THE FUTURE OF BRITAIN'S UPLAND WATERS

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H.A. Binney (Editors)

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Contents

Front cover photo: The Afon Hirant study site in Upland Wales. Catherine Duigan, Countryside Council for Wales.

The acid rain-acid waters debate in the UK – a brief history

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We now know that most upland waters in the UK are severely acidified. But until the 1980s we had little knowledge that acidification had indeed been occurring for over 100 years. The observations and prescient advice of Gorham in the 1950s, that the influence of acid rain "might be best sought in the high tarns, since they are most dependent upon rain for their nutrients..........and it is hoped that these matters will receive some attention in the future" (Gorham, 1958), went unheeded. The UK government of the time was pre-occupied with urban smogs and the need to reduce air pollution, especially smoke emissions, from cities. Indeed some of the remediation measures for urban air pollution introduced in the 1956 Clean Air Acts, including the building of new power stations equipped with tall chimneys, may have unintentionally aggravated the acid deposition problem by dispersing emissions further from their source. The alarm was eventually raised not in the UK but in Scandinavia when Svante Odén and the Swedish Government claimed that fishery loss in Sweden was caused by long-distance transported acidic compounds from industrial countries upwind, including the UK (Odén, 1968). Despite further papers in the early 1970s (e.g. Jensen and Snekvik, 1972; Almer *et al.*, 1974) providing supporting evidence for Odén's claim, the Central Electricity Generating Board (CEGB), UK's nationalised power utility at the time, was not convinced that there was a cause-effect relationship between acid rain and fish decline. In 1978 they convened a workshop, held in the Cally Hotel, Gatehouse-of-Fleet, to review the problem and the report from the meeting (Wood, 1978) forms the first substantive publication on the ecological effects of acid rain in the UK.

Shortly afterwards in a move partly designed to put pressure on the UK, Dick Wright and Arne Henriksen from the Norwegian Institute for Water Research (NIVA) in Oslo teamed up with Ron Harriman and Brian Morrison from the Freshwater Fisheries Laboratory in Pitlochry to assess the acidification status of lakes and streams in the Galloway region of south-west Scotland. Wright *et al.* (1980) subsequently concluded that "the Galloway area thus appears to be yet another region in which acidification of freshwaters has occurred because of deposition of strong acids from the atmosphere".

The CEGB, however, was not persuaded, and preferred explanations associated with land-use change and natural processes, alternatives that were bolstered by Rosenqvist's forceful promotion of land-use change and soil acidification as an explanation for fish decline in Norway (Rosenqvist, 1978a,b). In the UK the land-use argument was supported by the observation that afforested catchments had more acidic water and poorer fisheries compared with moorland

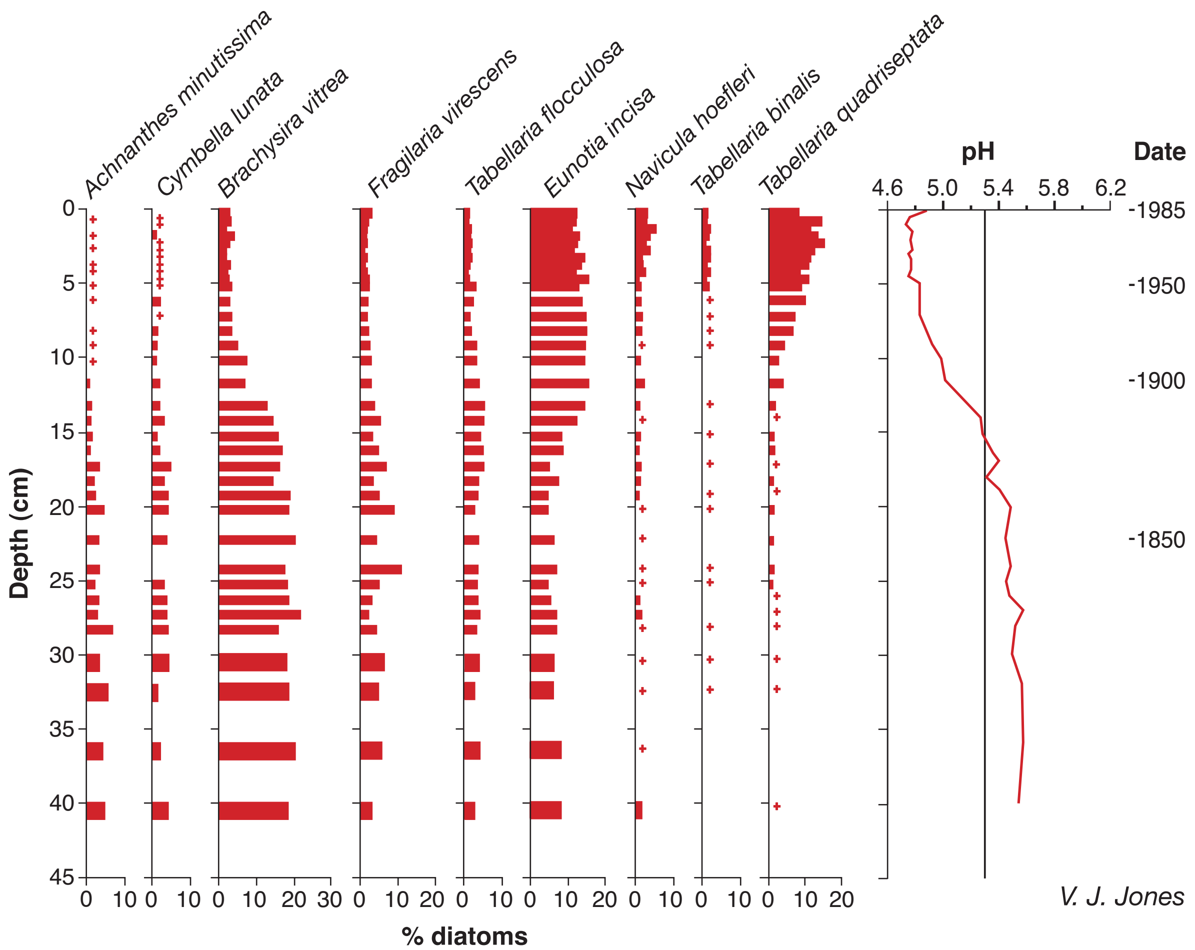


Figure 1: Diatom diagram for The Round Loch of Glenhead.

catchments (Harriman and Morrison, 1982; Stoner and Gee, 1985). The "natural process" case, that lake acidification was a long term (post-glacial) natural process, was put forward by Pennington (1984) supported by colleagues at the time in the NERC Institute of Freshwater Ecology, Ambleside.

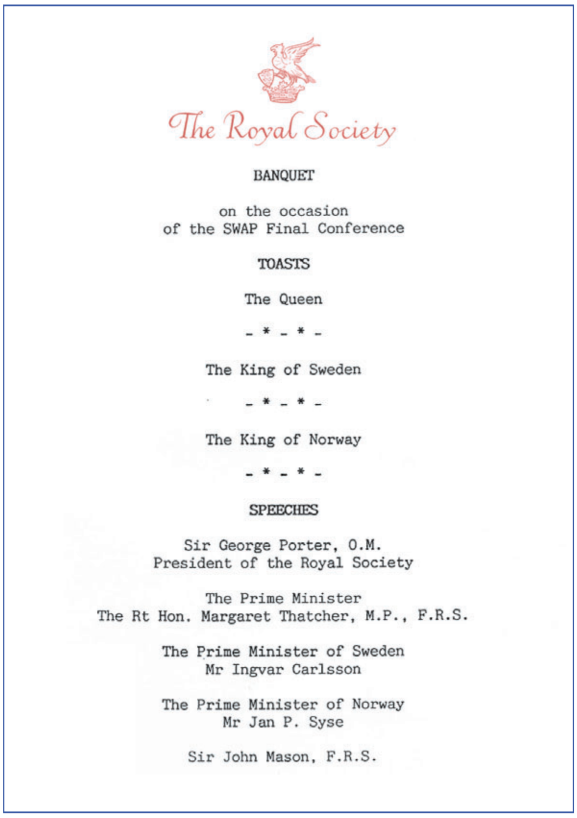
In the late 1970s the CEGB became aware of evidence for recent acidification coming from an examination of diatom assemblage changes in lake sediments in Norway by Frode Berge (Davis and Berge, 1980). In order to evaluate the Norwegian diatom results the CEGB funded Rick Battarbee and Roger Flower from University College London (UCL) to use a similar approach focussing on lakes in Galloway. Starting with multiple hypotheses for the cause of low-pH lakes in Galloway, we used diatom-pH transfer functions of dated sediment cores from two sites (the Round Loch of Glenhead and Loch Grannoch) to reconstruct the past pH of the lakes. Both lakes were very acidic (pH 4.5-4.7) and had a similar overall chemistry. They differed in that one, Loch Grannoch, had a recently (from 1963) afforested catchment and the other had a moorland catchment. We argued simply that if afforestation was the cause of low pH then Loch Grannoch should show evidence for increasing acidity after afforestation, and that if both sites had acidified naturally over the post-glacial period (approximately 11,500 years) then neither should show evidence of rapid change over the last 150 years. The results disproved both hypotheses as the Round Loch of Glenhead (Figure 1) showed evidence of rapid recent acidification (from about 1850) and the acidification at Loch Grannoch, the afforested site, began before, not after, afforestation (Flower and Battarbee, 1983). These results, and similar results from other sites, such as Loch Enoch, strongly supported the acid deposition hypothesis (e.g. Battarbee *et al.*, 1985).

Despite these data and other evidence, the CEGB maintained their rejection of acid deposition as the major cause of the problem and were unprepared to introduce sulphur dioxide removal technology into power stations. They promoted instead the view that surface water acidification, however caused, could be combatted locally and more cost effectively by liming. To demonstrate the approach in the UK they launched a major liming experiment at Loch Fleet in Galloway (Howells and Dalziel, 1992).

By the mid-1980s diplomatic relations between Britain and Norway became severely strained, with at one point the Norwegians reportedly threatening not to send to the UK the traditional gift of a Christmas tree for Trafalgar Square. Concerned that the CEGB and the Department of Energy, who were then leading the UK government's research programme into acid waters, might be on the wrong side of the argument and were damaging the UK's image overseas, the Department of Environment assumed responsibility for acid rain research and in 1984 launched a major initiative, headed by Bob Wilson, to assess the impact of acid deposition on all aspects of the UK natural environment including surface waters. Shortly afterwards the CEGB, headed by Sir Walter Marshall, together with the National Coal Board, headed by Sir Ian McGregor, aware that their in-house research and research sponsored by them was perceived to lack objectivity, provided a £5 million fund jointly to the Royal Society and to the Norwegian and Swedish national science academies to design a collaborative project involving Scandinavian and British scientists to resolve the issue once and for all. This became the Surface Water Acidification Programme (SWAP) launched in 1985, directed by Sir John Mason and chaired by Sir Richard Southwood, with a mission to report in 1990.

Somewhat to the surprise of the scientists involved in SWAP, the Thatcher government in 1987 announced acceptance of the Scandinavian position and made a commitment to reduce S emissions before the SWAP programme had been half-way completed. Later it became apparent that the change of heart was related to advice to Mrs Thatcher from Sir Walter Marshall who, on a tour of acidified lakes and forests in Norway and Sweden in 1987, had been persuaded by the Scandinavian evidence and returned convinced that emissions should be reduced, if only as a gesture. In addition Mrs Thatcher was in a mood to heed the advice, realising that resolving this problem would improve her green credentials before the forthcoming UK general election. However, so as not to undermine the SWAP programme completely, the UK government emphasised the need for the programme to continue both to provide a stronger scientific underpinning of the decisions already made and to provide a better scientific basis for future decisions on the size of reductions in sulphur dioxide that would ultimately need to be made.

SWAP duly reported with its findings at two Royal Society Discussion meetings in 1989 and 1990 respectively, the first devoted to the palaeolimnological evidence (Battarbee *et al.*, 1990) and the second covering the remaining programme (Mason, 1990). A dinner at the Royal Society in 1990 (Figure 2) attended by the three prime ministers heralded the formal end to hostilities.


The Royal Society

BANQUET

on the occasion
of the SWAP Final Conference

TOASTS

The Queen

- * - * -

The King of Sweden

- * - * -

The King of Norway

- * - * -

SPEECHES

Sir George Porter, O.M.
President of the Royal Society

The Prime Minister
The Rt Hon. Margaret Thatcher, M.P., F.R.S.

The Prime Minister of Sweden
Mr Ingvar Carlsson

The Prime Minister of Norway
Mr Jan P. Syse

Sir John Mason, F.R.S.

Figure 2: The 1990 SWAP banquet at the Royal Society.

By 1990 as a result of both the DoE national programme, the SWAP project, the continuing in-house work of the CEGB and a range of projects funded by the Natural Environment Research Council (NERC), it was possible to conclude that:

(i) all streams and lakes sensitive to acidification (i.e. those with low natural alkalinity) and receiving significant acid deposition had been acidified;
(ii) there was a dose-response relationship between S deposition and acidification;
(iii) high altitude sites were more at risk due to their thin soils and to enhanced deposition through the seeder-feeder mechanism (cf. Fowler *et al*., 1988);
(iv) streams were especially vulnerable to acidic episodes;
(v) afforestation itself was not a major cause of acidification, but forests were more effective at scavenging pollutant aerosols than moorland vegetation; and
(vi) acidification caused whole ecosystem changes, not just a decrease in fish populations.

The UK acceptance of the cause – effect relationship between acid deposition and surface water acidification in 1987 had several consequences for research on upland waters.

The first was the need to establish a monitoring network of streams and lakes across the UK to assess the responses of acidified waters to emission reductions. This led to the setting up of the UK Acid Waters Monitoring Network in 1988 (Patrick *et al*., 1991) consisting of 22 lake and stream sites throughout the UK comparing sites in areas of high and low acid deposition and, in some regions, comparing sites with moorland and afforested catchments. The network, now in its seventeenth year, was designed to monitor water chemistry, diatoms, macroinvertebrates, aquatic macrophytes and fish on a seasonal, annual or longer basis as appropriate. Results have been published every five years both in report form (Patrick *et al*., 1995; Monteith and Evans, 2000) and in the international peer-reviewed literature (e.g. Patrick *et al*., 1996, Monteith and Evans, 2005).

The second consequence was the acceptance by the UK of the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP) and its adoption of the "critical loads" approach (Nilsson and Grennfelt, 1988) to assess the extent and seriousness of acidification in all member states. For the UK this required the first ever systematic sampling of freshwater chemistry across the country on a grid square by grid square basis (Kreiser *et al*., 1995) to enable critical load maps to be made of the regions most sensitive to acidification. After comparison with maps of present-day acid deposition (taken as 1986-1988), the first critical load exceedance maps for freshwaters were generated by the DoE's Critical Loads Advisory Group (CLAG) in 1995 (CLAG Freshwaters, 1995) using both the steady state water chemistry (SSWC) model (Henriksen *et al*., 1992) and the diatom model (Battarbee *et al*., 1996).

A central purpose of the critical loads approach is its use as a policy tool, enabling exceedance maps to be generated not just for present conditions but also for the future under different scenarios of emission reductions. The first exceedance maps of this kind published in 1995 were for a comparison between the 1986-1988 sulphur deposition values and future scenarios based on a 70% reduction by 2005 and an 80% reduction by 2010 (CLAG Freshwaters, 1995). In the last 10 years there have been many updates of these maps using different critical load models and different critical values for acid neutralising capacity (ANC). The most recent maps are shown, courtesy of Chris Curtis, in Figure 3. They are based on the First-order Acidity Balance (FAB) model (Posch *et al*., 1997), a methodology that takes into account all the major processes controlling S and N export to surface waters. It is now the preferred critical load model for national and European mapping purposes. The critical ANC value used in the model has also been revised since the first maps were produced. In 1995, 0 μeq l^{-1} ANC was used (CLAG Freshwaters, 1995) as the value that in theory allowed for a 50% probability of undamaged brown trout populations being present at a site (Lien *et al*., 1992). Following extended discussion this value has now been revised upwards on the basis of improved knowledge to 20 μeq l^{-1} to afford better protection to fish populations and provide a more realistic target for the ecological restoration of upland waters based on their pre-acidification status (Curtis and Simpson, 2004).

The third major consequence of the UK decision to reduce S and N emissions in line with the UNECE CLRTAP was the further development of process driven dynamic models of acidification needed to assess time-scales of recovery based on different emission reduction scenarios. The model most favoured for this purpose is the MAGIC model (Cosby *et al*., 1985), developed for use in the UK by Paul Whitehead and Alan Jenkins of the then Institute of Hydrology, Wallingford. It has been used both to hindcast water chemistry, including comparisons with diatom-inferred pH reconstruction (Jenkins *et al*., 1990; Battarbee *et al*., 2005) and to forecast future water chemistry under different scenarios. In this context it has been applied both to streams and lakes, and to single sites and regions (Evans *et al*., 2001) and it has been used to assess impacts of land-use change (Cosby *et al*., 1990) as well as changes in S and N deposition (Evans *et al*., 2001).

The early emphasis on surface water acidification was on the effects of S deposition. Only in the mid-1990s (e.g. INDITE, 1994) was it realised that N deposition might also be important, potentially having an ecological impact on surface waters through both acidifying (Curtis *et al*., 2005) and enriching (Maberly *et al*., 2002) processes. Its acidifying role is probably the more important in upland waters. In many areas of high N deposition in the UK, principally the Pennines, North Wales, the Cumbrian Lake District and Galloway, nitrate will soon replace sulphate as the dominant acid anion (Curtis *et al*., 2005; this volume). In some regions attainment of critical load targets requires reductions of N as well as S; both steady state modelling using FAB (Curtis *et al*., 2005) and dynamic modelling using MAGIC (Curtis and Simpson, 2004) indicate threats of continued acidification if and when catchment soils become saturated with N. Much current research has been directed to this latter issue, essentially to understand the behaviour of deposited N in soils and the potential for N loss from soils to surface waters (Curtis *et al*., 2005).

Upland waters are contaminated not only by acid deposition but also by long range transported toxic substances, both metals and organic compounds. Research in the UK on these

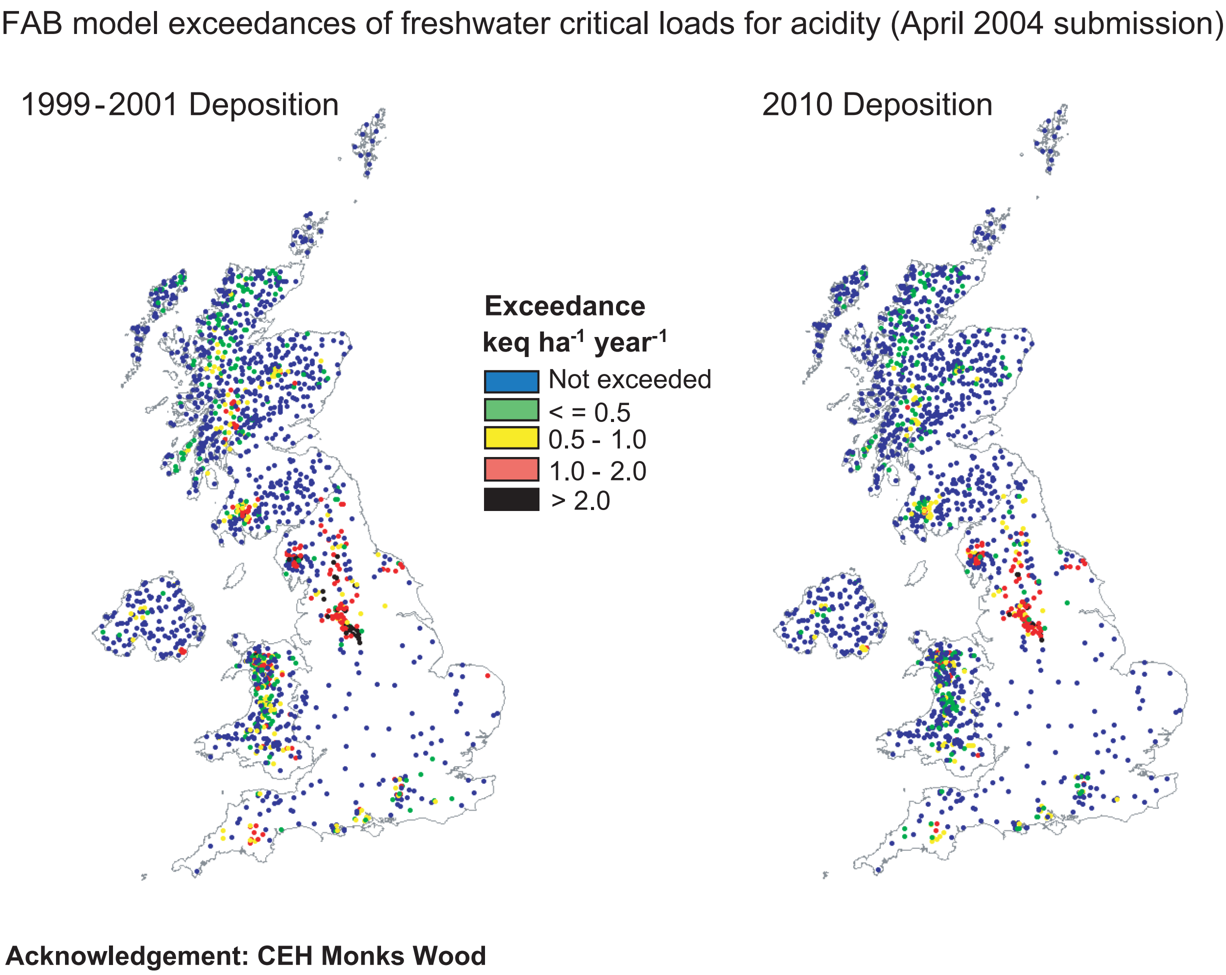


Figure 3: UK Critical load/exceedance maps - latest data submission to CCE for April 2004 with mixed ANC 20/0.

substances has not been as extensive as for acidity and there is considerable uncertainty about their space-time distribution and their physiological and ecological impact on aquatic biota. Studies of lake sediments (Rippey, 1990; Rose *et al*., 2001; Yang *et al*., 2002) clearly indicate that lead, mercury and other metals from anthropogenic sources have been gradually accumulating in upland areas since the nineteenth century, and brown trout populations in Lochnagar surveyed during the EU-MOLAR project show high levels of mercury contamination (Rosseland *et al*., 1997). A programme of trace metal monitoring, including mercury, in bulk deposition, lake water, aquatic macrophytes, aquatic invertebrates, zooplankton and catchment plants is being maintained in Lochnagar, and the EU-Eurolimpacs project will, in part, assess future threats to lake and stream ecosystems from mercury and other metals related to future global warming (www.eurolimpacs.ucl.ac.uk).

Results from the recently published 15 year report of the Acid Waters Monitoring Network (Monteith, this volume; Monteith & Evans, 2005) show that some recovery is now taking place at most sites throughout the country, following patterns in other countries (Skjelkvåle *et al*., 2005). A key question is whether current protocols and directives when fully implemented will lead to a more complete recovery to the "good ecological status" required by the EU Water Framework Directive. However, inspection of the data from the monitoring network show that changes are taking place in the chemistry and biology of upland waters that are not necessarily only driven by the reduction in acid deposition. For example, surface waters in afforested catchments behave differently from those in moorlands and dissolved organic carbon (DOC) concentrations are increasing at all sites in areas of both high and low acid deposition (Monteith and Evans, 2000). The reason for these increases is not known, although there are many hypotheses including global warming and recovery from acidification (Evans and Monteith, this volume; Evans *et al*., 2005). Clearly more attention needs to be given to the definition of "good ecological status" and its attainability in the context both of the adequacy of remediation measures and the influence of confounding factors, including land-use and climate change.

Science questions for today

Despite delays and difficulties in the 1980s the threat of sulphur deposition to upland waters in the UK and more widely in Europe has been successfully identified and to a large extent has been effectively managed by national governments. However, there are additional issues that require continued research and action. The key questions are:

- What will be the future trends in air pollutants, land-use and climate change that separately and in combination control upland water quality and aquatic biodiversity?
- Specifically, is there an increasing threat from N deposition, what is the relative importance of different sources of N and what processes lead to the release of N from catchments to

surface waters?

- What is the distribution and behaviour of toxic metal and organic substances in upland catchments, and what are their physiological and ecological impacts on aquatic biota?
- Why is DOC increasing in streams and lakes throughout the uplands?
- What is the impact of different forms of moorland land-use and management on nutrient and sediment loading to upland waters?
- What will be the impact of future climate change on upland waters, both directly and indirectly?
- How will changes in these factors influence recovery from acidification and the ecological targets set for upland waters?

The future health of upland waters in the UK depends not only on a continuing research programme to understand the impact of current and future ecological threats but also on a parallel review of policies needed to ensure the effective management of upland waters (see Murlis *et al*., this volume).

The following papers presented at a one day meeting on the future of Britain's upland waters at UCL review the current status of research in the ecology of upland waters in the UK. They stress the importance of upland waters (Duigan), threats from future trends in chemical climate (Fowler *et al.*) and land-use (Emmett *et al.*; Murlis *et al.*) and they report the results of recent research on the chemistry and biology of upland waters in the UK. These include evidence for recovery from acidification (Monteith), the role of N compounds (Curtis *et al.*), trends in DOC (Evans and Monteith) and the effects of heavy metal contamination (Tipping *et al.*). Alan Jenkins looks to the future based on MAGIC modelling, David Viner stresses the importance of understanding uncertainty in climate models, and Dick Wright provides a Norwegian perspective. The final discussion draws together the issues that need addressing by policy makers, managers and users of upland waters to ensure a more sustainable use of upland aquatic ecosystems in the future.

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Why do we care about upland waters?

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Introduction

The upland waters of Great Britain consist mainly of clear or brown water lakes and pools, and headwater streams. However, blanket bog and other water dependent habitats also merit inclusion in any consideration of the conservation importance of upland waters. In addition, mid-altitude and lowland water bodies may be strongly influenced by oligotrophic water from upland sources. All these intergrading habitats support key biodiversity resources.

Landscapes

Upland water bodies are mainly based on hard ancient rocks so their waters are naturally acid and nutrient poor. This geological substrate is overlain with glacial sediment and a range of soil types, including peat. Soils and waters are both further influenced by inputs from the atmosphere e.g. sea salt, nitrogen and sulphur. Together these environmental influences produce a high diversity of landforms which are of intrinsic value (e.g. for research and education, Figure 1) and often define the character of the landscape.

The designation of a National Park provides a means of recognising the environmental value of distinct landscapes. There are two main objectives arising from designation:

- to conserve and enhance the natural beauty, wildlife and cultural heritage of the National Parks; and
- to promote opportunities for the public understanding and enjoyment of the special qualities of the Parks.

For example, in the Lake District National Park, sixteen lakes, with a largely upland character, are said to be arranged like "the spokes of a wheel" to provide the landscape framework. Other organisations, such as the National Trust, also appreciate the landscape and biodiversity importance of lakes within their properties.

Figure 1: The Afon Hirant, a famous study site in upland Wales. (Photo - CCW).

Biodiversity Conservation

The upland hydromorphological and chemical environment supports a diversity of ecosystems making them important in terms of biodiversity conservation. In addition, dynamic processes and natural succession create a shifting range of habitat, often well represented in the British uplands. These processes can produce a stratigraphic record of the local environment, which adds to the conservation value (Ratcliffe, 1977). Upland waters are part of the national biodiversity resource. National freshwater classification schemes allow the recognition of ecologically distinct types (e.g. Holmes *et al*., 1999; Palmer, 1992; Palmer *et al*., 1992). Representatives of these types may qualify as Sites of Special Scientific Interest (SSSI) and this may lead to higher-level national conservation designations such as National Nature Reserves.

The media often report on the environmental condition of global biodiversity "hot spots" but upland waters should be considered equally important biodiversity "cold spots". High altitude isolated waters may have a naturally limited diversity of macro-flora and fauna; some may be naturally fishless. In some cases, unusual predator-prey relationships may develop. For example, in lochs in Wester Ross (Scotland) newts are the top predators in the food chain; in contrast to other local lochs where brown trout are the dominant predators (Ian Sime, SNH, pers. comm.). If fish are stocked to the newt lochs their unusual ecology will be irreversibly altered. The guidelines for the selection of biological SSSIs recognise this natural biodiversity limitation for high altitude/alpine waters (Nature Conservancy Council, 1989).

Jones *et al*. (2003a) carried out a very interesting analysis of the relationship between altitude and aquatic plant diversity. They used plant species lists from over 300 lakes in Cumbria. Species were assigned to attribute groups (i.e. species sharing similar morphological and life history traits). With altitude, they reported a decline in species richness, additive to the effect of area; a decline in attribute groups but the number of species per group increased; and high altitude species had stress tolerant traits.

Therefore, any conservation evaluation of upland waters should not discriminate against them on the basis of low biodiversity, since this is a natural characteristic of these systems.

International Conservation Legislation

Moving on from these academic considerations, it should be acknowledged that the UK Government has statutory obligations to protect specific upland habitats and species. Some of these responsibilities have come with the implementation of EU Directives:

- 1979: Directive on the Conservation of Wild Birds – *the Birds Directive*;
- 1992: Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora - *the Habitats Directive*.

Sites designated as Special Protection Areas (SPAs) and Special Areas of Conservation (SACs) respectively under these two Directives will form the Natura 2000 series extending across Europe.

For example, the red-throated diver (*Gavia stellata*) is protected by the Birds Directive. It has a northern holarctic distribution with north-west Scotland acting as a regional stronghold especially in Shetland, Orkney, the Western Isles, Sutherland and Wester Ross (Stroud *et al.*, 2001). The UK holds 30% of the breeding population in the EU. The breeding habitat is small water bodies within areas of open moorland. In Britain, the SPAs for this species have a high degree of naturalness, comprising blanket bog and wet/dry heath, interspersed with oligotrophic pools, lochans and lochs (Stroud *et al.*, 2001).

Under the Habitats Directive three major UK upland habitat types were included in Annex One, which requires the designation of SACs:

- oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea uniflorae* and/or of the *Isoëto-Nanojuncetea* (Figure 2);
- natural dystrophic lakes and ponds (Figure 3); and
- blanket bog.

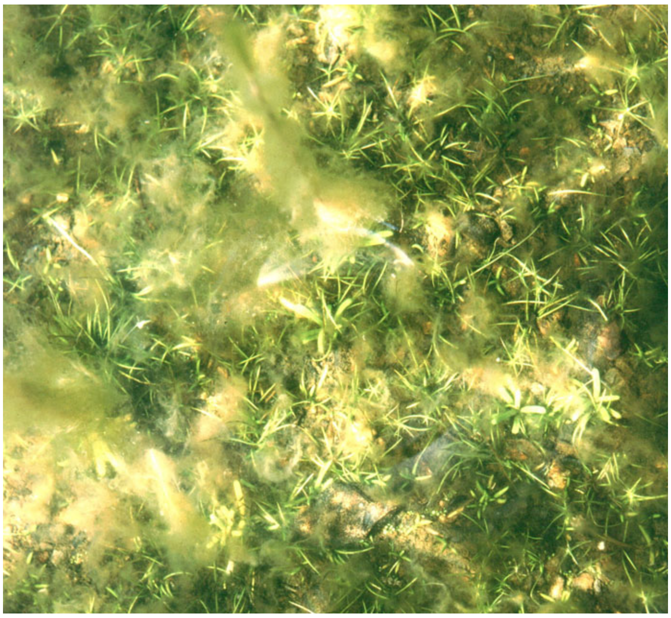

Figure 2: Typical *Littorella* dominated plant assemblage in an upland lake (Photo - CCW).

Three types of Alpine river were also listed in the Directive but they are not thought to occur in Britain. However, the UK SACs designated for water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation may include headwater streams. In addition, a number of freshwater species dependent on upland waters at least during part of their lifecycles, were also included in this Directive, and a selection of these are introduced below.

Oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea uniflorae* and/or of the *Isoëto-Nanojuncetea* are characterised by amphibious short perennial vegetation, with shoreweed *Littorella uniflora* being considered the defining component (Figure 2), in association with *Lobelia dortmanna*, *Potamogeton polygonifolius*, *Isoetes lacustris, Juncus bulbosus*, *Eleocharis acicularis*, *Myriophyllum alterniflorum* and *Sparganium angustifolium*. Most of these species are common components of the aquatic flora of standing waters in the mountainous regions of the British north-west.

Dystrophic lakes and ponds occur in association with blanket bogs. These water bodies are very acidic and poor in plant nutrients. Their water has a high humic acid content and is usually stained dark brown through exposure to peat. *Sphagnum* species are typically dominant and lesser bladderwort *Utricularia minor* is often found (Figure 3). The pools are naturally species-poor and a littoral zone is often absent. More detailed descriptions of these lake habitats and their supporting SACs in the UK can be found at http://www.jncc.gov.uk/Publications/JNCC312/default.htm. Blanket bogs are discussed below.

Figure 3: Small dystrophic peatland pool dominated by *Sphagnum* (Photo - CCW).

The processes that have come with the recent implementation of the Water Framework Directive are being advanced. At this stage a UK water body typology is being finalised. In the case of lakes, this typology exercise will produce a lake classification based on catchment geology, depth and size; some types appear to be associated with the uplands. The next step will be to describe reference conditions (i.e. high ecological status) for each type and it is hoped that some upland reference sites will be identified. In the longer term there will be a Government need to report on the environmental condition of both national and internationally important conservation sites based on a reliable monitoring methodology.

Biodiversity Action

In 1992 the UK Government signed the Convention on Biological Diversity at the Earth Summit in Rio de Janeiro. Subsequently Habitat Actions Plans (HAPs) and Species Action Plans (SAPs) were developed as a demonstration of commitment to this international conservation effort (see http://www.ukbap.org.uk/). This section will examine a number of upland associated habitats and species identified for biodiversity action.

At an early stage mesotrophic lakes (i.e. those in the middle of the trophic range) were identified as suitable subjects for a HAP. It was recognised that they are largely confined to the margins of upland areas in the north and west and they provide a good example of sites influenced by upland oligotrophic water supplies. Typically, mesotrophic lakes have nutrient levels of 0.3-0.65 mg N l^{-1} and 0.01-0.03 mg P l^{-1}. These lakes potentially have the highest macrophyte diversity of any lake type. Relative to other lake types, they contain a higher proportion of nationally scarce and rare aquatic plants. Fish communities in mesotrophic lakes are a mixture of coarse and salmonid species, but today there are few truly natural assemblages due to introduced species. A number of rare fish are associated with these lakes, for example vendace (*Coregonus albula*) in Bassenthwaite and gwyniad (*Coregonus lavaretus*) in Llyn Tegid.

The HAP for blanket bog covers the most extensive peatland habitat in UK (Jones *et al*., 2003b). This habitat is highly dependent on an abundant rainfall being widespread above 250m and with annual precipitation in excess of 1200mm. Its vegetation is characterised by having one or more species of *Sphagnum* prominent, and in combination with a number of ericoid species. Approximately 10-15% of the global resource of this habitat occurs in Britain and Ireland, with circa 1.48M ha estimated to occur in the UK alone (Jones *et al*., 2003b).

The freshwater pearl mussel (*Margaritifera margaritifera*) is the subject of a SAP and it has been fished for its pearls since pre-Roman times (Skinner *et al*., 2003). It is protected by the Wildlife and Countryside Act, the Habitats Directive and the Bern Convention. It is found in the Arctic and temperate regions of Western Europe where its range is declining sharply, including a 95-100% decline in known populations in central and southern Europe. It has an extremely complex lifecycle, including a larval phase that attaches to the gills of juvenile salmonids. It is known to prefer oligotrophic conditions with pH $\leq$ 7.5 and relatively low conductivity. These environmental conditions are consistent with British upland waters. Not surprisingly its UK stronghold is in Scotland, which also represents up to half of the world's remaining populations with active recruitment. However, at the current rate of population extinction these globally strategic Scottish populations may only persist for 25 years (Skinner *et al*., 2003).

The floating water plantain (*Luronium natans*) provides an example of a plant species covered by a SAP. It is also protected by the Wildlife and Countryside Act, Bern Convention and the Habitats Directive. This complex plant has a series of different reproductive strategies and metapopulations highly dependent on oligotrophic upland lakes (Kay *et al*., 1999; Lansdown and Wade, 2003). It is endemic to western and central Europe. In Wales, it has been recorded from 48 sites and 77% of these historic populations persist. It has also spread into canals on the Welsh borders and isozyme patterns indicate that the canal populations have originated from a native lake population connected to the canals by feeder streams (Kay *et al*., 1999).

Freshwater Fish

Maitland and Lyle (1991) carried out a review of the conservation status of freshwater fish in the British Isles. Eight species were identified as being under significant threat, including Arctic charr (*Salvelinus alpinus*) (Figure 4). This species has a holoarctic distribution, often found in large deep oligotrophic lakes in glaciated basins. Many separate populations exist in north-west Scotland (and Ireland) but there has been a steady loss of stocks further south. Charr is often found above 100m and it has been introduced to waters above 500m in Wales.

Figure 4: Male Arctic Charr (Photo - CCW).

Upland waters are also an important habitat for more common species of fish, such as brown trout (*Salmo trutta*) and Atlantic salmon (*Salmo salar*). This is illustrated by studies of the Cairngorm salmonid populations (Campbell, 1970; Gardiner and Mackay, 2002). Brown trout are ubiquitous throughout this area where they are generally found up to an altitude of 450m. A self-sustaining trout population was found in Dubh Lochan at Beinn a'Bhuird, 843m above sea level which may be a British altitude record for this species. Within this mountain range, Atlantic salmon spawn at altitudes of 450-500m, but limited spawning may take place at altitudes of 550m.

Recently the Environment Agency published a Trout and Grayling Fisheries Strategy (Environment Agency, 2003), stating that wild trout stocks will be given greater protection as part of Wild Fisheries Protection Zones. Within these zones stocking will not receive consent to avoid contamination of wild fisheries, to conserve genetically distinct populations and to avoid predation/competition risks in important nursery or spawning areas for trout and/or salmon. This demonstrates that the Agency appreciates the conservation importance of isolated fish stocks and it is probably safe to predict that upland waters will be a dominant feature of these protection zones.

Economic and intellectual asset

Fresh water provides a range of 'environmental services' to the general public. It is a raw material for food and drinks, and a water supply for domestic, industrial and agricultural purposes. It is used for hydropower generation and supports socio-economically important fish stocks and the aquaculture business. Holidays with a water focus are valuable components of the tourism and recreation industry. Fresh waters are of intrinsic value as an educational asset and the science of freshwater ecology has accrued a special academic legacy from the early studies of upland waters in Britain. In particular, when Professor Noel Hynes took up his post at the University of Liverpool in the early 1950s, he realised that the relatively limited invertebrate fauna of British streams made them ideal candidates for studying community structure. He chose the Afon Hirnant (Figure 1), a tributary of the River Dee in north Wales, as a study site because it was easily accessible at a range of altitudes, including the headwaters. His paper on "The invertebrate fauna of a Welsh Mountain Stream" became a citation classic (Hynes, 1961). Many of the insights into invertebrate community composition and structure derived from this study were subsequently incorporated into his textbook, "The Ecology of Running Waters" (Hynes, 1970).

Finally, lakes, waterfalls and rivers have an innate intrinsic appeal to the public and they have been a source of inspiration for writers, poets and artists, as well as scientists! It is easy to sympathise with the feelings and concerns of William Condry (1966) when he visited the remote mountain lakes of the Rhinog in Wales:
"*Gloywlyn, Morynion, Perfeddau, Cwm Hosan, Hywel, Ybi, Du and Dulyn: all these lakes lie close under the backbone of the Rhinog. Long may they remain, unexploited, tranquil and remote, for the spirit of man needs such retreats.*"

Figure 5: Llyn Cwmhosan in the Rhinogs, Wales. (Photo - CCW).

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The future chemical climate of the UK uplands

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Introduction

The interaction between the uplands and the atmosphere is at the heart of many current environmental issues in which uplands have been shown to be at particular risk. It is important to briefly outline the underlying physical processes which lead to the observed effects before describing the chemical climate of the uplands, current trends and provide an indication of the likely chemical climate of the UK uplands in the future.

Uplands provide an obstacle to airflow over the landscape, increasing the drag of terrestrial surfaces on the wind, forcing air to rise and cool as it ascends topography. The physical drag of mountains on the movement of wind results in increased turbulence and much larger wind speeds in the uplands. Generalising the effect is complicated by the three dimensional structure of the topography and its interaction with airflow. However, the tops of hills in the UK at elevations in the range 600m to 800m experience average winds a factor of between 2 and 4 times larger than lowlands in the same region (Chandler and Gregory, 1976). As turbulence is responsible for the vertical exchange of gases and particles throughout the atmosphere, terrestrial surfaces in the uplands are more effectively coupled to the atmosphere than lowland surfaces.

A consequence for exposure of the uplands to pollutants is that the chemical species which are deposited rapidly, due either to their chemical reactivity or the particle size, are deposited at much larger rates in the uplands. The uplands are therefore hot-spots for deposition of many pollutants, even at locations remote from major sources.

Wet deposition

An important effect of upland areas on physical climate is the orographic effect on precipitation. As air is forced to rise, it cools and relative humidity increases. With increasingly moist air a point is reached in mildly supersaturated air where water droplets form. The droplets form on aerosols through a process known as activation and, as air continues to ascend, the droplets grow with further condensation, increasing their liquid water content. This process leads to the formation of hill or orographic cloud in the uplands of northern Europe, and the washout of this low-level cloud by precipitation falling from higher-level cloud by seeder-feeder scavenging is the primary cause of the very high rainfall in the uplands close to the western coastline (Figure 1).

The seeder-feeder process leads to large increases in precipitation with altitude on the uplands of western Britain, from typical coastal values of about 1000mm to 3000mm on the tops of the mountains in Snowdonia, Cumbria and in the Western Highlands, as illustrated schematically in Figure 1b. Precipitation and wet deposition rates increase more slowly with altitude in the central and eastern mountains, with peak precipitation between 2000mm and 2500mm.

The UK uplands are generally remote from sources of pollutants, and in these areas, the bulk of pollutants are in the aerosol phase. The seeder-feeder scavenging process increases wet deposition of pollutants more than precipitation as the additional water scavenged from the orographic cloud forms on aerosols from low level air containing substantial quantities of the major pollutants (SO_4^{2-}, NO_3^-, NH_4^+).

Cloud deposition

The presence of hill cloud in the windy uplands presents an additional deposition process, unimportant elsewhere in the UK. The droplets of hill cloud are sufficiently large to deposit

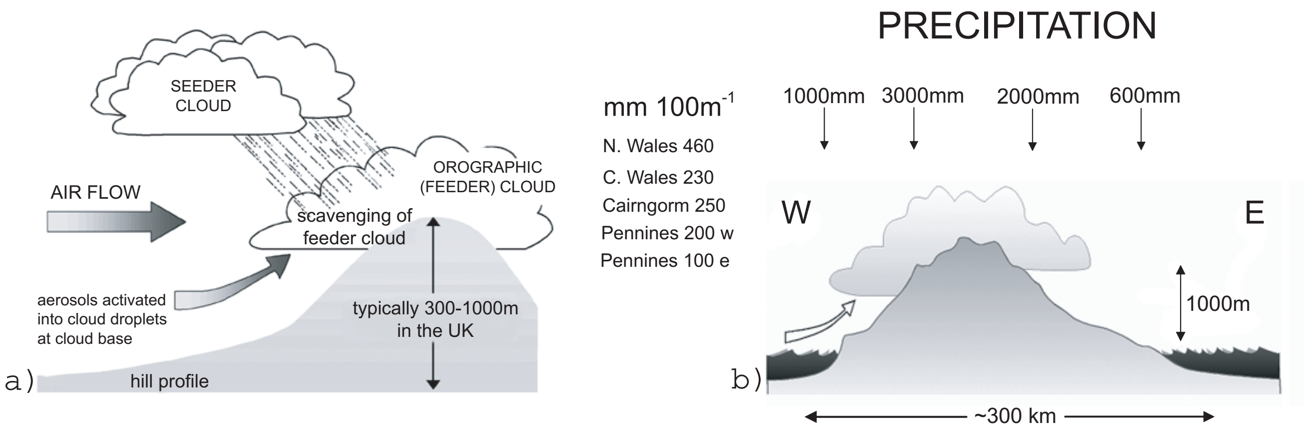

Figure 1: Schematic illustration of a) the seeder-feeder effect and b) precipitation amount from the west to the east of the UK.

rapidly on to vegetation and soil, unlike the aerosols on which the droplets formed. The rates of deposition of these droplets on to moorland and forest in the uplands have been shown to be close to the upper limits set by turbulent deposition processes, as illustrated in Figure 2, for droplets larger than about 4 μm radius from measurements using micrometeorological methods.

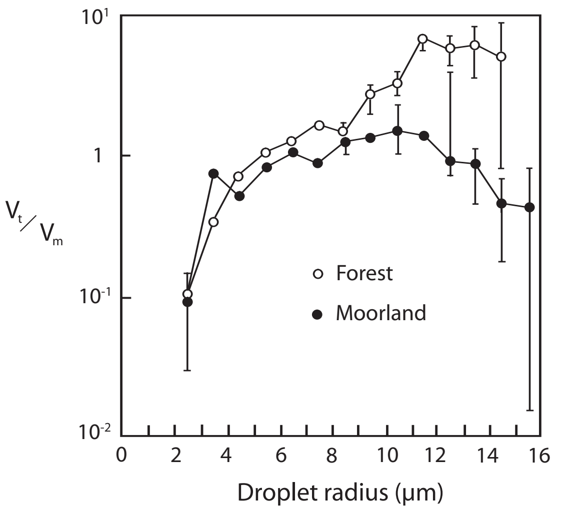

Figure 2: The ratio of turbulent deposition velocities for cloud droplets (V_t) and the upper limit for turbulent deposition (V_m), plotted against droplet size.

Ozone

Tropospheric ozone currently represents a threat to sensitive crops and semi-natural vegetation over large areas of Europe (NEGTAP, 2001). The threat posed has, in the past, been mainly due to peak concentrations, rather than the mean. The threshold concentrations for effects on many sensitive crops lie in the range 40 ppb to 50 ppb and the assessment methods have quantified the potential threat by quantifying the exposure to concentrations in excess of a Critical Level, expressed as exposure to an accumulated dose above a threshold concentration.

Current average concentrations of ozone in the UK are in the range 20-30 ppbV, with the largest average concentrations in the uplands and the smallest in urban areas and close to roads. Vehicles are responsible for about 50% of NO_2 and volatile organic compounds (VOCs), the precursor gases. Why should the uplands be exposed to the largest average concentrations of this potentially damaging pollutant? The photochemical formation of ozone requires short wavelength solar radiation, and is therefore confined to daylight hours; ambient concentrations at the surface represent the balance between photochemical production and losses at the surface by dry deposition. At night, terrestrial surfaces continue to remove ozone from surface air while the calmer conditions, caused by cooling at the surface and stratification of surface air, reduce vertical transport of ozone from higher levels. In these conditions concentrations at the surface decline during the hours of darkness in the lowlands. However, in the uplands, the windy conditions continue to provide an ozone supply from higher levels in the troposphere during the night, preventing the surface concentration declining as much during the night as is commonly observed at lowland locations. The effect of altitude on the average daily cycle in surface ozone concentration is illustrated in Figure 3.

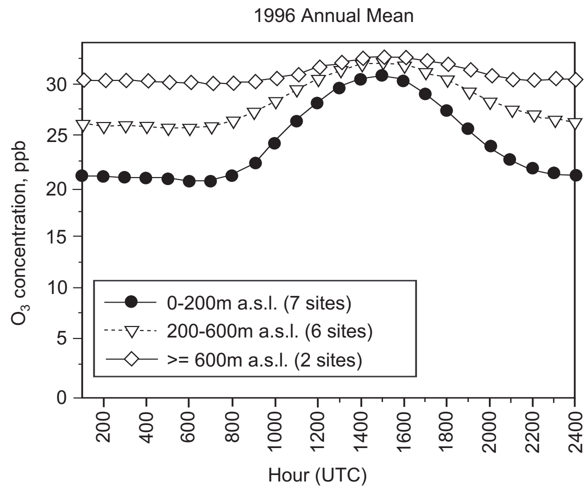

Figure 3: Annual Mean (1996) of the daily cycle of surface ozone concentration at three different altitude categories.

Control measures on the precursor pollutants (NO, NO_2 and VOC) have reduced peak concentrations by between 30 and 50 ppb in the UK during the last 14 years, and concentrations in excess of 80 ppb and 100 ppb are both becoming less common. However, as the peak concentrations have been declining, the background concentrations, largely from sources throughout the northern latitudes, have been increasing, as shown in Figure 4.

The gradual change in the ozone climate of the UK, with the background concentration rising and the peak values

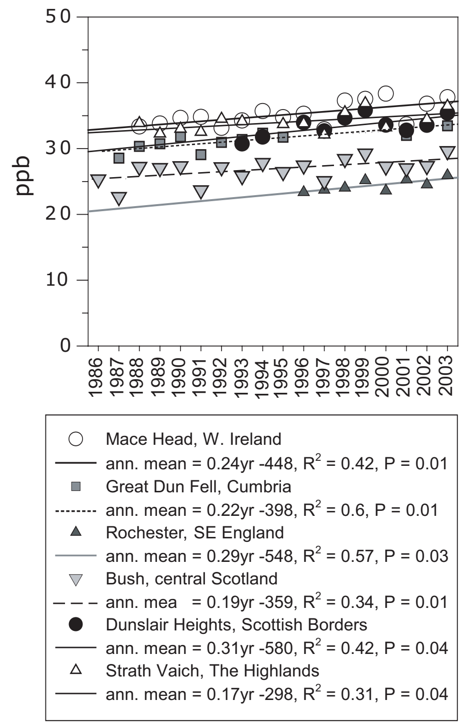

Figure 4: Statistically significant (at 5% level) upward trends in annual mean ozone concentrations at rural sites in British Isles. Only years with data capture over 75% are included. All sites are part of the UK National Monitoring Network (http://www.airquality.co.uk/) except Mace Head and Dunslair Heights which are maintained by Bristol University and CEH Edinburgh respectively.

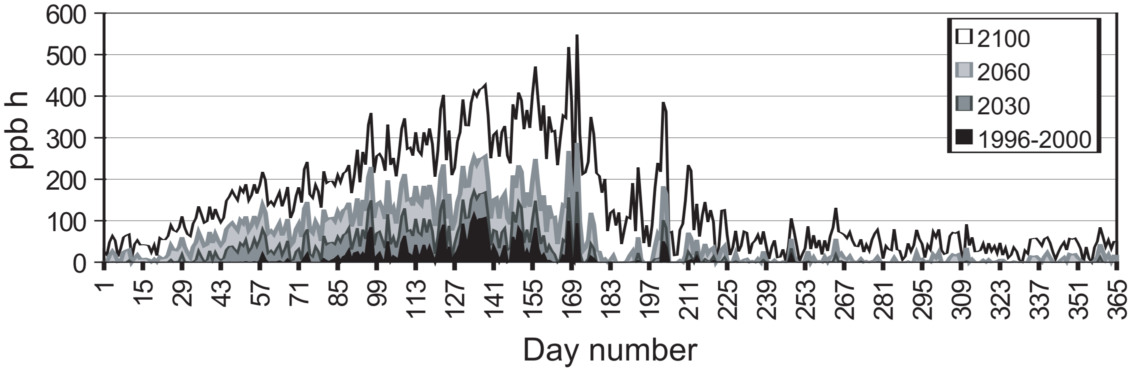

Figure 5: Current and future daily AOT40 values for Dunslair Heights (Scottish Borders). Future values are estimated using results from the STOCHEM (Stevenson *et al*., 2000) global model to enhance UK background ozone levels.

declining, places the focus for effects increasingly on the areas most sensitive to the growing background concentrations, the uplands. The semi-natural flora in these areas represent some of the ecosystems most at risk from the growing ozone background concentrations (Ashmore *et al*., 2003). To show the probable changes in the ozone climate of the UK uplands through the coming decades, a global chemistry-transport model has been used to simulate the effects of the expected changes in global emissions. The results for the UK have been used to estimate the accumulated exposure to concentrations in excess of 40 ppbV through the 21st century, as shown in Figure 5, from the work of Ashmore *et al*. (2002).

Trends in precipitation chemistry and expectations for the period 2005 –2015

The changes in emissions and deposition of acidity and sulphur over the last 20 years in the UK have been dramatic. The wet deposited acidity has halved over most of the country and the deposition of sulphur has also decreased on average over the UK by about 50%. Figures 6 and 7 show the declines in acidity and sulphur for four regions of the country in which the trends are regionally consistent. The grouping has been very useful to highlight a notable feature of the data, that different regions have experienced very different relative rates of change.

The total deposition of sulphur to the UK is partitioned into wet and dry deposition, and over the last 20 years the dry deposition has declined by 75% while wet deposition has declined by 47%, all during a period in which emissions throughout continental Europe and the UK declined by about 70%. The consequence of the changing partitioning of the deposition is that the majority of the decline in deposition occurred in the dry deposition dominated parts of the country. The uplands and especially the west coast uplands have benefited least from the reduced emissions of sulphur, and for many sites in the west the reduction in deposition is less than half of the reduction in emissions. The effects of shipping, especially in the eastern Atlantic, are particularly

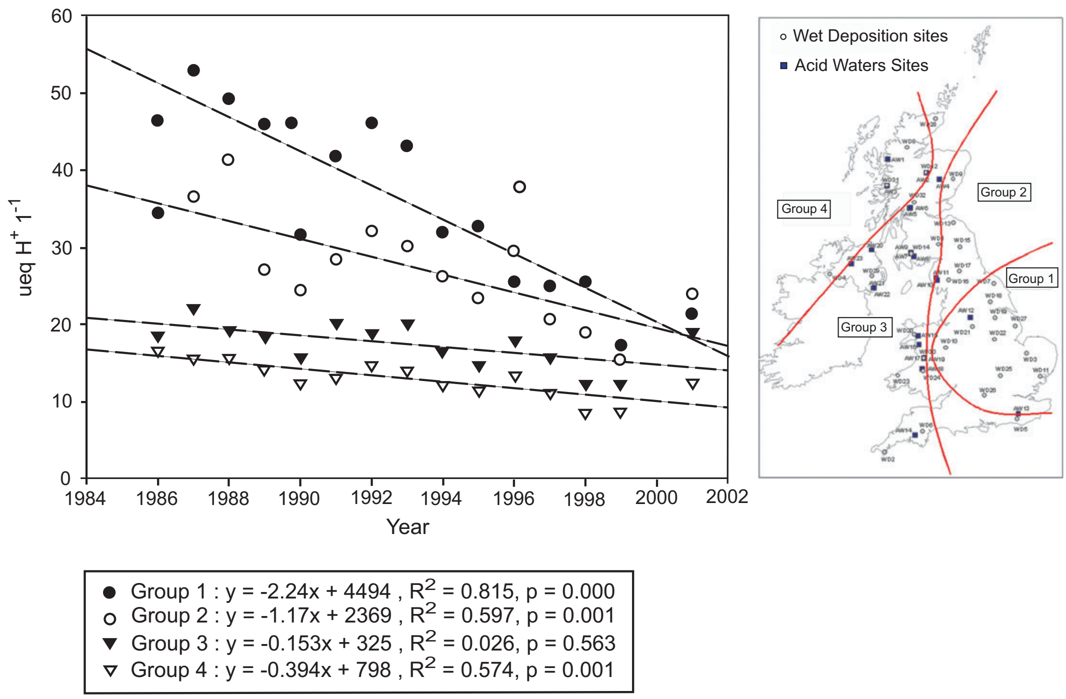

Figure 6: Trends in acidity in precipitation in the UK from 1986 to 2001. Grouping of sites according to the geographical distribution shown in the panel on the right.

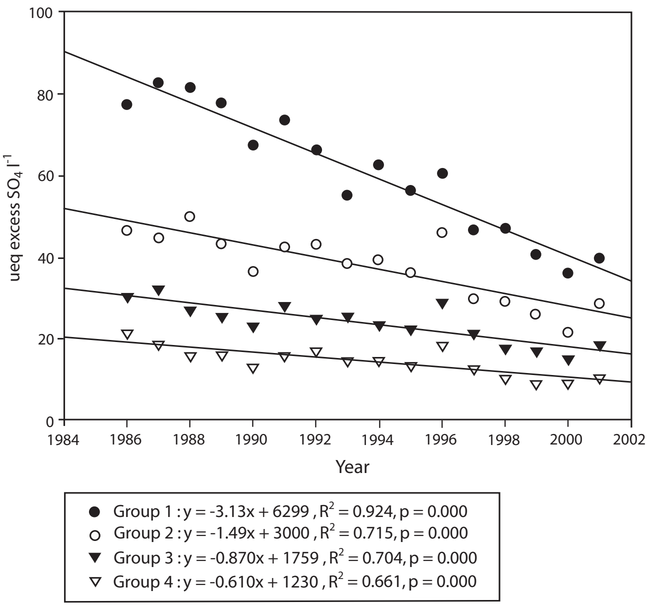

Figure 7: Trends in non-marine Sulphate in precipitation in the UK from 1986 to 2001. Grouping of sites according to the geographical distribution shown in Figure 6.

important and recent data suggest that changes in shipping emissions of sulphur are responsible for most of the observed non-linearity.

Future continued emission reductions in the UK will lead to smaller values in wet and dry deposition. However, with current emissions below 500 kTonnes S, by comparison with peak emissions of 3200 kTonnes S in 1979, most of the reduction has already taken place. Further reductions will also be strongly influenced by shipping sources, most of which are currently uncontrolled.

Nitrogen deposition

The emissions of nitrogen compounds in the UK have declined during the last decade, by approximately 45% for oxidized nitrogen and 10% for reduced nitrogen. However, for the uplands, the signal of reduced deposition is not detectable. Considering the total nitrogen deposition to the UK land surface, it is not possible to detect a clear downward trend so far (Fowler *et al.*, 2005) There is evidence that the transport distance of oxidized nitrogen has declined and that a larger fraction of emissions is deposited within the UK, but the data series for HNO_3 is currently too short to be confident of the historical budget. Thus the large deposition of both reduced and oxidized nitrogen in the uplands of the UK remains at values which exceed the pre-industrial values by more than an order of magnitude and shows little sign of change.

The spatial distribution of total nitrogen deposition in the UK is shown in Figure 8, and it is clear that the hot spots of nitrogen deposition include all the uplands, and especially Snowdonia, the Pennine hills and Cumbria, where nitrogen deposition ranges from 15 to 40 kg N ha^{-1} annually. The field evidence from the uplands shows that most of the deposited nitrogen from the last 100 years of deposition remains in the semi-natural soils and vegetation, with very few sites leaking significant quantities of the deposited N. The effects of the gradual sequestration of nitrogen by these soils are causing population changes in the flora (Smart *et al.*, 2004), and at some sites nitrogen is appearing in the run off surface water (Curtis *et al.*, this volume).

The prospects for the future nitrogen deposition in the UK uplands in the absence of further large reductions in emissions, especially of reduced nitrogen, is further accumulation of N in the soils and vegetation, and an increasing probability of NO_3^- breaking through into the surface water, leading to a renewed acidification of these sensitive fresh water systems.

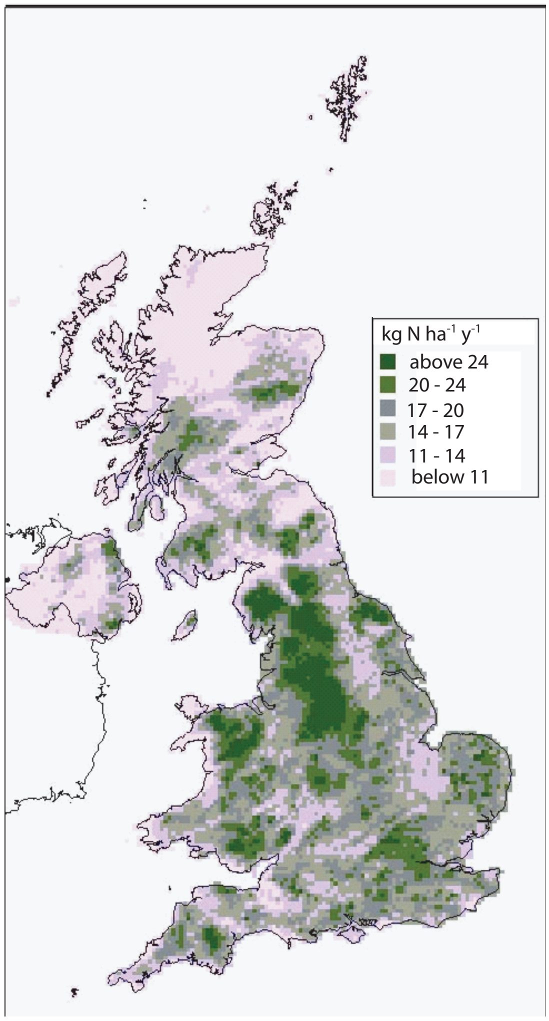

Figure 8: Total depostion of nitrogen in 2001.

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The sources of uncertainty in climate change impacts assessment studies

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Introduction

In assessments of future anthropogenic impacts on upland waters, one of the key issues is the role of climate. However, whilst we can look at the future in terms of certain pressures like land-use change or air pollution with some degree of spatial resolution and indeed accuracy, there is still huge uncertainty in future climate trends. Regional modelling and downscaling to specific sites may assume both the accuracy of the parent models and that different models are concordant; in this paper I highlight some of the key uncertainties associated with these assumptions.

For the basis of this paper, it is proposed that there are two primary reasons why climate change impacts assessments are performed. First; science driven impacts assessments are designed to study the sensitivity of a given system to changes in climate in order to understand linkages that may occur within a sector and/or between sectors. Science driven studies have, to date, made up the bulk of reported impacts assessments. They have, however, often overlooked the range of uncertainties that exist at a particular step within the process and the end results are not able to be quantitatively incorporated as feedbacks. Second; policy driven impacts assessments have been performed to enable decisions to be made about what adaptive and/or mitigative options are available for a particular stakeholder. Communicating the sources of uncertainty will enable the end-user/stakeholder to properly utilise the results of any impacts assessment and treat these as scenarios rather than predictions or forecasts.

Science driven studies did not previously need to incorporate uncertainty derived from the basic components of the climate to impact the policy system. They were more concerned with the understanding of how a system functioned. Many of these impacts assessments took results from one single climate change integration and used these as the next step on from applying arbitrary changes in climate parameters, i.e. the estimates of change from a single grid box from one or more Global Climate Model (GCM) integrations were used instead of arbitrary values. These climate change scenarios were then input into one impacts model and the results analysed. Often, in the absence of other sources of information, these results were used for informing stakeholders about future potential changes. There was as a result, therefore, no need or attempt to examine the actual sources and ranges of uncertainty that are present.

The main climate change inputs used in impacts assessments are the results from climate change experiments and accompanying observed climate data. These have been viewed by the impacts community as being "black box" in nature. It was seen by impacts assessors that there was no need to understand how GCMs operate, the range of forcing scenarios available or the sources of uncertainty associated with them. The process of climate change scenario data provision from the GCM community to the impacts community was often (especially during the First Intergovernmental Panel on Climate Change (IPCC) Assessment) seen as one way. The advent of interface projects, such as the Climate Impacts LINK Project which disseminates the results from the Hadley Centre, have helped to alleviate this perception and encouraged an understanding of the characteristics of data from GCM experiments

The sources of uncertainty

For the purposes of this qualitative assessment, the sources of uncertainty will be viewed from the traditional science driven approach to impacts assessment and then in more detail through the cascade of uncertainty that exits from emissions scenario through to the application of the results from a policy driven impacts assessment.

It is the purpose of this paper to present a qualitative description of the sources of uncertainty that exist in impacts assessments. The initial stage is the definition of an emissions scenario.

Source 1: Emissions scenarios
Uncertainties within this stage are derived from a number of sources. To construct an emissions scenario there is a need to define the way in which the socio-economic systems will develop in the future. This is possible, but the complex nature of these systems and their contrast to the behaviour of natural physically based systems which obey scientific laws, means that it is almost impossible to predict how the global socio-economic systems will develop in the future. It is from this basis alone that we must treat the results from further "down-the-line" in a climate change impacts assessment as being scenarios rather than predictions.

Through the IPCC's Special Report on Emissions Scenarios a range of world futures have been constructed. This process defined four "storylines" that describe possible future evolutions of the global socio-economic system:

SRES A1: a future world of very rapid economic growth, low population growth and rapid introduction of new and more efficient technology. Major underlying themes are economic and cultural convergence and capacity building, with a substantial reduction in regional differences in per capita income. In this world, people pursue personal wealth rather than environmental quality.

SRES A2: a very heterogeneous world. The underlying theme is that of strengthening regional cultural identities, with an emphasis on family values and local traditions, high population growth, and less concern for rapid economic development.

SRES B1: a convergent world with rapid change in economic structures, "dematerialization" and introduction of clean technologies. The emphasis is on global solutions to environmental and social sustainability, including concerted efforts for rapid technology development, dematerialization of the economy, and improving equity.

SRES B2: a world in which the emphasis is on local solutions to economic, social, and environmental sustainability. It is again a heterogeneous world with less rapid, and more diverse technological change but a strong emphasis on community initiative and social innovation to find local, rather than global solutions.

These futures are all very different, but must be treated as being equally plausible.

Source 2: Climate forcing
From the range of future scenarios, estimates for emissions of carbon dioxide and the other greenhouse gases are produced. For the SRES Scenarios these range from 500ppmv to 900 ppmv for 2100. If we take into account feedbacks from the carbon cycle this range becomes 550 to 1050 ppmv by 2100. Emissions are then converted by gas-cycle models to concentrations. At this stage the use of different gas-cycle models produces different end results, thus adding to the cascade of uncertainty.

Source 3: Global climate change
The scenarios of future concentrations of greenhouse gases are then used to derive how the climate system will respond. At a global scale two types of climate model are used; simple one-dimensional models and complex coupled ocean-atmosphere GCMs (e.g. HadCM3; Figures 1-2). The former due to their parsimonious computing requirements can be used to assess the full range of responses in the climate system as a result of different emissions scenarios. GCMs, on the other hand, have extensive computing requirements and as such only a limited range of forcing scenarios can be used in climate change experiments.

GCMs are the most powerful tool that we have available for assessing future global and regional climate change, but they are subject to uncertainties that need to be understood, listed below.

a) Errors in the initial state of the GCM. The initial boundary conditions that are used at the start of a GCM experiment are partly responsible for how a climate change integration evolves through time and for what its final outcome will be. The method used to overcome this uncertainty is to perform an ensemble of integrations that have the same forcing but different initial conditions.

b) Errors related to poor representation of unresolved scales. Many of the physical processes of the climate system operate at scales below the resolution (e.g., 2¡x3¡) of GCMs.

c) Errors related to instability of the climate system to small perturbations. The climate system at all scales is chaotic and is subject to unforeseen states caused by small internal changes. For example;

d) Constraints in the design of climate change experiments do not take into to account volcanic effects, which on a moderate scale (e.g. Pinatubo, 1992) can have a sub-decadal impact upon the climate system.

e) There are a range of different GCMs that represent the state-of-the-art in climate modelling. Most often within impacts assessments the results from one GCM integration using a given forcing scenario are used. Each individual GCM, however, has its own climate sensitivity (the equilibrium temperature change associated with a doubling of CO_2). The use of results from a single GCM will not capture the range of possible outcomes.

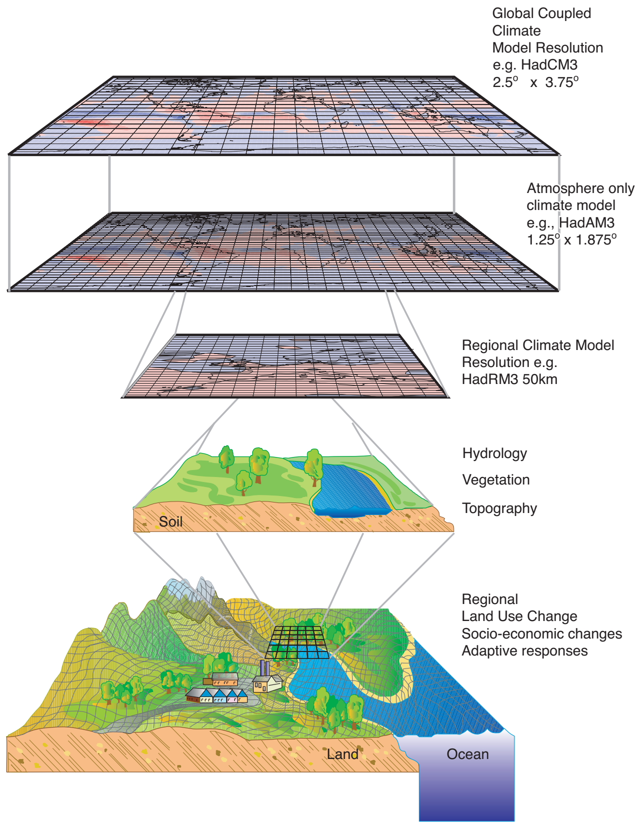

Figure 1: Resolving spatial scale issues in climate change scenario construction using a suite of nested climate models.

Source 4: Regional climate change
From the global scale most impacts assessments require regional or local scale climate information (Figure 1). As we "zoom-in" on a region we start to see increasing differences in the changes described by one GCM integration compared to another. For example in the US National Assessment, the results from two climate change integrations were used as the basis for wide ranging regional impacts assessments. These integrations, HadCM2GSa1 and CCCMa1GS, used the combined forcing of greenhouse gas and sulphate aerosols. As a result of the lower signal to noise ratio changes in these experiments there were contradictions in the climate change scenarios used. For example, over the central USA HadCM2GSa1 showed a wetting/drying and CCCMaGS showed the opposite.

In order to obtain data at a higher spatial and / or temporal resolution than obtained directly from the results of a given GCM experiment, there is a need to downscale (Figure 1). In order to employ a given downscaling technique there is often the requirement to have a high quality observational dataset.

Observational datasets themselves have incorporated into them uncertainties due to data errors and heterogeneities of measurement within a given region. The higher the spatial resolution, the greater the uncertainty. There has, however, been no attempt to date to fully assess the performance, reliability and subsequent uncertainties associated with the various downscaling methods. There are three broad methods that are used within the impacts community, described below.

a) Interpolation, applying an interpolation routine to the raw GCM data to produce a higher resolution version, or by applying the GCM resolution change fields to a high resolution baseline climatology.

b) Empirical, normally referred to as weather generators. There are two broad groups; first, circulation / weather typing, using relationships between surface climate and synoptic circulation and then applying these to future climates derived from GCM experiments, and second, stochastic weather generators that are used to perturb time series of observed data.

c) Dynamic methods, mainly using Regional Climate Models (RCMs) driven by boundary conditions, which are used to produce higher spatial resolution data over a given region (Figure 1). To date, one GCM has been used to drive one RCM as they are expensive on computing resources. This has introduced a new level of uncertainty, as differing RCMs forced by the same GCM will produce differing results, while other uncertainties such as domain size create uncertainty.

Source 5: Impacts model
Whilst not specifically covered by this paper, the use of a single impacts model (e.g. a hydrological model) for a given exposure sector does not cover the entire range of uncertainty. It is therefore advised, where possible, to force a number of comparable impacts models to investigate the range of sensitivity each has to climate data.

Source 6: Communication and interpretation of impacts assessments into policy decisions
The majority of climate change impacts assessments that have been performed to date have followed the science driven model and many of the integrated assessments of climate change have undertaken bench studies that have drawn upon these results. Translation and application of the results, therefore, from impacts assessments through to a policy decision regarding a given adaptive or mitigative strategy will produce different interpretations. It is important that the results and associated uncertainties are communicated directly by the scientists to the policy makers. There is a need, therefore, to:

a) fully inform stakeholders about the uncertainties embodied in assessments;
b) stress the robust aspect of the assessments; and
c) try to encompass probability.

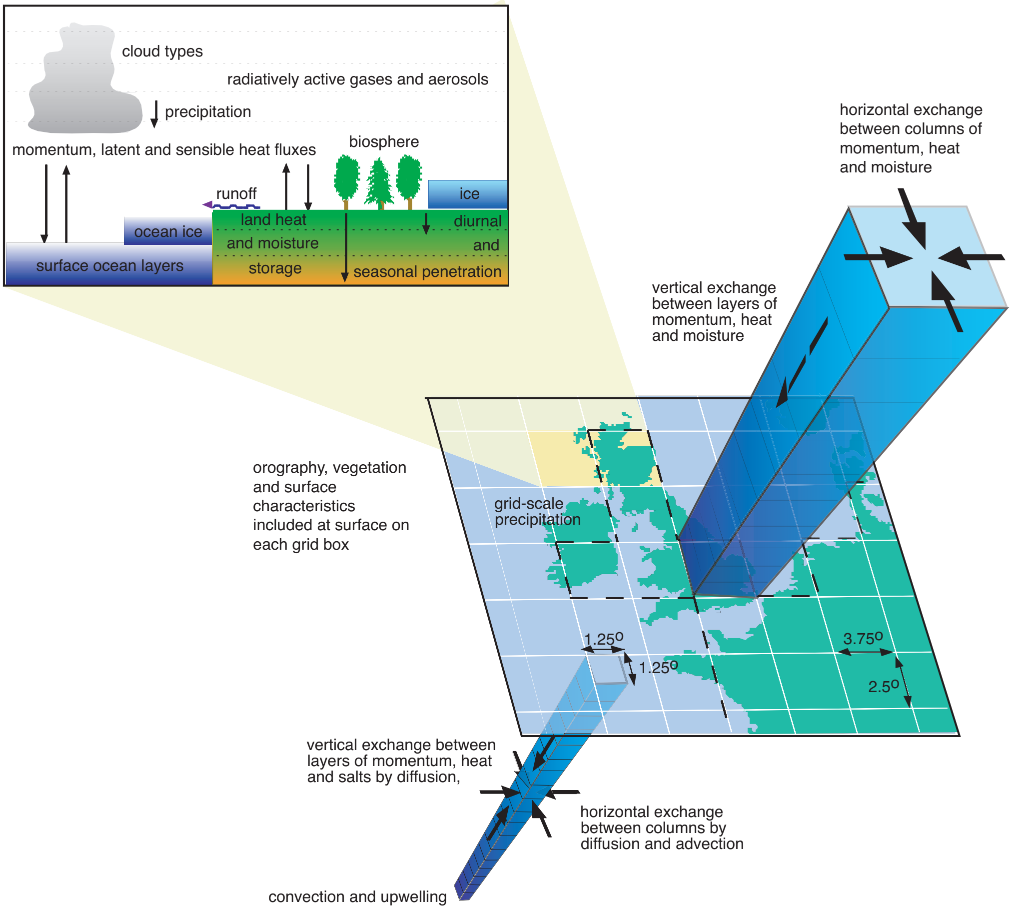

Figure 2: The conceptual structure of a coupled ocean-atmosphere GCM (e.g. HadCM3).

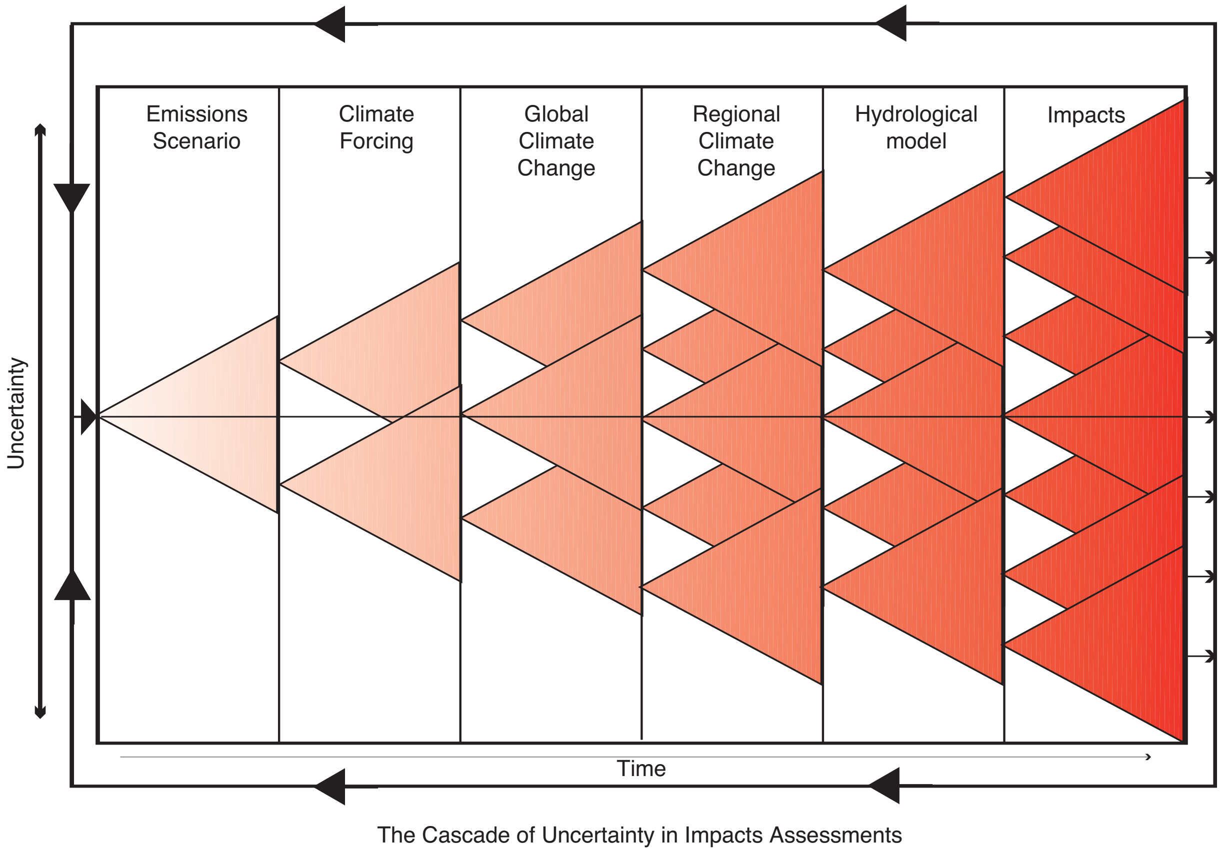

Figure 3: The cascade of uncertainty in impacts assessments.

Discussion and conclusions

This paper attempts to describe in a qualitative manner the sources of uncertainty that exist in climate change impacts assessments. The results from any given climate change assessment are only as good as the weakest link in the cascade of uncertainty (Figure 3). There is, therefore, a need to identify and encapsulate the range of uncertainties that exist within the results of an impacts assessment.

For upland water catchments, the models employed to predict future impacts on water quality or biological status (e.g. critical loads) have their own associated uncertainties, but also occur close to the bottom of the cascade of uncertainties in modelling climate impacts in terms of scale (Figure 3). Hence local uncertainties in, for example, future precipitation, temperature, hydrology and weathering rates resulting from "natural" variability, to the extent that this may have been captured in monitoring programmes during a historical period of climate change, are all magnified by uncertainties in future emissions and climate forcing scenarios at larger regional and global scales.

Overall, it is recommended that a range of emissions scenarios should be used to force a suite of GCMs in order to take into account the range of possible socio-economic futures. To incorporate the range of potential changes that may occur for a given region it is advised that the results from a number of GCM experiments are used. The use of just two integrations within the US National Assessment has shown how contradicting changes for a given region enhance the range of uncertainty and thus dilute any impacts results. To date there has not yet been a quantitative assessment about the differing merits of adopting a certain climate change scenario construction / downscaling methodology. Such an assessment would be a valuable tool in guiding impacts assessors.

Communication of the results from impacts assessments to the policy community is difficult and as yet there appears to be no suitable framework available to incorporate the sources of uncertainty, unless the above suggestions are taken into account when designing an impacts assessment.

Further reading

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Land-use change in upland catchments and effects on upland waters

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Introduction

Since the 1980s, the main threat to upland water quality has been perceived as acid rain (Wright, this volume). The only land-use practice implicated in contributing to declines in water quality has been plantation forestry, due to the scavenging effect of the canopy enhancing the deposition of acidic air pollutants. This focus has resulted in a surprising lack of data concerning the effects of the large increase in stock numbers which occurred in the latter half of the twentieth century on upland water quality, as highlighted in a review by Samson (1999). Here we briefly review the limited data available concerning the impact of grazing on upland water resources and consider proposed changes in both agricultural and forestry practices which may result in further changes in water quality in the uplands.

Changes in animal numbers

There has been a huge increase in animal grazing in the uplands of the UK in the last 50 years, driven by the low unit return from sheep and headage payments to farmers. For example, the numbers of sheep increased from approximately 22 million to 44 million between the 1940s and 1993 in the British Isles (Samson, 1999) although numbers have since dropped, partly due to the Foot and Mouth outbreak (2001) and partly due to the cut-back in subsidies (Environment Agency, 2004). In Wales alone there has been a three-fold increase to 11 million in the last 50 years and the increase has been seen across the European Union with a doubling of sheep numbers in the last 20 years. In addition to this large increase in animal numbers, there has been a move to heavier sheep breeds and increased use of pasture, particularly in the winter, which will have enhanced the environmental impacts of increased sheep numbers in upland and hill areas where 70% of the flocks graze (Samson, 1999). However, on top of these changes promoted by public subsidy, red deer have also increased, with a doubling of numbers between 1959 and 1989 to around 300,000 in Scotland alone (Staines *et al.*, 1995).

Impacts of grazing on soil

The impact of this increase in animal numbers will have been wide ranging, from changes in vegetation composition and structure, soil structure and function, and water flow and quality. For example, English Nature (2001) identified overgrazing as a major concern for the conservation of terrestrial plants and wildlife. For soils and water, if grazing pressure is too great for the carrying capacity of the land, impacts can include compaction of the soil, increased rainfall runoff, increased soil and riverbank erosion, reduced aquifer recharge and low river flows, and increased transfer of contaminants to water courses. However, much of the evidence on the effects of over-grazing on soils comes from studies in rangelands or the lowlands of the UK, with few published studies focussed on UK upland systems.

Why do we think there may be a problem in the uplands? Some evidence comes from monitoring work which seeks to identify long-term changes in the health of our soils. For example, in a recent report on the status, pressures and controls of soils in Wales there was an 11% loss of organic matter of soil in permanent pasture between 1980 to 1997, with the worst effects found on peaty, and thus presumably, upland soils (Reynolds *et al.*, 2002) (Figure 1). The cause of this decline may be complex, but overgrazing was identified as one likely contributing factor. This mirrors concerns expressed at the broader scale, with heavy trampling by animals and intensification of livestock identified as a pressure on soils in England and Wales by the Environment Agency (2004) and the Royal Commission on Environmental Pollution (1996) stating that over-grazing was responsible for 23% of soil degradation within Europe.

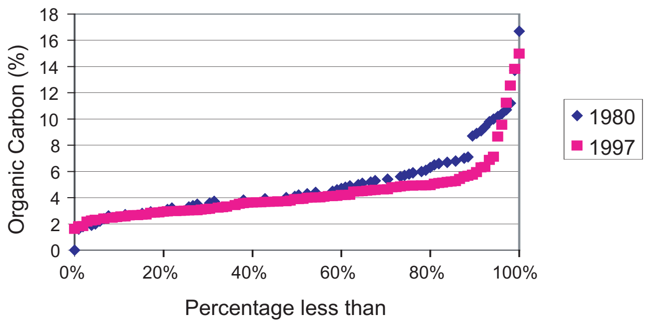

Figure 1: A cumulative frequency plot showing the change in % in organic content (0 - 15cm) in permanent grassland in Wales between 1980 and 1997 (Reynolds *et al.*, 2004).

Impacts of grazing on water movement

Direct evidence that stocking density affects soil structure and function and thus water movement in upland soils has been identified recently in a study by Carroll *et al.* (2004a) in mid Wales. Infiltration rates (the rate of water movement into the soil) were found to double in grazing paddocks established 15 years previously by ADAS Pwllperian where sheep numbers were reduced from 3.7 sheep per hectare to 1.9 sheep per hectare (Figure 2). This suggests that grazing even at these relatively low levels of stocking density can result in compaction of soil, thus lowering the infiltration rate. This is particularly worrying when considering that many parts of our uplands have between 3 and 8 sheep per hectare (DETR, 1998). In two other locations studied by Carroll *et al.* (2004a)

(Snowdon and Pontbren), experimental paddocks were not available and the variability in soil type masked any effect of grazing pressure on infiltration rates. This highlighted the importance of taking into account the tolerances of different soil types. Carroll *et al.* (2004a) compared their results with constant infiltration rates published by Berryman (1974) for a range of soil textures for non-compacted soils. Seven of the nine sites studied had infiltration rates lower than expected, indicating they had compaction problems to some degree. One conclusion from this study is that reduced infiltration in upland areas due to high stocking density may be having a greater contribution to high flow events than previously thought, due to widespread compaction problems and associated surface runoff.

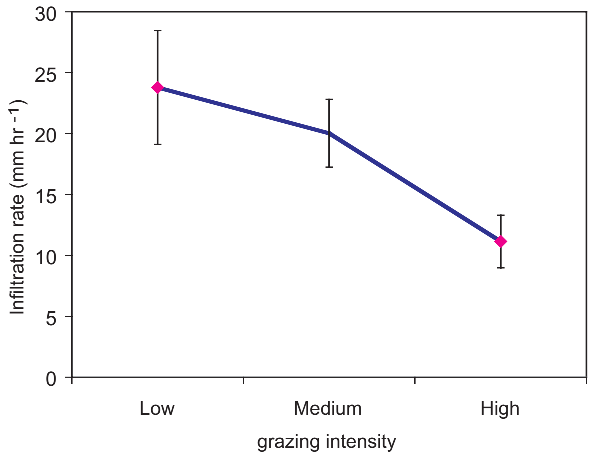

Figure 2: Infiltration rates using a double ring infiltrometer in 2 ha paddocks at ADAS Pwllperian with variable stocking density. High grazing = 3.7 sheep/ha, Low grazing = 1.8 sheep/ha. All grazing is imposed between April and October with no winter grazing (Redrawn from Carroll *et al.*, 2004a).

Impact of grazing on erosion

With lowered infiltration and increased runoff rates, the risk of erosion may increase, as may the risk of transfer of pathogens (e.g. oocyts from sheep dung) and other contaminants to waters. An in-depth study has been carried out in Blelham Tarn in the English Lake District that provides compelling evidence of accelerated erosion in this catchment which can be quantified and linked to an increase in stocking density (van der Post *et al.*, 1997). Using a wide range of techniques on frozen sediment cores, an exponential rise in sedimentation rates over the past 40 years was measured, which the authors could attribute to catchment erosion. This rapid rise in sediment yield closely paralleled the increase in sheep stocking density in the catchment (Figure 3). Increased sedimentation can have major ecological effects including smothering of river-bed gravels, thus harming aquatic plants, invertebrates and the eggs of fish. The four-fold increase in sediment accumulation rates since 1978 reported in the Blelham Tarn study is an important piece of evidence that quantifies the link between increased stocking and a decline in water quality. Climate change may further exacerbate this risk as the wetter winters, more intense rainfall and drier summers forecast will accelerate runoff and increase the erosion risk (Environment Agency, 2004).

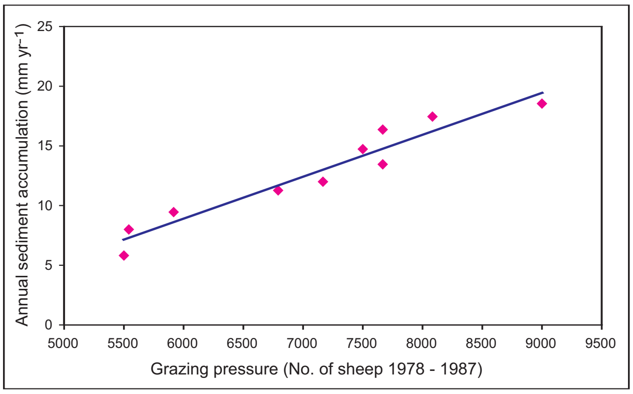

Figure 3: The relationship between sedimentation rate in Blelham Tarn and sheep numbers (van der Post *et al.*, 1997).

Impacts of grazing on the accumulation of carbon and nitrogen

Changes in water flowpaths and vegetation biomass and composition may affect the ability of upland pasture to sequester carbon and prevent transfer of deposited nitrogen from the atmosphere to waters. An attempt to quantify these processes comes again from the long term grazing paddocks at Pwllperian in mid Wales where water quality, soil carbon storage and ^{15}N retention have been quantified either side of fences separating paddocks of different stocking density (Emmett *et al.*, 2004). Results indicated that there was an increase in soil carbon content in upper soil horizons in paddocks with reduced stock numbers, due to the build-up of organic matter derived from dead and decaying plant material. As reported for grasslands in general (Conant *et al.*, 2001), this increase was only found in the upper soil horizons. In a review by Conant *et al.* (2002) changes are generally greatest in the first 40 years following improved grazing management, but are generally only found to be positive in wetter areas (>333mm pa). Further work is needed to quantify the potential for increased carbon accumulation more accurately if animal numbers are reduced in UK upland systems.

With respect to retention of reactive nitrogen deposited from the atmosphere, a ^{15}N study at this site revealed an increase in the retention capacity of deposited nitrogen when stocking density was reduced (Emmett *et al.*, 2004). This was found to be partly due to increased biomass of higher plants and partly due to increased retention by the soil, possibly due to the faster infiltration rates recorded by Carroll *et al.* (2004a) at the same site. This could lead to the conclusion that reducing stocking density could reduce the risk of transfer of pollutant nitrogen to water. However, monitoring of soil water quality suggested that there was a surprising trend for greater concentrations of nitrate in soil water when stock were removed, presumably due to the lower uptake demands by plants (Emmett *et al.*, 2004).

Changes in forestry strategies

New forestry strategies have been produced around the UK which promote a move away from single species, even-aged stands of exotic species for wood production to greater diversity in structure and use of forests (www.forestry.gov.uk). These changes include issues such as greater use of native species, continuous cover management practices and greater social interaction and involvement. These are all likely to have significant impacts on water resources although in some areas, such as the water use of native broadleaved species, empirical data on water use and quality are scarce (Environment Agency, 1998). A recent review of the impact of current forestry practices on peak flows and baseflows can be found in Robinson *et al.* (2003) and a summary of effects on water quality can be found in Nisbet (2001). New research clearly needs to be focussed around the likely change in species and felling practices together with studies into the greater use of trees in agricultural systems. These research needs will be driven by the Water Framework Directive which in upland areas is likely to require the control of diffuse pollution sources by both agriculture and forestry to control erosion, combat acidification and increase water retention to prevent damaging floods. A more integrated approach will be required at the catchment level and is likely to drive the formation of new agri-environment schemes.

The way ahead

Some studies are already highlighting some of the approaches which may be taken to minimise the adverse effects of land-use on upland water quality. One is the Upper Wharfedale Best Practice Project (Chalk, 2003) which is a partnership to demonstrate the principles, techniques and benefits of an integrated way of achieving good land and water management in an upland catchment area. One activity has included the blocking of moorland drainage channels, slowing peak flows and thus reducing erosion and flooding. There have also been benefits to stock and birds, suggesting that this approach may benefit both the environment and land managers alike. Other activities have included regeneration of riverside habitat by fencing and tree and shrub planting, lake restoration, hedging and coppicing and re-engineering of river features. These activities in turn helped the economy of the area by supporting over 50 rural businesses.

In another study, the potential advantages of greater use of trees in the agricultural landscape have been identified. A group of 10 farmers working 1000 hectares in the upper reaches of the River Severn in the Pontbren project are developing less intensive farming systems in mixed lowland/upland farms. Farmers are destocking and using hardier sheep breeds, restoring wetlands and streambanks, and planting trees to provide shelter and bedding through harvesting for wood chip. The trees are planted in 5m wide shelter belts using native tree species, thus replacing lost linear tree features. This is a bottom-up initiative which provides social and economic benefits for the farmers (e.g. saving in buying in straw for bedding through the use of woodchip) but will also deliver environmental benefits. For example, the farmers observed water sheeting off the grazed pasture under heavy rainfall disappearing into the soil when it passed into the recently planted shelterbelts. Infiltration measurements identified a 60-fold increase under 6 – 8 year-old trees with increases recorded even under 2 year-old planted trees (Carroll *et al.*, 2004b) (Figure 4). This increase was attributed to a change in the macro to micro-pore ratio of the soil due to tree roots and soil animal activity, as no ground preparation has taken place. The effect of excluding grazing animals from the planted area could also have contributed to this large increase in infiltration rates. Further work aims to quantify the relative contribution of reduced stocking alone and in combination with tree planting on water pathways and peak and base flows in streams. Wider benefits are also being investigated, including the socio-economic benefits to farmers and impacts on wildlife, reflecting the potential for this study to illustrate a more holistic approach to catchment management.

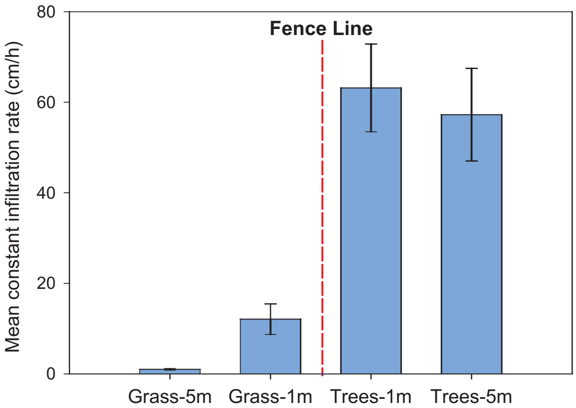

Figure 4: The change in infiltration rates 1m and 5m either side of a fence separating recently planted tree shelter belts (2 - 8 years) from the original grazed pasture (Carroll *et al.*, 2004b).

Conclusion

The issues and case studies outlined here highlight the need for more studies to identify the impact of stocking density, new styles of forestry and their spatial distribution within catchments on upland water resources. The start of CAP reform in 2003 and formation of new forestry strategies and agri-environment schemes makes this an urgent need for environmental scientists and managers alike if upland water resources are to be understood and managed more effectively in the future.

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Evidence for the recovery of freshwater lakes and streams in the UK from acidification, based on the analysis of data from the UK Acid Waters Monitoring Network (1988-2003)

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Atmospherically deposited acidic pollutants represent perhaps the most pernicious threat to the ecology of geologically sensitive upland surface waters in the UK. International concern over acidification has led to expensive acid emission controls in recent years and the environmental effects need to be audited. In the UK, the Acid Waters Monitoring Network (AWMN) is maintained for this purpose by the UK Department of Food and Rural Affairs. Established in 1988, the chemistry of run-off and the species composition of a range of biological groups from 22 acid-sensitive lakes and streams (Figure 1) have been routinely monitored ever since (Patrick *et al*., 1996). Analysis of the first 15 years of data from the AWMN has recently been completed and is due to be reported shortly. This paper summarises the main findings of this work, conducted by staff at University College London, the Centre for Ecology and Hydrology (CEH) Wallingford, CEH Bangor, CEH Dorset, CEH Edinburgh, the Department of Biological Sciences, Queen Mary College London and Fisheries Research Services, Pitlochry (see www.ukawmn.ucl.ac.uk).

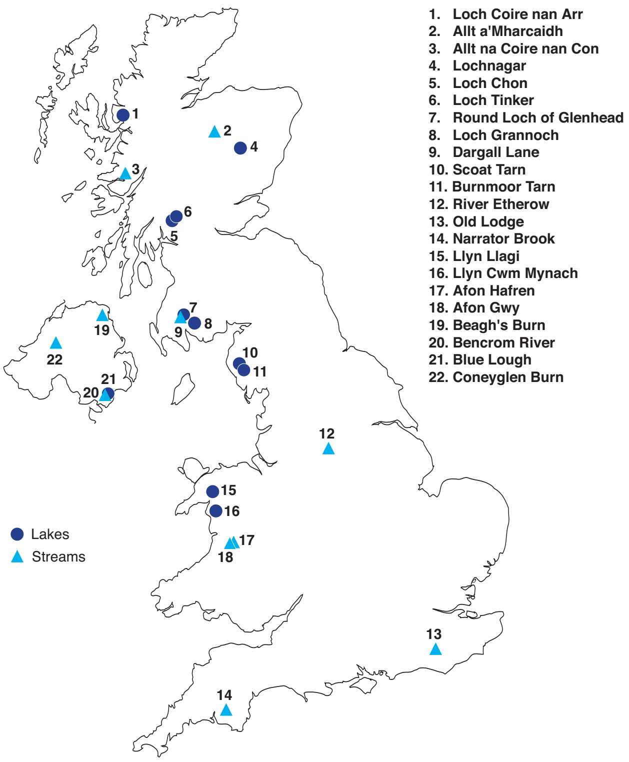

Figure 1: Location of UK Acid Waters Monitoring Network Sites.

Over the period of assessment (1988-2003), data from the UK's Acid Deposition Network (ADN) show that the deposition of sulphur (S) from non-marine sources has fallen by approximately 50% over much of the UK. The largest absolute reductions are centred over south-eastern and central England and fall in magnitude in northerly and westerly directions. Over the same period there has been a smaller reduction in nitrogen deposition, but for this pollutant statistically significant declines are mostly restricted to central England.

Trends in the chemistry of AWMN sites are illustrated in Figure 2. The decline in the estimated flux of non-marine sulphate (xSO_4) in deposition is reflected in the estimated flux of xSO_4 in run-off from AWMN sites with little evidence for any time lag. Significant declines in the xSO_4 concentration of run-off are observed in all but the north of Scotland where S deposition has historically been very low (Cooper and Jenkins, 2003). No trends have been observed in nitrate (NO_3) concentration, but a recent increase in spring maxima can be linked to a recent series of colder, drier winters associated with a relatively low North Atlantic Oscillation Index (Monteith *et al*., 2000). At some sites the recent elevation in NO_3 is of the same magnitude as the decline in xSO_4.

Declines in the SO_4 anion are predominantly balanced by declines in the concentration of base cations (particularly calcium and magnesium). However, 13 sites have experienced one or more of the following changes which are considered biologically favourable: an increase in pH; an increase in alkalinity; and a decline in labile (biologically toxic) aluminium. Acid neutralising capacity (ANC), a calculated expression of the acid-base balance, has been determined according to the calculation:

$$[\text{ANC}]\ (\mu\text{eq l}^{-1}) = [\text{Alkalinity}]\ (\mu\text{eq l}^{-1}) + F\ [\text{DOC}]\ (\text{mg l}^{-1}) - 3\ [\text{Labile Aluminium}]\ (\mu\text{mols l}^{-1})$$

(where F = 4.5 for pH <5.5 and 5.0 for pH>5.5)

ANC has increased at all 13 sites where change in one or more of pH, alkalinity and labile aluminium has been identified. The rate of increase in ANC is linearly related to the rate of decline in xSO_4 and those sites which fail to show any change in ANC are mostly those experiencing the smallest declines in xSO_4. Noise in acidity signals generated by climatic effects, particularly due to inter-annual variability in precipitation, sea-salt deposition (Evans *et al*., 2001) and nitrate leaching has masked chemical improvement at some sites where changes in xSO_4 are only modest.

Dissolved organic carbon (DOC) concentration has increased substantially at all 22 sites. The mechanisms for this increase, which is similar to that observed in other regions of the UNECE ICP Waters network, are as yet unclear and have been subject to much recent debate (see Evans and Monteith, this volume). However, work in progress seems to suggest a combined influence of declining S deposition and climatic effects (such as the rise in air temperature over the last three decades, the increased occurence of drought and/or changes in the amount of seasalt deposition). The tight link between xSO_4 and ANC trends suggests that changes in DOC may be

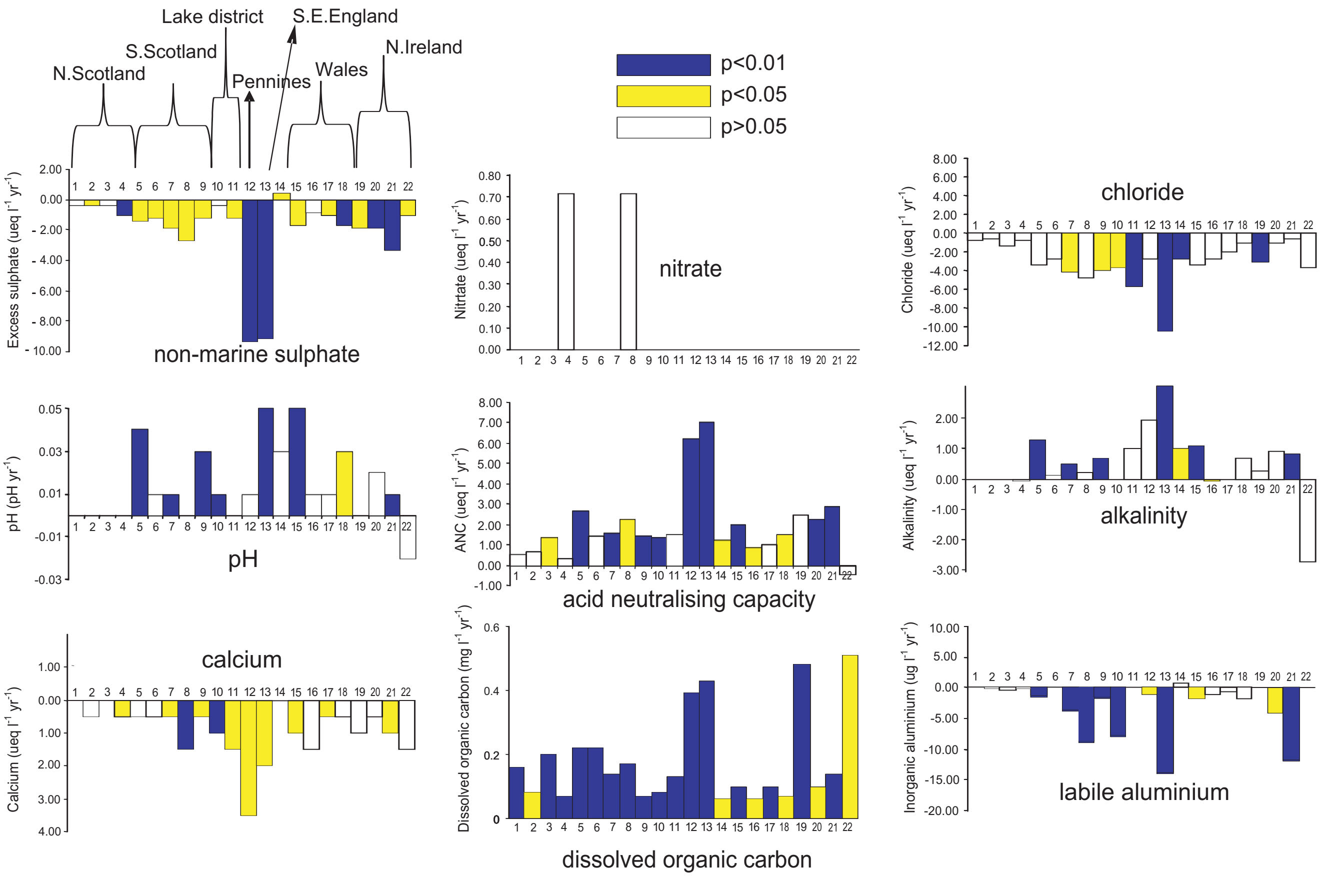

Figure 2: Trends in water chemistry determinands on the UK Acid Waters Monitoring Network based on the Seasonal Kendall Test and associated slope estimator.

integral to the recovery process. The decline in "strong" mineral acids may therefore be being partly balanced by an increase in "weak" organic acids and this should be biologically favourable. However, increases in DOC in north Scotland, where xSO_4 decline is negligible, can only be explained by climatic effects and may be leading to a net increase in acidity.

The species composition of epilithic diatom (single celled algae which grow attached to rock surfaces) and aquatic macroinvertebrate (largely representing insect larvae and beetles) samples has changed significantly at approximately half of all sites. There is a particularly strong spatial relationship between those sites showing trends in macroinvertebrates and those showing trends in ANC. Epilithic diatom communities have also changed at all sites which have undergone increases in pH and/or alkalinity with the exception of the three most acidic of these (all with a mean 15 year pH <5.0).

Where epilithic diatom communities have changed, acid-sensitive species have usually increased in proportion to acid-tolerant species. There is also evidence for an increase in the proportion of some known acid-sensitive macroinvertebrate taxa. Of potentially wider ecological significance there has been a notable increase in the representation of predatory animals; this is consistent with the hypothesis that lessening environmental stress tends to be accompanied by a lengthening of the aquatic food chain (Woodward and Hildrew, 2001). Acid-sensitive mosses and higher plants have been found for the first time in the last five years at several sites, most of which have undergone changes in alkalinity. In contrast, positive changes in salmonid density have only been identified at three of the most acidic sites on the AWMN, where juvenile brown trout have recently been detected for the first time. These changes can be linked to a rise in ANC at these sites above theoretical threshold levels for trout survival, with associated declines in labile aluminium concentration.

Observations of biological change are therefore consistent with a regional biological recovery signal. In the case of epilithic diatom communities, where acid-sensitive species have maintained a viable presence even when sites were in their most acidified state, the response to increasing pH or alkalinity appears to be almost immediate. Likewise, the apparent rapid return of acid-sensitive aquatic macrophytes species suggests these plants may have maintained a non-vegetative presence, e.g. in the form of seeds or spores. The prevalence of caddis fly species in the list of "expanding" macroinvertebrate taxa may reflect the relative motility of these winged insects. Generally, however, biological changes are gradual and it is possible that there may be physical and biological restrictions to successful re-establishment of species which may have been lost to these sites during decades of acidification. Other species may require considerably longer to return to sites in significant enough numbers to establish sustainable populations. Biological recovery may also be hampered by the sporadically disruptive effects of acidic episodes, particularly at times of high precipitation, and by competitive exclusion by dominant acid-tolerant species which are not adversely affected directly by falling acidity. As chemical improvement has only become detectable over the last five years, further monitoring is essential to determine the true extent of the biological response.

In summary therefore, the chemistry and ecology of lakes and streams in the AWMN appear to have responded rapidly to the emission control induced decline in xSO_4 deposition. ANC has increased in proportion to the rate of decline in xSO_4 with the dominant effect observed in a region stretching from south-east England to south-east Northern Ireland and central Scotland. An increase in organic acids and climatic variability may have partly masked responses in conventional measures of acidity status, i.e. pH and alkalinity, but the increase in DOC may also be integral to the recovery process. Gradual biological changes are apparent within the region, particularly where sites show a significant increase in ANC and are generally consistent with recovery responses. However the time-scale for full biological recovery is likely to lag chemical improvement and further monitoring is clearly necessary to determine the wider long-term effect of acid emission reductions. Our analysis also demonstrates the sensitivity of these systems to variability in climate and is beginning to shed light on how interactions between deposition reductions and progressive climate change might shape further recovery.

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Trends in the dissolved organic carbon content of UK upland waters

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Introduction and site description

Freshwater dissolved organic matter (DOM) is comprised primarily of complex, high molecular-weight humic substances. These humic substances form the primary control on water colour, contribute to water acidity, and provide a mechanism for transport of carbon, energy and organic nutrients within the aquatic system. Changes in the concentration of DOM, or the commonly measured dissolved organic carbon (DOC) component of DOM, are of great interest with regard to freshwater acidification, aquatic ecology, water supply and carbon cycling.

This paper focuses on the trends in DOC observed in the UK Acid Waters Monitoring Network (AWMN). The AWMN comprises 11 lakes and 11 streams in acid-sensitive areas of the UK, spanning a range of deposition, altitude, climate and land-use (i.e. moorland and coniferous forest). Co-ordinated chemical and biological monitoring of these sites began in 1988. Increasing DOC concentrations in AWMN lakes and streams were first identified after 10 years, at which time 16 of the 22 sites exhibited rising trends (Monteith and Evans, 2000). At this time, relatively few sites showed signs of chemical recovery from acidification, and observed increases were tentatively attributed to elevated temperatures, possibly through the increased production of relatively recalcitrant phenolic compounds at higher temperatures (Freeman *et al*., 2001). This paper provides an update on DOC trends in the Network after 15 years of monitoring, and discusses the potential drivers of observed changes. Note that a more extensive assessment of DOC trends in the AWMN, and elsewhere, is provided by Evans *et al*. (2005).

Results and Discussion

After 15 years of monitoring, trend analyses using the Seasonal Kendall Test (Davies *et al*., 2005) show that DOC concentrations have now risen at all 22 sites in the AWMN. This is the most consistent temporal change of any water quality determinand measured, and represents a remarkably uniform change in water quality across a wide spatial range. On average, DOC concentrations have risen by 0.18 mg^{-1} yr^{-1}, although a general pattern is observed in which trends are largest at sites with initially higher DOC (DOC trend = 0.06 x $DOC_{1988-1993}$, R^2 = 0.71). This implies a relatively uniform proportional increase, averaging 91% over the 15 year period.

Example DOC time series for six of the lakes (Figure 1) and six streams (Figure 2) illustrate the nature and extent of observed changes. The majority of sites show some seasonal cycle, and streams in particular exhibit episodic peaks, but at most sites increases appear to have been approximately linear, and sustained over the full 15 years. At many sites (e.g. Loch Coire nan Arr, Loch Chon, Round Loch of Glenhead, Allt na Coire nan Con, Old Lodge, Beaghs Burn) the highest recorded concentrations have occurred since 2001. Overall, therefore, there is little evidence to suggest that observed changes form part of a natural decadal scale cycle, as has been observed for other solutes such as seasalts (Evans *et al*., 2001) or nitrate (Monteith *et al*., 2000).

The processes controlling DOM production are complex. Freshwater concentrations may be influenced by biological processes of DOM production in soil solution, chemical

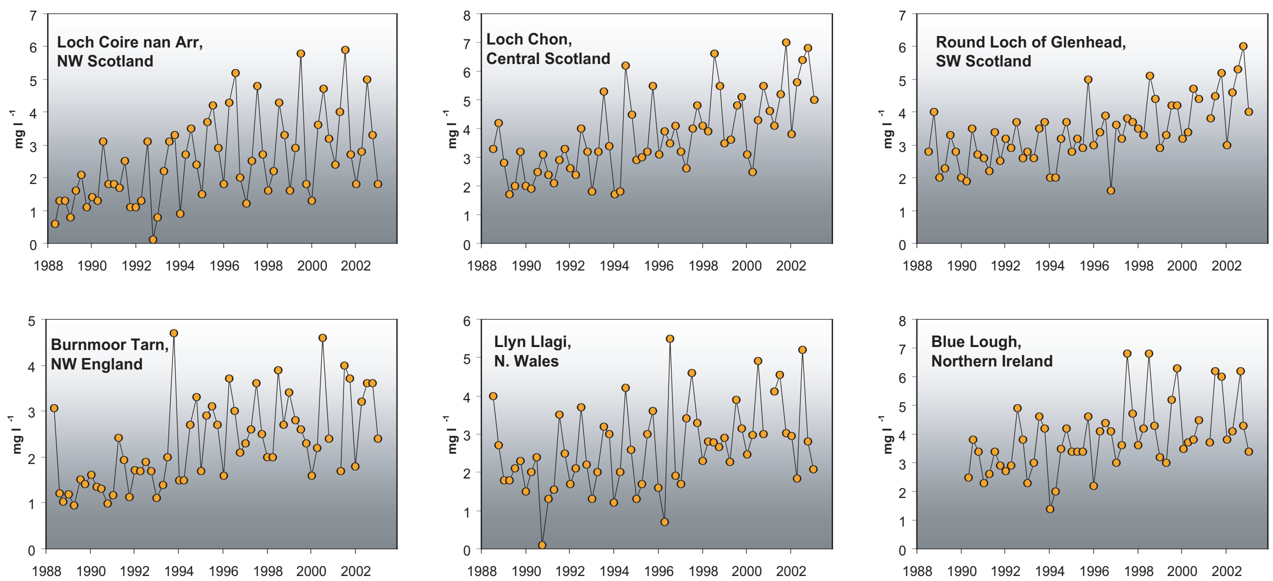

Figure 1: DOC time series for six AWMN lakes.

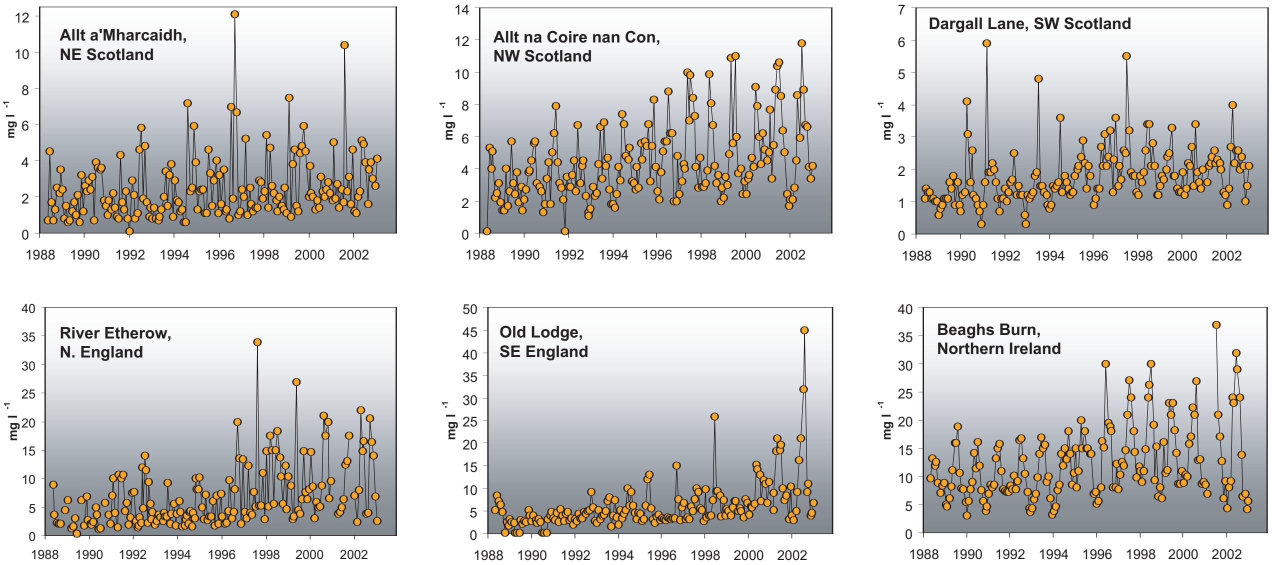

Figure 2: DOC time series for six AWMN streams.

controls on DOM solubility, physical processes transferring DOM from soils to streams and both biological and chemical removal or formation within the water column. Consequently, the range of potential environmental drivers of observed changes is extensive, and includes climatic factors (e.g. temperature, rainfall amount or timing), chemical factors (soil solution acidity or ionic strength) and local land-use factors (e.g. grazing, drainage, burning, afforestation). A full discussion of each of these potential drivers is provided in Evans *et al.* (2005). For the AWMN sites, it appears that some of these drivers can be discounted, at least for a proportion of sites. Firstly, although the UK uplands have been subject to substantial land-use changes in recent decades, including widespread afforestation, increased grazing levels in many areas, and the degradation or removal of drainage ditches in some peatlands (Emmett and Ferrier, this volume) there does not appear to be any one land-use change that could have affected all sites. For example, DOC trends have been comparable at paired moorland and afforested catchments, and amongst moorland catchments, land-use ranges from relatively high intensity sheep grazing to high altitude grouse-moorland, again with similar DOC trends at both extremes. Although drought-rewetting cycles have been identified as a possible driver of DOC trends (e.g. Watts *et al.*, 2001) and increasing winter/summer rainfall ratios were noted for a site in northern England by Burt *et al.* (1998), no clear evidence of changing hydrological patterns could be identified either from national rainfall indices, or from discharge and rainfall monitoring data at individual sites.

Based on the available data, the most plausible drivers of increasing DOC are considered to be rising temperatures and falling levels of acid deposition. Temperature levels have been more or less continuously elevated over the monitoring period relative to the preceding 30 years, on average by 0.65˚C. Sulphur deposition levels have fallen over the same period, particularly since 1995. Stepwise regression analysis (Evans *et al.*, 2005) indicates a correlation between DOC concentration and a temperature variable at 10 out of 11 lake sites. Of these, seven sites show a correlation with a $\geq$1 year mean temperature variable, suggesting that temperature contributed to long-term trend, while at the other three correlations are with <1 year temperature means only, and may simply reflect a seasonal correlation. As an indicator of changing acidification status, DOC increases were found to be correlated with decreases in either non-marine SO^4, or the sum of acid anions, at seven of the lakes, all located within areas of declining acid deposition. Correlations with pH were weaker, although this could be because increases in organic acidity have partially offset decreases in mineral acidity.

Overall, it is difficult to identify drivers of DOC increases on the basis of monitoring data alone, particularly since there have been clear changes in several potential drivers over the monitoring period. Although changes in temperature and S deposition appear the most likely causes of observed change, further process-based and experimental work is required to test these hypotheses. Additionally, given the uncertainties in the controlling processes, it is difficult to assess the wider significance of observed changes, in other words whether they represent i) part of a natural cycle; ii) a return to 'natural' conditions (e.g. with recovery from acidification); or iii) a shift towards conditions that have not previously occurred (e.g. due to climate change). In this respect, information on historic DOC change, for example through the development of palaeolimnological reconstruction techniques, would assist both in determining natural 'reference' conditions and in identifying drivers.

Regardless of mechanism, it is clear that DOC levels in UK upland waters have almost doubled since the late 1980s, representing perhaps the largest change in upland water quality over this period. The full consequences of this change have yet to be determined, but impacts are likely to be significant, including changes in aquatic flora and fauna in response to changing light, nutrient, energy and acidity levels; increased water treatment costs in peaty areas; and increased carbon (and associated metal) export from terrestrial stores to freshwater and marine systems.

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Nitrate and future achievement of "good ecological status" in upland waters

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Introduction

Upland waters in the UK provide some of our most unpolluted but sensitive ecosystems and often occur in remote areas of high conservation, amenity and aesthetic value. Despite their relative remoteness from the major sources of pollution associated with industry, urbanization and intensive agriculture in the lowlands, they are still vulnerable to the adverse effects of long-range air pollutants. One of the most insidious groups of pollutants comprises anthropogenic nitrogen (N) compounds derived mainly from fossil fuel combustion (generally oxidized forms; NO_x) and intensive agriculture (mainly reduced forms; NH_y). Because N is a key nutrient, its fate is tied up with biological cycles in both terrestrial and aquatic ecosystems, and obvious adverse effects may not occur for a long time under elevated deposition levels. Furthermore, biological cycles are sensitive to other disturbances such as climate change, so it may be very difficult to link deposition directly with adverse effects that can occur long after the onset of pollution and can be obscured by biological responses to other drivers of change.

The impacts of anthropogenic nitrogen deposition on upland waters and its relevance to the maintenance of "good ecological status" under the requirements of the EU Water Framework Directive are considered in terms of four key issues:

1. Nitrate and recovery from acidification
2. Nitrate as a nutrient in upland waters
3. Controls on nitrate leaching
4. Modelling future nitrate leaching

Nitrate and recovery from acidification

Historically, surface water acidification in the UK has been caused predominantly by anthropogenic sulphur (S) deposition. While elevated surface water nitrate concentrations linked to the deposition of oxidised and reduced nitrogen compounds have been recorded in several regions of the UK, the role of nitrate in acidification has always been secondary to that of sulphate in the great majority of sites. However, where nitrate leaching into surface waters does occur, there is an acidifying effect similar to that associated with sulphate leaching because of the accompanying hydrogen ions leached with the acid anions. A major difference between nitrate and sulphate leaching is that strong seasonal patterns are apparent in nitrate leaching because of biological controls in the terrestrial ecosystem on to which most deposition occurs. The links between nitrate, non-marine sulphate and pH on a seasonal basis at Easedale Tarn in the English Lake District are shown in Figure 1.

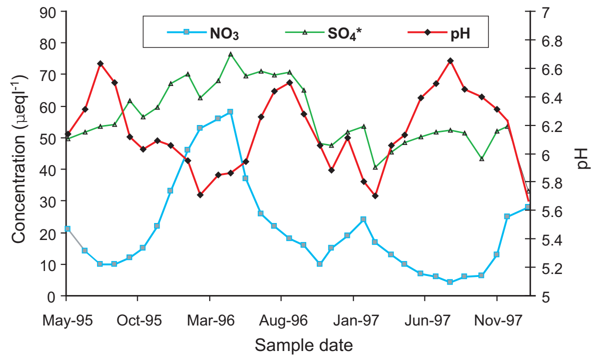

Figure 1: Seasonal variations in nitrate (NO_3), non-marine sulphate (SO_4*) and pH at Easedale Tarn in the English Lake District (concentrations in µeq l^{-1}).

Although S has historically been the dominant cause of acidification, data from the UK Acid Waters Monitoring Network (AWMN) clearly show that surface waters are responding rapidly to reductions in S deposition, and significant downward trends have been recorded in both S deposition and non-marine sulphate concentrations over the last decade (Cooper and Jenkins, 2003). No such trends have yet been observed in either the deposition of N compounds or surface water nitrate concentrations (Cooper, 2005), so the relative importance of nitrate has been increasing over the same period. If mean annual nitrate concentrations remain relatively constant over the next 10-20 years, then nitrate will become the most important acidifying anion in certain regions of the UK as non-marine sulphate concentrations continue to decline (Curtis *et al.*, 2005a).

On a simple extrapolation of current trends, AWMN sites where nitrate exceeds non-marine sulphate in 2010 (target year for implementation of EU National Emissions Ceiling Directive and UNECE Gothenburg Protocol) and 2016 (target year for achievement of "good ecological status" required by the EU Water Framework Directive) are shown in Figure 2. It can be seen that nitrate may prevent chemical and biological recovery in some acidified sites and result in a failure to achieve good ecological status. However, the period over which current trends may persist is unknown, given spatial and temporal changes in acid emissions in the future.

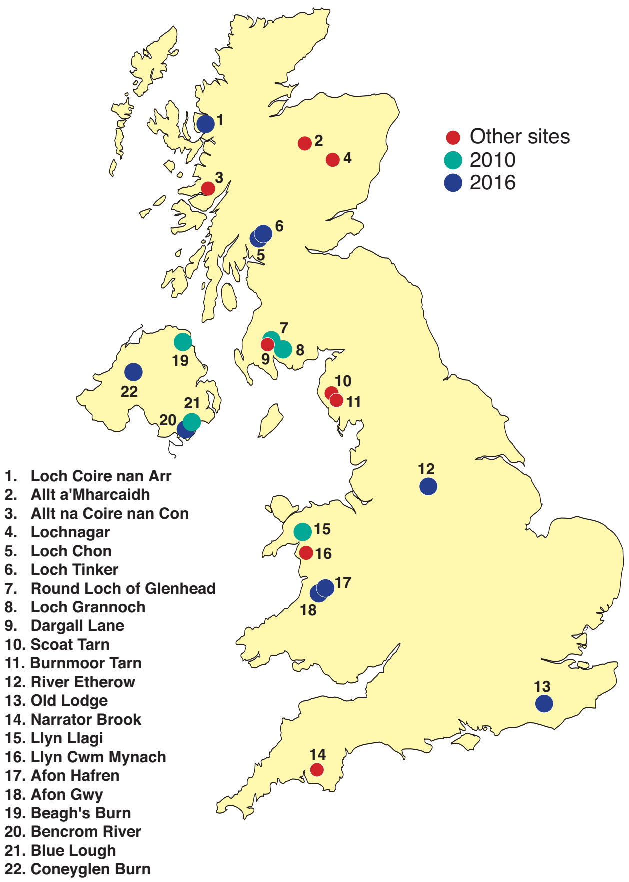

Figure 2: AWMN sites at which nitrate concentrations will exceed non-marine sulphate by 2010 (Gothenburg Protocol) and 2016 (target for WFD) if current chemical trends continue.

Furthermore, the process of N saturation may lead to increased nitrate leaching in future even under constant deposition levels (see below). A simple steady-state mass-balance model for S and N leaching, the First-order Acidity Balance (FAB) model (Posch *et al.*, 1997; Henriksen and Posch, 2001) has been applied to diverse freshwater datasets in the UK and indicates that at long-term steady-state, nitrate will greatly outweigh non-marine sulphate in contributing to acidification (Curtis *et al.*, 2005a). The timescale over which this change will occur is unknown.

Despite the uncertainties with future trends in sulphate and nitrate leaching, it is clear that N deposition is already contributing to acidification in some regions (Allott *et al.*, 1995; Curtis *et al.*, 2005a) and that it may overtake non-marine sulphate as the major agent of acidification in the next few years.

Nitrate as a nutrient in upland waters

Recent evidence (mostly in North America) has emerged suggesting that base poor upland/mountain aquatic ecosystems are demonstrating a response to atmospheric nitrogen deposition even at very low inputs (e.g. Findlay *et al.*, 1999; Baron *et al.*, 2000; Wolfe *et al.*, 2001; Fenn *et al.*, 2003; Sickman *et al.*, 2003), although some studies have been inconclusive (Burns, 2004). Unpublished research funded by NERC and the EU conducted by Monteith and Pla suggests an atmospheric pollution related link with recent biological change in Loch Coire Fionnaraich, north-west Scotland (Figure 3). This remote and relatively clean site has shown no evidence of an acidification response to S and N deposition. Preliminary results suggest that an observed change in the diatom record might therefore be linked to nutrient nitrogen enrichment, and similar trends appear to be seen in other remote areas of Europe (e.g. Pyrenees) as well as North America. Signals in the chrysophyte and zooplankton records are also suggested. Whether this is independent from or an interaction with climate change has not yet been ascertained. Furthermore, the interactions between acidification and eutrophication or oligotrophication are complex; for example, the aluminium released by acidification may inactivate P

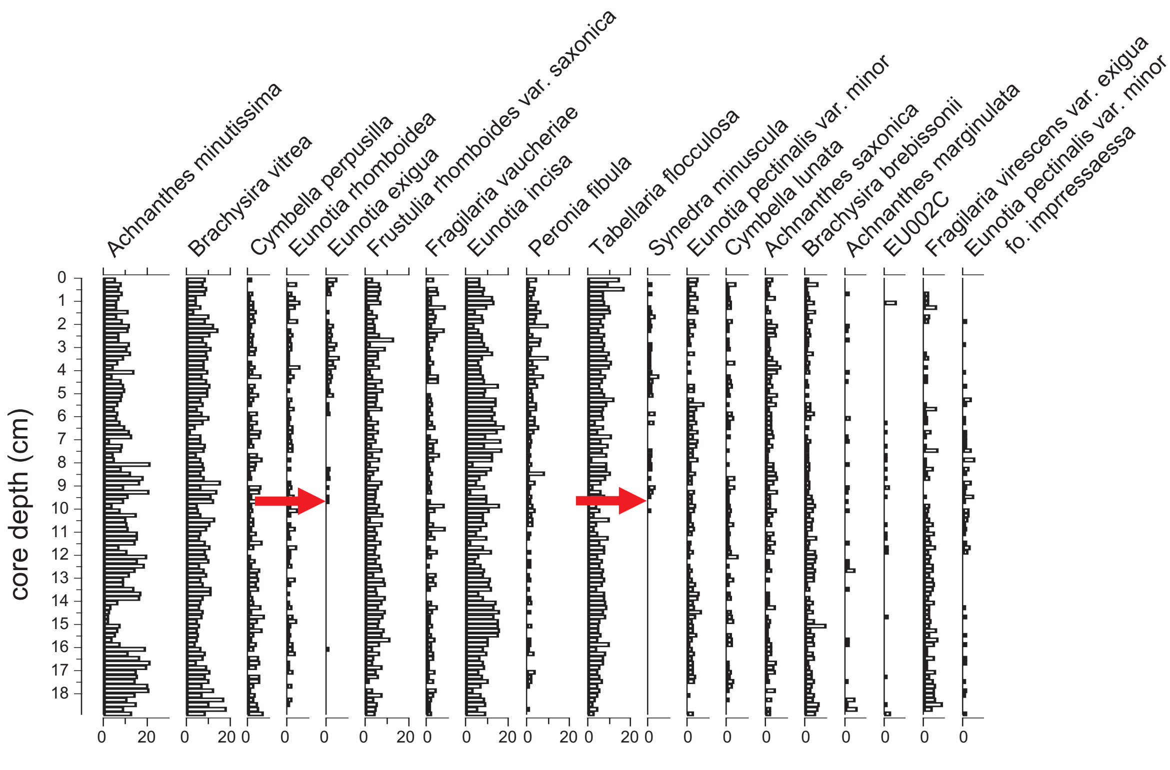

Figure 3: Diatom stratigraphy of Loch Coire Fionnaraich, north-west Scotland. Arrows indicate point of change.

through complexation (Kopáček *et al*., 2001), leading to P limitation. Other work in the UK uplands under the NERC Thematic Programme GANE (Global Nitrogen Enrichment) used phytoplankton bioassays to show that around three quarters of 30 study lakes are either N limited or co-limited by N and P (Maberly *et al*., 2002).

Hence for the first time there is direct evidence from the UK that there may be a widespread eutrophication effect of N deposition on upland waters, while the relative contribution of N deposition to eutrophication in the lowlands has never been properly quantified. Long-term shifts in diatom species composition in clean areas of north-west Scotland which mimic changes seen in other remote parts of the northern hemisphere could be related to changing N deposition, but the link is as yet unproven. A key issue for future research is therefore how N (as a eutrophier) is affecting the productivity and biodiversity of upland lakes in the UK. Palaeolimnology can provide a vital complementary approach to the assessment of current nutrient status through bioassays and chemical measurements.

Controls on nitrate leaching

While the importance of N as an agent of both acidification and possibly eutrophication has been demonstrated above, a major problem is understanding the fate of N deposition because most enters terrestrial biological cycles that are highly complex. In many parts of the UK, the deposition of total inorganic N (reduced plus oxidised compounds) is of comparable magnitude in equivalence terms to non-marine S deposition. The much lower observed concentrations of nitrate than non-marine sulphate reflect the importance of N as a nutrient in N-limited terrestrial ecosystems, with a very large proportion of N deposition being retained in the catchment soils and vegetation on to which it is deposited. N is rapidly taken up by vegetation or immobilised by microbial or abiotic processes in soils, as has been demonstrated in moorland systems in the UK uplands using the stable isotope ^{15}N (Curtis *et al*., 2005b). Biologically assimilated N is converted into organic compounds that may be very stable and be stored over long timescales in soil organic matter.

However, theoretical considerations suggest that there must be a finite capacity for terrestrial N retention. As N accumulates in the ecosystem, the N content of vegetation and soil organic matter increases and N saturation eventually occurs (Ågren and Bosatta, 1988; Aber *et al*., 1989). The carbon:nitrogen (C:N) ratio declines, potentially leading to enhanced mineralization, nitrification and ultimately nitrate leaching over timescales that are poorly understood but probably range from years to decades and beyond (Dise and Wright, 1995; Gundersen *et al*., 1998; Curtis *et al*., 2004). The C:N ratio has been used as a key control on nitrate leaching in dynamic acidification models (see below).

There is also evidence that where nitrate leaching occurs at present in the UK uplands, hydrological factors may be at least partly responsible, i.e. some observed nitrate has been transported directly from deposition to surface waters unchanged (Curtis *et al*., 2005b; Evans *et al*., 2004). Unlike microbial sources of nitrate controlled by C:N ratio, this atmospheric proportion of leached nitrate should respond very rapidly to changes in deposition at a rate determined only by the hydrological residence time of a catchment.

A third leaching pathway is a combination of the above two processes, whereby reduced biological demand for N in enriched terrestrial ecosystems allows the leaching of a proportion of deposition inputs at a rate that may be determined partly by hydrological flow rates. In reality, leached nitrate is probably derived from a combination of all these three pathways in systems with elevated N deposition. New research initiatives aim to determine the source of leached nitrate using the dual isotope approach (Durka *et al*., 1994; Spoelstra *et al*., 2001; Williard *et al*., 2001), which relies on the very different isotopic signatures (in terms of ^{18}O and ^{15}N) of atmospheric and microbial nitrate.

A final consideration is that the importance of temperature, sunlight and moisture to biological processes that influence nitrate retention means that climate change is likely to exert a major influence on both the degree and timescale of nitrate leaching. Decadal scale variations in nitrate leaching have been linked to the North Atlantic Oscillation (Monteith *et al*., 2000) and similar climatic factors may have accounted for some of the earlier reports of evidence for N saturation in rising surface water nitrate concentrations that have subsequently declined (Goodale *et al*., 2003). The leaching of organic compounds is also tightly linked to biological processes that are influenced by climate. Organic N leaching has been suggested as a mechanism for the maintenance of N limitation in low-deposition alpine ecosystems – the so-called "leaky faucet" hypothesis of Hood *et al*. (2003). Widespread trends of increasing organic carbon leaching have been linked to both climatic factors and recovery from acidification (Freeman *et al*., 2001), but potential interactions with biological N cycles are not well understood. The role of climate therefore provides a further challenge for understanding and modelling N saturation and leaching processes.

Modelling future nitrate leaching

Early critical loads models for total acidity took a simplistic approach to N deposition and assuming no knowledge of N retention and leaching processes, simply used the measured nitrate leaching flux to represent the proportion of N deposition that is leached and to add to the calculated critical load exceedance flux for S (Kämäri *et al*., 1992). This approach did not allow for testing scenarios of changing N deposition as the resultant nitrate leaching flux could not be calculated, so the mass-balance model FAB was designed specifically for this purpose (Posch *et al*., 1997; Henriksen and Posch, 2001). The FAB model uses default long-term, steady-state sink terms for the retention of N deposition based on immobilisation, denitrification, in-lake retention and biomass removal processes. Observed leaching bears little resemblance to that predicted by the FAB mass-balance, presumably because of a lack of steady-state, i.e. because terrestrial ecosystems are still accumulating N (Curtis *et al*., 1998). There are also too few data to improve on the literature based default sink terms used for site-specific model applications. The FAB model is therefore not intended to model temporal changes in nitrate leaching and only provides an estimate for a long-term steady-state situation that may be decades or centuries in the future.

In order to model the shorter-term changes in nitrate leaching under existing or future deposition loads, dynamic models are required. The key dynamic model used in acidification studies in the UK is MAGIC, which has in recent years been adapted to account for N dynamics (Ferrier *et al.*, 1995; Cosby *et al.*, 2001). The current formulation uses a linear interpolation of nitrate leaching between upper (no leaching) and lower (100% leaching) threshold values of soil C:N ratio. This approach was based initially on datasets and theoretical considerations from European forest ecosystems (Gundersen *et al.*, 1998) that may not be generally transferable to the UK uplands, but recent data from a small number of British moorland systems suggest that the approach may be justified in some regions (Curtis *et al.*, 2004).

Given the three potential nitrate leaching pathways described above, there is an evident need for more sophisticated approaches to modelling nitrate leaching under different deposition loads. Recent attempts range from the purely empirical models using catchment characteristics (Helliwell *et al.*, in prep) to more process-based approaches that define nitrate leaching zones based on catchment slope and soil type (Evans *et al.*, 2004). The catchment soil active carbon pool has also been proposed as a key factor for modelling nitrate leaching (Evans *et al.*, in press). Future modelling developments will inevitably have to account for the relative proportions of hydrologically and biologically controlled nitrate leaching and identify the most important catchment-specific factors. Isotopic tracer experiments may have a key role to play in unravelling the dominant leaching pathways for nitrate. Only with greater understanding of these pathways will we be able to predict the likely impacts of N deposition on the achievement of "good ecological status" in upland waters over the long term.

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Heavy metals in British upland waters

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Introduction

Aquatic life in upland waters may be subject to toxicity by heavy metals[1], following increased inputs as a result of human activities. Laboratory experiments have shown that dissolved heavy metals can exert a variety of toxic effects on aquatic organisms, including impairment of reproduction and mortality (Mance, 1987). Here, we report on recent and ongoing work aimed at understanding and predicting the biogeochemistry and ecotoxicity of nickel, copper, zinc, cadmium and lead. We do not consider mercury, although studies of that metal are also being performed (Yang *et al*., 2002), and more work will be required in the future. Neither do we consider the "light metal" aluminium, which has received much attention in acidification research, due to its mobilisation from soil and rock minerals by acid percolating waters. The toxic effects of aluminium are exerted through similar mechanisms to those of heavy metals, and ultimately a full assessment of the ecotoxicological aspects of upland water quality should include the combined effects of all metals, and also those of pH.

Sources of heavy metals to upland waters

The upland waters of the UK receive metals in catchment drainage water and from direct atmospheric deposition, the latter route being much more important for lakes than streams. Sources of metals to the catchment are atmospheric deposition and weathering, both natural and accelerated by mining activities. By combining recent direct measurements, historical information and evidence from peat cores and lake sediments, chronologies of metal deposition can be constructed, as exemplified in Figure 1.

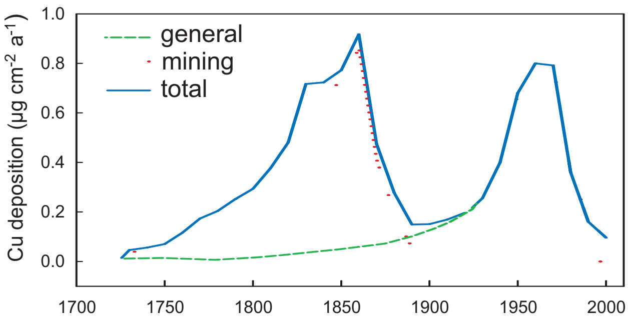

Figure 1: Estimated atmospheric deposition of copper in the S. Pennines, based upon UK production records (Schmitz, 1979), peat core data (Livett, 1988), direct measurements at rural UK sites since 1972 (Baker, 1999), and UK peat core and lake sediment data. The shape of the general trend is thought to apply to all regions of the UK, whereas the mining trend is localized.

To reach the surface water from the catchment, metals must be transported within the soil-rock system, where several factors determine the extent and rate of the metal movement. The principal control on transport is sorption to the natural organic matter and mineral surfaces of the soil solids. For cationic metals, such as Cu^{2+} and Cd^{2+}, sorption strength increases with pH, as competition by H^+ diminishes. Thus, metals tend to be more mobile at lower pH. Complexation by dissolved organic matter (DOM) counters the sorption process, and favours metal mobility. Anionic metals are controlled mainly by the amphoteric oxides of Al and Fe, and tend to be most strongly sorbed at low pH. Other significant processes include the incorporation of metals into mineral matrices, solubility control by carbonate precipitation and dissolution, the "burial" of metals in the anaerobic zones of peats, and the harvest removal of plants, especially trees, that have taken metal up from the soil.

Critical loads and critical limits

Within the UNECE Convention on Long-Range Transboundary Air Pollution, work is in progress to derive a critical loads, effects-based methodology for heavy metals. The basis of the approach is to define critical limits for soils and waters, i.e. the maximum concentrations of heavy metals that can be tolerated without causing significant ecosystem damage, and then to calculate the metal loadings that correspond to those concentrations, at steady-state. The approach then involves GIS-based mapping, firstly to ascertain the extent to which current deposition exceeds the critical load at different locations, and secondly to assess the current states of soils and waters with respect to heavy metals. The present focus in the UK critical loads work is on the cationic metals Ni, Cu, Zn, Cd and Pb.

As part of this effort, Lofts *et al*., (2004) analysed soil toxicity data for Cu, Zn, Cd and Pb, and showed that chronic toxic

[1] The term "heavy metals" is widely used but poorly defined (Duffus, 2002, Hodson, 2004). In the present context, the metals of interest fall within Groups 3-15 and Periods 4-6 of the Periodic Table (http://www.webelements.com), are known or suspected to be toxic at concentrations attainable in the natural environment, and are anthropogenic contaminants. The elements of most concern are V, Cr, Ni, Cu, Zn, Cd, As, Ag, Hg and Pb. Some of these metals are essential elements, others are toxic at sufficiently high concentrations, some exhibit both attributes.

effects could be related to (a) the concentration of free metal ion in the soil solution, and (b) the soil solution pH. The results are consistent with a toxicity-inducing interaction of the metal at a receptor possessed by the organism, and with competition by other cations (notably H^+); such interactions are central to the Free Ion Activity or Biotic Ligand Model (Paquin *et al.*, 2000). Thus, the higher is the proton concentration, or the lower is pH, the higher is the free metal ion concentration required to cause toxicity. Critical limit functions of the following type are obtained;

$$\log [M^{2+}]_{crit} = \boldsymbol{a}\ \mathrm{pH} + \boldsymbol{b} \qquad \textbf{(1)}$$

where $[M^{2+}]_{crit}$ is the critical free ion concentration, and ***a*** and ***b*** are constants (both negative).

Whereas there are many chronic toxicity data for a wide range of species in soils, the data for aquatic organisms are quite sparse. There are some indications that the soil critical limit functions may also apply to freshwaters, but further work is needed to confirm this proposition. Therefore to assess possible toxic effects of heavy metals in surface waters, the current recommendation from UNECE is to use "conventional" critical limits, expressed simply as total dissolved metal. The problem with such an approach is that no account is taken of the protective effects of other cations, nor of the influence of complexation by dissolved organic matter, which can significantly influence the free ion concentration.

Table 1 compares observed total dissolved concentrations of Cu, Zn, Cd and Pb with critical limit values, for a number of UK upland surface waters. The "conventional" limits are denoted by CL1, and do not vary with water chemistry. The values used here are from the Netherlands (van de Plassche *et al.*, 1997), since neither the UK nor the UNECE currently has recommended values for all four metals. The CL2 values correspond to the free-ion critical functions and depend upon water chemistry, firstly because of the pH-dependence of the free-ion limit (Equation 1), and secondly through the interactions of the free ion with other solutes. The total dissolved concentration is calculated, using the WHAM chemical speciation model (Tipping, 1994; 1998), by taking account of the interactions of the free metal ion with other solutes, notably OH^-, HCO_3^- and dissolved organic matter.

The data in Table 1 show that, for all four metals, current concentrations are nearly all within a factor of ten of the critical limits. Of the 56 cases (7 sites x 4 metals x 2 critical limits) considered, there are 9 exceedances, 7 of which are found for CL1 and 2 for CL2. Exceedances are found for both acid and alkaline waters, and for waters with both low and high DOC concentrations. Thus, we cannot generalise about the adverse effects of heavy metals simply in terms of water chemistry; the history of atmospheric metal deposition and/or mobilisation of catchment sources must also be taken into account. Overall, the data suggest that heavy metals are causing some ecosystem damage in upland waters.

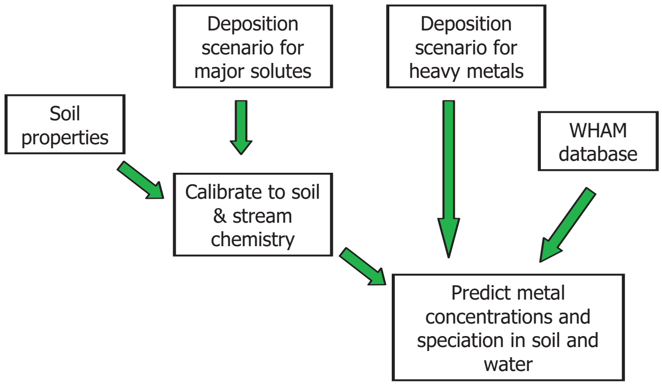

Figure 2: Application of CHUM-AM in the dynamic modelling of catchment chemistry. The model is first configured to simulate long-term acidification and recovery. Then metal behaviour is predicted, mainly on the basis of the sorption of metals by solid-phase natural organic matter, taking into account competition by the major cations (H^+, Mg^{2+}, Al^{3+}, Ca^{2+}) and complexation by dissolved organic matter.

Dynamic modelling with CHUM

The critical loads approach outlined above is based on metal concentrations at steady-state, but it is also important to

Stream	pH	DOC mg l^{-1}	Cu			Zn			Cd			Pb		
			obs	CL1	CL2	obs	CL1	CL2	obs	CL1	CL2	obs	CL1	CL2
Old Lodge stream Ashdown Forest	5.0	8.9	1.7	1.1	4.7	18	6.6	46	0.10	0.34	1.5	1.9	11	2.5
Gaitscale Gill Lake District	5.1	0.7	0.2	1.1	0.7	3.9	6.6	39	0.07	0.34	1.3	0.4	11	1.2
Pool X N Pennines	5.1	20.0	1.7	1.1	18	30	6.6	62	0.13	0.34	2.2	4.5	11	8.4
Lochnagar Cairngorms	5.4	1.4	0.1	1.1	0.9	4.9	6.6	33	0.04	0.34	1.1	0.6	11	0.8
River Etherow S Pennines	5.6	9.5	0.8	1.1	5.2	13	6.6	39	0.10	0.34	1.3	1.9	11	2.1
Gais Gill Howgill Fells	7.6	5.0	0.3	1.1	1.5	1.7	6.6	16	0.05	0.34	0.55	0.1	11	4.1
Cote Gill Pennines	8.1	2.2	0.8	1.1	0.4	38	6.6	11	0.20	0.34	0.34	3.4	11	1.6

Table 1: Some observed dissolved metal concentrations (µg l^{-1}) in upland surface waters, and comparisons with critical limits CL1 and CL2 (see text). Critical limit exceedances are indicated by underlining.

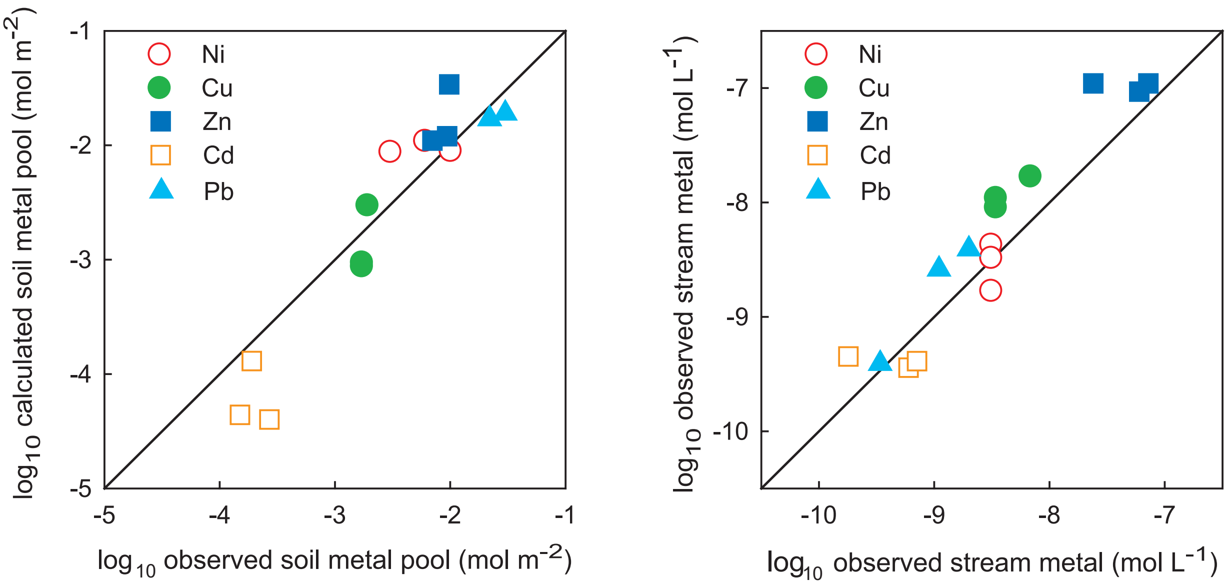

Figure 3: Summary of modelling results for three sub-catchments in the upper Duddon valley (Lake District). The stream waters had pH values of 5, 5.5 and 7 in 1998-9. By taking into account major-component and heavy metal chemistry and using a common deposition scenario from 1400 to the present, CHUM-AM is able approximately to simulate both soil pools and average stream water concentrations of heavy metals.

understand the timescales of change in metal concentrations, so that we can assess the time taken for steady state levels to be reached, from initially contaminated or uncontaminated states. To address this issue, we are carrying out dynamic modelling of a number of UK upland catchments, using a variant (CHUM-AM) of the Chemistry of the Uplands Model (Tipping, 1996). The key interactions in the model are between metals and soil organic matter. Other processes considered include weathering and interactions with DOM. By using the comprehensive chemical speciation model WHAM/Model VI (Tipping, 1994; 1998) within CHUM, it is possible to take into account the effects of changes in soil and water acidification status (Figure 2).

The dynamic modelling of heavy metals is hampered by the absence of observations covering a lengthy time period. Thus, while we have data on acidification covering several decades (e.g. NEGTAP, 2001), reliable metal determinations are only available for short periods in recent years (Neal *et al.*, 1997; Lawlor and Tipping, 2003), making it difficult to test the dynamic model by comparing time-series observations and simulations. Therefore, our approach has been to run the model and compare its simultaneous predictions of the soil pools and stream water concentrations of several heavy metals. The reasonable agreement obtained between observation and simulation (Figure 3) supports the assumptions about the key processes and historical metal deposition scenarios.

Figure 4 shows simulated variations in pH, dissolved Zn and dissolved Pb in the stream water of Gaitscale Gill in the Lake District, over the period 1500 to 2500, for two scenarios of

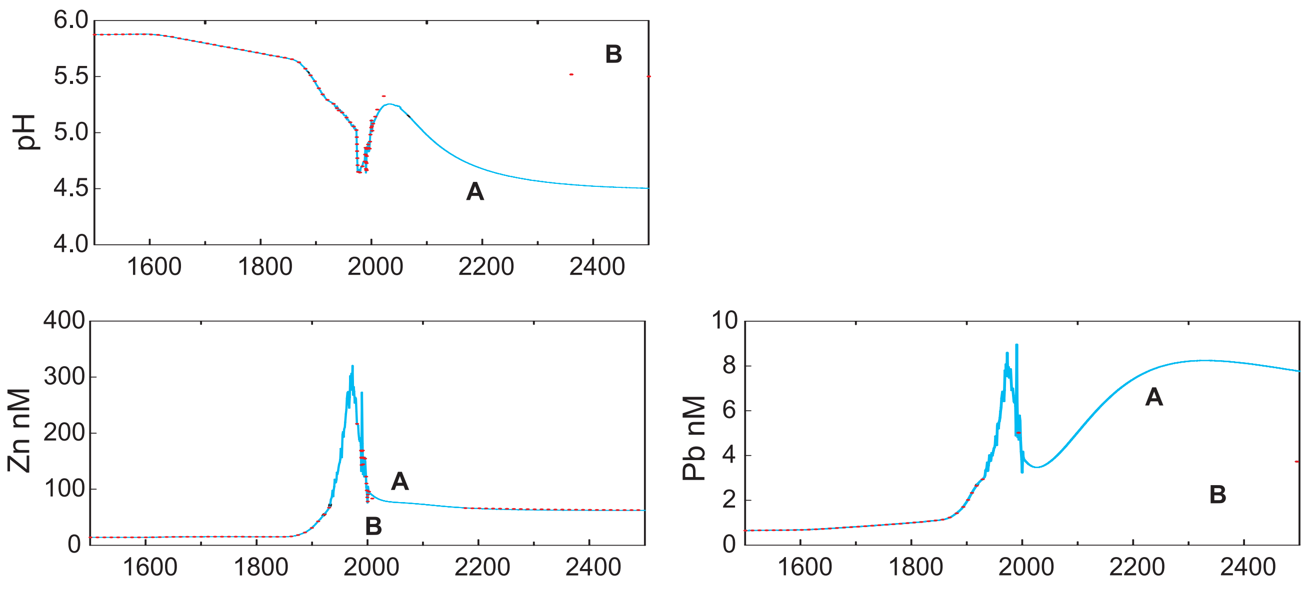

Figure 4: Simulations of stream water pH, dissolved Zn and dissolved Pb for Gaitscale Gill AD 1500 - 2500. Scenario A: N, S and heavy metals in deposition held constant from 2002 onwards; Scenario B: N and S deposition halved by 2022, metal deposition kept at 2002 value.

future acidification. The historical metal deposition at this site corresponds to the "general" shape shown in Figure 1. The stream was substantially acidified from 1800 to the late 20th century, and is now recovering because S deposition has declined.

Whether the recovery is sustained will depend upon future N deposition, the predicted re-acidification in Scenario A being due to nitrogen saturation and increased nitrate leaching. The concentrations of both Zn and Pb in the stream are calculated to have increased during the 20th century, and to have peaked in around 1970, at the time when stream acidity was at its highest. The concentration of Zn has now declined appreciably, and is forecast not to rise in the future, assuming deposition does not increase.

According to the model, much of the Zn that has been deposited on the catchment has leached out, because Zn is sorbed relatively weakly to the soil, especially at low pH. Thus, acidification promoted mobilisation of the metal at the time of its greatest input. The stream water Pb concentration also passed through a maximum at around 1970, because of the acid conditions. However, of the Pb that has been deposited on the catchment, more than 95% is still in the catchment soil, and the metal is continuing to accumulate, and to pass slowly downwards through the soil and thence to the stream. Thus, there is a large store of Pb available for mobilisation if re-acidification occurs, under Scenario A of Figure 4. Simulations of other metals show that Ni and Cd behave like Zn, while the retention and transport of Cu are closer to those of Pb.

The strong retention of Pb by the Gaitscale Gill soil means that the timescale of change for this metal is very long (Figure 4). However, the rate of change of Pb concentration and flux is substantially faster in a Pennine catchment that we are studying, where complexation by DOM hastens leaching. This difference underlines the difficulty of generalisation about metal behaviour. Our current project will finally report on dynamic modelling at 10 UK upland catchments, and it may then be possible to draw some wider conclusions about the past, present and future of heavy metals in British upland waters.

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The future of Britain's upland waters: the challenges for modelling

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Introduction

British upland waters have, since the early nineteenth century, been considered to be clean and plentiful and as such have been harnessed in many regions to provide public drinking water supplies. More recently, the recreational value of these pristine upland waters has been emphasized through the growth of tourism and fishing enterprises. It was not until the early 1970s that serious consideration was given to the possibility that these waters might be suffering pollution from atmospheric sources with dire ecological consequences, most notably through acidification. These considerations have over several decades led to the use of upland waters as a testing ground for hydrological models and, more recently, as a focus for development of chemical models to predict the future response of surface waters to changes in atmospheric emissions of acidifying compounds. It is the latter that is taken as a case study here to demonstrate how far we can predict the future acidification status of upland waters and the remaining challenges.

Recovery from acidification

Surface water acidification through the deposition of acidifying sulphur and nitrogen compounds emitted to the atmosphere, largely through fossil fuel combustion, is the most significant problem that the UK uplands has seen. The increased acidity caused changes in freshwater invertebrate and diatom communities and reduced salmonid fish populations in many regions with acid-sensitive bedrock geology. As international agreements were instigated and developed to combat this problem, so the emphasis moved towards answering the question of the degree of recovery from acidification that might be expected in response to a given emission reduction (NEGTAP, 2001). This prompted the development of dynamic hydro-chemical models.

Model requirements

The challenge facing modellers with respect to surface water acidification was to produce a model that could simulate the time development of changes in water chemistry (dynamic model) at the most appropriate spatial and temporal scales, including the key processes and that could, with reasonable data requirement, be applied widely. In the UK, the MAGIC model (Figure 1), has been tested and developed in fulfilment of these criteria (Cosby *et al.*, 2001).

Since water chemistry at any point in a drainage system reflects an integration of the whole catchment upstream, the most appropriate spatial scale for the model is the whole catchment. Recognizing the variability in catchment soils and vegetation, however, the catchment characteristics are "lumped" in the model into a single homogeneous "box". Temporally, the onset and recovery from acidification takes several decades and since it is primarily the long-term trend in mean chemistry status that is of interest rather than the short-term variation associated with changing rainfall inputs, the model runs on an annual time-step.

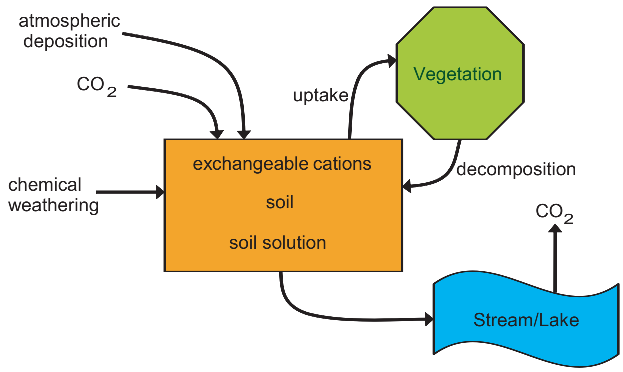

Figure 1: A schematic representation of the key flows and stores represented in the MAGIC model.

The model simulates soil solution chemistry and surface water chemistry to predict the monthly and annual average concentrations of the major ions in lakes and streams. MAGIC represents the catchment with aggregated, uniform soil compartments (one or two) and a surface water compartment that can be either a lake or a stream. MAGIC consists of (1) a section in which the concentrations of major ions are assumed to be governed by simultaneous reactions involving sulphate adsorption, cation exchange, dissolution-precipitation-speciation of aluminium and dissolution-speciation of inorganic and organic carbon, and (2) a mass balance section in which the flux of major ions to and from the soil is assumed to be controlled by atmospheric inputs, chemical weathering inputs, net uptake in biomass and losses to runoff. At the heart of MAGIC is the size of the pool of exchangeable base cations in the soil. As the fluxes to and from this pool change over time owing to changes in atmospheric deposition, the chemical equilibria between soil and soil solution shift to give changes in surface water chemistry. The degree and rate of change in surface water

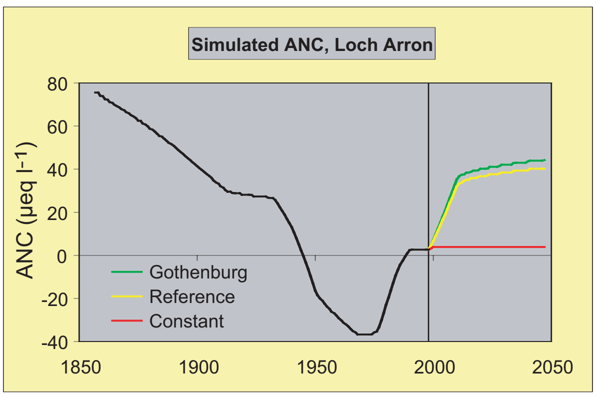

Figure 2: Simulated ANC at Loch Arron, 1850 - 2000 and future predictions under 3 emission reduction scenarios.

acidity thus depend both on flux factors and the inherent characteristics of the affected soils.

Future acidification status

When calibrated at an individual site (Jenkins *et al.*, 1997), MAGIC can reconstruct surface water chemistry to pre-industrial background conditions and predict forward under any scenario of change in atmospheric deposition (Figure 2). The model has been tested against long-term monitoring data in Europe and North America and has been shown to capture observed trends towards recovery from acidification (Figure 3). In the UK, MAGIC has now been calibrated to c.350 sites in seven acid-sensitive regions to explore the predicted response to acid deposition scenarios (Evans *et al.*, 2001).

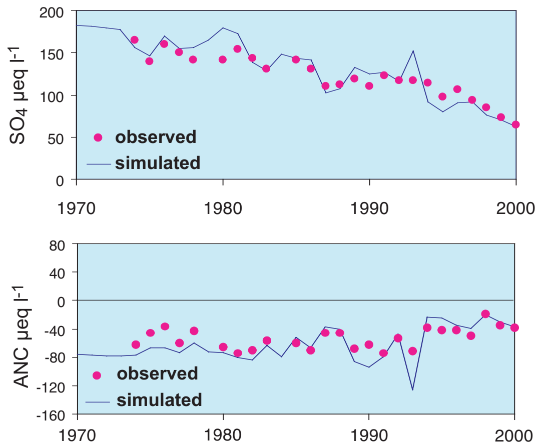

Figure 3: MAGIC model simulation and observed water chemistry at Birkenes, S. Norway, 1973 - 2000.

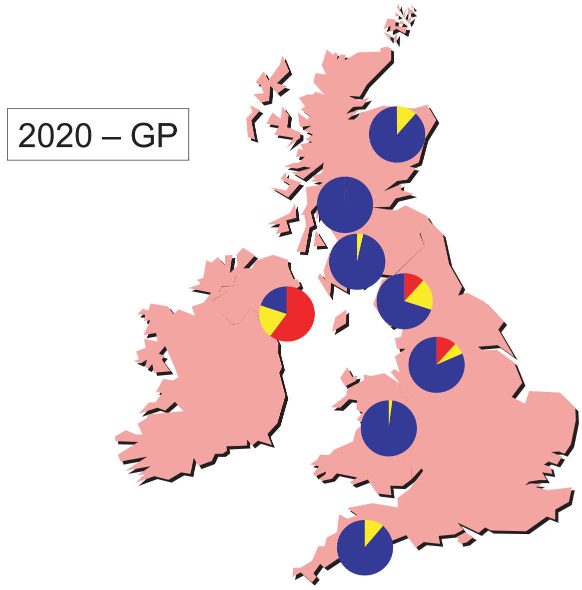

Figure 4: Regional ANC predictions in 2020 as percentages in 3 biologically relevant ANC classes in response to the Gothenburg Protocol. Red, ANC<0 (viable brown trout population unlikely), Yellow, ANC 0-20 and Blue, ANC>20 (healthy brown trout population likely).

Estimates of deposition across the UK in response to the agreed emission reductions under the Gothenburg Protocol have been calculated using the FRAME model. These regional deposition loadings for 2010 are assumed to be reached following a linear decline from the present level of deposition and to remain at that level thereafter. This is used as input to drive the model to make predictions. Predictions for 2020 indicate that significant acidification problems remain in the Mourne Mountains (Northern Ireland), the South Pennines (north-west England) and Lake District (north-west England) where ANC in c.15% - 50% of lakes remains below zero (Figure 4). In all other regions ANC is greater than zero although some regions are predicted to have a significant percentage of lakes with ANC below 20 μeq l^{-1} (Curtis and Simpson, 2004).

Immediate remaining challenges

Three key challenges remain with respect to modelling surface water acidification; specification of the processes controlling nitrogen dynamics, assessment of the potential impact of future climate change and application of the models in an appropriate format for use in future emission reduction negotiations.

With respect to N, most upland surface waters in the UK have low concentrations of nitrate (NO_3^-). It is widely held, however, that as N deposition continues in the future and the soil pool of N becomes enriched relative to available C, then immobilization of atmospheric N will be reduced resulting in increased leaching of NO_3^- (Jenkins and Cullen, 2001). This may offset the recovery from reduced S deposition in the longer term (Figure 5). This process, however, remains unobserved in time-series data from monitored catchments.

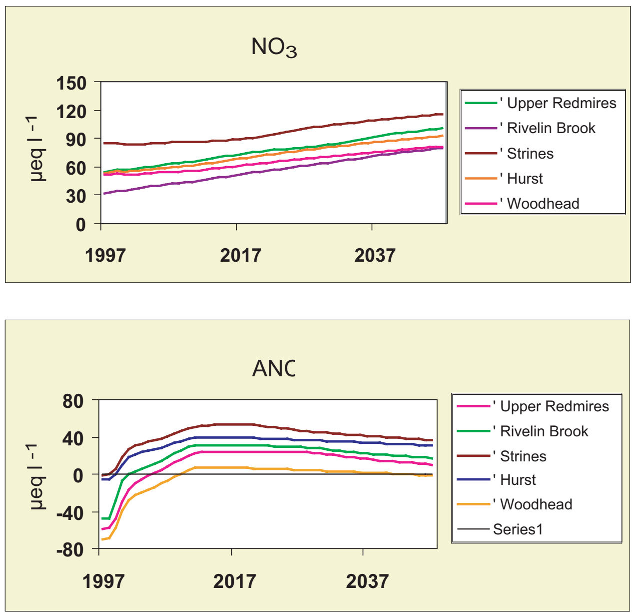

Figure 5: MAGIC simulations at 5 sites in the S. Pennines showing the impact of N saturation on ANC recovery.

The impact of future climate change is expected to lead to generally increased mean annual temperatures across most of the UK. This increase in temperature is likely to increase the mineralization of organic matter and further promote losses of inorganic N (Wright and Jenkins, 2001). This was clearly demonstrated experimentally by the CLIMEX project (Figure 6). This increased NO_3^- will have the effect of slowing recovery from acidification in response to reduced deposition, but is likely to be only a transient response.

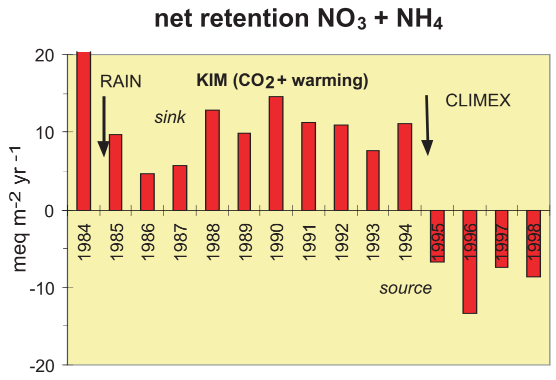

Figure 6: Net catchment fluxes of total N at the CLIMEX experiment in S. Norway. For the 4 years of experimental warming, the catchment became a net source of N rather than a net sink.

Application of dynamic models within the UN Convention requires the development of an approach which is consistent with that of critical loads. It is proposed that target load functions are calculated for specified years rather than on an equilibrium timescale (as in the critical load function). These target loads represent the S and N deposition that must be achieved in a specified time to achieve a specified target surface water chemistry in a given year (Jenkins *et al*., 2003).

Wider challenges

Beyond acidification, the upland waters of the UK are increasingly identified as being susceptible to other atmospheric derived pollutants, including heavy metals (especially mercury) and persistent organic compounds. Models for these pollutants are under construction and will require increased effort in data collection to adequately test and apply them. In addition, biological responses require that models account for short-term drivers of change, notably episodes of high and low flow which occur at very short timescales. Since hydrochemical pathways become increasingly important at short time steps (days or less) it is likely that more distributed catchment models will be required to tackle such issues. Such models require detailed description and understanding of hydrological processes and this remains as a significant challenge for the future.

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The future of Britain's upland waters: a view from Norway

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Introduction

Upland waters in Norway occur across 70% of the country and, as in the UK, they are an important natural resource for tourism, hunting, fishing and nature protection in national parks and nature reserves. They are also sensitive to environmental change caused by long-range transported air pollutants (acids, metals and persistent organic pollutants), land-use change and global scale drivers such as climate change.

Fully 30% of Norway's freshwaters have been affected by acidification (Overrein *et al*., 1980). Sulphur deposition peaked around 1970 while total nitrogen deposition peaked around 1990 (Figure 1). The resulting damage to fish populations, assessed with critical loads models, reached its widest extent around the late 1980s (Hesthagen *et al*., 1999). Although recovery is in full swing following international measures to reduce sulphur and nitrogen deposition (Skjelkvåle *et al*., 2001), critical loads in many sites will still be exceeded following full implementation of the Gothenburg Protocol to the UN/ECE Convention on Long-Range Transboundary Air Pollution (CLRTAP) (UNECE, 2002).

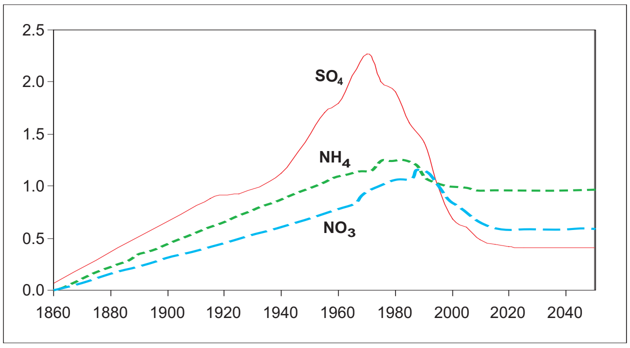

Figure 1: Reconstructed and predicted patterns in acid deposition in Norway. Deposition amounts are scaled relative to the year 1995.

In terms of upland waters, Norway has traditionally looked to the UK, which has a similar natural environment, similar pollution problems – and is where some of the pollution in Norway comes from! The close links between upland waters issues in the UK and Norway can be illustrated by considering a short history of acid deposition.

A short history of acid deposition in Norway and the UK

1900-1980: the Dark Ages

1900s: Populations of salmon and trout began to disappear in Norway.

1950s: A British lord cancelled his lease on salmon rights at the Tovdal River.

1960s: Acid rain was identified as the cause of acidification and damage to fish populations by Eville Gorham in the UK.

1970s: Empirical dose/response links were established between anthropogenic emissions, acid deposition, water chemistry and biological response.

1980-2000: the Age of Enlightenment

1979: Formal co-operation was established between Norway and the UK with a survey of acidified lochs in Galloway.

1984-89: The Royal Society co-ordinated the Surface Water Acidification Programme (SWAP), involving scientists from the UK, Norway and Sweden.

1985: Two famous ladies (Gro Harlem Bruntland and Margaret Thatcher) met in Oslo and discussed acid rain.

1985: The first of many EU research projects on acidification started, with UK and Scandinavian partners.

1990s: Parallel and co-ordinated monitoring was carried out at both national and international levels (e.g. ICP Waters and the 1995 Nordic Lake Survey which included Scotland and Wales).

1990s: Critical loads of acidity for surface waters were widely used in the UK and Scandinavia.

1990s: The first signs of recovery from acidification were recorded.

2000 onwards – the Post-modern Era

The new focus is on recovery in chemistry and biology, with many questions:

- How much recovery has occurred, where and when did it occur?
- What are the confounding factors?
- What revisions of LRTAP protocols may be needed?

Attention is now focusing on new threats:

- How does global change affect upland waters?
- What are the impacts of other air pollutants like heavy metals and POPs?

Recovery of surface waters from acidification

While critical loads models have been used to assess the spatial extent of potential acidification and recovery under different deposition levels in Norway as in the UK, the dynamic acidification model MAGIC has also been used in both countries to determine the timescales over which the damage occurred (Figure 2) and chemical recovery may be expected (Jenkins *et al*., 2003).

Predictions with the MAGIC model indicate that even after the implementation of the Gothenburg Protocol, a significant fraction of Norwegian freshwaters will still be too acidic to support fish populations. In this respect, upland areas in Norway are quite similar to those in the UK.

New threats and scientific challenges

While recovery from acidification in response to reductions in acid emissions is under way, new confounding factors have become apparent that may delay or prevent recovery in some areas. Future climate change will have unknown impacts but may affect the retention and loss of nitrogen. Again, common interests in the UK and Norway are apparent in large-scale

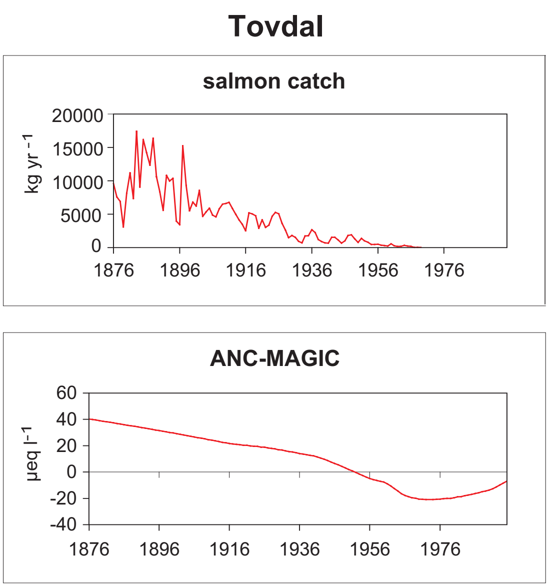

Figure 2: Historical salmon catch and reconstructed acid neutralizing capacity (ANC) using MAGIC in the Tovdal River, Norway (Kroglund *at al.*, 2002).

experiments such as CLIMEX in Norway (van Breemen *et al.*, 1998) and VULCAN in the UK (www.vulcanproject.com), which show that with increased temperature, nitrogen is mobilised from soil organic matter and lost to runoff and ultimately to coastal marine areas. Norwegian scientists have closely followed UK research such as the NERC Global Nitrogen Enrichment (GANE) thematic programme and the DEFRA funded Critical Loads of Acidity and Metals (CLAM) programme. Joint workshops on nitrogen issues with UK and Norwegian scientists have been held.

Data from the recent EU EMERGE programme which included both Norwegian and UK scientists have shown that pollution by heavy metals and persistent organic pollutants may be more severe than expected in remote upland waters. Particularly high levels of mercury were found in fish from Lake Ferguson in Greenland and Lochnagar in Scotland (Figure 3). The four Norwegian lakes on the left of Figure 3 show some of the lowest values.

Afforestation and abandonment of mountain farms have been the dominant forms of land-use change in upland areas of Norway during the past several decades. Effects of afforestation, well documented from numerous studies in the UK, are also apparent in Norway, albeit on a much smaller geographic scale. For freshwaters perhaps the most important effect is the exacerbation of seasalt episodes, and these are predicted to increase in frequency and severity with future global climate change.

View to the future

The UK, Norway and many other European countries with vulnerable upland ecosystems face similar environmental problems, and are all struggling to achieve European and other international environmental policy goals. Under the CLRTAP, there could be a revision of the Gothenburg Protocol for S and N emissions, while new critical loads protocols for heavy metals and POPs are under discussion. The EU Water Framework Directive requires the achievement of "good ecological status" by 2016. The Kyoto Protocol is relevant to upland waters with potential complex effects of climate change and interest in the role of upland ecosystems as carbon sources or sinks.

Particular efforts are therefore needed on three fronts:

1. National and international monitoring programmes must continue.
2. Further research is required on cause/effect and dose/response relationships for "new" pollutants like heavy metals and POPs.
3. Modelling efforts must be maintained to evaluate the risk of future damage in the context of global change.

My view from Norway is that these challenges are best met by close co-operation and communication between scientists

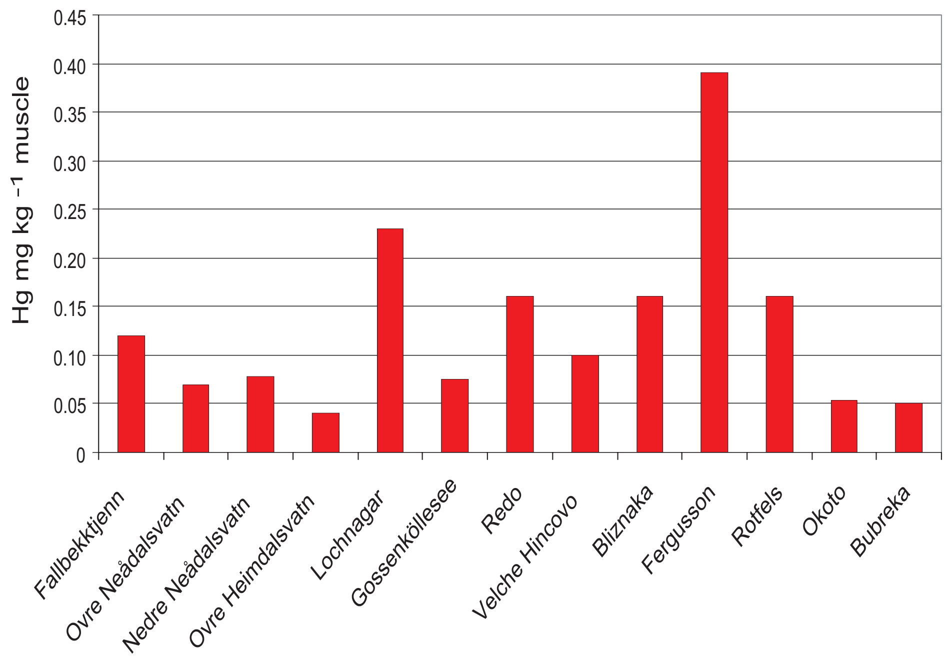

Figure 3: Levels of mercury found in salmonid fish in European mountain lakes (after Rosseland *et al.* In EMERGE, 2003).

and policy makers across national boundaries. Since the Viking times, Norway and the UK have had a tradition of such co-operation, and I hope that this will continue.

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The future of Britain's upland waters: the policy dimension

Group discussion – summary
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At the Conference, speakers presented evidence of the changes that are taking place in upland waters and an analysis of the forces that are now driving change. The future prospects were considered and the modelling tools for predicting future change were assessed.

Evidence from monitoring systems suggests that there have been reductions in some of the pressures on upland waters, e.g. from atmospheric inputs of sulphur and nitrogen, although it appears that the reduction of inputs has not been commensurate with the change in emissions. There is also evidence for recovery from acidification.

However, there are new pressures developing, from climate change and from changes in land-use. There are also unknowns, for example the impacts of the reform of the Common Agricultural Policy (CAP) on the state of waters.

The policy implications of this Conference, both for further policy development and for future work to improve the evidence base, were considered by a panel comprising Catherine Duigan (CCW), Alison Vipond (DEFRA), Judy Clark (UCL) and Tom Nisbet (Forest Research). John Murlis (UCL) chaired the discussion. He started by inviting each of the panel to give their own reflections on the policy messages that emerged during the day. In particular he asked panel members to address some specific questions:

1. To what extent are existing policies on air pollutants adequate with respect to S and N and how can N deposition be more effectively reduced?
2. How can land be better used and managed to minimise impacts on surface waters?
3. How can integrated ecological plans be developed to ensure appropriate protection or restoration of upland waters in the future, taking into account especially future climate change? Is the Water Framework Directive approach adequate?
4. Have the key agencies responsible for ecological planning been accurately identified, what are their respective roles and do they have the same objectives and priorities?
5. Are all stakeholders fully engaged with the issues and how well is the wider public represented?
6. What can be done to improve knowledge transfer between users, stakeholders and the wider public?

Alison Vipond (DEFRA Air Quality Division) described the Department's concerns. She reiterated the importance DEFRA attaches to good evidence of the current status of upland waters and about stock at risk. It was important, she said, to have good tools for assessing the future condition of these important parts of the UK landscape so that sensible targets could be set for recovery. Given the constraints on resources, it would be important to set priorities and to be sure about the chances of policy interventions delivering as required. She said that good communications between policy makers and all the stakeholders would be needed to ensure that the scope for policy intervention was fully understood and that expectations were managed in the light of what was possible.

There were concerns about the focus of European policy. The CAFE (Clean Air For Europe) project was almost entirely directed towards health effects of air pollution. Protection of sensitive ecosystems in remote areas may need further or different investments.

It would also be important to tackle some of the currently uncontrolled sources, for example ships.

Tom Nisbet (Forest Research) presented the following introductory paper.

Sustainable forestry and the protection of upland waters in the UK

Approximately 12% (2.8 million hectares) of the land area of the UK is under woodland or forest cover. More than half of this was created in the last century and most of the expansion involved conifer afforestation in upland areas. During the 1970s and early 1980s it became apparent that such a large-scale change in land use could have a major impact on upland waters. The major water quality concerns were recognised as being: increased turbidity and sedimentation due to the soil disturbance accompanying cultivation, drainage, road construction and harvesting operations; increased phosphate concentrations following aerial applications of fertiliser leading to eutrophication of standing waters; and the enhanced capture of acid deposition by forest canopies resulting in further acidification of surface waters.

In order to address these concerns, the Forestry Commission introduced their 'Forests and Water Guidelines' in 1988. At their core is a set of comprehensive advice on catchment planning, site planning and the conduct of all forest operations, informing forest owners and managers how forests influence the freshwater ecosystem and how operations should be carried out in order to protect and enhance the water environment. Common aspects are the need for meticulous planning and supervision of operations, the adoption of less disruptive practices, the careful matching of machinery to site conditions, varying the scale and timing of operations according to site sensitivity, the use of a wide range of protective measures, and the drawing-up of a contingency plan in case of the accidental spillage of chemicals or oils.

The Forests and Water Guidelines now form the principal document underpinning the protection of freshwaters within UK forests. They apply equally to the state and private forestry sectors and it is a condition of approval for forestry grants and plans that all operations meet the required standards. A joint committee conducts regular reviews with representatives from the water and forestry regulatory authorities and conservation agencies. Since their introduction there have been three revisions, in 1990, 1993 and 2003, to ensure that the Guidelines continue to reflect the most recent regulation, research and experience; the 4th edition covers Britain and Northern Ireland. The Forestry Commission routinely assesses compliance at a local level with the requirements of the Guidelines. Grant approvals, felling licences and forest plans are checked at various stages of development to ensure that the standards are being met.

Forestry as a land-use is now recognised by many as making a valuable contribution in the development of best planning and management practice to conserve upland waters. Attention is increasingly shifting from viewing forestry as a threat to exploring how it can be used to benefit the freshwater environment, such as by helping to counter flood flows, tackle diffuse water pollution, and improve aquatic and riparian habitats. A major challenge facing the forestry sector is the integration of the forestry management with other land-use practices during the development of River Basin Management Plans. This process should further promote the sustainable use of the UK's upland water resources.

In summary, the new policies in forestry management presented considerable opportunities for improving the environmental status of water catchments. In particular, the move towards afforestation with native broad leafed species meant the problems associated with coniferous trees would be reduced. There was also a trend towards use of lower slopes, leaving the more sensitive upper slopes to recover. Reductions in the use of pesticides and artificial fertilisers would discourage invasive species and help the recovery of native flora and fauna.

Judy Clark (UCL) said that DEFRA's commitment to public engagement was very welcome, but she sounded a note of caution. Although the engagement of stakeholders in the debate on the future of uplands waters seemed a good idea, in practice there were important considerations about what constituted a stakeholder, and how their positions were legitimised. In particular it would be important to be clear about the nature of the engagement, which positions should be represented and how. The idea of a "general public" was untenable in the face of evidence of many different constituencies with divergent views. What was urgently needed, however, was an improvement in the public understanding of a range of "generic issues" and in particular of ecological processes. The Water Framework Directive presented a major opportunity to develop methods of public and stakeholder engagement.

Catherine Duigan (Countryside Council for Wales) presented the following introductory remarks on the role of the CCW.

Role of the CCW

In many ways the role of the CCW reflects those of DEFRA regarding policy issues and upland waters.

- *CCW is the Government's advisor on all aspects of nature conservation in Wales; this role is reflected by English Nature and Scottish Natural Heritage in England and Scotland.*
- *CCW funds research on environmental concerns, including acid rain and climate change.*
- *CCW is involved in advising on policy development.*
- *At the moment CCW staff are heavily involved in the technical and policy development related to the Water Framework Directive.*
- *CCW has an additional unique role in managing Tir Gofal (an agri-environment scheme) for the Welsh Assembly Government.*
- *CCW is not the regulatory authority; this role rests with the Environment Agency (EA), (England and Wales), and the Scottish Environmental Protection Agency (SEPA).*

She spoke about the opportunities there were for "joined up" policy emerging from interactions between the different statutory agencies, for example the countryside agencies and the environment agencies (EA and SEPA), especially in meeting the requirements of the Water Framework Directive. She suggested that agri-environment schemes were a good example and had considerable potential. However, she suggested a gap in the policy community in the UK. In the US there was a powerful NGO focussing its efforts on fresh water. With the exception of ponds (e.g. Pond Conservation Trust) and fishing interests there was no equivalent in the UK, with the nearest being the RSPB.

In response to questions about how policy integration might be achieved to ensure that all the different pressures on upland waters might be managed together, Alison Vipond suggested acidification as an area of policy where a considerable degree of integration had been achieved. What was needed now, she said, was the extension of this to the other concerns, such as eutrophication. She repeated her invitation to the science communities to come forward with the evidence and suggestions for more integrated approaches.

David Fowler (CEH Edinburgh) highlighted the importance of deposition of reduced nitrogen, for example ammonia, in upland environments and said that there was little evidence of a decline in deposition. This was, he said, a difficult area of policy but progress was urgently needed. Alison Vipond replied that some progress had been achieved. Reduced nitrogen had been included in the UNECE Gothenburg Protocol and in the National Emissions Ceilings Directive and, although the requirements were not particularly demanding, they were at least a start. The problem of nutrient nitrogen in the environment was exacerbated by the continuing role of agricultural inputs of nitrogen.

Steve Head (Pond Conservation Trust) commented on the focus in the scientific assessment on large water bodies. Ponds, he said were an important part of the upland ecosystem and provided "stepping stones" for wildlife moving between larger water bodies. There was also the opportunity for engagement with local communities as they had a closer identity with local ponds than some of the more remote larger lakes.

The role of stakeholders with a close interest in the quality of lakes and rivers was emphasised by Steve Brooks (Natural History Museum). He suggested that fly fishermen, for example, would be a valuable source of information about trends in key species of fly and that their expertise could be enhanced through workshops.

Rick Battarbee (UCL) reminded the conference about the opportunities offered by the Water Framework Directive (WFD). By setting ecological objectives, the Directive would act as a powerful incentive for integrated policy making. Upland waters are part of important ecological systems and there is a great diversity of investigative work and in the groups working on them. Policy makers and their implementing authorities would benefit greatly from a more coherent approach to the collection and assessment of evidence about the condition of upland waters and to the development of effective policies to ensure their long-term protection.

Rick Battarbee suggested that an advisory group should be formed to include representatives of DEFRA and the devolved administrations, stakeholders and the scientific communities. This would provide immediate support to policy development on the WFD but could also take a longer view and ensure that the implementation of the WFD delivered benefits to these important but vulnerable parts of the UK landscape.

John Murlis concluded the discussion by reminding the Conference that significant change had flowed from the policy of earlier years. The AWMN showed how the pressures on upland waters from atmospheric pollution had fallen in response to the policy of reducing emissions from industry and from transport. There were signs that these changes were having a beneficial effect on the flora and fauna in the upland regions of the UK. However, there was clear evidence that the benefits were not in linear proportion to the reductions achieved in primary emissions. Furthermore, as the work of David Fowler showed, there were particular features of the upland areas, both in terms of the deposition of pollution and the sensitivity of the ecosystems, which meant that proportionally more reduction would be needed in pollution emissions and inputs to the environment to achieve the degree of protection required.

At the same time, the pressures on these areas were changing, in terms of the pollution climate, with nitrogen now a growing concern, the exacerbating effect of global warming, and in terms of human activity with changes in land-use.

This suggested that there was an urgent need for joined-up policy. Administrations and implementing bodies would have to work together and the Water Framework Directive provided a driver for this. The discussions had highlighted the need to engage closely with stakeholders, but, as Judy Clark reminded the Conference, this would have to be done in a well focussed way with adequate information for participants in the process. In particular it would be essential that the participants, and the different constituencies within the public, had a stronger understanding of the complex ecological processes at stake.

There were also other policy drivers and these might contribute constructively. For example, the reform of CAP would have significant (but as yet unknown) impact.

This suggested an urgent need for evidence to inform the policy process. It was encouraging to hear about the progress in modelling so that there could be improved confidence in scenario analysis of the future for upland waters, but there was an urgent need to assimilate this into policy analysis.

The collection and assessment of new evidence was a continuing requirement and the scientific communities would have a major role to play. The Conference demonstrated how the painstaking collection of data on the physical and biological dimensions of the problem had enabled an analysis of the pressures on upland waters and of the link between policy to reduce pollution and subsequent improvements in chemical and biological state of these areas. However, the analysis had also revealed how difficult it is to achieve beneficial change. There would be hard decisions ahead for policy makers and communities.

In particular, it was clear that nitrogen remained a problem. This would mean action to control emissions from power stations (effective controls on sulphur had delivered significant reductions, but in the UK nitrogen emissions from energy conversion remained high) and from road transport. New measures would be needed to tackle emissions from uncontrolled sources such as ships. The agricultural emission of reduced nitrogen, associated in part with grazing in upland areas, needed to be addressed urgently.

The future challenge would be to broaden the base of this analysis to include the human dimension, in terms of land-use and the concern that drives effective action. Communities would have to be involved but the means for doing this needed urgent development.

John Murlis thanked the sponsors and organisers of the Conference for a thoroughly useful and thought-provoking day. It had produced clear evidence that well directed policy worked and he hoped this would encourage DEFRA, the devolved administrations and the statutory bodies to redouble their efforts to ensure protection for these precious upland areas and to support the scientific communities that had so effectively delivered the evidence they needed for current policy.

ACKNOWLEDGEMENTS

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Rick Battarbee, Chris Curtis and Heather Binney
June 2005.

THE FUTURE OF BRITAIN'S UPLAND WATERS

Wednesday 21st April 2004

Department of Geography, UCL, 26 Bedford Way, London

Registration
10.00 – 10.30 Foyer, Department of Geography, UCL

Drivers of change in upland systems
10.30 Introduction: aims of the meeting and general introduction (*Rick Battarbee, ECRC, UCL*)
10.45 Why do we care about upland waters? (*Catherine Duigan, Countryside Council for Wales*)
11.10 Air pollutants in the uplands (*David Fowler, CEH Edinburgh*)
11.35 Climate change in the uplands (*David Viner, UEA*)
12.00 Land-use changes in upland catchments and effects on upland waters (*Bridget Emmett, CEH Bangor and Bob Ferrier, The Macaulay Institute*)

Lunch 12.25 – 13.30

Evidence of change in upland waters – effects on aquatic ecosystems
13.30 Recovery from S deposition (*Don Monteith, ECRC, UCL*)
13.50 Organics – trends in DOC (*Chris Evans, CEH Bangor*)
14.10 Nitrogen – acidification and nutrient N (*Chris Curtis, ECRC, UCL*)
14.30 Metals (*Ed Tipping, CEH Lancaster*)
14.50 Biological responses (*Steve Ormerod, University of Wales, Cardiff*)

Tea 15.10 – 15.40

15.40 The challenges for modelling (*Alan Jenkins, CEH Wallingford*)
16.00 A view from Norway (*Richard F. Wright, NIVA, Norway*)

Panel discussion: policy and management
Chair – John Murlis (UCL)

DEFRA (Alison Vipond)
Public and stakeholder involvement (Judy Clark, UCL)
Countryside Council for Wales (Catherine Duigan)
Forestry Commission (Tom Nisbet)

Close 17.30

DELEGATE LIST

Tim Allott	University of Manchester
Alona Armstrong	University of Durham
Jeremy Bailey	English Nature
Claire Bale	Cardiff University
Rick Battarbee	Environmental Change Research Centre, UCL
Aletta Bonn	Castleton Visitor Centre
Michelle Bromley	Forestry Commission Wales
Steve Brooks	Natural History Museum
Alastair Burn	English Nature
Heather Binney	Environmental Change Research Centre, UCL
Susan Casper	Environment Agency
Pippa Chapman	Leeds University
Judy Clark	University College London
Alan Clarke	University of Manchester
Stewart Clarke	English Nature
Bethan Clemence	ADAS
Sarah Clement	Durham University
Chris Curtis	Environmental Change Research Centre, UCL
Steve Daniels	University of Manchester
Daniel Donoghue	University of Durham
Catherine Duigan	Countryside Council for Wales
Robert Dunford	University of Durham
Bridget Emmett	Centre for Ecology & Hydrology, Bangor
Chris Evans	Centre for Ecology & Hydrology, Bangor
Bob Ferrier	Macaulay Land Use Research Institute
David Fowler	Centre for Ecology & Hydrology, Edinburgh
Alan George	Environment Agency
Jane Goodwin	DEFRA
Sian Griffiths	Cardiff University
Jane Hall	Centre for Ecology & Hydrology, Monks Wood
Phil Harding	Environment Agency, Nottingham
Stephen Head	Ponds Conservation Trust
Katherine Hearn	National Trust
Liz Heywood	Centre for Ecology & Hydrology, Monks Wood
Alan Hildrew	Queen Mary, University of London
Rebecca Humphrey	ADAS
Alan Jenkins	Centre for Ecology & Hydrology, Wallingford
Penny Johnes	Reading University
Martin Kernan	Environmental Change Research Centre, UCL
Renata Kowalik	University of Cardiff
Rene Larson	Natural History Museum
Alan Lawlor	Centre for Ecology & Hydrology, Lancaster
Mark Ledger	University of Birmingham
Bethan Lewis	Cardiff University
Stephen Lofts	Centre for Ecology & Hydrology, Lancaster
Claire Lorenc	Northumbrian Water
Nicola Lower	CEFAS, Lowestoft
Jane Lusardi	English Nature
Don Monteith	Environmental Change Research Centre, UCL
John Murlis	University College London
Julia Newbury	University of Gloucester
Pascale Nicolet	Ponds Conservation Trust
Tom Nisbet	Forestry Commission
Steve Ormerod	Cardiff University
Simon Patrick	ENSIS, University College London
Michael Payne	National Farmers Union
Jo-Anne Pitt	Environment Agency
William Purvis	Natural History Museum
Paul Raven	Environment Agency
Mick Rebane	English Nature
Neil Rose	Environmental Change Research Centre, UCL
James Rothwell	University of Manchester
Laura Shotbolt	University of Bradford
Gavin Simpson	Environmental Change Research Centre, UCL
Paul Sinnadurai	Brecon Beacons National Park Authority
Richard Skeffington	Reading University
Kate Snow	United Utilities
Phil Taylor	Lake District National Park
Ed Tipping	Centre for Ecology & Hydrology, Lancaster
Angus Tree	Environment Agency
Jacqui Ullyett	Centre for Ecology & Hydrology, Monks Wood
David Viner	University of East Anglia
Alison Vipond	DEFRA
Clare Whitfield	Joint Nature Conservation Committee
Richard F. Wright	NIVA, Norway

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