shown to contain large quantities of methane, under natural conditions very little of this makes its way to the surface of the bog to be released into the atmosphere.

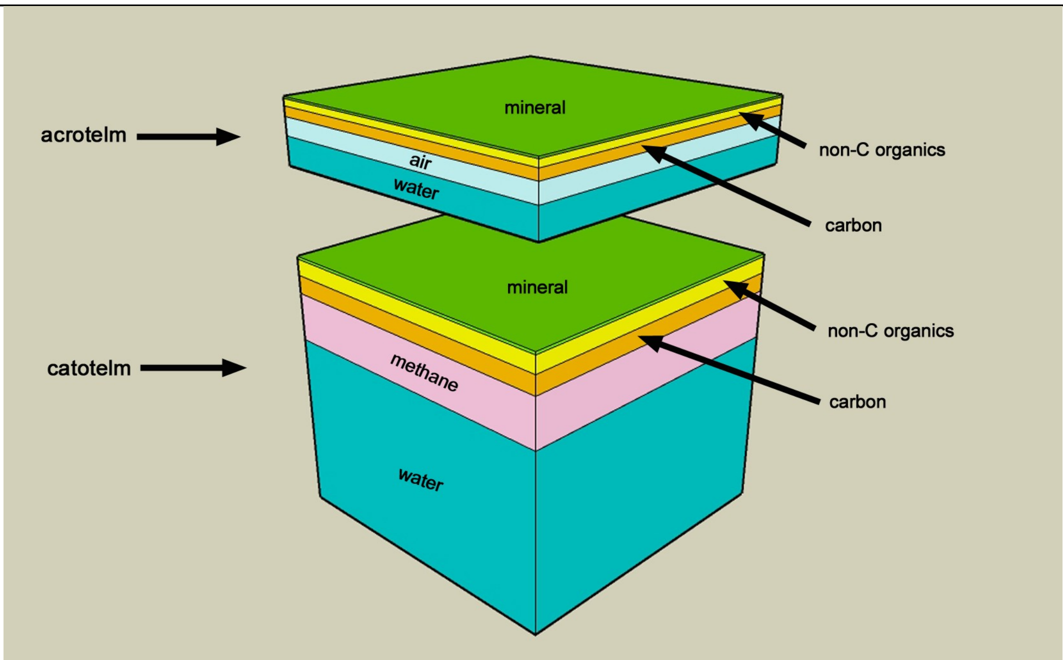


Figure 20. Quantities of material, including gas, contained within the acrotelm and catotelm of a 'natural cubic metre of peat'
Quantities of the different materials which go to make up a 'standard cubic metre of natural blanket mire. Indicated relative volumes (apart from gas) are approximately equivalent to the quantities in kg of each component. Note that both acrotelm and catotelm have a gas component, but in the acrotelm the gas is air, while in the catotelm the gas is methane.

6 Estimating the carbon stored within the peatlands of the UK

This section reviews various attempts to estimate the amount of carbon stored in UK peatlands; peat is an unusual carbon store in being continuous with few gaps; consequently even thin layers contain much carbon which is stored for almost geological timescales; the carbon stored in Sphagnum biomass may have been significantly under-estimated, and may equate to the carbon stored in conifer plantations on peat; initial estimates for carbon in UK peatlands gave 3,000 M t C; later estimates gave as much as 16,000 M t C in Scotland alone, but used atypical bulk density; other estimates based on more appropriate parameters give approximately 7,400 M t C for the UK; the UK carbon database records as standard only the uppermost 1 m for peat soils.

6.1 Peat - an unusual carbon store

There are three reasons why a peatbog is such a good carbon store compared to other habitats. Firstly, there are no (substantial) gaps in the peat. The peat mantle is a largely continuous layer of carbon-based material spread evenly over the landscape. Secondly, this mantle has a considerable thickness. Combining the continuous nature of the peat deposit with a significant thickness means that the total volume of this carbon-based matrix rapidly becomes very substantial indeed even though the amount of solid matter in any cubic centimetre may be comparatively low. Peatbogs are such rich carbon stores because they have both width and depth.

The third key characteristic of peatbogs, setting them apart from other habitats in terms of carbon storage, is that this thick, continuous mantle is maintained and even added to for periods far exceeding the steady-state storage capacity of most, if not all, other terrestrial ecosystems. An oak tree may live for 700 years and then take a further 700 years to decay completely (Kirby and Drake, 1993). The resulting timescale of almost 1,500 years for this complete cycle of growth-and-decay means that after 1,500 years there has been no net increase in above-ground biomass, no net increase in below-ground biomass, and only a relatively small amount of organic matter added to the soil store. Many bogs have been steadily adding to their long-term soil-carbon store for more than 3x this cycle-length and there is little reason to think that they will not continue adding to this store (albeit more slowly over time) for centuries if not millennia to come.

Clymo's (1992) conceptual model of bog growth suggests that even using models of growth in which bogs do reach a limit of accumulation, the oldest bogs today may still be only 2/3rds of the way towards their steady-state condition. Indeed some versions of his model have no such theoretical upper limit, although the accumulation rate does decline with time. For practical purposes it is sufficient to recognise that peat bogs are *very* long-term carbon stores on a scale not generally encountered within the terrestrial environment.

6.2 Published estimates of carbon storage

By bringing together information about the extent of the peat resource, the depth of the peat resource, and the carbon composition per unit volume of peat, it should be possible to begin assembling a picture of the total carbon store held in British or UK peat bogs. Several attempts to do just this have been undertaken since the early 1990s (prior to this, almost the sole interest in peatland carbon was its calorific value as a fuel).

6.2.1 Cannell, Dewar and Pyatt (1993)

Although it is a paper aimed primarily at exploring the question of whether conifer plantations on peat resulted in a net loss or gain of carbon, Cannell *et al*. (1993) also provide one of the earliest published attempts to provide figures for the total amount of carbon stored in British peatlands. The various measurements, parameters and formula used by Cannell *et al*. (1993) to calculate the amount of carbon stored in a unit volume of peat has already been described earlier in the present report.

Cannell *et al*. (1993) calculated that deep peats contained 0.47 kg C m^{-2} per centimetre thickness, while shallow peats (<45 cm) were calculated to contain 0.8 kg C m^{-2} per centimetre thickness. They estimated that deep peat covered more than 2.1 million hectares while shallow peats covered 3 million hectares. They also calculated that the average thickness of deep peat was 2.43 ±1.14 m.

Cannell *et al*. (1993) then multiplied their calculated carbon per unit volume by their estimated total surface area of peat bog in Britain and then by their calculated mean depth of peat. On this basis, they concluded that there may be 3000 Mt C (megatonnes; million tonnes) stored in British peatlands.

It is not, of course, as simple as that. Firstly, the margin of error given by Cannell *et al*. (1993) themselves for peat thickness is around 50% of the mean value used. In other words, peat depth varies substantially. There is also no way of determining whether the 302 samples (within FC holdings) used by Cannell *et al*. (1993) are typical of peat thickness and bulk density generally. Indeed there is an argument for suggesting that the very deepest peats may have been avoided during FC land purchasing. Both these points about the Cannell *et al*. (1993) synthesis are raised by Chapman *et al*. (2001) in their review of organic soil stocks in Scotland.

6.2.1.1 Units of carbon storage

Given that the main focus of material storage has until now been blocks of peat one cubic metre in size whereas carbon storage across whole countries involves very much larger units, a word about units might be useful here. As we have seen already, two fairly typical units for individual cubic metres of peat are kilogrammes of dry matter, or kilogrammes of carbon. It is important not to confuse the two, because one is roughly twice the mass of the other.

If the quantities relate to multiple cubic metres of peat, or involve areas of a hectare or more, the units may increase by 1,000 to become tonnes of carbon (t C). At the scale of a Scottish District, the figures are more likely to be megatonnes (millions of tonnes of carbon – Mt C). These are all fairly straightforward but it is important to note carefully which units are being used where. Values expressed in grammes per cubic centimetre may abruptly become megagrammes per cubic metre (Mg C m^{-3}, where a megagramme is a million grammes). A megagramme is more often expressed in the literature as a metric tonne of carbon: t C.

When dealing with a whole country, the values are most likely to be in millions of tonnes of carbon (Mt C), although the official SI unit is more accurately a Terragramme (Tg C). This is a somewhat cumbersome unit, and it is therefore common to find Mt C still being used in the scientific literature.

Mistakes can happen when units are being juggled. Cannell *et al*. (1993) conclude that Scottish deep peats contain 0.47 kg C m^{-2} per centimetre depth of peat. Cannell and Milne (1995) subsequently state that Cannell *et al*. (1993):

> *"...pointed out that deep ... peats in the British uplands contain about 4.7 g C ha^{-1} per centimetre depth."*

In fact Cannell *et al*. (1993) did not show this. Converting 0.47 kg C m^{-2} per centimetre depth to a weight of C per hectare produces 4.7 <u>tonnes</u> C ha^{-1}, not 4.7 <u>grammes</u> C ha^{-1} per centimetre depth. It seems that Cannell and Milne confused themselves by choosing to express the original statement in a different set of units. For these experienced workers to make such a slip merely emphasises the fact that very great care must be taken at all times when peat-carbon stores described by different authors are given at differing scales.

For large-area calculations, the present report will use tonnes and millions of tonnes of carbon (t C and Mt C).

6.2.2 Howard *et al.* (1995)

The next major review of carbon in the soils of Britain was undertaken by Howard *et al*. (1995). This analysis relied heavily upon the data held in the Soil Survey of England and Wales, and also the National Soils Database for Scotland. It was taken that the carbon content of each soil series was dependent upon the vegetation or ground cover, and this information was obtained from the ITE 1990 Countryside Survey and the ITE Land Classification.

The relatively limited number of actually-measured cores and depths for peat soils in both national soil-survey datasets has already been highlighted earlier in the present report, but as the National Soil Database did not hold data for peat bulk density, Howard *et al*. (1995) elected to use a standard bulk density of 0.35 g cm^{-3}, increasing to 0.5 g cm^{-3} at 5 m depth, for all Scottish peats, based on figures obtained by Burton and Hodgson (1987) for English lowland peats. The difficulties associated with selecting such a bulk density value have already been discussed in Section 4.3 and Section 5.1.3 of the present report, as has the advisability of assuming that bulk density increases with depth.

The consequences of using such values for bulk-density and for making fairly broad assumptions about peat depth were that Howard *et al*. (1995) estimated the total soil carbon in England and Wales to be 2,773 Mt C, while Scotland holds 19,000 Mt C. These figures are substantially larger than those given by Cannell *et al*. (1993) but it must be understood that the figures relate to *all* soils in Britain, not just peat. It is observed, however, that 75% of all soil carbon in Britain apparently resides in the peatlands of Scotland. This suggests that Howard *et al*. (1995) found 16,323 Mt C in Scottish peats alone.

6.2.3 Milne and Brown (1997)

The next major review of carbon in soils was undertaken by Milne and Brown (1997), but this review also attempted to provide the first comprehensive analysis of carbon stored in vegetation biomass.

Milne and Brown (1997) use much the same basis for their soil-carbon calculations as Howard *et al*. (1995), in the sense that they draw heavily on the data available from the Soil Survey of England and Wales, and on the National Soil Database for Scotland. The same fundamental limitations of small measured sample size thus apply to these data as when used by Howard *et al*. (1995), but Milne and Brown seek to refine the interpretation of mapped soil series in order to produce better estimates for soil carbon in each mapped square.

For Scottish peat, Milne and Brown (1997) also make use of the work by Cannell *et al.* (1993) to adjust the values of bulk density used by Howard *et al*. (1995), but they continue to assume that bulk density increases with depth and thus they employ a range of increasing bulk-density values with depth, just as Howard *et al*. (1995) had done. Whether this assumption is correct or not has already been discussed above.

The review of biomass was comprehensive in the sense that all forms of vegetation cover were considered. However, the main focus of attention was forest biomass, very much reflecting the perception of the day that forests represented the main terrestrial carbon sequestration systems.

Milne and Brown (1997) undertake a detailed analysis using Forestry Commission Census data to characterise the whole age-spectrum of coniferous and deciduous woodlands in Britain, and then combine these data with ITE Land Classes to produce values of biomass carbon for the total extent of each forest type. The values for non-forest types, on the other hand, were only calculated on the basis of broad estimates.

Assembling the two main stores of carbon - biomass and soils – Milne and Brown (1997) estimate that there is 113.8 Mt C in vegetation and 9,838 Mt C in soils within Great Britain. These figures come from all soils and vegetation, and it is not easy to separate out figures for peat in England and Wales. Milne and Brown (1997) do, however, provide figures for Scottish peats amounting to 4,523 Mt C (with an associated standard error of 2,287 Mt C). The total soil carbon for England and Wales is given as 2,890 Mt C, which means that, according to Milne and Brown (1997), the maximum possible total for peat soils in Britain would be 7,413 Mt C, though clearly a significant proportion of the English and Welsh soil carbon will not be associated with peat bog habitats.

The estimated biomass-carbon values for non-forest vegetation types used in these calculations are uniformly very low, ranging from 0–2 t C ha^{-1}, with only scrub vegetation rising as high as 10 t C ha^{-1}. In contrast, the calculated net carbon density (weighted by age and extent) for deciduous woodland ranged from 36-91 t C ha^{-1} and conifer plantations ranged from 4-35 t C ha^{-1}.

The biomass-carbon figures for forest carbon are arrived at through detailed analysis and calculation whereas the non-forest values are obtained by broad estimate only. In order to make an informed judgement about the balance of carbon storage in forest and peatland biomass, it is necessary to consider further the available evidence for carbon storage in the peatbog biomass.

6.2.3.1 *Sphagnum* and biomass carbon

Milne and Brown (1997) raise the important point that carbon is partitioned between biomass and soil, and emphasise that it is necessary to embrace both forms of carbon when undertaking a carbon inventory. The difficulty with such a seemingly simple proposition, however, is that there are surprisingly few published estimates for the biomass carbon of blanket mire landscapes.

Indeed for peatlands there is the additional complication that biomass is often measured simply as ‘aboveground biomass’, generally leaving the ground surface itself – *i.e.* the carpet of bryophytes which dominate the vegetation – to be excluded from such calculations. Thus Rydin and Jeglum (2006) provide a table of above-ground biomass in bogs and fens, and give above-ground values for ‘shrubs’, ‘herbs’ and ‘graminoids’, but then for ‘bryophytes’ they simply note ‘NA’ (data not available). Clymo and Hayward (1982) further muddy the waters by questioning the very applicability of the term ‘biomass’ to *Sphagnum*.

Milne and Brown (1997) provide their own estimates for the biomass-carbon stored in non-forest habitats. On this basis, the carbon content of bog biomass is given as 2 t C ha^{-1} whereas rough grassland and crops, for example, are each assigned a value of 1 t C ha^{-1}. It is not clear whether these estimates represent above-ground biomass alone. It is worth noting that these same values for biomass continue to be used in the latest official “Inventory and projections of UK emissions by sources, and removals by sinks, due to land use, land-use change and forestry” (Thompson, Mobbs and Milne, 2008).

A global review of carbon in live vegetation carried out by the Oak Ridge National Laboratory, USA (Olson *et al*., 1983), suggests instead that ‘non-wooded bog vegetation’ contains 20 t C ha^{-1}, a value 10x larger than that used by Milne and Brown (1997), although it is equally unclear whether this refers to above-ground biomass, above- and below-ground biomass, all biomass including carpet-forming mosses, or some other combination of features.

The explicit inclusion of biomass values by Milne and Brown (1997) for the various major habitats of the UK, highlights the fact that living plant tissue is an important part of an overall carbon budget. The nature of this biomass in terms of its appearance and structure may vary considerably between different habitats, but can be made more comparable by calculating the carbon density per unit area, as is done by Olson *et al.* (1983) and Milne and Brown (1997).

Immirzi *et al*. (1992) adopt a rather more visually-arresting approach to such comparisons. They condense the above-and below-ground biomass of various forest types by converting the biomass for each type into a uniform layer of wood (*i.e.* removing all the space between trunks and branches) and then calculating the different thicknesses of these wooden layers over an area of 1 hectare for each forest type. For riverine tropical forest they calculate the equivalent thickness of the forest biomass to be a layer of wood 20.1 cm thick (at a carbon density of 297-357 t C ha^{-1}), whereas for Douglas fir grown on a poor site the thickness would be only 7.1 cm (at a carbon density of 46-61 t C ha^{-1}).

The principles of this approach are also adopted by Cannell *et al*. (1993) when comparing the biomass carbon of a sitka spruce forest with an equivalent thickness of peat. They observe that a thickness of 36 cm of peat can be regarded as equivalent to the carbon content of a mature sitka spruce plantation at Yield Class 12. However, although Cannell *et al*. (1993) explore the carbon-biomass of the sitka plantation in some detail, they make no distinction in their calculation between peatbog biomass and peatbog soil.

It is not generally appreciated that the acrotelm layer in the ‘natural’ model of Figure 17 could reasonably be regarded as biomass. Clymo and Duckett (1986) demonstrate that re-growth of *Sphagnum* can occur from stems and stem fragments as much as 30 cm below the mire surface. Thus although much of the acrotelm might be classed as ‘dead plant litter’, there are clearly living parts of *Sphagnum* throughout this acrotelm thickness. Consequently the acrotelm could be thought of as living biomass much as a forest is made up of trees which consist of dead heartwood together with a relatively small amount of living bark and leaves.

Thus while Cannell *et al*. (1993) calculate a 36 cm thickness of peat to be equivalent to the carbon-biomass of a conifer plantation, a more direct comparison might instead involve distinguishing between the carbon contained within the living biomass of the acrotelm rather than simply the carbon stored within the catotelm peat. In this way, living biomass is compared with living biomass, rather than the living biomass of the forest being compared with the dead material of the peat soil.

Table 7 shows the amounts of carbon stored in different thicknesses of acrotelm on a *Sphagnum*-rich bog. It is presented as a worked example to make clear the derivation of carbon values for different thicknesses of acrotelm peat. The values used assume an average bulk density down the full thickness of the acrotelm to be 0.06 g cm^3, a mineral content of 3%, and a carbon content of 52% within what is largely a *Sphagnum* carpet but in which some vascular roots occur, as shown in Section 23, Appendix 3, Figure 63.

A bulk density of 0.06 g cm^3 integrated across the whole acrotelm means that the uppermost parts of the acrotelm have a bulk density of only 0.03 g cm^3, which is close to the lowest limits recorded for bulk densities in this layer. A denser sward of vascular plants than indicated in Section 23, Appendix 3, Figure 63 would increase this value for bulk density because the carbon from their root systems would be more densely packed than it is in *Sphagnum* plants.

Table 7. Quantities of carbon stored in differing thicknesses of acrotelm.
BD = bulk density, based on a BD of 0.03 g cm^3 rising to 0.1 g cm^3 at the base of the acrotelm. Values for % organic matter and % carbon content are based on the material being largely *Sphagnum* remains.

BD g cm^3	Thickness (cm)	Organic-matter %	Carbon %	g C per cm^3	g C m^2 per cm thickness	t C ha^{-1} per cm thickness
0.06	1	0.97	0.52	0.03	302.6	3.0
0.06	2	0.97	0.52	0.03	605.3	6.1
0.06	3	0.97	0.52	0.03	907.9	9.1
0.06	4	0.97	0.52	0.03	1210.6	12.1
0.06	5	0.97	0.52	0.03	1513.2	15.1
0.06	6	0.97	0.52	0.03	1815.8	18.2
0.06	7	0.97	0.52	0.03	2118.5	21.2
0.06	8	0.97	0.52	0.03	2421.1	24.2
0.06	9	0.97	0.52	0.03	2723.8	27.2
0.06	10	0.97	0.52	0.03	3026.4	30.3
0.06	11	0.97	0.52	0.03	3329.0	33.3
0.06	12	0.97	0.52	0.03	3631.7	36.3
0.06	13	0.97	0.52	0.03	3934.3	39.3
0.06	14	0.97	0.52	0.03	4237.0	42.4
0.06	15	0.97	0.52	0.03	4539.6	45.4
0.06	16	0.97	0.52	0.03	4842.2	48.4
0.06	17	0.97	0.52	0.03	5144.9	51.4
0.06	18	0.97	0.52	0.03	5447.5	54.5
0.06	19	0.97	0.52	0.03	5750.2	57.5
0.06	20	0.97	0.52	0.03	6052.8	60.5

From Table 7 it can be seen that an acrotelm depth of 15 cm gives a carbon total of 45.4 t C ha^{-1} for largely pure *Sphagnum*, while an acrotelm which is 20 cm deep contains 60.5 t C ha^{-1}. Given that Clymo and Duckett (1986) observed re-growth of *Sphagnum* stems from depths of more than 30 cm, there is a reasonable argument to make that the living biomass of a natural bog potentially extends to at least 20 cm depth.

There is also then the carbon stored in the above-ground biomass of the vascular-plant vegetation cover to consider. Such above-ground biomass figures for blanket mire vegetation in Britain are surprisingly rare, the main focus of attention for blanket mires generally having been productivity rather than standing biomass. Forrest and Smith (1975) give total above-ground biomass for only the vascular plants growing in various areas of blanket mire vegetation at Moor House, in the northern Pennines.

They record quantities of between 6–9.8 t ha^{-1}, but this is merely biomass weight, not carbon. Converting these figures to likely carbon values (multiplying by 0.4) produces vascular-plant carbon values for above-ground biomass of between 1.6 and 3.92 t C ha^{-1}, giving an average across the vegetation types of 2.76 t C ha^{-1}.

Garnett *et al*. (2001) assign values of above-ground vascular-plant biomass given by Forrest (1971), Forrest and Smith (1975) and Smith and Forrest (1978) to different blanket mire vegetation types which are then mapped within the Moor House study area. Allowing for the differing areas of the four blanket mire vegetation types mapped by Garnett *et al*. (2001), an average value of 3.18 t C ha^{-1} is obtained from their study for the above-ground vascular-plant cover of these blanket mire areas.

More recently, Ward *et al*. (2007) have calculated the biomass carbon in 'live shoots' of vegetation at Moor House, obtaining values of 1.57 t C ha^{-1} for un-burnt exclosures, while burnt and grazed plots were found to have 0.69 and 0.99 t C ha^{-1} respectively. However, the nature of the vegetation in the study plots at Moor House is an issue which is discussed in more detail under Section 18.4.1, Topic 7: (burning).

From this limited range of biomass estimates, it would seem that the biomass-carbon density of the above-ground vegetation in Moor House National Nature Reserve is in the region of 3 t C ha^{-1}. This could be added to the carbon content of a *Sphagnum* layer and thereby increase the total value for biomass carbon within the blanket bog system. An acrotelm biomass thickness of 15 cm, combined with the aboveground biomass, gives a possible total of 45.5 t C ha^{-1} + 3 t C ha^{-1} = 48.5 t C ha^{-1}.

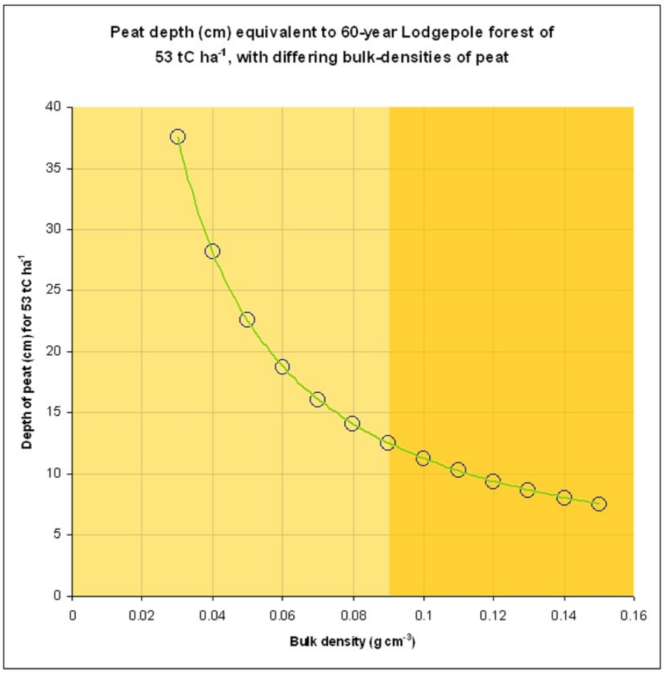
One unresolved part of this exercise is the contribution made by below-ground vascular-plant biomass. It is not clear whether the values given in the 'standard Clymo model' for acrotelm bulk density are essentially for a pure *Sphagnum* sward or whether these values include the contribution made by vascular-plant leaf-bases and roots. Smith and Forrest (1978) give a value of 0.2 t C ha^{-1} for the somewhat ill-defined below-ground biomass of blanket mire at Moor House, whereas Wallén (1992) gives explicit and detailed measurements of vascular-plant below-ground biomass for a raised bog in southern Sweden. Wallén (1992) gives values of 0.4 t C ha^{-1} for the uppermost 5 cm of the acrotelm, 7 t C ha^{-1} for a 15 cm depth of acrotelm, and 10 t C ha^{-1} to a depth of 25 cm.

It is not therefore clear whether these values might be *added* to the carbon values obtained from the 'standard Clymo model' plus the above-ground vascular-plant biomass, or whether these figures already form part of the earlier calculation. Further studies of above- and below-ground carbon-biomass in blanket mire systems might help to resolve this question.

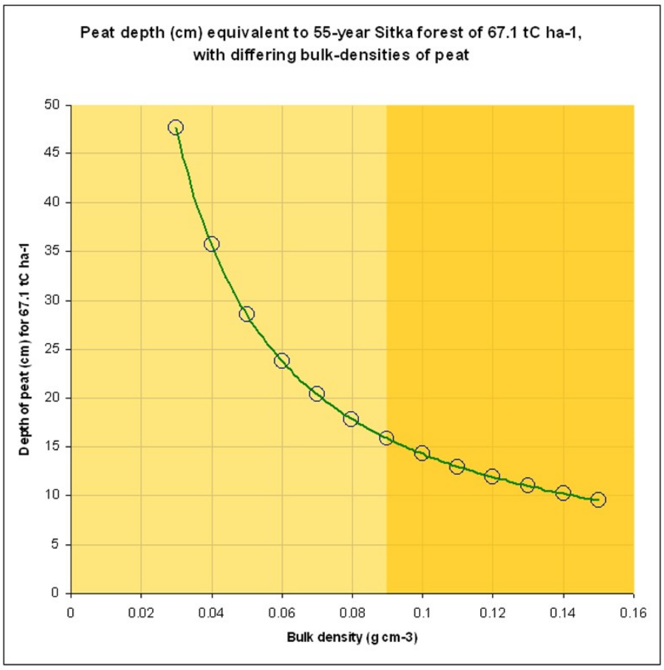
How do these figures compare with other habitats in the uplands – particularly upland conifer plantations, given that these are now often cited as providing significant carbon-sequestering capacity? Milne and Brown (1997), in their review of GB carbon in all types of vegetation and soils, underline this view by stating that woods and forests contain 80% of the carbon held in British vegetation. Taking the figure calculated earlier of 45.4 t C ha^{-1} in the biomass of a 15 cm thickness of acrotelm, plus 3 t C ha^{-1} for the above-ground vegetation, to give a total of 48.4 t C ha^{-1}, this equates to the estimated total (above- and below-ground) carbon density of 40-year sitka spruce or 60-year lodgepole pine plantations, according to forest figures provided by Milne and Brown (1997).

Considered from another perspective, Cannell and Milne (1995) provide carbon-density figures for stands of lodgepole pine and sitka spruce at stand maturity. Their figures suggest that a lodgepole pine plantation growing at Yield Class (YC) 8 would contain a carbon density of about 53 t C ha^{-1} at 62 years. For sitka spruce growing at YC12, the figures quoted by Cannell and Milne (1995) indicate that the total above- and below-ground carbon density at 58 years would be 67.1 t C ha^{-1}. It is thus possible to calculate the thickness of *Sphagnum* biomass that would be required to equal these carbon densities, using a range of bulk-density values for the acrotelm.

Figure 21 (a) and (b) illustrate this equivalence for both lodgepole pine and sitka spruce carbon-density values respectively. Figure 21 reveals that for lodgepole pine, a bulk-density value integrated across the acrotelm peat (0.06 g cm^{-3}) requires that the acrotelm be less than 20 cm thick to contain as much carbon per square metre as this mature lodgepole forest. For sitka spruce, the thickness is just under 25 cm. Figure 21 also make it clear that only relatively small increases in bulk density are needed to reduce significantly the acrotelm thickness required to match the forest carbon densities in both cases. The calculations of acrotelm thickness do not, of course, include the carbon content of the above-ground biomass of the blanket mire vegetation, which can add somewhere between 2-3 t C ha-1 to the total and thus reduce the required thickness of acrotelm biomass *pro rata*.



(a)



(b)

Figure 21 Depth of peat at particular bulk densities required to produce biomass carbon stocks equivalent to the biomass carbon of mature conifer plantations.
Using figures from Cannell and Milne (1995) for carbon content of lodgepole pine and sitka spruce plantations at felling age, it is possible to calculate the depth of acrotelm peat and bulk density required to produce a similar quantity of carbon within the acrotelm. Upper graph (a) indicates lodgepole-peat equivalents, while the lower graph (b) shows sitka-peat equivalents. Bulk densities typical of the acrotelm are shaded pale orange, while those more typical of the catotelm are shaded darker orange.

Furthermore Thompson *et al.* (2007) have identified that previous assumptions about the performance of conifer plantations have subsequently had to be revised downwards. The average Yield Class for Sitka is now thought to be YC11.2 rather than YC12, while figures for Lodgepole Pine may be nearer

Yield Class 7 rather than YC8. On this basis, the equivalent thicknesses of acrotelm biomass could be even smaller than those indicated here.

It would thus seem that there is a strong argument for a re-appraisal of figures used by both Milne and Brown (1997) and the current UK carbon-reporting process, in terms of the *biomass* contribution to carbon storage made by blanket bog systems. The distinction between peat and biomass is important because a peat store on its own cannot add further carbon – it can only lose carbon (if damaged). Biomass, on the other hand, has the potential to respond biologically to favourable conditions and add more carbon both to the biomass carbon store and the long-term catotelm carbon store.

6.2.4 Garnett *et al.* (2001)

A degree of confusion then creeps into the broader question of peat-carbon stores because, although Milne and Brown (1997) explicitly provide carbon values for blanket mire biomass and peat to 4 m thickness in their review of carbon in the vegetation and soils of Britain, it would seem that by 2001 the UK national C inventory was restricting itself only to consideration of the uppermost 100 cm of soil profiles. Garnett *et al*. (2001) undertake a detailed carbon inventory of Moor House National Nature Reserve, in the Pennines, and compare their results with those obtained from the UK inventory for the same area, explicitly commenting that the UK national carbon inventory estimates carbon to 100 cm only. Their final results, however, throw considerable doubt on values for carbon storage held by the UK national carbon inventory.

Garnett *et al.* (2001) note that 21 of the 22 x 1 km squares making up their study site are classed as Winter Hill soil series. Garnett *et al.* (2001) observe that the UK national carbon inventory assigns a value of 113,000 t C to any 1 km^2 belonging to the Winter Hill series, but they also note that the UK national inventory only contains data for the uppermost 100 cm of a soil. They therefore assume that the estimated 113,000 t C is based on the carbon held within this 100 cm thickness. Garnett *et al.* (2001) point out that to achieve this total within a 100 m thickness, the UK national inventory must use a bulk density of 0.23 g cm^{-3}, which is atypically high.

In contrast Garnett *et al*. (2001), using a much more reasonable bulk density of 0.1 g cm^{-3}, obtain by direct measurement an average measurement of 34,000 t C k for each Winter Hill km^2 within their study area, but they obtain this from only the uppermost 100 cm of their soil samples in order to maintain consistency with the UK national inventory. The differences are clearly considerable, the UK national carbon database giving values more than 3x the size of the total measured by Garnett *et al*. (2001). Garnett *et al*. (2001) explore a number of possible reasons for this discrepancy, and speculate that the cause may be the high bulk-density value of 0.23 g cm^{-3} used by the UK national carbon inventory.

An alternative explanation, however, is that the UK national carbon inventory does indeed *hold* data for soil series down to 100 cm depth, but bases its carbon calculations for any given soil series on the basis of an average peat thickness. For example, it uses an average thickness of 2.43 m for the Winter Hill soil series based on the study by Cannell *et al.* (1993). This would fit more comfortably with the process described in Milne and Brown (1997), and the calculated bulk density would indeed then be 0.1 g cm^{-3}, which everyone would agree as being a more reasonable value.

Continuing to assume, therefore, that the UK national carbon database in fact assigns a thickness of 2.43 m to the Winter Hill soil series and a bulk density of 0.1 g cm^{-3}, this would mean that a 1 km square of Winter Hill peatland would be predicted by the UK national carbon database to contain 114,000 t C. This is not identical to, but is close to, the value of 113,000 t C cited by Garnett *et al*. (2001). If this is indeed the explanation, then the more valid comparison should be between the 113,000 t C obtained from the UK national carbon database and the 85,000 t C contained within the total peat thickness recorded by Garnett *et al*. (2001) for the major bog vegetation type (Calluneto-Eriophoretum) at Moor House. Clearly these two values are much more comparable. The 28,000 t C km^{-1} difference (equivalent to 2.8 t C ha^{-1} or 280 g C m^{-2}) can probably be explained by areas of somewhat shallower peat within each 1 km square.Garnett *et al*. (2001) provide alternative explanations for these dramatic differences but these explanations involve values being held in the UK national carbon database which do not look entirely convincing. This exercise, and the measured figures obtained by Garnett et al. (2001), thus raise questions about the basis of the carbon values held in the UK national carbon database, the bulk-density values held in this database, and the soil thicknesses used for estimation of carbon stores. Furthermore, what Garnett *et al.* (2001) may also highlight are the problems which arise when constructing soil-carbon inventories which contain insufficient field-based measurements. The work also highlights the possibility of growing confusion about what the figures available from the UK national carbon inventory.

6.2.5 Bradley *et al.* (2005)

Bradley *et al*. (2005) present a summary of a soil-carbon and land-use database for the UK which has clearly been constructed around the key elements of the inventory described by Milne and Brown (1997). They, too, draw heavily on the information available from the Soil Survey of England and Wales and the National Soils Database for Scotland. As such, the analysis of Bradley *et al*. (2005) is just as constrained by the limited number of actually-measured records as have been all previous attempts to quantify soil carbon.

Specifically, Bradley *et al*. (2005) state that the organic carbon content of some 5,000 soil horizons were obtained from the Soil Survey of England and Wales Land Information System (LandIS), together with bulk-density values for 1,600 soil horizons. It should be noted that a soil core may contain several horizons, and thus the numbers given here represent a very much smaller number of actual soil cores. In practice this means that a little over 1,000 sites have measured data. The proportion of these sites relating to peat bog habitat is obviously going to be much smaller than this.

Bradley *et al.* (2005) supplement these data with information from a further 5,692 5 km x 5 km grid intersections across England and Wales. Organic carbon content for these locations was estimated using the LandIS database, however, so this process adds no further actually-measured figures to support the analysis. Bulk density was also predicted rather than measured.

For Scotland, Bradley *et al*. (2005) were able to use the organic carbon content of 26,000 horizons obtained from 9,400 profiles across all soil types described in the National Soils Database for Scotland. Available data for bulk density were, however, much more restricted, and Bradley *et al*. (2005) were obliged to use predictive pedotransfer functions to obtain values of bulk density for the full range of soil types and land-use conditions found in Scotland.

Data for Northern Ireland were based on a 5 km x 5 km grid of soil pits dug as part of the Northern Ireland Department of Agriculture and Rural Development soil survey. Almost all these pits were dug in pasture. Consequently it was necessary to rely on predictive transformations to estimate the properties of soils under arable, semi-natural or woodland conditions.

The most important feature of the database developed by Bradley *et al*. (2005) is that it now explicitly describes only the uppermost 1 m of the soil profile. Within this, it recognises a zone of 0-30 cm, and then a lower zone of 30-100 cm, but the database provides no information for any soil horizons which lie below 100 cm depth. This truncation of information is not explained, although it may reflect the fact that measurements of soil properties below 100 cm are quite rare, particularly in peat soils. Thus the assumptions used by Garnett *et al.* (2001) regarding the nature of the UK national carbon inventory have certainly been formalised by Bradley *et al*. (2005).

Given that some definitions of peat only begin at thicknesses of 1 m (*e.g*. the British Geological Survey definition of 'peat' on its Drift maps), the truncation of the database described by Bradley *et al*. (2005) at 1 m has substantial implications for the capacity of the database to measure the carbon store of peat soils. Indeed having enumerated the carbon stocks of the UK on the basis of the truncated database, Bradley *et al*. (2005) then explicitly note that previous studies (namely Milne and Brown, 1997) have demonstrated that an extra 3,248 Tg C lies below 100 cm in Scottish peats. Bradley *et al*. (2005) simply add this additional figure to the totals already obtained from their database to produce an overall total carbon content for peat and note that this raises the UK total to 7,814 Tg C.

No reference is made to the carbon stored in peat at depths greater than 100 cm in England, Wales or Northern Ireland, nor is there discussion about how such additional carbon storage might be incorporated into the database structure for England, Wales, Scotland or Northern Ireland. This would seem to be a significant gap in the story, especially as the database is described as:

> *"...putting the UK in a strong position to respond to the demands under the UNFCCC and the Kyoto Protocol for soil stock inventories."*

If Cannel *et al*. (1993) are correct in estimating that the average thickness of peat in Britain is 2.43 m, then the database assembled by Bradley *et al*. (2005) omits information for 60% of the UK carbon store. Compiling the figures for carbon stored only in the uppermost 100 cm of organic soils in England, Wales and Scotland, the data given by Bradley *et al*. (2005) indicate a total of 1,267 Mt C. This represents 42% of the 3,000 Mt C estimated by Cannell *et al.* (1993) for the total carbon store in British peatlands, and would thus seem entirely consistent with the idea that the database established by Bradley *et al.* (2005) can only catalogue 40% of the resource. However, simply relying on other sources to provide the carbon data for almost 60% of the store can hardly be regarded as one of the strengths of what is described as being the major UK national soil-carbon database. That the database is so constructed in order to make the data comparable with other soils databases across Europe does nothing to resolve this essential weakness in relation to deep-peat soils.

It is of some concern that the latest UK carbon reporting system (Thompson, 2008) explicitly states that figures for soil carbon have been restricted to a maximum depth of 1 m in order to harmonise soil data. If true, this would mean that the very substantial contribution made by the unusual thickness of peat bogs to the UK carbon soil-store has not been taken into account in UK reporting. This would not seem to be a desirable outcome from such an important national programme.

6.2.6 Dawson and Smith (2006)

Dawson and Smith (2006) summarise the inventory work of Milne and Brown (1997), but then highlight the mis-match between national inventory data for soil carbon and the measured data for a specific area obtained by Garnett *et al*. (2001), emphasising in particular the fact that the national carbon inventory values for carbon at Moor House were 3 times higher than those recorded by detailed field survey. Dawson and Smith (2006) present the database assembled by Bradley *et al*. (2005) as an information system capable of resolving such issues. They suggest that the revised database is more refined and appropriate to the task by showing that the Bradley *et al.* (2005) database results in a 24% reduction in the estimate of GB soil-carbon values compared to Milne and Brown (1997).

Such a reduction is not surprising, given that the Bradley *et al*. (2005) database holds no data for carbon stored in English and Welsh peat at depths of more than 100 cm. The surprise is that the reduction is not even greater. The reduction is not necessarily a sign that the Bradley *et al*. (2005) database is a more refined tool for use in the estimation of carbon stocks. If anything it appears to be a somewhat-blunted tool because it is unable to pick out the major characteristic of peat bogs – their extraordinary thickness – from other soil types because it explicitly excludes this unusual thickness. Dawson and Smith (2006) present the revised, reduced figures for total soil-carbon store in Britain from this truncated database. They add the lump-sum for Scottish peats deeper than 100 cm, obtained from Milne and Brown (1997) without comment about the implications for the database. The columns for peat greater than 100 cm in England and Wales are simply left blank.

6.2.7 The ECOSSE Report[3]

Chapman *et al*. (2001) review the many difficulties facing anyone attempting to estimate the carbon store held specifically within the UK's peatland soils, and summarise the key datasets created for Scotland up to 2001. They cite Robertson (1971), Birnie *et al*. (1991), Cannell *et al*. (1993) and Milne and Brown (1997). They also cite Howard *et al*. (1995), but conclude that the very high figure for the soil carbon store given by Howard *et al*. (1995) results from a rather crude approach to the mapping of peat, and also from the use of inappropriate values for the bulk density of Scottish blanket peat (0.35 g cm^{-3}).

The ECOSSE Report (2007) explicitly tackles what many reviews of soil carbon fail to address adequately, considering the effect of peat thickness on estimates of carbon storage. The ECOSSE Report (2007) consequently produces weighted average peat depths for different peatland soil units in Scotland and Wales. These weighted averages are derived from a wide range of sources, but it seems from the description given that the data relate largely to transects (rather than full 3-D surveys) or to single observations and have mostly been obtained from particular localities of interest rather than representing a generally distributed survey across the resource as a whole. In total, the ECOSSE Report (2007) obtains depth data of various types for 278 sites across Scotland. Given that the blanket mire resource potentially exceeds 4 million ha in Scotland (Milne and Brown, 1997), it is probably

[3] The ECOSSE 2 Report was released in late 2009 (Smith *et al*., 2009) but it was not possible, within existing time constraints, to provide a comprehensive review of ECOSSE 2 in the present report. It is important to note, however, that ECOSSE 2 usefully and informatively addresses a number of issues for Scottish peatlands raised in the present report. Inevitably, though, it also raises new questions.

reasonable to say that this scale of sampling is likely to provide only a general indication of peat depth. On this basis, the mapping resolution of 1:250,000 used by the ECOSSE Report (2007) for such data would seem appropriate. It does, however, mean that estimates of peat depth and consequent carbon storage are necessarily rather generalised.

The ECOSSE Report (2007) then calculates total carbon store for peat and organo-mineral soils in Scotland and Wales. To do so, the UK national carbon database (Bradley *et al*., 2005) is supplemented with additional data for peat and organo-mineral soils. The resulting totals (see Table 8 and Table 9) are not directly comparable between Wales and Scotland because different soil types are given for the two countries.

Furthermore any combined totals produced by other authors tend to combine England and Wales, rather than England and Scotland. Nonetheless, the total of 2,931 Mt C obtained for Scotland and Wales combined is almost equal to the 3,000 Mt C estimated for the whole 'peat and peaty' resource of Britain by Cannell *et al.* (1993). It would be reasonable to assume, on the basis of relative area calculations reviewed in Table 1 that England might contribute at least a further 600 Mt C, to bring the GB total to around 3,500 Mt C.

It is important to re-iterate that these figures are for peat and organo-mineral soils only. Neither do they use the cut-off thickness of 1 m which is one of the (less attractive) features of the UK national carbon inventory (Bradley *et al*., 2005). Consequently the figures cannot be compared directly with totals given by Milne and Brown (1997) and Bradley *et al*. (2005) because these latter authors list all soil carbon rather than focusing on the carbon associated only with peat and organo-mineral soils.

Table 8. Estimated stock of carbon (Mt C) held in Scottish peatlands.
Total carbon stock for peat and organo-mineral soils estimated by the ECOSSE Report (2007) for all thicknesses of peat in Scotland.

Scotland	**> 1 m thickness (Mt C)**	**1 m thickness or less (Mt C)**	**Total (Mt C)**
Peat	485	1,292	1,778
Organo-mineral	0	957	957
Totals	485	2,245	2,735

Table 9. Estimated stock of carbon (Mt C) held in Welsh peatlands.
Total carbon stock for peat and organo-mineral soils estimated by the ECOSSE Report (2007) for all thicknesses of peat in Wales.

Wales	**> 1 m thickness (Mt C)**	**1 m thickness or less (Mt C)**	**Total (Mt C)**
Peat	51.6	69.6	115.5
Organo-mineral	0	74.5	74.5
Totals	51.6	144.1	195.8

6.3 Total GB carbon store in peat – and associated uncertainties

6.3.1 GB peat-carbon store

While it might not be possible to make a direct comparison of figures between the ECOSSE Report (2007), Cannell *et al*. (1993) and Milne and Brown (1997), it is nevertheless interesting to note Milne and Brown's (1997) estimate that 'Scottish peat' contributes 4,523 Mt C to the total carbon store of Britain (unfortunately no such figure is given for England and Wales). This estimate by Milne and Brown (1997) is not truncated by the 1 m cut-off used later by Bradley *et al*. (2005), but it does use the rather suspect assumption that bulk density increases with depth.

Without knowing precisely how bulk density was assigned to which soil groups and at which depths, it is impossible to draw any conclusions about the comparability of the proposed 4, 523 Mt C for Scottish peat (Milne and Brown, 1997), the 2,931 Mt C for Scottish and Welsh peat (The ECOSSE Report, 2007) and the 3,000 Mt C estimated by Cannell *et al*. 1993) for British peats. All that can be said is that the estimate for Scottish peats by Milne and Brown (1997) is substantially higher than the value for Scottish peats obtained by the ECOSSE Report (2007) and the value for total GB peat estimated by Cannell *et al*. (1993).

It would seem reasonable to accept that the estimate by Cannell *et al*. (1993) of 3,000 Mt C in the peat soils of Britain can be taken as a minimum total. More recent values obtained as a result of field survey, field testing and re-analysis of existing datasets by the ECOSSE Report (2007) suggest that perhaps the value may be rather higher, approaching 3,500 Mt C. Even so, The ECOSSE Report (2007) is at pains to emphasise that the uncertainties remain large in all such estimates, and much more detailed work is required to improve such estimates. The very much larger sum obtained by Milne and Brown (1997) for stored carbon in Scottish peats illustrates how final totals can be dramatically altered by the use of differing assumptions about, for example, bulk density, or peat thickness. If a value of 121 Mt C for Northern Ireland is included (Cruickshank *et al*., 1998), the total minimum carbon store for UK peatlands becomes 3,121 Mt C.

6.3.2 Uncertainties and limitations to peat-carbon estimates

The major publications associated with national inventories for carbon auditing tend to provide far less detailed descriptions of the way in which uncertainties about peat thickness have been addressed than they do for the question of bulk density (*e.g*. Milne and Brown, 1997; Bradley *et al*., 2005). This would imply that bulk density is the more important parameter and therefore questions of uncertainty about bulk density are of paramount importance, but a closer examination of actually-measured values indicates that this is not the case.

Cannell *et al*. (1993) used a bulk density value of 0.1 g cm^{-3} to calculate their figure of carbon storage per unit depth in deep peat. They also noted that their average value for the thickness for deep blanket peat was 2.43 m +/-114 cm (*i.e.* the observed thickness for 'deep peat' ranged from 1.29 m to 3.57 m). It is possible then to use the formula presented by Cannell *et al*. (1993) to calculate the resulting range of carbon stores associated with this observed range of depths.

Using the mean value for peat thickness obtained by Cannell *et al*. (1993) of 2.43 m and the same formula as before, it is then possible to calculate the range of bulk densities required to produce this same calculated range of carbon stores. It emerges that this observed range of peat thickness is equivalent to peat bulk-density values which range from 0.01 to 0.328 g cm^{-3}. As we have already seen in the earlier exploration of peat bulk density, this range encompasses the majority of bulk density values cited for British blanket peats. In other words, within ranges that can be considered realistic, the scales of variability observed in both peat thickness *and* in bulk density can result in equivalent levels of variation in total carbon storage – neither parameter is pre-eminent.

This is not the impression obtained from the literature. Bulk density is acknowledged widely as a key factor in carbon storage, whereas peat thickness generally only features as an indicator of where bulk-density values have been obtained within the peat. It is not entirely clear why this imbalanced focus should have arisen. However, it has clearly influenced the way in which peatland carbon totals have been addressed in recent years – to the point where the UK national soil-carbon database standardises all figures for carbon storage to a maximum soil thickness of 1 m only, thereby explicitly excluding from the database content the very soil thicknesses typical of peatland habitats in the UK.

6.3.2.1 Thomlinson (2005) : an examplary study of uncertainty

In highlighting the uncertainties involved in the analysis of the soil-carbon store in Britain, and in explaining how these have been dealt with during the study, the ECOSSE Report (2007) probably does more than any other to make clear the limitations inherent in the peat-carbon inventory process so far. Yet the most lucidly open account of a national soil-carbon resource undertaken at the national level is that provided by Tomlinson (2005) for the Republic of Ireland. Here the uncertainties and limitations of available data are identified and described clearly, while the steps taken in response to these uncertainties and limitations are described in precise detail, enabling the reader to understand each step in the process.

The final soil-carbon estimate produced for the Republic of Ireland is not necessarily any more accurate than the estimates made by Milne and Brown (1997) or Bradley *et al*. (2005) for Britain of the UK. What is different is that the uncertainties and information-gaps are more transparent and thus more readily understood, while the assumptions and decisions made during the compilation of figures can be easily followed and, if necessary, adjusted appropriately by subsequent investigators.

Where there are gaps, uncertainties and limitations in the data which underpin major reviews, it is vitally important to have a clear statement of these deficiencies, for example making clear the number of actually-measured values which underpin selection of particular parameter values for general use. This degree of detail is unfortunately somewhat rare within large-scale estimates of carbon storage (whether in Britain or more widely). It is only by having such information that the reader can make an informed judgement about the degree of confidence which can be placed on any particular part of the inventory process. Moreover, this information is vital to an understanding of where further work is required in order to improve the quality of such estimates.

It is probably fair to say that such clarity of limitations does not yet exist for estimates of carbon held within British peats/soils. In particular, specific information would be valuable in relation to:

- number of thickness measurements obtained specifically from peat soils for England, Wales and Scotland;
- the number of soil cores taken in England, Wales and Scotland;
- the number of these profiles which have been taken from peat soils in each country;
- the number of bulk density values which have been obtained from these profiles in each country;
- within specifically peat soils, the number of bulk density measurements obtained from different depths within the peat profiles in each country;
- the number of bulk density measurements obtained from peat soils at depths below 1 m within each country;
- the number of measurements for ash/mineral content obtained from peat-soil profiles in each country, and from what depths these have been taken;
- the number of measurements for organic-matter content obtained from peat-soil profiles from each country, and from what depths these have been taken;
- the number of measurements for carbon content obtained from peat-soil profiles from each country, from what depths these have been taken, whether these have been calculated from measurements of organic-matter content or whether the carbon content was measured directly, and if the former, what assumptions were made about % carbon to % non-carbon organics.

It is also worth noting that any figure given for the total carbon content of a peatland area, whether it be for a district, country, or the UK, is comparatively meaningless unless accompanied by some definition of the associated parameters used. Thus a total of 16,000 Mt C for Scotland is only meaningful if associated with the area from which this figure has been derived, together with an explanation of the basis for this area, and a description of assumptions made about peat thickness and bulk density. It is not unusual to see figures cited without any of this essential information. Consequently it is difficult to assess the nature of any such figure in relation to other cited values of total carbon storage.

That said, all this discussion about carbon stores alone suggests an oddly static picture. Carbon stores in peat are indeed of over-riding concern because they are recognised as being special within the context of carbon cycling and global warming (while yet being largely overlooked by climate-change convention mechanisms). However, this carbon store exists only because an unusual imbalance exists in peatlands between the process of growth and decay, leading to accumulation of carbon into a long-term store. A clear understanding of the conditions and processes which govern the *balance* between growth and decay thus provides the key to understanding how best to maintain this carbon store and enable it to continue accumulating carbon. The next section of the present report consequently focuses on these dynamic processes.

7 Peat accumulation and decomposition

This is a long section which looks at the dynamics of peat accumulation and decay, and pathways of carbon accumulation and loss; acrotelm productivity may be very high but this must not be mistaken for peat accumulation – accumulation only occurs when material is passed from the acrotelm to the catotelm and the quantity is not exceeded by losses in the catotelm as a whole; residence-time in the acrotelm may be 80-100 years, but evidence is scanty; somewhere between 0.5-1 mm seems the most likely thickness passed to the catotelm annually in a living bog; Sphagnum is a particularly difficult material to decompose, and makes itself even more so by exuding compounds which immobilise microbial decomposers; vascular plants decompose much more readily, but not so readily when bathed in Sphagnum exudates; microtopography has a major influence on accumulation and decay, with hummocks accumulating more peat than hollows; though hollows are more productive, they decay faster; measurement of peat accumulation is easy over long time periods because the peat thickness can be measured and dated (LORCA/LARCA); measurements of current rates of accumulation (RERCA/ARCA) are very difficult, partly because the boundary between acrotelm and catotelm is indistinct, partly because of unknown residence-times in the acrotelm, and partly because the acrotelm may change over time in response to damage or climate shifts; a molecule of methane is now calculated to be 25x more powerful as a greenhouse gas than a molecule of CO_2, but this is over a 100-year period – over a 500-year period it is only approximately 9x more powerful (greenhouse warming potential GWP); estimated values of CO_2 uptake by blanket bogs vary from 30-70 g C m^{-2} yr^{-1}; methane-release from bogs is generally much lower, but varies between different elements of the microtopography, with hummocks emitting least and hollows emitting most; a Sphagnum carpet appears to limit CH_4 release, whereas some vascular plants in hollows and pools transport large quantities of CH_4 direct to the atmosphere; DOC release is increasing, but opinions are divided about why; the young age of carbon in DOC points to the possibility that it comes from freshly-decomposing vascular-plant cover rather than losses from the long-term carbon store.

This section of the present report focuses on the key processes by which peat maintains and possibly adds to its long-term carbon store while at the same time losing carbon in various forms through decomposition processes. Over the last 20 years or so, a great many studies have generated large numbers of measurements within this general field of study. It would be logical at this point to ask 'What values were obtained from these measurements?'. Such a question, however, is of considerably less importance than, 'What do these values represent?' While there are acknowledged and substantial questions about the extent of peat, its depth and its carbon density, when the focus instead turns to questions of carbon exchange between peat and its surrounding 'spheres' (atmosphere, hydrosphere, biosphere, geosphere) in a nutshell, there are many measurements, but surprisingly few certainties.

Accumulation occurs in the acrotelm when living vegetation adds biomass to the acrotelm layer. The controlling factors here are therefore those which influence conversion of gases, solutes and light into living structural components through the process of photosynthesis. The material so created then faces one of two fates – preservation, or decay. The majority of material is lost through decay. If material is instead to be preserved it cannot remain in the acrotelm because eventually oxidative processes will ensure its decay. Consequently preservation relies on material being transferred downwards into the anaerobic conditions of the catotelm.

The process of material accumulation into the catotelm is very different from accumulation in the acrotelm. Catotelm accumulation is entirely dependent upon a supply of already-dead plant material from the thin acrotelm layer above, because the catotelm is unable to produce material for itself. The primary part played by the catotelm in the process of accumulation is thus in the *preservation* of any material received from the acrotelm, because if the material is not preserved to any significant degree then there will be no accumulation. However, total preservation is not possible because even in the absence of oxygen a degree of anaerobic decomposition can occur. Consequently it is more realistic to talk of minimising the rate of decomposition in accumulating catotelm material, rather than simply preserving it.

Thus the acrotelm is the zone of primary accumulation, but not all of this material survives to the bottom of the acrotelm. A significant proportion succumbs to decomposition processes before it can pass downwards into the catotelm. Meanwhile the catotelm gains peat from the acrotelm, but is also losing material through slow anaerobic decomposition. It is the balance of this accumulation and loss that determines the carbon dynamics for any particular peatland.

In a peat bog which can be described as 'active' under the terms of the EU Habitats Directive, the surface layer includes living vegetation which is recognised as normally being associated with peat-forming conditions. The wording of the definition is necessarily ambiguous because it is actually very difficult to prove conclusively whether, at any specific instant in time, peat is actually forming. The pragmatic view therefore states that the presence of a significant amount of vegetation which typically forms peat is sufficient to qualify an areas as being 'active' bog.

The presence of such vegetation allows one to conclude that the peatland area is either currently accumulating peat or at least has the capability to do so even if it is not doing so at present. It is important to note that the definition is concerned only with the potential for peat formation. It does not concern itself with the question of whether the bog as a whole is gaining or losing carbon. Such mass-balance calculations play no part in the description and identification of 'active bog'.

7.1 Accumulation/decomposition in the acrotelm

The rate of organic-matter accumulation in the acrotelm is entirely dependent upon the vigour of peat-forming species within the living layer of vegetation. However, vigorous growth is no guarantee of vigorous peat accumulation. If the plant material produced is comparatively rich in nitrogen and phosphorus while at the same time being adequately supplied with oxygen, decomposition may keep pace with the rate of growth, leaving no organic matter to be transferred into the long-term store of the catotelm. Then again, on a bog surface where the acrotelm has been significantly altered by fire, or by trampling, there may be little living vegetation capable of providing fresh organic matter to the acrotelm in the first place.

Rates of plant growth (productivity) within the acrotelm have, over the years, been extensively reported within the scientific literature, at least in terms of overall productivity for the layer of living vegetation. Thus Forrest (1971) gives values for net annual production of cotton-grass/heather Pennine blanket bog as 635 g m^{-2} yr^{-1}, while Forrest and Smith (1975) give a value of 745 g m^{-2} yr^{-1} for similar vegetation on the same site, but this time including the productivity of *Sphagnum*. These figures compare quite favourably with temperate oak forest (900 g m^{-2} yr^{-1}) though rather less so with tropical rainforest (3,250 g m^{-2} yr^{-1}), the difference being, of course, that virtually 100% of the rainforest productivity will be recycled with little or no net gain in soil carbon, whereas a proportion of the Pennine productivity is almost certain to be added to the peat-carbon store.

More recently, Rydin and Jeglum (2006) have provided a synthesis for productivity of above-ground biomass in bogs, separating out productivity in the shrub layer from that of the herb/grass layer and the moss (bryophyte) layer. Their figures can be seen in Table 10.

Table 10. Productivity of differing above-ground layers in the vegetation of bogs (Rydin and Jeglum, 2006).

Component	Productivity (g m^{-2} yr^{-1})	
	Average	Range
Shrubs	180	43–338
Herbs and graminoids	16	3–34
Bryophytes	188	17-380

Whilst the above-ground biomass is relatively easy to see and quantify, it must be recognised that there is also a substantial component of productivity which is invisible, hidden below ground. This consists of roots, rhizomes, leaf-bases of sedges and grasses and scattered seeds. Charman (2002) highlights the fact that in a sedge fen, something approaching 90% of the living biomass may exist below ground, predominantly in the form of fine roots less than 1.5 mm in diameter. He goes on to summarise a range of productivity data for wetland types. For "Peat bog and wet tundra", he gives the following above-ground and below-ground ranges for productivity:

- above-ground : 42 – 1,118 (g m^{-2} yr^{-1})
- below-ground : 70 – 1,461 (g m^{-2} yr^{-1})

However, in the case of a peat bog the ground surface itself is often created by a bryophyte carpet. Thus whilst it can be seen from the data presented in Table 10 that shrubs, herbs and graminoids contribute significantly to above-ground productivity, the ground surface itself, in the form of the moss layer, may provide more than 50% of total annual productivity.

7.1.1 *Sphagnum* and acrotelm productivity

The acknowledged, principal peat-forming plant in Britain is the *Sphagnum* bog moss, but measuring productivity of this layer is not easy. Clymo (1970) suggests that it is neither possible nor meaningful to identify 'standing crop' in relation to a *Sphagnum* carpet because it is impossible to say where the living plant ends and the dead plant litter begins. He also identifies the difficulties involved in obtaining reliable values for *Sphagnum* growth without causing significant damage to the plant (and thus affecting sequential measurements).

Rydin and Jeglum (2006) observe that, in the 35 years or so since Clymo's (1970) review of methods, the most commonly-used technique for measuring the productivity of *Sphagnum* has become the 'cranked wire method', whereby a bent (cranked) stainless steel rod (wire) is inserted into the *Sphagnum* carpet and twisted to lock the bent end within the body of the carpet. Subsequent growth of the *Sphagnum* carpet is then measured against the vertical portion of the rod. They do emphasise, however, that considerable variation is to be expected in the data obtained and thus a substantial number of such samples is necessary "even in seemingly homogeneous moss carpet".

So far, the picture presented here fits fairly well with what we have been calling the 'standard Clymo (1992) model' of the acrotelm and catotelm, where plant material – assumed to be mainly *Sphagnum* - is created in the acrotelm and then undergoes a degree of decomposition until whatever is left passes down into the catotelm after a period of some 80 years. Detailed investigations into the spatial heterogeneity and chronology of the acrotelm, however, reveal a rather more complex picture.

Ohlson and Økland (1998) have looked at the age of peat at different depths within the acrotelm by examining the peat associated with stunted Scots pines (*Pinus sylvestris*) growing in a scattered pattern across an 800 m^{-2} area of a Norwegian raised bog. The trees were dated using tree-ring analysis, and thus the peat immediately beneath the root system (strictly, the root collar) could be dated with reasonable confidence. By way of background, Ohlson and Økland (1998) firstly highlight the fact that peat at 40 cm depth in boreal mires has already been shown to vary from 20 to 110 years old. For their particular site, they then demonstrate that the peat at 20 cm depth in different locations varies in age from anything between 7 years and 173 years old. The majority of these dated samples were taken from low or high hummocks. The average vertical growth rate in these hummocks during the last 10 years varied from 3 to 28 mm yr^{-1}, with the fastest rate being in a *Sphagnum* hummock which had increased in height by 23 <u>cm</u> in the last 15 years.

Such vigorous vertical growth was accompanied by dry-matter accumulation rates of between 100–900 g m^{-2} yr^{-1} over the most recent 10-year period (the uppermost value coming from the fast-growing *Sphagnum* hummock referred to above). A number of hummocks displayed an ability to sustain rates of more than 200 g carbon m^{-2} yr^{-1} over periods of up to 50 years. This equates to 2 t C ha^{-1} yr^{-1}. According to values presented by Cannell and Milne (1995) such a figure matches the net annual carbon flux over the first rotation for Norway spruce in central Europe, and for poplar plantations on agricultural land. The timescales for such *Sphagnum* growth rates are close to those of forest rotation times.

Ohlson and Økland (1998) note that the balance between carbon and nitrogen within the peat is not related to age if the peat is less than 40 years old. If significant decomposition had occurred during this period, then some form of age:C/N relationship would be observed. The fact that such a relationship was not found suggests that little decomposition of the *Sphagnum* material takes place until the material is at least 40 years old. Perhaps this time-interval represents the upper and central part of Clymo's (1992) 'standard' acrotelm model, where much of the original plant structure is clearly retained. Below this (and by association, after 40 years), decomposition becomes an increasingly significant factor until whatever remains finally passes to the catotelm.

When exactly this last step may occur seems to vary markedly from one small-scale location to the next. If, as they observe in places, peat at 7 cm depth is only 7 years old, this would suggest that there is a considerable acrotelm-depth beneath this point and that the peat is accumulating rapidly. In contrast, a location where the peat is already 173 years old at 20 cm depth suggests two alternative scenarios:

- firstly, that the acrotelm is comparatively thin at this location, and that material at 20 cm probably therefore consists of plant remains which passed into the catotelm some considerable time ago; or
- that some parts of the acrotelm simply produce *and* process material more slowly.

It is impossible to be sure that near-surface peat with an age of 173 years is catotelm peat unless the boundary between the acrotelm and the catotelm can be identified with some certainty. Unfortunately we have already earlier established that bulk density is not a reliable indicator of this boundary. While production of a residence curve for the bog water table would provide a valuable picture of the water-table behaviour and the probable position of the acrotelm-catotelm boundary, Ohlson and Økland (1998) take water table measurements across the site on only three occasions over the period of one month. It is thus not possible to produce any form of residence curve for their study area even for those locations which they sampled.

Consequently it is impossible to be sure what the results of Ohlson and Økland (1998) are telling us in terms of acrotelm dynamics. For example, does the acrotelm vary substantially in thickness from one part of the microtopography to the next? If so, does material take longer to pass through a thick acrotelm than a thin acrotelm? In other words, does material pass slowly through a thin acrotelm and more rapidly through a thick acrotelm, thus the age of material as it passes into the catotelm is the same whatever the thickness of acrotelm it has passed through?

It would thus be of the very greatest value to know precisely where the acrotelm-catotelm transition lay in relation to the kinds of observations provided by Ohlson and Økland (1998). What is a typical residence-time for the material of the acrotelm? It is important to know this because plant matter held in the acrotelm and capable of being regarded as biomass over periods of 100+ years becomes significant in terms of IPCC greenhouse gas balance and climate-change models because these use 100-year time periods as the default.

The length of time spent in the acrotelm is also a potentially-important question because, as Clymo and Hayward (1982) observe, the longer any material remains in the acrotelm, the longer it will be subject to oxidative decomposition. The more such material decomposes, the less carbon there is available for transfer to the catotelm. Only when material has passed into the environment of the catotelm is rapid oxidative decomposition no longer a threat.

Malmer and Wallén (1999) cite acrotelm residence-times of 80-100 years for three boreo-nemoral bogs in southern Sweden (*i.e.* bogs from a climate regime somewhat more Scandinavian than British blanket mire), though they also note that at various times in the past, acrotelm residence-times have been as brief as 30 years.

Ohlson and Økland (1998) note that “over longer intervals” [than 15 years] the rate of ‘vertical accumulation’ tends to decline towards a limit of only 1 mm per year (it is not clear whether this in fact refers to the acrotelm or catotelm). Such a figure offers a value more often quoted in relation to rates of overall mire growth. It would therefore seem that any vigorous acrotelm growth may indeed ultimately be subject to a considerable degree of decomposition through oxidative attack.

Indeed Malmer and Wallén (1999) observed in their boreo-nemoral bogs that acrotelm residence-times in hummocks and hollows were roughly similar, as were decay rates, also that it was the higher productivity of the hummocks which gave rise to the majority of accumulated peat, but that the longer acrotelm residence-times which currently prevail mean that the rate of catotelm peat accumulation is now lower than in the past.

Timescales of 80-100 years for such durations of oxidative attack in other habitats would normally mean that virtually all material had been lost, leaving none to pass to the catotelm. The unusual thing about bogs is obviously that some material does survive and *is* passed to the catotelm despite extended periods of oxidative attack. Possible reasons for such resistance to decay can be found in the particular nature of *Sphagnum*.

7.1.2 *Sphagnum* and decay

How does *Sphagnum*, such a relatively delicate plant compared to many of the vascular plants typical of bog vegetation, resist decay to such an extent that it is capable of accumulating peat at all? Is it something to do with micro-environmental conditions, or is *Sphagnum* itself inherently resistant to decay? The answer is: a little bit of both.

A thin surface film of water coats the *Sphagnum* plant when it lies above the bog water-table in a hummock (and thus when it is subject to the action of oxygen-reliant bacteria). This film is much easier to acidify than the bog water-table as a whole. Such high levels of acidity in this thin film, caused by active release of hydrogen ions into the film from the *Sphagnum*, give rise to conditions which are more extreme than many decomposer bacteria can tolerate (Glime, 2007). As a first line of defence, *Sphagnum* thus inhibits decomposition by manipulating the pH of its micro-environment, thereby partially paralysing the micro-organisms responsible for the breakdown of plant material.

Secondly, Glime (2007) highlights the fact that the chemical composition of *Sphagnum*, particularly in the form of phenols held within the cell walls, appears actively to inhibit bacterial action by blocking nitrogen uptake pathways in the micro-organisms. This anti-bacterial action, particularly by a pectin-like substance known as sphagnan, has been clearly demonstrated by Børsheim, Christensen and Painter (2001). Raw fish embedded in *Sphagnum* moss can be preserved almost indefinitely – a practice known to have been used on Viking voyages some 1,200 years ago.

Thus the *Sphagnum* litter actively resists breakdown by manipulating the chemical environment and by chemically shielding itself. This goes some way towards explaining why it decomposes significantly more slowly than the leaves of other plant species in other habitats - typically 10-20% mass loss per year for *Sphagnum* compared with 40-80% per year for vegetation in other habitats (van Breemen, 1995).

This specific type of chemical resistance to decay is clearly not essential for peat formation otherwise other types of peat deposit would not be possible. The action of waterlogging, acidity, and relative stagnation alone are themselves sufficient to permit a range of plant species to form peat of various types, but it seems that *Sphagnum* takes this a few stages further by making itself chemically unpalatable and exuding chemicals which, while not killing the decomposer bacteria, do nevertheless immobilise them rather effectively.

Consequently peat derived from, or at least rich in, *Sphagnum* fragments tends to be significantly less decomposed (humified) than peat resulting from most other plant materials. Such material will also tend to continue to resist decay for longer than the majority of other remaining plant materials. As such, *Sphagnum* peat contains a carbon store which may be retained for longer under a wider range of conditions than for many peat deposits. The remains of peat-forming species such as cotton grass or heather will be preserved as long as they are bathed in the anti-bacterial solution derived from associated *Sphagnum* plants, or as long as the water table remains sufficiently high to create anaerobic conditions. If the water table falls and exposes such material to oxygen, these fragments of higher plants appear to possess little defence from oxidative microbial decomposition and thus tend to undergo relatively rapid decomposition – unlike *Sphagnum* which is additionally protected by its chemical defences.

Wallén (1992) demonstrates the dramatically different fates of *Sphagnum*, heather (*Calluna vulgaris*) and cotton grass (*Eriophorum spp.*) material within the acrotelm. Heather and cotton grass productivity exceed that of *Sphagnum* more than ten-fold, with almost 80% of this extra material occurring below ground in the acrotelm itself, total annual biomass production amounting to 800 g m^{-2}. However, by the time all of this reaches the transition into the catotelm, only 30 g m^{-2} of this biomass remains and is transferred annually, and 99.5% of this is *Sphagnum*. A mere 1.5 g m^{-2} of the 720 g m^{-2} of biomass produced annually by the vascular plants is transferred into the catotelm, specifically in the form of cotton-grass roots and leaf sheaths (leaf bases).

Wallén (1992) observes that the enormous breakdown of vascular plant material is of great significance for the peat matrix because it adds considerable quantities of amorphous non-structural organic matter to the matrix. Such matter contributes significantly to the hydrological properties of the peat, particularly in terms of its low hydraulic conductivity. Such amorphous, vascular plant-derived material is almost certainly also a major factor influencing availability of dissolved and particulate organic matter from the peat. Clymo and Hayward (1982) do, however, show that the roots of hare's-tail cotton grass (*Eriophorum vaginatum*) are also highly resistant to decay, a fact widely attested to by layers in peat bogs which consist predominantly of cotton-grass fragments (*e.g.* Godwin, 1975; Barber, 1981; Charman, 1992).

It can generally thus be assumed that a peat bog in which the dominant vegetation cover is *Sphagnum* will release fewer decomposition products for a given level of water-table fluctuation than a peat bog in which the vegetation of the acrotelm (or haplotelm) layer is dominated by vascular-plant species such as cotton grasses (*Eriophorum*), deer grass (*Trichophorum cespitosum*) or heather (*Calluna vulgaris*). Indeed this would be the likely result even without the presence of sphagnans and other bacterial inhibitors, because *Sphagnum*-rich peat also contains a high proportion of pore space consisting largely of undecomposed hyaline cells containing nothing but water. Consequently the bulk density of stored material in *Sphagnum*-rich peat may often be less than that found in peat formed predominantly from, for example, cotton grass or heather.

The process of peat formation, as emphasised at the start of this section, is a result of three processes – accumulation in the acrotelm, decay in the acrotelm, and final transfer of the residue into the catotelm. Not all *Sphagnum* growth is equally productive nor do all *Sphagnum* species resist decay so effectively. Ohlson and Økland (1998) observe that accumulation of plant matter into the acrotelm displays a strong degree of spatial variation over quite small distances, and is closely linked to microtopography. Microtopes, nanotopes and vegetation are thus tightly bound to the net balance of peat accumulation.

7.1.3 Microtopography, accumulation and decay

Rydin and Jeglum (2006) highlight three particular characteristics of *Sphagnum* productivity:

- wetter parts of the microtopography (‘hummock-hollow pattern’) are more productive than the drier hummocks;
- productivity decreases with increasing latitude northwards;
- productivity increases with increasing oceanicity (*i.e.* increases westwards in Europe).

The latter two points are of only limited significance within the strictly British context, whereas the first point highlights a major universal interaction between microtopography and *Sphagnum* productivity. Clymo and Hayward (1982) investigated this effect by placing a range of *Sphagnum* species in three elements of the microtopography – hummock, lawn and pool. In almost every case, species showed highest productivity in the pools and lowest in the hummocks, even for species normally dominant within, and largely restricted to, the hummock element of the microtopography. Thus *Sphagnum capillifolium*, a hummock-forming species, grew better than any other species within the hummock environment, but its *highest* level of productivity was found in the pool environment.

This response is further emphasised by combining results obtained by Clymo (1970), as cited in Moore and Bellamy (1974), and Weltzin *et al*. (2001), as presented in Rydin and Jeglum (2006), for *Sphagnum* productivity levels within differing parts of the microtopography. It can be seen from Table 11 that the pool environment in general gives rise to consistently higher rates of productivity than either the somewhat drier lawn or the distinctly drier hummock environments.

Table 11. Productivity of different *Sphagnum* growth-forms or species grown in differing micro-topographic environments, based on data from Clymo (1970) and Weltzin *et al.* (2001).

Author	Species	Productivity (g m^{-2} yr^{-1})		
		Hummock	Lawn	Pool
Clymo (1970)	S. capillifolium	430	320	-
Clymo (1970)	S. cuspidatum	-	360	790
Clymo (1970)	S. papillosum	310	-	610
Clymo (1970)	S. recurvum	360	-	540
Weltzin *et al.* (2001)	3 Sphagnum spp.	162	236	311

It might thus appear reasonable to assume that pools and hollows, being sites of such vigorous growth, would rapidly fill with *Sphagnum* litter and rise above the much slower-growing hummocks to become hummocks themselves, while the almost moribund hummocks steadily degrade to become hollows. This image was indeed the basis of the 'hummock-hollow regeneration cycle', which Barber's (1981) work gently demolished.

The reason that slow growth in the hummocks and rapid growth in pools does *not* result in pools infilling and rising above the slower-growing hummocks is a rather complex story, encapsulated in two accounts of peat formation given by Belyea and Clymo (1998, 2001). One of the complicating factors is that published evidence contains contradictory statements about the relative growth of *Sphagnum* species in the differing parts of the microtopography. Thus Clymo and Hayward (1982) present evidence of field trials for *Sphagnum* species placed in differing micro-habitats in the north Pennines, which, as described earlier, clearly indicate that the rather delicate, typical species of hollows, *Sphagnum cuspidatum*, when grown in hollows has a net dry matter productivity substantially higher than that of *S. capillifolium* (a robust hummock-forming species) when grown in its typical hummock micro-habitat.

Belyea and Clymo (1998) note this same trend in *laboratory* experiments, but state that in *field* conditions the observed length-increase of different *Sphagnum* species was just the reverse, with hummock species showing the greatest incremental changes, although they do acknowledge that simple length increases may not equate to increases in total mass. They do, however, note that Wallén *et al*. (1988) record higher overall rates of *net* primary productivity from hummocks rather than hollows. This was partly because the hummock biomass included significant quantities of leaves and stems from higher plants above ground, while below ground there were even more substantial quantities of leaf-bases and roots, as highlighted by Charman (2002). Thus hollows and pools may induce rapid *Sphagnum* growth, but hummocks appear to have maximal overall net productivity.

The critical issues would therefore appear to be:

- the extent to which *Sphagnum* litter in <u>hollows</u> decomposes before it has the opportunity to be transferred into the catotelm;
- the extent to which *Sphagnum* litter in <u>hummocks</u> decomposes before it has the opportunity to be transferred to the catotelm;
- the extent to which <u>non-*Sphagnum*</u> litter in hummocks decomposes before it has the opportunity to be transferred to the catotelm.

The constant factor in all these questions appears to be the rate of decomposition, rather than the rate of growth.

In a wide-ranging review, Aerts *et al*. (1999) calculate a significantly lower decay-constant (% of material lost to decay each year) for *Sphagnum* plants (0.15) than for evergreen (0.3), deciduous (0.38) or grass/sedge (0.41) plant species. Meanwhile Johnson and Damman (1991) show that the typical hollow species *Sphagnum cuspidatum* consistently decays more readily than the hummock-forming *S. fuscum*, irrespective of whether the species is placed in hollow or hummock environments, or in oxic or anoxic conditions.

In other words, the hummock-forming species appears to be inherently more resistant to decay than the species typical of bog hollows. At the same time, the extra litter from higher plants decays comparatively quickly even within the hummock, leaving mainly the more resistant *Sphagnum* litter to pass down into the catotelm.

In some ways it appears intuitively obvious that the net balance between productivity and decay must be lower in hollows and pools than in hummocks, simply because hollows and pools so obviously form depressions in the bog surface. The difference in surface heights between the *Sphagnum papillosum* of a T1 low ridge (*sensu* Lindsay, Riggall and Burd, 1985) and the *Sphagnum cuspidatum* carpet of an A1 hollow may be no more than a few centimetres, but anyone who has sunk thigh-deep in such a hollow abruptly becomes only too conscious of the relative paucity of material filling the hollow. In the case of A3 drought-sensitive pools and A4 permanent pools, the distinction between the surrounding peat-forming ridges and the deep, un-vegetated pools is sufficiently stark to require little elaboration. Some bog pools have sheer sides which drop away through clear brown water to depths of 5 metres or more. Ivanov (1981) calls larger examples of these peat-based water bodies 'endotelmic lakes'.

In an elegant study which displays real sensitivity to the nuances of species behaviour and bog-ecosystem processes, Malmer and Wallén (1999) have explored the relationships between species composition, acrotelm thickness, litter residence-time and decay rates between different elements of the microtopography. They highlight first the fact that rapid peat accumulation across their sites in the past is clearly associated with presence of an almost unbroken cover of *Sphagnum* species typical of hummocks and hollows during such periods. In contrast, the bog surfaces of their two study sites today only have a positive mass balance (*i.e.* are accumulating peat) in the areas of *Sphagnum*-rich hummocks and lawns (T3 – T1 *sensu* Lindsay, Riggall and Burd, 1985) but these features today occur across a much-reduced area, with, for example, only 25-30% of hummocks now supporting *Sphagnum*.

Malmer and Wallén (1999) show that hummocks have consistently contributed a greater proportion of accumulated peat than the hollows during the past 1,000 years, despite the fact that overall decay has also evidently occurred at higher rates in the hummocks than hollows. This apparent contradiction can be explained by the fact that the hummocks contained more biomass and, more particularly, more vascular-plant biomass, than the hollows and therefore the levels of measured decomposition products (essentially nitrogen) are greater in the hummocks.

The critical point is that Malmer and Wallén (1999) show that *Sphagnum* decomposition in the hummocks is *less* than that in the hollows. The decline in *Sphagnum* cover over the last 200 years on these sites, particularly of hummock and lawn *Sphagnum*, is proposed as the reason for the observed decline in peat accumulation from 100 g m^{-2} yr^{-1} a millennium ago to the current estimated rate of only 10-20 g m^{-2} yr^{-1}. While all *Sphagnum* surfaces have declined – a phenomenon which Malmer and Wallén (1999) speculate may be due to landscape-scale drainage around the sites – the loss of *Sphagnum*-rich hummocks and lawns has the most marked impact on peat accumulation because these are the microtopographic features with the most evident positive mass balance.

In summary, therefore, while the wetter zones of the microtopography may be the sites of highest productivity, they also appear to be the sites of most rapid *decomposition*. Just like Alice's White Rabbit, despite all their vigorous activity, hollows and pools have little show for it at the end of the day. They remain somewhat poorer *net* producers of plant material (and thus of peat, and thus of carbon) than hummocks. As Belyea and Clymo so memorably put it:

> *"By analogy, one may imagine the hummocks as small but active dogs on leads, straining as far ahead as they can from the staid hollow that holds the leads in hand and controls the rate at which the convoy moves."*
>
> *Belyea and Clymo (1998)*

7.1.4 Decomposition and surface conditions

Looking in more detail at the process of decay in the acrotelm, Clymo (1965) investigated the process of *Sphagnum* breakdown in a blanket bog in the northern uplands of England (Moor House) and in a base-poor valley fen in the southern English lowlands (Thursley). He found several important things:

- some *Sphagnum* species decomposed more readily than others, even under the same conditions, with *Sphagnum papillosum* losing 2.5% original matter per year whereas *S. capillifolium* and *S. cuspidatum* lost 5% per year;breakdown rates were greatest in the surface layers whatever the *Sphagnum* species;
- decomposition was very much slower below a boundary that marked the distinction between oxidised sulphur in the upper layer and reduced sulphur (in the form of sulphides) in the lower layer – this largely equated to, but gave a clearer boundary line than, the position of the bog water-table and might thus be considered as a functional indicator of the boundary between acrotelm and catotelm;
- the roots and leaf-bases of several species of higher plant lay within the de-oxygenated sulphide zone and were thus largely preserved, whereas generally aerial parts were rapidly decomposed;
- decomposition occurred through the action of (mainly aerobic) bacteria and fungi alone, rather than being assisted by arthropods and other typical soil animals;
- very slow decomposition did occur in the sulphide zone, probably through the action of organisms using sulphur rather than oxygen to provide energy (in other words largely Archaean organisms);

- perhaps most significantly of all, in terms of local conditions and features, *Sphagnum* samples placed in a flushed region (a zone of distinct, though slow, water movement) decomposed so completely that the material was unusable for analysis.

Clymo and Hayward (1982) additionally show that *Sphagnum* contains remarkably low amounts of nitrogen, as do the below-ground parts of several other typical bog plants (*e.g.* hare's-tail cotton grass *Eriophorum vaginatum* and heather *Calluna vulgaris*), and that this is directly correlated with a slow rate of decomposition. Artificial enrichment with nitrogen results in faster decomposition rates. Thus it seems that *Sphagnum* and some plant parts to some degree 'starve' themselves of nitrogen in order to resist decay.

Goode (1970), while studying the dynamics of blanket bog within the Silver Flowe National Nature Reserve, in Dumfries and Galloway, found that small, low-lying sections of ridge separating bog pools at differing heights tended to form overflow runnels during periods of high water table. Over time, the slower rate of peat accumulation in these runnels meant that they were left behind as the ridge around them continued to grow. Once the ridges on the downslope pool had grown sufficiently for both pools to be at the same level, the pools would coalesce across these low-lying runnels, leaving the remainder of the intervening ridge as a line of isolated islands within the new, larger pool.

It can be taken from the above that:

- some species of *Sphagnum*, particularly terrestrial species, appear to resist decay more than those typical of aquatic conditions, this variation being at least partly based on the level of nitrogen in their tissues;
- (largely aerobic) microbes and fungi nevertheless do manage to overcome such defences to some extent; and
- overall peat accumulation depends on the quality and quantity of material that survives such attack and enters the sulphide zone – essentially the catotelm (Clymo, 1965; Moore and Bellamy, 1974); and
- zones of seepage, whether they be narrow, small-scale runnels or large valley fen systems, result in markedly reduced rates of net peat accumulation.

In addition to the significance of terrestrial *Sphagnum* species for peat formation, another key point therefore becomes clear. If the acrotelm layer is subject to *any* significant form of increased surface seepage – for example because it lies in a slight zone of water collection within the landform, or because there is slow seepage of groundwater from a sub-surface fault-line – the rate of decomposition within the acrotelm will increase dramatically. Signs of such zones in the field can sometimes be quite subtle, and thus it is all too easy to establish a research site which is designed to investigate ombrotrophic, peat bog processes yet the results are perhaps more applicable to minerotrophic, poor-fen conditions.

For example, the patterned ground of a ladder fen (Lindsay *et al*., 1988; Charman, 1994) does not look so different when viewed at ground level from the patterns formed across a patterned ombrotrophic blanket bog. Indeed the present author some decades ago contributed to a paper which identified a substantial area of patterned ground in the Outer Hebrides as a type of ombrotrophic blanket mire (Goode and Lindsay, 1979) but the present author has more recently recognised this same area as a large ladder fen (Lindsay and Freeman, 2008). However, the hydrological, oxygen and solute chemistry of ladder fens is markedly different from the conditions which prevail across ombrotrophic areas of blanket mire (Charman, 1993, 1995), and thus the processes of acrotelm growth and decay will also differ.

This is also an issue when peat-carbon studies are reported from Boreal regions such as Finland or Canada, because in the former case a great many of the sites investigated are actually patterned fens rather than bogs. In both countries, the long winter period means that the surface layers of the system are likely to be frozen and/or snow covered for several months, then in spring there is a very considerable period of surface-water flow resulting from snow-melt, followed by a relatively short growing season. All of these factors together mean that the hydrological, growth and decomposition conditions on Boreal mire systems are so different from the relatively ice-free conditions and almost-continuous growing periods characteristic of ombrotrophic British blanket mires that values obtained from such Boreal mires need to be considered with some care and considerable caution before applying them to British bog systems.

Ultimately, however, what separates peat bogs from non-peat habitats is that they do indeed transfer some undecomposed biomass from the rather turbulent environment of the acrotelm and pass it into long-term storage within the catotelm.

7.2 Accumulation and decay in the catotelm

The proof of the pudding, on this occasion, is in the peat. That is to say, the presence of a peat deposit demonstrates more clearly than anything the fact that there is, or has been, transfer of organic matter from the acrotelm to the catotelm, and that there has been preservation of at least some of this material in the catotelm.

One of the most notable features about the record of this transfer and partial preservation of material is the remarkably steady and consistent picture over thousands of years displayed by many peat bog systems. The simplest way of displaying the rate of transfer and preservation is by plotting peat thickness against age of the peat (a 'depth-age' plot).

Barber (1981) did so for Bolton Fell Moss, Cumbria, covering the most recent 2,000 year period and found that the resulting plotted line was almost, but not quite, a straight line, suggesting that the overall rate of accumulation had been something around 0.6 mm yr^{-1}. The line was in fact a gentle curve which indicated that the long-term rate of accumulation 2,000 years ago was a significantly lower 0.3 mm yr^{-1}, whereas the apparent rate during the most recent 500 years was 0.8 mm yr^{-1}.

These figures do not necessarily mean that there was slower peat accumulation in earlier times. The more likely explanation is that older material has steadily decomposed further and been lost over time compared to younger, more recently-added material. Quite simply, deeper material has had more time to decay.

Further complicating the picture is the evidence presented by Malmer and Wallén (2004) that Store Mosse in southern Sweden was transferring carbon to the catotelm 800 years ago at a rate of 53 g m^{-2} yr^{-1}, while 150 years ago the transfer rate had declined to 28 g m^{-2} yr^{-1}. They estimate that the present-day transfer rate is only 8 g m^{-2} yr^{-1}. Belyea and Malmer (2004) look at this phenomenon in more detail and identify that marked changes in carbon accumulation rates appear to be explained by shifts in the vegetation assemblage.

In other words, this is not a simple story involving CO_2 take-up by vegetation, processing through the acrotelm, and then steady transfer to the catotelm to be preserved indefinitely. Biotic factors can cause the pattern of carbon transfer into the catotelm to change markedly for a variety of reasons, while once in the catotelm there are decay processes which have proved rather difficult to model satisfactorily. Clymo (1992) and Clymo, Turunen and Tolonen (1998) have explored at length the basis of existing carbon accumulation models and demonstrate that the factor which leads to most debate is not the rate at which material is transferred to the catotelm but is instead the rate at which that material subsequently decays in the catotelm.

It is this balance between transfer of material to the catotelm and subsequent loss of this material within the catotelm which is the key to development and maintenance of a peat bog carbon store. Clearly in a mature tropical rainforest, or even a deciduous woodland, the numbers involved in such a (hypothetical) transfer would be very large but the result would be zero – decomposition of material matches the rate of input.

The balance in a peat bog is clearly not zero because material accumulates over long time periods as peat. The fact that this is a balance between opposing forces – deposition vs decay – means that the long-term (and even short-term) accumulation process is not as simple to describe as might be imagined.

7.2.1 LORCA/LARCA-RERCA-ARCA : rates of peat accumulation

The fact that differing rates of accumulation are apparently associated with different phases in bog growth has given rise to a range of terms which describe these various phases. It is as well to have a clear understanding of these terms because rates of peat accumulation should always be cited according to these terms, otherwise it is not clear what exactly is being described.

7.2.1.1 LORCA/LARCA

This term is known as the 'long-term apparent rate of carbon accumulation', and is measured simply by taking the mass (not the depth) of the whole peat thickness and dividing it by the total age of the peat from its origins to the present day. This is generally given as g m^{-2} yr^{-1}, and is relatively easy to calculate because only a single dated age is needed from the basal deposits, while the mass can be measured using consecutive cores taken from the entire profile, or estimated using a depth v bulk density relationship (with all the hazards of estimation which that entails).

The relationship is a straight line because it consists of only two points – the origin of the bog and the present-day surface. As we have seen above with Barber's (1981) data, however, although the LARCA provides a simple measure, it does not necessarily give the whole story. It is also important to be clear that comparison of LARCA values between sites is only really valid between similar peatland types and between peatlands of similar thickness. As we have seen above, accumulation rates in a seepage-influenced valley fen are likely to be very much lower than those found in a large watershed blanket bog. Thus a LARCA from a 5 m blanket mire in northern England is only really comparable with a LARCA from another 5 m blanket mire in the Pennines. Different thicknesses, or different geographies (England v Scotland), or different mire types (ombrotrophic watershed mire v flushed blanket mire slopes) will produce differing LARCA values.

7.2.1.2 RERCA

This term represents the 'recent rate of carbon accumulation' – 'recent' meaning within the last 100 years or so. It is measured as for LARCA, but from a more recent date in the peat. Thus the higher value obtained by Barber (1981) for the upper part of his depth-age curve could have been described by a RERCA value if bulk-density values had been available. RERCA gives a sense of the balance of accumulation in recent times, and can usefully be contrasted with values for LARCA to obtain some idea of decay rates deeper in the peat. As with LARCA, there are few, if any, determinations of RERCA for British blanket mires.

7.2.1.3 ARCA/TRACA

This term refers to the 'actual/true rate of carbon accumulation', measured as an instantaneous current rate of carbon accumulation within the catotelm. There are two important things to note about ARCA.

Firstly, ARCA should not be confused with the current rate at which material is added to the catotelm from the acrotelm. ARCA describes the *balance* between this newly-added material and the amount of decay which occurs throughout the peat thickness during the same time-interval. It is assumed from carbon accumulation models constructed by, for example, Clymo (1992) that the total accumulation rate of peat bogs declines with age because over time a greater quantity of catotelm decomposition balances the limited quantity of material added each year to the catotelm. Consequently carbon accumulation models show ARCA declining over time, with current ARCA estimates being only some 40% of LARCA values.

Secondly, ARCA is not actually measured, in the sense that the uppermost part of the catotelm is assessed for both its bulk density and its thickness, then the amount of decay throughout the catotelm is assessed, and from these measurements an ARCA value is obtained. ARCA values are more usually calculated from measurements of gas exchange associated with photosynthesis, respiration, aerobic decomposition and anaerobic decomposition (Rydin and Jeglum, 2006). Such calculations assume that all inputs and outputs are fully accounted for and are interpreted correctly. Clearly there is significant potential for this not to be the case.

7.2.1.4 "p*" – "Active" Rate of Deposition by the Acrotelm (ARDA)

It is curious that, unlike ARCA or LARCA, the rate at which material is transferred from acrotelm to catotelm has never been given a term to itself, other than the unwieldy "rate of addition of dry mass [to the catotelm] on an area basis" or the succinct but obscure parameter label "p*", both descriptions being given by Clymo (1992) in his peat accumulation models. Indeed Clymo *et al*. (1998) are themselves obliged to emphasise the distinction between "p*" and ARCA because they are clearly concerned that readers may confuse the two.

A suitable name for "p*" would be of considerable value because it represents something of very real importance within the terms of the EU Habitats Directive. For both raised bogs and blanket bogs, "active" bog is accorded priority status within the Habitats Directive, and "active" bog is defined as bog which supports a vegetation capable of producing peat. There is no requirement within the Directive for a bog to be a net accumulator of carbon. However, within the wider scientific peatland community there is also recognition that peatland accumulation models point to the fact that bogs become less effective at accumulating peat over time. This has led to the unfortunate conflation of ideas that bogs must inevitably become less "active" over time.

To see the error in this conflation, it is important to understand that a bog may be contributing just as much material from the acrotelm to the catotelm as it was 3,000 years ago but, because the peat deposit is now much thicker, the total amount of catotelm decomposition may mean that total *net* carbon accumulation (ARCA) is now quite low. Thus, as an object of carbon sequestration and storage, the bog is less efficient than it was 3,000 years ago. On the surface, however, the bog vegetation may be as vigorous and as capable of producing material for the catotelm as ever. The bog may therefore have a relatively low ARCA value but still have a healthy peat-forming vegetation contributing to a high value of "p*".

To be able to give this term "p*" a name – and it is proposed here to call it ARDA (active rate of deposition by the acrotelm) – would mean that it is then possible in discussions when carbon storage and "active" bog status are being discussed together to talk in conceptual terms at least about "active" bog having a value of ARDA greater than zero, even if ARCA values are low. In other words, from the perspective of the EU Habitats Directive, whatever the overall carbon balance of a bog might be, the acrotelm continues to be capable of contributing material to the catotelm, and is therefore "active", as long as its value of ARDA is positive.

In practical terms, of course, the measurement of ARDA would be highly complex and lends itself to experimental proof no more easily than ARCA. As such, the practical criterion for "active" status can continue to be the established definitions of "active" bog provided by the JNCC. The explicit recognition of the term ARDA, combined with the explanation that even a bog with zero ARCA may still be highly "active" because it has a positive ARDA value (*i.e.* the site was still laying down fresh peat even if its overall net carbon balance is zero), would nevertheless do much to undo the misunderstandings caused by the incorrect conflation of low LORCA/ARCA values and "non-active" status.

In fact Clymo (1992) observes that even the most conservative model of peat accumulation places the peat bog systems of Europe no further than 65% towards their limit of accumulation at the present time. Other less conservative models illustrated by Clymo (1992) suggest that in fact there is no limit – as long as the bog can continue to expand laterally, or is able to fuse with adjacent peat, the peat will continue to accumulate indefinitely, albeit at a slowly diminishing rate.

7.3 Peat accumulation, decomposition and gas exchange

Ultimately, it is the combination of growth and partial decay in the acrotelm, together with LARCA, RERCA and ARCA within the catotelm, which must be used to judge the significance of peatlands as carbon stores in comparison with other habitat types. Peatlands undeniably accumulate carbon, but they are also wetlands – anaerobic wetlands – and thus they are also capable of releasing methane from anaerobic decomposition processes. Methane is a much more potent greenhouse gas than carbon dioxide - the decomposition product most frequently associated with the majority of other non-peat habitats.

There is clearly a question of balance to be examined here. Do peat bogs accumulate sufficient carbon to offset their emissions of methane? What good is a huge carbon store if the very retention of that store necessarily involves release a gas which can significantly outweigh the greenhouse gas (GHG) benefits arising from the store?

The processes of peat accumulation and decay are intimately bound up with the gases which contribute to, or result from, these processes. The scale and significance of such gaseous exchange depends on a number of things including the number of pathways involved, the ease with which gases may pass along these pathways, and the relative potency of these gases in relation to climate change. Greenhouse 'potency', or to give it its technical term, 'greenhouse warming potential' (GWP) will be considered next because it is a rather more straightforward issue to deal with.

One final observation, however, needs to be made clear before moving on to consider GWP. A great many published figures for gaseous exchange from peatlands are based on studies carried out on Finnish peatland systems. It is important to recognise that fens greatly outnumber bogs in Finland (Ruuhijärvi, 1983; Aapala *et al.*, 1996), as indeed they do in many other parts of the world. This is reflected in the much higher proportion of published values for CO_2 or CH_4 fluxes from fens rather than bogs (*e.g.* Kettunen *et al.*, 2000; Saarnio, 2007; Minkkinen *et al.,* 2007a,b, and see Hargreaves, Milne and Cannell, 2003, and Nayak *et al*., 2008).

It is not always clear what type of site is being described in the Finnish literature because the Finns have their own site-type classification and often papers simply use the classification code. A summary of the main Finnish mire types can be found in Turunen (2008) and a more detailed treatment in Laine and Vasander (1996). The significance of this bias towards fens lies in the fact that fen peat decomposes more rapidly because there is more mineral influence and more water movement. Consequently CH_4 and CO_2 fluxes in these fenland systems are often very different from such fluxes in bog systems. It is thus important to be aware that these values may not be wholly relevant to British peat bog systems.

7.3.1 Carbon dioxide, methane and greenhouse warming potential (GWP)

Carbon dioxide is the most widely-recognised gas associated with the phenomenon of global warming and is most popularly linked with the idea of the 'greenhouse effect'. However, several other gases have a part to play in this story. Some of these, such as methane or nitrous oxide, have an even more potent greenhouse effect than CO_2, while others such as dimethyl sulfide may actually help to cool the planet.

Each of these gases has a different lifetime in the atmosphere, meaning that any given molecule of the gas will survive in the atmosphere for only a specified period of time before it breaks down (and must then be replaced by another molecule if the effect is to be maintained). Similarly, each gas has its own inherent capacity to warm or cool the atmosphere because of its particular physico-chemical characteristics. The atmospheric lifetime and 'warming potential' of these various gases are combined to create a value of 'global warming potential' (GWP) for each gas. It has been accepted by convention that the GWP for all such gases will be measured against the behaviour of CO_2; thus CO_2 has a GWP of 1.

For example, methane is currently considered to be 25x more powerful as a greenhouse gas than CO_2 because of its particular physico-chemical composition, but this does not mean that the GWP value for CH_4 is a simple value of 25. This is because CH_4 has a much shorter lifespan in the atmosphere than CO_2. Consequently any released quantity of CH_4 will steadily become less significant over time as more of the methane decomposes. Thus a GWP value should always be accompanied by a time-horizon (*e.g.* CH_4 GWP = 25, over 100-year time-horizon).

The relationship between GWP and time-horizon is extremely important but is not firmly established. Estimates for the atmospheric life-span of gases, for example, continue to be the subject of ongoing research. Thus in 2001 the CO_2 GWP=1 baseline was lowered by the IPCC, thereby raising the GWP values for other gases, as a result of recent findings (Rydin and Jeglum, 2006).

One important aspect to be aware of is that some GWP values cited in the literature are given on a molar basis and some are not. Molar values of GWP are smaller than non-molar values. Thus on a molar basis, the GWP of CH_4 is 9.1 (time-horizon 100 years), whereas on a non-molar basis the GWP of CH_4 is 25 (time horizon 100 years). The difference arises because a molar quantity of a compound is based on the molecular weight. Thus 1 mole of CO_2 can be compared directly with 1 mole of CH_4 and on this molar basis the CH_4 can be described as being 9.1x more powerful a greenhouse gas than CO_2 (time-horizon 100 years) However, 1 mole of CO_2 weighs 44 g whereas 1 mole of CH_4 weighs only

16 g. If similar masses of material are to be considered, then 44 g of CO_2 has a GWP of 1 (time-horizon 100 years), while 44 g of CH4 has a GWP of [9.1 x (44/16)], which is 25 (time-horizon 100 years).

The shorter the timescale considered, the greater the GWP of methane compared to CO_2. Whiting and Chanton (2001) demonstrate this relationship on a molar basis for CH_4 over differing timescales. It is thus possible to construct a table which considers the GWP of methane over different timescales and on a molar or mass basis (see Table 12).

Table 12. Greenhouse warming potentials (GWP) for methane (CH_4)
GWP values for methane based on differing time-horizons and expressed either on a molar or mass basis. Values used here on a molar basis are from Whiting and Chanton (2001), whereas figures for mass balance are based on updated figures. This is because GWP values have since been somewhat increased by the IPCC and so the mass-basis 100-year horizon for methane is now 25, as given here*, whereas Whiting and Chanton (2001) give it as 21 (see GWP Wikipedia website).

Time-horizon	Molar basis	Mass basis
20-years	21.8	72*
100 years (IPCC standard)	7.6	25*
500 years	2.6	7.6*

The values for GWP are of considerable significance when calculating the relative balances of gas exchange for a peatland. Even over a 100-year time-horizon, quantities for CH_4 on a mass basis must be multiplied by 25 (to use the latest IPCC values) in order to judge the GWP of such emissions. Such a multiplication factor is far from trivial. However, change the timescale to 500 years, which is a reasonable timeframe within the context of a peat bog carbon store, and the multiplication factor is reduced to less than 1/3 of this value.

7.3.2 Carbon-exchange pathways – carbon dioxide (CO_2)

Being the carbon-rich raw material for photosynthesis, CO_2 is the basis of plant growth. It is thus the means by which the acrotelm accumulates material and ultimately is the source of the small amount of material which is eventually passed down into the catotelm (ARDA). Without CO_2 and photosynthesis, peat cannot accumulate, although obviously it can only accumulate during the hours of daylight.

At the same time, CO_2 is also a product of oxidative breakdown, and is thus particularly a feature of night-time respiration within the living vegetation. It is also more generally a breakdown product when organic matter is decomposed in the presence of air. Consequently CO_2 is associated with the high levels of acrotelm decomposition observed by Wallén (1992) whereby 800 g m^{-2} of plant production in the acrotelm can be reduced to a mere 30 g m^{-2} of biomass by the time it is transferred to the catotelm.

Under natural conditions, O_2 would not normally be found in the catotelm because the conditions there are anaerobic. If the acrotelm has been damaged and transformed into a haplotelm allowing air to penetrate the catotelm, it then becomes possible for oxidative decomposition to occur within the catotelm peat, leading to what is known as 'oxidative wastage'. Under these conditions carbon is lost through two pathways:

- to the atmosphere as CO_2 (oxidative wastage);
- to the regional water-table as water-borne organic breakdown products (DOC, POC).

There is also the additional factor of water movement to consider. As highlighted earlier in relation to Clymo's (1965) work on *Sphagnum* decomposition, water movement can give rise to more oxygenated conditions simply by providing more oxygen per unit time. This movement can itself then also help both to remove water-borne organic breakdown products (generally referred to as dissolved organic carbon – DOC) and ensure their continual re-formation.

In terms of assessing the nature and scale of the CO_2 being taken up by, and being lost from, a bog system, it is thus of some importance to establish:

- whether the peat is vegetated and thus capable of adding CO_2 as biomass;
- whether the existing vegetation cover is capable of adding to the carbon store of the acrotelm;
- whether the existing vegetation is ultimately capable of adding material to the carbon store of the catotelm;
- whether the bog surface is an acrotelm or a haplotelm;
- the extent to which the catotelm may be penetrated by air;
- the extent to which the peat may lie in a zone of seepage.

7.3.2.1 Estimates for CO_2 storage – LARCA and ARCA

Theoretically, the measure of whether an undamaged bog presently absorbs more CO_2 than it loses can be found in the 'actual rate of carbon accumulation' (ARCA). The difficulty with such an approach until fairly recently was that, as discussed above, the ARCA is normally calculated using peatland growth models rather than by physically measuring any incremental changes in peat material. The ARCA value therefore relies heavily on the accuracy of the peatland growth models used as well as the actual employment of these models to produce site-based values. So far, such values have been created only for 'modelled' bog systems or for a range of Scandinavian sites (e.g. Clymo *et al*., 1998; Malmer and Wallén, 2004; Belyea and Malmer, 2004).

In perhaps the most extensive study of its kind, Clymo *et al*. (1998) explore the behaviour of peat accumulation models using a large dataset obtained from peatland sites distributed throughout Finland. Data from Canada are also used to test certain aspects of the model. Clymo *et al*. (1998) conclude that the three main versions of their model, each of which uses a different decay rate, indicate that for 310 Finnish bogs and fens considered by the model from southern Finland, the ARCA values range from 24.36 g C m^{-2} yr^{-1} to 37.2 g C m^{-2} yr^{-1}.

One alternative to modelled values of carbon accumulation is to use either RERCA or LARCA values for catotelm accumulation because these measurements can be obtained with the assistance of single radio-isotope measurements either from the start of the 'recent' period, or from the basal peat. Gorham (1991) does so for a global review of 'northern peatlands', and on the basis of 138 peat cores distributed through the Boreal and Sub-arctic Regions, he calculates a LARCA of 29 g C m^{-2} yr^{-1}. Using 38 peat cores from the Boreal Region alone, Gorham (1991) then calculates an ARCA value of 23 g C m^{-2} yr^{-1}.

In their review of peatlands and carbon balance, Immirzi *et al*. (1992) use values from Armentano and Menges (1986) and Adger *et al.* (1990) to suggest what might best be regarded as RERCA values for British peatlands of 48-70 g C m^{-2} yr^{-1}, although the basis of Adger *et al*.'s (1990) original values is not clear, and the values for Armentano and Menges (1986) are derived for western European peatlands as a whole.

Cannell *et al*. (1993) and Cannell and Milne (1995), in their reviews of peatland carbon and forestry in Britain, give rather broad ARCA/RERCA estimates of 40-70 g C m^{-2} yr^{-1} for 'northern peatlands', citing Clymo (1983) and Gorham (1991). The basis of these cited figures is also not entirely clear, given that both Gorham's (1991) LARCA and ARCA values lie well below the lowest value in this range.

Looking to the Boreal Region again, Tolonen and Turunen (1996) found average LARCA values of 24 g m^{-2} yr^{-1} for a range of Finnish bogs, while Turunen (2003) recorded LARCA values of 30-35 g m^{-2} yr^{-1} for raised bogs in southern Finland. The blanket mires of Britain are warmer and wetter than Finnish bogs, but this is a double-edged sword. Greater warmth and wetness can increase peat accumulation but warmth also significantly increases microbial decay, while increasing wetness means that more water movement, and thus more transport of solutes (including oxygen), occurs per unit time.

It is perhaps significant that Charman (2002), in his review of carbon accumulation rates, is obliged to fall back to age-depth curves in order to discuss measured values for peat accumulation in Britain. He notes that raised mires have been shown to accumulate at a rate of 1 mm yr^{-1} in Britain and observes that, according to Tallis (1995b), blanket mires have been more variable, ranging from 0.1-1.2 mm yr^{-1}. It is revealing to use the Cannell *et al*. (1993) formula together with their parameters, to calculate a form of LARCA value for the mid-point of 0.65 mm yr^{-1} from Charman's (2002) blanket mire values. This mean increase in thickness can be shown to be equivalent to 30.5 g C m^{-2} yr^{-1}, which is remarkably close to Turunen's (2003) measured LARCA figures of 30-35 g C m^{-2} yr^{-1} for Finnish raised bogs.

As an example of how convoluted the source of particular quoted figures for carbon accumulation becomes at times, Dawson and Smith (2006) cite Cannell (1999) and Clymo *et al*. (1998) as providing evidence for terrestrial carbon sequestration rates of 20-50 g C m^{-2} yr^{-1} for "UK undrained peatlands". Cannell (1999) himself cites Clymo *et al.* (1998) as his source of information about British peatlands, though also comment that there was discussion with Prof. Clymo about these values. Cannell (1999) consequently proposes carbon accumulation rates for undrained British peatlands in the range of 20–50 g C m^{-2} yr^{-1}, although the basis of these figures is not explained. Meanwhile the original study by Clymo *et al*. (1998) as we have seen from the description above is based on peatland sites in Finland together with some data from Canada, and sheds no specific light on British peatlands, undrained or not. Taking the comment by Dawson and Smith (2006) on face value, it would be reasonable to assume that carbon sequestration figures for UK undrained peatlands had been explicitly determined, whereas an audit trail of the source for such figures fails to reveal their basis or their origin.

7.3.2.2 Estimates for CO_2 storage – mass-balance studies

A more recent approach to estimating carbon accumulation in British peats does not involve measurement of the peat matrix itself, but relies instead on the measurement of inputs and outputs from the peatland to create a mass-balance calculation from which, amongst other things, the net increase in carbon storage may be calculated.

One important difference to recognise between these mass-balance studies and previous estimates based on actual measurement of the peat, is that LARCA, RERCA and ARCA values integrate all aspects of loss whether as DOM, particulates or gas. A clear incremental increase must mean that CO_2 has been sequestered from the atmosphere in greater quantity than carbon has been lost through all routes from the body of the bog.

In contrast, for the mass-balance approach the rate of CO_2 sequestration is just one parameter among many. Losses due to CH_4 release, or as DOC into surrounding watercourses, may be balanced against this rate in the mass-balance equation. The term 'carbon sequestration' in a mass-balance study may thus actually mean Clymo's (1992) parameter "p*", proposed above as ARDA. – the active rate of deposition from the acrotelm only. It is important to be clear about this when comparing terms and values from mass-balance studies with figures from other kinds of study.

Three such studies can be cited as examples of this trend. The first is a review of the total carbon budget for a blanket mire catchment in the Pennines (Worrall *et al*., 2003a). The second study involves a large lowland mire which is somewhat intermediate between raised mire and blanket mire, with measurements undertaken during 1995-1996 (Billett *et al*., 2004), while the third study involves the same site but with more extensive measurements taken during 2007-2008 (Dinsmore *et al*., in press).

The main experimental focus for Worrall *et al*. (2003a) was measurements of DOC, DIC and CH_4 within the Trout Beck catchment, Moor House, in the north Pennines. Consequently in assembling a total carbon budget for this catchment, several components in the budget were obtained from the literature. This included the net exchange of CO_2. The net carbon exchange values of Clymo *et al*. (1998) and Cannell *et al*. (1993) are cited, as are a number of CO_2 emission rates. For the purposes of the mass-balance model, Worrall *et al*. (2003a) select a value from the mid-point of the 40-70 g C m^{-2} yr^{-1} range proposed by Cannell *et al*. (1993) – namely 55 g C m^{-2} yr^{-1}.

This particular mass-balance study thus reverts to values obtained by LARCA/ARCA methods, but in applying this within the mass-balance approach, Worrall *et al*. (2003a) conclude that the Trout Beck catchment acts as a net sink for carbon, taking up 12.9 g C m^{-2} yr^{-1} (within a possible range of 6.4-27.9 g C m^{-2} yr^{-1}). Although rather low compared with the LARCA and ARCA values discussed earlier, it is quite conceivable that carbon accumulation varies from year to year and perhaps such relatively short periods of investigation (over one or two years) are capable of detecting this variation. Given the range of parameters involved, it is perhaps more remarkable that the mass-balance result should be so comparatively close to estimates of LARCA and ARCA.

Billett *et al*. (2004), on the other hand, assemble a total carbon budget for a catchment dominated by Auchencorth Moss, south of Edinburgh. The site is described as having been drained in the past, with a small area of current commercial peat mining on the western edge of the site. In fact although the area of commercial mining is described as extending across only 8 ha within the 335 ha catchment, close examination of the mining area reveals that this is likely to be an under-estimate. The edge of the peat-mining area runs along the 290 m contour evident on the map provided by Billett *et al.* (2004). As such, recent aerial photographs suggest that the likely extent of mining within the study catchment amounts to almost 40 ha. This represents nearly 12% of the study catchment and may therefore be more significant than is recognised in this particular mass-balance study, although the extent of peat mining may have expanded significantly since 1996. Nonetheless, Billett *et al*. (2004) acknowledge that the influence of drainage from the peat workings is a factor not determined during their study.

Land-atmospheric CO_2 fluxes were measured using an eddy covariance tower towards the downwind limit of the catchment (assuming a prevailing SW wind direction). It is also worth noting that the large expanse of bare, mined peat also lies some 2 km upwind of the tower and thus emissions tend to blow towards the tower. However, the gas footprint of such a tower is generally considered to be no larger than 500 m – 1 km. The influence of the exposed bare peat may thus be minimal.

Billett *et al*. (2004) recorded an overall net CO_2 input to the peat of 27.8 g C m^{-2} yr^{-1} for the catchment. This value, however, represents at most Clymo's (1992) "p*" or ARDA (active rate of deposition by the acrotelm), because against this Billett *et al*. (2004) balance carbon losses in streamwater and CH_4 emissions. Stream export of carbon from the site is noted by Billett et al. (2004) as being unusually high for a bog system, while methane emissions are taken from published literature rather than measured on-site. Consequently their effective value of ARCA (*i.e.* their net carbon balance) emerges as a net loss of carbon from the site, with 8.3 g C m^{-2} yr^{-1} being lost from the peatland.

Given the circumstances associated with the site, this is perhaps not surprising. The site has been intensively drained in the past, it is subject to active peat mining across a substantial part of the macrotope area, a large proportion of the macrotope crown of the bog on the southern margin of the catchment has conifer plantations planted on the deep peat, and the topography of the ground makes it likely that at least some of the water draining from the peat mining finds its way into the Black Burn, which is the main stream of the catchment. The pattern of regularly-spaced drains across the site also feeds into the Black Burn, further adding to the DOC and POC load received by this watercourse. It is worth observing that Billett *et al.* (2004) themselves note the high values for TOC (total organic carbon) in riverine C export. This TOC value is by far the most significant source of carbon loss, and almost exactly cancels out the measured amount of carbon sequestered as CO_2 by the site.

It is thus particularly interesting to note that more than 10 years later, Dinsmore *et al*. (in press) find that the same site now shows a significantly different carbon balance. In 2007, the Black Burn catchment had a positive carbon balance amounting to a net carbon gain of 101 g C m^{-2} yr^{-1}. A positive carbon balance was also observed in 2008, though at 38.2 g C m^{-2} yr^{-1} this was barely 1/3 of the carbon gains measured in 2007. The two most significant differences between Billett *et al*. (2004) and Dinsmore et al. (in press) are, firstly, that the annual measured uptake of carbon was between 3x and 5x higher in the recent study compared with values obtained for 1995-1996. Secondly, Dinsmore *et al.* (in press) measured on-site methane production, which proved to be unusually low, whereas Billett *et al.* (2004) used generic values for methane from published literature.

The one key set of values which remained the same between the two studies, and which are recognised by both studies as being highly significant factors in the overall carbon budget, are the measurements of carbon losses in the form of DOC and POC, which Billett *et al*. (2004) record simply as the combined TOC (total organic carbon). In both studies these values are unusually high for blanket mire catchments, Worrall *et al*. (2003a) for example using a range from 9.4 – 15 g C m^{-2} yr^{-1} for DOC export at Moor House, whereas Dinsmore *et al*. (in press) record an average of 18.6 g C m^{-2} yr^{-1} for 2007 and 32.2 g C m^{-2} yr^{-1} for 2008. A series of old drains runs eastwards all along the eastern margin of the commercial peat mining area towards the Black Burn, and the headwaters of the Black Burn approach to within 175 of the peat mining area through an area which is clearly part of this older agricultural drainage pattern. Furthermore, the area of peat mining draining eastwards towards the Black Burn now covers almost 40 ha of the study catchment.

If the high DOC values are in part derived from outflows from the peat-mining, this could mean that the net gain of carbon by the main bog system is even greater than indicated by Dinsmore *et al*. (in press). It is interesting to note that 2007 was somewhat wetter than 2008 yet the DOC output was substantially lower than in 2008. Wetter conditions generally mean more difficult, extended drying periods for the peat mining process and thus there is less overall extraction of peat – and thus, potentially, less DOC release. In the drier 2008, it may be that more peat was mined, explaining the higher DOC values. It would be interesting to correlate the peat mining activities at Auchencorth with DOC levels recorded in the Black Burn, combined with a detailed mapping of drainage lines feeding from the mined area towards the Black Burn.

The other surprising but potentially significant feature of the results obtained by Dinsmore *et al*. (in press) is the very low measured values for methane release from the bog. These values are an order of magnitude lower than those noted from other blanket bog sites. The fact that, as Dinsmore *et al*. (in press) observe, there are few open-water hollows or pools on the site but hummocks are frequent, may go some way to explaining this result, as will be discussed in Section 7.3.8.3.

Finally, it is worth noting the high rate of carbon sequestration observed in both 2007 and 2008 compared with the value obtained by Billett *et al*. (2004) a decade earlier. Although Auchencorth Moss was extensively drained for agricultural purposes several decades ago, it is clear that many of these drains are now choking up and infilling with *Sphagnum*-rich vegetation while much of the intervening ground now supports large hummocks of *Sphagnum* moss, hypnoid mosses and *Polytrichum* mosses. It is interesting to speculate whether in the past decade the drainage system has become so compromised by infilling of *Sphagnum* as a result of 'benign neglect' that the general water table is now capable of supporting an actively carbon-sequestering peatland vegetation.

Overall, therefore, this particular mass-balance study-site perhaps raises as many questions as it answers. Close examination of the relationship between the mining drainage outflows and study-catchment drainage would seem worthwhile, as would a study of the CO_2 emissions from the peat mining area in terms of their potential influence on eddy covariance values, simply to be sure that there is no influence there. Weather records, DOC and POC outputs and patterns of peat-mining operations might be fruitfully compared. The old but ubiquitous drains could also usefully be assessed in terms of their present hydrological effects and the character of the surrounding vegetation.

7.3.2.3 Estimates of CO_2 storage – a summary

It does not take long to become quite bemused by the various figures available for peat accumulation, especially as so many are in fact references to other works which are themselves references to other publications. The pragmatic response to this, just as we have already done with the values for peatland area in Table 1, Section 3.1 of the present report, is to set down the values in a table and identify the origins of these values (see Table 13).

In setting out the various figures in this way, a degree of consistency can be found to emerge, with estimates of LARCA generally ranging between 30 to 70 g C m^{-2} yr^{-1}, while ARCA values are lower (as would be expected) and mostly range from 6.4 to 50 g C m^{-2} yr^{-1}. The only negative value in Table 13 (i.e. an estimate of overall carbon loss rather than gain) is the mass-balance study by Billett *et al*. (2004), but possible reasons for this have already been discussed.

7.3.3 Mass-balance studies – areas of concern

Before moving on to consider other components of the carbon-balance equation, it is worth highlighting two particular areas of concern regarding the use of mass-balance studies in peatland carbon assessments.

Mass-balance studies are generally carried out on a catchment basis. This is understandable in the sense that any streamflow losses of carbon need to be collected within the relevant stream catchments. However, there is also a sense that this is a practical need which is being allowed to drive the system and something is being lost as a result. In particular, mass-balance work, because it is being driven by the traditional hydrologist's focus of *catchments* rather than the mire scientist's focus on *mire units* as the functional entities, tends to cut right through the centre of many mire systems, particularly those occupying watershed positions. The main areas of carbon storage are thus placed at the periphery of the study area and are bisected or fragmented by the boundary.

Table 13. Summary of values for carbon accumulation in peatlands.

Author	Basis for given values	Geographical range of, or use for, data	g C m^{-2} yr^{-1}	
			LARCA	ARCA
Armentano and Menges (1986)	Estimate	western Europe	48	
Adger *et al.* (1991)	No source specified	UK	70	
Gorham (1991)	Collated field records	Boreal Region	29	23
Immirzi *et al.* (1992)	Armentano and Menges (1986), Adger *et al.* (1991)	western Europe	48 – 70	
Cannell *et al.* (1993)	Clymo (1983), Gorham (1991)	UK	40 – 70	
Cannell and Milne (1995)	Clymo (1983), Gorham (1991), Immirzi *et al.* (1992)	UK	40 – 70	
Garnett (1998)	Field measurements	Moor House, Pennines	27	
Clymo *et al.* (1998)	Field measurements	Finland		24.4 – 37.2
Cannell (1999)	Clymo *et al.* (1998), discussions with R. Clymo	Finland (Britain?)	-	20 – 50
Anderson (2002)	Measured values	NW Scotland	4 – 73	
Worrall *et al.* (2003a)	Mass balance and some field measurements	Moor House, Pennines		6.4 – 27.9
Billett *et al.* (2004)	Mass balance with field measurements (damaged site)	Scottish Central Belt, Auchencorth Moss		-8.3 (loss)
Dinsmore *et al.* (in press)	Mass balance with field measurements (damaged site)	Scottish Central Belt, Auchencorth Moss		38.2 – 101.0 (gain)

The upland hydrologist thus quantifies the behaviour of several partial mire systems, rather than a single whole system. A mire ecologist, on the other hand, would most likely take the watershed mires as the central point of study and extend outwards from there to the river systems which mark the edge of the peat (the macrotope boundary). Thus the mire ecologist studies whole mire systems, but only considers partial catchments.

There is thus a systematic mis-match between the work of the mire ecologist and the upland hydrologist, yet both are essentially concerned with the carbon balance of the peatland system. It might be argued that this is not a significant issue because the hydrologist is concerned with the outputs of the system, which emerge through the catchment route, while the ecologist is concerned with the system itself.

Even in simple blanket mire landscapes, however, the practical result of these two differing approaches means that although several *parts* of mesotopes or macrotopes have been described, no complete blanket mire system (macrotope or mesotope), has been described for the UK in terms of mass-balance measurements. It is also worth noting that there is a third player to consider – the land manager – and land mangers may use landscape units which are sometimes defined by catchment boundary and sometimes defined by mire unit – mesotopes or macrotopes. In many cases the measurements obtained by the hydrologist or ecologist are ultimately intended to guide the decisions of such land managers, and the more integrated a picture which can be presented, the better for all concerned.

One solution might be for mass-balance investigations to focus first on the mire system and identify the elements of that system – macrotopes, mesotopes, microtopes – which play a role in defining the nature of the mire landscape under investigation. Having identified a macrotope, for example, as the central focus of the study, the catchments which contribute to that macrotope can also be delineated. The parts of these catchments which lie *within* the macrotope boundary contribute the mass-balance values of the macrotope, while the parts of the catchments which lie *outside* the macrotope provide the values which must be subtracted from the main outflow values in order to isolate only those values coming from the macrotope under investigation.

In practice, this would likely mean that mass-balance studies would need to carry out measurements across a significantly larger area of landscape than has traditionally been the case. Such additional work has resource implications, but if these resources are necessary in order to obtain a more peatland-focused set of mass-balance studies, then this must be recognised and funding bodies respond accordingly.

The second area of concern with regard to mass-balance studies is perhaps a rather more fundamental limitation. Mass-balance studies are predicated upon the assumption that measurements of the various mass-balance components taken over one or two years will provide a picture of the overall carbon balance of a peatland system.

This assumption does not take into account the dynamic and functional nature of the mire surface and its features. Microtopes and nanotopes change in response to changing conditions. Such conditions may be climatic, or in response to new drainage, or in response to the collapse and re-wetting of old drainage systems. As Barber (1981) and Ivanov (1981) have emphasised, drier conditions will tend to favour development of terrestrial T-zone features (*sensu* Lindsay, Riggall and Burd, 1985) such as ridges and hummocks, while wetter conditions favour the expansion of A-zone features such as hollows and pools.

Such shifts in surface pattern are largely (though not exclusively) the domain of acrotelm dynamics. The pattern of carbon sequestration in the acrotelm may be as much involved in re-shaping the nature of the acrotelm as it is in laying down peat for the catotelm. The retention or release of carbon at any one moment in time may or may not therefore reflect the longer-term balance of carbon which is available for transfer to the catotelm. This has major implications for mass-balance studies. Belyea and Clymo (2001) make the case very clearly:

> *"Short-term measurements of carbon dioxide exchange ... will also fail to provide accurate estimates of current rates of sequestration because carbon entering long-term storage in the catotelm cannot be distinguished from that contributing to short-term vertical adjustments in acrotelm thickness. Moreover, estimates of the overall greenhouse forcing will be compromised by climate-mediated shifts in the lateral extent and vertical height of microforms because the balance between the uptake of carbon dioxide and the release of methane depends very strongly on acrotelm thickness. All of these limitations in current approaches can be overcome only by including the feedback mechanisms that link carbon sequestration and hydrology to the dynamics of peatland microforms [nanotopes]."*

In other words, mass-balance studies cannot adequately interpret carbon-balance data unless the nature of the surface microtopography is adequately described and then measured over time. The element of time is also important for mass-balance studies because, as Alm *et al*. (1999), Evans *et al*. (1999) and Worrall, Burt and Adamson (2006b) observe, the effects of a single dry summer can have significant consequences for the carbon balance of a bog system.

7.3.4 Carbon-exchange pathways – methane (CH_4)

Methane is not easily created. Unlike the production of CO_2 which can be achieved using a single micro-organism, Svensson and Sundh (1992) highlight the fact that generation of CH_4 requires the combined efforts of three separate bacteria, each of which provides a part of the generation sequence. Methanogenic bacteria are extremely sensitive to oxygen, being unable to function when it is present. As such, they form part of the Archaeobacteria which first evolved during the early Archaean world.

Svensson and Sundh (1992) admit that the main pathways of methane production in peat remain largely unknown, although the general principles of what is known from other methanogenic habitats can still be applied.

While it is widely understood that carbon dioxide is both taken up from the atmosphere through photosynthesis and released back to the atmosphere through respiration, it is less well-known that methane (CH_4) can also be released from the peat or taken up from the atmosphere. Within waterlogged peat, methane is created as a product of breakdown. Whether this them moves upwards towards the atmosphere depends on a number of things:

- it can remain stored within the waterlogged peat as micro-bubbles (it is only sparingly soluble);
- it can be taken up by methanotrophic microbes as an energy source, with some entering the microbial biomass and some oxidised to CO_2 (which may or may not then be released into the atmosphere);
- it can be released directly into the atmosphere as methane.

Only the last two of these options have significant implications for greenhouse gas emissions, with the last option clearly being the most significant because the GWP for CH_4 is substantially larger than that for CO_2. Given these three possible fates, the important thing is therefore to establish the nature and scale of each option.

7.3.5 CH_4 storage (as micro-bubbles)

Svensson and Sundh (1992) emphasise the fact that one of the most important factors in controlling the rate of methanogenic breakdown of material is the chemical composition of that material. They illustrate the case with a study carried out on differing peatlands in the Hudson Bay Lowlands, examining the relative rates of CH_4 production between sites. It was expected that methane rates would be higher in bog sites compared with fens. This was because the bogs were richer in easily-digestible cellulose, and they contained much less lignin which is difficult to digest anaerobically.

The result proved to be quite the reverse; bogs produced less CH_4 than fens by at least a factor of 10. Factors such as differing nitrogen levels and positions of the water tables were considered as possible explanations for this unexpected result. However, it also seems highly likely that a key factor in creating the 'unpalatability' of the bog peat is the action of sphagnan, the pectin-like substance which acts both to make the *Sphagnum* structure unpalatable, but also inhibits nitrogen uptake in decomposer bacteria and thus causes them to fall into a condition of stasis. Where the peat also contains a quantity of non-lignified material derived from vascular plants, however, it is likely that methane production would increase significantly.

The end result of this process is that decomposition and associated CH_4 production is very slow in the catotelm. Levels of productivity are not sufficient to produce large bubbles of CH_4 which can then force their way to the surface as bubbles (ebullition). Production levels are only capable of creating many micro-bubbles of CH_4 which are then bound within the peat matrix by surface tension and simple physical entrapment.

This is a phenomenon exclusive to the catotelm, and helps to explain in part why water moves through the catotelm at speeds 100x slower than a snail's pace. Baird *et al.* (1997) present the case for this from a hydrological perspective while Brown and Overend (1993) and Brown (1995) present the microbial evidence. Brown (1995) emphasises that the large quantities of CH_4 involved in this catotelm store are not reflected in the rates of release observed from natural bogs. She therefore argues that this is evidence for a considerable degree of stability in the catotelm CH_4 store.

Brown (1995) observes that individual bubbles may be quite short-lived because they are consumed by methanotrophic microbial populations, but, because new bubbles are constantly forming, the net effect is the maintenance of a large CH_4 store, most of which never leaves the catotelm. However, some carbon must be lost by some route or another because otherwise it is not possible to explain the changes in age-depth curves for bogs, where older material now takes up less volume than younger material. Whether this loss occurs in the form of CH_4 from deep within the catotelm, or whether it is through the steady leaching from the catotelm of organic-matter decomposition products such as DOC, is not yet known.

7.3.6 CH_4 as an energy source

Methane is a breakdown product of organic matter under anaerobic conditions. It is thus generally associated with the catotelm rather than the acrotelm. It is perhaps not widely realised that methane production is also possible in the acrotelm – indeed may be more actively generated in the waterlogged parts of the acrotelm.

Brown (1995) observes that 45% of active methane-producing microbial populations were found to occur in the acrotelm of the Mer Bleue Bog, Canada, and that these acrotelm methanogens were far more active producers of methane than the methanogen population which occurred in the catotelm. She goes on to observe, however, that the low rates of observed methane-release from the Mer Bleue Bog indicate the presence of an equally-vigorous methane-oxidising microbial population in the acrotelm. It is, however, worth noting that limited CH_4 oxidation can occur in the absence of oxygen by instead using electron donors such as nitrate, ferric or sulphate ions.

Svensson and Sundh (1992) point out that methane oxidising organisms require both methane and air, and this combination is to be found most regularly at the seasonal mean position of the bog water table. Consequently the most vigorous methane-oxidising bacteria are indeed likely be found at this level – effectively in the acrotelm – as described by Brown (1995).

Clymo and Pearce (1995) note this highly active layer of methanogens and methane oxidisers within the lower layers of the acrotelm, and highlight the fact that this zone may be no more than 1-2 cm thick. They speculate that the eventual emission of CH_4 from the bog surface may be controlled as much by the actions of the methane oxidisers in this acrotelm zone as by the methanogens in the catotelm and lower acrotelm.

This rapid oxidation of methane in the acrotelm is not a simple chemical reaction between air and CH_4 to produce CO_2 and water, otherwise all methane would simply oxidise as soon as it emerged from the water table and there would be no concerns about methane-release from peat bog systems (or from paddy fields, or cows, or the many other sources of methane currently under widespread debate). Methane is both created and oxidised within the acrotelm as an active process controlled by differing microbial populations which gain energy as a result of their particular roles. Svensson and Sundh (1992) suggest that a close coupling exists between these processes within the acrotelm.

The important point about this active zone of CH_4 production and oxidation is that it is controlled by living microbial populations. The precise nature of these populations has not yet been determined, although Edwards *et al*. (1998) provide some valuable insights into the microbial population structure to be found in the acrotelm and upper catotelm. However, Edwards *et al*. (1998) observe that the microbial population is complex and several species they identify have not been previously described. Consequently the conditions necessary for the long-term maintenance of these populations is not yet known. All that can be reasonably assumed for the moment is that anything which disrupts the natural condition of the acrotelm is also likely to have a significant effect on these microbial populations and their influence on CH_4 flux.

7.3.7 CH_4 oxidation and release of CO_2 to the atmosphere

Under natural conditions, oxidation of CH_4 and release of resultant CO_2 into the atmosphere is very much an activity of the acrotelm. The highly active zone of methanogenesis and CH_4 oxygenation identified by Svensson and Sundh (1992), Brown and Overend (1993), and Brown (1995) within the lower parts of the acrotelm has already been described, but a number of other studies have also investigated the relationship between CH_4 release and surface conditions.

Clymo and Reddaway (1971) present data for methane production at Moor House, in the north Pennines. They found that annual CH_4 production rates for differing parts of the microtopography ranged from 1.3, 5.3 and 9.3 g m^{-2} yr^{-1} in hummocks, lawns and pools respectively.

Clymo and Pearce (1995) study CH_4 emissions at Ellergower Moss, SW Scotland, and note that the efflux of CH_4 from the top of the catotelm into the acrotelm was recorded as 1.1 μmol m^{-2} hr^{-1}. They then note that CH_4 emissions from hummocks were found to be 23 μmol m^{-2} hr^{-1}, while from hollows the value was almost three times as much, at 62 μmol m^{-2} hr^{-1}.

Frenzel and Karofeld (2000) have studied in some detail the CH_4 emissions from a hollow-ridge microtopography on a raised bog in Estonia. They found, firstly, that T3 hummocks (*sensu* Lindsay, Riggall and Burd, 1985) actually consumed CH_4, albeit at a low rate. They identified two forms of hollow, identifiable as the A1 *Sphagnum* hollow and the A2 mud-bottom hollow (*sensu* Lindsay, Riggall and Burd, 1985).

Frenzel and Karofeld (2000) found that CH_4 production in these hollows was very much higher than the observed emissions of CH_4, leading them to conclude that 99% of CH_4 production was oxidised before emission. Fine-scale studies revealed that the extremely effective methane-oxidation layer in the A2 mud-bottom hollows was only 2.5 mm thick. In A1 *Sphagnum* hollows, they found no trace of CH_4 down to the base of the green parts of the *Sphagnum cuspidatum* carpet at a depth of 7 cm, but below this the CH_4 level steadily increased. Frenzel and Karofeld (2000) therefore concluded that CH_4 oxidation was being driven by photosynthetically-derived oxygen, rather than by penetration of air into the *Sphagnum* carpet.

Whalen and Reeburgh (2000) found consumption of atmospheric methane on a 'dry' bog mesotope within the Lamata Bog wetland complex in Alaska, while their 'wet' site (which is more similar to extreme poor fen than truly ombrotrophic bog) emitted 69 mg CH_4 m^{-2} day^{-1}. They estimated that nevertheless 55% of CH_4 produced in the lower parts of the acrotelm of the wet site was oxidised before being released to the atmosphere because observed CH_4 concentrations in these lower parts were much higher than the rate of CH_4 emission. They also observed that their oxidation rates were low compared to a number of other studies which cited 70% - 90% oxidation of available CH_4 before release. Whalen and Reeburgh (2000) noted that their atypically low oxidation estimate was based on only a single observation, and more detailed investigation methods would almost certainly produce a revised estimate.

7.3.8 Direct methane (CH_4) emissions to the atmosphere

It is worth noting at the start that in recent years, measurements of methane emissions from peatlands have often been undertaken by sampling in very considerable detail over relatively short periods of time, either using enclosed chambers or eddy covariance masts. Relatively few studies have been undertaken over entire 12-month periods and thus it is rather unusual to encounter annual estimates for methane emission such as those given by Clymo and Reddaway (1971). In general, figures are now quoted in terms of μmol CH_4 m^{-2} h^{-1}, although as with Whalen and Reeburgh (2000) above, figures will also be found in mg CH_4 m^{-2} day^{-1}, or even mmol CH_4-C m^{-2} day^{-1}. This raises a number of issues:

- the period during which measurements are taken is obviously not typical for conditions throughout the year, and thus the duration and timing of the measurement period are both of some significance;
- measurements per hour obtained from a limited period during the year obviously cannot simply be multiplied-up to create a value for annual methane production because methane emissions are likely to vary throughout the year;
- equally, measurements given for CH_4 production per day cannot simply be divided by 24 to produce equivalent units per hour, because CH_4 production is not uniform throughout the day, being markedly lower at night because the temperature is lower (Hargreaves and Fowler, 1998; Kettunen *et al*., 2000), but sampling from closed chambers is sometimes carried out during daylight hours only;
- comparison between sites is difficult if the sampling periods are not coincident (further complicated by inconsistent use of units);
- biological factors which are present during some parts of the year but not others (such as emergence, or die-back, of aerial leaves above the water surface) may result in 'step changes' in values which cannot be predicted or detected at certain times of the year.

7.3.8.1 Hargreaves and Fowler (1998)

Hargreaves and Fowler (1998) studied the CH_4 emissions at Loch More in Caithness using an eddy covariance mast positioned downwind from a substantial bog-pool system dominated by A3 and A4 deep pools. The area around the mast was divided into sectors and the proportion of pools which occurred in each sector was calculated. The water table was also measured from eight locations within each sector at the start of the 15-day study with a view to relating CH_4 emissions to water level. Unfortunately, although a very considerable amount of technical detail is given for the eddy covariance studies, no information is provided about the way in which the water table was measured.

This is to be regretted because each nanotope element – hummock, high ridge, low ridge – will give a different value for exactly the same overall water table, and these differences may be as much as 20 cm. Furthermore, Goode (1970) found that the weight of the researcher close to a water-level tube or pit can significantly alter the reading obtained, particularly when the peat is soft, as it tends to be in bog-pool systems. He thus constructed a remote-arm measuring device for accurate recording of water tables on the bog pool systems of the Silver Flowe NNR. The absence of details about the way in which the water-level data were obtained by Hargreaves and Fowler (1998), combined with the fact that the water levels were measured only once, is a cause for concern given that the paper is described as quantifying the relationship between water table, temperature and CH_4 emissions. Clearly, therefore, accurate and meaningful measurements of water table would need to be a key part of the study.

Notwithstanding questions over the way in which the hydrology of the site was measured, Hargreaves and Fowler (1998) measure CH_4 emissions over a 15-day period. They then construct a model of CH_4 emissions vs temperature obtained from a location in the same region in order to convert their relatively short sampling period in May into a simulated series of emissions over a 12-month period. From this they produce an annual estimate of 6.9 g CH_4 m^{-2} yr^{-1} for CH_4 emissions from their study site in northern Scotland. They also note that they obtained a close correlation between extent of pools, height of water table, and level of CH_4 emissions.

While in some ways an elegant approach, the model employed by Hargreaves and Fowler (1998) to estimate annual emission should be viewed in the light of certain constraints:

- the relatively short experimental period coincided with the end of an extreme drought period, and thus the observed pattern of CH_4 emissions might not reflect the norm, simply because certain biological thresholds may have been affected (*e.g.* recovery of *Sphagnum* carpets from desiccation);
- use of a weather station located some distance from the actual study site means that local and microclimate effects cannot easily be catered for;
- the combination of drought in early May and the relatively early period of the summer during which the work was undertaken on such a northerly site means that the vegetation may not have been as fully-developed within the growing season as it might have been in other years.

These issues are raised simply as examples of the factors which can influence the results of this or any such survey, but they are issues which must be considered when discussing the results of such work.

7.3.8.2 The ECOSSE Report (2007)

The ECOSSE Report (2007) describes a set of CH_4-emission studies undertaken on three blanket mire sites which have already been described in the present report in terms of bulk-density values. One site (Hafren) was in mid-Wales, a second (Glensaugh) was on the eastern limit of the Grampians near Aberdeen, while the third site (Ullapool) was located on an area of blanket mire some way north of Ullapool.

The ECOSSE Report (2007) found that the site in Wales was a net sink for CH_4, while the two Scottish sites were both CH_4 sources. The site at Ullapool in particular showed a highly variable response, producing some very high emission values for CH_4 over a 12-month period. these values were almost 4x those recorded from the two other sites, and an order of magnitude larger than the site at Hafren. The highest value approached 4,000 μg CH_4 m^{-2} hr^{-1}, while even the yearly average was 1,053 μg CH_4 m^{-2} hr^{-1}, both figures being extraordinarily high compared to other published rates for CH4 emissions from British peat bog sites on an hourly basis.

A closer examination of the Ullapool site (Figure 22) reveals that it is rather complex, with old abandoned peat cuttings, a mineral mound protruding through the peat and from which there will be run-off down into the peat, and a very distinct zone of seepage which can be expected to give rise to poor-fen conditions. Given that fenland systems typically have much higher values for CH_4 emission than do bogs, it is a reasonable question to ask whether in fact the sampled peat for the Ullapool site may have been subject to seepage influences? The ECOSSE Report (2007) does not comment specifically on the very high values obtained for CH_4 emissions at Ullapool.

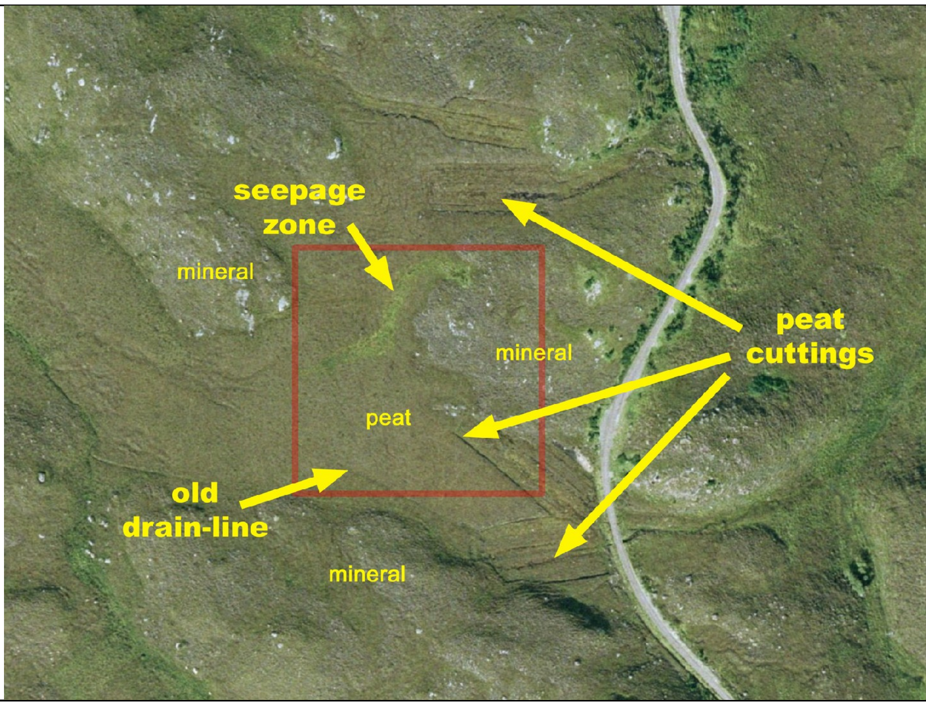


Figure 22. Study area for the ECOSSE Report methane emission studies at Ullapool.
A red square with sides of 100 m in length has been centred on the grid reference given by the ECOSSE Report for its CH_4 study-site north of Ullapool. It can be seen that the ground here is rather complex, with mineral ground protruding through the peat, old peat cutting banks, a old drain line, and a clear zone of seepage close to the given grid reference.

Aerial photograph © getmapping.com

7.3.8.3 CH_4, microtopography and vegetation

What becomes very evident when working through the literature describing CH_4-emission studies is that data tend to fall into two broad groups. There are those studies, often regional overviews, which provide single emission figures for sites, and there are those which investigate CH_4 emissions at the small scale on particular sites, linking emission values to individual parts of the microtopography. These smaller-scale studies often reveal substantial differences in emission rates between features which may be separated by no more than a metre or so. Holden (2005a) emphasises the importance of recognising such small-scale variation and its significance in building up a broader picture of peatland site processes. It is revealing to bring together a number of published values for CH_4 emissions from different sites, particularly where the different elements of the microtopography have been distinguished. This information is set out in Table 14.

It is clear from Table 14 that hummocks emit much less CH_4 than hollows, often by a factor of 10. Indeed hummocks are clearly capable of absorbing CH_4 from the atmosphere under some circumstances. This pattern accords with the well-established general relationship whereby higher water levels in peatlands give rise to exponentially higher levels of CH_4 emission (Martikainen *et al*., 1995; Laine and Vasander, 1996). Curiously, however, the level of CH_4 emission from the 'pool' noted by MacDonald *et al*. (1998) is lower even than the rates recorded from the hummock zone. If high water tables give rise to high CH_4 emission rates, one might expect the pool nanotope to produce the highest emission levels of all. This is clearly not the case. Consequently the relationship established by Hargreaves and Fowler (1998) between pool extent, water table depth and methane emissions perhaps requires closer examination.

Table 14. Range of methane (CH_4) emission rates from differing bog sites.
Figures for CH_4 emission are given for whole sites (or regional areas), or for individual features of the microtopography within sites. All sites are raised or blanket bog. The column for 'vegetation' lists the CH_4 emission rates recorded for individual plant stems or groups of stems for particular plant species – bog bean (*Menyanthes trifoliata*), or hare's tail cotton grass (*Eriophorum vaginatum*). Negative values mean that CH_4 has been absorbed rather than released.

Author	Location	CH_4 emissions µmol CH_4 m^{-2} hr^{-1}				
		Site	Hummock	Hollow	Pool	Vegetation
Frenzel & Karofeld (2000)	Männikjärve Bog, Estonia		-1.3	16.2	20	300
MacDonald *et al.* (1998)	Loch More, Caithness		17.5	128.8	14.5	176.6
Clymo and Pearce (1995)	Ellergower Moss, SW Scotland		23	62		
Fowler *et al.* (1995)	Loch More, Caithness	14.7				
Beverland *et al.* (1996)	Loch More, Caithness	23 (-70 to +120)				
Hargreaves & Fowler (1998)	Loch More, Caithness	39				
Beswick *et al.* (1998)	Northern Scotland	48				
Watson & Nedwell (1998)	Ellergower Moss, Dumfries		0.336*	2.3*		
Watson & Nedwell (1998)	Great Dun Fell, Cumbria		-1.28*	3.14*		

* rates expressed as mmol C m-2 day-1

The explanation is simple, and the clue lies in the very high CH_4 emission values given for 'vegetation'. A deep, permanent A4 pool contains little but water. The pool base may be 3 or 4 metres below the pool surface but is generally devoid of vegetation. The upper column of water is oxygenated, and thus any CH_4 released from the anaerobic peat base has the opportunity to be oxidised before reaching the pool surface unless this occurs as large-scale ebullition (bubble formation), which happens relatively rarely. Consequently an A4 pool, though by definition having a high water table, is not necessarily a good source of CH_4 emissions.

Where a pool or a hollow contains a stand of bog bean (*Menyanthes trifoliata*), however, the aerenchyma tissue within the bog bean can act as a rapid-transport system enabling CH_4 to travel from the anaerobic peat to the atmosphere without passing through a microbial oxidising zone. In hollow-stemmed species such as common reed (*Phragmites australis*) this is often referred to as a 'shunt', whereas species with gas-transport systems using diffusion such as bog bean, are better regarded as 'methane-transport species'. MacDonald *et al*. (1998) illustrate the vascular structures within bog bean which permit the species to act in this way.

Frenzel and Karofeld (2000) demonstrate the effectiveness of methane-transport species even more clearly by switching the system off. They noted very high levels of CH_4 emissions from A1 and A2 hollows when stands of Rannoch rush (*Scheuchzeria palustris*) or hare's-tail cotton grass (*Eriophorum vaginatum*) were present, and identified these stands as the source of such emissions. Frenzel and Karofeld (2000) therefore cut the aerial parts off down to ground level and found an immediate and dramatic drop in CH_4 emissions down to levels of only 16 µmol CH_4 m^{-2} hr^{-1} compared to the levels of 300 µmol CH_4 m^{-2} hr^{-1} recorded prior to cutting the vegetation. In other words, the *presence* of only a few stands of such vegetation can increase CH_4 emissions by more than 200x.

Such results highlight the importance of Weber's (1902) observations about the growth-forms of vascular plants such as hare's tail cotton grass (*Eriophorum vaginatum*) and deer grass (*Trichophorum cespitosum*). Where these species form dense tussocks within damaged, former areas of hollow, they may represent large methane sources, whereas loose, scattered swards of such species within an undamaged terrestrial zone may be much less significant as methane emitters.

The very low emissions of CH_4 observed by Dinsmore *et al.* (in press) for Auchencorth Moss, mentioned earlier in Section 7.3.2.2, may reflect the preponderance of terrestrial *Sphagnum* species and the virtual absence of water-filled hollows or pools on Auchencorth Moss. The site thus contrasts significantly with the pool-patterned system at Loch More, for example.

7.3.8.4 Gray (2005)

This PhD study provides an excellent, critical review of peat and methane emissions published to that point, specifically looking at the question of methane fluxes in blanket mire systems. Gray (2005) brings together the most comprehensive synthesis of greenhouse-gas emission studies published to date, and highlights a number of important issues:

- that estimates of carbon stores must take account of microtopography because the presence of pools, for example, reduces the store – consequently carbon estimates which ignore pool systems will tend to be over-estimates;while water table has been shown to be strongly linked to CH_4 production and it may therefore be expected that vegetation patterns would provide a useful proxy for CH_4 release, no single consistent system for defining the spatial scales or descriptive nature of bog vegetation has been applied by the various studies reviewed;
- only eight CO_2 studies relevant to British bogs were found by Gray (2005), involving six sites, and of these six sites, four were blanket mire and two were raised mire – with Loch More (Caithness blanket mire) and Ellergower Moss (Dumfries & Galloway raised mire) supporting more than one study;while 19 studies of CH_4 relevant to British bogs were found by Gray (2005), only nine blanket mire sites, one raised mire, and a soligenous (fen) mire were studied, meaning that a significant proportion of evidence comes from repeated studies on a relatively limited number of sites – with again Loch More (Caithness blanket mire) and Ellergower Moss (Dumfries & Galloway raised mire) featuring heavily;
- using the ‘sink/source’ model devised by Whiting and Chanton (2001), Gray (2005) tentatively suggests that the existing evidence points to British bog systems being at worst ‘greenhouse neutral’ but more probably ‘greenhouse friendly’ as it seems that they are generally sinks for carbon, the tentative nature of his conclusions arising from the extremely limited number of studies and sites involved, particularly in relation to CO_2 flux – indeed for blanket mire, Gray (2005) concludes that there is only a single study which gives field measurements of CO_2 flux (Hargreaves *et al.*, 2003), although Gray (2005) highlights the fact that even this study relies on modelled climate data from some considerable distance form the site;
- Gray (2005) questions whether laboratory-based studies of greenhouse-gas flux can really describe what happens under natural conditions in the field, and thus suggests that figures of greenhouse-gas flux obtained in this way would be treated with the utmost caution – a point also made by Brandyk *et al.* (2002) who express similar concerns about laboratory measurements of hydraulic conductivity in peat;
- even where regional studies have been undertaken, as in the case of Beswick *et al.* (1998), Gray (2005) points out that these have been of extremely short duration and are therefore hardly representative of the whole annual picture, particularly as Gray (2005) highlights the fact that the few existing ground-based studies have rarely measured greenhouse-gas flux during the winter months;
- Gray (2005) also highlights the fact that remarkably little information is given about the management of these various study sites (in effect, the concept of management here includes the ‘condition’ of the sites) and thus it is difficult to correlate the published greenhouse-gas emission values with the character, land-use pattern and condition of the ground involved – indeed Gray (2005) questions the use of terms such as ‘pristine’ used in certain studies because he believes (quite justifiably) that most, if not all, UK bogs have a history of significant human impact;
- Gray (2005) explains very clearly the way in which the limited number of studies into both CO_2 and CH_4 flux from UK peatland systems means that for the purposes of UK carbon reporting it has been necessary, though highly unsatisfactory, to extrapolate from small study areas to whole countries, and from a few days’ measurements to the whole span of seasonal conditions prevailing over a year.

Given the context outlined above, Gray (2005) therefore undertook a range of field studies on blanket mires in two differing geographical regions in order to provide a series of greenhouse-gas emission values for blanket mires which have experienced differing land-use histories and thus display differing levels of condition. The sites consist of Hard Hill within Moor House National Nature Reserve in the north Pennines, and a series of blanket mire systems based around Forsinard in Sutherland. A long-established multi-plot, multi-treatment experimental site at Hard Hill was used as the basis of the work in the Pennines, while nine areas of blanket mire, each subject to differing management practices, were selected in Sutherland.

Gray (2005) first attempted to establish a link between land management (or mire condition) and the National Vegetation Classification (NVC) by undertaking a detailed survey and analysis of the vegetation found at each site or study plot. The most significant thing to come out of this particular component of the study was that it proved impossible to distinguish management/condition regimes on the basis of the NVC. Gray (2005) concludes that:

> *"...the mire NVC may be insensitive to difference in management (at least at the Hard Hill site) even when clear statistical evidence of treatment effects on species composition is present."*
>
> Gray (2005), p.122

The failure of the NVC to make such distinctions at Forsinard as well gives rise to the following observation:

> *"As the NVC does not differentiate the Hard Hill treatments or the Forsinard sites, the NVC classifications also offer little in gauging how representative these sites are in terms of management. Given the uncertainties of the geographical spread of blanket bog management, the insensitivity of the NVC to differences in management is unfortunate since it may have allowed indications of management on a wider scale."*
>
> Gray (2005), pp. 124-125

This discovery and conclusion are of the utmost importance for most work (whether it be scientific research or various forms of land management) currently undertaken on UK peat bog systems. It lies at the heart of much that is apparently confusing or contradictory about the current understanding of such systems and how they work. The NVC for peat bog habitats, as currently constructed, is often too coarse a tool by itself to distinguish between differing conditions of bog habitat even when there is clear statistical evidence that vegetation stands differ significantly from each other. In other words, the differences are there but the NVC is unable to see them.

The NVC looks *only* at vegetation. Indeed Rodwell (1991) states explicitly in the introduction to the NVC Mires volume that features such as microtopography were not included in the construction of the system. It is not possible to separate out the vegetation from the microtopography in a bog system. Certainly if this were done for the bogs of Tierra del Fuego they would all be assigned to virtually a single vegetation type because they consists almost entirely of *Sphagnum magellanicum* monocultures, yet these monocultures display the same diversity of features and response to management impacts as do the bogs of the northern hemisphere. The surface *structure* of a bog – microtope and nanotope – is as important as the vegetation composition because in such species-poor environments it is these features which provide the key to difference, whether these differences be from natural causes of from land-management practices.

It is therefore to be much regretted that Gray (2005), though carrying out extremely detailed analyses of *vegetation* structure, did not also devote time and effort to the structures associated with the hydro-morphological hierarchy of mesotope-microtope-nanotope because such effort would have paid rich dividends. Indeed the absence of such attention significantly limits the analysis and interpretation of Gray's (2005) field experiments.

The assessment of CO_2 and CH_4 emissions was carried out in Sutherland, firstly on a sub-set of the study sites to establish whether any clear relationship could be identified between climatic variables and gas flux, then all nine study sites were used to determine whether gas fluxes were influenced by land management/bog condition. Static chambers were used because they give a direct measurement of the gas flux from specific locations, whereas Gray (2005) observes that gas-flux measurements based on atmospheric sampling using an eddy covariance mast can only give an overall picture of the whole upwind sampling area. Thus the spatial resolution, in terms of gas-exchange from specific vegetation and surface features, is very much greater when using static chambers compared to what is possible from eddy covariance measurements.

The vegetation of the study plots as a whole are described and analysed, but unfortunately, having opted to use static chambers and specifically referred to the fine-scale resolution in terms of vegetation which this approach brings, Gray (2005) then provides no systematic information about either the vegetation or the microtopography actually enclosed *within* these chambers. In passing, he mentions that one 'plot' consisted of a *Racomitrium lanuginosum* hummock, and a photograph is given of another couple of chambers dominated by heather (*Calluna vulgaris*).

A canoncial correspondence analysis (CCA) plot given by Gray (2005) shows various trends plotted against vegetation composition. This may be evidence of variation *within* the chambers themselves, or it may instead be a vegetation description of the overall homogeneous stand within which the chambers were placed. It is impossible to say.

This lack of information about the detailed nature of the bog surface being sampled is unfortunate, given that several previous studies had already identified the fact that certain vascular plants are vigorous emitters of CH_4 while nanotope elements such as T3 hummocks can actually reduce CH_4 emissions. The descriptive approach used results in the absence of critical information needed to evaluate the gas-flux data.

There are other methodological issues. Gray (2005) found that the highest emissions of CH_4 were obtained from the 'relatively intact' sites of Maol Donn and Sletill, averaging 85.71 and 34.56 mg CH_4 m^{-2} day^{-1} respectively. These are very high values, with the Maol Donn value being more than 4x the values normally associated with general blanket bog areas. Several points should be borne in mind when considering these results:

- These two sites were shown by penetrometer readings to possess the softest peat. Gray (2005) did not use boardwalk when sampling the chambers; in fact he specifically mentions the damage done to the ground around the chambers. Such soft peats are most likely to undergo compression beneath the weight of the observer and such compression can force CH_4 from the peat matrix. Although Gray (2005) states that pulses can be detected by comparing measurements over time, CH_4 was measured by filling a flask at 10-minute intervals. Each time a flask was used the peat will have been compressed and therefore every reading may be associated with CH_4 release. It is perhaps not a coincidence that the highest CH_4 readings were obtained from the softest site.
- The Maol Donn and Sletill sites, being relatively intact and softest, may also possess higher proportions of species which act as a CH_4-transport, but it is impossible to say whether or not this is the case because information given about the vegetation in relation to the chambers themselves is ambiguous.
- At the vegetation level, a canonical correspondence analysis (CCA) of the recorded species plotted in relation to the key axes indicates a close correlation between CH_4 flux and species such as *Sphagnum cuspidatum* and common cotton grass (*Eriophorum angustifolium*), the former indicating the presence of A1 hollows within which the latter can act as a significant CH_4-shunt. The relative abundance of these species in the various chambers is not, however, made clear.
- The presence or absence of particular nanotope elements within the chambers is likely to influence significantly the levels of CH_4 flux, as discussed above in Section 7.3.8.3. Gray (2005) mentions the issue in relation to volume calculations and the hummock-hollow nature of bog surfaces, but provides no information about the nanotopes specifically sampled within the chambers.
- Meanwhile at the mesotope level, the Maol Donn site is described by Gray (2005) as 'relatively natural', but close examination of the site using aerial photographs reveals that this is not really the case. Figure 23 shows that large proportions of the site margins have been ploughed for forestry, presumably during the private forestry boom of the 1980s. The trees now appear to have been removed and it is likely that the ploughing furrows have been blocked (though this is not clear from the aerial photograph). Clearly the mesotope has undergone some fairly significant impacts here. How the ploughing and then (possible) blocking of the ploughing furrows may affect CH_4 fluxes is not entirely clear, but it is possible that they are temporarily elevated because the site is re-wetting. A map of the mesotope main features would have highlighted the presence of such features.

- At the microtope level, Figure 23 also shows that the 100 m position given by Gray (2005) for Maol Donn lies in the centre of a very large ‘soakway’ flush system which flows from the bog to the north. GPS positions to 12-figure accuracy obtainable form a GPS unit would provide confidence that none of the sample plots actually lay in this flushed system, but if they did then the higher CH_4 emissions for this site would be entirely consistent with such a zone. A map of the microtope patterns would have helped to ensure that this zone was indeed avoided.

Thus while Gray (2005) provides a valuable review of greenhouse-gas flux literature and highlights a number of key gaps in existing knowledge and concerns about certain methodological issues, he himself introduces additional levels of confusion and uncertainty in relation to his own field methodology. Whilst the levels of CH_4 emissions he noted are unusually high, there may in fact be fairly simple methodological reasons why this is so.

7.3.8.5 Laine, Wilson, Kelly and Byrne (2007) – an exemplary study

In one of the most detailed investigations so far undertaken into blanket mire surface patterns, vegetation and methane emission, Laine, Wilson, Kiely and Byrne (2007) studied the rates of CH_4 emission from an area of blanket mire in Co. Kerry, Ireland. Glencar bog possesses a range of nanotopes described by Laine *et al.* (2007) as hummocks, high lawn, low lawn and hollows. From the descriptions provided they appear to be equivalent to Lindsay, Riggall and Burd’s (1985) T3 hummocks, T2 high ridge, T1 low ridge and either A2 mud-bottom hollows or A3 drought-sensitive pools.

Laine *et al*. (2007) took regular readings of methane emissions from each distinct nanotope over a period of a year, with some plots from each nanotope being sampled for a further 12 months. Boardwalk was used to prevent disturbance to the peat, and water levels in perforated pipes were sampled from the boardwalk whenever the CH_4 samples were taken. Soil temperature at 20 cm depth was found to provide a good reflection of a wide range of temperature readings obtained from different peat depths.

Perhaps most significantly, Laine *et al.* (2007) also calculated an index for the leaf area of vascular plants (VGA_{AER}) known to be capable of transporting CH_4 through their tissues. This VGA_{AER} included species such as common cotton grass (*Eriophorum angustifolium*), hare’s-tail cotton grass (*E. vaginatum*), bog bean (*Menyanthes trifoliata*) and deer grass (*Trichophorum cespitosum*).

Overall, Laine *et al.* (2007) observed an exponential relationship between water table and CH_4 emissions. They found that CH_4 emissions were lowest and showed least variability in the HU/T3 hummocks where the median water table was more than 10 cm below the hummock surface. The high and low lawns (HL/T2, LL/T1) emitted more CH_4 than the hummocks, and displayed substantially more variation, with the low lawn (LL/T1) having the higher variation.

The hollows showed a dramatic differentiation between hollows containing few methane-transporting vascular plants (*i.e*. having a low VGA_{AER}) and those possessing a substantial plant community. Hollows with a low VGA_{AER} emitted CH_4 at the same rate as hummocks, whereas hollows with a high VGA_{AER} had CH_4 emissions which were almost 5x times higher.

Interestingly, Laine *et al*. (2007) did not find any relationship between VGA_{AER} in the terrestrial parts of the microtope. If anything, lawns (T2/T1) appear to be slightly negatively correlated, though not significantly so. It would thus appear that methane-transporting vascular plants have very little part to play in the release of CH_4 from terrestrial zones.

A very strong and significant correlation between VGA_{AER} and hollows, on the other hand, emphasises the major part played by vascular plants in the release of CH_4 from hollows. The hollows on this site are described as having only a sparse cover of *Sphagnum*, but Laine *et al.* (2007) note that Frenzel and Karofeld (2000) observed a significant decrease in CH_4 emissions when hollows possessed a cover of *Sphagnum*.

The values of CH_4 emission recorded by Laine *et al.* (2007) for the various elements of the microtopography can be seen in Table 15. Both daily and annual emission estimates are provided, based on regression models constructed from the range of data obtained during the three calendar years of the study.

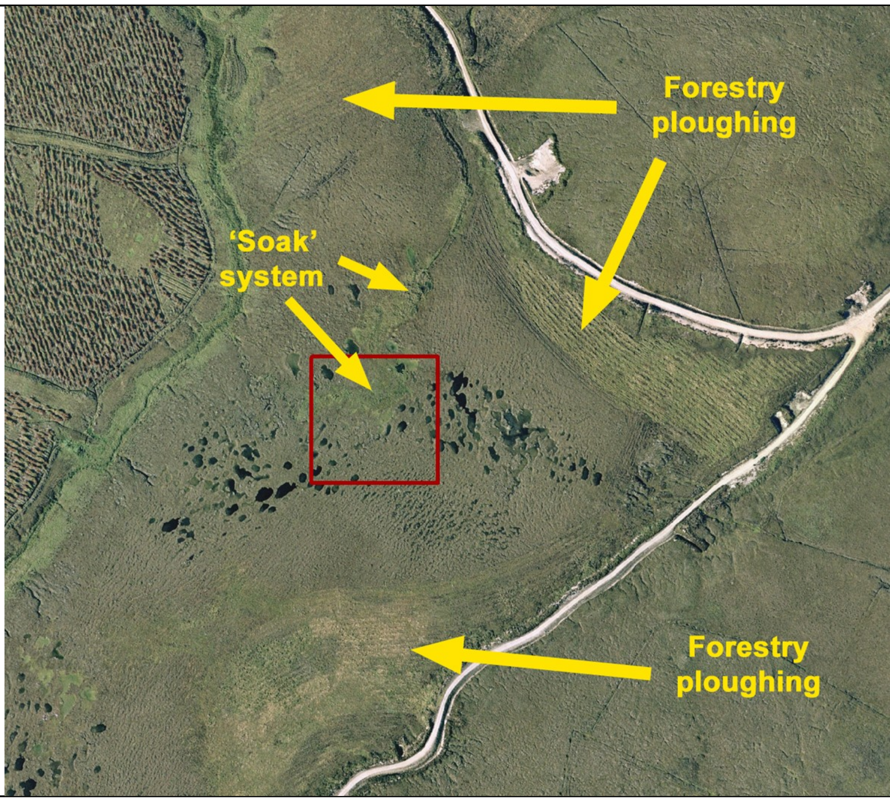


Figure 23. Maol Donn gas-flux study site, Sutherland used by Gray (2005).
The patterned blanket mire at Maol Donn lies to the left of the two white roadways. Areas of former forestry ploughing within the Maol Donn mesotope are indicated by the large orange arrows. The 100 m square given as the location of the site is shown in dark red. Smaller orange arrows point to the green 'soakway' flush system which occupies a large proportion of the red 100 m 'sample site' square before running off-site to the north.
Aerial photograph © getmapping.com

Table 15. Methane emissions estimated by Laine *et al.* (2007) from differing parts of the microtopography of Glencar blanket bog, Co. Kerry, Ireland.

Nanotope	**Annual CH_4 emissions ($g\ CH_4\ m^{-2}\ yr^{-1}$)**	**Daily CH_4 emissions ($mg\ CH_4\ m^{-2}\ day^{-1}$)**		**Median water level (cm)**
		Average	**Range**	
HU (T3 hummock)	3.3	11.8	0.1 – 64.1	-13
HL (T2 high ridge	5.8	19.2	0.0 – 72.2	-5
LL (T1 low ridge)	6.1	20.9	0.1 – 101.4	-1
HO1 (non-vegetated A2 hollow)	3.5	11.6	1.7 – 31.8	3
HO2-3 (vegetated A2 hollows)	13.0	50.4	0.3 – 263.0	5

It can be seen from Table 15 that the estimated annual CH_4 emissions from all except vegetated hollows are quite low, ranging from 3.3-6.1 g CH_4 m^{-2} yr^{-1}. This is somewhat less than the 6.9 g CH_4 m^{-2} yr^{-1} recorded by Hargreaves and Fowler (1998). Vegetated hollows, on the other hand, emit an estimated

13 g CH_4 m^{-2} yr^{-1}, which is more than double the rate for other elements of the microtopography. Clearly, the extent of hollows and pools vegetated with methane-transporting vascular plants is a key factor in determining the likely emission rates from a bog.

Daily emission rates in Table 15 are somewhat similar to values noted by MacDonald *et al*. (1998) for hummocks (6.72 mg CH_4 m^{-2} day^{-1}), un-vegetated pools (5.57 mg CH_4 m^{-2} day^{-1}), and pools or hollows with vascular plants (49.5 – 67.4 mg CH_4 m^{-2} day^{-1}), assuming a simplistic multiplication from hours to days, which is not entirely valid. Indeed one thing which Table 15 makes clear is that simple multiplication of daily CH_4 emission rates does not provide an annual rate of emissions.

7.3.8.6 CH_4, Pennine blanket mires and hare's-tail cotton grass

This then raises interesting questions about the blanket mires of the Pennines, where, as Rodwell (1991) describes, vast tracts of the blanket mire landscape is dominated by tussocks of hares' tail cotton grass (*Eriophorum vaginatum*) – almost certainly not because this is the natural vegetation cover, but because a history of burning and atmospheric pollution has rendered it all-but impossible for *Sphagnum* species to survive in any quantity.

In the absence of such fire damage and pollution, it would be very interesting to see whether a resurgent *Sphagnum*-dominated community would result in significant reductions in the levels of methane currently released from Pennine blanket peats.

If hares' tail cotton grass (*Eriophorum vaginatum*) in former *Sphagnum* hollows were to act as a significant shunt, then this could release large quantities of CH_4 from the Pennine moorlands. It appears, however, that while growth of hares' tail cotton grass in A1 hollows provides a substantial CH_4 shunt, when the species grows in terrestrial T-zones this shunt mechanism may not operate. It is not clear why this should be, unless it is because the root systems are not in standing water. The same question arises in relation to deer grass (*Trichophorum cespitosum*), which acted as a shunt in A1 hollows for Frenzel and Karofeld (2000). In the north and west of Britain deer grass is one of the most ubiquitous species on blanket mires, and forms tussocks when the ground is damaged. Does it act as a shunt or not in such circumstances?

Indeed the relationship between water table and CH_4 emissions described by Martikainen *et al.* (1995) and Laine and Vasander (1996) referred to above is probably not a wholly-direct relationship at all. It may instead reflect the pattern of vegetation development in peatland systems whereby peatlands with higher water levels, such as fen swamps, have a much greater proportion of aerenchyma-rich vegetation capable of providing a CH_4 shunt or CH_4-transport system. For example, Ding *et al*. (2003) have investigated in some detail the way in which tall-sedge fen provides a very considerable CH_4 shunt within a Chinese fen swamp, indicating that it is through such systems that fens release substantially more CH_4 than the emissions form bogs.

The CH_4-transport system is not the entire story, of course, because fen swamps also tend to contain material which is more easily digested by anaerobic microbial populations than peat rich in *Sphagnum*. Consequently the story of CH_4 emissions from blanket mire systems is closely bound up with:

- the proportion of bog peat derived from *Sphagnum*, thereby rendering the peat less 'palatable' to methanogenic microbial populations;
- the proportion of the bog surface covered by living *Sphagnum*, thereby providing an actively-oxidising zone as CH_4 moves towards the bog surface;
- the proportion of the bog surface dominated by 'T' zones rather than 'A' zones (*sensu* Lindsay, Riggall and Burd, 1985) because *Sphagnum*-rich T zones even have the potential to sequester CH_4 from the atmosphere, while A zones have higher water tables and thus shallower active-oxidation layers, plus an increased proportion of aerenchyma-rich species such as bog bean (*Menyanthes trifoliata*);
- the vegetation condition of any A-zone features, because hollows dominated by a vigorous *Sphagnum* carpet still produce relatively small quantities of CH_4, while deeper pools are not large emitters of CH_4 unless they support stands of bog bean (*Menyanthes trifoliata*);

- the abundance of aerenchyma-rich species capable of providing a CH4 shunt, including species such as bog bean (*Menyanthes trifoliata*), hare's-tail cotton grass (*Eriophorum vaginatum*) – particularly in its tussock form, deer grass (*Trichophorum cespitosum*) – again particularly in its tussock form, and common cotton grass (*Eriophorum angustifolium*) – which has no tussock form, but can grow in extensive open swards.

7.3.9 Carbon-exchange pathways – water-borne carbon (DOC, POC)

Not all carbon exchange on a peat bog occurs in gaseous form. A sometimes very substantial quantity of carbon is also lost as dissolved and particulate matter carried out of the system along watercourses. This, of course, is part of the origin for the catchment-centred approach adopted by mass-balance studies.

While a certain amount of dissolved inorganic carbon (DIC) enters a peat bog as dilute carbonic acid in rainfall and another small proportion enters as dissolved organic carbon, the major movements of carbon in solution or as particles occur as losses to the peatland system in the form of dissolved organic carbon (DOC) and particulate organic carbon (POC). In reality not all DOC is actually dissolved. It occurs as a complex mixture of organic chemical species, some of which are in solution and some of which occur as colloids, and has defied current attempts to identify the chemical nature of the components. It is not even known precisely how DOC is formed. All that is known is that water seeping from peat brings with it this jumble of organic material in a form which often colours the water yellow or brown.

In terms of carbon balance, figures obtained for LARCA, RERCA and ARCA have implicitly accounted for losses via DOC or POC because the necessary measurements of peat thickness or accumulated mass are made on the body of peat which, as it were, remains after all such losses have occurred. Thus although such losses have been taken into account, they have not been explicitly visible in such calculations.

It is only now that such losses have become so very visible in the everyday world – because drinking water has changed colour, and many new reservoirs are having to be built – that attention has become focused on these forms of carbon loss from peatlands.

7.3.9.1 Particulate organic carbon (POC)

Particulate organic matter (POC) differs from DOC mainly in the size of the particles involved; there is in fact a continuum from DOC to POC and the two are only separated arbitrarily by the size of filter needed to trap their un-dissolved matter (Worrall *et al*. 2003a). One major difference between POC and DOC is that POC can be also removed by the action of wind as well as water, particularly in areas of eroded peatland where exposed faces of bare peat can dry out to a loose powder under the combined effects of sun and wind.

Indeed POC is generally a feature of peatland systems where there has been some form of damage which exposes bare catotelm peat, or where erosion has taken hold across the site. High levels of POC have been a problem for the Finnish peat mining industry for decades. POC outputs from natural, undamaged, non-eroding peat bogs, on the other hand, tend to be very low. This means, however, that POC is a major factor for many British blanket mire landscapes because erosion is such a widespread feature of the British upland scene (Osvald, 1949; Taylor, 1983; Tallis Meade and Hulme, 1997; Evans and Warburton, 2007).

POC is also a major source of concern to water utility companies because the accumulation of POC as sediment in reservoirs can substantially reduce the storage capacity of a reservoir, and can also cause major clogging of the outlet flumes. It has been estimated that many reservoirs in Britain have lost at least 1/3 of their water-storage capacity because of accumulated POC sediments, while White, Labadz and Butcher (1996) have estimated that Yorkshire Water has been obliged to spend £74 million in constructing new reservoirs to compensate for loss of capacity in existing reservoirs.

7.3.9.2 Dissolved organic carbon (DOC)

Increases in water colour have become the symbol of a major problem for water utility companies in recent years. DOC and water colour have become the subject of increasingly strict European Union regulation, while consumer concerns about the perceived quality of drinking water if it is not completely clear have forced water utility companies to invest heavily in expensive water treatment and in funding research to identify the possible causes of DOC and associated colour increases (Glaves and Haycock, 2005). Freeman *et al*. (2001) highlighted an observed 65% rise in DOC concentrations over a 12-year period in waters draining peat-dominated catchments within the UK.

Other studies have shown similar trends in recent years (e.g. Worrall, Burt and Adamson, 2004; Worrall and Burt, 2005), although there appears to be little evidence for levels of DOC in stream waters extending back to the inter-war years, for example. Consequently it is difficult to know just how long this has been an issue. There is a danger that because records exist back to the mid-1960s and increases in DOC have been shown since then, the problem is seen as a recent phenomenon. This may not be the case.

The fundamental difficulty with DOC is that its origin has yet to be identified, although Wallage *et al*. (2006) provide a valuable summary of what is known. McDonnell *et al*. (2001) observe that:

> *"Humic substances are so complex that their mechanisms of production and even their definition are still causing speculation."*

Ground-conditions leading to increased levels of DOC have been identified in a number of cases (White *et al*., 2004), but the actual processes by which DOC is created are still subject to considerable speculation. Freeman *et al*. (2001) attribute their observed rise in DOC to increasing global temperatures, and demonstrate that the activity of the enzyme phenol oxidase, which is associated with carbon loss from peat, can be stimulated by increasing temperatures. Unfortunately they did not also provide details of what the 'peat soil' used in such tests consisted of – was it highly humified and rich in vascular-plant tissue, or was it *Sphagnum*-rich peat?

Worrall, Burt and Adamson (2004) pursue the question of DOC and temperature further and suggest that increased temperatures alone cannot explain the observed levels of DOC increase. They acknowledge that changes in land management may play a major part in controlling DOC release, but also present the idea of an 'enzyme latch' mechanism based around much the same phenyl oxidases proposed by Freeman *et al*. (2001). It is suggested that water-table draw-down causes breakdown of the phenolic compounds which normally make peat so resistant to decay. Without the protection of these phenols, peat decomposes to produce DOC until such time as new phenolic compounds have been formed after a period of some 3-4 years. The idea is plausible, but remains unproven.

Worrall and Burt (2005) then attempt to quantify the rate at which DOC will be released from peat-covered catchments up to 2010, based on past trends, detailed modelling of acrotelm and enzyme-latch processes, and predicted climate patterns. They predict an increase in DOC from peatland areas of 0.48 tonnes C km^{-2} of peatland area yr^{-1}. On this basis, Worrall and Burt (2005) suggest that the DOC flux from the UK might be as high as 1Mt C by 2010.

A different perspective on the DOC debate is provided by Monteith *et al.* (2007). Following a study of DOC levels across a large proportion of the Boreal Region both in Europe and North America, they propose that observed increases in DOC can be wholly explained by recent decreases in atmospheric sulphate (SO_4^{2-}) pollution levels or, in certain coastal regions, reduced chloride levels as a result of reduced oceanic storm events. Monteith *et al*. (2007) thus propose that the current period of increased DOC represents a period of recovery to pre-industrial levels of DOC. Of course it is not easy to prove (or disprove) this last point because there are no records of DOC from pre-industrial times, although some evidence available from central Europe offers strong corroborative evidence from at least one site (D. Monteith, pers. comm.). Perhaps some proxy record of pre-industrial DOC exists, but if it does it has not been found yet.

The significance of the explanation proposed by Monteith *et al*. (2007) is that blanket mire systems are currently undergoing change while they emerge from the stresses imposed by high levels of sulphate deposition. If this does indeed represent a return to DOC levels present prior to industrialisation, it might be expected that the rise in DOC will eventually plateau at pre-industrial levels. Indeed it is possible to speculate that if high sulphate levels have removed the *Sphagnum* cover from the Pennine blanket peats, a decrease in these sulphate levels might see a return of this *Sphagnum* cover at the expense of the vascular plants which dominate the present vegetation cover. If DOC outputs from

Sphagnum-rich vegetation are low, then ultimately the current rise in DOC might not merely plateau, but then fall to a new level as *Sphagnum* becomes re-established.

If, instead, the rise in DOC is linked to rising temperatures as suggested by Freeman *et al*. (2001), there will be no evidence of a plateau and no subsequent fall – DOC will simply continue to rise with rising temperatures.

Worrall (pers. comm.) accepts that Monteith *et al.* (2007) may have correctly identified part of the explanation for rising DOC levels but does not feel that this is the whole story, particularly as Monteith *et al*. (2007) themselves identify that the picture in the UK is somewhat anomalous compared to the findings of the study from elsewhere. Perhaps a closer consideration of the processes of decomposition and the botanical composition of material subject to such decomposition can shed further light on the question.

We have already see in Section 7.1.2 above that *Sphagnum* has particularly effective methods of rendering itself unpalatable to decomposer micro-organisms. We have also learned that the majority of material produced by vascular plants is decomposed in the acrotelm, leaving *Sphagnum* as the dominant material to be passed down into the catotelm. During the decomposition of this vascular-plant material, the breakdown products may be released as gas, or as complex organic decomposition products which contribute to the low hydraulic conductivity of the lower acrotelm (and possibly the catotelm). It is possible that these complex organic breakdown products are the primary source of DOC.

If Fontaine *et al*. (2007) are correct in their observations that, in the absence of younger organic matter to activate decomposer populations, old soil organic matter (SOM) tends to be highly resistant to decay, it may be that the surface layers of a bog are the primary sources of DOC because the acrotelm (or haplotelm surface) contains the bulk of the fresh organic matter. Acknowledging that DOC is a breakdown product of plant material, it is possible to see from Clymo (1983) that figures for the decomposition rates of material such as cotton grass leaves (*Eriophorum spp.*) are significantly higher than for the decomposition rates of *Sphagnum* plants, while the larger-leafed material of species such as bog asphodel (*Narthecium ossifragum*) or cloudberry (*Rubus chamaemorus*) may decay 4-5x more rapidly than *Sphagnum*.

Wallén (1992) highlights the fact that in the two ombrotrophic mires he studied, the majority of vascular plant material had decayed by the time it reached the transition between acrotelm and catotelm. Only *Sphagnum* and cotton grass roots (*Eriophorum spp.*) remained in any quantity to be passed down into the catotelm. As noted earlier in Section 7.1.2 of the present report, Wallén (1992) observed that, in his sites, 99.5% of the material passed down to the catotelm consisted of *Sphagnum*, whereas 770 g m^{-2} of the 800 g m^{-2} annual vegetation production were lost each year through breakdown into gas or soluble carbon-rich material.

There is thus strong evidence pointing to the acrotelm as the primary site of breakdown products within the peat column. In addition, Worrall *et al*. (2006a) observe that, in the Trout Beck catchment of Moor House National Nature Reserve, DOC production is most likely derived from the uppermost layers of the peat soil. Specifically, they note that DOC appears to be almost exclusively derived from the uppermost 10 cm of the soil, and that this does not change substantially even following extended periods of drought.

This suggests that DOC production may be largely confined to the surface layers of peat which are rich in vascular-plant matter. On a natural site this surface layer may be an acrotelm in which freshly-dead vascular material undergoes breakdown to be released as CO_2 and DOC. Alternatively, on a damaged site this surface layer may be a haplotelm where fresh litter from the vegetation provides part of the DOC output, with the remainder coming from catotelm peat which has been penetrated by the now more vigorous root systems of vascular plants such as heather (*Calluna vulgaris*), thereby stimulating decomposition of this older, catotelm peat.

The link between 'damaged' bog and DOC is convincingly argued by Yallop and Clutterbuck (2009) in relation to burning in blanket mire landscapes to improve sporting and agricultural productivity. The very clear coincidence demonstrated between scale of burning and DOC production provides yet another possible mechanism for the observed rise in DOC release from such landscapes, especially as Yallop *et al*. (2006) have demonstrated the very substantial rise in such burning management which has occurred in recent decades.

A link between DOC and burning does not, however, rule out the possible link between increased vascular-plant cover and elevated levels of DOC. The two are not mutually exclusive mechanisms. Burning is generally inimical to *Sphagnum* growth, although under certain tightly-controlled conditions not often found in the UK blanket mire environment (frozen ground, low wind speeds, adequate manpower), this may not be the case. Where burning does occur, it tends to diminish *Sphagnum* cover to the benefit of vascular plant cover, with species such as common cotton grass (*Eriophorum vaginatum*) and heather (*Calluna vulgaris*) responding particularly quickly.

Yallop *et al*. (2006) acknowledge that the colour on aerial photos of their most recent burn category (Class 1) varies according to the presence of species other than *Calluna vulgaris* (*Calluna* being the exclusive focus of their study). In addition, of course, a fire removes much of the vegetation cover but the below-ground parts of all species remain, either to regenerate or to decompose. Consequently it may still be possible both to reconcile burning with increases in DOC and for this DOC to be derived from the living vegetation or recent litter.

The general review above concerning the origins of DOC is significant because it poses the question: is rising DOC an issue in terms of carbon loss from the long-term store held in the peat, or is DOC a breakdown-product of fresh plant litter? The former gives rise to clear and direct concerns about loss of the long-term carbon store, whereas the latter may have implications for the quantity of carbon available to enter the long-term store from the acrotelm, or it may not – the quantities involved may be insignificant compared to the overall scale of litter breakdown.

A further factor needs to be recognised at this point. 'DOC' is a very broad term which embraces a wide range of differing organic substances which may differ from each other in fundamental ways. Thus Wallage *et al*. (2006) note a change in the chemical composition of DOC following ditch blocking. It is altered from a composition which is difficult for water utility companies to treat, into a form of DOC which is more amenable to water treatment. Whether these differing forms come from differing sources – one from fresh litter, another from the catotelm peat – has yet to be determined.

If it is indeed the case that DOC is essentially a breakdown product of vascular-plant tissues in the acrotelm or the surface layers of a haplotelmic bog, a number of (largely testable) consequences would follow:

- blanket mires dominated by a vascular-plant canopy rather than an open *Sphagnum* carpet would be associated with higher levels of DOC;conversely, areas of blanket mire dominated by open *Sphagnum* carpets with a thinly scattered vascular-plant community would be associated with low levels of DOC;
- DOC levels in areas of blanket mire currently dominated by vascular plants rather than *Sphagnum* might be reduced if land-management practices were adopted to encourage more *Sphagnum* growth at the expense of a dense vascular-plant cover;
- land-management practices which tend to encourage a vascular-plant cover at the expense of *Sphagnum* would tend to be associated with higher levels of DOC;
- areas of eroded blanket mire which tend to be characterised by areas of dry bog vegetation dominated by vascular plants would tend to be associated with high DOC levels, but if *Sphagnum* colonisation or the erosion gullies can be encouraged then DOC levels might be expected to decline;
- areas of extremely eroded blanket bog which consists largely of bare peat with little vascular vegetation might be expected to have relatively low release of DOC (though very high POC losses);
- the carbon in DOC would generally be young, having been derived from relatively recent vegetation, whereas if DOC were a product of catotelm decay then the carbon in DOC would be much older;
- haplotelmic blanket mire may give rise to DOC with a bimodal age distribution (*i.e*. showing two relatively distinct ages of carbon – one from the current plant litter, the other from breakdown of catotelm peat which now forms the bog surface), whereas blanket mire with a natural acrotelm might be expected to have a unimodal age distribution because most DOC loss would be from the acrotelm only;
- the widespread rise in DOC production recorded by Freeman *et al*. (2001), Worrall *et al*. (2003a, b, c) and even Monteith *et al.* (2007), over recent decades might thus in part be

explained by a combination of three related factors: firstly, high sulphate pollution levels are generally agreed to have contributed to loss of *Sphagnum* from the Pennines, and loss of *Sphagnum* encourages the development of vascular-plant cover; secondly, elevated temperatures may be leading to greater lowering of water tables and thus greater growth of vascular plants because their root systems are less constrained by the water table, and, lastly, greater growth of these same vascular plants may be stimulated by increasing levels of CO_2 in the atmosphere; all three factors, either individually or in combination, would lead to larger quantities of vascular-plant biomass to be decomposed each year and thus greater DOC release without any loss in the underlying long-term carbon store.

There is evidence to support some of these propositions already. Thus Holden *et al.* (2007a) cite evidence from radiocarbon work which indicates that DOC from peatlands is mostly derived from the top-most layers of peat and from the vegetation. Similarly, work associated with the ECOSSE Report (2007) has measured the age of carbon in DOC from a variety of soils and land-uses in North Wales and found that the carbon is generally less than 50 years old. This pattern was more clearly linked to organic soils than to the carbon released from mineral soils.

It would be of considerable interest to know the extent to which easily verifiable facts could be used to test these various propositions. It could have profound implications for the management of many blanket mire landscapes if it were found that the most cost-effective water-treatment plant for DOC might simply consist of *Sphagnum*, growing *in-situ* in the blanket mires which supplied the reservoirs of the water-utility companies.

7.4 The balance between CO_2, CH_4 and dissolved/particulate carbon in blanket mires

Having reviewed the conditions which give rise to sequestration and release of CO_2 and CH_4, and considered the types of values recorded for these processes, it is now possible to consider the overall balance between CO_2 sequestration, CH_4 release, and loss of DOC and POC by water or wind.

In an ideal world, it would be possible to present figures for this balance from differing parts of the surface pattern within:

- a natural blanket mire which is largely dominated by *Sphagnum*-rich T-zones (*sensu* Lindsay, Riggall and Burd, 1985) and thus has few 'holes' in the peat caused by A-zone hollows or pools – representing probably the most extensive form of blanket mire in Britain;
- a natural blanket mire dominated by a mosaic of *Sphagnum*-rich T-zones and shallow A zones – also widespread in Britain, but more localised within a given mesotope;
- a natural blanket mire dominated by a mosaic of *Sphagnum*-rich T-zones and deep A3/4 pools – largely restricted to localised parts of mesotopes within western and northern Scotland.

Unfortunately such descriptions are not yet possible, though Clymo and Pearce (1995) certainly describe the process by which a mass balance this might be undertaken in this way for Ellergower Moss, giving weighted averages for gas-exchange from hummocks and hollows based on the proportional areas that each covers across the site.

One of the very practical reasons that construction of such a microtopography-based model cannot yet be undertaken is that figures for CO_2-carbon accumulation have tended to be given on an annual basis whereas values for CH_4 release have tended to be given on an hourly or daily basis. Simple multiplication of such values up to a yearly basis is not possible because of diurnal and seasonal variations in CH_4 release. Consequently direct comparison of two critical factors cannot easily be achieved on the basis of existing published literature and for the level of detail desired.

A crude form of comparison is nevertheless possible, given broad figures for carbon accumulation and some specific values for CH_4 emissions (see Table 16). From Table 16 we see that the hummock level at Moor House may have a strong positive balance – in other words, hummocks absorb such large quantities of carbon that they are able to counteract the relatively low levels of CH_4 which they emit. It is worth noting that the hummock level is rather abundant at Moor House. The hummock level at Ellergower Moss is also strongly greenhouse cooling.

Table 16. Balance between rates for carbon accumulation and methane emissions.
Sites have been selected because yearly CH_4 emission values are available. Balanced against these emissions are approximate values of LARCA/ARCA carbon accumulation values representing net peat accumulation which by its nature takes into account losses to DOC and POC. Appropriate values of LARCA/ARCA have been estimated on the basis of microtopography. CH_4 emission values are converted their GWP value calculated on the basis of the IPCC GWP value of 25 for CH_4 over a 100-year period. The balance is given as a negative value if the bog is greenhouse cooling despite its CH_4 emissions, while the value is positive if the GWP CH_4 values render the bog to be greenhouse warming overall.

	Site/feature	**net CO_2 sequestration g CO_2 m^{-2} y^{r-1}**	**CH_4 emissions g CH_4 m^{-2} yr^{-1}**	**CH_4 GWP (100 yrs)**	**Balance net GWP: (-) cooling (+) warming**
Moor House (Clymo & Reddaway, 1971)	hummock	201.7	1.3	32.5	-169.2
Moor House (Clymo & Reddaway, 1971)	lawn	165	5.3	132.5	-32.5
Moor House (Clymo & Reddaway, 1971)	pool	0	9.3	232.5	+232.5
Ellergower Moss (Clymo and Pearce, 1995)	hummock	201.7	3.2	80	-127.7
Ellergower Moss (Clymo and Pearce, 1995)	hollow	128.3	8.68	217	+88.7
Loch More (Hargreaves and Fowler, 1998)	site	201.7	6.88	170	-31.7

Hollows and pools are obviously the major source of a GWP imbalance, but this is perhaps not so surprising when these areas are identified specifically by MacDonald *et al*. (1998), Lloyd *et al*. (1998), Arah and Stephen (1998), and Frenzel and Karofeld (2000) as prime locations for vascular plants such as bog bean (*Menyanthes trifoliata*) which provide a CH_4 shunt to the atmosphere. However, such features are rather restricted in their distribution across individual mire sites even in the north and west of Scotland, although in certain hyper-oceanic mires such as Claish and Kentra Mosses, Argyll, the area of open water pools within each mesotope probably matches or maybe even exceeds the extent of T-zone ridges and hummocks, as can be seen on the cover photograph of Rodwell (1991).

Interestingly, at the site level, Loch More in Caithness emerges from this crude approximation as being moderately greenhouse cooling, its GWP CH_4 emissions being higher than those for lawn and hummock at both Moor House and Ellergower Moss, but not so high that they exceed the sequestering capacity of the site. However, the study at Loch More involved sampling at the downwind edge of a large A3/A4 pool system (*sensu* Lindsay, Riggall and Burd, 1985) and thus the effect of bog-bean shunts for CH_4 are likely to be higher for this particular microtope area than for the mesotope as a whole. MacDonald *et al*. (1998) certainly highlight the very much higher levels of CH_4 emission from areas of bog bean.

An alternative approach to determining the GWP balance of a blanket mire system involves re-examination of the mass-balance method for estimating the overall carbon balance. Such an approach inevitably focuses on catchments – in this case two groups of catchments – both dominated by peat soils. Notwithstanding the concerns expressed earlier about certain aspects of mass-balance approaches and some of the specific site issues involved, it is nevertheless revealing to compare the results from the three mass-balance studies examined earlier – one for Moor House, and two set of measurements for Auchencorth Moss (see Table 17).

Table 17. Comparison of two mass-balance studies for peatland catchments.
The two mass-balance studies of Worrall *et al.* (2003a), Billett *et al.* (2004) and Dinsmore *et al*. (in press) compared against each other. Pink-shaded rows highlight substantial differences in values between sites/studies. Figures given in brackets indicate that this figure was not calculated in the particular study, but a component or composite value was calculated. Values are in g C m^{-2} yr^{-1}, and a negative value means that this quantity is lost from the peatland.

	Worrall *et al.* (2003a)	**Billett *et al.* (2004)**	**Dinsmore *et al.* (in press)**
	Moor House, Pennines	**Auchencorth Moss, nr Edinburgh**	**Auchencorth Moss, nr Edinburgh**
Condition of site	**moderate erosion, scattered drainage**	**100% drained, some peat mining**	**100% drained, possible recovery some peat mining**
net CO_2 exchange/ARCA	55	-	-
net CO_2 exchange	-	27.8	114.8
DOC	-9.4	(-26.9)	-25.4
POC	-19.9	(-1.4)	-3.6
TOC	-29.3	-28.3	-29.0
Rainfall DIC	1.1	-	-
Rainfall DOC	3.1	3.1	1.36
CH_4	-7.1	-4.1	-0.32
Dissolved CO_2	-3.8	-0.9	-1.32
Dissolved Inorganic Carbon (DIC)	-5.9	-1.2	-
Weathering DIC	1.8	-	-
Stream surface CO_2 loss	-	-4.6	-12.7
Total	**+14.9**	**-8.2**	**+69.6**

The majority of parameters recorded for both sites display largely similar values, including even the rather high values for total organic carbon (TOC) recorded for both sites in stream waters. The key difference between the sites lies in the way in which this TOC value is constructed.

Thus at Moor House, Worrall *et al.* (2003a) record rather low values of DOC but very high values for particulate carbon (POC). In contrast, Billett *et al.* (2004) record very low levels of POC (they estimate only 5% of TOC), but very high levels of DOC. Given that Moor House has extensive (though not intensive) erosion throughout the catchment, the high values for POC are not surprising. Auchencorth Moss, meanwhile, has little evident erosion but has drains at regular spacing throughout the site, and some of the drainage waters from the peat-mining operation may also find their way through to the Black Burn monitoring stream.

At least in the broad balance of values for parameters, it would seem that the mass-balance approach has highlighted key areas of site dynamics and functioning in both cases. Such attention to DOC and POC would not have formed part of a 'simple' LARCA or ARCA study and may thus have been missed.

The question of whether the final balance equations are correct to give Auchencorth Moss a net loss of carbon while Moor House appears still to be accumulating carbon, is not an easy issue to judge from the data provided. A simplistic view might conclude that Moor House has a positive carbon balance and is thus a living blanket bog continuing to accumulate and store carbon from the atmosphere. Auchencorth Moss, on the other hand, is clearly a degraded site which has lost its capacity to store carbon and is itself now losing its long-held store.

Notwithstanding the concerns raised earlier about some of the values obtained for Auchencorth Moss, the simplistic view given above fails to recognise one of the key unresolved features of these two mass-balance studies – namely the nature and status of the CO_2-exchange values.

In the case of Moor House, the CO_2-exchange value taken by Worrall *et al*. (2003) from the literature appears in that literature to be derived from an estimate of ARCA (or perhaps LARCA). If so, then all the values for parameters such as POC, DIC and rainfall DOC play no part in the balance equation because the ARCA value has already assimilated such losses and gains. This would mean that the only factors to consider for Moor House would be the ARCA value balanced against the GWP for the recorded CH_4 emissions. Doing so gives a final carbon balance figure of -108 GWP over a 100-year time-frame. The point is not that this figure might be more correct – it may or it may not be. The point is that there is not sufficient clarity about the nature of the original net CO_2-exchange value and how it is to be appropriately used in this mass-balance exercise.

Equally, the net CO_2-exchange value given by Billett *et al*. (2004) is a measure of CO_2 absorbed by the peatland during a year, balanced by the amount of CO_2 emitted by the peatland during the same year. The problem with these values is that there is no way of knowing the relationship between these exchanges of CO_2 and the quantity of material actually being passed down from the acrotelm for long-term storage in the catotelm. As Belyea and Clymo (2001) observe, this exchange of CO_2 may all relate to processes within the acrotelm. Any apparent losses or gains of carbon today may be balanced by long-term shifts in the structure and carbon content of the acrotelm in 20 or 50 years' time, with perhaps no actual changes in the catotelm (the real source of net carbon storage) during this period. Substantial differences in mass-balance can also result from differences in climatic conditions between years, as demonstrated by the two years described by Dinsmore *et al*. (in press).

The same argument can be applied to figures for carbon loss such as DOC. High DOC values may indicate significant losses from the catotelm carbon-store, but it may equally indicate a re-structuring of the acrotelm to a thinner form involving less T3 hummocks and more T1/T2 ridges (*sensu* Lindsay, Riggall and Burd, 1985), the acrotelm as it were ridding itself of much extraneous carbon-rich litter accumulated during a period of dominance by vascular plants, but during which these plants contributed little in the way of material to the catotelm. This process may take several decades, and during this time any measurements of DOC might indicate a high rate of loss even though the catotelm may now steadily be gaining carbon once again.

The problem, in other words, with mass-balance approaches is that they can provide only a brief and relatively static 'snapshot' of the bog at any moment. Observed values then tend to be assessed outside the context of the overall dynamic of the bog system in its present state, and incorrect conclusion then drawn from these values about the carbon balance of the bog.

The only way to overcome the problem is to turn the mass-balance 'snapshot' into a mass-balance 'movie', by continuing mass-balance measurements over longer periods of time. Without this context, mass-balance studies may confuse more than they enlighten. For example, Billett *et al*. (2004) comment, in relation to their conclusion that Auchencorth Moss is losing its carbon store, that :

> *"Many peatlands may be reaching a stage in their natural development in which C accumulation since the last ice age has slowed down or ceased, such that peatlands become sources rather than sinks for C."*

In fact the best current carbon accumulation models for peatlands do not suggest this. Clymo *et al*. (1998) point out that all models used in their study point to continued carbon accumulation in peatlands for millennia, and that the sequestering efficiency of "long established bogs in the concentric bog region of Finland may now be only 50% to 80%". This is very different from the idea that bogs such as Auchencorth Moss may have reached their natural limit of carbon accumulation. Rather than look to natural end-points, it might be more fruitful to consider the evidence for comprehensive drainage and substantial peat mining, to say nothing of the forestry plantation covering the crown of the bog just on the southern margin of the mesotope-bisecting boundary of the Auchencorth catchment study area.

Indeed the subsequent mass-balance study by Dinsmore et al. (in press) on the same site a decade later not only reveals substantial differences from the values obtained by Billett *et al*. (2004), but indicates that Auchencorth Moss may in fact be undergoing a period of significant net carbon accumulation. The question even here, of course, is whether the values obtained by Dinsmore *et al*. (in press) represent an actual period of carbon transfer to long-term storage in the catotelm, or whether the figures reflect a damaged system in which the acrotelm is re-establishing itself and the carbon accumulation values have little to do yet with long-term carbon storage.

8 Carbon accumulation in natural UK peat bogs – a summary

Synthesising everything reviewed so far in terms of carbon accumulation and storage within a natural peat bog system and using values gathered from a wide range of sources, Figure 24 illustrates a standard natural cubic metre of peat, with estimated associated values. It is assumed that a natural bog surface has a largely continuous cover of *Sphagnum* forming hummocks, ridges and hollows. Pools are not described by this simple model because they are clearly not significantly peat-forming.

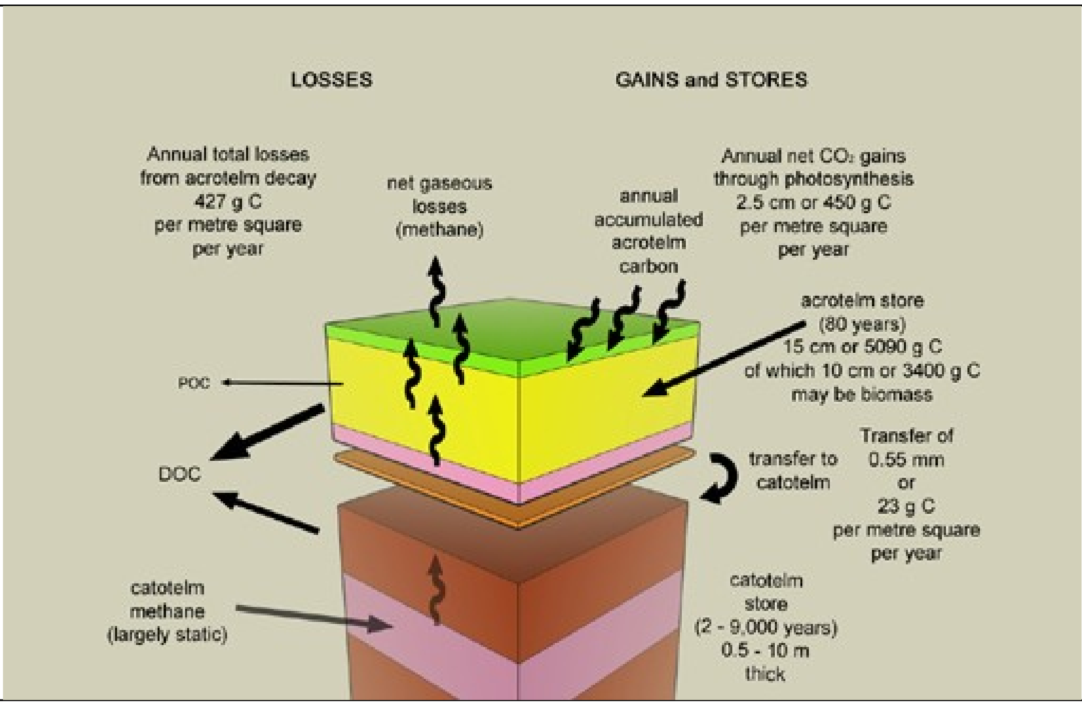


Figure 24. Quantities of material movement in a 'standard cubic metre of peat' for a natural peat bog in Britain.
The standard cubic metre of peat consists of an acrotelm and a catotelm layer. A certain amount of material is added each year to the catotelm mainly through photosynthesis, but over an 80-year period most of this is lost through decay. A small proportion of material surviving this 80-year journey is transferred annually to the catotelm. Methane occurs in both the acrotelm and the catotelm, but may be largely static in the catotelm. Although CO_2 is lost from the acrotelm, overall, more is gained, so the only net gaseous loss is CH_4. In a natural peat bog, dissolved organic carbon (DOC) is a product largely of vascular-plant breakdown in the acrotelm, supplemented by a much smaller quantity from breakdown of material in the catotelm. In a natural peat bog system, particulate organic carbon (POC) is released in only very small quantities. Note that biomass may amount to 3.4 kg C m^{-2} or 34 tC ha^{-1}.

Illustration R A Lindsay

Assuming an annual gain of 2.5 cm at the acrotelm surface, and assuming a low bulk density of 0.035 g cm^{-3} for this material, this represents an annual carbon gain of 450 g C. Total annual losses represent 427 g C leaving an annual transfer to the catotelm of 23 g C. At a bulk density of 0.12 g cm^{-3} this represents a thickness of 0.55 mm added each year to the catotelm.

8.1 GWP-carbon balance

Taking values for C-accumulation and CH_4 release mid-way between hummock and lawn from Table 16 for Moor House, northern Pennines, gives values of 50 g C m^{-2} yr^{-1} for C-accumulation and 2.8 g CH_4-C

m^{-2} yr^{-1} for CH_4 release. Converting this figure for CH_4 into a GWP for a 100 time-horizon gives a net uptake (*i.e.* cooling effect) of +90 g CO_2-eq m^{-2} yr^{-1} (GWP 100 yrs). This contrasts with the average GWP value of +351 g CO_2-eq m^{-2} yr^{-1} (GWP 100 years) obtained by Dinsmore *et al*. (in press) for Auchencorth Moss.

Auchencorth lies at an altitude of 280 m and shows clear evidence of recovery from extensive drainage, with much vigorous *Sphagnum* growth. In contrast, Moor House has had a history of damage from burning, erosion and atmospheric pollution, it lies at an altitude of 670 m and thus has a shorter growing season than Auchencorth Moss, and is not currently subject to any significant form of recovery 'stimulus'. As such, the differences in carbon gains between the two sites appear reasonable.

Furthermore, Moor House is something of a haplotelmic blanket mire relatively rich in tussock-forming hare's-tail cotton grass (*Eriophorum vaginatum*) and thus cannot be taken as a good example of our 'cubic metre of natural peat'. There is the distinct possibility that the 'lawn' community is dominated by a vegetation which specifically provides a CH_4-transport system. A less disturbed community dominated by *Sphagnum* and thus minimising tussock formation in *Eriophorum vaginatum* may well emit smaller quantities of CH_4, as indicated by Frenzel and Karofeld (2000).

Indeed Frenzel and Karofeld (2000) noted reductions in CH_4 emissions of 94% on removal of the *Eriophorum vaginatum* stands compared to *Sphagnum*-dominated surfaces in their study plots. On this basis, the potential CH_4-emission value for a Moor House richer in *Sphagnum* than is currently the case, could fall to 0.18 g CH_4 m^{-2} y^{r-1} to give a 100-year GWP of +277 g CO_2-eq m^{-2} yr^{-1}. This would exceed the GHG gains obtained at Auchencorth Moss for 2008, but would remain substantially less than the gains recorded at Auchencorth in 2007.

8.2 Biomass storage

The value given earlier in Section 6.2.3.1 of 48 t C m^{-2} for possible biomass carbon-storage (equivalent to 4.8 kg C m^{-2}) within the acrotelm is the same order of magnitude as the global estimate (2 kg C m^{-2}) for the biomass of open bogs provided by Olson *et al*. (1983), particularly as the latter say that values can be as high as 6 kg C m^{-2}. However, both these sets of values are an order of magnitude larger than the 0.2 kg C m^{-2} estimated for the biomass of UK blanket bogs by Milne and Brown (1997). The figure given by Milne and Brown (1997) is the one currently used in UK carbon reporting.

There is clearly a substantial disparity between the official UK figures for the carbon content of peat-bog biomass given by Milne and Brown (1997) and the figures presented here. This difference may arise simply because the thickness of the living moss layer has not been recognised as biomass by Milne and Brown (1997). However, as observed earlier in Section 6.2.3.1, the key feature of living biomass compared with the relatively inert peat-soil deposit is that the biomass has the continuing potential to sequester further carbon whereas the peat soil does not.

8.3 Limits to the illustrated, idealised 'cubic metre of peat'

The values suggested here are best approximations from an amalgam of what is nevertheless a rather limited set of information. In any case, the acrotelm values are likely to change significantly from 1 m^2 to the next because values associated with a T3 hummock (*sensu* Lindsay, Riggall and Burd, 1985) will undoubtedly differ from those of a T1 low ridge or an A1 *Sphagnum* hollow. The value of acrotelm transfer is likely to be somewhat similar for all T-zone nanotopes, but may be smaller for hollows and will undoubtedly be much lower or even zero for T3/T4 pools.

It is suggested here, based on what little evidence currently exists, that the majority of CH_4 release has its origins in the acrotelm rather than the catotelm, and the same is proposed for DOC. Given that POC is predominantly associated with loss of material from bare peat, it is assumed that POC losses are generally very low in natural bogs, although wave action at the margins of steep-sided A4 pools may contribute some POC on those sites where A4 pools are a feature (Osvald, 1949).

There are complicating factors such as root penetration into the catotelm, and possible transport of dissolved material including CH_4 (Charman *et al*. 1994), heavy metals (Shotyk, 1988), and presumably DOC, from upper layers to areas deeper in the peat, but these are beyond the capacity of this simple illustration to quantify and illustrate.

In many ways, the idealised 'cubic metre of peat' poses more questions than it answers, but hopefully these questions will assist in the design of future research efforts into this complex subject.

8.4 'Vascular-plant cycle' and '*Sphagnum* cycle'

One of the consistent themes through much of the description given above about the various processes which govern the carbon-sequestering and storage system is that many of the more dynamic processes appear to be, or at least are proposed here as being, governed by the presence of vascular-plant material. Thus most sequestration and respiration, DOC production, CH_4 emissions – all are linked more closely to vascular-plant processes than they are to *Sphagnum*. The one process in which the *Sphagnum* sward appears to exceed the capacities of the vascular-plant sward is in resisting decay sufficiently to be capable of passing substantially greater quantities of preserved remains down into the catotelm as peat.

It is important to emphasise that the acrotelm is more than simply an area of dynamic water movement, a role for which it is most commonly known. The acrotelm is also the zone in which vascular-plant material is processed so rapidly and in such a variety of ways that very little survives to be passed down into the catotelm as peat. It is therefore proposed here that the vegetation of a peat bog can be thought of as two distinct, though interlinked, cycles moving at different speeds rather like the hare and the tortoise of the fable. Both are heading towards the same end-point, but one, here called the 'vascular-plant cycle', does so in a state of busy haste with little to show for it by the end of the cycle, while the other, termed here the '*Sphagnum* cycle', plods steadily towards the completion of the cycle without much direct involvement in methane or DOC production. By the end of the cycle, however, the *Sphagnum* cycle still possesses a significant quantity of material which can be passed down to long-term storage in the catotelm. These two cycles are illustrated conceptually in Figure 25.

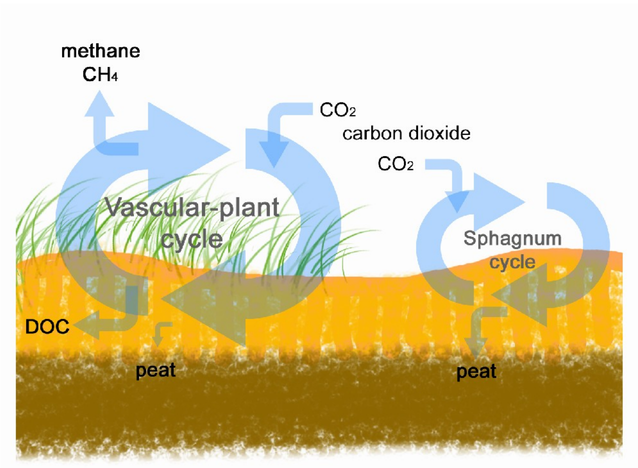


Figure 25. 'Vascular-plant cycle' and '*Sphagnum* cycle'.
This diagram illustrates the two major cycles of the peatland ecosystem. The two blue circles represent the cycle of CO_2 in the form of photosynthetic uptake and respiration release. The vascular-plant circle is larger because more material is involved. Both cycles lie partly within the acrotelm (orange shading), and partly in the atmosphere. The vascular-plant cycle provides the main quantities of methane and DOC, largely from within the acrotelm, and contributes a very small quantity of material to the catotelm (brown shading). The *Sphagnum* cycle contributes very little to methane and DOC production, but provides the major input of carbon-rich material from the acrotelm to the catotelm.

Illustration R A Lindsay

In both cases the main cycles represent sequestration and respiration of CO_2. The major outputs from the vascular-plant cycle are methane and DOC, with a very small amount of material passed into the long-term carbon store of the catotelm.

Meanwhile the *Sphagnum* cycle contributes nothing of any significance to any major off-site outputs. Its main contribution to the system is the relatively large amount of material passed down into long-term storage in the catotelm. Figure 25 is undoubtedly a highly simplified view of several much more complicated processes, but it is presented here on the basis that it nevertheless captures the essence of several key peat-bog carbon processes.

PART 3

CARBON AND PEAT BOGS – DISCUSSION TOPICS

9 DISCUSSION TOPIC 1a Drainage of peat bog systems – the carbon balance

This Discussion Topic is substantially longer than most of the other Discussion Topics because drainage underpins much of what happens on a peatland site. Thus it forms part of agricultural land-claim, it is the major action in site preparation for forestry, and it is an essential part of site preparation and maintenance for peat extraction. All of these activities will be considered in their own Discussion Topics later in the present report, but the underlying issues of drainage will be dealt with here in Discussion Topic 1.

The extraordinarily high water content of peat soils is a widely recognised feature of such soils, but, almost contrarily, so too is the very slow rate of water movement through the peat matrix. Thus while a peat bog soil may contain fewer solids and more water by weight than a jellyfish, this water is known to move through the peat at a rate (as observed earlier in the present report) more than 100x slower than the speed of a snail. Speeds of less than 1 cm per day are not unusual.

These features are not, however, what makes peat bog soils so unusual. Other soils have high water contents or slow rates of water movement. Peat bog soils are not like other soils in terms of one very particular aspect, and this aspect is almost wholly a product of biology. It is not therefore the physical or chemical response of a peat bog soil to drainage which are so distinctive. What separates these soils from all other types of soil is the fact that a peat bog soil is a biological product which is capable of displaying a range of significant biological responses to drainage. Some of these biological responses can give rise to feedback processes which have the potential to counteract the effects of drainage and return the system to some form of equilibrium state. Few, if any, other soils have this responsive capacity.

9.1 Biological aspects of, and responses to, drainage

The biological responses of a peat bog to drainage arise from two key factors. Firstly, the peat soil itself is almost entirely biological in origin, being little more than the semi-decomposed remains of once-living vegetation. Secondly, the current vegetation is not merely the continuing source of fresh peat soil, the particular characteristics of any vegetation assemblage also gives rise to particular hydrological conditions within the structure of the peat bog – change the vegetation assemblage and the hydrological characteristics change. Thus the vegetation has the capacity to respond to, and alter, hydrological conditions if so dictated by changing circumstances.

9.1.1 Drainage and the peat matrix

Remove the water from any soil other than peat and although structural or chemical changes may occur the soil matrix largely remains *in-situ*. Remove the water from a peat soil and the soil will slowly but steadily vanish, much of it quite literally into thin air as a result of the oxygen-fuelled energetic decomposition processes which the water had hitherto prevented.

This characteristic has important implications for the processes associated with peatland drainage but it is an aspect which is often overlooked when investigating the impact of drainage on peatland sites. A commonly-stated view of peatland drainage is that the effects of drains are felt over distances of only 5 – 10 m, but this view fails to take into account a number of important responses described by Hobbs (1986), including that of oxidative wastage. Together, these responses can result in effects which may extend much further than the distances generally associated with peatland drainage.

9.1.1.1 Primary (1°) consolidation

The first, immediate effect of drainage on a peat bog is the loss of water held loosely within the large spaces which exist between particles (interstitial water). In the upper layer of the acrotelm these spaces are large but they become steadily smaller down through the acrotelm. Within the catotelm, such spaces are generally small but Holden and Burt (2002) have identified that certain types spaces (macropores and peat pipes) can be substantial and relatively widespread within the catotelm. While some of these features appear to be a natural part of the bog sub-surface structure, Holden (2005b) has demonstrated that such features become more frequent when peat is drained. Furthermore, drainage has the capacity to remove water rapidly from such interstitial spaces and macropores.

This loss of water manifests itself in rapid subsidence of the peat surface because the interstitial spaces collapse as water progressively escapes from the peat. In damaged haplotelmic blanket mires the peat is already fairly dense (with dry bulk densities of 0.2 g cm^{-3} or more) and thus the amount of primary consolidation following drainage is relatively limited, whereas in wet, natural mires this primary consolidation can be substantial and dramatic.

9.1.1.2 Secondary (2°) compression

Material suspended in air weighs much more than the same material suspended in water. Consequently as water is lost from the peat through primary consolidation a significant quantity of material formerly floating largely weightless within the bog water-table steadily emerges from the falling water table to become a substantial load on the peat column beneath. This additional weight on the peat column steadily forces more water out from the interstitial spaces of the peat as the particles become more tightly pressed together. As more water is lost so the water table falls further and the exposed load becomes even greater. The peat surface continues to subside but now the process is driven largely by secondary compression rather than primary consolidation.

Secondary compression continues as long as there is a load on the peat column. The UN Food and Agriculture Organisation (UN FAO peat drainage website) state that every 1 cm draw-down of the water table in a peat soil results in an increased load of 1 g cm^{-2} (*i.e*. 10 kg m^{-2}) on the peat beneath. This value seems too high for blanket bog (the UN FAO document is concerned with tropical peat swamps), but the fundamental process still applies – any material (and the water which it contains) left above a falling water table will impose a load on the peat beneath. Moreover this is without considering the increased weight which may arise from increased shrub or tree growth as a result of drainage. Eggelsmann (1975) additionally emphasises that subsidence occurs within the whole peat column *beneath* the base of a drain, not just around the drain – a phenomenon which is not widely recognised.

Holden, Chapman and Labadz (2004) highlight the effect of drainage on low flows in peat catchments. They suggest that observed increases in such flows result from slow but steady de-watering of the peat when drainage is carried out within a catchment. They comment on the often-cited drainage study carried out on an experimental blanket-bog plot at Glenamoy in Ireland, reported on by Burke (1975), and point to the fact that the catchment was estimated to be losing 1000 mm of water per year through such slow de-watering. It would be reasonable to attribute much of this loss to the process of secondary compression.

9.1.1.3 Oxidative wastage

The third process resulting from water loss is largely unique to peat soils and the fact that they are composed almost entirely of organic matter. Once oxygen penetrates the catotelm of the system, this formerly anoxic catotelm peat begins to oxidise and decompose rapidly. Carbon is lost from the system as CO_2 and in the form of other more complex breakdown products such as DOC.

This process of oxidative wastage continues to occur as long as oxygen penetrates the catotelm peat. A number of examples of long-term secondary compression and oxidative wastage can be found across western Europe but the most famous example in Britain is the Holme Fen Post at Holme Fen National Nature Reserve in Cambridgeshire. In 1852 this iron pillar was embedded into Holme Fen raised bog so that its cap was level with the surface of the raised bog. In the same year, the adjacent Whittlesea Mere (at the time England's second-largest freshwater lake) was drained in one of the final acts in the wholesale conversion of the former Cambridgeshire Fenlands into agricultural land (see the Great fen Project website). The top of the pillar now lies some 4 m above the present, intensively-drained, peatbog surface and this peat surface continues to sink, indicating an overall average subsidence rate of some 2.5 cm per year since 1852 (though a second post, erected some years later, shows that subsidence was very much faster in the early years after drainage).

Immirzi *et al*. (1992) provide a valuable review of the evidence for subsidence and oxidative wastage in peatlands from various parts of the world, and highlight the fact that the rate of subsidence and oxidative wastage is also linked to latitude (and thereby climate). Thus drains cut into northern blanket mire appear not to result in such dramatic rates of secondary compression and oxidative wastage as observed in the lowlands of Cambridgeshire, perhaps because the higher rainfall in blanket mire areas enables the peat to remain more saturated despite the effects of drainage. However, it is important to inject a strong cautionary note here, because few drained areas have the advantage of iron pillars indicating the former level of the peat surface. It is all too easy to assume that little has changed in relation to a drain, but without figures for the position of the bog surface and the former condition of the mire surface before the drain was dug, it is impossible to be sure of subsequent effects.

Braekke (1987) estimated the loss of height associated with forestry drainage on a peatland site in Norway based on the increased concentration of phosphorus in the peat profile, while Anderson *et al*. (1992) used the relative differences in bulk density within surface layers to estimate the original level of the peat surface in an area of afforested blanket mire. Both approaches thus rely on proxy data, together with a defined set of assumptions, to estimate the fall in peat-surface rather than providing direct measurements of such a fall. There is thus an element of doubt about the actual rate of subsidence in both cases.

9.1.2 Subsidence and oxidation – consolidation or loss?

The important distinction between consolidation and oxidative wastage is that the former simply results in a more compressed peat and carbon store, whereas the latter represents actual loss of peat and carbon to the atmosphere. For the estimation of carbon budgets, this distinction is clearly of fundamental significance and the measured separation of these two processes becomes an issue of some importance.

On the face of it, determining the relative contributions of consolidation and oxidation in a drained peatland system should be relatively straightforward. Changes in ground level can be measured, changes in bulk density can be measured, and the quantity of CO_2 can also be measured. The quantity of CO_2 released indicates how much of the subsidence has been caused by oxidative wastage. Thus the remainder of the fall in ground surface and the measured increase in bulk density must be a result of consolidation. Unfortunately it is not a simple as that.

Clearly in the case of the Holme Fen Post in Cambridgeshire, the observed 4 m fall in ground level is unlikely to have resulted entirely from consolidation and consequent increase in bulk density. It seems likely that a considerable quantity of peat has been lost to oxidation in this case. Schothorst (1977) observed that some 50% of surface lowering in a drained fen peatland in the Netherlands resulted from oxidation of the drained peat rather than because of slumping, consolidation and compression.

Most examples of drainage, however, particularly those in blanket mire areas, do not have such dramatic evidence of shrinkage as seen in the lowland environment of Holme Fen. Intuitively, however, it would seem logical that catotelm peat newly-exposed to oxygen would undergo at least some oxidation even in blanket mire environments, but the critical question is – how much oxidation can be expected? Presumably not the same proportion as observed by Schothorst (1977) for fen peat.

9.1.2.1 Kasimir-Klemedtsson *et al.* (1997) – a comparison of estimates

The difficulties involved in distinguishing between consolidation and oxidation are reviewed in some detail by Kasimir-Klemedtsson, Klemedtsson, Berglund, Martikainen, Silvola and Oenema (1997). The problems begin with measured changes in surface height. It is only possible to make accurate estimates of such changes if accurate measured of ground surface were obtained *prior* to drainage. For British bog systems (and indeed for most other drained peatland systems around the world) this pre-drainage information is rarely available.

The difficulties continue with bulk density, because again it is only possible to obtain accurate measures of the change in bulk density if the bulk density was measured before drainage took place. Again, pre-drainage data are rare.

Finally, there is the difficulty of obtaining an estimate of CO_2 losses from the drained soil. Although it would seem on the face of it rather easy to measure the loss of CO_2 from a drained peat soil, the problem arises from the fact that not all CO_2 released from the peat is a result of oxidative wastage. Some CO_2 is produced by oxidation of the drained organic matter via the respiration of soil organisms, some by roots, some by mycorrhizae, and if there is a vegetated surface then some will be released by the respiration of new organic matter.

In a large-scale review of oxidative carbon losses from peat soils subject to agricultural use, Kasimir-Klemedtsson *et al*. (1997) consequently use three different approaches in their attempts to identify actual losses from oxidative wastage. Their first approach uses rate of subsidence, and they assume that 70% of subsidence is attributable to oxidation, while also assuming single values of bulk density for the three countries involved. Their second approach uses a climate-based model to calculate likely oxidative losses. Their third approach involved measuring CO_2 flux and then assuming that root respiration provided 38% of this flux.

The three approaches used by Kasimir-Klemedtsson *et al*. (1997) produced results which varies substantially between methods. In some cases the different approaches resulted in values which were 100% or 200% different, while in certain cases there was more than an order of magnitude difference. Such variation probably reflects the underlying uncertainty in the assumptions used in the three methods, but also emphasises the more general difficulties associated with the accurate determination of oxidative losses.

9.1.3 Surface subsidence and observed water-table draw-down

The overall results of the various surface-subsidence processes outlined above are that, firstly, the effective width of a drain may be increased, sometimes substantially, as peat adjacent to the drain undergoes consolidation, compression and oxidative wastage. Depending on the slope on which the drain has been dug, the depth to which the drain itself subsides, and the water content and bulk density of the surrounding peat, this process of subsidence may spread for some distance from the drain itself. On a raised bog, Lindsay *et al*. (1988) have shown that this effect can extend for distances of 100 m or more (see Figure 26).

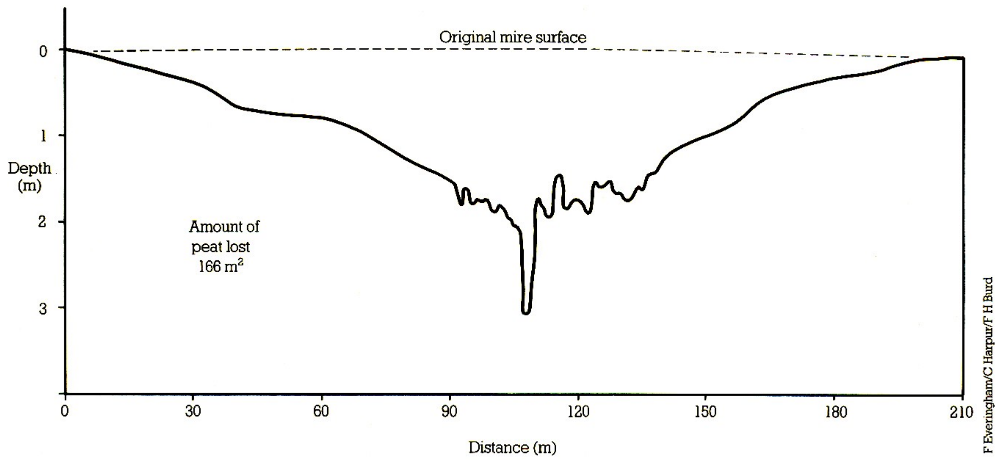


Figure 26. Slumping, subsidence and oxidation associated with a drain cut in Wedholme Flow, a raised bog in northern Cumbria.
A cross-sectional view of a drain cut into Wedholme Flow, Cumbria, some 100 years ago. The gradually sloping profiles on either side of the drain (which has been repeatedly cleaned out) indicate the combined impact and extent of consolidation and oxidative wastage. The horizontal scale has been compressed relative to the vertical scale to illustrate more clearly the extent of surface lowering.

Adapted from Lindsay *et al*. (1988)

The second important effect to arise from the combined processes of consolidation, compression and oxidative wastage is that drains *appear* to result in very limited water-table draw-down. However, if the water table falls, then consolidation, compression and oxidative wastage mean that the peat surface will itself also fall, in effect chasing the water table downwards.

Consequently there is never a time when the peat surface and the water table are separated by a very great distance except very close to the ditch. Both water table *and* peat surface are likely to have sunk towards the mineral sub-base to varying degrees following drainage. A simple measurement of the distance between the water table and the bog surface would give little or no indication of this.

This modification by drainage of the ground surface itself is generally overlooked when the effects of drains in peat are being examined. Measurements of water-table draw-down in relation to drains are most usually taken with reference to the peat surface rather than to a fixed datum point such as a metal rod anchored in the basal sediments. The effect of a drain on the water table may thus appear to be minimal, even though both components – peat surface and water table – are steadily sinking.

The lack of fixed datum points for many peatland drainage studies represents a significant information-gap and means that great caution should be exercised when considering the evidence from such studies. Indeed there is surprisingly little published information about ground-surface changes in relation to blanket mire systems and drainage, given the considerable interest in the issue over the years.

It would be reasonable to expect such morphological changes to be smaller in many blanket mires simply because, whether through land-use practices or natural decomposition processes, the peat of blanket mires is often already denser than is typically found in reasonably natural raised mires. It should not be assumed that this is invariably the case, however. There are still significant areas of blanket mire with very high water contents and low bulk densities, and it could be expected that these areas would react dramatically to drainage. Such a reaction may still, however, be less dramatic than in lowland raised bogs because there is a greater rainfall supply in blanket mire regions to replenish the water removed by drainage.

In blanket mires there is the added complication that slope often has a much greater influence on hydrological processes than is the case for the relatively gentle gradients associated with raised mires. Thus the impact of a drain cutting across a hill slope is unlikely to result in slumping of the peat by 1-2 m within an adjacent zone of some 10-20 m if in fact the slope itself falls by more than this over a similar horizontal distance.

9.1.4 Acrotelm, catotelm and peatland drainage

Water-table measurements in drained peat bogs may thus often fail to give the full picture of drainage in relation to draw-down of both the ground-surface *and* the water table relative to the mineral sub-soil. However, a substantial body of literature has also been published for peat bogs stating that draw-down of the water table relative to the bog surface is still very limited and localised in its effect. This may be how it is described, but a closer look at some of the often-cited studies (*e.g.* Boelter, 1972; Stewart and Lance, 1983, 1991; Gilman, 1994) reveals a more complex picture than is perhaps suggested by these studies.

9.1.4.1 'Surface waters' and 'groundwater'

To some extent the view that drainage has only a very limited effect on peatland hydrology arises from the perceptions and expectations of those undertaking the drainage; drainage is undertaken in order to increase the capacity of the land to support greater human use – generally agricultural use. Consequently the perception of what is 'significant' draw-down is influenced by whether the agricultural productivity of the ground has been improved. However, agricultural improvement and ecological impact are two very different things. The critical difference lies in the true nature of what are broadly termed 'surface waters' and 'groundwater'.

Boelter (1972) illustrates two draw-down profiles of bog water-tables associated with a ditch in a bog where the peat is moderately decomposed (Marcell Bog). The two profiles show, firstly, the effect of lowering the water level in the ditch, and secondly, the effect on the water table profile of wet conditions and dry conditions. These two water-table profiles have been much quoted and copied over the years, but what is not usually highlighted is that Boelter (1972) makes a clear distinction between a thin uppermost ‘fibric’ layer and the main body of peat which he describes as ‘hemic’. These two layers clearly represent the acrotelm (fibric layer) and the catotelm (hemic layer).

Boelter’s (1972) main focus is the effect of the ditch on drawdown within the lower, catotelm layer of Marcell Bog. He demonstrates that drawdown in this layer is restricted to a zone varying between 5 m and some 20 m alongside the ditch. Beyond this distance, there is little or no drawdown into the catotelm. Lowering the water level in the ditch barely affects the zone of draw-down. A month of dry weather, however, extended this zone from approximately 5 m to something over 20 m in width within the catotelm.

The effect on the upper, acrotelm layer is, however, rarely commented upon. Boelter (1972) observes that when the water table is in the acrotelm layer the high hydraulic conductivity of this layer offers little resistance to water flow and thus water flows rapidly out of the system. He also notes that during periods of high water levels, the water table at a distance of 50 m from the drain is ‘only 0.09 m higher than at 2.5 m’. Boelter (1972) evidently regards a difference of 9 cm height in the water level to be almost insignificant, but within the context of an acrotelm thickness this is a substantial difference. Boelter (1972) makes it clear that he regards the effect of drainage to be largely restricted to the acrotelm:

> *“The major effect of the ditch in the Marcell bog has been to hasten somewhat the flow of water from the surface or near-surface fibric horizons during high water conditions.”*
>
> Boelter (1972), pp. 336-337

Anyone contemplating the question of bog drainage and its possible impacts is strongly urged to read Ivanov’s (1981) account of the process. In places it is necessarily a mathematical treatment of the subject, but the description is sufficiently clear and lucid to mean that a full grasp of the mathematics is not necessary to understand the key processes involved. It is to be regretted that this book is not more widely known and used because it sheds a clear and unambiguous light on the subject. The reason for its obscurity is almost certainly because it has been out of print for some time, is extraordinarily difficult to obtain even on the second-hand book market, and possibly many potential readers (and users) have been put off by the mathematical treatment of the subject.

Ivanov (1981) gives a clear, quantitative picture of why the acrotelm is so easily de-watered. He gives detailed values for hydraulic conductivity (water transmission) at centimetre intervals through the acrotelm layer of several peatland types. The transmission rates he gives for the uppermost layers of the acrotelm associated with a 20 cm-high hummock are so high that it is difficult to present a meaningful graph of the change in rates down the acrotelm profile. Consequently Figure 27 shows the transmission rates only for the acrotelm to a height of 4 cm above the water table in the hummock. Speed of water flow in the hummock at 16 cm above the water table is equivalent to a typical walking pace (5 kph). Even with the slower scale of speed used in Figure 27, the catotelm peat appears to have a flow rate of zero because the catotelm flow is so slow that values cannot be seen at this scale.

Figure 27 thus highlights the fact that zones *above* the bog water table have such high transmission rates that drainage may not affect the hydrological regime to any great degree. Even under natural conditions, hummocks shed their water extremely rapidly. Meanwhile T1 low-ridge areas close to the water table are adapted to high water levels but also have high transmission rates. These, and shallow carpets of aquatic *Sphagnum cuspidatum*, are potentially the microform element most vulnerable to drainage, and indeed these do seem to be the elements which are least abundant in drained areas.

Stewart and Lance (1983, 1991) demonstrate that agricultural productivity of blanket mire is not improved by moorland drainage. They similarly conclude that moorland drainage merely removes excess surface water and that drainage has little effect on groundwater. Here again, the effect of drainage is seen as impacting only the surface layer of the bog – but the significance of this for acrotelm dynamics is not explored. The picture is slightly complicated in the case of Stewart and Lance (1983, 1991) because much of their study site was probably haplotelmic – i.e. lacking a true acrotelm.

The same distinction between ‘surface water’ and ‘groundwater’ can be found in Gilman (1994), where he explicitly separates the effects of drainage on ‘surface and near-surface water’ from the effects on ‘groundwater’. It seems reasonably clear that Gilman’s (1994) account of ‘surface water’ behaviour refers to water in the acrotelm, while his observations about ‘groundwater’ relate to the behaviour of the water table in the catotelm.

Gilman (1994) then goes on to point out that:

> *"The layered structure of some mires [essentially bogs], in which a zone of high permeability occurs near the surface, can extend the influence of the ditch, and the cutting of drains across a previously undisturbed peat expanse will draw down the water table permanently into the lower layer, the catotelm..."*
>
> Gilman (1994), p.96

Thus Gilman (1994) explicitly indicates that drainage can permanently empty the acrotelm. All of these often-cited works specifically identify that drainage has a significant impact on the acrotelm, but these same works are most commonly cited as demonstrating that drainage of peat is only capable of drawing down the water table over distances of between 2-20 m. While this is true for the catotelm, it is demonstrably not true for the acrotelm. However, this distinction is rarely, if ever, reported.

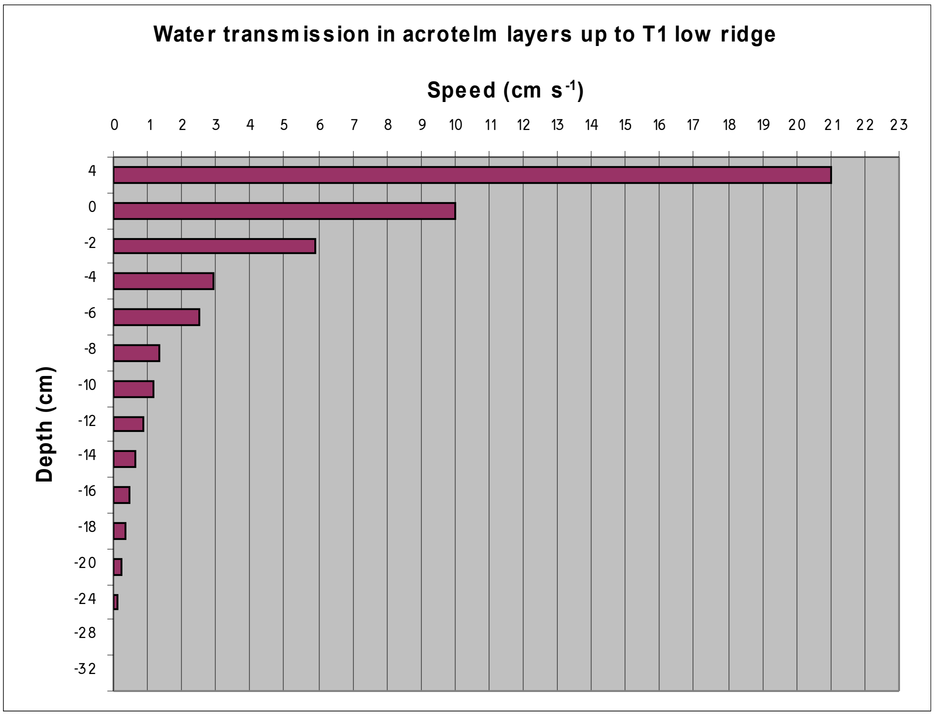


Figure 27. Rates of water transmission within different layers of the acrotelm
Based on rates of water transmission (hydraulic conductivity, k_0) given by Ivanov (1981) for a *Sphagnum-Eriophorum*-sedge bog with low microtopography, these graphs show the speed with which water is capable of moving laterally through each defined section of the acrotelm within a 20 cm-high hummock. The graph shows values only for heights to 4 cm above the water table within the hummock because speeds of water movement at higher levels in the hummock are so large that all other values would be invisible using the necessary scale. A fast snail moves at 0.2 cm per second, while a walking pace of 5 kph is equivalent to 139 cm per second.

9.1.4.2 Acrotelm de-watering (draining)

The simple picture of a permanently-empty acrotelm described by Gilman (1994) is not, of course, what is seen in the field. During periods of rain the water table rises within the acrotelm, perhaps even to the surface of the bog. The effect of drainage is therefore masked for a while after a rain event. Once rainfall ceases, however, the impacts of the drains become increasingly obvious again. As Boelter (1972), Stuart and Lance (1983, 1991) and Gilman (1994) observe, drains cause water held in the surface layers to be lost more rapidly than in the un-drained condition. Thus the acrotelm is de-watered more rapidly than before.

Holden, Evans, Burt and Horton (2006) studied the pattern of water movement in several blanket mire catchments and were able to compare the behaviour of an undrained catchment with one which had been drained. In the un-drained catchment, 99% of water flow was found to occur either at the surface of the bog or within the uppermost 10 cm. In the drained catchment, Holden *et al*. (2006) found that only 74% of water flow occurred within these zones. The remaining 26% of water flow occurred at peat depths of between 10 cm and 100 cm – most likely within the catotelm. In other words, in the presence of drainage, up to a quarter of all throughflow no longer reacted directly, if at all, with the living surface.

The direct implications of this altered behaviour can be illustrated using water-table data from Cors Caron raised bog in mid-Wales. Figure 28 shows values recorded for the maximum and minimum heights of the bog water table at two locations on Cors Caron each month over a period of 12 months. These measurements are obtained using an instrument called a Walrag (Water-level range gauge – Bragg *et al*., 1994). Walrag 2 is located 135 m from the cut edge of the bog, while Walrag 3 is located only 80 m from the same edge. A cut edge such as this acts, in effect, like a single-sided drain.

It can be seen from Figure 28 that in the driest conditions the water table falls further at Walrag 3 than it does at Walrag 2 (a fall of 35 cm as opposed to 23 cm). This difference in behaviour is notable in its own right, but perhaps even more significant is the response during periods of maximum wetness. Whereas the water table at Walrag 2 rises to the bog surface during the wettest periods, the water table at Walrag 3 fails to reach the bog surface, its maximum height remaining 5 cm below the surface. Consequently at Walrag 3, which lies 50 m closer to the cut edge of the bog, there is never a time when the surface vegetation is fully saturated. This accords with Gilman's (1994) observation that drainage may cause the acrotelm of a bog to become permanently de-watered.

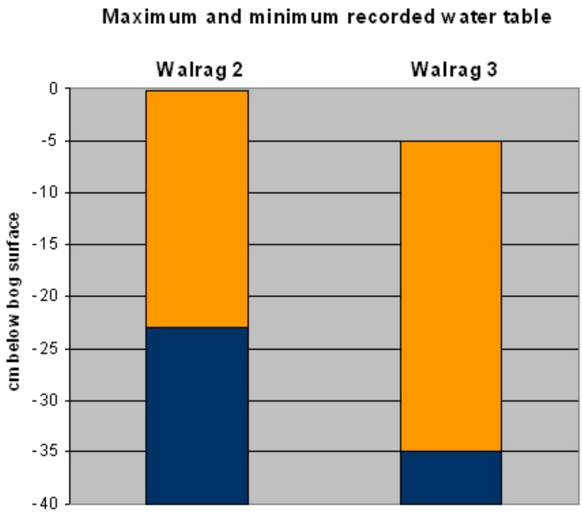


Figure 28. Behaviour of water table at particular locations on Cors Caron NNR raised bog, mid-Wales – maximum and minimum water levels.

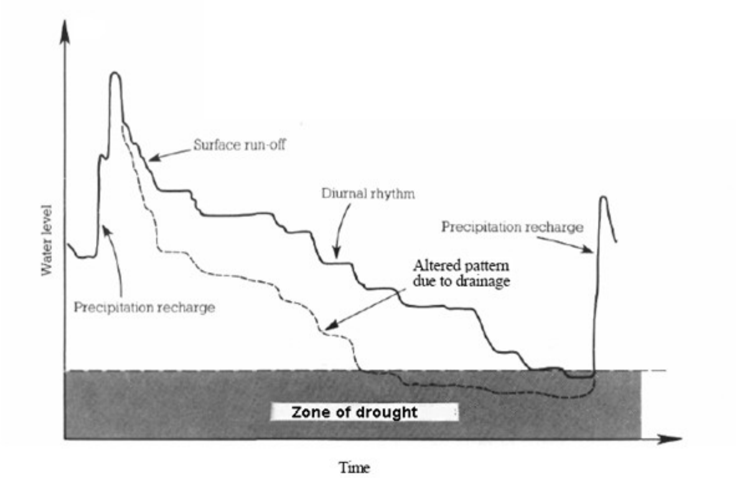
The two bar-graphs display data showing the maximum height attained by the water table (uppermost limit of orange block) and the minimum height attained by the water table (i.e. maximum extent of oxygen penetration – lowermost limit of orange block) for two locations on Cors Caron. Below the maximum limit of oxygen penetration the peat remains constantly waterlogged (dark blue shading). Walrag 2 is located 135 m from peat cuttings at the edge of the bog, while Walrag 3 is located only 80 m from these cuttings.

Data supplied courtesy of Countryside Council for Wales

The more detailed behaviour of the water table under such circumstances can be examined by considering a continuous water-table trace obtained from a blanket mire in Argyll. Coladoir Bog NNR is an area of oceanic blanket mire, and the water-table trace obtained for a 15-day period can be seen in Figure 29.

The trace begins with a day or so of heavy rain, during which time the water table rises sharply. There is then a period of run-off before the traces enters a step-wise series where the water table falls during the day but shows little fall at night. This continues for some days until, just as the surface vegetation begins to experience conditions of drought, there is another rain event and the water table rises rapidly again. A second generalised trace has been added to the graph. This second trace represents the pattern of water table behaviour when drains remove water from the acrotelm more rapidly. It can be seen that run-off is greater and thus the water table in the acrotelm falls more rapidly. Consequently the surface vegetation begins to experience drought conditions sooner and so drought conditions must be endured for longer periods than before. The implications of this for ecological processes are potentially highly significant, but before exploring the nature of such changes it is again worth highlighting the fact that, for many blanket mires, slope is also a critical factor in determining the possible extent of drainage impacts.

Figure 29. Behaviour of water table at Coladoir Bog NNR, Isle of Mull, Argyll - a continuous record.



The continuous solid line represents the recorded pattern of water-table behaviour in a T2 high ridge on Coladoir Bog over a 15-day period. Towards the end of this period it can be seen that the water table has fallen to such a degree and for such a long period that certain *Sphagnum* species would by now be suffering from drought damage. The falling dashed line represents a hypothetical water-table pattern in the presence of a drain. The water table experiences greater initial run-off before entering a period of diurnal rhythm. Consequently the water table falls into the drought zone more rapidly.

Unpublished data from the Nature Conservancy Council

9.1.4.3 Slope and acrotelm de-watering – the 'topographic index'

Holden (2005a) emphasises the fact that the topographic context of a drain (*i.e*. its position within the landscape) is a major factor influencing the degree of direct hydrological impact which any given drain will have on a blanket mire system. He describes the way in which 'topographic index' values can be constructed for the length of a drain, based on slope angle and the area of slope which feeds into each section of drain. This set of values can then be used to create a map of the blanket mire terrain and drainage impacts. The map indicates the extent and degree of hydrological change experienced by areas of ground lying downslope from any drains.

Such drains intercept surface, near-surface and sometimes even deeper seepage and re-direct this water down the lines of the drains. Ground lying below these drains is thus deprived of this water and consequently experiences drier conditions than before (more particularly, has a reduced propensity to saturation). The distances over which such impacts may be felt can be considerable, depending on the slope and immediate catchment size. Holden's (2005) map of a study area in Upper Wharfedale shows that individual drains may have impacts which can extend as far as 400 m or more downslope. Figure 30 shows a similar exercise undertaken by Holden (2009) for Mickleton Moor in the northern Pennines.

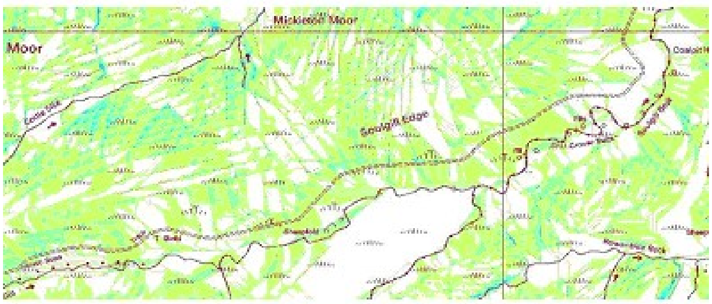


Figure 30. Topographic index calculated for blanket mire at Mickleton Moor in the northern Pennines.
Green shading indicates areas predicted to be drier downslope from individual drains in a blanket bog landscape, according to a calculated topographic index model derived from topographic data for the site. The extent of the map is approximately 1.6 km x 0.75 km.

Reproduced with kind permission from Holden (2009)

This ground is criss-crossed by a large number of drainage ditches, each of which has its own calculated area of downslope impact, shaded green. While some sections of drain have only a small effect (narrow green shading), other drains or sections of drains have wide swathes of green shading downslope, indicating a more extensive impact.

It is important to recognise that the topographic index model presented by Holden (2005a, 2009) is based on landform characteristics alone and offers no immediate insight into the nature of ecological change which may result from such hydrological impacts, but it is reasonable to assume that there would indeed be ecological consequences from such change.

9.1.4.4 Drainage and biotic change : Coulson, Butterfield and Henderson (1990)

One of the most detailed studies of biotic changes yet to have been carried out into the effects of drainage on blanket mires was undertaken by Coulson, Butterfield and Henderson (1990) on four sites in the Pennines – Moor House, Cumbria, Waskerley, Co. Durham, and Oxnop and Gunnerside, Yorkshire. Coulson *et al.* (1990) investigated the effects on water table, soil water-content, vegetation, invertebrates, decomposition rates and the nutrient content of heather, within areas which had been drained (moor-gripped) at intervals of between 16-30 m. The drains had been dug between 8 and 30 years previously, and all were functioning effectively. It is worth noting that Coulson *et al.* (1990) make no mention of boardwalks associated with their dipwell measurements, and as the sites were measured at monthly intervals it might be expected that some impact from visitor pressure would be felt. Coulson *et al.* (1990) noted that several of the dipwell holes collapsed during the study.

Coulson *et al.* (1990) found no effect on soil water-content in the grips at Moor House, but some effect at the lower-altitude Waskerley site. Soil water-content is, however, more a measure of effects on the catotelm than on the acrotelm. Measurement of the water table indicated that draw-down was more pronounced at lower altitude, although all sites showed a lowering of water table immediately downslope from a drain. At the mid-point between ditches, the draw-down at the high-altitude Moor House was only 3.3 mm, whereas at lower altitude differences of 107 mm were recorded.

Coulson *et al.* (1990) found that there was some reduction in *Sphagnum* cover downslope from ditches at lower altitude but a very marked increase in the cover of grasses downslope from a drain. In contrast, no significant vegetation change was noted associated with the drains at the higher-altitude Moor House, although it is worth noting that the vegetation consisted only of heather (*Calluna vulgaris*), crowberry (*Empetrum nigrum*) and hare's-tail cotton grass (*Eriophorum vaginatum*). There was no moss cover at all at Moor House.

At high altitude, Coulson *et al.* (1990) found no differences in the invertebrate populations above and below drains, but at lower altitude there was significant change downslope from the drains. Similarly, there was no significant difference above or below the ditches at Moor House in terms of measured decomposition rates, whereas at lower altitude a significant difference was detected immediately downslope from the drains.

Coulson *et al*. (1990) conclude that at high altitude the constant supply of precipitation reduces the effectiveness of drains, whereas at lower altitude the impact of drainage is more pronounced. This is almost certainly the case and matches with the conclusions of Braekke (1983). However, from this study by Coulson et al. (1990) it is impossible to say what the effect of drainage at high altitude might be on a natural expanse of blanket mire because the study is all based on effects observable some time after drainage has occurred – in the case of Moor House after 30 years of drainage. It is worth noting that the study site at Moor House possessed no *Sphagnum* cover and clearly represented something more akin to Rodwell's (1991) 'dreary prospect' of endless M20 blanket mire mixed with the heather cover of M19 blanket mire. It would therefore appear that Coulson *et al*. (1990) have the same methodological problems of Stewart and Lance (1983) in studying a bog system which has already undergone major change to the point where the system as a whole has altered, rather than displaying immediate effects close to drain margins.

It is significant that the distances between the drains given by Coulson *et al.* (1990) fall within the range of drain-spacings studied by Braekke (1983), with the largest drain-spacing noted by Coulson *et al*. (1990) still being associated with a maximum between-drain draw-down of more than 70 cm in the Norwegian studies. This is clearly a much greater draw-down than is typical of a natural site, and could be expected to give rise to a vegetation much more typical of heath than bog.

Unfortunately Coulson *et al.* (1990) did not include studies into the macrofossil remains stored in the peat archive of their study sites. Such evidence could have gone some way to demonstrating conclusively whether the vegetation which existed prior to drainage at, for example, Moor House was significantly different from that found today. The opportunity to investigate former vegetation cover at study sites is one of the peculiar features of peatland systems, but this opportunity has rarely been used in studying habitat change in relation to human impact, being instead largely restricted to studies of vegetation responses to climate change. This ecological response to human impact is explored in more detail in the next section, below.

9.1.5 Ecological response to drainage

Given that the majority of the drainage effects described so far occur within a vertical zone which may be no more than 20-30 cm thick, the overall change in average water-table position of a drained bog may be quite small, but the effect of even a small decrease in saturation (or conversely, increase in drought conditions) can have significant ecological implications.

Indeed Ivanov (1981) emphasises that it only takes a very small change (of 4-5 cm) in the average water table of a bog to bring about substantial alterations to the composition of bog moss communities. Thus the effects of drainage may not lead inevitably to wholesale loss of bog vegetation; there may instead simply be a shift in the composition of the existing vegetation assemblage.

Such shifts in species composition occur naturally in bog vegetation whenever there is a shift in the climate. As discussed in Appendix 3, Section 24.1 of the present report there is clear evidence from the peat archive that, during the last 2,000-8,000 years, bog surfaces have on repeated occasions changed their vegetation and surface patterns in response to changes in climate. What may be less widely understood is that the effects of a drying climate are not so different from the effects of drainage. Consequently it might be expected that drainage would bring about changes which are at least similar, if not identical, to those which have been recorded in relation to climate change. Indeed one of the difficulties confronting palaeo-ecologists looking at evidence in peat bogs for climate change during the last 2,000-4,000 years is that a distinct change from wetter to drier conditions in the archive of a bog may indicate a shift in climate, but it may instead indicate the effect of human drainage and land-claim – a process which continues on peat bogs to the present day.

While it may be a headache for the palaeo-ecologist, this similarity of responses is of considerable benefit to the modern peatland ecologist because it provides an insight into the kinds of change which might be expected on a peat bog in response to drainage. Thus Barber (1981) and subsequent workers have demonstrated that during dry phases in the climate, peat bogs have tended to show expansion of T3 hummocks and T2 high ridges (*sensu* Lindsay, Riggall and Burd, 1985) at the expense of A1 hollows and T1 low ridges. Indeed there are periods of time when A1 hollows are entirely absent and even T1 low ridges appear to be rare features. These periods correspond with the driest climate phases in recent millennia.

Goode (1970, 1973) and Ivanov (1981) provide the eco-hydrological explanation of this response by describing how ridges of peat, alternating with water-filled hollows, can provide extremely fine control of water flow through the peat system simply by altering the proportions of ridge and hollow. If, in addition, the ridges and hollows can vary their hydraulic properties by changing the nature of the vegetation assemblages which dominate each of these components, an even finer level of control can be achieved. There is of course no sense of 'control' involved in this process – feedback processes mean that drier conditions tend to favour ridge communities which thus expand, and even drier conditions favour particular vegetation assemblages which produce denser peat on these ridges. Thus the rate of water loss from the system as a whole is reduced still further.

Much the same ecological feedback processes might be expected in response to drainage. Woike and Schmatzler (1980) illustrate just such a sequence in relation to various stages of drainage in a raised bog. This sequence has been adapted for blanket bog conditions in Figure 31, the main difference being that in lowland environments the final stages of drainage often involve colonisation by woodland, whereas in the oceanic blanket mires of Britain and Ireland this rarely if ever occurs.

Four stages of drainage can be identified from Figure 31. With a relatively limited fall in the average water table, a *Sphagnum*-rich vegetation is retained but the microtope pattern becomes simpler, with the loss of A1 hollows, while the species particularly associated with such hollows are also lost. The bog may still look vigorous and actively-growing, but it has a simpler structure and a more limited *Sphagnum* assemblage than before. Further lowering of the water table leads to loss of any remaining complexity in the microtopography, with the surface becoming a smooth, moss-covered layer in which only *Sphagnum* species typical of T3 high hummocks are found – generally *S. capillifolium*. The cover of higher plants is often dominated by dwarf shrubs. In places the moss cover is lost, leaving pockets of haplotelmic mire surface.

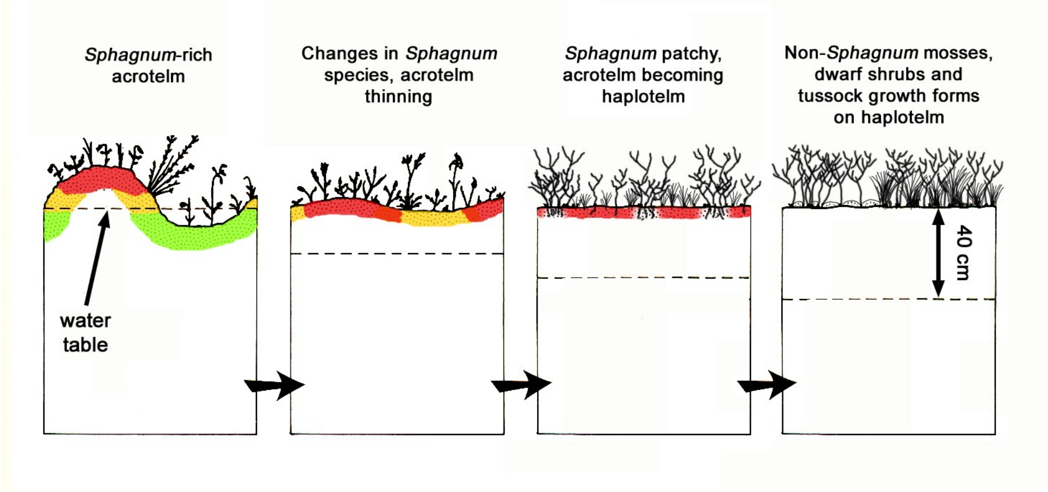


Figure 31. Various stages of change resulting from progressive drainage of blanket mire habitat.
The effect of a steadily falling water table on the microtopography and vegetation of a blanket mire is likely to occur as a continuum of change rather than as distinct stages, but the four stages illustrated here show the key elements of change. Firstly the microtopography becomes simplified with the loss of A1 *Sphagnum* hollows and the species associated with these features are lost too. Next the *Sphagnum* carpet becomes reduced to a smooth monoculture of *S. capillifolium* with patches of bare ground, beneath a sward of dwarf shrubs. Finally the *Sphagnum* carpet is lost altogether, creating a haplotelmic surface which may support non-*Sphagnum* mosses together with dwarf shrubs or tussock-forming species.

Adapted from Woike and Schmatzler (1980)

In the final stages, the *Sphagnum* cover is lost completely to form a haplotelmic surface. The former *Sphagnum* carpet may be replaced by non-*Sphagnum* mosses such as *Hypnum jutlandicum* while the sward of higher plants may be dominated by dwarf shrubs (where *Hypnum* is dominant) or by tussock growth forms of species such as hare's tail cotton grass (*Eriophorum vaginatum*) or deer grass (*Trichophorum cespitosum*) where the moss cover is patchy or absent. Woolly hair moss (*Racomitrium lanuginosum*) may also dominate large patches of ground to the exclusion of most other species. Precisely this pattern of vegetation-shift within the microtopography is described by Komulainen *et al.* (1999) for a drained bog in southern Finland. After 30 years of drainage the A1 hollows were becoming overgrown with typical hummock species such as heather (*Calluna vulgaris*), but when the drains were dammed the hummock species rapidly retreated back to the hummock-tops and typical hollow species re-established themselves.

It is important to understand, however, that the sequence shown in Figure 31 only readily applies to a microtopography ranging from T3 hummocks to A2 mud-bottom hollows (*sensu* Lindsay, Riggall and Burd, 1985). This is because, as Ivanov (1981) points out, deeper pools, such as A3 and A4 pools, and what Ivanov (1981) terms 'bog lakes', become progressively too deep to be able to respond to drying conditions by infilling. They represent relatively fixed structures within the peat landscape and are therefore less able to adapt their structure if conditions become drier, especially as deeper pools have relatively little or no vegetation to assist in creating this structural transformation. Ivanov (1981) observes that large bog lakes are more likely to suffer collapse and erosion if conditions become substantially drier.

Evidence that drying within the terrestrial zone may occur while deeper pools persist can be found in a number of places where wet pool systems have, for example, been surrounded by afforestation. Many examples can be seen in the Flow Country, but a particularly clear example can also be found in the Slieve Aughty Mountains in Co. Galway. A small pool system possessing A3 pools has been completely surrounded by a conifer plantation for about 30 years (see Figure 32).

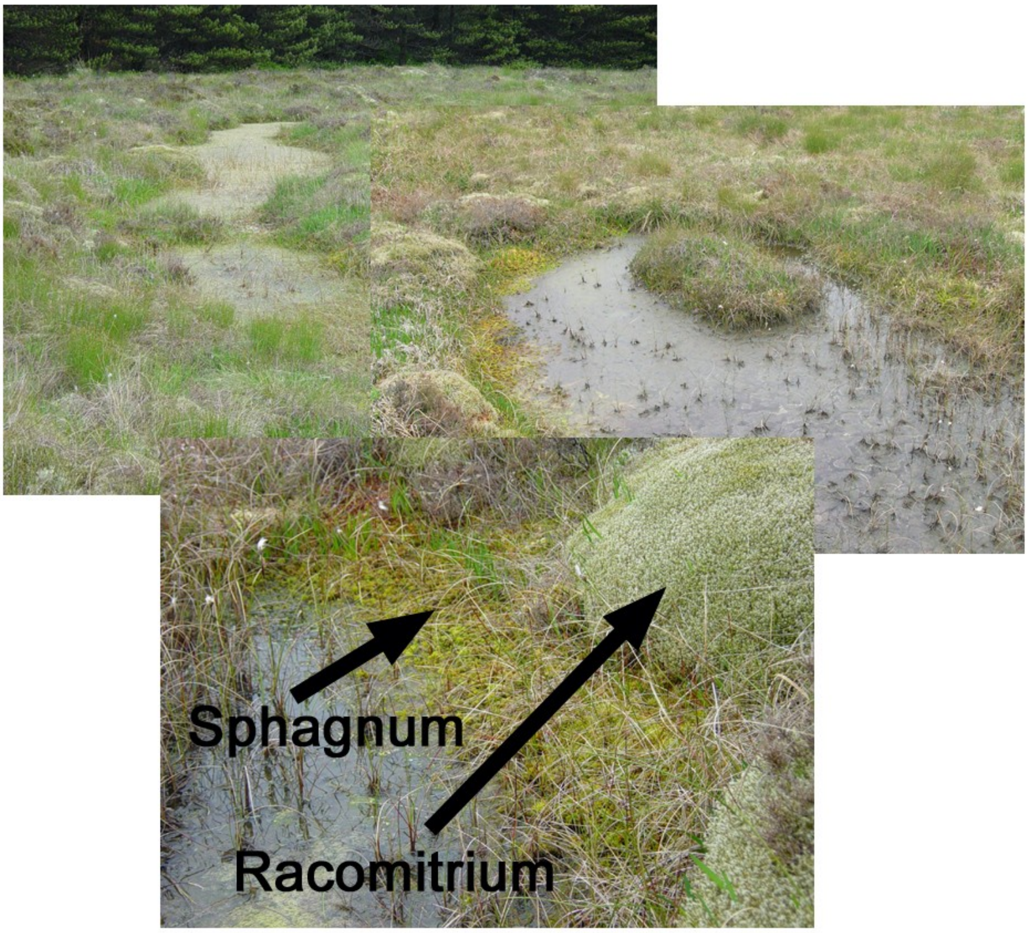


Figure 32. Photo composite of an A3 pool system on Cashlaundrumlahan, in the Slieve Aughty Mountains, Co. Galway
The upper photographs show the general nature of the ground, the proximity of the forest plantation, and the characteristic shape of A3 pools in this small pool system completely surrounded by a forest plantation. The upper-right photograph shows a pool which is now very shallow, with a base of 'mud-cracks', suggesting that for periods of the year the pool is now dry. The yellow fringe around the margin of the pool in both the upper-right and lower-centre photographs consists of pool *Sphagnum* species such as *S. cuspidatum*, while the lower-centre photograph shows how the woolly hair moss (*Racomitrium lanuginosum*) now dominates the majority of the terrestrial parts.

R A Lindsay 2004

The pools remain but increasingly resemble A2 mud-bottom hollows, while the terrestrial zone is now almost completely dominated by woolly hair moss (*Racomitrium lanuginosum*), with only a fringe of *Sphagnum* persisting around the pool margins. That the *Racomitrium*-rich areas were formerly *Sphagnum*-dominated is easily demonstrated – a hand plunged into the *Racomitrium* hummocks will find abundant *Sphagnum* remains just beneath the layer of *Racomitrium*.

This particular characteristic of peatland systems – that they preserve a record of what existed in the past – offers a very useful tool for investigating the long-term impacts of drainage but it is a tool which has been surprisingly little-used. Almost all studies of the vegetation record stored within the peat have been used either to investigate processes of bog development and natural successional sequences, or to shed light on the pattern of past climate change.

One notable exception to this focus in palaeo-ecological research is a study undertaken by Hughes *et al*. (2007). The reasons for the decline in cover of typical bog vegetation, and specifically the decline and eventual loss of the major peat-forming *Sphagnum austinii* (formerly *S. imbricatum*), are investigated in terms of possible human impacts.

Hughes *et al*. (2007) identify that increases in purple moor grass (*Molinia caerulea*) and birch (*Betula pubescens*) at the expense of typical bog species may be linked to increased nitrogen deposition. In contrast, the general replacement of wet bog vegetation by hummock and dry ridge communities across Rhos Goch Common is linked to the impacts of peripheral peat cutting and consequent general de-watering of the site. Meanwhile the loss of *S. austinii* from Cors Caron (Tregaron Bog) is shown to be temporarily coincident with the intensification of surrounding land use which began in Roman times.

Relatively little effort, however, has been devoted to the explicit investigation of more recent changes in the preserved palaeo-archive of peat bog systems where drainage systems are known to have been dug and can be dated, at least approximately, from maps of a known date. Casual observations made while investigating other aspects of the peat archive suggest that this would be an extremely useful area for investigation, but an area which has so far remained somewhat neglected.

9.1.6 Cracking, piping and slope-failure

Once a bog surface reaches the third stage in Figure 31, there is the likelihood that significant physical changes will occur to the peat matrix. They may occur earlier than this, but such effects become more probable by Stage 3, and are almost inevitable by Stage 4. The wetter the peat was in its undrained state, the more marked such changes will be. This is because as the peat shrinks there is no longer sufficient organic material to fill the space originally occupied by water in the undrained condition. Consequently substantial cracks and voids begin to appear in the peat matrix, and these can act in a variety of ways to channel yet more water from the peat.

Holden (2005a) has identified that the phenomenon of peat ‘piping’, whereby convoluted subterranean pipes develop within the peat matrix, is twice as frequent beneath a peat surface which has been drained and this effect increases over time. Holden and Burt (2002) show that these pipes can contribute significantly to the overall hydrological budget of a bog, providing a more rapid means of water loss than is possible through general surface and sub-surface seepage.
Cracks can also provide a means by which large surface flows during heavy rainfall events may be channelled deep into the body of the peat matrix.

This is even more the case when drains are present because cracking *beneath* the base of the drain is a common phenomenon. Such cracking occurs during extended dry periods, but once such cracks have formed they are difficult to eradicate and will tend to open whenever conditions are dry. This is of particular relevance in the summer months when dry periods may be broken by powerful convective storms with heavy rainfall.

Warburton, Holden and Mills (2004) point out that this combination of conditions can lead to water being channelled down to the interface between the peat and the underlying mineral sub-soil. Lubrication of this interface by such channelled rainwater can trigger slope failure, leading to a peat avalanche, or peat-slide. Warburton *et al.* (2004) and Lindsay and Bragg (2004) identify a considerable number of such bogslides from around the world and highlight the fact that there is often evidence of prior drainage, and conclude that such drainage can predispose peat-covered slopes to slope failure.

A peat avalanche, or bogslide, clearly involves disruption of a substantial quantity of peat. In the case of the very large bogslide at Derrybrien, Co. Galway, the slide was estimated by AGEC (2004) to have displaced and disrupted 450,000 m^3 of peat (Lindsay and Bragg, 2004). Not all of this will eventually be oxidised and lost, but there are substantial areas of ground where relatively thin layers of peat have already vanished through oxidation and particulate loss, and much organic mater which entered the main river system will since have been oxidised by the flow and turbulence of the river.

Essentially, drainage reduces the level of waterlogging in the peat and thus exposes the peat *in-situ* to greater penetration of oxygen – with consequent carbon release – or alternatively it renders the peat more prone to mass movement and slope failure with consequent oxidation of the peat which has suffered disruption as a result. It is also worth adding that the obligations of the EU Water Framework Directive are relevant here, because the downstream effects of peatslides can be substantial and may be felt over an extended period of time.

9.2 Carbon balance of drainage

Given that the purpose of drainage is to reduce waterlogging but the process of carbon sequestration and long-term retention depends on waterlogging, there is clearly a direct conflict between these two processes. The extent to which differing degrees of drainage may actually halt carbon sequestration and even lead to carbon loss is not yet well understood. Drainage impacts equivalent to Stage 1 in Figure 31 (above) may not in fact halt carbon sequestration but instead change the species which are undertaking this sequestration. In addition, increases in macropores (Holden and Burt, 2002) and small-scale cracking facilitates greater penetration by air, while increases in larger scale peat-piping are now known to occur in drained areas of blanket bog (Holden, 2005b). However, not enough is known about these processes to be able to predict the possible effects on the net carbon balance with any certainty.

More intensive drainage, where the peat-forming vegetation is lost and the bog begins to develop a haplotelm rather than a catotelm, is associated with a number of identified changes in relation to carbon sequestration. A considerable amount of work has been undertaken in relation to this in Finland. However, it must be remembered that a large proportion of Finnish study sites are fens rather than bogs, and the bogs are generally frozen for several months of the year and thus unable to sequester carbon (though they can still decompose), while the oceanic raised and blanket mires of Britain and Ireland have a much smaller period of vegetation dormancy or even, in some years in more western regions, perhaps no dormant period at all.

The essential carbon-balance processes associated with drainage can be summarised as increased oxidation of the peat matrix leading to CO_2 release to the atmosphere and DOC loss in water outflows. Balanced against this is the potential to reduce CH_4 emissions because the methane can be more readily oxidised before it escapes to the atmosphere. Give that CH_4 is 25x more powerful a greenhouse gas than CO_2, the oxidation of one molecule of CH_4 as a result of drainage means that the system is able to release up to around 9 additional molecules of CO_2, also as a result of drainage, and still remain in carbon credit. The critical things to resolve, therefore, are the extent to which carbon sequestration is disrupted by drainage, and the scale of CO_2 release following drainage, compared with the rates of CH_4 release in the natural state. It is also important to establish whether methane continues to be released even when a site is drained.

9.2.1 Drainage and carbon sequestration

The question of whether carbon continues to be sequestered long-term in a drained peat bog is probably the most difficult question to answer. This is because, as Belyea and Clymo (2001) observe, the carbon taken up in the acrotelm may spend a considerable period of time in the acrotelm – often many decades – and it is only when this is passed into the catotelm for long-term storage that the carbon can be considered to have been successfully sequestered. Determining whether or not this is the case requires studies which run over extended periods of time, and there have been few, if any, studies which can provide such information.

In practice, it is possible to distinguish vegetation communities which are normally considered to be peat-forming, and this is what the European Commission adopts as a pragmatic definition of 'active' bog, along with the recognition that microtopography is an essential part of a living mire surface (European Commission, 2003; and see EU Habitats Directive definitions : JNCC website). Where a bog supports a vegetation which is not normally considered to be peat-forming (in effect a haplotelmic bog), it is generally assumed that little or no carbon sequestration is occurring.

In any carbon-balance study, therefore, it is important to note that while a bog which is now haplotelmic may be actively losing carbon in the form of CO_2 and DOC, the bog will generally also have lost its original capacity to absorb carbon. The loss of this carbon sink must therefore be added to the values of active carbon loss to produce a full picture of loss. As discussed earlier in Section 7.3.2 of the present report, estimates of the natural carbon-sequestration rate vary between roughly 30 and 70 g C m^{-2} yr^{-1}. The lower part of this range is obtained from boreal raised bogs while the uppermost value is given specifically for British blanket mires (Adger *et al*., 1991; Turunen, 2003). It is important to understand that *sequestration* rates concern the rate at which material is added to the catotelm. This must be balanced against *decomposition* within the whole thickness of the catotelm in order to produce an overall net carbon balance. Consequently the higher rates of sequestration noted for oceanic blanket mires compared to boreal mires do not mean that blanket mires necessarily have the thicker peat deposit.

Thus a blanket mire which is now reduced to a haplotelmic state can be regarded as already 'losing' some 50-70 g C m^{-2} yr^{-1} which is no longer being sequestered, before considering losses in the form of CO_2 and DOC release resulting from this damage. In fact the 'standard natural cubic metre of peat' used in the present report (see Section 8) is clearly rather conservative in this respect because it assumes a natural sequestration rate of only 23 g C m^{-2} yr^{-1}. As we shall see in the next section below, this quantity of carbon is very much smaller than the amount of carbon potentially lost as CO_2 through intensive drainage, but this loss of sequestration capacity is by no means trivial and should not be overlooked.

9.2.2 Drainage and carbon dioxide release

A considerable quantity of literature exists about this subject, but the extent to which this literature clearly and specifically addresses the effect of drainage on CO_2 release from oceanic raised and blanket bogs, rather than the more broadly-defined 'western European peatlands', or on Scandinavian 'boreal mires', is less clear. Indeed the problem is further complicated by the fact that many studies which are described as having been carried out on 'natural' or 'pristine' bogs have in fact been undertaken on drained sites.

The scale of this problem has been highlighted by the International Mire Conservation Group (IMCG) in relation to sites in Finland. During its 12th Biennial Conference held in Finland in 2006, the IMCG did not see a single 'pristine' mire, even within national parks, although the Conference had been assured that many of these sites were pristine (see Resolution to Finland, IMCG website). This same problem occurs in many carbon-balance papers where sites which are assumed to be 'natural' are not. Thus, for example, Hargreaves, Milne and Cannell (2003) examine the carbon balance of conifer plantations on peat (a paper explored in more detail in Discussion Topic 4 of the present report). Auchencorth Moss is included in this study as an 'undisturbed' site, although in fact it has an extensive network of drains across much of the site.

This is by no means an isolated example and highlights a real problem for researchers. Ingram's (1982) paper describing the ground-water mound theory for raised bogs informs much of our present understanding about the natural hydro-morphology of raised bogs. His study was based on Dun Moss, however, and this raised bog has a very substantial ditch which cuts deeply into the site. Indeed Lindsay and Immirzi (1996) have identified the fact that no examples of undamaged lowland raised bog survive in Britain – every site has suffered some form of drainage whether in the form of marginal peat removal, drainage of the surrounding lagg fen, or the actual cutting of drains across the site. Development of the Cuthbertson plough in 1945 has also led to quite remarkably widespread drainage of peat soils throughout the uplands of Britain since that time. In addition, long-established drainage linked to the demarcation of community boundaries, management of water for quern mills, and the widespread cutting of peat banks, combine to add an even older pattern of drainage to such blanket mire landscapes. Many papers describing the hydrological and gas-exchange characteristics of raised bogs or blanket mires fail to recognise, or at least do not sufficiently acknowledge, the extent to which the study sites are already affected by existing drainage.

This confusion about what is 'natural' and what is 'drained' highlights a very considerable problem in linking peatland drainage to quantities of carbon released. Mapping every drain line across a site will give a picture of overall drainage intensity, but drainage effects caused by peat cuttings on the site margin, or by the unseen pattern of peat pipes within the peat body, or by the network of cracks which develop in the peat, all serve to complicate the pattern of drainage in ways which are difficult to quantify. Further complications are added by the fact that individual parts of the peat surface, such as A1 hollows or T1 low ridge, will be more sensitive to drainage effects than will T2 high ridge or T3 hummocks. Thus even if a site is completely drained in an exactly regular fashion, it still remains very difficult to quantify the degree of drainage impacts associated with any specific study plot.

The intensity of drainage at a specific location can be assessed fairly readily by measuring the water table, but then the natural water table shows variation over time and this natural variation must first be determined. If the site is already damaged, it is not possible to do this. Furthermore, water-table measurements must inevitably be taken from the specific location of the water-recording instruments, yet the water table may vary by several centimetres, and behave rather differently, over quite short horizontal distances on a bog surface because of the variations in microtopography. It is thus important that water-table measurements and gas-exchange measurements are taken from similar elements of the microtopography if a direct link between water-table and gas exchange is to be made.

Where gas-exchange measurements are obtained using small chambers and these chambers are placed sufficiently close to the water table instruments, it is possible to be confident that both sets of instruments are experiencing the same water table. In the case of eddy covariance measurements, however, gas exchange is measured using a mast which samples the overall pattern of gas exchange from, potentially, rather a large area. In this case the link between intensity of drainage and CO_2 release is much less direct because the gas is measured as a general emission whereas the water table is measured at one or more specific points and these points may or may not reflect the general behaviour of the water table across the site, particularly if there are drains, areas of peat cutting or other forms of disturbance.

All of these various factors mean that considerable caution must be applied when attempting to compare the relative CO_2 (and CH_4) balances described for 'natural' sites and 'drained' sites. Putting aside, therefore, the distinction between 'natural' and drained sites, the relationship between drainage and CO_2 release has been the subject of many studies during the last 20 years or so.

Some of the earliest of this work was undertaken by Armantano and Menges (1986), who brought together a large amount of data from around the world in order to produce a synthesis of carbon losses from peatlands resulting from anthropogenic impacts. Immirzi *et al*. (1992) provided a further analysis based on the work of Armentano and Menges (1986) but also incorporating additional evidence which gave the review more of a UK focus. However, neither review attempted to relate carbon losses to a measured intensity of drainage or to any measured water tables – the peat soil was simply considered to be 'drained'.

Armentano and Menges (1986) give a regional figure for western European peatland of 920 g C m^{-2} yr^{-1}, while Immirzi *et al*. (1992) give an estimated value of 1100 g C m^{-2} yr^{-1} for lowland peat in Britain. It would probably be reasonable to assume that losses in blanket mire areas would be smaller because the wetness of climate slows down the processes of slumping and oxidation.

There has also been a great deal of work carried out in Finland during the last two decades, looking at gaseous exchange and land-use impacts on peatland systems, although the main focus of such work has been on methane rather than carbon dioxide. Nevertheless, three studies looking at soil respiration (*i.e*. CO_2 release) in 'virgin' and drained peatlands merit closer examination.

9.2.2.1 Silvola, Alm, Ahlholm, Nykänen and Martikainen (1996)

One of the most extensive and widely-reported early studies of Finnish mires and gaseous balance (Silvola *et al*., 1996) looked at a range of Finnish mire types, some of which were described as drained, others as 'virgin' untouched sites. Measurements of CO_2 flux were obtained using closed chambers from 6-12 locations on each site, and water-table measurements were obtained for these locations on each occasion that gas flux was measured.

Silvola *et al.* (1996) noted that about 90% of the carbon fixed annually by the acrotelm of a mire will be re-released. Thus only a 10% increase in this quantity will mean that the mire is no longer accumulating carbon, meaning that quite small changes to the system might become highly significant. They measured the quantities of CO_2 re-released from a variety of mire types ranging from sedge fen to *Sphagnum* bog and compared these values of CO_2 release with measurements of water table and soil temperature.

Of the 'virgin' sites, Silvola *et al*. (1996) found that the open *Sphagnum*-dominated bog sites re-released the smallest amount of CO_2, with values of 52-144 g CO_2-C m^{-2} yr^{-1}, while wooded bogs rich in hare's-tail cotton grass (*Eriophorum vaginatum*) released between 292-339 g CO_2-C m^{-1} yr^{-1}, which was more CO_2 than any other undisturbed site type. They noted that the *Sphagnum*-dominated site already had a rather low water table, but that it maintained its low levels of emission even if the water table was lowered slightly more.

Given that Silvola *et al*. (1996) themselves noted the unusually low water table recorded for the open *Sphagnum* bog, and the undoubted propensity for trees and cotton grass to grow more vigorously on drying sites, it is tempting to question whether these 'virgin' sites are truly undisturbed or whether these, like so many reportedly 'pristine' sites shown to the IMCG 2006 Conference, have all suffered at least some drainage effects. It is thus perhaps unwise to assume that these figures give a true picture of emissions from ‘virgin’ or ‘pristine’ sites, but the value of this study is what it tells us about CO_2 emissions and changes in water levels.

Thus Silvola *et al.* (1996) found that a reduction in water-table values was almost linearly related to an increase in CO_2 emissions. A decrease in the water table of 20 cm resulted in a doubling of CO_2 emissions. A fall of 30 cm resulted in losses of between 465-606 g CO_2-C m^{-2} yr^{-1}, although if the water table fell by more than 30 cm there was no further increase in CO_2 emissions. They noted that this level of CO_2 loss greatly exceeds the primary productivity of boreal forested peatlands and would require a level of productivity equivalent to temperate deciduous forest to make good such drainage losses.

Overall, across the range of mire sites, Silvola *et al.* (1996) found that a 1 cm lowering of the water table resulted in an average increase of 9.5 g CO_2-C m^{-2} yr^{-1} in CO_2 emissions. Something else noted by Silvola *et al.* (1996), however, was that these emissions were sensitive to temperature change and that the wettest sites were most sensitive to such change. It was estimated that a rise in temperature of 2-4°C could increase CO_2 emissions by 30-60%.

9.2.2.2 Minkkinen and Laine (1998)

This study made use of detailed measurements taken in the 1930s when an extensive programme of mire drainage was instigated. During the original drainage programme the lines of proposed ditches were carefully mapped and peat thickness measurements taken along these lines. Minkkinen and Laine (1998) were able to re-find these original drain lines and repeat the original peat-thickness measurements for a large sample of sites and site types throughout Finland. From this, they hoped to compare the carbon stored within the mire prior to drainage with the carbon store which had developed over the 60-year interval since drainage.

It is important to understand several things about this study. Firstly, all the site-types involved were fens rather than bogs. The closest type to oceanic ombrotrophic bog consisted of a nutrient-poor fen dominated by *Sphagnum papillosum* (called LkR by Minkkinen and Laine). Further discussion here will thus be restricted to this one site type.

Secondly, a limit of 2 m peat thickness was placed on the selection of sites for the repeat study, as this thickness is most typical of peat which is subject to forest drainage in Finland.

Thirdly, the original measurements from the 1930s did not include any measurement of bulk density, carbon density or total carbon and thus there was no direct way of knowing how much carbon was contained in the peat profile prior to drainage. Minkkinen and Laine (1998) therefore measured the carbon density in cores taken from 188 undrained mires of similar types to the drained mires and then used a regression model to predict what the original carbon density of the peat in the drained sites would have been prior to drainage. This regression model was little more than 50% reliable and resulted in a large degree of uncertainty.

Within the context of these limitations, Minkkinen and Laine (1998) found that all site types together showed an average subsidence of 22 cm, with the LkR sites showing least subsidence, ranging from 16-19 cm. They also found that all sites showed an increase in total carbon following 60 years of drainage, with the LkR sites generally showing the largest increase, ranging from 100-275 g C m^{-2} yr^{-1} for the 60-year period.

Minkkinen and Laine (1998) thus conclude that the majority of subsidence is caused by dewatering and compression of the peat by the increased growth of trees and shrubs, and that oxidative wastage plays only a relatively minor part in the observed fall in the ground surface. They also attribute the gain in carbon to the increased biomass produced by the trees, shrubs and their roots (particularly large quantities of fine roots in the case of LkR sites), this increased growth being stimulated by lowering of the water table.

It is worth highlighting several points at this stage:

- Minkkinen and Laine state that they included within their measurements of modern peat thickness fresh growths of non-*Sphagnum* mosses which had replaced the original *Sphagnum* cover; consequently they admit that subsidence of the original peat surface may have been substantially greater than they recorded;

- although the data indicate significant increases in carbon content for LkR sites, the values are accompanied by large elements of uncertainty arising from the original regression model used to predict former carbon densities;
- the tree-stand volume for LkR sites is substantially smaller than that measured for the other site types, suggesting that the nutrient-poor LkR sites may not lose so much carbon through oxidation because the trees are rather small, but it would be instructive to see what the carbon balance on such sites would be if the tree cover were encouraged to grow more vigorously, or the study were left to run until the trees on LkR were more substantial.

In summary, therefore, although Minkkinen and Laine (1998) recorded an increase in total carbon following 60 years of drainage on the relatively thin peat of a solute-poor fen, it is not clear that this could be directly translated into conditions on a blanket mire site in Britain. Furthermore the gains can only be seen as temporary because eventually the oxidative loss of peat would inevitably exceed the accumulation and storage capacity of the forest. Certainly Turunen (2008), in a review of Finnish peatlands and carbon storage, suggests that the gains noted by Minkkinen and Laine (1998) are temporary.

9.2.2.3 Minkkinen, Laine, Shurpali, Mäkiranta, Alm and Penttilä (2007a)

This study looks at the effect of forestry drainage on three peatland sites, the most relevant of which is a treed bog in southern Finland which was first drained in 1915, then again in 1933 and 1954, the final forest cover having been commercially thinned twice prior to the start of the study described here. The specific focus of the work was the rate of soil respiration on this drained forest peatland.

Minkkinen *et al*. (2007a) found that rates of CO_2 loss from the peat in the most ombrotrophic part of the drained bog amounted to 248 g CO_2-C m^{-2} yr^{-1}. This is substantially lower than the values obtained by Silvola *et al*. (1996) for intensively-drained peatland. However, Minkkinen *et al*. (2007a) point out that their study specifically excluded the respiration from living roots within the peat whereas the measurements made by Silvola *et al*. (1996) include such root respiration. It has been estimated that between 10%-40% of CO_2 released from drained peat may come from the respiration of living roots. If the values obtained by Silvola *et al*. (1996) for drained ombrotrophic sites are reduced by this amount, their values for CO_2 losses are similar to those obtained by Minkkinen *et al*. (2007a).

To summarise the Finnish research, therefore:

- lowering the water table by 1 cm increases CO_2 release by 9.5 g CO_2-C m^{-2} yr^{-1} including root respiration, or approximately 7.1 g CO_2-C m^{-2} yr^{-1} if root respiration is excluded;
- increasing the temperature by 1°C increases CO_2 release by approximately 15%;
- intensive drainage of bog sites may increase the total carbon store by stimulating greater biomass growth on lowland bogs, although the gains would be temporary, whereas it is unlikely that the biomass increase on blanket mires would compensate for the oxidative losses of CO2;
- for an intensively-drained bog (in the Boreal region) where the water table sits at approximately 30 cm below the bog surface, losses of CO_2 specifically through the oxidation of peat may be around 250 g CO_2-C m^{-2} yr^{-1}.

9.2.2.4 Relationship between Finnish studies and the British context

The direct relevance of the Finnish work to the UK must be tempered by the fact that the UK lies further south than Finland, and thus temperatures within blanket peat are likely to be higher than those recorded for the Finnish sites. This has immediate implications in terms of the temperature-dependent response shown by CO_2 emissions. Higher temperatures in Britain are generally expected to induce higher rates of CO_2 emission – though this may not be such a clear-cut relationship as is often portrayed (see Section 9.2.2.5 below).

The fact that Britain has an oceanic climate also means that carbon sequestration is possible for almost the entire year, whereas in Finland the snow cover and freezing conditions prevent sequestration for several months of the year. Equally, drained sites in oceanic Britain have the potential to sustain oxidative losses for most of the year. Consequently in the course of a year the undamaged bogs of Britain have the potential to sequester more carbon while the loss of carbon through drainage may be greater than in Finnish sites. This combination of lost sequestering capacity and increased CO_2 losses may thus amount to a significantly greater overall loss than is indicated by these Finnish studies.

The particular sequence of events described by Minkkinen and Laine (1998) is relevant to the lowlands of Britain in that drainage of lowland raised bogs does indeed tend to lead to woodland growth. Consequently there is potential for increased biomass of the kind described for Finnish nutrient-poor mires – rather slow-growing, somewhat poor growth which does not induce large oxidative losses from the peat until a stage is reached where the tree cover becomes sufficiently dense to induce significant oxidative losses.

This contrasts sharply with what is likely in blanket peat, however, because even intensive drainage of such areas does not induce spontaneous colonisation by trees. There are various reasons for this, including the lack of a local seed bank, grazing pressures from hares, sheep and deer, and the propensity for at least the parts of the blanket mire landscape with the deepest peat to create conditions which are simply too hostile for successful tree growth under natural conditions.

Consequently it is unlikely that an area of drained blanket mire would develop the quantity of biomass (in the form of increased cover of dwarf shrubs and/or moor grasses) necessary to offset the losses of CO_2 caused by oxidation of the peat. It is more likely that intensive drainage would lead to increased losses of particulate organic carbon (POC), dissolved organic carbon (DOC) and possibly induce peatland erosion, while even the increased carbon sequestration of a forest plantation may not be sufficient to offset the combined losses resulting from peat oxidation and these additional carbon losses.

It is worth noting that in both the studies involving Minkkinen, there are no actual measurements of water table. Consequently the only study of the three which specifically attempts to relate CO_2 emissions to intensity of drainage is Silvola *et al*. (1996). There have been few studies since then which have attempted to relate the scale of CO_2 emissions to intensity of drainage on peatland sites. Hargreaves *et al*. (2003) provide one of the most detailed studies of CO_2 balance and drainage intensity on British bogs in recent years, but their measure of drainage intensity is based on varying ages of forest cover – no specific measurements of water table were taken from any of the sites.

Billett *et al*. (2004), on the other hand, use a mass-balance approach to assess the hydrological budget and CO_2 emissions from Auchencorth Moss, south of Edinburgh. Here again, there are no measurements of water table from the site. Stream discharge and precipitation data were instead used to model the overall hydrological budget, although not with any intention of relating drainage intensity to measured CO_2 emissions.

9.2.2.5 Drainage, temperature and CO_2 release from the catotelm

One final aspect of CO_2 release in relation to drainage is worth highlighting here. Fontaine, Barot, Barré, Bdioui, Mary and Rumpel (2007), investigated the decomposition characteristics of deeply-buried soil organic matter (SOM) and found that such material was remarkably resistant to decomposition even under supposedly favourable conditions. It is normally assumed, for example, that increasing temperatures will give rise to increasing levels of CO_2 release because the whole peat column is oxidising more readily, but Fontaine *et al.* did not find such a clear relationship when they specifically investigated old SOM. This led them to question the often-stated belief that increasing global temperatures will necessarily lead to greater release of old carbon stores. They propose instead that the majority of old, deeply-buried SOM is actually strongly resistant to decomposition even under conditions of increasing temperature and possibly even increased oxygen availability.

They then demonstrate, however, that if old SOM is mixed with fresh, young SOM, decomposition of the old SOM occurs readily – apparently because the younger SOM provides an energy source for the activation of microbial decomposer populations. Thus if fresh organic matter can be introduced to the peat of the catotelm by, for example, forestry ploughing or by moor-gripping, or if drainage encourages the penetration of catotelm peat by living root systems, then the introduction of this young SOM enables the older catotelm SOM to decompose readily.

If true, the implications of the work by Fontaine *et al*. (2007) for peatland drainage are rather illuminating because it would suggest that one of the most effective actions of a drain is the way in which it encourages plant species such as heather (*Calluna vulgaris*) to root more deeply into the peat. In doing so, this provides more fresh organic matter at depth and thereby assists the catotelm peat to decompose more easily. It may not be the air-penetration as such which is the major stimulus to oxidative wastage, but the presence of fresh root material penetrating to ever greater depths. Field-based evidence for increasing emissions of CO_2 from peatlands with increasing temperature may therefore simply reflect more rapid decomposition of organic matter in the acrotelm, rather than representing loss from the long-term catotelm store.

In a further twist to this story, Charman, Aravena and Warner (1994) identify the fact that gas samples obtained from a boreal forested peatland in north-eastern Ontario yielded ages which were anything up to 2000 years younger than the peat from which the samples were obtained. Charman *et al.* (1994) present strong evidence to suggest that the gas is derived from DOC which has migrated downward through the profile. Whether this younger material has the potential to stimulate decomposition of the surrounding older organic matter is not known. Clearly more work on this topic would be of considerable value.

9.2.3 Drainage and methane release

It is generally assumed that drainage will for the most part eliminate methane emissions from peatlands because the methane is oxidised when air penetrates more deeply into the peat. While CH_4 emissions are certainly reduced in the presence of drainage, it would be wrong to assume that they are altogether eliminated.

9.2.3.1 Acrotelm, catotelm, drainage, and CH_4 emissions

It is worth recalling that there appear to be two main sites for methane production in a natural bog – deep within the catotelm, and also towards the base of the acrotelm (Clymo and Pearce, 1995). The latter site seems to be by far the more active of the two, although this zone also contains vigorous methane oxidisers. It is also important to recognise that methane does not normally oxidise spontaneously on contact with air. Even the much-feared 'fire-damp' of miners, and the 'will-o'-the-wisp' flames seen at night on some fen sites, is more likely due to the presence of phosphine (PH_4) rather than methane alone (S. Chapman pers. comm.) because phosphine can form explosive mixtures with air, and can self-ignite.

Consequently increased air-penetration *per se* does not mean that methane production will be reduced. The increased quantity of air must in some way activate microbial populations which are then capable of oxidising the methane. However, the catotelm is not particularly noted for methane-oxidising microbial populations – that is, populations which can produce CO_2 in the presence of air. The microbial populations more normally associated with the catotelm are those which can use CH_4 as an energy source for glycolysis. These organisms then rely on various anaerobic pathways rather than the oxygen-fuelled Krebbs cycle to complete the essential metabolic loop which finally enables them to undertake further glycolysis. It seems that many of these organisms are obligate anaerobes (*i.e.* they cannot function in the presence of oxygen) so an opportunistic metabolic switch to the aerobic Krebbs cycle seems unlikely should air become available.

It has already been observed in Section 7.3.4, however, that evidence exists to suggest that relatively little CH_4 escapes from the catotelm under natural conditions. The CH_4 which is produced may for the most part be taken up by the obligate anaerobic methanotrophs discussed above (Svensson and Sundh, 1992; Brown and Overend, 1993; Brown, 1995; Baird *et al*., 1997).

It is therefore perhaps important to be clear about the main source of methane-release in bogs under both natural conditions and following drainage. If the catotelm should be breached by drainage and drying in such a way that does not require the methane to pass through a layer of aerobic methane-oxidising microbial populations before release, there are few populations within the catotelm which are able to make use of the increased oxygen to oxidise methane before its release. The CH_4 stored in the catotelm may thus be released to the atmosphere unchanged. Such breaches may take the form of deep cracks which are often associated with drainage. The more intensive the drainage the more extensive and deeper the cracks will be.

9.2.3.2 Methane emissions from differing forms of intensive drainage

Kasimir-Klemedtsson *et al*. (1997) provide some information about the scale of change in methane emissions following intensive drainage of peat by reviewing information available for cereal fields and grasslands on peat soils in Finland, Sweden and the Netherlands. They cite values of around 1.1-1.6 kg CH_4 ha^{-1} yr^{-1} for release of CH_4 by these agricultural habitats, and 100-280 kg CH_4 ha^{-1} yr^{-1} for undrained peatlands in Finland and the Netherlands. It should be recognised, however, that the majority of mire systems in both countries are fens rather than bogs, and fens have much higher rates of CH_4 production than bogs under natural conditions. By way of contrast, Kasimir-Klemedtsson *et al*. (1997) also cite rates of CH_4 release from an 'undrained bog' in Wales, giving values of only 4-22 kg CH_4 ha^{-1} yr^{-1}. Unfortunately they do not give any comparative values for drained bog in Britain.

Cleary, Roulet and Moore (2005) summarise the gas-exchange characteristics of peatland sites in Canada subject to commercial peat extraction. They note that in general, the industry tends to favour bogs as extraction sites so the summarised data for Canada largely describes bog systems. Cleary *et al.* (2005) observe that natural bogs typical of those used by the extraction industry emit around 5.3 g CH_4 m^{-2} yr^{-1}, which is substantially more than the Welsh bog cited by Kasimir-Klemedtsson *et al*. (1997), but rather similar to the value of 6.9 g CH_4 m^{-2} yr^{-1} estimated by Hargreaves and Fowler (1998) for a blanket bog in northern Scotland. It is also a little on the high side compared with the average of values for British bogs given in Table 16, Section 7.4, of the present report, but it should be recalled that bogs in Canada have a long period of snow cover during the winter and anaerobic decomposition with no accompanying oxidation can be significant during this period.

Cleary *et al*. (2005) give figures of 1.87 g CH_4 m^{-2} yr^{-1} for the methane emissions from peatlands (bogs) under commercial extraction in Canada. This is a very much higher figure than the emissions cited by Kasimir-Klemedtsson *et al.* (1997) for Dutch or Finnish agricultural peatlands. It is also perhaps higher than many might expect from such an intensively-drained bog system particularly when compared with the emission rates given for the natural condition, emissions of CH_4 continuing at 35% of natural rates even under such drainage.

Returning to Finnish sites, and noting again that caution must be exercised when considering overall values from Finland because of the predominance of fenland systems, an extensive review of greenhouse gas emissions from Finnish peatlands has been undertaken by Alm *et al*. (2007). From this review, they give CH_4 emission rates for a range of land-use types. The values they give can be seen in Table 18.

Table 18. Summary of CH_4 emission rates for Finnish peatland sites under differing land uses, according to Alm *et al.* (2007).

Land use	Annual CH_4 emission rates ($g\ CH_4\ m^{-2}\ yr^{-1}$)
Commercial peat extraction, milling fields	0.1
Commercial peat extraction, milling fields <u>including</u> emissions from drains	7.23
Commercial peat extraction, stockpiles	19.48
Agriculture : grass	1.27
Agriculture : cereals	-0.43
Agriculture : fallow	0.41
Peatland forestry (dwarf-shrub type)	1.9
Peatland forestry (dwarf-shrub type) <u>including</u> emissions from drains	2.1

It is interesting to note that Alm *et al*. (2007) give even higher CH_4 emission rates (7.23 g CH_4 m^{-2} yr^{-1}) from areas of commercial peat extraction than Cleary *et al*. (2005). It is also worth noting that their values include emissions of CH_4 from the ditches, as is the fact that the major part of CH_4 emissions on commercial extraction fields is from the drains rather than the milling strips themselves. It can be seen that drains also add significantly to the CH_4 emissions from peatland forestry. Whether these emissions are released from the catotelm, or whether they are a result of anaerobic processes in surface layers when drains become choked with vegetation re-growth, is not clear from these data.

Minkkinen, Penttilä and Laine (2007b), recording the CH_4 emissions from forestry-drained sites cited in Alm *et al*. (2007), ranging from -0.75 – 3 g CH_4-C m^{-2} yr^{-1}, noted that the highest CH_4 emissions were found on sites where tree growth was poorest, or where the sites had only recently been drained. It seems that a threshold of around 135 m^3 ha^{-1} must be reached in the tree-stand volume before CH_4 levels fall substantially because the trees themselves are significantly adding to the drainage effect.

Alm *et al.* (2007) record the lowest values of CH_4 emissions (in fact there is CH_4 take-up) from cereal fields. To enable cereals to grow it is necessary to have a good-sized and dependable aerobic layer for the root zone, and thus it is necessary to ensure that the water table is held constantly at a low level in the peat. Under such conditions it is possible for aerobic oxidising microbial populations to develop abundantly within the root zone and thus prevent any CH_4 emissions. Other agricultural conditions such as permanent grass or fallow land do not require such a constantly-lowered water table and thus we find that there is often still some CH_4 release.

In their review of factors to quantify when considering the carbon balance of windfarms constructed on peat, Nayak, Miller, Nolan, Smith and Smith (2008) provide a brief review of CH_4 emission values from peatlands in both the natural and drained state. Their values are all expressed in terms of emission rates per day and thus it is not possible to compare values directly with the annual rates provided in the present report.

As discussed earlier in Section 7.3.8, Laine *et al*. (2007) provide a particularly useful set of CH_4 emission values for an Irish blanket mire over a period of a whole year, giving both annual and daily emission rates for methane. Their data show that daily rates vary between 0.1 and 64.1 mg CH_4 m^{-2} day^{-1}, with a daily mean of 11.8 mg CH_4 m^{-2} day^{-1}. The annual emission rate is given as 3.3 g CH_4 m^{-2} yr^{-1}. It is important to note that a simple multiplication of the mean daily rate by 365 days produces a value of 4.3 g CH_4 m^{-2} yr^{-1}, which is some 25% larger than the actually-measured annual emission rate.

Although various authors have attempted to provide a model of the relationship between the position of the bog water table and CH_4 emissions, almost all studies associated with oceanic bog systems have been undertaken on relatively natural sites, rather than exploring the effect of drainage on methane release. Thus Hargreaves and Fowler (1998) present a correlation between water table and CH_4 emissions at the natural pool system of Loch More, while Laine *et al*. (2007) show that this relationship appears to be exponential at their blanket bog in Ireland. Both studies, however, are concerned with sites which have not been subject to drainage. Whether such a correlation would still hold under drained conditions is not clear.

9.2.4 Drainage and POC losses

Drainage is capable of increasing particulate organic carbon (POC) losses in various important ways. Firstly, Holden, Gascoign and Bosanko (2007b) have shown that drains with gradients of more than 4° in blanket peat tend to be self-scouring. Consequently such drains may erode peat material from the drain base and sides until such time as the drain-base reaches the mineral sub-soil. Scouring may then begin on this mineral soil.

Drains may also initiate erosion of blanket mire. Various examples exist where drains can be seen to have resulted in upslope erosion into an area of blanket mire. The resulting erosion clearly brings with it extensive losses of POC.

Drainage tends to stimulate growth of more combustible plant species such as heather (*Calluna vulgaris*) along the drain margins, as well as reducing the quantity of surface and acrotelm water. Consequently bog surfaces can become a much higher fire risk and more prone to accidental (or natural) fires. Such fires typically produce much ash, which is washed away by rain, and much bare peat, which is then subject to rain-scour and potential erosion.

Drainage of bog systems is usually undertaken to remove surface water from wetter areas in order to improve the ground for stock grazing. Such formerly-wet areas are generally soft and dominated by vegetation which is not capable of withstanding any significant grazing pressure. As more animals move onto such areas following drainage, there is a danger that the surviving wet vegetation will be destroyed by trampling, resulting in steadily-expanding patches of erosion.

9.2.5 Drainage and DOC losses

While a range of evidence points to some form of link between the drainage of bog systems and an associated rise in DOC release, this link is not as firmly established as is generally thought. Holden, Chapman and Labadz (2004), for example, highlight a range of apparently conflicting evidence in which some studies actually point to a fall in DOC following drainage. Of particular note is the fact that Moore (1987) found no difference in DOC losses from sites described as 'undisturbed' and sites subject to commercial peat extraction.

If DOC is tightly linked to the abundance of vascular plants in the vegetation, this is what might be expected. Undisturbed sites are rich in *Sphagnum* with a relatively limited range of vascular plants to provide material for DOC production. Commercial peat extraction sites have very little vegetation although what there is around the margins tends to consist of vascular plants. Both sites might thus be expected to produce relatively low levels of DOC.

Balanced against this, however, is the evidence obtained by Glatzel, Kalbitza, Dalva and Moore (2003), who recorded high levels of DOC release from a commercial peat-extraction site where vacuum harvesting (peat milling) was used. Data for DOC were also obtained for a range of other peat surfaces, including somewhat drained shrub-dominated bog, re-wetted block-cut peat, abandoned block-cut peat, and three experimental restoration plots.

Of these various sites, the plot with the lowest level of DOC production was the milled restoration site where restoration commenced in 1997 and the vegetation is now dominated by large cotton grass tussocks (*Eriophorum vaginatum spp. spissum*). The greatest variability in DOC release was observed in the re-wetted block-cut plot. The plots with the highest degree of humification (decomposition) were the vacuum harvested and abandoned plots. This may then explain the high DOC values obtained from the vacuum harvested site because the milling process artificially aerates and mills the surface layer of the peat matrix, thereby making decomposition of the matrix much easier.

Worrall, Gibson and Burt (2007b) cite studies which have shown fairly clear correlations between the extent of drainage and the concentration of DOC in waters flowing from drained bog systems (Clausen, 1980; Mitchell and McDonald, 1995). Meanwhile Wallage, Holden and McDonald (2006) obtained direct measurements of DOC from three areas of blanket bog, one of which had been drained, and they noted significantly higher DOC levels from the drained site compared to the other two study areas.

When the effect of drainage on DOC release is examined in any detail, almost invariably the main focus of attention is the draw-down of the bog water-table and consequent increased aeration of the peat soil (*e.g.* Holden *et al*., 2004; Holden, 2005a,b; Wallage *et al*., 2006; Worrall *et al*., 2006a; Worrall *et al*., 2007a,b). Mechanisms for DOC production and release are considered in terms of increased oxidation leading to breakdown of the peat matrix – in other words, the effect of drainage on decomposition.

Little, if any, attention has been devoted instead to the effect of drainage on growth – specifically the growth of the vegetation in response to such drainage. Vegetation is able to respond rapidly to environmental changes. Aerial biomass can increase substantially and rapidly if conditions permit, as can root systems. In times of stress, material can be re-allocated to different parts of the plant, or material can simply be shed – potentially then to form DOC. Changes in the competitive ability of particular species, brought on by altered environmental conditions, can produce alterations in species composition, but can also give rise to significant changes in biomass for an area.

It is worth noting that any increase in vascular-plant biomass is likely to occur at the expense of the biomass of *Sphagnum*. Consequently the overall biomass of the vegetation cover may remain the same or even decline, but the important thing in terms of DOC production is that the proportion of vascular-plant tissue within the biomass has increased. Meanwhile, more active and extensive root systems can greatly enhance decomposition processes within the acrotelm. If the water table is drawn down into the catotelm along a drain margin, vigorous root growth of species such as heather (*Calluna vulgaris*) can stimulate the decomposition of recalcitrant catotelm peat in the manner suggested by Fontaine *et al.* (2007).

This biological response at the vegetation scale (rather than at the microbial scale) is likely to be significant if the main source of DOC is indeed from a combination of the present vegetation and, in effect, the litter layer of the acrotelm or haplotelm, yet this response remains largely un-reported. Given the scale of DOC-increase described by, for example, Wallage *et al.* (2006) it would require only a relatively subtle shift in the biomass and composition of the vegetation, rather than a significant increase in oxidation of catotelm peat, to generate such an increase.

Though their study does make more reference to vegetation character than many studies of DOC and peatland hydrology, the level of information provided by Wallage *et al*. (2006) about the vegetation cover of their three study sites is, unfortunately, not sufficient to be able to determine whether any significant vegetation differences exists between the sites.

Lowered water levels may be the driver for all of these changes, but as Fontaine *et al*. (2007) observe, increased aeration *per-se* may not result in greater decomposition of deeper catotelm peat or even the more recalcitrant material towards the base of the acrotelm. This point is even more significant when dealing with a haplotelmic blanket mire because, potentially, all the peat on such a site is recalcitrant because there is no acrotelm. It is only with the addition of fresh roots that this resistant catotelm material will readily decompose, adding to the DOC released from the litter of whatever vegetation cover there might be.

There is thus a potentially strong argument for looking more closely at the relationship between DOC, drainage, water table and the vegetation. Some of the apparent inconsistencies in the relationship between drainage and DOC may prove to be explainable in terms of particular responses by the vegetation.

9.2.6 Drainage and local climate effects

One final effect of drainage has been little reported, but this may be because the effect has not been looked for. It is the potential of peatland drainage to alter the local climate. Venäläinen, Rontu and Solantie (1999) investigated this effect in a heavily-drained part of Finland. Based on earlier evidence that night-time temperatures had been reduced in areas of intense drainage, they simulated this effect and found that there was indeed a reasonable basis for this suggested reduction in local night-time temperatures.

It might be illuminating to carry out a similar study on areas which were formerly rich in peatlands but which are now intensely drained, such as the Somerset Levels, the Cambridgeshire Fens, the Lancashire coastal plain and the Forth Valley.

Given the high water content of blanket peat soils, it is perhaps also worth noting the suggestion by Hossain (in press) and Hossain, Jeyachandran and Pielke (2009) that the presence of large water bodies resulting from dam construction may give rise to changes in regional precipitation patterns. They observe that the idea of manipulating local or regional rainfall through dam construction dates back as far as the 1930s, but they then offer evidence to suggest that the existence of a dam and its associated water body may influence both the average rainfall and the pattern of extreme precipitation events.

It would be interesting to know whether a reverse pattern is associated with extensive blanket mire areas which have been intensively drained – for example in the afforested regions of the Flow Country.

9.3 Overall carbon-balance model for drainage

The overall picture of carbon exchange in relation to drainage might thus be summarised as (see Figure 33):

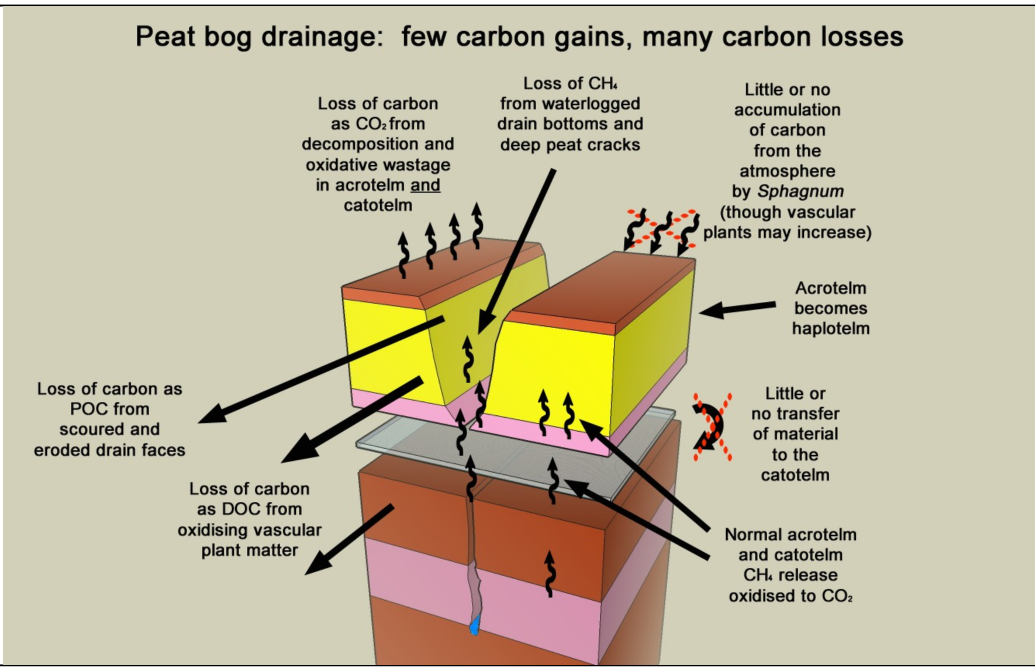


Figure 33. Carbon-balance model of drainage for a standard cubic metre of peat
While drainage may give rise to increased growth of vascular plants at the bog surface, the loss of *Sphagnum* cover represents a much greater loss of biomass carbon than any gains which might be made from the vascular plants. The increased quantity of vascular-plant material in the surface layers provides much greater opportunities for DOC production and release, while the bare peat of the drain sides can more readily lose POC through physical erosion of the exposed peat. Failure to provide a supply of relatively recalcitrant peat material from vegetation growth can mean that there is little or no transfer of material from acrotelm to catotelm into long-term storage. Deep cracks created by peat shrinkage mean that methane can escape directly to the atmosphere while allowing heavy rain to penetrate deep into the catotelm, with possible consequences for slope stability.
R A Lindsay 2004

- an acrotelm which is more usually empty of water than full;
- between 10-50 m of a drain (and possibly much further), a vegetation cover develops in which the natural bog vegetation and patterns move towards an assemblage typical of drier climatic conditions, while the ground surface may gradually sink through the process of consolidation;
- within 2-5 m of a drain, the vegetation ceases to be peat-forming, loses many of the characteristics of an acrotelm, and generally becomes dominated by vascular plants, while the ground surface sinks along with the falling water table as a result of consolidation and oxidative wastage and thus the water table shows only limited draw-down *relative to the ground surface*;
- the acrotelm may show an increased biomass of vascular plants but a reduction in *Sphagnum* biomass, and given that *Sphagnum* biomass occurs at such high carbon density, this loss of *Sphagnum* to vascular plants – even vascular plants such as birch or young conifers – generally means that the carbon held in living biomass declines dramatically; the increased biomass of vascular plants is balanced by the fact that this vascular-plant material is more easily decomposed and thus there may be little or no net accumulation of carbon from the atmosphere;
- indeed if the acrotelm is lost and the bog becomes a vegetated haplotelmic system, there may be a long-term net loss of carbon from the system because older catotelm carbon is lost from the catotelm when decomposer microbial populations are re-energised by a supply of fresh organic matter in the form of new root systems;
- increased quantities of decomposing vascular plant tissues release greater quantities of CO_2 and DOC;

- if the supply of recalcitrant *Sphagnum*-rich material to the base of the acrotelm quite literally dries up, there is then no material to pass from the acrotelm to the catotelm and thus additions to the long-term carbon store cease;
- water-filled cracks in the peat enable methane from the base of the acrotelm and deep within the catotelm to escape directly to the atmosphere;
- drains with gradients of more than 4° will tend to remain open and release significant quantities of particulate organic carbon (POC) as the drain-sides undergo scouring, whereas drains with lower gradients will tend to scour until they choke up naturally, although repeated cleaning of such drains will maintain high levels of POC;
- drainage creates conditions in which the stability of the peat mass may be compromised, pre-disposing the peat to slope failure and mass movements.

9.4 Research needs

Several areas which would benefit from further research can be identified:

9.4.1 Acrotelm response to drainage

The bulk of peatland drainage work has concentrated on evident draw-down within the catotelm and has largely ignored effects in the acrotelm. Detailed studies of water-table are urgently required, particularly focusing on maximum and minimum levels, duration of draw-down, and changes to the acrotelm structure in terms of microtopes and nanotopes.

This area of investigation work should also include a closer examination of the 'topographic index' modelling developed at the University of Leeds, to investigate precisely what the field effects are in relation to the scale of hydrological change predicted by the model. This should include detailed water-level studies as described above, but also detailed examination of microtope, nanotope and vegetation effects.

9.4.2 Surface-profile changes in response to drainage

Any future work on peatland drainage must ensure that an accurate measure is made of surface changes resulting from drainage. This would include not only measurements of surface levels (noting the nanotope elements from which these measurements are taken) but also measuring any changes in bulk density. Measurements of surface level and bulk density should be carried out over sufficient distances or depths to detect the limits of change.

9.4.3 Oxidative wastage rates in the catotelm

Studies of peatland drainage must ensure that appropriate measurements are obtained prior to the drainage taking place. This includes an accurate description and measurement of surface microtopography and vegetation, as well as bulk density and CO_2 flux under a range of weather conditions and seasons.

9.4.4 Evidence for drainage impacts in the peat archive

The recent peat archive of areas where drainage has already occurred could be examined for evidence of recent change in vegetation communities. At its simplest, such work could simply note any changes from *Sphagnum*-rich material to macrofossil remains containing few if any *Sphagnum* fragments. It would be important to ensure that such archive sampling extends over an area sufficient to detect the limits of apparent change.

9.4.5 Evidence of change from remote sensing

At its simplest, this work could consist of detailed examination of aerial photographs from dates prior to drainage (or at least of a known date even if it is after drainage) and then comparing the microtope pattern with that visible in more recent aerial photography. This could then be ground-truthed by field survey of present vegetation and microtopes, together with analysis of the recent peat archive, as described above. Some progress is being made in terms of the remote-sensing of water content, and detailed microtope/vegetation analysis using multi-spectral scanners or LIDAR, and this could also be pursued, though again it would be essential to have accompanying and sufficiently accurate ground-truthing of microtopes and vegetation.

9.4.6 CO_2 and CH_4 flux rates correlated with changes in microtopography and vegetation

Although a considerable volume of work has been published concerning the general release of CO_2 and CH_4 from peatland systems, a much smaller number of studies have looked at the relationship between gaseous carbon exchange and microtopes and nanotopes. Virtually nothing has been published describing the relationship between microtope, nanotope, vegetation change and gaseous carbon exchange in response to drainage. Such work requires detailed mapping of the microtopes, nanotopes and vegetation assemblages prior to drainage, together with the gaseous carbon exchange in relation to these, and then careful monitoring of all these factors following drainage.

9.4.7 Age-estimates for CO_2, CH_4 and DOC to determine the origin of emissions/release

Some information already exists regarding the radiometric age of carbon in CO_2, CH_4 and DOC, and this information suggests that the bulk of DOC at least is derived from surface vegetation and upper layers of the acrotelm. Detailed studies of all three sources of carbon loss should aim to identify the proportions of differing ages of carbon in CO_2, CH_4 and DOC in order to determine the proportions of these carbon losses from differing depths within the peat. This work is still required on natural systems, but should then be applied to drained systems as well in order to identify the influence of cracks, consolidation, oxidative wastage, and changes in microtope, nanotope and vegetation.

9.4.8 Relationship between *Sphagnum*, vascular plants and DOC release

Given the economic importance to water-utility companies, it would seem a high priority to identify the relationship between *Sphagnum* cover, the cover of vascular plants, and the rate of DOC release. It would be valuable to identify whether DOC release is most directly linked to aeration of the peat through drainage and other forms of disturbance, or whether it is instead tightly coupled to the relative abundance of vascular-plant material. If it is found to be the latter, then it would be useful to establish the extent to which DOC results from decomposition of the vascular-plant material itself, and the extent to which such material also stimulates loss of older catotelm peat by re-energising microbial decomposer populations within the catotelm.

10 DISCUSSION TOPIC 1b Restoration of drained peat bog systems – the carbon balance

After almost 300 years of scientific investigation into the best ways to drain peat bog ecosystems, very considerable amounts of time, effort and money have in recent years been devoted to the best ways of undoing this drainage by blocking the drains and attempting to restore the peat bog habitat.

Major initiatives such as Moors for the Future, Peatscapes and a number of other projects and programmes established, often with substantial EU support, by the UK Government, the RSPB and other bodies, have undertaken a very considerable amount of work in attempting to restore damaged blanket bog through, amongst other things, the blocking of drains. Much of this work has been carried out at least in part because of concern about the damage which drainage causes to long-term carbon stocks held within the extensive blanket mire regions.

Such activities are not restricted to the UK. Germany has a long history of mire restoration (Eigner and Schmatzler, 1980), while recent work in Austria and Switzerland ranging from ditch blocking, dam construction, forest removal and detailed monitoring, has seen a dramatic transformation of many formerly-degrading peatbog systems. Steiner (2005) gives detailed descriptions and illustrations of this work, some of it building on the fundamentals of Ingram (1982 and Bragg (1995) and the practical guidance of Stoneman and Brooks (1997), but some work takes restoration and monitoring to a new level of scientific complexity.

Concern has been raised more recently, however, about the true carbon balance of such activities. Evidence has been put forward to suggest that restoring drained blanket bog may actually be greenhouse-harmful because large amounts of methane have been described as emerging from blocked drainage systems. Water utility companies have also expressed concern that blocking of drains may lead to an increase in, or change in the composition of, DOC from such restored areas. Any review of the evidence for such effects, however, should first consider the nature of the restoration process because different approaches to restoration have different potential consequences.

10.1 Restoration processes for drained blanket bog

As Gilman (1994) observes, the higher conveying capacity of straight, smooth-sided drains compared with the undulations and varying resistance to water movement typically encountered in the surface layers of a peatland means that drains are able to remove surface water more rapidly, limit the degree of water-table rise, and prevent anoxic conditions from developing in the surface layers.

The first priority of restoration must therefore be to reduce the conveying capacity of such drains back to a rate which reflects the original rate of water movement through and over the surface layers of the bog as closely as possible. A variety of ways have been used to achieve this.

Holden *et al.* (2004) additionally highlight the fact that such a slowing of water movement is not in fact the objective of restoration *per se*, but rather it represents a means by which a more typical bog vegetation may be re-established and thus ultimately the ecological functioning of the system may be restored. Holden *et al.* (2004) observe that in some cases the active restoration of the water regime alone may not be enough to create a naturally-functioning system, and active conservation management may be required in order to encourage the re-development of an appropriate vegetation cover to reach this final state.

Holden (2009) has produced an extremely informative and succinct review of ditch blocking in blanket mire landscapes, focusing particularly on research into such activities in the Pennines. He emphasises that the theory and practice of ditch blocking should not focus only on the behaviour of water. The vegetation also has a vital part to play in the resulting dynamics of ditch blocking, and may hold the key to whether any given ditch-blocking programme is perceived as being successful or not. For example, an increase in *Sphagnum* or purple moor grass (*Molinia caerulea*) following ditch-blocking appears to result in reduced levels of DOC.

Similarly, Baird, Holden and Chapman (2009), in a review of methane emissions from peatlands, highlight the fact that restoration programmes which involve ditch-blocking on bog systems tend to create ribbons of open water behind the dams. They point out that these bodies of water can be significant emitters of methane as long as they remain open water. Baird *et al.* (2009) therefore recommend that restoration programmes should be designed specifically to avoid the creation of such ribbons of water.

However, this recommendation does not take into account the tendency for such narrow ribbons to become colonised by *Sphagnum* within a period of 5-10 years. This is perhaps because the potential for supply from local sources of *Sphagnum* is so sparse in the Pennines (an evident regional focus within the review. Other less *Sphagnum*-poor areas, however, display rapid colonisation of such small open-water patches by *Sphagnum*.

Meanwhile authors such as Frenzel and Karofeld (2000) and Bortoluzzi *et al*. (2006) record low levels of methane production from hollows dominated only by *Sphagnum* – Frenzel and Karofeld (2000) having actually cut off the aerial parts of vascular plants to remove the previously-measured strong 'carbon shunt' effect provided by these aerial parts. It is also worth noting that both MacDonald *et al*. (1998) and Laine *et al*. (2007) recorded low levels of methane release from *un-vegetated* hollows (features which are in many ways similar to flooded drains). Methane release from such features was no higher than those recorded from hummocks.

In other words, while it may be true that flooded ditches *may* (though not necessarily) produce relatively high levels of methane, the presence of bog bean (*Menyanthes trifoliata*) or common cotton grass (*Eriophorum angustifolium*) in the flooded drain may result in significant methane release. The faster this vascular-type of vegetation can be replaced with a *Sphagnum* carpet, the sooner these emissions are likely to fall. Ideally, it would appear that a restoration programme would involve not merely the blocking of ditches to prevent water movement, but also the seeding of *Sphagnum* to hasten the establishment of a *Sphagnum* carpet and prevent colonisation by vascular plants.

10.1.1 Drain restoration techniques : flooding (terrestrialisation) and re-wetting (paludification)

10.1.1.1 Dams

The most obvious way to reverse the effect of an individual drain is to block the drain channel with a dam. On sloping blanket mire, however, the ponding behind such a dam is likely to be limited in extent upslope along the drain. Consequently if a long sloping drain is to be re-flooded along its whole length, a great many dams may be needed. While this number of dams may not be a problem in terms of bog processes, it is likely to be a major practical issue in terms of cost, materials and labour. Holden (2005a) thus proposes that use of hydro-topographic models such as his 'topographic index' can help to pin-point the most important drains, and sections of drain, which can be dammed to best effect using the resources available.

Dams may consist of impermeable corrugated plastic sheeting, boards of marine ply, tightly-packed wooden posts, or plugs of peat. What they all do with varying degrees of success is hold a proportion of surface run-off back behind the dam to create an area of low-energy ponded water. This ponding provides two things:

- maintenance of a low-energy area of ponded water in the section of ditch immediately upslope from the dam, this ponded water having the potential to encourage development of aquatic *Sphagnum* carpets (terrestrialisation) along the line of the drain itself;
- maintenance of higher water levels in the section of ditch immediately upslope from the dam, thereby elevating water levels in adjacent areas of peat and inducing re-wetting (paludification) of these adjacent areas – with the consequent potential for development of largely terrestrial and hummock-forming species of *Sphagnum* bog moss.

The terrestrialisation process is the more immediately evident of the two processes, with typically bright green lines of former ditches now choked with *Sphagnum recurvum* or *S. cuspidatum*. The benefit of having *Sphagnum* carpets developing in the ditch-line is that the *Sphagnum* then provides some hydrological control on the loss of water from the ditch (*Sphagnum* typically bleaches when dry and thus reflects a great deal of solar energy, reducing evapotranspiration from the moss surface and potential cracking of the underlying peat). Presence of a *Sphagnum* carpet also helps to diffuse water flow within the ditch, reducing scouring effects during periods of heavy rain.

In terms of peat formation, however, aquatic *Sphagnum* species are poor peat-formers and thus it can take a considerable period of time before the line of the ditch is lost completely. Even then, the *S. recurvum*/*cuspidatum* peat filling the ditch is likely to be of a markedly different nature and thus with different hydrological properties from the peat immediately surrounding it.

The paludification process, on the other hand, makes itself evident in more subtle ways. It is expressed as a shift from a drier vegetation assemblage along the margins of the ditch to an assemblage which increasingly resembles an actively-growing bog vegetation. Paludification has two characteristics which terrestrialisation cannot match.

Firstly, as ground paludifies and becomes more *Sphagnum* dominated, this new *Sphagnum* cover can itself then paludify adjacent areas and so increasingly expand the area of paludification, whereas terrestrialisation is a process entirely restricted to the ditch-line.

Secondly, paludification tends to encourage terrestrial *Sphagnum* species, particularly *S. capillifolium*, *S. papillosum* and *S. magellanicum*, which are typical of T3 hummocks, T2 high ridges and T1 low ridges (*sensu* Lindsay, Riggall and Burd, 1985). These species are more typical of the general bog surface than aquatic *Sphagnum* species, and they are also associated with lower emissions of CH_4 than those typical for *S. cuspidatum*/*recurvum* carpets.

It is important to understand that, because the ditch sides are likely to have slumped by at least a few centimetres within a short time of the drain being dug, the top of the drain channel is unlikely to be at the same level as the surface of the bog. Indeed the effective width of the drain will now be the original cut channel plus the width of any compression and oxidation either side of the drain (see Figure 34).

Figure 34. Long-term profile changes in blanket mire drainage

(Left) A freshly-dug drain across an area of deep peat. The turf has been cast aside to the left of the drain. Slumping along the drain-line because the peat is so fluid can already be seen in the ragged shape of the drain-line. (Right) A long-established drain on deep blanket peat. The sides have slumped and in places have collapsed, but the drain continues to function. The effective physical width of this drain is clearly now larger than originally dug. Careful topographic measurements would be required to determine the extent of slumping and thus the effective dimensions of this drain.

Photo (left) J B Ratcliffe; Photo (right) R A Lindsay

Consequently if a dam is installed to be level with the top of the cut channel only, any ponding will have much less potential to paludify the surrounding bog surface than a dam which is installed with sufficient width to include the zone of slumping, and with sufficient height to match the levels of the surrounding bog surface. Careful topographic measurements may be necessary to ensure this is achieved, because if the entire acrotelm is only 10 cm thick, a slumping of only 5 cm can mean that only half the acrotelm at most can be re-wetted.

10.1.1.2 'Filters' – e.g. heather bales, conifer waste

Heather bales are not designed to act as impermeable barriers which will pond water behind them. They instead function as filters which trap peat sediment and thereby encourage the drain to choke up and infill with this loose material.

As such, the restoration process does not explicitly encourage terrestrialisation by aquatic *Sphagnum* species, although these may be the first to colonise if the accumulated peat sediment is particularly liquid. It is equally possible, however, for terrestrial *Sphagnum* species to colonise such areas along the ditch bottom or along the ditch sides if the central parts of the peat material are too liquid.

Under such conditions, paludification of the adjacent areas of peat would only occur once the drain had infilled to a significant extent, but this is likely to occur more rapidly if colonised by terrestrial rather than aquatic *Sphagnum* species.

Theoretically, because this approach does not involve ponding and thus has limited opportunities for terrestrialisation while paludification is largely restricted to the ditch-line, it would seem that use of heather bales or similar semi-permeable filters offers only limited value.

In practice, however, the bales are likely to become choked fairly quickly and will then act more as a low-efficiency dam in rather similar fashion to peat dams – which are fine when conditions remain wet, but have a tendency to develop cracks and leak somewhat during extended periods of dry weather.

The slower process of ponding and re-wetting associated with the use of 'filters' such as heather bales means that slumping of the ditch sides, and thus lateral expansion of the ditch, is likely to be more significant when using this method.

10.1.1.3 Infilling of drains

Sometimes the option exists to roll back into the drain the ridge of peat turf dug to make the drain in the first place. Given that peat shrinkage can be considerable in the first few months after drainage, the peat turf itself is likely to have shrunk by a substantial degree unless it can be turned back within a month or two of having been dug.

More usually, this turf has been lying on the bog surface for some years and has thus had much time to shrink and oxidise. That said, the drain itself may well have shrunk and slumped if it has not been maintained during the intervening years. Braekke (1983) found that some drains cut in soft peat had shrunk from between 30 cm – 1.1 m deep to only 18 cm - 29 cm deep in 10 years.

Nevertheless, it can usually be assumed that a turf turned back into a drain will no longer fill that drain. If this is the chosen restoration strategy, it is likely that the drain will remain as a shallower furrow. The somewhat undulating nature of the turned-back turf will reduce the ability of this furrow to convey water away quickly, and may well tend to encourage paludification of both this surface and the furrow sides. It is unlikely that terrestrialisation would feature prominently in such a scenario unless dams were added as well.

10.1.1.4 Restoration by 'benign neglect'

It is obvious from Figure 34 that in soft peat a drain can lose its clean and straight profile within a matter of months through differential subsidence and soil movement, while if maintenance is not carried out regularly in succeeding years, the original shape can be all-but-lost amidst various encroachments and collapses of both peat and vegetation.

This tendency of ditch profiles to change over time can lead quite inadvertently to blockage, infilling and restoration by simple neglect. Holden *et al*. (2004) refer to this accidental management as 'benign neglect', and observe that in some cases this may be the most cost-effective way of achieving significant improvements in blanket mire habitat condition.

Holden, Gascoign and Bosanko (2007b) examine the underlying mechanisms behind this natural tendency to infill, and they reveal that drains with gradients of less than 4° tended to infill whereas drains with gradients of more than 4° were generally capable of maintaining an open channel. Once drain scouring had exposed the mineral sub-surface, drains tended not to develop a vegetation cover on the drain floor. Drains which were almost entirely shaded showed vigorous development of vegetation along the drain floor but between 90% and 60% cover quite the opposite occurred – little re-vegetation was observed. If shading was below 60%, re-vegetation again became common.

Thus the work by Holden *et al.* (2007b) identifies that 'benign neglect' may act in a positive way, adding to any active conservation management programme involving ditch-blocking. Their findings suggest a set of criteria which might be used to prioritise conservation action. On drains which have slopes of less than 4°, benign neglect may already be resulting in much of what is intended from a ditch-blocking programme, whereas drains with steeper gradients, and particularly those which are have already eroded to sub-soil or which are close to doing so, may require rapid active management in order to put them on the road to recovery.

10.2 Carbon responses to drain-blocking in blanket mires

Raising water tables by installing dams, weirs or sluices is by far the most common method of restoring bog systems generally and blanket mires in particular. Consequently this present review of responses to restoration management of drained bog will focus almost exclusively on such an approach, with its accompanying processes of terrestrialisation and paludification.

It is important to recognise that both terrestrialisation and paludification have a part to play in the restoration process. That terrestrialisation occurs is evident from the bright green lines of *Sphagnum recurvum*/*cuspidatum* picking out former drain lines, whilst the effect of paludification on at least the water table behaviour is clearly demonstrated by continuous water-level records for a sloping bog in Austria following dam installation. An extensive drainage network across the site was dammed in the summer of 2000 and the surrounding peat almost immediately began to show marked paludification response (see Figure 35).

Similarly, data gathered by Frank Mawby at Wedholme Flow, Cumbria, and presented by Holden *et al*. (2004), shows that damming of an intensive drainage network associated with commercial peat cutting clearly raised the water table in the adjacent peat. This paludification produced a much more stable water table having clear affinities with the behaviour of a natural bog water-table.

The water-table response to ditch-blocking therefore seems fairly clear. What, then, of the carbon balance resulting from this change in water-table behaviour?

10.2.1 Gaseous carbon responses to drain blocking – CO_2 and CH_4

Intuitively, given that oxidation of peat results in increased release of CO_2 while the highest CH_4 emission rates are associated with high water levels (Martikainen *et al*., 1995; Silvola *et al*. 1996; Laine and Vasander, 1996; Laine *et al*., 2007), it would seem likely that ditch-blocking will tend to reduce CO_2 emissions but increase emissions of CH_4.

10.2.1.1 Drain-blocking and CO_2 flux

While there are several published studies which have examined the CO_2 balance resulting from the re-wetting of commercial peat cuttings, there are fewer reports of the balance associated with the damming of drained (rather than cut-over) bogs.

It is important to draw a very clear distinction between blocking drains on a bog system, particularly a blanket mire, and the re-wetting of a commercially-worked peat extraction site because the two are rarely comparable. A drained blanket mire will generally retain an original mire surface which is broken only at intervals by the lines of the ditches. A commercially-worked extraction site has a surface with an artificially-created topography and either a complex vegetation pattern reflecting past extraction methods, or a completely bare worked surface. The re-wetting of such surfaces raises a great many more issues than are generally relevant to the blocking of drains on a blanket mire system.

Interest in demonstrating the restoration possibilities arising from commercial peat extraction has led to relatively extensive research into the re-wetting of such ground. In contrast, comparatively little research attention has been devoted to the carbon balance of drain-blocking within drained bog landscapes. In one of the most detailed and comprehensive of such studies still undertaken to date, however, Komulainen *et al*. (1999) give a clear picture of the CO_2 response to drain-blocking on a bog system and the ecological processes associated with that response.

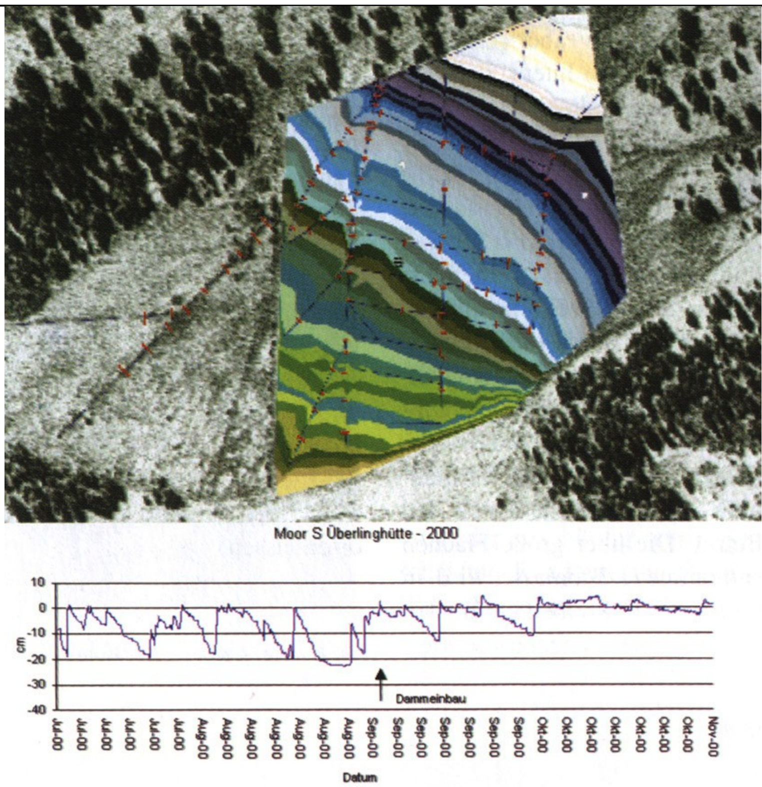


Figure 35. Paludification response to drain blocking on an Austrian sloping bog.
Plan view (upper diagram) of a mire which lies south of Überlinghütte, Austria, which was formerly extensively drained. Surface contours of the site are indicated in coloured banding, indicating the sloping nature of the bog. Dotted lines indicate drains, and red dots indicate dams installed in the summer of 2000. The graph (lower diagram) shows the response of the water table, with the point when the dams were installed indicated by an arrow. It can be seen that the water table rose and became comparatively stable following dam installation.
Illustration courtesy of G M Steiner

Komulainen *et al.* (1999) investigate a bog and a fen in southern Finland, the two sites having been drained 30 and 45 years ago respectively. For the purposes of the present review, only the results from the bog site will be considered, though it should also be borne in mind that even southern Finland is markedly more continental in climate than the blanket mire regions of Britain. The bog site had an area which was re-wetted by damming a drainage network consisting of drains at 30 m spacing, but part of the area was left in the drained state as a control. It is thus worth noting that Ingram's (1982) ground-water mound theory would suggest that the two experimental areas are not wholly independent of each other hydrologically (unless they are on two separate peat lenses, but no mention is made of this possibility).

Komulainen *et al.* (1999) explicitly sample areas of low hummock, lawn and hollow (T3/2, T1 and A1 – *sensu* Lindsay, Riggall and Burd, 1985) and provide useful descriptions of the vegetation found in these three nanotopes both prior to installing the dams and then again two years after dam installation. They similarly provide measurements of soil respiration, soil-and-vegetation respiration, gross CO_2 uptake and net CO_2 exchange for the two pre- and post-damming time periods.

What emerges from the study by Komulainen *et al*. (1999) is that damming the drains raises the water table by some 20 cm. Soil respiration was significantly reduced after 2 years of re-wetting, while the rate of total respiration (including respiration by the vegetation) was significantly reduced in A1 hollows and T1 lawns after 2 years, but the respiration rate in the T2/3 hummock remained unchanged. This was because vigorous vascular plants such as heather (*Calluna vulgaris*) had become established in the lower levels of the microtopography but the high water table now meant that such species could no longer survive at these levels in the microtopography. Their reduced biomass produced less CO_2 from respiration.

Rates of gross CO_2 uptake also declined significantly in the A1 hollows but increased significantly in the T2/3 hummock for the same reasons – 'hummock' species at A1 hollow level were no longer able to photosynthesise effectively when water tables were raised, while increased growth of typical 'hummock' vascular-pant species on the T2/3 hummock led to greater uptake of CO_2 in that part of the microtopography. In the T1 lawns, rates of gross CO_2 uptake were unaffected by re-wetting, and it was presumed by Komulainen *et al.* (1999) that this was because losses of lichen biomass were matched by an increase in *Sphagnum* biomass. Thus the species composition in the T1 lawns was altered by re-wetting but rates of CO_2 uptake were not.

Despite substantially higher rates of productivity, over the growing season the T2/3 hummock nevertheless showed an overall net accumulation of CO_2-C which was 50% lower than that observed in the A1 hollows. This was because most of the vigorous vascular-plant growth in the hummock decomposed readily. In contrast, comparatively little of the CO_2 taken up by the A1 hollows was subsequently lost to decomposition because vascular-plants were by now comparatively rare at this level and most CO_2 was taken up into relatively long-term storage by the increasingly-vigorous *Sphagnum* carpet.

Ignoring for a moment, however, the cover of vascular plants and considering only the *Sphagnum* layer of the T2/3 hummock, according to the model of vigorous hummock growth proposed by Belyea and Clymo (1998) it is likely that the value for net CO_2-C accumulation in the hummock over the growing season would then be significantly higher than the equivalent values obtained for the A1 hollows.

Overall, then, Komulainen *et al.* (1999) showed, firstly, that blocking the ditches raised the water table by some 20 cm. This then resulted in a sharp decline in respiration from all but the hummocks. There was also an overall and generally significant increase in CO_2-C sequestration (101 g CO_2-C m^{-2} in the hollows, and 54 g CO_2-C m^{-2} in the hummock over the growing season) whereas the drained control site displayed little capacity to sequester CO_2.

It is also worth noting that 50-70% of measured respiration in the re-wetted site came from leaves, roots, and microbial populations using root exudates – in other words, a high proportion of decomposition was associated with young material rather than, as is often assumed, 'old carbon' released from the peat-carbon store by flooding.

10.2.1.2 Drain blocking and CH_4 flux

When considering whether or not to re-wet an area of peat bog, concern is often expressed about the possibility that methane emissions will rise as the water table rises and that this CH_4, with its much greater Greenhouse Warming Potential, may cancel out or even overwhelm any benefits gained from reduced CO_2 emissions and possible CO_2 sequestration. Again, this is a somewhat intuitive response because it has been shown that a direct relationship exists between water-table depth and methane emissions (Laine and Vasander, 1996) - higher water levels produce higher CH_4 emissions.

The remarkable thing about this particular concern is that there is relatively little evidence to support it. This is not to say that the concern is not valid – it may be, given what we know about CH_4 and water tables – but, as Colls (2006) observes, no published evidence is currently available regarding the pattern of CH_4 emissions from any drained British bogs subject to restoration by ditch-blocking. Gray (2005) had intended to investigate this relationship at the Cross Lochs, Sutherland, but in the event was unable to do so. A certain amount of information exists for methane emissions from restored cut-over peatlands in Finland and Canada, but, as with the pattern of CO_2 fluxes described above, a commercially-worked peat extraction site is a very different prospect from an area of blanket mire cut only by a series of moor-grip drains.

This is an obvious gap in the existing body of knowledge, although the work currently being carried out in Sutherland as part of the programme of research to inform UK carbon reporting should provide some valuable data (Levy, Billet and Crowe, 2008). Nevertheless, even in the absence of more specific information it is possible to apply certain known characteristics of methane production to the question of re-wetting drained bogs.

It has been repeatedly demonstrated that vascular plants growing within hollows and pools act as major conduits for CH_4 transport directly to the atmosphere (e.g. MacDonald *et al*., 1998; Laine *et al*., 2007). This ‘methane shunt’ by-passes the oxidative processes available within the upper layers of the peat column and can therefore result in CH_4 emissions which are an order of magnitude larger than those recorded from the general bog surface.

Conversely, as discussed earlier in Section 10.1, the findings of Frenzel and Karofeld (2000) and Bortoluzzi *et al*. (2006) suggest that a continuous carpet of *Sphagnum*, even if it is a very thin carpet, can provide very substantial levels of CH_4 oxidation prior to release into the atmosphere. Vigorous *Sphagnum* carpets tends to present something of a challenge to vascular plant species. Thus it is common to find lower densities of vascular plants, growing with reduced vigour, within such carpets. Similarly, as shown by MacDonald *et al*. (1998) and Laine *et al*. (2007), flooded areas of bare peat do not *per se* release large quantities of methane.

Interestingly, the same vascular plants growing on terrestrial parts of the bog surface (T-zones, *sensu* Lindsay, Riggall and Burd, 1985), appear not to act so readily as methane shunts. Thus while Frenzel and Karofeld (2000) found that hare’s-tail cotton grass (*Eriophorum vaginatum*), for example, was a major emitter of CH_4 when growing in an A1 hollow, this same species has not been identified as a major source of CH_4 when growing as part of the general T-zone bog vegetation. Laine *et al.* (2007) found no correlation between CH_4 emissions and the vascular green leaf area index (VGA_{AER}) of several CH_4-conveying species in the T-zones of their site, whereas a good correlation was observed in the A1/A2 hollows, lending weight to the idea that it is the *combination* of microtopography and vascular-plant species which provide the methane shunt.

If indeed it is necessary for aquatic-zone (A-zone, *sensu* Lindsay, Riggall and Burd, 1985) microtopography to be combined with methane-transporting vascular plants before the methane-shunt mechanism works effectively, this has potentially significant relevance to the question of drained bogs and their re-wetting. As discussed in Section 10.2.1.1 above, Komulainen *et al*. (1999) found that drainage encouraged normally-terrestrial species such as heather (*Calluna vulgaris*) and hare’s-tail cotton grass (*Eriophorum vaginatum*) to establish themselves within A-zone hollows. Such an increase in abundance of potentially CH_4-transporting species in response to drainage may thus *increase* CH_4 emissions from hollows via the methane-shunt mechanism. On the other hand, re-wetting of the bog by ditch-blocking allows the typical *Sphagnum*-rich hollow communities to re-assert themselves and thereby reduce or displace the stands of CH_4-transporting vascular-plant assemblages.

Looking at the likely sequence of events when a drain is blocked, the first stage will generally result in the accumulation of pools behind the upslope face of each dam. The water will be devoid of vegetation and the drain sides may consist only of bare-peat faces. Under such conditions, CH_4 emissions are likely to be very low unless methane is released as bubbles from the ditch floor, which is a distinct though episodic possibility. If the ditch has already re-vegetated to some extent, it is possible that anaerobic decomposition of the flooded vegetation will release CH_4, as observed by Kelly *et al*. (1997) when peatland vegetation was inundated to create a reservoir. The quantities of inundated vegetation likely to be involved in a typically narrow ditch are likely to be rather small, however.

Within a year or two of ditch-blocking, it is possible for species such as common cotton grass (*Eriophorum angustifolium*) or bog bean (*Menyanthes trifoliata*) to become established. During this phase it is likely that such vascular-plant stands will begin to operate as methane shunts, but this possibility is limited by two other factors, and in time is likely to be mitigated by a third factor.

Firstly, if the initial coloniser of the ponded water is *Sphagnum* rather than cotton grass or bog bean, this would probably limit the potential for, and vigour of, any subsequent development of cotton grass or bog bean stands within the ditch. Development of such a *Sphagnum* carpet (most likely *S. recurvum* or *S. cuspidatum*) would provide a CH_4-oxidising layer at the water surface, thereby significantly reducing any CH_4 emissions which might otherwise have emerged from the drain line.

Secondly, a certain amount of evidence exists to suggest that prolonged oxidation of peat can all-but remove methane-production potential from the peat profile. Kettunen *et al.* (1999) observe that drought conditions extending for more than two months can effectively eliminate methane-production sites from the peat. Even if water tables are raised again, it can therefore take a considerable period of time before methane production is re-established. Indeed Tuittila *et al.* (2000) found that methane production on a restored site remained at a lower state than natural background emissions for several years, albeit their site was a cut-over bog rather than a drained blanket mire. If, therefore, during these months or years of recovering methanogen populations a floating *Sphagnum* mat can begin to develop across the water surface (and *Sphagnum recurvum* can colonise with remarkable speed), any methane eventually released may be oxidised by the *Sphagnum* mat.

Finally, whether or not *Sphagnum* is the first coloniser of a flooded drain-line, it will almost certainly become the dominant species within 5-10 years provided there are no serious issues of water quality or exceptionally high-energy water flows. Consequently even if methane-shunt communities of cotton grass or bog bean do become established soon after re-wetting, this is likely to be a short-term successional phase which will be replaced by a *Sphagnum* carpet within only a few years.

Figure 36 shows the extent to which a sheet-steel dam installed in a drain at Blawhorn Moss, Midlothian, had become completely overwhelmed by *Sphagnum* growth within 10 years of the dam being installed. It can also be seen from Figure 36 that a significant population of common cotton grass (*Eriophorum angustifolium*) continues to grow within the *Sphagnum* sward, and it would be very interesting to know the extent to which this cotton grass sward continues to act as a methane-shunt through the *Sphagnum* carpet.

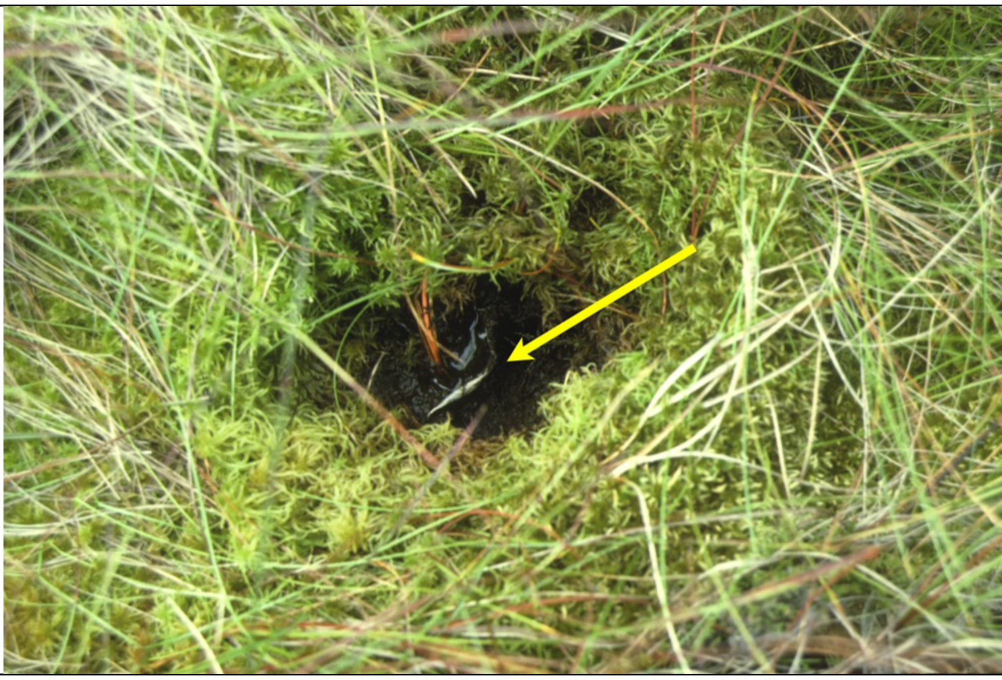


Figure 36. Overwhelmed sheet-steel dam in drain at Blawhorn Moss, Midlothian.
The dense *Sphagnum* carpet which now fills a former drain-line at Blawhorn Moss NNR, Midlothian, has been pulled back to reveal the top of a sheet-steel dam (arrowed) which was installed 10 years earlier into a drain-line which, at that time, was devoid of vegetation.

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There is currently insufficient evidence to suggest that drain-blocking would result in any greater levels of CH_4 emission than the 6.9 g CH_4 m^{-2} yr^{-1} estimated by Hargreaves and Fowler (1998) for Loch More blanket mire in Caithness – a patterned blanket mire where a number of pools were shown to be large sources of CH_4 because bog bean (*Menyanthes trifoliata*) was providing a substantial methane-shunt. These levels of emission may indeed be rather high for much of the blanket mire south of the Scottish Central Belt, given that bog bean is not a common component of blanket mire vegetation south of this line, and given that Laine *et al.* (2007) generally recorded lower CH_4 emission values for a blanket bog in western Ireland. Indeed Dinsmore *et al*. (in press) recorded methane emissions an order of magnitude lower from Auchencorth Moss, also lying within the Scottish Central Belt, than those recorded by Laine *et al*. (2007) for Ireland.

Exceptions to the pattern of events described above include conditions where the peat has cracked significantly as a result of shrinkage and drying caused by the drainage, and such cracks either render it impossible to install reasonably water-tight dams, or the cracks provide alternative routes for CH_4 deep within the peat to escape directly to the atmosphere. Holden (2005a,b) emphasises that macropores, cracks and soil pipes may play a substantial part in transport of gaseous carbon from blanket mire systems but this pathway has yet to be investigated in any detail.

10.2.2 Drain-blocking and DOC production

In economic terms, there is no doubt that one of the most anxiously-debated topics in relation to the re-wetting of drained blanket mire is the question of whether damming drainage systems will reduce or increase levels of DOC released from the system. Water utility companies are already deeply concerned about current levels of DOC production from blanket mire catchments and are understandably anxious that restoration methods designed to reduce this problem may actually either increase levels of DOC or change its nature in such a way that current treatment systems will need to be re-designed.

Two key studies have been undertaken on blanket peat in northern England, looking at the short-term and somewhat longer-term effects of ditch-blocking on release of DOC (Worrall, Armstrong and Holden, 2007b; Wallage, Holden and McDonald, 2006), while Worrall, Gibson and Burt (2007a) have investigated the potential for modelling DOC release in response to ditch-blocking.

10.2.2.1 Short-term response of DOC to ditch-blocking : Worrall, Armstrong and Holden (2007b)

This study investigated the effect of ditch-blocking on DOC release for a 10-month period following installation of several different dam types into a range of drain-lines within the Forest of Bowland in the north-west of England. Several sets of repeat data were obtained for conditions prior to dam installation, then the study site was visited weekly between mid-March and late October 2005.

In part, this study was designed to investigate the relative performance of differing drain-blocking methods, but the results showed little difference between the various approaches used. The difference between blocked and unblocked drains was significant, however. Water flowing within blocked drains always had more colour than water in unblocked drains, and the difference was generally a doubling of colour in the blocked drain compared with the un-blocked drain. This increase in water colour was also evident at the catchment scale, although the observed increase at the catchment outlet was relatively small.

Drain blocking was shown by Worrall *et al.* (2007b) to raise water levels in the adjacent peat and therefore it was initially assumed that this increase in anaerobic conditions would result in reduced breakdown of plant material into DOC. The fact that water colour (and thus DOC) increased in those ditches which had been blocked was something of a surprise, and a number of possible explanations are offered by Worrall *et al.* (2007b):

- DOC already in the peat may have been flushed out by the rising water table – although the scale of water-table rise seems unlikely to have any more effect on pre-existing DOC than would direct flushing by rainfall;

- increased DOC production may have occurred in response to reduced levels of sulphate in the soil water, although Worrall *et al*. (2007b) suggest that changes in specific absorbance mean that this is unlikely to be the explanation;
- the suggestion favoured by Worrall *et al.* (2007b) is that a process termed an 'enzyme-latch' mechanism exists within the peat, this process controlling DOC production, but it is a process which can be disabled or switched off during dry periods but then requires a significant period of time for the enzyme system to re-activate itself after re-wetting.

A fourth, little-discussed, possibility is that drainage has encouraged the growth of vascular plants across formerly-wet ground (otherwise why originally put drains there?) as described by Komulainen *et al*. (1999) for their drained bog site in southern Finland. When the drains are finally blocked this large standing crop of vascular-plant material is compromised by rising water levels. Significant parts (most likely fine roots) then begin to die and decompose to release DOC. Komulainen *et al.* (1999) certainly observe such die-back in response to rising water levels in their study, and it seems reasonable to assume that some degree of die-back would occur here too. DOC is known to arise mainly from fresh carbon, and thus if die-back of living vegetation is likely to be occurring, it might be simplest to assume that the observed increase in DOC arises from this source until proven otherwise.

10.2.2.2 Medium-term response of DOC to ditch-blocking : Wallage, Holden and McDonald (2006)

A large proportion of the Oughtershaw Beck catchment, which forms part of the upper reaches of Wharfedale, northern England, was drained in the 1960s but some of the drainage network was then blocked in 1999. Wallage *et al.* (2006) set out three experimental areas within this catchment to investigate the pattern of DOC release from an area of 'intact' peat, a drained area, and an area of drain-blocking. The study thus examined the impact on DOC of drainage after a 40-year period and the effects of drain-blocking after a five-year period.

The three study areas were selected to be as similar to each other as possible, and it is suggested by Wallage *et al*. (2006) that the vegetation across the three areas was similar because there were no obvious differences in species diversity between them. Given that one area had been subject to drainage for 40 years, one had been subject to elevated water tables for 5 years, and one had remained 'intact', it seems rather unlikely that the three areas would have similar vegetation other than at the very broadest scale. At the very least, it might be expected that the drained area would show increased biomass even if the species composition was largely similar, but Wallage *et al.* (2006) do not explore this or other possible sources of biotic difference between the study sites.

Notwithstanding the lack of detailed ecological information about the three study areas, Wallage *et al.* (2006) found significant differences in DOC release and water colour between the three sites. DOC and water colour were both highest in the drained site, where the majority of DOC and colour appeared to be produced at depths of between 10 cm and 40 cm with a marked peak at 20 cm depth, this peak being 40% higher than release of DOC and colour at the same depth from the 'intact' site.

Drain-blocking reduced DOC and water colour by 69% and 62% respectively compared with the drained site. Somewhat unexpectedly, however, the drain-blocked site produced DOC values which were 52% lower than those from the 'intact' area. Wallage *et al.* (2006) noted that the nature of the DOC produced from the drain-blocked site appeared to come from relatively humified peat, being rich in darker humic acids rather than the paler fulvic acids typical of the DOC from the 'intact' site.

Wallage *et al.* (2006) therefore investigated the peat depth at which DOC was produced in the three experimental areas. All three areas showed a general increase in DOC and water colour with depth, with the 'intact' site displaying a stepwise increase at 10 cm, suggesting that the upper 10 cm of the peat column represented the acrotelm in this plot. Maximum DOC values were obtained at 20 cm depth from the drained peat, whereas the drain-blocked peat showed a curious *minimum* at 10 cm depth.

Wallage *et al.* (2006) conclude that, at least after 5 years, drain-blocking is a very effective means of reducing DOC and water colour. They make no observations about when the transition from the elevated DOC levels noted by Worrall *et al.* (2007b) to reduced levels might occur. Wallage *et al.* (2006) suggest that the lower levels of DOC emerging from the drain-blocked site compared with the 'intact' site indicate that the blocked site has in some way exhausted its store of DOC pre-cursors.

While this might be one explanation, it is also possible that the 'intact' site is not so intact. If it is assumed for a moment that the 'intact' site is in fact somewhat degraded and is releasing elevated levels of DOC as a consequence, it is quite possible to imagine that, after five years of elevated water levels, the drain-blocked site might be in better habitat condition than the supposedly 'intact' site.

In the drained site, lowered water tables mean that the plant biomass is able to root more deeply into the peat before encountering the lowered water table. Vigorous growth of vascular plants encouraged by the thick zone of aerated peat along the margins of the drains results in a rich root-mass, with the finest (and therefore most easily decomposed) roots at around 20 cm depth, thereby giving a maximal value for DOC production at this level.

In the drain-blocked site, formerly-vigorous vascular-plant communities can be expected to have become much less vigorous and suffered die-back over time, thereby releasing elevated levels of DOC for a period after drain-blocking. If elevated water tables have subsequently encouraged the steady development of a more vigorous *Sphagnum* carpet at the expense of vascular-plant biomass, the surface layers down to 10 cm or so are likely to have only limited potential to produce DOC. The suggestion of such a layer is almost suggested by Figure 2 of Wallage *et al.* (2007).

Beneath this *Sphagnum*-rich layer the peat remains relatively rich in root and leaf-base remnants from the former vascular-plant cover, albeit now somewhat resistant to decomposition because they are increasingly bathed in the microbial inhibitor sphagnan. The change in DOC character within the drain-blocked area could be explained by the death of these deeper roots as their associated aerial parts have died off, leaving a complex mixture of freshly-dead roots mixed with older carbon. Fontaine *et al.* (2007) suggest that such a mixture is likely to stimulate decomposition of 'older' carbon. It would have been good to know the age of the DOC from the drain-blocked site.

This alternative scenario is presented as a way of emphasising the potential value of having detailed information about the vegetation, microtopography, and other biotic factors when trying to interpret hydro-chemical (or indeed hydro-topographical) data from peatland systems. Suffice it so say that the work of Wallage *et al.* (2007) clearly demonstrate that the longer-term effect of drain-blocking is a substantial reduction in DOC release, although at this stage in the re-wetting sequence it seems that the DOC is of a different kind from that released by drained blanket bog.

If the biotic link described above is to any degree correct, then it might be reasonable to expect that DOC levels would fall even further once the last of the vascular-plant root systems have finally decomposed or become immobilised within the catotelm peat. The nature of the DOC should also return to the paler fulvic acid type when DOC is once again largely a product of vascular-plant decomposition in the surface layers.

10.2.2.3 Modelling the impact of drain blocking : Worrall, Gibson and Burt (2007a)

A model of past and future DOC trends for the Trout Beck catchment at Moor House National Nature Reserve, in the northern Pennines, was constructed by Worrall *et al.* (2007a) in order to provide some idea of possible future capacity needs for water treatment works in the area. The rather short record of water colour at Moor House was extended by using a much longer record from the Broken Scar treatment works at Darlington and then extrapolating this back for the Trout Beck catchment to 1971.

Worrall *et al.* (2007a) then use a single colour-calibration curve to calculate past DOC values, although Wallage *et al.* (2006) highlight the fact that such single-curve calibrations can be misleading even if used only within a single site. It is thus not clear how closely the predicted levels of DOC within the model reflect the actual past levels of DOC.

A model for predicting future levels of DOC is then assembled, based partly on a model of water-table and drainage behaviour for deep peat in Norway constructed by Braekke (1983). This Braekke model is described by Worrall *et al.* (2007a) as being based on 40 catchments throughout Norway, although in fact only five locations were involved, all based in north-central Norway, with generally two or three experimental plots at each to make a total of 11 experimental plots. Braekke (1983) restricted his measurements to the north-central Norwegian growing season, which he considered to run from 1 June to 30 September.

Worrall *et al.* (2007a) state that 'differences in water-tables between drained and undrained peat occurred only in the summer months' but Braekke (1983) gives no data for water tables outside the growing season, so it is not possible to know whether differences existed or not. What is fairly certain, however, is that the growing season at Trout Beck extends for a longer period than 1 June to 30 September. There are thus some concerns about the basis of the subsequent model constructed by Worrall *et al.* (2007a).

Notwithstanding these concerns about the model, Worrall *et al.* (2007a) construct a predicted historical record for the Trout Beck catchment and then run the model forward to see what DOC values emerge to the year 2012. In 100 cycles of this model, 77 runs show a steady rise in DOC release from the Trout Beck catchment up to 2012 while 23 runs predict a decrease. This pattern of results is then used to conclude that 'for a pristine catchment the DOC concentration will continue to rise at an average of 3% per year', thus although the model predicts that drain-blocking can reduce DOC release, it is also concluded that such actions can 'only be considered a medium term strategy' in the face of rising DOC release from pristine catchments.

The difficulty with these assertions and conclusions is that the Trout Beck catchment cannot be considered 'pristine'. It is a highly modified area of blanket mire (as is much of the northern Pennines) and cannot be used as an example of how mires in this area would behave if they were in the pristine state. However, by calling it 'pristine' this leads to the conclusion that this is the best that can be achieved. If in fact by appropriate management the Trout Beck catchment could be turned into something much closer to a naturally functioning active blanket mire it is possible – indeed one might almost say likely – that the picture of DOC release would be significantly different.

One of the other key factors missing from the model constructed by Worrall *et al*. (2007a) is a significant biotic component. It is stated that drier conditions will cause the acrotelm/catotelm boundary to be lowered into the peat, but this is not at all necessarily the case. Recognising that the peat archive for the past 8,000 years shows how bog systems have continued growing remarkably consistently through major shifts in climate, it is possible to point to the biotic response whereby a shift in the vegetation to that more typical of hummock tops can maintain water levels at more or less the same position in the peat but provide a vegetation more tolerant of such drier conditions.

It is this biotic feedback process which has provided bog systems with a resilience to climate changes which is often overlooked. It is only because the Trout Beck catchment, and Moor House as a whole, are so comparatively degraded that the typical responses of this biotic resilience are not as evident as they might be. Nonetheless, a model which incorporates such biotic feedback responses is more likely to mirror underlying, fundamental habitat trends and consequent hydro-chemical responses.

10.3 Research needs

Several areas which would benefit from further research can be identified:

10.3.1 Drain blocking – the vegetation response

While much hydro-topographic hydro-chemical information is now being gathered in association with hydrological management and drain-blocking on raised and blanket mire systems, what is much less-well documented nowadays, however, is the biotic component of such work, together with its important feedback processes. The hydrological responses to drainage, and to drainage restoration, are being investigated and modelled in considerable detail but the quantity of equivalently-detailed biotic (vegetation/ecological) data is all the more remarkable for its absence than for its presence.

In terms of the possible increased release of either methane and DOC, for example, this lack of vegetation detail is both significant and surprising. By far the most substantial emitters of CH_4 from a bog surface have been shown to be vascular plants acting as methane shunts (*e.g.* MacDonald *et al*., 1998; Laine *et al*., 2007), while there is evidence to suggest that *Sphagnum*-rich carpets can be substantial inhibitors of methane release (*e.g.* Frenzel and Karofeld, 2000). Equally, it is widely acknowledged that the main source of DOC is generally the younger carbon within the surface layers of a bog (*e.g.* Holden *et al*., 2007a), and this carbon is directly linked to the vegetation which is its source. Meanwhile Fontaine *et al*. (2007) provide evidence that decomposition of older, deeper organic matter can be stimulated by the addition of younger carbon-rich material such as root systems.

There would thus appear to be a strong argument for assuming that the detailed vegetation composition of a drained bog, in terms of its species composition, its quantity of biomass, the proportion of vascular plants to bryophytes, and the extent to which vascular plant roots penetrate the peat, might be of considerable significance to the question of whether drain-blocking is greenhouse-friendly or greenhouse-harmful. It is all the more puzzling, therefore, to understand why such work is not being done. It should clearly form a high priority for future work, not as studies of biotic responses in isolation, but as fundamentally integrated parts of all forms of investigation into peat bog drainage and drainage restoration.

10.3.2 Relationship between terrestrialisation, paludification and restoration strategies

The work should make a clear distinction between, on the one hand, what is effectively terrestrialising *Sphagnum* such as the dense mats of *Sphagnum recurvum* and *S. cuspidatum* in erosion gullies which are infilling, and on the other hand, bog surfaces where the *Sphagnum* species and microtopography are those characteristic of a natural bog surface. Such species are also characteristic of conditions where re-development of a peat-forming surface is occurring through paludification rather than terrestrialisation.

In other words, DOC production should be examined in the general peat surface *between* blocked drains, rather than simply in the blocked drains or gullies themselves. This is because the general peat surface will become increasingly paludified following ditch-blocking, while the blocked drains themselves will infill through terrestrialisation. The species which re-develop on the general peat surface will tend to be terrestrial (T-zone) *Sphagnum* species such as *S. capillifolium*, *S. papillosum* and *S. magellanicum* which, being terrestrial species, are able to form peat more rapidly than aquatic-zone species such as *Sphagnum cuspidatum* or *S. recurvum*.

10.3.3 Relationship between CO2 flux and drain-blocking

There is a clear need for more CO_2-flux studies on UK bog systems generally, as highlighted by Gray (2005), but as part of this there is a particular need to investigate the CO_2 flux associated with drain blocking on UK bogs in general, but especially so on blanket mire systems given their greater extent and the widespread nature of 'moorland' drainage.

10.3.4 Relationship between CH4 flux, drainage and drain-blocking

There is a clear need for more CH_4-flux studies on UK bog systems generally, as highlighted by Gray (2005), but given the widely-expressed concerns that re-wetting of bog systems will lead to significantly greater emissions of CH_4, as part of this there is a particular need to investigate the CH_4 flux associated with drain blocking on UK bogs.

11 DISCUSSION TOPIC 2 Windfarms on peat

The construction of windfarms on peat has raised important questions about the relative carbon balance between the carbon savings of the windfarm and the loss of carbon storage and sequestration from the peat as a result of construction. The greenhouse gas impact of windfarms on peatlands depends very much on the type of peatland involved and the scale, location design and management of the windfarm.

At one end of the spectrum, well planned developments under certain peatland conditions can limit the carbon loss through careful design and layout of the built infrastructure, combined with high quality peatland restoration management as part of the development. At the other extreme, poorly-planned and managed developments involving large areas of construction across extensive areas of active blanket bog have the potential to cause significant carbon loss, as well as a range of other impacts. One of the most dramatic responses to human actions shown by a blanket mire system in living memory was the bogslide which occurred close to the village of Derrybrien, Co. Galway, in October 2003 (Lindsay and Bragg, 2004). The bogslide occurred while Ireland's largest windfarm was being constructed across the summit and upper slopes of Cashlaundrumlahan, above Derrybrien. The liquid peat flowed for several days, blocking a road into the village. Much of the material travelled more than 20 km down a local river system into Lough Cutra, where it killed an estimated 50,000 fish (approximately 50% of the fish population of the lake) and halted plans to turn Lough Cutra into the water supply for the adjacent town of Gort.

Such extreme events have occurred on at least two other occasions in Ireland within the last 5 years. Significantly, however, there have been no cases of peat failure reported for any windfarm development in England, Scotland or Wales. The fact that the British windfarm industry has remained free of such major incidents may indicate that areas having a potentially high risk of slope-failure can be identified and avoided by careful planning of development.

Understanding precisely how the various elements of a windfarm affect the local peatland carbon dynamics under different situations requires further study but many of the basic principles concerning drainage and the impact on peatland functioning and carbon loss can be applied. Further testing through effective monitoring would undoubtedly help in future planning and design of windfarms.

Observations made in this section of the present report are further informed by:

- the author's own visits to a number of windfarms on deep peat in Britain, Ireland and Spain;
- the contents of several Environmental Impact Statements provided as part of the planning process by various developers seeking consent to build windfarms on peat soils in Britain; and
- by the author's field survey work on several of these sites.

It is interesting to note that the majority of material prepared by both developers and objectors when a windfarm development is proposed concerns the turbines. This is perhaps understandable in the sense that the turbines are the most evidently-visible part of the development, and they also represent the largest part of the manufacturing and assembly process. However, from a peatland-habitat perspective, the turbines are a relatively minor part of the potential impact. By far the most extensive part of the infrastructure, and the part of the development with the largest potential for impact on the habitat, is the need to construct roads across the peat to enable every turbine to be constructed and serviced throughout the life of the windfarm. Indeed the stated intention for most developments is that the road system would not be removed but would remain in place when the windfarm is decommissioned.

11.1 Windfarm roads

From a carbon perspective, therefore, the key questions for a windfarm proposal are those concerning the likely impact of a ribbon development passing through the peat bog landscape. The roadway must be capable of being used safely at all times by heavy construction and maintenance equipment, yet in many cases must cross peat which may be 2-3 m deep (in some cases as much as 5 m) and will generally cut across the natural direction of water movement through and over the peat.

These requirements pose significant challenges, partly of an engineering nature, but probably more significantly of an eco-hydrological nature. The engineering challenges have been recognised and are being addressed in a variety of ways by the windfarm industry, but the eco-hydrological issues are technically more challenging and are only at the earliest stages of being addressed.

11.1.1 Floating roads

If the road is to cross peat which is more than 1 m deep, the favoured approach of developers now is to use the 'floating road' technique, in which a geotextile is laid on the bog surface, crushed rock laid on that, with the possibility of further geotextile layers being interbedded with further layers of crushed rock until a road thickness of some 0.5 m has been achieved.

This method is described by developers as having limited impacts on the peat and the carbon store because no peat is removed and the road will 'float' on the bog surface and thereby disrupt the hydrology of the bog to only a limited degree.

However, there is virtually no documented research, let alone published research, examining the ecological impacts of such roads. Indeed remarkably little has even been published about the mechanics of constructing floating roads.

One key document is that produced by Forestry Civil Engineering in 2006 (MacCulloch, 2006) which is a synthesis of much hard-won experience and expertise in the subject. Perhaps the most useful aspect of MacCulloch (2006) is the way in which it emphasises the uncertainties which still exist even within the engineering. Thus it states in the introduction:

> *"Road construction within a peatland constitutes a major challenge to the designer and contractor. As the characteristics of peat can differ enormously across a deposit, and even within a few metres, few definitive rules exist to assist the engineer. This variability makes the construction of a risk free, low volume/low cost road within a peatland, unrealistic. All involved with the construction of roads over peat must be aware that failure is to be expected and a process must be in place to minimise and manage the impact of any such failure."*
>
> *MacCulloch, 2006, Foreward*

It is important to understand that 'failure' here does not mean a bogslide of a scale to match the Derrybrien slide. 'Failure' more often means a small zone of slippage or peat movement over an area of just a few metres, or an unacceptable degree of slumping on short sections of a constructed road. More importantly, the term 'failure' relates to the construction process rather than the possible impacts on the eco-hydrology of the blanket bog system. MacCulloch (2006) has little to say on this latter aspect largely because so little research has so far been undertaken into the long-term eco-hydrological impacts of floating roads.

Use of such floating roads almost inevitably leads to disruption of surface and near-surface flow, and reduces flow through the peat because of compression and consequent reduction in hydraulic conductivity within the peat. Figure 37 shows the combination of factors associated with the disruption of surface flow.

The scale of impact may be as great as that modelled by Holden (2005a) for moor grips, as discussed in Section 9.1.4.3, Discussion Topic 1a, depending on whether or not the concentrated outflows from the culverts diffuse evenly across the bog surface without creating channelled erosion. Such even, diffuse outflows would seem unlikely, given that most natural flush systems lead into channelled flow. It is already possible to point to some examples of culverts where channelled flow has eroded peat at the culvert outflow, at least over short distances. However, the evidence has not yet been gathered to show whether, over the life of a windfarm, culvert outflows can maintain diffuse flows or whether, at least in some cases, they eventually lead to channelled flow.

The impacts of a floating road on sub-surface peat can also be expected to result in sub-surface ponding. A number of Environmental Impact Statements for windfarm developments have stated that water flow through peat is very slow and thus ponding at depth would not be an issue. However, ponding will still occur at depth, it will just happen more slowly than surface ponding. If, on the other hand, there are sub-surface pipes in the peat this ponding might even occur more rapidly than surface ponding.

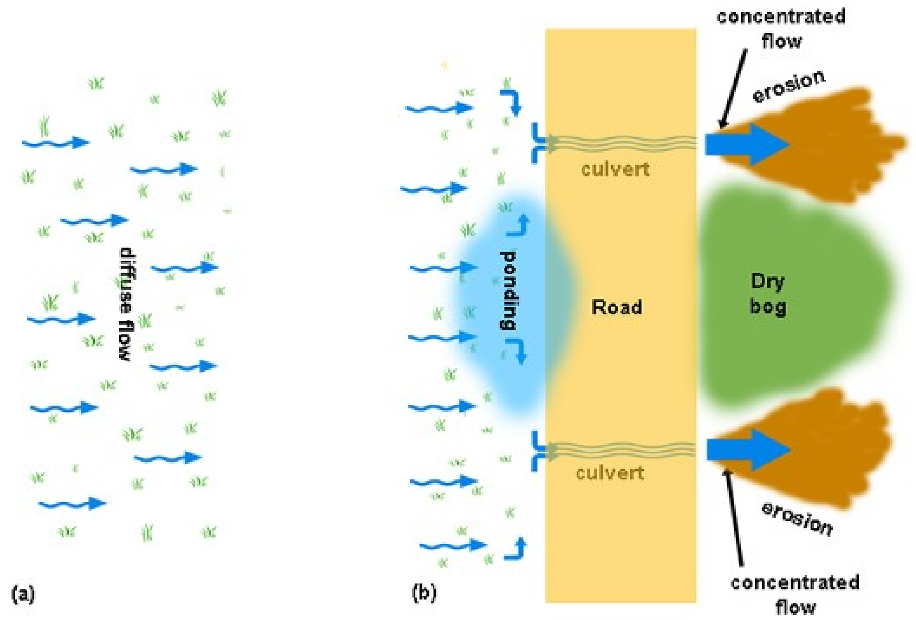


Figure 37. Disruption of surface seepage by floating roads
(a) The normal pattern of diffuse flow across a blanket bog surface. (b) Where a floating road cuts across the lines of flow, water is often ponded on the upslope side of the road unless directed to a culvert by a road-side drain. Parts of the bog downslope from the road will become drier even if culverts are present, but if the culverts induce channel formation through surface erosion (particularly during storm events), most of the downslope area will be rendered drier than before.

Illustration R A Lindsay

Sub-surface ponding is caused by compression of the peat by primary consolidation and secondary compression resulting from the increased load provided by the roadway itself, and also from the use of the roadway by vehicles. Again, little work has been undertaken to investigate this effect. However, road settlement following construction is acknowledged as an issue for at least some sections of floating road (MacCulloch, 2006), while studies have revealed the problems of settlement experienced by normal trunk roads when crossing peat deposits (Nichol and Farmer, 1998). Indeed, in the wider engineering community the problems of settlement when constructing on peat are widely recognised. These issues have been summarised by Hobbs (1986) and, more recently and in more depth, by Bell (2000).

Some evidence also exists to suggest that, even without a road surface, regular use of a quad bike across a bog system may result in changes to bulk density deeper in the peat. Figure 38 shows Transect 2 from the work of Shotbolt *et al*. (1998) at Bad á Cheo, as already seen in Figure 13 of the present report, but this time with an arrow to indicate an evident band of greater bulk density extending down through the whole of the surface 1 m of peat. Shotbolt *et al*. (1998) offer possible explanations for this band, but also note that it corresponds to the route used regularly by a forest worker on a quad bike. The weight of a floating road would represent a greater and more sustained load on the peat than the occasional passage of a quad bike.

Surface blockage and sub-surface compression would both therefore tend to result in short- and long-term ponding on the upslope side of any floating road with potentially consequent drying of the peat downslope for the road. Furthermore, compression and consolidation of the peat causes the road surface to sink steadily into the bog and thus cause waterlogging problems for the road surface itself. A waterlogged running surface is an unstable surface for service and repair vehicles and is thus an unacceptable hazard. Such areas require ongoing maintenance. Specifically, such areas of settlement require the addition of supplementary road-surface material. This adds further to the load acting on the peat beneath and so further secondary compression occurs, potentially leading to yet further sinking of the road surface and so a positive feedback cycle becomes established.

It has been suggested by some Environmental Impact Statements that tracks can be engineered with layers of graded stone to mimic the hydraulic conductivity associated with seepage of water across the bog (*e.g.* Todmorden Moor, Reaps Moss and Crook Hill, Southern Pennines : Sinclair Knight Merz, 2008). In practice this may not be such a realistic idea. It is obviously possible to engineer a road initially with the appropriate grade of material which may (or may not) mimic the surface hydrology, but as the road undergoes consolidation and compression the layers of graded stone become distorted and buried, while fine material from the road surface is washed down into the matrix and blocks the pore spaces, thereby altering the designed hydrological nature of the road.

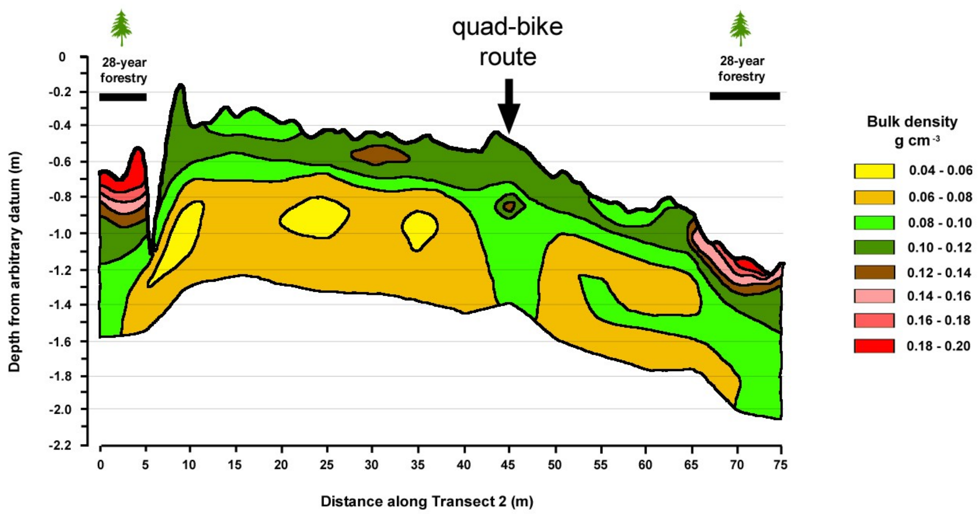


Figure 38. Possible increase in bulk density at Bad á Cheo by use of quad bike.
This is the same Transect 2 as shown in Figure 13 of the present report, but with the regular route of a quad-bike also indicated. It can be seen that this point corresponds to an increase in bulk density which extends down through the whole of the measured topmost 1 m of the bog.

Adapted from Shotbolt *et al.* (1998)

11.1.2 Excavated roads

Where peat is less than 1 m deep, windfarm developments generally propose excavation to create a running surface on the underlying sub-soil. It may be possible to fill the road trench entirely with stone, resulting in a running surface which is level with the surface of the bog. Clearly for any reasonable length of road this would have significant implications for the volumes of fill material required. This material may then act as a drain if it is more conductive than the surrounding peat, or as a dam if it is less conductive than the peat. It is unlikely that the fill material would exactly match the properties of the surrounding peat. It is more likely that it will become a barrier to water movement, or a line of preferential water movement.

Where an excavated road is not back-filled to the bog surface, such roads are, in effect, very wide drains cut into the peat. They therefore have all the impact issues associated with moorland drains, but also potentially cause increased sediment loading because the sub-soil (and the surfacing material) which was formerly protected by peat is now directly exposed to the heavy and prolonged rain events which inevitably occur from time to time in blanket mire areas.

Sediment loading into surrounding water-courses spans a range of conditions, from the low, chronic levels of small-scale outwash during 364 days of the year, to the loads induced by a single high-rainfall storm on the 365th day which moves large quantities of sediment into adjacent water-courses. The impacts of such events are difficult to document unless a continuous monitoring system is established, and such systems have rarely been installed on stream systems associated with wind farm road networks. Consequently available information about the ecological impacts of road/track systems on upland stream systems is quite limited. However, Greave and Gilvear (2008/9) have obtained clear evidence for elevated sediment and DOC loads coming from the Braes of Doune windfarm in central Scotland. In particular, they observe that these elevated sediment loads do not appear to be restricted to a construction-related pulse only, but appear to represent a sustained source of loss from the site.

One of the issues to arise from the excavation of tracks is the question of what to do with the resulting stock of peat. It is often proposed that it should be spread somewhere else as a management scheme, but this is not likely to develop anything similar to the bog vegetation lost by excavation in any meaningful timescale. In addition, such peat is unlikely to lie within the natural bog water table, and is thus most probably going to oxidise and be released as carbon dioxide to the atmosphere.

11.2 Carbon savings for windfarms on peat (Nayak *et al.*, 2008)

The purpose of windfarm construction is obviously to reduce UK carbon emissions, but when such windfarms are constructed on peat there is an inevitable loss of carbon from the long-term carbon store. The critical question is just how large is that carbon loss in relation to the carbon saving achieved by operation of the windfarm?

The most recent and undoubtedly most comprehensive assessment of this question to date is the carbon calculator created by Nayak *et al.* (2008). This consists of a multi-worksheet Excel spreadsheet together with an accompanying report. A number of issues raised by these two documents merit further discussion. Overall, however, the Nayak *et al.* (2008) model is a useful means of exploring possible carbon-balance outcomes but the key to its widespread adoption and use will be the testing and establishment of robust parameters and the determination of realistic values for sites.

11.2.1 Revision of worked example values

Firstly, a worked example is provided by Nayak *et al.* (2008) as part of the documentation. The values from this worked example are now increasingly appearing in windfarm development documents under the description of 'default' values. However, the values given in the worked example are very specific to one particular windfarm development and many of the values are not appropriate to other developments (J. Smith pers. comm.).

11.2.2 Grid emission savings

The critical value to be derived from the Excel model is the windfarm payback time, but this value is provided on the basis of three scenarios depending on whether the grid-emission savings are calculated on the basis of coal fired, fossil fuel or grid-mix power generation. Nayak *et al.* (2008) question the SNH guidance on this, suggesting that the 430 g CO_2 kWh^{-1} used is 'overly pessimistic' and that a more realistic value would be that proposed by the British Wind Energy Association (BWEA).

However, a ruling by the Advertising Standards Authority (ASA) in December 2008 resulted in the BWEA agreeing to use the same value proposed by SNH, rather than the higher value suggested by Nayak *et al*. (2008). This generally increases the payback times by between 41% and 100% relative to the other options given by Nayak *et al*. (2008).

11.2.3 Extent of drainage around drainage features

Nayak *et al.* (2006) review a number of published studies of peatland drainage, including Boelter (1972) and Gilman (1994), and provide a formula for calculating drainage distance based on hydraulic conductivity values, but they emphasise that this model is only suitable for level sites with uniform peat and that the majority of sites will not meet these criteria. Their worked example uses a distance of 100 m. This seems like a reasonable starting point, given that Nayak *et al.* (2008) subsequently suggest that the drainage distance associated with a sloping site might be 200 m. Holden (2005a), of course, indicates that drainage distances could be considerably more, as has already been discussed in Section 9.1.4.3 above.

11.2.4 Peat landslide hazard

It is stated by Nayak *et al.* (2008) that the model assumes a zero loss from peatslide events because developers will be expected to follow the Scottish Government guidance of managing peatslide risk. This guidance does not *remove* peatslide risk. As Nayak *et al.* (2006) themselves observe, this guidance 'should lead to a reduced likelihood of peat landslides occurring.

However, if this carbon calculator is to be used to inform the planning process prior to granting of consent, it might be argued that this section of the calculator should attempt to reflect more closely the recommended approach set out in the Scottish Executive guidance on peatslide risk assessment (Scottish Executive, 2006). The decision-tree used to determine whether any significant peatslide risk exists at all (essentially presence of peat, and slopes of more than 2°), could readily be incorporated into the spreadsheet.

The likelihood of 'hazard' and the potential scale of 'exposure' (essentially potential risk and potential scale of impact) could be likewise incorporated. Given that both are assessed in a semi-quantitative way and are then assembled in a matrix within the Scottish Executive guidance (Scottish Executive, 2006), this approach would be amenable to incorporation within the Nayak *et al*. (2008) spreadsheet format and could be done so in a more meaningful way than the current 'Peatslide' element of the spreadsheet. Such incorporation would also give useful guidance to developers in terms of how 'hazard' and 'exposure' can be employed meaningfully within the EIA process.

11.3 Research needs

Several areas which would benefit from further research can be identified:

11.3.1 Engineering properties of peat

It is becoming increasingly acknowledged by soil engineers that the traditional tools for measuring the strength of mineral soils do not work reliably in peat (Creighton, 2006; Dykes and Kirk, 2006; Yang and Dykes, 2006). There is therefore an evident need for further research into the most effective ways of measuring peat strength. and its behaviour under differing loads and under differing weather and climatic conditions. This research would then feed into revised development guidance.

11.3.2 Eco-hydrological studies of floating roads

There is an almost complete absence of published, peer-reviewed scientific literature describing the eco-hydrological impacts of floating roads (and indeed excavated and rock-fill roads) on the blanket mire landscape. This poses significant challenges for the planning process because there is currently little in the scientific literature which can guide decision-makers in relation to the likely eco-hydrological impacts of these structures. Consequently there is an urgent need to establish best-practice studies of differing road types in differing conditions. These studies should be undertaken within the context of microtopes, mesotopes and catchments, and should record the impacts at the scale of microtope, nanotope and vegetation. For this purpose, vegetation description would need to be at a finer level than that provided by the NVC. Hydrological parameters such as overland flow, near-surface flow, catotelm flow, and macropore and pipe flow should also be measured, as should the engineering properties of the road structure and the peat, over time.

11.3.3 Development of the Nayak *et al*. (2008) carbon model

While representing a valuable tool with which to assess the potential carbon budget of a peat-based development, a number of parameters and assumptions which form part of the model developed by Nayak *et al*. (2008) would benefit from further testing and discussion.

12 DISCUSSION TOPIC 3a
Forestry on peat – the carbon balance

12.1 Forestry on peat : ground conditions

Conifer trees do not generally colonise areas of blanket mire naturally in Britain. Areas of conifer plantation on blanket mire show little tendency to self-seed into the surrounding landscape. This is because soil-moisture levels are so high while nitrogen, phosphorus and potassium levels are so low. Grazing by hares, deer and sheep undoubtedly also play their part.

If conifers are to grow successfully on areas of blanket peat more than 50 cm – 1 m deep, the ground must be specially prepared for them and then they must be planted. Such ground preparation is designed specifically to lower water tables and to raise nutrient levels in the peat. Both actions have considerable potential significance for the carbon balance of peat bog systems.

12.1.1 Drainage for forestry

The general impacts of drainage on peat bog systems has already been reviewed in Discussion Topic 1 of the present report, but the drainage required for forestry on raised or blanket peat represents a different scale of intensity compared to moorland drainage undertaken for typical upland agricultural purposes. Wood (1974) highlights the fact that the deep peats of northern Scotland had been considered unplantable until the 1940s, but between the late 1940s and mid-1950s an extensive series of experimental plantations had been established using the innovative Cuthbertson plough in combination with new tree species, particularly lodgepole pine (*Pinus contorta*). Wood (1974) comments that this species was found to be the only feasible crop on the deepest, wettest peats. It has since been shown that, in part, this is because lodgepole pine is capable of drying out the peat around it through highly effective evapotranspiration and is therefore now used extensively for this purpose as a 'nurse' crop for the more commercially valuable sitka spruce.

12.1.1.1 Water table and forest drainage: Braekke (1983)

Braekke (1983) reviews the process of drainage for forestry on peatlands and comments that the primary aim of drainage is the creation of an aerated zone for the tree roots no matter how wet the weather. He identifies that the typical practical objective of such drainage is a water table which is maintained at somewhere between 30-60 cm at a point mid-way between drains, but he also points out that because the water table forms an ellipse rather than a straight-line gradient between drains, a simple measurement of the water table in the drains and then at the mid-point between them does not tell the whole story.

The ellipsoid shape of the water table means that the water table is relatively high across a substantial proportion of the inter-drain space and thus biologically, for the root systems of the trees to function well, it is necessary to lower the overall water table somewhat more than might be suggested by a simple straight-line model of the water table. This greater level of drainage is termed the *biological drainage norm* as opposed to the *hydro-technical drainage norm* by Braekke (1983), and he observes that this value seems to be around 32 cm for young Scots pine (*Pinus sylvestris*) but 40 cm for Norway spruce (*Picea abies*). For both species, the biological drainage norm increases to between 40-60 cm at, fittingly, 40-60 years of age (presumably because the weight of timber pushes the root system deeper into the peat, though this is not explicitly stated).

Braekke (1983) describes an experiment involving a total of 11 study plots distributed across five locations in northern and central Norway. The locations lie between latitudes 66°57' N and 69°14' N, which are equivalent to the northern coast of Iceland and Jan Mayen Island in the Greenland Sea respectively. They are thus northern Boreal in character, although one site (Borgmyra) lies on the more oceanic Lofoten Islands and might thus be considered to have some affinities with northern Scottish blanket mire.

The experiment described by Braekke (1983) involved measurements of water table under differing levels of drainage intensity both in terms of drain depth and drain frequency (distance between drains). Measurements were taken through the growing season only (1 June – 30 September). Drainage intensity was additionally varied by adding shallow ploughing furrows at either 2 m or 4 m intervals between the main drains in some experimental plots. These ploughing furrows are typical of British peatland forestry. The seedling tree is planted on the upturned ridge from the furrow in order to give the seedling root system a useful, additional aerated zone of soil above the drained bog water-table.

From this work, Braekke (1983) concluded that drains deeper than 1 m did not significantly increase the drainage effect, although in practice drains must be dug deeper than this in order to retain the desired depth of drain following three or four years' of subsidence (in some cases drains had lost more than 75% of their original depth after this period).

Distance between drains rather than drain depth was found to be a more important factor in determining water-table draw-down. This distance was found to be influenced to a substantial degree by level of rainfall. Thus if the June-September rainfall total for a site is 250 mm, drains can be spaced at 30 m in order to achieve a 'biological drainage norm' water-table depth of 30 cm, whereas if the June-September rainfall is 700 mm, the drains must be spaced at 7.5 m to achieve the same effect. This mirrors the findings of Coulson *et al*. (1990) in terms of drainage impacts at differing blanket mire elevations in Britain, as discussed in Discussion Topic 1 of the present report.

Perhaps most surprising of all, however, is Braekke's (1983) conclusion that 'furrowing had little or no effect on mean water table level'. In fact this comment is subsequently qualified by observing that shallow furrowing had no effect on water tables specifically when the mean water table was held lower than 30 cm by drainage. This is because the water table was already below the base of the furrows. After 10 years they had subsided to a depth of only 17 cm and were thus unable to contribute significantly to the drainage process. Braekke (1983) states that furrows must retain a depth of at least 30 cm if they are to influence the 'biological drainage norm' of a 30 cm water-table depth on a forested peatland site.

Braekke (1983) also qualifies his observations about the limited effects of furrows in relation to his most oceanic site, at Borgmyra. He notes that the permeability of the bog at Borgmyra was very low, and under these conditions a ditch spacing of anything between 6 m and 20 m did little to achieve the desired draw-down of 30 cm in the bog water table. In these circumstances the presence or absence of furrowing was much more influential. Braekke (1983) therefore recommends that, in order to maintain a water-table depth of 30 cm throughout the growing season, any smooth areas of low-permeability bog should have drain-spacings of 20 m with furrows of 30 cm depth at 4 m intervals, while low-permeability bog with hummocks and hollows should have a drain-spacing of between 10-12 m with furrows at 2-4 m intervals.

This view of peatland hydrology is of course entirely focused on obtaining a 'biological drainage norm' of 30 cm depth in order to grow trees successfully on deep peat. Whether the drains and furrows have an effect on the original vegetation of the bog is not the issue. Indeed no information is provided about the original condition of this, or any of the other study sites investigated by Braekke (1983). Borgmyra is described as 'coastal heather bog' but no data are given for the site prior to the drains and furrows being dug. The potential implications of such intensive drainage on the carbon balance of the site are clearly substantial but without any baseline data it is difficult to judge how much effect the ploughing of drains and furrows might have had.

The one clear piece of evidence for a change in the carbon balance lies in the figures for surface subsidence across the drained parts of Borgmyra, obtained because the ground surface was carefully surveyed prior to drainage. Braekke (1983) notes that this subsidence ranged from 3 cm to 10 cm over the 10-year study period, with the largest degree of subsidence found on the deepest parts of the peat (or at least on the most level part of the bog, which is generally where the deepest peat occurs). Thus it appears that on well-humified peat up to 10 cm of subsidence can occur as a result of intensive forestry-style drainage in a period of 10 years, even without trees. The key thing in terms of carbon balance, of course, is the proportion of this subsidence which results from compression and the proportion which results from oxidative wastage. Before exploring this further, however, it is worth just looking in a little more detail at one of the other features to emerge from this study.

Braekke (1983) states that furrowing had little effect on the *mean* water table, but a closer examination of maximum and minimum values for the water table reveals a rather different pattern of behaviour between furrowed and unfurrowed bog at Borgmyra. The water table was measured at 10-day intervals between 1 June and 30 September, and a number of associated factors were also recorded for these study plots (see Table 19).

Table 19. Comparison of maximum and minimum values for lowering of water table, and associated parameters, provided by Braekke (1983) for two drained study plots, one with ploughing furrows, one without, on Borgmyra, Lofoten Islands, Norway.

	Maximum lowering		**Minimum lowering**	
	Un-furrowed	**Furrows**	**Un-furrowed**	**Furrows**
Mean water-table level (cm) 1 June – 30 Sept	73.8	75.8	2.9	1.7
Water-table level (cm) 1 June	62.0	85.0	0	0
Ditch distance (m)	31.4	32.6	5.0	6.0
Permeability (cm/24 hr)	6.1	6.5	2.8	0.7
Surface slope (cm/m)	4.4	6.7	0.2	0.1

It can be seen from Table 19 that in general the water table sinks lower and does not rise as high in the furrowed plot as it does in the un-furrowed plot, despite the fact that the furrowed plot has more widely-spaced drains, the minimum permeability of the furrowed plot is 4x less than that of the un-furrowed plot, and the furrowed plot lies on a slope with a maximum slope-angle only half that of the un-furrowed plot. These site factors would all tend to increase wetness in the furrowed site, but under extremes of both wetness and dryness, the furrowed site tends to be the drier of the two.

Braekke (1983) himself highlights the importance of furrowing within the low-permeability Borgmyra site by presenting water-table data for three ditch-spacings on the furrowed study plot for a dry year and a wet year. The water-table range is of interest here, and can be seen in Figure 39.

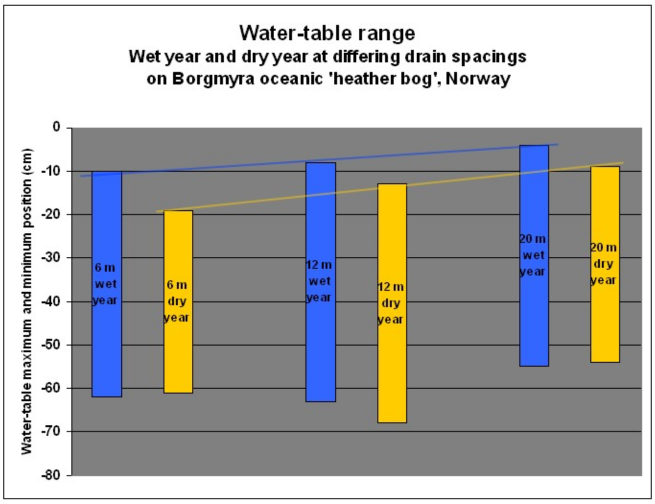


Figure 39. Water table ranges on a drained, furrowed site (Borgmyra, Norway) with differing drain distances. Coloured bars represent water-table ranges recorded by Braekke (1983) on Borgmyra 'heather bog', Norway, when subject to differing drainage intensities, the distances between drains being indicated on each coloured bar. All study plots have also been furrow-ploughed at 4 m intervals. Blue bars represent the water-table range during a 'wet' year, while orange bars represent the water-level range in a 'dry' year. General trend lines are shown for the highest water-level positions in the wet and dry years across differing drainage intensities.

It is immediately apparent from Figure 39 that even in a wet year at maximal drain spacing (20 m), the water table rises to within 4 cm of the bog surface but never actually reaches it. At minimum drain spacing (6 m) in a dry year, the water table never comes closer than 19 cm to the bog surface. In all cases, the lowest water-tables lie between 50-70 cm, which exceed even the deepest thickness of a normal acrotelm. Consequently it can be assumed that catotelm peat is being subject to aeration here, even without the presence of trees. If this gives rise to vigorous growth of deep-rooted heather (*Calluna vulgaris*) – as the description of the bog implies – the root systems could provide the fresh organic matter necessary to stimulate decomposition of old catotelm peat, as described by Fontaine *et al*. (2007).

It is also worth noting that, under these defined conditions of rainfall and permeability, the trends in maximal water-table height given in Figure 39 also suggest that, under wet conditions, a drain-spacing of some 25 m would at least permit the water table to rise to the bog surface occasionally. In a dry year, however, it seems that a drain-spacing of around 35 m would be required for this to occur.

It would thus appear evident that the style and intensity of drainage normally associated with forestry on peat, whether this be in northern Norway or northern England, involves a lowering of the water table to an extent which can be assumed to cause slumping and potentially oxidative wastage of the peat soil (specifically the catotelm). Braekke (1983) records a maximum of 9 cm for such slumping on this low-permeability site but does not attempt to identify how much of this might be due to oxidative loss of carbon from the peat (oxidative wastage). In a second study, however, Braekke (1987) attempts to do precisely that.

12.2 Forestry on peat: consolidation, compression and oxidative wastage

Braekke (1983) found evidence that a low-permeability bog site such as Borgmyra could show up to 9 cm of subsidence after 10 years of intensive drainage. This can be contrasted at one extreme by the Holme Fen Post in Cambridgeshire. Over a period of 150 years this post has demonstrated a 4 m lowering of the ground surface in what was undoubtedly once a lowland raised bog. Borgmyra indicates a slumping rate of 0.9 cm yr^{-1} whereas the extreme conditions of Holme Fen give a rate of 2.67 cm yr^{-1}.

It is generally assumed that blanket mire peat is more decomposed (humified) than raised bog peat and will therefore tend to show much lower rates of slumping – as indicated by Borgmyra. While this is often true for many (though by no means all) areas of blanket peat in England and Wales, much blanket mire in Scotland has a relatively low bulk density not so different from that found in many lowland raised bogs. Consequently it is unhelpful, indeed it can be positively misleading, to make a sharp distinction between the likely response to drainage shown by raised bog peat and blanket mire peat, particularly where intensive drainage is involved. Two studies illustrate the way in which raised and blanket mire systems can show much the same response to afforestation.

12.2.1 Forest impacts over a 26-year period: Braekke (1987)

In this study, Braekke (1987) investigated two sites, one of which was originally an open lowland raised bog located in southern Norway some 45 km south of Oslo (Sem bog). The bog was drained at 30 m spacing in 1957 and then fertilized in 1959. A naturally-seeded Scots pine (*Pinus sylvestris*) woodland subsequently developed across the site. A detailed description for the site was obtained prior to these actions taking place. The site was then investigated by Braekke (1987) in 1983, some 26 years after drainage.

Braekke (1987) firstly notes that the bulk density of the peat at 0-10cm, 10-20 cm and 20-40 cm has hardly changed over the intervening 26 years. Given that the peat now contains a substantial proportion of fine tree roots which are significantly denser than the original peat, Braekke (1987) assumes that this indicates a considerable decrease in the density of the original peat matrix. This might be explained by an increase in the proportion of macropores and cracks within the peat – in other words, there is now less peat and more air in the matrix.

Braekke (1987) then identifies that the surface of the bog was found to have subsided by about 70 cm between 1957 and 1983. This equates to a rate of 2.69 cm yr^{-1}, almost identical to that observed at Holme Fen in Cambridgeshire. Braekke (1987) calculates the concentration of phosphorus within the surface layers of the peat and uses this to reconstruct the surface thickness – now 40 cm thick – as originally having been 58 cm thick (Braekke actually says 62 cm, but this does not fit with his own figures).

Braekke (1987) proposes that to explain the total 70 cm fall in ground surface, 35% of this fall occurred within the uppermost 62 cm of the bog, while 19% occurred below this depth. This translates into a loss of 251 g C m^{-2} yr^{-1} from the uppermost 62 cm of the original bog surface, then a further (approximately) 60 g C m^{-2} yr^{-1} from the thickness of the bog deeper than 62 cm. Total losses therefore amount to 311 g C m^{-2} yr^{-1} from the whole 3.25 m peat column, although Braekke (1987) emphasises that even this figure is likely to be an under-estimate because his method of calculating subsidence tends to under-estimate the true extent of subsidence.

It is worth pointing out, however, that Braekke (1987), when he uses phosphorus levels in the peat to reconstruct original thicknesses, appears not to have taken into account the input rates of aerial phosphorus. This could significantly reduce his estimate of peat loss.

12.2.2 Forest impacts over a 28-year period: Shotbolt *et al.* (1998)

The work undertaken by Shotbolt *et al*. (1998) at Bad á Cheo, Caithness, in relation to bulk density profiles for a northern blanket mire has already been discussed in Section 4.3.1.7 of the present report. The main focus of their work, however, was the subsidence which had evidently occurred in the peat following planting of the Bad á Cheo experimental forest. They were able to compare the present surface with a set of benchmarks established in 1966, two years prior to the planting of the forest. The level of slumping across the Bad á Cheo peatland during the intervening 28 years could thus be assessed.

Certain limitations in the method used by Shotbolt *et al.* (1998) need to be recognised at the outset, however. The original measurements taken in 1966 were based on the intersections of a 50 m x 50 m grid from which a detailed contour map was produced linked to certain fixed metal posts. Shotbolt *et al.* (1998) had only the original metal posts and a contour map with contours at 1 ft (30 cm) intervals. They were unable to repeat the exact grid locations and were unable to survey points deep in the forest. Consequently the ‘original’ 1966 bog surface at each of their 1998 sample points had to be interpolated from the 1966 contour map, which introduces the possibility of interpolation errors. These errors could (in a realistic ‘worst-case’) be as large as 15 cm, but could certainly be in the range of 5-10 cm on a fairly regular basis.

Furthermore the 1966 contour map gives a generalised bog surface with no indication of microtopography. It is not clear whether the generalised 1966 surface represents the highest, or the lowest, levels of the surface microtopography or some undefined position in between. This alone can introduce height differences of 20-30 cm, and the uncertainty is further increased by the fact that Shotbolt *et al.* (1998) do not themselves define the basis of their own surface measurements in terms of the microtopography – consistently on the highest, or the lowest, position or somewhere in between. This is important given both the tussocky nature of vegetation associated with drained peat and also because a transect used as part of the study runs out from the forest across a T3, T2, T1, A1, A3 pool system (*sensu* Lindsay, Riggall and Burd, 1985) having substantial microtopographic variation (see Figure 40).

Measurements of the ground surface within the site were also taken in 1987, and although the same methodo-logical issues are relevant, this does at least gives some opportunity to look at the rates of change over time, rather than having only a starting-point and an end-point.

Bad á Cheo has a total rainfall of 930 mm and a summer rainfall (June – Sept.) of 300 mm. This summer rainfall is very similar to that of the pine-covered raised bog studied by Braekke (1987) south of Oslo, although winter conditions are rather different on the two sites. Given this evident similarity, it is interesting to compare the response to drainage and afforestation of Bad á Cheo with that noted by Braekke (1987) for Sem bog in Norway, especially as the study periods are of similar duration (approximately 27 years).

Shotbolt *et al.* (1998) observe that the trees have dried, compressed and ultimately cracked the peat. The most marked subsidence occurred beneath the forest plots themselves. Ground levels fell by up to 45 cm in the first 19 years, and then by as much as 80 cm in total by 1996. The overall maximum, minimum and average rates of subsidence were fairly linear over the study period, making it reasonably easy to predict that by the end of the likely rotation period of 35 years for this particular lodgepole-sitka mix in Caithness, total subsidence beneath the trees could be as much as 100 cm, although the plantation average may be closer to 80 cm.

This scale of shrinkage is interesting because it matches very closely the rate observed by Braekke (1987) at Sem bog, a lowland raised bog. Lowland raised bogs are generally thought of as having much lower bulk densities than typical blanket mires (and thus the lowland sites would be more prone to shrinkage and compression). However, the values of bulk density for Bad á Cheo observed by Shotbolt *et al.* (1998) are not so different from those characteristic of a lowland raised bog and are much lower than values more usually thought of as typical of British blanket mire. As observed earlier when discussing bulk density in Section 4.3.1.7, it can be seen from Figure 41 that significant parts of the peat profile along Shotbolt *et al.*'s (1998) Transect 1 have bulk densities of 0.06-0.08 g cm^{-3} and even values as low as 0.04 g cm^{-3}.

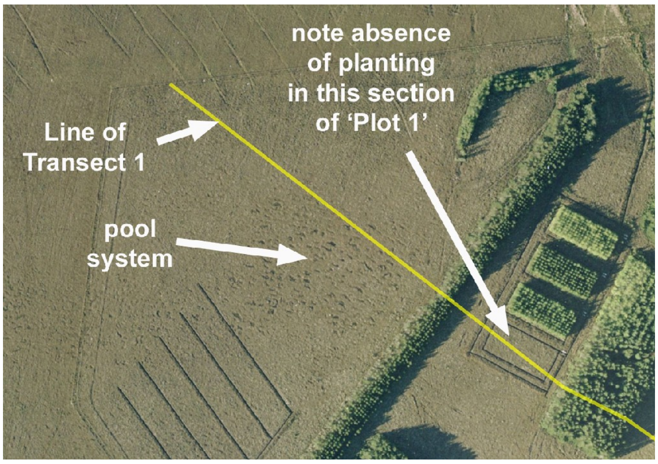


Figure 40. Transect 1 detail at Bad á Cheo, Caithness.
Aerial photograph showing position of Transect 1 in relation to pool system and small experimental plot (part of 'Plot 1') left unplanted in 1987, although Shotbolt *et al.* (1998) describe this as 'planted in 1987'. Part, but not all of it, was planted in 1987. Older trees planted in 1968 are visible (bottom right), as is the 1968 shelterbelt.

Caution must therefore be applied to any assumptions made about the likely response of an given blanket mire to forest drainage. If the blanket peat is soft, as is the case at Bad á Cheo, shrinkage and consequent subsidence might be substantial. If the peat is denser, with a bulk density similar to that of the blanket mire studied by Braekke (1983) at Borgmyra, subsidence would be more limited. Clearly some blanket mires have bulk densities more usually associated with raised mires and this fact, combined with the physical weight and evapo-transpirative capacity of the trees, can result in very substantial subsidence beneath a conifer plantation. Indeed Shotbolt *et al.* (1998) observe, as did Braekke (1983) for his site, that subsidence extended deeper into the peat than the 90 cm surface layer which formed the basis of their study, but, unlike Braekke (1983) they do not provide any estimates for the scale of this deeper subsidence.

The scale of subsidence beneath the forest plots is very evident from Figure 41, but so, too, is the fact that the ground adjacent to the forest plots is also drawn down to form a slope towards the forest stands. Interestingly, Shotbolt *et al.* (1998) are unable to show a relationship between proximity to trees and water-table draw-down because, as they observe, the fact that the ground surface is also sinking means that as fast as the water table falls, the ground surface follows it down (as discussed in the earlier Section 9.1.3 of the present report). Measurements of the water table *relative* to the ground surface alone might thus lead to the erroneous conclusion that the trees had not affected the bog water-table at all. That said, the water table *within* the forested plots was found to be an average of -55 cm whereas the water table outside these plots was very significantly higher, at 20 cm.

Shotbolt *et al.* (1998) note that both bulk density and water content of the peat are very significantly correlated with proximity to the forested plots, indicating that in fact the trees have de-watered adjacent ground and caused it to become denser through compression and decomposition. They suggest that subsidence has probably not extended further than 50 m from the forest plots, but then create a regression model from their data to examine this. From the regression model they conclude that subsidence does not extend beyond 30 m from forested plots.

In fact the regression model presented by Shotbolt *et al.* (1998) does not match with the plotted data, which clearly show that subsidence occurs up to 40 m from the forest edges, with one or two points extending much further. More significantly, however, Shotbolt *et al.* (1998) point out that the rate at which subsidence effects are spreading outwards from the forest plots is increasing. They therefore conclude that at a rotation maturity of 35 years the effects of subsidence will have extended as much as 40 m from the forest edges.

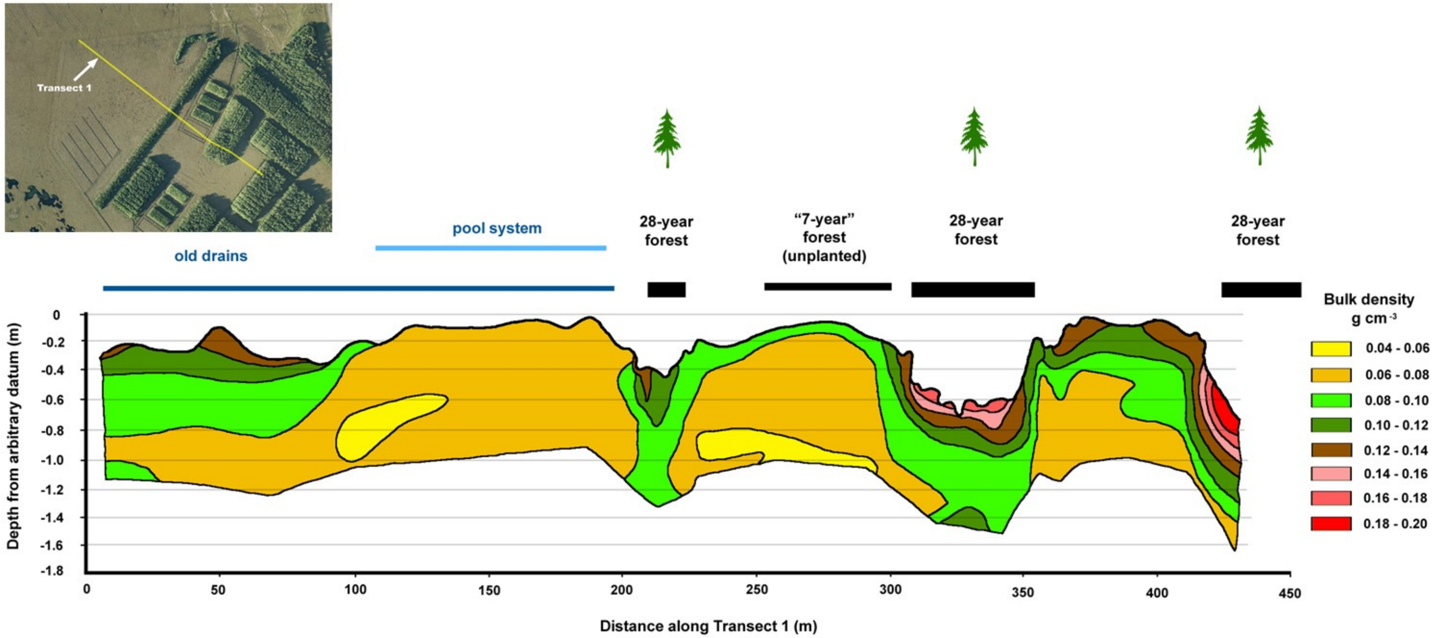


Figure 41. Transect 1 at Bad á Cheo, Caithness.
Aerial photograph showing position of Transect 1 in relation to pool system and small experimental plot (part of 'Plot 1') left unplanted in 1987, although Shotbolt *et al.* (1998) describe this as 'planted in 1987'. Part, but not all of it, was planted in 1987. Older trees planted in 1968 are visible (bottom right), as is the 1968 shelterbelt. The substantial subsidence which has occurred beneath the 28-year forest plots is obvious, as is the evident impact on the pattern of bulk density within the peat. Note that although the ground at '7-year' forest is not planted, the surface bulk density is affected by the draw-down caused by the two adjacent blocks of 28-year forest. Note also the gradient from the pools to the shelterbelt.

Diagram adapted from Shotbolt *et al.* (1998); aerial photograph © getmapping.com

In plotting this value along with the two rates for subsidence expansion obtained in 1987 and 1996, it would seem that Shotbolt *et al.* (1998) have rather under-estimated the distance over which subsidence may spread. A value of 40 m produces a slightly sigmoid curve (see Figure 42a) suggesting that the rate of subsidence expansion will decrease as the trees increase in size. This seems unlikely, given that the evapotranspirative and interception impacts, and the weight of timber, will be so much greater when the trees approach 40 years old. A sigmoid curve would presumably be appropriate in the long term, but a levelling off could be expected some time after the trees had reached maturity.

Two alternative curves have therefore been generated , based on the four measured data points, and which reflect the increasing tendency for subsidence as the trees mature. These curves are shown in Figure 42b. The curves suggest that subsidence might extend anywhere between 50 and 80 m by the end of the forest rotation.

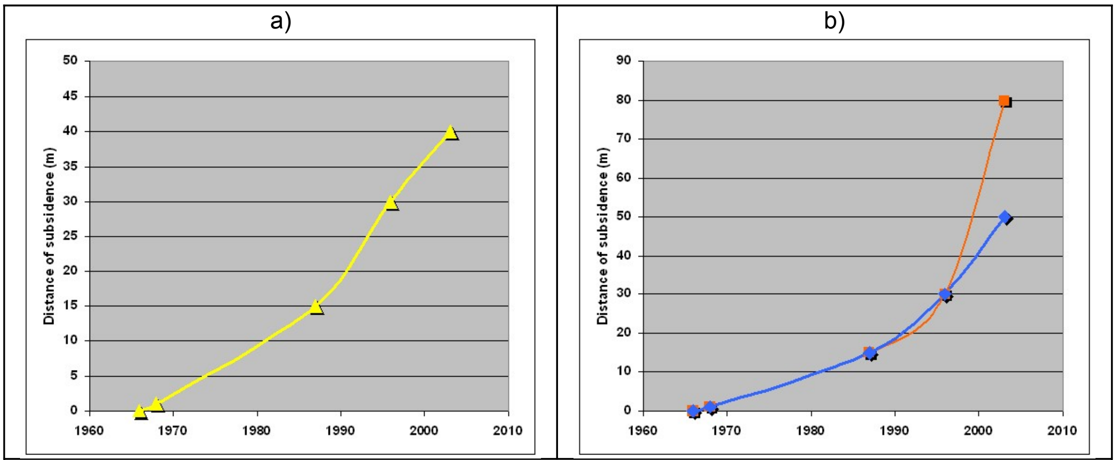


Figure 42. Expansion of subsidence during 35-year forest rotation.
Graphs based on the values obtained in 1966, 1987 and 1996 for expansion of subsidence outwards from forested plots. Plot (a) shows the measured values combined with the final limit of 40 m for 2003 predicted by Shotbolt *et al.* (1998). Plot (b) shows two curves, the blue line indicating the minimum curvature required to maintain the smooth curve of the data, while the orange line shows something approaching the maximum possible curvature consistent with the measured data. These two curves suggest that subsidence may extend to somewhere between 50 m and 80 m from the forest plots by 2003 (end of 35-year rotation).

On the basis of their prediction, Shotbolt *et al*. (1998) observe that peatland features of nature conservation value might require a buffer from plantation areas of at least 40 m for the forest rotation. In fact the evidence from Figure 42 suggests that this buffer may need to be at least 50-60 m if direct subsidence effects are to be avoided – although of course hydrological effects could be expected to extend beyond even that, as discussed in Section 9, Discussion Topic 1 of the present report. Shotbolt *et al.* (1998) also allude to the possible effect of subsequent forest rotations, suggesting that these would probably then require an even greater margin of safety.

The question of what this observed subsidence consists of – is it all simply compression, or all oxidation, or a mixture of the two – is not specifically addressed by Shotbolt *et al.* (1998). They note that substantial cracking of the peat had developed by 1996, and that this clearly allowed oxygen penetration deeper into the peat matrix. They observe that a thickness of some 35 cm of peat had entered the pool of oxidisable peat, although this does not take into account the fact that some peat may already have been lost through oxidative wastage and thus any peat lost in this way should be added to the 35 cm of oxidisable peat.

The total measure of between 60-80 cm subsidence beneath the trees at Bad á Cheo is similar to that observed by Braekke (1983) at Sem bog in Norway. There, he calculated that "*physical compaction was more than equalled by dry-matter loss*", though such comments must be read with caution, given questions raised about his assumptions with regard to phosphorus concentrations in the peat.

Braekke (1983), and subsequently Cannell *et al.* (1993), and the present report above (Section 12.2.1), provide calculated figures suggesting that carbon losses may have been in excess of 300 g C m^{-2} yr^{-1} on

Sem bog. If the same is true for Bad á Cheo, this has substantial implications for the carbon balance of such plantings on this type of peat. With carbon losses such as this, Cannell *et al.* (1993) recognise that any such forest plantation on peat would be rendered greenhouse 'harmful' almost immediately the forest is established and the position simply gets worse in subsequent years.

It is worth noting that Chapman and Thurlow (1996) recorded CO_2 emissions from forested parts of Bad á Cheo which were on average 90% higher than similar emissions measured from the open bog environment, with low values of 10 mg C m^{-2} h^{-1} from the bog in March to high values of 187 mg C m^{-2} yr^{-1} from the forest in July. It is not possible to distinguish root respiration from peat oxidation in this study, but it is clear that there are substantially-increased emissions of CO_2 from the forested peat compared to the open bog. Chapman and Thurlow (1996) were unable to detect any CH_4 emissions from the forested area, although they recorded rates of up to 1.05 mg CH_4-C m^{-2} h^{-1} from the open bog during the peak emission period of September.

There is one other factor to consider in relation to the overall zone of forest impact. Over the lifetime of a forest block, it seems fairly clear that on this type of peat a zone of subsidence will develop. This zone may be somewhere between 40-60 m wide (perhaps more) and there are no trees growing within this zone to compensate for any oxidative wastage which may be occurring there. If the forest block is very large, close to circular, and very clearly producing a net long-term carbon benefit, this edge-effect may be unimportant relative to impacts within the forest block itself. If, however, any one of these parameters is not the case, then the carbon release from this boundary zone may take on greater significance.

Also of relevance are the possible effects of downslope drying, as described in Holden's (2005a) 'topographic index' model (discussed above in Section 9.1.4.3 of the present report), together with the additional possibility of carbon losses through increased peat-pipe development as also described by Holden (2005a). All these potential factors together highlight the potential complexities which must be addressed in order to understand fully the question of peat, carbon and forestry.

12.2.3 Compression or oxidation? Anderson *et al.* (1992, 2000)

12.2.3.1 Anderson *et al.* (1992): back-calculating scale of subsidence

Anderson, Pyatt, Sayers, Blackhall and Robinson (1992) tackled the question of oxidative loss employing a rather different approach. They used as their baseline the ground survey of Bad á Cheo which had been undertaken in 1966 prior to its planting with conifers in 1968. Anderson *et al*. (1992) surveyed the same area some 25 years later, in 1990, using a set of transects along which they took measurements of peat thickness and samples of peat for volumetric analysis down the length of the peat core. They observed that the forested area was 0.4-0.6 m lower than the ground surface in an unplanted ride. The question, of course, is how much of this evident surface slumping results from oxidation of the peat?

Anderson *et al*. (1992) estimated the original thicknesses of the peat beneath the tress and shelterbelt using a surface modelled from the 1966 grid of thickness values. Anderson *et al.* (1992) note that these 1966 values were only recorded to the nearest 30 cm – specifically, as explained in more detail in Shotbolt *et al*. (1998), the contours of this modelled surface were spaced at 30 cm intervals and thus values between these had to be interpolated, with inevitable potential for error.

Consequently there is a degree of uncertainty which Anderson *et al*. (1992) estimate to be +/- 20 cm associated with this modelled surface. The transects established by Anderson *et al*. (1992), and the cores taken along those transects, were not taken solely at measured node-points from the 1966 grid because the 1966 nodes were at 50 m intervals whereas the measurements in the 1990s were taken at 1 m intervals. There is thus additional uncertainty in relation to any comparison between the 1966 and 1990s data because there is no 1966 information about the peat thickness between the 50 m nodes. Finally, the measurements of peat thickness taken in the 1990s themselves have an uncertainty of +/- 10 cm (*i.e.* total potential uncertainty of 20 cm).

Anderson *et al*. (1992) took cores from two locations along each of their three transects, one core in the centre of the unplanted ride, the other 10 m into the forest block. From these they obtained values for the relative volumes of peat solids and water at (in effect) 15 cm intervals down the length of each core. They then calculated the possible original volumes of peat for the cores beneath the trees on the basis of the volumes obtained in the cores taken from the rides, matching "the originally equivalent layer of the unplanted peat". It is not made clear how layers were identified as equivalent.

On this basis, although the ground surface beneath the forest plot is currently between 0.4-0.6 m lower than the surface of the unplanted ride, Anderson *et al.* (1992) calculated that by adding the required volumes of water back into the peat, as indicated by the volumetric calculations, the cores in the planted area would expand to the point where the present ground surface in the forest plot would sit 2 cm higher than the modelled 1966 surface (although, again, this increase cannot be proven because of the statistical uncertainty in the data). In other words, these calculations indicate that there was little direct evidence for oxidative wastage – put the water back into the peat and the bog is as deep as ever with no apparent loss of material.

Several points need to be considered in relation to this result:

- The volumetric adjustments were performed using the volumes obtained from the unplanted ride, but the peat in the ride itself is likely to have experienced drainage effects from the adjacent forest. If the peat in the rides has lost material to oxidative wastage, this would not then feature in the volumetric adjustments to the forest cores.
- The trees in the forest plot shed branches, twigs and needles, potentially adding significant thickness to the ground surface beneath the forest each year, but it is not clear how much organic matter had been added to the peat beneath the trees in this way, whereas of course no such material will have been added to the peat in the ride. Consequently the equivalence of layers may not be as straightforward as suggested. The original peat in the forest plot may thus represent only a part of the profile used by Anderson *et al*. (1992) in their calculations and may thus expand back to a lower height using their methods and assumptions.
- As a result of oxidative wastage the unplanted ride may be lower than it would naturally be, and the forest plot may be lower still through the same process, but with its greater fall partially masked by forest litter. This could only be determined by analysis of the material making up the peat cores in the forest plot, while there is no way to judge the extent to which the unplanted ride has lost thickness (or failed to gain thickness) because of oxidative wastage. The bulk-density values given by Shotbolt *et al*. (1998) only give a picture of compression and provide no clues about the quantity of peat lost to oxidative wastage.
- The measurements described by Anderson *et al.* (1992) were undertaken in 1990. Shotbolt *et al.* (1998) note that when measurements of the ground surface at Bad á Cheo were taken in 1987 by Pyatt *et al.* (1992), much of the ground surface outside the forest plots themselves had actually risen to a higher level than that recorded in 1966. Indeed Anderson *et al.* (1992) also commented on the fact that the unplanted ride which formed part of their study had risen compared to the 1966 measurements. Meanwhile the forest plots themselves had sunk by some 45 cm by 1990. When Shotbolt *et al.* (1998) came to measure these same areas, however, their data show that the unplanted ride had sunk by some 5-10 cm below the ground level of 1966 and had fallen some 10-20 cm from the elevated levels noted in 1987. In other words, substantial additional subsidence appears to have occurred throughout the site between 1987/90 and 1996. Shotbolt *et al.* (1998) also comment on the much-increased degree of peat cracking which occurred between 1987 and 1996. It is possible, therefore, that the work of Anderson *et al.* (1992) was simply too early in the drying process to detect significant oxidative losses.

Anderson *et al.* (1992) themselves observe that the scale of uncertainty in the 1966 and 1990s peat-depth surveys means that loss of peat to oxidation must be at least 30 cm before this study would be able to detect oxidative losses. Given the considerations reviewed above, the uncertainties might be even greater than this. What this study emphasises particularly clearly is the very real need to have accurate pre-drainage measurements if changes due to drainage are to be calculated with any certainty.

12.2.3.2 Anderson et al. (2000): detailed monitoring of impacts

Anderson, Ray and Pyatt (2000), in setting up a further study at Bad á Cheo, address many of the methodological limitations of the studies reviewed above. A set of study plots is established within the forest (there are no study plots outside the forest). Four forest plots which had remained as 'control' unplanted study areas were each sub-divided into four sub-plots, one of which remained as an unplanted 'control. The remaining sub-plots were then ploughed with a double-mouldboard plough and planted with sitka spruce, lodgepole pine, or a mixture.

Within these study plots, permanent markers were embedded into the mineral sub-surface strata to provide fixed reference points with which to measure peat subsidence. There is therefore no reliance on the interpretation of earlier peat-thickness measurements, and these permanent markers have been installed in such a way that subsidence at differing depths within the peat can be measured.

In addition to the detailed fixed markers for measuring physical changes in the peat, Anderson *et al*. (2000) also obtain continuous measurements of precipitation, run-off, depth to water table, and sediment deposition. On a methodological point, no mention is made of boardwalk for taking the dipwell measurements. In addition, as Anderson *et al*. (2000) acknowledge, the fact that the study plots all lie within the forest area, well within the slumping-distance already recorded by Shotbolt *et al*. (1998), means that although these plots are all described as former unplanted 'control' plots, the prior effect of the adjacent afforested plots on the study areas must be recognised, as must the ongoing effects on the unplanted 'control' sub-plots.

These are perhaps the two potential area of weakness, or at least limitations, in the data obtained by Anderson *et al*. (2000). It would also have been valuable to have a detailed description of the vegetation and microtopography in the plots before the experiments began, and ongoing descriptions of these from the 'control' plots during the course of the monitoring programme. These limitations aside, there is much which is exemplary about this study.

Anderson *et al*. (2000) present the first five years of this detailed study:

- Run-off was found to be greater in the unplanted control plots, but only in the spring and summer. This suggests that the drains and trees take up or intercept a greater proportion of the precipitation inputs than the open bog, but the capacity of the open 'control' plots to take up and hold precipitation would be influenced by the proportion of *Sphagnum* within the vegetation. No information is provided for this.
- Base flows (essentially outflows during extended periods of dry weather) were higher from the unplanted control plots than from the planted plots in two of the three measured years. This is an interesting result within the context of the debate about whether blanket bogs provide higher base flows than mineral ground. Again, a critical factor is likely to be the presence or absence of *Sphagnum*.
- Water tables were lower in the afforested plots. Mean water levels in the control plots were 12 cm below the bog surface while in forested plots the water table had fallen to 19 cm below the surface. This fall of 7 cm would be regarded as relatively insignificant to a drainage engineer, but Ivanov (1981, p.199) emphasises that a fall of only a few centimetres in the mean annual water table of a mire can result in significant changes to mire plant communities ("for several varieties of moss cover it is less than 4-5 cm").
- All plots showed a degree of subsidence associated with the perimeter ditch, highlighting the fact that the 'control' plots are not strictly free from the influence of surrounding afforestation treatments. However, on average the control plots subsided 3.7 cm less than the planted plots. What is particularly interesting is that the highest points of the ridges in the forested plots subsided by 18.5 cm while the shallow drain bottoms (furrows) subsided by 5.5 cm, reflecting the problem highlighted by Braekke (1983) that ploughing furrows must be dug deeper than their final design depth in order to achieve desired levels of draw-down.
- Subsidence occurred rapidly during the first year, amounting to 50% of total change in surface level, then slowed to the point where after 5 years there was only measurable subsidence during the summer months. This response reflects the well-established initial phase of primary consolidation where water is lost from large pore spaces – a response which typically occurs within a period of up to a year. Secondary compression and oxidation then become the major drivers in subsequent years.
- Significantly, subsidence was found to occur at all levels within the peat thickness, not just in the surface layers. Shrinkage was absent (in the control plot) or minimal in the uppermost parts of the peat profile, the main shrinkage having occurring at depth in all plots. This accords with the observations of Eggelsmann (1975) concerning secondary compression affecting deep layers, not just those immediately associated with the drain sides, as discussed in Section 9.1.1.2 of the present report.

Anderson *et al*. (2000) conclude that the levels of observed shrinkage can be explained almost entirely by consolidation and compression and that subsidence through loss of peat to oxidative wastage was negligible. Direct measurements of CO_2 loss did not form part of the monitoring regime, however, and so conclusions about carbon flux must be based on changes in ground level and changes to the water budget of the peat. While it may indeed be the case that this measured water budget can explain the observed subsidence in these first five years, as Anderson *et al*. (2000) observe, subsidence is likely to accelerate once the trees in the forested plots approach canopy closure. This acceleration is due to greatly-increased interception and a substantial increase in evapotranspiration from the tree canopy.

Probably the most important aspect of the study described by Anderson *et al*. (2000) is that it continues to be monitored, and in another 10 or 20 years' time will have gathered vitally important information about the response of peat soils to afforestation. The study could be significantly enhanced if it were also to include carbon-flux measurements, the vegetation and microtopography were to be described in detail, and the dipwells were to be measured via boardwalk.

12.2.4 Forest drainage and subsidence – oxidation vs compression

It is perhaps worth re-iterating at this point that the conditions prevailing at Bad á Cheo, both in terms of summer climate, forest drainage, development of forest cover and subsequent rates of surface subsidence, are similar to those noted by Braekke (1987) for Sem bog near Oslo. The scale of surface subsidence observed at Bad á Cheo over a similar length of time is also very similar to that observed by Braekke (1987) for Sem bog.

It would seem, from the evidence reviewed so far, that oxidative losses from peat beneath conifer plantations might involve rates of loss amounting to some 300 g C m^{-2} yr^{-1} once the forest is well established, but that such losses may be substantially lower than this during the early stages, any evident subsidence being attributable largely to de-watering of the peat rather than through oxidative loss.

12.3 Forestry on peat : carbon balance

The present report has highlighted the fact that UK assumptions about the carbon content of biomass in oceanic peat bogs may benefit from further critical scrutiny. The values used in UK carbon reporting for bog habitats are generally an order of magnitude lower than figures published for 'cool open bogs' in what is widely regarded as the most comprehensive global review of vegetation biomass (Olson *et al*., 1983) so far undertaken. Indeed this global review is cited by Milne and Brown (1997). However, the paper by Milne and Brown (1997) continues to form the basis of peat bog biomass values for UK national carbon inventory work and the most recent UK carbon-reporting exercise.

The present report has also collated a range of information about carbon content from various sources. Using what appear to be reasonable values, the present report has calculated that the surface layer (acrotelm) of a relatively natural British peat bog may indeed contain a quantity of carbon which is of the same order of magnitude as Olson *et al*. (1983 – and subsequent updates). As such, it seems possible – or at least the possibility seems worth investigating – that the carbon held in the biomass of natural, *Sphagnum*-rich raised and blanket mires in Britain might be substantially higher than is currently realised, and may in fact be equivalent to the biomass-carbon stored in, for example, lodgepole pine plantations grown on deep peat as they approach harvesting age.

This possibility gives a somewhat unexpected twist to the much-discussed question of the carbon implications of forestry on peat bogs. The question has been raised many times, and continues to be the focus of much forest-based research. Four publications in particular have contributed to this debate in the last two decades – namely Cannell, Dewar and Pyatt (1993), Cannell (1999), Hargreaves, Milne and Cannell (2003) and Colls (2006).

12.3.1 Cannell, Dewar and Pyatt (1993)

This paper considers whether planting conifers on peat bog in Britain results in a net loss or a net gain of carbon. They acknowledge that precise answers were not then possible for all aspects of the question, so their paper was based on a number of assumed scenarios.

12.3.1.1 Carbon-balance values to set the scene

Cannell *et al.* (1993) observed that reliable figures for CH_4 emissions from British peatlands were not then available, so they took values from other “temperate (non-Arctic) regions” and suggested that mid-summer emissions may be anything between 40-400 mg CH_4 m^{-2} day^{-1}. No reference is given for the source of these figures, although the reference to “non-Arctic” regions suggests that perhaps the figures are from the Boreal Region and most likely from Finland. Earlier sections of the present report have highlighted the fact that figures from Finland tend to be biased towards fens, and that values of CH_4 emissions from fens are generally significantly higher than those from bogs.

Cannell *et al.* (1993) calculate that the annual emission rates of methane from blanket bogs may thus be 4-40 g m^{-2} y^{r-1}. This can be compared with the values given by Worrall *et al.* (2003a) of 2-15 g m^{-2} yr^{-1}, Billett *et al.* (2004) of 5.5 g m^{-2} yr^{-1}, and of 6.9 g m^{-2} yr^{-1} calculated for Loch More peatland, Caithness, by Hargreaves and Fowler (1998). Chapman and Thurlow (1996) meanwhile obtained values of 4.9 g CH_4 m^{-2} yr^{-1} from the open bog at Bad á Cheo. Clearly all these values suggest that only the lower part of the range used by Cannell *et al.* (1993) might be relevant to the British blanket bog scene. Indeed the figures obtained by Dinsmore *et al*. (in press) suggest that for some bog sites even the lowest part of this range is an order of magnitude too high.

Cannell *et al.* (1993) observe that their suggested figures of CO_2 sequestration by peatland sites range from 40–70 g C m^{-1} yr^{-1}. Although these figures are much higher than the level of CH_4 emissions, Cannell *et al*. (1993) considered that probably they result in no net gain in terms of greenhouse because the GWP of 23 (as was used then) for CH_4 over a 100-year time-horizon means that both gases cancel each other out.

In discussing sequestering rates of CO_2, Cannell *et al*. (1993) note that “some deep peat systems may have reached an equilibrium state” whereby they are no longer net accumulators of peat. As has been discussed earlier in Sections 7.2 and 7.3.2.2, existing peat growth models suggest that peat bogs may be only 2/3 of the way to such a growth limit, if indeed there is a growth limit (some models give no limits to growth, merely a steadily-declining rate of accumulation). The suggestion that some peatlands may already have stopped growing because they have reached a natural limit is thus not supported by present evidence or thinking.

Cannell *et al*. (1993) discuss the carbon accumulation processes associated with conifer plantations, and observe that part of the carbon accumulation occurs in the trees themselves, but part is also provided to the soil in the form of litter. They suggest that the final equilibrium total after 100-150 years of carbon from forest litter accumulation, at 9,400 g C m^{-2}, is equivalent to the estimated rate of carbon accumulation provided by British peat bog vegetation over the same period.

Beneath the acrotelm layer in a peat bog lies a substantial quantity of carbon stored within the peat. The critical question is thus whether conifer plantations growing on this peat cause sufficient oxidation of this deeper carbon store to nullify the accumulation of carbon in the trees.

12.3.1.2 Forest plantations and loss of peat thickness

The equilibrium total for all carbon stored in the soil, litter, trees and forest products, is calculated by Cannel *et al*. (1993) for sitka spruce plantations at Yield Class 12 to be 16.7 kg C m-2. Given that the commoner species grown on deep peat is lodgepole pine, which accumulates significantly less carbon overall, it is perhaps not a fair test of the question which Cannell *et al.* (1993) attempt to answer. Furthermore, a recent estimates of forest stocks has shown that the average Yield Class for sitka spruce across Britain is YC11 rather than YC12, and it is quite likely that the lower-performing sitka plantations are those growing on deep peat. Consequently the baseline against which carbon losses from the peat are to be judged may have been set significantly higher by Cannell *et al.* (1998) than real-life experience would justify. Cannell *et al.* (1993) calculate, for example, that a single Yield Class unit represents 0.5 kg C m^{-2}. Consequently the true equilibrium total for sitka spruce in Britain (based on the lower Yield Class 11) is thus more likely to be 16.2 kg C m^{-2} rather than 16.7 kg C m^{-2}.

Cannell *et al*. (1998) use their formula for carbon storage – the same formula used throughout the present report – to calculate that the equilibrium total for sitka spruce in terms of a thickness of peat is 35.5 cm. In other words, if 35.5 cm of peat is lost through oxidation by the plantation, there has been no overall gain in carbon storage resulting from the planting these trees on the peat. In fact, if the average Yield Class is 11 rather than 12, this critical thickness is reduced by 1 cm, to 34.5 cm.

Of course this is not the whole story, because during site preparation the ground will have been ploughed, resulting in substantial increases in carbon loss as POC from the bare peat and potentially even larger losses of DOC because what was originally the acrotelm layer is now highly aerated and subject to much more rapid water outflows during periods of rain. Both factors are likely to increase very significantly the breakdown of vascular-plant tissues in what is now becoming a haplotelm, and these breakdown products will be released as DOC. Cannell *et al*. (1993) make no comment about pathways for carbon loss other than oxidative wastage to CO_2.

Given the limited focus adopted by Cannell *et al*. (1993) in terms of carbon losses, what, then is the evidence for oxidative loss beneath conifer plantations on peat? Cannell *et al.* (1993) note that Braekke (1987) recorded loss of dry matter on a peat bog in Norway colonised by Scots pine following drainage. Braekke (1987) calculated that over a 27-year period, the bog surface had subsided by 70 cm, but using concentrations of phosphorus in the peat he calculated that actual oxidative losses amounted to a thickness of 58 cm.

Cannell *et al.* (1993) point out that this is not the whole story because some oxidative wastage will have happened at deeper levels in the peat than tested by Braekke (1987), so the total height loss due to oxidative wastage would have been *more* than 58 cm. For the (claimed under-estimated) figure calculated by Braekke (1987) the quantity of carbon lost is equal to 250 g C m^{-2} yr^{-1}. Set against this must be Russell Anderson's questioning of Braekke's (1987) assumptions about background phosphorus inputs to the mire, which may significantly change these estimates of loss (Russell Anderson, pers. comm.).

12.3.1.3 Modelling of carbon gains and losses

Cannell *et al.* (1993) then set out four scenarios involving differing rates of peat oxidation and loss. Their four rates of loss are 50, 100, 200 and 300 g C m^{-2} yr^{-1}. Modelling the carbon accumulated by a sitka spruce plantation over several planting cycles and matching this against the four rates of carbon loss, they demonstrate the way in which the various scenarios play out over a 300-year period. Some scenarios involve gains in carbon relative to the original peatland for a time, others do not. Table 20 shows the point of null gain for the four rates of loss, together with the rate of 250 g C m^{-2} yr^{-1} calculated by Braekke (1987).

Table 20 shows that though there are carbon gains after 100 years for the carbon-loss rates of 50 and 100 g C m^{-2} yr^{-1}, by a rate of 150 g C m^{-2} yr^{-1} there is no such gain. At rates of loss greater than 150 g C m^{-2} yr^{-1}, there is an overall loss of carbon from the system after 100 years, and for a loss-rate of 300 g C m^{-2} yr^{-1} there is never any carbon gain. For the loss-rate of 250 g C m^{-2} yr^{-1} apparently observed by Braekke (1987), the system gains carbon until about 60 years after planting, then loses increasingly substantial quantities of carbon thereafter.

Cannell *et al.* (1993) recognise that, according to their model, carbon gains from planting sitka spruce on deep peat will only occur within the IPCC 100-year timeframe if the trees grow at Yield Class 12 and the rate of carbon loss from the peat through oxidative wastage is less than 100 g C m^{-2} yr^{-1}. They advocate further research into the rates of peat oxidation beneath conifer plantations, given the loss-rate of 250 g C m^{-2} yr^{-1} apparently observed by Braekke (1987).

It is worth re-iterating in relation to these modelled results that they take no account of losses caused by ploughing and planting at each rotation, of which there will be at least two such events during the IPCC 100-year timeframe – when the plantation is first established, and then again after 60 years at the 2nd rotation. During each of these events, significant amounts of carbon will be lost as POC and DOC, and also through oxidative wastage of the overturned ploughing turves. Furthermore POC and DOC will continue to be lost throughout the plantation rotation.

These forestry-induced losses must be balanced against the losses of DOC and POC which would have been lost from the unplanted bog site. Any net increase in loss resulting from afforestation would tend to reduce still further the modelled values. Consequently the null point for all rates of carbon loss indicated by Table 20 would need to take this net balance of losses into account.

Table 20. Scenarios for differing rates of C-loss beneath plantations on peat.
Based on Figure 4 of Cannell *et al*. (1993), four scenarios for differing rates of carbon loss from the peat beneath a sitka spruce plantation are given here. The models were run by Cannell *et al.* (1993) over a 300-year simulation period. The 'Zero benefit point' is the point in the model at which that rate of carbon loss means that the forest exactly compensates for the loss of carbon for the peat. 'Overall carbon after 100 years' represents the gains or losses in carbon indicated by the four models after 100 years (the standard IPCC time for carbon reporting). A fifth rate of loss has been added based on the loss-rate of 250 g C m^{-2} yr^{-1} calculated by Braekke (1987) as an example of actual field-based losses.

Rates of carbon loss from peat (g C m^{-2} yr^{-1})	Zero benefit point (years after planting)	Overall carbon gains (+) or losses (-) after 100 years (g C m^{-2} yr^{-1})
50	300+	+12
100	140	+6
200	75	-4
250 (Braekke, 1987)	60	-10
300	0	-15

12.3.2 Cannell (1999)

In this paper, Cannell (1999) addresses a number of questions in relation to forests, carbon storage and, amongst other things, the carbon balance of plantations on peat. Cannell (1999) gives figures for carbon accumulation in undrained UK peatlands of 0.2–0.5 tC ha^{-1} yr^{-1}. The cited source for these figures is Clymo *et al*. (1998), although that paper describes only Finnish and Canadian mires, although the reference is coupled with a Clymo 'personal communication'. The basis for these figures for the UK context is therefore not clear, particularly as the values for carbon accumulation modelled in Clymo *et al*. (1998) are quite different from those given here by Cannell (1999).

Notwithstanding the uncertain origin of the figures for carbon accumulation in UK bogs, Cannell (1999) states that if trees are planted on peat, the system will show a gain in carbon only until approximately 20-40 cm of peat has been lost through oxidation. Cannell (1999) states that using current estimates, the system would remain in carbon credit for more than 100 years, but that in the longer term an afforested bog system would experience a net loss of carbon because of continued oxidation of the peat.

12.3.3 Hargreaves, Milne and Cannell (2003)

This is a paper which appears to follow-up the recommendation of Cannell *et al*. (1993) that more field measurements and experimental work should be undertaken to establish rates of peat oxidation beneath conifer plantations. It describes a study of three conifer plantations on peat and one peatland site described as 'undisturbed peat'.

The afforested peatlands are Mindork Moss, Newton Stewart, and Bealach Burn and Channain Forest, both in Sutherland. The 'undisturbed peatland' is Auchencorth Moss, near Edinburgh. As discussed earlier in the present report, Billett *et al*. (2004) and Dinsmore *et al*. (in press) describe Auchencorth Moss as being extensively drained and also subject to commercial peat mining. The range of sites chosen by Hargreaves *et al*. (2003) are described as showing a chrono-sequence of forest development from undrained bog through to Mindork Moss, which was planted 26 years ago. Clearly, the 'undrained bog' Auchencorth Moss does not form a good baseline because it is already drained. The regularity of this drainage can be seen in Figure 43 for the part of the site given by Hargreaves *et al*. (2003) as the location of the site (and possibly the location of the instrumentation).

12.3.3.1 Methodological issues

It is firstly significant that Hargreaves *et al*. (2003) record an unusually low rate of carbon accumulation from their 'undisturbed peatland' (Auchencorth Moss). They record values of 0.25 t C ha^{-1} yr^{-1}, and they highlight the fact that this value is significantly lower than the UK mean rate of 0.4–0.7 t C ha^{-1} yr^{-1} reported by Immirzi *et al*. (1992). Hargreaves *et al*. (2003) do observe, however, that their rate is in line with carbon accumulation values from boreal peatlands.

As noted earlier in the present report, carbon accumulation values from the Boreal Region, particularly Finland, tend to have a heavy bias towards fenland systems, which characteristically have lower carbon-accumulation rates than bogs. The carbon accumulation rate recorded for Auchencorth Moss is undoubtedly low, but this is hardly surprising given that the site is so comprehensively drained, is in parts is being mined for peat, and the crown of the deepest area is dominated by a conifer plantation. In their Figure 5, Hargreaves *et al.* (2003) specifically describe the baseline site as being "undisturbed peatland before ploughing", which it clearly is not, but the values for Auchencorth Moss are used as the starting-point for a carbon accumulation model constructed for a 26-year period. It seems reasonable to question the use of Auchencorth Moss as the 'natural baseline' against which the afforested sites are to be measured.

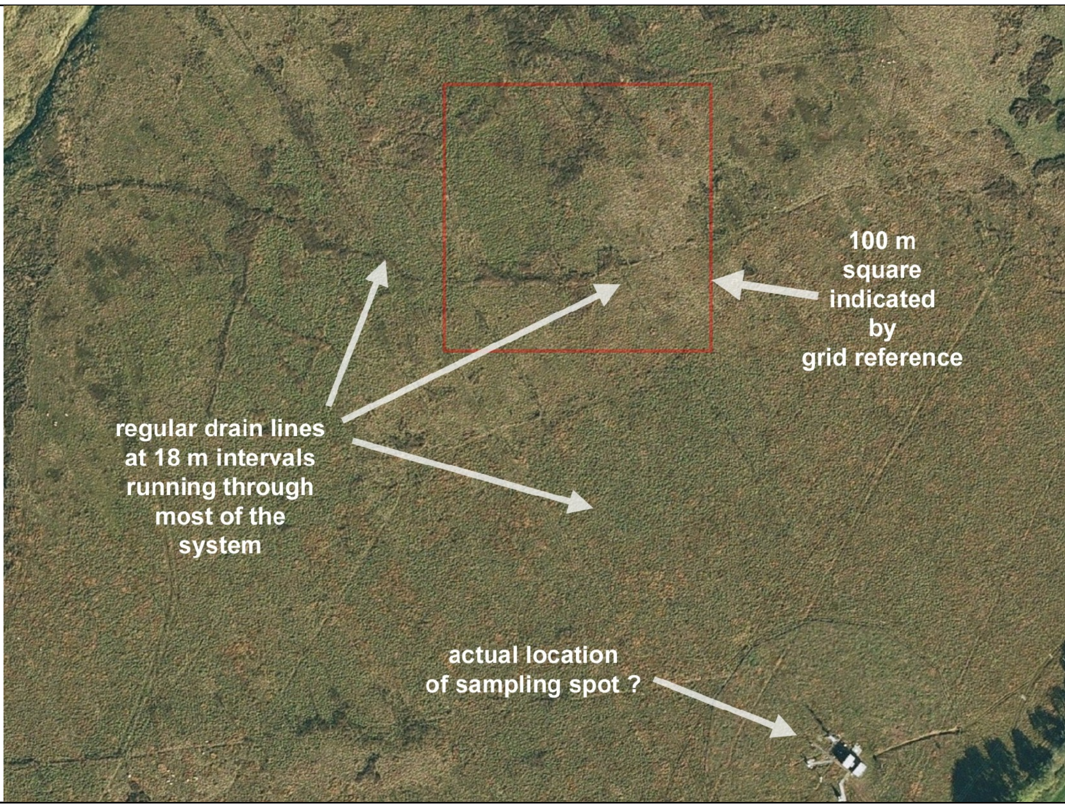


Figure 43. Aerial view of Auchencorth Moss, and possible location of instruments.
An aerial view of Auchencorth Moss at the grid reference given by Hargreaves *et al*. (2003). The actual grid reference location is bounded by a red box with sides of 100 m. Long-established, regularly-spaced drains can be seen cutting across the whole area. The objects to the bottom-right of the image may be the instrumentation for the study by Hargreaves *et al.* (2003). Unfortunately it is not certain that this is the case because no location information is given for the instrumentation by Hargreaves *et al.* (2003).

There are further concerns about the 26-year plantation Mindork Moss, which is described as "complete tree cover with little ground vegetation", and the plantation is described as a "mature spruce stand". This cannot be the case because the forest at Mindork Moss is less than half-way to its felling age of 65 years. The most substantial oxidative effects on the peat caused by 'mature' trees will not be felt for another 30 years at least. Indeed it is very clear that the forest cover of Mindork Moss is a very long way from being 'mature forest'.

Figure 44 is an aerial photograph of the grid reference given by Hargreaves *et al*. (2003). From Figure 44 it is evident that there has been no canopy closure for significant parts of the plantation. In parts there is no tree growth at all, leaving patches of remnant bog vegetation. This does not accord with the description of Mindork Moss given as “complete tree cover with little ground vegetation”. In addition, the uneven nature of the canopy for much of the plantation immediately to the south-east and east of the grid reference may well pose problems for eddy covariance measurements because the system assumes a relatively uniform surface. Hargreaves *et al*. (2003) describe the sites as being “reasonably uniform in most directions up to 1 km”. This does not seem to be entirely the case at Mindork Moss.

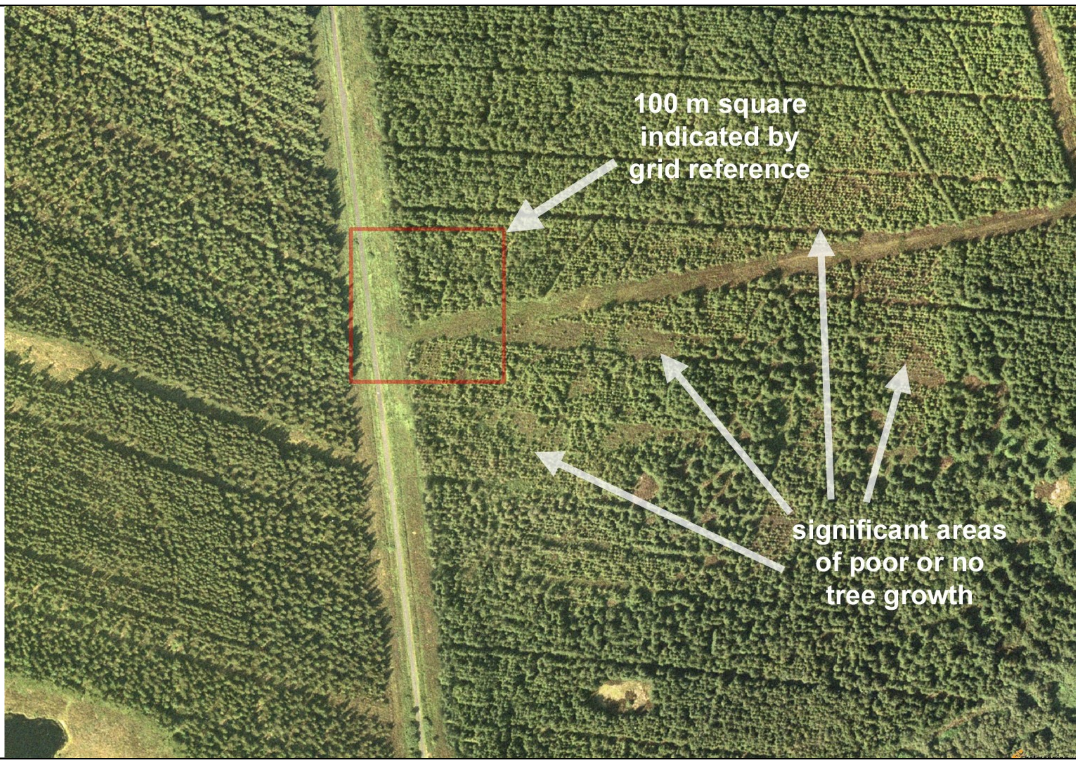


Figure 44. Aerial view of Mindork Moss, Newton Stewart.
An aerial view of the location given by Hargreaves *et al*. (2003) for Mindork Moss. The location of the grid reference is bounded by a red box with sides of 100 m. This may be the location of the instrumentation, but no information is given by Hargreaves *et al*. (2003). It can be seen that the forest to the south-east and east of the grid reference has not grown well, indeed has not grown at all in some places, leaving patches of ground covered by remnant mire vegetation. The uneven nature of the canopy may be a problem for eddy covariance readings.
Aerial photograph © Getmapping.com

12.3.3.2 Carbon accumulation model

The model constructed by Hargreaves *et al*. (2003) takes the rate of carbon accumulation at Auchencorth Moss as the starting point for the model, and then calculates the overall carbon gain or loss by the system as a whole. They also model the accumulation of carbon within the trees.

The diagrams used by Hargreaves *et al.* (2003) are quite difficult to interpret as presented. Part of the problem is that carbon gains are shown in the negative zone of the y-axis, while losses lie above the x-axis in the positive zone of the y-axis. This is somewhat counter-intuitive in terms of gains and losses and makes the graph less easy to read than it might be. Consequently Figure 45 displays the same data rotated through 180° so that the gains in carbon are displayed above the x-axis (intuitively positive accumulation) and losses are below (intuitively negative losses).

Hargreaves *et al*. (2003) and Figure 45 show a curve with a number of measured points and several interpolated points for the total net carbon balance of the system as a whole. Also displayed on the same graph are the modelled carbon accumulation curves for the carbon accumulating in the trees. If the trees accumulate *more* carbon than is actually recorded for the *system as a whole*, this means that some carbon must have been lost from the system as a whole (*i.e.* by oxidation of the peat store) to nullify some of the carbon gains made by the trees. Equally if the system as a whole records *more* carbon than is stored in the trees, this must be explained by continued accumulation of carbon as freshly accumulated peat adding to the quantity of carbon being stored in the trees.

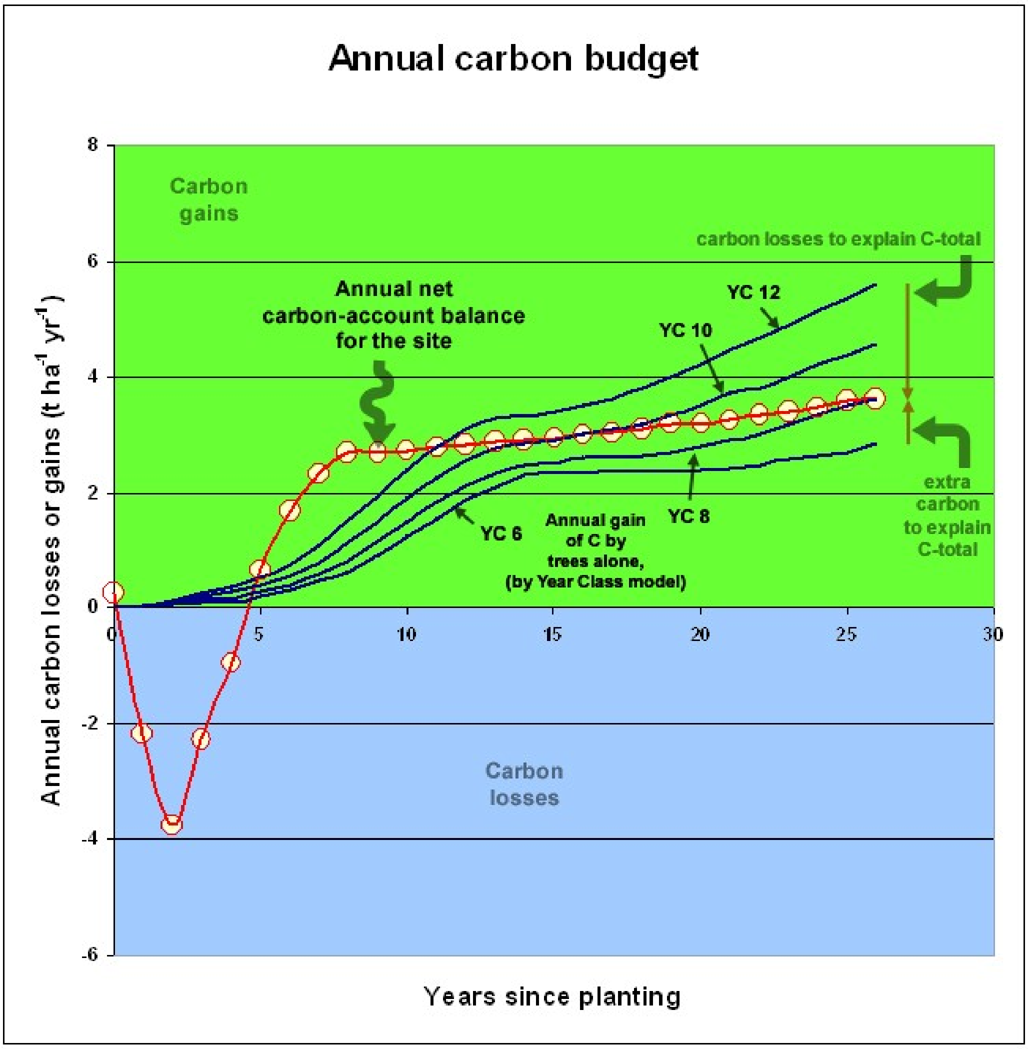


Figure 45. Carbon balance model constructed for 26 years of forestry on peat by Hargreaves *et al.* (2003). The net accumulated carbon for the afforested peatland site is shown by the red line. Dark blue lines represent modelled carbon-accumulation curves for the timber of conifer forests growing at differing Yield Classes. The pale blue lower zone of the graph is when the net accumulation of carbon within the site is negative (*i.e.* lost to the atmosphere), the green upper zone of the graph is where the net carbon accumulation of carbon on the site is positive (*i.e.* the site gains carbon). Where the red line lies below a modelled Yield Class line, the site is losing carbon through oxidation even though the trees may be gaining carbon, the difference between the lines showing how much carbon is lost. Where the red line is above a modelled Yield Class line, the site is gaining carbon faster than can be explained by the growing trees, and must therefore still be forming peat in remnant areas of vegetation.

Adapted from Hargreaves *et al.* (2003)

Thus the easiest way to interpret Figure 45 is firstly to recognise that when the red-circle-line lies in the blue-shaded sector the system is losing carbon, whereas when it lies in the green-shaded sector there is an overall carbon gain. However, if any part of the dark-blue lines (which represent *predicted* forest gains) lie *above* the overall system red-circle-line, then the system has failed to meet its predicted carbon accumulation rate, which indicates that there have been some unaccounted losses somewhere. These losses are caused by oxidation of the peat. If, on the other hand, any part of the blue lines lie *below* the red-circle-line, then there has been a *gain* of carbon to the system in excess of that predicted by the forest-carbon model. In other words, remnant bog vegetation has possibly continued to add carbon to the system *in addition* to carbon sequestered by the forest.

Looking firstly at the red-circle-line in isolation, it is clear that there is a period at the start of the plantation cycle where there is substantial loss of carbon as a result of ploughing and consequent drainage. By Year 5 the system is coming back to its 'natural' starting point, and then shows fairly steep increases in carbon accumulation up to Year 7. After that the net accumulation rate becomes much slower, rising from around 2.8 t C ha^{-1} yr^{-1} at Year 7 to 3.5 t C ha^{-1} yr^{-1} by Year 26. While these figures are much higher than the 0.23 t C ha^{-1} yr^{-1} transferred into the catotelm according to the 'standard cubic metre of natural peat' described earlier in the present report, the critical question, as identified by Cannell *et al*. (1993), is what level of oxidation is occurring to the long-term carbon store beneath the forest?

As observed above, when the red-circle-line of net carbon balance for the site lies *above* the blue line of carbon accumulation for any given Yield Class of tree, the bog must be adding carbon on top of that accumulated by the trees. Where the red-circle-line lies *below* the blue line for any given Yield Class of tree, the bog has lost carbon through oxidative wastage, the amount lost being the difference between the two lines at that point.

It can be seen that until around Year 10 the bog continues to add carbon in the form of shrubs and grasses (and possibly some surviving *Sphagnum*). After Year 10, Yield Class 12 trees begin to result in losses to the bog carbon store through oxidation. Yield Class 10 trees do the same after about Year 17, and Yield Class 6 reach this critical point at Year 26. Interestingly, Yield Class 6 continues to see the bog adding carbon to the store. This is probably because tree growth in Yield Class 6 is so poor that significant areas of bog vegetation still survive and continue to add small amounts of carbon to the bog system. Yield Class 6 is not a great contributor of tree carbon, however.

By comparing for any given year the four blue Yield Class curves with the red-circle-line of net carbon balance, it is possible to calculate the losses or gains of carbon within that year for a given Yield Class. Thus in Year 20, Yield Class 12 trees should have resulted in a total system gain of approximately 4.5 t C ha^{-1}. In fact the system only gained around 3.2 t C ha^{-1} in that year, and thus there must have been oxidative losses equivalent to 1.3 t C ha^{-1}.

Thus after 26 years, Yield Class 12 is causing losses of 2 t C ha^{-1} yr^{-1} (the difference at Year 26 between the system carbon balance and the carbon curve for Yield Class 12) and this quantity is clearly rising, and rising more steeply, each year. According to Cannell *et al*. (1993), 2 t C ha^{-1} yr^{-1} is equivalent to a peat thickness of 4 mm.

The values of carbon gain or loss in Hargreaves *et al*. (2003) of course only go as far as Year 26 and are based on a site where canopy closure has not yet been achieved on significant parts of the site. Thus the drainage effects on the peat associated with a closed canopy of mature trees cannot yet be seen in this experiment. It is possible, however, to take the data obtained so far and use the values of rates of loss already established to Year 26 to make some general predictions about the likely course of events up to harvesting at year 60. This prediction can be seen in Figure 46.

The two curves in Figure 46 beyond Year 26 are based on either a constantly increasing rate of loss, or a rate which shows exponential deceleration with time (*i.e.* the rate of loss declines with time). It might be argued that the straight-line increasing rate of loss is the more realistic, given the increasing capacity of maturing trees to intercept and transpire water. However a decelerating rate of loss is also offered here, with the observation that the true picture is likely to be somewhere between these two scenarios, within the yellow-shaded zone of Figure 46.

It is particularly interesting to note that the losses exceed 3 t C ha^{-1} yr^{-1} somewhere between Year 31 and Year 45. This value of 3 t C ha^{-1} yr^{-1} is equivalent to the worst-case 300 g C m^{-2} yr^{-1} scenario modelled by Cannell *et al.* (1993). Under the curve displayed for a steady increase in the rate of loss, this value then increases until it reaches 700 g C m^{-2} yr^{-1} by Year 60. Even under the conservative exponential curve, annual rates of loss greater than 300 g C m^{-2} yr^{-1} can be expected for more than 15 years. Given that Cannell *et al.* (1993) equate 500 g C m^{-2} to a 1 cm thickness of peat, the decelerating curve amounts to a total lost thickness of some 25 cm, while the curve showing a steady increase in the rate of loss amounts to some 35 cm of peat lost.

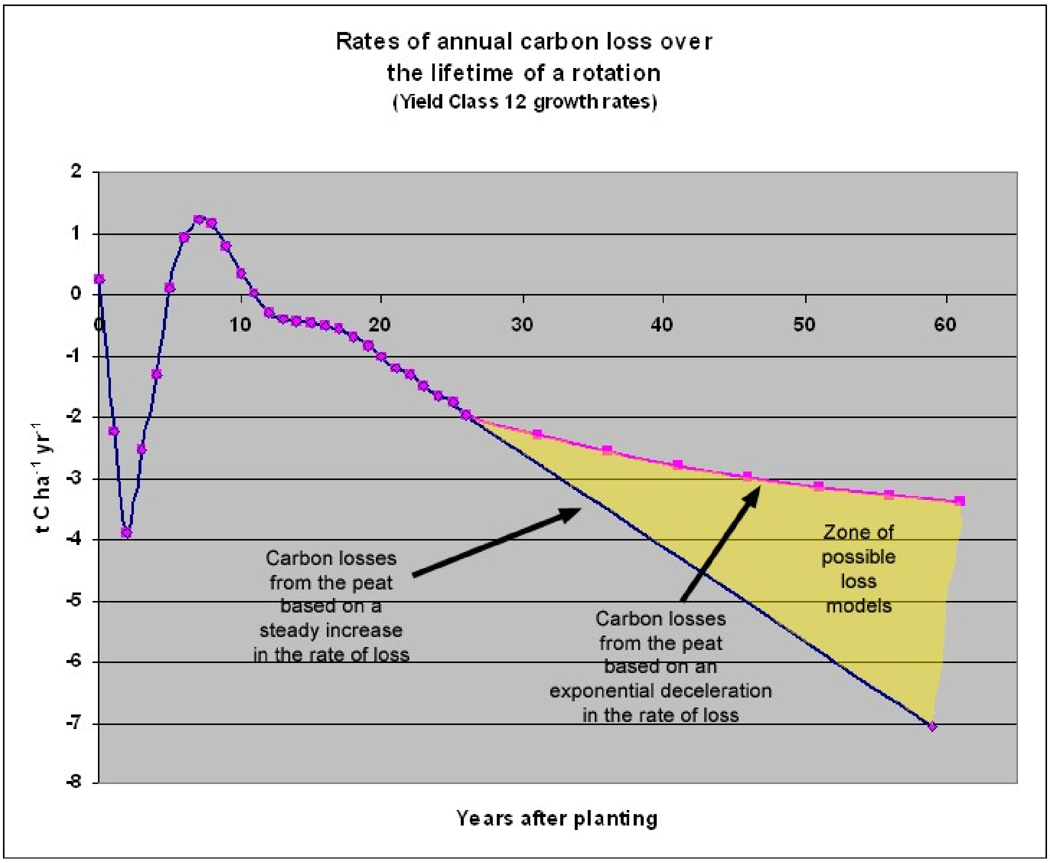


Figure 46. Carbon balance over 60 years for Yield Class 12 trees planted on peat.
The graph is initially based on the data provided by Hargreaves *et al.* (2003), calculating the difference between net carbon store of the system and the carbon store in Yield Class 12 trees, this difference representing loss of carbon from the peat. After Year 26, the rate of loss is described by two curves. The first is a constantly-increasing rate of loss for the remainder of the plantation rotation. The second curve describes a decelerating rate of loss based on an exponential decline in the rate of loss over time. The yellow zone between these two curves is the zone occupied by rates of loss between these two defined scenarios.

This is almost precisely the loss of peat thickness which Cannell *et al*. (1993) identify as resulting in no net benefit from the forestry. Their value does not take into account carbon losses as DOC and POC, which may reduce still further the thickness required to nullify any gains made by the conifer plantation. It would therefore seem possible that over a full rotation of 60 years, the losses of carbon from the peat may cancel out the gains made by the plantation forestry.

The total loss of thickness calculated with this model for the period covering only Year 0 to Year 26 amounts to 3.97 cm, which is actually not so different from the estimate of 1 cm loss calculated by Hargreaves *et al.* (2003). The critical point is that their calculation ends just as the trees really begin to affect the peat. Thus their subsequent calculations and statements do not reflect the likely scenario over a whole rotation.

One further point is that Hargreaves *et al*. (2003) assume that CH_4 emissions cease once the ground is ploughed and supporting tree growth. This may not be a valid assumption because methane can continue to be released from forest plantations on peat, particularly if the trees are not growing as well as they might (Minkkinen *et al*., 2007b). It can escape from within the peat via cracks, and any areas of poor tree growth may continue to support a modified bog vegetation which is often characteristically rich in tussocks of hare's-tail cotton grass (*Eriophorum vaginatum*) which can act as a very effective CH_4-transport system to the atmosphere when in wet hollows.

13 DISCUSSION TOPIC 3b Restoration of afforested peat – the carbon balance

13.1 Peatlands, forests and carbon stores

Given the various national and international obligations towards conserving and restoring peatlands, it is legitimate to ask about carbon benefit in transforming conifer forests on peat soils back to living blanket mire.

A cautionary note in relation to such thinking has already been sounded in this present report in terms of how much carbon may be stored in the living biomass of actively-growing bog systems. Indeed some fairly simple calculations are highly relevant to this question and prove to be quite revealing.

Milne & Brown (1997) give the total area of conifers in Britain as 1,364,600 ha. They review a range of estimates for the carbon content of these conifer plantations and finally settle on a value of 29 million tonnes of carbon. This equates to a carbon density of 21 t C ha^{-1}. In terms of plantations on peat, however, the main species are lodgepole pine and sitka spruce. The average carbon density for these two together, taking into account their relative area and age distribution in Britain at the time of Milne and Brown's survey, was estimated to be 10.3 t C ha^{-1}. Cannell *et al.* (1993) estimated that 190,000 ha of deep peat had been planted with conifers by 1980 (thus excluding the large expansion onto deep peat in the 1980s during the life of the Forestry Grant Scheme). Assuming that this 190,000 represents the area actually planted and that there are no unplantable gaps included within this, the area of lodgepole and sitka together holds 1,957,000 t C.

Let us assume that a *Sphagnum* carpet instead develops on an equivalent area of deep peat, and that this carpet has a bulk density of 0.03 g cm^{-3} (i.e. a fairly soft carpet). This *Sphagnum* carpet equates to 1.54 t C ha^{-1} per centimetre thickness. Across the whole area of 190,000 ha, this amounts to 307,800 t C per centimetre thickness. What thickness of *Sphagnum* is required to equal the carbon stored in the area of conifer plantations? It emerges that a layer of *Sphagnum* only 6.4 cm thick would be required to store an equivalent quantity of carbon. Such a thickness is probably close to the thickness of an initial colonising layer when *Sphagnum* species such as *S. papillosum*, *S. magellanicum* re-establishes themselves on a formerly-damaged peat surface. In other words, if a *Sphagnum* carpet were to be re-established across a total area of 190,000 ha across Britain, this would have the capacity to match the carbon stored in all the lodgepole pine and sitka spruce plantations growing on peat, as listed by Cannell *et al.* (1993) on the basis of plantings to around 1980.

However, the main expansion of forestry onto peat after 1980 was stimulated by the Forestry Grant Scheme and a high proportion of such planting occurred in the Flow Country of Caithness and Sutherland. Lindsay *et al.* (1988) highlight the fact that by 1987 a total of 67,000 ha of the original peatland area in the Flow Country had been planted with conifers. Assuming that this was all planted post-1980 (which it was not, as shown by the much older plantings at Bad á Cheo), this area can be added to the 190,000 ha given by Cannel *et al.* (1993). Assuming that a further 67,000 ha had been planted on peat during the 1980s and 1990s across the rest of Britain, this gives a final, almost certainly over-generous total of 324,000 ha for conifer planting on deep peat soils in Britain. At a carbon density of 10.3 t C ha^{-1}, this represents 3,337,200 t C contained within conifer plantations on deep peat in Britain.

What extra thickness of *Sphagnum* would be required to provide as much carbon as the estimated total area of conifer plantings on deep peat given above? A bulk density of 0.03 g cm^{-3} can again be assumed for the colonising *Sphagnum* carpet, again giving 1.54 t C ha^{-1} per centimetre thickness. If it is assumed that this carpet of *Sphagnum* is conservatively 3 cm thick, the 3 cm layer contains 4.62 t C ha^{-1}. This means that an area of recolonising *Sphagnum* amounting to only 722,338 ha would be sufficient to match the carbon currently stored in all afforested blanket mire in Britain. This is substantially less than the total target area already identified for blanket mire restoration (845,000 ha) given in the UK Biodiversity Habitat Action Plan.

The simple question of carbon storage is not the whole picture, of course. Wetter bogs have been widely reported to show higher emissions of CH_4 than drier sites. Re-wetting of formerly forested bogs, either by simply removing the trees or by ditch-blocking as well, clearly has the potential to increase methane emissions. In addition, decomposition of any woody material left on the bog surface is likely to give rise to increased emissions of CO_2.

The critical question here is whether the levels of methane and carbon dioxide released as a result of such restoration work mean that the carbon balance of the recovering bog is better or worse than in the bog before it was afforested, or than when it was in the afforested state.

The latter condition has been explored extensively in Section 12.3.3 of the present report, and is clearly time-dependent. The overall picture of what happens to the carbon balance, when a bog which has been forested is then subject to restoration management, remains to be investigated in detail. The one significant study relevant to the British blanket mire environment is that undertaken by Colls (2006). This work merits close examination.

13.2 Colls (2006)

This PhD involves the study of several ecosystems and the carbon fluxes associated with restoration of these habitats. The study includes a consideration of the carbon fluxes associated with the restoration of blanket mire from mixed lodgepole pine and sitka spruce plantation forestry which was planted on peat only 15 years ago.

Colls (2006) creates a decomposition model to investigate whether felling the trees and leaving them where they lie creates such a degree of CO_2 release that re-establishment of the peat-forming vegetation would not be able to compensate for these losses combined with resultant methane emissions if the bog is re-wetted.

The decomposition model derived by Colls (2006) is based on differing rates of decomposition for differing forest fractions (branches, needles, etc.) and for differing layers – the soil surface, the acrotelm and the catotelm. She notes that the forest decomposition model uses decay rates obtained from Palviainen *et al*. (cited as 2002 but in fact 2004), which is described as a study of forest decomposition on a peatland site. Close examination of Palviainen *et al*. (2004) reveals that in fact the site had only a very thin organic layer of a few centimetres, and is described by the authors as a haplic podzol rather than a peatland.

Colls (2006) cites Hargreaves *et al.* (2003), observing that they found much lower losses of peat carbon than expected beneath an afforested bog. The methodological questions concerning Hargreaves *et al.* (2003) have already been explored in some detail above. Colls (2006) also states that Hargreaves *et al.* (2003) conclude that forests on peatland in the UK have been net carbon sinks for the last 50 years. What they actually state is: "The conclusion *may be* that, over the past 50 years and throughout most of the 20th Century", UK forests on peat have been a net sink. This conclusion is based solely on their own study. However, Hargreaves *et al*. (2003) themselves admit, in relation to their estimate of only 1 cm loss over 26 years:

> *"...This estimate is the difference between two large uncertain numbers (the total net C exchange and net tree gain) and so is subject to considerable uncertainty."*
> *Hargreaves et al. (2003) p.313*

This important caveat is not reflected in the comments by Colls (2006) about the findings of Hargreaves *et al.* (2003), nor are the scales of carbon loss presented by, for example, Braekke (1987), considered as alternative evidence for the effects of forestry operations on peat.

Colls (2006), constructs three decomposition models – one for the bog surface, one for material in the acrotelm, and one for decomposition of tree material in the catotelm. Each of these models is then examined separately and rates of CO_2 emission are presented on the basis of all the material being on the surface, or all being in the acrotelm.

In doing so, Colls (2006) concludes that with a C-sink of 0.1 t C ha^{-1} yr^{-1}, it would take 73 years for the bog to sequester sufficient carbon to compensate for the carbon released by the decomposition of the felled trees. This rate of C-sink is low compared with the value (0.32 t C ha^{-1} yr^{-1}) for annual carbon transfer from acrotelm to catotelm given in the 'standard cubic metre of natural peat' developed in the present report. Colls (2006) also presents a model with a C-sink rate of 0.5 t C ha^{-1} yr^{-1}, and in this scenario she calculates that it would take 15 years for the bog to absorb sufficient carbon to compensate for the carbon released by decomposition of sitka spruce and 9 years for lodgepole pine.

The main problem with the model used by Colls (2006) is that material will not decompose in only one layer. A more realistic decomposition model would involve partitioning the decomposing material between the bog surface, the acrotelm and the catotelm.

The present report re-works the model developed by Colls (2006) to do precisely this. It uses the same decay rates (despite the doubt that some of these are for 'peatland'), and partitions material on the basis of:

- 75% decomposes on the surface;
- 24% decomposes in the acrotelm, and
- 1% decomposes in the catotelm.

This re-worked model produces the picture of decay and accumulation shown in Figure 47. The pale lines represent rates of net carbon accumulation under differing rates of carbon sequestration. The dark brown curve at the bottom of the graph shows the losses of carbon from forest decomposition. It can be seen from Figure 47 that if only the CO_2 losses from tree decomposition are considered, all but the lowest carbon accumulation rate exceed the losses at all stages. Only an accumulation rate of 0.1 t C ha^{-1} yr^{-1} fails to exceed CO_2 decomposition losses initially, but after 42 years it does then exceed those losses.

Essentially, the re-worked model shows that under all scenarios except the slowest peat-accumulation rate, the process of restoration makes net gains in CO_2-carbon from the very start, at Year 1, compared to estimates by Colls (2006) of, for example, 15 years for sitka spruce to decay on the surface at a peat accumulation rate of 0.5 t C ha^{-1} yr^{-1}. For the lowest rate of peat accumulation (0.1 t C ha^{-1} yr^{-1}), Colls (2006) calculates that the system would not be in net carbon gain for up to 79 years. This re-worked model indicates that using this slowest rate of carbon accumulation, net carbon gain over decomposition losses is achieved after about 42 years.

If the effect of methane is then considered, the picture changes significantly because of the strong GWP of methane. Colls (2006) does not model this. However, two curves (purple and orange) are provided in Figure 47 which represent the GWP of all carbon losses, the purple curve being the GWP on a 100-year timescale, the purple showing the 500-year timescale. The rate of CH_4 release (0.057 t CH_4 ha^{-1} yr^{-1}) used in this model is higher than the value mentioned by Colls (2006), who appears to use a rate of 0.052 t CH_4 ha^{-1} yr^{-1} (and gives this a GWP of 0.3 t C ha^{-1} yr^{-1} on a 100-year time-horizon). However, it is assumed that the forest growth is in a condition and a stage which is similar to the peatland forests described by Minkkinen *et al*. (2007b) from which 0.01 t CH_4 ha^{-1} yr^{-1} continue to be released, and thus the extra CH_4 released by re-wetting over-and-above this amounts to 0.47 t CH_4 ha^{-1} yr^{-1}.

It can be seen that the curve for the highest rate of peat accumulation exceeds even the GWP (100 year) curve, and thus the system would be greenhouse cooling despite its methane emissions. Even the next-highest rate of accumulation (0.4 t C ha^{-1} yr^{-1}) is close to matching the GWP (100 year) curve. Conversely, only the two lowest rates of carbon accumulation (0.1 and 0.2 t C ha^{-1} yr^{-1}) fail to exceed the orange 500-year GWP curve within a 50-year timeframe, although the 0.2 t C ha^{-1} yr^{-1} rate would exceed the GWP curve after about 53 years.

As more of the ground becomes dominated by *Sphagnum* at the restored site, it could be expected that CH_4 emission levels may fall significantly. On this basis it seems possible that the 100-year CO_2/CH_4 emission line in Figure 47 would begin to fall significantly after a period of some 10 years, as *Sphagnum* becomes increasingly established on the site. With a relatively modest reduction in CH_4 emissions, the red 100-year GWP time-line might be brought down to the level of the current 500-year GWP time-line shown in Figure 47 – possibly even lower.

One final point worth making in relation to the Colls (2006) study concerns the scale of recovery and *Sphagnum* growth necessary to compensate for the biomass carbon held within this mixed stand of trees. Earlier in the present report, it has been demonstrated that it is possible to calculate the thickness of peat necessary to match the amount of carbon stored within a conifer plantation. Using the

50:50 mix of sitka spruce and lodgepole pine described for the study site used by Colls (2006), it is possible to show the thickness of peat needed to match the forest biomass-carbon of the Colls study under scenarios for differing bulk densities of the peat.

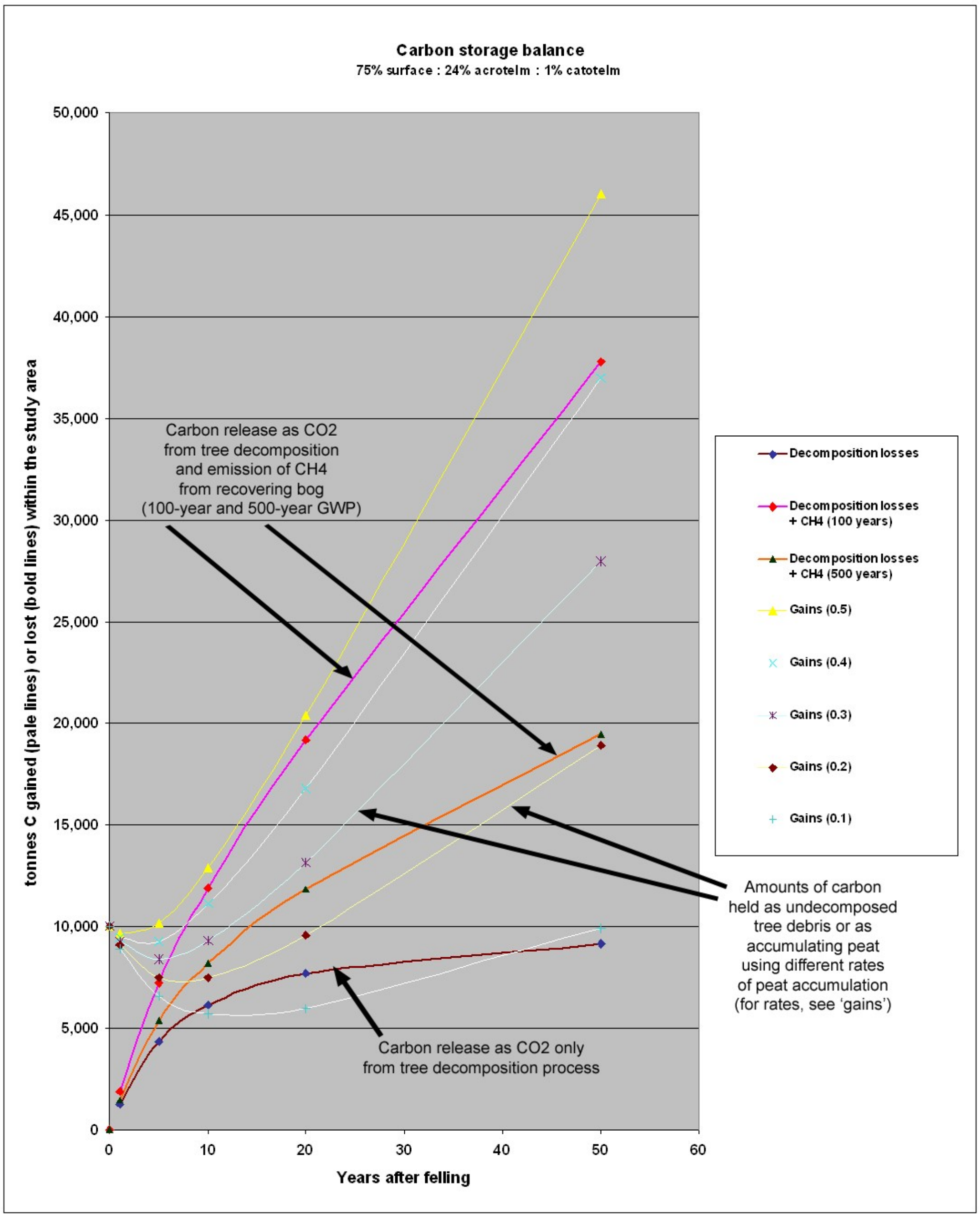


Figure 47. Carbon accumulation rates and carbon losses for 15-year felled forestry
A re-working of the decomposition model constructed by Colls (2006). In this model the decomposing tree material is partitioned between the bog surface, the acrotelm and the catotelm on the basis of 75%, 24% and 1% respectively. Pale-coloured lines (Gains) represent the quantity of undecomposed forest material held in the three layers, together with the accumulation of peat based on the 5 given rates of carbon accumulation (0.1 tC ha^{-1} yr^{-1} to 0.5 tC ha^{-1} yr^{-1}). Balanced against these gains are the indicated losses of carbon as CO_2 from decomposition, and the release of carbon as CH_4 from the bog surface. When a line of carbon loss is higher than any given line of carbon gain, the site is suffering a net loss of carbon. Two GWP timeframes are given (100 years and 500 years). Rates of CH_4 release are based on amount released by the bog (0.03 tC ha^{-1} yr^{-1}) in excess of that released by the poorly-performing plantation (0.01 tC ha^{-1} yr^{-1} : Minkkinen *et al*., 2007b). If *Sphagnum* colonisation does occur, CH_4 release may be expected to diminish significantly from the figures used to create these curves.

This model is shown in Figure 48, from which it can be seen that, even at the lowest bulk density of 0.03 g cm^{-3}, a thickness of only 4.5 cm of *Sphagnum* is required to match the carbon stored in the forest. At a bulk density of 0.05 g cm^{-3}, the thickness required is only 2.6 cm. It does not therefore require prodigious scales of recovery on the part of *Sphagnum* before the *Sphagnum* carpet has matched the carbon stored in the forest during its 15-year growth. Just the thinnest layer of a re-established *Sphagnum* carpet has the potential to match the accumulated carbon of the plantation. On the evidence from such studies as Frenzel and Karofeld (2000) and Bortoluzzi *et al*. (2006), such a carpet could have the added benefit of reducing emissions of CH_4 from the re-wetting peat. This is why *Sphagnum* may be such a potentially important factor in the restoration and maintenance of bog systems.

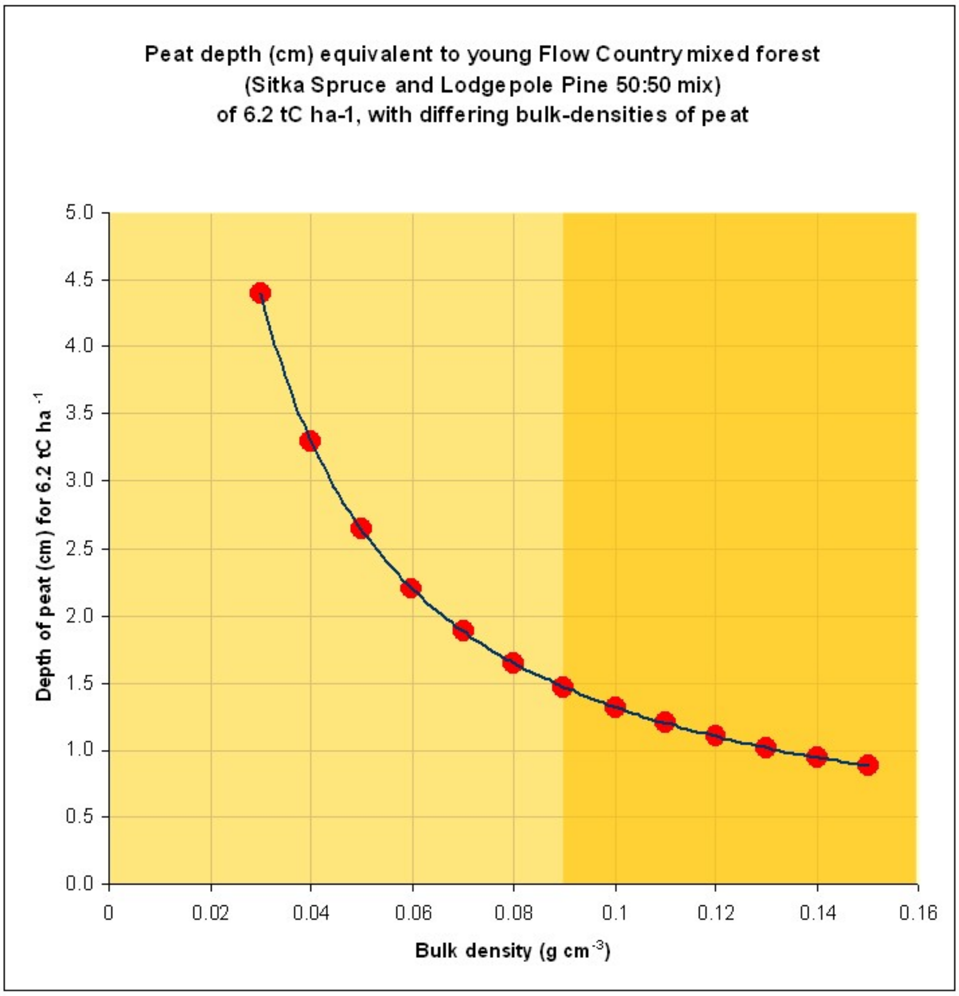


Figure 48. Thickness-equivalence of a *Sphagnum* carpet vs forest carbon biomass for a 15-year mixed forest.
This graph shows the thickness of *Sphagnum* necessary to match the carbon stored as biomass in a 15-year conifer plantation comprising a 50:50 mixture of sitka spruce and lodgepole pine. Differing values of bulk density for the *Sphagnum* are given, ranging from those typical of a very open, rapidly-growing *Sphagnum* carpet (0.03 g cm-3) to a denser *Sphagnum* typical of the centre of hummocks (0.09 g cm-3). The model continues on into bulk density values more typical of catotelm peat (this zone shaded dark orange). Thus it can be seen that a bulk density of 0.09 g cm-3 can match the carbon stored in the 15-year-old forest by generating a *Sphagnum* thickness of only 1.5 cm.

14 DISCUSSION TOPIC 4 Hydro-electric schemes in peat-covered catchments

Almost the whole of Scotland receives its drinking water from reservoirs lying within catchments which are dominated by peat soils. In addition, Scotland creates a larger proportion of renewable energy from hydro-power schemes than any other part of Britain. With government policies worldwide increasingly encouraging the development of renewable energy, there is likely to be growing interest in the potential for ever greater expansion of hydro-power because this has traditionally been seen as a relatively 'clean' energy source.

Closer examination of the carbon dynamics of reservoirs in recent years, however and particularly those which are created on peat or within peat-dominated catchments, has begun to reveal that reservoirs and hydro-power can have significant carbon consequences..

In terms of carbon stores and balances, there are four major issues in relation to reservoirs created on or within peat-dominated landscapes. Two of these issues are common to all reservoirs associated with peat whether in the Amazon Basin, the Canadian muskeg, or an upland valley in Britain, while two are more particularly associated with the British scene.

These issues are:

- gaseous release from flooded vegetation and peat soils;
- gaseous release from peat washed into the reservoir;
- reduced storage capacity and turbine blockage.

14.1 Gaseous release from flooded vegetation and soils

In recent years there has been growing recognition that where hydro-power schemes are associated with peatland systems that there are various pathways by which gaseous carbon may be released from reservoir surfaces. The problem arises when carbon-rich peat material becomes inundated, either when the reservoir is constructed, or when peat material is eroded and washed down into the reservoir system.

Construction of new reservoirs, or expansion of existing reservoirs, gives rise to gaseous-carbon release when areas of peatland habitat are flooded and the carbon-rich vegetation and peat soils then begin to decompose, often under anaerobic conditions. This is most often an issue when the area to be flooded is relatively level terrain rich in peatland systems associated with the river or stream system which will be dammed.

14.1.1 Kelly *et al.* (1997) : Canadian reservoir experiment

Kelly *et al*. (1997) studied the carbon balance of the flooding process in the Canadian muskeg peatland (relatively thin peatland fed for part of the year by snow-melt). They found that whereas in the natural state the wetland appeared to be a small net carbon sink (-6.6 g C m^{-2} yr^{-1}), once the area was flooded it became a substantial carbon source (+130 g C m^{-2} yr^{-1}). The entire wetland area emitted much more CO_2 than before (from -4.5 g CO_2-C m^{-2} yr^{-1} to +120 g CO_2-C m^{-2} yr^{-1}) as the former peatland vegetation and peat soil decomposed, while methane emissions rose from +0.7 g CH_4 m^{-2} yr^{-1} to +11.9 g CH_4 m^{-2} yr^{-1}. The most marked increase in CH_4 emission rates occurred within the area of flooded peatland, with values rising 60x, from 0.17 g CH_4 m^{-2} yr^{-1} to 10.5 g CH_4 m^{-2} yr^{-1}. In absolute terms, however, the area of open water represented by the original pond on the site changed from an emitter of 3.7 g CH_4 m^{-2} yr^{-1} to 20 g CH_4 m^{-2} yr^{-1}.

Kelly *et al.* (1997) noted that the peatland area in particular had lost its carbon sequestering ability and that the former peat land vegetation and soils were undergoing more rapid decomposition than before, but they also highlighted the fact there was a substantial increase in anaerobic conditions caused by the flooding. Under natural conditions the peatland vegetation itself had provided an oxidation pathway for much methane produced within the peat, meaning that relatively little methane escaped to the atmosphere. Once the area was flooded this oxidation pathway no longer existed and, although some oxidation occurred within the water column, much more CH_4 was rising to the water surface un-oxidised before release into the atmosphere.

The flooding of the former peatland area resulted in substantially increased emissions of both CO_2 and CH_4. The increase in the latter was caused by the inundation and consequent destruction of methane-oxidising layers in the peatland vegetation. With a Greenhouse Warming Potential (GWP = 100 yrs) of 25 for CH_4, this extra methane emitted from the former peatland area amounts to 500 g CO_2 equivalents m^{-2} yr^{-1}. Given the much greater GWP of methane, this increased release of CH_4 represents an impact more than 100% greater again than the impact of the increased CO_2 emissions from the peatland. The same basic pattern also applies to the area of the former pond.

Kelly *et al.* (1997) recommend two basic principles for the minimisation of carbon release associated with reservoir construction, namely, keep the area of inundated land to a minimum, and flood as little peatland habitat as possible.

14.2 Gaseous carbon balance for reservoirs within a peat-dominated catchment

Many reservoirs in Britain are long-established and thus questions of what happens when a new reservoir is created on peat-dominated ground have little relevance to such reservoirs. Almost all reservoirs, old or new, in the north and west of Britain are, however, constructed within catchments dominated by peatland soils. The consequences of this for the long-term carbon balance of such reservoirs can be considerable.

14.2.1 Huttunen *et al.* (2003) : reservoirs in Finland

Huttunen *et al.* (2003) investigated the gaseous carbon balance of several lakes and two reservoirs in Finland, the reservoirs having been created on a mixture of peatland and forest more than 30 years ago. They found that all water bodies emitted CO_2 and CH_4, but the reservoirs had the highest average CO_2 emissions, whereas methane emissions were linked to the trophic status of the water body – high levels of nitrogen and phosphorus were associated with high CH_4 emission levels.

Reservoir Lokka involved the flooding of ground which was 75% peatland, 25% upland forest, while Reservoir Porttipahta inundated land in which peatlands covered only 45% of the area. The methane emissions from Reservoir Lokka were consistently higher than those from Reservoir Porttipahta, and displayed a range which far exceeded that shown by Reservoir Porttipahta. Huttunen *et al.* (2003) noted that release of CH_4 in bubble form (ebullition) was a significant source of CH_4 from Reservoir Lokka whereas ebullition is not associated with CO_2 because it dissolves in the water column. When released as bubbles, CH_4 largely by-passes the oxidation processes in the bottom sediments and water column and is thus released as CH_4 to the atmosphere. Such ebullition is episodic and can thus be difficult to measure accurately.

Huttunen *et al.* (2003) observed that both CH_4 and CO_2 production were dependent upon the supply of fresh organic matter. In an earlier study, Huttunen *et al*. (2002) had previously identified that much of the carbon in CH_4 emitted from Reservoir Lokka was quite young. They suggested that most of this CH_4 was being produced by the anaerobic decomposition of flooded vegetation and primary productivity in the water column, rather than by decomposition of the older carbon-rich peat-soil deposits. Huttunen *et al*. (2003), in contrast, observed that CO_2 production appeared to be more closely associated with a constantly renewed supply of organic matter washed into the reservoir from the surrounding catchment (allochthanous material). In both cases, this meant that levels of CH_4 and CO_2 were still high some decades after the reservoirs were originally created and there was no sign that these levels would decline at any stage in the foreseeable future.

The shallow-water zone (littoral zone) of lakes and reservoirs was identified by Huttunen *et al*. (2003) as making one of the most important contributions to CH_4 balance in such systems, the more extensive this zone the higher the CH_4 emissions are likely to be. This accords with the findings of Kelly *et al.* (1997).

Huttunen *et al*. (2003) concluded that CO_2 emissions were highest from lakes and reservoirs which contained a high proportion of peatland soils within their catchment, whereas shallow-water zones within a lake or reservoir were important for CH_4 emissions. Flooded peat soils themselves made only a small contribution to carbon emissions, although they obviously had the potential to maintain such emissions over a long period of time.

14.2.2 Reservoirs in the British context

Within the British context, as already mentioned above, reservoirs have tended to be based on the principle of 'narrow and deep' rather than 'shallow and wide'. Consequently the proportion of peatland habitat lost to reservoir construction has generally been rather limited. It can therefore be assumed that CH_4 and CO_2 emissions from the flooding of such habitat in Britain are equally limited.

In contrast, when there is any peatland habitat within the catchment the steady accumulation of fresh organic matter into British reservoirs is almost guaranteed, given the degree of damage and erosion which is typical of the British blanket mire landscape. This steady supply of organic matter seems likely to ensure that reservoirs in peatland catchments will, if they have anaerobic conditions in the bottom sediments, release significant quantities of CH_4 either through diffusion or ebullition, and do so for the foreseeable future. Equally, if conditions are oxic throughout the water column and in the bottom sediments, reservoirs in peatland catchments are likely to be significant sources of CO_2 emissions, also for the foreseeable future.

14.2.3 Global concern about reservoirs and peat soils

An increasing number of large hydro-power schemes have been proposed in recent years for various parts of the world. These schemes have tended to focus on the flooding of areas which are considered to have low economic value and low population densities (and which are thus very often extensive peatland systems). The 'greenhouse-beneficial' nature of such schemes has increasingly come to be questioned and has led to growing debate about many of the issues raised by researchers such as Kelly *et al*. (1997) and Huttunen *et al*. (2003).

Articles taking a global perspective, such as Graham-Rowe (2005), highlight the fact that substantial international debate is now taking place about hydro-power and its greenhouse status. For example the 2nd Workshop on Greenhouse Status of Freshwater Reservoirs, held in Brazil in 2007, identified a number of aspects which remain poorly-studied and understood (see Reservoir Workshop website). Many of these areas of uncertainty would have relevance to the carbon budgets of a reservoir constructed on peat, or in a peatland catchment, within Britain.

14.3 Sedimentation rates and loss of reservoir capacity

A certain amount of contradictory and rather confusing evidence exists regarding the rates at which reservoirs in peat-dominated catchments are losing capacity because peat is being washed into the reservoirs from the catchment.

For example, various references are made to White *et al.* (1997) in the DEFRA project report "Research on the quantification and causes of upland erosion" (DEFRA Science & Research Projects website – soil erosion), citing the fact that peatland erosion has cost Yorkshire Water £74 million because new reservoirs have been needed to compensate for the loss of capacity in existing reservoirs.

White *et al.* (1997) make no reference to such a sum and indicate instead that capacity at Strines Reservoir, in the southern Pennines, has lost little more than 10% of its capacity since 1869. Such a long-term sedimentation rate (116 t km^{-2} yr^{-1}) is described as rather high compared to most other UK upland reservoirs but rather low compared to sedimentation rates observed in the Pennines (125 t km^{-2} yr^{-1}).

The shorter-term sedimentation rate for Strines Reservoir measured by White *et al.* (1997) for the period spanning 1956 to the present, however, indicates a substantial increase in sedimentation during the last 50 years. This increase amounts to 220 t km^{-2} yr^{-1} compared with 70 t km^{-2} yr^{-1} for the period 1869-1956 and is attributed to increased incision of peatland erosion gullies in the catchment. Thus long-term sedimentation rates give a different picture from that gained by looking only at the recent past.

The financial costs to Yorkshire Water resulting from sedimentation and loss of capacity in fact come from White *et al.* (1996), and perhaps reflects the impacts of recent sedimentation rates rather than long-term rates. More recently, Yeloff *et al.* (2005) have examined sedimentation rates in March Haigh Reservoir in West Yorkshire and found sedimentation rates as low as 2-28 t km^{-2} yr^{-1}. These rates indicated a marked increase from the early 1960s but then showed a peak between the mid-1970s and mid-1980s. Since then, rates have declined.

In fact as White *et al.* (1997) observe, the estimation of sedimentation rates in reservoir systems is a difficult process with many potential sources of error. Thus the results may well rely on the accuracy of original topographic surveys carried out at the time of reservoir construction, or the original mapping of soils within the area to be flooded. A project for Yorkshire Water carried out by the University of Leeds (see Bathymetry – University of Leeds website) highlights the fact that most water companies have relatively little information about sedimentation rates, and describes how multi-beam sonar will be combined with previous surveys carried out on Gouthwaite reservoir, North Yorkshire, to provide up-to-date-estimates of sedimentation rates for Yorkshire Water.

Such work will, of course, for the present rely heavily on the accuracy of the earlier studies, but as more sonar scans are taken over time, this approach may provide a good picture of current sedimentation rates. However, it seems clear that for the moment the question of sedimentation rates and loss of reservoir capacity is an issue of considerable concern to the water utility companies but one which has a relatively limited basis in terms of robustly-measured data.

15 DISCUSSION TOPIC 5 Peat bogs and climate change

15.1 Peat bogs – responsive systems to climate change

Peat bogs are in many ways more closely linked to issues of climate change than any other habitat in Britain. Indeed peat bogs provided some of the earliest clues to the fact that substantially differing climates had prevailed across Britain and Ireland not only over periods of geological time but also in the relatively recent past. The archives of preserved plant material dating back over the last 9,000 years allowed a picture to be built up of the changing vegetation forming the bog over time, with layers of pine stumps within the peat being taken as indications of drier climatic conditions in the past (Geikie, 1866; Lewis, 1905, 1906, 1907).

Of wider significance, however, was the collection of pollen which was also preserved within the peat. This pollen came not just from species growing on the bog itself, but from plant species growing in the wider landscape around the bog, in some instances such pollen having travelled very considerable distances. The richness of information provided by such pollen records gave rise to a new science – palynology (the study of the pollen archive) - and it was largely from this new science that a picture of Britain's, indeed Europe's, changing vegetation and climate for the last 12,000 years was assembled (Godwin, 1975).

It is thus worth highlighting at the outset of this section that the peat bogs of Britain have demonstrably developed, grown and survived over periods of anything between 2,000 years and 9,000 years despite a series of fairly major and occasionally rather dramatic climatic shifts. They have done so because natural peat bogs are highly responsive systems. The living vegetation provides a biological response to climate change. It is possible to point to the evidence of Barber (1981) and Belyea and Malmer (2004) for this, as well as to the homoeostatic mechanisms described by Goode (1973), Ivanov (1981) and Barber (1981).

The drainage sequence described by Woike and Schmatzler (1980) has already been discussed and illustrated in Section 9.1.5, Discussion Topic 1, but it is worth reproducing their illustration again here as Figure 49 because it also illustrates the homoeostatic nature of bog vegetation to climate change. The sequence in Figure 49 shows a bog surface which initially consists of hummocks, ridges and hollows. If the climate becomes warmer <u>and</u> rainfall patterns result in a drop in water table (note: temperature and low rainfall are not necessarily tightly coupled), the vegetation and microtopography become dominated by peat-forming *Sphagnum* species of the hummock and high ridge zone only, the hollow species being lost.

If conditions become even drier, the moss layer may consist entirely of hummock species. Finally, if conditions become drier still, the moss layer may be lost entirely and the vegetation consist only of dwarf shrubs and other vascular plants. This is thought to have been the case where one or more 'stillstand' phases have been found in the peat of some bogs, particularly in northern Germany, and also where pine and birch stumps can be seen within the peat of many blanket bogs in Britain, and which so interested Geikie (1866) and Lewis (1905, 1906, 1907).

During wetter phases in the climate, instead of the left-to-right drying sequence described above, the trend in Figure 49 would be from right-to-left. Thus the wetter elements of the microtopography expand, with the result that hummock and high-ridge (T3 and T2) species are steadily replaced by low-ridge and hollow (T1 and A1) species.

15.2 Climate-change predictions and peat bog systems

The latest climate models for the UK have been assembled by the UK Climate Impacts Programme, and are presented via their website as a series of three scenarios based on differing levels of greenhouse gas emissions – low, medium and high emissions. Predictions are given for the 2020s, the 2050s and the 2080s (UK Climate Impacts Programme UKCP09 website). The models include mapped predictions of % change for factors such as temperature, maximum summer temperature, rainfall, humidity and winter mean precipitation.

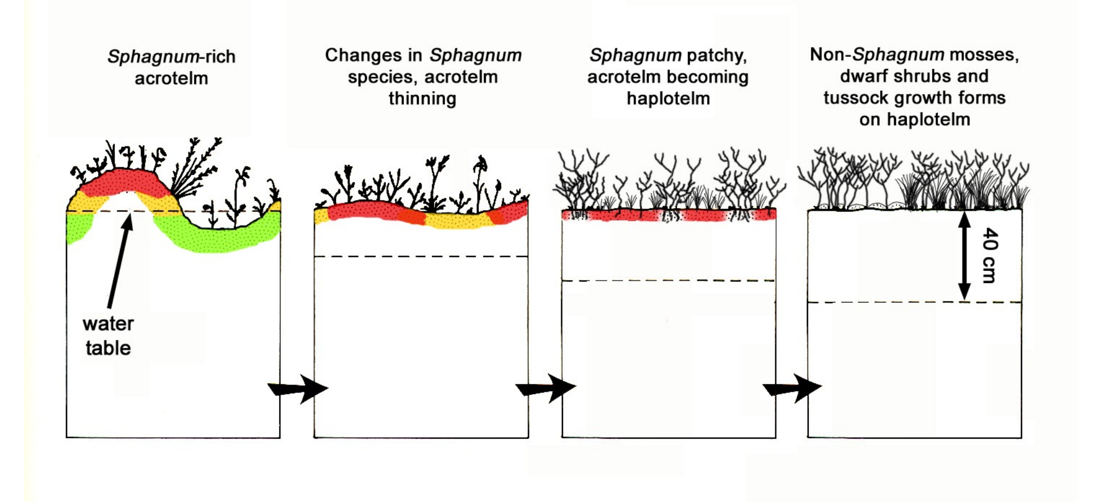


Figure 49. Various stages of change resulting from progressive climate-warming in blanket mire habitat. The effect of steadily rising temperatures and falling water table on the microtopography and vegetation of a blanket mire is likely to occur as a continuum of change rather than as distinct stages, but the four stages illustrated here show the key elements of change. Firstly the microtopography becomes simplified with the loss of A1 *Sphagnum* hollows and the species associated with these features are lost too. The bog surface instead becomes dominated by high-ridge and hummock-forming *Sphagnum* species which are vigorous formers of peat, while vascular plants become more numerous. If temperatures continue to rise <u>and</u> water levels fall, the *Sphagnum* carpet eventually becomes reduced to a smooth monoculture of *S. capillifolium* with patches of bare ground, beneath a sward of dwarf shrubs whose root systems introduce provide fresh carbon to the catotelm, leading to loss of stored carbon. Finally the *Sphagnum* carpet is lost altogether, creating a haplotelmic surface which may support non-*Sphagnum* mosses together with dwarf shrubs or tussock-forming species.

Adapted from Woike and Schmatzler (1980)

It might be expected that the various maps would show a general trend in effects from the south/south east up to the north of Britain, with the most intense % changes in warming and drying being felt in the south. In fact this is not inevitably the case. For example, under a high-emissions scenario in the 2080s, the warmest day of summer is predicted to increase by at least 2°C in London but by at least 4°C in Fort William. Under a similar scenario for the 2080s, London is very unlikely to experience an *increase* in summer precipitation of more than 10%, while central Sutherland is very unlikely to experience a *reduction* of more than 10% in summer precipitation. Looking at change in mean winter precipitation, the high-emissions scenario for the 2050s indicates that precipitation increases are unlikely to be less than +10% for the blanket mire areas of Britain, but are unlikely to be less than +20% for London.

Notwithstanding these slightly counter-intuitive results, the overall predictions for UK climate change summarised by Hulme *et al*. (2002) as increased generally-rising temperatures, summer and autumn drought, year-round increase in rainfall intensity, an extended growing season and reduced frequency of frost, are largely confirmed by the latest UKCP09 models. These trends have been linked to concerns about potential viability and increased degradation of (particularly) the blanket mire landscapes of the UK. Early global peatland concerns were raised by Gorham (1991), who identified the potential for rising temperatures to increase respiration losses faster than photosynthetic capture of carbon, particularly under conditions of falling water tables, as well as the release of CH_4 from formerly-frozen permafrost peats, an increase in woody vegetation, and an increased risk of fire on drier peatlands.

Specific concerns about the possible impacts on UK blanket mires have since cited increased desiccation of the peat due to increased summer drought, greater loss of material through erosion as a result, particularly if there is increased storminess, increased incidences of fire (Holden *et al*., 2007a), increased dangers of peatslides (Evans and Warburton, 2007), increased levels of DOC (Freeman *et* *al*., 2001;   Worrall *et al*., 2004), reduced rates of carbon storage, increased methane release and shifts in vegetation patterns (Heathwaite, 1993).

These are undoubtedly all responses which might occur across many of Britain's blanket mires under conditions of greater rising temperatures, summer drought and generally increased storminess, because the majority of blanket mire systems in Britain are haplotelmic and therefore do no have a vegetation cover which is capable of responding in the manner shown in Figure 49. Indeed McCarthy *et al*. (2001) discuss models developed by Hilbert *et al*. (1998) which suggest that bog systems "will adjust relatively quickly to perturbations in moisture storage". These models, however, assume that a living acrotelm is present.

It would thus appear that an urgent priority should be actions devoted to the re-establishment of acrotelm conditions on as wide a range of blanket mire systems as possible if the worst potential effects of climate change are to be avoided. That said, to what extent would an acrotelm help under predicted climate conditions? Would it not merely pass from left-to-right through the sequence shown in Figure 49 and lead ultimately to the same haplotelmic conditions which currently prevail?

15.3 Blanket mire and climate

Lindsay *et al*. (1988) present a rather simple qualitative 'climate envelope' for blanket mire formation and maintenance, based on a number of parameters including precipitation, number of rain days, humidity and cloud cover. Interestingly, however, the parameter which is most generally cited in the literature concerned with bog growth, development and distribution is precipitation. Thus Berry and Butt (2002) describe Thorne and Hatfield Moors as raised bogs which are "marginal to bog formation" because they lie in an area with rainfall of less than 600 mm yr^{-1}, while Holden, Chapman and Labadz (2004) describe the same sites as "close to the threshold of rainfall required for *Sphagnum* growth". It is worth pointing out that the raised bog at Holme Fen, Cambridgeshire, despite being located in an area of even lower rainfall than Thorne and Hatfield Moors, was a vigorously-growing *Sphagnum*-rich bog until the adjacent Whittlesea Mere was drained in 1855.

While rainfall and days on which rain falls are both undoubtedly important factors in the development and maintenance of bog systems, the other parameters making up the 'climate envelope' for the Lindsay *et al*. (1988) model are often overlooked but may be of much greater significance than is generally realised. In particular, local atmospheric humidity, cloud cover and mist or fog may play a much-undervalued part in maintaining *Sphagnum*-rich systems.

15.3.1 'Occult precipitation' and *Sphagnum*

Sphagnum has no cuticle, and can therefore absorb moisture directly through its leaf surface. Thus even dry cloudless weather does not necessarily mean that *Sphagnum* will dry out. This is because hot dry weather is often accompanied by heavy falls of dew and mists during the night. *Sphagnum* is able to take advantage of such 'occult precipitation' to replenish its physiological water.

An idea of the potential volumes of moisture available to *Sphagnum* in this way can be obtained from Figure 50, which shows the quantity of cloud moisture which has condensed on vegetation during an episode of thick mist in the central Pennines. Such volumes would not feature in measurements of rainfall, or even in the number of rain days.

The significance of such inputs for the water balance of *Sphagnum* has yet to be established for British bog systems. However, there is evidence that such water is readily taken up. Indeed this was one of the problems when *Sphagnum* was subjected to atmospheric pollution in the Pennines in former years – it scavenged occult precipitation very efficiently. Unfortunately pollution levels also become concentrated within such occult droplets and so the *Sphagnum* cover of the southern Pennines was particularly hard-hit by this process. Since atmospheric pollution levels in the Pennine region have diminished, this particular constraint to *Sphagnum* growth may now be much reduced.

15.3.2 Cloud cover

Solar radiation is reduced by approximately 25% even when it passes through a clear sky, but cloud cover can dramatically reduce the level of irradiance, depending on the type of clouds involved. At noon, cirrus can reduce the total received solar radiation to 60% transmitted light, whereas at lower solar altitudes this can fall as low as 40%. Even at noon, a thick layer of nimbostratus can reduce the quantity of transmitted light to 10% of total solar radiation, while at lower solar angles this can be reduced to less than 5% of received solar radiation (Gates, 1980, p.124).

Thicker and more regular the cloud cover, while lowering photosynthetic rates, also reduces the rate of water loss through evapotranspiration which, in *Sphagnum*, is an important consideration because, as discussed above, the *Sphagnum* leaf has no cuticle. Thus while *Sphagnum* may be able to take up occult precipitation at night and in the early morning as a result of dew fall, during a sunny day *Sphagnum* has only limited means of controlling water loss.

Figure 50. 'Occult precipitation' (mist/fog) collecting on vegetation in the central Pennines.
Mist collecting on the cuticle of various vascular plants within Pennine blanket bog. Lacking a cuticle, *Sphagnum* is in contrast capable of taking up this moisture directly into its hyaline cells through open pores. The significance of such occult inputs, over-and-above measured rainfall, has been relatively little-studied in terms of *Sphagnum* water balance.

Photo R A Lindsay

Cloud cover not only provides the potential for direct water inputs when the cloud descends over the blanket mire landscape. By attenuating solar radiation to greater or lesser extents the cloud cover also reduces the potential rates of water loss during daylight hours. It is worth noting the observation by Goudie and Brunsden (1994) that Great Dun Fell, in the Pennines, sits in fog for 2/3 of the year.

Warmer predicted temperatures may be relevant here because warmer air can hold more moisture. When this rises it meets cooler air and tends to form clouds. The regular noon-day deluge of the tropics is ample evidence of that process, but potentially just as significant for the continued health of bog systems is probably the amount of cloud cover. Unfortunately climate models are notoriously poor at modelling cloud cover.

Interestingly, the UKCR09 model provides predictions for summer and winter cloud cover, and the high-emissions scenario for the 2050s suggests that areas south of Britain's blanket mires will experience a reduction in cloud cover not *less than* 10-20%, whereas the blanket mire areas of Britain will experience a reduction of not less than 0-10%. Alternatively, under different probability settings, the 2050s are described as being very unlikely to experience an *increase* in cloud cover of *more than* 10% across the whole of Britain, while much the same is predicted for Scotland in the 2080s. The Cairngorms, southern Scotland and all of England and Wales are, in contrast, described as very unlikely to experience a decrease in cloud cover of more than 10%.

Overall, then, the worst-case UKCR09 models do not suggest a substantial reduction in cloud cover, and there may even be increases in cover. If cloud cover does increase, evapotranspiration is substantially reduced and there is also the potential for increased rainfall. Rising temperatures and increased cloud cover therefore have the potential to re-create the warmer but wetter conditions of the Atlantic period some 6,000 years ago in Britain. This was a period of vigorous bog growth.

15.3.3 Atmospheric humidity

The UKCR09 models suggest that under the worst-case, high emissions scenario, atmospheric humidity is likely to diminish by at least 5% across all blanket mire areas during the 2050s, and by at least 10% across the English and Welsh blanket mires by the 2080s. This represents a relatively modest change, but one which is predicted even under the low-emissions scenario. Changes in cloud cover could offset this small reduction in humidity.

15.3.4 Increasing 'oceanicity'

Crawford (1997) discusses the ecological implications of increased oceanicity in Britain, defined on the basis of increasingly warm winters. He considers that such a trend is likely to encourage blanket bog growth, and discusses the implications of similar trends in oceanicity on the possible expansion of bog systems at the expense of boreal forests around the Russian Arctic ocean.

It is therefore of interest to note that evidence is subsequently put forward by Crawford, Jeffree and Rees (2003) to suggest that in oceanic areas a period of invigorated peat growth does indeed appear to be occurring. They describe evidence for a southward retreat of the boreal tree-line in the face of increased development of bogs and wet heath around the Arctic Ocean. They attribute this expansion of paludified habitats to a period of warmer winters and cool, rather wet summers – a pattern, it is worth noting, which is not dissimilar from recent years in Britain.

15.4 Species 'climate envelope'

While the response of the habitat as a whole can be considered in terms of predicted climate trends, it is perhaps somewhat easier to consider the responses of individual species. Berry and Butt (2002) do this for a range of species which occur at Thorne and Hatfield Moors, both raised bogs on the flood-plain of the Humber Estuary. Berry and Butt (2002) create the climate envelopes of several bog species under high- and low-emission scenarios for the 2020s and 2050s, based on the UKCIP98 climate-change predictions.

Of particular interest is the climate envelope created by Berry and Butt (2002) for *Sphagnum papillosum*. This species is now one of the main peat-forming species in British blanket bogs. The current distribution of the species is mapped by Berry and Butt (2002), and then various scenarios are run on the climate envelope constructed for the species.

What emerges from this analysis is that even under the worst-case scenario, *Sphagnum papillosum* would still be capable of forming peat virtually to the south coast of England. Under the 2050s low scenario, *S. papillosum* would be capable of forming peat anywhere in Britain. This matches with the observation made earlier about active bog being present at Holme Fen until human drainage impacts in 1852 reduced the bog to a birch woodland. More importantly for the blanket bogs of Britain, it would seem that this key peat-forming species would remain ecologically viable within blanket mire areas under the worst-case scenarios of the 2050s.

15.5 Climate change and peat accumulation

Examination of the peat archive contained within Britain's peat bogs reveals a rather interesting feature which has potentially significant implications for the relationship between peat bog growth and climate change. It is clear from the archive that many of the fastest periods of peat accumulation have been when the vegetation has been rich in *Sphagnum* species typical of the terrestrial zone – ridges and hummocks – whereas periods in which pool species such as *S. cuspidatum* predominate are periods of relatively slow peat accumulation.

The greater resistance to decomposition displayed by species such as *S. papillosum* compared to *S. cuspidatum* have been discussed earlier in the present report, but the implications of this for the impacts have climate change have not yet been explored. The responsive nature of the acrotelm and its component species to changes in climate have been discussed above in relation to Figure 49, and from this it is clear that warmer, drier conditions would tend to result in an increased abundance of ridge and hummock Sphagnum species. These are species which tend to result in more rapid peat accumulation. Consequently it is possible that warmer, drier conditions might even lead to an *increase* in peat accumulation. This is a proposal for which there may be some evidence in Section 19.6.3.2, Discussion Topic 8.

15.6 Actions and research needs

15.6.1 Restoration of acrotelm

Given that bogs with a functioning microtopography, vegetation and acrotelm have demonstrably survived through warm dry climate periods in the past by altering the composition of the living vegetation, it would be reasonable to suggest that programmes designed to produce widespread restoration of such functioning systems would render the UK peat bog resource better able to respond to climate changes than would be the case if the resource remains in its present generally haplotelmic condition.

15.6.2 Refinement, development and application of Barber's (1981) 'phasic' model of peat bog response to climate change

Further analysis of the stratigraphic record stored within peat bogs sited in differing climate regions of the UK can provide a greater degree of understanding about how peat bogs have responded to past climate change. In particular, there have been very few studies involving the three-dimensional reconstruction of the stratigraphic archive from multiple, linked peat cores – Barber (1981) himself providing one of the few examples ever produced. If cores are sufficiently closely-spaced, such three-dimensional reconstruction can provide a much clearer picture of the actual surface character for any given time-period, and can also then reveal the dynamics over time.

Such studies can be relatively crude in the sense that the major macrofossils alone can form the main basis of analysis, identifying whether *Sphagnum* leaves are terrestrial or aquatic, hummock-formers or species of lower ridges, together with the relative abundance of non-*Sphagnum* bryophytes, heather, cotton grasses and deer grass remains.

In addition, and linked to such studies, linked examination of aerial photography and the recent stratigraphic history using very short cores, if taken from a sufficiently wide area, could provide some indication of any obvious trends in present bog dynamics. Extensive infilling of pool systems with vegetation might suggest that the bogs are responding to warming and drying effects.

15.6.3 Refinement of habitat and species climate-envelopes

Further development of the species-climate-envelopes devised by Berry and Butt (2002) would be of value as a means of identifying species which might be at risk of loss from UK bog systems. Equally, more refined development of habitat-climate-envelopes for raised and blanket bog systems would help to identify the potential nature of change in differing regions of the UK.

In particular, detailed studies are required into the relative importance of occult precipitation within the overall water budgets of individual *Sphagnum* plants, and the budgets of bog systems as a whole. These studies would need to be linked to the detailed study of local, regional and national patterns of occult inputs – namely presence and quantities of dew, mist and fog – together with the temporal patterns these display. Additional factors to consider in this work would be patterns of cloud cover and cloud types, together with patterns of atmospheric humidity.

15.6.4 Climate change and peatland hazards

A better understanding is required of the processes and mechanisms which lead to slope-failure in blanket mires, linked closely to a detailed assessment of associated habitat condition. Currently, habitat features of potential significance for slope-failure can be identified by field ecologists, but these features are not yet adequately recognised or incorporated within engineering models of slope stability.

An assessment of peatslide risk across the UK blanket mire resource, as is currently being undertaken in the Republic of Ireland, would help to identify areas potentially and particularly at risk under changing climate conditions, particularly if linked to models of potential storminess and intense rainfall events.

Similarly, an assessment of potential fire risk across UK peatlands would be useful, particularly if linked to models of potential soil-water deficit and drought periods. This could assist in identifying areas potentially most at risk from either accidental fires or from fires caused by lightning during convective summer storms.

15.6.5 Climate change, vascular-plant cover and DOC

Further investigations are required into the source of DOC, based on age and chemical composition (and recognising that 'DOC' is a broad term embracing a wide range of carbon-based compounds). In particular, further investigations are required into the relationship between rising DOC and declining atmospheric pollution, DOC and vascular-plant litter, and DOC and *Sphagnum*.

It can be assumed that warmer conditions and elevated levels of CO2 will give rise to more vigorous plant growth. Whether future climate conditions will encourage *Sphagnum* growth at the expense of vascular-plant cover, or *vice-versa*, might be an important factor in determining future levels of DOC in upland stream systems. What can be fairly safely assumed, however, is that warmer, drier conditions will encourage more vigorous growth of vascular plants on the currently-dominant haplotelmic bogs of the UK, with potentially significant effects on DOC levels.

However, at present there are few, if any regular measurements being made of blanket mire plant biomass. Even ECN (Environmental Change Network) sites do not currently measure plant biomass. There would therefore seem to be a strong argument for initiating regular measurements of vascular-plant biomass across a range of peat bog sites in the UK, and linking these measurements to DOC levels coming from the catchment.

15.6.6 Climate change, water management schemes and lowland bogs

Most lowland raised bogs are surrounded by agricultural land which, in most cases, was formerly some form of fenland. These fenlands have been drained, leaving the ground-water mound of the raised bog perched somewhat high and dry compared to the natural condition.

In the Netherlands, agricultural fields surrounding remnant raised bogs are being transformed into limited flood-plain systems through financial incentives and land purchase. A similar concept is under way in Britain with The Great Fen Project (see Great Fen Project website). Lindsay and Immirzi (1996) identified that every lowland raised bog in Britain had been subject to some form of significant hydrological disruption, the commonest cause being agricultural drainage. Given the likely climate trend towards warming and possible drying in the parts of the UK characterised by lowland raised bogs, water-management schemes could be drawn up for land associated with all surviving raised bogs in order to provide them with the necessary natural hydrological resilience to respond to warmer and drier conditions.

16 DISCUSSION TOPIC 6a Commercial peat extraction and the carbon balance

Commercial peat extraction for use in energy production removes carbon from a bog most directly as the raw peat which forms the commercial product or the source of energy. The amount of carbon lost is directly related to the quantity of peat extracted. As such, the losses of carbon from these activities can be calculated, at least approximately, if the volumes extracted, the bulk density and the carbon content of the peat are known. Earlier sections of the present report explain why even this apparently simple calculation can only provide an approximate calculation of such direct carbon losses.

In a life-cycle analysis of the Canadian peat industry, Cleary, Roulet and Moore (2005) estimate that decomposition of the extracted peat represents 71% of the total carbon cost associated with commercial peat extraction, while loss of carbon-fixing potential from land-use change accounts for 15%. The large distances involved in Canada mean that transport of the product contributes 10% to the carbon cost, whereas this would generally be a smaller proportion in Europe.

Extraction of peat is also associated with a series of indirect carbon emissions, however. The cut over bog surface continues to release carbon and the neighbouring uncut areas of the bog will also release carbon. Some of these emissions are gaseous, and occur through the processes described in Section 9.2, Discussion Topic 1 in relation to peatland drainage. Other losses are particulate and occur at a range of scales, encompassing both particulate organic carbon (POC) and dissolved organic carbon (DOC).

16.1 Gaseous carbon balance and peat extraction

Saarnio *et al.* (2007) undertook a study and review of carbon fluxes from boreal mires in order to provide a background to the fluxes associated with the peat-extraction industry. They highlight the fact that there are few published figures for the CO_2 balance of mires, and also that figures must be read with caution because mires typically display considerable variation in terms of broad mire mesotope type – bog or fen – as well as at the scale of microtope and nanotope (which they term broadly as 'microsite'). They also suggest a number of ways in which modelling of carbon fluxes might be improved in future:

- less use of process-based modelling;
- better hydrological models;
- better measures of the photosynthetic surface-area and character of the vegetation;
- long-term observations of both biotic and abiotic factors.

Glatzel, Basiliko and Moore (2004) compared potential CO_2 and CH_4 production rates in natural, cut-over and restored sites in Canada using incubation methods to determine the potential gaseous production from such sites. While incubation methods are of interest, they must be considered with a great deal of caution because the peat is subjected to a number of altered conditions which are not typical of those when it is *in-situ*. Glatzel *et al*. (2004) themselves accept that the incubation process imposes a number of atypical conditions on the samples. Waddington, Rotenburg and Warren (2001) also use incubation methods to investigate potential gaseous exchange from cut-over bogs, and the same cautionary note applies.

Both Glatzel *et al.* (2004) and Waddington *et al.* (2001) found higher CO_2 production rates in aerobic conditions (0.02–1.05 mg CO_2 g^{-1} day^{-1}) compared to anaerobic conditions (0.01–0.29 mg CO_2 g^{-1} day^{-1}), and higher production in the peat from the natural site compared with peat from the cut-over site. These results are unlikely to reflect the carbon flux of the peat *in-situ* because the peat of an extraction site is more likely to be undergoing oxidation than the peat of a 'pristine' site. Such incubation studies thus reveal the theoretical potential for carbon fluxes but not the actual fluxes found in the field. Gray (2005) specifically highlights this distinction, while Nayak *et al.* (2008) demonstrate the sometimes very substantial differences between values obtained from incubation and field-based studies in their review of carbon fluxes from a range of mire types and conditions.

Tuittila *et al.* (2000), Yli-Petäys *et al.* (2007)and Mäkiranta *et al.* (2007) instead used static-chamber methods to investigate the actual rates of CO_2 and CH_4 flux from cut-over sites in Finland, although these cut-over sites were in a variety of conditions and states of restoration or re-vegetation. As such, the values obtained for the first two studies are more appropriate to the issues of gaseous exchange and cut-over restoration than of actively-worked peat extraction sites, and will thus be considered in Discussion Topic 6b – Restoration of cut-over bogs.

Cleary *et al*. (2005) cite a number of sources giving measured values of gaseous carbon flux on actively-worked sites, these values ranging from 250-289 g CO_2-C m^{-2} yr^{-1} and 1.4-5.0 g CH_4-C m^{-2} yr^{-1}. In contrast Mäkiranta *et al.* (2007) found CO_2 release-rates of 276–479 g CO_2-C m^{-2} yr^{-1} from cut-over sites which had then been afforested. Mäkiranta *et al.* (2007) further conclude that afforestation of cut-over sites results in a net loss of carbon from the system for which the trees are unable to compensate. Both these sets of values can be compared with the net carbon accumulation rates of 30–35 g CO_2-C m^{-2} yr^{-1} cited by Turunen (2003) for natural Finnish bogs.

16.2 Particulate carbon losses from peat-extraction sites – POC and DOC

16.2.1 Losses of POC through aerial transport

As discussed earlier in the present report, the distinction between particulate organic carbon (POC) and dissolved organic carbon (DOC) is merely one of particle/molecule size. However, POC may be transported by wind or water, whereas by definition DOC is only water-borne. The distinction is important for peat-extraction sites because wind-borne peat is a significant factor in commercial peat-extraction operations, particularly when peat milling is the extraction method employed.

The scale of carbon losses associated with wind-borne POC is demonstrated by Tissari *et al*. (2006), who investigated fine-particle emissions from two commercially-worked bogs in central Finland. Tissari *et al.* (2006) used a Fugitive Dust Model (FDM) to model the overall pattern of wind-borne particulate losses from the two bogs, using field measurements to populate the model with realistic values. They found that the milling operation, which consists of several distinct stages, showed substantial variation in the scale of particulate loss between the different stages.

The two operations most evidently associated with very high losses were transporting and unloading the milled peat onto the main stockpile, and then re-shaping this stockpile prior to its transfer into wagons from removal and use. These two stages, and the amount of peat dust lifted into the air, can be seen in Figure 51.

Tissari *et al.* (2006) found that emission rates during unloading of peat and re-shaping of the stockpile could be one or two orders of magnitude greater than those associated with other aspects of the milling operation. Overall emissions from milling itself were lowest, amounting to 1 kg ha^{-1} $harvest^{-1}$, but when the other operations associated with milling were included the total rose to between 6 and 15.9 kg ha^{-1} $harvest^{-1}$, depending on the harvesting method used. The peat is typically dried to 30%-60% water content before lifting from the field, and a peat-milling field may have at least 4-6 harvests per season. Consequently the quantities of carbon involved for a 100 ha site may amount to 2.9 t C in a single harvesting season.

Although Tissari *et al*. (2006) found that the highest concentrations of aerial POC were restricted to distances of 200 m around particular operations, their data make it clear that concentrations of greater than 100 μg m^{-3} were recorded at distances of 1,500 m, and these measurements do not include emissions from stockpile shaping – the dustiest operation of all. Given that the EU daily limit-value is 50 μg m^{-3}, this is clearly an issue.

Tissari *et al.* (2006) also note that POC can be lost from peat-milling fields through direct wind erosion, but for this to happen the wind-speed must possess enough energy to lift the peat particles into the air. This threshold is approximately 3 m s^{-1}, which barely a ‘gentle breeze’ on the Beaufort Scale (Force 3).

Given that oceanic regions generally have stronger winds than Finland, and that Tissari *et al.* (2006) note the greater ease with which more highly-decomposed peat is wind-transported, peat milling in Britain and Ireland may experience significant losses through aerial transport, particularly in blanket mire areas.

Figure 51. Loss of POC from a commercial peat-milling field, Nutberry Moss, Dumfries and Galloway
The quantity of particulate organic carbon (POC) potentially lost into the surrounding landscape and watercourses on a dry, windy day during peat milling can be gauged from these photographs of transport to the main stockpile, then re-shaping of the stockpile, at Nutberry Moss, Dumfries and Galloway.

Photo R A Lindsay

16.2.2 Water-borne losses of POC from peat-extraction sites

The release of water-borne sediment from commercially-worked peatland sites has been an issue of some considerable discussion, debate and concern for many years in Finland. Kløve (2000) summarises the history of research and successive attempts to reduce or eliminate the problem, pointing out that official national targets were set for a reduction of 65% in suspended sediment from 1993 levels by 2005.

Kløve (2000) goes on to describe a large-scale experiment set up to test a combination approaches designed to reduce suspended sediment loads from peat-mining areas. The study looks at three techniques in particular – peak runoff control, floodplain filtering, and sedimentation ponds. Kløve (2000) makes a distinction between exceptional runoff events and more general runoff, one such exceptional event in Finland being snowmelt in the Spring.

16.2.2.1 Peak runoff control

This consists of dams which have multiple outlet pipes of differing diameters, the smallest diameter being positioned to permit lowest flows to pass, while the largest diameter outlet is positioned close to the top of the dam to permit large outflows when water-levels are high. In effect, these peak-runoff control dams mirror the variable conductivity of the acrotelm, permitting slow flow near the base of the acrotelm but allowing for large flows when the bog water-table is near the acrotelm surface.

The principle behind such structures is that suspended sediment is largely derived from loose channel-bed material which, if the current is sufficiently strong, can be lifted and transported down through the drainage system. It is not clear exactly what this threshold should be for any given site, but Kløve (2000) assumes a runoff intensity value of 60 mm day^{-1} as his threshold.

During snowmelt these structures were found to be capable of removing up to 95% of suspended sediment, whereas during the summer and autumn months in general, this method reduced suspended sediment by 53%-88%.

16.2.2.2 Floodplain filtering

Small floodplains of 30 m x 30 m were constructed and mire vegetation was transplanted into these. When water levels rise above a certain threshold in the feeder ditch, the water and sediment spills out across the mini-floodplain and sediment is trapped amongst the vegetation.

Sediment loads at snow melt could be reduced by up to 79% using these structures, and by up to 76% for the general summer and autumn months.

16.2.2.3 Sedimentation ponds

These ponds were quite large, being 60 m long and 8 m wide. During the general summer and autumn period they were capable of removing up to 68% of the sediment load.

Kløve (2000) found that runoff control measures on highly humified sites (which would be the case for many blanket mire areas) do not reduce the peak flow rate but instead slow down the speed at which the peak arrives. Nonetheless they resulted in a dramatic reduction in suspended sediment during exceptional events. They were sometimes less effective during periods of low flow, however. Similarly, the constructed floodplains were only partially successful in reducing suspended sediment. Kløve (2000) therefore concludes that the use of sedimentation ponds continues to be necessary even if peak flood control and mini-floodplain measures are introduced.

He recommends that outflows from peat mining areas should first be subject to peak flow control measures, then passed into constructed floodplains. The water is next passed into sedimentation ponds before finally being released from the peat-mining site. This sequence is described as being capable of meeting the national targets for suspended sediment – though even meeting these targets does not mean that outflow waters are free from suspended sediment. Their suspended sediment loads are significantly reduced from originally very high levels, but they are not eliminated altogether.

Given the complex pattern of sediment control described by Kløve (2000) in order to reduce but not eliminate sediment loadings, it is perhaps instructive to look at the sediment-control measures currently being employed at a peat-milling site in southern Scotland (see Figure 52).

16.2.3 DOC losses and commercial peat mining

Carbon loss in the form of DOC from commercial peat-extraction sites has not been extensively investigated, but Glatzel *et al.* (2003) report the results of DOC studies at several sites in various stages of commercial working, abandonment or restoration. They found that DOC levels varied from 35-625 mg C l^{-1} between the various sites, the highest levels (and these are extraordinarily high) being associated with re-wetted block-cut ground.

In general, they found that actively-worked milling fields generated DOC levels of between 50-180 mg C l^{-1}, indicating a potentially significant increase in DOC levels compared to the 60 mg C l^{-1} recorded by Glatzel *et al.* (2003) for an undisturbed bog.

Figure 52. Sediment pond at Auchencorth Moss peat-extraction site, Midlothian
Water from the peat-mining site enters at the left of the photograph and then exits via the evident outflow left-of-centre in the photograph. The accumulation of sediment in the centre of the pond is also obvious, but so too is the dark brown water flowing directly from inflow to outflow in the centre-left of the photograph.
Photo R A Lindsay

16.3 Actions and research needs

See Section 17, Topic 6b.

17 DISCUSSION TOPIC 6b Carbon balance and the restoration of commercial peat extraction sites

A great deal of work, funded by the commercial peat industry itself, has been undertaken into possible approaches to peatland restoration following commercial peat mining. These investigations have taken place most notably in Germany during the 1970s and 1980s (*e.g.* Eigner and Schmatzler, 1980), and more recently in Austria and Canada (*e.g.* Steiner, 2005; Rochefort *et al*., 2003). While the earlier German work was largely concerned with hydrological control, the Canadian work has concentrated particularly on the conditions required to encourage re-growth of *Sphagnum* across former commercial peat workings (Rochefort, 2000)

The carbon balance of the Canadian re-growth experiments is complicated by the fact that a straw mulch is used as a protective layer for the introduced *Sphagnum* and this straw layer releases much carbon as it decomposes. Determining the carbon balance of the *Sphagnum* itself is thus not as straightforward as it might be, although of course the carbon from the straw would have been released rapidly even if it had not been used in such mulching experiments. As such, the carbon from the straw is not relevant to the debate about the carbon balance of peatland restoration because it is on such a short and relatively closed cycle.

One of the most frequently-cited concerns about re-wetting and restoration of bog systems is the increased levels of methane which may result from elevated water tables. While a clear relationship between height of water table and methane release has been shown by, for example, Laine and Vasander (1996), it does not inevitably follow that raising water levels and re-wetting bog systems will result in substantially greater emissions of methane. In part, the type of vegetation being restored plays a critical part in determining the scale of CH4 emissions, but it is also a question of timescale.

17.1 Sphagnum regeneration and carbon exchange

17.1.1 Sphagnum regeneration and water tables

Money (1995) observed that water tables can be too high when attempting to restore *Sphagnum* cover. Deeper water tends to damage and wash away colonising *Sphagnum* plants through wave action and flooding, while Grosvernier, Matthey and Buttler (1995) and Sliva and Pfadenhauer (1999) identified the important role of vascular plants in providing protection and stability to *Sphagnum* during the re-colonising process.

Tuittila, Vasander and Laine (2003) describe an experiment on a Finnish peatland site using many of the same principles which underpin the Canadian restoration work, using macerated *Sphagnum* but without a straw mulch. They investigate in particular the role of *Sphagnum* capitula (dense bud-filled head of the *Sphagnum* plant) in re-growth of a *Sphagnum* cover because, as they observe, restoration aims to create a self-sustaining system which re-establishes the carbon-sink function typical of peatland systems, and the best indicator of this state is considered to be the presence of a living *Sphagnum* carpet.

Tuittila *et al*. (2003) demonstrate that using a high percentage of *Sphagnum* capitula in the layer of macerated *Sphagnum* spread over the experimental site induced 10x more *Sphagnum* cover than a layer without capitula. Significantly, in terms of carbon exchange and in particular methane flux, they also found that the best rate of growth was obtained at water tables equivalent to those in T3 hummocks (*sensu* Lindsay, Riggall and Burd, 1985), rather than in the wetter conditions of an A1 hollow (*sensu* Lindsay, Riggall and Burd, 1985), despite the fact that the species used (*Sphagnum angustifolium*) is more typically found in or close to the level of A1 *Sphagnum* hollows. Hummock water tables are generally associated with low methane release, or even methane uptake, suggesting that this form of restoration need not result in elevated CH_4 emissions, as is often feared.

17.1.2 Gaseous carbon-exchange and re-wetting of cut-over sites

Tuittila *et al*. (2000) measured the methane dynamics of a cut-over site which was undergoing restoration management involving re-wetting of the site. They found that CH_4 emissions increased on re-wetting but that, even with this increase, emissions remained lower than those found on undisturbed sites.

Mäkiranta *et al*. (2007) observe that the peat left behind for restoration after commercial working has ceased may be several 1000 years old, and the microbial populations necessary for CH_4 production may no longer be present in the peat. Consequently the process of re-wetting alone may not result in substantial release of methane from cut-over sites. Indeed Waddington, Rotenberg and Warren (2001) emphasise that re-wetting of cut-over sites is important as a means of preventing temperature increases in, and consequent CO_2 losses from, the dry peat.

Yli-Petäys *et al*. (2007) noted that various accounts of cut-over restoration programmes describe late-stage restoration as being a significant carbon sink. Yli-Petäys *et al.* (2007) themselves studied the carbon-exchange balance of Aitoneva mire, central Finland, as this site had undergone more than five decades of spontaneous re-wetting and re-vegetation following cessation of peat cutting in 1948. During this time the peat-cutting 'hollows' – formed by the long trenches created as peat was cut in blocks from the site – had infilled with four distinct *Sphagnum*-dominated vegetation stands which spanned a gradient from extremely wet (*Sphagnum pulchrum* stand) to rather dry (*S. papillosum* stand).

The wettest stands on Aitoneva were found to be the most productive and were net sinks of carbon during at least one year of the 2-year study. Although all stands showed much the same rate of photosynthesis, differences between them arose from differing respiration rates, these being dominated by levels of respiration from the vascular plant cover in comparison to rates associated with the *Sphagnum* carpet.

Overall, the CO_2-C balance of the *Sphagnum* trenches on Aitoneva was essentially zero because winter respiration balanced summer growth. Yli-Petäys *et al.* (2007) observe that weather conditions were particularly unfavourable during the study period, with exceptionally warm springs accompanied by night frosts sufficient to reduce vascular-plant emergence significantly (it is worth noting that the growing period for bogs in Britain is virtually year-round, whereas in Finland the winter months are essentially periods of respiration only, with no growth). The warm spring also led to significant release of CH_4, although the wettest *Sphagnum* stands were sufficiently large C-sinks to balance these CH_4 emissions and still remain in overall carbon credit.

Yli-Petäys *et al*. (2007) conclude that such recolonising trenches originating from block-cut extraction show a carbon balance which lies within the parameters often cited for natural mire systems. They do not find evidence for such areas being large carbon sinks, but suggest that restoration of milled extraction sites begins from a different starting-point, and the trajectory of restoration is rather different. It is suggested that this may explain why evidence has emerged from other sites that, in the long term, restoration of commercially worked peatland sites can result in the development of large carbon sinks.

17.2 Actions and research needs

17.2.1 Rates of peat oxidation

Evidence suggests that the major carbon-cost of commercial peat extraction is the oxidation of the extracted peat (71% according to calculations in Canada). Remarkably, the rate of oxidative loss from exposed, extracted peat under differing environmental conditions is rather poorly documented. Studies of peat-oxidation rates under differing conditions are therefore urgently required.

17.2.2 GHG flux and peat-extraction sites

The limited number of studies looking at the GHG balance of peat bogs in Britain (or indeed across most of Europe) means that it is difficult to put any measurements of GHG flux obtained from cut-over bogs into any wider context. Further GHG-flux studies are needed from both uncut and commercially-extracted peat bogs. Incubation measurements of GHG-flux potential should be treated with caution because this potential may not be realised in the field.

17.2.3 GHG flux and peat-extraction sites

Although particulate losses of peat in water systems is well documented for Finland, measurements for extraction sites in Britain are more limited, while aerial losses of particulate carbon have been little-studied. More widespread studies of POC loss, both water-borne and airborne, are required within the British context.

Commercial peat extraction sites which are now subject to restoration management tend to consist either of old-style 'baulk and hollow' cuttings, or extensive milled peat fields, or sometimes both. On the basis of relatively limited experimental evidence, the GHG-flux experience for these two types of extraction appears to differ substantially, with milled-peat surfaces more easily demonstrating carbon gains following restoration management. However, the evidence-base for this is very limited.

Estimates for the extent of 'baulk and hollow' cuttings, and milled peat fields, should be made across Britain to determine the potential extent of these two starting-points in the restoration process. Furthermore, a survey and analysis of existing bog regeneration, whether managed or spontaneous, within abandoned commercial workings could usefully be undertaken in order to assess the sequence of recovery processes demonstrated on such sites, and combined with GHG-flux, biodiversity, water quality and DOC measurements.

Detailed studies, incorporating GHG-flux, biodiversity, water quality and DOC, and building on the best of the restoration techniques described from Britain, Ireland, continental Europe, and Canada in particular, could also be usefully established on both abandoned 'baulk and hollow' and milled peat fields.

18 DISCUSSION TOPIC 7
Burning and peat bog systems

Bog peat consists almost entirely of plant mater. Consequently just like wood it can be burnt as a fuel. Two thousand years ago, Pliny noted in rather disparaging terms that natives of the Frisian Islands 'dug the soil' to burn, from which we can probably assume that the use of fuel-peat was an established practice as far back as pre-historic times. Although it burns well it is not a good fuel because the carbon density is so low. Anyone who has cut, stacked and burnt peat knows only too well the speed with which it burns (especially the surface 'mossy' peat) and consequently the huge quantities which must be amassed in order to provide a year's supply.

Though not the best of fuels, peat nevertheless burns readily when sufficiently dry. Indeed fire on peat bogs can be a natural phenomenon, being caused by lightning strikes during convective storms following long dry periods of weather. These natural events occur with long repeat times, estimated to be in the order of 500 years for boreal peatlands. Human-induced burning may also have influenced the character of British and Irish blanket mires since Mesolithic times. Fire is a well-established and documented tool in Mesolithic hunting, either chasing target animals to ambush, or encouraging grazing animals to feed in high densities on fresh post-fire growth. However, the regularity of such human-induced fires has probably never been so great as it has been during the last two centuries.

This rise in the burning of blanket mires, in particular, is contemporaneous with, and to some extent explained by, a number of significant social events. In the 18th Century pressure to increase productivity of hill ground for sheep led to an increase in burning of blanket mires to produce fresh growth of vascular plants, particularly early in the spring.

With the rise in sporting management in the 19th Century many blanket mire areas were included in burning regimes adopted for game interests. This burning activity intensified at times when estates had less manpower to control the fires. Mackay and Tallis (1996) cite the difficulties experienced by sporting estates after World War I in obtaining sufficient manpower to maintain traditional moorland management methods. They conclude that this resulted in widespread adoption across the Forest of Bowland of poor management practices, including burning regimes that were less carefully-controlled than in the past. They identify in particular a catastrophic (and probably unintentional) wildfire which took place around 1921 as one of the key factors contributing to the present eroded condition of blanket peats in the area.

The most recent factor influencing the fire history of blanket mire regions in Britain has been the support system provided to upland agricultural holdings in 'less favoured areas'. The support system has until recent years been based on the number of sheep in a holding, which inevitably encourages a tendency towards high densities and potential overstocking on the hill ground. Regular burning in the spring to provide an 'early bite' for these increased numbers of hill stock has also had a significant impact on the condition of Britain's blanket mires.

The principle of burning management in upland areas is now so deeply entrenched that it is the one large-scale management tool which remains sufficiently ubiquitous throughout the uplands that official guidelines are deemed necessary to provide constraints and best-practice guidance for upland land-manager.

18.1 Burning and peatland management

Much guidance concerning the use of fire management on bogs has in the past emphasised the importance of only undertaking such an activity when weather and ground-surface conditions are right, and that the fire be controlled to produce a quick, light burn which does not affect the ground layer. In Germany, Eigner and Schmatzler (1980) have provided very specific parameters for peatland fire-management, including the condition that bogs should only be burnt when the ground is frozen. The various stringent conditions set out by Eigner and Schmatzler (1980) can often be difficult to meet in oceanic Britain.

The difficulties involved in providing the right controls in conjunction with the right conditions mean that official guidance in Britain now states that blanket bog should not be burnt. This recommendation, however, introduces its own uncertainties and ambiguities, because the critical issue then becomes "what is the definition of blanket mire"? The guidance states that blanket mire with more than 70%

heather cover can in fact be burnt, but it might be argued that it is precisely this type of blanket mire which should <u>not</u> be burnt.

While rocky knolls with thin peat or no peat stand out in a blanket mire landscape as areas of natural heather domination, heather-dominated blanket mire is usually a product of management which is causing the peat to dry out. Holden (2005a,b) points out, for example, that heather-dominated blanket mire has a greater concentration of sub-surface peat pipes, and the suggestion is that such piping indicates a drying of the peat mass. If blanket mire has an un-naturally high cover of heather, burning of such heather is likely to do one, or both, of two things:

- burning regenerates the heather, making it more vigorous and thus aiding yet further in the drying out of the peat and discouraging *Sphagnum* re-colonisation;
- tall heather is far more likely to result in a hot fire which can kill the ground layer and any *Sphagnum* which is re-colonising and re-wetting the ground.

Agreeing, or recommending, that such heather-dominated blanket mire may be burnt simply perpetuates the un-natural dominance of heather on such areas and could undermine efforts to develop a more characteristic bog vegetation. Perpetuation of such heather-dominated areas probably ensures that these parts of the blanket mire landscape remain emitters of carbon for the foreseeable future.

The effects of burning on blanket peat have been described and summarised by a number of authors, including Lindsay *et al*. (1988), while Maltby, Legg and Proctor (1990) provide a detailed account of changes to blanket peat soils and vegetation in the North York Moors as a result of a severe fire in the summer of 1976. Tucker (2003) also provides an in-depth review of the impacts of fire on the upland environment.

18.2 Extent and frequency of burning management

What is very clear from the literature and from extensive field experience is that burning has been, and continues to be, one of the most widespread of human impacts on the blanket mire systems of Britain. Large swathes of hare's-tail cotton grass (*Eriophorum vaginatum*) in the Pennines and eastern Scotland, purple moor grass (*Molinia caerulea*) in NW England and SW Scotland, and deer grass (*Trichophorum cespitosum*) in northern Scotland, attest to the ubiquitous and extensive nature of burning across all blanket mire landscapes.

McVean and Ratcliffe (1962), in their authoritative monograph on the vegetation of the Scottish Highlands, comment on the extensive evidence for fire throughout the blanket mires of the region.

> *"…Bogs bearing Trichophoreto-Eriophoretum typicum [one of the most characteristic vegetation types of Scottish blanket mire] often show fire degeneration more by changes in floristic composition than by active peat wastage ... Such changes involve a decrease in Sphagnum cover, increased tussock formation in the vascular plants, and often the development of large mounds of Racomitrium lanuginosum on the drying bog ... Around the south-east end of Loch Meadie in Sutherland ... a continuous carpet of Racomitrium occupies at least one square kilometre of moorland leaving only a sparse scattering of Calluna vulgaris and Trichophorum cespitosum except in those places that have escaped the full effects of fires ...*
>
> *... During a dry spell in spring and summer it is possible to burn even the wettest Sphagnum-dominated bog and this surface disturbance combined with marginal interference probably explains the degeneration of many pool and hummock complexes. Although some examples, such as the Strathy Bog described by Pearsall (1956), have escaped serious interference, the majority have been burned and show every stage of drying and wastage...*
>
> *McVean and Ratcliffe (1962), p.110*

Tucker (2003) provides a valuable and wide-ranging review of burning as a management tool in the uplands. In drawing together the evidence from a number of studies which provide evidence for burning at various times in the past, Tucker (2003) sheds a revealing light on the pattern of fire frequency in the uplands of Britain over the last 11,000 years. He notes that the fire frequency of Swedish boreal forests is around 100-150 years in more moist sites, but also notes that fire frequencies during the Boreal Period in Britain were probably higher because conditions were warm and dry then.

Reviewing a number of studies, mainly from Scotland, Tucker (2003) then reviews the evidence for fire frequency in the uplands during the last 10,000 years. It is important to recognise that although at least the earliest evidence of fire comes from the stratigraphy of peat bogs and thus the fire frequency can be taken to relate to fires on the bog, later figures refer more generally to 'the uplands', or to 'grouse moors', and these may or may not equate to blanket mire habitat. The Grampian Region of Scotland in particular has extensive heather moors which are not blanket bog (or are barely so), while other parts of Scotland which typically support more extensive tracts of blanket mire also contain significant expanses (particularly on markedly sloping ground) of thinner organo-mineral soils which support heather but which are not blanket bog.

Tucker (2000) notes that fire-return times for England over the last 200 years are not well documented, but the charcoal archive in the peat enables an estimate to be made of earlier, prehistoric fire events. On this basis, he notes that fires on Dartmoor during the Mesolithic Period appear to be almost continuous, while in Sutherland, NW Scotland, the record for 4,000 years ago (4,000 BP), suggests that there was a fire-return frequency of some 32.5 years, although some evidence points to a frequency of between 2 and 25 years within the associated areas of pine forest.

Tallis and Livett (1994) record a fire-return frequency of some 165 years in the 2,500 years between the pre-Roman Iron Age and the present day, according to the charcoal record of Alport Moor in the Peak District. Anderson (1997) summarises a variety of documented fire events in the Peak District during the 1800s and 1900s, some caused by management, some by accident, some by deliberate arson.

Anderson (1986) also looks at the scale of more recent accidental fires within the Peak District National Park, and then summarises a range of data for such fires (Anderson, 1997) to show that the most extensive fires occur during long dry summers, and that 72% of fires are associated with paths, roads and other areas of open access. With the new 'right to roam' legislation for open ground, this set of findings has potentially significant implications for the blanket mires of the Peak District and other similar areas if summers are indeed to become warmer and drier in the future.

Lindsay *et al*. (1988) describe the condition of 399 blanket mire sites in the Flow Country of Caithness and Sutherland, having first dismissed 84 sites on the basis of aerial-photo evidence as being too damaged to be worthy of visiting. Of the 399 sites visited, 66% had signs of fire damage. More than 10% of sites were severely burnt, having a blackened, greasy surface through which only deer grass (*Trichophorum cespitosum*) and hare's-tail cotton grass (*Eriophorum vaginatum*) grew as fresh shoots.

Yallop *et al*. (2006) have used aerial photography across a 2% sample of the English uplands to assess the scale and frequency of fire events between the 1970s and 2000. They found a substantial increase in the extent of burning, rising from 15.1% of samples in the 1970s to 29.7% in 2000. They calculated fire return-times and found that these varied between 14 and 25 years, with a median return-time of 20.1 years.

Bringing these various data together into Table 21, it can be seen that the average fire return-times all lie within a range of 10-100 years, and that there is even evidence from return-times of a similar order of magnitude from as long ago as 4,000 years. It is interesting that for the Peak District, the record for Alport Moor suggests a much longer mean return-time of 167 years for the whole period since the pre-Roman Iron Age. This may either reflect the background rate of natural fires caused by lighting during dry weather, or perhaps a strict regime of common and manorial land management, as referred to by Anderson (1997).

18.3 Recovery times after fire

Given the catalogue of fire-return times shown in Table 21, it is then interesting to consider the likely fire-recovery times for the blanket mire habitat. Lindsay and Ross (1994) provide evidence of recovery after a severe fire on Glasson Moss National Nature Reserve, a lowland raised bog in northern England. Their results indicate that, even under relatively benign climate conditions, *Sphagnum* is not likely to re-establish vigorous growth until 10-20 years after serious fire damage, assuming that there are surviving remnants of *Sphagnum* from which new shoots can develop. Growth from axial buds on *Sphagnum* stems can take several years before new shoots reach the surface and begin to re-establish a *Sphagnum* carpet. There is thus a lag period of some years after a fire during which the *Sphagnum* carpet is re-assembling itself. If the fire was sufficiently hot to kill even the axial buds, re-establishment is likely to take much longer. The present author visited Glasson Moss again in 2008, more than 30 years after the fire, and found a vegetation surface which still looked distinctly fire-damaged. It was clear that a considerable period of further time would be required before the site would begin to resemble its pre-fire state.

Table 21. Fire return-times (in years) for blanket mires in Scotland and England
The data have been assembled from Tucker (2003), Yallop *et al*. (2006), and Tallis and Livett (1994).

Period	Scotland	England	Purpose
9,700 – 8,300 BP	-	[continuous] Dartmoor	hunting, forest removal
4,000 BP	2 – 25 / 32.5 Sutherland	-	hunting? sheep grazing?
2,500 BP		167 Alport Moor	?
1800 – 1850s	10	10?	sheep grazing
1850 – 1873	100 (max)	100 (max)?	grouse, sheep grazing
1875 – 1900	30 – 40	30 – 40?	grouse, sheep grazing
1900s – 1950s	60 – 70?	60 – 70?	less grouse management
1950s – 1990s	30 – 40+	7 - 12	sheep grazing (grouse in Scotland)
2000	-	4 – 20	grouse

The data from Lindsay and Ross (1994) come from a lowland site where the climate is relatively warm and humid and thus conducive to *Sphagnum* growth. Climate conditions in blanket bog regions are much harsher, and recovery rates are thus significantly slower. Maltby *et al*. (1990) emphasise the fact that across extensive areas of their fire-damaged site in the North York Moors, there was little evidence of recovery 10 years after the fire. McVean and Ratcliffe (1962) emphasise the way in which increasing altitude and latitude reduce temperatures and shorten the growing season for various species and vegetation types of the Scottish Highlands. Furthermore, McVean and Ratcliffe (1962) and Hunter and Grant (1971) emphasise that wind exposure also reduces effective temperatures and growth rates in western Scotland.

Lindsay *et al*. (1988) provide a review of the physical, biological and ecological impacts of fires on peatlands, and discuss a number of aspects of the recovery process. They also highlight the fact that Wein and MacLean (1983) give the natural return-time for fires in the peatlands of Nova Scotia as being between 400 and 500 years. Such potential timescales for recovery do not look so unlikely if the scale of potential damage to the peat system caused by severe fires is also considered. Anderson (1997) presents a range of information emphasising both the dramatic and long-term impacts which can result from fire damage.

Whelan (1995) illustrates the way in which fire frequency can have a rapid and dramatic effect on recovery times following fire. Citing the work of Zedler *et al*. (1983) on Californian chaparral vegetation, he shows that two fires in consecutive years have a major effect on the recovery process compared with the usual recovery sequence. The two successive fires resulted in the mortality of species which would normally have recovered after a single fire, and also significantly depleted the soil seed-store because young germinating plants were killed before they were able to produce replacement seeds. The frequency of *Adenostoma fascicularis* was reduced to 10% frequency in twice-burnt plots compared with 60-70% frequency in plots burnt only once. Whelan (1995) goes on to discuss in some detail the concept of 'fire regime'.

With this in mind, it is worth looking at the fire return-times given in Table 21 again, but this time within the context of potential ecosystem-recovery times. Figure 53 shows the figures from Table 21 set against the natural return-time given by Wein and MacLean (1983) for Nova Scotia peatlands. It is clear from Figure 53 that even if the ecosystem-recovery time is 100 years, the present pattern of fire-return times is far shorter than this.

Kuhry (1994) analyses the fire return-times for *Sphagnum*-dominated boreal peatlands in various parts of Canada. He notes that recovery times for such systems requires "a few decades" in order to re-establish the original vegetation, but also highlights the fact that this is not the whole story. Even if the vegetation has returned to something similar to that which existed before the fire, there will still have been effects on system components and processes which mean that the peatland has not re-established its original carbon budget.

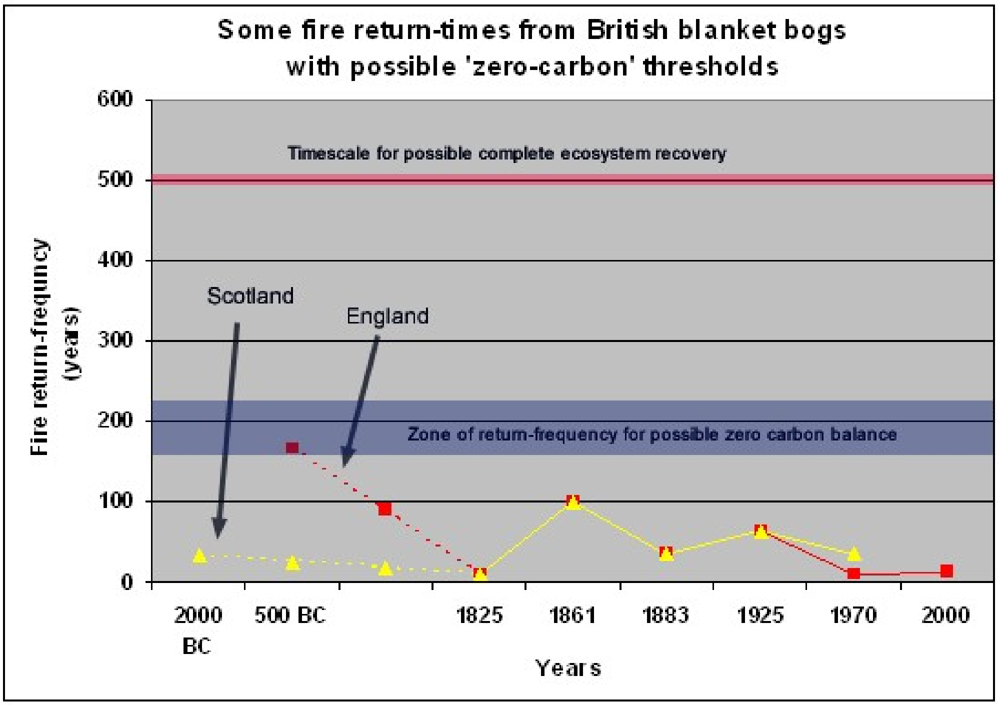


Figure 53. Fire return-times for various upland areas in Scotland and England, with an indication of potential recovery time.
Values for fire return-times have been assembled from Tucker (2003), Yallop et al. (2006) and Tallis and Livett (1994). The values are the same as those shown in Table 21 of the present report. The yellow line represents data for Scotland, the red line shows data for England. The red and yellow dotted lines represent the break in the time-line between pre-historic records and those for the 1820s. The dark-blue shaded zone represents the return-frequency identified by Kuhry (1994) as the threshold beyond which fire results in net carbon losses from the site. Anything *below* this line would be in net carbon loss. The red line at 500 years return-frequency represents the natural return-time recorded for bogs in Nova Scotia by Wein and MacLean (1983), and may represent the timescale required for full ecosystem recovery.

Kuhry (1994) identifies that in the last 2,500 years, the mean fire return-time for these boreal *Sphagnum*-rich peatlands is 1,150 years. Even the shortest return-time within the group of sites under study – Buffalo Narrows, Saskatchewan – is 400 years. While such return-times do not necessarily reflect the vegetation recovery time, Kuhry (1994) demonstrates that for the system carbon budget, these timescales are very important. He calculates that, if the frequency of fires increases to somewhere between 5x and 7x the average fire frequency for these sites (*i.e.* 164 - 230 years), then the long-term carbon accumulation rate would be reduced to zero. Return-times shorter than this would mean that, overall, the site was moving into carbon deficit.

Consequently, given that the current fire return-time for British pea bogs appears to be anywhere between 4 and 40 years, the present cycle of burning on British blanket mires would appear to be based, by a very substantial margin, on too short a timescale to retain overall carbon stocks.

Maintenance of the carbon cycle and store may thus require something closer to 200 years, or perhaps even longer, depending on the scale of fire damage, rather than the current 20-year cycle. Such extended timescales are rarely discussed or contemplated when the question of fire management is being considered. This is because fire management has been increasingly driven by the 25-year cycle of heather growth and senescence. Yet heather is not, or at least *should* not be, the dominant plant of blanket bog systems even in places such as the Peak District. The keystone species for such areas is *Sphagnum*, which has no need of a burning cycle – indeed any sort of burning regime is likely to cause distinct harm to any existing *Sphagnum* sward and render the ground more hostile to recolonisation by *Sphagnum* species.

18.4 Burning and the carbon balance

Intuitively, fire reduces the carbon store of a bog. When peat is burnt as a fuel the carbon stored within the peat is released to provide heat via the oxidation of the carbon to CO_2. This CO_2 is then lost to the atmosphere.

The burning of Indonesia's tropical swamp forests has increasingly been recognised as a major source of CO_2 emissions because the bulk of the fires consume the peat soils of the swamp forests and destruction of these soils represent a much greater carbon loss than the carbon lost from the forest itself (Page *et al*., 2002). Fires on peat therefore appear to be inimical to the maintenance of long-term peatland-carbon stores.

Some recent research has questioned this intuitive assumption. Several recent studies into burning, grazing and carbon losses explore the possibility that some aspects of the blanket-mire carbon-balance may even benefit from fire management, or at least suffer losses which are insignificant in the overall carbon budget of the system. This work merits close attention because it considers burning and blanket mire management in ways which appear to contradict the growing belief that blanket mires should not be burned because the resulting carbon losses are too great.

As discussed above, the long-term carbon balance for boreal *Sphagnum* mires identified by Kuhry (1994) in Canada indicates the very long timescales necessary to recoup carbon losses from all possible routes. A mean fire-frequency of 1,150 years for these sites is derived from a set of fire-frequency values ranging from 400 yrs to 1,790 years, and the carbon-recovery time thus cannot be less than 164 years if even the site with shortest cycle is to stay in carbon-credit.

18.4.1 Burning studies in the Pennines

A long-term monitoring experiment was established at Moor House, north Pennines, in 1954 by the Nature Conservancy in order to look at the various impacts of, and varying interactions between, burning and grazing on blanket mire. Four blocks of what is generally described as 'heather moorland' were delineated on Hard Hill within the Trout Beck catchment of Moor House. These four blocks were divided into six sub-plots. Some plots were fenced to exclude sheep grazing, and some plots were burnt on an experimental basis, the combination of treatments allowing a comparison to be made between burning, grazing, and burning and grazing combined. All plots were burnt in 1954 but some have not been burnt since. Others have been burnt on a 10-year rotation, while others have been burnt on a 20-year cycle.

In recent years a series of inter-related studies has used these Hard Hill plots to investigate the relationship between burning, grazing and carbon balance within blanket mire systems. In general these studies have involved measurements of water table, water chemistry, DOC levels and, most recently, the hydraulic conductivity and carbon content of the peat. Although there are issues common to all these studies, each is sufficiently distinct to merit specific comment.

18.4.1.1 Methodological concerns about Hard Hill studies

In a vegetation study undertaken by Marrs, Bravington and Rawes (1988) into the effect of grazing and exclusion of grazing in these study plots, the title of the paper by Marrs *et al*. (1988) is "Long-term vegetation change in the *Juncus squarrosus* grassland at Moor House, northern England". This is revealing because *Juncus squarrosus* grassland is not blanket bog. Indeed the *J. squarrosus*-characterised sub-community of deer grass/cotton grass (*Trichophorum cespitosum-Eriophorum vaginatum*) blanket mire is more generally associated with the transition from blanket mire to wet heath (Rodwell, 1991). Marrs *et al.* (1988) also refer to mat grass (*Nardus stricta*), sheep's fescue (*Festuca ovina*) and heath woodrush (*Luzula multiflora*) which are all more characteristic of thinner peats and peaty heathlands. It is not therefore clear what precisely is being investigated within the Hard Hill plots.

The second important thing to recognise about these studies is that the measurement process of the studies themselves have the potential to influence the results obtained in two distinct and important ways. Marrs *et al*. (1988) comment on the detailed and intensive (annual) monitoring of Breckland grasslands by Watt (1981) but observe that such intensive monitoring at the Hard Hill plots would not be possible nor desirable because the vegetation is so sensitive to trampling. They make the point that the fragility of the vegetation can easily result in a study which largely records the effect of monitoring rather than the effects of the experiment itself. Lindsay *et al*. (2003) and Lindsay and Freeman (2006) describe the considerable lengths to which it is necessary to go if monitoring of peat bog sites is to avoid the problem of trampling disturbance.

None of the studies described below mention the use of boardwalk or other measures to avoid causing trampling impacts to the vegetation, although the dipwells were visited every two weeks. Marrs *et al.* (1988) consider that sampling once a year is too frequent a disturbance for such fragile vegetation. There is thus the possibility that the vegetation in the study plots may have been affected by such frequent visits, particularly around the dipwells, and that consequently the water-table behaviour may have been altered. This is particularly the case for *Sphagnum*-rich vegetation, which is both extremely sensitive to trampling and highly capable of influencing water-table characteristics.

A further source of potentially observer-influenced error also relates to the absence of boardwalks. On peat of 1 m or more, the simple weight of an observed standing next to a dipwell can alter the water level in the dipwell, sometimes quite significantly. Goode (1970) found differences of up to 20 cm in his studies of blanket mire in SW Scotland, which led him to use boardwalk and devise a simple but effective remote probe for measuring the water levels in his dipwells. Anyone testing the level of water in the peat, or the emissions from the peat, needs to take this compression-effect into account (ideally by using boardwalk).

18.4.1.2 Garnett, Ineson and Stevenson (2000): burning and carbon accumulation

Garnett *et al*. (2000) use the Hard Hill experimental plots to determine the effect of light grazing and a 10-year burning cycle on carbon accumulation rates of the bog. They conclude that light grazing has no effect on the accumulation rate, but that over a 30-year period a 10-year burning rotation results in a reduced rate of carbon of carbon storage.

18.4.1.3 Ward *et al.* (2007): burning and carbon stocks

Ward *et al.* (2007) investigate the relationship between a 10-year burning cycle, low levels of grazing, a combination of the two, and the carbon stored in above- and below-ground biomass. They also measure carbon fluxes in the form of CO_2, CH_4 and DOC..

Ward *et al*. (2007) find that burning on a 10-year rotation reduces the above-ground carbon by 56%, whereas grazing resulted in a reduction of 22% in above-ground carbon. Grazing had no effect on below-ground carbon stocks, but the 10-year burning cycle reduced the carbon stored in surface horizons by 60% (F and H horizons – partially and wholly decomposed litter horizons). Deeper horizons were not affected.

Proportionally in terms of carbon density this loss is very small when compared to the carbon store of the whole peat thickness. However, within the dynamics of the acrotelm, this loss could represent a significant proportion of material which would otherwise have contributed to long-term transfer of material into the catotelm. Calculated over a 10-year time period, the annual C-losses from above- and below-ground stocks amounts to 25.5 g C m^{-2} yr^{-1}. This is almost exactly the amount calculated to transfer annually into the catotelm in 'standard natural cubic metre of peat' in Section 8, Figure 24 of the present report.

Ward *et al*. (2007) also found that burning increased overall uptake of CO_2 for photosynthesis, although these values were more significantly influenced by seasonal climatic conditions. Uptake was also found in response to grazing, but the effect was not as large.

One other thing to emerge from the study by Ward *et al.* (2007) was the response of the plant species-groups to burning and grazing. In particular, the moss layer was found to be highly sensitive to the 10-year burning cycle, being almost entirely absent from these plots. The general trend relating treatments to moss cover was:

no burning ----- ungrazed ----- grazed ----- 10-year burn
(most moss) (least moss)

Thus the bryophyte (moss) layer is particularly sensitive to burning but is also affected by grazing, mainly through trampling. Given that in a functioning bog the majority of material passed from the acrotelm to the catotelm is moss, the absence of moss from the burnt plots is significant.

18.4.1.4 Worrall, Armstrong and Adamson (2007c)

This study examines the effect of burning and grazing on water-table depth and water chemistry. Water table was measured using dipwells of at least 90 cm depth, recorded at 2-week intervals between June and September. Water chemistry analysis was performed on samples taken from the dipwells.

Worrall *et al*. (2007c) observe that the plots burnt on a 10-year burning cycle have higher water tables than either those without burning or those on a 20-year cycle. They note that Garnett *et a*l. (2000) found reduced peat accumulation in the 10-year cycle plots, despite the fact that the water tables were highest in these plots. Worrall *et al*. (2007c) observe that the differences in water table "may be due to differences in the presence of *Sphagnum* mosses", but provide no information about the *Sphagnum* cover or species composition in the various study plots. It is therefore difficult to make any judgement about the relationship between burning, peat accumulation, water table and *Sphagnum* from this study.

18.4.1.5 Worrall and Adamson (2007)

This study looks at the chemistry of water taken from dipwells in the Hard Hill study plots to determine whether the chemical signature of the bog water indicates structural changes within the peat. Many of the same issues apply to this study in terms of fragility of the vegetation. Significantly, Worrall and Adamson (2007) cite both Mallik, Gimmingham and Rahman (1984) and Mallik and Fitzpatrick (1996) in terms of the effect of burning on the properties of peat and the duration of such effects, but both these cited studies are really concerned with dry and wet heath rather than blanket mire.

Worrall and Adamson (2007) observe that burnt plots and unburnt plots differ because the latter have shrubby heather. Worrall and Adamson (2007) themselves conclude that the 10-year experimental burning cycle is too short to permit proper vegetation recovery.

18.4.1.6 Clay, Worrall and Fraser (2009a)

This study considers the effects of burning and grazing on DOC release from the Hard Hill study plots. The study found that unburnt plots showed the greatest range of DOC release. Plots with a 20-year burning cycle showed lowest absorbance (water colour – taken to be a proxy for DOC) values while plots with a 10-year cycle showed largest absorbance values. However, DOC values were not significantly different between treatments, other than peaks for short periods after burning.

Clay *et al*. (2009a) conclude that burning does not affect DOC concentrations in runoff waters except for some weeks after a burn. They note that DOC and water colour were generally lower in surface runoff water compared with water in the peat. They also concluded that water colour was lower in plots with a 20-year burning cycle. From this, they go on to suggest that longer burning rotations may help to reduce water colour coming from blanket mire systems.

The difficulty with these conclusions is that no information is given about the nature of the vegetation in the various plots, despite the fact that Clay *et al*. (2009a) acknowledge the possible impact of differing vegetation types on the various parameters under investigation. The fact that DOC and absorbance vary so much within the unburnt plots would suggest that there may be significant vegetation differences which might help to explain this source of variation, but the issue is not explored.

18.4.1.7 Clay, Worrall, Clark and Fraser (2009b)

This study concludes that sites with managed burning produce greater run-off and that run-off contains lower concentrations of DOC. The paper does not provide any measured results for DOC so it is not clear where the observation about DOC comes from. However, the higher run-off observed from burnt plots would be consistent with a bog surface which has developed a surface layer which is rich in carbonised materials, bitumens and waxes as a result of burning, thereby making the surface layers more water repellent (Clymo, 1983; Chistjakov *et al.*, 1983).

18.4.1.8 Worrall and Clay (2009)

This poster describes investigations into the carbon content of peat when subjected to combinations of burning and grazing. The poster gives an account of the production of 'biochar' from burnt litter and biomass, and describes this as low-volume, high carbon-content material which can be used to store carbon.

Values are given for the carbon content above a known horizon from the three burning treatments of no burn (actually burnt 50 years ago), a 20-year burning cycle and a 10-year burning cycle. The carbon value for the 20-year cycle is somewhat larger than for the other two treatments. Although the difference is described as 'not significant', the poster concludes that the 20-year burning cycle appears to store more carbon than the other treatments.

The argument for biochar seems to be that a concentrated layer of carbon is deposited at the peat surface every 20 years or so. This carbon, being largely charcoal, is highly resistant to decay (10,000 years or more). The vegetation of vascular plants is allowed to grow on a 20-year cycle and at the end of each cycle a new layer of charcoal is laid down. Over time, a sequence of carbon-rich layers is deposited and subsequently stored, potentially for millennia.

Considered in the abstract, therefore, biochar may have the potential to offer much the same long-term carbon-storage process as a living peat bog. In practice, however, the potential benefits may not be large, whereas the potential dangers may be significant. Work by Farage *et al*. (2009) in nearby Wensleydale, and studies by Yallop and Clutterbuck (2009) put the potential benefits into a useful context, as discussed below.

18.4.1.9 Farage *et al*. (2009) : Mossdale Moor, Wensleydale

Farage *et al*. (2009) examine the carbon losses associated with the burning of heather moorland on a peaty gley soil. The organic layer of their study plots was generally somewhere between 0.3 m and 0.5 m thick, although some areas were as thin as 0.1 m and others reached thicknesses of 0.8 m. The area is therefore not strictly blanket bog, and is thus not directly analogous with the study plots at Moor House, although the vascular-plant cover on the two sites probably does not differ greatly.

Three stages of heather growth were identified – burnt within the last month, burnt 12-15 years ago, and not burnt during 25 years of growth. Fresh fire patches in each of these categories were used to compare the carbon losses resulting from fire compared with the carbon stored in the vegetation within the remainder of that age/growth-category.

Farage *et al*. (2009) calculate that carbon losses arising directly from the burning of vegetation amount to 100-200 g C m^{-2}. They point out that when these losses are spread across the 15 year timescale of the burning rotation and compared with published figures for other annual carbon losses from the system such as respiration and methane emissions, the loss due to combustion of vegetation represents less than 10% of all losses. Farage *et al*. (2009) therefore suggest that well-managed burning on an upland heather moorland may not have the major detrimental effect which is often assumed. They do observe, however, that there are risks involved in the use of fire as a management tool, and that fire which damages peat soils causes release of carbon from the long-term carbon store and harms the natural biodiversity of the system.

Of particular relevance to the question of biochar, however, are the volumes of material consumed in such fire management. Farage *et al*. (2009) calculate that the quantity of vegetation consumed in a managed fire on an upland heath amounts to 100-200 g C m^{-2}, and that this therefore represents approximately 7.5 g C m^{-2} yr^{-1} on a 20-year burning cycle (the timescale favoured by Worrall and Clay : undated poster). This represents only 1/3 of the quantity of carbon transferred annually from the acrotelm to the catotelm according to the rather conservative figures given in the 'standard natural cubic metre of peat' shown in Figure 24. In other words, given a choice between the use of normal acrotelm dynamics or of biochar to accumulate long-term carbon, the acrotelm option is potentially 3x more effective than biochar, according to the data obtained by Farage *et al*. (2009).

In addition, the observation by Farage *et al*. (2009) concerning the potentially damaging effects of fire on the natural biodiversity of an organic system is an important one, because there is always the potential for damage to the peat-forming system when fire is used. Accidents will happen from time to time, but even if there are no accidents, it is inevitable that there will be localised hot spots during the burning process, for example on hummocks.

Fire is harmful to hummock-forming *Sphagnum* species in particular because heat rises, fire naturally burns upwards, and the hummocks tend to support more growth of woody tissue which will burn hotter, and for longer. If fire is established as a regular management tool on a blanket bog site, there is therefore the danger that hummock-forming *Sphagnum* species will be lost from the system, or prevented from becoming established on sites where a more natural bog vegetation is being encouraged to develop. Underpinning all of these considerations, of course, must be a clear awareness of the long ecosystem-recovery times required for the blanket bog habitat should fire cause serious damage.

18.4.1.10 Yallop and Clutterbuck (2009) : burning and DOC

In a study which embraced 50 catchments in the south-central Pennines, Yallop and Clutterbuck (2009) examined the relationship between DOC export and a variety of catchment features such as % peat cover, extent of moorland drainage, and extent of burning within each catchment. They found that a good correlation could be identified between the extent of peat within a catchment and the level of DOC export – the greater the area of peat, the greater the levels of exported DOC.

This relationship between peat cover and DOC export has been widely reported by others, but far more unexpected was the even closer correlation observed by Yallop and Clutterbuck (2009) between the extent of burning within the catchment and the levels of DOC export. This relationship was sufficiently strong to over-ride or at least modify the established link between extent of peat soils and DOC export. The strongest correlations were observed between DOC and 'Class 1' burn areas (0-4 years old), then with 'Class 2' burns (3-8 years old).

It is perhaps unfortunate that the only catchment completely dominated by blanket bog and with, perhaps coincidentally, the highest proportion of 'Class 3 & 4' burns (Keighley Moor) did not have a complete data run and could not therefore be included in the calculation of the annual mean DOC levels. Nonetheless the evident relationship observed between younger burn scars and levels of DOC export suggests that the question of whether burning increases or decreases DOC requires much further work.

As to whether the DOC is derived from relatively young plant litter or from the long-term carbon store, Yallop and Clutterbuck (2009) note that they observed low levels of DOC from catchments in the North York Moors, which had little peat but which had suffered burning. From this, Yallop and Clutterbuck (2009) conclude that the exported DOC is most likely derived from the carbon store in the catotelm.

Then again, a young fire undoubtedly results in the subsequent decomposition of many roots and leaf-bases left within the surface layer (acrotelm/haplotelm) of the peat. If the particular conditions of this layer contribute to the formation of DOC from such breakdown products, the observations made by Yallop and Clutterbuck (2009) do not necessarily conflict with the idea of DOC as a product of vascular-plant litter rather than being derived from the catotelm store.

Complicating this story is the fact that DOC occurs in several different forms, some hydrophobic, some hydrophilic, and it may be that differing forms are produced by differing processes – indeed it would seem quite likely that this could be the case. One way to shed further light on this question would be to measure the age of the carbon in the DOC. Young carbon would suggest that DOC is a product of the acrotelm/haplotelm; old carbon would indicate that DOC is most likely being produced in the catotelm.

18.5 Fire, carbon and *Sphagnum*

Ultimately, the key to understanding the impact of fire on peat bog systems, particularly blanket bogs, depends on two things:

- an appreciation of likely ecosystem-recovery times; and
- an understanding of the differing effect fire has on the two key cycles – the *Sphagnum* cycle and the vascular-plant cycle – discussed earlier in Section 8.4 and Figure 25.

As long as a fire impacts only on the vascular-plant cycle of a blanket bog, the ecosystem is likely to show fairly rapid recovery and remain relatively stable. There may, however, be implications for factors such as DOC export and methane emissions (see Figure 54).

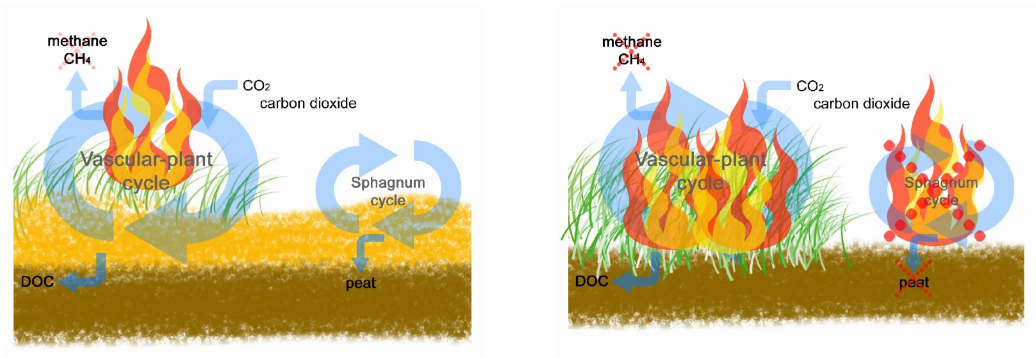


Figure 54. Fire in the vascular-plant cycle, and in the *Sphagnum* cycle.
The diagram on the left illustrates the fact that if a fire is restricted to the above-ground parts of the vascular-plant cover, the main peat-forming process within the *Sphagnum* cycle remains unaffected. DOC export may continue from vascular-plant roots and methane emissions may diminish somewhat, but peat accumulation continues. If fire spreads into the *Sphagnum* layer (right), the major peat-forming *Sphagnum* cycle is damaged or destroyed. If the acrotelm (orange shading) were to be lost as a result of a fire which burns deep into the peat, then the catotelm peat (brown shading) would become exposed to a range of powerful erosive forces.

If, on the other hand, a fire should disrupt or break the *Sphagnum* cycle, the system is likely to become subject to a series of important changes which can result in a degenerative sequence which may take decades or even centuries to reverse.

Thus loss of *Sphagnum* means that the peat surface becomes directly exposed to the weathering and erosive forces of rain, frost, sun, wind and physical disturbance by grazing animals. Even a micro-erosion complex such as that shown in Appendix 3, Figure 85 may take many decades, if not centuries, to re-establish a living acrotelm, while formation of deeper erosion gullies may require millennial periods of regeneration to infill the gullies once (if) the surface begins to re-colonise with *Sphagnum*.

18.6 Actions and research needs

18.6.1 Burning, *Sphagnum* and heather

Fire tends to favour vascular-plant cover at the expense of *Sphagnum* cover, with heather being encouraged in the east of Britain while in the west and north, species such as deer grass and purple moor grass are encouraged. The majority of research into fire and vegetation dynamics has tended to focus on the relationship between fire and heather and/or grassland management, rather than the relationship between fire and *Sphagnum* cover.

The relationship between deep peat, *Sphagnum* and heather cover should be examined with a view to determining likely *long-term* trajectories for the ecological character of areas under differing burning regimes and changing climatic conditions. In particular, the relationship between fire and *Sphagnum*, rather than fire and heather, should be given specific, detailed attention.

Studies of burning, whether in terms of fire frequency or in terms of carbon balance, should include detailed descriptions of the ecological character and ecological condition of the bog system, including in particular details of the *Sphagnum* cover, its species composition, and the microtopography of the bog surfaces involved.

18.6.2 Recovery times and sequences after fire

The timescales of recovery in blanket mire environments are poorly documented, but the little evidence which does exist for peatbog systems in both lowland and upland environments suggests that recovery times are substantially under-estimated, particularly in the harsher blanket mire environment. Recovery times to original ecological condition following fire should be investigated, particularly in a range of blanket mire environments.

Detailed palaeo-archival studies can shed light on the natural dynamics of fire events and the record of vegetation recovery following fire. Such studies should be undertaken across the range of available time-spans, and in a range of geographical areas, from the Mesolithic Period to the present day.

18.6.3 Fire, carbon balance and DOC

Confusing and conflicting messages have been emerging recently about the effect of fire on the carbon store of, and DOC-release from, blanket bogs. This confusion appears to result in part because the areas under investigation are not adequately described in terms of their ecological character and ecological condition.

Future GHG-flux and DOC studies in blanket mire habitats should ensure that detailed descriptions are obtained of the ecological character and ecological condition of the bog system, including in particular details of the *Sphagnum* cover, its species composition, and the microtopography of the bog surfaces involved.

The age of the carbon involved in both GHG-flux and DOC studies should be determined in an attempt to identify whether these losses are from the carbon store or from recent plant litter and root systems.

19 DISCUSSION TOPIC 8
Erosion in blanket bogs

19.1 Introduction

Before plant life began colonising dry land during the late Silurian Period some 420 million years ago, the Earth's surface was subject to direct and continuous attack by rain, frost, wind and sun. Erosion rates in those times must have been very considerable. However, with the development of a terrestrial plant cover the mineral soil became increasingly protected from direct erosive attack by a blanket of vegetation. If anything, this protective mat probably increased the rate of chemical and physical weathering (*i.e.* breakdown of the mineral substrate), but rates of erosion (*i.e.* physical transport of this broken material) will have been substantially reduced. Thus rainforests now protect easily erodable laterite oxisols from the heavy tropical rains, while continuous grassland swards protect prairie soils from powerful winds.

Peatlands represent probably the most extreme form of such a protective vegetation mantle. Many peatland systems in the northern temperate and boreal zones overlie extensive tracts of glacial till which was deposited during earlier ice ages. In particular, the blanket mires of Britain, Ireland, Norway and the Faeroes protect whole landscapes of mixed glacial till which would otherwise be subject to the continuous and often heavy rains characteristic of such oceanic regions.

Without the protective mantle of blanket mire, many such till deposits would soon be eroded to bedrock. The peat-forming process has a natural tendency to reduce these destructive forces through the action of biological growth, *Sphagnum* carpets forming a protective continuous layer which is well structured to minimise and diffuse the power of these forces, particularly heavy rain (see Appendix 3, Figure 63).

It is all the more extraordinary, therefore, to find that these blanket mires themselves should be so strongly associated with erosion. One of the most striking things about British and Irish blanket peat is the extent to which severe erosion dominates so much of the resource. Erosion in places such as Kinder Scout in the Peak District mean that only isolated slabs of peat remain, the wide gullies between representing a huge volume of peat which has simply been lost from the hill-top (see Figure 55). To quote Verona Conway:

> *"Few people can see severe peat erosion for the first time without a sense of astonishment, and the curious-minded will wonder why there is any peat still left when it is obviously exposed to such destructive forces."*
>
> *Conway (1954) p.117*

Do images such as Figure 55 mean that blanket peat is not, in fact, capable of resisting the erosive forces typical of such upland conditions? Does blanket mire face the same inevitable fate as the underlying glacial till would have done had the blanket mire not formed some 3,000+ years ago? Lilly *et al*. (2009) appear to suggest so when they state that:

> *"...once upland peatland erosion is initiated, in many instances, there is an inexorable tendency towards almost complete loss of the accumulated peat over time. There are limited cost-effective options available to redress this trend."*
>
> *Lilly et al. (2009) p.iv*

It is worth observing that evidence presented later in this section suggests that this may be an overly-pessimistic view of peatland erosion dynamics.

Perhaps just as surprising as finding that this naturally-protective system should itself be subject to such severe erosion is the fact that we simply do not know whether such blanket peat erosion is natural or not. More than 100 years after the first scientific accounts of peat bog erosion, a straw poll of peatland scientists and conservationists attending a conference about peatland erosion found that the majority believed such erosion could be natural. A subsequent questionnaire, however, resulted in only 28% of respondents considering that natural causes (including climate change) were the primary cause of peatland erosion (Immirzi, Meade and Ramsay, 1997).

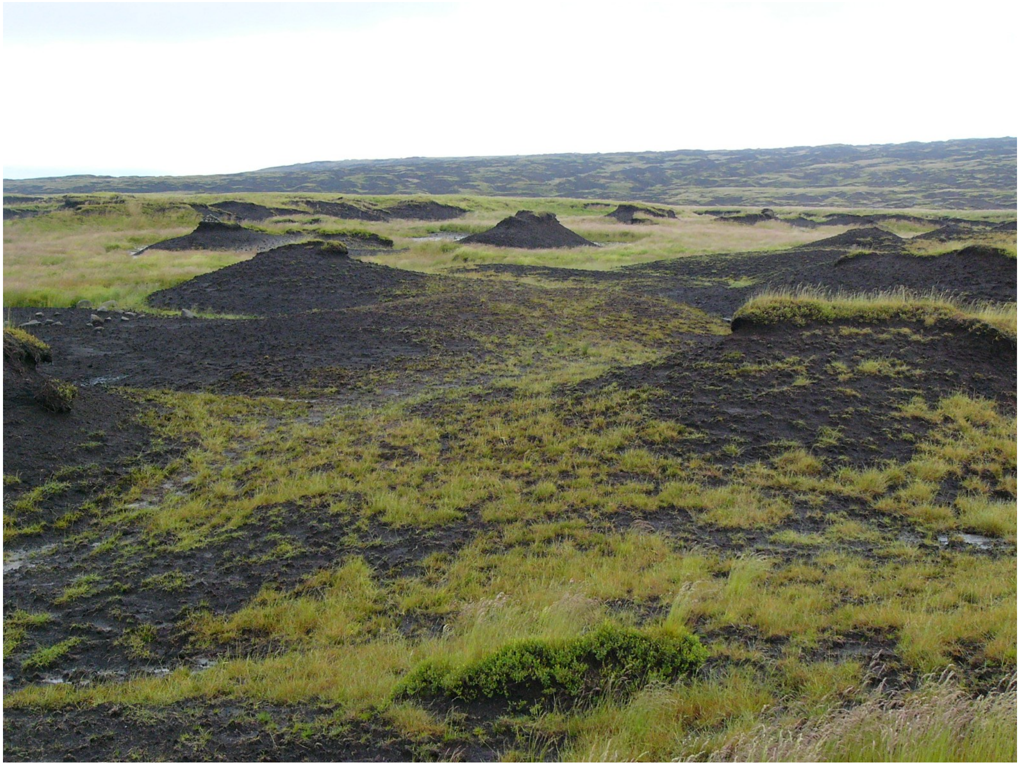


Figure 55. Blanket mire erosion at Kinder Scout, Peak District.
Remnants of the original peat thickness can be seen as isolated islands, surrounded by expanses of ground which have now largely lost their peatland cover. The low-lying channel running from bottom left to top right has almost reached the glacial till which underlies the peat. The huge quantity of peat lost from this watershed ridge through erosion is sufficiently obvious to require little further comment.

Photo R A Lindsay

The deep-seated belief that peatland erosion may be a natural process has an interesting history in the scientific literature – a history worth exploring in order to understand what is commonly assumed about peatland erosion. Firstly, however, it would be useful to establish the extent and severity of such erosion in the UK.

19.2 The extent of blanket mire erosion in the UK

There are no complete, unified figures for the extent of eroding blanket bog in Britain. Based on an estimated extent of 2,250,000 ha for the extent of blanket mires in Britain, Tallis (1998) suggests that 350,000 ha (16%) is eroded. It is worth noting that the estimate of the total resource used by Tallis (1998) falls within the main body of estimates given for the UK shown in Section 3.1, Figure 3 of the present report, and would therefore appear reasonable. His estimate for the extent of erosion, however, is based on an amalgamation of various sources, few of which are directly comparable.

In the absence of consistent figures for peatland erosion across the UK, it is necessary to piece together information for particular countries, regions or studies, and then compare equivalent data where possible.

The identification and mapping of erosion also represents something of a challenge because erosion can occur to varying degrees and take various forms. Thus deep E2 gullies of the type illustrated in Appendix 3, Figure 84 are easy to see on aerial photographs (though interestingly erosion gullies are not marked on Ordnance Survey maps). If this is just a single gully running across an expanse of unbroken ground as indeed is the case in Figure 84, how is this mapped or measured in comparison with a slope which is riven with such gullies? Grieve, Davidson and Gordon (1995) discuss such difficulties in relation to their review of erosion in the uplands of Scotland.

The issue becomes even more complicated in the case of E1 regenerating gullies because while many such gullies are reasonably evident on aerial photographs, examples of the type illustrated in Appendix 3, Figure 83, are less so.

Finally there is the question of Em micro-erosion (see Appendix 3, Figure 85). To what extent should this be included within maps and estimates of erosion, especially where there has been a degree of vegetation recovery?

19.2.1 Bower (1960, 1961) : a classification of erosion

In a study and review of erosion in the Pennines, Bower (1960, 1961) identified that peat erosion occurred in a variety of ways through the action of water or because there was mass movement of the peat. She recognised three differing types of erosion driven by the action of water: dissection, sheet erosion and erosion of marginal peat faces. She considered dissection erosion to be the most important of these.

Bower (1960, 1961) defined two major types of dissection erosion based on characteristics of erosion gullies which were evident from aerial photographs. She identified 'Type 1' erosion as a type restricted to ground which is level or very gently sloping. It consists of a maze of interconnected gullies and was found to occur in association with peat deposits of at least 1.75 m depth. Bower (1961) described Type 1 erosion as a pattern apparently derived from the breakdown of hummock-pool systems formed on watershed summits. Type 1 erosion therefore tended to occur at higher altitude than Type 2 erosion.

'Type 2' erosion consisted of individual gullies which ran fairly straight down-slope and having few, if any, linkages with other gullies. Where there are many such gullies on a slope they tend to run parallel with other rather than connecting to each other. Type 2 erosion was found most commonly on ground where the slope was greater than 5°, but generally showed no particular constraints in terms of peat depth or landscape location.

These two classes of erosion have since been used in a number of erosion studies. Bower (1961) herself only recognised erosion (or in her terminology 'dissection') where two or more gullies lay closer together than 110 m (120 yds). Individual isolated gullies were not therefore included in her study.

19.2.2 The extent and severity of peat erosion in England

19.2.2.1 Bower (1961) : the Pennine chain

Bower (1961) provides relatively detailed maps of the blanket mires covering the whole of the Pennine chain indicating which areas have Type 1 gullies, which have Type 2 gullies, and also which areas remain free from erosion. Unfortunately she does not provide any area-estimates for these categories. All that can be deduced is that the area of un-eroded peat probably exceeds the area of Type 1 and Type 2 erosion combined, and that Type 2 appears to be somewhat more extensive than Type 1 erosion.

19.2.2.2 Anderson and Tallis (1981) : southern Pennines

Anderson and Tallis (1981) estimate the extent of erosion within the southern Pennines – one of the study regions mapped by Bower (1961) – and conclude that, of the 30,000 ha of blanket peat within this area, something in the order of 22,500 ha (75%) is eroded. This proportion is very much higher than the 16% given by Tallis (1998) as the proportion of eroded blanket mire within the resource as a whole, although Tallis (1985) comments that the scale of erosion in the southern Pennines is "probably without parallel elsewhere in Britain". As we shall see, there may be at least one other part of Britain with equivalent levels of erosion.

19.2.2.3 Burt and Labadz (1990) : southern Pennines

In an overview of peat erosion in the southern Pennines, Burt and Labadz (1990) cite figures for peatland erosion assembled for the Peak District National Park, giving:

- 3,940 ha of uneroded peat (26% of the total peat area);
- 1,960 ha of Type 1 erosion (13%);
- 9,230 ha of Type 2 erosion (61%); and additionally
- 1,700 ha (11%) of the above total consisted of bare peat.

These figures suggest that erosion occurs across 74% of the blanket mire resource within the National Park, and that a significant proportion of bare peat exists as a result.

Ivanov (1981, p.215) comments that the fibrous nature of peat, when in a state of limited or moderate decomposition, makes it highly resistant to water erosion compared to most mineral soils. The presence of bare peat *per se* does not therefore mean that the bare areas will erode rapidly, but as it dries out and becomes more decomposed such peat will become progressively less capable of resisting the erosive forces of water, wind and ice.

As for the severity of this erosion, Burt and Labadz (1990) calculate that the peat thickness removed annually from gullies is between 0.6 and 1.6 mm^{-1} yr^{-1}. At a bulk density of 0.1 g cm^{3}, the average rate of peat loss is equivalent to a carbon loss of 53 g C m^{-2} yr^{-1}, or 0.53 t C ha^{-1} yr^{-1}. This is rather more than 2x the carbon-accumulation rate assumed in the 'standard natural cubic metre of peat' presented in Section 8, Figure 24. In other words, the figures presented suggest that eroding gullies are losing peat 2x faster than it would be possible to replace even in a natural bog.

An eroding bog is not likely to be accumulating peat at this rate, however, because the gullies act as drains and extensive areas will therefore be subject to water-table draw-down. The significance of such losses for the carbon balance of the bog is that the 'carbon replenishment times' for these eroding gullies are likely to be at least twice the span of time over which these loss rates have been sustained. Clearly, the sooner any actively-eroding E2 gullies can be transformed into E1 regenerating gullies, the better for the carbon balance of the site.

19.2.2.4 Tallis (1997) : a southern Pennine overview of severity

In his review of what is known about the history, pattern and causes of erosion in the southern Pennines, Tallis (1997) examines the processes of blanket mire erosion but does not give figures for the extent of erosion, either for the southern Pennines or for British blanket mires as a whole. He does, however, present figures for the rate of peat removal in different microtope-nanotope settings, and these offer an even more thought-provoking picture than the values given by Burt and Labadz (1990) above.

The annual rates for surface lowering given by Tallis (1997) are:

- gently-sloping bare peat 28.7 mm yr^{-1} (16.0-53.1
- sides of gullies 11.9 mm yr^{-1} (7.8-20.4)
- gully floors 5.5 mm yr^{-1} (2.5-8.1)

The most immediately-obvious feature of these values is that they are mostly an order of magnitude larger than the figures given by Burt and Labadz (1990) for the same general area. This may be explained by the fact that Burt and Labadz (1990) presented calculations based on carbon outflows from the system into streams and reservoirs, and they themselves highlighted the fact that significant quantities of mobilised peat could be missed using such methods. The bulk of the values which form the basis of the figures given by Tallis (1997) were derived from direct measurement of surface lowering. As such, Tallis (1997) probably gives a more reliable idea of the true picture.

It is worth noting, for example, that Birnie (1993) recorded rates of surface lowering for areas of bare peat in Shetland which span the values given by Tallis (1997) for bare peat, Birnie (1993) noting values of between 15-37 mm yr^{-1}.

The implications of these figures for the carbon budget of an eroding blanket bog are considerable. Assuming a dry bulk density of 0.1 g cm^3 (which may be an under-estimate for such drained peat), the figures given above can be transposed into quantities of carbon loss as follows:

- gently-sloping bare peat 1392 g C m^{-2} yr^{-1}
- sides of gullies 577 g C m^{-2} yr^{-1}
- gully floors 267 g C m^{-2} yr^{-1}

Thus even the lowest loss-rates, from the gully floor, represent a 10-fold flow of carbon from the system compared to what a natural system might accumulate. If the bog were storing carbon at the high rate observed by Dinsmore *et al*. (in press) at Auchencorth Moss, the gully floors would still be losing carbon more than 2.5x faster than the bog was storing it.

The sides of the gullies (perhaps rather counter-intuitively) are losing peat at twice the rate of the gully floors, and 23x faster than the accumulation rate of the 'standard natural cubic metre of peat' shown in Section 8, Figure 24. Most thought-provoking of all is the fact that gently-sloping areas of bare peat are losing carbon 55x faster than this natural system can accumulate peat. The loss is 15x faster even than the rapid rate of carbon accumulation noted by Dinsmore *et al*. (in press) for Auchencorth Moss.

Essentially, such rates of carbon loss cannot be matched by known accumulation rates. The scale of the gully and peat-flat system compared to the non-eroding parts of the bog is a critical issue, as is the fact that many parts of the non-eroding bog may not be accumulating carbon because of drainage effects from the gullies. It therefore seems that in order simply to match the carbon losses from erosion, an expanse of bog still capable of reasonably natural carbon accumulation would need to cover an area which was at least 10x the area of the gully floors, 23x the area of the gully sides, and 55x the area of any wide expanses of bare peat.

19.2.2.5 Garnett and Adamson (1997) : Moor House, north Pennines

Garnett and Adamson (1997) provide an overview of erosion within Moor House National Nature Reserve in the northern Pennines, estimating that just under 10% (173 ha) of the blanket mire in the NNR is actively eroding, just less than 12% (207 ha) is erosion which is re-vegetating, and 74% (1,381 ha) is described as 'pristine'. It is interesting to note that the mean slope-angle for this pristine blanket mire is greater than the slopes for the two eroded categories, suggesting that the picture is very much as Bower (1961) recorded for the Pennines as a whole – namely that the summits and immediately-surrounding slopes are most eroded, while steeper areas of peat on the lower flanks of the hills are not so eroded.

It seems likely that Garnett and Adamson (1997) restrict their definition of 'eroding' to obvious presence of gullies, much as Bower (1960, 1961) did for her survey. It is thus likely that much ground which could be classed as 'Em' micro-erosion exists within the area defined by Garnett and Adamson (1997) as 'pristine'. Certainly the present author's experience of survey at Moor House suggests that little could be described as truly 'pristine' whereas much could be described as either micro-erosion or recovering micro-erosion. This distinction is important in carbon terms because a bog which is genuinely pristine is likely to have a different carbon mass-balance from one which is in various states of micro-erosion.

19.2.3 The extent and severity of peat erosion in Wales

Yeo (1997) discusses the extent of blanket peat in Wales, and although he does not give a figure for the amount of eroded bog, he does comment that gully erosion is very localised on Berwyn and Migneint in the northern part of the mid-Wales uplands, but becomes steadily more intense towards the south. In the Brecon Beacons at the southern limit of these uplands, more than 80% of the peat above 457 m (1500 ft) was found to be eroded.

Rates of erosion have been measured for Pumlumon in mid-Wales, ranging from 0.3 to 1.32 mm^{-1} yr^{-1}, and are thus only slightly lower than those recorded by Burt and Labadz (1990) for the Peak District. The average loss equates to 39 g C m^{-2} yr^{-1}.

19.2.4 The extent and severity of peat erosion in Scotland

19.2.4.1 Hulme and Blyth (1985) : extreme rates of peat erosion

While the erosion rates discussed by Burt and Labadz (1990) and Yeo (1997) reflect the overall rates of loss which may occur within a peat-dominated catchment, Hulme and Blyth (1985) witnessed an example of specific loss which emphasises the significance that a single event can have within the overall carbon balance of a site.

Hulme and Blyth (1985) were undertaking survey on the island of Yell, in the Shetlands, when a powerful convective storm occurred following a long period of dry weather. As a result of the dry weather the bottoms of most erosion gullies had developed a crust of dry peat which had then broken into polygonal plates in the manner of cracked, dry mud.

With the onset of the storm with its heavy rainfall, these erosion gullies rapidly filled with water and within 30 minutes the plates of peat were observed to lift and float off down the gullies, carried by the flow of water. The photographs provided (albeit with their Fig. 2 and Fig. 3 transposed) show the process clearly and dramatically. Hulme and Blyth (1985) estimated that a thickness of up to 20 mm had been lost in this way from the gully bottoms within the space of an hour. This represents a loss of up to 970 g C m^{-2} in this one event.

As Hulme and Blyth (1985) observe, such events, though infrequent, may represent the most significant mechanism by which peat (and thus carbon) is lost from the eroding bog. With climate models predicting more frequent periods of dry weather followed by storms, this may become an even more significant factor in the carbon budget of eroding sites and thus there may be even greater incentive to stabilise and re-vegetate eroding gullies.

19.2.4.2 Grieve *et al*. (1994, 1995) : review of upland erosion in Scotland

Grieve *et al*. (1994, 1995) examined a 20% sample of the Scottish uplands using aerial photographs, looking specifically for erosional features. They found evidence of such features in 12% of their study areas. Within this eroded ground, peat erosion accounted for 50% of the observed erosion.

Grieve *et al*. (1994, 1995) found that erosion was most extensive within the Monadhliath Mountains, but most intensive in the eastern Grampians and eastern Southern Uplands. It is important to note that Shetland was not included in the sampling programme. Grieve *et al*. (1994, 1995) also observed that eroded peat, gullied areas, debris flows and sheet erosion all occurred most extensively at altitudes above 550 m. This link between peat erosion and altitude was also noted by Bower (1961, 1962) in the Pennines.

19.2.4.3 Coupar, Immirzi and Reid (1997) : regional patterns of erosion

Coupar *et al*. (1997) provide a regionally-based review of peat erosion in Scotland, highlighting the differences between Scottish regions in terms of extent and intensity of erosion. Coupar *et al*. (1997) consider only blocks of blanket mire greater than 10 ha, but they assess the extent of gully erosion within each of these blocks. Their survey embraced the whole of Scotland.

The results of this survey reveal that 36% of extensive Scottish blanket peat shows signs of gully erosion. The survey highlights the fact that not only is Shetland the most eroded Scottish region, but the proportion of eroded ground (77% of the peat in Shetland) matches that of the worst areas of erosion in the Pennines, in particular the erosion in the Peak District. The next most eroded areas were found to be Grampian and Tayside, each with 49% of the blanket peat showing signs of gully erosion, while the Scottish Borders were next with 36%. Perhaps surprisingly, the west side of Scotland was generally associated with lower values, despite having more peat and higher rainfall.

Coupar *et al*. (1997) also highlight the fact that there can be considerable variation within a region, demonstrating that, in Highland Region, 76% of extensive blanket mire was eroded, whereas in Skye and Lochalsh the value was 16% and in Caithness it was a mere 0.2% of the blanket mire resource.

As Coupar *et al*. (1997) note themselves, their survey focuses only on the obvious signs of erosion and cannot provide information about less intense forms of erosion such as micro-erosion. Nonetheless, their data make it clear that high-altitude blanket mires are more likely to show significant signs of erosion than low-altitude examples, although this relationship becomes less clear in Shetland.

19.2.4.4 Lilly *et al*. (2009) : erosion review for Scotland (and Northern Ireland)

Lilly *et al.* (2009) record the fact that 34% of blanket bog in Scotland was classed by the Land Cover of Scotland 1988 (LCS88) as being in an eroded state, and re-state the findings of Grieve *et al*. (1994) that 50% of erosion recorded within a 20% sample of the Scottish uplands was associated with blanket mire.

Lilly *et al*. (2009) also summarise a range of data for Northern Ireland, noting that 14% of the peatlands in Northern Ireland have signs of erosion.

19.2.5 Extent and rate of blanket mire erosion : a summary

Overall, then, it is clear that gully-style erosion is an extensive feature of many blanket mire areas in both Britain and Ireland, although there is considerable regional variation. Blanket bog at high altitude is more likely to be eroded than bog at lower altitude, but the extent of less evident forms of erosion such as micro-erosion remains largely unknown.

Rates of peat loss from erosion gullies can range from 2x the (rather conservative) carbon-sequestering rate used in the present report for a natural bog, to rates which may rise to more than 40x this sequestering rate during storm events, and potentially reaching rates of 55x the carbon-sequestering rate on wide expanses of bare peat. Although erosion gullies cover only a proportion of the ground and therefore the overall particulate losses from a blanket mire may be constrained by this, the impact of gullies on the surrounding bog can still be extensive. This is particularly the case with Type 1 erosion, or with Type 2 erosion where the gullies are closely spaced, because the gullies will act as drains and result in water-table draw-down within the intervening un-eroded bog, thereby leading to further carbon losses.

19.3 The origins of 'peatland erosion as a natural process'

19.3.1 Natural erosion – a key question for conservation

As Pearsall, (1950), Lindsay *et al*. (1988), Lindsay (1995), Tallis (1995a) and Evans and Warburton (2007) point out, a profound conceptual question sits at the heart of ideas about peatland erosion. If erosion is natural, is it appropriate to talk of, and think of, eroded blanket mire as 'degraded'? Would one describe the famously-eroding sea cliffs of Happisburgh as a 'severely degraded coastal system'? Dynamic, perhaps, but 'degraded' does not seem the appropriate word to use.

Perhaps more importantly, if blanket bog erosion is natural, might not attempts to reverse this natural process through expensive conservation management be inappropriate? If there is a natural cycle of erosion and re-growth, as hinted at by Bower (1962), it would seem that any conservation action designed to break this cycle is akin to expensive coastal-defence works which are designed to prevent the natural movement of coastal sediments from particular sections of coast. All this does, of course, is ultimately to cause more rapid erosion from other sections of coast lying within the coastal sediment circulation-cell. If erosion is natural, should not conservation bodies be identifying types, and stages, of erosion and actively conserving these as 'type' examples of the natural cycle?

Conversely, if a 'natural' cycle of erosion and re-growth does *not* exist and the vast majority of erosion is anthropogenic in origin, then the very extensive areas of blanket mire erosion in the UK and Ireland should be viewed in a very different light. If they truly represent 'degraded' habitat then the potential losses in terms of biodiversity, ecosystem services and carbon from such damaged areas make positive conservation action to restore these areas and prevent further erosion an extremely high priority.

The difficulty at present is that mixed messages exist about the questions posed above. When discussing the causes of erosion, conclusions are almost always prefaced by comments to the effect that blanket mire erosion is probably, fundamentally, a natural process but one which can also be triggered or exacerbated by certain land-management actions or other factors. The suggestion that erosion may in some fundamental way be natural has a tendency to undermine arguments for urgent positive conservation action. Equally, when concerted conservation action is undertaken and eroded systems are described as 'severely degraded', this does not sit logically with the idea that blanket bog erosion is considered by some to be fundamentally driven by natural processes.

This dilemma arises from three sets of circumstances. Firstly, we do not yet know what the key triggers for blanket bog erosion are - or rather 'were' in many cases. As Lilly *et al*. (2009) point out, because much erosion appears to be centuries old, the origins of such erosion may need to be sought from the past rather than the present landscape. Secondly, these triggers may prove to be a combination of things, some of which may be natural processes, so there may genuinely be a mixed message in the explanation.

Finally, and possibly the key source of this dilemma, is an idea which was developed almost a century ago and which has shown extraordinary persistence as a concept despite the absence of clear, unequivocal supporting evidence, either then or now. This idea has been so persistent that it has virtually become the default option in relation to eroding blanket mire, with other options regarded largely as possible contributory factors. This long-held idea states that blanket bogs are in some way the architects of their own destruction, either through their own internal processes or because natural external forces will ultimately bring about their collapse.

This idea has taken various forms over time, and has also separated into two key themes: the processes which *initiate* or *trigger* blanket mire erosion, and the processes which then *drive* the erosion process. It is probably fair to say that there is widespread agreement about the driving factors, but the initial triggers of erosion continue to stimulate much debate, particularly about whether these triggers are natural or not.

Given that the question of 'naturalness' is such a key issue for conservation, it is worth looking at the development of the ideas about blanket mire erosion over the last 50-100 years because it is still possible to encounter any or all of these arguments today.

19.3.2 Weber (1902) : hummocks and pools as constructive features

More than 100 years ago, Weber (1902) recognised that the pattern of pools and hummocks which he observed scattered across the Augstumal raised bog represented an arrangement of features which could respond to climate change by shifts in their relative proportions. He likened this to a pulsating organism responding to its environment. Weber's (1902) deeply perceptive observations are explored in more detail in Appendix 3, Section 24.1, but for the present it is sufficient to note that he saw the pattern of pools, ridges and hummocks as a reactive, positive and constructive mechanism which provided resilience to the bog system in the face of changing climatic conditions.

19.3.3 Osvald (1923) : hummocks and pools as destructive features

Twenty years after Weber (1902) had observed the hummock-hollow pattern of the Augstumal mire and seen a dynamic, pulsating organism, Osvald (1923) looked at the hummock-hollow patterns of Komosse in Sweden and saw only destruction and degradation. Osvald (1923) set out a degenerative view of the hummock-hollow pattern which has since proved extraordinarily durable, even though it is now known to be incorrect.

Again, the details of this are explored in Appendix 3, Section 24.1, but essentially Osvald (1923) saw pools as sites of hummocks which had grown too high above the bog water table and so subsequently degenerated. Equally these pools had the potential to grow vigorously into hummocks but these too, in time, would degenerate into pools again. This 'hummock-hollow regeneration complex' appears to have coloured all of Osvald's subsequent thinking because almost wherever he looked he saw signs of natural and inevitable degeneration or breakdown.

His subsequent account of Norwegian 'oceanic raised mire' (more correctly oceanic blanket mire) dwells on the natural breakdown processes (as he saw them) in these mire systems (Osvald, 1925), and he developed these ideas further in the company of Sir Arthur Tansley and Sir Harry Godwin during a field excursion through the blanket mires of Britain and Ireland (Osvald, 1949).

In particular, Osvald (1925, 1949) saw evidence for pool collapse when intervening ridges were eroded by wind-waves, while in other cases pools collapsed due to the development of underground drainage channels. In his views and ideas, which were then embraced enthusiastically by Tansley and Godwin, the phenomenon of blanket mire erosion and subsequent collapse was inevitable and natural.

19.3.4 Pearsall (1950) : internal collapse and unstable growth

Pearsall (1950) does not cite Osvald, though he does cite Tansley (1939), in his overview of peat erosion in British blanket mires. Pearsall (1950) considers that peat erosion can be caused by a variety of mechanisms. He describes how head-ward erosion of stream gullies can steadily eat back into the peat which covers watershed ridges, causing the peat increasingly to drain into such headwater streams. Pearsall (1950) also cites internal peatland drainage systems which can disappear underground and subsequently cause collapse of the overlying peat. He additionally describes how some peatland sites can be de-stabilised when they are super-saturated with water and can then 'burst' in a bog flow.

Lastly, Pearsall (1950) describes how periods of rapid peat accumulation during wet phases in the climate probably cause areas of blanket peat to over-step their physical boundaries and thus result in breakdown and erosion of this wet peat. Pearsall (1950) offers specific examples for the collapse of internal drainage and for bog bursts, but gives no examples of site which have overstepped their physical boundaries.

Interestingly, Pearsall (1950) is also the first to observe that there are differences of opinion about whether peatland erosion is a natural process of landscape erosion, or whether it results from drying caused either by a changing climate or by human action such as burning or drainage.

19.3.5 Conway (1954) : vigorous unstable growth

In the introduction to her analysis of several peat cores taken from blanket bog sites in the Kinder Scout area of the southern Pennines, Conway (1954) addresses the question of why these areas are so eroded. She is quite clear in her reasons. She observes that the uppermost layers of most deep deposits in the area consists of soft peat which has clearly accumulated during a period of rapid peat growth in the last 2,500 years. This peat is described as evidently unstable and liable to collapse through mechanisms described by Pearsall (1950) "so that no more need be said here." Except that Pearsall (1950) presents no specific evidence to support the idea of vigorous growth leading to collapse.

Conway (1954) goes on to describe how the wet climate since the early Iron Age has led to such rapid peat accumulation that mechanical breakdown of the peat on convex watershed ridges becomes inevitable. Rupture of the vegetation surface by drainage channels in the wetter peat leads eventually to exposure and erosion of the deeper peats, resulting in complete erosive breakdown of the system.

This is a potentially plausible description of what may have happened, but no evidence is presented to support it, although Conway (1954) states that "the results of pollen analysis have confirmed this conclusion". In fact the pollen work simply confirms that the climate was wetter and that the peat accumulated rapidly. There is no evidence for the proposed breakdown mechanisms as described.

19.3.6 Bower (1960, 1961, 1962) : studies and reviews of blanket mire erosion in the Pennines

The studies by Bower (1960, 1961, 1962), already referred to in Section 19.2 above, lead her to conclude that the prevalence of extensive erosion at the highest altitudes indicates a close link to climate. She proposes that the wetter climate at these altitudes gives rise to more rapid peat accumulation which then reaches the limits of stability proposed by Conway (1954) more rapidly than other parts of the peat mantle.

Although acknowledging that biotic factors are sometimes cited as the cause of erosion, Bower (1961) finds no convincing evidence of a link between signs of biotic influence and the observed pattern of erosion. She does not examine evidence of fire damage in the recent peat archive, nor does she consider potential recovery times from damage at these altitudes, but Bower (1961) does note that burning can have a "particularly drastic effect" on the system because loss of vegetation cover exposes the underlying peat to the erosive forces of rain, wind and ice. Overall, however, Bower (1961) concludes that climate is the main driver of peatland erosion in the Pennines.

In her often-cited reviews of blanket bog erosion, Bower (1960, 1962) maintains the argument that climate, landform and geology are the main drivers of erosion, rather than biotic influences. However, in considering the fact that most blanket mire systems today show a different, drier character compared to the condition of the peat just a short distance below the bog surface, Bower (1962) acknowledges that this may be due to human impacts during recent times. She considers the suggestion made by both C.B. Crampton and W.H. Pearsall that climate change is the cause, but admits that there is little evidence for this while also admitting that there is ample evidence for human impact.

Bower (1962) then considers the mechanisms proposed by Conway (1954) – namely rapid peat accumulation during wet periods leading to instability and collapse of the peat mass into a series of drainage gullies. Bower (1962) believes that her own evidence, which shows that the most extensive erosion is located on the deepest peats, supports this idea.

Bower (1962) then considers the evidence for drier conditions resulting from climate change, and such drying causing a failure in the peat-forming process, but cannot find convincing evidence for this. She then considers the possibility that biotic factors may have initiated erosion, and makes the following rather curiously lukewarm observation: "This idea [of biotic impacts causing erosion] is mentioned by few writers but appears to be favoured by some research workers." Bower (1962) reviews the history of human activity in the Pennine region, but finds it difficult to identify any links to erosion because the dates of erosion and the dates of many activities are unknown.

She does consider burning in some depth, and acknowledges that burning can "make erosion more liable to occur" and "must encourage erosion", although she also observes that it is difficult to distinguish between the effects of burning and grazing because the two are so often linked.

Bower (1962) also observes that some evidence exists to suggest that erosion may have been present as far back as Viking times, pointing to the possibility that at least some erosion may be of considerable age and therefore whatever triggered it may be centuries old. This, of course, comes back to the question of recovery-times for blanket bog and any form of impact.

Bower (1962) concludes that either natural instability or biotic influences, or both together, are more likely candidates than climate change in explaining the extent and distribution of blanket mire erosion. She favours the idea that blanket bogs have an inherently unstable end-point and that although biotic influences may have dramatic effects on occasion, it is rather inevitable that erosion will begin at some point.

It is clear, therefore, that the idea of inevitable collapse and disintegration is a strong one, but it is worth highlighting the fact that none of the authors reviewed so far has presented a specific example where such a natural 'end-point' has demonstrably led to ecosystem collapse. The mechanism is repeated time and time again, and the apparent logic of the idea is clearly attractive, but no concrete evidence is presented to support the idea.

19.3.7 Taylor (1983) : peatlands in the UK

In what was (and still should be) regarded as a definitive global review of the peatland habitat in its time (Gore, 1983), the peatlands of the UK were reviewed by Taylor (1983). He discusses the phenomenon of erosion, and presents some clear illustrations of Type 1 erosion from Moor House National Nature Reserve. He considers that "mires are inherently unstable systems" which accumulate peat in positions within the landscape to such depth that they eventually become extremely vulnerable to factors which can trigger erosion.

Taylor (1983) suggests that the saturated nature of blanket bogs, being located on hill summits, renders them liable to self-induced hydrological rupture and collapse either as some form of mass-movement or through the development of internal drainage systems within pool and hummock systems - he cites Bower (1960) as describing these hummock and pool systems as "lines of weakness".

Taylor (1983) also discusses a range of other possible mechanisms which might trigger erosion, including peat-slides, collapse of sub-surface peat-pipes, burning, grazing and air pollution. However, he regards these as localised in their effects compared with the "universal natural processes" of hydrological instability.

19.3.8 Tallis (1985) : erosion and mass movement

Tallis (1985) used evidence of slumped peat blocks found in stream-courses, together with measurements of peat accumulation in several adjacent areas of blanket mire in the Pennines, to re-construct the onset of an erosion phase which he dates to somewhere between 1,000 and 1,200 years before present (BP). He concludes that erosion in this case was initiated by instability at the blanket bog margins which then led to gully development upslope across the main mire surface.

Though someone who has often promoted natural instability as a cause of peat erosion within the recent literature, Tallis (1985) then admits that the absence of clear, widespread evidence for instability features or events elsewhere in the region is something of a difficulty for this argument, given the widespread nature of blanket mire erosion. He is therefore careful to avoid the suggestion that instability might be the only cause of peat erosion and invariably qualifies his conclusions with cautionary observations.

19.3.9 Tallis (1987) : 'erosion, fire and flood'

Tallis (1987) examines the history of events at Holme Moss, in the southern Pennines, and concludes that an initial phase of erosion was stimulated by forest clearance around the margin of the site during the 11th Century, resulting in the establishment of Type 2 gullies. He describes the development of a *Sphagnum*-rich bog across western parts of the site during the 16th Century, but then concludes that the eastern part of Holme Moss suffered a catastrophic fire in the 1700s, and links the effects of this to a huge bogslide which occurred in 1777.

Tallis (1987) describes the present erosion patterns on Holme Moss as resulting from a mixture of natural and human-induced factors which have shaped the site over the last 1,000 years. Tallis (1987) includes, as one of these natural factors, the inherent vulnerability of a peat mantle when draped over increasingly steep slopes. He links this vulnerability to the fact that heavy rain can cause slope failure (a peatslide) in such relatively steeply-sloping peat.

19.3.10 Tallis (1995b) : stream erosion and forest clearance

Tallis (1995b) collates a range of dates for the initiation of gully erosion from a range of blanket mire sites across Britain and Ireland. These dates indicate that there are three distinct phases of initiation, with the two types of gully erosion described above by Bower (1961) in some cases being present at the same site but being initiated at different times.

Tallis (1995b) then suggests that the earliest group of dates, spanning the period from 4,500 BP to 2,000 BP, may reflect the cutting back of rejuvinated streams onto watershed plateaux following forest clearance on hill-slopes, the headwaters of these streams eating into the drainage pattern of mires which dominated the plateau summits. For the more recent dates of initiation, he speculates that these may be linked to Roman forest clearance, post-mediaeval sheep farming, or natural instability in the peat. He observes that none can be discounted as explanations, but equally none can be taken as the clear cause.

If the explanation of headwater erosion is indeed the explanation for the earliest erosion events, it is not possible to say from Tallis's (1995b) analysis whether such headwater erosion would eventually have happened anyway, even with the forest cover left intact. However, it is interesting to note that such watershed mires in Tierra del Fuego, Argentina, show no signs of such stream-induced erosion, as discussed further below in Section 19.4.3.

Thus far, the idea that blanket bog has an innately unstable end-point has formed a central theme of the evidence reviewed. However, certain lines of evidence now raise significant questions about the concept of natural instability as postulated by Osvald (1949), Pearsall (1950), Conway (1954), Bower (1962) and even Tallis (1985).

Before examining these new lines of evidence, it would perhaps be helpful to summarise the various ideas which have been proposed to explain the *initiation* of blanket mire erosion. These various ideas are summarised in Table 22 and are considered further in subsequent sections.

19.4 Peatland erosion and innate stability

19.4.1 Evans and Warburton (2007) : slope stability

The logically persuasive argument that peat cannot accumulate indefinitely on a plateau summit or a hill slope has influenced thinking about peatland erosion for decades. However, the physics of this situation has proved to be somewhat unexpected.

Evans and Warburton (2007) describe a form of slope-stability analysis carried out by Carling (1986) to test the stability or otherwise of blanket peat draped over typical hill slopes. The calculations undertaken by Carling (1986) indicated that the peat was stable even on slope angles which significantly exceed those on which blanket bog is known to form.

In other words, within the known limits of blanket bog formation, it would seem that there may be no stage at which a blanket bog inevitably and fundamentally becomes unstable because of the underlying slopes. This is an important finding, because the idea of inevitable instability has been a common central theme running through much that has been written about in relation to blanket bogs and erosion.

19.4.2 Hydrological constraints to peat growth

In his description of the 'hummock-hollow regeneration complex' at Komosse, Osvald (1923) describes how hummocks grow up beyond the ability of the bog water table to maintain an adequate water supply to the hummock vegetation and so the hummock begins to dry out and then decay, eventually to become a hollow. This idea of growth beyond hydrological supply was also extended to whole bog systems, leading Tansley (1939) to describe the way in which a raised bog would eventually grow beyond its domed water table and become a woodland.

It is likely that Tansley (1939) had in mind such sites as Holme Fen, Cambridgeshire, or Westhay Moor and Shapwick Heath in the Somerset Levels, when he wrote this successional account, but what he did not take into account was the fact that these sites are heavily drained fragments of former raised bogs. The trees are there because of the drainage, not because the bog has grown beyond its capacity to maintain itself.

It is interesting to see Taylor (1983) repeating this idea of a bog system accumulating so great a peat thickness that the accumulated peat can no longer be supplied by the bog water table. Consequently (so the theory goes) the surface layer dries out and possibly suffers a degree of crumpling or collapse.

Table 22. Summary of possible mechanisms triggering blanket mire erosion, according to a range of authors.

Author	**Year**	**Growth above water table**	**Growth beyond slope limits**	**Headward erosion of streams**	**Internal hydrological collapse (piping)**	**Upland climate**	**Climate change**	**Mass movement (peatslide)**	**Burning**	**Grazing**	**Air pollution**
Osvald	1949	Yes	Yes		Yes						
Pearsall	1950		Yes	Yes	Yes		Yes	Yes	Yes	Yes	
Conway	1954		Yes		Yes			Yes			
Bower	1960, 1961, 1962	Unlikely	Yes		Yes	Yes	No evidence	Yes	Yes		Yes
McVean & Ratcliffe	1962								Yes		
Ivanov	1981				Yes						
Taylor	1983	Yes		Yes	Yes	Yes	Yes	Yes	Yes		
Tallis	1985, 1987, 1995a, 1995b, 1997 1998		Yes	Yes	Yes		Yes	Yes	Yes	Yes	Yes
Lindsay et al.	1988				Yes				Yes		
Maltby et al.	1990								Yes		
Stevenson et al.	1990		No				Yes		Yes	Yes	
Birnie and Hulme	1990									Yes	
Grieve et al.	1994					Yes				Yes	
Mackay and Tallis	1996								Yes		
Rhodes and Stevenson	1997						Yes		Yes		
Anderson	1997								Yes		
Yeloff et al.	2006								Yes	Yes	
Evans and Warburton	2007			Yes			Yes	Yes	Yes		

In fact the general consensus now is that hummocks and bog systems do not over-reach themselves in this way. Tallis (1987) presents the argument that, as bog systems reach critical limits, decomposition rates increase and accumulation rates decrease, thereby preventing the peat mass from exceeding these critical limits. Belyea and Clymo (1998, 2001) meanwhile demonstrate that hummocks have a very finely-tuned hydrology which does not involve them growing beyond their water supply.

This 'growth to dryness' idea has persisted since the time of Osvald (1923) and has influenced ideas about fundamental stability in bog systems, but has in recent times proved to be an incorrect interpretation of how the bog system functions.

19.4.3 Peat-forest landscapes : trees as 'soil-nails'

Conway (1954) suggested that initiation of blanket peat erosion in the southern Pennines might have been encouraged by forest removal around the blanket mire margin. In contrast, the investigation of mass movement and marginal instability undertaken by Tallis (1985) concerned fragmentation of the blanket mire margin when blocks of peat fell away towards the drainage stream at the mire edge. Then there is the initiation of Iron Age peat erosion which, as discussed above, Tallis (1995) suggested might be caused by rejuvination of marginal streams as a result of forest clearance.

Figure 56. Blanket peat draped over hill summit, northern Skye.
The thin capping of blanket peat covering the summit of this hill-slope can be readily seen, as can the abrupt unstable edge to the peat, and the steep nature of the ground over which this peat margin is draped. The area along the peat margin would probably formerly have been wooded.

Photo R A Lindsay

It is certainly possible in many parts of Britain to find examples of blanket peat where the peat ends abruptly with a bare peat face as the ground falls away steeply downhill. Figure 56 shows an example from the Isle of Skye, but such scenes are particularly common in, for example, the Peak District, and involve less steep mineral slopes than the one illustrated from Skye.

Clearly the peat margin visible in Figure 56 is not stable. Parts have evidently already broken off and slid away down the steep slope, while others look as if they could follow suit in the not very distant future. Fragmentation of the blanket bog margin results in effects which are not just felt at the margin alone. The drainage impacts are likely to have knock-on effects back up into the deeper peat which cloaks the hill summit.

In some ways, this could be precisely the scene envisaged by Tallis (1995) when Iron Age communities began to clear the hill-slopes of their forest cover. Formerly-wooded slopes became bare, exposing the blanket bog margins on the hill summits to the forces of slope-failure and mass movement.

This scene can be contrasted with three other blanket mire landscapes, one in Tierra del Fuego, one in the Queen Charlotte Islands, near Vancouver, western Canada, and one in Nova Scotia, eastern Canada.

The eastern part of Tierra del Fuego has extensive tracts of blanket mire, but the steeper slopes are densely wooded. Figure 57 shows the scene from within a wooded slope looking out over areas of coastal blanket mire, and then the second image illustrates the open patterned ground which dominates the watershed summit.

The important point about Figure 57 is that there is no abrupt edge to the peat. Towards the edge of the watershed plateau the peat becomes steadily thinner and scattered stunted trees appear. Closer to the plateau edge the peat becomes thinner still and the trees become taller. Eventually the peat becomes sufficiently thin for the trees to root through the peat into the mineral soil beneath. These trees then act like anchors or, to use an engineer's term, 'soil nails' (which are often used by engineers to keep potentially unstable slopes in place).

Figure 57. Blanket mire in Tierra del Fuego, Argentina.
The view on the left is taken within the forest which covers the steeper hill-slopes near Moat in Tierra del Fuego, Argentina. In the distance, areas of open mire can be seen as pale brown patches. The scene on the right is the patterned bog which covers the watershed summit. In the background it is possible to see a few stunted trees which represent the fringe of the forest cover at the edge of the main bog system where it falls steeply away downslope. Note that there is no sign of erosion in the watershed mire system despite its high altitude.
Photo R A Lindsay

Two similar scenes can be viewed using Google Maps, in the Queen Charlotte Islands off the Canadian coast near Vancouver, and in the Cape Breton Highlands National Park in Nova Scotia. Firstly, put the Queen Charlotte Islands as the destination into Google Maps, then zoom in on the Naikoon Provincial Park in the north-east corner of the islands. Switching to satellite view and zooming in on the main brown areas then reveals a maze of blanket mire systems with black bog pools, interlaced with a filigree of forests which occupy the higher, steeper, mineral ridges of ground.

Next, put Cape Breton Highlands National Park into the destination for Google Maps. Switch to satellite view and zoom in on the name of the park. Keep zooming in until the image switches to an aerial photograph. A series of pool-dominated areas will be evident (the water in the pools showing black). Zooming in still further and slightly to the left of the park name, at full zoom the photo reveals that the green areas interlaced between the peatland systems are dominated by relatively scrubby, short woodland.

It is quite likely that many of Britain's blanket mire areas looked remarkably similar to both these images prior to wholesale forest clearance during the period between Neolithic and Anglo-Saxon times. Tallis (1987) gives a number of slope profiles for the edge of Holme Moss, in the southern Pennines, and it takes no great leap of the imagination to picture the steeper parts of the slopes shown in Figure 4 of Tallis (1987) in a wooded condition. The slopes and peat thicknesses involved offer no real problems to woodland colonisation:

- 10° slopes have 1.5 m peat thickness;
- 15° slopes have 1 m thickness;
- 20° slopes have 0.75 m thickness;
- 25° slopes have 0.5 m thickness.

Whereas modern commercial afforestation of bog systems, even on the bog margins, is an activity which is likely to result in significant hydrological disruption to the mire system because of the ploughing required and the species involved, encouragement of natural regeneration and sensitive planting of native species on the hill slopes around the margins of many blanket mire systems may actually benefit the mire systems, particularly where there are concerns about marginal erosion and slope instability.

Of course such a change to the long-established open landscapes of these hill slopes and their associated peat-draped summits raises a number of controversial issues for the open-ground communities of these slopes, and for the character of the landscape itself. It would nevertheless be useful to explore and debate these issues if there is the potential to improve the ecological condition of many blanket mire systems.

19.5 Natural hydrological instability

There is one form of natural hydrological instability within raised and blanket mires which has been identified in much of the literature referred to above, and which has in recent years become the subject of much intensive research – namely the presence of natural pipes and associated 'sink-holes' within peat bog systems. These occur in both raised bogs and blanket bogs but are more common in, and more characteristic of, the latter. The pipes which are referred to in the context of erosion, development of an internal drainage system, and eventual collapse, are the same type which have become the focus of recent research – most notably by Holden (2005b). However, what is not generally realised is that there are two types of peat pipe. There is another type of pipe which is essentially a constructive feature within the peat body, and is not associated with collapse and erosion.

19.5.1 'Destructive' and 'constructive' peat pipes

19.5.1.1 'Destructive' peat pipes

Osvald (1925, Plate 10b) illustrates a striking example of a bog pool in which the water level has clearly fallen a considerable distance – far further than could be explained simply as draw-down during dry weather conditions. Such dramatically-lowered pools are generally a sign of a 'sink hole' forming an entrance to a destructive peat-pipe system.

Despite the fact that these peat-pipe systems have been known about for more than 70 years, and are referred to by Osvald (1949), Pearsall (1950) – as internal drainage systems – and Lindsay *et al.* (1988), it is only recently that Holden and Burt (2002) have provided the first detailed account of these features and their hydrological relationship with the surrounding blanket mire. Holden (2005a) has emphasised their significance in water, gas and particulate carbon transport from the mire system.

Where a pipe entrance, in the form of a sink hole, occurs within a bog pool pattern, the pools closest to the sink-hole suffer water-table draw-down partly caused by the fact that the ground level sinks towards the pipe entrance in the manner of a wide funnel. Thus a pool with an entrance to a peat pipe may be 1 m or 1.5 m lower than the surrounding bog surface. Holden and Burt (2002) observe that on some occasions the roof of such pipes may collapse, thereby producing ready passage of surface water into the pipe channel at various points along its length.

It is not known how these pipes form, but they appear to be a natural feature in the sense that they have been found in relatively natural bog systems. However, Holden (2005b) has shown that the incidence of pipes within the peat increases substantially if the peat is drained. It is not yet clear how these structures relate to natural hydrological processes and to the development of internal drainage systems leading to erosion a particular form of erosion, but there must be such a link.

Erosion systems typical of peat pipes and sink holes have been found in the extensive blanket mires which dominate the eastern limit of Tierra del Fuego (Dr. Ab Grootjans, pers. comm.), and similar erosion patterns can be seen in the blanket mires of eastern and western Canada. These resulting erosional systems are unlike either the Type 1 and Type 2 erosion patterns described by Bower (1960, 1961), and do not appear to result in the same wholesale breakdown to the system typical of these two patterns. This may be because, as Holden and Burt (2002) observe, peat pipes generally follow rather convoluted routes and often appear to become blocked. Consequently there may be phases when a pipe is acting as a drain for the system, but if it then becomes blocked it can raise water levels in the bog pool and in the surrounding bog surface.

19.5.1.2 Constructive peat pipes

An alternative form of peat pipe has been recognised by Lindsay and Freeman (2008). This form of pipe appears to result from peat growth rather than peat breakdown, and is thus not associated with erosion. Consequently not all 'sink holes' are a sign of system collapse – sometimes they can be a sign of an expanding, healthy bog system.

These are considered to be constructive features because they follow lines of streams which run along the mineral sub-base, but are where the peat has grown to such a thickness either side of the stream that the two banks are able to join and form an archway over the stream channel. In time, virtually the entire stream channel can become hidden, leaving only the faintest trace which often manifests itself as a scattered line of sink holes (see Figure 58). They have yet to be described in detail, though Lindsay and Freeman (2008) illustrate several examples and emphasise that they are widespread in the blanket mires of north-west Britain.

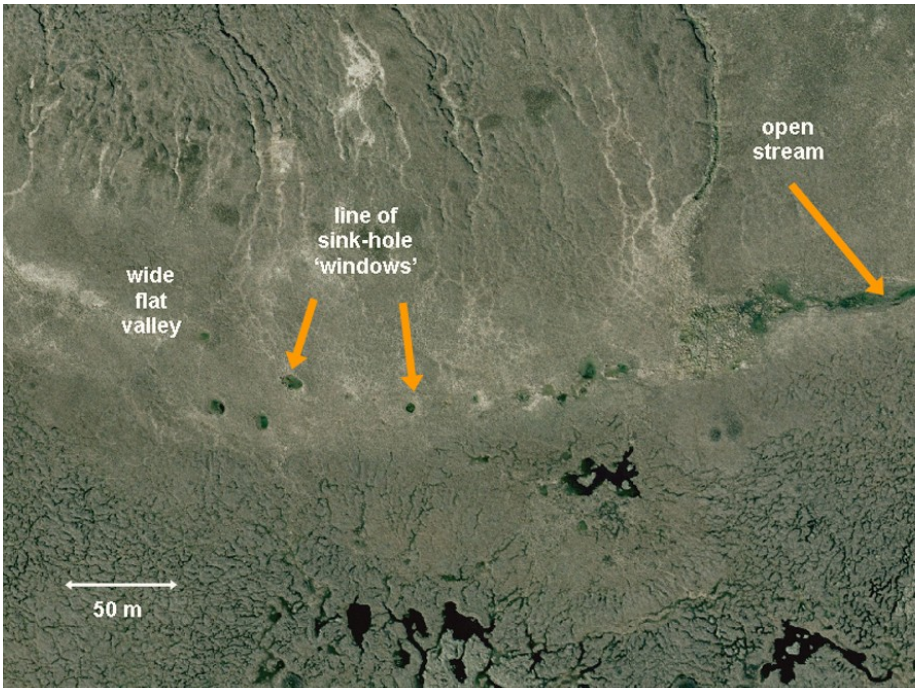


Figure 58. Aerial photograph of Lewis blanket bog, Outer Hebrides, showing line of 'constructive' sink holes indicating presence of stream running along the mineral sub-base beneath the peat. The stream eventually emerges from the peat pipe at the right of the photograph.

19.6 Anthropogenic triggers of blanket mire erosion : fire, grazing and pollution

19.6.1 Blanket mire erosion in a global context

It is a point of some curiosity that the severe and extensive erosion seen across so much of the British and Irish blanket mire resource appears to be restricted to these islands (Evans and Warburton, 2007, p.15). Erosion can be found in other blanket mire areas such as those in Tierra del Fuego, but it is the type of erosion discussed above in relation to destructive and constructive peat pipes, rather than the type of intense network and gully erosion seen in most British and Irish examples of erosion.

It is not entirely clear why this should be. The blanket mires of the Cantabrian Mountains in Spain's north-western Galicia sit on broad mountain-tops as high as those in Britain and Ireland, and in warmer conditions, but they do not show the erosion so characteristic of British and Irish watershed-summit bogs. Indeed they are remarkably free from signs of erosion. Given the slopes on which they occur, these Spanish blanket mires (and they are true blanket mires) also face the same potential issues of mass movement described by Tallis (1987) for British blanket mires. The blanket mires of both western and eastern seaboards of Canada experience harsher winters than those experienced by British and Irish blanket bogs, almost certainly harsher than those of the Little Ice Age in Britain, yet they have none of the intense gully systems of British and Irish blanket bogs. Spanish, Canadian and Tierra del Fuegan blanket mire systems have all experienced past climate changes to the same degree as British and Irish blanket bogs – possibly more so – but they appear to have done so without becoming subject to intense gullying.

It is perhaps significant that one other part of the world where not only is there blanket mire, but this blanket mire subject to fairly intense erosion, should be the Falkland Islands. These islands have a land-management system imported from Britain – a management system based on hill-sheep farming. In Spain the livestock on the blanket mires is traditionally cattle and horses, with no burning. In Tierra del Fuego there was no livestock farming until 100 years ago, and after that for many decades it was cattle only the margins of the lowland blanket mires.

Evans and Warburton (2007) comment that the effects of a fire are local and therefore burning is unlikely to explain the widespread scale of erosion seen in Britain and Ireland. However, seen from a global perspective, the frequency and intensity of burning and of sheep grazing on British and Irish blanket mires might indeed be viewed as a localised phenomenon – one which separates these mires from similar blanket mires elsewhere in the world.

19.6.2 Blanket mires and grazing

Deer, hares and potentially aurochs were the native mammalian grazers of blanket mires. Their role in assisting the development of blanket mire by grazing tree seedlings growing on more waterlogged ground is still the subject of ongoing research. By around 5,500 BP, however, sheep and goats had been introduced from the continent as part of the Neolithic farming revolution.

Sheep graze much more intensively, and have sharper hooves which can break up soft ground more easily that the wide hooves of deer or cattle. Hares, of course, leave no discernable footprint. The intensity of sheep husbandry in the millennia prior to the first written records is poorly understood, but for at least the last 1,000 years, sheep have been the main form of animal husbandry on the blanket mires of Britain. At various times during this 1,000 years, wool has been a major source of wealth for Britain, though precisely what impact the wool industry had on the blanket mire landscape during these periods is again not well documented. Even in recent times, documented figures for stocking rates do not necessarily give a true picture of the grazing pressure on the blanket mire systems themselves, because the animals may spend much of their time on the margins of the deep peat in the more nutritious flush systems, on areas of thin peat and along stream margins.

19.6.2.1 Grazing experiments at Moor House NNR

Rawes and Hobbs (1979) describe the results of an exclosure experiment at Moor House National Nature Reserve, north Pennines, in which the normal grazing level of 0.12 sheep per hectare (on the blanket bog) was removed from a number of plots and the vegetation examined after 7 years, 18 years and 21 years. In general, they found that removal of grazing led to an increase in heather (*Calluna vulgaris*) and lichen (*Cladonia impexa*) at the expense of hare's-tail cotton grass (*Eriophorum vaginatum*).

One of the study plots was established in a hummock-pool complex (more strictly a T2/T1:A1/A2 complex). Removal of grazing was found to result in the expansion of the lowest *Sphagnum papillosum*-rich terrestrial zone (T1 low ridge) in particular, at the expense of the *Sphagnum cuspidatum* A1 hollow and the bare-peat/common cotton grass (*Eriophorum angustifolium*) A2 'mud-bottom' hollow. This may reflect the release of trampling pressure on the T1 zone, which tends to be the softest of the terrestrial zones and is the easiest to walk on because it is low lying. Trampling of this zone therefore tends to compress it down into the A1/A2 hollow zone. Similar 'squashing' of the normal zones has been observed on the bogs of Galicia in NW Spain, where the hollows are firm and the hummocks rise directly from these hollows without a T1 zone.

Interestingly, Rawes and Hobbs (1979) also point out that mid-summer rainfall had been 12% lower for the previous 12 years, and thus they speculate that the observed increase in *Sphagnum papillosum* at the expense of the pool zone might be a response to a drier climate. This is precisely the kind of change that would be predicted by the climate phasic model of Barber (1981) if conditions were indeed becoming drier overall.

Rawes and Hobbs (1979) also found that heavy grazing produced a rapid decline in heather (*Calluna vulgaris*) cover and a rise in the two cotton grasses *Eriophorum vaginatum* and *E. angustifolium*. The moss cover, particularly *Sphagnum*, was significantly reduced under the heavy-grazing regime (equivalent to 3.4 sheep ha^{-1}). Over time this vegetation did not recover to its original state, but remained in its new form with steadily decreasing bryophyte cover and increasing amounts of bare peat. of the normal background grazing level, they conclude that the observed 0.12 sheep ha^{-1} on the blanket bog could be increased 3-fold to 0.37 sheep ha^{-1} without changing the vegetation composition of the bog.

19.6.2.2 Sheep grazing and blanket mire damage: Hulme and Birnie (1997)

Hulme and Birnie (1997) review the relationship between blanket mire vegetation and sheep stocking-levels, taking into account the required biomass offtake by the sheep and the level of grazing which the habitat can sustain. Birnie and Hulme (1990) had already identified that the recommended stocking levels in Shetland were causing evident damage to the blanket bogs of Shetland.

Using the "Hill Grazing Management Model" devised by Grant and Armstrong (1993), they identify first that cattle numbers had declined since World War 2, larger breeds of sheep were now generally being used, and they were being grazed on the hill all year round rather than just during the summer months. Using the current grazing regime, therefore, Hulme and Birnie (1997) calculate that grazing levels of between 0.5 and 1 sheep ha^{-1} are the maximum level allowable if the blanket bog vegetation is not to suffer damage.

Hulme and Birnie (1997) emphasise that this calculation is based on a number of simplistic assumptions. It would therefore be better to err on the side of caution, but it is interesting that the lower level, especially if given a margin of error, is quite similar to the figure obtained by Rawes and Hobbs (1979) for the grazing tolerance of the vegetation at Moor House NNR.

19.6.2.3 Exclusion of cattle from mossbeds (bogs) in Victoria, Australia

McDougall (1989) describes a monitoring programme which examines the effect of excluding cattle from a 'mossbed' (bog) on the Bobong High Plains, Victoria, Australia. A large mossbed was fenced to exclude cattle in 1948, while an adjacent mossbed remained open to cattle grazing. Both mossbeds were examined on a sporadic basis, then in 1979 a botanical survey was undertaken of both mossbeds, with limited repeats in 1982 and 1987. Casual observations had found no marked changes until 1966, but by then the extensive areas of bare peat had begun to re-vegetate.

The detailed study in 1979 was based on a 4x4 m grid covering the whole area of *Sphagnum* within the fenced mossbed. The ungrazed mossbed was found to have an unusually high and largely continuous cover of *Sphagnum*, and when compared to the grazed mossbed it recorded significantly higher frequencies of high cover values for *Sphagnum*.

Structurally, the grazed mossbed formed an interconnecting series of runnels which fed into the main outflow stream, whereas the ungrazed mossbed retained water within the site in distinct hollows. Consequently the site remained wetter and provided water of a higher quality because the outflow had passed through the *Sphagnum* carpets rather than flowing swiftly along bare-peat runnels.

Little change was found between the vegetation recorded in 1979 and 1987. McDougall (1989) speculates that either change is very slow, in which case more obvious changes may require the next survey to be undertaken in 1999 or later. Alternatively, the site may have completed its recovery and already have reached a stable state by 1979 – *i.e.* the damage required a recovery period of some 30 years.

19.6.3 Blanket mires, erosion and burning

A substantial amount of information on blanket mires and burning has already been reviewed and considered in Section 18, Topic 7. However, in relation to the specific question of burning and erosion there are several additional aspects which can usefully be explored here. This is particularly so, given that in Table 22 the one factor which appears more frequently than any other as a potential trigger for erosion is fire. It might be argued that this weighting towards burning simply reflects the particular papers included within the table, but in fact any review of blanket mire erosion will identify burning as one of the potential triggers – in at least certain specific cases.

In fact, virtually every review of the topic identifies specific instances where erosion has been caused by fire damage. Indeed it is possible to put it more strongly than that. Of all the various mechanisms identified as possible triggers for the mesotope-and macrotope-wide scale of erosion (uniquely) observed in the British and Irish blanket mire habitat, burning is the *only* documented factor to have demonstrably resulted in extensive erosion.

All the other potential trigger mechanisms listed in Table 22 undoubtedly have potential to cause erosion, but none has been conclusively demonstrated (other than perhaps very occasional bog bursts). This is a little curious, since Occam's Razor would suggest that the demonstrable cause of something should logically be taken as the most evident general mechanism except where good evidence is presented to show otherwise.

Not only do most reviewers of erosion and its causes consistently list burning as a potential trigger, it is usually described thus: "causing severe damage", "a catastrophic fire", "more recent fires have been much more obviously devastating", "a scorched and arid plain", "Burning seems to be very important in modifying and weakening vegetation cover....*Sphagnum* is severely damaged", "fire can produce dramatic and rapid erosion of upland peats" (Tallis, 1987; Mackay and Tallis, 1996; Anderson, 1997; Bower, 1962; Evans and Warburton, 2007).

The argument against fire being the prime trigger for blanket mire erosion appears to be that it is not sufficiently ubiquitous, that charcoal in the peat archive is not always synchronous with erosion events, and that the scale of fire evidence does not match with the apparent scale of erosion (*e.g.* Tallis, 1985; Evans and Warburton, 2007; Grieve *et al*., 1994; Rhodes and Stevenson, 1997).

The arguments that burning is not sufficiently ubiquitous and that it is not always synchronous depend to a considerable degree on the response-time of the ecosystem to a burning event. Burning is one of the most ubiquitous features to be found within a peat core from a British blanket mire. Somewhere in almost very core there are signs of burning. Indeed Dr Sarah Crowe has undertaken coring on the top of Knockfin Heights in Caithness at the behest of the RSPB to investigate just this point. It was assumed that no-one would have bothered to burn such a high-level plateau, and it was therefore a considerable surprise to find that there were several very clear records of fire within the core (DR Sarah Crowe, pers. comm.). Such a find not indicate that someone has deliberately set out to burn this high-level watershed plateau, but fires generally burn up-slope and thus there would be a tendency for wildfires to climb up to this plateau.

Recovery or response-time is also important here, because a fire may not cause erosion immediately. It may combine with the pressures from grazing, or from acid precipitation, whereby the bare, incipiently-eroding surface is kept un-vegetated by these additional factors. Thus it is not suggested that burning alone might be the single trigger to explain such widespread erosion; it *is* suggested, however, that its demonstrable ubiquity, its proven capacity to render a peat surface "catastrophically" damaged, its ability to affect substantial areas, its evident presence far back into the peat archive, and its capacity to render any of the other potential factors more potent, should encourage a re-appraisal of burning as a prime cause of blanket mire erosion in Britain and Ireland.

Two examples will suffice to show how burning can interact with other factors to create conditions for erosion to be triggered and can at the same time create a form of dependence-cycle which makes the position increasingly worse over time.

19.6.3.1 Burning and grazing experiments at Moor House NNR

The grazing experiments described by Rawes and Hobbs (1979) in the previous section are actually a multi-factorial study looking at the impact of both grazing and burning at different levels. Thus they considered grazing on its own, burning on its own, and the effect of grazing and burning together on the blanket bog vegetation of Moor House NNR, in the northern Pennines.

Rawes and Hobbs (1979) examined the effect of burning on a 10-year cycle, a 20-year cycle, and a fire at the start of the experiment (in 1954). The vegetation was recorded after 7 years and 19 years, then certain species were recorded again after 22 years.

They found that on the 10-year rotation plot, the vegetation was reduced to a simple species list dominated by hare's-tail cotton grass (*Eriophorum vaginatum*) tussocks, with bare peat showing in more than 50% of the plot. After 18 years, the short-rotation burn combined with grazing created the most extensive areas of bare peat.

This result is significant not merely for the fact that much bare peat has been created, but Rawes and Hobbs (1979) make the point that heather (*Calluna vulgaris*) does not set seed successfully. It therefore it relies almost wholly on vegetative reproduction, which it achieves by setting out stems and shoots into a *Sphagnum* carpet. Under these conditions, heather does not go through the classic 25-year cycle of growth, maturity and collapse because old stem material is constantly being smothered by upward growth of fresh *Sphagnum*. This is exactly the same process as described and illustrated for deer grass (*Trichophorum cespitosum*) in Appendix 3, Figure 87.

If a fire cycle is established which destroys the *Sphagnum* layer, the heather no longer has a benign medium in which to reproduce vegetatively and so the heather stems slowly elongate until, after 25 years, they collapse. In order to prevent the heather from reaching this stage it becomes necessary to burn the heather, but doing so is likely to kill off any colonising *Sphagnum*. Consequently the peat surface becomes drier and more hostile and more prone to serious fire damage, but burning is the only way to keep the heather vigorous now – unless a period of no burning is accepted which is sufficiently long to permit a new *Sphagnum* carpet to develop. Once it has developed, land-management effort can be reduced because burning is no longer required. Carry on burning, however, and it becomes a relentless cycle.

19.6.3.2 Blanket mire erosion, Sphagnum and Racomitrium lanuginosum

Tallis (1995a) describes a detailed study of peat stratigraphy carried out at Over Wood Moss, near Alport Moor in the southern Pennines. The site lies between Kinder Scout and Bleaklow. The purpose of the study was to investigate any relationship observable within the peat between *Sphagnum* remains, the remains of woolly hair moss (*Racomitrium lanuginosum*), signs of erosion, and possible triggers of erosion.

The study also provides an opportunity to draw a range of potential erosion factors together and present a somewhat more burning-focused picture than the potential description of events given by Tallis (1995a). Tallis (1995a) proceeds on the basis of certain perfectly reasonable assumptions:

- that *Racomitrium* requires low water tables but high atmospheric humidity;
- that lowered water tables necessary for its establishment result either from an earlier dry climate, or that the peat is dry despite a wet climate because erosion is draining water from the peat;
- that the combined presence of *Sphagnum* and *Racomitrium* suggests that there is at least some erosion present on the site.

The one curious assumption which Tallis (1995a) makes is that Over Wood Moss is not an eroded site. Examination of detailed aerial photographs for the area would suggest that it is an extensively eroded site which has shown substantial vegetation recovery (see Figure 59). Alport Moor, just to the south, is undoubtedly un-eroded on its crown, but Over Wood Moss appears to have gone through a substantial erosion phase in the past. This would appear to fit with the data obtained by Tallis (1995a).

Tallis (1995a) takes a peat core of the uppermost 50 cm of peat from Over Wood Moss. From this he then generates a frequency histogram for the leaf-remains (macrofossil remains) of *Sphagnum* and *Racomitrium*, the frequency of charcoal remains, and the frequency of pollen for crowberry (*Empetrum*)/bilberry (*Vaccinium*), ribwort plantain (*Plantago lanceolata*), and Cyperaceae – sedges, essentially the cotton grasses (*Eriophorum*). Two carbon dates are obtained from a similar core at the nearby Alport Moor and these are then transposed to the Over Wood Moss core. In addition, the pollen records for ribwort plantain (*Plantago*) can provide certain approximate dates as a general guide to timings in other parts of the 50 cm core.

To summarise, Tallis (1995a) finds that *Racomitrium* is present in the peat column between about 1250 to 1450 AD, then it disappears only to re-appear in the mid-17th Century. He concludes that the presence of *Racomitrium* on a non-eroded site indicates that the long dry period of climate of the Early Mediaeval Warm Period (EMWP), which ends around 1350 AD primed the peat to dry out, which then enabled *Racomitrium* to become established. Conditions then became too wet for the *Racomitrium* at the height of the Little Ice Age (LIA), which covers the period from around 1350 to 1850 AD, but then conditions dried out sufficiently for *Racomitrium* to re-establish itself briefly before acid rain rendered the bog vegetation moss-free.

While Tallis (1995a) acknowledges the presence of charcoal and the possible effects of burning, he regards the primary signal for *Sphagnum* and *Racomitrium* to be one related to climate. Although agreeing with him to a certain extent, the somewhat different interpretation presented below takes a more fire-focused interpretation. The data are shown in Figure 60, with the key points numbered 1 – 9.

The sequence of events suggested above differs from that given by Tallis (1995a) in three important respects. Firstly, it recognises that there is significant erosion within Over Wood Moss. Secondly, it identifies a phase of very considerable re-vegetation within the eroded areas. Finally, and most importantly in terms of describing the relationship between fire and erosion, it identifies a specific period of sustained burning between 1100 AD and 1200 AD as the trigger for erosion. Over a period of 150 years this erosion becomes increasingly severe.

The shift from the EMWP to the Little Ice Age is acknowledged as having encouraged substantial expansion of *Racomitrium*, but because the site was already sufficiently eroded to permit *Racomitrium* to establish, the initial colonisation occurred before this climate shift had become strongly established. The erosion may thus have been a response to *fire*, not to climate change.

One final thing to note from these results. From the three known and two probable dates identified within the archive of this 50 cm peat column (1040, 1450, present day surface, and possibly 1200 and 1500), it would appear that peat accumulation during the Early Mediaeval Warm Period (EMWP) was almost twice the rate of that which occurred during the Little Ice Age (LIA) and on up to the present day. The thickness of peat is 24 cm for the 350 years between 1140 AD and 1450 AD, whereas the thickness is only 20 cm for the succeeding 540 years.

This would imply a rate of 0.69 mm yr^{-1} for the EMWP and only 0.37 mm yr^{-1} for the LIA and the modern era – almost twice the rate during the earlier warm period - which has obvious implications for the possible response of the site to current and future climate trends. Also relevant is the fact that, as Belyea and Clymo (1998) observe, hummocks produce peat more rapidly than hollows.

According to Barber's (1981) phasic theory, hummock and high-ridge would tend to be the dominant nanotopes during drier phases of the climate, and thus such a phase may be linked to faster, rather than slower, peat formation. Such an idea would need careful testing to determine the extent to which it is a true reflection of the story at Over Wood Moss.

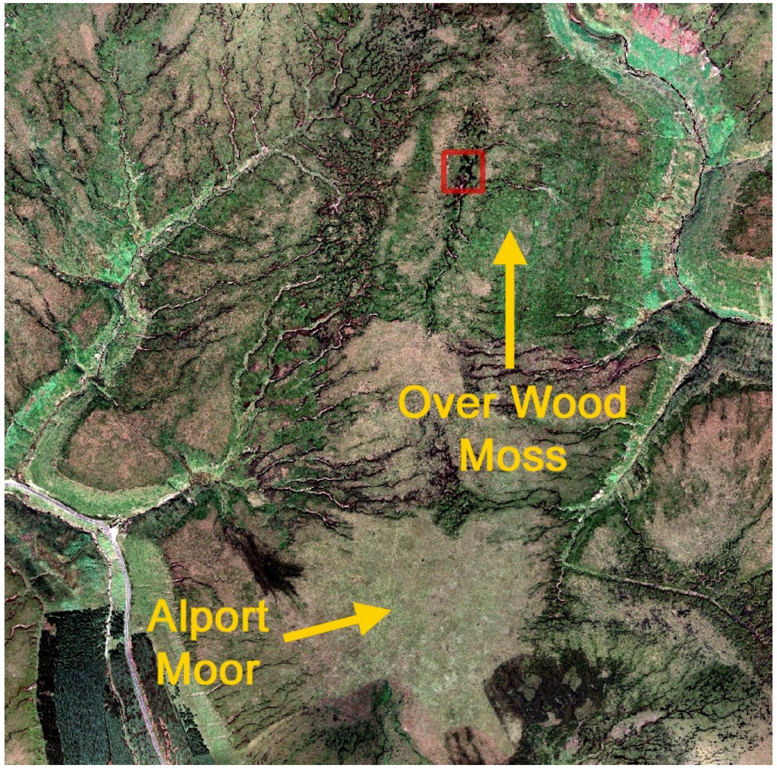
Complicating such interpretation is the fact that subsequent drying caused by adjacent erosion may have caused existing peat thicknesses to undergo consolidation, compression and oxidation, thus obscuring original rates of accumulation. Thicknesses of peat may also have been lost during the LIA or the modern period directly through erosion.

Nonetheless, the difference in apparent accumulation rate between these two periods is sufficiently striking to warrant further investigations into comparative *rates* of peat accumulation during different climate phases on sites where erosion or drainage have not been issues.

19.7 Actions and research needs

19.7.1 Erosion and fire

The initiating causes of blanket mire erosion are still not known, although a number of theories have been put forward over the years. It has not yet been established whether erosion is a natural part of blanket mire dynamics, essentially an end-point of a natural process or cycle, or is instead (or in part) a process initiated by human-induced processes. It is therefore not clear whether erosion is a sign that action is needed or certain actions be curtailed, or whether the erosion process should be allowed to follow its natural course much as coastal erosion is now being accepted as the price of living on an island.



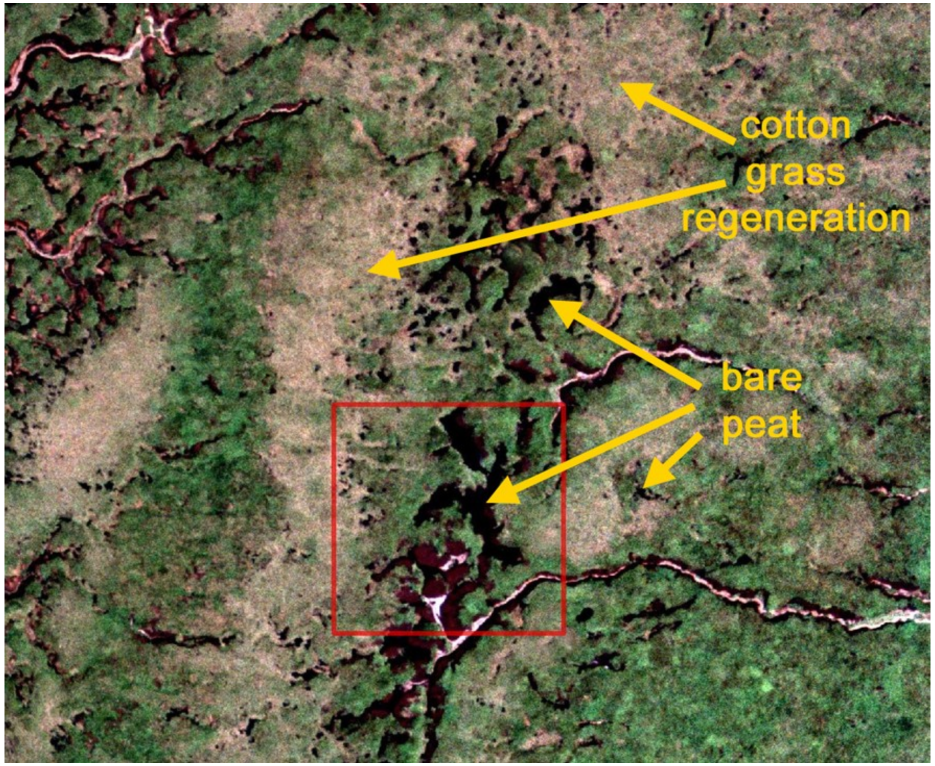
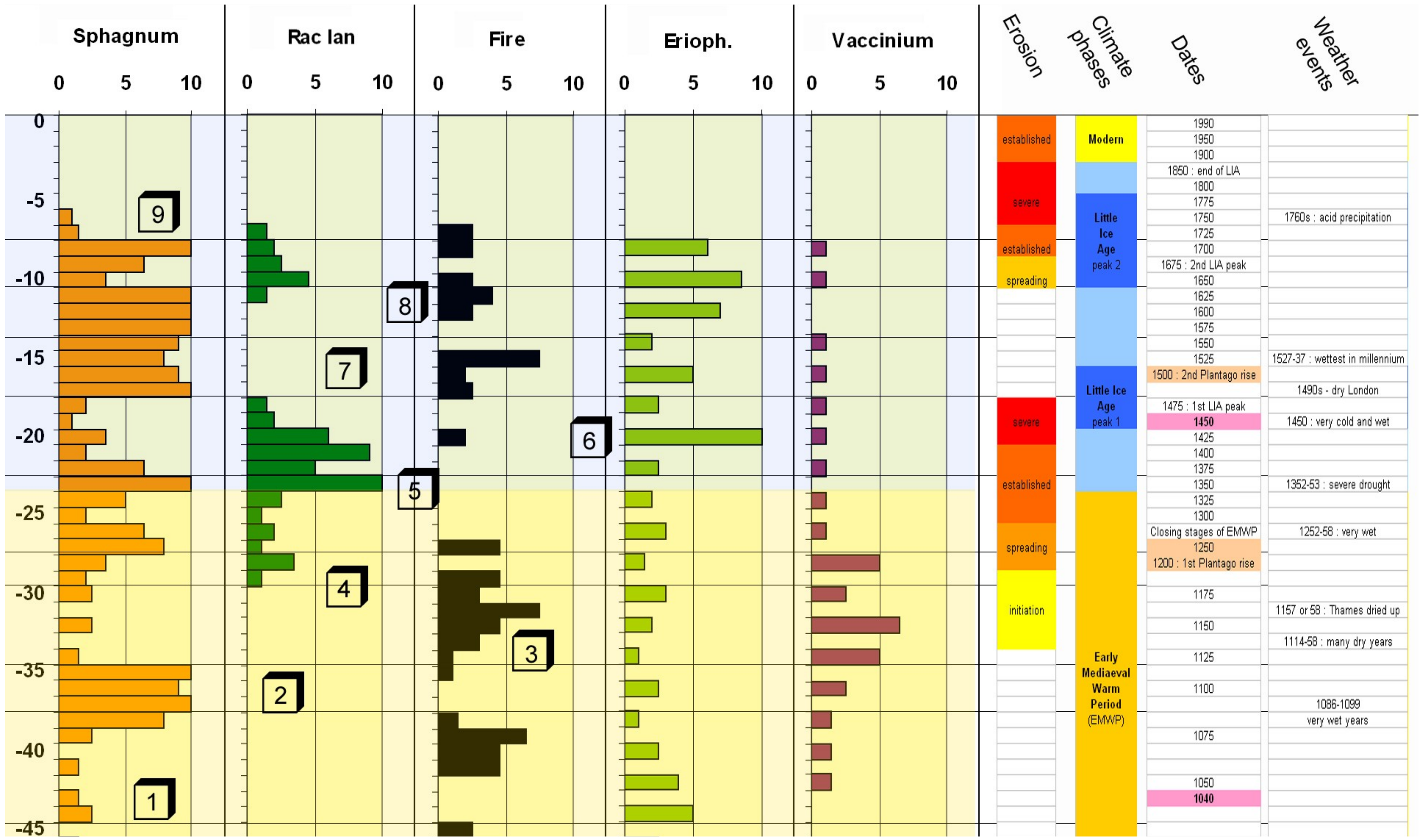


Figure 59. Aerial photograph of Tallis (1995a) study site: Over Wood Moss, south Pennines.
The aerial photograph on the left shows Over Wood Moss in relation to the location of Alport Moor, which is Tallis's other study site in this area. It can be seen from the relatively smooth brown/green colouration that Alport Moor is not eroded, at least over the main dome of the mire unit (mesotope). Over Wood Moss lies to the north, with the OS grid reference given by Tallis (1995a) indicated by a 100 m red square. The right-hand photograph gives a detailed view of Over Wood Moss, again with a red square indicating the OS grid reference provided by Tallis (1995a). It can be seen from the many dark-brown/black areas indicated on this photograph that Over Wood Moss is rather eroded. Indeed it appears to have been very extensively eroded in the past but much of this has no re-vegetated with cotton-grass. These re-vegetated areas can be seen by their distinctive mottled fawn-and-green appearance, The green represents old erosion haggs, whereas the fawn is cotton grass which has infilled the former erosion gullies between the haggs.



Sphagnum
Rac lan
Fire
Erioph.
Vaccinium
Erosion
Climate phases
Dates
Weather events
0 5 10
0
-5
-10
-15
-20
-25
-30
-35
-40
-45
1
2
3
4
5
6
7
8
9
established
severe
established
spreading
severe
established
spreading
initiation
Modern
Little Ice Age peak 2
Little Ice Age peak 1
Early Mediaeval Warm Period (EMWP)
1990
1950
1900
1850 : end of LIA
1800
1775
1750
1725
1700
1675 : 2nd LIA peak
1650
1625
1600
1575
1550
1525
1500 : 2nd Plantago rise
1475 : 1st LIA peak
1450
1425
1400
1375
1350
1325
1300
Closing stages of EMWP
1250
1200 : 1st Plantago rise
1175
1150
1125
1100
1075
1050
1040
1760s : acid precipitation
1527-37 : wettest in millennium
1490s - dry London
1450 : very cold and wet
1352-53 : severe drought
1252-58 : very wet
1157 or 58 : Thames dried up
1114-58 : many dry years
1086-1099
very wet years

Figure 60. Analysis of certain plant remains from Over Wood Moss, south Pennines.
Plant remains (macrofossils) of *Sphagnum* and *Racomitrium* (Rac lan), together with charcoal particles ('Fire'), and pollen counts for cotton grasses (*Erioph*.) and bilberry and crowberry (Vaccinium) within the top 50 cm of the peat. Scores for *Sphagnum* and *Racomitrium* are actually 1-100%, but all scores have been simplified to highlight the relative changes in values over time. 'Erosion' represents a summary of possible erosion intensities. 'Climate phases' highlight in particular the Early Mediaeval Warm Period (EMWP) in gold, and the Little Ice Age (LIA) in blue. These two colours have also been faintly run across into the species data, to highlight the key shift from the EMWP to the LIA. Suggested dates are provided in the 'Dates' column. Note the apparently greater rate of peat accumulation during the EMWP. The fixed C^{14} dates are highlighted in pink with bold numerals. Information in the 'Weather' column is provided from Historical Weather Events website. Boxed numbers refer to the numbered descriptive sequence provided in the text below.

Adapted from Tallis (1995a)

1. Comfortably within the EMWP, with presumably fairly warm temperatures therefore, but there is also regular burning. Recovery time between fires is anything from 12 years to 50 years and therefore there is limited opportunity for *Sphagnum* recovery.

2. A brief period without fire, but sufficient for *Sphagnum* to become fully dominant on the bog surface. The period between 1086-1099 was extremely wet, and may have helped rapid recovery from previous fire damage.

3. A sharp rise in sustained burning removes the *Sphagnum* cover and encourages development of a bilberry/crowberry moor. During this period there are extended runs of very dry years. The sustained impact of burning may have initiated the beginnings of an erosion complex at this stage.

4. With a slow decline from the EMWP into the LIA and therefore potentially becoming cooler and wetter, conditions are just suitable for the colonisation of the developing erosion complex by *Racomitrium*. Continued burning exacerbates the erosion process and suppresses recovery of *Sphagnum*.

5. Transition from the EMWP to the cool wet conditions of the LIA. Conditions are much improved for *Racomitrium*, especially as the erosion complex is now well developed. *Sphagnum* begins to recover thanks to the much wetter climate, while crowberry (*Empetrum*) and bilberry (*Vaccinium*) are suppressed by these conditions.

6. Ongoing severe erosion causes *Sphagnum* to decline, and a fire in the 1420s does not help. However, by the 1450s there has been extensive re-vegetation of erosion gullies by cotton grass (*Eriophorum*), encouraged by the wet conditions. This paludifies the surrounding peat haggs, making them increasingly less suitable for *Racomitrium*. This is aided by the relatively long period with few, if any, fire events.

7. Continued re-vegetation of gullies by cotton grass sees complete *Sphagnum* recovery and dominance during what are described as the wettest years of this millennium. There are also some serious fires which disturb the *Sphagnum* cover a little, but it is able to recover rapidly. The erosion complex is regenerating so vigorously that there is no place for *Racomitrium*.

8. A period of burning is sufficiently sustained and damaging to remove much of the *Sphagnum* cover as well as re-starting erosion, and so *Racomitrium* becomes re-established within the erosion complex.

9. A combination of acid deposition and burning removes the bryophyte carpet completely.

Evidence suggests that significant erosion has occurred at a number of times in the past though always within the time-span when Britain was populated, though the evidence also suggests that erosion in more recent times has been particularly marked. Precisely what initiated these erosion events has not been clearly identified.

Although several theories have been proposed to explain the onset of erosion, the only directly-observed cause has been burning; all other explanations have been inferred or have been proposed as potential mechanisms based on the available evidence. Fire is thus the only demonstrable initiator of erosion. Despite this, the weight of opinion appears to favour natural processes or the impact of grazing as the initiating agent.More widespread and more detailed studies of lake sediments within peatland catchments should be carried out, in particular investigating the timing of erosion events, the presence of charcoal. These studies should be accompanied by associated palaeo-archival investigation of the blanket peat in the catchment to correlate between lake and catchment the patterns of burning, recovery times identifiable from the succeeding vegetation archive, and evidence of erosion.

19.7.2 Erosion and recovery rates

The relationship between fire events (either documented or visible in the recent palaeo-archive), fire-age, ecosystem-recovery times and presence and extent of erosion has not been adequately investigated, perhaps leading to unwarranted assumptions about the lack of a relationship between ‘recent’ fire-evidence in the peat archive and the present eroded state.

A survey could be undertaken using historical aerial photographs and modern aerial coverage to identify examples of evident breakdown and loss of bog pool systems in parts of the blanket mire landscape where pool systems are typical or frequent (*e.g.* the Flow Country). Any such examples could then be visited and an assessment made of evidence for fire and/or heavy trampling. The recent peat archive should be examined to determine whether there is evidence for fire damage between the date of the original photograph and the present day.

Detailed studies based on palaeo-archival work and documented evidence of recent fires in a number of geographical areas and climatic conditions should be undertaken to determine evidence for recovery rates following fire events under differing conditions.

Peat cores taken in a grid across sections of blanket mire could be analysed in detail for their palaeo-archival record of fire events, looking in particular for evidence of associated erosion events and subsequent recovery and infilling of gullies. Such work would involve high-resolution 3-D palaeo-archival analysis combined with carbon-dating of the various profiles in order to determine any obvious age-discontinuities.

19.7.3 Erosion and grazing/trampling

While trampling by red deer, cattle and even horses may be a cause of peatland erosion, sheep have been more usually identified as the causal agent of erosion, but there has been insufficient analysis of the relationship between the introduction of sheep to Britain, together with the subsequent variation in their numbers, and the observed pattern of erosion present in the palaeo-archive of, for example, lakes in blanket mire catchments.

The inter-relationship between recorded grazing levels, fire events (either documented or visible in the recent palaeo-archive), recovery times from burning, and the initiation of erosion (rather than simply the intensification of existing erosion), should be investigated. To this end, detailed work is required into the relationship between the introduction of sheep to Britain, historical records of sheep numbers over the millennia, and the observed pattern of erosion present in the palaeo-archive of, for example, lakes in blanket mire catchments.

19.7.4 Blanket mire erosion in a global context

The relationship between burning, grazing and erosion in other parts of the world where blanket mire can be found should be the subject of a systematic investigation.

PART 4

REFERENCES AND APPENDICES

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21 APPENDIX 1: Origins of the confusing and sometimes contradictory evidence for carbon, peat and global warming.

The difficulties associated with peat, carbon and global warming potential begin with, but are far from restricted to, confusion arising from the fact that there is little consistency in the units used by different researchers to present their results. Authors of different papers use different units of measurement even though describing similar things, and some authors even switch to different units within their own papers without due warning. In theory, use of differing units would be fine, just so long as a conversion is also provided to some 'standard' unit. In practice, however, even conversion factors do not give a satisfactory solution. It is all too easy to miss an abrupt switch from g **CH4** m^{-2} yr^{-1} to g **CH_4-C** m^{-2} yr^{-1}, or at least to give pause for thought on encountering the latter. The resulting uncertainty thus obliges the now-unsettled reader to retrace his/her steps to the previous mention of this unit to check whether it was based on methane only, or methane carbon.

This confusion is compounded by the apparent lack of determination amongst those working on peat and carbon to ensure that, by using appropriate units, their data can be readily compared with other published data across the spectrum of interests. Thus some figures for methane emissions may be given in the form of **moles m^{-2} yr^{-1}** (weight divided by molecular weight per square metre per year), others quote **µmoles CH_4 m^{-2} h^{-1}** (micro-moles of methane per square metre per hour), others use **mg CH4 m^{-2} day^{-1}** (milligrammes of methane per square metre per day), while others present their data in the form of **t CH_4-C m^{-2} a^{-1}** (tonnes of methane-derived carbon per square metre per year).

In part, this reflects the scales and durations of measurement from particular experiments. A short study lasting a few days may generate figures which are best expressed in µmoles per day, whilst a study over a whole year can more easily express data in terms of grammes per year. Nevertheless this does not preclude authors from additionally providing conversions to a standard set of units designed to permit comparison between relevant studies. Without such standardisation and harmonisation it can be extremely difficult, if not impossible, to put the results obtained from any single study into a wider context.

There may be fundamental difficulties in converting to certain standardised units of measurement. For example, rates per *hour* may not be readily converted into rates per *year* because there may be significant seasonal variation in the response. In such a case, simple multiplication by hours and days to convert the data into rates per year would not be appropriate. However, in the absence of any reference to standardised units, or of any comment about the constraints associated with conversion to such units, readers may not have sufficient understanding of the processes being measured to recognise that a simple mathematical conversion from hours to years will not suffice.

Furthermore, readers may lack the knowledge to perform some conversions, or be so daunted by the mixture of units confronting them that they simply give up. Worst of all, from the reader's point of view, are those cases where a change in units is simply not spotted or understood, and the confusion resulting from such seemingly-conflicting values appears to be un-resolvable.

The following, taken from an official IPCC document, is typical of the problem:

> *"Using a Global Warming Potential (GWP) of 21 for CH_4, emissions of ~1.7 g CH_4 m^{-2} yr^{-1} will offset the CO_2 sink equivalent to a 0.1 Mg C ha^{-1} yr^{-1} accumulation of organic matter."*
>
> *Watson et al. website*

...so **grammes** are compared to **mega-grammes** - *i.e.* a metric **tonne** of carbon. Meanwhile the **CH_4 molecule** is directly compared with **CO_2-carbon only**, and a **square metre** is compared to a **hectare**. The only unit that remains constant in the IPCC statement is time, which is a year. This jumble of units occurs to no reasonable purpose in the sentence, and is a recipe for confusion rather than clarity. Everything could have been expressed in grammes of carbon for each molecule, and in square metres, per year. In its present form this guidance (for that is its purpose) can hardly be described as the most convenient ready-reckoner. Part of the reason for such un-necessary obscurity may lie in the tendency for carbon-balance scientists to use one set of units when describing methane studies, and another set when describing carbon dioxide. Given that scientific publications are written to communicate and illuminate, this is not an ideal state of affairs, especially when methane and carbon dioxide figures must so often be compared against each other.

A confusion of units, however, is not the most important issue to have emerged from the present work. Some confusion arises simply because of errors of calculation within certain published works. Other confusion arises because figures for differing sets of environmental conditions are used inter-changeably, or are compared without sufficient emphasis being given to the differences between such systems. There are, for example, examples in the literature where figures obtained for 'wetland' systems (which may or may not be peatlands) are compared directly with figures for 'boreal peatlands'. Equally, figures from such 'boreal peatlands' (which have a significant period of freezing during the winter months followed by a substantial spring snow-melt) are compared with data from Scottish blanket mires where there may be little or no period of extended freezing. Such comparison may be necessary because no other relevant data are available, but it is essential when doing so to highlight the substantial contrasts in conditions which exist between the two types of site. The highlighting of such difference is often absent from published literature.

Most significantly of all, however, and a surprisingly widespread issue, is the fact that insufficient care has been taken in identifying and describing the nature of the ground being investigated. In some instances, this amounts to the simple example of a habitat being described as 'peatland' when it is not. In others, the detailed nature of the ground under investigation plays a potentially important part in determining the nature of the data obtained, but this necessary level of detail is missing from both the description of, and thinking behind, the investigation.

Holden (2005a) highlights very clearly a number of ways in which small-scale factors can also substantially influence the data obtained from studies of peatland hydrology, carbon flux and restoration. Despite this, a significant proportion of the literature available in relation to the carbon dynamics of peatland systems is derived from studies in which the particular conditions prevailing on the study sites are not adequately described, or are not entirely as described.

Such identifiable differences between research-site descriptions and the actual conditions on the ground appear likely to have substantial significance for current accounts of carbon-balance in relation to British peatlands. This is all the more so when a further consequence of Holden's (2005a) observation about the significance of small-scale and local conditions is appreciated. During the course of the present review, what has become evident is that figures for the carbon balance of peatland systems in Britain are based on a remarkably small number of research studies and an even smaller number of research sites. Given that some of these have, in addition, not been adequately described or characterised, the whole basis of carbon-budgeting for peatlands in Britain (and probably Ireland) at present appears to resemble an inverted pyramid balanced on a remarkably small plinth.

22 APPENDIX 2: The Peatland Habitat – essential knowledge for working with peatlands

Peat consists of semi-decayed plant remains (strictly speaking, peat also contains bacteria, fungi and, to a much lesser extent, animals, but the vast majority of organic tissue is plant-based). To be classed as peat, these remains must have developed *in-situ* through plant growth ('autochthanous' deposits), rather than having been carried from some other location and deposited there by agents such as wind or water currents ('allochthanous' deposits). As we shall also see in the course of the present document, this definition has interesting, important, and rather unexpected consequences, particularly for the peatlands of western Britain and Ireland.

22.1 Peatlands as wetlands

The fact that any substantial plant debris exists as peat for any length of time at all is unusual, because most terrestrial ecosystems recycle virtually all of their components quite meticulously. Thus a fallen leaf in a rainforest may be recycled within days, and even a large fallen tree in a British oakwood may be reduced to its component nutrients within 200 years or so. Continued presence, and indeed accumulation, of semi-decayed plant fragments in the form of peat depends on the action of the second central character in the story – namely water.

Aerobic decomposition of organic matter (*i.e.* decomposition in the presence of oxygen) requires very considerable quantities of oxygen. This is because the constituent chemicals of plant matter – essentially lignin and cellulose – consist of long chains and rings dominated by carbon atoms. Every one of the many carbon atoms in these enormous molecules must be linked to two oxygen atoms if the complex of interlinked rings and chains is to be broken up and completely oxidised into individual molecules of carbon dioxide (CO_2).

Given the large number of carbon atoms involved, it is evident that aerobic decomposition of peat will require much oxygen. However, peat-forming plants grow in waterlogged conditions. This is a critical condition, because oxygen diffuses through water 10,000 times more slowly than it moves through air (Clymo and Hayward, 1982). Consequently the speed with which the available oxygen is used up by the disintegrating carbon-rich chains quickly exceeds the rate at which fresh oxygen can diffuse through the water to re-supply aerobic decomposition processes. As a result, the bacteria and fungi that use oxygen to bring about this chemical transformation quickly become starved of oxygen and cease to function, leaving only those organisms able to function in anaerobic conditions to continue the decomposition process. This transformation in conditions reflects (albeit in reverse) the very history of life on Earth itself.

Thus in wetlands, the growing aerial parts of the vegetation occupy a vigorous, modern, oxygen-rich environment, but the dead, decomposing remains of last season's growth may pass downwards into in a waterlogged, oxygen-free, Archaean-type environment . Indeed many of the micro-organisms causing decomposition in these anaerobic conditions are now termed Archaea (*e.g.* Basiliko *et al*., 2003; and see Archaea website). Such organisms are not able to keep pace entirely with the supply of fresh material from the now-dead and decomposing aerial parts. Consequently a proportion of the dead plant matter steadily accumulates as a peat deposit.

These conditions prevail as long as the peat-accumulating zone remains waterlogged. Should oxygen be allowed to penetrate this zone, by, for example, lowering the water-table through drainage, rapid aerobic decomposition can once more take place. There is, however, an important distinction to be made between a peat wetland which is still capable of laying down peat and a peat deposit on which a completely different habitat type, such as an agricultural grass ley or even a field of carrots, has been established. A mire is a peat-forming wetland, whereas a peatland in which carrots are being grown is merely a peat deposit, or a peat soil, which nonetheless continues to hold substantial quantities of carbon as long as the peat exists.

The elements thus assembled so far have revealed the important role of the third character in the story – namely decomposer micro-organisms. Dead plant matter provided with an abundance of oxygen will not, of itself, undergo significant levels of chemical oxidation. If decomposer organisms cannot function, no decomposition takes place. Supply this dead plant-matter with just sufficient water, however, and the aerobic micro-organisms can quickly digest such material. On the other hand, over-do this by providing too much water, and oxygen cannot reach the aerobic micro-organisms in sufficient quantity for them to function. Such an over-supply of water is typical in wetland conditions, and means that Archaean micro-organisms operating at low energy levels become the dominant decomposer organisms, but necessarily using the much slower process of anaerobic decomposition.

22.2 Life on Earth – a question of energy

The story of life on Earth consists of two key stages. Firstly, there was the origin of self-replicating, self-sustaining entities that we know as life, thought to have emerged some 3.4 billion years ago. Life was slow, sluggish and small in this Archaean world. At its most advanced form it consisted of photosynthetic bacteria growing in colonies known as stromatolites. Life was slow and sluggish because the atmosphere was dominated by gases such as carbon dioxide (CO_2) and methane (CH_4), which provided only rather feeble amounts of metabolic energy. The much more reactive oxygen, generated as a photosynthetic waste-product by the stromatolites, quickly became bound to dissolved iron in the seas and thus simply fell to the sea-floor as rust. Finally, some 2 billion years ago, the seas ran out of soluble iron. Consequently both the seas and the atmosphere became increasingly rich in oxygen.

Though oxygen is technically a highly-reactive poison, new life-forms eventually evolved that could harness this energetic molecule. As a result, some 700 million years ago, active complex life began to appear, fuelled by this much more effective source of energy. In modern times, oxygen-starved decomposition and oxygen-fuelled decomposition mirror these two great phases in life on Earth – slow and sluggish giving way to rapid and energetic. Most modern decomposition is of the latter sort because oxygen is now so readily available from the atmosphere. However, there are still conditions (such as on the abyssal ocean plain, or in the main body of a peat bog) where conditions reflect the original Archaean world of low-energy processes taking place at rates very much slower than those found in the oxygenated surface layers of both these ecosystems.

The fundamental basis for the interchange between carbon stores in peat and carbon stores in the atmosphere is thus based on the relationship between:

- the rate and nature of organic-matter supply;
- the quantity and nature of available water present;
- the consequent vigour of the aerobic microbial population;
- the rate and nature of aerobic decomposition;
- the rate and nature of anaerobic decomposition; and
- the resulting overall rate of organic-matter decomposition.

This series of relationships is obviously fairly complex in its own right, but unfortunately each element in the relationship is itself influenced by a wide range of factors which ultimately have a bearing on the character of each element. There is also the fact that these element interacts with each other. It is this complexity-built-on-complexity that goes a considerable way to explaining the wide variety of findings obtained, and the wide variety of conclusions drawn - many of them apparently contradictory - within the existing scientific literature concerning carbon, peat and global-warming potential.

23 Appendix 3: The fundamental architecture of peat-forming systems

It is important to understand that peat can form under a wide variety of conditions, and at a range of different scales - from small saxifrage-rich fens which have developed round a single spring-head to whole landscapes draped in blanket mire created by sedges and *Sphagnum* bog moss; from southern-hemisphere 'patterned' fens largely devoid of water in the dry season to tropical peat-swamp forests where the peat consists largely of tree roots and fallen branches.

Where a peat deposit demonstrably exists, there is no real debate about whether at the particular location in question a peat-forming system exists (or at least has existed in the past). At some point, this locality has clearly been actively sequestering CO_2 from the atmosphere and has remained sufficiently waterlogged to retain this CO_2 in the form of a peat deposit to the present day. Debate today tends to centre on the question of whether the area continues to sequester CO_2 and lay this down as fresh peat. Indeed, even if it is doing so, is the peat deposit beneath the freshly-growing surface at the same time disintegrating and releasing long-stored carbon, thereby cancelling out, or even exceeding, the sequestering capacity of this apparently vigorous system?

An area of peat on which the current surface is now a rye-grass ley, or a permanent cattle-grazed pasture, is evidently not actively laying down peat and is almost certainly losing its stored peat through the process of oxidative decomposition. It is still, however, a 'peatland'. In contrast, a 'mire' is the accepted terms for a system that is considered to be still peat-forming.

23.1 Mire systems – defined by their source of water

There are two broad types of mire – those where waterlogging is created by groundwater, and those where waterlogging results from direct precipitation inputs. The former, groundwater-fed mires are called 'fens', whereas the latter precipitation-fed sites are called 'bogs'. It is often difficult to say whether a 'peatland' which now consists of a green agricultural field with grazing cattle was originally a fen or a bog, because much of the original peat may have been lost, leaving only the lower parts of the peat deposit. Thus such areas are best referred to simply as 'peatland' with a note of what the remnant peat indicates as a possible former mire type, and this type can then be used as a guide for any initial restoration efforts.

23.1.1 Fens – fed by groundwater

The water of fenland systems is influenced largely by the local geology and terrain. Thus fens which have formed along rivers in wooded chalk landscapes tend to have high calcium levels, low levels of nitrate, and water regimes governed by river levels and flood patterns. Fens formed in enclosed basins on clay and surrounded by intensively-farmed agricultural land tend to have high levels of nitrate and phosphate, but very stable water levels. Fens formed in shallow valleys on acidic rock such as Lower Greensand may have a water chemistry not so different from rainwater. Their slow rate of water seepage combines with the low level of nutrients to an extent that the system almost resembles the conditions and vegetation found on a true bog system. This similarity is mere illusion, however. Change the nutrient regime of the catchment by adding a leaking slurry tank somewhere within the catchment and the difference between this kind of system and a true bog will quickly become apparent.

23.1.2 Bogs – fed directly by precipitation

Bogs form where the peat thickness is sufficient to separate the living vegetation from both the underlying mineral ground *and* from the associated mineral ground-water table. This happens most often in areas of moderately regular, year-round precipitation, and in lowland Britain can result in large domes of peat that rise more than 10 m above the surrounding landscape. For obvious reasons these are known as 'raised bogs'.

23.1.3 Blanket mires – complex mire systems

Where cloud-cover and regular precipitation are sufficiently frequent, rain-fed peatlands can become much more extensive than the relatively simple, isolated 'lenses' of lowland raised bogs. The more humid climate means that peat can form a more-or-less complete mantle across entire landscapes. This is known as 'blanket mire'. The fact that this peat mantle drapes itself across the landscape means that there are important difference between 'classic' raised mire and blanket mire. In the former, the domed shape of the raised mire is almost entirely due to the accumulated peat deposit, whereas much of the landform of a blanket mire reflects both depth of peat and the slopes and gradients of the underlying mineral ground. The thickness of peat may smooth out some of the undulations in the landscape, but the landform profile remains largely determined by the shape of the underlying mineral terrain.

Consequently the landform of a blanket *mire* is generally much more complex than that of a raised mire. Indeed it may embrace a wide variety of peat-forming conditions including emerging spring fens, zones of evident surface-water seepage ('flushes'), wide slopes down which there is less-evident surface flushing. There may be 'patterned' fens, areas that resemble raised mire domes, and many more peat-forming systems, as well as areas of thinner peat more correctly described as 'wet heath'. These all form an interconnected mosaic of extremely variable habitat conditions, but are often simply referred to in scientific literature as 'blanket bog' but more accurately termed 'blanket mire' because not all the peat-forming systems making up the peatland expanse are bogs. It is vital to have a clear picture of the precise nature of any given blanket mire complex being investigated if a meaningful picture is to be obtained of its carbon budget.

23.1.4 Bogs – a system with two layers

Despite the sometimes substantial differences between raised mires and blanket mires, one thing they generally share is the fact that the peat in a bog tends to exist as two distinct layers. The upper layer is thin – anywhere from a few centimetres to 75 cm - and represents both the layer of living plant material and the zone of water-table fluctuation (and thus of direct oxygen penetration). This layer is known as the 'acrotelm', and represents the narrow zone of relatively rapid oxidative decomposition in the bog.

The acrotelm typically consists of a moss carpet, thus giving rise to a structure with loosely-packed vertical plant stems and small but quite stiff side-branches, which together create a relatively open lattice structure in the upper parts near the bog surface. Bryophytes differ from higher plants in dying upwards from the base, their apical regions continuing ever upwards as their dead remains are left behind and below them. Thus while the uppermost part of the living *Sphagnum* carpet is an open, structured environment, only 10 cm or 20 cm below the surface this structure and order begins to break down as the dead branches collapse and stems begin to weaken and fail (Clymo, 1992).

Below the base of this increasingly disordered acrotelm layer is the remaining thickness of peat, which may be 10 metres or even more. This generally much thicker layer of peat is known as the 'catotelm'. It consists of plant remains that are now largely broken into small, non-oriented fragments, creating a remarkably amorphous structure. This catotelm peat always lies beneath the water table and is thus the zone of the oxygen-free Archaean world. The catotelm thickness gives the raised mire system its domed shape, and is what smothers or smoothes the shape of the underlying mineral landform in blanket mire landscapes.

The catotelm thickness increases steadily over millennia as small increments of peat are added to it from the acrotelm each year. An addition of only 1 mm per year means that the catotelm can attain a prodigious thickness over a period of 10,000 years – specifically 10,000 mm (*i.e.* 10 m). Something not always understood about this often very deep catotelm peat is that there is no re-working of the accumulated layers by worms, as there would be in mineral soils. This is because there are no earthworms in peat soils, only very small nematode worms. Consequently the thin accumulated layers generally remain stacked in chronological order, which can be very convenient when attempting to re-construct past events which may be recorded in some way within the peat archive.

23.2 Peat – highly variable in origin and composition

It should be clear from the various descriptions given above that peat is actually a highly variable material, in the sense that it can be created by remarkably differing sets of original plant materials. In Australia it can be formed by the root-mass of the horsetail-like plant *Empodisma*, in Tierra del Fuego the dense saxifrage-like cushion plants *Donatia* and *Astelia* are significant peat formers. In Indonesia, peat occurs as waterlogged accumulations of tree roots and branches, while in the fens of Eastern England, peat can be formed by the dense accumulations of leaf-bases and roots of saw-sedge. What is perhaps the most remarkable thing about these differing forms of peat is that they all have some underlying similarities in their properties – essentially a rather amorphous structure containing many empty spaces but nonetheless a structure that resists rapid water movement through it.

Clearly there must also be sufficient water within the peat to give rise to anaerobic conditions and thus re-create the Archaean, oxygen-devoid environment necessary for peat to accumulate. However, in the case of many fen systems and some parts of blanket mires, this water brings with it many other things. It may contain high concentrations of soluble minerals derived from the regional or local catchment – in other words, from ground-water inputs. It may also contain significant quantities of sands, clays or silts provided by river sediments or eroding mineral ground, particularly if the peat-forming system is, for example, a flood-plain fen. Even some raised bogs contain narrow lenses of such sands, clays or silts where they have in the past been subject to adjacent riverine flooding or marine incursions caused by sea-level rises.

Furthermore, the atmosphere provides more than merely water as inputs – it may bring dusts from local or even very distant sources, and ash from volcanic eruptions. Thus some bogs have relatively high calcium levels because dust is blown from nearby limestone cliffs, while there is now a well-defined zone in European peats marking the Chernobyl disaster, and other fine ash layers can be linked to specific volcanic eruptions. In Japan, Kamchatka and Iceland, volcanic ash is such a major component of many peatland systems that the character of the vegetation is transformed (Hotes, 2004). Equally, bogs formed on the Atlantic seaboard of Europe – particularly the extensive blanket mires of Britain and Ireland – may receive significant quantities of marine solutes blown inland by westerly gales (Moore and Bellamy, 1974). Even without this, rain itself is slightly acid because it reacts with atmospheric carbon dioxide to produce the weak acid carbonic acid and associated carbonate ions. In the absence of any other mineral inputs, therefore, the peat will at least contain carbonate ions – which consist of a carbon atom linked to oxygen atoms, and so this universal form of carbon input to the world's peatlands constitutes part of what is termed DIC (dissolved inorganic carbon) inputs.

Recognising that many peatland systems have significant mineral inputs, the reality is that 'raw' peat soils lie at one end of a continuum. At one extreme are conditions where the peat consists almost entirely of organic matter derived from what dissolves in precipitation or is absorbed as a gas (largely carbon, oxygen and hydrogen). Then there are conditions where significant amounts of mineral matter are mixed with the organic material to produce an organo-mineral soil. Finally there are those conditions where only a small proportion of organic matter exists and the soil consists almost entirely of mineral components. It is important to identify the position on this continuum at which the soil-mineral mixture can be called 'peat'. There is, however, no absolute or universally-agreed position for this threshold.

Joosten and Clarke (2002) and Rydin and Jeglum (2006) highlight the wide range of values for % organic matter used by different authors to separate 'peat' from categories such as 'peaty organic soils' or 'peaty muck'. On this basis, the required % organic matter for 'peat' may lie anywhere between 5% to 80%, which is a huge range, but it is important to emphasise that these differing thresholds have been adopted for different purposes – specifically, as Joosten and Clarke (2002) emphasise, for different *uses* of peat. For any given sample of 'organic soil', however, it is universally agreed that the % mineral content of ombrotrophic bog peat is generally extremely low, the peat solids consisting almost entirely of organic matter. This has important implications for density measurements ('bulk density') of bog peat, because organic matter is relatively light compared to grains of mineral soil or fragments of rock, and, as we shall see, bulk density plays a key part in the peat-carbon story.

Given these several potential sources of variability, what will be described initially in the present report is a 'typical' example of simple British raised bog peat. This represents a peatland fed only by atmospheric precipitation falling directly onto the living peat surface and whose general surface is raised sufficiently above the surrounding mineral ground-water table to be well clear of any influence from that source. There have been no major inundations by flood water in the past, nor have there been there any significant inputs of dust or marine salt spray.

23.3 Peat – 'mostly water and holes'

The above discussion about the % composition of mineral and organic matter can seem rather small beer, however, when it is recognised that peat, or at least bog peat, is generally around 90-95% water, by weight. It is not unusual for it to reach as much as 98% water by weight. It is thus often said that bog peat has less solids than milk - indeed the wettest bog peats have a higher % water-content than a jellyfish. From this perspective, even differences of between 5% and 80% for the organic/mineral fractions amount in total to less than 10% (often much less) of the total *in-situ* peat mass. Water represents by far the major component by weight *and* by volume within the main body of ombrotrophic bog peat. This is because the mineral component is very small, the organic matter component is very light, and this combined solid matter is both arranged as, and consists largely of, an open latticework of holes.

The water in bog peat distributes itself within this latticework in four ways:

- water filling large voids between individual moss stems and plant fragments - 'interstitial' water - (Clymo and Hayward, 1982);
- water occupying 'macropore' spaces greater than 1 mm between peat fragments ('matrix' water) (Holden and Burt, 2003);
- water occupying spaces of 1 mm down to 1 μm between fragments, or which is tightly attached as a thin film to the peat fragments ('intra-particle' water, 'surface-film', or 'capillary' water) (Clymo and Hayward, 1982), and
- water within smaller spaces and water contained within the cells of the peat fragments ('intracellular' water) (Clymo and Hayward, 1982; Hobbs, 1986).

Clymo and Hayward (1982) characterise the behaviour of the different forms of water by observing that a hole dug in peat will fill fairly readily with interstitial and macropore water. Drain away this water, however, and the hole does not then fill with intra-particle or intracellular water. They vividly liken this to a jelly, which like a peat soil may be totally water-saturated, but holes and depressions created in the surface by a jelly mould do not spontaneously fill with this tightly-bound water. Instead of leaking from the gelatine matrix and flooding the surface microtopography of the jelly, the water remains tightly bound within the matrix and the jelly retains its shape.

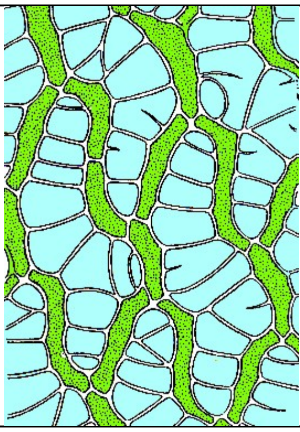

Figure 61 Sphagnum cell structure
Living cells with chlorophyll are shaded green. Dead hyaline cells, filled only with water, are shaded light blue. Note spiral thickening visible as cross walls in hyaline cells.
Adapted from Guerra and Cros (2004).

The last but nonetheless substantial form of water, intracellular water, highlights the fact that even the peat fragments themselves consist mainly of storage spaces, or holes, which can be seen in drawings of typical *Sphagnum* bog moss cells.

The photosynthetic ('chlorophyllous') cells in Figure 61 can be seen squeezed into long narrow strips between large, dead, empty cells known as 'hyaline cells'. These hyaline cells are used purely for water storage. Water enters or exits through the occasional pores visible in the hyaline cells.

The cells have spiral thickening to prevent total collapse when not filled with water. This spiral thickening is visible as the diagonal lines crossing each cell in Figure 61. Meanwhile the stronger surrounding framework of living chlorophyllous cells provides rigidity to the whole structure, in effect providing a stiff internal mesh to the open geotextile construction (Clymo and Hayward, 1982).

But peat, even bog peat, consists of more than just *Sphagnum* bog-moss fragments. It often contains a high proportion of much denser roots and leaf bases from higher plants. Nonetheless, because a bog is a wetland, the main plant species to occur here are all termed 'hydrophytes' – that is, plants adapted to wet places. One of the defining characteristics of hydrophytes is that they contain significant quantities of aerenchyma cells. These are essentially large empty cells allowing oxygen to be transported to parts of a plant where waterlogging would otherwise cause these parts to die because there is not enough oxygen in the surrounding environment.

In plants such as the common cotton grass (*Eriophorum angustifolium*), aerenchyma cells are sufficiently large for a particular family of beetles, the donacid or 'jewel' beetles, to live in the aerenchyma cells. The iridescent wing cases of these beetles can often be found preserved in the peat (see Figure 62).

Figure 62. Donacid wing cases.
Donacid wing cases (elytra) preserved in the peat of a raised mire in Galicia, NW Spain. The elytra can be seen as iridescent blue objects lying with the fibrous peat matrix which consists largely of cotton-grass (*Eriophorum angustifolium*) leaf sheaths. This particular sample came from a depth of approximately 2 m.

Photo R A Lindsay

The presence of aerenchyma is significant for two reasons. Firstly it means that much of the non-*Sphagnum* peat also consists of spaces or holes, further reducing the amount of solid matter within the peat. Secondly, this aerenchyma system has the potential to play an important part in transporting gases both ways between peat and atmosphere.

According to van Breemen (1995), the living layer of a *Sphagnum*-dominated bog is up to 98% pore space. This pore space consists of the gaps between *Sphagnum* plants, gaps between the overlapping leaves of each plant, and up to 20% is taken up by the spaces contained within the hyaline cells which form the major part of each leaf. In other words, only 2% of this living *Sphagnum* layer has any possibility of contributing directly to the carbon story – although the spaces themselves can obviously play a part in storing and acting as a conduit for gaseous materials.

The breakdown of this open, space-filled lattice-work, and the consequently dramatic decline in the ease with which water moves through the peat, is probably one of the most important steps in the whole story of peat and carbon, because without it there would be no major waterlogging, there would be no dense accumulations of carbon-rich matter, indeed it is quite possible that there would be little or no accumulation of peat at all. This breakdown is such a key step that it is important to consider it, and all its implications, in some detail if the peat-carbon relationship is to be understood clearly. The story of the transition from an ordered living surface consisting mostly of empty space, to a relatively dense and amorphous peat matrix beneath, is the story of the acrotelm-catotelm relationship.

23.4 Acrotelm, catotelm, decomposition and waterlogging

As described above, the upper, living parts of a *Sphagnum* carpet are generally arranged in a highly structured way whereby their branches and stems form an open geotextile-like structure through which water can flow very easily. When measuring the hydraulic conductivity of fresh, undecomposed bog peats, Boelter (1965, 1972) gives conductivities of 33-140 m day^{-1} (metres per day), while Gilman (1994) observes that on some occasions permeabilities can be too large to measure in these undecomposed surface layers because there is little or no resistance to water flow.

If plant material in the acrotelm did not undergo a degree of progressive breakdown and decomposition within the upper acrotelm layer, then bogs would consist of nothing but acrotelm – probably a fairly thin acrotelm at that. This is because water would flow through, and then out of, the open acrotelm matrix too freely to create a permanently-waterlogged layer resulting from greatly reduced water movement. It is the presence of this dense, permanently-waterlogged layer at the base of the acrotelm that causes peatlands to create their most characteristic and eponymous product – namely, peat. Without a zone of reduced water movement there would be no lower, waterlogged layer in which carbon-rich peat could become concentrated. 'Bogs' would instead be thin layers of *Sphagnum* moss lying over a mineral base, fairly swiftly shedding whatever water came their way much as any other vegetation type does. The breakdown of this open matrix is thus critical to the formation of the main peat carbon store. The ease and speed with which this breakdown occurs in the acrotelm also has a profound effect on the nature of the final carbon store in the catotelm.

Clymo (1992) provides a pictorial model of the acrotelm to illustrate the architecture of this layer. Figure 63 is based on Clymo's (1992) diagram of acrotelm structure. In addition to showing the somewhat abrupt structural collapse of the *Sphagnum* plants described by Clymo (1992), Figure 63 also highlights the fact that the roots of species such as cotton grass (*Eriophorum* spp.) penetrate some way into the catotelm. This can create localised pockets of somewhat more aerobic conditions because oxygen can diffuse from the aerenchyma tissue into the immediately adjacent peat.

For the majority of the acrotelm, water is able to move both vertically and laterally through the open lattice-work (evident in the upper parts of Figure 63) with comparative ease, and thus the water table tends to respond quite rapidly to precipitation events. The closer the water table rises to the surface the more easily it can flow laterally away through the lattice (Bragg, 1982, and see Figure 63). Thus during periods of heavy rain the upper part of the acrotelm is able to shift water rapidly through (or across, if bog pools and hollows are present) the surface layers and away downslope.

The thin layer of living vegetation is thus subject to a fluctuating water table, as is a good proportion of the dead plant litter consisting of dead *Sphagnum* branches and leaves, the leaf-bases of plants such as cotton grass, and the roots of many shallow-rooted species. All of this acrotelm material contains carbon, and all is subject to (largely) aerobic decay. Towards the base of the acrotelm, however, and on into the peat deposit below, the story is very different. Any plant material reaching this point enters the Archaean world of the catotelm.

The process of breakdown, collapse and compression towards the base of the acrotelm, as illustrated in Figure 63, means that the spacing between particles becomes much reduced and thus it becomes increasingly difficult for water to pass through the matrix. Clymo (1992) observes that the rate of water movement is heavily influenced by the size of channel through which it must flow. Resistance increases by a power of 4 as channel size decreases. This can mean a 256x increase in resistance to water flow through this denser peat matrix.

The progressive collapse of *Sphagnum* structure significantly reduces the amount of space between individual plants and between individual leaves. The majority of hyaline cells *within* the *Sphagnum* leaf structures, however, remain intact. Consequently towards the bottom of the acrotelm the hyaline cells now contribute as much as 70% of the surviving pore space remaining in the increasingly dense peat matrix (van Breemen, 1995). The point about this pore space is that most of it is going nowhere because the hyaline-cells are not interconnected. Water in such cells may still be lost through evapotranspiration, or gained through absorption directly from the peat-matrix water supply, but what these hyaline pore spaces do *not* do is form an interconnected transport system. Such pore spaces are likely to be filled with water but this water is, in effect, locked up in the individual cells. Rather like the inmates of a prison, entry and exit from a cell is controlled, whilst movement between cells is not normally permitted.

Water does not flow easily in any direction through this denser layer, but there is very slow downward movement of water resulting from the relentless pull of gravity. However, the rate of flow is so slow that downward seepage from the acrotelm can replenish such losses relatively easily. With a rate of water movement 256x slower than that of the acrotelm, adequate re-supply of water from the acrotelm is virtually guaranteed for the catotelm provided there is at least occasional precipitation.

Boelter (1972) gives conductivity values which range from 62-0.02 m day^{-1} for catotelm peat as an example of the way in which water movement is reduced to no more than a slow seepage through the catotelm peat matrix. van Breemen (1995) cites even slower figures, ranging from 8.64 m day^{-1} to as low as 0.0009 m day^{-1} (equivalent to 10^{-7} cm sec^{-1}).

Consequently the catotelm remains almost entirely immune from the fluctuating conditions of rain and sun experienced by the acrotelm. The slow and steady trickle-down supply of water from the compacted base of the acrotelm keeps the catotelm in a constantly-waterlogged state even during long dry spells in the summer. Thus although the catotelm is composed almost entirely of water (by weight), most of the water itself is so tightly bound up in the catotelm peat matrix that its escape from this embrace involves speeds so sluggish as to make actual molluscs appear quite lively by comparison[4]. Any additional water input over-and-above what is needed to replenish the small quantity of gravity-driven loss tends therefore to collect on top of this waterlogged layer, within the acrotelm, and either flows away laterally through/across the bog surface, or is returned to the atmosphere by evaporation or by transpiration via the living plants.

The only oxygen available in the slowly draining, waterlogged catotelm environment is found in localised pockets around the living roots of aerenchyma-rich plants such as the common cotton grass (*Eriophorum angustifolium*). Where their deep-roots penetrate some way into the catotelm, small amounts of oxygen may be released into the peat by diffusion. These isolated pockets occur only within the root zone of the deepest-rooting species – i.e. only within the uppermost 50 cm or so of the catotelm. The major part of the catotelm is thus much more like the ancient Archaean world, being devoid of both air and the highly reactive oxygen molecule. Any plant material in the catotelm will be attacked only by the slow actions of anaerobic micro-organisms. These break down the long-chain carbon molecules to produce methane (CH_4), rather than the carbon dioxide (CO_2) which would result if oxygen were available.

There are two distinct (though related) aspects of peatlands which are significant in terms of carbon cycling and climate change. Firstly there is the question of carbon *balance* – namely, how much carbon is sequestered from the atmosphere by the living vegetation compared with the amount of carbon released by breakdown of that same vegetation when it dies. This dynamic process is true of all vegetated terrestrial ecosystems, and, precisely because it is so dynamic, may display considerable variability from one year to the next - a series of poor summers can significantly reduce productivity for a time. Nonetheless it is this process on which the Kyoto Protocol has principally focused.

What sets peatlands significantly apart from other ecosystems, however, is their capacity for carbon *storage*. These stores do undergo changes with time and there is thus a dynamic element to them, but the changes involved over a year or even a decade are generally quite small compared to the scale of stored material. The carbon store of a peat-dominated landscape thus tends to be treated more as a static feature of the peat, a figure for which there is ultimately a single correct and measurable answer. The extent to which this is true will be examined in due course, but before doing so it is necessary to introduce one other fundamental aspect of the peat bog environment – namely the small-scale surface patterns which are such a characteristic part of the peat bog habitat.

What follows, therefore, is a somewhat extended but necessary diversion into the functional structure of peatland ecosystems because, as we shall see, an understanding of the way in which multi-scaled structure provides an integrated and functional ecosystem is absolutely essential to an understanding of the carbon balance for any peatland.

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[4] *"A garden snail named Archie, owned by Carl Branhorn of Pott Row, England, covered a 13 inch course in 2 minutes (0.0028 m/s) at the 1995 World Snail Racing Championships, held in Longhan, England. The Guinness Book of World Records 1998. Stanford, CT: Guinness, 1997: 144."* This is equal to 234 m day^{-1}, more than 1 million times faster than the slowest catotelm seepage rates, and more typical of speeds associated with acrotelm seepage (See Speed of a Snail website)

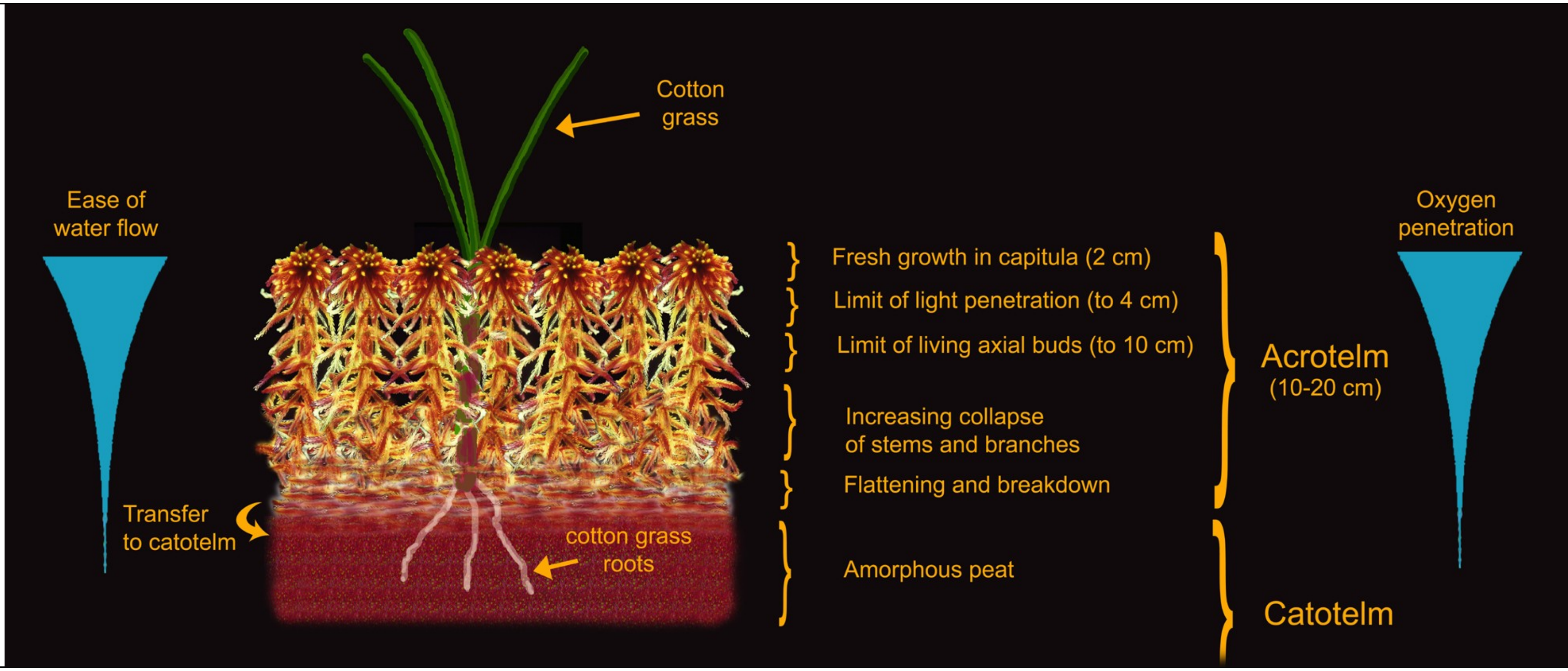

Figure 63. Acrotelm structure and properties.
The essential structure of the acrotelm is illustrated in the generalised photo of a *Sphagnum* carpet, within which is growing a single individual of common cotton grass (*Eriophorum angustifolium*). The different layers within the acrotelm, as described by Clymo (1983), are shown here. Also shown on the far left of the diagram is a representation of the changing rate at which water can flow through this surface layer, while on the far right of the diagram is a representation of the extent to which air can penetrate this layer.

Based on Clymo (1983, 1992)

24 Peatlands and pattern: the fine-scale architecture of peatland systems

Peat bogs are generally recognised and referred to as entities – Featherbed Moss, Blar nam Faioleag, Moor House. In the lowlands, such entities tend to occur as single, discrete units of peat in the form of raised bogs (though as we shall see, in their natural state they are neither as single nor as discrete as is generally thought). In the uplands, these named entities may represent individual lenses of peat (*e.g.* Featherbed Moss) which form part of an interconnected peat mantle, or the name may refer to the whole interconnected mantle itself (*e.g.* Moor House). While the name of peat bog generally refers to an entire and more-or-less distinct landscape unit, even the earliest descriptions of peat bogs in modern times (*e.g.* Weber, 1902 : Couwenberg and Joosten [transl.], 2002) have recognised that one of the most characteristic and distinctive features of a peat bog is the architecture of its surface layer. This architecture has become almost universally known as 'hummock-hollow', generally thought of as an intimate mosaic of small-scale, rather dry raised mounds of moss rising to 40 cm or so (the hummocks) between which are equally small-scale, water-filled depressions in the bog surface (the hollows).

In reality, this popular vision is true only in the sense that bog surfaces do broadly consist of two structural components, one elevated and relatively dry, the other lying at a lower elevation and being relatively wetter. This two-part structure is the essential key to understanding a great many aspects of peat bog and carbon balance processes. It is unfortunate, therefore, that the entire basis of this two-part structure was misinterpreted at an early stage. This led to the widespread dissemination and acceptance of a model which completely failed to appreciate the underlying relationship between the surface structure and the peat bog system. The unfortunate legacy of this theory continues to be felt today.

24.1 The 'hummock-hollow regeneration cycle' vs. 'phasic change'

In the years immediately before and after World War 2, field excursions and discussions between the Swedish peatland specialist Hugo Osvald and the leading British ecologists Arthur Tansley and Harry Godwin led to the development and promulgation of a model describing peatland surface dynamics based on the idea of ecological succession. The model assumed that bog hummocks grew up so far beyond the bog water table that they eventually became too dry for peat growth, and would thus decay and collapse to form new hollows. Hollows, meanwhile, would infill rapidly with bog moss (in the manner of infilling shallow lakes – a favourite example of succession) and grow sufficiently vigorously to form new hummocks. In this way, hummocks became hollows and *vice versa* in a never-ending cycle of small-scale dynamic succession and collapse forming a mosaic across the bog surface (Osvald, 1949; Tansley, 1939; Godwin and Conway, 1939).

This model was sufficiently persuasive to be widely accepted, and can still be found in many general descriptions of peat bog systems today. It was, unfortunately, wrong. It also, equally unfortunately, diverted attention away from an earlier theory of bog pool dynamics which has a most modern and timely relevance today:

> *"...the unambiguous relation between pools, particularly at the edge of the plateau, and hummocks is explained. Both are consequences of an alternation of wetter and drier periods. Numerous pools will become overgrown during later dry periods. Yet some will survive such periods and become deeper with continuing growth of the surrounding peatland area. I therefore view the deepest peatland pools as the oldest. The pools are evidence, as hummocks also seem to be, of the constant change the raised bog surface shows under the varying influence of long-term weather on the vegetation and the peat. As long as the natural vegetation remains, the raised bog somewhat resembles a slowly pulsating organism reacting to outside influences in a particular fashion of its own. Only when it is destroyed do stasis and permanent decay begin."*
>
> *Weber, 1902 : Couwenberg and Joosten [English transl.], 2002*

Thus, 50 years before the 'hummock-hollow regeneration cycle' was devised, Weber was suggesting that the main cause of pool infilling was not natural succession[5] but a response of the bog system to changes in climate patterns, whereby pools infilled during dry phases in the climate and hummocks became less abundant during wet phases. It was only finally with the painstaking work of Barber (1981) that the Osvald/Tansley theory of hummock-hollow regeneration was shown to have little real basis in evidence. He demonstrated instead that hummocks and hollows respond to shifts in climate much as Weber (1902) had proposed 80 years earlier. There was no cyclical degeneration of hummocks to hollows and growth of hollows into hummocks – the hummocks merely became squeezed by expanding hollows when the climate moved into a wet phase, then the hummocks in turn expanded to squeeze the hollows during drier phases in the climate.

This 'phasic change' model described by Barber (1981) was immediately given added support with the English translation from the original Russian of K.E. Ivanov's classic work "*Water Movement in Mirelands*". In this, Ivanov (1981) describes how the surface hydrology of a bog is controlled by the interplay between two structural elements, one element being significantly more permeable to water than the other element. He dubbed this the 'strip-ridge' structure. By varying the proportions of these two contrasting 'strips' of surface structure across a bog, the system is capable of maintaining a relatively constant hydrological regime over several thousand years in the face of dramatic climate shifts.

Thus the small-scale structures (microtopography) found across a bog surface are not merely rather curious localised examples of a successional cycle with no relevance to the wider bog environment, as suggested by the Osvald/Tansley model of the hummock-hollow regeneration cycle. The microtopography of a bog is absolutely fundamental to the long-term maintenance of the bog system because it by means of these small-scale structures that the whole system is able to respond to wider changes in the surrounding environment.

It is therefore important to be clear about the precise nature of these small-scale structures – to be unambiguous about what the microtopography of a bog actually consists of. As we shall see later, there are also other reasons for needing to be clear about the nature of the microtopography, reasons which have direct relevance to carbon balance measurements on bog systems.

Unfortunately and almost universally, this is one of the most poorly-described features of peat bog systems. While 'hummock' tends to generate a reasonably consistent image in people's minds, even this is open to a substantial degree of mistaken identity and misinterpretation. The term 'hollow' is even more fraught with difficulties of interpretation, but even this pales beside the almost complete failure to devise a vocabulary and classification system capable of describing the patterns which these various structural elements ultimately create – yet these patterns are the critical step whereby individual structures can work collectively to provide a homoeostatic system for the bog environment. Yet if these structures and patterns are not adequately described, how can judgements be made about the significance of these for published research results? As Weber (1902) observes in relation to the need for accurate descriptions of bog vegetation:

> *"Unfortunately, the interpretation of the observations of others, which would have been of the utmost value to me when discriminating the generally valid from the accidental, is often uncertain or even impossible because the primary and secondary state of the peatland vegetation has either not been distinguished at all or only too vaguely."*
>
> *Weber, 1902 : Couwenberg and Joosten [English transl.], 2002*

In this case, instead of Weber's "primary and secondary state of the peatland vegetation", read: "microtopography". Weber himself is generally very precise in his use of terminology, but even he uses several different words for the entity 'pool' (Couwenberg and Joosten, 2002). Such inconsistent naming of the individual small-scale structures which form the bog surface is not a problem restricted to Weber (1902). It has been a common theme throughout the peatland literature of the past century, and the consequences of this for the peat-carbon story are significant.

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[5] Ecological succession was not proposed as a formal theory until more than a decade after Weber's 1902 publication (Clements, 1916), but the general ideas of progressive ecological change had been recognised long before 'Clementsian succession' was set out as a formal, integrated and all-embracing model – a model which soon came under vigorous and sustained attack and which remains controversial even today.

The small-scale structures which may or may not be hummocks and hollows combine collectively in various ways to produce areas of distinctive surface pattern. It is actually these surface patterns (rather than the individual small-scale structures) which often reveal an area to be a peatland, particularly when viewed from above in an aircraft, or from aerial photographs or a satellite image. These patterns are themselves arranged in particular relationships with the overall morphology of the peatland unit, as was presciently identified by Weber (1902) more than a century ago. The distinct nature of each pattern group can also provide a valuable insight into the nature and type of each individual peatland unit as a whole; certain patterns are more typical of bogs, while others are more usually associated with fens. Where these units are interconnected to form a continuous peatland complex at the landscape scale, as is the case for blanket mires, the overall complex can then be characterised by the assemblage of individual peatland units making up the complex.

This integrated arrangement of peatland systems, interlinking small-scale structural features through increasingly larger-scale structural entities which all ultimately form part of a coherent landscape unit, is fundamental to an understanding of how almost all peatland environments work, from the smallest spring-head fen to the largest blanket mire landscape. Fenland peat systems have the added complication of the catchment from which the bulk of their water is obtained, but for blanket mire landscapes the major part of the whole system consists of two fundamental components - the interlinked hierarchy of peatland structures, and the sky (from where the major water inputs are derived).

Ivanov (1981) provided the basic descriptive framework for this integrated hierarchy, but the framework has since been developed and somewhat expanded by Lindsay *et al*. (1988), Lindsay (1995) and Couwenberg and Joosten (1998). It is also worth noting that national peatland programmes in other countries (*e.g.* Zoltai and Pollett, 1983, for Canada; Moen, 1985, for Norway) have devised somewhat similar models based on a hierarchy which closely matches that of the basic Ivanov (1981) system, lending strength to the argument that the concept of a hydro-morphological hierarchy reflects a fundamental truth about the way in which peatland ecosystems are organised.

The scientific value of using the system, and its long scientific pedigree, is perhaps most clearly set out by Charman (2002):

> *"A systematic description of landforms is crucial to the understanding of fundamental processes in peatlands. The classifications based on hydromorphological criteria presented in Chapter 1 depend heavily on a consideration of size and shape, and the basis for these ideas is enshrined in the Russian school of peatland science most closely associated with, and accessible through, the work of K.E. Ivanov. The book "Water Movement in Mirelands" (Ivanov, 1981), made available in English by Thompson and Ingram, summarises the approaches to peatland hydrology in post-war Russia and is perhaps best known for its influential presentation of broad concepts of peat landform units. The ideas presented by Ivanov are based on earlier work (principally that of E.A. Galkina) and emphasise the importance of scale in consideration of peat landforms through the use of the terms microtope, mesotope and macrotope."*
>
> *Charman (2002), p.26*

Having reviewed the long gestation, and the significance, of this hierarchy and its associated patterns and small-scale structures referred to earlier, perhaps now is the moment to see in detail what they consist of and actually look like.

24.2 Hydro-morphological hierarchy of peatland ecosystems

To describe the hierarchical system which forms the essential architecture of peatland systems, it is easiest to begin at the highest level – the landscape scale – and work downwards through the hierarchy to the finest level of structural detail. The hierarchy is summarised in Figure 64, which should be viewed in conjunction with the descriptions given below.

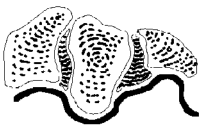
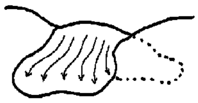
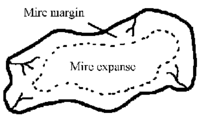
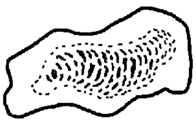
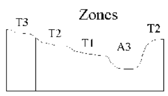
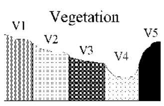
Feature	Hierarchical level	Description	Hydrological relationship	Utility for classification and evaluation
	Macrotope	Assemblage of hydrologically linked mire units	Individual bog units hydrologically linked via intervening fens and stream-courses	Identification of boundary for minimum, hydro-logically sound, conservation unit
	Mesotope morphology	Distinct, recog-nisable hydro-topographic unit.	Inputs of rainfall, outputs of seepage, drainage and evapo-transpiration	Identification of individual, recognisable units for comparison
	Mesotope sub-sectors	Distinction between mire-margin and mire expanse.	Broad patterns of water movement within the mesotope, from high ground to low ground	Recognition of 'core' and 'marginal' zones; in Europe, the margin often partly removed
	Microtope	Repeated surface patterns - *e.g.* pool system.	Surface pattern reflects hydrology of acrotelm layer and overall mire gradient	Identification of naturalness; source of comparative diversity
	Nanotope	Individual surface features (*e.g.* hummock, pool)	Small-scale water movements within the acrotelm	Source of niches for individual species; comparison of diver-sity and damage
	Vegetation	Distribution of vegetation within surface structures.	Ultimate control of acrotelm and surface water movement	Source of comparative diversity; indicator of "naturalness"

Figure 64. Hydro-morphological hierarchy of mire systems
Hierarchical relationship between the various functional levels of peat bog systems, from the large-scale concept of mire landscapes [macrotopes], to the smallest structural level of hummock or hollow [nanotope], and the hydrological relationships that operate at each of these levels. Colour shading indicates hierarchy level.
Adapted from Lindsay *et al.* (1988).

24.2.1 Macrotope

This represents a single continuous expanse of peat soil. It may vary considerably in thickness but it is an essentially unbroken mantle. The boundaries of a macrotope are set at the edge of the peat mantle, where mineral soil outcrops, rivers, lakes or streams lying on mineral sub-soil represent a clear break in the mantle. Lindsay *et al.* (1988) have demonstrated this process for the Flow Country of Caithness and Sutherland, northern Scotland, while Lindsay and Freeman (2008) do the same for the northern part of the Isle of Lewis, Outer Hebrides (in fact both publications demonstrate the practical utility of the entire hierarchy). Evans and Warburton (2007), meanwhile, put the concept of the macrotope (and other levels of the hierarchy) to use in a geomorphological synthesis of blanket mire erosion.

Some macrotopes are truly vast (Couwenberg and Joosten, 1998), but in a blanket mire this is rarely the case. In its largest form, a blanket mire macrotope may extend for tens of kilometres along broad hill ridges. However, it is more usual to find that major stream-courses and outcrops of mineral soil restrict macrotopes to a maximum size of between 10 km and 15 km in diameter (Lindsay *et al*. 1988; Lindsay *et al*., 2003), though they may often be smaller than this (*e.g.* Lindsay and Freeman, 2008).

A macrotope may consist of several identifiable bog entities – *e.g.* Featherbed Moss, Alport Moor and Over Wood Moss – along with a series of discrete minerotrophic (fen) systems which link these named bog entities into a continuous peat mantle and which may also have their own names – *e.g.* Rough Syke, or Foxtor Mires (the real-life basis for the Great Grimpen Mire in Sherlock Holmes and the Hound of the Baskervilles) . In addition there are also likely to be numerous smaller 'flush' systems forming minerotrophic pockets throughout the peat mantle (see Figure 65).

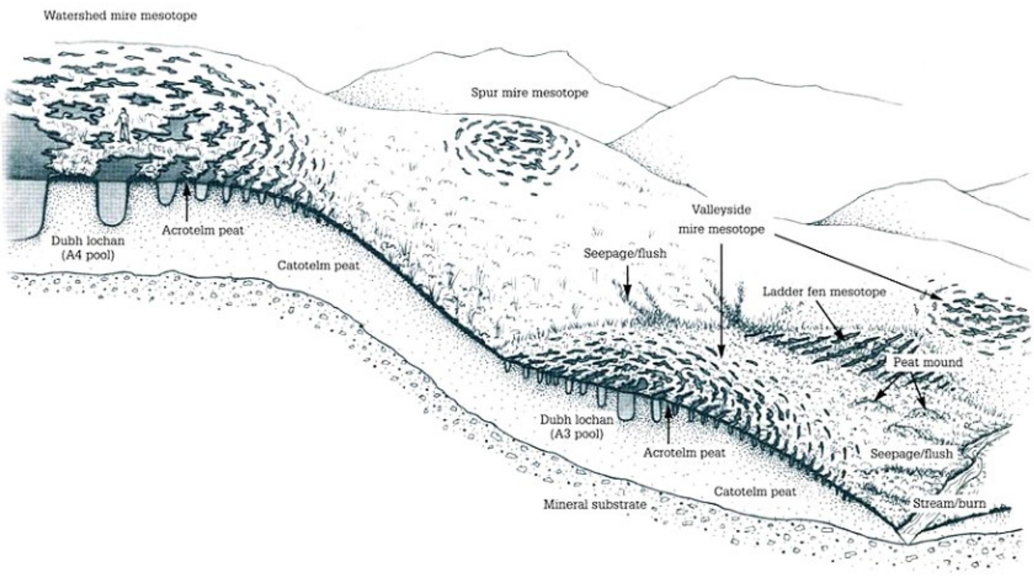


Figure 65. Schematic representation of blanket mire
Showing the relationship between peat depth and slope of the underlying mineral substrate, as well as the physical (and associated hydrological) inter-relationships between the various structural components of a blanket bog complex [or macrotope]. Terms are explained in Section 24.2. (Depth of peat exaggerated compared to horizontal scale).

Adapted from Lindsay *et al*. (1988)

The macrotope is the entity which can be conserved with some confidence in the sense that adjacent land-use change *beyond the boundary feature* will have at most only minor eco-hydrological effects on the conserved area. This is because the macrotope is a discrete and distinct hydrological entity, separated from adjacent peatland expanses by non-peat features. Agricultural drainage, or afforestation, of one macrotope is unlikely to have major hydrological implications for an adjacent macrotope – although it may have important implications for the movement and concentration of deer, or fell walkers. What happens to the boundary feature, however, may have profound implications for the macrotope, particularly if it is a lake or river. Draining of an adjacent lake, or damming of a boundary river, is likely to have a significant effect on the associated macrotope(s).

It is worth here highlighting the fact that although raised bogs in the lowlands of Britain may consist of a single raised lens of peat, in their natural state they nevertheless almost always form part of a macrotope. This is because in the natural state the bog margin – the 'lagg fen', as it is known – would normally consist of a fen-peat system which may be a relatively limited minerotrophic system, or may be of such an extent that it also embraces other raised bogs in the vicinity. Loss of this fenland component from the macrotope means that the remaining components (the associated raised bogs) are now likely to be hydrologically compromised even if they themselves have suffered little or no direct human disturbance. The model which explains why the raised bogs are likely to be thus compromised is called the 'ground water mound theory' and was developed by Hugh Ingram in the early 1980s (Ingram, 1982).

It is not necessary to go into the details of this model here, but the fact that almost all raised bogs in Britain have probably been compromised at the macrotope level by loss of surrounding lagg fen does have important implications for the carbon balance of these raised bogs. A simplified version of the problem is therefore perhaps in order.

Take a bowl and place a sponge in this bowl. Fill the bowl with water until the whole height of the sponge is saturated. [The part of the sponge standing proud of the water surface will be saturated because of upward capillary rise, whereas a bog is saturated by downward flow of precipitation, but no matter; for the purposes of this example, the end result is the same.] Now remove some water from the bowl so that the level of the water surrounding the sponge is lowered. This is equivalent to partial drainage of the surrounding lagg fen. It will be observed that the crown of the sponge is now drier than before. Remove the water in the bowl altogether (*i.e.* remove the lagg fen and turn it into intensively-drained agricultural fields). The sponge will now become very much drier. This is the current condition of almost all lowland raised bogs in Britain (indeed, in almost all of Western Europe), exacerbated by the fact that in most cases the bog has also been cut away at the margins, sometimes by a substantial amount.

Figure 66 contrasts the lagg fen of a relatively natural raised bog in Latvia with the margin of a typical raised bog in Britain.

In the British example the carbon balance of the macrotope system is likely to be significantly affected. This is partly because the fenland element of the macrotope has been lost and this fen peat is now oxidising because it is being drained, and partly because the raised bog element of the macrotope is hydrologically compromised. The bog is therefore likely to be shifting from a system of carbon gain towards a condition of carbon loss through long-term oxidative decomposition of the peat – a process recognised as long ago as the 17th Century (King, 1685; Weber, 1902; Bragg, 1995; Ginzler, 1997).

24.2.2 Mesotope

The mesotope is represented by the individual peatland unit which is often sufficiently distinct for it to be given a name – Featherbed Moss, Munsary Dubh Lochs, Offerance Moss (see Figure 66, above). In the case of an ombrotrophic mire unit, this entity consists of a mound of peat which is in some way domed, although the dome may be highly eccentric (*i.e.* substantially offset from the centre of the mire unit). It is this domed structure which ensures that the bog is indeed an ombrotrophic bog in contrast to the adjacent 'ladder fen' in Figure 65, which is clearly fed by ground/surface water.

In blanket mires particularly, the doming may sometimes reflect the shape of the underlying mineral ground. Thus watershed (Lindsay, 1995) or summit (Evans and Warburton, 2007) mires tend to be draped across the watershed ridge of the underlying landscape and therefore tend to dominate the highest points of this landscape whether or not the peat itself is 'domed' (though generally it is).

Ivanov (1981) emphasises that these individual mire units can be broadly characterised by their distinctive patterns of surface-water flow, which are themselves based on the overall surface morphology of the system. Ivanov (1981) demonstrates very clearly the way in which the mapping of surface 'flow lines' can be used in such characterisation (see Figure 67).

The Nature Conservancy Council (NCC) (1989) and Lindsay (1995) also use flow lines to characterise the major bog mesotope types found in British blanket mires (Figure 68). Charman (2002) and Evans and Warburton (2007) use the method to demonstrate how simple flow-line diagrams can be used to identify different topographic types of peatland systems. Lindsay and Freeman (2008) illustrate the use of the method for all mesotopes found within two macrotopes in the blanket mires of the Isle of Lewis, Outer Hebrides.

Figure 66. Comparison of lagg fen zones on raised mires in Latvia and Scotland.
(Top) View along the lagg fen of Teiçi raised bog nature reserve, Latvia. The raised bog is to the right, with stunted tree growth on the mire margin, while the mineral ground to the left is dominated by tall pine forest. Note the wide, flooded lagg fen separating the bog from the mineral ground. **(Bottom)** Offerance Moss raised bog, Forth Valley, Scotland. Note the canalised lagg fen, now effectively a drain, running between Offerance Moss, which is the brown and straw-coloured area in the centre of the picture, and the afforested raised bog just partially visible on the right of the picture. Note also the sharply-defined edge of the bog where parts have been lost to agricultural land-claim, fertilizer treatment and under-drainage. Several drains have also clearly been cut into the body of the bog itself.

Photos © Richard Lindsay/Lorne Gill

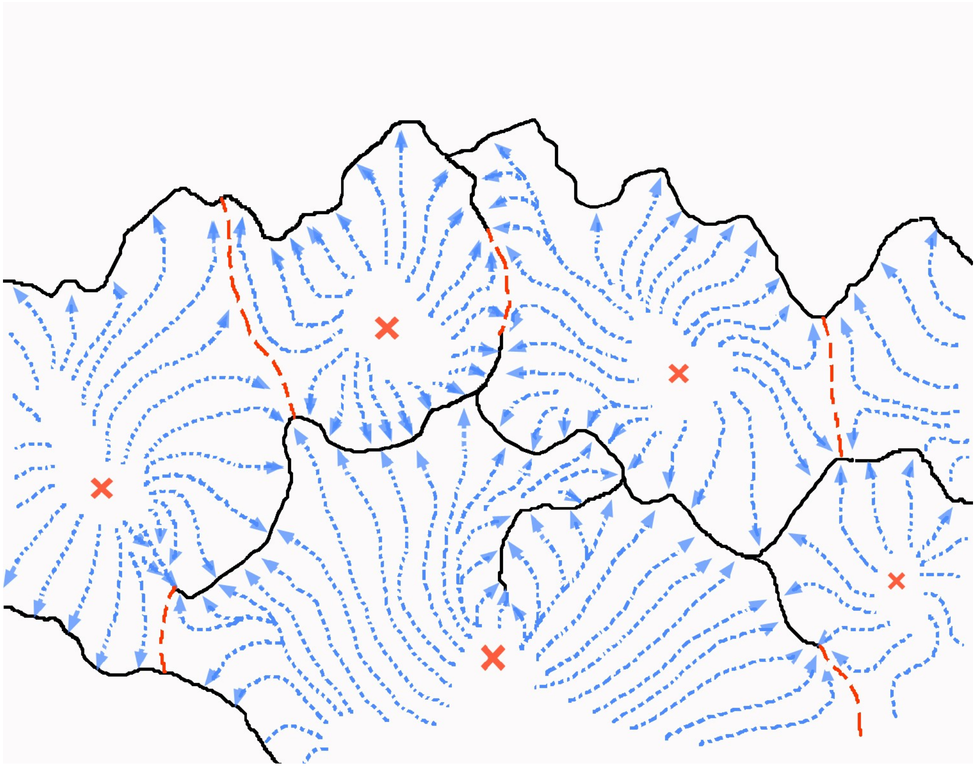


Figure 67. Mapping of peatland flow lines and mesotopes.
An example of a flowline map produced using the methods set out by Ivanov (1981) for a five-mesotope ombrotrophic mire complex. Flowlines are represented by the dashed lines with arrowheads, streams/rivers are shown as black lines, and mesotope boundaries not formed by streams are shown as red dashed lines. The watershed summit of each mesotope is indicated by a red cross. Note how the mesotopes can be readily distinguished, thanks to the pattern of flow lines. Different flow-line patterns are associated with different mesotope types (see Figure 68).

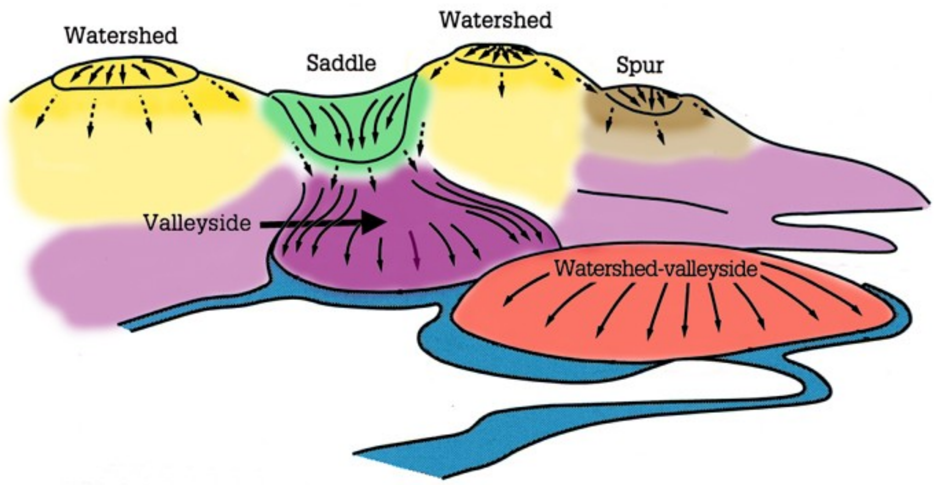
Based on Ivanov (1981).

One important additional distinction which can be made at the mesotope scale is the sub-division of the mire unit, or mesotope, into zones affected by marginal influences and those which are largely free from such influence.

Sjörs (1948) referred to these two distinct areas of ground, logically enough, as 'mire margin, and 'mire expanse'. Weber (1902) had earlier called these zones 'marginal slope' and 'the plateau'. The distinction between these two zones is far more evident in lowland raised bogs than in blanket mires, so we will not dwell further on the issue here, other than to observe that the carbon balance of the two zones is likely to be quite different, which must be borne in mind when selecting sites for investigation.

An apparently trivial, though associated, point about these two zones is the very real difficulty of judging distances when walking across a mire because there are few objects which can give a sense of scale. This, combined with the subtle changes associated with the mire-margin, can mean that it is all too easy for a research plot to be selected in the field without realising that the ground is still subject to significant marginal influence. It is much better to select study sites using aerial photographs and then create these as waypoints in a GPS to locate the sites in the sure knowledge that the sample site is located well within the mire expanse.

Such a technique was of course not available prior to the development of GPS. Consequently any papers published prior to about 1990 will have relied on traditional navigation techniques. This is not necessarily less accurate than using GPS, particularly if detailed aerial photographs were used as navigation aids. Nevertheless the possibility that mire margins may feature unduly in the locations of study sites must be recognised – particularly as the mire margin is also pragmatically convenient in terms of accessibility if large, heavy equipment must be transported to the study location.



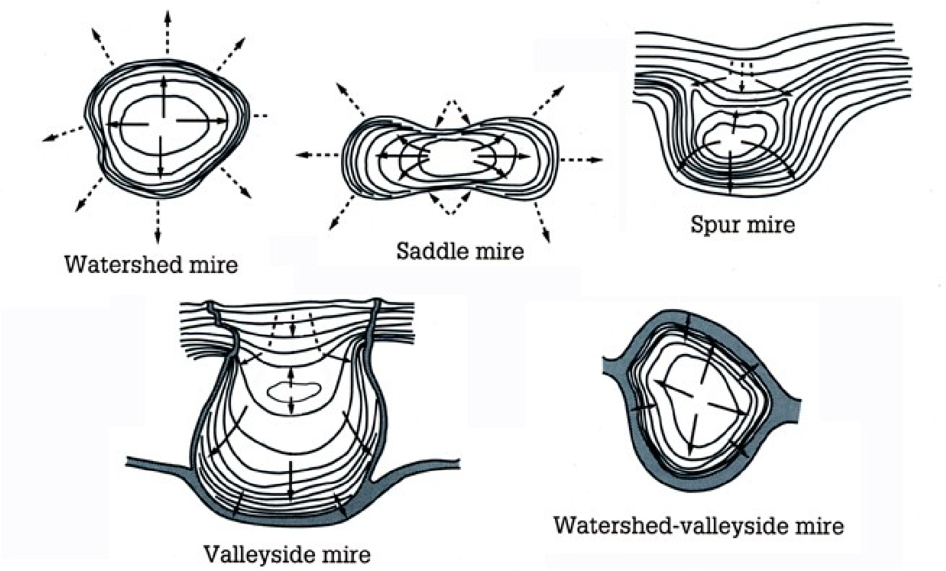


Figure 68. Characterisation of UK blanket bog mesotopes using flow lines.
(Top) Depiction of a schematic blanket mire landscape illustrating the locations taken up within that landscape by particular blanket bog mesotopes. Arrows demonstrate direction of surface-water flow. **(Bottom)** Blanket bog mesotopes viewed from above, displaying surface contours and the general direction of surface flow lines.

Adapted from Lindsay (1995)

24.2.3 Microtopes

The most well-known feature of the peat bog environment may be its hummocks and hollows, but without doubt the most distinctive feature of a peat bog – indeed the most distinctive feature for a very large proportion of peatlands – is the presence of patterns created by these, or several other, characteristic surface structures. It is by such patterns that peatlands can be identified from aircraft, aerial photographs, or even from space.

Almost certainly, if peatlands were to exist on another planet, they would betray their presence by displaying similar surface patterns. These patterns are known as microtopes. Ivanov (1981) describes microtopes as:

> *"...the elementary geographical units out of which mires are built."*
>
> *Ivanov (1981) p.82*

Ivanov (1981) also observes that:

> *"Any mire massif [peatland unit] which forms a meso- or microtope is a systematically constructed whole consisting of a number of microtopes ... which differ from each other in the composition and structure of their plant cover, and, correspondingly, in the physical properties of their acrotelms. Calculations of their hydrometeorological regimes cannot be based upon the results of research into the physical properties of their plant cover and peat deposits unless experimental descriptions of their properties are linked to definite types of microtopes."*
>
> *Ivanov (1981), p.82*

Microtope patterns are in effect created by two structural elements - the 'strip-ridge' structures described by Ivanov (1981), and the means by which peatlands achieve stability in the face of constant environmental change. It is generally assumed that the two distinct elements consist of 'dry' hummocks/ridges, and 'wet' hollows/pools. As Couwenberg and Joosten (2005) demonstrate, however, the critical condition of the model is merely that one element is *more* conductive than the other. Thus there may be no 'wet' elements such as hollows or pools at all; the pattern may consist only of 'dry' elements and 'less-dry' elements.

Failure to understand this essential part of peatland structure has led many to conclude that pattern is only found where there are hollows or pools. As these are generally only found across the mire expanse, microtope patterns are often assumed to be restricted to areas of the mire expanse. This is not the case. Every part of a peatland is characterised by a microtope pattern. Sometimes this pattern is very simple, sometimes it is highly complex. The essential characteristic, however, is that microtopes result from differential growth of peat-forming species which thereby giving rise to surface structures differing in their ability to transmit water.

24.2.3.1 Microtopes and climate: the pattern of patterns

Figure 69 demonstrates something of the variety of microtope pattern types which may be encountered in Britain. Three key points are associated with these examples. Firstly, it is clear that some patterns are more evident than others, at least from ground level. Secondly, it is often the case that patterns are very difficult to discern or at least characterise from ground level. Even where pattern is evident, it is very rare that the *shape* of the microtope is evident from ground level. When viewed from above (using an aerial photograph, or simply viewing from a high vantage point) their nature become very much clearer (e.g. Figure 69 bottom left and bottom right).

One thing evident from Figure 69 is that not all examples of pattern given here contain obvious examples of 'wet' and dry' structures (e.g. Figure 69 : top left). However, all examples do at least contain structures which can be described as 'dry' and 'less dry'. Another associated feature is that the amount of water (or 'wet structure') varies from example to example.

The fundamental idea that a wet climate gives rise to expanded hollows and pools, while dry climates favour ridges and hummocks, has already been discussed in relation to the debate over the 'hummock-hollow regeneration' model and Barber's (1981) subsequent 'climate phase change' model. The consequences of the 'climate phase change' model are immediately apparent even in today's climate because of course the north-west of Scotland is wetter than the north-west of England. Consequently Barber's (1981) model would predict that bogs in the far north-west of Scotland should have larger hollows and pools ('wet phase' structures) than those in the southern Lake District of England.

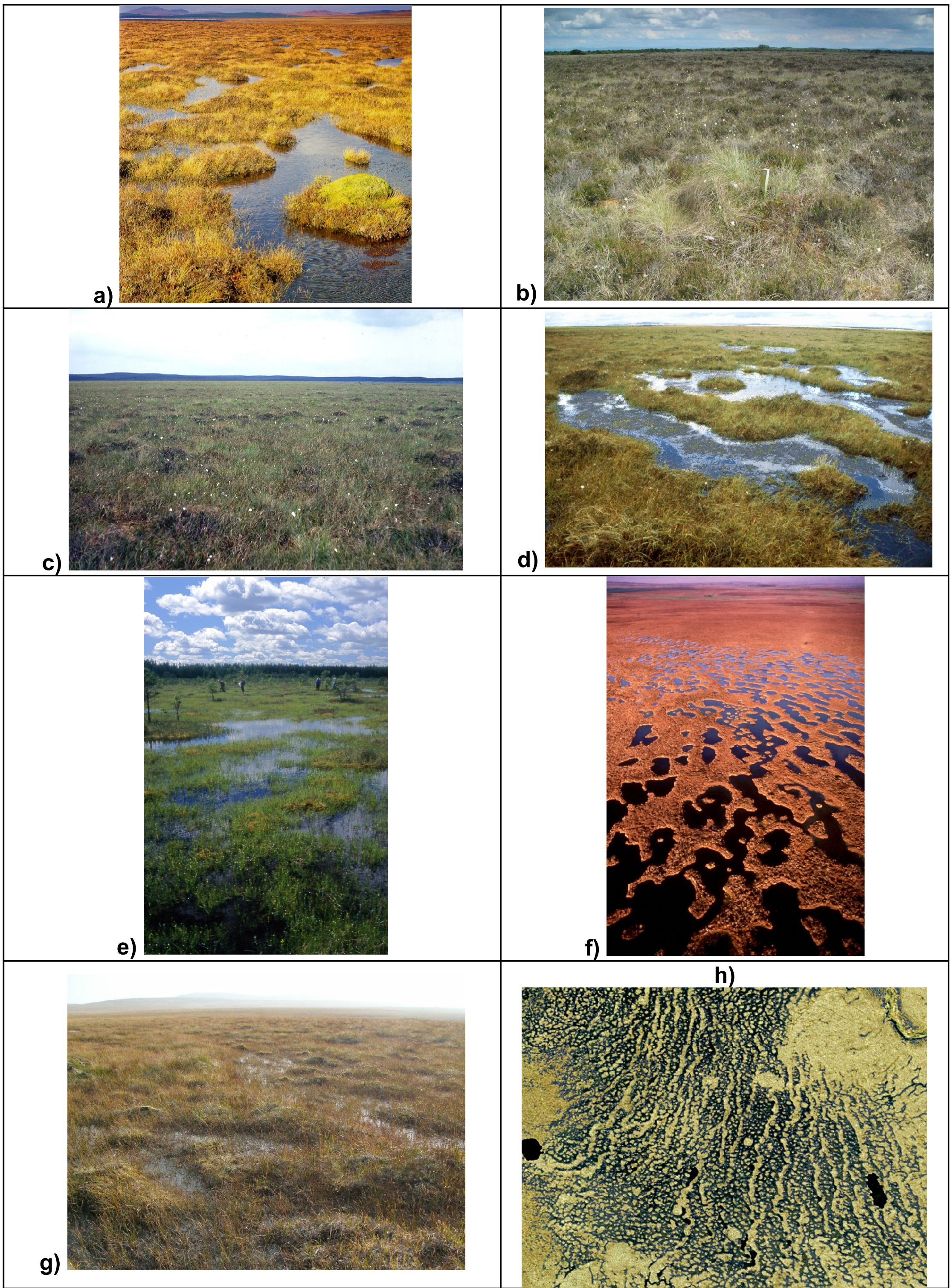


Figure 69. A selection of microtope patterns found on raised and blanket mires.
This collection of images illustrates some of the microtope patterns which may be found on raised and blanket mires. The images at **(b)** and **(c)** demonstrate that microtope patterns do not necessarily involve open water or even evident hollows. The image at **(e)** is from a raised bog in Latvia, all others are from Britain. Image **(g)** illustrates the ground view of a ladder fen within blanket mire on the Isle of Lewis; Image **(h)** shows the same area when viewed from an altitude of 500 m, at which point the microtope pattern becomes very clear.

In a wide-ranging survey of Britain's boglands, this is precisely what Lindsay, Riggall and Burd (1985) and Lindsay *et al*. (1988) were able to demonstrate – the *maximum* scale of hollow/pool formation increased in proportion with the recorded number of 'wet days' (a day with at least 1 mm recorded precipitation). This relationship can be seen in Figure 70.

Figure 70 also brings us to the second important characteristic of the microtope pattern, which is the way in which these patterns are spatially arranged on a site. Aerial assessment of surface patterns on bogs across Britain soon reveals that there is a pattern to the patterns.

Typically the most evident and extreme patterns occur on the deepest peat, while microtopes associated with shallow peat (*i.e.* generally towards the edges of individual mire units/mesotopes) are much simpler and more subdued, at least in terms of what is visible from an aerial photograph.

It is these more subdued areas which are generally regarded as having no pattern, but closer examination of such areas reveals that this is not the case. Central areas of the bog may display the maximal degree of patterning, but all other parts of the bog do possess a microtope pattern, each microtope being characteristic of the conditions prevailing on that part of the site.

The scale of 'wet' and 'dry structures on a peatland is not determined solely by climate. There are two factors at work; the first is climate, the second is slope.

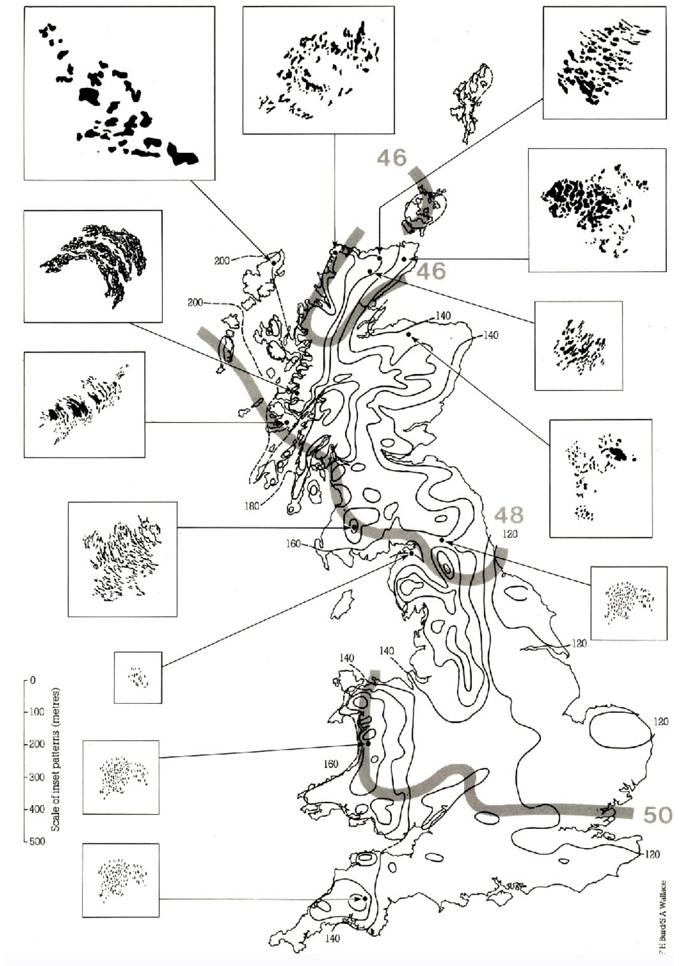


Figure 70. Relationship between maximal microtope pattern and climate.
The small inset boxes show the maximum degree of microtope patterning identified for bogs in each given part of Britain. In the inset boxes, black represents hollows or pools. Also shown as narrow black lines are the boundaries between ranges of 'wet days' (1mm precipitation in 24 hours) across Britain, while the grey-shaded lines indicate average mean daily temperatures (in °Farenheit).

Taken from Lindsay *et al*. (1988)

24.2.3.2 Microtopes and slope

Indeed the reason for stating so explicitly in relation to Figure 70 that wet days relate to the *maximum* scale of hollows/pool formation is that gradient also plays an important part in determining the scale of these structures on any given peatland site (even many fen peatlands). As a corollary to Barber's (1981) and Ivanov's (1981) picture of a 'strip-ridge' mechanism in bogs for adapting to changing climate, Goode (1970, 1973) demonstrated that this same strip-ridge ('wet' element-'dry' element) provides a mechanism for maintaining hydrological stability across differing angles of slope in a peatland.

Where the gradient is very gentle, a level water surface can be maintained across a wide area with only narrow ridges of peat being required to separate each large pool from the next pool downslope. Where the gradient of the overall slope is steeper, the ground is only able to fall within the ‘dry’ elements of the microtope pattern - *i.e.* within the solid ridges separating each pool (because obviously the water level in pools cannot slope). Furthermore, the overall slope does not allow room for wide, flat pools to form. Water can extend along a contour line but it cannot extend back upslope. Consequently the ‘wet’ elements of the microtope on steeper slopes tend to become elongated along the contours but are much narrower in the direction of slope.

Where the gradient becomes sufficiently steep, there is no longer any room for the true ‘wet’ element. The pattern instead becomes a strip-ridge structure which alternates between ‘dry’ elements and ‘less-dry’ elements. If the slope becomes too steep and/or the peat becomes too thin, the microtope may even be reduced to a single element – generally that of the hummock level. It is important to understand, however, that this zone of ‘dry’ and ‘less-dry’ elements, or even a zone of a single element, is still a microtope – in this case characterised by a very simple pattern structure.

The consequence of this response to slope can be seen in Figure 71, which shows the general pattern taken up by ‘wet’ and ‘dry’ elements, as predicted by Goode (1970, 1973). This prediction can be compared with Figure 72, which is an actual area of blanket mire in northern Scotland. At least three microtopes are visible here, one of which contains no evident ‘wet’ element.

The nature of the ground in a peatland – essentially the nature of the microtopes –has important implications for the overall carbon balance of a site, as we shall see, particularly as these patterns are long-lived, demonstrably persisting for thousands of years in some cases. On a smaller scale, the results obtained from specific research locations within a peatland will be influenced by the type of microtope found in that location, as well as (potentially) the proportion and nature of other microtopes in the general vicinity. Consequently it is extremely important that the pattern of microtopes be mapped and characterised around any carbon-balance research location. This is rarely done. Or at least it is rarely done adequately, in part because of the third major issue associated with microtopes; currently no language or vocabulary exists for describing microtopes except at the very crudest level.

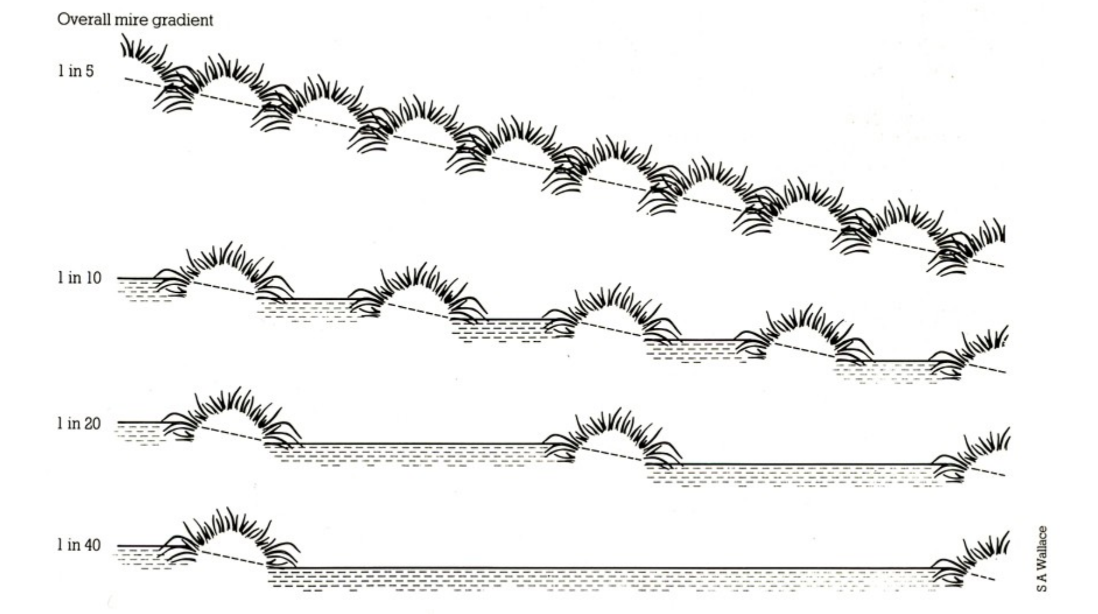


Figure 71. Relationship between microtope pattern and gradient.
Generalised illustration of the relationship between microtope pattern and slope. Where the gradient is steep, the wetter element of the pattern tends to be reduced in width. Where the slope is steepest, there may be no aquatic element at all, merely ‘dry’ and ‘less dry’ elements. Where the slope is very shallow, the wetter element may become the dominant feature.

Taken from Lindsay *et al*. (1988)

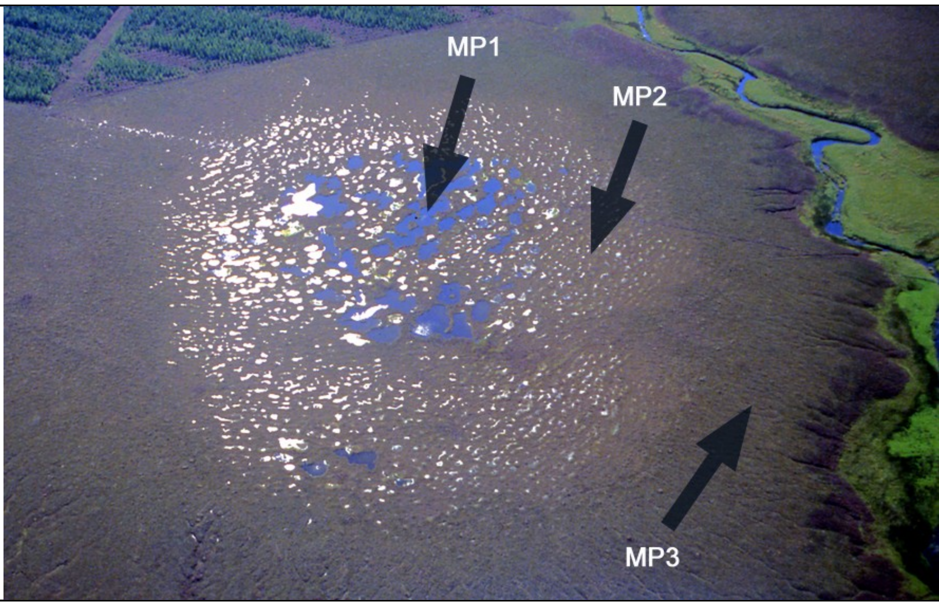


Figure 72. Microtope patterns on a watershed/valleyside mire.
On the crown of the dome of this watershed/valleyside mesotope, the gradient is very shallow indeed. Here the pools are large and wide (MP1). Where the slope increases somewhat beyond the crown of the bog, the pools and hollows become narrower, but stretched along the contours (MP2). .Towards the margin of the bog where the slope is greatest, the gradient is so steep that there are no hollows or pools; the microtope consists entirely of hummocks and ridges (MP3).

Photo © Richard Lindsay

24.2.3.3 Microtope characterisation

Most languages are poorly adapted to describing pattern beyond very simple geometric shapes. Characterising and classifying any complex pattern is almost as difficult as attempting to explain the character and classification of colour to someone who is blind. While a considerable quantity of literature has been devoted to the description of the structural elements which together create microtope patterns on peatland systems, remarkably little has been published about the types of pattern thereby created. The literature is also not lacking in models designed to demonstrate the way in which peatland patterns develop, yet descriptions and classifications of the precise pattern types which emerge from these models are notable mainly for their absence.

Imagine trying to express in words, or even in categories, the patterns evident in Figure 69 and Figure 72. The systems have not yet been invented to categorise and describe these shapes in any agreed, coherent, and internally consistent manner. This absence of a vocabulary represents a significant problem for peatland science in general, because one of the key features of a peatland – the one most readily identifiable even from space – cannot actually be described other than in the crudest terms. Thus microtope patterns have been described as ‘rounded pools’, ‘linear pools’ ‘arcuate pools/ridges’, ‘anastomosing’ (all interconnected), and little else. Some descriptions go no further than noting that ‘a pool system was present’, or ‘the surface was dominated by hummocks and hollows’, but give little other information. Further investigation sometimes reveals that even these descriptions are not entirely to be relied upon because ‘pool systems’ turn out to be hollows rather than pools, while ‘hummock-hollow’ system are often found to contain no hollows as such, merely undulating surfaces made up from ‘dry’ elements and ‘less-dry’ elements.

This absence of a vocabulary is profoundly unhelpful. It means that many researchers avoid descriptions of the microtope at all, while others, confused by the ambiguity of existing literature, present descriptions which merely add to the confusion rather than enlighten the reader. As far back as Weber (1902) it has been possible to indicate the boundaries of different microtopes and even to map them (Ivanov, 1981; Göransson *et al.*, 1983; Lindsay and Freeman, 2008), because our visual ability is quite sensitive to differences in pattern. Unfortunately our words cannot keep pace with our eyes. Our language has not yet evolved sufficiently to provide adequate descriptions of different pattern systems.

In the hydro-morphological hierarchy of peatland function, the microtope is the key level associated with the maintenance of stability of a peatland in the face of climate change (and in response to changing hydrological gradients as induced by, for example, drainage). As such, it can be assumed that the microtope is more intimately linked to the carbon balance of a peatland system than any other level in the hierarchy, yet researchers have not merely failed to describe it in their studies, they have been *unable* to describe it.

This major lacuna in the tools of peatland research is unlikely to be filled in the immediate future. Pattern recognition, artificial intelligence and fractal geometry all offer potential for the development of classification and descriptive systems, but any system so devised would need to be fed with, and tested against, an enormous range of example patterns from peatland systems before any agreed system could be adopted. Despite this, the work should clearly be undertaken as an urgent priority. For the immediately foreseeable future, however, it must be recognised that any published or ongoing work investigating the peat-carbon balance does so within the context of this very significant level of uncertainty.

Finally, an inconsequential point on the face of it, but a point which is nonetheless worth making. It is surprisingly easy to walk past a region of extraordinarily striking microtope pattern without seeing it. Even from distances of only 20 or 30 m, often little can be seen of even quite large and dramatic microtope patterns. This is partly because eye-height gives such a shallow angle of view compared to the relatively level expanse of the peat landscape, and partly because the vertical range of microtope patterns themselves is so small.

It is therefore absolutely essential that any assessment of microtope pattern is made with the aid of aerial photographs. It is virtually impossible to make an assessment without such assistance. To illustrate this, Figure 73 compares a typical example of a ground view contrasted with an aerial photo view of the same location.

Figure 73. Microtopes and aerial viewpoints
The image on the left illustrates what it looks like to walk within 30 m or so of the pool system so clearly visible in the oblique aerial photograph of the same area displayed on the right. From ground level, even at a distance of only 30 m, the pool system is barely visible. At a distance of 50 m or so, it would have been invisible to a ground-based observer. Certainly there is no indication of just how large the pool system is, nor of its overall pattern, unless the pattern is viewed as a whole using an aerial view.

Walking only 20 m further away from the just-visible bog pools in Figure 73 (left) would have meant that the dramatic pool system shown in Figure 73 (right) would have been completely missed. Indeed it might be said that an aerial photo assessment should be mandatory for any blanket mire research site. Today, with the availability of information sources such as Google Earth, this is not generally a problem, but it is worth bearing in mind that earlier research did not have this luxury. Aerial photography at the appropriate scale and cost was not always available for a site. This, too, must be borne in mind when interpreting previously-published research, particularly early work, where site descriptions may not always be what they seem.

24.2.4 Nanotopes

In order to maintain the consistency of terminology so far employed in the peatland hydro-morphological hierarchy, the term nanotope has been proposed by Couwenberg and Joosten (1998) for the range of small-scale structures from which surface patterns (microtopes) are assembled. Ivanov (1981) originally termed these individual features 'microforms', so either term may be encountered in the literature. More usually, however, neither term is used and descriptions fall back on the rather unfortunate legacy of the 'hummock-hollow regeneration cycle'.

The Osvald/Tansley 'hummock-hollow regeneration cycle', though now regarded as having only limited basis in fact, has tended to dominate peatland thinking over the second half of the 20th Century. This has led to an entirely unfounded idea, at least in the mind of the non-specialist, that the individual small-scale structures which collectively create the bog surface consist largely of 'hummocks' and 'hollows'. Even the original Osvald/Tansley model identified more than two structural elements, but this has tended to be lost in the dominance of the names in the model title.

A simple, two-element system does in fact work fairly well as the crude basis for the Ivanov-Barber (1981) strip-ridge model of how bogs respond to climate change, and for Goode's (1970, 1973) model of how increasing hydraulic gradients influence pattern. However, the nature of the two elements used in these models, and indeed the nature of a great many more elements, extends well beyond the limited concept of 'hummock' and 'hollow'.

Around the same time as the 'hummock-hollow cycle' was being advanced, Sjörs (1948) provided fairly detailed descriptions of several small-scale structures he had identified and which together created the patterns on the mire system he was studying. His work is generally credited with having brought together a number of terms for the microtopography of a bog into some degree of consistent format, at least in terms of the ecological, hydrological and structural character of such features, if not always in the names he gave to these features. His collection of terms included:

- hummock, which accords with most ideas of what a hummock should be;
- lawn, which describes a terrestrial zone lying only a small distance above the water table;
- carpet, which is a zone dominated by aquatic Sphagna, and most closely equates to the Osvald/Tansley idea of a 'hollow';
- mud-bottom hollow, which is an odd term, given that there is no mud in a peatland, only peat, but it refers to a hollow which has no *Sphagnum* carpet, just a submerged peat surface.

Lindsay, Riggall and Burd (1985) reviewed these and a great variety of other terms employed in a range of literature to describe the microtopography of bog systems and found, essentially, that one man's 'lawn' is another man's 'carpet' – in other words, many established terms were used in altogether different senses by different authors. Lindsay, Riggall and Burd (1985) also recognised a somewhat wider range of microtopography than described by Sjörs (1948) because they were concerned with the more complex peatland structures of the blanket mire environment, but these additional structures also had a range of conflicting terminology attached to them.

In an effort to bring some sort of order and consistency to the terminology of peatland microtopography, Lindsay, Riggall and Burd (1985) reviewed the various terms used in the peatland literature and proposed assigning simple codes to the many structures which were so widely recognised but so rarely labelled consistently. Authors could continue to use any name they wish, but the name should be prefaced by the appropriate code. This coding system, and the identification of structures, was further developed, with explanatory diagrams, in Lindsay *et al*. (1988), for the Nature Conservancy Council (1989), for the JNCC (1994), and was illustrated with photographs in Lindsay (1995) [his Plates 5 to 8]. Lindsay *et al*. (2003) then added one more important structure to the range so far identified, to produce the final spectrum (so far) of small-scale surface structures characteristic of bog systems in the temperate zones of both hemispheres.

24.2.4.1 Summary of nanotope features

The coding system proposed by Lindsay, Riggall and Burd (1985) firstly separates nanotopes into either terrestrial (T) or aquatic (A) features. They are then numbered sequentially in their vertical distance away from the average water table. Thus T1 and A1 are the terrestrial and aquatic features respectively which lie closest to the position of the average water table. A diagrammatic version of this system is given in Figure 74, along with some examples of the way in which particular plant-species groups distribute themselves within these zones.

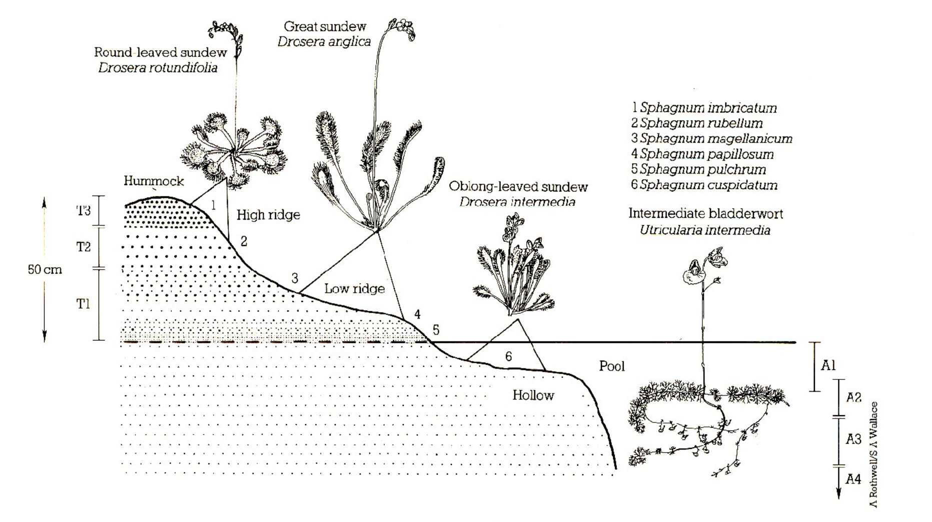


Figure 74. Generalised zonation of nanotope features found on bog systems.
The location and range of nanotope zones is here displayed in relation to the position of the average bog water table, together with the codes suggested by Lindsay, Riggall and Burd (1985) for each zone. Also indicated are the typical associations of *Sphagnum* species, and of Britain's native carnivorous bog plants (largely sundews), with these various zones.
Taken from Lindsay *et al.* (1988)

24.2.4.2 T1 low ridge

Described by Sjörs (1948) as 'lawn', this nanotope:

- occupies a vertical range of 0 cm to 15 cm (approximately) above the average water table;
- is generally the richest zone for characteristic mire species, and generally has more cross-leaved heath (*Erica tetralix*) than heather (*Calluna vulgaris*);
- tends to support the great sundew (*Drosera anglica*) rather than the round-leaved sundew (*D. rotundifolia*), particularly at the lower edge of this zone, but both can occur within the nanotope;
- within narrow limits displays a highly fluctuating water table (because it receives water from all higher nanotopes and loses it to aquatic nanotopes;
- is generally the 'softest' of the T-zones (i.e. least dense);
- generally has the highest proportion of *Sphagnum*;
- is common on mires which have suffered little human impact.

Figure 75. T1 low ridge
Typical *Sphagnum*-rich species assemblage.
Photo © David Goode

24.2.4.3 T2 high ridge

This nanotope:

- occupies the vertical zone of 15 cm to 25 cm (approximately) above the average water table;
- is characterised by bog species and some species from heathland – notably heather (*Calluna vulgaris*), which here can become co-dominant with cross-leaved heath (*Erica tetralix*);
- tends to support round-leaved sundew (*Drosera rotundifolia*) rather than the great sundew (*D. anglica*), but both can occur;
- is the predominant nanotope in microtopes of the mire margin, for blanket bog where there is a distinct slope, and for bogs which have suffered some degree of human impact;
- *Sphagnum* cover is variable, and thus the surface is often quite firm.

Figure 76. T2 high ridge
An example of rather dry T2 with a dense dwarf-shrub sward comprising a 50:50 mix of cross-leaved heath and heather. The *Sphagnum* cover beneath may be discontinuous.
Photo R A Lindsay

24.2.4.4 T3 hummock

This is the nanotope on which there is most agreement. T3 hummocks:

Figure 77. T3 hummock
Hummock-forming *Sphagnum fuscum*, with heather and round-leaved sundew.
Photo R A Lindsay

- occupy the vertical zone of 25 cm to 1 m (approximately) above the average water table;
- are characterised by some bog species and several heathland species – notably heather (*Calluna vulgaris*), which here can become the dominant dwarf shrub (albeit, in a very stunted form);
- other dwarf shrubs such as crowberry (*Empetrum nigrum*) and bearberry (*Arctostaphylos uva-ursi*) also tend to grow at the T3 hummock level;
- often supports the round-leaved sundew (*Drosera rotundifolia*) but rarely other sundew species;
- are scattered or almost absent on very wet bogs, but occur more frequently within microtopes of the mire margin, while on bogs which are subject to water-table draw-down (from drainage, for example), extremely high and very dense, tight hummocks may form;
- are invariably formed by moss growth;
- are generally formed by *Sphagnum* species but woolly-hair moss (*Racomitrium lanuginosum*) can also form T3 hummocks;
- if made from woolly-hair moss (*Racomitrium lanuginosum*) are more abundant on bogs which have been damaged, although on some sites in the wettest parts of Britain such hummocks appear to be natural;
- are not formed by other species of moss, though these mosses may be found covering a hummock (this generally indicates a degenerating hummock);

- if formed by woolly-hair moss, can sometimes reveal by sampling the heart of the hummock, that the hummock was once a *Sphagnum* hummock.

24.2.4.5 T4 erosion hagg

Peatland erosion is a much-debated topic. There is no doubt that it is a widespread phenomenon. Where it occurs the ground tends to be dominated by drier nanotopes. While T2 high ridge and T3 hummocks are common in erosion complexes, they are joined by, and sometimes dominated by, hagg tops which represent vestiges of the original bog surface left highest and driest by encircling erosion gullies. T4 erosion haggs:

- occupy the vertical zone of 0.75 m to 1 m+ (approximately) above the average water table;
- are often characterised by woolly-hair moss (*Racomitrium lanuginosum*) and several heathland species, but generally also still support one or two bog species;
- occasionally supports round-leaved sundew (*Drosera rotundifolia*) but no other sundew species;
- dwarf shrub species are common, as are lichens, particularly of the *Cladonia* group.

Figure 78. T4 erosion haggs
The large dry mounds with pale woolly-hair moss are surrounded by erosion gullies, and have a vegetation showing affinities to heathland.
Photo R A Lindsay

24.2.4.6 T5 peat mounds

This is a nanotope which is restricted to the far north and west of Scotland. They are large mounds of peat anything up to 20 m in diameter and rising as much as 3 m above the general bog surface. The origins of these features remains unclear.

24.2.4.7 A1 *Sphagnum* hollows

These may or may not refer to the 'hollows' which are presented by Tansley (1939), Godwin and Conway (1939), and Osvald (1949) as central to the 'hummock-hollow regeneration' model. What is certain is that Sjörs described A1 *Sphagnum* hollows as 'carpets'. This nanotope:

- occupies the vertical zone of 0 cm to $^{-}$10 cm (approximately) below the average water table;
- is often rather species poor, being dominated by a more-or-less continuous carpet of the aquatic *Sphagnum cuspidatum*;
- may support individual open stands of white-beaked sedge (*Rhynchospora alba*), common spike-rush (*Eleocharis multicaulis*), common cotton grass (*Eriophorum angustifolium*), bog bean (*Menyanthes trifoliata*), or great sundew (*Drosera anglica*);
- often gives the appearance of a solid green carpet because no actual water is visible;
- cannot support the weight of an adult stepping onto the *Sphagnum* carpet;

Figure 79. A1 hollows
The yellow-green expanse of *Sphagnum cuspidatum* looks firm but cannot be walked on.
Photo R A Lindsay

- is more commonly found in the bogs of England, Wales and eastern Scotland.

24.2.4.8 A2 mud-bottom hollows

A misnomer, as discussed above, but such is the respect accorded to the late Hugo Sjörs that the name has stuck. This nanotope:

Figure 80. A2 mud-bottom hollow
Long-leaved sundew, white-beaked sedge and spike rush all growing in the dark, decomposing mass of plant litter forming the base of a mud-bottom hollow.
Photo R A Lindsay

- occurs as shallow pools no more than 20 cm deep;
- has a base which is sufficiently solid to stand on, and consists either of bare peat (the 'mud-bottom') or of a dense mass of semi-decomposed purple moor grass leaves (*Molinia caerulea*) - in the latter case, the hollow may be deeper than 20 cm, but the mat settles firmly at around 20 cm depth;
- has no dense, continuous mat of *Sphagnum*, merely scattered pockets of either *S. subsecundum* or *S. cuspidatum*;
- is most characteristically associated with short, dense, cherry-red mats of oblong-leaved sundew (*Drosera intermedia*), but other species include common spike-rush (*Eleocharis multicaulis*), white-beaked sedge (*Rhynchospora alba*), brown-beaked sedge (*R. fusca*), great sundew (*Drosera anglica*), bog bean (*Menyanthes trifoliata*) and common cotton grass (*Eriophorum angustifolium*);
- tends to occur in the far west of Britain, but particularly towards the north-west of Scotland.

24.2.4.9 A3 drought-sensitive pools

These may also have formed part of the original 'hummock-hollow' model, but the descriptions given by Godwin and Conway (1939) are somewhat ambiguous on this point. This nanotope:

- occupies the vertical zone of $^{-}20$ cm to $^{-}40$ cm (approximately) below the average water table;
- has an evident expanse of open water, though there may also be loose floating mats of *Sphagnum cuspidatum* or *S. subsecundum*;
- generally only supports stands of bog bean (*Menyanthes trifoliata*) or common cotton grass (*Eriophorum angustifolium*);
- is now completely restricted to Scotland, though it may once have occurred on the raised bogs of Wales and north-west England.

Figure 81. A3 drought-sensitive pools
The pools here can be seen to support loose mats of *Sphagnum* and open water.
Photo R A Lindsay

24.2.4.10 A4 permanent pools

This is probably the only part of the nanotope range which faces real challenges in responding to climate change in the manner described by the Barber (1981) 'phase change' model. These pools are permanent structures which can expand as the peat accumulates and surface gradients close to the crown of the bog are reduced, as shown by Goode (1970), but contraction is a more difficult process, as Ivanov (1981) describes.

This nanotope:

- is represented by pools which may be 3 to 4 m deep, extending almost to the mineral sub-base, with solid, vertical walls to the pool;
- consists largely of an open water body in which some detritus and a range of aquatic invertebrates can be found;
- may have floating columns of *Sphagnum cuspidatum* or bog bean (*Menyanthes trifoliata*), but little else in the way of vegetation;
- is found only in Scotland, in the far west and north.

Figure 82. A4 permanent pools
These are deep lake-like structures but still formed by the peat body. The sides of the narrow ridge in this photo descend vertically for 3 m into the pools, although the ridge is only 50 cm wide.
Photo R A Lindsay

24.2.4.11 Erosional nanotope forms

Originally Bower (1960), then most recently Evans and Warburton (2007) have (along with many other authors in the intervening years) described and categorised the geomorphological characteristics of peatland erosion. Evans and Warburton (2007) present a particularly helpful hierarchical approach which looks at erosional forms "from landscape to micro-relief". Their 'microscale' observations and tentative classification based on small-scale erosional mechanisms arguably go to a finer level of detail than the nanotope, while their 'mesoscale' is clearly a description of erosional microtope patterns, as is the classification proposed by Bower (1960).

There is no specific classification proposed in either Bower (1960) or Evans and Warburton (2007) for the shapes of individual gullies, merely their assembled pattern. Lindsay *et al*. (1988) likewise did not attempt a classification of gully shapes, other than to separate out three distinct types of erosion gully, two of which feature in Evans and Warburton (2007). These three types are described below, together with a final associated nanotope structure which is almost entirely overlooked in the literature but which may ultimately prove to be one of the most widespread of all nanotope types.

Considerable debate still surrounds the question of whether erosion is a natural process or is a result of human action, but a conference convened a decade ago specifically to consider blanket mire erosion in Britain concluded that erosional forms were important, whatever their origin (Tallis, Meade and Hulme, 1997). Given the enormous extent of eroded blanket mire in Britain and Ireland, it is thus logical that the nanotope forms associated with erosion should be provided for within the overall hydro-morphological hierarchy.

24.2.4.12 E1 regenerating erosion gullies

This nanotope is used to represent *any* erosion gully, or section of gully, which is now showing evident signs of re-vegetation and potentially renewed peat formation. Typically the gully walls may still be eroding but the gully floor now supports a vegetation which is normally peat-forming. The opposite of this is also possible, whereby gully sides are sufficiently re-vegetated almost to be smothering the gully floor, but still the gully floor is eroding, although usually this only applies to narrow gullies.

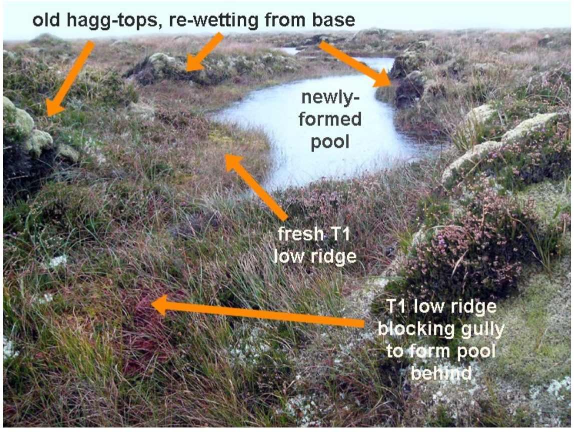


Figure 83. E1 regenerating erosion gullies
This is a well-developed example of regeneration within a former erosion gully in the blanket bog of Lewis, Outer Hebrides. The original gully has become blocked by the *Sphagnum* growing in the left foreground, resulting in a pool of water being ponded back within the gully. The pool is now terrestrialising with *Sphagnum*, and the surrounding dry ridges and hagg tops are becoming more paludified with the rise in local water table. More steeply-sloping examples may not produce pools, but the underlying effects on the surrounding bog are the same.

Photo R A Lindsay

Lindsay *et al*. (1988) found re-vegetating erosion gullies to be widespread in the Flow Country but somewhat restricted to higher altitudes, while Evans and Warburton (2007) emphasise that half of all erosion patterns surveyed in upland England were found to show a significant degree of re-vegetation.

Crowe (2007) provides a valuable picture of the rates and scales of re-vegetation, and of potential peat formation, within the E1 erosion gullies of the south Pennines. From this, it is clear that the gullies are not going to infill completely and therefore disappear even within decadal timescales, but many of the functions and services typical of active, natural bog are probably already returning thanks to fresh bog growth within the E1 gullies.

This E1 nanotope is also important because it is explicitly identified by the JNCC as an indicator that blanket mire may be considered 'active' and thus worthy of 'priority status' under the terms of the EU Habitats Directive (Active blanket mire - erosion : JNCC website).

24.2.4.13 E2 active erosion gullies (formerly TA2 : *sensu* Lindsay *et al.*, 1988)

Figure 84. E2 active erosion gully
This deep erosion gully on the slopes of Carn-nan-Tri-Tighearnan, Highland Region, has cut down through almost 2 m of peat to the mineral sub-soil and is now eroding this glacial till, with few signs of re-vegetation.

Photo R A Lindsay

This nanotope is allocated to any erosion gully, or part of a gully, which is still actively eroding across its whole cross section – *i.e.* there is no sign of significant re-vegetation in any part of the gully at this point.

Some gullies are no wider than cut moorland drains, others are enormously wide and deep, while in other cases the landscape consists almost entirely of gully with a few remnant upstanding haggs scattered like islands across the area. All are classed (in nanotope terms) as E2 actively-eroding gullies (the microtopes in these contrasting examples should, however, differ quite markedly).

There is little to say here about E2 gullies, other than to highlight the fact that Lindsay *et al*. (1988) found such active erosion to be more frequent in the west and in the high ground of the Flow Country, while presumably Evans and Warburton's (2007) observation about the prevalence of re-vegetating blanket bog means that 50% of eroding blanket mire in upland England is not re-vegetating.

24.2.4.14 Em1/2 micro-erosion

(= 'microbroken' of Lindsay *et al*., 1988)

Lindsay *et al*. (1988) and Lindsay (1995) highlight and illustrate a condition of the bog surface which appears to be on the cusp: it may be a pre-cursor to large-scale erosion, or it may instead be capable of re-establishing a peat-forming vegetation. It is clear that the bog surface has been subject to some form of impact – most likely burning – and the normal range of nanotopes and microtopes do not apply.

'Micro-erosion' is actually a microtope pattern rather than an individual nanotope structure, but the small depressions associated with such micro-erosion do not sit comfortably within the concept of a true erosion gully. Consequently it is proposed that these be classed as Em1 and Em2, depending on whether they show signs of recovery.

Figure 85. Em micro-erosion
The surface has clearly suffered some disturbance – possibly fire – but is not yet eroding into deep gullies.
Photo R A Lindsay

24.2.4.15 Tk tussock

Areas of ground which were formerly Em1/2 micro-erosion but which have undergone significant vegetation recovery, or areas which were once peat cuttings and have since revegetated with hare's-tail cotton grass (*Eriophorum vaginatum*), have a characteristic feel to them.
Anyone walking across such areas soon discovers a further structural feature of micro-erosion: dense lumps which are easily felt through the soles of one's boots. These are not small moss hummocks because they are too solid for that; they are in fact tussocks.

The Tk 'tussock' nanotope is widespread in micro-erosion microtope patterns, but it is also found across a great many other peat bog surfaces where the *Sphagnum* layer has been lost at some stage (by fire, drainage or atmospheric pollution, for example). The tussock form is sometimes highly visible, but on a surface which has re-developed a vegetation cover, albeit a modified vegetation, it may not be so immediately evident. If it is there, though, it will make its presence felt – quite literally – through the soles of one's boots as hard lumps which may, in the case of hare's-tail cotton grass, occur as closely-packed steep-sided lumps almost 50 cm tall which make walking through them a misery.

Figure 86. Tk tussock
An example of a bog dominated by low tussock forms, also with evidence of micro-erosion.
Photo R A Lindsay

Weber (1902), however, gives a delightfully clear explanation of these solid lumps. He points out that deer sedge (deer grass : *Trichophorum cespitosum*) and hare's tail cotton grass (*Eriophorum vaginatum*) produce below-ground shoots, technically known as sympodia. When a new side shoot begins to grow it has 3-4 sections of stem (internodes) which, in a fast-growing *Sphagnum* carpet, extend rapidly upwards towards the light in order to avoid being smothered by the growth of *Sphagnum*. Consequently both deer grass and hare's-tail cotton grass tend to grow as open swards of individual stems when in vigorous *Sphagnum* surfaces (see Figure 87).

Where there is no *Sphagnum*, however, deer grass and hare's-tail cotton grass have no reason to waste precious resources on rapid extension of these internodes. The internodes for each year instead remain very short. In time, this results in a tightly compacted tussock (see Figure 87). An added benefit for the plant is that such dense tussocks are better able to resist fire – which is often a cause of the original *Sphagnum* loss. Fire becomes more likely on a peat surface which is not maintained in a wet condition by a living layer of *Sphagnum*. This is particularly so because the species which tend to increase under dry conditions (*e.g.* heather) are often highly flammable and produce much litter to act as tinder. The leaf litter of both deer grass and hare's-tail cotton grass burns readily, but the tightly-packed leaf bases and underground shoots do not. The end result is that the tussock form of these species enables them to perform better after fire than many other species, whilst fire damage itself actually intensifies the tussock growth form – to the continued advantage of the species.

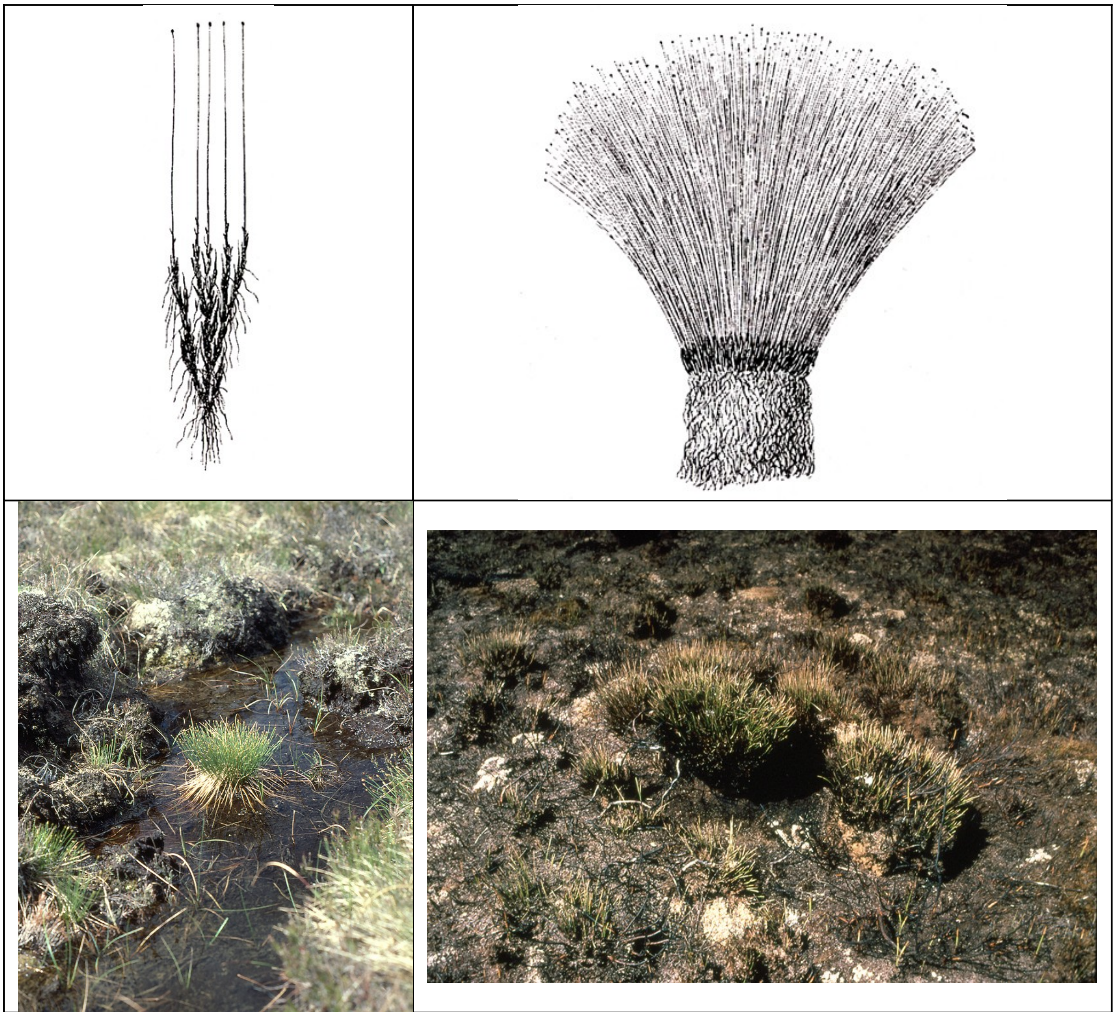


Figure 87. Tussock growth form
The upper two diagrams are taken from Weber (1902), and illustrate the way in which deer grass elongates its stem sections (sympodia) significantly to keep pace with growth of a vigorous *Sphagnum* carpet and thus produces individual stems growing through this carpet, while such elongation is not necessary when there is no *Sphagnum* carpet and so all stems bunch together into a tussock. The lower two diagrams show (on the left) just such a deer grass tussock illustrated by Weber (1902), while on the right a hare's-tail cotton grass tussock has developed after repeated fire damage. These feel distinctively 'lumpy' underfoot.

24.3 Microtope patterns – assembling the nanotopes

The many nanotope descriptions given above can perhaps best be synthesised by taking an actual area of blanket bog and illustrating the way in which these nanotopes create a jigsaw of elements which then together make up the microtope pattern. Such an example is provided in Figure 88.

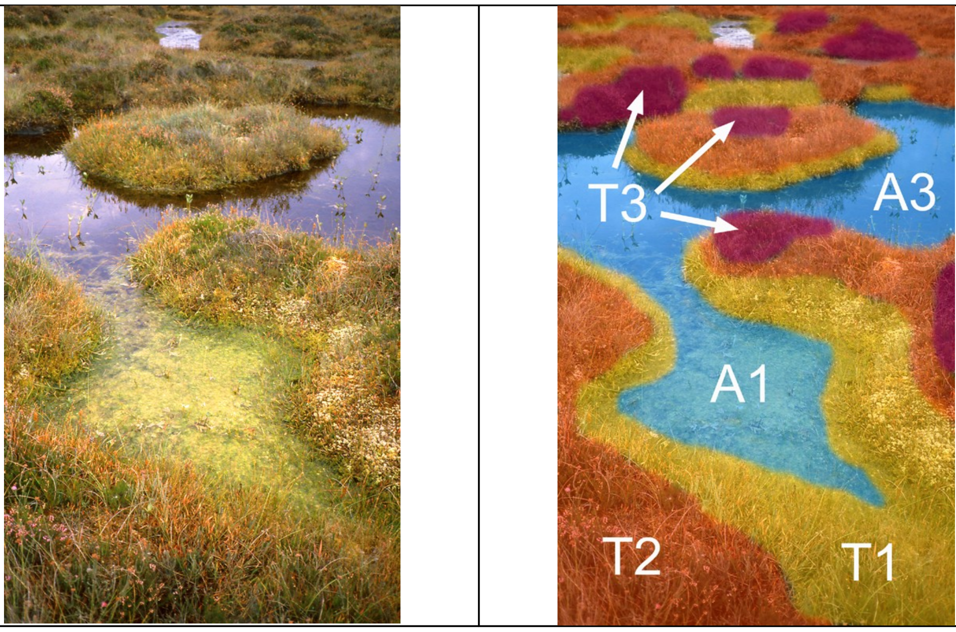


Figure 88. Nanotope distribution within a blanket bog surface
On the left is a photograph showing the detail of a microtope pattern displayed by a blanket bog area in the Flow Country of northern Scotland. On the right is a colour-coded layer over the same photograph, showing the distribution of nanotopes making up this pattern. The area marked as an A1 hollow in this case occurs as an offshoot of a larger A3 pool. This is because *Sphagnum cuspidatum* here forms a dense, firm carpet (though not quite firm enough to stand on). More usually, A1 hollows occur as discrete entities in their own right, while very loose, wind-blown collections of *S. cuspidatum* in larger A3 or A4 pools are generally simply regarded as part of the larger A3 or A4 structure. Note how the major part of the terrestrial bog surface appears to consist of T2 high ridge.

Photo R A Lindsay

To summarise the significance of microtopes and nanotopes:

- every part of a bog has a microtope pattern;
- this pattern may be simple or complex, but generally consists of contrasting drier and wetter elements, though the wetter elements are not necessarily aquatic;
- the pattern oscillates in rhythm with changes in climate;
- the pattern is long-lived, often demonstrably present (in oscillating form) for millennia;
- in general, aquatic elements (hollows, pools) tend to remain as aquatic elements, while terrestrial elements (ridges, hummocks) remain terrestrial;
- the pattern gives a very clear indication of gradient and relative rate of water movement;
- the pattern is made up from individual structural elements called nanotopes;
- nanotopes range from high hummocks and dense tussocks to *Sphagnum*-rich ridges and very deep pools;
- each nanotope is characteristic of a particular moisture range (and consequent peat density) in the peat at that specific point.