physically, there will be an increased threat to overall biodiversity of shallow lakes.

Studies on experimental mesocosms support this scenario. Experiments have shown increased release of phosphorus from sediments (McKee *et al.,* 2003) and changes in animal communities as temperature increases. There have been short-term increases in plant biomass and an extension of plant growing season and a tendency for exotics like *Lagarosiphon major* to take over the communities (McKee *et al.*, 2002). At a 3°C rise above ambient temperature in 1999-2001, fish-kills (of sticklebacks, (*Gasterosteus aculeatus*)) were few but a 4°C rise in 2006 meant extensive fish kills as even ambient temperatures soared to 27°C (H. Feuchtmayr, D. Atkinson, I. Harvey, R.J. Moran, B. Moss, pers. comm.) At 27°C, male stickleback care behaviour is impaired and many fewer young are raised even if the fish survive deoxygenation in the warm water (K. Hopkins, A.B. Gill and B. Moss, pers. comm.) With fish deaths, zooplankton populations may increase for a time but filamentous algae or lemnids often take over and shade out submerged plant communities. The fish usually used in experiments are sticklebacks but carp are more thermally tolerant and higher temperatures will not be lethal for them. An increase in carp dominance seems very likely in shallow lakes. Restrictions on carp stocking and movement, indeed carp reduction programmes, are likely to be needed to ensure continuation of reasonable biodiversity, unless conditions become so severe that any fish community is welcome rather than none at all.

Prognosis

The prognosis for shallow lake systems in the face of warming is poor. In the short to medium terms there will be at least a severe deterioration of lowland freshwaters. In the longer term, there could be improvements but these will depend on a very rigorous application of the spirit of the Water Framework Directive and substantially greater international attempts to limit absolutely our carbon emissions. In turn this seems unlikely without major revisions to economic philosophies and a rejection of growth economies. More likely, and possibly more certain than the ecological predictions we are able to make, will be a desperate, and ultimately failing, attempt to maintain the current economic system of the west through protectionism, military action and repeal of environmental legislation. It may be that we will be condemned to a hot and concrete landscape dominated by water hyacinths, prickly pears, common carp, gulls, foxes, rats and rabbits. That is avoidable, but problems that are fundamental require solutions that are profound. There can be no business as usual if any sort of equability is to persist.

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Climate change and aquatic ecosystems in the UK: science, policy and management

Issues affecting upland water quality: Climate change, acidity, nitrogen and water colour

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Introduction

Upland landscapes, and the rivers and lakes that drain them, have been greatly affected by human activity. Deforestation, the eradication of larger predators and the spread of domestic livestock have altered the landscape over millennia. In the last two centuries, upland management has expanded and intensified through increased livestock densities, liming, land drainage, heathland burning (to support grouse shooting) and afforestation with exotic conifer species. Simultaneously, upland ecosystems have been exposed to elevated levels of long-range air pollutants, including sulphur (S) compounds emitted during fossil fuel burning, and nitrogen (N) compounds originating from fossil fuel burning and agriculture. Deposition of these compounds as 'acid rain' has led to the widespread acidification of soils and waters in many upland areas, and to eutrophication of naturally N-limited terrestrial and aquatic ecosystems. While efforts are now being made to limit the impact of land-management on many upland ecosystems and international agreements such as the Gothenburg Protocol have greatly reduced rates of S (and to a lesser extent N) deposition, there is increasing concern that anthropogenic climate change may either reduce the degree of ecosystem recovery following the alleviation of these other pressures, or at worst cause further ecosystem degradation. Here, we review a number of the potential impacts of climate change on upland water quality, and attempt to set these in the wider context of other anthropogenic pressures. The paper is structured around four elements of the biogeochemical cycle in upland ecosystems, namely acidity, sulphur, nitrogen, and finally dissolved organic carbon (DOC), the major determinant of water colour.

Projected climate changes in the UK uplands

Projections made for the UK Climate Impacts Programme (UKCIP, Hulme *et al.*, 2002) suggest that by 2100 UK temperatures will have increased by 1 to 5°C. Warming is expected to be greater in the southeast than in the northwest. An average UK increase of 0.4 to 0.9°C has been recorded since 1914, leading to a 30-day increase in thermal growing season since 1900. Summer precipitation is projected to decrease by 50% by 2100, whereas winter precipitation is expected to increase by 30% over the same period, but with less falling as snow. There is evidence that this trend towards greater seasonality in precipitation has already begun, with a greater proportion of rainfall having occurred during more intense winter events over the last 30 years relative to the preceding 240 years (Hulme *et al.*, 2002). The trend towards wetter winters is most pronounced in northern and western Britain (where most of the upland area is located), and this appears to be reflected in an increasing incidence of high river flows in these areas (Cannell *et al.*, 1999; Werritty, 2002).

To an extent, increasing winter 'storminess' can be associated with a shift towards more positive values of the North Atlantic Oscillation (NAO) Index, a large scale measure of climate which strongly influences UK climate. Positive values are associated with wetter, windier and milder winter conditions, and negative values with colder and dryer winters. Positive values of the NAO Index are also associated with increased deposition of marine salts onto coastal catchments, due to entrainment of marine aerosols into the atmosphere from breaking waves during periods of sustained high winds (Evans *et al.*, 2001). The NAO Index itself is also expected to increase significantly over the next century, becoming almost continuously positive by 2100 (Hulme *et al.*, 2002).

Based on current projections, some key predictions for climate change which have the potential to impact on upland water quality may be identified as:
(i) higher temperatures;
(ii) increased incidence and severity of summer drought;
(iii) wetter conditions during winter;
(iv) increased frequency and magnitude of winter high flows;
(v) reduced snowpack development; and
(vi) increased occurrence of winter sea-salt deposition events.

These predicted changes form the basis of the assessment undertaken in the following sections.

Surface water acidity and climate

Anthropogenic acidification has had a detrimental impact on the chemistry and biology of upland lakes and streams in the UK and elsewhere. The main driver of acidification to date has been S deposition, which peaked in the 1960s and 70s, but has since declined dramatically (by 50% since 1986 alone) in response to a succession of emissions control protocols agreed under the framework of the UNECE Convention on Long-Range Transboundary Air Pollution. The 22 lakes and streams comprising the UK Acid Waters Monitoring Network, which has been in operation since 1988, now show clear chemical recovery from acidification in response to the decline in acid deposition (Davies *et al.*, 2005), although biological recovery remains harder to detect (Monteith *et al.*, 2005). During the monitoring period, however, substantial inter-year variability in water chemistry was found to be linked to climatic fluctuations. These included fluctuations in sea-salt deposition, with higher loadings associated with a peak in the NAO Index having caused short-term acidification in the early 1990s (Evans *et al.*, 2001). Nitrate and sulphate concentrations were also found to be strongly influenced by climatic factors, which are discussed below. The stability of invertebrate communities in acid-sensitive streams has also been found to be sensitive to variations in the NAO Index (Bradley and Ormerod, 2001). The potential for climate change to impact on recovery from acidification has been widely noted (e.g. Skjelkvåle *et al.*, 2003; Wright *et al.*, 2005).

On a shorter-term event basis, climate and acidification are linked during 'acid episodes'. These are transient periods of elevated acidity that primarily affect streams (e.g. Figure 6.1) (Wright, 2007) and, to a lesser degree, smaller lakes. Acidic episodes are crucially important in terms of stream biota, as it is the severity of chemical extremes, rather than average conditions, which typically determines biological damage such as fish kills (e.g. Baker *et al.*, 1990; Hindar *et al.*, 1994) and loss of invertebrate species (e.g. Kowalik and Ormerod, 2006). Episodes can be caused by a variety of different drivers, including high rainfall events, snowmelt, sea-salt deposition events, sulphate flushes after droughts, and nitrate flushes after freezing events (e.g. Davies *et al.*, 1992; Evans *et al.*, 2007; Wright, 2007). A consistent feature of all these drivers, however, is that they are associated with some form of climatic extreme. Although these climatic events can be considered natural, their impact on stream acidity is invariably exacerbated by pre-existing acidification of the system. With climatic extremes such as summer drought and winter rainfall predicted to become more pronounced in future, some impact on episodic acidity appears likely.

A number of studies have attempted to predict the impact of climate change on recovery from acidification. Wright *et al.* (2006) used the MAGIC (Model of Acidification of Groundwater in Catchments) model to examine the sensitivity of future mean acidity to a range of projected climatic changes at 14 sites in Europe and North America. Sensitivity was highly variable both among different drivers and between sites, with climatic effects on organic acid leaching and N retention identified as those requiring the greatest focus. A modelling study at the Afon Gwy monitoring catchment at Plynlimon, Wales (Evans, 2005) also suggested that increased organic acid leaching could have a major impact on soil and river acidity. In the same study, simulated increases in sea-salt deposition were predicted to have a limited impact on mean stream acidity, although the expected increase in the frequency and severity of sea-salt deposition events could be of greater importance on an episodic timescale. At the same site, an attempt to predict the severity of future high-flow driven acid episodes (Evans *et al.*, 2007) suggests that these are declining in severity as S deposition is reduced (Figure 6.2). Increases in the magnitude of high flow events appear unlikely to counteract more than a small proportion of this improvement: it is therefore expected that high flow-driven episodes may become more common during winter in response to climate change, but not necessarily more severe. Snowmelt events, which are a major cause of acid episodes in areas subject to large annual snowpack accumulations (e.g. Laudon *et al.*, 2004) are less important in most of the UK, where snow accumulations are smaller and shorter-lived. Projected decreases in winter snowfall will further reduce the influence of snowmelt on runoff chemistry.

Sulphur and climate

The majority of UK upland catchments are characterised by mineral and organo-mineral soils, developed since the last glaciation. These soils have little capacity to store incoming S either through sulphate adsorption or sulphate reduction. As a consequence, sulphate export from mineral soil-dominated catchments is typically equal to or (if geological S sources are present in soils or bedrock) greater than the deposition input (Figure 6.3a). With little short-term S storage in the catchment, the major control on surface water sulphate export is thus the S input, and this in turn is primarily controlled by S emissions. Climatic influences are therefore likely to be restricted to controls on the location and timing of S deposition; previous studies have shown increased S deposition in (predominantly western) upland catchments during periods of easterly airflow, and reduced deposition in periods of westerly airflow. A greater prevalence of westerly conditions during winter, associated with the projected increase in the North Atlantic Oscillation, might therefore be expected to lead to a slight reduction in the S loading to acid-sensitive western catchments.

In wetland areas, such as the blanket peats that cover large areas of the British uplands, anaerobic conditions in water-saturated soils permit much greater S storage, via reduction to organic S compounds and inorganic sulphides. Sulphate export in surface waters may therefore be significantly lower than the deposition input (Figure 6.3b). Long-term burial of reduced S in accumulating peat can represent a substantial buffer against the anthropogenic acidification of peats, as illustrated by peat soil solution data from the Moor House Environmental Change Network research site in the North Pennines (Figure 6.4), at which non-marine sulphate concentrations in soil solution in a typical year are close to zero. However, the storage of reduced S in peatlands is highly dependent on the maintenance of water-saturated, anaerobic conditions. Severe droughts, such as those observed during 1995 and 2003 at Moor House, lower the peat water table, allowing oxygen to enter the soil and re-oxidise reduced S compounds to sulphate (Adamson *et al.*, 2001). This can generate extreme levels of acidity which, if flushed from the soil, can cause major acid episodes in runoff, together with mobilisation of toxic metals (Tipping *et al.*, 2003). Although a long-term assessment of sulphate-driven acid episodes at Birkenes indicates that episode severity has decreased since the 1970s as the

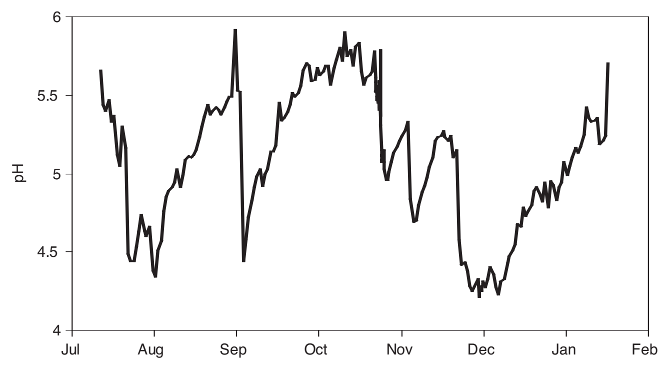

Figure 6.1 Examples of acidic episodes affecting stream chemistry at Birkenes, Norway (Wright, 2007)

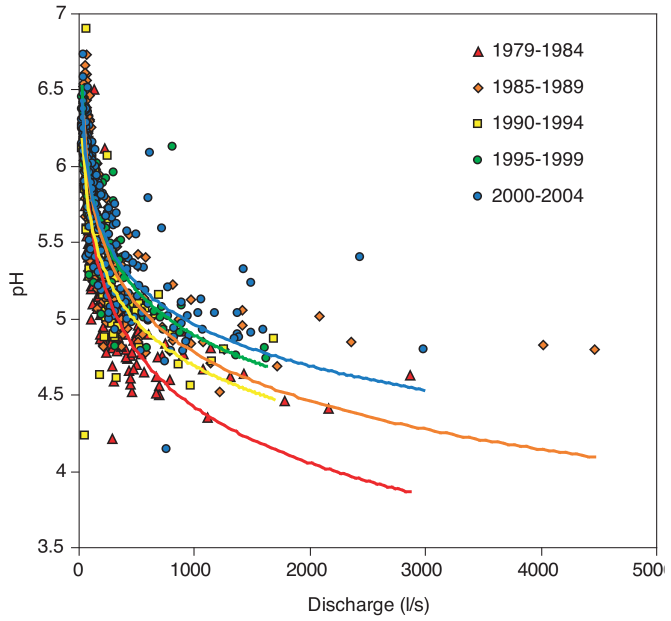

Figure 6.2 Long-term decreases in severity of discharge-driven acid episodes at the Afon Gwy catchment (after Evans** et al.*, ***2007)

rate of S input has declined (Wright, 2007), a modelling study for an Ontario wetland catchment by Aherne *et al.*, (2006) suggested that repeated drought events, even at current levels of drought frequency, would be sufficient to severely retard recovery from acidification. Repeated droughts, together with reduced S deposition, will lead to the gradual depletion of peat S stores (Tipping *et al.*, 2003), but in more polluted regions this process may take many decades. Peat catchments containing large stores of anthropogenic S must therefore be considered highly sensitive to a projected increase in the frequency and severity of summer droughts, which could lead to the destabilisation of these stores and consequently to an increased incidence of biologically damaging post-drought stream acidification events.

Nitrogen and climate

UK semi-natural terrestrial ecosystems are predominantly N limited, and any additional N from deposition tends to be taken up by the biota, and strongly retained within the soil. As a result, nitrate export to surface waters is typically much lower than N deposition (Figure 6.3). However, 'N saturation' theory suggests that sustained supply of N in excess of biological demand will ultimately lead to elevated nitrate leaching to surface waters, with associated (and potentially severe) acidification and eutrophication impacts. Plot and catchment-scale experimental N addition studies (e.g. Moldan *et al.*, 2005) confirm that N saturation can occur, and although temporal trends

are difficult to detect over the timescale of available monitoring data, levels of surface water nitrate are clearly elevated in regions of high N deposition (e.g. Curtis *et al.*, 2005).

N in terrestrial ecosystems is tightly cycled. Any climatic event which disrupts biological cycling is likely to result in nitrate leaching. In the UK this has been most clearly observed following soil freezing events, which typically occur when the winter NAO Index is negative (Monteith *et al.*, 2000; Davies *et al.*, 2005). Such events are likely to become less frequent under future climate change. However, nitrate flushes also occur after droughts (e.g. Adamson *et al.*, 1998), and these events may increase in frequency. Extreme rain events may also transport nitrate directly to surface waters, where water bypasses biological sinks within the soil, e.g. as overland flow. Again, these events may be more common in future. Of greatest overall concern, however, is the long-term stability of the soil organic matter pool, as this contains most of the N accumulated over more than a century of elevated deposition. The Norwegian CLIMEX study, in which a small catchment was exposed to elevated temperatures and CO_2, showed a marked increase in N mineralisation from the soil, leading to elevated nitrate leaching and effectively turning the catchment from an N sink to an N source (Wright and Jenkins, 2001; Figure 6.5). Such a response to climate warming across the UK uplands would clearly have grave consequences for the acidification and eutrophication of upland waters.

Dissolved organic carbon (DOC), water colour and climate

DOC represents a large part of the carbon export of many upland catchments and is the major source of water colour. It is a significant component of the upland carbon balance, contributes significant costs to water treatment and impacts on aquatic ecosystems by altering light regime, nutrient transport, acidity, and metal transport and toxicity. Since the late 1980s, surface water DOC concentrations have approximately doubled across a large proportion of the UK (Freeman *et al.*, 2001; Worrall *et al.*, 2004a; Evans *et al.*, 2005). The reasons for these increases have not been fully resolved, but there is growing evidence that a significant, and perhaps primary, driver has been the reduction in S deposition and subsequent recovery from acidification, which has increased the solubility of organic matter (Evans *et al.*, 2006). As an integral part of the upland carbon cycle, however, there is little question that climate-related factors also impact on DOC export. Rising temperatures, by increasing organic matter decomposition rates, may lead to increased DOC production (e.g. Freeman *et al.*, 2001; Evans *et al.*, 2006). However, field evidence for a warming contribution to DOC increases is inconclusive, and laboratory data suggest that observed warming of around 0.6°C since the late 1980s can only account for a small part of the observed rise in DOC. Elevated atmospheric CO_2 itself has also been shown to increase DOC by accelerating plant production (Freeman *et al.*, 2004), although again this probably accounts for a relatively small proportion of the DOC increase to date. Finally, hydrological changes may impact on the magnitude and/or timing of DOC release. DOC peaks often occur during high flows, as more runoff water is routed through shallow organic soil horizons. Larger or more frequent high flows might therefore increase DOC release, although this effect is likely to be greatest for summer high flows, whereas more frequent high flows in winter (as predicted in climate forecasts) may be expected to have less impact, since DOC concentrations are generally low at this time. Droughts appear to have a strong effect on DOC release, generally decreasing DOC concentrations during the drought period itself, with increases observed thereafter (Hughes *et al.*, 1997; Clark *et al.*, 2005; Figure 6.6). These drought-rewetting cycles have been identified in some long-term DOC and water colour records, with a step-change in DOC observed after the droughts of 1976 and 1995 interpreted as possible drivers of long term increases (Watts *et al.*, 2001; Worrall *et al.*, 2004b). Overall, it is possible that climatic changes have contributed to the DOC increases observed to date, and probable that they will contribute to further DOC changes in the future. However it must be emphasised that other factors (S deposition, land management and possibly N deposition) are believed to have had as much, or more, influence on DOC trends during the last 30 years, and cannot be ignored in predicting future changes.

Conclusions

There is little doubt that climate change will have significant effects on water quality in upland lakes and streams. These effects are complex and are very unlikely to be uniformly detrimental, or indeed uniformly beneficial, to aquatic chemical and biological status. Furthermore, climatic effects on water quality must not be considered in isolation from the other natural and anthropogenic drivers of change in upland systems, including the effects of atmospheric pollutants, and local factors linked to land management. Perhaps most importantly, it must be recognised that many UK upland waters are currently undergoing rapid change as they recover from the effects of many decades of acid deposition; current chemical status cannot therefore be considered a natural baseline from which

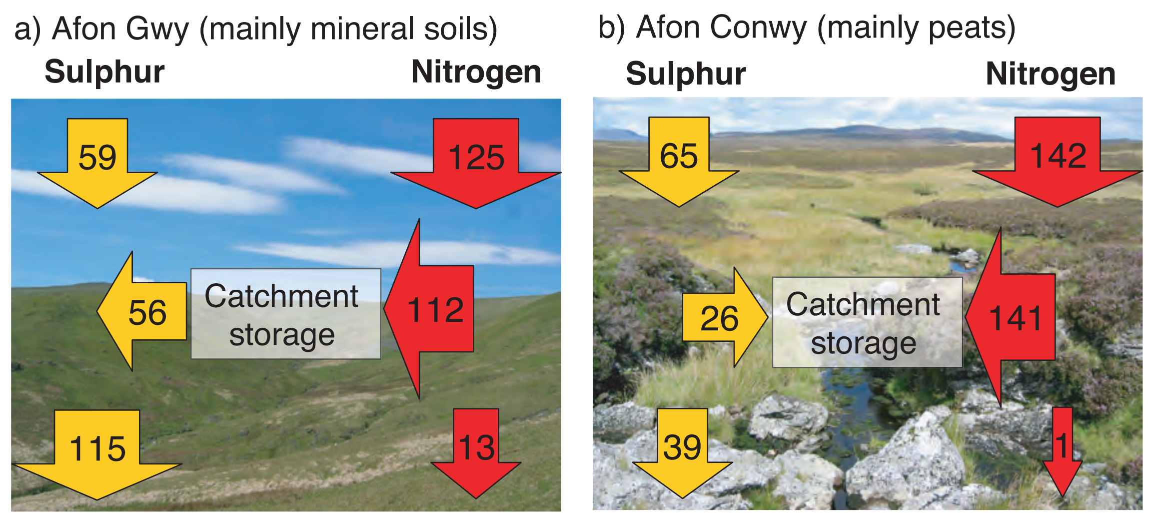

Figure 6.3 Sulphur and nitrogen fluxes in upland catchments.
All fluxes are in mmol m^{-2} yr^{-1}. Deposition inputs are based on 5 km CEH Edinburgh modelled deposition for 1998-2000, stream outputs on the average concentrations of SO_4 and NO_3 in samples collected over the same period, and average stream discharge. Fluxes to and from storage simply represent the net difference between measured inputs and outputs (i.e. assume no other significant inputs or outputs)

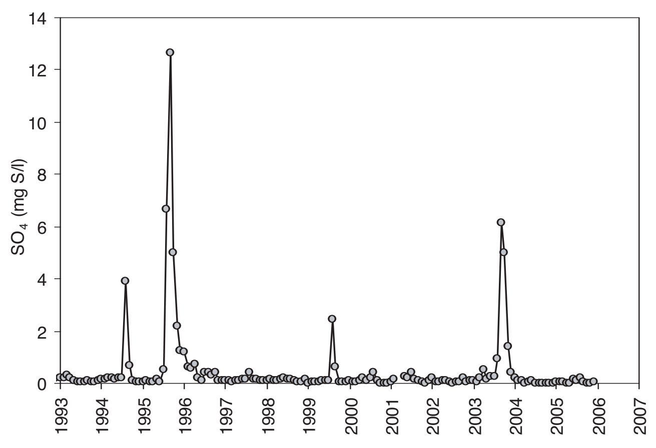

Figure 6.4 Sulphate concentrations in peat soil solution at Moor House, Northern England. Near-zero concentrations during most of the record result from sulphate reduction in the peat, peaks result from re-oxidation of this sulphur during droughts. Data were provided by the Environmental Change Network

to assess the future impacts of climate change. In addition, upland systems remain at long-term risk from the effects of continuing elevated N deposition, and in some cases from land-management practices that may not be compatible with long-term ecosystem sustainability.

To a large extent, future sensitivity of surface waters to detrimental climate change impacts may occur due to interactions with other anthropogenic pressures. Perhaps most importantly, increases in temperature and in climatic extremes such as droughts, may reduce the capacity of upland catchments to store, and thereby mitigate, the detrimental impacts of pollutants including nitrogen, sulphur, and also heavy metals and organic pollutants. Climate change-induced release of pollutant stores accumulated over a century or more could have severe implications for the acidity, toxicity and nutrient status of upland waters. The relative contribution of climate change and acid deposition to increases in DOC and water colour remains unresolved; since DOC increase due to climate change may be considered as ecosystem degradation, and DOC increase due to recovery from acidification as ecosystem recovery, it is clearly essential that this question is resolved in order to appropriately manage and adapt to continuing change. Finally, this study has focused primarily on the effects of climate change on water quality, but there is also evidence that climate change may impact directly on aquatic biota; recent work by Durance and Ormerod (2007) indicates that rising water temperatures, in the absence of a change in water quality, could cause a loss of invertebrate species diversity. Again, however, the direct effects of warming in that study appear to be influenced by interactions with other drivers, with the greatest warming-induced species losses occurring in streams not already degraded by acidification.

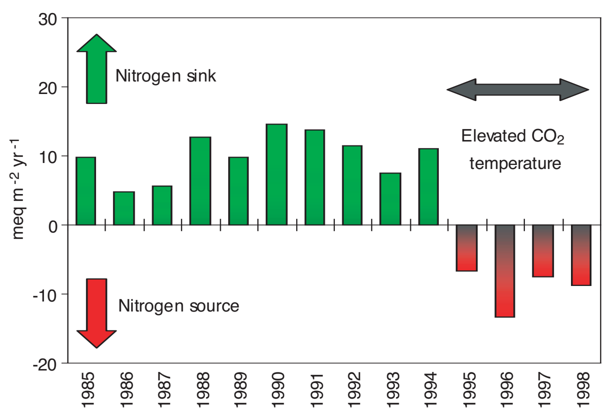

Figure 6.5 Nitrogen balance of the 'CLIMEX' experimental catchment in southern Norway, showing a shift from an N sink to an N source in response to elevated CO₂ and temperature during 1995-1998 (after Wright and Jenkins, 2001)

Overall, it is clear that effective management and protection of our upland waters requires a holistic approach, in which the potential impacts of climate change are addressed in conjunction with, and based on an adequate understanding of, the other anthopogenic pressures to which these ecosystems have been, and continue to be, subjected.

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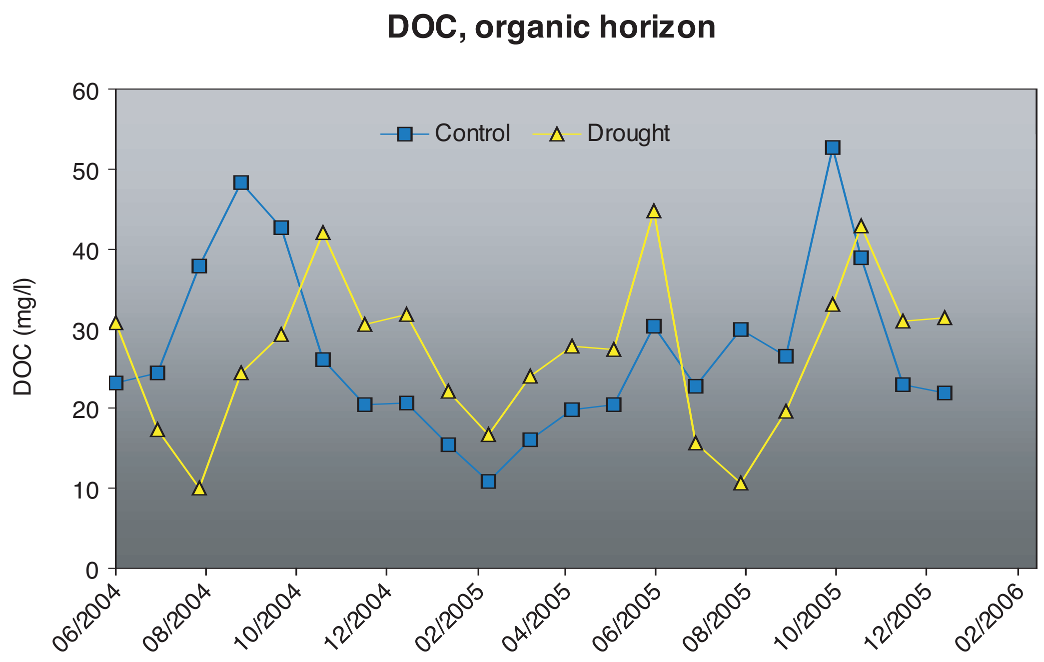

Figure 6.6 DOC response to repeated experimental summer drought at the 'Climoor' plot-scale experiment, Wales, showing a reduction in DOC concentration during the drought period, and higher concentrations following re-wetting (Centre for Ecology and Hydrology, unpublished data)

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