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Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration

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Abstract: Peatlands have been subject to artificial drainage for centuries. This drainage has been in response to agricultural demand, forestry, horticultural and energy properties of peat and alleviation of flood risk. However, there are several environmental problems associated with drainage of peatlands. This paper describes the nature of these problems and examines the evidence for changes in hydrological and hydrochemical processes associated with these changes. Traditional black-box water balance approaches demonstrate little about wetland dynamics and therefore the science of catchment response to peat drainage is poorly understood. It is crucial that a more process-based approach be adopted within peatland ecosystems. The environmental problems associated with peat drainage have led, in part, to a recent reversal in attitudes to peatlands and we have seen a move towards wetland restoration. However, a detailed understanding of hydrological, hydrochemical and ecological process-interactions will be fundamental if we are to adequately restore degraded peatlands, preserve those that are still intact and understand the impacts of such management actions at the catchment scale.

Key words: afforestation, blanket peat, drainage, moorland gripping, peat, water table, wetland restoration.

I Introduction

Peat is decaying organic matter that has accumulated under saturated conditions. Formation of peat therefore occurs in areas of positive water balance. Peatlands are more likely to form in regions with high precipitation excess, such as upland areas of

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the temperate and boreal zones or in lowland areas where shallow gradients, impermeable substrates or topographic convergence maintain saturation. Classification of peatland types is generally related to two fundamental factors: source of nutrients and source of water. Bogs are ombrotrophic peatlands dependent on precipitation for water and nutrient supply, whereas minerotrophic peatlands or fens are reliant on groundwater for water and nutrient supply (Johnson and Dunham, 1963). Bogs are therefore highly acidic (pH <4) and contain low amounts of calcium and magnesuim, whereas minertrophic peats are less acidic and tend to be base rich.

In England and Wales peat is classified as a deposit of at least 30 cm depth (50 cm in Scotland) containing more than 50% organic carbon (Johnson and Dunham, 1963). This definition is arbitrary as there is no clear break between a highly organic mineral soil (e.g., podzol) and an almost purely organic *Sphagnum* peat (Clymo, 1983). However, from this definition it is possible to say that 2.9 million ha or 13% of Britain is covered in peat, most (2.6 million ha) of which is in Scotland (Milne and Brown, 1997). This represents less than 1% of the 350 million ha of the northern peatlands that mainly occupy the boreal and subarctic zones (Gorham, 1991). In Britain the dominant peatland is blanket bog, which occurs on the gentle slopes of upland plateaux, ridges and benches and is primarily supplied with water and nutrients in the form of precipitation. Blanket peat is usually considered to be hydrologically disconnected from the underlying mineral layer. The British blanket peatlands represent around 10–15% of the world's blanket peat resource (Tallis, 1998). In some areas there are raised bogs where the peat has grown into a dome with a halo of lagg fen, overlying level mineral terrain or an infilled basin (Bragg and Tallis, 2001). However, Lindsay (1995) and Charman (2002) suggest that raised bogs and blanket bogs are simply end-points of an ecological continuum. Britain is also covered in approximately 6.1 million ha of peaty gley and peaty podzol soils that can be classified as shallow peats (Milne and Brown, 1997). There are now few areas of lowland Britain covered by extensive peat deposits, with the exception of the Somerset Levels and Cambridgeshire Fens; drainage for agriculture and peat-cutting for fuel and horticulture have reduced their extent (Burt, 1995).

The relative position of the water table within the peat ultimately controls the balance between accumulation and decomposition and therefore its stability. Peat is therefore very sensitive to changes in hydrology that may be brought about by climate or land use change. Greater aeration above the water table increases decomposition in unsaturated conditions relative to saturated conditions below, so having fundamental implications for properties and attributes above and below the water table. Three of the main land management practices to have resulted in changes to peatland water tables in Britain and elsewhere in the world are those of moorland ditching, pumped removal of water from fens and afforestation. However, several problems have been associated with these drainage activities; some of these problems were recognized as early as 1862 when Bailey-Denton discussed the uncertainty related to the effects of pipes and ditches on river flow. Moorland drainage is often blamed for increased flooding in UK rivers (e.g., Lane, 2001). There are also problems related to water quality, erosion and ecosystem destruction. This paper attempts to shed light on the nature and extent of these problems and review the progress made in understanding hydrological and hydrochemical processes associated with drainage of peats. The paper will firstly give an overview of peat drainage practice before reviewing the literature to show that artificial drainage of peatlands is unsustainable. The paper will then discuss the future needs for wetland research and peatland restoration; our understanding of many hydrological and hydrochemical processes associated with peat drainage is still poor yet the processes may have crucial implications for global environmental change given that peatlands act as an important terrestrial carbon store.

II History and extent of drainage

Many European countries have witnessed vast amounts of artificial peatland drainage including The Netherlands, Finland, Russia, Ireland and the UK. In Ireland drainage of peats and gleys has been reported since 1809 (Common, 1970; Wilcock, 1979). Most of the Irish peat drainage was associated with the aim of reducing flooding but drainage schemes altered and accelerated after the second world war owing to the need to increase livestock production in upland farms (Stephens and Symons, 1956; Common, 1970). In Northern Ireland there are only 169 km^2 of intact peat left compared with 1190 km^2 of total peatland (Cooper *et al.*, 1991). In New Zealand where peat soils cover more than 180 000 ha, peatlands were extensively drained for farmland in the 1970s with little regard to their ecological or environmental value (Bowler, 1980).

Britain is one of the most extensively drained lands in Europe (Baldock, 1984) and drainage of peatlands has played a fundamental role in the history of British farming (Williams, 1995). More than half the agricultural activity in Britain occurs on land that has been drained (Newson, 1992). Land drainage commenced before Roman times and there are records of it in Domesday (Darby, 1956). In Britain drainage took off in the seventeenth century accompanying land tenure, enclosure and reclamation of the Anglian Fens. In the following hundred years, peat shrinkage and subsidence associated with the pumped removal of water from the fens meant that more and more water had to be removed to render the drainage works useful (Cole, 1976). Until the twentieth century most drainage activity had focused on 'improving' fenlands for agriculture by lowering the water table. After 1900 drainage was also directed towards flood alleviation; expansion in ditching, tile draining and channelization activity was huge. The 'feed Britain' post-second world war era saw government grants for expansion in drainage works paid at 70%, particularly in agriculturally marginal upland areas. It was in the 1960s and 1970s that most of the upland drainage of blanket peats took place, particularly in the English Pennines. The peak rate of drainage is estimated to be 100 000 ha yr^{-1} in 1970 (Green, 1973; Robinson and Armstrong, 1988). Economic incentives for upland drainage were not limited to the twentieth century. In the mid-eighteenth century Turner (1757) provided a cost-benefit analysis of moorland drainage. His essay, which also showed that the peat bogs of upland Britain were not remnants of recessional deposit left after the 'Great Deluge', suggested a three-phase model for 'improving moorland' involving cutting open surface drains, adding sand and earth to the surface and the establishment of twitch grass.

The Cuthbertson plough was developed in the 1930s and has been used to create steep sided, open ditches (commonly called 'grips' in northern England) that are traditional for draining 1.5 million ha of blanket peatland in upland Britain (Stewart and Lance, 1983). The drains are often contoured or in a 'herring-bone' shape with short lateral feeder ditches collecting into a central ditch. Single isolated ditches are sometimes used for tapping springs or other natural seepages (Stewart and Lance, 1983). Moorland draining was carried out with the purpose of lowering the water table and removing surface water to improve the vegetation for grazing and game. Partly this drainage was to improve the quality of grazing and partly to remove the hazard to stock (Ratcliffe and Oswald, 1988). However, Stewart and Lance (1983) demonstrated that there was no evidence that peatland draining fulfils the claims made for it. Grouse populations do not seem to have increased and whilst drains are the cue for increases in stocking density there is little evidence that the moors can sustain large increases. Thus Newson (1992) suggested that upland drainage was backed by very limited rationale. As such the economic benefits are very low and yet the potential environmental effects high (Newson and Robinson, 1983). In general there has been very little research into artificial drainage of hill areas. In particular hydrological monitoring and process-based measurement has been poor. This is surprising given that large sums of money have been spent on draining the slopes (and that large sums are planned to be spent on peatland restoration the future).

In addition to drainage for agricultural use, about 15 million ha of northern peatlands and wetlands have been drained for forestry, mainly in northern and eastern Europe and the British Isles (Paavilainen and Paivanen, 1995). In Britain, about 190 000 ha of deep peatland and 315 000 ha of shallow peats have been afforested with coniferous plantations since 1945 (Cannel *et al.*, 1993). However, in order to ensure successful establishment of trees on peat soils, the water table must first be lowered. In Scandinavia, Finland, Russia, Canada, Ireland and Britain, drainage by a combination of closely spaced plough furrows and deep (usually 0.5–2 m) but more widely spaced ditches has taken place. The result is frequently a change in runoff production from the hillslopes both in the short term while the drains are active (David and Ledger, 1988; Prevost *et al.*, 1999; Anderson *et al.*, 2000) and in the long term when the forest establishes. From the time of canopy closure, the increased interception of rainfall leads to greater evaporation by the trees and enhanced evapotranspiration that encourages drying of the peat and the development of shrinkage cracks. In Finland, 5.7 million ha of peatlands have been drained, so that now one-quarter of the country's forested land consists of drained peatland (Laiho *et al.*, 1998). In Scotland 25% of Caithness and Sutherland peatlands have been affected by differing intensities of drainage associated with afforestation (Ratcliffe and Oswald, 1988). This area recently became the focus of major conservation protest and international condemnation (Charman, 2002).

III Impact of peat drainage on catchment hydrology

Conway and Millar (1960) were the first to examine experimentally the effects of moorland drainage on the hydrological response of peatland catchments. They reported results from four small (2 ha) moorland catchments at Moor House in the English north Pennines; two had natural drainage channels and two had artificial networks of moorland drains. They concluded that runoff production in blanket peat was extremely rapid especially where hillslopes had a dense gully network, had been burned or were artificially drained giving an increased sensitivity of runoff response to storm rainfall with peak flows both higher and earlier. In contrast, relatively uneroded subcatchments exhibited a smoother storm hydrograph with greater lag times and the water balance calculations suggested that uneroded hillslopes could retain significantly more water than drained, eroded or burnt basins. This paper was inferred by many to have therefore suggested that moorland drainage increases flooding downstream and reduces the water storage capacity of the hillslopes. However, a number of other small investigations followed, some showing conflicting and some showing corroborating results. Burke (1967) investigated water balances in drained peatland at Glenamoy, Ireland. In contrast with results from Moor House, runoff tended to be quicker from the undrained part of the bog with the water table very close to the surface. In the drained bog the water table was often 45–60 cm deep and runoff from the catchment was much slower. The reason given for this was that in the drained catchment most of the runoff flowed underground to the drains whereas in the undrained catchment runoff was generated at the surface and could be transmitted much quicker from the catchment. Similar results were reported for German peatlands by Baden and Egglesmann (1970). Runoff:rainfall ratios from the undrained Glenamoy catchment were only 23.4% compared with 79.2% from the drained catchment (Burke, 1975a; b). This is a remarkable difference and demonstrates the importance of enhanced understanding of the effect of land management practices on the hydrology of peatlands. Indeed Burke suggested that his evidence had important implications for catchment management: 'The results also indicate that if widespread drainage is undertaken in the area, beneficial effects on stream and river flow will follow. Floods will be reduced in frequency and amount and summer flow of streams will be increased in the short-term' (Burke, 1975a: 176).

McDonald (1973), however, noted that whilst the results from Conway and Millar (1960) and Burke (1967) seemed to be in direct contrast there was a lack of comparability between the study catchments. The peat at Glenamoy was more *Sphagnum*-rich than Moor House and a limited number of measurements showed that hydraulic conductivities were generally an order of magnitude higher at the less decomposed Irish site. McDonald (1973) suggested that drainage of one peat type will have a different effect on runoff–rainfall relationships than drainage of another peat type and as such the use of the broad term 'peat' has been misplaced. McDonald (1973) placed great emphasis on the importance of peat type but he also noted that drainage patterns were crucial. Robinson (1980) pointed out that at Moor House the drains were 0.5 m deep and 14 m apart, compared with Glenamoy where they are about twice as deep and four times closer together. Thus Robinson (1985) suggests that drain density was the most important difference between Moor House and Glenamoy. Of course Burke (1967) had already established that drain density was an important factor at Glenamoy showing that water table was only affected within 2 m of the drains. Since the aim of any drainage work was to lower water table, a drain spacing of 4 m was therefore required. Thus, the low hydraulic conductivity of peatlands frequently renders drainage operation unsuccessful or uneconomic because extremely close ditch spacing is required in order to significantly lower the water table, although this will depend on the properties of the peat (Huikari, 1968; Boelter, 1972; Hudson and Roberts, 1982). Conway and Millar (1960) had never established whether their drains significantly affected water table and thus a 14-m spacing was established without recourse to soil properties. Stewart and Lance (1991) later showed that water table was only affected within 0.5 m of the Moor House ditches. It is clear that both ditch network design and soil properties are important in determining the effects of artificial drainage on water storage and runoff generation from a peatland.

Ahti (1980) found that flood peaks increased drastically after ditching and peaks increased as ditch spacing decreased. For Burke (1967, 1975a, b), however, closer ditch spacing would result in a greater effect on water table, increased temporary storage and a subdued runoff response to rainfall with lower flood peaks. Clearly the effects are more complex depending on local site conditions. Comparison of Burke's (1975a) hydrographs (in particular the ones from his undrained plot) with other published hydrographs from intact moorland areas suggests that the Glenamoy catchments are not typical. The smooth delayed flow does not compare well with many upland peat catchments where a much more flashy flow regime would be expected (e.g., Bay, 1969; Gardiner, 1983; Labadz, 1988; Burt *et al*., 1990, 1997; Evans *et al*., 1999; Holden and Burt, 2000). Turner (1757: 31) noted 'Before draining and improving peat bogs . . . it will be necessary to examine the nature and properties of peat itself, which is in the nature of a sponge; for if a dry piece is put in water it will absorb double its weight'. However, we now know that many bog peats *do not* typically act like 'sponges' as Turner (1757) and many others since have assumed. Rather, baseflows are poorly maintained and runoff generated very quickly from the near-saturated hillslopes. However, Turner (1757) makes a useful point about examining the properties of the peat before drainage. We will see below that very few took heed of this advice and we therefore know very little about peatland process and why peatlands respond to drainage in such disparate ways.

Table 1 provides a list of papers that have examined hydrological response to artificial drainage in peatlands. Typically these are all water balance approaches and they either simply present the results with limited explanation or provide some explanations but have no corroborating field evidence for them. There have been few instances of hydrological process-based measurement within the catchments themselves. The papers all provide similar conflicting and corroborating results as illustrated by the comparison between Moor House and Glenamoy discussed above. These papers concentrated solely on the effects of drainage on river flow or on how well drainage activity could be utilized to 'improve' the land, often with a blatant disregard for ecological sustainability. A classic illustration is provided by Institute of Hydrology (1972) who assessed the work of Conway and Millar (1960), Hill Farming Research Organization (1964) and Burke (1967). In a remarkable ecologically unfriendly statement, the Institute of Hydrology (1972: 19) concluded against Conway and Millar that: '. . . in the short term, a drained upland or lowland peat may be a better "sponge" than an intact mire surface. All long-term planning of peat covered catchments must take into account whether it is better to have bare bedrock or an undrained mire'.

Many other studies since the Institute of Hydrology report have shown that drainage increases flood peaks but with a similarly 'anti-green' edge. Robinson (1980, 1986) found that for the Coalburn catchment in Northumberland, ditching increased peak flows (a 40% increase in the unit hydrograph peak – Robinson, 1986). Annual runoff increased by 5% even though rainfall was less after drainage in the catchment. Robinson (1980) suggested that drained moorland is better for reservoirs, especially during summer, than forested or undrained moorland, as the increase in annual flow mainly occurs through maintenance of summer low flows at a higher level.

Several studies have been based on examination of river flow and water balance at the large catchment-scale rather than at the hillslope or plot scale. Lewis (1957) suggested that land drainage had a 'noticeable affect' on flood discharge into the

Table 1 Reported hydrological effects of peatland drainage

	Affect on temporary storage	Affect on flood peak	Affect on annual runoff	Quantitative assessment*	Processes measured (other than stream flow)	Process discussion
Lewis (1957)	↓	↑	↑	C	X	storage
Oliver (1958)		↑		C	X	storage
Howe and Rodda (1960)		↑	↑	X	X	X
Conway and Millar (1960)	↓	↑	↑	H	X	storage burning
Mustonen (1964)		↑		H	X	X
Burke (1967)	↑	↓	↑	H	water table	storage
Howe *et al*. (1967)		↑	↑	C	X	drainage density
Baden and Egglesmann (1970)	↑	↓		H	X	storage overland flow
Institute of Hydrology (1972)		↑	↑	C	X	storage
Moklyak *et al*. (1975)	↑↓	↑↓	↑↓	C	X	YES – lots
Heikurainen (1968)	↑	↓		H	X	X
Ahti (1980)	↓	↑		H	X	drainage density, overland flow
Robinson (1980, 1986)	↓	↑	↑	H	X	YES – lots
Newson and Robinson (1983)		↓	↑	C	X	Catchment characteristics
Guertin *et al*. (1987)		↑		X	X	X
Gunn and Walker (2000)	↓	↑	↑	H	X	Vegetation changes

Notes:
*C, large catchment data within which some parts of the catchment have been artificially drained; H, small subcatchment or artificially drained hillslope monitored.

reservoirs on the Alwen catchment. Oliver (1958) also suggested that river regime at Learmouth had changed because of hill drainage in the upper Eye and Humbie catchments. Howe and Rodda (1960) observed qualitatively that plough ditching and drainage associated with forests in the Ystwyth catchment expedited runoff. Howe *et al*. (1967) examined changes to flooding in the Severn and Wye catchments. The Severn has witnessed significant afforestation with accompanying peat drainage. The Coweeta and Wagon Wheel gap studies in the USA have provided context for how afforested catchments could be expected to behave (e.g., Hoover, 1944; Hursh, 1951; Croft and Hoover, 1951) but the paired catchment experiments at the time were generally water balance studies and referred to yield rather than peak flows which did not seem to decrease. After the floods of 1946, 1947 and 1948 in the Severn Valley public opinion was roused against the drainage and afforestation schemes in the catchment. Howe *et al*. (1967) estimated that increases in drainage density brought about by moorland drainage are likely to have resulted in the increased flood peaks in the Severn and agreed with Conway and Millar (1960) that drained peatlands were more sensitive to rainfall with increased flood peaks and shorter lag times. Thus, for the River Severn the trigger mechanism for flooding was considered to be the increased incidence of intense storm events but concomitant land use changes had aggravated the problem of flooding in mid-Wales. Generally these catchment-scale studies suffer from poor data availability and thus conclusions tend to be rather piecemeal or anecdotal. Often river flow records are not available for periods before or during drainage operations and fail to cope with high flow measurement. Institute of Hydrology (1972) discussed the Brenig catchment, for which a good series of 40 years of records could be compared with the period 1960–65, which was when 40% of the catchment was ploughed and drained by the Forestry Commission. Annual streamflow was found to have increased by 10% with daily flows up by 2.5 $m^3 s^{-1}$.

Moklyak *et al*. (1975) present quantitative evidence from a peatland area in the Ukraine showing that drainage can both reduce and increase total runoff from peatlands within the same area. Like the Brenig study, the Moklyak *et al*. (1975) paper is rare because river flow was monitored before and after drainage operations. Out of five catchments investigated, three had reduced annual runoff and flood peaks following drainage, one had an increase in annual runoff and flood peak and one catchment had no significant change in flow regime. There was inconclusive evidence for any explanations for these phenomena although at least Moklyak *et al*. (1975) attempted to place emphasis on the potential processes responsible. They suggest, in line with McDonald (1973), that the peat type and drainage technique used were important determinants. Decreases in flood and annual runoff may come about following drainage because of a reduction in hydraulic conductivity, loss of surface runoff by storage in the upper peat layers, flow loss by storage on soil slopes and depressions caused by subsidence, increased evaporation related to changes in vegetation and use of sluices or canals that store water and increase evaporation. Flow increases may have been caused by increased direct precipitation in drainage channels, temporary flow increases by straightening, deepening and clearance of vegetation from streams and ditches, decreased evapotranspiration from drained but uncultivated land, an increase in surface and groundwater slopes, an increase in exposure of previously confined aquifers and artesian waters and increased drainage of previously closed marshy systems.

Robinson (1986) attempted to evaluate some physical mechanisms causing changes in yields at Coalburn. Many of these were similar to those discussed by Moklyak *et al*. (1975) but Robinson (1986) was able to discount many of the potential mechanisms for the increase in flood peaks (Table 2). The increase in drainage density was seen as the most important factor. Noteably the drainage density at Coalburn was naturally high at 3.5 km km^{-2} and was increased 60-fold by draining.

The effects of ditching may depend on where in the catchment the disturbance takes place. For example, drainage of part of a catchment may result in delayed runoff from hillslopes where peak flows normally occur before the catchment peak. The result could be that drainage increases the peak discharge in the catchment because the timings of the catchment and drained subcatchment peak flows correspond. Hence, even though drainage may result in a reduction in the flood peak at the hillslope-scale the net result may be an increase at the catchment-scale depending on where in the catchment the drainage operations took place and how that part of the catchment responds. No work has been done on this aspect of peatland hydrology and clearly a catchment modelling approach is required. Higgs (1987) suggested River Severn flood events have increased over the past 60 years and that these are directly related to variation in heavy rainfall since 1920. However, between 1968 and 1985 there had been a decrease in flood magnitude and frequency related to land use change. Drainage and afforestation had resulted in more flashy flow in the upper reaches of the catchment but the effects of the land use change varied according to location in the catchment. Thus, in a larger catchment, drainage schemes in headwater regions may have different consequences on the flooding regime compared with floodplain schemes through the effects of flood wave synchronization (Higgs, 1987).

Table 2 Processes discussed by Robinson (1980) that could account for changes in flow regime (increased annual runoff and flood peak) at Coalburn

Reason	Evaluation	Decision
A decrease in soil moisture would lead to a temporary flow increase while water drained from wetter area and turf ridges	The changes did not decrease over time	X
Drier soil would result in decreased evapotranspiration and hence runoff increase	But this would lead to shrinkage and there was little evidence of this at Coalburn	X
Drains occupy 10% of area and hence direct channel precipitation would increase	Yes but only during a storm and yet medium flows are most affected at Coalburn	X
Bare soil area would increase and evapotranspiration would decrease and hence runoff could increase	Rapid revegetation of the turf mounds and disturbed surfaces occurred	X
Increase in drainage density removing surface water from the catchment more quickly	Robinson thought this was best reason	✓

IV Impact of peat drainage on soil properties

1 Hydrological implications

Many drained peatland catchments exhibit increases in low flows. Robinson (1985) suggests that there is not enough evidence to support the idea that drainage decreased low flows at Moor House as suggested by Conway and Millar (1960) and hence agrees with the increases reported by Baden and Egglesmann (1970), Mustomen and Seuna (1971), Heikurainen *et al*. (1978), Robinson (1980) and Ahti (1980). The increase in low flows has sometimes been attributed to catchment 'dewatering'. The drained Glenamoy catchment was estimated to lose 1000 mm of water per year (Burke, 1975a) through slow drainage of the peat. While lowering of the water table increased short-term (storm-event) water storage and made the runoff response to rainfall less sensitive, in the medium term water was being lost from the catchment. This, of course, was partially the intention but in peatlands this has often been found to be unsustainable because of associated feedback mechanisms. In the long term, as peatlands dewater they are also liable to subside and decompose so that the temporary increase in water storage capacity may be lost and the catchment may start to behave in a more flashy way and increase the flood risk. Burke's study was not maintained over a sufficient length of time to establish whether these effects occurred at Glenamoy, but certainly relaxation times are an important element that have been ignored in most peat drainage studies. Robinson (1986) suggests that at Coalburn the 20% increase in the peak of the six-hour unit hydrograph in the first five years after ditching was reduced by half after ten years.

Drainage of the fens has been associated with severe shrinkage and decomposition of the peat such that large pumping operations have had to be implemented to keep apace with the subsidence of the soil surface. The shrinkage occurs because, as the water table is lowered, the upper peat collapses causing bulk density to increase by up to 63% in the upper 40 cm within a few years of drainage (Silins and Rothwell, 1998). The subsidence is associated with physical breakdown and consolidation of dry peat in surface layers and accelerated mineralization of organic matter (Egglesmann, 1975). The subsidence is also associated with the collapse of readily drainable macropores (Silins and Rothwell, 1998) which are ordinarily important pathways for runoff generation in peat (Baird, 1997; Holden *et al*., 2001). The dry surface increases capillary action resulting in more water being removed from the subsurface layers. Hence the whole peat mass dries more and shrinks, since peat tends to be 90% water by mass and 300% by volume (Hobbs, 1986). Anderson *et al*. (1995) investigated the effects of afforestation on blanket peat water tables, finding that shallow ploughing significantly lowered the water table followed by subsidence of the ground surface by a few centimetres as a result of consolidation of the peat at all depths. With shrinkage and consolidation, drain life is severely reduced (Prus-Chacinski, 1962) and many mires change topographical shape around drains. Surface 'wastage' (or decomposition) is also increased as bacterial aerobic action more readily decomposes the near-surface soil that is no longer anaerobic (Prus-Chacinski, 1962, Ivanov, 1981). Once peat dries it often becomes hydrophobic and cannot regain its initial moisture content (Egglesmann *et al*., 1993). Subsidence and irreversible drying of peats has been noted as a problem following drainage in New Zealand (Bowler, 1980). At Waikato 50 cm subsidence was measured in the 18 months following drainage. Holden and Burt (2002c) found permanent structural changes to blanket peats in the north Pennines subject to drought simulation in the laboratory. This lead to changes in the hydrological routing of water through the peat tested.

For catchments where drainage of peat decreases the flood response from disturbed hillslopes this is because the soil, catchment and ditch characteristics have enabled water tables to fall and thus the desired response of the slope to drainage is achieved. However, a fall in water table is often accompanied by increased peat decomposition at the surface and in subsidence of the peat mass. Thus the drainage operation becomes unsustainable. In other areas where drainage seems to increase flood response from a catchment, this tends to be where ditches have a very limited effect on water table. Thus, the ditches simply act to increase the speed at which surface storm water can escape from the catchment as storage properties are not significantly altered. In these cases the drainage activity has not succeeded in achieving its underlying objectives, even in the short term, and may cause problems downstream.

2 Chemical implications

The lowering of the water table following drainage leads to a number of processes taking place within the peat that affects both its physical and chemical properties. The major impact of drainage is the lowering of the water table that leads to an increase in the air-filled porosity of the peat, which in turn affects microbial processes and thus decomposition rates. The oxygen allows aerobic decomposition to take place, which occurs at a rate about 50 times faster than anaerobic decomposition (Clymo, 1983). The oxygen also enhances the mineralization of nutrients, particularly the carbon-bound nitrogen and sulphur and the organically bound phosphorus. The top metre of deep organic soils can contain as much as 20 000 kg nitrogen (N), 10 000 kg sulphur (S), 500 kg phosphorus (P) and 500 000 kg of carbon (C) (Miller *et al*., 1996) so even an increase in mineralization of just 1% yr^{-1} has the potential to generate large losses of these elements. The loss of nutrients may in turn affect the fertility of peat. For example, De Mars *et al*. (1996) found that drainage of a Polish fen resulted in P and potassium (K) limitation as a result of aeration of topsoil, accelerated decomposition and increased nutrient release.

Drainage and subsequent lowering of the water table has been hypothesized to change peatlands from C sinks to C sources to the atmosphere as a result of increased oxidation of organic matter. Laine and Minkkinen (1996) investigated the post-drainage change in the peat C stores by determining the bulk density and C content of peat profiles along a transect from the undrained part to drained part of a mire in Finland. They found that the differences between the undrained and drained peat C stores indicated that the accumulation of C had been 35 g C m^{-2} yr^{-1} greater in the undrained site over the 30 years since drainage. In contrast, a study of 273 forested peatlands in Finland 60 years after they had been drained reported that on average the peat surface had subsided 22 ± 17 cm, the C density had increased by 26 ± 15 kg m^{-3} and the C stores had increased by 5.9 ± 14.4 kg m^{-2} after drainage (Minkkinen and Laine, 1998). Domish *et al*. (1998) suggested that increased organic C flows from tree stands into the soil and consequent retention in the peat accounts for the increase in C storage in drained, forested peat soils. However, we are unaware of any studies that have investigated the impact of drainage on C storage in moorland peats.

A number of studies have observed that the exchangeable cation content in drained peats is lower than in undisturbed peats and total concentrations of N and P often increase whereas K always decreases in the topsoil (0–20 cm) of peat after drainage (e.g., Laiho *et al.*, 1998; Sundstrom *et al.*, 2000). For example, Sundstrom *et al.* (2000) observed that drainage with 60-m ditch spacing in Sweden led to an increase in concentration of total N and P, a decrease in concentrations of total K, calcium (Ca) and magnesium (Mg) and had little effect on soil pH. Because of the increase in bulk density of the peat, the total amounts (kg ha^{-1}) of N and P showed an even greater increase, whereas the drained peat contained only 25–40% of the K that were present in the topsoil of the undrained peat (Sundstrom *et al.*, 2000). In Canada, Wells and Williams (1996) investigated the impact of ditch spacing on soil nutrients in both bog and fen peats. They observed that in bog peats bulk density, total N concentrations (mg g^{-1}) and total contents (kg ha^{-1}) of N, P, K, Ca and iron (Fe) were significantly higher in the 3-m ditch spacing compared with the 15-m ditch spacing. They concluded that increases in total nutrient contents in drained bog peats could be attributed mainly to increased bulk density. In contrast, they observed that bulk density and most nutrient contents of fen peats were not significantly affected by drainage.

The increase in total N concentrations (mg g^{-1}) observed in the topsoil of peat after drainage is due to an increase in the retention of N by microbial immobilization as the plant residues in the peat decompose and total N is increased per unit volume of peat (Wells and Williams, 1996), which also results in a lowering of the C:N ratio. However, many studies have also observed that drainage and lowering of the water table results in an increase in N mineralization (Williams, 1974; Williams and Wheatley, 1988), in response to an increase in oxygen and the number of ammonifying and nitrifying bacteria. For example, Williams and Wheatley (1988) observed that on lowering the water table from 0 to 50 cm the mean content of available mineral N in the peat profile increased by a factor of 1.5. The response of N mineralization to water table lowering, however, is not always predictable. For example, Williams (1974) observed that lowering the water table to 18 cm significantly decreased the amount of N mineralized in the top 10 cm of peat but that further lowering of the water table to 34 cm increased mineralization in the top 10 cm.

Mineralization–immobilization responses of soil N to peatland drainage depend largely on the change in peat decomposition rate, which is regulated by environmental and substrate factors. Environmental factors include temperature, redox potential and pH. Substrate factors include stage of decomposition, organic matter quality, nutrient content, chemistry of the soil solution and the presence of chemical and biological inhibitors to microbial activity. Although lowering the water table should eliminate poor aeration as the foremost limitation to mineralization, the improved aeration may have little impact on mineralization rates if temperature, pH or nutritional constraints still inhibit microbial activity. For example, Humphrey and Pluth (1996) observed that N mineralization rates did not respond to drainage in peat at pH 4.0 but were significantly stimulated in peat at pH 7.2. Updegraff *et al.* (1995) observed that aerobic N mineralization was at least twice as high as anaerobic mineralization in bog peats but not in sedge soils, and thus suggested that the sensitivity of N mineralization to aeration status depended on substrate characteristics related to the quality and quantity of organic matter. These studies therefore suggest large heterogeneity of N dynamics to drainage across the landscape depending on the interacting influence of environmental and substrate factors.

V Impact of peat drainage on water chemistry

As well as changes in runoff generation and soil properties, installation of drainage ditches has an impact on water chemistry. Sometimes where drainage appears to have little effect on catchment hydrological regime it can have significant effects on soil and drainage water quality (e.g., Ministry of Agriculture, Food and Fisheries (MAFF), 1980). Many studies have observed that installation of drainage ditches usually increases the leaching of nutrients. For example, large increases in ammonium (NH4) concentrations have been observed following drainage (Lundin, 1991; Sallantaus, 1995; Miller *et al*., 1996) and lowering of water table (Adamson *et al*., 2000) in blanket peat, but only small changes in nitrate (NO_3) concentrations. This suggests that while the organisms for ammonification benefited from drainage, those responsible for nitrification did not do so to the same extent. However, increased NO3 losses along with base cations have been reported from less acidic peats (Burt *et al*., 1990; Lundin, 1991; Freeman *et al*., 1993).

Sallantaus (1995) observed a net loss of Ca, Mg and K from drained catchments compared with undrained catchments, where inputs and outputs of the these nutrients were more or less balanced. Astrom *et al*. (2001) observed that forest ditching resulted in an increase in concentrations of suspended sediment, Ca, Mg, manganese (Mn) and aluminium (Al), a decrease in total organic C (TOC) and an increase in pH from 4.4 to 5.4 in stream water. In Scotland, Miller *et al*. (1996) observed initial increases in NH4-N and silica (Si) owing to losses from the exposed peat in the drains.

Studies that have investigated the impact of drainage on dissolved organic C (DOC) concentrations (and hence water colour) have observed contradictory results. Drained peat soils have been found to have more humus compounds and substances that are readily hydrolysed and thus runoff quality from the catchments is likely to be altered. Edwards *et al*. (1987) found that drained catchments produced much more discoloured (DOC-rich) water than undrained catchments. Clausen (1980) provided evidence that disturbed Minnesota peats produced higher concentrations of water colour, suspended sediment, K, Fe, Al and sodium (Na) with a reduction in pH than undisturbed catchments. In contrast, Moore (1987) observed only minor changes in stream DOC concentrations in drained and harvested bogs, compared with undisturbed peatlands in southern Quebec. Adamson *et al*. (1998, 2000) noted a decline in DOC and dissolved organic N (DON) in soil solution at 10-cm depth when the water table declined to 40 cm below the peat surface. Chapman *et al*. (1999) also observed significantly lower concentrations of DOC and DON in streams flowing through peaty podzols drained for forestry compared with steams flowing through undrained moorland.

Different results have been observed where drainage ditches penetrate the mineral soil beneath the peat. For example, Robinson (1980) found that the order of concentration of Na>Ca>Mg>K in drainage water changed to Ca>Na>Mg>K. Exposure of the underlying boulder clay at the base of the artificial ditches was used as a causal mechanism but there were no measurements of any processes. Reynolds and Hughes (1989) observed that the mineral soil exposed on the base of forest ditches acted as a source of Al. Astrom *et al*. (2001) also observed that mineral soils (till) exposed at the base and side of the ditch were sources of Al and Mn to stream water. They also suggested that the observed decrease in TOC and H^+ concentration was most likely due to immobilization in the mineral soil exposed in the base of the ditch. Forest drainage is often associated with the acidification of surface waters (Miller *et al*., 1990), however a number of studies observed an increase in the pH of drainage water, which has often been attributed to contact with mineral soil in the drainage ditches, although Paavilainen and Paivanen (1995) attributed pH increases in several studies to interception of more neutral groundwater after drainage.

In Canada, Prevost *et al*. (1999) investigated the impact of drainage on soil solution collected from 20 and 40-cm depth and at 1.5, 5 and 15 m from the centre of each ditch. They observed that the solute content of soil solution was enhanced by drainage, with the effect generally proportional to ditch closeness for S and Mg, while increases in N, Na, K and Ca were mainly observed within 5 m of the ditch and at 20-cm depth. This increase in solutes was associated with slight decreases in pH and coincided with an increase in soil temperature, a decrease in moisture content and accelerated decomposition rates observed within the top 30 cm and close to the ditches where water-table drawdown was greatest (Prevost *et al*., 1999).

Adamson *et al*. (2000) investigated the impact of water table drawdown in blanket peat on soil solution composition during a drought period at Moor House nature reserve in northern Britain. They observed a large increase in sulphate (SO_4), Na, Mg, Ca, NH_4 and H^+ concentrations at10-cm depth when the water table dropped to 40 cm below the surface of the peat. For 83% of the year water table is within 5 cm of the surface at the Moor House sampling site (Evans *et al*., 1999). During this time anaerobic conditions exist in most of the peat profile and anaerobic bacteria converts SO_4 to H_2S. However, when water table falls, aerobic conditions exist within the peat and the H_2S is oxidized to dissociated H_2SO_4, which generated the observed increase in SO_4 and H^+ in the soil solution at Moor House (Adamson *et al*., 2000). It is likely that some of the H^+ ions replace other cations on exchange sites resulting in the marked increase in Na, Mg and Ca concentrations. Freeman *et al*. (1993) manipulated water tables on laboratory peat columns collected from a valley bottom wetland in mid-Wales and also observed a large increase in concentrations of SO_4, as well as NO_3, DOC, Na, Cl, Fe and Mg. Calcium was the only solute to show a slower rate of release.

In fen peats, water is often pumped from the land, which results in the rapid lowering of the water table and transfer of solutes from peat to ditch. In Somerset, Heathwaite (1987) observed that SO_4 concentrations were at least three times higher in pumped-drained ditches compared with watercourses and that Ca and Mg concentrations were at least twice as high in pumped ditches.

Green (1974) noted that decreases in downstream water quality following drainage installation could often be associated not directly with ditching but the activities surrounding it, such as increased use of fertilizers. For example, although ditching will create conditions favourable to microbial activity and the release of nutrients, some studies show that the amounts of N released are insufficient for optimum tree growth (Williams, 1974; Williams and Wheatly, 1988). Hence, fertilizers are usually required to establish plantations on blanket peat. Liming and or fertilizers have been added to many upland peats in Britain while some areas have been ploughed and reseeded with grasses (Newbould, 1980). Many studies, especially in Scandinavia and Finland, have investigated losses of P and K from forest fertilization (Karsisto, 1970; Kaunisto and Mailanen, 1992). In Scotland, Miller *et al*. (1996) reported losses of 1–2 kg P ha^{-1} (of the 58 kg P ha^{-1} applied) and 25–35 kg K ha^{-1} (of the 108 kg K ha^{-1} applied) in drainage water in the year after application and noted the growth of moss and algae in the main drainage channels.

The majority of studies that have investigated the impact of artificial ditching on water chemistry have observed changes in solute concentrations and fluxes in the short term. However, the duration of the drainage effects on water chemistry is not known, as few studies have continued monitoring for more than 5 years. In addition, most studies have monitored the chemistry of drainage water rather than the soil solution, and few studies have linked these measurements to soil processes. Therefore it is not known in detail to what extent and by which mechanisms various solutes are released and leached in artificially drained catchments. Compared to forested peatlands, there is little information on the impacts of drainage on water chemistry in moorland peatlands.

VI Impacts of peat drainage on erosion

In some areas ditching can lead to severe degradation of wetland soils. Mayfield and Pearson (1972) noted that some ditches in Derbyshire have been known to erode severely in places, quickly becoming deep, wide channels and supplying large amounts of peat material to the channel system. Drains cut to 50 cm depth may erode to several metres. Institute of Hydrology (1972) reported that many peat drains can be highly erodible and there were serious problems in the Tywi forest, mid-Wales. At Coalburn, Robinson (1980) showed that sediment concentrations increased by two orders of magnitude during the drainage period and took several years to stabilize. Concentrations were still several times greater than pre-drainage levels after 5 years (although there was only a short pre-drainage calibration period). Sediment removal from drainage ditches can lead to ecological problems downstream. Burt *et al*. (1983) investigated pre-afforestation drainage in the southern Pennines. There was a marked increase in suspended sediment following ploughing that caused major pollution of a local reservoir and plentiful supply of storm sediment. In the Ribble and Hadden catchments, northern England, the Salmon catches fell during the 8 years following drainage from 1400 yr^{-1} to 380 yr^{-1}, while in the nearby Lune, where there had been no drainage, catches remained stable (Stewart, 1963). In the River Nuorittajoki in northern Finland, Laine (2001) observed that the recapture rates of stocked yearling salmon were lower in riffles receiving high inputs of particulate matter from drained peatlands than in riffles receiving a considerably smaller loading. In addition, the size of the salmon was inversely related to the estimated particulate matter load to the riffle. Changes to flow regime, sediment flux and masking of gravel bed spawning grounds by fine organic sediment makes the salmon redds unstable. However, little is known about the full impact of drainage on sediment movement or ecology in upland areas and more work needs to be done in this area.

Often in upland areas moorland burning accompanies drainage. The burning is designed to encourage new shoots of *Calluna* and *Eriophorum* for grazing and game. Burned bare peat areas can rapidly erode, particularly around drains where increased runoff across the burned peat surface increases particle entrainment both on the intact peat surface and within the drain networks themselves. Often grazing increases are associated with drainage activities yet often the moorland cannot sustain great increases in stocking densities. Rawes and Hobbs (1979) found that for north Pennine peats grazing densities over 0.55 ha^{-1} removed the *Calluna* cover and instigated erosion.

Moorland drainage has also been linked to slope instability. Mass movements of peat, usually reported as bog bursts or peat slides have been well documented over the last 150 years (Warburton *et al*., 2004). These mass movements transport vast quantities of material from slopes and some peat slides have been known to be larger than 1 km^2. Many peat mass movements in both the UK and Ireland have occurred in conjunction with artificial drainage where failure occurs along the artificial drainage line. Ditches are often found at the margin of failure scars and have been cited as possible contributors to failure and subsequent mass movement (Tomlinson, 1981; Wilson and Hegarty, 1993; Dykes and Kirk, 2001; Warburton *et al*., 2003).

VII Ecological protection

Even as late as 1984, Finn *et al*. were trying to measure hydraulic conductivity in peats so that ditch designs could be more adequately developed to lower the water table as far as possible. However, the recent greening of UK public policy and demonstration of the limited rationale behind moorland drainage, combined with perception of increased flooding has resulted in a complete reversal of attitudes towards artificial drainage of the uplands. Wetland environments are now appreciated for their habitats and as a valuable carbon store (Royal Society, 1993). Drained moorlands often lose their bog pools and their associated ecology (Ratcliffe and Oswald, 1988) and peat subsidence and wastage is seen as a major problem. Horticultural alternatives to peat are also being sought. In the 1990s UK drainage authorities were given conservation guidelines set out in Acts of Parliament (see Institution of Civil Engineers, 1993). The UK Environment Agency now has an environmental duty to further the conservation and enhancement of natural beauty and flora, fauna, geological or physiological features of special interest. For example, it is a stated objective of nature conservation agencies to provide for the future sustainability of raised mire habitat (Joint Native Conservation Committee (JNCC), 1994) with 'active raised bogs' and 'degraded raised bogs capable of regeneration' listed under the EC Habitats and Species Directive (1992) as priority habitats. Department of Transport and the Regions (DETR; 1999) produced a report indicating UK obligations in peatlands under European Law and the need for SSSI designation and more stringent control of peat extraction. The report discusses the need for rehabilitation and restoration of wetlands.

However, Maltby (1997) has emphasized that peatland ecosystems are not very resilient to stress in terms of water relations, suggesting that the biodiversity assemblage is highly vulnerable to perturbation. Bragg and Tallis (2001) similarly suggest that peatland vegetation may alter in response even to very small changes in water level and or water chemistry. Therefore it may not be a simple task to restore a disturbed peatland. Nevertheless, peatland degradation has been perceived as reversible. However, changes to peat pH and nutrient status as a result of drainage or fertilizers added to the peat in association with artificial drainage can make ecological restoration difficult.

VIII Peatland restoration

1 Approaches

Peatland restoration most commonly takes two forms. First the re-establishment of high water tables and secondly the recolonization of important peat-forming species such as *Sphagnum*. Schouwenaars (1993) suggests that ecologically *Sphagnum* is essential for peat growth and restoration and hence water tables must be maintained at a high level without great fluctuation. Where drainage has resulted in water table lowering and changes to peat properties there is a necessity to reconstruct the water storage capacity of the peat in order to allow *Sphagnum* to regrow and survive. Refilling of drains with strongly humified peat has been suggested (Egglesmann, 1988). The primary aim of the hydrological management of damaged and fragmentary peats is normally to minimize water loss through a strategy of ditch blockage or through some attempt at sealing the boundary of the mire to prevent the loss of water. Most attempts at restoration to date have concentrated their efforts within the boundary of the peatland area and often within the boundary designated for nature conservation, which may be considerably smaller than the original peat extent. Only in recent years have workers considered approaches using buffer zones outside the area of peat and beginning to think about integrated catchment management. Techniques have been applied at a wide variety of scales and costs, often without detailed monitoring to assess the effectiveness of the works.

Many restoration projects have concerned the reclamation of drained sites by means of deliberate ditch blockage. At Wedholme Flow, Cumbria, UK, a strategy of small ditch blockages using either peat plugs with a polythene membrane or tin sheets was employed (Mawby, 1995). Monitoring of peat anchors showed that the peat surface rose following damming. Figure 1 illustrates the behaviour of the undisturbed part and the drained part of the bog at Wedholme as a mean response of around 15 dipwells per site. Both intact and drained peat dipwells experienced a cyclical fluctuation in water table depth, with maximum depths experienced in summer during relatively dry conditions, and minimum water table depth in the winter. Although both intact and drained peat dipwells show a similar pattern, the amplitude of variation in water table depth is much smaller in the undisturbed peat than in the drained peat. For the period before the commencement of damming, both sites appear to exhibit a slow decline in water tables from March to August/September, followed by a relatively faster rise to a stable winter level. The programme of restoration commenced in January 1992, and Figure 1 demonstrates an almost immediate response with a high degree of correspondence between winter and early spring data for both sites. For the first summer after restoration commenced this correspondence broke down and water levels on the drained peatland still experienced a much steeper decline than those of the intact peat. Despite this decline, the drained peat water table levels did not fall back to the minimum levels experienced in previous years (a minimum mean value of –0.41 m compared with –0.53 m and –0.52 m for 1990 and 1991, respectively), whereas water levels in the intact peat dipwells fell to a level very close to those of the previous two years (a mean minimum of –0.24 m compared with –0.24 m and –0.23 m). After 1993 the disturbed water table corresponds well with that in the undisturbed part of the peatland. So water table recovery in peatlands following ditch blocking can be relatively rapid. However, that is not to say that vegetation or hydrochemical recovery will follow.

Price (1997) tested a range of water management approaches that attempted to ameliorate conditions limiting *Sphagnum* regeneration in North America. Water table depth was found not to be a good indicator of water availability at the peat surface because of decomposition of the surface layers. Simply blocking ditches caused good water table recovery during the wet spring period, but the water table recession was much faster and greater than in an undisturbed area. Price (1997) suggests more aggressive management techniques such as creating open reservoirs and using straw mulch (which increased soil moisture by 10–15%), in addition to blocking ditches to recreate a water table regime comparable with that in a natural area. Gunn and Walker (2000) studied the impacts of peat extraction, ditching and ditch blocking on runoff at Cuilcagh, near Enniskillin, Ireland. Intensive ditch blocking reduced the flashy nature of the flow from open ditches and produced a response similar to that of undisturbed bog. The extra discharge from the drained catchment, which came from increases to winter low flows linked to vegetation destruction, was reduced in the blocked area. It may often be necessary to seed vegetation on the surface of a damaged bog in addition to hydrological restoration and protection of existing vegetation. *Sphagnum* diaspores, for example, can be spread across the surface of the bog. These may need additional protection by mulching to enable establishment (Price *et al.*, 1998). Often peat and plastic ditch plugs are unsuitable for ditch blocking where slopes are steep and ditch waters scour around the plugs. *Calluna* bails are being used in some upland peats (e.g., at Halton-Lea-Fell, Cumbria, UK) where the seed bank and nutrients are local (cf. straw bails). These allow water to flow through the bails, but slow the velocity and allow sediment to slowly accumulate. The aim is to avoid further scour erosion around the ditch plugs and allow the ditch to slowly infill with sediment and vegetation.

2 Thresholds of recovery and nonlinear trajectories

Lindsay and Immirzi (1996) note that there are boundary conditions beyond which peatlands cannot be restored. For example, they suggest that a suitable depth of peat left *in situ* is often required, particularly if that peat is only supplied with water and nutrients by rainwater. Podschlud (1988) showed that the best chance of recovery was where the former, upper, slightly humified peat layer was still intact. Once the peat starts to regenerate it will eventually become self-sustaining and artificial water tables will no longer be needed. The general quality of a peatland is assessed by the degree to which it has remained capable of active peat growth (JNCC, 1994). This requires the continued existence of sufficient hydrological integrity of the peatland complex. Immirzi *et al.* (1992) suggest that only peatlands that are sufficiently hydrologically intact can form more peat. Thus, an essential element of any approach to wetland restoration is the assessment of damage, or threat of damage, to hydrological conditions, together with consideration of appropriate options for remediation.

The hydrological condition of a raised peat system, for example, is largely a product

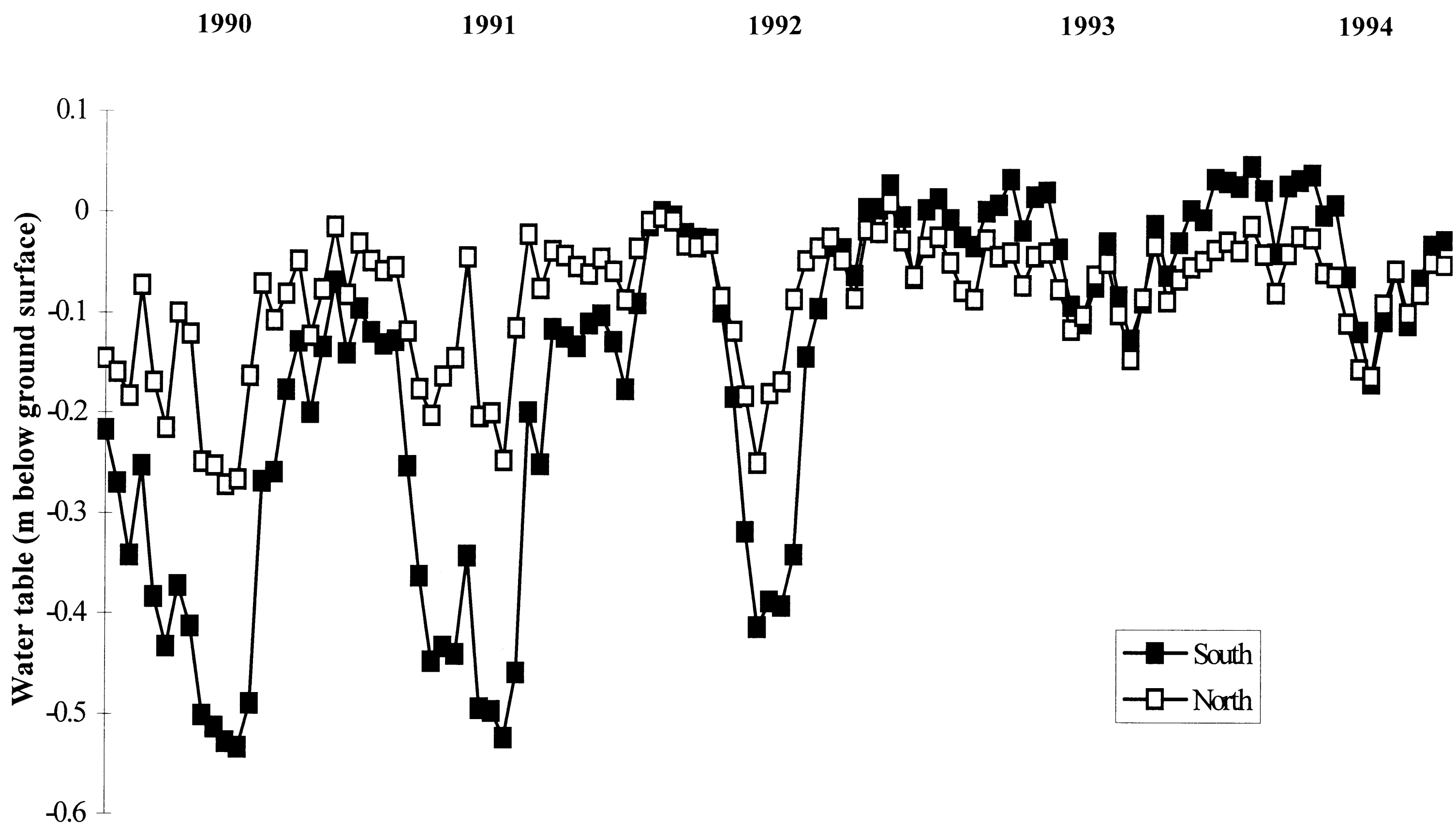

Figure 1 Mean water table depth 1990–1994 at Wedholme Flow, for North (intact) and South (cutover, restored during 1992) parts of the peatland
Source: data of Mawby, after White and Butcher (1994)

of the balance between two factors: the effective rainfall input into the system and the losses of water through evaporation, surface and subsurface runoff. In practice managers are clearly not able to control the rainfall input but it is important to stress that the degree of rainfall will control the sensitivity of the peatland to any damage. Those peatlands, such as Thorne and Hatfield Moors in South Yorkshire, UK, which are close to the threshold of rainfall required for *Sphagnum* growth will be more sensitive to drainage since there is less replenishment of the system. It is of note that much of the existing management strategy with regard to restoration of lowland raised peatlands is based on the hydrology of peats in their undisturbed state and associated with the ground water mound theory of Ingram (1982). The relationship between hydrological conditions in an undisturbed peatland and those within an artificially drained peatland, however, exhibit significant differences, as discussed by Eggelsmann *et al*. (1993): (a) the fragmentary nature of residual peat structures in a cut-over mire does not allow the creation of a ground water mound in any recognizable form; (b) the rapid transfer of water through the ditch systems to the edge of the mire acts as a significant control on general water table levels within the mire; (c) as a result of the increased area in which rapid drainage is taking place, hydraulic gradients in the peat are likely to be significantly greater than in an undisturbed system; (d) the drying of peat over time may increase hydraulic conductivity. Desiccation cracks within the peat may allow a far higher overall hydraulic conductivity than would normally be the case in an undisturbed mire. The increased heterogeneity in the hydraulic conductivity across the mire is of great significance where flow predictions are made, particularly if a distributed model is to be used (Holden and Burt, 2003b).

Holden *et al*. (2001) and Holden and Burt (2002a) showed that macropores and soil pipes were significant pathways for water movement in blanket peat. Once a ditch has been dug the peat can become exposed to weathering through freeze-thaw activity and summer desiccation. This appears to promote cracking and hence macroporosity on ditch slopes. An important feature of hydrological changes to peat is that they are often irreversible. MAFF (1978) noticed that experimental lowland peat drainage systems were often associated with increased soil cracking and fissuring during dry weather and that these fissures could persist through the following winter and for years to come. Hence blocking of ditches may result in more water entering through cracks and macropore networks through the ditch sides. This may promote development of subsurface pipe networks through turbulent action within the macropore networks once a ditch becomes filled with water. Soil pipes are commonly found in peatland catchments (Jones *et al*., 1997, Holden and Burt, 2002a; Holden *et al*., 2002). These pipes and macropores are able to rapidly transmit water to deeper layers within the peat mass than through the peat matrix. This is important because most water movement within peats tends to occur through the upper layers and very little runoff is generated from deep within the peat except via soil pipes (Holden and Burt, 2002b). With more water reaching deeper peat layers much more quickly following drainage or drought (Holden and Burt, 2002a), Warburton *et al*. (2004) suggest that this may result in changes to the hydrochemistry of runoff waters and may also result in a reduction of frictional strength within the lower peat layers or at the peat–substrate interface. Blocking of ditches in peats has been cited as a possible cause of slope failure owing to increased pressure in the drainage ditches (Wilson *et al*., 1996).

Natural revegetation of ditches and disturbed peatlands has been observed. If ditches are not maintained they can fill in with vegetation and sediment, losing their effectiveness in water removal (e.g., Fisher *et al*., 1996). Indeed, this 'benign neglect' of ditches may be one of the simplest management strategies proposed to return peats to favourable condition. Van Strien *et al*. (1991) suggested that reduction in the frequency of ditch cleansing will have a beneficial effect upon species richness. Robertson *et al*. (1968) noted that drains in a Lanark bog had 'ceased to function' owing to regrowth of *Sphagnum*, such that they can now only be detected by careful inspection. Mayfield and Pearson (1972) also noted that re-colonization of artifical drainage can be rapid where peat formation is contemporaneously in progress. Ditches in Bleaklow infilled rapidly when not maintained and disturbed peat can regenerate without intervention as witnessed in the north Pennines where extensive revegetation has taken place since the 1960s. Wilcock (1979) demonstrated that channel and ditch clearances in upland peats were only temporarily effective in withdrawing water from storage and that net annual replenishment starts within two years as revegetation of the ditches takes place. Wilcock estimated that it would take approximately 12 years for full recovery of Glenullin bog, northeast Ireland. Stewart and Lance (1991) noted that drain channels may remain bare for many years especially when they are overhanging with *Calluna*, but on flat and gently sloping ground the channels eventually fill with vegetation. Infilling often starts where peat has slumped onto the drain floor and is colonized by mosses and later by rushes and sedges. If unshaded the floor should regrow with *Sphagnum*. The tendency of drains to infill depends on the type of material forming the floor, the slope angle and hence the resistance to scouring. Van Seters and Price (2001), working on a naturally regenerated cut-over bog in Quebec, found that *Sphagnum* had not re-established even after 25 years from abandonment of peat working. They concluded that, without suitable management such as ditch blocking, *Sphagnum* regeneration may never occur. Natural healing of ditches only seems to occur in certain locations, particularly on gentle slopes and in peats with extremely low hydraulic conductivities. This is the case in the peats in the north Pennines at the Conway and Millar (1960) study site. The only major problem at the site is knickpoint erosion at the grip network confluence. Thus, it will necessary to establish management protocols to ensure that before ditch-blocking schemes are implemented it is determined that they are actually necessary. It may be that only small parts of the artificial drainage network need to be treated such as those on steeper slopes or where several drains connect.

IX Future needs

Currently in Britain, organizations such as English Nature or the National Parks are heavily investing in ditch-blocking restoration schemes. The River Swale is an important tributary of the Ouse that flows through York and has been subject to some recent severe flooding. Part of the River Swale Regeneration Project aims to examine the relative roles of climate change and land use change in exacerbating the downstream flood risk. In 1997, English Nature undertook the blocking of several areas of artificially drained moorland in the Swale headwaters. English Nature's main interests lie with the Upland Heath Habitat Plan and promotion of biodiversity in the British uplands. While hydrology is central to ecological restoration peatlands (Schouwenaars, 1993; Price, 1997) unfortunately no hydrological or hydrochemical monitoring of the blocked or unblocked sites was undertaken, which means that we have little information on the wider success of these projects. The Yorkshire Dales National Park, in partnership with English Nature, the Environment Agency and The National Trust, are now looking to block several more areas of moorland drains in the region. While historic problems with the data record make establishment of climate or drainage effects on river regime difficult, only new instrumentation coupled with process-based monitoring of a range of grip blocking schemes will allow development of cost-effective integrated catchment management tools and improve understanding of process.

In Upper Wharfedale, north Yorkshire, work is currently underway to provide process-based monitoring of artificial drainage and restoration practices. The study (see McDonald *et al.*, 2003) is assessing the impact of management strategies on water quality, quantity and sediment delivery. Rather than relying on traditional water balance and river regime investigations the project will involve much more process-based work at a smaller scale coupled to catchment and hillslope-scale monitoring to examine the hydrological processes and the feedback mechanisms related to water quality and changes to soil properties. In catchments where flooding is a problem one of the aims of ditch blocking as seen by management authorities is to reduce the flashy nature of the open ditches and produce a more subdued hydrograph response. However, as Evans *et al*. (1999) and Holden and Burt (2003a), show intact peat catchments can produce very flashy runoff anyway. The effects of ditch blocking may therefore be relatively small on the river hydrograph but important on hillslope flow routing, water quality and sediment release. There are other problems surrounding those areas where artificial drainage has resulted in decreased storm peaks downstream – will blocking the ditches cause increases in downstream flood peaks because of changes to tributary synchronosity? Again, effects will depend not only on soil and drainage properties but also on where in the catchment the land management change takes place. The development of integrated models that can be applied to a range of upland catchments to predict the effects of spatially localized changes in management practice such as afforestation, deforestation, ditch blocking and changes to grazing intensities will be of enormous benefit. One of the major problems associated with UK peat restoration is the lack of maps of artificial drainage; there are often no available records. Here LIDAR (LIght Detection And Ranging) flights are proving useful as the filtering algorithms are improving, so that we can now very quickly identify ditches on hillslopes with a precision of a few centimetres This is enabling production of highly accurate digital elevation models that can be coupled to hydrological models. These high resolution models should be able to predict which ditches or ditch networks are the most important ones to target for blocking.

X Conclusions

Most of the studies associated with artificial drainage of peats have been black-box water balance studies with limited measurement of the hydrological processes. At the same time it is clear that hydrological studies can be used to demonstrate problems and help sustain and extend wetland sites (Newson, 1992). Across the UK the cutting of peatland drains has almost ceased. However, there are still areas of the UK where peat cutting is actively pursued (e.g., Isle of Skye, Caithness and Sutherland) and in many parts of the world peat is still highly valued for its horticultural and fuel-burning value. The UK government policy now discourages afforestation on land with peat over 1 m deep but further planting and associated drainage is still likely to occur on shallower peats (Andersen *et al*., 2000). However, new drainage schemes should take into account best-practice recommendations that have been incorporated into revisions (1991, 1993) of the *Forest and water guidelines* (Forestry Commission, 1988) and predictive models that are currently being developed (McDonald *et al*., 2003). It is advised that: cross-drains should discharge into vegetated areas and not directly into water courses; drains should be cut with a gradient less than 2° to prevent bed scour; and the spacing of cross-drains should be reduced (Carling *et al*., 2001).

Wetlands are complex systems where multiple processes operate in combination. A significant amount of work towards ecological restoration has taken place in wetland areas but a great deal of this work has been carried out on a pragmatic or even an *ad hoc* basis. This reflects the urgency of the requirement to protect important sites and the frequent shortfalls in available funding. Whilst there is a body of knowledge relating to the hydrological processes of peatlands, too often managers, through time and resource constraints, have been required to act with only a limited understanding of the functioning of their particular site. Often, when ecological restoration is attempted, several interventions are employed at the same time. Restoration work has often been completed with limited prior monitoring, and it has therefore been difficult to sustain scientific assessments for a sufficient time period in order to evaluate success (Carpenter and Lathrop, 1999) or to disentangle the precise effects of particular interventions. Often wetland landscapes have such disparate relaxation times that process–responses are difficult to identify. Burt (1994) stresses the importance of long-term observation of the natural environment as a basis for environmental policies. Many laudable results have been achieved by the hard work and detailed 'on the ground' knowledge of managers such as Mawby (1995) but there remain many sites where restoration has been a hit-and-miss affair, where time and money has been wasted because the hydrological functioning of the system has been poorly understood.

Artificial drainage rarely occurs in isolation; burning, grazing, afforestation, fertilization can all accompany drainage. Thus the effectiveness of any restoration strategy does not rest solely on the restoration technique adopted but on how well integrated the catchment management schemes are and how well we understand the interacting mechanisms. Nonlinear restoration strategies are often needed and much more work is required to examine the hydrological and hydrochemical processes surrounding artificial drainage and peatland restoration.

Our final point concerns the question of 'restoration to what?' The climate today is different from that when many peatlands began to form in the early Holocene. Therefore a peatland restored in today's climate may well develop on an entirely different trajectory than peatlands did a few thousand years ago. When 'restoring' wetlands do we simply want to maintain 'current ecological functions' (Charman, 2002) or do we want to allow wetland ecosystems and their hydrochemistries to develop in new directions? The latter may not be avoidable. Judging the success of peatland restoration must then depend on our perception of peatland functions and process understanding.

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