Corrigendum

Since the publication of this paper an error has been discovered in Box 1. The illustration (figure 4) is an image of land slippage near Cowms Rocks in the Alport valley and not the nearby Cowms rock landslide studied by Johnson and Vaughan (1989). This corrected copy of the paper has removed figure 4 and made minor changes to the text to correct this error. The author would like to apologise for any inconvenience caused by this mistake.
Natural changes in upland landscapes

Martin Evans

Introduction

Upland landscapes in the Britain are often perceived as the last wild lands on an increasingly overcrowded island. Whilst the uplands undoubtedly play an important recreational role as an escape from the urban experience, further enhanced recently by increased access under the Countryside Rights of Way Act, the notion of a wild land is untenable. The common classification of the British uplands as semi-natural ecosystems acknowledges the important role of human activity in creating and maintaining contemporary upland landscapes. In 2001, during the UK outbreak of foot and mouth disease, concern was expressed that the loss of hefted flocks of Herdwick sheep (flocks habituated to a particular upland area) would lead to dramatic reductions in grazing pressure and scrub invasion, causing the loss of the ‘landscapes of Wordsworth’ (BBC News 2001). At the same time the counter view that the ‘scrub invasion’ represented the precursor to a desirable return to natural woodland was also advanced (Holdgate, 2001). Central to this discussion is the fact that the upland landscapes of the Britain represent a delicate balance of the processes of natural ecosystem change and both intended and unintended anthropogenic pressures on the system. Contemporary upland landscapes result from the interaction of land
management and the natural biophysical environment. In this context there are three fundamental reasons why any assessment of these systems should be rooted in the natural ecosystem processes. First, effective management or manipulation of upland landscapes necessarily depends on a clear understanding of the processes of natural change both to ensure the longer term sustainability of management solutions but also to avoid unintended consequences. Second, a clear understanding of rates of natural change is a necessary context to assessment of the significance of anthropogenically forced change. Finally, whilst the uplands represent cultural landscapes so that the definition of desirable end points for land management lies in large part within the spheres of politics, economics and public taste, scientific understanding of the ‘natural’ system can inform the discussion through assessment of local environmental history, and assessment of what landscape states are biophysically achievable through management and conservation. The aim of this chapter is to outline dominant natural trajectories of change in upland systems, and, through a case study of eroding peatlands, to demonstrate that an understanding of both the longer term context of natural change, and shorter term natural responses to disturbance is central to effective management of upland systems.

**Definition and classification of Natural changes**

At Holocene timescales\(^1\) two main modes of landscape change can be identified. Intrinsic changes are a function of the natural evolution of system function with time whilst extrinsic changes result from changing boundary conditions due to environmental change (Table 1). Extrinsic change therefore encompasses the response to anthropogenic forcing as well as to natural environmental change. Under the

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\(^1\) The Holocene spans the last 11500 years or the last 10000 uncalibrated radiocarbon years. Uncalibrated radiocarbon ages diverge from calendar years before the mid-Holocene so that early Holocene radiocarbon ages are 1000-1500 years younger than the true calendar age. Most dates in this chapter are reported as uncalibrated radiocarbon ages honouring the original sources and are identified as years BP. Where calibrated ages are reported they are identified as cal. BP.
heading of natural change this chapter will therefore consider the natural evolution of biophysical systems, responses to natural climate change, and the natural regeneration of disturbed systems.

**Long Term Changes in Upland Landscapes**

The notion of a climax vegetation (Clements 1916) or a soil type (Jenny 1941) in equilibrium with contemporary environmental conditions is deeply ingrained in the literature of environmental science and has significantly influenced thinking on upland landscape change at longer timescales. The uplands of the UK were either glaciated or subjected to intense periglacial conditions during the last glaciation so that the development of contemporary soil and vegetation systems has a clearly defined starting point at the end of the last glacial period. The retreat of the ice sheets left large areas of land mantled in barren glacial till and fluvio-glacial deposits. Work on contemporary glacier forefields has demonstrated that processes of soil formation and re-vegetation progress rapidly to produce a more stable vegetated landscape (Matthews, 1992). However, the composition of climax vegetation (across most of upland Britain with the exception of the highest peaks this means forest cover) evolved over a longer period because of the time required for the migration of tree species from refugia south of the glaciation limit (Birks, 1989) (Figure 1).

The idea of a naturally controlled long term trajectory for the development of upland landscapes is also commonplace in the literature of upland catchments and freshwaters. The classic work by Pearsall (1921) on the development of the Lakes of the English Lake District was based on the premise that progressive weathering and soil development in the Lake catchments during the Holocene drove
development from ‘primitive’ lakes with low pH and low base cation concentrations to ‘evolved’ lakes with higher pH and base cation concentrations and a more diverse and abundant flora. Recent work, however, has cast some doubt on this notion of the unidirectional evolution of upland waters. Engstrom et al. (2000) studied a chronosequence of lakes exposed by glacial recession in Glacier Bay Alaska and demonstrated that over a 10000 year chrono-sequence older lakes are more acid, more dilute and have higher dissolved organic contents (Figure 2). These changes are linked to soil development and terrestrial succession in the lake catchments which promote carbonate leaching, humus accumulation and changes in runoff pathways. Engstrom et al. suggest that this pattern may be typical of many lakes in deglaciated areas of the cool temperate zone. The soil development trends they describe including accumulation of humic material and hard pan development are common to many areas of upland Britain. In areas of high rainfall, high rates of leaching produce podzolic profiles and often eventually produce iron pans. These layers impede drainage such that the climax soil type is a histosol: either a peaty podzol or blanket peatland. In wetter areas of northern and western Britain blanket peat development occurred relatively rapidly after deglaciation in favoured locations. Basal dates from below blanket peats of between 9000 and 7000 BP are common (Tallis, 1991; Bennett et al. 1992; Fossit, 1996). In many areas, however, later dates of onset of peat development particularly clustered around circa 5000 BP are associated with human intervention in the system (Moore 1973; 1975). Mesolithic or Neolithic remains are often associated with basal peats (Caulfield, 1978; O’Connell, 1986; Mills et al. 1994) and Moore suggested that the expansion of peatlands across upland Britain was due to waterlogging connected to forest clearance.

The concept of climax soils and indeed climax vegetation cover is reasonable as long as the boundary conditions of the environmental system (e.g. climate, topography, human intervention) remain constant. Indeed by making this assumption the degree of soil
Development can be a useful relative dating tool in landscapes where alternative dating is not available (Birkeland 1999). However, the processes of soil development and the establishment of climax soil and vegetation cover occur over extended periods whereas, particularly in high energy upland environments, periods of landscape instability (see box 1) can lead to rapid rejuvenation of land surfaces. Harvey et al. (1984) describe soil chronosequences from the uplands of the Howgill fells in northwest England where mature podzols are observed on soils inferred to be stable throughout the Holocene but where much more immature soils are typical of valley floors where lateral instability of the river channel causes erosion and deposition and periodic rejuvenation of soils. Such disturbance of land surfaces also has a significant impact on local vegetation cover. In a dynamic landscape the surface vegetation will represent a mosaic of successional stages. Therefore understanding of vegetation succession in the uplands is important not only to the long-term development of vegetation systems but also to the contemporary patterns and dynamics. Miles et al. (2001) emphasised the importance of vegetation recovery from disturbance or ‘secondary succession’ as a control on the vegetation mosaic of upland Scotland although they note that the pattern of the last 50 years in Scotland has been for such change to be a response to changes in land management.

Perhaps the most fundamental shift in boundary conditions occurs with climate change. The Holocene has seen significant variations in climate with consequent effects on vegetation cover. The direct control of climate on vegetation patterns and bog ecosystem function is the basis for the numerous palaeoecological approaches to reconstructing climate. Some of the most sensitive climate records come from upland peat. Plant macrofossils, peat humification records and testate amoebae from peatlands have all been used to reconstruct Holocene climate (e.g. Barber et al. 1994; Chambers et al. 1997; Woodland et al. 1998; Hughes et al. 2000) and particularly to identify wet and dry shifts in bog surface condition. Chiverell (2001) applied multiple methods to reconstruct hydrological changes from an ombrotrophic bog site in the North York Moors and identified changes to cooler and wetter climate at 1690-1401 cal. BP, 1400-1300
cal. BP, 1280-970 cal. BP, 600-500 cal. BP, 550-330 cal. BP, and 250-150 cal. BP. Macklin and Lewin (2003) compared wet shifts identified from bog records with radiocarbon-dated alluvial records of major flooding and suggested that wet periods around 8000 BP (Hughes et al., 2000) and since 4000 BP are associated with major flood episodes. They also identified, during the last 4000 years, an increase in the sensitivity of fluvial systems to climate change due to significant human modification of the land cover. Patterns of upland landscape disturbance are therefore closely associated with climate change through the climatic control on river flooding in addition to the direct control of vegetation by climate.

In the last 20 years the notion of unidirectional ‘development’ of soils has been increasingly challenged (Johnson et al., 1990; Huggett, 1998). Huggett argues that in parallel with a shift in emphasis in the vegetation literature away from simplistic applications of successional theory, temporal change in soil properties should be regarded as ‘soil evolution’ a term encompassing the possibility of soil development, retrogression, or rejuvenation and the possibility of multiple stable states. Essentially Huggett argues for a rebalancing of the emphasis on immanent processes of landscape development and configurational changes (sensu Simpson, 1963). Some changes in upland systems do occur as a result of the physical, chemical or biological development of stable parts of the landscape driven by natural processes, however, the intensity of the disturbing forces in upland environments both natural and anthropogenic and the sensitivity of upland systems mean that much of the form of the contemporary landscape is inherited as an integration of past impacts. It is more appropriate to regard these upland landscapes as being in dynamic equilibrium and resilient to human impacts; given sufficient time and the removal of the stress, they will revert to a ‘natural state’. Figure 3 is one way to visualise this dynamic equilibrium with natural processes of succession and soil development tending to drive the system to the right of the diagram, and disturbance,
anthropogenic or otherwise, driving the system towards the left. The end points of the spectrum are not fixed but vary according to climatic conditions and the particular history of a given landscape.

Defining the appropriate point on this spectrum for a given upland landscape is the crux of the Lake District Foot and Mouth debate with which this chapter was introduced.

Understanding the dynamic processes that control this equilibrium is central to scientifically based upland management.

**Box 1  Rapid change in upland landscapes: Deep-seated landsliding**

Not all natural change in the uplands is gradual; in numerous locations across the UK rapid change has occurred as a result of deep-seated landsliding. Major landslides have occurred across the uplands of Britain including South Wales, (Bentley and Siddle, 1996) North Yorkshire (Simmons and Cundhill, 1974; Waltham and Forster, 1999), the Lake District (Wilson, 2005) and the south Pennines (Johnson,1980; Johnson and Vaughan, 1983:1989). The south Pennines are one of the largest concentrations of deep seated landslides including the largest inland landslide complex at Alport Castles in the Upper Derwent Valley. These landslides have been well studied and provide a good example of highly localised and high-magnitude natural drivers of upland change. Such landslide events are not stochastic but may be driven by extreme rainfall events, by seismic activity, by fluvial undercutting, or by long-term weathering related changes in rock mass strength (Johnson; 1965, Skempton et al. 1989; Johnson and Vaughan 1989; Wilson, 2007), However, the complexity of interaction of the potential causes of large-scale landsliding is such that although landslide risk can be regionalised on the
basis of susceptible lithologies, prediction of landslide occurrence is problematic. The magnitude of the local effects of deep-seated landsliding on the local environment is extreme. Remodelling of the slope profile through slope failure can lead to subsequent instability so that the initial stochastic impetus for change drives further slope evolution through retrogressive landsliding. Skempton et al. (1998) suggest that the time period required for stabilisation of deep-seated landsliding in the Peak District is up to 8ka. During this period secondary landsliding regrades the slope to a series of successively more stable configurations with the magnitude of hydrological forcing required to trigger further movement increasing progressively. More recent work on the Mam Tor landslide (Arkwright et al., 2003) has however demonstrated that slip rates in the period 1991 – 2002 of 15cm yr\(^{-1}\) are three times the long term average movement rates reported by Skempton et al.(1998). Arkwright et al. demonstrated that movement rates of the landslide correlate with antecedent rainfall and suggest that groundwater levels are the dominant control on slippage. They ascribe recent more rapid movement to climatic wetting after the Medieval Warm Period. The implication is that for the Mam Tor slide which failed 3200-4000 BP whilst there may have been change in the hydrological threshold to movement due to progressive stabilisation the slope system is still highly sensitive to small changes in climate so that climate change is the major control on stability. Dixon and Brook (2007) analysed the probable stability of the Mam Tor slide with respect to predicted climate change and suggested that increased seasonality and increased winter variability of rainfall will increase instability but that reduced overall rainfall may to some degree counteract this effect. Land surface stability in landslide areas is therefore conditioned not just by the initial slippage but also by the ongoing interaction of slope processes with extrinsic change in climate.
A typical south Pennine failure which has been studied in detail (Johnson and Vaughan, 1989) is the Cowms Rock failure. This landslide 1.3 km² is amongst the largest of the south Pennine landslides; it has not been precisely dated but is suggested to be older than 5000 BP (Johnson and Vaughan, 1989). Johnson and Vaughan suggest that the preconditions for failure were established through rock mass creep under periglacial conditions during the Devensian Glacial and they estimate failure to have occurred in the early Holocene triggered by fluvial undercutting of the toe slope. Landsliding on this scale has a dramatic effect on the local landscape. The effects of major failure are likely to include temporary damming of the river and establishment of a new local base level upstream of the landslide deposit. Three phases of landslip impact on the landscape can be identified: 1) the initial instantaneous morphological change associated with failure, 2) transient changes as the landscape adjusts to the new form including vegetation succession and channel adjustments to modified base level (10¹ – 10² years) 3) longer term effects of ongoing slope instability at timescales of 10³ years.

Case study: Natural processes in peatland systems

The themes explored in the first part of this chapter, of intrinsic and extrinsic change, gradual and rapid change, natural responses to disturbance and the dynamic equilibrium of upland landscapes are well exemplified through a case study of upland peatland systems. Upland peatlands cover 8% of the land area of the UK (3.3% in England and Wales and 13% in Scotland) (Tallis et al., 1997; Hamilton et al., 1997; Holden et al., 2006). Most of this area is covered by blanket peatland and Britain and Ireland support 15% of the total world resource of this land cover type (Tallis et al. 1997). The peatlands of the UK, particularly those of England and Wales have been heavily
impacted such that much of the peatland area shows evidence of peat erosion (30-74% of peatland affected by gullying depending on region (Tallis, 1997a; Evans and Warburton, 2007).

Peatlands exist because permanently high water tables limit litter decomposition causing accumulation of partially decomposed organic matter (peat). The local water balance is therefore central to the continued maintenance of peatlands with the ombrotrophic peatlands, typical of the uplands, confined to areas where there is an excess of precipitation over evapotranspiration. In the case of blanket peatlands Lindsay (1998) identifies threshold conditions for their existence of greater than 1000 mm of rainfall, greater than 160 days of rain, and mean July temperature less than 15°C.; conditions which describe much of upland Britain. Excess precipitation is removed either as runoff or through evaporation. Figure 4 is a representation of the water balance of a typical ombrotrophic mire. It illustrates that the major loss of water from storage in deep peat is evaporation. Rates of lateral drainage are very low due to the very low hydraulic conductivity of the catotelm which is typically in the range of $10^{-5} \text{ – } 10^{-8} \text{ m/s}$ (Evans and Warburton 2007) Reported rates of evaporation from ombrotrophic mires are in the range 1.1-3.8 mmd$^{-1}$ (Evans and Warburton 2007). Ombrotrophic mires are characterised by perennially high water tables. For example, in blanket peatlands from northern England, Evans et al. (1999) report water tables in the upper 5 cm of the peat profile 83% of the time. Even during drought periods water table is rarely more than 50 cm below the surface and the storage capacity of upper peat layers is such that after a drought water tables are often restored to near surface conditions within a single storm (Figure 5). The high water tables common to blanket peatlands mean that they are systems highly productive of runoff through the development of saturation overland flow. Runoff is concentrated at the surface or in the upper acrotelm layer of the peat which has higher hydraulic conductivity. Holden and Burt (2003a and b) demonstrated that for a North Pennine
peatland, overland flow and near-surface stormflow account for 96 per cent of runoff generation under storm conditions. Low catotelm hydraulic conductivities mean that there is very little lateral drainage from deep peat to maintain streamflow during low flow periods so that peatland stream systems have characteristically flashy stream hydrographs (e.g. figure 5) with total runoff dominated by stormflow events.

The extensive development of overland flow on peatland surfaces generates significant erosive energy at the peatland surface. The peat surface is bound together by the roots of the vegetation cover, which provides resistance to surface erosion. In circumstances where the vegetation layer is weakened by overgrazing, fire, desiccation, trampling or pollution impacts, physical erosion of the surface may result. The extensive erosion of the peatlands of the UK and Ireland dates largely to the last millennium (Evans and Warburton, 2007) and has been variously ascribed to many of these causes (Shimwell, 1974; Tallis, 1985, 1995, 1997 a and b; Anderson, 1986) and also associated with climate changes during the Little Ice Age causing increased runoff and storminess (Rhodes and Stevenson, 1997). Tallis (1987) describes how, for a particular location in the southern Pennines at Holme Moss a history of accumulating stresses on the peatland surface produced the dramatic erosion characteristic of the contemporary system.

Eroding peatlands are amongst the most actively eroding landscapes in the UK. Sediment loads in eroding catchments range up to 265 t km$^{-2}$ a$^{-1}$ (Evans et al., 2006; Evans and Warburton, 2007). The surface impact of these rates is large because of the low density of peat which means that the volume of material removed is significant. Once bare peat is exposed, rates of erosion are relatively rapid with recorded rates of surface recession of up to 74 mm a$^{-1}$ (Phillips et al., 1981). Erosion of the bare peat is facilitated by preparation of the
peat surface through processes of frost action and desiccation (Francis, 1990; Labadz et al., 1991) such that the rate of sediment production from eroding peat catchments can be modelled effectively from climate data (Yang, 2005).

The spatial patterning of peat erosion produces the dramatic erosional landscapes of degrading peatlands. Figure 6 is a peat land-system model representing the range of characteristic erosional forms and processes found on eroding peatlands.

Catchment vegetation cover is important not only in defining the bare eroded areas but also as a control on delivery of eroded material to the main stream system and its export from the catchment. Evans and Warburton (2005) demonstrated a circa 60% reduction in sediment yield from a north Pennine catchment over a 40-year period and argued that the reduction was linked to re-vegetation of the floors of eroding gullies so that eroded material was stored on gully floors and sediment delivery to the main channel was very low. Evans et al. (2006) compared severely eroding southern Pennine sites with this north Pennine catchment and proposed the conceptual model of peatland sediment flux illustrated in figure 7.

Clearly there is an intimate link between processes of erosion and re-vegetation in degrading peatland systems. Areas such as the North Pennines show evidence of extensive past erosion but also very significant degrees of re-vegetation. In contrast the southern Pennines have larger areas of bare eroding peat, although there is evidence of recent re-vegetation (Evans et al. 2005), figure 8. Traditionally the extensive bare peat of the southern Pennine peatlands has been explained by the heavily impacted nature of the system as documented by Tallis (1995, 1997). However, the differences in degree of re-vegetation could equally be explained not as variations in the rate of erosion but as variations in the degree of re-vegetation. If the higher rates of pollution in the southern Pennines act to suppress re-vegetation then this would cause the accumulation in the landscape of erosional features without any change in the rate of their
generation. A complete understanding of landscapes of degraded peatlands is therefore dependent on a clear scientific understanding of the processes of recovery and regeneration as well as the processes of erosion.

Natural re-vegetation of eroding peatlands is widely reported in the literature (Phillips, 1954; Bowler and Bradshaw, 1985; Large and Hamilton, 1991; Cooper and Loftus, 2001, Evans and Warburton, 2005) but has been relatively little studied. The most comprehensive work on gully re-vegetation had been carried out in the southern Pennines (Crowe, 2007). This demonstrates that two cottongrass species, *Eriophorum angustifolium* and *Eriophorum vaginatum*, are central to the natural reestablishment of vegetation cover in eroding gullies. Evidence from repeat aerial photography and from macrofossil analysis of peat cores from re-vegetated gully floors suggest that the cottongrass pioneer species are succeeded by a more diverse flora including *Sphagnum, Juncus, and Polytrichum*. These observations are consistent with work on the regeneration of peat cutting sites which suggests that *E. vaginatum* cover provides surface stability and enhanced humidity sufficient to support *Sphagnum* establishment and succession to more diverse cover (Rochefort, 2000; Lavoie et al. 2003). The timescale for succession from bare gully floors to a more diverse wet bog flora appears to be of the order of 20-40 years with complete vegetation cover of the gully floors within 5-10 years (Evans et al., 2005; Crowe, 2007). Therefore, where suitable conditions exist, natural re-vegetation rapidly stabilises eroded systems and significantly reduces the rates of sediment loss from the system. Further research is required to establish precisely the conditions necessary for natural re-vegetation to occur. Sufficiently low local slopes to allow deposition of eroded peat as a substrate for cottongrass growth, and a local source for the vegetative spread of cottongrass appear to be essential (Evans et al., 2005; Crowe, 2007). These conditions can occur through the natural development of the gully system, either in locations where gully bank collapse causes local constriction of the gully, or in the later stages of gully
development where gully width has increased to the extent that the channel wanders on the gully floors, promoting lateral deposition (Evans and Warburton 2007). Crowe (2007) has demonstrated that slightly different successional sequences are associated with particular geomorphic contexts (gully forms). This analysis of the interactions of the erosional system and the natural vegetation succession is a good example of the importance of understanding natural processes as an input to restoration practice. Restoration approaches such as gully blocking, appropriately applied, have the potential artificially to create conditions suitable for re-vegetation and therefore have the potential to initiate relatively rapid and sustainable re-vegetation of eroded sites. Because upland geomorphic systems are naturally dynamic it is of particular importance that landscape restoration works with the natural processes of system recovery in order to create restored landscapes which are stable under contemporary conditions. The importance of understanding natural ecosystem recovery as a guide to what a suitable reference point for a restored landscape might be is considered further in the next section.

**Natural processes and upland management**

Restoration of adverse human impacts on upland systems requires an understanding of natural trajectories of recovery. Restoration efforts that work to promote natural recovery mechanisms are more likely to be sustainable in the long term and more likely to restore ecosystem structure and function.
Fundamental to landscape restoration efforts is a clear understanding of the aims of the restoration project and in particular the desired end point. Practical restoration of degraded uplands requires a reference condition of the system both as a guide to suitable restoration practice and as a benchmark against which to reference the success of varying restoration approaches. At its simplest the desired endpoint might be a return of the system to notionally ‘pristine’ conditions that existed prior to significant human impact. Such a notion is incorporated into the recent European Water Framework Directive with the requirement that freshwaters have ‘good’ ecological status relative to the reference conditions by 2016 (Kallis and Butler, 2001). Palaeoecological techniques have been widely applied to define pristine conditions because of the difficulty of identifying unimpacted sites in highly modified environments (Bennion et al., 2004). However, in semi-natural environments such as those typical of most of the UK uplands the notion of a pre-impact condition is problematic. Similarly the ‘pristine’ state of upland landscapes may not be an appropriate end point for restoration efforts. This is clearly exemplified through consideration of attempts to re-vegetate the eroded peatlands of the southern Pennines over the last 30 years (Tallis and Yalden, 1983; Anderson, et al., 1997; Evans et al. 2005). Dobson et al. (1997) distinguish between restoration and reclamation, where restoration involves a return to a pre-impact landscape condition with both ecosystem structure and function restored, whereas reclamation means an instrumentalist approach to ecosystem manipulation that promotes restoration of ecosystem function. In blanket peat systems restoration to a pristine state is often perceived to require a return to typical peat domes with a diverse sphagnum peat vegetation, perennially high water tables and active peat growth and carbon fixation. However, other states of the bog ecosystem occur naturally with phases of sedge domination common, and even tree growth occurring during warm periods (Chambers, 1997). The historical pressures on many British upland mires require that restoration efforts consider alternative stable states of upland mires. In systems where there has been significant erosion, particularly where gullying is extensive and water table is drawn down by
gully drainage, it is questionable whether a return to pristine sphagnum dominated conditions is a feasible medium-term target for restoration. The approach taken to re-vegetation in the southern Pennines has focussed on the establishment of an initial grass cover which stabilises the surface in order to allow colonisation of moorland species. The initial re-vegetation can be regarded as reclamation in that it restores some measure of ecosystem function in terms of limiting erosion and promoting carbon sequestration. The timescale for complete restoration of these systems, particularly where extensive gully erosion has occurred, is close to the time taken for initial peat formation (in excess of 5000 years) since a return to near pristine conditions would entail infilling of gullies with new peat growth. Work on natural re-vegetation of eroding systems (e.g. Clement, 2005; Crowe, 2007) suggests a more appropriate medium-term target for restoration is achieving the structure of naturally re-vegetating gully systems that support a diverse wet bog vegetation in gully bottoms. Complete re-vegetation to this condition is achievable on timescales of the order of 20-40 years (Evans and Warburton, 2005; Clement, 2005; Crowe, 2007)

Conclusions

Upland landscapes of the UK are relatively high-energy systems subject to significant natural disturbance through mass movement, severe temperatures and flashy runoff regimes. They are also semi-natural systems so that ecosystems are additionally stressed by pollution, grazing, fire (both managed and wildfire), and recreational impacts. Upland landscapes are naturally a mosaic of landcovers affected by these various impacts at a range of timescales and the intensity of change in these patches is increased through human
impacts. Short term change associated with this dynamic mosaic are superimposed on long term trajectories of change in upland systems
driven by processes of plant migration, climate change and soil development.
The primary conservation aim in upland systems should be limiting the stresses on the system. However, in a dynamic semi-natural
system conservation should also focus on managing patch dynamics. Understanding of natural processes is essential to informed
management. In addition the character of natural system recovery should be an important area of study since restoration to conditions
akin to natural recovery is an achievable aim within the timescales and budget constraints of practical conservation, whereas a return to
pristine conditions may represent only a hypothetical goal. The management of upland landscapes in the 21st Century during a period of
potential climatic instability provides a significant challenge. Natural system responses to change may provide the best guide to what
constitutes ‘achievable restoration’ of degraded systems.
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Figures

**Fig 1** The postglacial spread of key upland tree taxa across Britain (After Birks, 1989)
**Fig 2** Chemical evolution of lake basins in Glacier Bay (after Engstrom *et al.*, 2000)
**Fig 3** Dynamic equilibrium of upland landscapes
**Fig 4** Representation of the water budget of ombrotrophic peatlands. The size of the arrows indicates the relative magnitude of the fluxes.
**Fig 5** Water table and runoff data from the Trout Beck catchment of the Moor House National Nature reserve in the Northern Pennines, UK. Data from a storm of 6/7/95. Note that during the initial rainfall event, water table rises but that runoff response is delayed until the second rainfall pulse when water table is close to the surface. (After Evans *et al.*, 1999)
**Fig 6** The peatland-system representing the potential range and connectivity of geomorphological features found on a typical eroding mire. (After Evans and Warburton 2007)
**Fig 7** A Conceptual model of sediment delivery in eroding peatlands (After Evans *et al.*, 2006)
**Fig 8** The blanket peat erosion mosaic. Examples of eroding and re-vegetating blanket peat found within a 1 km radius in the southern Pennines. a) Active gully erosion. b) partial re-vegetation of a broad mineral gully floor by *Eriophorum vaginatum*. c) recent re-vegetation of a peat floored gully by *Eriophorum angustifolium*. d) Complete re-vegetation with a diverse wet bog flora including *Sphagnum*.

Tables

**Table 1** Classification of the major agents of Holocene change in upland landscapes
<table>
<thead>
<tr>
<th>Upland Environmental Change</th>
<th>Type</th>
<th>Timescale</th>
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<td>Soil development</td>
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</tr>
<tr>
<td>Evolution of fresh waters</td>
<td>Intrinsic</td>
<td>$10^3$ years</td>
</tr>
<tr>
<td>Vegetation succession</td>
<td>Intrinsic</td>
<td>$10^3$ years</td>
</tr>
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<td>Post glacial</td>
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<td>$10^1 - 10^2$ years</td>
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<td>Climate change</td>
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<td>Soil instability</td>
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Table 1