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SUMMARY OF RESEARCH UNDER DETR CONTRACT "ACIDIFICATION OF FRESHWATERS: THE ROLE OF NITROGEN AND THE PROSPECTS FOR RECOVERY"
EPG/1/3/117

WORK PACKAGE 2: RECOVERY

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Contents

Hysteresis and problems in detecting biological recovery

1) In the UK, the effects of acidification have been unique in combining impacts on ecosystems that are both large-scale and long-term: existing studies suggest that effects of this type require proportionately long time scales for recovery (i.e. decades-centuries).

2) Since ecosystem structures and/or functions might not return to pre-disturbance conditions within practically measurable timescales, "recovery" should, at best, be considered as damage reversal towards some target endpoint rather than true repair or restoration.

3) Understanding of functional indicators of recovery (e.g. production, decomposition, predator-prey interactions) is particularly weak.

4) In aquatic systems, several factors could cause a time lag in recovery, including the persistence of acid episodes, organism dispersal, the need for recovery across the whole food web to restore functional integrity, and intrinsic limits on population rates of recovery.

5) Liming experiments at Llyn Brianne have demonstrated that hysteresis in recovery after liming reflects limited persistence rather than restricted dispersal, and could be due to the continued effects of acid episodes.

6) In a study of 70 Welsh streams, indices of pH and Al episodicity significantly improved regression models that predicted biological status over models based on modal chemistry values alone.

7) Evaluation of data from upland Welsh streams (Welsh Acid Waters Survey) found that large variations between years in the persistence of invertebrate communities are not entirely stochastic, instead varying with environmental fluctuations, in particular the NAO, which might confound or obscure recovery from acidification.

Defining targets and management for recovery

1) Should forecasts and rates of change due to deposition reduction be limited or slow, interventionist management may be considered by some agencies, e.g. liming, species re-introductions, or agri-environment schemes.

2) The Water Framework Directive is likely to drive targets and prescriptions that require us to establish reference conditions for surface waters prior to acidification. It will also require management strategies that promote recovery. A range of uncertainties currently affect our abilities in this regard.

3) In order to evaluate future recovery, Flower *et al.* (1997) have proposed a palaeolimnological technique for defining targets for the recovery of acidified surface waters, based on the technique of analogue matching of lake sediment diatom assemblages.

4) Analogue matching has been successfully applied to several acidified lakes, and modern analogue systems defined for the pre-impact (pre-acidification) status of these impacted sites. An advantage of the approach is that it can provide recovery targets for both chemical and biological status of acidified lakes.

5) A new approach was applied to 8 AWMN lakes using both diatoms and cladocera, and close modern analogues were identified for 7 out of 8 AWMN lakes studied, with the majority of the analogues located in North and Northwest Scotland, areas of low sulphur and nitrogen deposition.

6) Comparison of the chemistry and catchment characteristics of the AWMN lakes to those of the modern analogues showed that the modern analogues had higher lake water pH and alkalinity levels and lower aluminium concentrations, but very similar ionic strength and calcium concentrations.

7) These results indicate that the analogue matching approach using diatom and cladoceran remains is a simple, robust and reliable method of identifying modern analogues for acidified lakes in upland areas of the UK, which provide suitable reference conditions for pristine upland lake ecosystems and may be used to evaluate progress made towards restoration in acidified lakes.

Chemical and biological trends

1) An assessment of long-term Galloway data shows that the time-scale and start /end point of the time series has a major bearing on conclusions regarding recovery. Step-changes in the deposition pattern and climatic variations have directly influenced the recovery signal in these sensitive lochs and streams. Nevertheless, these waters are currently at their healthiest point of recovery since 1978.

2) Recent trends in sedimentary diatom assemblages in five acidified lochs in Galloway indicate that two sites are recovering strongly while one shows no evidence of recovery. At the remaining two sites, slight species changes could not be confidently ascribed to a recovery response. Here, the influence of inter-annual variation in climatic factors on acidity during the growth season may currently outweigh any benefit of a reduction in anthropogenic acidity.

Chemical and biological modelling of recovery

1) The CLAG/CLAM chemical-biological database was expanded and used to develop predictive models for three biological targets, the diatom *Achnanthes minutissima*, the invertebrate *Baetis rhodani* and an acid tolerant diatom assemblage, using pH and ANC.

2) The fitted logistic regression models can predict the occurrence of the target organisms with an internal success rate of c. 70%, and provide a means of predicting the probability of occurrence of the target organisms from modelled hindcasts and forecasts.

3) Application of the models to the CLAM national chemical database using an SSWC model hindcast of baseline (pre-industrial), measured present, and FAB modelled future (Gothenburg scenario) ANC indicates substantial biological change in the areas of known acidification (SW England, Wales, Cumbria, Pennines and large parts of Scotland, but virtually complete biological recovery (with no timescale given) under the Gothenburg scenario.

4) Biological predictions for five MAGIC5 regions (Cairngorms, Galloway, Pennines, Wales, Lake District) show similar patterns of biological change between baseline and present, with varying degrees of loss of sensitive taxa. Predictions for recovery under the Gothenburg scenario indicate that only partial recovery will take place by 2050, and for some regions sensitive taxa are predicted only to return to approximately half of the sites inhabited prior to the onset of acidification.

5) The biological models provide a simple but robust way to convert hydrochemical hindcasts and forecasts into measures of biological change, and in particular, to assess biological damage under different emission reduction scenarios. There is now a need to validate the models using independent field data, and to increase data coverage in some areas to extend the models to other biological groups e.g. macrophytes.

6) Site-specific MAGIC7 applications to the AWMN sites clearly demonstrate the benefit of the emission reductions in both S and N under the Gothenburg Protocol over making no further emission reductions, but the ‘best’ and ‘worst’ cases for N leaching have a less significant impact on predicted ANC up to 2041 than the agreed S reductions.

7) Regional simulations with MAGIC7 in Galloway, Wales and the south Pennines show marked variability between regions, with potential increases in N leaching having significant implications for recovery only in the Pennines.

8) MAGIC7 simulations carried out under different forestry scenarios for the Galloway dataset indicate that immediate replanting of forest after harvesting (worst case) severely curtails recovery in surface water ANC under the REF scenario, while reversion to moorland after harvesting (best case) leads to significant recovery in ANC.

Task 2.1: Continue monitoring for trends at Galloway sites

Task 2.1.1: Chemical and biological trends in Galloway, Southwest Scotland

R.Harriman

2.1.1 Chemical and Biological trends in Galloway, south-west Scotland

Ron Harriman

Background

Studies of the ecological impacts of acidic deposition in upland areas of Galloway commenced in 1978 when 22 lochs and 27 streams were surveyed to provide fish population and chemical data (Harriman *et a.l,* 1987). Chemical sampling progressed on a random basis prior to 1988 and thereafter on a monthly or three-monthly basis while fish population surveys in 1978/79,1984 and 1994 involved systematic netting in lochs and quantitative electrofishing in streams. Angling catch records from a number of sites provided an historical perspective of changes in fish populations while continuous records were available for lochs Riecawr and Macaterick from 1911 to 1999. An experimental reintroduction of brown trout into Loch Enoch commenced in 1995 to assess the extent of chemical recovery (Collen *et al.*, 2000). A preliminary assessment of chemical trends for 5 high-elevation lochs between 1978 - 1994 (Harriman *et al.*, 1995) revealed a significant reduction in non-marine sulphate and an increase in pH and alkalinity. However, for Loch Grannoch, a lower elevation loch with a large proportion of coniferous forest in its catchment, there was no improvement in pH and Alkalinity despite a significant decline in non-marine sulphate.

Chemical trends

Trends in chemical data were tested statistically using the Seasonal Kendall (SK) test which provided an estimate of the mean annual change and the significance of the trend. Supporting evidence for the significance of the trends was obtained using a randomised Linear Regression (LR) technique. At all sites for virtually all determinants any significant trend obtained using the SK test was matched by a significant LR trend.

17 sites were used in the analysis of the 1978-99 data (15 lochs and 2 streams) of which the catchments of 5 lochs (Riecawr, Macaterick, Grannoch, Harrow and Minnoch) and a stream (Green Burn) contain a large proportion of coniferous forest. Following a visual inspection of the data it became clear that the pattern of response to reduced S deposition was not linear but followed a more ‘stepped’ type of response. Consequently, trend analysis for 4 of the most sensitive sites was extended to cover the period 1988-98, to match the 10 year analysis of the AWMN data, and the 1995-1999 period to evaluate the most recent changes.

Trends for the 1978-99 period

For all sites, total $S0_4^{2-}$ and non-marine $S0_4^{2-}$ declined significantly ($p<0.05$) averaging 2.7 and 2.2 $\mu eql^{-1}yr^{-1}$ respectively. There was a significant increase in pH at all sites (+0.015 pH units yr^{-1}) while alkalinity increased significantly (+ 0.62 $\mu eql^{-1}yr^{-1}$) except at the forested sites of Green Burn and Loch Grannoch.

Toxic forms of Al (Al-L) declined significantly at all sites but changes were greater at loch sites than stream sites. Conversely, non-toxic (organic) forms of Al (AL-NL) increased significantly, in line with increasing concentrations of dissolved organic carbon (DOC) which in many cases resulted in a non-significant change in Total Al.

Concentrations of NO_3^- showed a small but significant increase at loch sites but not for streams whereas the patterns of base cation change were extremely variable but with no overall trend. Marine salt (Na, Cl) concentrations varied considerably and although there was a slight downward trend over the period it was non-significant.

Trends for the 1988-98 period

Because the results of the above analysis differ, although not unexpectedly, from that of the Acid Waters Monitoring Network (AWMN) the data for the equivalent 10 year period were evaluated for four sites, three of which were also included in the AWMN (Grannoch, Round Loch of Glenhead and Dargall Lane). Fewer significant trends were obtained for this period and although non-marine SO_4^{2-} declined at three of the four sites only the Dargall Lane site showed a significant increase in both pH and alkalinity. In general, the rate of decline/ increase for most parameters was lower than the 1978-99 period while NO_3^- declined at Grannoch but increased at Enoch and Round Loch of Glenhead. Marine salt concentrations declined significantly at all sites except Grannoch where the decline was not significant.

Trends for the 1995-1999 period

The patterns of recovery for this most recent period indicate a faster rate of change but with some reversal of previous changes. For example, marine salt concentrations show a large and significant increase while NO_3^- concentrations declined significantly. This sharp decline in NO_3^- appears to be the result of especially high values in the 1995-97 period caused by increased N mobilisation following the long dry summers. Correspondingly high SO_4^{2-} concentrations were also recorded during the same period resulting in a significantly large decline in non-marine sulphate at all four sites. Only at Enoch and Round loch of Glenhead were significant increases in pH and alkalinity recorded whereas large reductions in Al-L were found at all sites, although the reduction at Dargall Lane was not significant

It is evident from this assessment that the time-scale and start /end point of this time series has a major bearing on conclusions regarding recovery. Step-changes in the deposition pattern and climatic variations have directly influenced the recovery signal in these sensitive lochs and streams. Nevertheless, these waters are currently at their healthiest point of recovery since 1978.

Task 2.1.2/2.1.3: An examination of evidence for biological recovery in the acidified lochs of Galloway

D.T. Monteith, R.J. Flower, T.E.H. Allott, G.L. Simpson, and A. Kreiser

2.1.2 & 2.1.3: An examination of evidence for biological recovery in the acidified lochs of Galloway

D.T. Monteith, R.J.Flower, T.E.H. Allott, G.L. Simpson and A.Kreiser

Summary

1. This report focusses primarily on evidence for the recovery of diatom populations in five acidified lochs (Loch Narroch, Round Loch of the Dungeon, Loch Grannoch, Round Loch of Glenhead and Loch Enoch), in the Galloway region of south-west Scotland, in response to reductions in sulphate deposition over the last two decades. Water chemistry monitoring has shown that all sites other than Loch Grannoch have undergone a recent amelioration in acidity.

2. Diatom analysis was carried out on sediment samples from annually sampled sediment traps (representing the last 10 years) and ^{210}Pb dated sediment cores taken around 1990. Trap and core data were integrated, allowing high temporal resolution trends in diatom assemblages over the last decade to be examined in the context of the longer-term change. In a complementary study, new sediment cores were taken from Round Loch of Glenhead and Loch Enoch, and the most recent sediments extruded at 2 mm intervals and also subjected to diatom analysis.

3. Principal Component Analysis (PCA) of the integrated core-trap datasets showed that small species reversals had occurred at four of the lochs over the past two decades. Loch Grannoch, the only afforested loch, showed no reversal. A PCA of the top of the new sediment core from Round Loch of Glenhead also provided evidence of a recent species reversal, but this was not apparent in the new Loch Enoch core.

4. The direct gradient analysis approach of Canonical Correspondence Analysis, and the diatom-pH reconstruction method of Weighted Averaging Partial Least Squares (both based on the SWAP diatom-chemistry model), applied to the integrated core-trap datasets, demonstrated that recent species changes in Loch Narroch and the Round Loch of the Dungeon were strongly indicative of a response to declining acidity, providing clear evidence for biological recovery. In contrast the slight species changes identified by PCA at Round Loch of Glenhead and Loch Enoch could not be confidently attributed to a response to declining acidity. It is possible that these species reversals are as yet too small to translate as acidity responses according to the SWAP model, but it is also feasible that they represent a response to other environmental or biotic changes.

5. The findings of the analysis of the new sediment cores from Round Loch of Glenhead and Loch Enoch were in broad agreement with the core-trap study. A diatom pH reconstruction of the new Round Loch of Glenhead core provided tentative evidence for recent improvement, but the degree of change was well within the error estimates for the models. No recent rise in diatom inferred pH was detected in the new core from Loch Enoch.

6. Differences in the extent of recovery between sites may relate to catchment characteristics and sensitivity to climatic effects. The apparent continuation of acidification at Loch Grannoch is supported by recent water chemistry. This could result from a continued

depletion of base-cations derived from a heavy application of rock-phosphate fertilizer as part of forestry management in the mid 1980s. The sediment trap assemblages of Round Loch of Glenhead and Loch Enoch show large inter-annual variability which could be driven by hydrological variation over the spring growing period. Inter-annual variability in spring chemistry may be large relative to any deposition driven improvement in water quality. Gradual recovery in the flora is therefore still expected but further years of monitoring are necessary before a clear response to declining acidity can be demonstrated.

7. An analysis of cladoceran remains in the new core from Loch Enoch revealed substantial changes in species and a drop in diversity over the period of acidification. This illustrates the wider ecological influences of acidification on these systems, and also demonstrates the broader potential of the sediment record for reconstructing and monitoring acidified ecosystems.

8. This report demonstrates the power of a combined sediment core – sediment trap approach for the assessment of recent biological change against a longer-term perspective. Similar approaches could be applied to other biological groups which leave fossil remains (including chironomids and cladocera) and would provide a more comprehensive assessment of the extent of ecosystem recovery in these chemically improving systems.

1 Introduction

During the 1980s, sediment core diatom analysis played a crucial role in revealing the cause, timing and spatial extent of freshwater acidification across the UK. Diatom assemblages in sediment cores from acid-sensitive lakes in high deposition areas consistently showed species trends which were indicative of a change to more acid conditions over the last 150 years. Towards the end of the decade, new analysis of sediment core tops from the Round Loch of Glenhead (in the Galloway region of south-west Scotland) suggested that a small reversal had occurred in the species assemblage (Battarbee *et al.*, 1988; Allott *et al.*, 1992). Such a change was consistent with the reduction in non-marine sulphate (xSO_4) measured in deposition at nearby Eskdalemuir, and evidence for a slight improvement in the water quality of a group of Galloway lochs over the same period (Harriman pers. comm.). The changes were therefore attributed to the early stages of biological recovery, the first reported for the UK.

The timing of these findings coincided with the onset of regular chemical and biological monitoring of the Round Loch of Glenhead and 21 other acid-sensitive sites within the UK Acid Waters Monitoring Network (UKAWMN). A report on the first 10 years of UKAWMN data (Monteith and Evans, 2000) suggested that the earlier reductions in deposition and the reported chemical improvement of acidified waters in the UK did not continue into the 1990s. Over the period 1988-1998 there was little evidence for a decline in xSO_4 concentration or acidity at most sites, including those in Galloway. Trends indicative of improvement were observed in epilithic diatom and macroinvertebrate communities at some sites, but in most cases these could be explained by changes in rainfall over the winter and spring, and the occurrence of sea-salt deposition events (linked to a falling North Atlantic Oscillation Index between 1991 and 1997), which can influence short-term variability in acidity.

However, a new study of UKAWMN water chemistry trends to March 2000 (Evans and Monteith, 2001) now conclusively shows a regional scale decline in xSO_4 concentration and acidity from around 1995. More than 10 years after the first evidence that biological recovery was underway at the Round Loch of Glenhead, and given the recent widespread chemical improvements, it is pertinent to ask whether the diatom recovery in this loch has been sustained, and whether biological responses are occurring on a regional scale (in this case across 5 acidified lochs in Galloway, south-west Scotland).

Two approaches have been taken to answer these questions under the CLAM contract:
First, ^{210}Pb dated sediment samples from cores taken from five Galloway lochs around 1990 were integrated with sediment trap samples collected annually each year thereafter. Although small discrepancies might arise as a result of differences in the type of sediment receptor (i.e. trap or lake bottom), this approach has the advantage that samples for the decade of particular interest (i.e. the last ten years) are precisely dated. Unlike the core samples, trap samples are temporally discreet, i.e. there is no smoothing by bioturbation or other sediment mixing processes, although they may still be vulnerable to redeposition of older sediments. In a complementary approach, high temporal resolution diatom analysis has been conducted on two new sediment cores taken from the Round Loch of Glenhead and another strongly acidified loch, Loch Enoch, taken during the summer of 2000. These cores are unlikely to provide the degree of temporal resolution provided by the trap study, but will avoid the potential problem of core-trap compatibility, and, as more temporally smoothed records, could provide a better indication of decadal scale time-trends.

Additionally, a second core taken from Loch Enoch in 2000 has been cursorily screened for cladoceran remains, to investigate the potential for applying this technique to recovery studies. Cladocera are microscopic crustaceans and a member of the zooplankton component of lake ecosystems. They are found in two distinct habitats; the littoral zone and the pelagic or open water zone (Scourfield & Harding, 1966). The Cladocera possess an exoskeleton of chitin that surrounds and protects the soft body parts of the animal. Upon death this exoskeleton dissarticulates into its component parts. These remains are readily preserved in lake sediments, particularly those of the littoral dwelling species of the family Chydoridae and the planktonic Bosminidae (Frey, 1962). The past community composition of a lake can be determined by studying the remains of the cladocera preserved in lake sediments.

2. Methods

The project involves five upland lochs, the Round Loch of Glenhead, Loch Enoch, Loch Narroch, Round Loch of the Dungeon and Loch Grannoch, all situated in Galloway, southeast Scotland (Table 1). All sites have previously been shown to have acidified using diatom inferred pH reconstruction techniques on ^{210}Pb dated cores, and results have been reported by Patrick *et al.* (1995) and Allott *et al.* (1992). Diatom data for these cores were downloaded from the ECRC database, Amphora.

2.1 Sediment core-trap study

Sediment traps (two arrays per site, each containing 3 replicate traps) were first deployed in all five lochs in 1991, and samples have been collected from most sites in most years. Each year, sediment from one replicate trap from each loch was subjected to standard diatom analysis techniques. 500 diatom valves from each sample were counted and identified to species level using x1000 phase contrast microscopy. Diatom taxonomy follows the conventions of the

Surface Water Acidification Programme (SWAP) (see Stevenson *et al.,* 1991). For each site, species frequency data for each year of trap exposure were then appended to frequency data for ^{210}Pb dated sediment core samples. The resulting datasets therefore represent diatom frequency counts for a series of dated samples, taken from sediment cores (to approximately 1990) and sediment traps (from around 1991 to 2000). Taxonomy between the core and trap diatom data was harmonised. ECRC core codes, ^{210}Pb dates of the basal samples and the number of core and trap samples comprising each dataset are presented in Table 1.

Table 1 Sample information

Site	**ECRC core code**	**Date of core surface**	**^{210}Pb date of core base**	*No. samples*	
				core	**traps**
R.L. Glenhead	K86	1990	1790*	26	10
Loch Enoch	ENO6	1990	1920	17	9
Loch Narroch	NARR3	1990	1886	21	9
R.L Dungeon	RLD1	1991	1861	17	9
Loch Grannoch	GRAN89/1	1989	1925	19	8

* indirectly dated by cross-referencing with other dated cores from the site

For each site, data were transformed to percentages and only those diatom species which occurred in at least one sample at more than 2% abundance were excluded from all further analysis. Relative abundances of the most commonly occurring taxa are presented in Figures 1a-e. All subsequent statistical analysis was carried out using the multivariate analysis software, CANOCO (ter Braak, 1990a,b).

Preliminary statistical analysis of the datasets, using DCA (Detrended Correspondence Analysis), revealed limited gradient lengths (i.e. less than 3.0) for each of the five lochs. This implies that species turnover is low, and that these data are suitable for linear analysis methods. The significance of individual Principal Component axes was tested by applying the Broken Stick model (Jackson, 1993). The first three axes were deemed significant for Loch Enoch, but only Axes 1 and 2 were significant at the other four sites.

Subsequent analysis was conducted at three levels:

1: The datasets were analysed individually using the indirect gradient analysis approach Principal Component Analysis (PCA). PCA provides a highly sensitive measure of between-sample variance in the species assemblage and is ideally suited for detecting small species reversals in a series of samples. PCAs were constructed using the sediment core data only. Sediment trap data were then added as “passive” samples. Axis scores for these samples are calculated in the analysis although they do not influence the PCA structure. In this way the influence of possible systematic differences in species assemblages between core and trap samples are minimised. Time series of sample scores on PCA Axis 1 are plotted in Figure 2.

2: Combined sediment core and trap data for individual sites were placed as passive samples in the ordination space of a Canonical Correspondence Analysis (CCA) of the Surface Waters Acidification Project (SWAP) water chemistry-diatom dataset (Stevenson *et al.*, 1990). CCA is a direct gradient analysis approach, in which the ordination of sites is constrained by linear combinations of a defined set of environmental variables. CCA assumes unimodal species responses and is appropriate in this case since gradient lengths are large. The SWAP species data, representing surface sediment samples for 131 sites, were constrained by three chemical variables (pH, total aluminium and DOC) recorded for the sites around the time of sediment sampling. Again, passive samples have no influence on the structure of the ordination, while their sample scores are calculated for each axis according to their diatom assemblages. This method should isolate the pH effect on the assemblage, and should, therefore, provide a better indication of the cause of any recent change in the datasets. Time tracks of the core/trap samples on the first two CCA axes are presented in Figures 3a-e, while time series for CCA Axis 1 scores (i.e. the acidity axis) are presented in Figures 4a-b.

3: Diatom-inferred pH (DI-pH) for each sample in each dataset was calculated by Partial Least Squares Weighted Averaging (WAPLS) (ter Braak & Juggins, 1993). As with CCA, WAPLS is an iterative, weighted averaging method and we have again used the SWAP water chemistry-diatom dataset as the model. Unlike the CCA approach however, pH is the only explanatory variable used. This approach provides a more readily interpretable measure of species change (i.e. the pH response rather than ordination Axis Scores) while ignoring possible covariance between pH and other variables. Compared with the CCA, the WAPLS SWAP-pH model incorporates over 30 additional sites, i.e., the 167 for which data are available for both diatoms and pH.

2.2 New sediment core study

300 mm long sediment cores were taken from the deepest point of the Round Loch of Glenhead (KYKA) and the eastern basin of Loch Enoch (ENO10) in May 2000 using a Glew gravity corer (Glew, 1989). The cores were extruded on site at 2 mm intervals, transported to the laboratory and stored at 4°C. Contiguous samples from the top 32 mm of the Round Loch of Glenhead and 120 mm of the Loch Enoch core were prepared for diatom analysis using standard procedures (Battarbee, 1986), and 300 valves were counted from each sample using x1000 phase contrast microscopy. Diatom taxonomy follows the conventions of the Surface Water Acidification Programme (SWAP) (see Stevenson *et al.,* 1991).

Variation in diatom assemblages within these uppermost sediments was analysed using two ordination approaches. First, principal components analysis (PCA) was undertaken to highlight the main gradients of variation in the diatom assemblages. PCA was implemented using CANOCO 3.10 (ter Braak, 1990a, 1990b) and the broken stick model (see Jackson, 1993) was used to test the significance of individual PCA axes. Second, diatom-inferred pH (DI-pH) was calculated by weighted averaging for each sample using the SWAP diatom-pH transfer function (Birks *et al.,* 1990; Stevenson *et al.,*1991).

2.3. Cladoceran fossil analysis on the new Loch Enoch core

4 samples from a second core from Loch Enoch (ENO12) were processed in hot 10% KOH, washed, sieved and the residue plated on microscope slides for the counting of cladoceran remains. The 4 samples reflect the major periods in the recent history of Loch Enoch; pre-acidification (32.8-33.0 cm), onset of acidification (10.0-10.2 cm), strongly acidified (5.0-5.2 cm), present day (0.0-0.2 cm).

3. Results

3.1 Sediment core-trap study

3.1.1 Qualitative assessment of species changes:

Time trends in the integrated sediment core and trap data are presented in Figures 1a-e. With the possible exception of Loch Grannoch, diatom assemblages toward the top of the sediment cores (i.e. the most recent core sediment) closely match those of the assemblages in the sediment trap samples. This suggests that direct comparisons of core and trap samples are valid. For Loch Grannoch there are discrepancies between the two sample types; the sediment core samples are relatively diverse, while the trap samples are more dominated by one species (*Eunotia incisa*). Discrepancies are also apparent for samples for 1991 and 1992 from Loch Narroch. This could be due to contamination of the sediment by substantial numbers of caddisfly larva cases, which provide an additional substrate for *Eunotia incisa* communities (Flower, pers. comm.) and were not removed from the sample in these years.

Diatom assemblages from all five sites are characteristic of oligotrophic, acidic lakes. Recent sediment samples from all sites contain significant proportions of *Eunotia incisa*, *Frustulia rhomboides* var. *saxonica* and *Tabellaria quadriseptata*, all of which have pH optima of 5.2 or less, while *Brachysira vitrea* (pH optima 5.9) is relatively abundant in all sites other than Loch Enoch.

All five sites show a gradual increase in the proportion of the acidobiontic species *T. quadriseptata* (pH optima 4.9) towards the most recent sediment at the core top. This feature is characteristic of cores from many acidified lakes (Flower *et al.* 1987). In the Round Loch of Glenhead, Round Loch of the Dungeon and Loch Grannoch, this increase is accompanied by a large decline in *B. vitrea* among other species. At Loch Enoch and Loch Narroch the clearest decline has been in *E. incisa* (pH optima 5.1*)*. It is likely that the oldest sample from Loch Enoch (c. 1920) is still too recent to capture the initial stages of acidification at this site. Loch Narroch appears to have been naturally quite acidic, presumably due to relatively high concentration of organic acids in this relatively high DOC loch.

There is evidence for a decline in *T. quadriseptata* in sediment deposited within the last 20 years for all sites other than Loch Grannoch. At Loch Narroch the proportion of this species begins to decline in the upper layers of the sediment core, c. 1980, while at the Round Loch of Glenhead, Round Loch of the Dungeon and Loch Enoch, the decline occurs within the sediment trap record, i.e. post-1994. In most cases this decline has been accompanied by an increase in *E.incisa*. Although there are problems in relating absolute proportions of species in the core and trap samples from Loch Grannoch, the gradual increasing trend in *T. quadriseptata* abundance since the 1950s in the sediment core has been continued in the sediment trap samples, at least up to 1998.

Figure 1a. Frequency of occurrence of diatom taxa in a ^{210}Pb dated sediment core and annually collected sediment trap samples for the Round Loch of Glenhead. The dotted line marks the transition between core and trap data

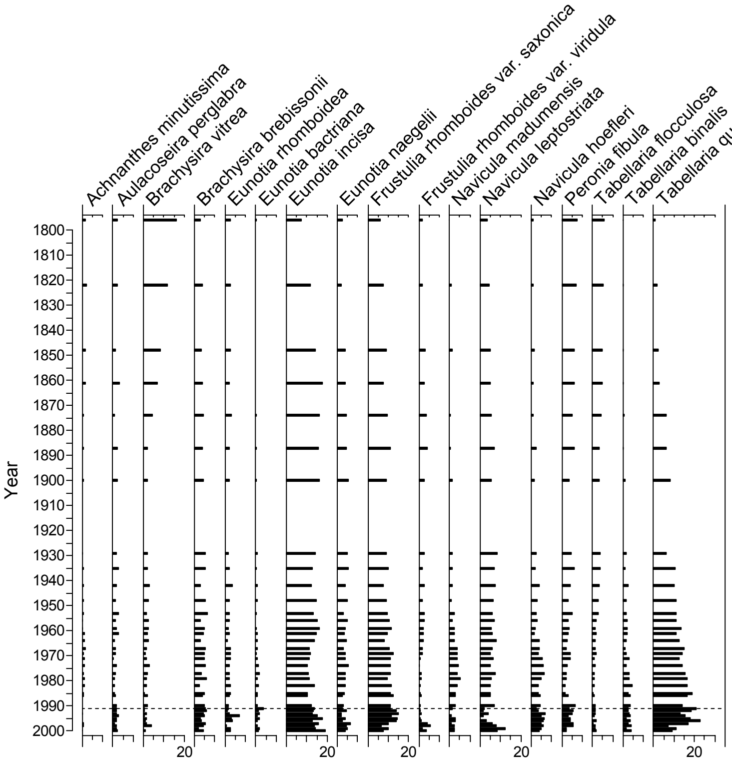


Figure 1b. Frequency of occurrence of diatom taxa in a ^{210}Pb dated sediment core and annually collected sediment trap samples for Loch Enoch. The dotted line marks the transition between core and trap data

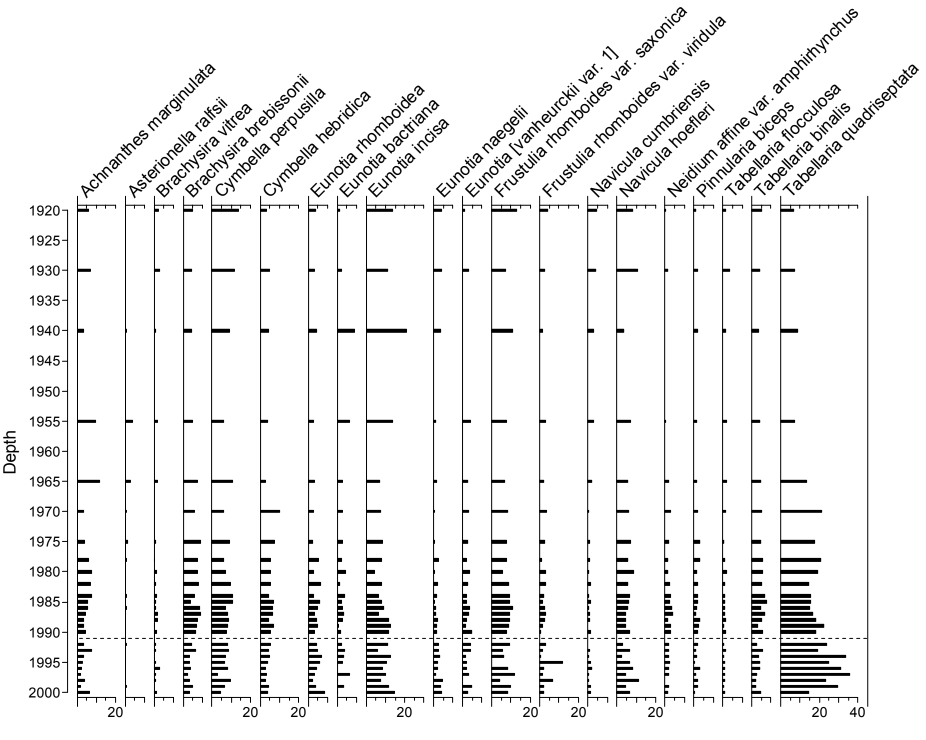


Figure 1c. Frequency of occurrence of diatom taxa in a ^{210}Pb dated sediment core and annually collected sediment trap samples for Loch Narroch. The dotted line marks the transition between core and trap data

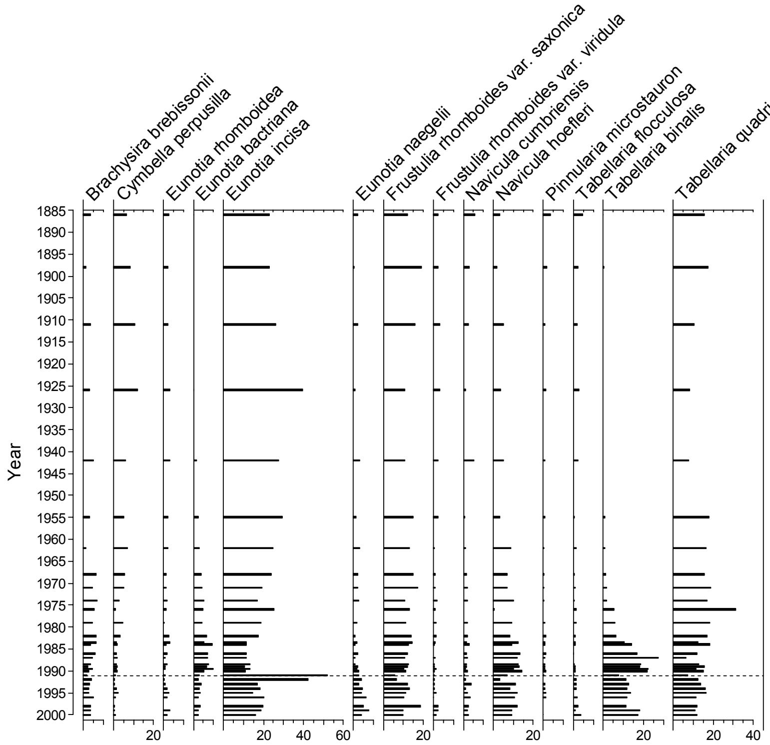


Figure 1d. Frequency of occurrence of diatom taxa in a ^{210}Pb dated sediment core and annually collected sediment trap samples for Round Loch of the Dungeon. The dotted line marks the transition between core and trap data

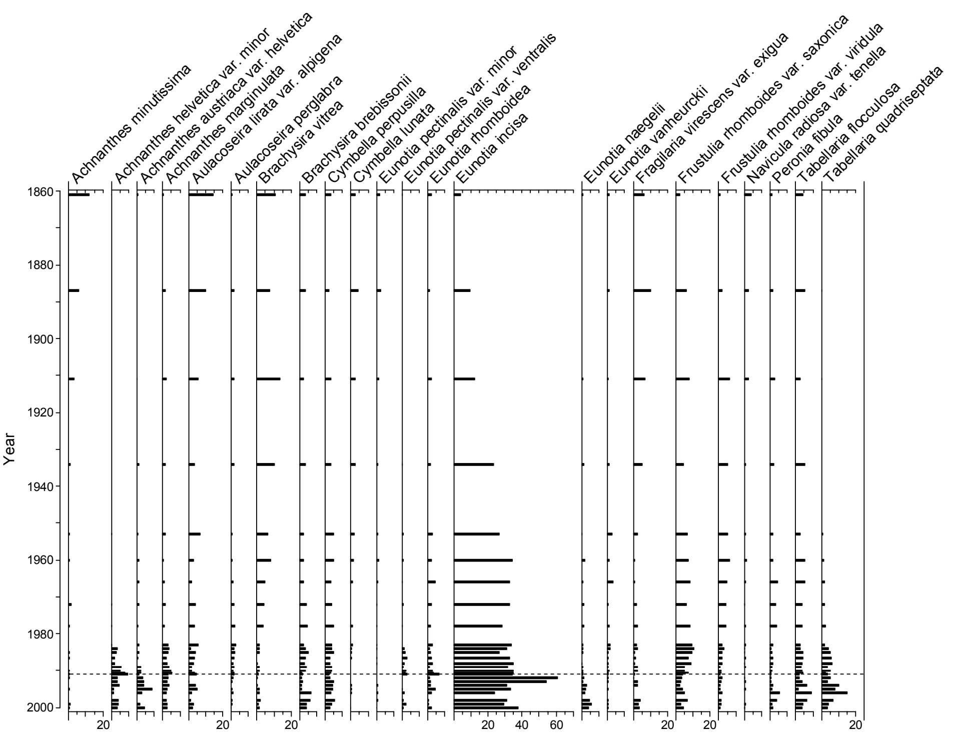
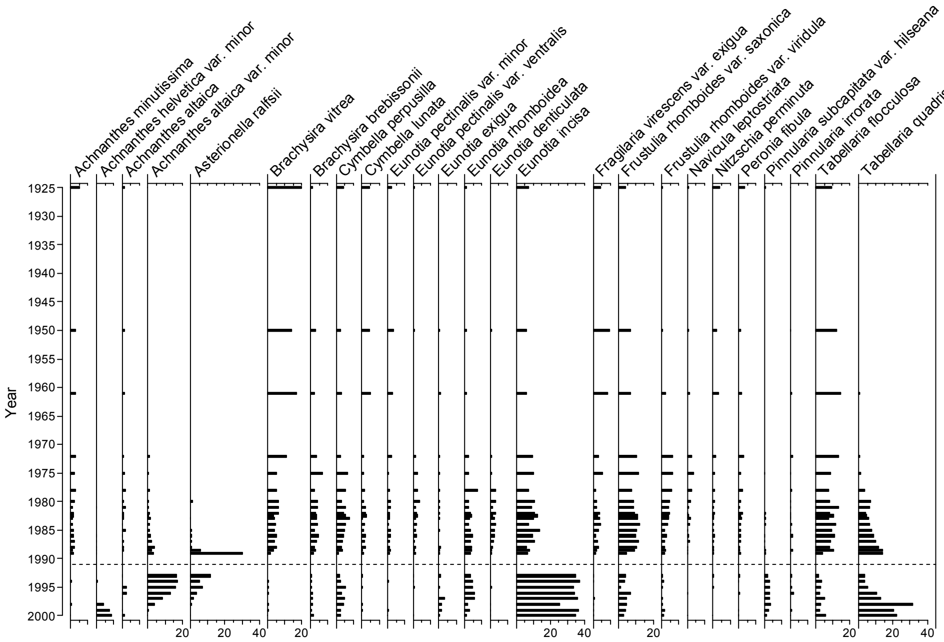


Figure 1e. Frequency of occurrence of diatom taxa in a ^{210}Pb dated sediment core and annually collected sediment trap samples for Loch Grannoch. The dotted line marks the transition between core and trap data



3.1.2 PCA ordination

PCA Axis 1 Species Scores, which describe the strength of the correlation between species abundance and the principal axis of variation in the species data, are presented for key taxa in Table 2. This demonstrates that for the sediment core data for all sites other than Loch Narroch, the abundance of *T. quadriseptata* (pH optimum 4.9) is tightly correlated with Axis 1 (large negative scores). In Loch Narroch, *T. binalis*, characteristic of even more acid conditions (pH optimum 4.7), shows the strongest negative relationship*. Brachysira vitrea* (pH optimum 5.9) and *Cymbella lunata* (pH optimum 5.7), and other species with relatively high pH optima, generally show the reciprocal relationship, i.e. they have large positive scores on Axis 1 at most sites. Loch Narroch again provides the exception, as changes in the proportion of the acidophilous species *E. incisa* (pH optimum 5.1) and *T. flocculosa* (pH optimum 5.4) balance changes in the extremely acid species.

At all sites, therefore, PCA Axis 1 appears to relate to an acidity gradient. In all cases, species with negative scores have a preference for more acidic conditions. By inference, the lower sample scores on PCA Axis 1 should represent the more acid conditions at the site.

Table 2 Correlations between the relative abundance of key species and Axis 1 of Principal Components Analysis of site specific datasets.

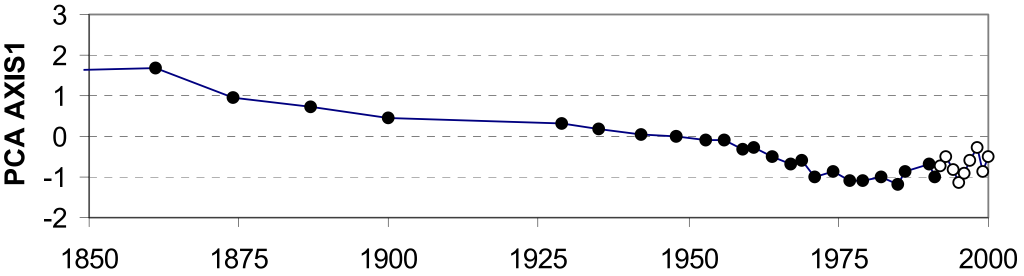
Species/Site	**Loch Enoch**	**Round Loch Glenhead**	**Loch Narroch**	**Round Loch Dungeon**	**Loch Grannoch**
Brachysira vitrea	0.60	0.90		0.92	0.95
Cymbella lunata		0.73		0.92	0.68
Eunotia incisa	0.65	0.15	0.88	-0.80	-0.30
Tabellaria flocculosa	0.69	0.93	0.77	0.40	0.54
Tabelleria binalis	-0.37	-0.90	-0.98		-0.88
Tabellaria quadriseptata	-0.91	-0.97	-0.02	-0.93	-0.97

Time series for PCA Axis 1 sample scores (Figure 2) show a progressive reduction in samples scores toward the sediment core tops, indicating the acidification process. Inclusion of sediment trap data in the PCA as passive samples provides an indication of a reversal in the species assemblage of four sites, with a gradual trend toward more positive sample scores. At Loch Grannoch however, sample scores for sediment trap samples remain as negative as the most extreme scores for the sediment core samples. Four out of five sites therefore provide evidence for the onset of recovery using this approach. Between-year variability in individual sample scores makes it difficult to precisely define the date of inflection but the most extreme negative scores generally occur around 1990.

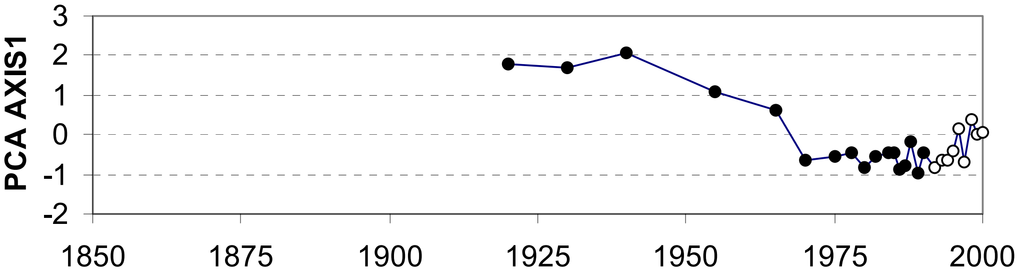
Caution should however be exercised in the interpretation of the PCA data. Although the species scores make it clear that the first axis is likely to be closely correlated with acidity, it may also be indicative of other chemical changes, since pH is likely to have

Figure 2. Time series of PCA Axis 1 Sample Scores for the 5 Galloway Lochs. Sediment trap samples (open circles) were treated passively

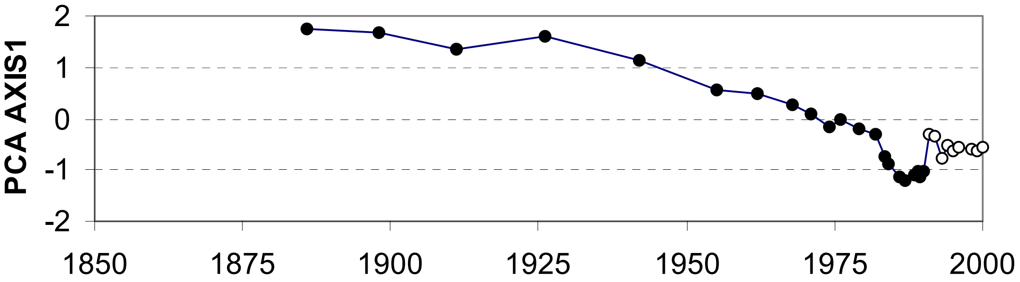
(b) Round Loch of Glenhead



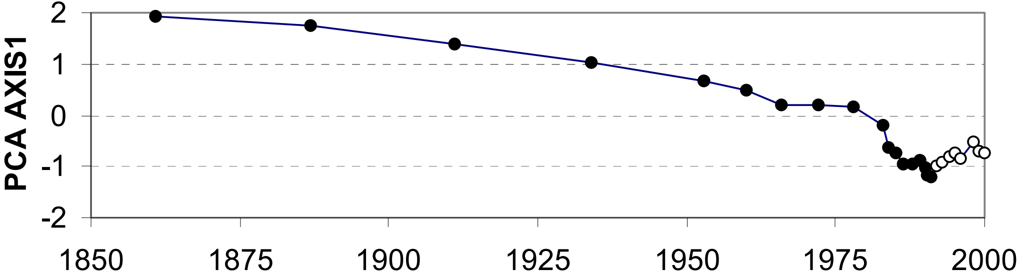
(a) Loch Enoch



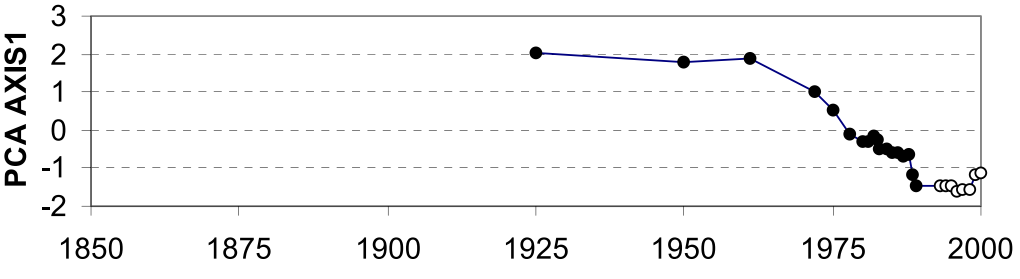
(c) Loch Narroch



(d) Round Loch of the Dungeon



(e) Loch Grannoch



covaried with other chemical properties during the course of acidification. For example, several studies have suggested that DOC concentrations roughly mirrored the pH decline at acidifying lakes. Importantly, data from the UKAWMN now suggest a national scale increase in DOC, which may be entirely independent of changing mineral acidity (Monteith & Evans, 2000).

3.1.3 CCA ordination

3.1.3.1 Assessment of bi-plots

The placing of samples from the integrated datasets in SWAP diatom-chemistry CCA ordination space should provide a clearer view of how species changes relate specifically to changes in pH. Correlations between the first three CCA axes and the three constraining variables are given in Table 3. This shows that Axis 1 is closely associated with pH, while Axis 2 is primarily related to DOC. Total aluminium is best correlated with the third axis.

Table 3 Correlations between SWAP CCA Axes and the 3 chemical variables

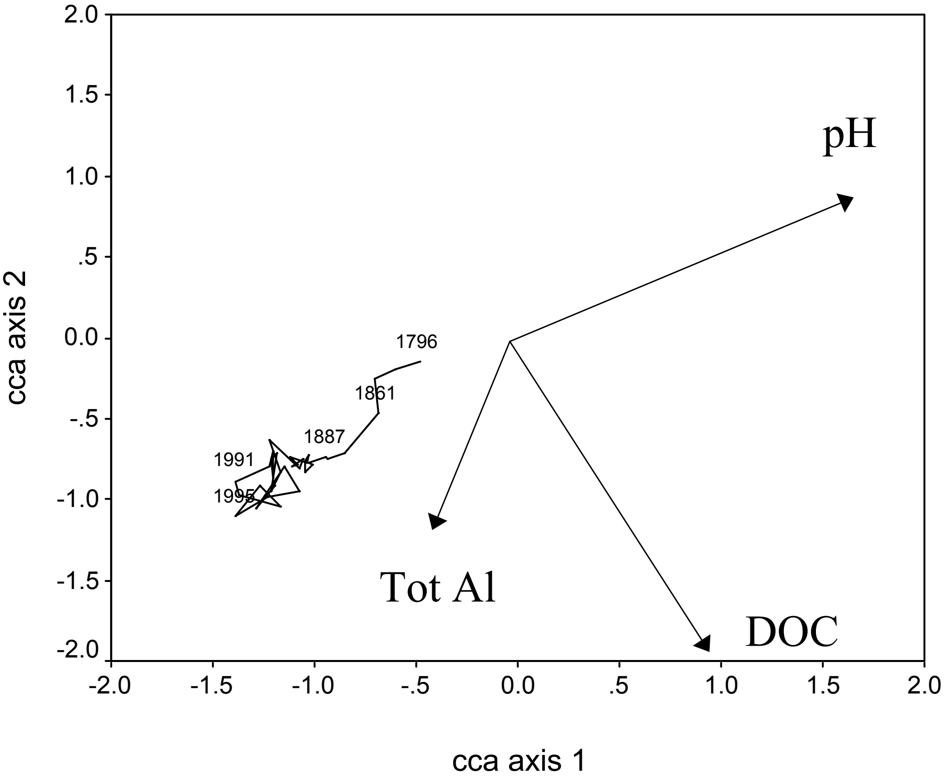
variable	CCA Axis 1	CCA Axis 2	CCA Axis 3
pH	0.83	0.32	0.05
DOC	0.46	-0.73	-0.07
Total Aluminium	-0.22	-0.44	-0.58

Time tracks of core and trap samples on the first two CCA axes for individual lochs are presented in Figures 3a-e. The diagram vectors indicate the direction of maximum change in the three chemical variables between the 131 sites in the SWAP ordination (not shown). Time tracks for the more recent core samples and all trap samples up to the year 2000 are also shown at higher resolution.

Long-term acidification phases, indicated by a shift toward the bottom-left of the ordination diagrams, are clear for Round Loch of Glenhead, Round Loch of the Dungeon and Loch Grannoch in the low resolution figures. Loch Narroch also shows acidification, but over a shorter range, supporting the suggestion that this site was naturally relatively acidic prior to anthropogenic acidification. The time track for Loch Enoch shows no evidence for a trend in pH since the date of the basal sample (1920) although it does suggest considerable short-term variability in pH.

The high resolution figures for Round Loch of Glenhead, Round Loch of the Dungeon and Loch Narroch are broadly supportive of the PCA evidence that very recent changes, consistent with a recovery response, have occurred. In each of these cases, samples from the very top of the sediment core and the early years of sediment traps (c. 1988-1992) lie at the extreme left of the diagrams, indicating that the most acid assemblages occurred at this time. Sediment trap samples representing the last three or four years lie significantly to the right of the most acid samples.

Figure 3a. Round Loch of Glenhead K86 sediment core and sediment trap diatom samples plotted as passive samples within the ordination space of a Canonical Correspondence Analysis (CCA) of diatom and chemistry data for the SWAP dataset. i) low resolution



ii) high resolution

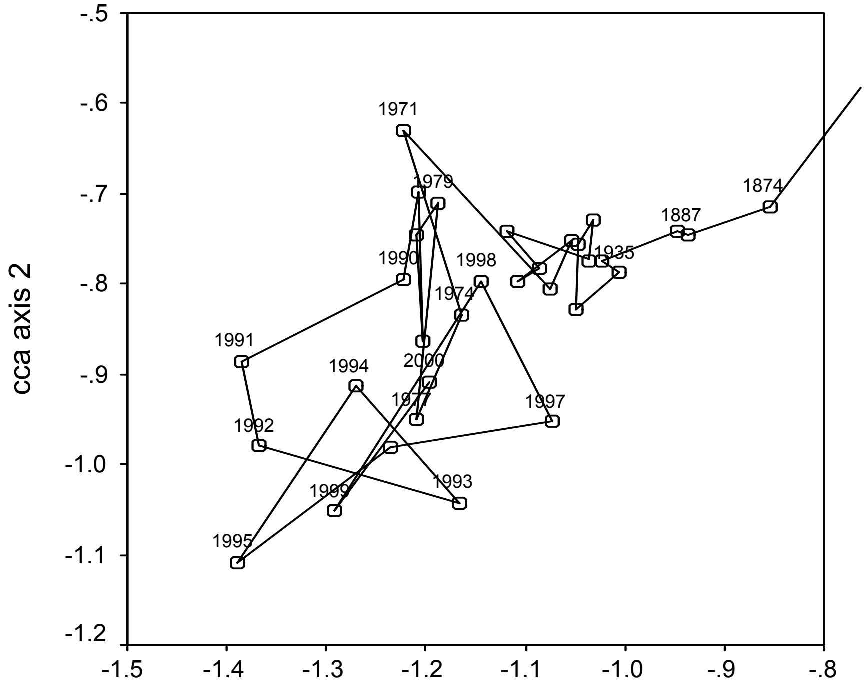
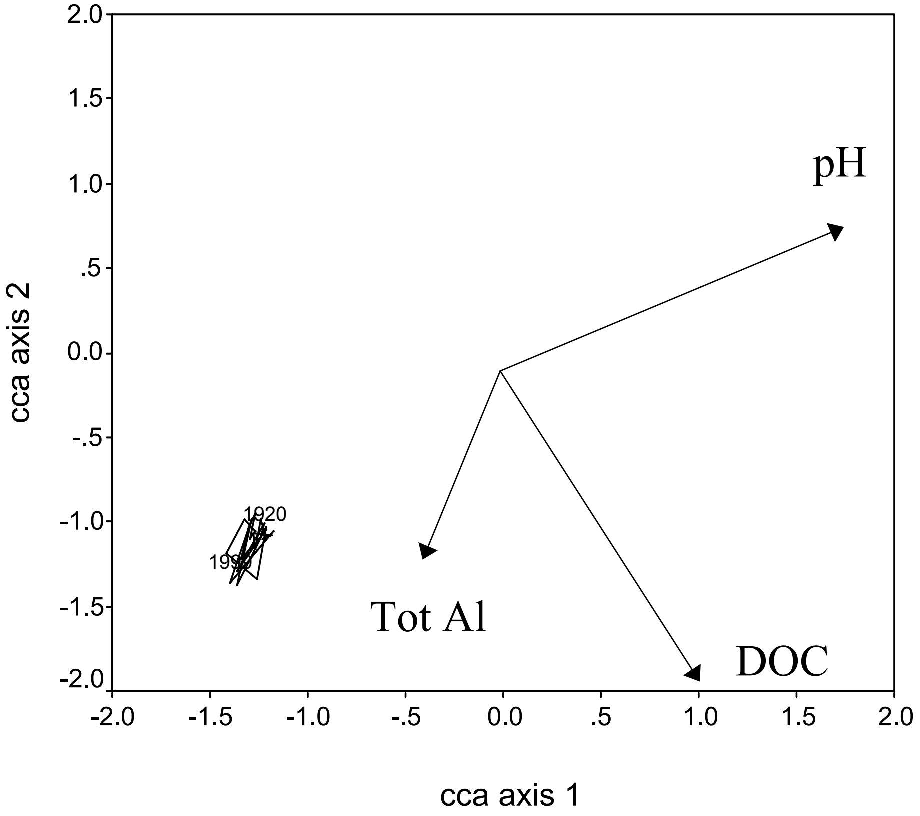


Figure 3b. Loch Enoch sediment core and sediment trap diatom samples plotted as passive samples within the ordination space of a Canonical Correspondence Analysis (CCA) of diatom and chemistry data for the SWAP dataset.

i) low resolution



ii) high resolution

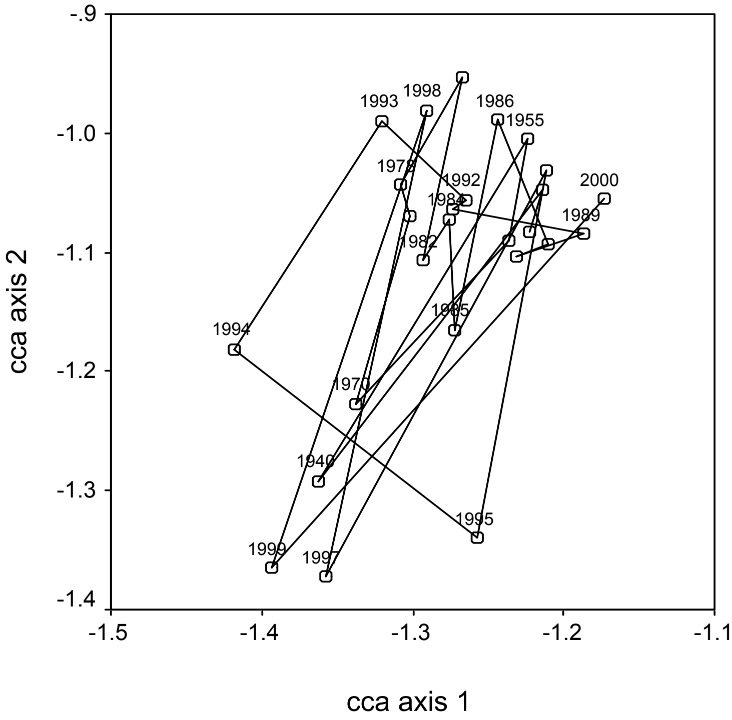
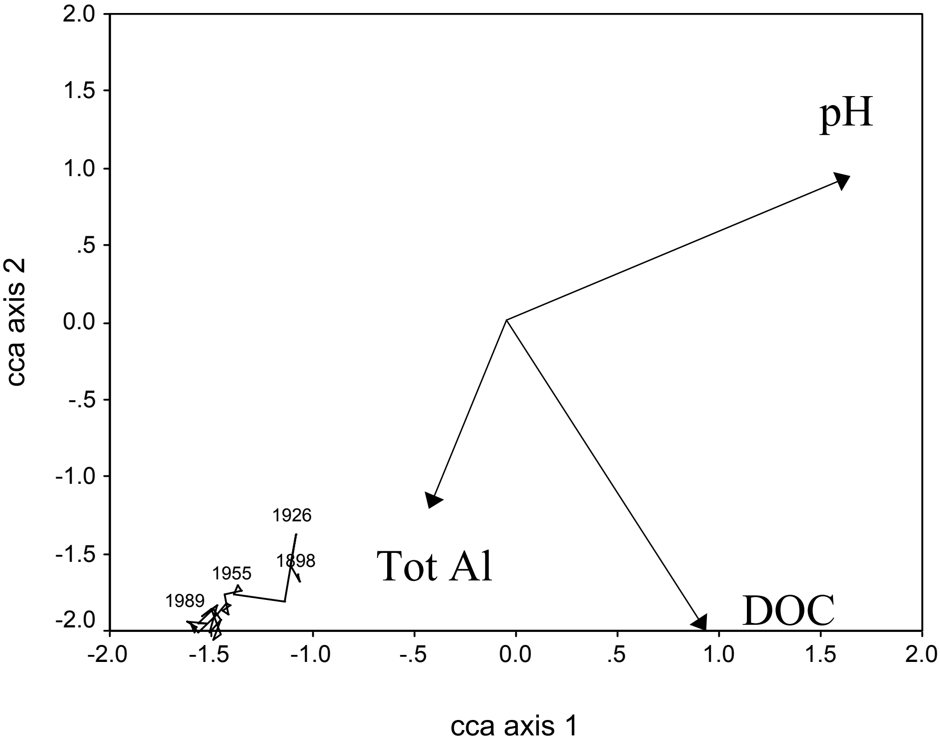


Figure 3c. Loch Narroch sediment core and sediment trap diatom samples plotted as passive samples within the ordination space of a Canonical Correspondence Analysis (CCA) of diatom and chemistry data for the SWAP dataset.

i) low resolution



ii) high resolution

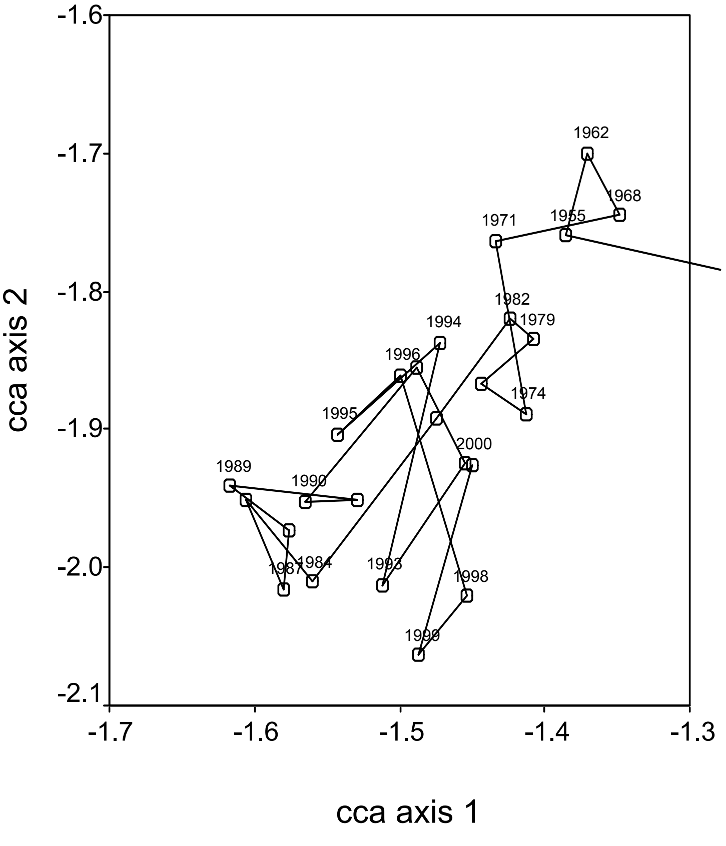
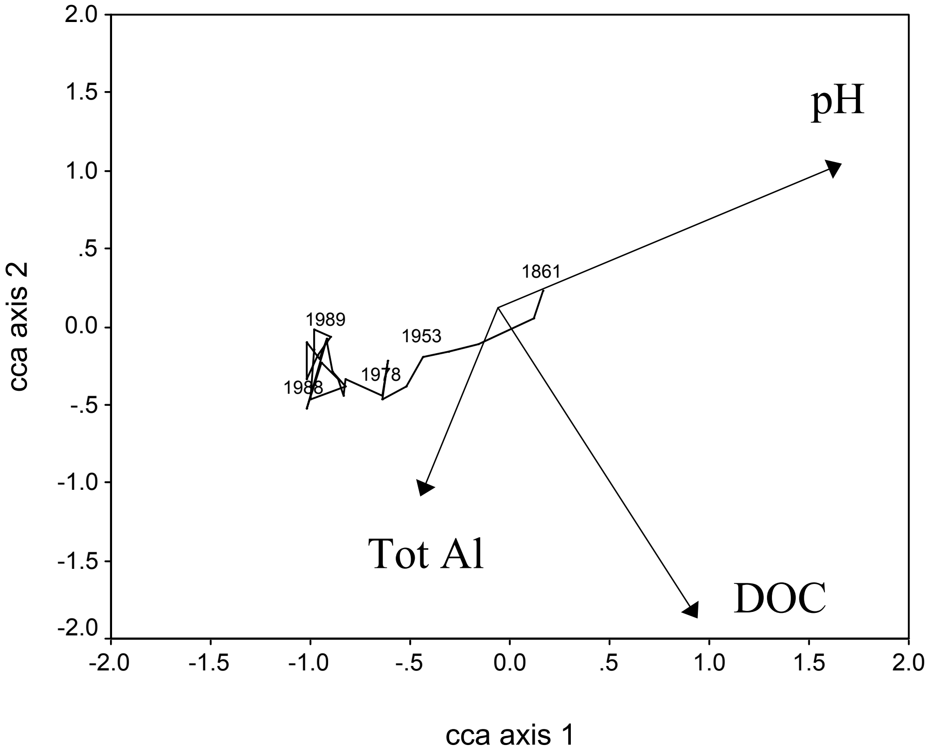


Figure 3d. Round Loch of the Dungeon sediment core and sediment trap diatom samples plotted as passive samples within the ordination space of a Canonical Correspondence Analysis (CCA) of diatom and chemistry data for the SWAP dataset.

i) low resolution



ii) high resolution

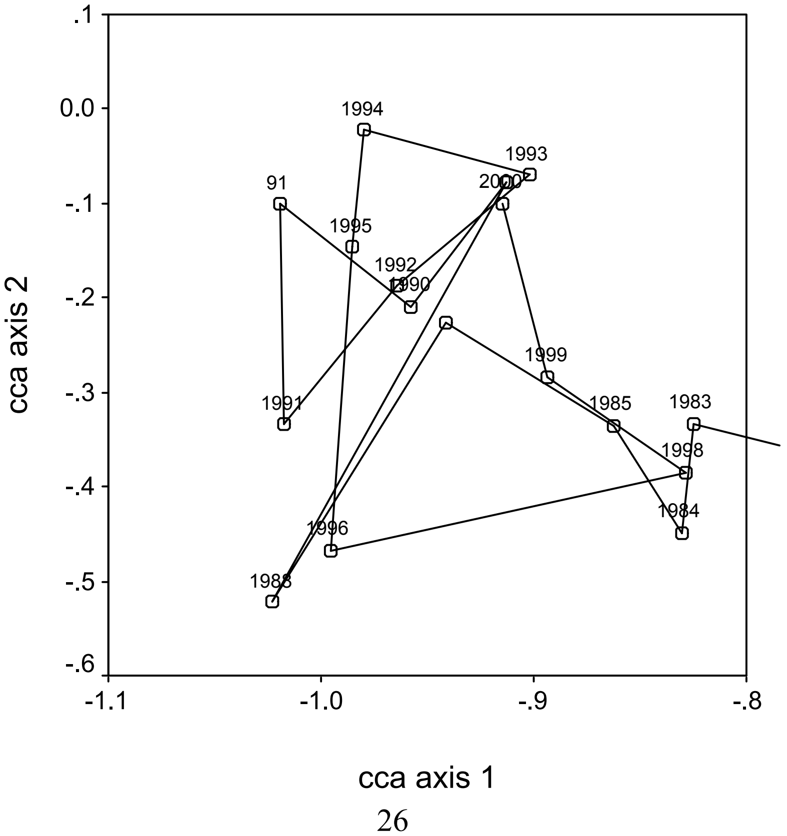
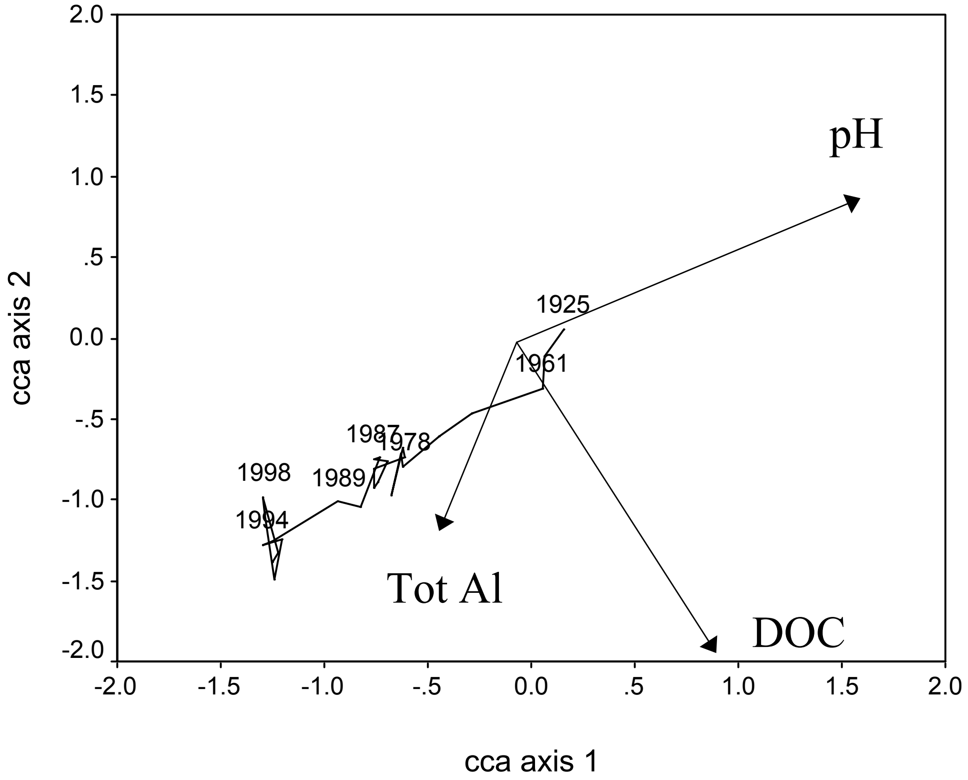
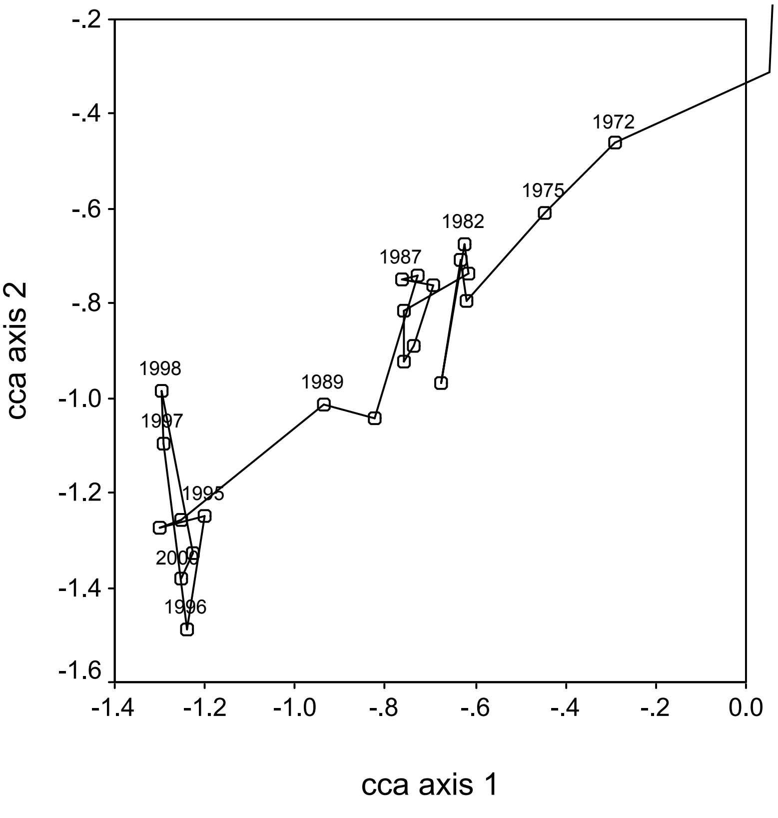


Figure 3e. Loch Grannoch sediment core and sediment trap diatom samples plotted as passive samples within the ordination space of a Canonical Correspondence Analysis (CCA) of diatom and chemistry data for the SWAP dataset.

i) low resolution



ii) high resolution



The Loch Grannoch sediment trap samples all have very similar Axis 1 scores and these lie at the extreme left end of the diagram. Again this supports the PCA evidence for the lack of recovery in Loch Grannoch. Considerable between-year variation is apparent on Axis 2, suggesting recent DOC influence on the assemblage.

No temporal structure is discernible in the CCA time-track for Loch Enoch. Although the trap assemblage for 2000 plots as the least acid of all samples that for 1999 is one of the most acid. These results contrast with the trend in PCA Axis 1 scores which indicate a reversal in the assemblage since 1995, with a possible gradual upward trend since the early 1980s. It appears that while the PCA Axis 1 scores of the most recent samples has been strongly influenced by a decline in *T. quadriseptata*, this species has been replaced by species with very similar optima, such as *Navicula hoefleri* (pH optimum 4.9) and *Eunotia naegelii* (pH optimum 5.0). In addition *Cymbella perpusilla*, which has a relatively high optimum (pH optimum 5.2) has declined over the same period. The cumulative effect of these species changes has been the replacement of one very acid assemblage with another, at least according to our understanding of the pH preferences of these species. Despite the partial species reversal in Loch Enoch sediment since the mid-1990s, therefore, we have no evidence to date that this represents a recovery response.

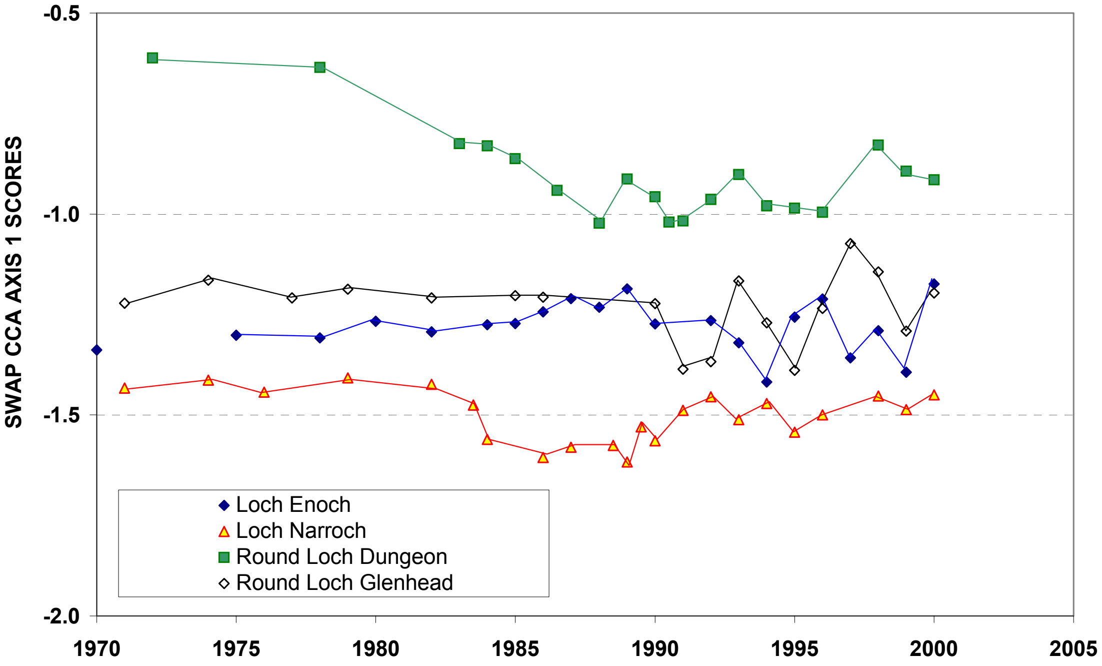
3.1.3.2 Between-site comparison of CCA Axis 1 scores

Time series for CCA Axis 1 scores (the pH-related axis) are compared on two time-scales in Figures 4a and 4b. Unlike the output from the PCA analyses these time trends can be directly compared, since all datasets have been placed passively within the same ordination. Post-1970 changes for all sites other than Loch Grannoch, which shows no evidence of recovery, are presented in Figure 4b. The assemblages of Loch Narroch and Round Loch of the Dungeon show an inflection in the late 1980s. Both sites have shown persistent axis score increases since.

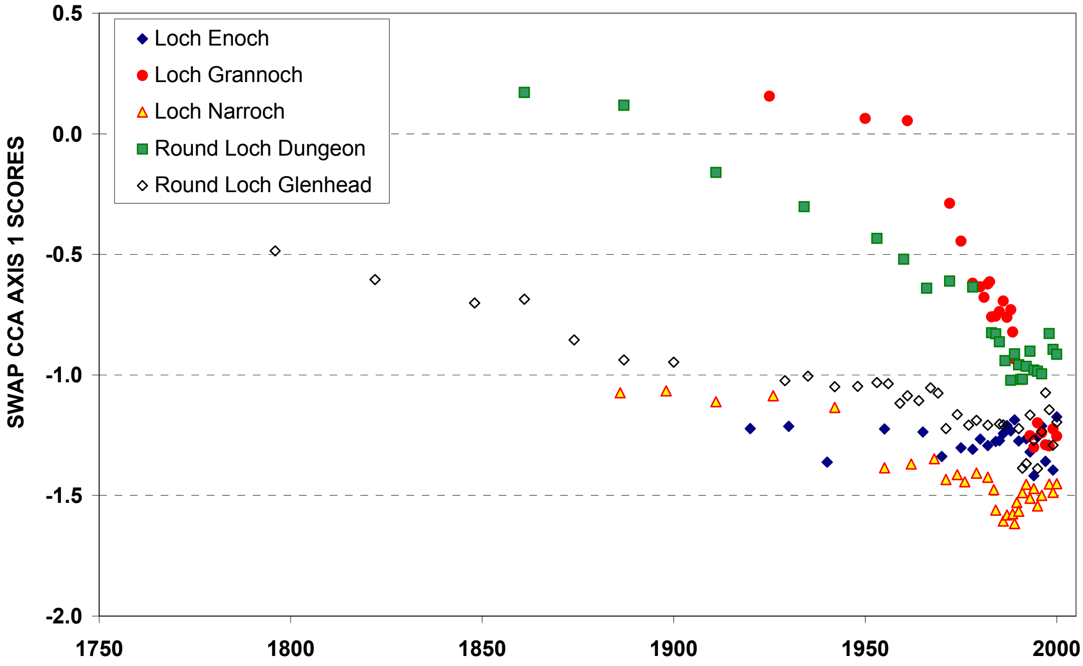
Round Loch of Glenhead also shows slight evidence of a recent upward trend on CCA Axis 1 but inter-annual variation between sediment trap samples is considerable and the trap sample for 1995 has the most negative score for any in the dataset. The last ten years of samples from the Round Loch of Glenhead therefore show a general increase in short term maxima, but assemblages in the more acid years indicate an acidity as low as any in the history of the site. This fluctuation is consistent with the variation in pH of spring water samples analysed by the UKAWMN over the same period, and supports evidence from other sites that the assemblage in annually collected samples is dominated by taxa which bloom in spring. However, pH of samples from Round Loch of Glenhead over the past decade are anti-correlated with the amount of antecedent rainfall as a result of a base-cation dilution effect. It therefore follows that diatom changes over the past decade may partly reflect a hydrological influence on loch acidity. In this respect it is interesting to note that the least negative CCA Axis 1 scores in the trap data are for 1997 and 1998, years in which spring rainfall (total January to April) were particularly low (see Figure 5). Apparent recent recovery in the diatom assemblage of Round Loch of Glenhead may therefore, at least in part, be climatically driven.

Figure 4. CCA Axis 1 scores of sediment core and sediment trap samples from 5 Galloway lochs

(a) all data

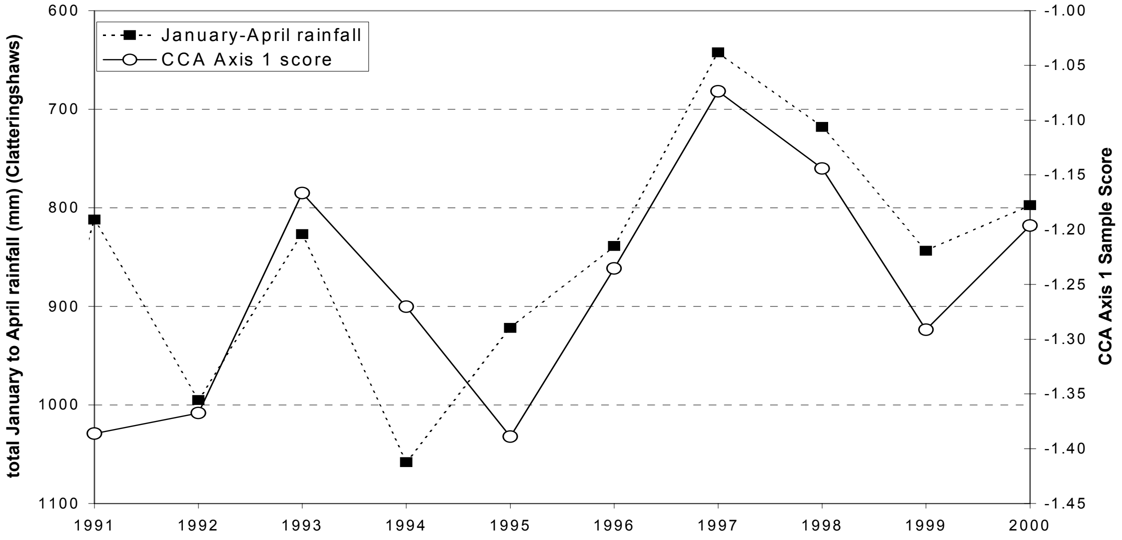


(b) post 1970 data only (Loch Grannoch excluded)



Further time is therefore required before we can adequately separate the effects of natural variability and chemical change at this site.

Figure 5 Inter-annual variation in CCA Axis 1 sample scores for sediment trap samples from the Round Loch of Glenhead and mean January to April rainfall for the nearby weather station (Clatteringshaws) (inverted scale)

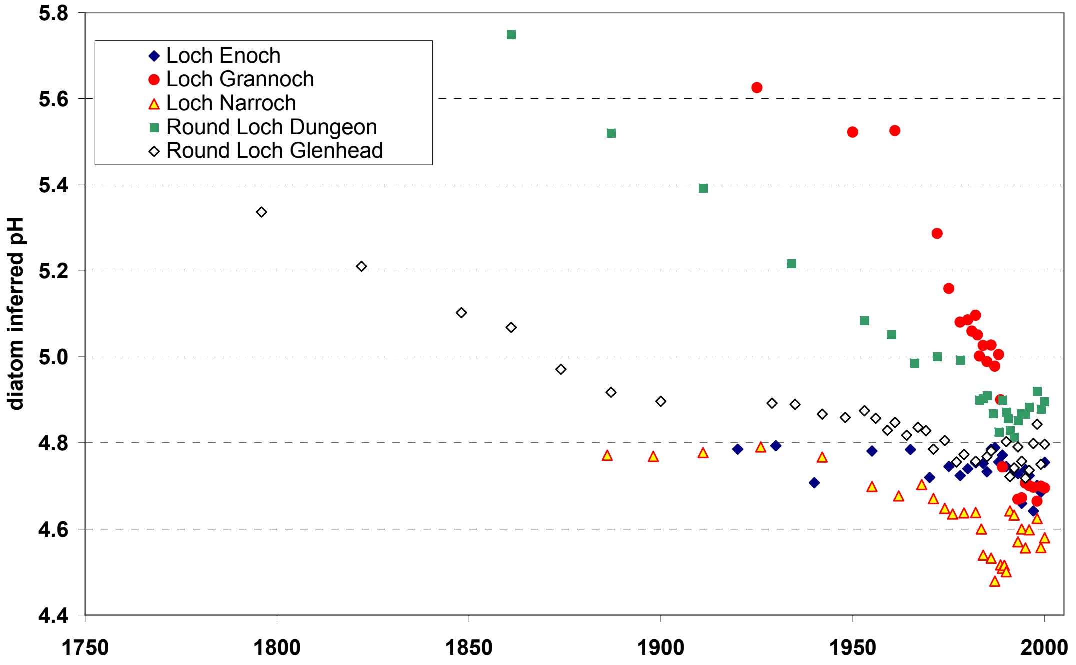


Large between-year Axis 1 variation is also apparent for the sediment trap samples from Loch Enoch. Three samples in the last six years comprised particularly acid assemblages. The apparent small reversal seen in the PCA output is clearly not repeated as an increase in the CCA Axis 1 score. These trap data are therefore in agreement with the results of the new Loch Enoch sediment core analysis (Flower *et al.*, this report).

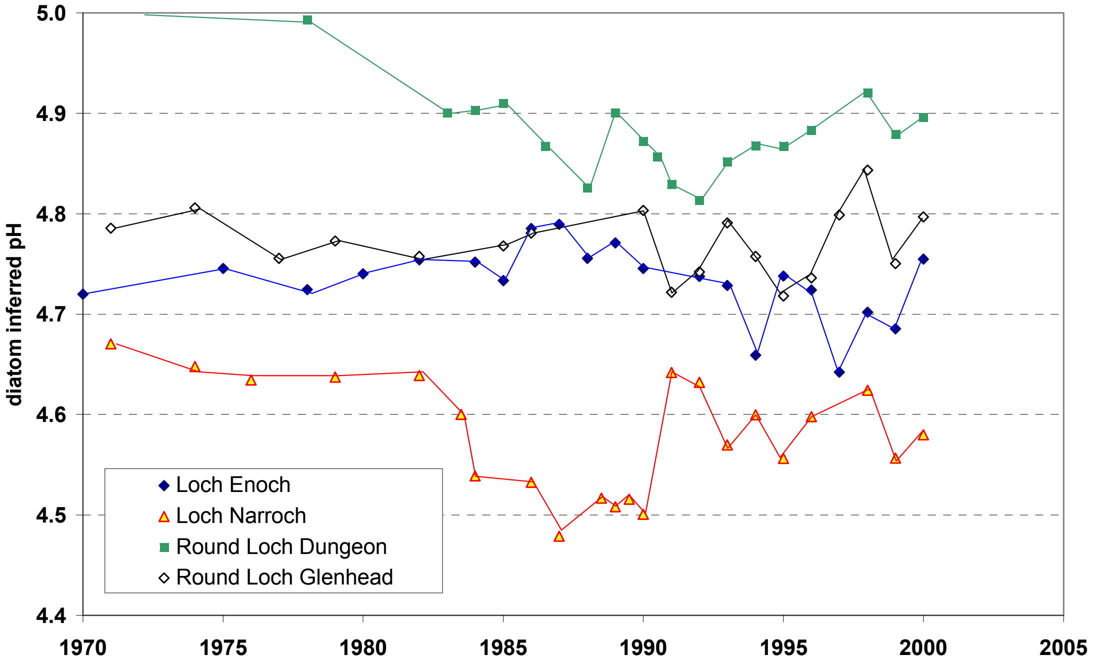
3.1.4 pH reconstructions

WAPLS pH reconstructions for all sites are presented in Figure 6a and 6b. Unsurprisingly, the diatom inferred pH time series are broadly similar to the CCA Axis 1 time-series in Figures 4a and 4b. The plots indicate the magnitude of acidification which has occurred over the time represented by the available sediment record at all sites other than Loch Enoch, and emphasise the rapid recent acidification of Loch Grannoch. Changes indicative of rising pH in the recent record are clear for Round Loch of the Dungeon and Loch Narroch, while the absence of recovery in Loch Grannoch is equally clear. Large inter-annual fluctuations in diatom inferred pH are evident in the sediment trap record of Round Loch of Glenhead and Loch Enoch. There is little evidence for increasing pH at Round Loch of Glenhead, although as with the CCA Axis 1 scores, samples from 1997 and 1998 are indicative of less acid conditions.

Figure 6. Diatom inferred pH of sediment core and sediment trap samples from 5 Galloway lochs



b) post-1970 data only (Loch Grannoch excluded)



3.2 New sediment core study

Previous studies have shown that sediment cores from the deep water in the Round Loch of Glenhead and Loch Enoch typically have sediment accumulation rates of 1.0 – 1.5 mm yr^{-1} (Allott, 1991; Allott *et al.,* 1992). The top 32 mm of the Round Loch core (KYKA) is therefore likely to represent the last 20 – 30 years of accumulated sediment, while the 120 mm record from Loch Enoch (ENO10) is likely to cover 70-80 years. Both records should therefore cover the period of measured chemical recovery (i.e. 1980 to the present) (Monteith & Evans, 2000).

3.2.1 Qualitative assessment of species changes

In common with the sediment trap samples from the Round Loch of Glenhead, the diatom assemblages within the uppermost sediments of KYKA are dominated by the acidobiontic *Tabellaria quadriseptata* and the acidophilous *Eunotia incisa* and *Navicula leptostriata*. Other common species include the acidobiontic taxa *Tabellaria binalis*, *Navicula hoefleri* and *Navicula madumensis* and the acidophilous *Brachysira brebissonii*, *Eunotia naegelii*, *Frustulia rhomboides* var. *saxonica*, *Peronia fibula*, and *Tabellaria flocculosa*. The data are noisy with 131 taxa present and many zero values. Despite this variation, however, there is general trend of a recent reduction in the relative abundances of *T. quadriseptata* and *N. madumensis*, and an increase in the relative abundances of *E. naegelii* and *N. leptostriata* taxa. The former two species are highly tolerant of low pH conditions (Stevenson *et al.,* 1991). Declines in the abundances of these species were a key feature of the uppermost sediments of sediment cores taken from the RLGH in 1989-90 following the modest increase in loch-water pH observed in the 1980s (Allott *et al.,* 1992). The trend in the latter species corresponds to observed post-1996 increases in its abundance in both epilithic and sediment trap diatom assemblages (Monteith & Evans, 2000).

Samples from the Loch Enoch (ENO10) core are also very similar floristically to sediment trap samples, and have much in common with the Round Loch core, the main exceptions being the absence of *N. leptostriata* and the relative abundance of *Cymbella perpusilla.* The frequency of *T. quadriseptata* (one of the most acid-loving species) in the top 4 mm of sediment is as high as it has ever been. Frequencies of the slightly less acid species, *E. incisa*, remain remarkably constant over the upper 50 mm of the core. Although ENO10 was not dated radiometrically, we inferred from previous dated cores that a peak in *Asterionella ralfsii* at 45-50 mm depth dates to the 1950s. The core is therefore conformable with earlier cores taken from this loch, and it is clear that sediment deposited over the last decade (upper ~ 20 mm) shows no sign of recovery of diatom species more characteristic of less acid conditions.

3.2.2 PCA ordination

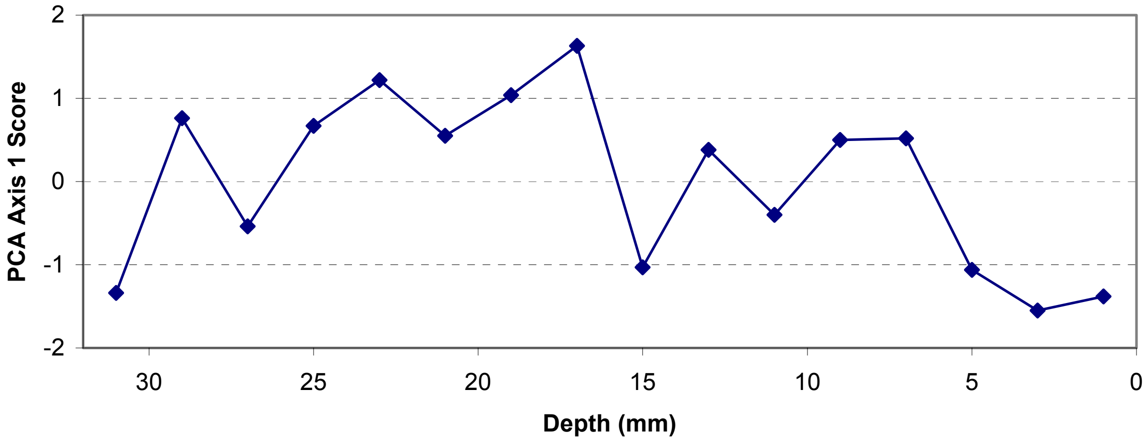
A PCA of the Round Loch core revealed a single statistically significant axis of variation in the diatom assemblages (PCA axis 1). This axis explains 25.2% of the variation in the diatom abundances. This figure is relatively high given the noise within the data. High axis 1 scores are associated with relatively high abundances of *N. madumensis* and *T. quadriseptata* (Table 4). Low axis 1 scores are associated with relatively high abundances of *N. leptostriata* and *E. naegelii*.

Table 4 PCA axis 1 Species scores for the Round Loch of Glenhead core (KYKA)

Taxon	**Axis 1 Score**	**Species Fit (%)**
Navicula leptostriata	- 0.85	71.0
Navicula madumensis	+ 0.83	69.1
Tabellaria quadriseptata	+ 0.64	40.5
Eunotia naegelii	- 0.62	37.8

The trend in PCA axis 1 scores is shown in Figure 7. This reveals a general pattern of low scores in the basal sample, higher scores between 30 and 16 mm, and lower scores between 16 mm and the core top. There is a clear trend of decreasing scores between 20 mm and the uppermost sediments, with possible step changes at 16 mm and 6 mm. This trend represents a decline in the relative abundances of *T. quadriseptata* and *N. madumensis* an increase in the relative abundances of *E. naegelii* and *N. leptostriata*. Although the diatom data are noisy, they are therefore consistent with a diatom response to increasing loch-water pH.

Figure 7 PCA axis 1 scores for the Round Loch of Glenhead core (KYKA)



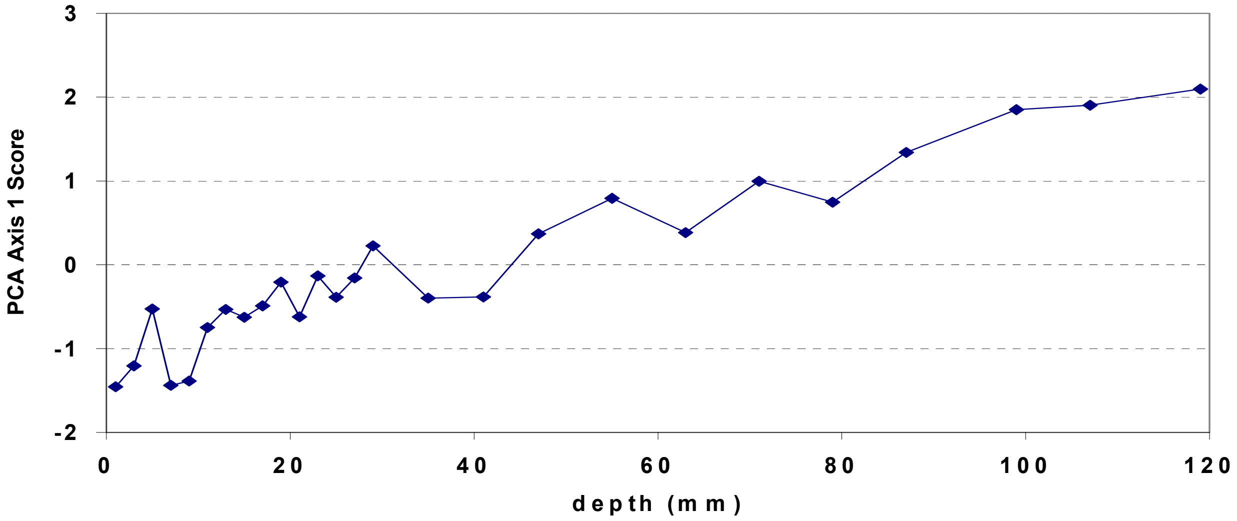
PCA of the new core from Loch Enoch also showed only 1 significant axis of variation.

Commonly occuring species with the highest correlations with PCA Axis 1 are given in Table 5. Given the strong relationship between *T.quadriseptata* and the first axis it is clear that the long-term downward decline in the Axis 1 score (Figure 8) is likely to reflect the acidification process. Although it would appear that this trend halted by 40 mm depth there is little evidence for a marked species reversal since. In contrast to the sediment trap record there is no indication of a recent decline in the frequency of *T. quadriseptata*.

Table 5 PCA axis 1 Species scores for the Loch Enoch core (ENO10)

Taxon	**Axis 1 Score**	**Species Fit (%)**
Tabellaria quadriseptata	-0.95	89.79
Eunotia naegelii	0.83	68.46
Cymbella perpusilla	0.81	66.14
Eunotia incisa	0.66	43.95

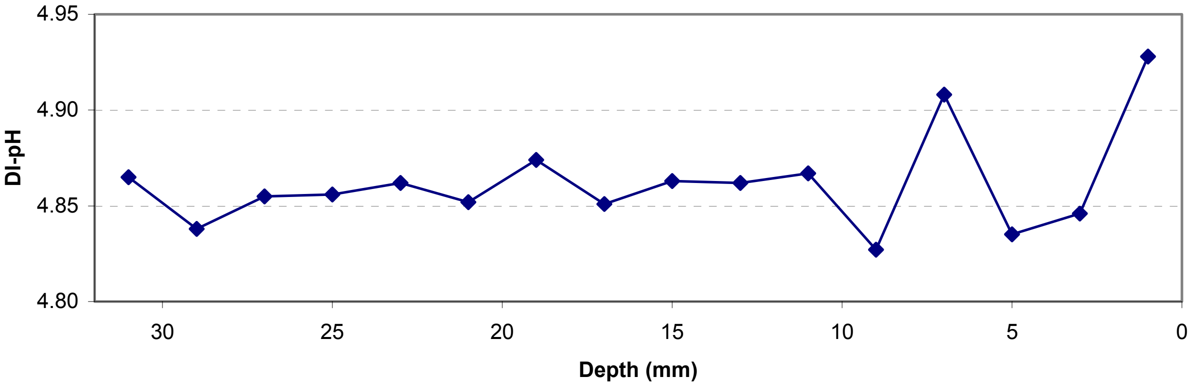
Figure 8 PCA axis 1 scores for the Loch Enoch core (ENO10)



3.2.3 pH reconstructions

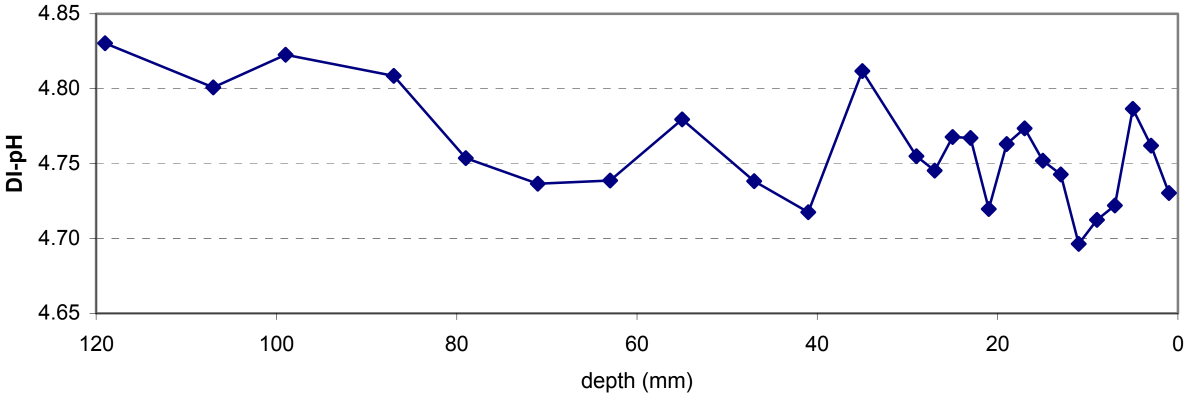
The diatom-inferred pH (DI-pH) values for the Round Loch core (KYKA) are shown in Figure 9. There is a slight trend of increasing DI-pH, with the highest DI-pH values found in the uppermost sediments. However, the gradient of change is very slight, while the data are again noisy and not statistically significant, and must therefore be interpreted with caution. Values of DI-pH are particularly variable in the uppermost 10 mm of the core.

Figure 9 Diatom-inferred pH for the Round Loch of Glenhead core (KYKA)



The diatom-inferred pH record for the Loch Enoch core (ENO10) (Figure 10) suggests slight acidification over the last few decades, but provides no evidence of recovery at the top of the core. This is entirely consistent with the findings of the core–trap study which identified large inter-annual variability over the last decade but no evidence of a trend toward less acid assemblages.

Figure 10 Diatom-inferred pH for the Loch Enoch core (ENO10)



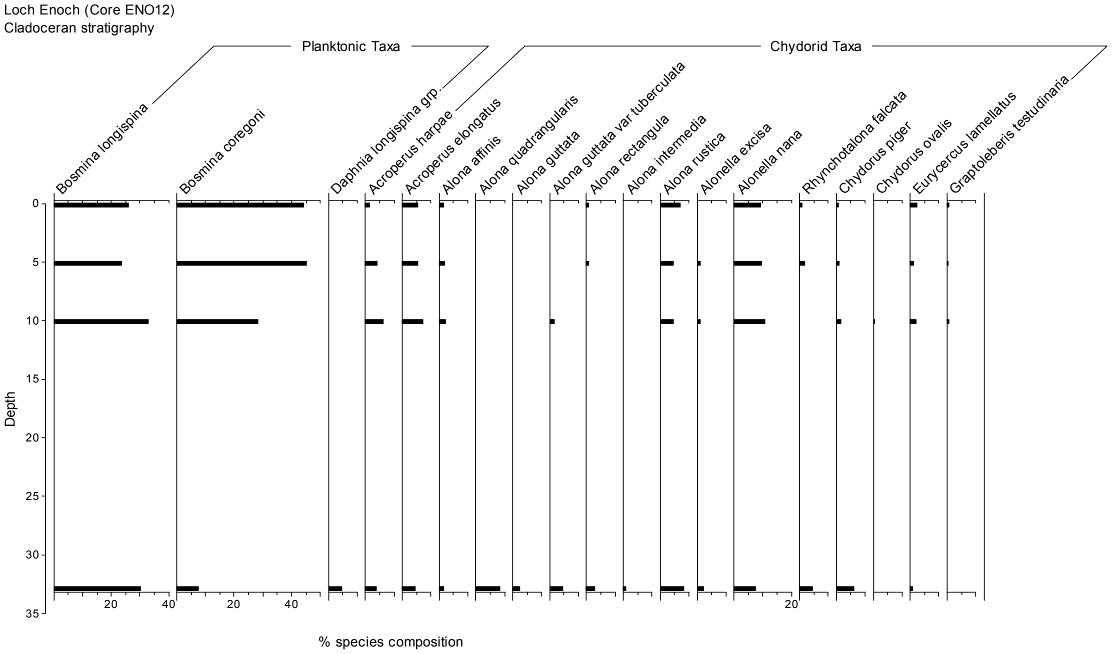
3.3 Cladoceran counts from ENO12

Cladoceran fossil counts for the four samples from ENO12 are presented in Figure 11. The pre-acidification sample (32.8-33.0 cm) indicates that a diverse cladoceran community existed in Loch Enoch. This community is typical of many upland oligotrophic lakes. Three planktonic species were found in this sample, *Bosmina longispina*, *Bosmina* c.f. *coregoni* and *Daphnia* c.f. *longispina* grp.. While the former two species are charactersitic of acid, oligotrophic lakes, *Daphnia* are acid-sensitive and their occurrence therefore indicates conditions were much less acidic than today. Furthermore the presence of *Daphnia* and other large body cladocerans (e.g. *Eurycercus lamellatus*) suggests that zooplanktivorous fish were either absent or present in low numbers.

This diverse flora has disappeared from the 10 cm and subsequent samples, the acid sensitive *Daphnia* are absent and the dominance of the planktonic Bosminidae *(B. longispina* and *B. coregoni*) are in stark contrast to the pre-acidification sample. *B. longispina* and *B. coregoni* are typical of acid, upland systems. 4 species found in the pre-acidification sample from Loch Enoch are not found above 10-cm (*Daphnia longispina* grp, *Alona quadrangularis*, *Alona guttata*, *Alona intermedia*). Of the chydorid taxa, only those species commonly found in acid waters remain (e.g. *Alonella nana*, *Alona rustica*, *Acroperus elongatus*, and *Acroperus harpae*).

The 5-cm level in the period of strong acidification shows a further increase in the dominance of the planktonic Bosminid species and declining proportions of the Chydoridae in the cladoceran assemblage. There also appears to be a shift towards *B. coregoni* as the dominant member of the planktonic cladocera. The taxonomy *of B. longispina* and *B. coregoni* is complex, however, and it is unclear whether the two taxa are indeed separate species (c.f. Hellsten & Sundberg, 2000). The two taxa may be morphotypes of a single species (Freyer, 1993) or the shift in species may be the result of a morphological response to increasing lake acidity or shifts in predator dynamics as the loch acidified.

Figure 11 Cladoceran counts from ENO12



The uppermost sample shows little recent change in the cladoceran fauna of Loch Enoch. The present day cladoceran community is an impoverished assemblage typical of acid, oligotrophic, upland surface waters. Planktonic Bosminid species and a depleted chydorid community of acid tolerant taxa today dominate Loch Enoch.

These results are in accordance with those indicated by the diatom results in this report, showing no signs of recovery in the cladoceran fauna of Loch Enoch. However, it must be noted that the samples counted are only a skeleton stratigraphy. Further damage to the cladoceran fauna between 5 cm and the surface may have occurred and the surface sediment assemblage may show some improvement if this is the case. This cannot be proved or disproved with the current data. This work does however illustrate the potential of such analysis for broadening our understanding of ecosystem damage and recovery of upland acidified lakes.

4. Discussion

The premise for this work is that some chemical recovery has occurred in the Galloway lochs as a consequence of emissions reductions, but it is worth considering the chemical evidence in more detail. Samples have been analysed for water chemistry from all five lochs included in this analysis since 1979 although continuous (i.e. regular) monitoring has only been undertaken since around 1987. pH trends show a common pattern, with the most acid samples (n=2 or 3) generally occurring in 1979, slightly less acid samples around 1984, a plateau from around 1988 to 1994 and then a relatively rapid increase from 1995 to the present (Harriman pers. comm). Although the depressed pH of samples in 1984 can in part be explained by sea-salt effects (Cl concentration is very high at this time), the 1979 pH values provide persuasive evidence for an amelioration in acidity over the last 20 years. This is certainly consistent with the observed fall in non-marine sulphate deposition at Eskdalemuir over the same period. Continuous monitoring does provide irrefutable evidence of a substantial decline in non-marine sulphate since the mid-1990s and this has been accompanied by a clear rise in pH at many sites, including all sites examined in this report with the exception of Loch Grannoch.

Acid impacted freshwater ecosystems are unlikely to show linear recovery responses to improvements in water chemistry. Acid-sensitive primary producers, such as diatoms, (which may have survived in local refugia, in resting stages, or as a reduced population), might be expected to respond relatively rapidly, however, further up the food chain recovery is likely to be more dependent on a complex interaction of biotic and abiotic factors, distance from refugia and dispersal/migration mechanisms. Substantial time lags (hysteresis) are possible before conditions are favourable for the re-establishment of many organisms. This report has concentrated solely on identifying responses in one biological group. As a major group of primary producers in acid-sensitive systems, diatoms have a role in underpinning the freshwater ecosystem as a whole (although it is as yet unclear how important the replacement of one species by another may be for grazing organisms). While a diatom response does not necessarily imply that other biological improvements are occurring, it is possible that a reversal in the microflora is a prerequisite for changes further up the food chain. In addition to their role as sensitive indicators of changing acidity, diatoms may therefore be viewed as indicators of the potential for recovery of the wider ecosystem.

The diatom assemblages of Loch Narroch and Round Loch of the Dungeon show clear changes toward communities which were formerly common (prior to acidification). The specific changes at these two sites can be clearly linked to falling acidity and thus provide powerful evidence that biological recovery, at least at this trophic level, is now occurring. It would be interesting to investigate whether the populations of other biological groups, such as benthic chironomids and zooplankton, remains of which also preserve well in sediments, have also improved in these lochs. Such work is necessary before we can be confident that ecosystem scale recovery is taking place.

The absence of any recovery in Loch Grannoch is consistent with the water chemistry data of this UKAWMN site. Over the 1988-1998 period Loch Grannoch was the only site to show a further decline in pH. Earlier palaeoecological investigations have revealed a possible diatom response to a very large rock phosphate application in the catchment in the mid-1980s in the course of forestry management. It is therefore possible that the recent trend in water chemistry and the biological response reflects a gradual decline in base-cations derived from this fertiliser.

Slight species reversals have occurred in Round Loch of Glenhead and Loch Enoch. However it has not been possible to convincingly link these with an acidity response using direct ordination and weighted averaging methods. For the Round Loch of Glenhead, changes in the diatom community since the mid-1990s are consistent with the rise in pH measured over the same period (Evans and Monteith, 2001). Inter-annual variability in the assemblage is, however, considerable and can be linked to climatic factors. Further samples will be necessary to demonstrate whether the recent upward trend is sustainable.

The results for Round Loch of Glenhead suggest that the recovery identified earlier by Battarbee *et al.* (1988) and Allott *et al.* (1992) has not been sustained as the sediment trap samples in the early 1990s are indicative of the most acid conditions ever experienced at the site. However, it is important to take into account the difference in temporal resolution of the two sample types. The scale of inter-annual variability observed in the assemblages of the sediment trap samples would be diminished within a sediment core subject to bioturbation and other smoothing processes. We cannot therefore rule out the possibility that the most acidic assemblages were deposited in the mid-1980s, since the species composition of the sediment from this time could have been "diluted" by that deposited before and/or since.

The apparent recent, but very slight, species reversal in Loch Enoch does not translate into a response to rising pH according to our understanding of the pH optima of the dominant species in the assemblages. This change could therefore represent:

1. a species response to an unknown environmental factor.
2. a response to a small decline in acidity which is too fine to be quantified in terms of pH using the SWAP species – water chemistry relationship.
3. a response to a small decline in acidity which differs from the expected response determined from the spatial species – acidity relationship of the SWAP model.

Further years of monitoring are clearly necessary before we can properly understand and test the significance of these recent changes.

5. Conclusion

Within our restricted region of five lochs, recent sedimentary diatom trends can be classified in three categories:

1) Loch Grannoch - no chemical improvement – no change in the diatom assemblage
2) Round Loch of the Dungeon & Loch Narroch – rising pH (at least since the mid 1990s)– strong, linear diatom reversal indicative of a response to recovery from acidification (since the mid-1980s).
3) Round Loch of Glenhead and Loch Enoch – rising pH (at least since the mid-1990s) – weaker non-linear diatom change – not directly attributable to chemical recovery to date.

While the absence of biological response in Loch Grannoch is possibly related to forestry practice, it is feasible that the difference in response between the other two categories may also reflect catchment characteristics. Round Loch of the Dungeon and Loch Narroch have relatively small catchments and loch chemistry is likely to relate closely to deposition chemistry. The larger catchments of the remaining two lochs make them more dependent on soil-ion exchange processes including sulphur desorption and hydrogen ion release following sea-salt events, and on hydrological variations in the contribution of base-flow to surface-flow. Inter-annual variation in these factors is likely to be strongest during the winter and spring, which may also be the time of maximum diatom productivity. Climatic influences on acidity at this time of year may therefore have been large relative to the decline in anthropogenic pollutant acidity. It is also possible that improvement in other biotic or abiotic factors are necessary before the diatom flora of these sites can to return towards a pre-acidification state. For example, the toxicity of other atmospherically derived pollutants, such as heavy metals, could be preventing the re-establishment of sensitive species. Hysteresis in the diatom communities of these two lochs is therefore perhaps not surprising and further time may be necessary before recovery responses become obvious.

It is clear from this exercise that the analysis of the contents of annually sampled sediment traps in conjunction with sediment from dated cores provides a powerful method for placing recent biological changes in a longer term context. Sediment traps continue to be deployed in the five Galloway lochs, in addition to the 9 other lakes on the UKAWMN. Although we have only attempted to examine trends in diatoms, similar approaches could be taken for a variety of other biological groups which leave fossil remains, and this would provide us with a better understanding of recovery processes across the broader ecosystem.

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Task 2.2: Run the MAGIC7 model in both site-specific and regional modes

C. Evans, R. Ferrier, R. Helliwell, and A. Jenkins

2.2 Run the MAGIC 7 model in both site specific and regional modes

2.2.1. Site Specific MAGIC 7 application to AWMN sites

The MAGIC model was calibrated previously to the 22 sites in the UK AWMN in 1995 using the mean surface water chemistry from 1988 – 1994 to define the calibration targets and 1991 as the 'present day' reference year (Jenkins *et al.* 1997). The calibrated suite of models was used to assess the changes in surface water chemistry to 2041 in response to the Oslo Protocol. These calibrations have been up-dated where possible to a 'present day' reference year of 2000 and used to assess the changes in surface water chemistry in response to the Gothenburg Protocol. Best (MAGIC5) and worst (MAGIC7) cases of N retention are modelled although this is not possible at several sites since soil C/N data are not available. In addition, annual deposition fluxes have been calculated from adjacent deposition collectors in the ADMN and used to drive the model from 1988 – 1998 to enable comparison with the annual mean runoff chemistry from the AWMN and so providing an independent test of the model.

Data Sources and Model Calibration

Catchment physical data, including soil chemistry and physical characteristics, were unchanged from the earlier calibrations. Forest management information, including felling and planting operations, was up-dated to 2000. At forested sites, future predictions assume felling of each stand at 50 years age and immediate replanting. Where possible, new soil samples were collected from the upper organic layers and analysed to estimate the C and N content. These data were not available for Old Lodge, Narrator Brook, Burnmoor Tarn, Scoat Tarn, River Etherow and the four sites in N. Ireland.

Calibration of the model to 'present day' requires an estimate of mean annual chemistry. From the recent analysis of the AWMN chemistry data (Monteith and Evans, 2000), the recent trends towards decreased xSO_4 preclude the use of mean data for the whole monitoring period as a calibration target. Consequently, the mean of the 1988 – 1993 period, when no significant trends were observed, was used for calibration. Wet deposition data for the same period was calculated from the nearest site in the ADMN and appropriate sea-salt and dry deposition factors were calculated for each site for Cl^- and SO_4^{2-}, respectively, on the basis of conservativity. Chloride in the model has no adsorbed phase and SO_4^{2-} adsorption is assumed to be low and currently at maximum capacity. Total deposition of NO_3^- and NH_4^+ was derived from the UK 95-97 deposition data. The observed trend in wet deposition of xSO_4 from 1990 was used to scale the deposition to 2000 (Smith *et al.*, 2000) for each site.

For model testing over the period 1988 – 1998, annual mean deposition chemistry from adjacent ADMN collectors was used. Measured concentrations of NO_3^- and NH_4^+ in wet deposition were enhanced in each year by the ratio of total N deposition 95/97 for the relevant 5 x 5 km square to wet N deposition 95/97. Annual rainfall totals from the ADMN gauges were used and evapotranspiration was calculated on the basis of percentage mature forest cover in the catchment (moorland = 10%, mature forest = 25%) and assumed constant in each year.

Future forecasts were run for two scenarios; the Gothenburg Protocol and constant deposition at present level. Future deposition estimates for S and N for the Gothenburg Protocol were derived from HARM model outputs. Deposition reduction factors were derived from HARM

11.5/FRAME as the ratio of HARM 2010: HARM 1995. In addition, two different model structures were used to describe future N dynamics. The best case assumes that the current retention of N will continue into the future as a proportion of input (MAGIC5). The worst case utilises a finite store of N within the soil and an empirical relationship to describe the proportion of N leakage in relation to the store size and the deposition flux (MAGIC7).

Modelling Results

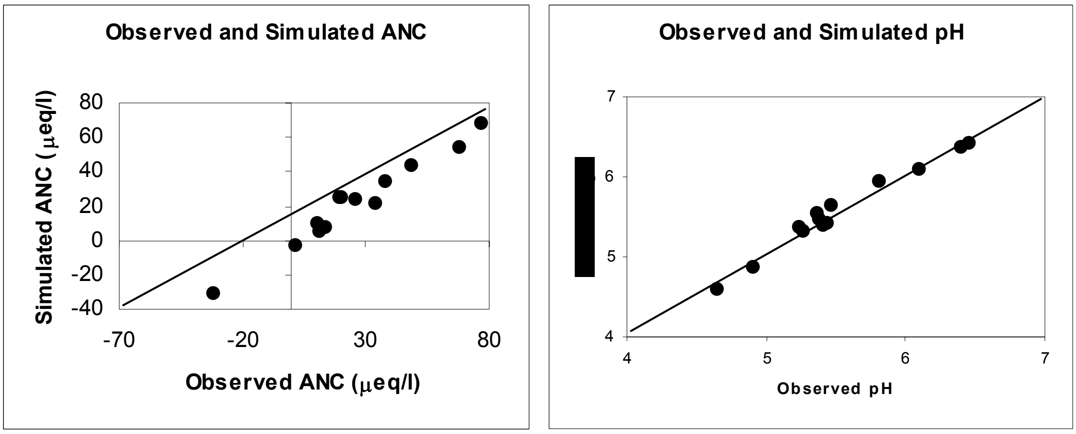
The calibrated model successfully matches mean 1988-1992 major ion chemistry (Table 2.2.1.1) and pH (Figure 2.2.1.1b) at all sites and there is no systematic bias in the calibrated determinands. Comparison with observed ANC (not calibrated) also shows broad consistency from site to site (Figure 2.2.1.1a). Inconsistencies between observed and simulated data at individual sites may in part be attributed to the method used to calculate the mean annual ANC. This is calculated from the mean annual ion concentrations rather than as a mean of the ANC of each sample taken within the year. Nevertheless, the observed relationship between pH and ANC across the suite of sites (Figure 2.2.1.1) is closely matched by the model simulations.

Table 2.2.1.1. Calibrated vs observed chemistry, AWMN sites used in MAGIC application

Site	**Ca**		**Mg**		**SO_4**		**NO_3**		**ANC**		**pH**	
	Obs	**Calib**	**Obs**	**Calib**	**Obs**	**Calib**	**Obs**	**Calib**	**Obs**	**Calib**	**Obs**	**Calib**
Loch Coire nan Arr	43	43	73	73	41	42	3	3	48.5	44.4	6.40	6.38
Allt a'Mharcaidh	43	43	34	37	44	49	2	2	67.9	53.8	6.45	6.44
Allt na Coire nan Con	59	61	81	82	62	63	5	5	38.1	34.7	5.81	5.95
Lochnagar	29	30	37	38	61	64	11	10	11.0	6.0	5.40	5.41
Loch Chon	76	78	56	57	72	74	10	8	25.9	23.5	5.47	5.65
Loch Tinker	86	87	52	50	53	53	2	2	76.6	68.1	6.10	6.09
Round Loch of Glenhead	35	36	56	57	68	71	5	5	1.5	-2.5	4.90	4.88
Loch Grannoch	52	52	66	68	98	102	14	13	-32.2	-31.0	4.65	4.60
Dargall Lane	52	55	66	66	82	86	11	9	10.5	9.6	5.44	5.43
Llyn Llagi	67	66	58	58	62	64	11	10	18.9	25.4	5.23	5.38
Llyn Cwm Mynach	95	98	80	82	88	91	22	22	20.4	25.1	5.38	5.48
Afon Hafren	48	52	78	78	83	87	21	19	13.8	7.2	5.27	5.33
Afon Gwy	49	49	60	60	67	67	11	9	34.1	21.7	5.36	5.55

The pattern of change of ANC, pH and $SO_4^{2-} + NO_3^-$ (the major acidic anions) through time at each site (Figure 2.2.1.2a-m) shows the historical build up of acid anions (SAA) in surface water to c.1970. (Note that the historical SAA fluxes at Loch Coire nan Arr and Allt na Coire nan Con are relatively low since these are located in the North West of Scotland. At all sites except Loch Coire nan Arr and Allt a'Mharcaidh, this is predicted to have promoted some degree of surface water acidification. At Loch Coire nan Arr in NW Scotland, a site regarded as an un-acidified reference, no acidification is predicted to have occurred and reconstructed background ANC is well within the variation of presently observed values. Similarly, at Allt a'Mharcaidh in NE Scotland, simulated historical pH and ANC fall temporarily below the range of present day values but pH is not predicted to have been below 6.0. At these two sites, therefore, little change is predicted into the future. At Allt na Coire nan Con, the impact of forestry drives the deposition of SAA in the recent past and to 2020. Beyond 2020, re-planting causes further acidification but this result should be interpreted in the light of the need to re-examine the impacts of afforestation practices.

Figure 2.2.1.1 Simulated versus observed ANC and pH, AWMN sites



Of the other sites, since the 1970s, all acidified sites are predicted to have recovered to some extent in response to a significant and rapid decline in SAA concentrations to present day. This decrease is in line with the 60% reduction in UK SO_2 emissions since 1970 and the 70% reduction in European emissions. The extent of chemical recovery reflects the sensitivity of the site and the degree to which it was acidified.

Comparison of the model simulations against the observed annual runoff chemistry and using the observed annual deposition for 1988 – 1998 to drive the model (Figure 2.2.1.3a-d) gives generally good results. The range of variation in ANC and pH at most sites is generally well matched by the model. At some sites, short term increases or decreases in observed ANC and pH beyond the range of the modelled values are a response to the annual variation in Cl deposition (e.g. two years of very low Cl deposition in the Galloway region). The implication is that the deposition chemistry is not reflected in the annual mean stream chemistry. This could result from very rapid response of runoff chemistry to deposition which is effectively ‘missed’ by the monthly/quarterly sampling regime or of very slow responses being ‘averaged’ over calendar years. Continued testing of the model against those observed time-series will promote increased reliability of model predictions.

Future predictions show an immediate increase in SAA under the constant deposition scenario. This results from the assumptions relating to N cycling within MAGIC7 i.e. the worst case whereby soil C/N is already <= CN_{UP}. Under the deposition reductions associated with the Gothenburg Protocol, SAA declines to 2010 in both best and worst case models. Thereafter, in the worst case, N leaching increases whereas in the best case, N leaching remains constant into the future. The deviation in ANC between best and worst cases, therefore, is entirely the result of N leaching.

The time-series predictions must be interpreted with regard to the observed variation in annual mean chemistry. The model simulations into the future (beyond 1998) are driven by a ‘smoothed’ deposition trajectory and so do not reflect the variation in annual deposition that has been observed between 1988 and 1998. Since much of this variation in annual deposition reflects changes in rainfall totals, it is likely that the ‘smoothed’ deposition reduction in the future will, in reality, reflect a similar variability. The relationship between variability in deposition flux and variability in mean surface water chemistry is unlikely to be linear but it is not currently known how the variation about the long-term mean will change as the sites recover from acidification.

The simulations for the whole suite of sites clearly demonstrate the benefit of the emission reductions in both S and N under the Gothenburg Protocol over making no further emission reductions. Under constant deposition and assuming the worst case of N leakage NO_3^- concentrations increase (Figure 2.2.1.4), as a result of the assumptions made relating to decreased N immobilisation as the soil C/N decreases through time. SO_4^{2-} concentrations remain stable (Figure 2.2.1.5), although with some variation at the forested sites as the canopy growth influences dry deposition. As a result, ANC (Figure 2.2.1.6) and pH (Figure 2.2.1.7) are further decreased at all sites.

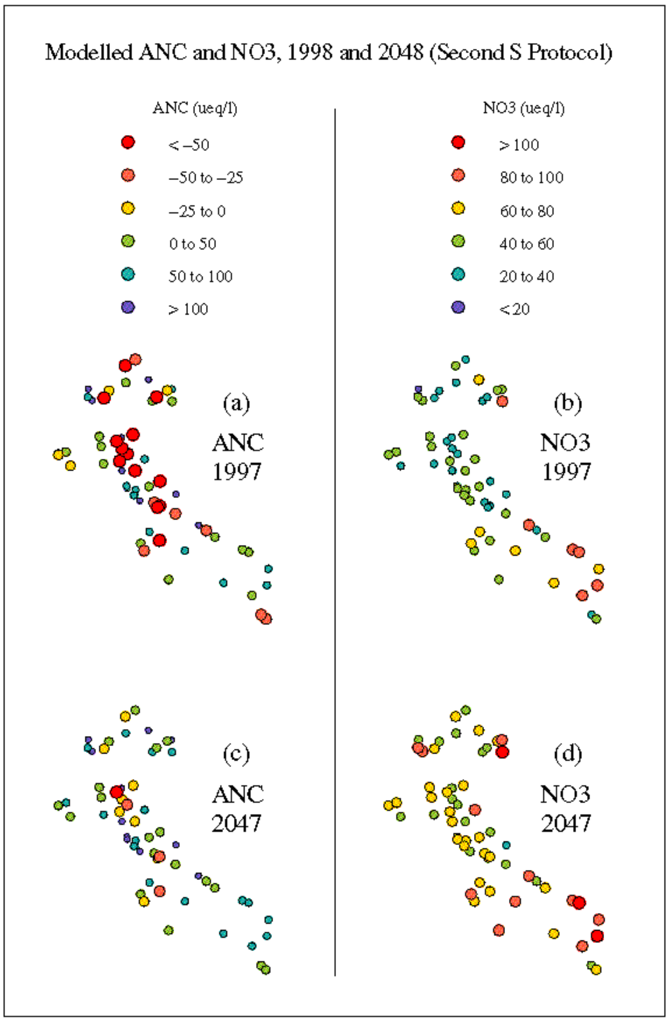
Under the Gothenburg Protocol, SO_4^{2-} concentrations decrease significantly at all sites (Figure 2.2.1.5). This leads to a dramatic improvement in ANC which, as expected, is most marked at sites with the lowest ANC (Figure 2.2.1.6). The ‘best’ and ‘worst’ cases for N leaching, however, have a less significant impact on predicted ANC than the agreed S reductions. This is because NO_3^- concentrations at all sites are currently relatively low and even at a worst case assumption for decreased N immobilisation, the decrease in N is sufficient to keep the NO_3^- leaching at roughly its current level. This emphasises the importance of actually achieving the agreed S and N emission reductions to promote chemical recovery from acidification over the next 20 years. The fact that both ‘best’ and ‘worst’ cases for N leaching provide rather similar ANC simulation to 2010 further indicates that it is the sharp decline in S deposition under the Gothenburg Protocol that is mainly driving the model response. This further emphasises the need to achieve the S emission reductions as agreed. Uncertainty over the future retention of N remains, however, and beyond 2020 decreasing catchment retention could lead to NO_3^- becoming the dominant anion and potentially leading to further acidification in the longer term.

2.2.2. Regional MAGIC 7 applications

Surface water chemistry data have been collected at a large number of sites in acid sensitive regions of the UK as part of this programme, as well as the CEH/MLURI Dynamic Modelling contract for DETR. The application of MAGIC 5 to five regions (Cairngorms, Galloway, Lake District, Dartmoor and Wales) and of MAGIC 7 to the South Pennines, is described in a report prepared under the Dynamic Modelling contract (Evans *et al.*, in press). This study was undertaken in order to assess the regional-scale impacts of different future deposition scenarios, and was based primarily on the simpler N dynamics of MAGIC 5, while MAGIC 7 has remained under development within the current programme.

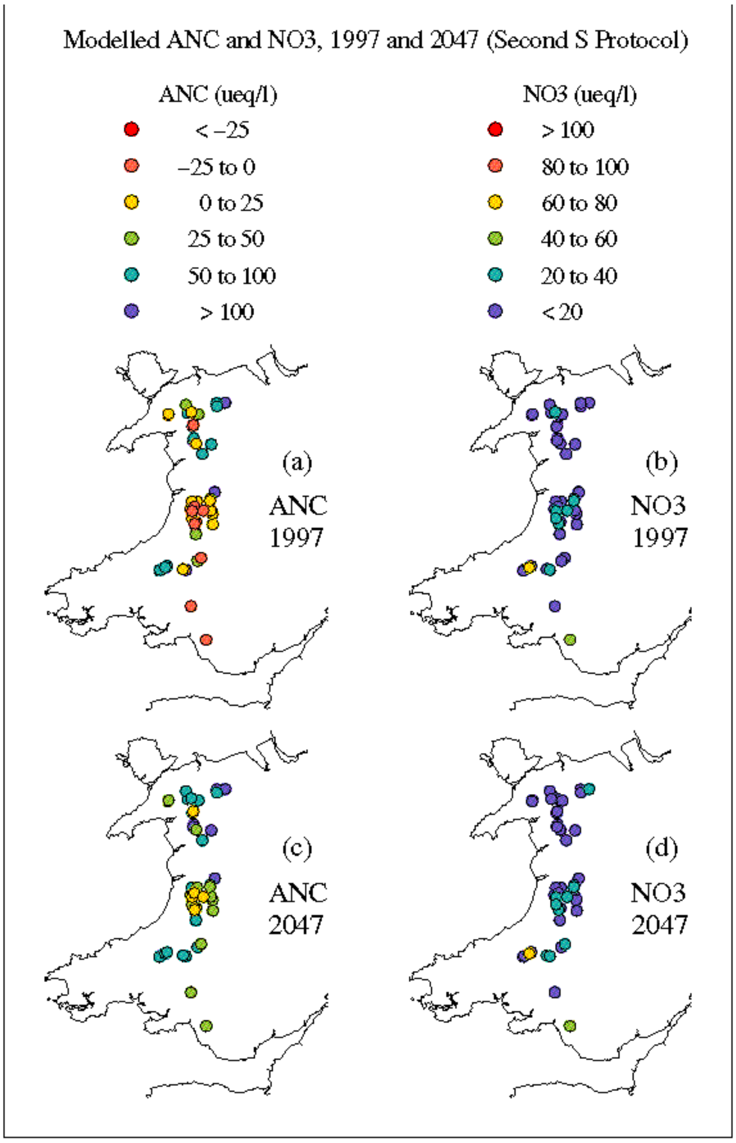
Preliminary regional applications of MAGIC 7 have been undertaken for three regions where soil C and N data were also available; Galloway, the South Pennines and Wales (Section 1.5.1). Modelling procedures were the same as those described in the preceding section, except that calibration was based on a single spring sample (South Pennines, Galloway) or one year of monthly samples (Wales). Future changes in surface water chemistry have been modelled using HARM deposition forecasts for the Second Sulphur Protocol (‘Reference Scenario’ – representing emissions reduction prior to the Gothenburg Protocol), and modelled ANC and NO_3^- for the present day and 2050 are shown in Figures 2.2.2.1 to 2.2.2.3.

Figure 2.2.2.1 MAGIC 7 application to the South Pennines



Results of these simulations show marked variability between regions. In the most highly impacted South Pennine region, 24 of the 58 calibrated sites currently have a negative ANC, but with the large SO_4^{2-} reductions forecast under the Reference Scenario, this is predicted to reduce to 10 sites. However, NO_3^- concentrations, which are already high in the region (median 41 µeq/l) are predicted to rise significantly, exceeding 60 µeq/l at 35 sites by 2050. These NO_3^- increases will thus significantly offset recovery due to falling SO_4^{2-}, and in the longer term, as N leaching increases further, lead to some re-acidification.

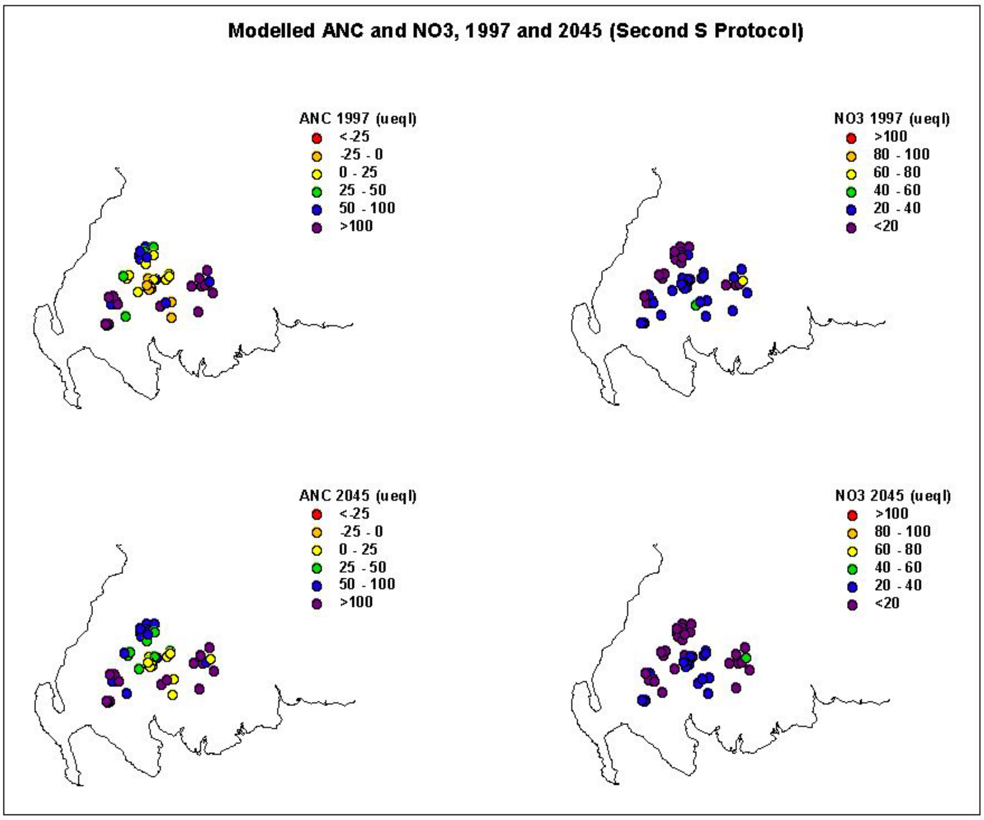
Figure 2.2.2.2. MAGIC 7 Application to Wales.



In Wales, although acidification is less severe than in the South Pennines, many sites throughout the study region currently have low or negative ANC. By 2050, it is predicted that almost all sites will have a positive ANC, with low-ANC (< 25 µeq/l) streams largely restricted to central Wales. In contrast to the South Pennines, NO_3^- currently makes a relatively minor contribution to stream acidity, and this situation is not predicted to change markedly, with most streams retaining concentrations below 20 µeq/l in 2050.

The situation in Galloway is fairly similar to that in Wales; acidic conditions are currently observed in the central part of the region, whilst NO_3^- concentrations are low to moderate. Forecasts suggest substantial recovery by 2045, with a halving of the number of lochs with a negative ANC. Again, this recovery is driven mainly by reductions in SO_4^{2-}, with relatively minor changes in NO_3^- concentrations.

Figure 2.2.2.3. MAGIC 7 application to Galloway



Task 2.4: Palaeolimnological analogues in defining target assemblages for the recovery of acidified surface waters: a desk study

G.L. Simpson

2.4 Palaeolimnological analogues in defining target assemblages for the recovery of acidified surface waters: a desk study

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Key Words: surface water acidification, palaeolimnology, restoration, analogue matching

Executive Summary

1) Palaeolimnological techniques have been widely employed to study lake acidification. This approach has been central in testing the cause-effect relationship between acid deposition and lake acidification, and in assessing the magnitude and extent of surface water acidification across the UK.

2) Most of these palaeolimnological applications have been based on diatom analysis, and the use of diatom-pH transfer functions to make reconstructions of hydrochemical change in upland lakes associated with acidification.

3) Following the signing of the Second Sulphur Protocol, attention is now focusing on emissions reductions and the reversibility of surface waters acidification. There is a clear need for criteria against which to evaluate the recovery process.

4) In order to evaluate future recovery, Flower *et al.* (1997) have proposed a palaeolimnological technique for defining targets for the recovery of acidified surface waters. This is based on the technique of analogue matching of lake sediment diatom assemblages. Multivariate statistical methods are used to identify modern analogues for the pre-acidification diatom assemblages of acidified lakes. The chemical and biological status of modern analogue lakes can then potentially provide recovery targets for acidified systems.

5) This approach has been successfully applied to several acidified lakes, and modern analogue systems defined for the pre-impact (pre-acidification) status of these impacted sites. An advantage of the approach is that it can provide recovery targets for both chemical and biological status of acidified lakes.

6) Modern analogue matching as currently applied makes several key assumptions:

   a) that analogue matches based on a single biological group (diatoms) effectively represent the hydrochemical and biological variation of low alkalinity systems;

   b) that the modern data set used to identify modern analogues contains the range of hydrochemical conditions represented by the fossil assemblages;

   c) that a suitable stable 'baseline' (pre-impact) status can be defined.

7) Prior to more comprehensive application of the modern analogue approach to acidified lakes in Britain, these assumptions require evaluation. Three studies are proposed:

   a) Extension of the current modern lake dataset used for analogue matching by the inclusion of minimally impacted low alkalinity sites from northern Scotland.

   b) Development of the current technique by including two more fossil groups (chironomids and cladocera) in the modern surface sediment dataset used in the matching procedure. This will allow the assumption that diatoms represent wider ecosystem variation to be tested, and should result in more robust analogue matches.

   c) A study of hydrochemical and biological variation in the pre-acidification conditions of acidified lakes through high-resolution palaeolimnological study of selected Acid Waters Monitoring Network lakes. This will allow the stability of baseline (pre-acidification) conditions to be evaluated.

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Acid deposition, surface water acidification and palaeolimnology

The acidification of surface waters via atmospheric contamination by strong acid-forming compounds has been a major environmental issue for the last 20 years or so. Angus Smith (1852 cited in Battarbee et al. 1988) first coined the term 'acid rain' when he used it to describe the effect of coal combustion on air and precipitation chemistry at various industrial sites in the UK. The effect this would have on lake ecosystems was first discussed by Gorham (1958). Since then numerous studies have been published which have investigated the problems associated with acid deposition and have confirmed that lake acidification has taken place in large areas across Europe and North America (see for example Battarbee & Charles 1986). In a review of the available data on surface water acidification, Charles et al. (1990) draw upon examples from North America, the United Kingdom, Scandinavia and Central Europe (see Table 1). Studies have shown extensive acidification of many lakes within these regions. For example, the Palaeoecological Investigation of Recent Lake Acidification (PIRLA) (Charles & Whitehead, 1986) was a large-scale project that attempted to assess the extent and cause of acidification in North America. In the Adirondack Mountains National Park, New York, all 11 clearwater lakes from the region sampled as part of PIRLA where contemporary pH was below 5.5 have acidified as a result of the deposition of strong acids to their catchments.

In the UK, Battarbee et al. (1988) identified a range of lakes impacted by atmospheric contamination resulting in increased acidity levels. Acidification of UK surface waters was identified in central and northern Wales at sites on Lower Palaeozoic Sedimentary and Metamorphic rocks. Acidification was also identified in Cumbria, on Borrowdale volcanic strata, and in Scotland, Southeast of Loch Ness on granitic geology. There are also numerous highly acidic ponds and lakes in the English Pennines. Since this work, further studies have shown that lakes have acidified in areas previously thought to be unaffected by atmospheric deposition, such as lakes on the Cairngorm plateau (Jones *et al.* 1993) and remote sites in the north of Scotland (Allott *et al.* 1995).

Palaeolimnological studies (Charles & Whitehead, 1986; Battarbee & Renberg, 1990) have proved conclusively that acid deposition has resulted in the widespread acidification of surface waters. The timing of acidification in lakes has always occurred after the onset of the industrial revolution and the record of carbonaceous particle deposition of lake sediments clearly demonstrates atmospheric contamination from industrial sources prior to acidification (Rose, 1996; Rose *et al.* 1995). Other studies have provided understanding of the emission, transport, and deposition of pollutants and the processes of atmospheric chemistry (RGAR, 1997).

Table 1: Selected regions showing the extent of surface water acidification throughout North America and Northern Europe. (Main data source: Charles et al., 1990)

Geographic Region	Local Area	Lakes Acidified ?	Comments
United States of America	Adirondacks	Yes	11 clearwater lakes with pH <5.5. Lake with pH >6.0 have not acidified or have done slightly
	New England	Yes - recently	Trends are unclear
	Northern Great Lakes	Little or none Post 1900 pH decline in 4 Wisconsin lakes	Lakes are generally seepage lakes with low alkalinity (-38 to 80 μeq l^{-1}) and function differently to those of the Adirondacks
	North Florida	Little or none	12% of regions lakes have pH <5.0. 4 lakes studied were naturally acidic due to low cation exchange capacity and base saturation of catchment soils
Canada	Eastern Canada	Yes	Studies biased towards lakes close to point source emissions, e.g. Sudbury, Ontario
UK	Central & Northern Wales	Yes	Strong acidification (1-1.5 pH units in some lakes)
	Cumbria	Yes	An area of high acid deposition, though acidified lakes restricted to geologically sensitive areas, e.g. Borrowdale volcanic strata
	Southeast Scotland	Yes	On granite bedrock
	Northern Scotland	Yes	Sites that were thought to be relatively unaffected by acid deposition have been found to have acidified recently (e.g. Allott *et al.* 1995)
Rest of Europe	Scandinavia	Yes	Large scale loss of fish populations and damage to forests- many acidified lakes
	The Netherlands	Yes	
	Denmark	Yes	
	Germany	Yes	

International Emissions Reductions

The acceptance of the cause-effect relationship between acid deposition and acidification has led to international efforts to reduce acid emissions. In 1979, the Convention on Long-Range Transboundary Air Pollution (LRTAP) was adopted to implement measures to reduce the levels of sulphate emissions from industrial sources. Currently there are 40 signatory countries to LRTAP, which has identified principles for international co-operation and for the abatement of the emissions of pollutants.

Under the LRTAP convention, a number of protocols have been developed that commit member states to certain abatement actions. The early protocols limited emissions of sulphur dioxide (the 1985 Helsinki Protocol) and oxides of nitrogen (the 1988 Sofia Protocol). All countries were required to cap their emissions or reduce them in relation to a given reference year (see

Table 2).

In 1994, the Second Protocol on the Further Reduction of Sulphur Emissions was adopted. This, the Oslo Protocol, introduced a new method of setting reduction targets by incorporating critical loads, an effects based approach to deposition (Bull, 1995). Critical loads maps represent the sensitivity of ecosystems to given levels of modelled acid deposition. Using modelled deposition across Europe for given abatement strategies, critical load exceedence maps indicate the effect on ecosystems for that level of abatement. Setting abatement targets for international protocols now involves the benefits to ecosystems that a given reduction will involve, and through Integrated Assessment Modelling cost effective abatement strategies have been developed.

Currently a further Protocol is being developed for emissions of nitrogen oxides. This protocol will be ‘multi-pollutant/multi-effect’, as sources of nitrogen oxides include acid deposition, ammonia and ozone, and cause a range of environmental disturbances (e.g. acidification, eutrophication, and direct effects to human and vegetation health (RGAR, 1997).

The UK, whilst not signing up to the Helsinki Protocol, has met with the protocol’s requirements by reducing emissions of sulphur dioxide by 35% (RGAR, 1997). A national strategy has also been developed to outline the ways in which the UK will meet its obligations to the Oslo protocol. In the light of these international reductions in air pollutants, the emphasis is now shifting towards the recovery of acidified surface waters. There is now a need to investigate the process of recovery in acidified surface waters to evaluate and model the response of surface waters to reduced acid deposition. Given that the protocols arranged under the LRTAP convention are having success in reducing the level of acid deposition to catchments across Europe and North America the role of recovery is now central to the acid deposition debate.

The aim of this report is to evaluate the modern analogue approach to setting targets for the recovery of acidified surface waters. The approach uses palaeolimnological and multivariate techniques to compare one aspect of the flora or fauna of an acidified lake with the flora and fauna of other lakes. To identify potential targets for recovery, the technique allows the selection of a baseline or pre-impact condition for an acidified lake and the comparison of this to the modern flora and fauna of other lakes. Lakes identified using the technique are known as modern analogues. Identification of modern analogue sites potentially allows chemical and biological targets to be set for recovery.

Table 2: Current protocols developed as part of the LRTAP convention to curb the emissions of sulphur dioxide and oxides of nitrogen implicated in the acidification of surface waters across Europe and North America

Protocol	Year Adopted	Year in Force	Requirements of the Protocol
EMEP	1984	1988	The Protocol on Long-Term Financing of the Co-operative Programme for Monitoring and Evaluation of Long-Range Transmission of Air Pollutants in Europe. Collates information on deposition and emission inventories supplied by member states, and develops transport models for pollutants.
Helsinki	1985	1987	The Protocol on the Reduction of Sulphur Emissions or Their Transboundary Fluxes by at least 30%. Committed parties to a 30% cut in SO_2 emissions by 1993 based on 1980 levels.
Sofia	1988	1991	The Protocol concerning the Control of Emissions of Nitrogen Oxides or Their Transboundary Fluxes. Commits member states to bring back NO_x emissions to their 1987 levels by 1994
Oslo	1994	Not Yet	The Second Protocol on the Further Reduction of Sulphur Emissions. Requires different % reductions from member states based upon an effects based concept know as the critical loads approach. The UK is required to reduce sulphur emissions by 80% against 1980 levels, by 2010.

The structure of this report is as follows. The contribution of palaeolimnology to the acidification debate is described in Chapter 2. Palaeolimnology has been instrumental in confirming the cause-effect relationship between acidification and deposition across Europe and North America. Chapter 2 also describes the techniques used to reconstruct hydrochemical variables from fossil data, and reviews the biological groups preserved in fossil records. The concept of ecosystems restoration is discussed in Chapter 3. Chapter 4 describes the technique of analogue matching and describes a preliminary application of the approach for northern European lakes developed by Flower et al. (1997).

The report also considers the question of what makes a suitable restoration target given that in the UK there are very few pristine ecosystems. Human activities have influenced the natural landscape for thousands of years. There is also increasing evidence that acidification may be influenced by climate change in remote alpine lakes. The degree to which climate change has impacted upland lakes in Europe and the UK in general is addressed. Chapter 5 details the available data in this area, as well as outlining the other assumptions of the modern analogue approach.

Chapter 6 briefly outlines ways in which the method can be applied and improved, and how the validity of the assumptions of the approach can be tested.

Recovery from acidification will be, in most cases, restricted to natural recovery. Identifying a period when recovery has taken place, to the satisfaction of both the environmental groups and those with economic interests, is required if recovery can be defined as being complete. The modern analogue approach and palaeolimnology allows us an opportunity to investigate recovery in acidified systems, and to evaluate the success of emission reductions.

Describing the biological target for recovery is more difficult. Directly, palaeolimnology is limited to identifying communities of those organisms that preserve well in sediments. Palaeolimnology, however, can help to select modern lakes that are analogous to the pre-impact conditions of presently acidified lakes. When coupled with the detailed knowledge of aquatic ecosystems gained from investigating analogue lake systems, the tools required for setting targets for recovery and evaluating the progress made towards those targets would then be at our disposal.

Chapter 2: Palaeolimnology

What is Palaeolimnology?

Palaeolimnology is the study of the history of lakes. It is concerned with how lakes have changed over time and with understanding those changes, primarily through an assessment of the historical record contained in the sediments of lakes. Palaeolimnology has been used extensively to study the development of lake systems (ontogeny).

Over the last few decades numerous sampling and analytical techniques have been developed that enable palaeolimnologists to obtain sediment cores containing undisturbed sediment records (e.g. Glew, 1991, Charles et al., 1994). These can then be analysed for a variety of fossil organisms and geochemical markers using appropriate laboratory methods (Berglund, 1986). Dating techniques using ^{210}Pb (Appleby et al., 1986) and other radiometric methodologies (Charles et al., 1994) can be used to apply chronologies to the changes observed in the sediment record. This allows the timing, magnitude and rate of change to be determined.

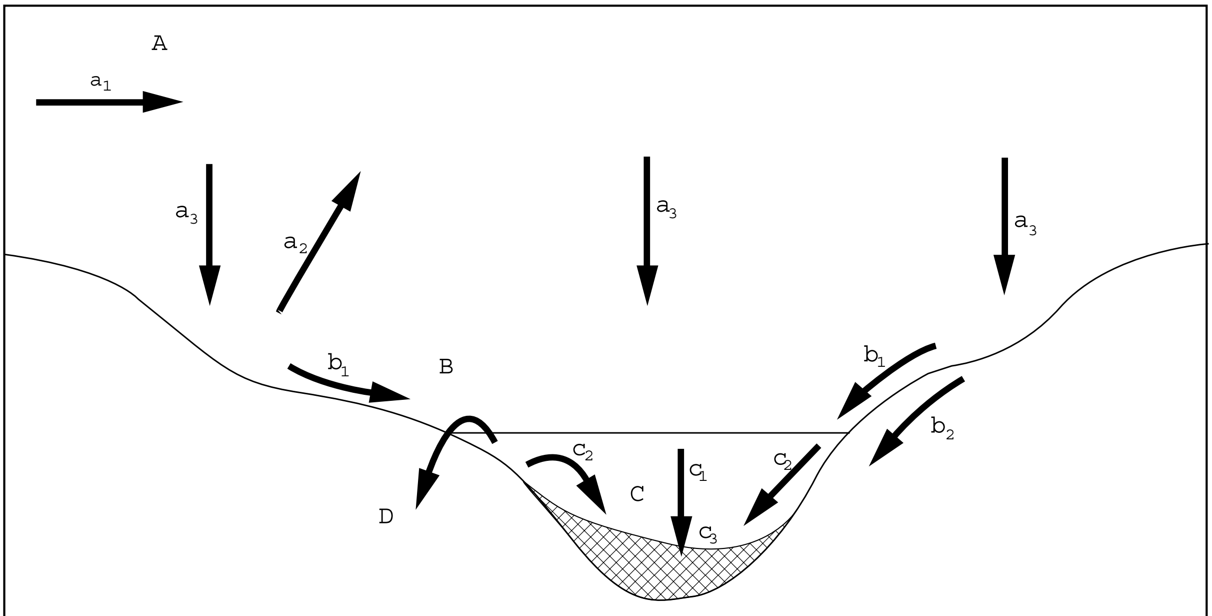
Sediments found within lakes are derived from three main sources; atmospheric inputs, catchment inputs and lake inputs (Figure 1. Atmospheric sources include wet and dry deposition both from sources within the catchment and those from outside. Inflows from streams and groundwater contributions make up the catchment component, and the planktonic, benthic and littoral flora and fauna of the lake contribute to the lake input fraction (Battarbee et al., 1988).

Organisms that are preserved in the sediment of a lake, such as diatoms or cladocera, can be analysed for changes in community structure over time. Assessment of the modern distribution of these organisms can be related to the hydrochemistry of the waterbody, and optima and tolerance ranges for individual species can be calculated (Battarbee, 1991). These optima and tolerance ranges can be applied to the fossil communities by means of transfer functions (e.g. see Battarbee, 1991; Bennion, 1994; Birks et al., 1990b; Charles et al., 1994; Dixit & Smol, 1994). These enable changes in the hydrochemistry of a lake to be determined from the fossil community assemblages deposited to the sediment at that time (Anderson, 1995).

These techniques are now widely used within palaeolimnology. They have been used to determine changes in waterbodies related to many hydrochemical variables, including acidity (pH) (Dixit et al., 1992; Charles et al., 1990; Allott et al., 1995; Cumming et al., 1994; Dixit et al., 1992), total phosphorus (TP) (Bennion, 1994; Dixit & Smol, 1994), Dissolved Organic Carbon (DOC) (Kingston & Birks, 1990), salinity (Fritz, 1990), aluminium (Al) (Dixit et al., 1992), as well as enabling changes in climate and temperature to be inferred from the sediment record (Lotter et al., 1997).

Analyses of the sediment record for contamination by pollution indicators seen as heavy metals (Pb, zinc etc.) (Davis et al., 1983) spherical carbonaceous particles (SCP) (Rose, 1996; Rose et al., 1994) and inorganic ash spheres (IAS) (Rose, 1996), have allowed the causal mechanisms for pollution induced changes to be determined, especially in the case of the acute acidification of sensitive surface waters in Northern Europe and North America.

Figure 1: Schematic cross-section of a lake and its catchment showing sources and pathways of material found in lake sediments (from Battarbee et al., 1988).



A. Atmospheric inputs

a_1 sources outside catchment
a_2 sources within catchment
a_3 wet and dry deposition to lake and catchment surface (radioactive nuclides, S, PAH, trace metals, fly ash, pollen and spores)

B. Catchment inputs

b_1 inflow streams (silts, clays, organic detritus, pollen and spores, solutes)
b_2 ground water (precipitates eg. Fe, Mn)

C. Lake inputs

c_1 plankton (diatoms, chrysophytes, cladocera)
c_2 littoral flora/fauna (diatoms, macrophyte spores and pollen, chydorid cladocera, chironomidae
c_3 benthic flora/fauna (diatoms, chironomidae)

D. Outflow losses

Palaeolimnological techniques have been used to assess changes in both the biology and, more recently, the chemistry of lakes. They have been used over timescales ranging from millions of years (e.g. the study of Lake Baikal) to sub-decadal (e.g. Allott et al., 1992). The techniques used in palaeolimnology have also been used in monitoring projects attempting to evaluate current changes in aquatic systems. Dixit and Smol (1994), present four transfer functions that can be used to reconstruct the hydrochemistry of lakes from the enumeration of siliceous microfossils from lake sediments. In this way, inferences about the current, changing, hydrochemical nature of surface waters can be made using the biological record of organisms contained within recent lake sediments.

The short-time series used in many recent ecological studies are often unsuitable for monitoring studies (Smol 1995). Natural variability is difficult or impossible to distinguish from data of insufficient quality or time span. Without such data, environmental managers often have difficulty in determining the trajectory or the causes of degradation, let alone the likely effects of recovery or targets for their mitagatory efforts (Smol, 1995). Smol (1992) argues that the task of assessing ecosystem health is made harder without suitable and available long-term data. Palaeolimnological data provides such long-term environmental, hydrochemical, lithological and biological data, enabling the baseline (pre-impact) conditions of a surface water to be established (see Chapter 3). These baseline states of surface waters provide invaluable information on the degree of change present at a site. Where anthropogenic pollution is suspected or known to have degraded a surface water, targets for the restoration of the site can be established with regard to these baseline conditions.

Palaeolimnological studies are often based upon studies of the fossil remains of biological indicators, the aquatic flora and fauna living in a lake. Palaeolimnological sediment records of change are averaged over time, the species composition dynamically in tune with the predominate or average hydrochemical or physical conditions of the lake. Short-term cycles, such as diurnal or diel changes, can influence the water chemistry or the biological sample taken during spot sampling measures (Anderson & Battarbee, 1994). However, the historical record of lake sediments is time averaged, and, therefore, does not contain the degree of noise inherent in spot sampling records.

The wide variety of organisms preserved in the sediments and the range of chemical, lithological and biological assays that can be applied to the sediment allows for a huge range of experiments and hypotheses to be tested using palaeolimnological techniques.

Palaeolimnology and lake acidification

Palaeolimnology played a key role in the study of lake acidification. During the early 1980s, there was considerable debate surrounding the cause of fish stock declines in upland lakes. Many scientists, especially those from Scandinavia where the effects of acid deposition were first described, believed that acid emissions were to blame for the recent acidification of surface waters in Europe and North America. A number of alternative hypotheses were formulated, however, in what was a highly charged scientific and political debate.

One such claim was that acid lakes were the result of long-term, natural acidification processes (Pennington, 1984). It was claimed, that after lakes were formed when the glaciers retreated and climate warmed at the start of the Holocene, lakes acidified as weathering of soils led to the progressive leaching of base cations. Soils would gradually acidify and the subsequent runoff would slowly lower the pH of lakes. The basis behind this hypothesis was that there are many lakes that are presently acid but that had no historical record of fish stocks, evidence for long-term acidification.

Other claims surrounded the large-scale changes in land-use within upland areas of Northern Europe (Rosenquist 1978). Decreased grazing in lake catchments may have led to an increase in heathland vegetation and with it an increase in acid soils. Evidence from Sweden (Renberg et al. 1993a,b) has shown considerable influence on lake hydrochemistry by changes in land-use. Many lakes in southern Sweden were show, by hydrochemical reconstruction using a diatom-pH transfer function, to have an alkaline phase prior to the onset of acidification in the 19th Century. The alkaline phase was the result of land-use practices that no longer exist in modern day Sweden.

More credibility was placed upon claims that recent acidification of surface waters was the result of recent afforestation. Streams draining from afforested catchments have been shown to be more acidic than those draining non-afforested areas (e.g. Harriman & Morrison, 1982; Stoner & Gee, 1985). Many of the afforested regions of the UK are located in areas of high sensitivity to acid deposition with base-poor bedrock, slow weathering rates and high rainfall (Kinniburgh & Edmunds, 1986). Consequently, many areas of afforested upland Britain have low acid buffering capacities. Tree growth processes (Nilsson et al., 1982), enhanced scavenging and foliar uptake of sulphur dioxide (Lindberg & Garten, 1998), as well as land improvement measures prior to tree planting (Hornung & Newson, 1986), have all been proposed as mechanisms which promote acidification.

Palaeolimnological studies have demonstrated that, whilst afforestation can result in enhanced acidification, it cannot account for the widespread, rapid acidification of European surface waters. Kreiser et al. (1990) studied four Scottish lochs, two with afforested catchments (Loch Chon and Loch Doilet) and two with moorland catchments (Loch Tinker and Lochan Dubh). Loch Tinker and Loch Chon are located in the Trossachs region of Scotland, an area of high acid deposition, where as Loch Doilet and Lochan Dubh are located in an area that receives low levels of acid deposition. Kreiser et al. (1990) demonstrated that acidification occurred gradually from c. 1850 at Loch Tinker. When the site was afforested in the 1950s, there was an increase in the rate of acidification. The most rapid period of change occurs in the 1960s following canopy closure when pH fell from 5.8 pH units to 5.2. Loch Tinker shows some signs of early decline in the planktonic Cyclotella flora of the lake, an early biological indication of acidification. There is further acidification up to the 1930s, but the acute, recent (post 1960) acidification and associated diatom changes shown to occur in Loch Chon do not occur in Loch Tinker. There was no overall change in inferred pH or diatom composition above a depth of 8cm (1930) in the sediment core. pH reconstruction at Loch Tinker shows that pH fell from 6.6 to 5.7 by 1930, but then fluctuated around 5.6-5.7 until the present day. Carbonaceous particle records from the two lochs show that both were receiving high levels of atmospheric contamination by the 1940s, which is prior to the afforestation of Loch Tinker.

The study by Kreiser et al. (1990) demonstrated that in areas of high deposition afforestation can lead to further (or more severe) acidification of soils and surface waters. The tree canopy increases scavenging of particles from the atmosphere, which can enhance the flux of atmospherically derived acidity to the catchment. Afforestation in the catchment of Loch Chon had such an effect leading to further acidification after c. 1960s. Loch Tinker stopped acidifying after c. 1930, even though it was experiencing similar levels of deposition as Loch Chon. The diatom sub-fossil flora of Loch Doilet, the other Loch in the study with an afforested catchment, does not indicate any further acidification following afforestation. Loch Doilet receives lower levels of deposition than Loch Chon and the scavenging effect of the canopy has had little or no effect of the chemistry of Loch Doilet. The two Lochs with moorland catchments both acidified after c. 1850, precluding afforestation as the primary cause of surface water acidification.

Palaeolimnological techniques were also used to evaluate the claims that acidification of surface waters was the result of deposition of atmospherically derived acids to lakes and the catchments draining into them (e.g. Battarbee et al., 1990). Recent lake acidification is strongly correlated with evidence of atmospheric contamination from lake sediments (Battarbee, 1990). Records of carbonaceous particles document the impact of atmospheric contamination to surface waters (Rose et al., 1995; Rose, 1996). Created during high-temperature burning of fossil fuels in power stations, carbonaceous particles first enter the sediment record during the nineteenth century as the industrial revolution took hold in the UK. Carbonaceous particle concentrations drastically increase in the post-war period (1950s), reaching a peak around 1970. Concentrations decline after the 1970s where oil availability was reduced during the Oil Crisis and because of international efforts to curb emissions in the late 1970s and early 1980s.

Battarbee (1990) demonstrates that the trend in surface water acidification seen in UK lakes parallels the trends in carbonaceous particle concentration identified from the sediment record. The first indication of atmospheric contamination occurs around c.1850 and rarely before 1800 in UK lakes. This is also the period where a response of the diatom flora to the contamination is first observed. Contamination levels increase rapidly after c. 1940, and the diatom flora shows a rapid change to more acid tolerant species after this time. Lakes that have little evidence of atmopsheric contamination in the sediment record, such as Loch Corrie nan Arr (Battarbee, 1990), are also those that have not acidified, for those areas that receive high loadings of atmospheric deposition acute acidification of the surface waters has been identified from the sediment record (Battarbee, 1990)

Palaeolimnology has played a crucial role in determining the cause and effects of lake acidification. Without the high resolution historical record contained in the sediment of lake basins and interpreted by palaeolimnologists it would have been particularly difficult or impossible to prove the role emissions from power stations played in acidifying surface waters.

Since SWAP and PIRLA, the emphasis has shifted towards identifying chemical and biological recovery from acidification. Dixit et al. (1989) showed clear recovery in Swan Lake using chrysophytes, a siliceous type of algae (see below). Swan Lake, situated near to the Sudbury smelters in Ontario, had acidified because of the large amount of SO_2 and toxic metal pollution emitted from the smelters. Dixit et al. (1992) demonstrated similar recovery in three other Sudbury lakes, this time using both diatom and chrysophyte remains. Recovery was attributed to the reduction in emissions from the Sudbury smelters where emissions of SO_2 had fallen by 50%. This reduction has been attributed to a reduction in the activity at some of the smelters and improvements in emissions control and taller stacks at the others. Evidence of recovery in sites remote from point source pollution was described by Allott et al. (1992) for the Round Loch of Glenhead. In cores with high sedimentation rates recent reversal of the diatom floristic trend towards dominance by a few acid tolerant forms was seen. These studies indicate that recovery from acidification should happen in lake systems following emissions reductions, though detecting this may be more difficult in sites remote from point-source emissions. It may take many years before significant changes in the hydrochemistry and biology of lakes are observed.

Hydrochemical Reconstruction using Palaeolimnology

It has long been recognised that by studying the autecology of biological indicators for various hydrochemical variables you can classify specific groups of organisms that are found together under similar hydrochemical conditions. Taking this a step further allows us to determine the hydrochemistry of surface waters through an analysis of the organisms living in them. Earlier attempts at hydrochemical reconstruction were based on qualitative assessment of species. In recent years, however, more sophisticated statistical techniques have been used to provide accurate, robust hydrochemical reconstructions from fossil records that have a firm ecological basis.

The Surface Waters Acidification Project (SWAP) (Mason, 1990) and the Palaeoecological Investigation of Recent Lake Acidification (PIRLA) (Charles & Whitehead, 1986) projects developed new methods of reconstructing hydrochemical variables from the diatom species assemblages found within the sediment record.

Early approaches to pH reconstruction had been made qualitatively using Hustedt's pH classification system using diatoms. Nygaard (1956, cited in Battarbee et al., 1986) further enhanced this approach by developing indices. These indices were based on ratios of the percentages of diatom valves in Hustedt's pH categories. Meriläinen (1967) further developed quantitative approaches for reconstructing the acidity of surface waters using the relationship between the $\log^{10}$ of index values (e.g. index α) and the measurements of lake water pH using regression analysis. The slope and intercept of the regression equation are then used to predict lake pH. Renberg and Hellberg (1982) derived Index B, again based on pH categories, which can be used to predict pH. Index B uses more information than Index α, and is less reliant on alkaline taxa that are rare or absent in acid lakes. Multiple linear regression of the optima and tolerance ranges of biological indicators for pH has also been used successfully to reconstruct the historical record of pH change from sedimentary records (Charles, 1985; Flower, 1986).

Whilst multiple regression has been useful in providing accurate pH reconstructions, it has two major drawbacks; the technique is not ecologically realistic and, because class-interval groups are used, individual species data may be ignored.

The technique that was developed for SWAP and PIRLA (e.g. Birks et al., 1990b) was based around weighted averaging (WA) regression and calibration (ter Braak, 1987). Weighted averaging regression and calibration overcome many of the problems of other calibration methods (Korsman & Birks, 1996). WA assumes a unimodal relationship for the response of species to explanatory variables. This is considered to be a sound ecological assumption (ter Braak & Prentice, 1988). Other regression and calibration methods assume linear responses to explanatory variables and, therefore, do not represent ecological functioning as well as WA. WA also maximises the covariance between the diatom data and the measured environmental variables (Korsman & Birks, 1996). This is the same approach used in direct gradient analyses such as canonical correspondence analysis (CCA). Indirect gradient methods attempt to maximise the variance only with the diatom data and, consequently, some information may be lost when only the first few components are used for regression. If more components are used, multi-collinearity problems are introduced to the analysis (Korsman & Birks, 1996).

There are some weaknesses in WA, but these have recently been addressed with the development of a new variation of WA; Weighted Averaging Partial Least Squares (PLS) regression. WA-PLS uses the residual structure of the species data within the regression and calibration procedure to improve the predictions made using the technique (ter Braak & Juggins, 1993). Simple WA fails to accommodate this extra data and the predictive power of the WA calibration models produced using this method may not be as accurate as those developed using WA-PLS.

The calibration approach involves a two-step process. Firstly the relationship between the species data and the measured environmental variables (the predictor variables) is established using WA or WA-PLS regression. The relationships derived from the regression technique are then regressed or calibrated against the fossil data to predict pH from the fossil assemblages using inverse WA or WA-PLS regression.

To test the transfer function model, the predicted results are compared to a set of observed data. Strictly, the model should be tested against independent data, not against the data from which the model was derived. However, Birks et al. (1990b) demonstrated the techniques of 'Bootstrapping' or 'leave-one-out jack-knifing' that estimate the true error of the model. These methods achieve this by taking a sub-sample of the training set to compare with the observed data, thus forming an independent test of a model's predictive power. These techniques are computer intensive, with c. 1000 calibrations being run to test the error of prediction in a transfer function. Bootstrapping is especially suitable as the whole dataset is used to test the model.

Sub-fossil remains found in lake sediments

Palaeolimnological techniques are dependent upon the preservation of organisms in the sediments of lake systems. Not all conditions are suitable for the preservation of organisms and the fossils found in lakes can be dissolved or broken (Cameron, 1995; Flower, 1993). Some groups of organisms are not preserved within the sediment record. Preservation can also be dependant on the environmental characteristics of the lake. For example, ostracods are not found in surface waters that have low calcium concentrations because in these conditions, their ability to grow a strong calcite shell is impaired and they are less protected against predation. However, a number of important species groups are reliably preserved in the sediments of low alkalinity lakes. It is upon these organisms that the majority of the development of the techniques has taken place. These groups are described below with examples of their use in palaeolimnological studies.

Diatoms (Bacillariophyceae)

Diatoms are single celled, golden-brown algae characterised by an external siliceous cell wall, or frustule, in which silicic acid has been dehydrated and polymerised to form silica particles (Wetzel, 1983). This frustule consists of two overlapping valves connected by bands of silica known as girdle bands. The frustule is often highly ornamented with various species exhibiting different employment of features. These include the raphe (a slit in the length, or part of, the cell), the pseudoraphe (a depression in the axial areas of the cell wall) or striae (lines on the valve face composed of holes of complex structure within the cell wall). It is this ornamentation on the valve and its general shape and nature that form the basis of all taxonomic separation of diatom species (Barber & Haworth, 1981). Cell morphology varies between genera and species, and even allows varieties and forms of species to be distinguished.

Diatoms are found almost everywhere that light and moisture occur, including virtually all marine, brackish and freshwater environments, as well as soils, ice, and attached to rocks and other substrates within spray and splash zones near water. Diatoms live singly or form colonies, usually secreting a mucilaginous material that covers the frustule and allows the diatom cells to attach to one another or to the benthos (Wetzel, 1983). Diatoms live on a variety of substrates, as well as in the plankton of lakes. They occupy habitats on rocks and stones (the epilithon), on aquatic macrophytes (epiphyton), sand grains (epipsammon), the sediment (epipelon) and on mosses (epibryon).

Due to the siliceous nature of the frustule, it is resistant to a certain degree of chemical attack and is usually well preserved within lake sediments. The relative ease with which diatom samples can be taken and analysed, and the variety and beauty of the various forms of frustule, have resulted in diatoms being a long studied member of the limnic flora (Battarbee et al., 1986). Diatom autecology has largely been described in the literature and observations regarding tolerances and optimal abundances for a variety of hydrochemical variables have been made. Diatoms have been demonstrated to respond to a number of environmental and hydrochemical variables (e.g. Battarbee, 1984; Dixit & Smol, 1994), including those not related to the acidified nature of many lakes in Northern Europe and North America, such as salinity (Fritz, 1990) and TP (Bennion, 1994; Engstrom et al., 1985).

The response of the diatom community to acidification has been examined via inferences from the sediment record and from laboratory and whole lake experiments. It has been demonstrated that diatoms respond markedly and quickly to changes in pH (e.g. Flower & Battarbee, 1983; Battarbee, 1984; Flower, 1986), alkalinity (Charles et al. 1994), Al and DOC (Kingston & Birks, 1990). These variables represent hydrochemical components that change during the acidifying process.

Chrysophytes (Chrysophyceae)

Chrysophytes are another type of algae, found in the plankton, and are characterised by a golden brown colour. Most of the Chrysophycean algae are unicellular organisms possessing one or two flagella (Wetzel, 1983). Many species do not posses a cell wall and are bounded instead by a cytoplasmic membrane or are covered by a coating of tiny siliceous or calcareous plates and scales. These siliceous scales are well preserved in the sediment and form the basis of species identification being distinguishable to species level.

Chrysophytes have been used as biological indicators (Kristiansen, 1986) and in palaeolimnological investigations including studies of pH change and lake acidification (e.g. Siver & Hamer, 1990; Facher & Schmidt, 1996). For example, Smol et al. (1984) demonstrated the use of Mallomonadacean microfossils in the analyses of past changes in lake acidity. The genera *Mallomonas* are a group of flagellated algae of the class Chrysophyceae. In the upper sediments of a core from Deep Lake, Adirondacks, USA, the contribution of *M. crassisquama* to the total (%) scale count decreased, whilst the contributions of *M. hindonii* and *M. hamata* increased, the former becoming the dominant source of chrysophycean scales to the sediment. This shift in species contributions to the total scale count was attributed to a progression from circumneutral conditions to those of lowered pH. From the 38 lakes in the Adirondacks assessed by Smol (1980) for Mallomonadaceae microfossils, *M. hindonii* and *M. hamata* were found to be indicative of acidic lakes, whilst *M. crassisquama* was rarely found in lakes with a pH <5.0. Smol et al. (1984) substantiated their hypothesis of a response in the mallomonadaceae microfossils to increasing acidity with an analysis of the diatom microflora of Deep Lake, which showed a similar response to the decreasing pH.

Dixit et al. (1992) used changes in the chrysophyte and diatom communities to infer long-term trends in pH and metals in three lakes in the Sudbury region of Ontario, Canada. The chrysophyte community was shown to respond more quickly and more markedly to acidification than the diatom communities in the lakes. Vernal blooming and euplanktonic chrysophytes tracking spring depressions in pH are thought to account for the closeness of the chrysophyte inferred data to the observed pH trend (op cit.). Chrysophytes may infer lower pH values than those of diatom inferred data because the chrysophytes respond to the spring water chemistry when pH values are lower and aluminium levels higher, whilst diatoms infer average, ice free conditions (op cit.).

Chironomids (Chironomidae)

Adult chironomids are delicately built, long-legged, frequently brightly coloured members of the non-biting midge group. The family Chironomidae is typified by a thorax that overhangs the headpiece of the adult chironomid. They commonly form swarms hovering about water bodies at dusk.

Nearly all chironomid larvae are aquatic. They have worm like bodies with a pair of pro-legs on the prothorax and another pair on the caudal segment, with the penultimate segment sometimes bearing filamentous gills. They are found in a variety of colours, with some species (Chironominae) adopting a ‘blood-red’ colour; the result of haemoglobin-like pigments, which allow the larva to survive in oxygen depleted environments.

Chironomids are found in two habitats within lakes; the littoral benthic and the offshore, or profundal, benthic. Thienenmann (1920 cited in Frey, 1988) described a successional series of lakes based upon chironomid taxa from the offshore communities. At one end were the oligotrophic lakes represented by the Tanytarsus community, and at the other were the lakes with low dissolved oxygen levels, where communities of Chironomus were found. Today, oligotrophic lakes are characterised by a *Tanytarsus lugens* community because in Thienenmann’s original work complications arose as the genus Tanytarsus primarily occupies the shallower water inshore, with only a few, specialised species occupying the offshore areas (Frey, 1988). Most of the littoral species don’t move into the profundal range at all. This is probably due to low water temperatures in the profundal benthic region (Frey, 1988).

The remains of chironomid larvae found in lake sediments are derived from two sources; the actual remains of dead larvae and their molt stages (Frey, 1988). The chironomid larvae have four stages, the skin and head capsule are split allowing the larvae to eject a stage, thus allowing it to continue growing. The type of species will determine the degree of preservation of the head capsule in lake sediments(Brooks, pers comm). Species from the subfamily Tanypodinae have more robust head capsules and are usually found intact. The head capsules of species from the Orthocladiinae, on the other hand have a more brittle suture on the central axis and are often found to be split in the sediment.

Chironomids are useful as environmental indicators (Seather, 1979). The head capsules are chitinous and are found abundantly in the sediments of most lakes and can be identified to generic level. The chironomid fauna of lakes is species rich, which allows the transfer function approach to be applied for environmental reconstruction.

Chironomid remains have been largely used to infer changes in climate on a Holocene timescale and to construct chironomid-based climate prediction transfer functions (Walker et al., 1991; Walker et al., 1991; Wilson et al., 1993; Lotter et al., 1997). Lotter et al. (1998) used chironomids as part of a multi-proxy study of nutrients in the Swiss Alps. They found a strong relationship between chironomid communities and TP, with their predictive model using chironomids having a $r^2 = 0.68$ (op cit.).

Chironomids have also been used to investigate lake acidification (Henrikson & Oscarson, 1985; Henrikson et al., 1982), and studies have attempted to look at the responses of chironomid populations to increasing acidity (Griffiths, 1992). However, chironomids may not be responding to changes in acidity, rather to degrees of environmental stress (Brooks, pers comm.). The removal of acid sensitive species allows other species to dominate the chironomid fauna. When pH returns to higher levels (e.g. through liming, Brooks, pers comm.) these (so called) acid tolerant species remain in the sediment record after liming, even though the pH is now much higher. Therefore, the response shown by some chironomid species appears to be not just a response to hydrochemical change but to some sort of disturbance criteria.

Cladocera

Cladocera, or water fleas, are small crustaceans. The thorax of a cladoceran is enclosed within a folded oval carapace, and bears five or six pairs of limbs. Some species of cladoceran occur all the year round, others only develop from eggs during warmer periods or more-favourable conditions. The females reproduce rapidly, producing 'summer' eggs that require no fertilisation from the male. These eggs develop quickly from the brood pouch contained behind the carapace (Wetzel, 1983). When winter approaches or unfavourable conditions arrive, the females produce 'winter' eggs that require fertilisation from the males. These 'winter' or 'resting' eggs have a thick shell and can persist in the sediment for many years until conditions are suitable for their development. These eggs are resistant to freezing and desiccation, and are easily transportable.

Cladocera are represented in lake sediments by a variety of body parts. Cladoceran species can be identified by differentiating between the morphology of headshields, shells, post-abdomens, claws, and ephippia (Scourfield & Harding, 1958), which accumulate in the sediment. Cladocera remains recovered from lake sediments are primarily found to be derived from the littoral-benthic dwelling family Chydoridae (Nilssen & Sandøy, 1990). The planktonic species are usually restricted to *Bosmina* spp. Sedimentary records of Cladocera have previously been used to show the development from oligotrophy to eutropy in lakes (Nilssen & Sandøy, 1990). Clear changes in the species composition of the Cladoceran assemblages, where there is distinct replacement of species within the Chydoridae and the genus Bosmina, can be related to changes in fish predation pressure and lake trophy (e.g. Kerfoot, 1974; Boucherle & Züllig, 1983).

The direct response of the cladocera to acidification is more problematic. Interpreting changes in species assemblages and abundances over small changes in pH is complex, and detailed knowledge of cladoceran autecology is poorly understood below pH 5.5 (Nilssen and Sandøy, 1990). Changes in species composition often related to acidification could reflect indirect effects, resulting from changes in predation pressure and vegetation type and cover that are brought about via changes in pH.

A number of palaeolimnological studies have, however, demonstrated changes in cladoceran assemblages related to acidification (Steinberg et al., 1988; Uimonen-Simola & Tolonen, 1987;

Nilssen & Sandøy, 1990). Increasing acidity has been demonstrated to effect changes in community interactions, the loss of acid sensitive species and of species richness as a whole, and changes in total individual numbers of cladocera (Paterson, 1994).

Other fossil groups found in lake sediments

The fossil organisms described above are not the only biological indicators that are preserved in the sediment record of lakes, although they are the most widely studied groups in acid lakes. Lakes support a rich and diverse flora of living organisms and many of these are represented in the sediment record.

In well-buffered systems the calcite, bivalve shells of the group ostracoda are preserved. These shells have been used to infer changes in lake water O_2, salinity and other climate related variables (DeDecker et al. 1988). In low alkaline systems mandibles from the genus Chaoborus have been used to effectively infer changes in fish predation pressure, and presence and absence of fish populations (Uutala 1990 & Uutala et al. 1994). A number of other algal groups are preserved in the sediment as non-siliceous fossils. These include the Pediastrum (Chlorococcales) group, which are the best preserved of the other algal groups (Charles et al., 1994). In suitable and very rare conditions, however, green, blue-green and dinoflagellate vegetative cells and resting spores can also be found (Livingston, 1980). The reproductive structures of charophytes (oospores) are also preserved. Biochemical fossils such as photosynthetic pigments are also being investigated to attempt to trace past changes in algal and bacterial populations that do not leave reliable morphological fossils (Leavitt et al. 1989).

Little work has been done, however, to assess the degree to which many of these fossil organisms quantitatively and qualitatively represent the structure and productivity of the ecological community from which they were derived (Battarbee, 1991). Lack of representation stems from problems in the preservation of the organism post mortem (taphonomy) and because whole biotic groups appear to be missing or greatly underrepresented in the sediment record (op cit.).

From an ecological standpoint restoration of aquatic ecosystems should represent the restoration of biological activity: achieving working ecosystems in which macrophytes, zooplankton, plankton and other aquatic fauna are functioning within their normal range of activity.

Freshwaters are perturbed by two different sets of impacts (Battarbee, 1997); contamination of the surface waters from a diverse range of pollutants and habitat disturbance such as physical alterations to the shoreline or catchment (e.g. hydroelectric dam construction). How pollution and habitat disturbance interact to determine the nature of environmental change in freshwaters needs to be considered if the restoration of freshwaters is to be attained. Battarbee (1997) illustrates the relationship between pollution effects and habitat disturbance in the form of a naturalness matrix (see Figure 2)

Ecosystems can be related to each other in terms of their structure and their function. The number of species and the organisational complexity of the ecosystem define ecosystem structure. Ecosystem function is a combination of the biomass and nutrient content of that ecosystem (Bradshaw, 1984). Degradation of an ecosystem will result in the reduction of ecosystem function or structure or both. Bradshaw (op cit.) represents this in the form of a diagram (Figure 3). Natural ecosystem processes will move a degraded ecosystem along a theoretical pathway back to its original state. This process will take a long time to complete and can only take place if the forcing that led to the degradation has been removed. If this has not been removed further degradation of the ecosystem may take place. Moving the ecosystem along the pathway to the original state artificially is the act of restoration. Bradshaw (op cit., & 1996) proposes a number of options for where restoration may not be possible; the original state of the ecosystem may be the result of human land use practices that are now outdated and no longer practised, for example. These options are rehabilitation and replacement. Rehabilitation is progress made towards the original state that is not complete. Ecosystem function and structure have been improved, but the pre-disturbance conditions have not yet been achieved. Alternatively, another ecosystem can be substituted for the degraded one. The substitute ecosystem is generally less complex than the original state of the degraded one. This process is known as replacement. True restoration may be unrealistic in many situations. It may be expensive or inappropriate. In these circumstances, rehabilitation or replacement may be better suited to providing an ecosystem more valuable in function and complexity than the degraded ecosystem.

Bradshaw (1996) has also proposed that by aiming at ecosystem restoration we are setting too high a target for ourselves. In current ecological terminology, the word ‘ecosystem’ describes both the biological and the non-biological elements that occur together in a given area. Restoration of an ‘ecosystem’ then should revolve around restoring the function, structure, and the interaction of the whole system. Consequently, it is now common to talk about ‘habitat’ restoration where the emphasis is placed upon restoring the ‘place’ where organisms live (the habitat) rather than the processes of a degraded ecosystem.

Good ecological restoration entails negotiating the best possible outcome for a specific site based on ecological knowledge and the diverse perspectives of interested stakeholders (Higgs, 1997). It should be noted then, that simply restoring a disturbed ecosystem to its former state

Figure 2: Naturalness matrix showing: (1) hypothetical change over baseline for three lakes (a, b, & c): (2) a 'state-changed' classification system related to change over baseline, with 1=most changed, 9=least changed and 10=pristine (from Battarbee, 1997).

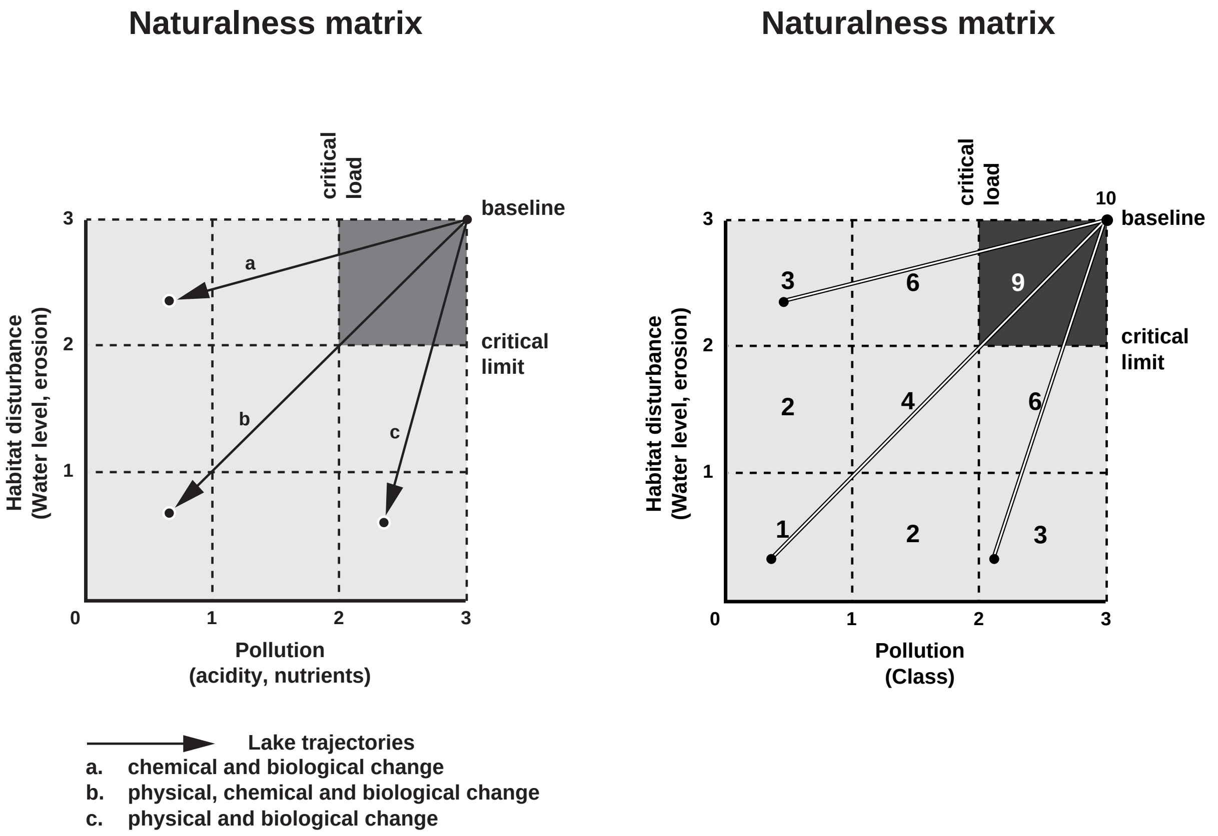
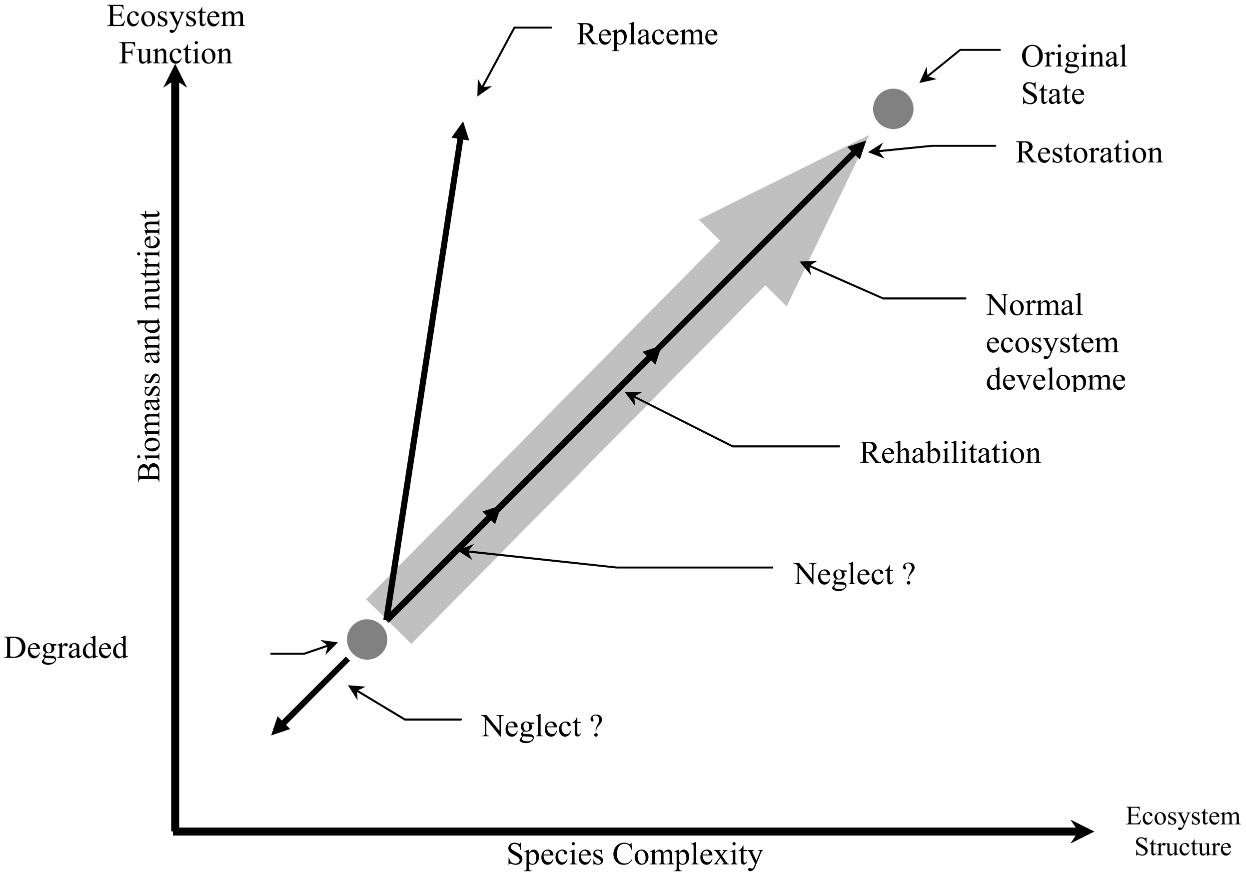


Figure 3: Changes in ecosystem function and structure for a hypothetical degraded ecosystem. Note the various forms of 'restoration' and the changes associated with true restoration (after Bradshaw, 1984).



does not always form the most appropriate use of the available funds or the most appropriate restored community.

The use of baseline conditions as restoration targets or reference states

Most surface waters in Scotland, Wales and Northern England have existed for around 10 000 years. They were formed as glaciers retreated when the climate began to warm at the end of the last glacial. During this period natural climatological, geological, geomorphological and biological change has taken place. More recently, anthropogenic changes have begun to affect UK surface waters, most notably the eutrophication and acidification of lakes and rivers. Catchment processes or the hydrochemistry of certain lakes may have buffered some or all of the effects of change. Other, more sensitive, surface waters may have been altered by the natural and anthropogenic changes they have been exposed to.

The potential influence of natural factors has clear implications when we consider defining the baseline state of degraded (acidified) surface waters. The baseline state of the surface water could be defined as the state prior to the onset of acidification, or alternatively as the state immediately following the retreat of the glaciers and the lake formed. The potential effect of climate change on lake water acidity is discussed in more detail below. Such natural responses complicate the process of defining baseline state for acidified lakes, let alone describing the hydrochemical and biological status of such a state.

Bradshaw (1984) used the term ‘original state’ to define the pre-impact state of an ecosystem. In palaeolimnology, this state can be described as the ‘Baseline State’ of surface waters. The baseline or pristine state of a surface water is that which combines an absence of habitat disturbance and an absence of pollution. It can be argued that this represents the highest quality of surface water that is attainable in the UK, and that these conditions should be the main goal of any restoration project.

The baseline can also be thought of as the state prior to a specific impact. In the UK, evidence of acidification is rarely found before the 1850s. This period can be defined as the baseline for acidification but does not necessarily reflect the state where there has been no anthropogenic influence on the lake. In this way baseline conditions can be used to define future hydrochemical and biological reference conditions, or if unobtainable, due to irreversible change, alternative targets that approximate natural conditions can be defined (Battarbee, 1997). In such cases, some of the most significant baseline biological characteristics can be defined from the sediment record contained within the lake.

Hydrochemical and Biological Targets for Restoration

Studies of disturbance in lakes often revolve around the decline or accumulation of a certain chemical variable or suite of associated variables in the lake water (for example, total phosphorus (TP) (Bennion, 1994), pH (Flower & Battarbee, 1983) and Al and DOC (Kingston & Birks, 1990)). Targets could be set with reference to certain hydrochemical or biological criteria. Reconstructing the hydrochemical characteristics of surface waters has often been achieved through the use of indicator species (e.g. Birks et al., 1990b); the hydrological change being defined by the response of a particular group of organisms to that change. Other studies (e.g. Henriksen et al., 1990) have used point source measurements to determine the degree of change between studies in surface water hydrochemistry, or to identify those surface waters that are sensitive to environmental change.

Consequently, because of this need to quantify biological change in response to environmental change for hydrochemical reconstruction, the biological response to acidification is fairly well understood (e.g. Havas & Rosseland, 1995). The effects of lake acidification have long been identified, with the major loss of fish populations and aquatic vegetation within acidified systems being the most immediately identifiable of responses of the system to decreasing pH (Uutala, 1990; Kingston et al., 1992; Havas & Rosseland, 1995). Other studies have looked at the effects on algal populations, such as diatoms or chrysophytes (Battarbee, 1984), whilst other authors have described the change in insect (Brodin & Gransberg, 1993) or zooplankton (Havas & Rosseland, 1995) populations associated with acidification.

The return of fish stocks to lakes is an easily identifiable, ecological (biological) target for scientists, politicians and the environmentally aware public to focus upon, as well as being an issue of great economic importance. The reliance upon this sort of target, and the use of solely hydrochemical parameters to define it, has slowed progress towards adopting relevant biological reference states that address the whole lake system rather than a small part of a lake's ecosystem. Henriksen et al. (1990) sampled many lakes across Norway for the 1986 1000-lakes survey. The project was designed to determine the surface water chemistry and the status of fish populations of lakes from acid sensitive areas of Norway. The data was used to identify changes in water chemistry and fish status since 1974 and 1975 (c.f. Henriksen et al., 1990) and to act as baseline characteristics against which the effects of emission's reductions (e.g. LRTAP protocols) could be judged. The only biological aspect incorporated in the 1000-lakes survey was the fish status of the surveyed lakes. It was only later, through further study of the other biological groups found in aquatic systems, that the degree to which other organisms were affected by acidification was identified (Havas & Rosseland, 1995).

Liming: an example of lake acidification restoration

A key technique employed by those attempting to restore acidified surface waters has been to lime either the water directly or the catchment that drains into the acidified lake (Renberg & Hultberg, 1992). Although the impetus behind such restoration methods has been a biological one it is still essentially a hydrochemical approach in nature. The aim of liming is an elevation of the pH of the water (and the increase in acid neutralising capacity (ANC) and the decrease in Al toxicity that go with it) that will allow fish populations to be re-introduced or tore-colonise.

Each year approximately 300 000 tons of powdered limestone is used to treat lakes, rivers and wetlands in order to increase the pH of surface waters across the globe (Henriksen et al., 1995). This costs between 40-50 million US$ per year (Henriksen et al., 1995). There are over 11 000 lakes that are routinely limed in Sweden and Norway alone.

Originally, the main aim of liming was to restore fish stocks to impacted lakes for economic and recreational fishing (Henriksen et al., 1995). These aims have now been broadened to encompass the preservation and recovery of biodiversity and human health. The official aim of the liming projects in Sweden (Svensson et al., 1995) and Norway (Romundstad & Sandøy, 1995) are shown in Table 3.
In the UK there has been far less use of liming to mitigate the impacts of acid deposition (c.f. Howells, 1995). At Loch Fleet, Southwest Scotland, an extensive study of liming applications and the biological and chemical responses to the treatment was begun in 1986. Around 450

tonnes of $CaCO_3$ was applied to various parts of the catchment, and the water quality of the loch quickly improved following the treatment. pH rose to 6-7, calcium to 4 mg l^{-1} and labile Al was greatly reduced (Howells, 1995).

Table 3: The primary aims of liming of surface waters in Sweden and Norway (Source: Muniz, 1991).

Sweden	The biological aims are to detoxicate the water for the preservation or recolonisation of the natural flora and fauna
	The chemical aims are to raise the pH above 6.0 and alkalinity to levels exceeding 0.1 meq l^{-1}
Norway	To improve the conditions for recreational fishing
	To preserve biological diversity

The water quality proved suitable for the return of trout through restocking, and 18 months after initial treatment the restocked fish have shown good growth and fecundity (op cit.). There have also been changes in the invertebrate fauna of Loch Fleet, a decline in predatory beetles (thought to be due to fish predation) and the return of some acid-sensitive species. Flower et al. (1990) conducted a palaeolimnological study of Loch Fleet using diatoms to determine the biological response of this important group to the liming treatment. They concluded that the pH of the lake had risen to a level that had never been reached throughout the history of the lake (Flower et al., 1990; Anderson et al., 1986). The diatom taxa present after liming contain species that were previously rare or absent from the diatom sedimentary record.

This demonstrates the crux of the problem with liming as a restoration tool. Far from restoring the pre-acidification flora and fauna of acidified lakes the results show that conditions created in lakes after liming are often different to any condition previously seen in the lakes through the palaeolimnological record. This cannot be considered restoration in the sense of Bradshaw, rather it is replacement. In essence a degraded ecosystem is being replaced by one that restores a target organism to the treated lake but which does not reflect the original state of the lake in terms of structure or function. This may be an appropriate target for which to aim if the biological target is the restoration of fish stocks to impacted lakes. However, it should not be taken to represent the original state of the degraded ecosystem. For true restoration to be achieved the biology of the original state of the degraded ecosystem must be restored. Identifying such targets or reference states, therefore, should involve biological as well as hydrochemical variables and should be related to conditions that are similar to those previously exhibited during the history of any given lake. Palaeolimnology can define such targets or reference states in terms of the hydrochemistry and selected biological indicators.

Defining baseline states and restoration targets

To define a baseline or reference state for a surface water hydrochemical reconstructions have traditionally been used to provide information about the pre-disturbance hydrochemistry of surface waters. There are two potential ways that hydrochemical reconstructions can be made; dynamic modelling and transfer functions. Dynamic modelling uses information about the hydrological and chemical influxes to a catchment and other variables such as soil CO_2 partial pressures, soil porosity, bulk density, and cation exchange capacity, to model hydrochemical changes for given deposition patterns. One such dynamic model is MAGIC (Model of Acidification of Groundwaters in Catchments) (Cosby et al., 1985a,b). Using known historical deposition levels (usually only for the last decade or so) the models can be used to hindcast the historical hydrochemistry of surface waters. However, most often the models are used to predict future hydrochemical targets for given levels of emission abatement such as the LRTAP conventions discussed above. The other main way in which hydrochemical reconstructions are made is using palaeolimnological techniques and hydrochemical transfer function. As discussed above, certain biological indicators (usually the siliceous algae, diatoms and chrysophytes) have strong responses to hydrochemical variables, primarily pH, phosphorus and salinity. These can be quantified, and using weighted averaging calibration, used to infer the historical hydrochemistry of surface waters from the sediment record. This then acts as a record of the hydrochemistry of a surface water. Then a pre-disturbance period can then be identified and the hydrochemistry inferred from the sub-fossil remains of the sediment record.

The use of the palaeolimnological record to set biological targets for restoration is considered in the next chapter.

Chapter 4: Setting Biological Targets for Ecosystem Restoration Using Palaeolimnology and the Modern Analogue Approach

One approach to defining biological targets or reference states using palaeolimnological data is the modern analogue approach. The modern analogue approach uses robust statistical methods to compare fossil assemblages to modern assemblages. This approach was developed by Flower et al. (1997).

Modern Analogues

Modern Analogues are lakes that have modern chemistries and biological communities that match the past conditions of a lake. Defining analogue sites can provide the chance to examine the ecology of selected lakes so that the biological characteristics of the analogue lakes can be used to define biological targets or reference states for recovery from acidification. Space-for-time substitution is the basis of selecting sites as modern analogues. This assumes, however, that the aquatic communities currently existing in sites closely resemble those that existed in similar but now acidified habitats (Flower et al., 1997). The sites and the biological communities they contain can vary greatly through subtle differences in geology, climate or other biogeographic and chemical variables.

The methodology behind modern analogues has its foundations in the field of palynology (Overpeck et al., 1985). Fossil pollen spectra derived from palaeoecological studies have been compared to the pollen spectra retrieved from pollen traps and taken from modern sites. This has allowed palynologists to match modern forest ecosystems to those that existed in the past (e.g. Wright, 1967). From this, they could interpret the type of ecosystem and vegetation that once existed and evaluate the changes that have brought the ecosystem to that which is found today. Early studies used qualitative matches of the various pollen spectra. Modern studies use statistical methods to quantify the degree to which modern sites match the fossil flora (c.f. Overpeck et al., 1985). These statistical methods are known as dissimilarity coefficients.

Dissimilarity coefficients measure the difference between multivariate (e.g. species abundances) samples such as those produced in palaeolimnological studies. Dissimilarity coefficients have a number of advantages:

1. They are a precise method of comparison between samples;
2. The process may be automated or computerised;
3. Dissimilarity Coefficients allow the opportunity to calibrate the dissimilarity scale in terms of the underlying biological or environmental differences.

Other, multivariate, approaches also exist (e.g. using principle components analysis or canonical variates analysis) but these do not directly compare the degree of analogy that dissimilarity coefficients measure.

Prentice (1980) suggests that there are three types of dissimilarity coefficient, the simple or unweighted, the equal weight, and the signal-to-noise measures. Simple methods tend to be heavily influenced by taxa that have wide ranges, and the rare types have little influence upon the results. Equal weight measures upweight the rare taxa and downweight the common species. A potential problem is that these tests may give extra weight to the insignificant, yet potentially noisy, taxa. The method of upweighting is dependent upon the statistical test being used. Signal-to-noise measures avoid this by weighting the data so that the signal component of the difference between fossil and modern samples is emphasised at the expense of the noise component. The problem with signal-to-noise measures is that they are scale dependent (Webb et al., 1978; Delcourt et al., 1984). Equal weight measures may prove more appropriate if certain minor taxa are judged important to the analysis. Overpeck et al. (1985) tested various dissimilarity coefficients for application in analogue matching tests. They found that the signal-to-noise methods provided the more robust analysis because rare taxa are down-weighted and the floristic signal drawn out. Their results also showed that the three signal-to-noise measures they tested all provided similar results.

There has been very limited use of modern analogues, especially in any quantitative way, within the field of palaeolimnology to tackle the issues of recovery in acidified lake systems.

An exception is the work of Flower et al. (1997), who identified modern analogues for two Scottish lochs from a modern dataset of lakes from northern Europe. Flower et al. (1997) used the squared chi-square distance measure in their study of diatom based modern analogues for the pre-acidification status of recently acidified lakes in northern Europe. They compared the fossil diatom assemblages in two acidified lakes from the Galloway region of Scotland (the Round Loch of Glenhead and Loch Dee) with modern surface sediment samples from over 200 lakes in Britain, Ireland, Sweden and Norway. The squared chi-square analysis identified several modern analogue sites within the dataset. Loch Teanga in the Hebrides and Lough Claggan, Ireland had the most similar diatom floras to the pre-acidification floras of the Round Loch of Glenhead and Loch Dee respectively. These sites, however, can not be considered as being pristine sites (i.e. sites that have not been impacted by atmospheric deposition). Loch Teanga and Lough Claggan are located in areas of low to moderate acid deposition (0.4-0.8 g S m^2 yr^{-1}) (Flower et al., 1997; Battarbee et al., 1988; Flower et al., 1994). Although the diatom communities of these lakes do not show any response to this level of deposition other biological communities may have been affected by deposition or trace metal contamination. Flower et al. (1997) also looked at the other hydrochemical determinants of the lakes selected as analogue matches for the Round Loch of Glenhead and Loch Dee. The Round Loch of Glenhead had no modern analogues that had a similar lake-water calcium concentration. Loch Dee, however, had some analogues with similar calcium concentrations (e.g. Loch Howie, Scotland, and Lough Brockagh, Donegal) and these sites may represent better analogue sites than Lough Claggan.

The approach has since been applied to other Acid Waters Monitoring Network Sites (Allott, pers. comm.).

With the adoption of robust statistical techniques, the application of modern analogues to identifying potential targets for the restoration of acidified surface waters has become a potentially powerful tool.

The approach used by Flower et al. (1997) raised a couple of interesting points regarding the suitability of analogue sites selected in this manner. These will be discussed further in the following section.

Chapter 5: Key Assumptions and Problems of the Modern Analogue Approach

The modern analogue approach makes a number of assumptions about the data used and the theoretical concepts of analogue matching (c.f. Flower et al., 1997). These assumptions need to be tested before analogue matching techniques are applied to the study of recovery, as they are inherent in understanding the validity of substituting space for time in biological studies.

The key assumptions used in the modern analogue approach are:

1. The fossil group (or groups) used for analogue matching effectively represents ecosystem variation.
2. The modern surface sediment dataset contains suitable analogue lakes for the acidified lakes that will be matched with them (e.g. that there are enough pristine, high calcium concentration sites to provide an adequate range of matches).
3. The hydrochemistry and biological communities of analogue sites accurately represent the pre-acidification conditions of acidified lakes.

Do diatoms effectively represent ecosystem variation?

It has to be assumed that the fossil group used in analogue matching (e.g. diatoms in the Flower et al., (1997) approach) is representative of the whole trophic cascade and variation in other biological groups found in aquatic systems (e.g. other algal groups, aquatic macrophytes, other invertebrate groups, or fish). Furthermore, the variation in the fossil record is assumed to accurately reflect changes occurring to the whole biology of a given lake. This has to be assumed because not all groups leave remains that are preserved in the sediment record of lakes (see Palaeolimnology section earlier). Therefore, analysing change in other biotic groups is impossible without these inferences unless historical records are available.

It is well known that diatoms respond quickly and with definite pattern to changing lake-water acidity. Given this information, it is likely that a matching process based solely on sub-fossil diatom data is selecting analogues with similar pH characteristics to those inferred from the surface sediments of presently acidified lakes rather than on the whole hydrochemical signature. It is likely, therefore, to expect that the results do not necessarily reflect whole ecosystem characteristics.

Does the modern data set contain a large enough range of sites from a range of hydrochemical characteristics?

In the Flower et al. (1997) application, the lakes selected as the closest analogues to the Round Loch of Glenhead and Loch Dee were lakes that had, somewhat, been impacted by acid deposition. They were from areas receiving low to moderate acid deposition. Flower et al. (1997) suggest that this was due to there being insufficient pristine sites in their dataset. Another problem they encountered was that the two selected analogue lakes had very different hydrochemistries apart from their pH levels (as inferred from the diatom data). Consequently, it is important to assume that the dataset being used for analogue matching has sufficient geographical range and hydrochemical scope for the matching process to be worthwhile.

Flower et al. (1997) recognise that there is a need to incorporate surface sediment data from more pristine sites. In their dataset, there were very few pristine sites, the majority found in northern Norway. Including more pristine sites would improve the chances of pristine lakes being selected as modern analogues. This is problematic because there are very few areas of Northern Europe that can be considered pristine as there are large areas of northern Europe that have been impacted somewhat by acid deposition.

Can baseline conditions be adequately defined?

For the process of space-for-time substitution to be accepted as an adequate model, it must be assumed that the hydrochemistry and biology of analogue lakes are a true reflection of pre-acidification conditions observed in acid lakes. This assumes that anthropogenic acidification is the only or the major forcing mechanism affecting environmental change upon acidified lakes in the UK over the Holocene period. This also assumes that climate change or other anthropogenic influence (e.g. habitat disturbance) since pre-acidification times (c. AD 1850) has had negligible effects upon the overall hydrochemical and the biological functioning of the lake systems.

There is some degree of evidence that climate changes and other anthropogenic influences (e.g. catchment disturbance or land-use change) have had profound impacts on the hydrochemistry and biology of some studied lakes across northern Europe.

Natural variability in surface water systems

Nature is inherently dynamic: A constantly changing or fluctuating climate has been a major driving force in determining the present day distribution of ecosystems and the physical appearance of the landscape that they occupy. Among this background of climate change and natural variability anthropogenic activities have influenced and changed ecosystems, predominately since the industrial revolution, but also during the forest clearances of the Neolithic period and later clearances as land was prepared for agricultural use (Roberts, 1998).

An important consideration in the ecological restoration of acidified sites is the degree to which the pre-acidification hydrochemical and ecological conditions may have changed. There are three main themes that need addressing when considering the fluctuating hydrochemical and ecological conditions of surface waters, the climate effect, catchment disturbance and land-use change effects, and the apparent stability of UK lakes prior to anthropogenic acidification.

Climate

For mountain lakes, Skjelkvåle & Wright (1998) suggest that palaeolimnological analogues may be of little use because future climate changes are likely to cause ‘conditions never previously experienced on earth, such as high atmospheric CO_2 levels and high UV-B radiation.’ (Ibid. pp-285). A number of other studies also describe the likely effects of a changing climate for acid lakes (e.g. Schindler et al., 1996; Psenner & Schmidt, 1992; Leavitt et al., 1997). These effects are summarised in Table 4. Such dynamism would compromise the applicability of the modern analogue approach and question whether targets for recovery from anthropogenic acidification are realistic goals.

To assess the impact or influence that fluctuations in climate have had upon lakes, long core stratigraphies that contain past records of climate fluctuation are required. Analysis of the cores for changes in biological assemblages over time and dating of these changes can answer questions surrounding whether upland acid waters were chemically and ecologically stable prior to anthropogenic acidification or are continually changing, dynamic systems.

Psenner & Schmidt (1992) have demonstrated a relationship between colder air temperatures and lower pH of surface waters in two soft-water, high-altitude lakes in the central Alps. The inferred pH data from diatom analysis and correlation with the temperature record was also supported by the results of loss-on-ignition (LOI) and Fe/Mn ratio analysis of the lake sediment. LOI and Fe/Mn are surrogates for the organic content of the sediment. Psenner & Schmidt (1992) found three distinct peaks in the LOI and Fe/Mn data that correspond roughly to the temperature peaks of 1820, 1860, and 1900. Prior to the onset of anthropogenic acidification in the late 19th and early 20th centuries there was a distinct coupling between climate and biogeochemical processes in the two alpine lakes. This coupling has progressively broken down over the last 100-150 years as the deposition of strong acids to surface waters has had greater influence on the biogeochemical processes of the lakes than the fluctuating climate. This has clear implications for assessment of recovery in systems that have climate/biogeochemical coupling. Psenner & Schmidt (1992) suggest that recovery observed in some systems, by a levelling off of the inferred pH decline, may be due to rising temperatures and global warming as much as that due to decreasing acid deposition.

Catchment disturbance and land-use change

For Swedish lakes, Renberg (1990) and co-workers (Renberg et al., 1993a,b) have produced long-core stratigraphies and have analysed the diatom profiles of a great number of lakes. The results show a number of features characteristic of diatom-inferred pH profiles from some of the acidified lakes in Northern and Southern Sweden. Swedish lakes are generally base rich, well buffered and mesotrophic immediately following the end of glaciation. Following this a period of slow natural acidification takes place where progressive leaching and loss of base cations from the soils leads to soil acidification and dilution of the Acid Neutralising Capacity (ANC) of run-off entering the lakes (Renberg, 1993b). The pH of the Swedish lakes immediately following retreat of the ice sheets has been shown to be around 7.0 and the slow process of natural acidification increased acidity to c. pH 5.5.

Acidification continued up until ca. 2300 BP when suddenly, and across a geographic range, the pH of lakes rose quickly to c. 6.5. This has been attributed to a shift in land-use in the catchments of many Swedish lakes known to have taken place around this period of history because of an expansion in the agrarian culture of the Iron Age. Increases in pH have been correlated with the recession of natural forest, expansion of shrub vegetation, increased frequency of cereals and weeds, increased concentrations of charcoal and LOI values, indicating increased soil erosion (Renberg, 1993b). Cultural alkalisation continued until the industrialisation of the 1900s. Many lakes in Sweden have acidified considerably since the 1950s; many lakes becoming severely acidified with permanently reduced pH values of between 4.5 and 5.0.

Renberg et al. (1993a) also identify a recent liming period in many of the acidified Swedish lakes. Liming has been adopted as a widespread restorative measure in over 6000 lakes (Renberg, 1993a). Liming characteristically increases pH to levels above those of the immediate post-glacial, and the resultant diatom flora is quite unlike anything that has been found in the post-glacial history of the Swedish lakes.

Table 4: Possible effects on lake-water chemistry from different possible effects of Climate (Source: Skjelkvåle and Wright, 1998)

Environmental driving variable	Primary Effect	Secondary Effect	Biological impact	Type of Study	References
Climatic warming	Increased water temperature reduced water turnover	higher concentrations of solutes	decreased summer habitat for cold stenothermic organisms	empirical data, 20-year natural climatic cycle (Canada)	Schindler et al., 1990
Climatic warming	Increased water temperature		changes in fish yield	modelling study (Canada)	Minns & Moore, 1992
Climatic warming	Increased water temperature	increased alkalinity generation	changes in diatom communities	palaeolimnological study (Austria)	Psenner & Schmidt, 1992
Stratospheric ozone depletion	Increased UV-B		reduction in photosynthesis and growth of diatoms	mesocosom experiment	Bothwell et al., 1994
Climate warming and acid deposition	Increased mineralization of soil organic matter	increased NO_3 and acidification		large-scale experiment CLIMEX (Norway) empirical data (Norway)	Lükewille and Wright, 1997, & Wright, 1998 Lydersen, 1995
Climate warming and acid deposition	Decreased DOC concentrations	increased light penetration (UV-B)	photoinhibition of phytoplankton	empirical data (Norway)	Schindler, Bayley & Parker, 1996, & Schindler et al., 1996
More strong winds (Storms)	More 'sea-salt episodes' in lakes in coastal areas	'acidic episodes' in acidified areas	damage to aquatic biota and fish kills	empirical data (Norway)	Hindar et al., 1994 Hindar et al., 1995

From this type of study, it is clear that considerable anthropogenic influence on Swedish lakes has been experienced since around 2300 BP. This has implications for any attempt to restore these acidified lakes to a former state. The problem now facing environmental managers is that if they are to restore lakes to pre-acidification states then they will be attempting to return lakes to conditions that were inherently dependant upon the cultural changes in land-use associated with the expansion of agriculture in the Iron Age. This type of land-use no longer exists in Sweden so there is little hope of restoring lakes to such a status. Renberg et al. (1993a) have extrapolated the predicted pH response to continued natural acidification given that none of the changes in land-use or deposition chemistry of the last 2300 years had taken place. Arguably, this provides a suitable target that could be attainable given pollution abatement strategies already in place or being negotiated. This is a theoretical attempt to express the expected trajectory following the overriding trend in natural acidification over the previous 10 000 years following deglaciation. However, this provides no more a representative target than the pre-acidification state of surface waters and there is no guarantee that this is a more attainable target given possible changes in the hydrochemical, physical, biological and climatological variables since 2300 BP.

Pre-acidification chemical and biological variation in UK lakes

Atkinson and Haworth (1990) provide a similar study this time of two sites in the UK, Devoke Water in the Lake District, Cumbria, and Loch Sionascaig, Northwest Scotland. Similar trends of slow natural acidification to those found in Swedish lakes were found following de-glaciation. However, both lakes are not as sensitive to acid deposition as other upland sites in the UK and the Swedish lakes, and have suffered much less disturbance. Devoke Water has recently (post 1850) begun to acidify to a point where there are considerable biological changes within the waterbody. Loch Sionascaig, on the other hand, is a more well-buffered site and has a stable inferred pH profile for the past few thousand years. In neither of these two sites was a cultural alkalisation period identified. This suggests that upland UK systems are relatively stable during the Holocene compared to their Swedish counterparts. Further studies also demonstrate the apparent stability of UK upland surface waters prior to the onset of anthropogenic acidification. Jones et al. (1986) presented a diatom-based reconstruction of pH for a Holocene sediment core from the Round Loch of Glenhead, Galloway, Scotland. The diatom stratigraphy of that core showed little change other than that associated with the acidification of the lake shortly after the end of the last glacial, and the recent, anthropogenic acidification. Birks (1996) also demonstrates stability in the Round Loch of Glenhead using rate-of-change analysis to assess the degree of variability in diatom species throughout a Holocene sequence. The only significant period of change was found to be in the last 200 years that is associated with the anthropogenic acidification of the loch (Allott et al., 1992), and the immediate post-glacial as base cation rich soils began to leach, thus influencing the ionic composition of run-off.

Given the relative stability of UK upland acidified sites it should be reasonable to assume that the pre-acidification status of a given lake is an attainable target for restoration measures to attempt to recreate. However, the apparent stability of UK upland surface waters may be misleading. Only a few sites have been studied in the UK and the resolution of the studies has been biased towards detecting acidification in the upper layers of the sediment record. There has been little high-resolution analysis of pre-1800AD sequences.

Palaeolimnological techniques have been widely employed to study lake acidification. This approach has been central in testing the cause-effect relationship between acid deposition and lake acidification, and in assessing the magnitude and extent of surface water acidification across the UK.

Most of these palaeolimnological applications have been based on diatom analysis, and the use of diatom-pH transfer functions to make reconstructions of hydrochemical change in upland lakes associated with acidification.

Following the signing of the Second Sulphur Protocol, attention is now focusing on emissions reductions and the reversibility of surface waters acidification. There is a clear need for criteria against which to evaluate the recovery process.

In order to evaluate future recovery, Flower *et al.* (1997) have proposed a palaeolimnological technique for defining targets for the recovery of acidified surface waters. This is based on the technique of analogue matching of lake sediment diatom assemblages. Multivariate statistical methods are used to identify modern analogues for the pre-acidification diatom assemblages of acidified lakes. The chemical and biological status of modern analogue lakes can then potentially provide recovery targets for acidified systems.

This approach has been successfully applied to several acidified lakes, and modern analogue systems defined for the pre-impact (pre-acidification) status of these impacted sites. An advantage of the approach is that it can provide recovery targets for both chemical and biological status of acidified lakes.

Modern analogue matching as currently applied makes several key assumptions:

1. that analogue matches based on a single biological group (diatoms) effectively represent the hydrochemical and biological variation of low alkalinity systems;

2. that the modern data set used to identify modern analogues contains the range of hydrochemical conditions represented by the fossil assemblages;

3. that a suitable stable ‘baseline’ (pre-impact) status can be defined.

Prior to more comprehensive application of the modern analogue approach to acidified lakes in Britain, these assumptions require evaluation. Three studies are proposed:

1. Extension of the current modern lake dataset used for analogue matching by the inclusion of minimally impacted low alkalinity sites from northern Scotland.

2. Development of the current technique by including two more fossil groups (chironomids and cladocera) in the modern surface sediment dataset used in the matching procedure. This will allow the assumption that diatoms represent wider ecosystem variation to be tested, and should result in more robust analogue matches.

3. A study of hydrochemical and biological variation in the pre-acidification conditions of acidified lakes through high-resolution palaeolimnological study of selected Acid Waters Monitoring Network lakes. This will allow the stability of baseline (pre-acidification) conditions to be evaluated.

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Task 2.5: Biological targets for recovery from lake acidification: Developing the analogue matching procedure

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2.5 Biological targets for recovery from lake acidification: Developing the analogue matching procedure

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Summary

Recovery in acidified upland lakes is the focus of much of the current work on lake acidification. Measuring and modelling recovery in the light of emission reduction protocols are important in evaluating the effectiveness of pollution abatement strategies. Evaluating recovery of acidified lakes towards a target based on their pre-acidification status is a more difficult task as there are very few biological records for these remote upland systems. The analogue matching approach can be used to identify modern analogues for the pre-acidification status of lakes using the sub-fossil remains of the diatoms and the cladocera; these analogues can then be used in the formulation of recovery targets.

An 83-lake modern training set was created, with full diatom, cladoceran, water chemistry and catchment data for each lake, selected to reflect the range of likely conditions found in upland systems in the 1800s. Fossil samples from 8 AWMN lakes were chosen to represent pre-acidification conditions in those lakes, and the closest analogues in the modern training set were identified by means of the squared chord distance measure.

Close modern analogues were identified for 7 of the 8 AWMN lakes. The majority of these modern analogues are located in North and Northwest Scotland; areas of low sulphur and nitrogen deposition. Comparison of the chemistry and catchment characteristics of the AWMN lakes to those of the modern analogues showed that the modern analogues had higher lake water pH and alkalinity levels and lower

aluminium concentrations. Ionic strength and calcium concentrations in the modern analogue lakes were also very similar to observed values in the AWMN lakes.

These results indicate that the analogue matching approach using diatom and cladoceran remains is a simple, robust and reliable method of identifying modern analogues for acidified lakes in upland areas of the UK. The modern analogues provide suitable reference conditions for pristine upland lake ecosystems and may be used to evaluate progress made towards restoration in acidified lakes.

Introduction

The acidification of sensitive surface waters through the deposition of strong acids has had serious impacts on the biological communities supported by these surface waters (Muniz, 1991). Using palaeolimnological techniques the cause of this acidification was shown to be the result of emissions of acid forming sulphur compounds from industrial sources (e.g. Charles & Whitehead, 1986; Battarbee & Renberg, 1990). As a result of these findings emission reduction protocols were adopted across Europe. These protocols have led to dramatic decreases in the levels of sulphur deposition throughout the UK and beyond (RGAR, 1997).

The focus of the work within the study of lake acidification has now shifted towards monitoring acidified systems for signs of recovery in the light of the reductions in acid deposition (Monteith & Evans, 2000). For us to be able to monitor the progress made towards recovery we need to be able to define a target community composition with which the present state can be compared. Ideally, this target would reflect the community composition of an acidified lake prior to the onset of acidification (Flower *et al.*, 1997). However, the majority of acidified lakes are found in remote locations and few, if any, have records of their biological communities prior to acidification.

The historical record of past environments, as recorded in lake sediments, is perhaps the only record of past community composition available for use in setting biological targets for recovery from acidification. Yet, here the record is limited to those organisms that leave identifiable remains in the sediments of lakes. The process of analogue matching makes use of the historical record of community composition recorded in lake sediments by comparing the historical record to the contemporary record found in the surface sediments of a range of modern reference lakes. By using a range of sediment slices it is possible to identify a series of targets that may represent total restoration of the lake or other targets that fall along the path towards total restoration, but which may reflect more appropriate or attainable intermediate targets in the short term.

Analogue matching is a statistical technique that compares the degree of floristic or faunistic similarity between a fossil sample taken from a lake and a range of modern samples selected so as to reflect the range of likely conditions found prior to disturbance (Overpeck *et al.*, 1985). It is assumed that those samples selected from the modern lakes as being the most similar to the fossil sample will also have similar community composition in those species which do not leave identifiable remains in lake sediments.

Analogue matching has mainly been used by pollen analysts where fossil pollen spectra retrieved from lake or bog sediments were compared to modern pollen spectra taken from a range of habitats. By identifying the most similar modern pollen spectra to each fossil pollen spectrum it is possible to infer environmental conditions in the past from those found today where the modern analogue spectra were recovered.

Flower *et al.* (1997) first demonstrated the application of the modern analogue approach to the issue of restoration in acidified lakes. Pre-acidification diatom sub-fossil remains from two acidified lakes were compared with the diatom sub-fossil communities of a modern training set of lakes from Northern Europe. Flower *et al.* (1997) found that the technique identified a number of lakes in the modern training set that were suitable modern analogues for the pre-acidification diatom communities of the two acidified lochs. However, many of the modern analogue lakes selected were shown to have a dissimilar water chemistry properties to those of the acidified lochs. In particular, the most similar modern analogue lakes had higher lake-water calcium levels than those observed in the acidified lakes today. Furthermore, the calcium content of the lake water in these modern analogue lakes would have afforded them a high degree of protection from acidification, so that it is unlikely that they would have acidified.

Flower *et al.* (1997) showed that the analogue matching approach had the potential to be used as a tool for setting biological targets or identifying reference states for acidified lakes but that there were some underlying problems in the approach. Simpson (2000) addressed these problems and potential solutions were suggested. The work reported here is an attempt to develop the analogue matching approach, following the recommendations of Simpson (2000), to provide a more robust method of identifying appropriate reference conditions for acidified, upland surface waters in the United Kingdom.

The approach described here combines the sub-fossil remains of the diatoms with those of the cladocera, a group of microscopic crustaceans, into a single, reliable and robust methodology for determining reference conditions in acidified, upland surface waters.

The diatom community response to pH is particularly strong (e.g. Round, 1990), masking subtle differences in the community composition amongst lakes that may be due to other environmental factors important for other biological groups found in acid sensitive surface waters.

The cladocera are members of the zooplankton. They form one of the primary trophic pathways in lake ecosystems, transferring energy generated by the primary producers up the food chain to higher trophic levels such as invertebrates and fish. Major environmental factors that influence the distribution of the cladocera are substrate type, presence of fish or invertebrate predators, presence, abundance and type of aquatic macrophyte assemblages (Quade, 1969), climatic factors (temperature etc.) (e.g. Hofmann, 2000) and water chemistry (pH, nutrients etc.) (e.g. Nilssen & Sandøy, 1990). This contrasts with the overwhelming dominance of water quality, in particular lake-water pH and aluminium concentrations, in determining the distribution of the diatom communities in oligotrophic, acid sensitive, upland lake ecosystems.

Cladoceran distribution patterns are heavily influenced by a range of environmental factors, the majority of which do not leave an identifiable signature in the sediment record. These environmental factors are important for a range of other species groups, such as fish, aquatic macrophytes, and invertebrates. By incorporating the cladocera into the analogue matching procedure the criteria for selecting an appropriate modern analogue should reflect a wider range of important environmental factors than by using the diatoms alone.

Studied sites and methods

A total of 83 lakes were selected from an initial dataset of 163 lakes to form the modern training set used in the analogue matching approach (Figure 1). Lakes were selected to reflect the range of likely hydrochemical and biological conditions found in acid-sensitive, upland areas of the UK.

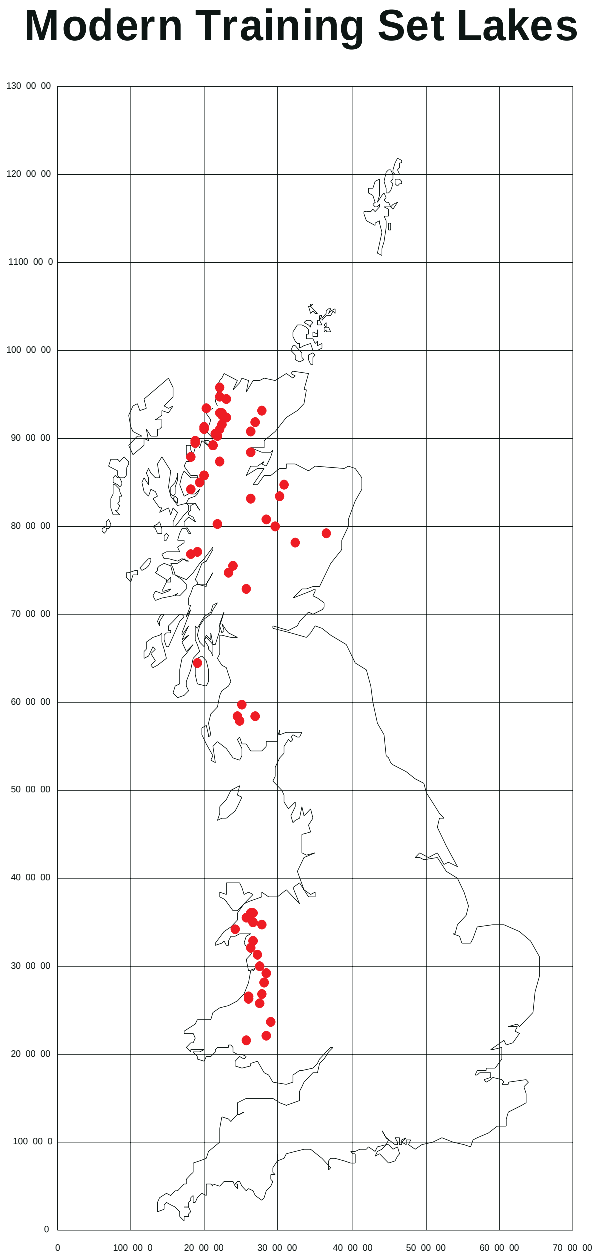
Previously counted diatom samples were extracted from the ECRC’s database, Amphora, as were catchment and water chemistry data for each of the 83 modern training set lakes.

The cladocera remains were enumerated from dried sediment samples (c.f. Simpson unpublished data) using a standard deflocculation of the samples in hot 10% KOH followed by washing and sieving (37 μm mesh size) and subsequent mounting of a known amount of the residue onto a microscope slide. The remains were identified using the standard reference works (Frey, 1959; 1960; 1962; 1965; Goulden & Frey, 1963; Floßner, 1972). At least 200 remains from each sample were enumerated where possible, but some samples had very low numbers of remains per cm^3 of sediment and subsequently fewer remains were enumerated from these.

The diatom and cladoceran counts for selected Acid Waters Monitoring Network (AWMN) fossil samples were collected in the same manner.

Ordination techniques were used to explore the relationships between the diatoms and the cladocera and the environmental variables (ter Braak & Verdenschot, 1995).

Figure 1: Map showing the locations of the 83-lake modern training set.



These analyses identify the main environmental variables that determine the diatom and cladoceran distributions respectively. The ordinations were performed using the CANOCO for Windows computer program. The environmental data (except pH, Total Organic Carbon and maximum altitude) were log transformed prior to analysis in order to reduce the effect of outlier samples and to stabilise the variance in the data.

Analogue matching was performed using the ANALOG computer program, kindly provided by Prof. John Birks, University of Bergen, Norway and ECRC, London, UK. The squared chord distance measure was used to determine the degree of similarity amongst the modern training set samples and between the fossil samples and the modern training set. The squared chord distance between samples *i* and *j* is given by

$$d_{ij} = \sum_k \left(p_{ik}^{1/2} - p_{jk}^{1/2}\right)^2$$

where d_{ij} is the squared chord distance (or dissimilarity coefficient, DC) between samples *i* and *j*, and is equal to the sum of differences in the proportion *p* of species *k* between *i* and *j*. A close analogue is a modern sample with a DC value between it and a fossil sample equal to or smaller than the extreme 10th percentile of the distribution of DC values between all samples in the modern training set (c.f. Birks, 1995).

Results

The results of the analogue matching are not reproduced here in detail. Rather, comparison between the present day measured water chemistry for the acidified lakes and that inferred from the selected modern analogues was made and these data are presented in Table 1. The inferred water chemistry and catchment data values show a degree of closeness to the mean and range of the measured water chemistry for a range of determinants, most significantly in calcium concentrations, and catchment morphology of the AWMN lakes. The inferred pH and alkalinity values are higher

Table 1: Modern analogue chemistry and catchment properties. (DC = Dissimilarity Coefficient, DC-1 = Inverse DC used as weights, WA = weighted averages, Alk1 = Alkalinity, Allab, Almon & Altot = Labile, Monomeric & Total Aluminium, Ca = Calcium, Cl = Chloride, Cond = Conductivity, EqAlk = Equivalent Alkalinity, K = Potassium, Mg = Magnesium, Na = Sodium, NO3 = Nitrate, SO4 = Sulphate and TOC = Total Organic Carbon)

Loch Coire nan Arr			μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ S l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	H^{+}	μ eq l^{-1}	mg l^{-1}
Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
CWEL	0.3948	2.5329	37.00	2.00	3.00	4.00	89.00	192.00	36.00	29.00	7.00	46.00	175.00	2.74	6.35	80.00	1.30
ARR	0.4067	2.4588	55.17	2.00	24.00	8.67	43.33	219.17	39.17	49.06	13.33	59.17	216.17	3.00	6.30	38.17	2.17
LNEI	0.4279	2.3370	60.30	0.00	8.00	8.00	38.00	303.00	17.00	41.00	8.00	55.00	262.00	2.00	5.74	41.00	3.40
CLAI	0.4300	2.3256	60.30	2.00	11.00	13.00	37.00	245.50	33.00	34.00	7.50	34.00	193.50	0.00	6.13	31.00	2.20
LAI	0.4408	2.2686	20.75	2.67	21.67	18.25	41.25	130.00	24.75	12.81	6.25	31.00	132.00	2.75	5.79	45.50	3.10
WA		11.9229	46.77	1.74	13.42	10.21	50.36	218.00	30.20	33.38	8.46	45.29	195.97	2.12	6.07	47.61	2.41
Loch Coire nan Arr		Mean		3.10	13.70		42.50	257.70	39.20	37.80	8.50	60.80	232.20	2.90	6.39	40.80	2.20
Measured		Max		7.00	33.00		70.00	664.80	85.00	89.00	15.40	158.30	495.70	7.90	6.95	56.30	5.20
		Min		<2.5	<2.5		17.50	123.90	21.00	4.00	2.60	25.00	130.40	<1.4	5.75	27.10	<0.1

Blue Lough

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
CADH	0.4299	2.3261	6.50	3.50	16.00	19.50	49.00	354.20	52.00	-2.00	9.00	64.00	288.00	0.00	5.14	55.50	5.35
ACH	0.4522	2.2114	12.25	3.00	14.00	6.60	41.80	198.40	36.40	4.33	7.00	32.60	210.40	1.25	5.24	36.60	2.75
WA		4.5375	9.30	3.26	15.03	13.21	45.49	278.27	44.40	1.08	8.03	48.70	250.18	0.61	5.19	46.29	4.08
Blue Lough		Mean		286.90	90.20		40.00	275.80	55.90	-22.80	12.80	60.00	257.00	28.60	4.69	94.80	3.50
Measured		Max		470.00	394.00		98.00	400.00	73.00	-4.00	25.40	91.70	369.60	72.90	5.11	118.80	6.80
		Min		72.00	18.00		16.50	152.10	36.00	-33.00	7.70	33.30	121.70	10.00	4.51	35.40	1.40

Burnmoor Tarn

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
No analogues within the 10 percentile cut-off of the distribution of DC values between samples in the modern training set																	
MEO4B	0.4625	2.1622	17	0	4	4	86	148	32	7	6	61	143	3.06	5.8	71	0.8
Burnmoor Tarn		Mean		3.00	8.10		89.00	209.90	41.60	47.80	8.50	65.00	192.20	5.00	6.48	81.50	2.00
Measured		Max		14.00	42.00		118.50	287.30	54.00	96.00	15.40	83.30	243.50	12.90	6.99	118.80	4.70
		Min		<2.5	<2.5		32.00	129.60	23.00	-44.00	2.60	33.30	126.10	<1.4	4.38	56.30	0.90

Llyn Llagi

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
ARR	0.4091	2.4444	55.2	2	24	8.67	43.33	219.2	39.17	49.06	13.3	59.17	216.2	3	6.3	38.2	2.17
LOSG	0.4354	2.2967	18.5	3.5	30	33.5	44.5	146.5	25	3	5	30.5	122	0	5.5	21	6.6
CLYD	0.4581	2.1829	62.4	7.33	8.33	16.25	55.9	134	29.25	33.93	5.31	48	123.9	7.63	6.2	65.8	0.48
WA		6.9241	45.30	4.18	21.05	19.30	47.68	168.21	31.34	29.01	8.04	46.14	155.84	3.46	6.00	41.17	3.11
Llyn Llagi		Mean		39.70	35.60		52.50	193.80	31.20	5.60	6.20	46.70	168.70	10.00	5.34	61.00	2.40
Measured		Max		159.00	80.00		94.00	377.50	58.00	6.30	19.20	75.00	291.30	38.60	6.30	81.30	5.50
		Min		<2.5	<2.5		31.00	98.60	13.00	4.78	2.60	25.00	100.00	2.10	4.78	39.60	<0.10

Lochnagar

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
LOCH	0.3523	2.8385	60.3	0	5	5	22.5	260	34.5	15.5	10	39	204.5	0	6	35.5	1.05
GLAS	0.3919	2.5517	61.9	10.33	1.67	13	71.25	126.9	28.25	33.45	7.8	43.35	130.1	7.63	6.3	57.3	0.5
CLYD	0.3930	2.5445	62.4	7.33	8.33	16.25	55.9	134	29.25	33.93	5.31	48	123.9	7.63	6.2	65.8	0.48
WA		7.9347	61.5	5.673	4.997	11.18	48.89	176.8	30.81	27.183	7.79	43.29	154.7	4.9	6.1	52.2	0.69
Lochnagar		Mean		25.50	16.50		29.00	89.30	21.80	0.60	7.40	33.30	93.90	15.70	5.33	57.70	1.10
Measured		Max		137.00	41.00		50.00	166.20	35.00	12.00	12.80	58.30	173.90	30.70	5.81	85.40	3.40
		Min		< 2.5	< 2.5		21.50	50.70	4.00	-10.00	2.60	25.00	69.60	< 1.4	4.95	45.80	0.20

Table 1: Modern analogue chemistry and catchment properties. (DC = Dissimilarity Coefficient, DC-1 = Inverse DC used as weights, WA = weighted averages, Alk1 = Alkalinity, Allab, Almon & Altot = Labile, Monomeric & Total Aluminium, Ca = Calcium, Cl = Chloride, Cond = Conductivity, EqAlk = Equivalent Alkalinity, K = Potassium, Mg = Magnesium, Na = Sodium, NO3 = Nitrate, SO4 = Sulphate and TOC = Total Organic Carbon)

Llyn Cwm Mynach			μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ S l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	μ eq l^{-1}	H^{+}	μ eq l^{-1}	mg l^{-1}
Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
LOSG	0.2714	3.6846	18.5	3.5	30	33.5	44.5	146.5	25	3	5	30.5	122	0	5.5	21	6.6
FEOI	0.3621	2.7617	60.3	1.25	12.25	13.5	58.25	498.8	70	35.75	11.5	89.25	400	0	6.1	50.3	4.55
TINK	0.3624	2.7594	31.3	4.67	18	19	79.25	141.8	29.25	24.17	8	38.75	121	5	5.7	64.3	3.6
DUBH	0.3634	2.7518	60.3	2.25	9.25	11.5	59	741.3	96.5	9	13.3	112.5	585.8	0.25	5.6	75	3.25
NEUN	0.3762	2.6582	60.3	1	5.75	6.75	30.75	383.8	53.5	7.75	8.25	61.5	312.5	0	5.7	40.3	3.08
TEAN	0.3904	2.5615	20	14.52	22.75	0	110	1160	16.3	12.04	23	255	981	7	5.7	233	3.3
ARR	0.4088	2.4462	55.2	2	24	8.67	43.33	219.2	39.17	49.06	13.3	59.17	216.2	3	6.3	38.2	2.17
CLAI	0.4126	2.4237	60.30	2.00	11.00	13.00	37.00	245.50	33.00	34.00	7.50	34.00	193.50	0.00	6.13	31.00	2.20
ACH	0.4377	2.2847	12.25	3.00	14.00	6.60	41.80	198.40	36.40	4.33	7.00	32.60	210.40	1.25	5.24	36.60	2.75
DCAL	0.4380	2.2831	60.3	3.6	33.2	36.8	59.4	1006	129.8	-11	17.8	155.8	772.2	0	4.9	78.8	6.75
GLOY	0.4402	2.2717	25	7	14	21	88	232	44	15	7	75	217	0.48	5.9	88	2.8
LACH	0.4423	2.2609	32	17	38	55	110	155	33	32	7	50	139	1	5.9	71	5.8
CREI	0.4569	2.1887	60.3	15	16.25	31.25	70	1069	138.5	6.5	20.8	163.8	828.5	0.75	5.5	109	3.38
WA		33.3360	42.19	5.62	19.18	19.65	62.93	463.04	55.46	16.81	11.22	86.73	381.13	1.43	5.71	70.03	3.95

Llyn Cwm Mynach		Mean		65.20	52.30		70.00	304.20	46.00	4.60	5.60	63.30	268.30	10.00	5.37	86.00	2.60
Measured		Max		291.00	158.00		128.00	518.30	72.00	34.40	9.70	100.00	404.30	30.70	6.30	154.20	10.70
		Min		<2.5	<2.5		21.50	143.70	24.00	-21.00	2.60	33.30	173.90	2.10	4.70	58.30	<0.1

Round Loch of Glenhead

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
LNEI	0.3501	2.8563	60.30	0.00	8.00	8.00	38.00	303.00	17.00	41.00	8.00	55.00	262.00	2.00	5.74	41.00	3.40
DUBH	0.3521	2.8401	60.3	2.25	9.25	11.5	59	741.3	96.5	9	13.3	112.5	585.8	0.25	5.6	75	3.25
DOI	0.3612	2.7685	22.2	7	47.67	27.33	54	230.8	43.83	15.43	12.5	60.5	238.7	6.4	5.8	58.8	2.4
LACH	0.3677	2.7196	32	17	38	55	110	155	33	32	7	50	139	1	5.9	71	5.8
LAI	0.3706	2.6983	20.8	2.67	21.67	18.25	41.25	130	24.75	12.81	6.25	31	132	2.75	5.8	45.5	3.1
CLAI	0.3741	2.6731	60.30	2.00	11.00	13.00	37.00	245.50	33.00	34.00	7.50	34.00	193.50	0.00	6.13	31.00	2.20
HIR	0.4030	2.4814	14	7.3	18	25	74	182	35	5	6	65	171	63	5.6	77	3.1
ARR	0.4201	2.3804	55.2	2	24	8.67	43.33	219.2	39.17	49.06	13.3	59.17	216.2	3	6.3	38.2	2.17
TINK	0.4263	2.3458	31.3	4.67	18	19	79.25	141.8	29.25	24.17	8	38.75	121	5	5.7	64.3	3.6
LOCH	0.4307	2.3218	60.3	0	5	5	22.5	260	34.5	15.5	10	39	204.5	0	6	35.5	1.05
CLYD	0.4465	2.2396	62.4	7.33	8.33	16.25	55.9	134	29.25	33.93	5.31	48	123.9	7.63	6.2	65.8	0.48
CADH	0.4537	2.2041	6.50	3.50	16.00	19.50	49.00	354.20	52.00	-2.00	9.00	64.00	288.00	0.00	5.14	55.50	5.35
LOSG	0.4544	2.2007	18.5	3.5	30	33.5	44.5	146.5	25	3	5	30.5	122	0	5.5	21	6.6
WA		32.7298	39.22	4.59	19.80	20.06	54.72	255.15	38.27	21.32	8.63	53.59	220.06	6.92	5.80	52.63	3.26

Round Loch of Glenhead		Mean		60.20	34.90		33.00	195.20	36.70	-12.20	8.20	45.80	174.30	7.10	4.90	67.50	3.00
Measured		Max		111.00	70.00		42.00	298.60	49.00	6.00	12.80	66.70	247.80	24.30	5.21	114.60	5.00
		Min		9.00	16.00		25.00	121.10	28.00	-22.00	2.60	33.30	130.40	<1.4	4.72	45.80	1.60

Scoat Tarn

Sample Code	DC	DC-1	Alk1	Allab	Almon	Altot	Ca	Cl	Cond	EqAlk	K	Mg	Na	NO3	pH	SO4	TOC
EDNO	0.3222	3.1037	25.3	54.50	23.00	78.33	38.07	150.43	28.67	0.00	5.20	28.87	134.43	7.63	5.15	59.70	1.43

Scoat Tarn		Mean		108.90	12.60		32.50	185.60	35.10	-8.40	7.90	48.30	163.90	21.40	4.99	60.80	0.90
Measured		Max		293.80	76.00		48.00	326.80	49.00	6.00	15.40	75.00	265.20	47.90	5.23	72.90	2.70
		Min		2.50	3.00		23.00	118.30	24.00	-26.00	2.60	33.30	126.10	5.70	4.57	35.40	<0.1

Table 1: Continued. (Afforest = % of catchment afforested, Area:Dep = Lake Area to Depth ratio, CA:LA = Catchment Area to Lake Area ratio, CArea = Catchment Area, LAlt = Lake Altitude, LArea = Lake Area, MaxAlt = Maximum Altitude, MLDepth = Mean Lake Depth, & NCR = Net Catchment Relief)

Loch Coire nan Arr		%			ha	m	m	m	m	m
Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
CWEL		0.00	2.36	24.39	2073.00	150.00	85.00	1085.00	36.00	935.00
ARR		50.00	0.92	81.55	897.00	125.00	11.00	896.00	12.00	771.00
LNEI		0.00	0.88	7.04	197.00	355.00	28.00	933.00	32.00	578.00
CLAI		10.00	2.92	63.17	5243.00	101.00	83.00	925.00	28.40	824.00
LAI		0.00	12.13	24.60	11637.00	280.00	473.00	1116.00	39.00	836.00
WA		12.26	3.74	40.38	3900.84	200.20	132.00	990.92	29.35	790.72
Loch Coire nan Arr	Measured	1.00	0.97	77.30	897.00	125.00	11.60		12.00	771.00

Blue Lough

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
CADH		0.00	0.73	7.00	28.00	398.00	4.00	421.00	5.50	23.00
ACH		0.00	1.87	14.82	476.00	305.00	26.21	568.50	9.20	278.60
WA		0.00	1.29	10.81	246.34	352.68	14.82	492.89	7.30	147.57
Blue Lough	Measured	0.00	0.42	19.90	47.90	340.00	2.10		5.00	363.00

Burnmoor Tarn

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
No Analogues within the 10 percentile cut-off of the distribution of DC values between samples in the modern training set										
MEO4B		0.00	0.30	7.07	99.00	470.00	14.00	568.50	47.00	278.60
Burnmoor Tarn	Measured	0.00	1.85	9.40	226.00	252.00	24.00		13.00	350.00

Llyn Llagi

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
ARR		50.00	0.92	81.55	897.00	125.00	11.00	896.00	12.00	771.00
LOSG		0.00	3.00	11.22	101.00	237.00	9.00	510.00	3.00	273.00
CLYD		0.00	0.17	10.00	10.00	660.00	1.00	947.00	6.00	287.00
WA		17.65	1.37	35.66	353.32	330.82	7.18	784.04	7.12	453.22
Llyn Llagi	Measured	0.00	0.34	27.70	157.00	380.00	5.67		16.50	298.00

Lochnagar

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
LOCH		0.00	1.00	9.94	159.00	517.00	16.00	1062.00	19.50	545.00
GLAS		0.00	0.15	20.00	20.00	520.00	1.00	945.00	6.50	425.00
CLYD		0.00	0.17	10.00	10.00	660.00	1.00	947.00	6.00	287.00
WA		0.00	0.46	13.19	66.52	563.82	6.37	987.50	10.99	423.67
Lochnagar	Measured	0.00	0.38	9.37	91.90	785.00	9.80		26.00	370.00

Table 1: Continued. (Afforest = % of catchment afforested, Area:Dep = Lake Area to Depth ratio, CA:LA = Catchment Area to Lake Area ratio, CArea = Catchment Area, LAlt = Lake Altitude, LArea = Lake Area, MaxAlt = Maximum Altitude, MLDepth = Mean Lake Depth, & NCR = Net Catchment Relief)

Llyn Cwm Mynach		%			ha	m	m	m	m	m
Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
LOSG		0.00	3.00	11.22	101.00	237.00	9.00	510.00	3.00	273.00
FEOI		20.00	2.14	10.67	32.00	115.00	3.00	202.00	1.40	87.00
TINK		0.00	1.12	10.18	112.00	420.00	11.00	700.00	9.80	280.00
DUBH		0.00	0.67	13.33	160.00	88.00	12.00	420.00	18.00	332.00
NEUN		0.00	0.94	11.00	33.00	162.00	3.00	212.00	3.20	50.00
TEAN		5.00	0.43	3.33	30.00	25.00	9.00	47.00	21.00	22.00
ARR		50.00	0.92	81.55	897.00	125.00	11.00	896.00	12.00	771.00
CLAI		10.00	2.92	63.17	5243.00	101.00	83.00	925.00	28.40	824.00
ACH		0.00	1.87	14.82	476.00	305.00	26.21	568.50	9.20	278.60
DCAL		0.00	0.87	37.50	75.00	20.00	2.00	102.00	2.30	82.00
GLOY		0.00	0.42	21.00	63.00	380.00	3.00	568.50	7.20	278.60
LACH		0.00	1.12	28.18	310.00	285.00	11.00	765.00	9.80	480.00
CREI		0.00	0.40	4.75	19.00	94.00	4.00	162.00	10.00	68.00
WA		6.44	1.36	23.02	552.56	182.78	14.06	466.44	10.16	290.84
Llyn Cwm Mynach	Measured	55.00	0.54	25.90	152.50	285.00	5.90		11.00	395.00

Round Loch of Glenhead

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
LNEI		0.00	0.88	7.04	197.00	355.00	28.00	933.00	32.00	578.00
DUBH		0.00	0.67	13.33	160.00	88.00	12.00	420.00	18.00	332.00
DOI		40.00	3.15	62.06	3289.00	10.00	53.00	800.00	16.80	790.00
LACH		0.00	1.12	28.18	310.00	285.00	11.00	765.00	9.80	480.00
LAI		0.00	12.13	24.60	11637.00	280.00	473.00	1116.00	39.00	836.00
CLAI		10.00	2.92	63.17	5243.00	101.00	83.00	925.00	28.40	824.00
HIR		0.00	0.57	3.58	17.90	435.00	5.00	455.00	8.80	20.00
ARR		50.00	0.92	81.55	897.00	125.00	11.00	896.00	12.00	771.00
TINK		0.00	1.12	10.18	112.00	420.00	11.00	700.00	9.80	280.00
LOCH		0.00	1.00	9.94	159.00	517.00	16.00	1062.00	19.50	545.00
CLYD		0.00	0.17	10.00	10.00	660.00	1.00	947.00	6.00	287.00
CADH		0.00	0.73	7.00	28.00	398.00	4.00	421.00	5.50	23.00
LOSG		0.00	3.00	11.22	101.00	237.00	9.00	510.00	3.00	273.00
WA		7.84	2.26	26.10	1817.89	291.22	58.70	769.27	16.73	478.04
Round Loch of Glenhead	Measured	0.00	0.93	7.50	95.10	295.00	12.50		13.50	236.00

Scoat Tarn

Sample Code		Afforest	Area:Dep	CA:LA	CArea	LAlt	LArea	MaxAlt	MLDepth	NCR
EDNO		1.00	1.87	14.82	476.00	500.00	26.21	607.00	13.49	278.60
Scoat Tarn	Measured	0.00	0.26	18.20	95.00	602.00	5.20		20.00	239.00

than present day mean values and aluminium concentrations are inferred as being lower than present.

These results demonstrate that the method is performing with a greater degree of reliability than the original method that used diatoms as the sole biological group. It is worth noting that the AWMN lakes exhibit a wide range in the water chemistry values measured throughout the period of monitoring. The inferred water chemistry values are based on no more than 4, and often only 1, sample. That the inferred values fall within the observed range of water chemistry values for the AWMN lakes would suggest that the selected modern analogue lakes are indeed suitable modern analogues.

Constrained ordination of the cladoceran and diatom data with the environmental variables illustrates that different environmental variables are important in determining the species composition of the cladocera and the diatoms respectively. Forward selection procedures were performed to select the minimally adequate set of environmental variables that best described the species distributions over the 83 lake modern data set for both the cladocera and the diatoms. P-values were adjusted for multiple simultaneous tests by means of a Bonferroni type correction to the required P value, where P-required = P / test number (e.g. P-required for the second test = 0.05 / 2 = 0.025). The forward selection procedures indicate those environmental variables that explain independent and statistically significant (estimated by Monte Carlo permutation procedures) amounts of the variation in the diatom and cladoceran data respectively. These results are shown in Table 2.

pH, Total Organic Carbon (TOC), Maximum Lake Depth (MLDep), Maximum Altitude (Max Alt), Calcium (Ca), Monomeric Aluminium (Almon), and Net Catchment Relief (NCR) were the variables that best explained the diatom distributions across the modern training set. TOC, pH, MLDep, and Chloride (Cl) were the environmental variables that best explained the cladoceran distribution. Whilst there is similarity in these responses the relative magnitude of explanatory power of the variables differs between the diatoms and the cladocera. Furthermore, certain environmental variables are unique in explaining the distribution of the diatoms or the cladocera (Cl, Ca, Almon, and NCR).

Table 2: Results of the forward selection procedures, showing those environmental variables that account for a statistically significant amount of the variance in the diatom, cladocera and joint data sets

Cladocera

Variable	F	P-value	P-required	λ	%
TOC	4.553	0.001	0.05	0.053	5.3
pH	2.988	0.002	0.025	0.034	3.4
MLDepth	2.6687	0.002	0.017	0.030	3.0
Cl	2.276	0.012	0.013	0.025	2.5
Total				**0.142**	**14.20**

Diatoms

Variable	F	P-value	P-required	λ	%
pH	8.207	0.001	0.05	0.247	8.63
TOC	3.323	0.001	0.025	0.097	3.39
MLDepth	2.663	0.001	0.017	0.076	2.66
MaxAlt	2.388	0.001	0.013	0.067	2.34
Ca	2.316	0.001	0.010	0.064	2.24
Almon	2.165	0.001	0.008	0.059	2.06
Total				**0.610**	**21.32**

Cladocera and Diatoms

Variable	F	P-value	P-required	λ	%
pH	8.977	0.001	0.05	0.100	10.0
MLDepth	4.584	0.001	0.025	0.048	4.8
TOC	3.581	0.001	0.017	0.037	3.7
MaxAlt	2.710	0.001	0.013	0.027	2.7
Ca	2.602	0.001	0.010	0.026	2.6
Cl	1.924	0.004	0.008	0.019	1.9
K	1.784	0.004	0.007	0.017	1.7
CA:LA	1.845	0.004	0.006	0.018	1.8
Total				**0.292**	**29.20**

Table 3: Ordination results for the cladocera (RDA), diatoms (CCA), and joint (RDA) data sets, constrained by the environmental variables. λ = eigen value.

Cladocera

Axis	1	2	3	4	Total Variance
λ	0.070	0.034	0.028	0.010	1.000
Species Environment Correlation	0.686	0.707	0.633	0.453	
Cumulative % of Species data	7.0	10.4	13.2	14.2	
Cumulative % of Species Environment Correlation	49.3	73.2	92.7	100.0	
	Sum of all unconstrained λ				1.000
	Sum of all canonical λ				0.142

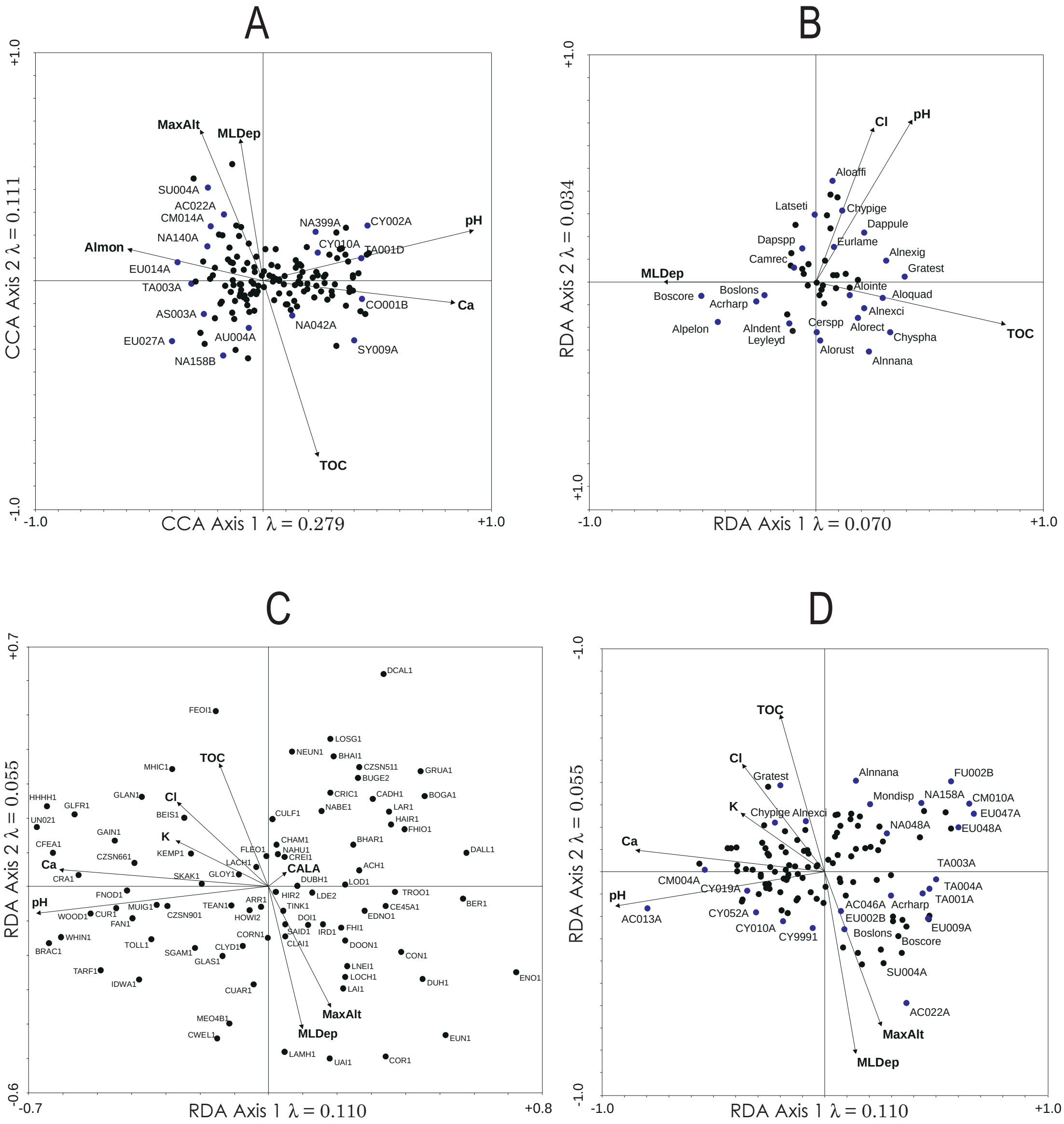
Diatoms

Axis	1	2	3	4	Total Variance
λ	0.279	0.111	0.086	0.059	2.682
Species Environment Correlation	0.922	0.848	0.777	0.752	
Cumulative % of Species data	10.4	14.5	17.7	19.9	
Cumulative % of Species Environment Correlation	45.8	63.9	77.9	87.6	
	Sum of all unconstrained λ				2.682
	Sum of all canonical λ				0.610

Cladocera and Diatoms

Axis	1	2	3	4	Total Variance
λ	0.110	0.055	0.041	0.032	1.000
Species Environment Correlation	0.921	0.848	0.782	0.773	
Cumulative % of Species data	11.0	16.5	20.5	23.8	
Cumulative % of Species Environment Correlation	37.7	56.5	70.4	81.5	
	Sum of all unconstrained λ				1.000
	Sum of all canonical λ				0.292

Figure 2: Biplots showing the results of the ordination analyses. A = joint plot for the diatom data set, B = species environment correlation biplot for the cladoceran data set, C = Euclidean distance biplot of the joint data set, D = species environment correlation biplot for the joint data set.



These results have been plotted in the form of ordination biplots and are shown in Figure 2. Figure 2A and Figure 2B show the species environment correlation biplot for the diatom and cladoceran data respectively. Here the ordination results have been scaled so that the relationships between the species and the environmental variables are emphasised. Only those environmental variables shown to be contributing a statistically significant and unique amount of explanatory power to the ordination models, as assessed through forward selection procedures, are shown.

In these biplots, the positions of the species are related to the environmental variables most influential in determining the species distribution. The environmental variables are depicted in the form of biplot arrows. The length of each arrow denotes the importance of that variable in explaining the species distribution. The angles between arrows reflect the degree of correlation between environmental variables. Small, acute angles indicate close, positive correlations between environmental variables. Conversely, large, obtuse angles indicate close, negative correlations between environmental variables. Biplot arrows that fall perpendicular to each other indicate zero correlation between the respective environmental variables. The angle between a biplot arrow and each axis is an indication of the degree of correlation between the environmental variable and that axis. The eigen values, a measure of the amount of total variance in the dataset explained by each axis, are shown together with the percentage of the total variance described by each axis.

Figure 2A shows the species environment biplot for the diatom species data. The first axis is primarily an acidity gradient contrasting species found in low pH and high aluminium environments on the left of the biplot with species found in high pH low aluminium conditions on the right. The second axis is a contrast between species found at high TOC concentrations at the bottom of the plot with those found at low TOC concentrations at the top. Furthermore, this second axis also contrasts between species found in deeper lakes at higher altitudes and those found primarily in shallow lakes at lower altitudes.

Figure 2B shows the species environment biplot for the cladoceran data. Here axis 1 represents the contrast between species found in deep, clear water lakes (low TOC) to the left of the diagram with species found in shallower lakes with high TOC concentrations to the right. The second axis is more closely related to the acidity gradient, contrasting those species located towards the top of the biplot associated with high pH values and those taxa located towards the bottom of the plot, which are generally associated with more acidic (lower pH) environments.

Figure 2C and Figure 2D show the main patterns of variation in the combined diatom and cladocera data. The ordination depicted in Figure 2C is scaled so as to best represent the similarity between sites based on the species composition of those sites. Only those taxa occurring in sites with abundance greater than 2% of the total for that site have been included in the ordination. Sites plotted close together have similar diatom and cladoceran assemblages whilst those positioned far apart will have few species in common. Figure 2C shows the primary response gradient to be one associated with acidity; lakes that fall to the left of the plot will be those with higher lake-water pH values in the data set, whilst those that fall on the right of the biplot will have low lake-water pH. The second axis contrasts between those deep, high altitude, clear water lakes that fall towards the bottom of the plot and the shallow lakes found at lower altitudes with high TOC concentrations at the top of the plot.

Figure 2D is a species environment biplot showing the results from the ordination of the combined diatom and cladoceran data, this time scaled to emphasise the relationships between the environmental variables and the species. This biplot reflects the patterns shown in the ordination biplots of the individual groups (diatom and cladocera) and also indicates that both diatom and cladoceran species responses are important across the dataset. Species that fall towards the edges of the biplot are better fitted in terms of their environmental niche and illustrate strong responses to the environmental variables. It is important to note that both diatom and cladoceran taxa fall towards the edges of the biplot and it is not just the diatom taxa that are highlighting the differences between sites shown in Figure 2C. This is an important point and, when taken in consideration with the other results presented here and the following discussion, points towards the reasons for the improvement in performance of the diatom and cladoceran approach over the diatom approach previously reported (Flower *et al.* 1997).

Discussion

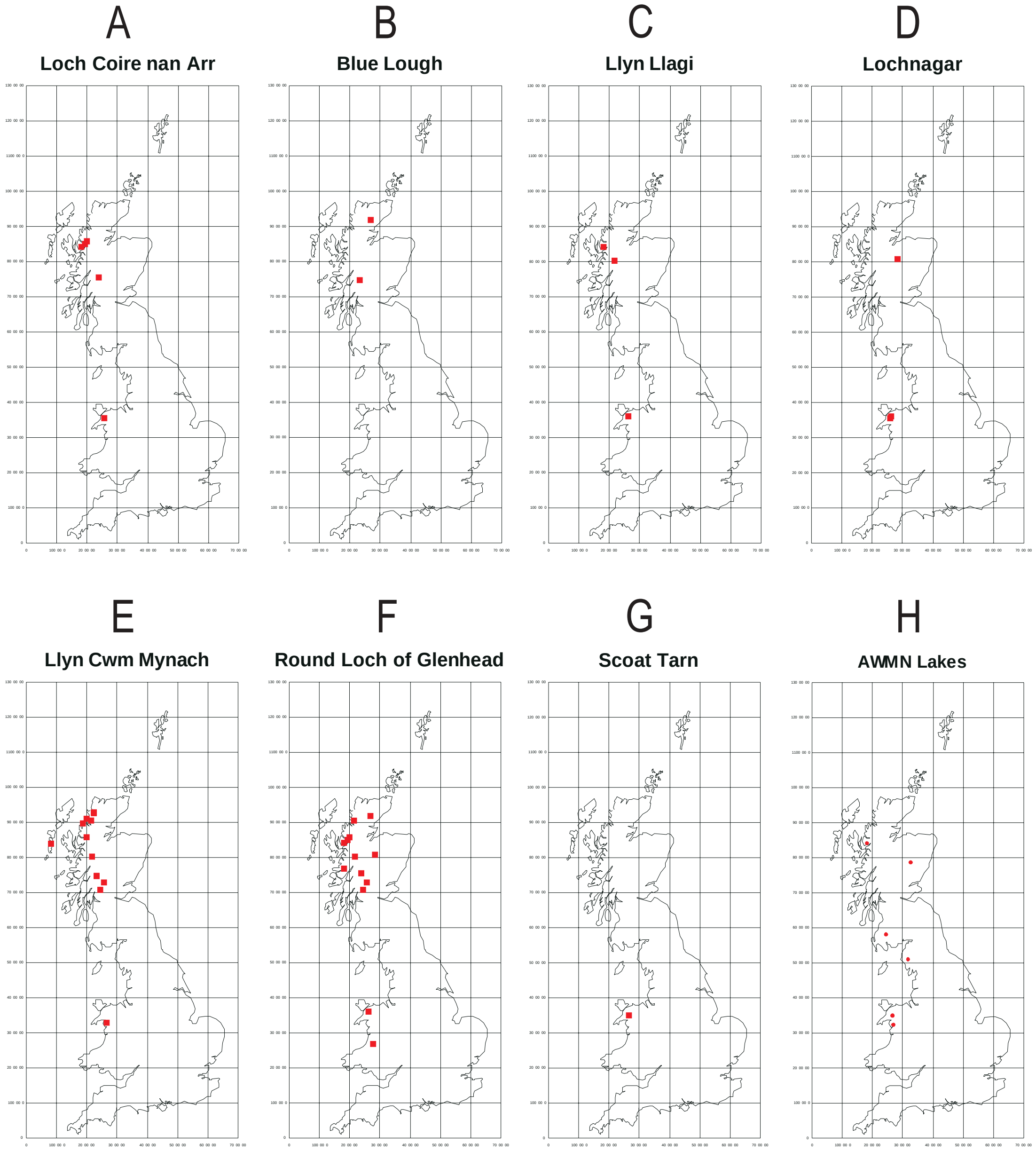
The results described above demonstrate that the improved analogue matching procedure is capable of identifying appropriate modern analogues, both in terms of the cladoceran and diatom assemblages and in measured water chemistry and catchment morphology. These modern analogues may be used to provide sensible reference conditions for acidified, oligotrophic upland surface waters in the UK.

The locations of the close modern analogues for the pre-acidification status of the selected AWMN lakes listed in Table 1 above are shown in Figure 3. The majority of the analogues are found in North and Northwest Scotland, an area of the UK receiving relatively low levels of sulphate deposition. This is encouraging because if any area of the UK were likely to contain modern analogues for acidified lakes it would be these clean areas of North and Northwest Scotland. A number of analogues are found in areas of Wales on bedrock with greater buffering capacity against acid deposition. These analogues are located in areas of relatively high deposition indicating that they may be inappropriate modern analogues in the sense of restoration targets. However these results suggest that the diatom and cladoceran communities supported by lakes located on low sensitivity geology are very similar to the types of communities supported by lakes on sensitive geology prior to acidification.

It is important to explain this improvement in performance and to describe the mechanisms by which the addition of the cladoceran species distribution data to the modern training set has improved the matching procedure. These mechanisms can be explained in both theoretical ecological terms and through empirical interpretation of the species response patterns described by the series of ordinations performed above.

Cladoceran distributions have been shown to be dependent on a range of environmental factors. Presence and abundance of aquatic macrophyte beds is a particularly important environmental factor. Quade (1969) showed that particular cladoceran species are associated with certain species of aquatic macrophytes. Whilst this may be an over simplification of a complex ecosystem, there is little doubt that aquatic macrophytes are important in providing exploitable habitats and refugia for cladocerans. Predation on cladocerans by zooplanktivorous fish is also reflected in the distribution patterns of cladocerans. Large bodied cladocerans are eaten in preference to smaller bodied cladocerans (Brooks & Dodson, 1965). Aquatic macrophytes can provide refugia from predation by zooplanktivorous fish for those cladocerans that can actively migrate from open water into the littoral zone. There is a clear interrelationship between aquatic macrophytes, fish, and cladoceran distribution patterns.

Figure 3: Maps showing the locations of the modern analogue lakes for each of the AWMN lakes (A-G), and the locations of the AWMN lakes used in this study (H).



Water chemistry also plays an important role in determining cladoceran distribution patterns. Cladoceran sub-fossil remains have been used to infer changes in pH using the pH preference of particular species groupings (Steinberg *et al.*, 1988). Whilst it has been shown by some workers that there is a relationship between cladocera and pH, other workers have had limited success with this approach. Clearly the cladoceran response to pH is a much more complex process than a simple response to increased acidity. Concomitant changes in other species groups (e.g. aquatic macrophytes, algae, invertebrates, and fish) during acidification will obviously complicate the cladoceran response to changes in acidity. Other workers have shown that the availability of nutrients is another important factor. For example *Chydorus sphaericus* is rarely found in waters of low ionic strength (Freyer 1993). *Chydorus piger*, in contrast, is regularly found in oligotrophic, upland waters.

These distribution patterns contrast with the clear and strong relationship between diatoms and pH. This is illustrated in the results shown in Figure 2A.

The empirical information on the species distribution patterns shown in the ordination biplots adds further information to the distribution patterns of the cladocerans and diatoms. It is clear that the measured environmental variables are influencing the distribution patterns of the two groups in different ways.

As has been shown in previous studies the main environmental variables that explain the distribution pattern of the diatom species are those associated with lake water acidity, in particular pH.

In the cladocera, this acidity gradient plays a less important role in determining cladoceran distributions. The extent of the pelagic area, altitude and water clarity are more important in determining the distribution of the cladoceran species than variables associated directly with acidity. This pattern can be explained through knowledge of the ecology of the cladocera. The greater the water depth of a lake, the greater the area of habitat available for exploitation by the planktonic species (e.g. *Bosmina coregoni* and *Bosmina longispina* amongst others). Shallower lakes may have a greater proportion of littoral area available as an exploitable habitat for the chydorid taxa (e.g. species from the genera *Alona*, *Alonella*, *Chydorus* and

Acroperus, amongst others). The altitude effect on the cladoceran species is most likely a complex gradient including temperature effects and constraints on dispersal. Certain species of cladocera are know to be associated with tolerance of cold conditions whilst others are restricted to more clement climatic conditions. Whilst there are no truly cold tolerant cladocera, certain species (e.g. *Acroperus harpae*) are able to tolerate cold climatic conditions are will therefore experience competitive advantages in these situations.

Water clarity is important to cladocera in many ways. Many fish and invertebrate predators rely upon visual clues to hunt for food. Clear water lakes provide little opportunity for those cladocera that are not capable of migrating into refugia to avoid predators. Waters with high TOC values will provide some degree of protection against predation for those taxa that do not actively avoid predation. Ultra violet radiation is particularly harmful to the cladocera, yet the degree of harm is species dependent (Rautio & Korhola, 2001). High TOC provides protection by attenuating harmful UV radiation. This has the effect of increasing the availability of exploitable habitats in shallow lakes that would otherwise be exposed to high levels of radiation.

To illustrate how the combined cladocera and diatom modern training set has improved the fit of the modern analogue lakes to the fossil assemblages, analogue matching procedures were run using cladocera alone, diatoms alone, and diatoms and cladocera together. The results are presented in Table 4 and Table 5 for Loch Coire nan Arr and the Round Loch of Glenhead respectively. These results show how the combined diatom and cladoceran approach filters the selected modern analogues so that those lakes identified as close modern analogues under the combined model are the lakes most likely to be similar to the conditions prior to acidification in Loch Coire nan Arr. and the Round Loch of Glenhead.

Table 4: Results for separate analogue matching procedures for the cladoceran, diatom and joint data sets for Loch Coire nan Arr, Scotland. Highlighted analogues are those identified using the joint data set. The samples are listed in order of their similarity to the fossil sample.

Fossil Sample	Cladocera Matches		Diatom Matches	Both
Loch Coire nan Arr 15-16 cm. sample	CADH	GLOY	**ARR**	**CWEL**
	FHIO	HOWI	**CLAI**	**ARR**
	TROO	TINK	DOI	**LNEI**
	DOON	HHHH	**LNEI**	**CLAI**
	LAI	LDE	SKAK	**LAI**
	HAIR	WHIN	**CWEL**	
	MEO4B	IDWA	CORN	
	LOD	CON	LACH	
	CRIC	LACH	BHAR	
	ACH	DALL	CFEA	
	FHI	CLYD		
	DUH	NEUN		
	GLAS	**CLAI**		
	WOOD	CREI		
	LAMH	DOI		
	CZSN90	DCAL		
	ENO	CZSN99		
	COR	LOSG		
	EDNO	BOGA		
	GRUA	GAIN		
	CWEL	**ARR**		
	LNEI	FLEO		

Table 5: Results for separate analogue matching procedures for the cladoceran, diatom and joint data sets for the Round Loch of Glenhead, Scotland. Highlighted analogues are those identified using the joint data set. The samples are listed in order of their similarity to the fossil sample.

Fossil Sample	Cladocera Matches		Diatom Matches	Both
Round Loch of Glenhead 29-30 cm. Sample	**CADH**	GAIN	CORN	**LNEI**
	SGAM	ACH	**DUBH**	**DUBH**
	FLEO	BOGA	CHAM	**DOI**
	EDNO	SKAK	**ARR**	**LACH**
	DOI	LAMH	**LNEI**	**LAI**
	LDE2	GLOY	CLAI	**CLAI**
	LNEI	CULF	**LACH**	**HIR2**
	HIR2	CRIC	BAHR	**ARR**
	IDWA	EUN	TEAN	**TINK**
	HAIR	DUH	FEOI	**LOCH**
	TARF	LOD	**CLYD**	**CLYD**
	CREI	CON	**LAI**	**CADH**
	GLAS	MUIG	**TINK**	**LOSG**
	LAI	BHAI	**LOSG**	
	LACH	TEAN	**LOCH**	
	CLAI	ENO	NEUN	
	MHIC	TROO	**DOI**	
	CZSN66	**LOSG**	LOD	
	FHIO	**CLYD**	**HIR2**	
	NAHU	**ARR**	LAMH	
	DUBH	CHAM		
	HOWI	NEUN		
	LOCH	GRUA		
	TINK	COR		
	IRD	CWEL		
	GAIN	FHI		
	IRD			

The other immediately apparent feature of these results is the high number of matches when the cladocera are used alone in the analogue matching procedure. This is due to the restricted cladoceran fauna found in oligotrophic upland surface waters. This restricted fauna is fairly cosmopolitan across the range of sites in the modern data set, yet the subtle differences in cladoceran community composition exert a great deal of influence on the modern analogues selected when using both the diatom and cladoceran data in the procedure.

The 83-lake modern training set contains appropriate modern analogues for strongly acidified lakes, such as the Round Loch of Glenhead and Llyn Llagi, as well as those lakes that have not acidified to the same degree, such as Loch Coire nan Arr, that are either better buffered on in areas of lower sulphur deposition. Also odd lakes (e.g. Blue Lough, which has been strongly acidic throughout its history) can be accommodated in the approach and suitable modern analogues can be found in the dataset.

Of the eight AWMN lakes that have been analogue matched for, the only lake without a suitable, close, modern analogue in the 83-lake modern training set is Burnmoor Tarn. This lake is much better buffered, has not acidified, and supports a very different assemblage of diatoms and cladocera. The modern training set contains very few, if any, lakes that are even remotely similar to Burnmoor Tarn, and here the concept of a modern analogue for a non-acidified lake becomes somewhat spurious.

Conclusions

The development of the analogue matching approach to include information on the cladoceran species composition alongside the diatom species composition has been shown to be a robust and reliable method for identifying reference conditions for acidified upland surface waters.

The approach has identified suitable modern analogues for 7 of the 8 studied AWMN lakes. The main criticism of the original approach was that modern analogues were very different in the calcium content of their lake-water than that of the acidified lakes. The results presented here show that the use of two proxies “filters” the selected analogues so that close modern analogues have similar chemical properties to the acidified lakes.

It is assumed that the cladoceran and diatom species assemblages are indicative of the wider biological structure and functioning of upland, oligotrophic, acid sensitive surface waters. Further work is required to assess the suitability of these modern analogues in terms of those biological groups that do not leave identifiable remains in the sediments of lakes (e.g. aquatic macrophytes and fish).

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Appendix 1

Site and Sample Codes referred to in the report and the corresponding Lake information

Sample	Site Code	Name	Country	Grid	Easting	Northing
ACH1	ACH	Loch na Achlaise	SCO	NN	310	480
ARR1	ARR	Loch Coire nan Arr	SCO	NG	808	422
BEIS1	BEIS	Loch na Beiste	SCO	NG	885	943
BER1	BER	Llyn Berwyn	CYM	SN	743	568
BHAI1	BHAI	Loch Coire a Bhaic	SCO	NC	247	295
BHAR1	BHAR	Loch Bharranch	SCO	NG	977	575
BOGA1	BOGA	Loch nam Badan Boga	SCO	NH	99	930
BRAC1	BRAC	Loch nam Brac	SCO	NC	179	480
BUGE2	BUGE	Llyn Bugeilyn	CYM	SN	822	923
CADH1	CADH	Loch Dubh Cadhafuaraich	SCO	NC	682	183
CE45A1	CE45A	Llynnoedd Ieuan	CYM	SN	794	812
CFEA1	CFEA	Loch na Claise Ferna	SCO	NC	201	468
CHAM1	CHAM	Loch a Cham Alltain	SCO	NC	283	446
CLAI1	CLAI	Loch Clair	SCO	NG	999	574
CLYD1	CLYD	Llyn Clyd	CYM	SH	635	597
CON1	CON	Llyn Conwy	CYM	SH	780	463
COR1	COR	Loch Coire an Lochan	SCO	NH	943	4
CORN1	CORN	Loch Bealach Cornaidh	SCO	NC	208	282
CRA1	CRA	Loch Craggie	SCO	NC	625	75
CREI1	CREI	Loch na Creige Duibhe	SCO	NC	5	118
CRIC1	CRIC	Loch na Cric	SCO	NC	166	37
CUAR1	CUAR	Loch na Cuaran	SCO	NC	292	238
CULF1	CULF	Loch Cul Fraioch	SCO	NC	25	330
CUR1	CUR	Loch na Curra	SCO	NG	823	800
CWEL1	CWEL	Llyn Cwellyn	CYM	SH	560	549
CZSN511	CZSN51	Llyn Llech Owain	CYM	SN	569	151
CZSN661	CZSN66	Llyn Eiddwen	CYM	SN	606	670
CZSN901	CZSN90	Llyn Fach	CYM	SN	905	370
DALL1	DALL	Loch Dallas	SCO	NJ	92	475
DCAL1	DCAL	Loch Dubh Camas an	SCO	NG	871	972
DOI1	DOI	Loch Doilet	SCO	NM	808	678
DOON1	DOON	Loch Doon	SCO	NX	495	985
DUBH1	DUBH	Lochan an Dubha	SCO	NC	147	55
DUH1	DUH	Dubh Loch	SCO	NO	238	828
EDNO1	EDNO	Llyn Edno	CYM	SH	663	497
ENO1	ENO	Loch Enoch	SCO	NX	445	851
EUN1	EUN	Loch nan Eun	SCO	NO	230	854
FAN1	FAN	Llyn y Fan Fawr	CYM	SN	831	216
FEOI1	FEOI	Lochan Feoir	SCO	NC	229	252
FHI1	FHI	Coire Fhionn Lochan	SCO	NR	902	459
FHIO1	FHIO	Lochan Fhionnlaidh	SCO	NC	191	103
FLEO1	FLEO	Loch Fleodach Coire	SCO	NC	275	248
FNOD1	FNOD	Llyn Fanod	CYM	SN	603	643
GAIN1	GAIN	Loch na Gaineimh	SCO	NC	765	304
GLAN1	GLAN	Llyn Glanmerin	CYM	SN	755	991

Sample	Site Code	Name	Country	Grid	Easting	Northing
GLAS1	GLAS	Llyn Glas	CYM	SH	601	547
GLFR1	GLFR	Llyn Glasfryn	CYM	SH	402	422
GLOY1	GLOY	Gloyw Llyn	CYM	SH	646	299
GRUA1	GRUA	Loch na Gruagaich	SCO	NC	243	158
HAIR1	HAIR	Loch na h-Airbhe	SCO	NH	103	924
HHHH1	HHHH	Un Named H	SCO	NO	653	909
HIR2	HIR	Llyn Hir	CYM	SN	789	675
HOWI2	HOWI	Loch Howie	SCO	NX	697	834
IDWA1	IDWA	Llyn Idwal	CYM	SH	645	596
IRD1	IRD	Llyn Irddyn	CYM	SH	630	220
KEMP1	KEMP	Loch Kemp	SCO	NH	612	323
LACH1	LACH	Lochan Lairig Cheile	SCO	NN	558	278
LAI1	LAI	Loch Laidon	SCO	NN	380	542
LAMH1	LAMH	Loch a Mhadaidh	SCO	NH	199	732
LAR1	LAR	Loch na Larach	SCO	NC	214	583
LDE2	LDE	Loch Dee	SCO	NX	470	790
LNEI1	LNEI	Loch Nan Eion	SCO	NG	925	508
LOCH1	LOCH	Loch Toll an Lochain	SCO	NH	832	74
LOD1	LOD	Lochan Dubh	SCO	NM	895	710
LOSG1	LOSG	Loch Bad an Losguiun	SCO	NH	158	38
MEO4B1	MEO4B	Llyn Cau	CYM	SH	716	125
MHIC1	MHIC	Loch Mich Leoid	SCO	NJ	8	347
MUIG1	MUIG	Loch Muigbhlaraidh	SCO	NH	635	830
NABE1	NABE	Loch na Beiste	SCO	NC	4	125
NAHU1	NAHU	Loch Bealach na h-Uidhe	SCO	NC	264	256
NEUN1	NEUN	Loch na Eun	SCO	NC	232	298
SAID1	SAID	Loch coire na Saidhe	SCO	NC	450	360
SGAM1	SGAM	Loch Sgamhaiu	SCO	NH	100	530
SKAK1	SKAK	Loch Skae	SCO	NX	710	837
TARF1	TARF	Loch Tarff	SCO	NH	425	100
TEAN1	TEAN	Loch Teanga	SCO	NF	818	383
TINK1	TINK	Loch Tinker	SCO	NN	445	68
TOLL1	TOLL	Loch Tollaidh	SCO	NG	841	785
TROO1	TROO	Loch Trool	SCO	NX	412	798
UAI1	UAI	Lochan Uaine	SCO	NO	1	981
UN021	UN02	Un-Named	SCO	NC	168	478
WHIN1	WHIN	Loch Whinyeon	SCO	NX	625	608
WOOD1	WOOD	Loch Woodhall	SCO	NX	673	675

Task 2.6 & 2.8: The CLAM biological - chemical database: the development and application of biological models to predict taxon distributions from SSWC and MAGIC hydrochemical models

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Summary

1. A combined database has been constructed using existing and new datasets to provide information on diatom, macrophyte and invertebrate distribution in relation to a range of water chemistry variables for 459 samples. This new database greatly expands the number of taxa and range of chemical gradients available in previous individual datasets. As such the new database captures taxon distributions more fully and allows modelling along the full range of taxon distribution along multiple chemical gradients.

2. Community ordination analysis using detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) indicates that the subset of chemical variables available for all samples (pH, alkalinity, conductivity, Ca, Cl, SO_4, and total monomeric aluminium) effectively account for the dominant biological gradients in the diatom and invertebrate datasets.

3. Logistic regression response models fitted to individual diatom and invertebrate taxa with 20 or more occurrences indicate that all 50 selected diatom taxa, and 31 of 39 selected invertebrate taxa show statistically significant responses to pH. These results are used to identify potential indicator taxa with varying degrees of tolerance to lowered pH. Although there is a statistically significant relationship between macrophyte community composition and pH, the macrophyte dataset is at present too small (28 sites) to allow the modelling of individual taxa.

4. Predictive models are derived for three biological targets, the diatom *Achnanthes minutissima*, the invertebrate *Baetis rhodani* and an acid tolerant diatom assemblage. *Achnanthes minutissima* and *Baetis rhodani* are both common in streams above pH 6.0, and are among the first taxa to decline in response to lowered pH. The acid tolerant diatom group is characterised by *Eunotia exigua*, an acidobiontic taxon commonly found in acidified waters.

5. The fitted logistic regression models can predict the occurrence of the target organisms with an internal success rate of c. 70%, and provide a means of predicting the probability of occurrence of the target organisms from modelled hindcasts and forecasts.

6. Application of the models to the CLAM national chemical database using the SSWC model provides biological predictions for baseline (pre-acidification), present, and future (Gothenburg scenario) conditions. Comparisons of the mapped biological predictions for baseline and present indicate substantial biological change in the areas of known acidification: SW England, Wales, Cumbria, Pennines and large parts of Scotland. The loss of sensitive taxa predicted for sites in the north of Scotland are probably the result of errors in the calculation of non-marine ANC for sites with high inputs of sea-salt. Comparison of predictions for baseline and future steady state ANC suggest virtually complete biological recovery under the Gothenburg scenario.

7. Biological predictions for five MAGIC 5 regions (Cairngorms, Galloway, Pennines, Wales, Lake District) show similar patterns of biological change between baseline and present, with loss of sensitive taxa from 18% of sites in the Cairngorms through to loss from 60% of sites in the Pennines. Predictions for recovery under the Gothenburg scenario indicate that only partial recovery will take place by 2050, and for some regions sensitive taxa are predicted only to return to approximately half of the sites inhabited prior to the onset of acidification.

8. The biological models provide a simple but robust way to convert hydrochemical hindcasts and forecasts into measures of biological change, and in particular, to assess biological damage under different emission reduction scenarios. However, there is a need to validate the models using independent field data, and a need to reconcile the differences in hydrochemical predictions between the SSWC model and MAGIC 5.

1. Introduction

The main goal of emission reduction is to promote the recovery of impacted ecosystems. For acidified surface waters, recovery is taken to mean the re-establishment of a healthy flora and fauna, or a return to the taxonomic structure and function of the pre-impacted system. To achieve this implies knowledge of (a) the biological status of pre-impacted waters, and (b) the chemical conditions necessary to promote a return to the pre-acidification biological targets. Furthermore, to assess the biological damage resulting from a particular level of critical load exceedance we also need (c) information on the relationship between the occurrence of a wide range of biological taxa and their key chemical determinants.

For lake sites palaeolimnological analyses can yield important information on the past occurrence of some biological groups (see Section 2.4, this report), but this technique is labour intensive and cannot be applied to large numbers of sites. For stream sites the taxonomic composition of analogue sites located in non-acidified regions can provide some clues to the pre-acidification biology of impacted sites, but chemical and physical differences between targets and analogues makes the selection of the latter difficult (see Section 2.5, this report). An alternative approach to addressing (a) and (b) above is to model the relationship between biology and chemistry in order to (i) predict the key components of the ecosystem under pre-impacted conditions, using hindcasts of pre-acidification chemistry, and (ii) measure the biological deviation from target conditions (i.e. damage) under different reduction scenarios (i.e. (c) above). These relationships could be generated by field or laboratory experiment, but in either case the generation of sufficient data to relate the large number of potential biological targets to the full range of water chemistry parameters would be a vast undertaking. An alternative method is to derive empirical statistical relationships between biological status and chemical conditions using field survey data collected from a range of sites spanning the appropriate chemical gradients (Ormerod 1994).

To this end, an important aspect of the current contract has been the development of a high quality chemical – biological database focused on diatoms, invertebrates and macrophytes, and on the use of this database to derive statistical models for predicting the occurrence of key taxa using measured and modelled chemical data. These groups of organisms are key structural components of freshwater ecosystems and are extremely sensitive to changes in acidity (Battarbee 1984; Farmer 1990; Lien *et al.* 1996; Ormerod & Edwards 1987). They are thus excellent “indicator” groups for assessing and modelling acidification status of freshwaters and for assessing biological damage as a result of critical load exceedance.

2. Database Development

2.1 Data Sources

The CLAM chemical – biological database has been assembled by merging existing datasets with new data collected during the current contract. All data are required to meet the AQC criteria concerning sampling techniques, chemical and biological analyses used in the UKAWMN (Patrick *et al.* 1991). After screening, six datasets with matching chemical, diatom and invertebrate data, and two datasets with matching chemical and macrophyte data, have been compiled and entered into the database. Table 1 lists the source datasets and their pH ranges.

All chemical and biological data are stored as a series of tables in a relational database managed using Microsoft Access2000 software. All biological data are stored using the analyst’s original determinations and taxonomic nomenclature. However, as different analysts frequently work to different taxonomic levels, and use different naming conventions for diatom and invertebrate data, a process of taxonomic harmonisation was carried out prior to merging the datasets (cf. Munro *et al.* 1991). This involved screening the individual taxon lists for synonyms and differing levels of taxonomic resolution between datasets. Taxa were then renamed, or grouped at a higher taxonomic level, to produce a single, harmonised taxon list for the combined dataset.

Table 1 Data sources for the CLAM biological – chemical database

Dataset	*Source*	*Sampling date*	*Site type*	*N*	*mean pH*	*Min pH*	*max pH*
Scot Base	Scottish Baseline Survey (Doughty 1989)	1986	Streams	144	6.70	4.42	8.49
UKAWMN	UKAWMN Survey Data Stream sites (Patrick et al. 1991)	1990-92	Streams	33	5.52	4.53	6.68
IF / OF	Inflow / outflow validation study (this contract)	1996	Streams	31	5.97	3.70	7.05
Sea Salt	Sea salt study (this contract)	1996	Streams	17	5.77	4.89	6.98
WAWS	Welsh Acid Waters Resurvey	1995	Streams	118	6.21	4.0	8.29
RSPB	RSPB survey (Allott & Rose 1993)	1992-94	Streams	88	6.54	5.02	8.46
CCW	CCW Macrophyte survey	1998	Lakes	17	5.88	4.73	6.71
UKAWMN 2	UKAWMN Survey Data Lake sites (macrophyte data)	1998	Lakes	11	5.44	4.61	6.48
Total				**459**	**6.35**	**3.7**	**8.49**

2.2 Dataset summary

The combined CLAG database contains a total of 459 biological samples with associated chemical data. Samples cover the range 3.7 to 8.5 pH units, although most fall between pH 5.0 and 7.0 (Figure 1).

Samples derive from all the main acidified areas of England, Scotland and Wales, and the database also includes material from soft-water sites in the low deposition areas of northern Scotland.

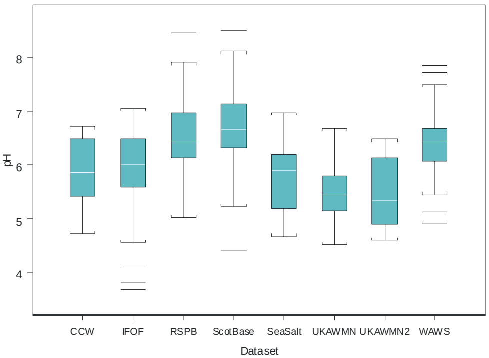


Figure 1: Mean annual pH by dataset

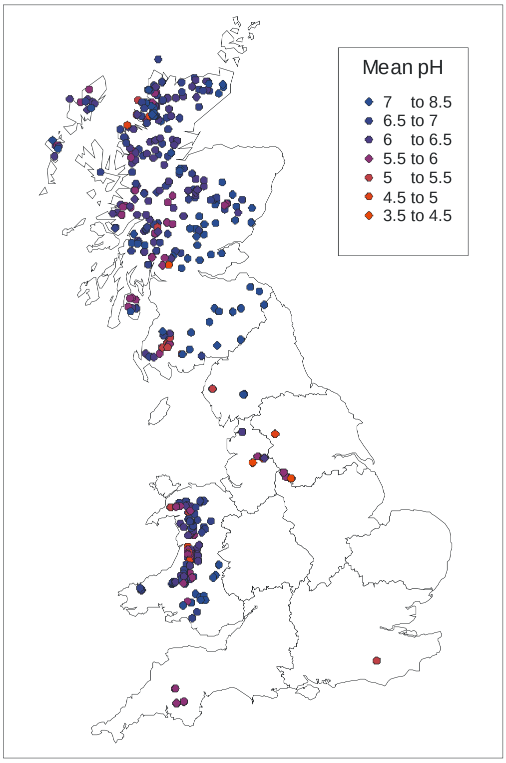


Figure 2: CLAM biological database site locations and pH

Table 2 lists the subset of chemical variables available for all samples in the database. Table 3 lists the total number of diatom, invertebrate and macrophyte taxa and samples, recorded after harmonisation. Many of these taxa occurred only rarely so rare forms (defined as taxa with less than two occurrences and a maximum relative abundance of less than 2% (diatoms) or 1% (invertebrates) in any single sample), or macrophyte taxa with a cover maximum score of less than 3, were deleted prior to statistical analyses.

Table 2 List of chemical variables available for all samples

Chemical parameters
pH
Alkalinity
Conductivity
Calcium
Chloride
Sulphate
Total monomeric aluminium

Table 3 Numbers of diatom and invertebrate taxa in the merged dataset

Biological Group	***Numbers of taxa***		***Numbers of samples***
	Total	*After deletion of rare taxa*	
Diatoms	320	158	452
Invertebrates	260	122	421
Macrophytes	28	21	28

Figure 3 shows frequency histograms summarising the distribution of the main chemical parameters. Figures show that by combining individual regional datasets the combined biological database now includes samples from a wide geographical range that span the main chemical gradients important in determining taxon distribution.

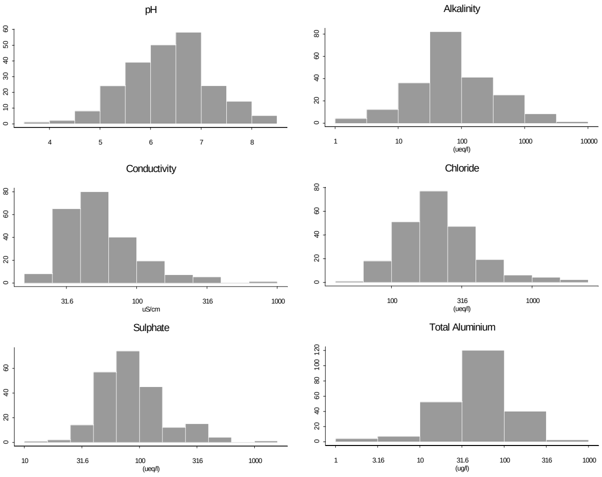


Figure 3 Frequency histograms summarising the distribution of the main chemical parameters

Biological organisms respond in complex ways to multiple chemical gradients. A first step in modelling these responses is to understand the relationships between different chemical variables. Figures 3 and 4 illustrate these relationships for the combined database. The principal component biplot shown in Figure 4 captures 75% of the total variation in the water chemistry variables and effectively display the main chemical gradients and inter-relationships in the dataset. Examining Figures 3 and 4, the following observations can be made:

1. As expected, pH and (log-transformed) alkalinity are highly positively correlated. It will be difficult or impossible to separate the effects of these variables on species' distributions using this dataset.

2. Total monomeric aluminium is only weakly negatively correlated with pH.

3. Sulphate is strongly correlated with conductivity, and both variables are weakly correlated to pH.

4. Calcium is strongly correlated to both pH and conductivity, especially at pH > 6.5.

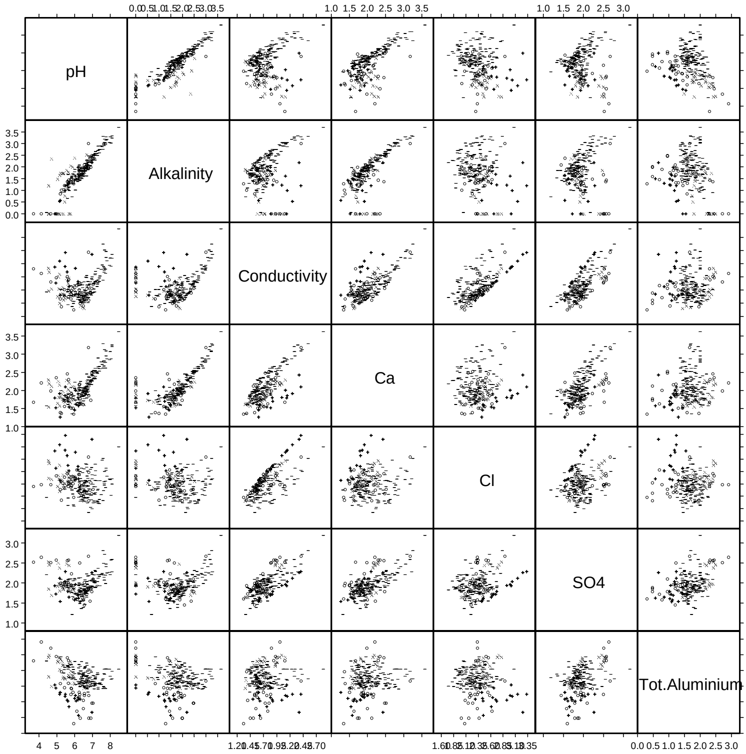


Figure 4 Scatterplot matrix show relationships between chemical variables

3. Modelling species – chemistry relationships

This section presents results of analyses of diatom, macrophyte and invertebrate community and species' response to chemical gradients. In developing predictive models of species responses we require that the biological data be related in a statistically significant way to the chemical or physical predictor variables of interest. A first step in model development is therefore to assess (1) which chemical variables are important in determining taxon distributions and so should be included in any predictive model, and (2) which taxa show a statistically significant relationship to stream (diatoms and invertebrates) or lake (macrophytes) hydrochemistry.

All chemical variables except pH were $\log_{10}$ transformed prior to analysis as they showed highly skewed distributions. Abundances of diatom and invertebrate taxa were expressed as relative proportions to the total assemblage.

3.1 How well do the chemical variables explain the distributions and abundance of diatom and invertebrate communities?

In developing predictive models of species' response along chemical gradients we assume that we have measured and included the main chemical variables important in influencing species' distributions. This assumption can be tested using the multivariate statistical techniques of detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) (Jongman *et al*. 1995). DCA is an indirect ordination method that quantifies the variance in species data accounted for by the dominant biological gradients according to a model of unimodal species distribution along environmental gradients. CCA is a direct ordination method that quantifies the variance in species data accounted for by biological gradients that are linear combinations of environmental (chemical) variables. Comparing the eigenvalues of a DCA and CCA gives an assessment of the degree to which the supplied chemical variables account for the dominant biological gradients in the species datasets. The number of significant environmental variables included in the CCA was determined using forward selection (see section 3.2).

Table 4 DCA and CCA eigenvalues and gradient lengths for diatom and invertebrate datasets

Dataset		**DCA**			**CCA**	
	Axis number	***Eigenvalue***	***%***	***Gradient length***	***Eigenvalue***	***%***
Diatoms	1	0.735	9.1	5.96	0.518	6.4
	2	0.496	6.2	4.35	0.259	3.2
	3	0.341	4.2	3.21	0.111	1.4
Macrophytes	1	0.248	18.9	3.27	0.149	11.4
	2	0.171	12.0	1.93	0.096	7.3
	3	0.093	5.1	2.18	N/A	N/A
Invertebrates	1	0.493	8.6	4.11	0.278	4.8
	2	0.357	6.2	3.70	0.161	2.8
	3	0.280	4.9	3.12	0.083	1.5

Table 4 lists the eigenvalues and gradient lengths for the three datasets for DCA and CCA. For diatoms, the main floristic gradient of the DCA (axis 1) accounts for 9.1% of the total variation in the diatom data. This variance explained is much lower than that obtained in ordinations of, for example, chemical data and is common for noisy, species rich, biological datasets. The equivalent figure for the CCA is 6.4%, indicating that the supplied chemical data accounts for about 70% of the total explainable variation of the main floristic gradient. For the second and third axes the chemical data only explains c. 50% and 33% of the explainable variation respectively. This pattern is similar for the macrophyte and invertebrate datasets. For the macrophytes the chemical data account for c. 60% of the explainable variation on the first axis, but only 56% for the second axis. For the invertebrate data the chemical data account for c. 55% of the explainable variation on the first axis, but only 45% and 31% for the second and third axes respectively. Thus, for all three datasets we conclude that the chemical data adequately accounts for the dominant biological gradient, but that there are secondary biological gradients that are poorly accounted for, in a statistical sense, by the measured chemical variables.

Diatoms and macrophytes show large gradient lengths (> 4 SD units), indicating a complete turnover in species along the gradients represented by the first two ordination axes. The shorter gradient (3.3 SD units) for the macrophyte data is not surprising given the much smaller size of this dataset.

3.2 Which chemical variables are significantly related to diatom and invertebrate community distribution?

Chemical and environmental datasets often contain redundant information in the form of (1) variables that show no significant relationship to biological data, and (2) variables that are highly correlated with other chemical variables. CCA with forward selection was therefore used to identify such redundancies and to identify a subset of variables that are significantly related to diatom and invertebrate community distribution. The significance of each variable for inclusion in the model was assessed using a Monte-Carlo permutation test, 999 permutations.

Table 5 Results of variable selection in forward selection CCA

	Diatoms		*Macrophytes*		*Invertebrates*	
Order of selection	***Variable***	***p-value***	***Variable***	***p-value***	***Variable***	***p-value***
1	pH	0.001	pH	0.001	pH	0.001
2	Tot. Mon. Al	0.001	Calcium	0.009	Tot. Mon. Al	0.001
3	Conductivity	0.001			Conductivity	0.001
4	Chloride	0.001			Sulphate	0.001
5	Alkalinity	0.001			Calcium	0.001
6					Alkalinity	0.001

Table 5 lists the results of the forward selection CCA analyses. Five (diatoms), three (macrophyte) and six (invertebrates) chemical variables make significant contributions in explaining the variance of the biological data. As expected, pH is the most important chemical variable but interestingly, pH, total monomeric aluminium and conductivity are selected in the same order for both the diatom and invertebrate datasets, highlighting importance of these variables in explaining both diatom and invertebrate distributions (e.g. Birks *et al*. 1991; Ormerod *et al*. 1988). That only two variables are significant in explaining the distribution and abundance of the macrophyte communities is again not surprising given the small size of this dataset.

Table 6 lists results of a series of CCAs and partial CCAs conducted to partition the total variance in the diatom and invertebrate datasets into components represented by the three key chemical variables identified above: namely pH, total monomeric aluminium and conductivity. The small size of the macrophyte dataset precludes this analysis. The total variance explained is the fraction of the total variance in the diatom and invertebrate datasets that is explained, in a statistical sense, by that variable alone. The unique variance explained is that fraction of the total variance explained by a variable, when the effects of interactions with other correlated variables have been removed, or partialled out. P-values for the unique contribution are also listed and all are highly significant. Clearly, if we propose to model species response to a chemical variable or gradient then we should require that that variable make a significant ***unique*** contribution in explaining the variance in the species data. The gradient length lists the length of each chemical gradient in species SD units, and gives a measure of biological turnover, or beta diversity, along that gradient.

Table 6 Results of variance partitioning CCA (see text for explanation)

Dataset	***Variable***	***Gradient length***	***Variance explained***		
			Total	***Unique***	***p-value***
Diatoms	pH	4.05	6.2%	3.4%	0.001
	Tot. Mon. Al.	2.45	2.8%	1.8%	0.001
	Conductivity	2.18	2.3%	1.8%	0.001
Invertebrates	pH	2.58	3.6%	2.6%	0.001
	Tot. Mon. Al.	1.70	1.8%	1.8%	0.001
	Conductivity	1.77	1.7%	0.9%	0.001

Values for total variance explained range from 6.2% to 1.7% and are relatively low, but as noted above, are typical of noisy, species rich datasets. All explained total variances are highly significant (p = 0.001) as assessed using a permutation test. The unique variances account for up to 50-100% of the total variance and are also all highly significant. Thus we conclude that pH, total monomeric aluminium and conductivity all make significant unique contributions to explaining the variance in the biological datasets, and the attempts at deriving predictive models relating the distribution of individual taxa to these variables are justified.

The gradient length of pH is much larger than the other two variables, highlighting the overall importance of this variable in influencing species distribution.

3.3 Which diatom, macrophyte and invertebrate taxa show statistically significant relationships to stream water acidity?

In Section 3.2 pH, total monomeric aluminium and conductivity were highlighted as important chemical variables that are related to diatom and invertebrate distributions, and that make significant and unique contributions to explaining the variance in the biological datasets. The next stage in developing predictive models of species responses is to identify which taxa show statistically significant relationships with these variables. In this section we use Huisman-Olff-Fresco (HOF) hierarchical models for testing and quantifying taxon response to pH.

Previous attempts at modelling species responses along environmental gradients have used logistic regression to fit linear monotonic or symmetric Gaussian response functions (e.g. ter Braak & Looman 1986; Juggins *et al*. 1995). However, many taxa showed skewed, non-symmetric responses, and Huisman *et al*. (1993) proposed a set of models that add a skewed response to a hierarchy that also includes monotonic and symmetric responses (Oksanen 1997). Thus, it is possible to fit a hierarchy of models to the taxon distribution using a logistic regression framework, and to test whether (1) a taxon has a significant relationship with the chemical variable, and (2) the nature of that relationship: sigmoid (monotonic), symmetric, or skewed.

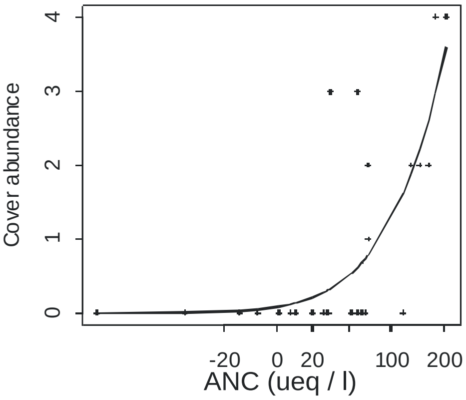


Figure 6 Response of *Nitella* spp. to ANC

HOF models were fitted to all taxa macrophyte taxa and to all diatom and invertebrate taxa with 20 or more occurrences. One macrophyte taxon, *Nitella* spp., showed a significant response to pH. This genus is well-know as being acid-sensitive, and is rarely found below 50 µeq l^{-1} ANC (see Figure 6). The lack of significant fit to other taxa is probably a function of the small size of this dataset. Existing ecological data and visual inspection of the species response plots suggest that lake water acidity is a major determinant controlling the distribution of many macrophyte taxa. However, it is not possible at present to derive statistically significant relationships using this dataset. Results from the diatoms and invertebrate datasets suggest that it may be possible to quantify these relations with a bigger dataset.

Results for diatoms and invertebrates are listed in Appendices 1 and 2 respectively. Table 7 presents a summary of these analyses and lists the number of taxa showing significant monotonic, symmetric and skewed responses, or no significant response to pH. The distributions of selected taxa along the pH gradient, together with their modelled responses are shown in Appendices 3 and 4.

Table 7 Number of taxa showing different responses to pH

Response	***Diatoms (N = 50)***	***Invertebrates (N = 39)***
No significant response	0	8
Monotonic	4	9
Symmetric	33	16
Skewed	13	6

The scatter around the fitted response curves display the “noise” common in such datasets. However, results show that many taxa exhibit significant responses to pH, and despite this noise it is possible to fit a variety of response curves to these taxon distributions.

Table 7 shows that, surprisingly, all diatom taxa exhibit significant response to pH, with the majority of common diatom taxa (defined as those present in 20 or more samples) having unimodal responses and a clear optimum between pH 4 and 8. Using the data given in Appendices 1 and 3 it is possible to identify a sequence of potential indicator taxa having increasing tolerance to lowered pH (Table 8).

Table 8 List of potential diatom indicator taxa

Rarely found below pH 7	G. olivaceoides, G. clevei and C. sinuata
Rarely found below pH 6.5	A. lanceolata, H. arcus, C. ventricosa
Rarely found below pH 6	G. parvulum, S. acus v. angustissima
Rarely found below pH 5.5	A. minutissima, B. virea, E. pectinalis v minor fo impressa
Rarely found below pH 5	B brebissonii, E. incisa, E. naegelii, P fibula, T flocculosa
Common below pH 5	E. exigua

Results for the invertebrate dataset show that 8 taxa exhibit no significant response to pH, and only 22 show a symmetric or skewed unimodal response. However, examining the pattern of responses displayed in Appendix 4, a list of potential invertebrate indicator taxa can also be identified (Table 9). The lower proportion of unimodal distributions exhibited by invertebrate taxa, as compared to diatoms, is probably a function of the smaller turnover along the pH gradient of this group (see section 3.2).

Table 9 List of potential invertebrate indicator taxa

Rarely found below pH 7	G. pulex
Rarely found below pH 6	Baetis spp., D. cephalotes, P. bipuctata
Rarely found below pH 5.5	Rhithrogena spp., P. flavomaculatus
Rarely found below pH 5	B. risi
Common below pH 5	Nemoura spp., L. hippopus

Tables 8 and 9 identify a range of potential indicator taxa that show statistically significant responses to pH, together with a qualitative description of their distribution. The fitted responses also provide a quantitative description of taxon distribution that can be used to predict the probability of occurrence, or relative abundance of each taxon at particular chemical values.

3.4 Classification and prediction of diatom and invertebrate communities

In addition to modelling the response of individual taxa to chemical gradients, it is also possible to model the occurrence of biological assemblages or communities. To achieve this the diatom and invertebrate datasets have been classified using two-way indicator species analysis (TWINSPAN; Hill, 1979). Results of this cluster analysis allowed the identification of a total of five diatom and five invertebrate assemblage groups. Tables 10 and 11 summarise each group according to their dominant taxa.

Table 10 Description of TWINSPAN diatom groups according to dominant taxa

Group No.	Dominant Taxa
1	*Eunotia exigua, Achnanthes austriaca var. helvetica*
2	*Eunotia vanheurckii var. intermedia, Eunotia exigua, Peronia fibula*
3	*Tabellaria flocculosa, Eunotia exigua, Achnanthes minutissima*
4	*Achnanthes minutissima, Diatoma hyemale*
5	*Achnanthes minutissima, Cocconeis placentula, Gomphonema spp.*

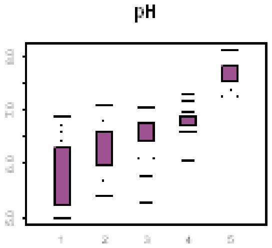
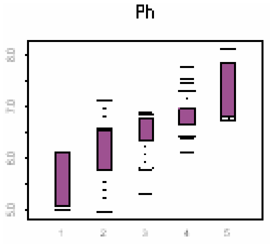


Table 11 Description of TWINSPAN invertebrate groups according to dominant taxa

Group No.	Dominant Taxa
1	*Amphinemura sulcicollis, Leuctra nigra*
2	*Leuctra inermis, SIMIULIIDAE, Amphinemura sulcicollis*
3	*Isoperla grammatica, Leuctra inermis, CHIRONOMIDAE*
4	*Baetis rhodani, Rithrogena semicolorata, Leuctra inermis*
5	*Rithrogena semicolorata, Baetis rhodani, Gammarus pulex*



The strong relationship between diatom and invertebrate communities and pH shown in the boxplots can be modelled in the same way as for individual taxa: in this case it is the probability of occurrence of a particular assemblage group, or community, that is modelled. Figure 7 shows the fitted response curves for the five diatom and five invertebrate assemblage groups. All have statistically significant relationships with pH ($p < 0.01$).

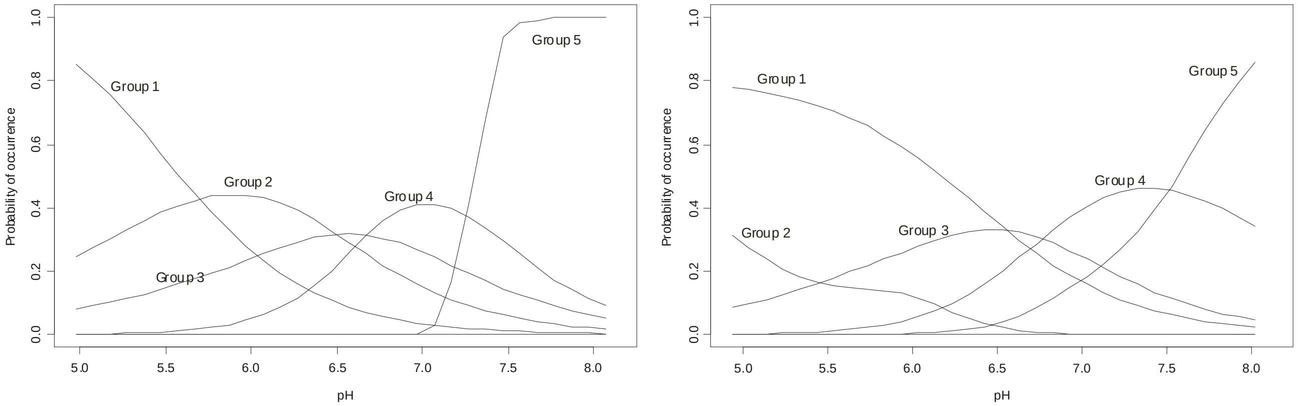


Figure 7 Probability of occurrence of diatom (left) and invertebrate (right) assemblages as a function of stream water pH

3.5 *Predictive models for key diatom and invertebrate taxa and assemblages*

The main aim of the species - chemistry modelling is to develop models that allow us to predict the likely biological status of pristine, unimpacted waters, and to measure the amount of change or damage, in biological terms, that has occurred since the onset of acidification, and that will remain under different emission reduction scenarios. The inputs of such models are the chemical predictions from steady-state or dynamical hydro-chemical models that attempt to hindcast or forecast site chemistry under different boundary conditions. There is therefore a pragmatic requirement that the biological model be derived for the same chemical parameters that are output from the hydro-chemical models.

In the above analysis pH was used as a simple measure of acidity because it can best account for the variation observed in the biological datasets. However, both the steady state and dynamic models are formulated to predict ANC. ANC and pH are usually strongly correlated, and although both variables are uniquely significant in explaining variation in both diatom and invertebrate datasets, the two variables are confounded, and it is difficult to separate their causal effects in these datasets. Given the strong correlation between ANC and pH, the evidence that ANC is significantly related to biological distributions after the effects of pH have been accounted for, and the need to link biological and hydrochemical models, subsequent predictive models have been developed with ANC, rather than pH as an driving variable.

In Section 3.3 and 3.4 it was shown that a large number of individual diatom and invertebrate taxa, and assemblage types, have significant relationships with stream water acidity: predictive models could be developed for each of these taxa. However, the aim of the present study is to develop models that can be applied to a large number of sites for a synoptic study, the results of which should be easy to map and visualise. To this end we have developed predictive models for just two taxa and one assemblage:

1. Occurrence of the diatom *Achnanthes minutissima*.
2. Occurrence of the macroinvertebrate *Baetis rhodani*.
3. Occurrence of a diatom assemblage characteristic of acidified waters (Diatom group 1 in Table 1).

Achnanthes minutissima and *Baetis rhodani* are both common in streams above pH 6.0, and are among the first taxa to decline in response to lowered pH (e.g. Battarbee *et al*. 1988; Raddum *et al*. 1994). Diatom Group 1 is characterised by *Eunotia exigua*, an acidobiontic taxon commonly found in acidified waters. Taken together these indicators provide a general indication of ecosystem health, and are thus important and appropriate for use in assessing the overall biological status of soft waters.

Figure 8 shows the distribution of these two taxa and two assemblage groups in relation to ANC. The modelled response curves, derived using logistic regression, are also shown. The regression coefficients from the fitted models can be used to predict the occurrence of a taxon or assemblage group, given any particular measured or modelled value of ANC.

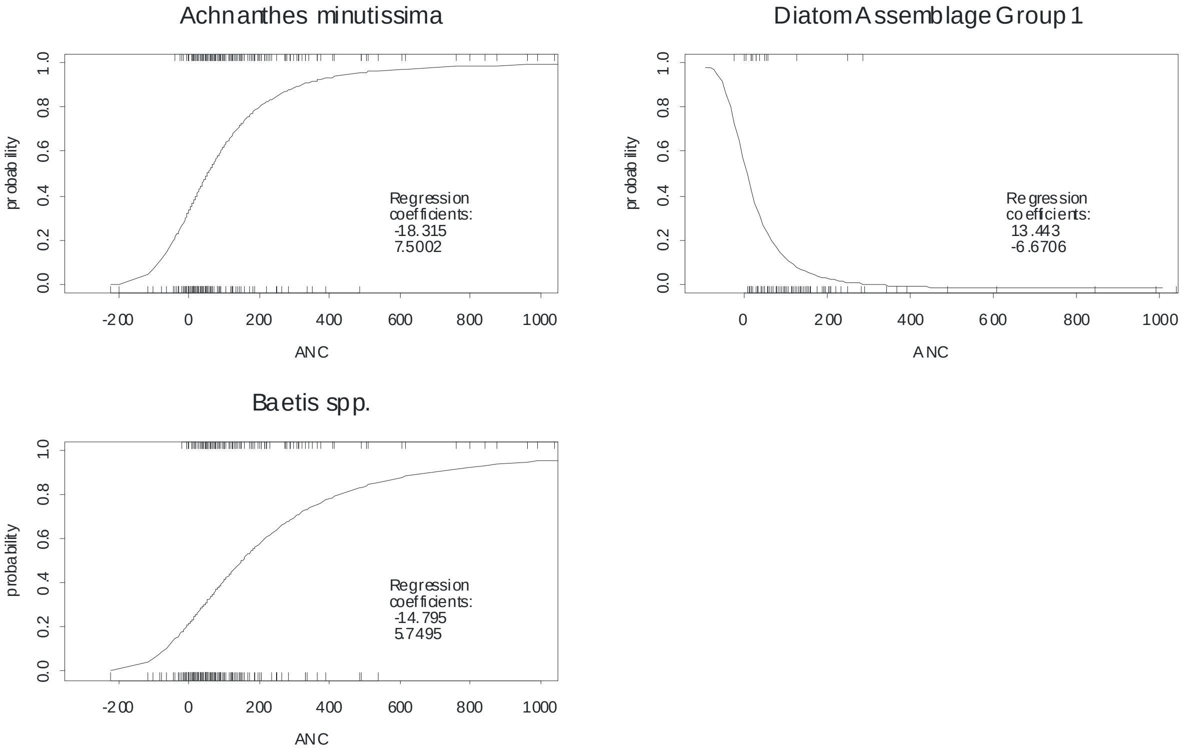


Figure 8 Distribution of selected diatom and invertebrate taxa / assemblages, with fitted response curves

The performance of the models is assessed on the basis of the proportion of sites where the presence / absence of the organism or assemblage is predicted correctly. With logistic regression and other classification models one has to make the choice of a predicted probability to use as the presence / absence threshold. In many cases this is taken as 0.5, although many have shown that this may not produce an overall optimal discrimination (e.g. Manel *et al*. 1999). We have therefore used ROC plots (Zweig & Campbell 1993) to determine the probability threshold that produces the best predictive model, in terms of correctly predicted absences and presences.

Table 12 Performance statistics for the three logistic regression models

Model	***Threshold probability***	***% Absences predicted correctly***	***% Presences predicted correctly***	***Overall % predicted correctly***
Achnanthes minutissima	0.5	73	70	71
Baetis rhodani	0.35	70	66	68
Diatom Group 1	0.35	77	72	76

Performance statistics for the three models are listed in Table 12. For the diatom taxon model, a threshold probability of 0.5 gave the most robust model, while probabilities of 0.35 were chosen for the invertebrate and diatom group models. The performance of all three models is similar, with overall and presence / absence success rates of 68-77%. These rates are high, and indicate that the models are sufficiently robust to be used to predict the occurrence of these organisms from field and / or modelled ANC determinations.

4. Linking biological and hydrochemical models

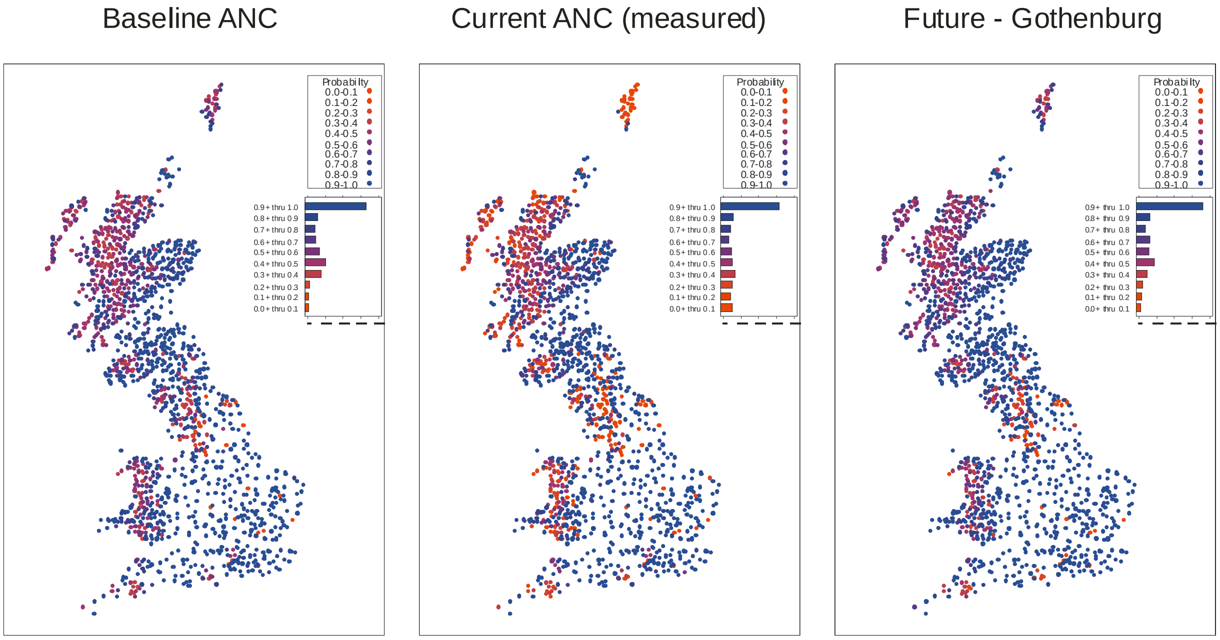
Inherent in the critical loads approach is the concept of biological change or damage that occurs once a critical load has been exceeded. In assessing the impact of critical load exceedance it is often difficult to identify biological communities that are unambiguously indicative of a “damaged” biota. This is because the same or very similar biological communities can occur at a moderately buffered site that has been strongly acidified and at a weakly buffered, naturally acidic, low ANC site that has only been mildly acidified. Without knowledge of the pre-acidification flora and fauna of a site it is therefore very difficult to assess biological damage using field survey data. Without such knowledge of the pre-acidification biota, and in the absence of other information from palaeolimnology or analogue sites, we use an approach that links measured and modelled ANC values to the biological models to predict pre-acidification, current and future biota, and to assess the spatial distribution and magnitude of biological change as a result of critical load exceedance.

Two different modelling exercises have carried out. The first uses chemical data from the CLAM national chemical database for current measured ANC, together with pre-industrial ANC calculated using the SSWC model, and future ANC calculated using the SSWC model under the Gothenburg scenario.

The second modelling exercise uses MAGIC 5 predictions of baseline, current and future ANC (under the Gothenburg scenario) for regional datasets (Cairngorms, Galloway, English Lake District, Pennines and Wales).

4.1 Biological predictions for the national chemical database

Biological predictions using the CLAM national chemical database are shown graphically in Figures 9-14, and summarised in Table 13. Figures 9-11 show maps of the predicted probability of occurrence of each biological target organism under baseline (pre-industrial), current and future (Gothenburg – HARM 11.5/FRAME modelled deposition for 2010, see NEGTAP Report) conditions. These figures encapsulate the spatial distribution of the shifts in probability of occurrence for the different time periods, and clearly show the loss of the acid sensitive taxa and expansion of the acid tolerant diatom group between baseline and the present, and the recovery, or partial recovery under the Gothenburg protocol.


Baseline ANC

Current ANC (measured)

Future - Gothenburg

Figure 9: Probability of occurrence of *Achnanthes minutissima* under pre-acidification, present, and Gothenburg modelled ANC.

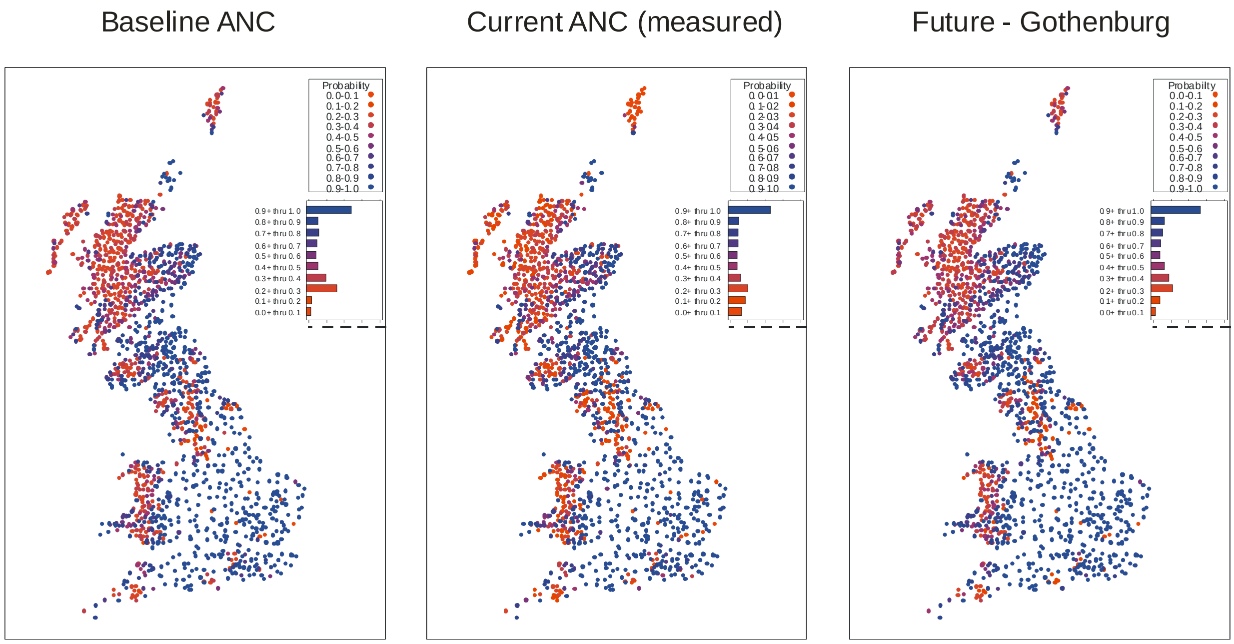


Figure 10: Probability of occurrence of *Baetis rhodani* under pre-acidification, present, and Gothenburg modelled ANC.

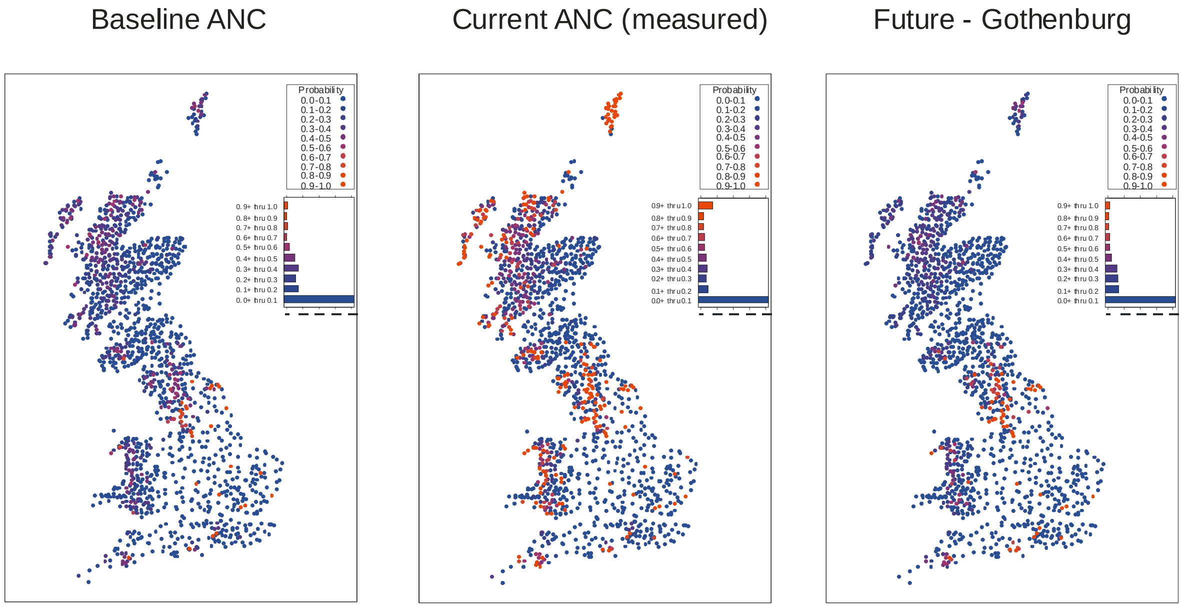
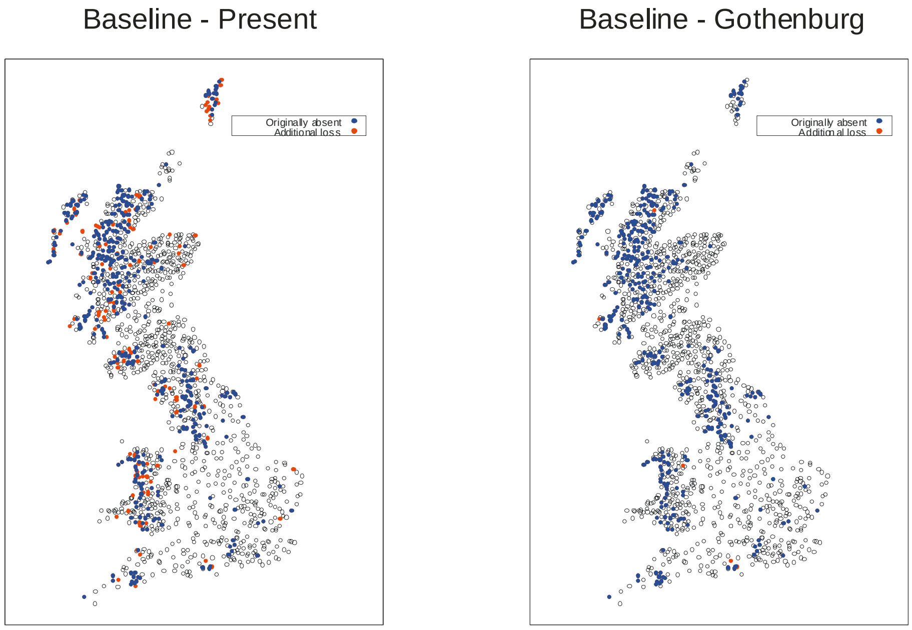


Figure 11: Probability of occurrence of Diatom Group 1 under pre-acidification, present, and Gothenburg modelled ANC.

Figures 12-14 summarise the changes depicted in the previous figures and summarise the change in the predicted occurrence of each target between baseline and the present, and between baseline and future conditions. Table 13 provides summary statistics for these distributions.



Baseline - Present

Baseline - Gothenburg

Figure 12: Change in the distribution of *Achnanthes minutissima between* pre-acidification, present, and Gothenburg modelled ANC. See text for explanation.

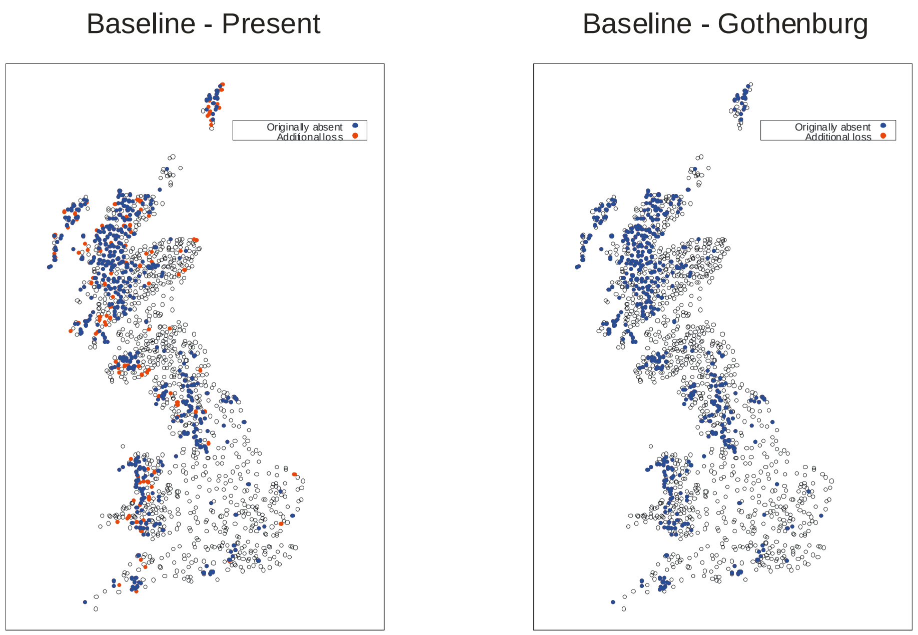


Figure 13: Change in the distribution of *Baetis rhodani between* pre-acidification, present, and Gothenburg modelled ANC. See text for explanation.

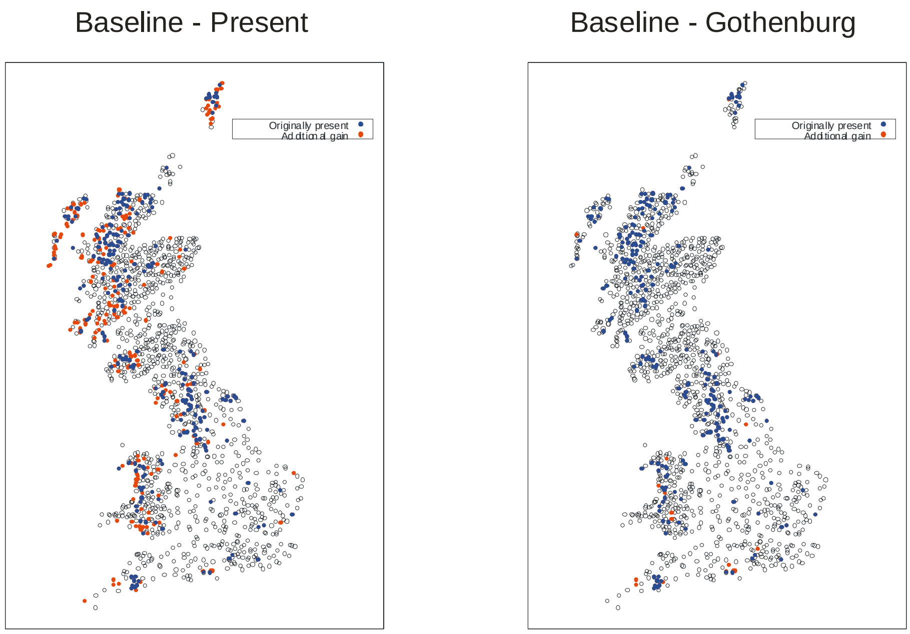


Figure 14: Change in the distribution of Diatom Group 1 *between* pre-acidification, present, and Gothenburg modelled ANC. See text for explanation.

Table 13 Number of sites with predicted occurrences for each target organism for baseline, current and future ANC. Figures in brackets indicate change from baseline). Total number of sites = 1468.

Target organism	***Baseline ANC***	***Current ANC***	***Future ANC***
Achnanthes minutissima	1080	958 (-124)	1133 (53)
Baetis rhodani	1022	910 (-112)	1082 (60)
Diatom Group 1	240	434 (194)	190 (-50)

In Figures 12-14 and Table 13 a comparison of the baseline and current distributions summarises the biological change due to acidification. Results indicate that *A. minutissima* has been lost from 124 sites, and *B. rhodani* lost from 112 sites since the mid 19th century, and that Diatom Group1 has expanded its distribution to an additional 194 sites. These figures represent profound biological change across 8 – 13%

of the total national dataset, with impacted sites clustered in regions of known acidification: SW England, Wales, Pennines, Cumbria, and large parts of Scotland. For many regions these predictions are consistent with information on current diatom and invertebrate communities from contemporary field survey data. However, the loss of taxa predicted for the far north of Scotland is unexpected, and probably a result of errors in the calculation of non-marine ANC for sites with high inputs of sea-salts.

Predictions for future ANC based on the Gothenburg scenario suggest an almost complete recovery: ANC has increased sufficiently to allow return of the target organisms to virtually all sites inhabited prior to the onset of acidification. Predicted future distributions do in fact suggest an increase in the range of *A. minutissima* and *B. rhodani*, and a reduction in the distribution of Diatom Group 1 from baseline conditions. Again, this is also probably due to an error in the SSWC ANC at sites with high sea salt inputs.

4.2 Biological predictions for the MAGIC 5 regional datasets

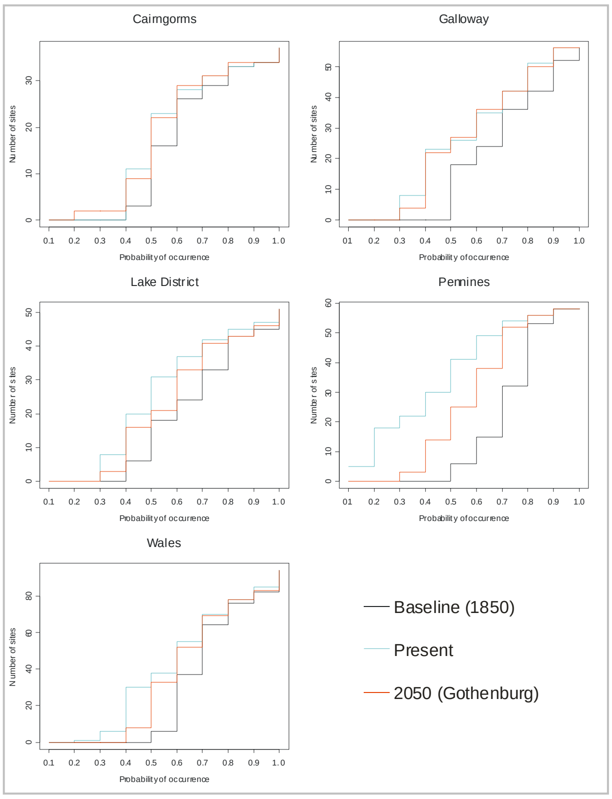
Biological predictions for the MAGIC 5 regional datasets are shown in Figure 14 and Table 14. The former shows cumulative frequency plots of probabilities of occurrence for each biological target for three time periods (1850, present and 2050 under the Gothenburg scenario). Table 14 summaries these distributions and lists the numbers of sites with predicted occurrences for each dataset.

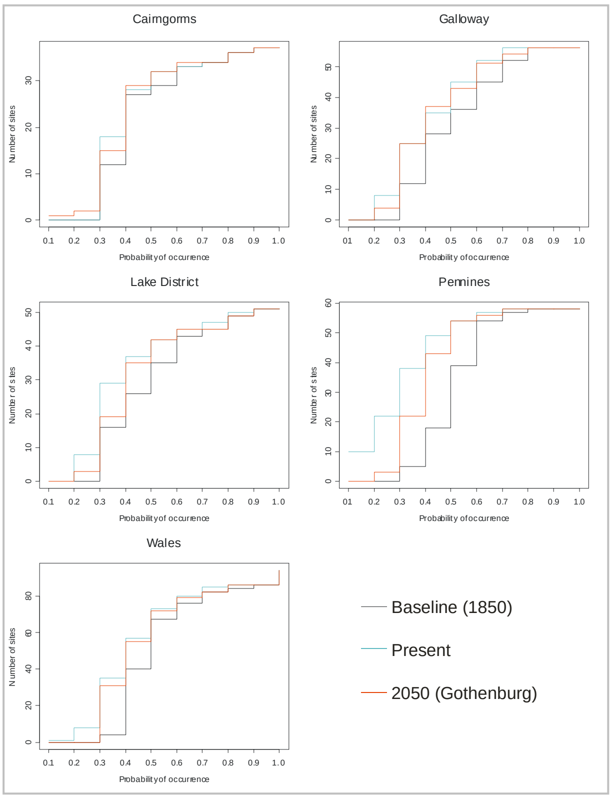
All regions exhibit significant biological change between baseline and present conditions, although the magnitude of change varies greatly between regions. Results suggest that the Cairngorms are the least impacted, with *A. minutissma* lost from 7 sites (18%), and *B. rhodani* lost from 3 (8%) since the onset of acidification. The Pennines have undergone most change, with *A. minutissima* lost from 35 sites (60%), and *B. rhodani* lost from 34 (58%). In all regions the acid tolerant diatom group has expanded its distribution, by 6 sites (16%) in the Cairngorms, 31 sites (53%) in the Pennines, and 31 sites (33%) in Wales.

Table 14 also summaries results for predicted biological recovery under the Gothenburg scenario. Again, the pattern of recovery varies greatly between regions: the Cairngorms and Galloway show little change between present and future predictions for all three biological targets, while other regions show some limited recovery for different groups. For example, in the Lake District results suggest that *A. minutissima* will return to 30 of its previously inhabited 33 sites, but that there will be more limited recovery for *B. rhodani* populations, which will still be absent from 8 (16%) of previously inhabited sites. In the Pennines, *A. minutissima* is expected to return to 33 of a previously inhabited 52 sites, while *B. rhodani* will recover at only 27 of the previous 49 sites. Similarly, Diatom Group 1 shows little change in Galloway or the Cairngorms, while there is a limited reduction in its distribution in Pennines and Wales where is disappears from approximately half the sites.

4.3 Discussion

The models developed in Section 3.5 are empirical statistical models. They predict the likely probability of occurrence of an organism based on ambient mean ANC. They are, in effect, steady state models - they take no account of additional chemical, habitat or biological factors that influence biological distributions, nor do they allow for hysteresis or lags in recolonisation. Unreported analyses demonstrate that the predictive ability of the models can be improved by including additional explanatory factors, but the improvements are small, and the additional chemical and habitat data needed for model application often not available. Despite these limitations the models do provide a robust means of predicting the occurrence of key organisms from a single ANC value. As such the models provide a simple means of converting hydrochemical hindcasts and forecasts into measures of biological change. In particular, they allow the assessment of biological damage due to acidification, and the likely recovery under different emission reduction scenarios.





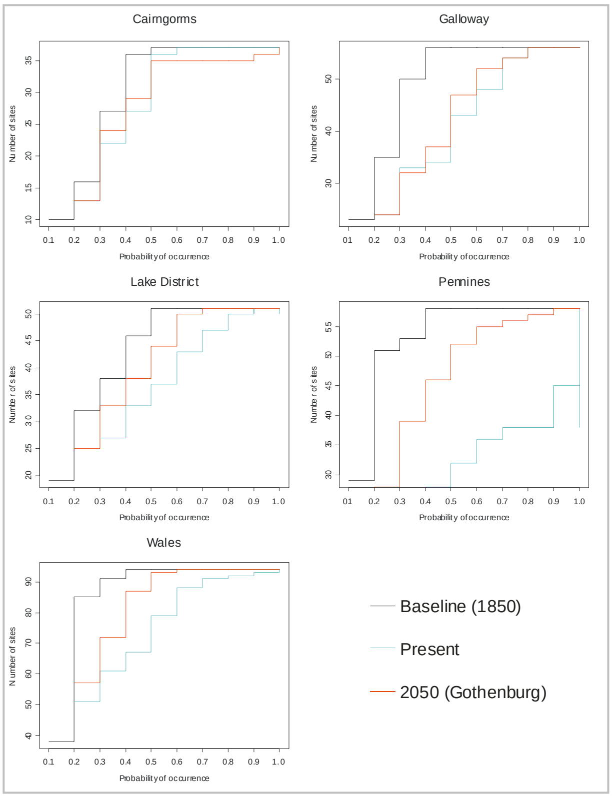


Figure 15: Cumulative frequency distributions of probabilities of occurrence for Achnanthes minutissima (top left), Baetis rhodani (top right, and Diatom Group 1 (bottom left), for MAGIC 5 datasets.

However, the biological predictions are only as good and the hydrochemical data they are based on. The above analyses based on MAGIC 5 and the SSWC models and revealed similar patterns of impacts and biological damage, but differ in their predictions of likely recovery: SSWC suggests virtually complete recovery under the Gothenburg scenario, where MAGIC 5 predicts that substantial biological damage will remain after 2050. These differences are partly explained by the assumptions of steady state in the former, but clearly there are differences in the hydrochemical predictions that need to be reconciled.

Table 14 MAGIC 5 Biological predictions, showing total number of sites and number of sites with predicted occurrences for each target organism for baseline, current and future ANC. Figures in brackets show percentages of the total number of sites.

Target organism / Region	*Number of Sites*	*Baseline ANC*	*Current ANC*	*Future (2050 - Gothenburg) ANC*
Achnanthes minutissima				
Cairngorms	37	21 (57)	14 (38)	15 (41)
Galloway	56	38 (68)	30 (54)	29 (52)
Lake District	51	33 (65)	20 (39)	30 (59)
Pennines	58	52 (90)	17 (29)	33 (57)
Wales	94	88 (94)	56 (60)	61 (65)
Baetis rhodani				
Cairngorms	37	14 (38)	11 (30)	11 (30)
Galloway	56	35 (63)	23 (41)	23 (41)
Lake District	51	32 (63)	16 (31)	24 (47)
Pennines	58	49 (84)	15 (26)	27 (47)
Wales	94	84 (89)	50 (53)	55 (59)
Diatom Group 1				
Cairngorms	37	5 (14)	11 (30)	12 (32)
Galloway	56	2 (4)	23 (41)	24 (43)
Lake District	51	10 (20)	20 (39)	16 (31)
Pennines	58	1 (2)	32 (55)	16 (28)
Wales	94	0 (0)	31 (33)	11 (12)

5. Conclusions

1. A combined database has been constructed using existing and new datasets to provide information on diatom, macrophyte and invertebrate distribution in relation to a range of water chemistry variables for 459 samples. This new database greatly expands the number of taxa and range of chemical gradients available in previous individual datasets. As such the new database captures taxon distributions more fully and allows modelling along the full range of taxon distribution along multiple chemical gradients.

2. Community ordination analysis using detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) indicate that the subset of chemical variables available for all samples (pH, alkalinity, conductivity, Ca, Cl, SO_4, and total monomeric aluminium) effectively account for the dominant biological gradients in the diatom and invertebrate datasets.

3. Forward-selected CCA with Monte-Carlo permutation testing revealed that pH, total monomeric aluminium and conductivity, in that order, are the most important variables in accounting for the observed diatom and invertebrate distributions. Partial CCAs indicate that all three environmental variables account for significant total, and unique fractions of the total, variance in the biological datasets, justifying their inclusion as explanatory variables in predictive models of species distribution.

4. Logistic regression response models fitted to individual diatom and invertebrate taxa with 20 or more occurrences indicate that all 50 selected diatom taxa, and 31 of 39 selected invertebrate taxa show statistically significant responses to pH. These results are used to identify potential indicator taxa with varying degrees of tolerance to lowered pH.

5. Although there is a statistically significant relationship between macrophyte community composition and pH, the macrophyte dataset is at present too small (28 sites) to allow the modelling of individual taxa.

6. Predictive models are derived for three biological targets, the diatom *Achnanthes minutissima*, the invertebrate *Baetis rhodani* and an acid tolerant diatom assemblage. *Achnanthes minutissima* and *Baetis rhodani* are both common in streams above pH 6.0, and are among the first taxa to decline in response to lowered pH. The acid tolerant diatom group is characterised by *Eunotia exigua*, an acidobiontic taxon commonly found in acidified waters.

7. The fitted logistic regression models can predict the occurrence of the target organisms with an internal success rate of c. 70%, and provide a means of predicting the probability of occurrence of the target organisms from modelled hindcasts and forecasts.

8. Application of the models to the CLAM national chemical database using the SSWC model provides biological predictions for baseline (pre-acidification), present, and future (Gothenburg scenario) conditions. Comparisons of the mapped biological predictions for baseline and present indicate substantial biological change in the areas of known acidification: SW England, Wales, Cumbria, Pennines and large parts of Scotland. The loss of sensitive taxa predicted for sites in the north of Scotland are probably the result of errors in the calculation of non-marine ANC for sites with high inputs of sea-salt. Comparison of predictions for baseline and future steady state ANC suggests virtually complete biological recovery under the Gothenburg scenario.

9. Biological predictions for five MAGIC 5 regions (Cairngorms, Galloway, Pennines, Wales, Lake District) show similar patterns of biological change between baseline and present, with loss of sensitive taxa from 18% of sites in the Cairngorms through to loss from 60% of sites in the Pennines. Predictions for recovery under the Gothenburg scenario indicate that only partial recovery will take place by 2050, and for some regions sensitive taxa are predicted only to return to approximately half of the sites inhabited prior to the onset of acidification.

10. The biological models provide a simple but robust way to convert hydrochemical hindcasts and forecasts into measures of biological change, and in particular, to assess biological damage under different emission reduction scenarios. However, there is a need to validate the models using independent field data, and a need to reconcile the differences in hydrochemical predictions between the SSWC model and MAGIC 5.

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7. Appendices

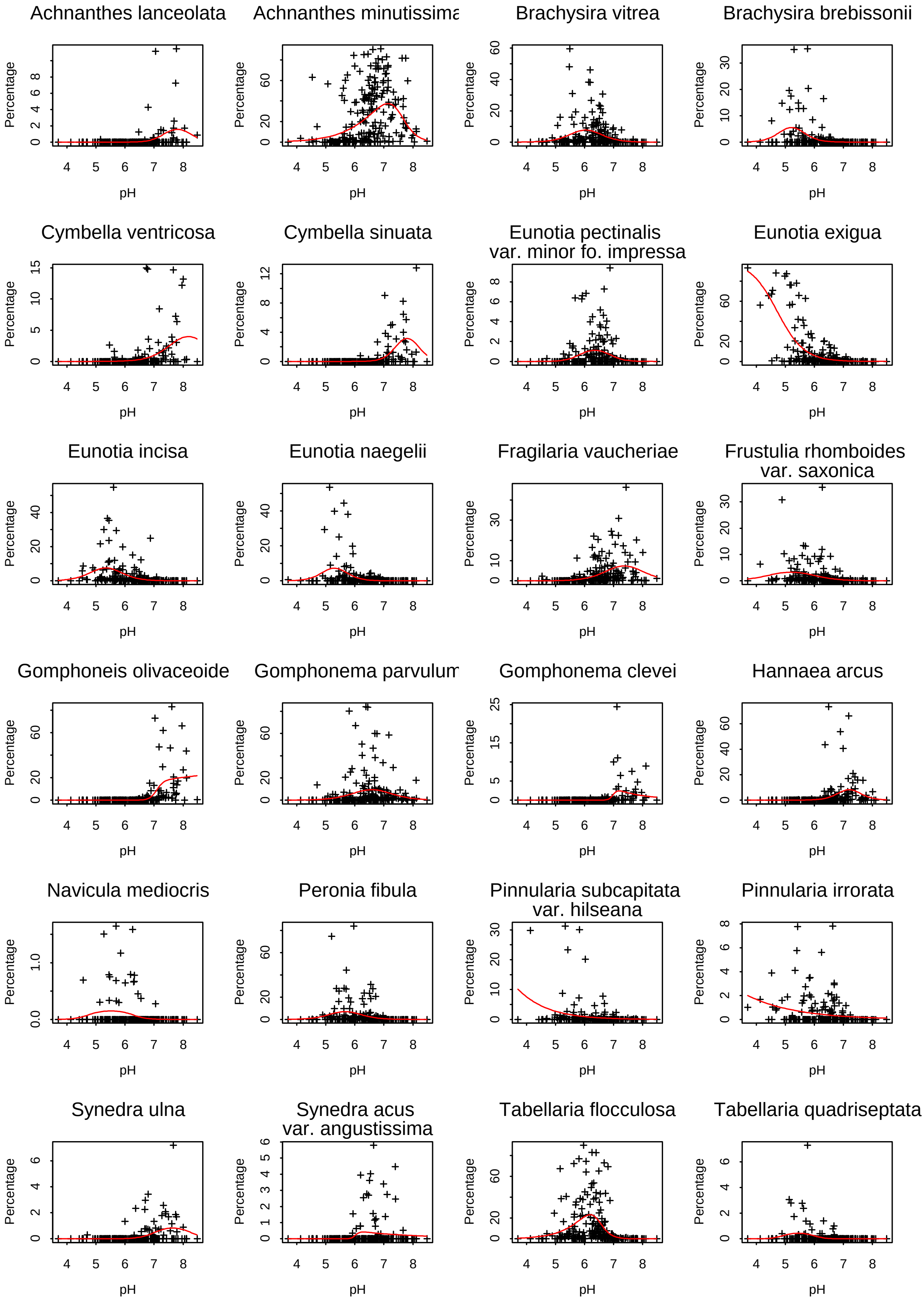
Appendix 1 List of diatom taxa with 20 or more occurrences, showing maximum relative abundance, number of occurrences, and nature of modelled response to pH (NS = not significant).

TaxonName	*Max*	*N*	*Response*
Achnanthes lanceolata	11.5	24	Symmetric
Achnanthes minutissima	90.9	199	Skewed
Achnanthes austriaca var. minor	44.0	33	Symmetric
Achnanthes austriaca var. helvetica	18.5	68	Symmetric
Achnanthes marginulata	44.5	38	Skewed
Achnanthes saxonica	84.7	59	Skewed
Achnanthes detha	8.9	25	Skewed
Brachysira vitrea	59.6	149	Symmetric
Brachysira brebissonii	35.4	72	Symmetric
Cymbella ventricosa	15.0	43	Symmetric
Cymbella sinuata	12.8	33	Symmetric
Cymbella microcephala	15.5	28	Symmetric
Cymbella minuta	19.2	47	Symmetric
Cymbella lunata	8.6	75	Symmetric
Cocconeis placentula var. euglypta	58.0	33	Skewed
Diatoma elongatum var. tenue	18.3	27	Symmetric
Diatoma hyemale var. mesodon	44.1	46	Symmetric
Eunotia pectinalis var. pectinalis	9.1	55	Symmetric
Eunotia pectinalis var. minor	12.3	82	Symmetric
Eunotia pectinalis var. minor fo. impressa	9.4	77	Symmetric
Eunotia exigua var. exigua	99.0	154	Monotonic
Eunotia rhomboidea	50.0	110	Symmetric
Eunotia denticulata	10.7	28	Skewed
Eunotia meisteri	6.7	25	Symmetric
Eunotia incisa	54.7	132	Symmetric
Eunotia naegelii	53.5	92	Symmetric
Eunotia curvata	29.5	80	Skewed
Eunotia vanheurckii var. intermedia	61.6	49	Skewed
Eunotia minutissima	33.9	39	Symmetric
Fragilaria virescens var. exigua	26.1	32	Symmetric
Fragilaria vaucheriae	46.3	132	Symmetric
Frustulia rhomboides var. saxonica	50.0	127	Symmetric
Frustulia rhomboides var. viridula	4.0	35	Symmetric
Gomphoneis olivaceoides	83.1	54	Skewed
Gomphonema gracile	27.0	64	Symmetric
Gomphonema parvulum	84.2	133	Symmetric
Gomphonema clevei	24.5	32	Skewed
Hannaea arcus	73.3	66	Symmetric
Meridion circulare	58.1	65	Skewed
Navicula mediocris	2.3	30	Symmetric
Nitzschia perminuta	5.2	54	Monotonic
Peronia fibula	84.0	103	Symmetric
Pinnularia subcapitata var. hilseana	31.3	64	Monotonic
Pinnularia irrorata	7.8	66	Monotonic
Synedra acus	50.1	67	Symmetric
Synedra acus var. angustissima	5.8	25	Skewed
Synedra minuscula	52.2	126	Symmetric
Tabellaria flocculosa	89.7	198	Skewed
Tabellaria quadriseptata	47.0	30	Symmetric

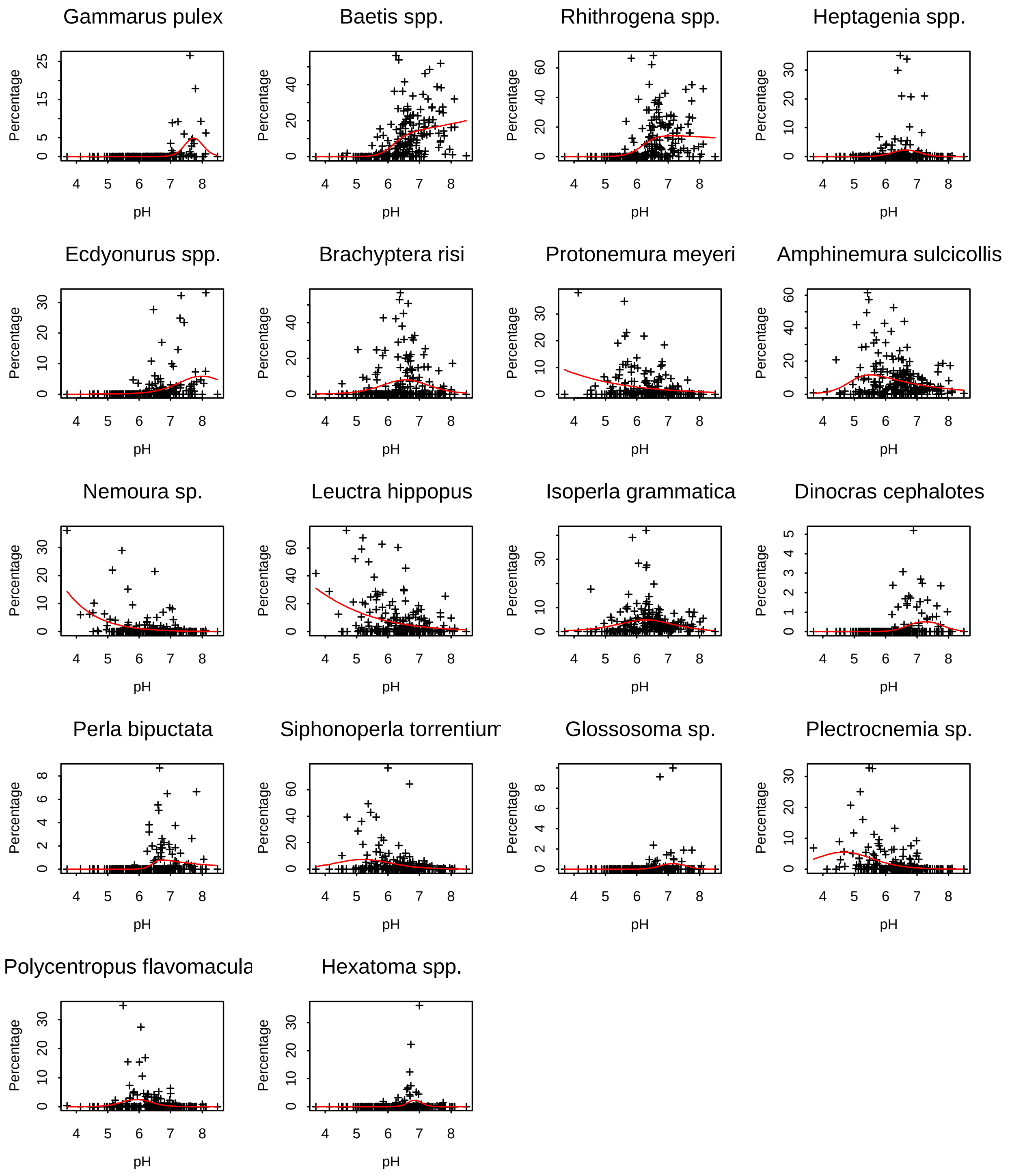
Appendix 2 List of invertebrate taxa with 20 or more occurrences, showing maximum relative abundance, number of occurrences, and nature of modelled response to pH (NS = not significant).

TaxonName	***Max***	***N***	***Response***
Pisidium sp.	37.7	29	Symmetric
OLIGOCHAETA	72.7	156	Symmetric
Gammarus pulex	26.6	22	Symmetric
Baetis spp.	56.3	152	Skewed
Rhithrogena spp.	68.5	112	Skewed
Heptagenia spp.	35.1	53	Symmetric
Ecdyonurus spp.	33.1	66	Symmetric
Paraleptophlebia submarginata	10.4	26	NS
Brachyptera risi	56.7	96	Symmetric
Protonemura praecox	10.5	30	NS
Protonemura meyeri	37.9	113	Monotonic
Amphinemura sulcicollis	61.4	179	Skewed
Nemoura spp.	36.0	63	Monotonic
Leuctra inermis	69.5	156	NS
Leuctra hippopus	72.5	138	Monotonic
Leuctra nigra	65.4	36	Symmetric
Perlodes microcephala	9.1	54	NS
Diura bicaudata	7.4	22	Symmetric
Isoperla grammatica	42.0	171	Skewed
Dinocras cephalotes	5.2	33	Symmetric
Perla bipuctata	8.7	44	Skewed
Siphonoperla torrentium	76.5	119	Symmetric
Chloroperla tripunctata	10.4	70	Symmetric
Elmis aenea	35.6	62	NS
Limnius volckmari	45.5	99	Monotonic
Oulimnius spp.	23.4	36	Monotonic
Rhyacophila sp.	14.7	130	Monotonic
Glossosoma sp.	10.0	26	Symmetric
Plectrocnemia sp.	32.8	90	Symmetric
Polycentropus flavomaculatus	34.9	68	Symmetric
Hydropsyche spp.	24.5	107	NS
LIMNEPHILIDAE undet.	16.7	27	Monotonic
Potamophylax spp.	20.2	70	NS
TIPULIDAE	43.2	66	Symmetric
Dicranota spp.	11.1	74	Skewed
HEXATOMA SPP	36.2	27	Symmetric
CHIRONOMIDAE	85.5	173	NS
SIMULIIDAE	97.4	150	Monotonic
EMPIDIDAE	4.2	30	Monotonic

Appendix 3 Selected diatom taxa, showing percent abundance vs. pH and fitted response curves. See text for details.



Appendix 4 Selected invertebrate taxa, showing percent abundance vs. pH and fitted response curves. See text for details.



Task 2.7.1: Assessing hysteresis in biological response to chemical recovery from acidification: the case of limed streams at Llyn Brianne

D.C. Bradley & S.J. Ormerod

SUMMARY

1. There is a need to assess the factors that might delay biological recovery in streams recovering from acidification, but real chemical recovery has so far been insufficient to satisfy this need. However, catchment liming causes accelerated deacidification and provides an opportunity to assess the speed of accompanying biological response. Using the replicated basin-scale experiment at Llyn Brianne, we assessed chemical and biological effects for 10 years after the catchments of three acidified Welsh streams were limed in 1987/88.
2. Stream chemistry was measured weekly to monthly, and macroinvertebrates monitored annually, between 1985 and 1998. Biological change through time was assessed from the abundance and taxon richness of invertebrates. We paid particular attention to 18 species known to be acid-sensitive. The effects of liming were assessed by comparing chemical and biological trends among the three replicate limed streams, three acid reference streams and two naturally circumneutral streams.
3. Following single lime applications, acid-base chemistry in treated streams changed significantly. High mean pH (> 6), increased calcium (> 2.5 mg L^{-1}) and low aluminium (< 0.1 mg L^{-1}) persisted throughout the 10 years following liming.
4. The effects of liming on invertebrates were modest. Acid sensitive taxa increased significantly in abundance in limed streams, but only during 2 years following treatment. Significant effects on richness were more sustained, but on average added only 2-3 acid-sensitive species to the treated streams, roughly one-third of their average richness in adjacent circumneutral streams. Only the mayfly *Baetis rhodani* and the stonefly *Brachyptera risi* occurred significantly more often in limed streams after treatment than before it.
5. Despite these modest effects on invertebrates, nearly 80% of the total pool of acid-sensitive species has occurred at least once in the limed streams in the 10 years since treatment. This pattern of occurrence suggests that the delayed colonization of limed streams by acid-sensitive taxa reflects limited persistence rather than restricted dispersal. We present evidence to show that episodes of low pH continued to affect acid-sensitive taxa even after liming. At Llyn Brianne, hysteresis in streams recovering from acidification is real, but might reflect the continued effects of episodes rather than other biological limits.

Introduction

Marked ecological changes in acidified surface waters have been reported since the 1970s, and much research is now focussed on recovery and methods to restore or protect sensitive locations (e.g. Ormerod *et al.* 1990). Causative treatment by reducing acidifying emissions is the primary goal, but future outcomes from this strategy are still uncertain and recovery may be slow (Hildrew & Ormerod, 1995; Sandøy & Romundstad, 1995; Weatherley *et al.* 1995). Symptomatic treatment, involving the addition of neutralizing agents such as powdered limestone to affected waterbodies or their catchments, is the only realistic remedy in the short-term, and has become a widespread practice in Europe (e.g. Weatherley, 1988; Svenson *et al.* 1995).

Although often a central strategy in the treatment of acidification, some of the effects of liming on stream biota remain unclear. Liming sometimes provides adequate chemical conditions for acid-sensitive taxa (e.g. Lingdell & Engblom, 1995; Hindar *et al.*, 1996; Clayton & Menendez, 1996; Schofield & Keleher, 1996; Simmons & Cieslewicz, 1996a, 1996b), but biological responses on other occasions have fallen short of expectations, particularly among acid-sensitive invertebrates (e.g. Appelberg *et al.*, 1995; Fjellheim & Raddum, 1992, 1995; Rundle, Weatherley & Ormerod 1995; Clayton & Menendez, 1996; Eggleton, Morgan & Pennington, 1996; Simmons & Doyle, 1996). In the latter case, authors have speculated about which factors other than acid-base chemistry might limit or obscure species' colonisation: (i) factors such as climate may affect colonisation, for example by causing continued acid episodes (Weatherley & Ormerod, 1992; Rundle, Weatherley & Ormerod, 1995); (ii) biological changes resulting from liming might be hidden against a background of natural population variations (Weatherley & Ormerod, 1992) or other natural factors such as predation (Schofield & Keleher, 1996); (iii) the dispersal abilities of acid-sensitive taxa may limit their colonisation of de-acidified streams (Weatherley & Ormerod, 1992; Rundle, Weatherley & Ormerod, 1995); iv) liming can cause negative impacts on streams, such as fine $CaCO_3$ deposited locally on the stream benthos (Fjellheim & Raddum, 1995; Clayton & Menendez, 1996), or toxic conditions in mixing zones between limed and acid waters (Larsen & Hesthagen, 1995); v) habitat factors such as food quality or structural physiography might prevent some invertebrates from colonizing limed streams. Among these possibilities, habitat assessments and transplantation experiments show that the last hypothesis can be dismissed for the sites involved in this study (Z. Masters *et al.* unpubl. data). Hypothesis iv) can be discounted where lime applications are made to the watershed rather than directly to the stream. However, the other three hypotheses remain active areas of consideration.

With many liming experiments running for only short durations, the lack of long-term data prevents these alternative explanations from being adequately evaluated (e.g. Weatherley & Ormerod, 1992; Simmons & Doyle, 1996). We therefore assessed, for 10 years post-liming, the long-term changes in stream chemistry and invertebrate fauna that have occurred in three formerly acidified streams that were limed in 1987/88. We compared trends in limed streams with three unmanipulated acid reference streams and two naturally circumneutral streams. In so doing, our evaluations allow an assessment of the extent to which hysteresis affects biological recovery in streams where de-acidification has been substantial. So far, no other stream has recovered as substantially in ANC as the effects due to liming, and so these experimental data have real value in indicating biological lags that might occur in other systems.

This section of the report extends by 6 years a previous assessment of liming effects on these same streams (Rundle, Weatherley & Ormerod, 1995). Our expectation was that continued high pH and base-cation concentration might have allowed a further period for dispersal and colonisation. We also wished to assess trends over a timescale long enough to account for natural background fluctuations. Substantial year-to-year variation can occur among stream invertebrate communities (Weatherley & Ormerod, 1990; Lancaster *et al.* 1996), and at Llyn Brianne such changes closely reflect the North Atlantic Oscillation (D. Bradley & S. J. Ormerod unpubl. data).

Methods

Study area

As part of a large-scale project aimed at assessing the causes, consequences and management of acidification, eleven streams in the upper Tywi catchment near Llyn Brianne reservoir, Mid-Wales (52° 8’N 3° 45’W) have been monitored annually from

1985 to the present (Stoner, Wade & Gee, 1984, Weatherley & Ormerod, 1987). The streams are all second or third order, with catchments of 15-253 ha, altitudes of 215-410m and varying land-use (Rutt, Weatherley & Ormerod, 1989). Monthly mean temperatures in the streams range from 0-20 °C and mean annual rainfall is c. 1900mm (Weatherley & Ormerod, 1990). The base-poor soils give rise to soft runoff (mean total hardness 3.9-18.8 mg $CaCO_3\ L^{-1}$) which is susceptible to acidification (mean pH 4.6-6.9) (Rutt, Weatherley & Ormerod, 1989). Three of the experimental catchments were artificially limed with a single addition of powdered $CaCO_3$ in 1987/88. CI5, draining moorland, was treated in September 1987 with the addition of 9 t $CaCO_3$ ha^{-1} to the entire catchment (0.34 km^2). Another moorland catchment, CI2, was treated in June 1988 with the addition of 15-20 t $CaCO_3$ ha^{-1} targeting the hydrological source areas (0.05 km^2). LI4, draining plantation forest of sitka spruce (*Picea sichensis* Carriere) was treated in November/December 1987 with the addition of 25 t $CaCO_3$ ha^{-1}, also targeting the source areas (0.04 km^2) (Edwards, Gee & Stoner, 1990; Weatherley & Ormerod, 1992; Donald & Gee, 1992). For this work, we consider the three limed streams; three adjacent, unmanipulated acid reference streams at LI1 (acid forest), CI1 and CI4 (acid moorland); and two naturally circumneutral reference streams at LI6 (moorland) and LI7 (moorland) (Table 1). Pre-treatment biological differences between these three stream types have been described previously (Weatherley & Ormerod, 1987; Rundle, Weatherley & Ormerod, 1995).

Biological sampling

Benthic macroinvertebrates were sampled once every March/April from 1985 to 1998 (except 1991, when no funding was available) in all streams using a standardised three-minute kick-sampling procedure (Weatherley & Ormerod, 1987). At each site a two-minute sample was taken from mid-channel riffle habitat and a one-minute sample was taken from stream margins (Weatherley & Ormerod, 1987). Samples were preserved in 70% alcohol for storage. Following identification and enumeration to the lowest feasible taxonomic level, riffle and margin samples from each site were combined as a total for the purposes of this analysis.

Table 1. Site characteristics and chemical data pre- (1985-1987) and post-liming (1988-1998) for the experimental streams at Llyn Brianne. Data on catchment area are from Edwards, Gee & Stoner (1990).

Site	Treatment	Catchment land use	Catchment area (ha)	Mean pH pre-liming	Mean pH post-liming	Mean Al pre-liming	Mean Al post-liming
LI4	Limed 1987	Conif. afforestation	32	5.1	6.2	0.19	0.08
CI2	Limed 1988	Moorland	59	5.0	6.1	0.15	0.11
CI5	Limed 1987	Moorland	33	5.2	6.2	0.15	0.06
LI1	Acid reference	Conif. afforestation	253	4.9	4.9	0.39	0.40
CI1	Acid reference	Moorland	15	5.2	5.2	0.10	0.10
CI4	Acid reference	Moorland	49	5.2	5.5	0.12	0.12
LI6	Circumneutral reference	Moorland	68	6.8	6.9	0.06	0.05
LI7	Circumneutral reference	Moorland	72	6.9	6.9	0.04	0.04

Physicochemical variables

Stream chemistry was monitored weekly-monthly from 6 January 1984 to 24 November 1998. Weatherley & Ormerod (1987) detailed analytical procedures for determining pH, aluminium (mg L^{-1}), calcium (mg L^{-1}) and hardness (mg L^{-1}) concentrations. Mean annual values were calculated for each chemical determinand. In addition, we calculated pH ranges and minima, and aluminium ranges and maxima for the winter (October-March inclusive), to give an indication of the severity of episodic events since this period directly preceded the annual invertebrate samples. The detection of chemical extremes is dependent on sampling frequencies, so that monthly protocols like ours that are not continuous are estimates of episodicity rather than absolute measures (Brewin *et al.* 1996).

Statistical analysis

The experiment involved sampling for periods before and after liming, and so was suited to the BACI analytical design (= before-after-control-impact) proposed by Stewart-Oaten, Murdoch & Parker (1986). BACI designs are increasingly accepted as a means of overcoming problems in analyses that might otherwise be pseudoreplicated (Hurd, Perry & Perry, 1996; Solazzi *et al.* 2000). However, our study had an additional advantage in that it involved replicate reference and treatment catchments. It therefore offered opportunity for a multiple BACI (= MBACI) design, suited to repeated measures analysis of variance (= ANOVA; Paine, 1996; Niemi *et al.* 1999). We chose the latter option, following the general model summarised by Paine (1996). Our overall approach was to compare changes in limed streams with those in acid references, and then assess subsequently whether any significant changes due to liming led to similarity with the circumneutral reference streams. Since stream CI2 was not limed until 1988 all data for this year were omitted from statistical analyses

For stream chemistry, total abundance and taxon richness, we assessed variations due to treatments (limed or unlimed), replicate streams (within treatments), times within replicates (i.e. pre- versus post-liming years) and time x treatment interactions. The latter interactions were key to testing statistically the hypothesis that any changes in chemistry or invertebrates were caused by liming (Paine 1996; Rundle, Weatherley & Ormerod, 1995). Assessments of abundance involved either all taxa, or alternatively they were confined to the eighteen most acid-sensitive taxa at Llyn Brianne that have been previously identified from ordination (Weatherley & Ormerod, 1990, Rundle, Weatherley & Ormerod, 1995). These taxa have shown a consistent affinity for circumneutral conditions (Rundle, Weatherley & Ormerod, 1995). Prior to all these analyses, we logarithmically transformed abundances and the concentrations of any determinand that required it to remove heteroscedasticity and satisfy analytical assumptions.

In a further permutation of the repeated measures analysis, we wished to examine whether any changes due to liming were restricted to the immediate post-liming years. For this, we separated the time component into three periods respectively involving the pre-liming period (3 years), immediately post-liming (3 years) and the remaining period (7 years). Following Green (1993), we used contrasts to identify which of these three periods were significantly different. Otherwise, all other variations were examined as outlined above.

To assess which taxa might have been involved in any changes due to liming, we used chi-square tests to compare the frequency of occurrence of individual acid-sensitive taxa in limed streams before and after treatment. We also assessed the cumulative percentage of the total pool of species that occurred in all stream types through time. Measures of the abundance and richness of acid sensitive taxa in limed streams were related to mean and extreme stream chemistry using rank correlation.

Results.

Stream chemistry

Following liming, acid-base status in the limed streams changed substantially while values in acid reference streams remained constant. Mean annual pH increased from 5-5.1 before liming to 6.1-6.2; mean annual aluminium concentrations decreased from 0.15-0.18 to 0.05-0.11 mg L^{-1}, and calcium concentrations increased from 0.8-2.0 to 2.4-4.5 mg L^{-1}. Despite some inter-annual variations, increased pH and calcium concentrations, and reduced aluminium, have persisted since treatment (Fig. 1). Significant interactions between time (pre- and post-liming) and treatment for limed and acid reference streams for each of these determinands revealed clear and highly significant liming effects (Table 2). After liming, calcium concentrations in the limed streams was not significantly different from naturally circumneutral streams, although average pH ($F_{1,16}$, = 41.2, $P < 0.001$) remained significantly lower, and aluminium concentrations were significantly higher ($F_{1,16} = 8.5$, $P < 0.05$) than those in circumneutral streams.

Figure 1. Mean annual chemistry (± SE) between 1985-98 for replicate groups of limed (♦), acid reference (■) and circumneutral streams (▶) at Llyn Brianne, mid Wales. a) pH; b) aluminium (mg L^{-1}); c) calcium (mg L^{-1}). Arrows indicate lime additions.

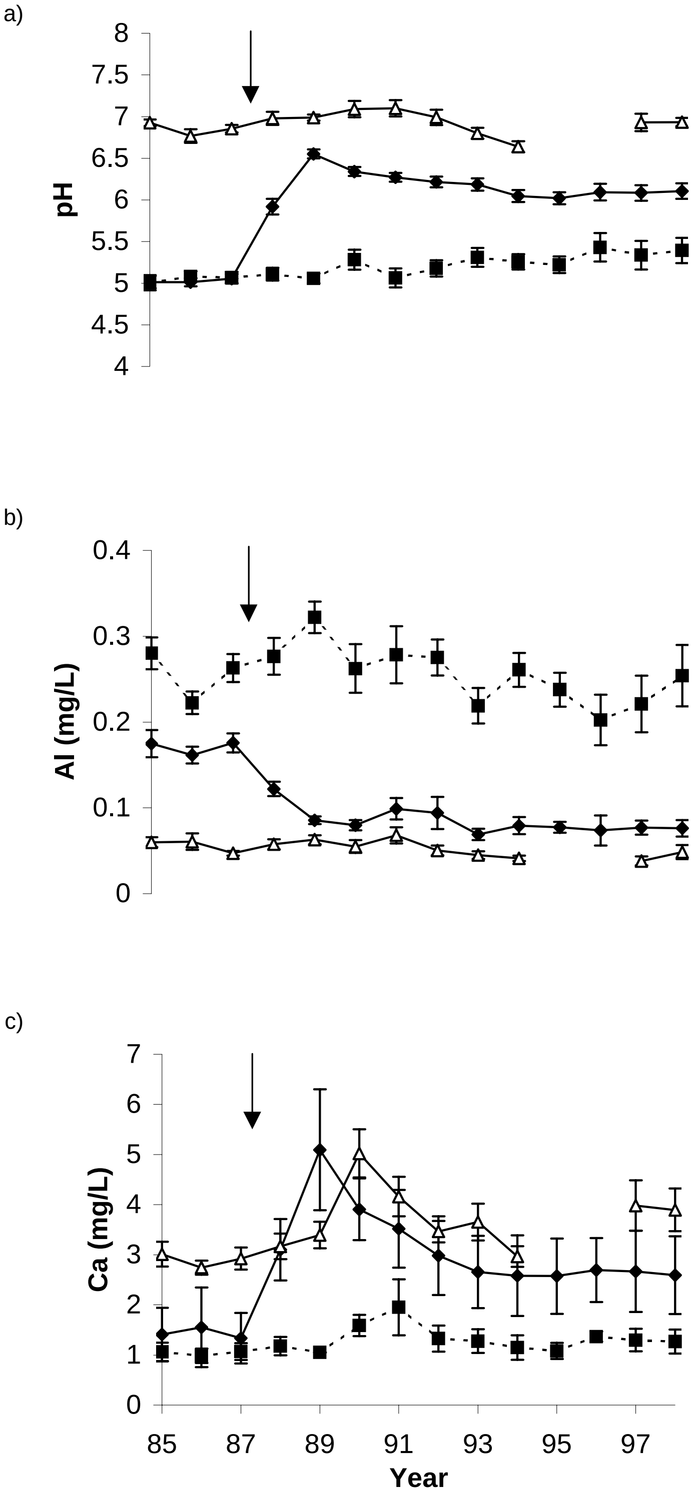


Table 2. Sources of variation in pH, aluminium, and calcium concentrations in streams at Llyn Brianne between 1985 and 1998 assessed using repeated measures ANOVAs (after Paine, 1996). Variations due to treatment (limed vs acid reference streams), times (before and after lime addition), replicate streams (within treatments) and time x treatment interactions were all assessed (See Fig 1 for data). *** = $P < 0.001$

Source of variation	d.f.	pH	Calcium	Aluminium
Treatments	1,55	42.5***	53.6***	81.3***
Replicates within treatment	3,55	7.5***	36.4***	97.0***
Time	1,55	47.4***	46.3***	3.8
Time x treatment interaction	1,55	47.4***	31.0***	14.5***

Macroinvertebrates

Whole community

Although the total abundance of all macroinvertebrates varied significantly through time ($F_{1,64} = 4.33$, $P < 0.05$), changes were contemporaneous in both acid reference and limed streams, and there were no significant treatment effects (time x treatment interaction $F_{1,64} = 0.03$, N.S.). Nor were there any significant temporal variations from any source on total taxon richness.

Acid-sensitive taxa

Variations among acid sensitive taxa contrasted from the pattern shown by all taxa (Fig. 2). Their abundance and taxon richness in limed streams increased after treatment, but not acid reference streams. Liming effects were apparent from the significant time x treatment interactions (Table 3). For abundance, the effect was short-lived, and significant increases over pre-liming values were confined to the limed streams and to the immediate post liming period (Fig. 2; one way ANOVA with Tukey range tests, $P < 0.05$). Moreover, averaged across post-liming years, the abundance of acid sensitive taxa in the limed streams in any year never reached more than 10% (± 9% 95% C.I., $n = 10$ years) of their abundance in naturally circumneutral streams.

For the richness of acid sensitive taxa, the effects of liming were more sustained, increasing significantly from pre-treatment years into both the immediate post-treatment period and also the remaining years of the study (Fig. 2; one way ANOVA with Tukey range tests, $P < 0.05$). Nevertheless, as with abundance, the absolute magnitude of the effect has been modest; limed streams in any one year held on average only 2.7-3.5 acid sensitive species, or 31% (± 8% 95% C.I., $n = 10$ years) of their richness measured contemporaneously in circumneutral streams. Of all eighteen taxa identified as acid-sensitive, only the mayfly *Baetis rhodani* (Picteti) ($\chi^2_1 = 4.034$, $P = <0.05$) and the stonefly *Brachyptera risi* (Morton) ($\chi^2_1 = 5.781$, $P = <0.05$) occurred significantly more often in post-liming samples than in pre-liming samples in limed streams. There were no similar changes between pre- and post-liming years for any taxa in the reference streams.

Despite the apparently modest response in overall richness to liming, nearly 80% of the acid-sensitive taxa at Llyn Brianne occurred in the limed streams on at least one occasion, and 60% occurred twice or more following treatment. These values were intermediate between corresponding values for acid reference streams and circumneutral streams (Fig. 3a, b). Acid-sensitive taxa occurred on three or more occasions in limed streams at rates barely discernable from acid reference streams (Fig 3c). Acid-sensitive taxa therefore appeared to have dispersed sporadically into both limed and acid streams, but not persisted. Their abundance and richness in limed streams was significantly reduced in winters with low minimum pH across all years of study, and for richness this effect was also in post-liming years alone (Fig. 4a; b). Richness was significantly lower in years with a large pH range in winter (Fig. 4c). Periods of acid discharge might therefore have removed acid-sensitive taxa even after liming.

Discussion

One of the clearest positive results to emerge from this study was in the duration over which single lime applications to whole catchments or hydrological sources at Llyn Brianne were effective in raising stream pH (Fig. 1). Where logistics or conservation constraints permit, such catchment applications are widely favoured over other liming methods to combat acidification. Postulated advantages are that even the most upstream headwaters then receive base-rich runoff, while the technology is more straightforward than direct lime application to the stream (e.g. Hindar *et al.* 1996). More subtle advantages arise because the dissolution of $CaCO_3$ is enhanced at the higher pCO_2 in soil waters over stream waters, while potentially toxic aluminium is also retained in catchment soils or wetlands (Cirmo & Driscoll, 1996). Our results reveal the latter effect very clearly (Fig. 1). To add to these advantages, we can now confirm predictions made elsewhere that even single lime applications can have effects at upland British sites that endure for over a decade (Dalziel, Wilson & Proctor, 1994). Catchment budgets provide an indication of why liming effects have been so long-lasting (Ormerod & Edwards, 1985): discharge from streams at Llyn Brianne, after evapotranspiration of incident precipitation, is in the region of 13600 m^3 ha^{-1} $year^{-1}$ so that background catchment losses of calcium prior to liming at concentrations of 1.3 mg L^{-1} would have been around 17.6 kg ha^{-1} $year^{-1}$. At calcium concentrations after liming that were 3.8 mg L^{-1} higher than before liming (Fig. 1), catchment losses of calcium must have increased initially on average by 51.7 kg ha^{-1} $year^{-1}$. Concentrations steadily fell thereafter to 1.2 mg Ca L^{-1} above background, giving calcium losses of around 16.3 kg ha^{-1} $year^{-1}$ during the period from 1993-1998. Total post-liming losses of additional calcium between 1988-1998 (i.e. not accounted for in background concentrations) must therefore have been in the region of 7.6 t in LI4, 7.9t in CI5 and 14.1t in CI2, or respectively 11, 26 and 73% of the amounts of calcium initially applied. These loss rates are slower than those often assumed in agricultural lime applications (1 t lime ha^{-1} $year^{-1}$, or 240 kg Ca ha^{-1} $year^{-1}$; Ormerod & Edwards, 1985). This difference probably reflects applications at Llyn Brianne that were largely targeted to wetland areas. Additionally, agricultural applications usually involve repeat lime applications on a 5-yearly cycle, rather than single applications after which calcium loss rates decline through time.

The long-term nature of the effects of liming at Llyn Brianne appears clear with respect to mean stream chemistry. By contrast, the postulated advantages of catchment liming in protecting streams from acid episodes was not evident from our data (see Fig. 4). Continued episodes or periods of low pH can continue after both direct stream liming (Menendez, Clayton & Zurbuch, 1996) and watershed liming

(Miller *et al.* 1995; Newton *et al.* 1996), with several corollaries. First, episodes represent one of the strongest explanations for limited biological response to liming in this study (see below). Second, we recommend that assessments of the occurrence of episodes, with chemical sampling regimes of suitable frequency and duration, should form part of any post-liming evaluation. Third, there may well be a need to increase the intensity of lime applications to offset the risk of episodes (*cf* Miller *et al.* 1995), and this might require a trade-off against the risk of damage caused by liming to some terrestrial ecosystems (Hindar *et al.* 1996).

Many studies have reported the biological effects of liming over relatively short timescales (e.g. Diamond *et al.* 1992; Fjellheim & Raddum, 1992; Rundle, Weatherley & Ormerod, 1995; Clayton & Menendez, 1996), with little known about the longer-term ecological effects. Although empirical models have allowed comparisons of the likely long-term outcomes of reduced deposition (Ormerod *et al.* 1988) and catchment liming (Ormerod, *et al.* 1990), their applications to real data have revealed some limitations of this approach (Weatherley & Ormerod, 1992; Rundle, Weatherley & Ormerod, 1995). Simmons & Doyle (1996) recommended that adequate assessment of the effects of stream liming requires data from 10 years or more post-liming, closer to the length of our data set. Our results indicate that, whereas the effects of catchment liming on acid-base status were substantial and prolonged, effects on invertebrates were modest. First, increases in abundance in limed streams were confined to acid sensitive taxa and were short-lived. Second, liming added on average only 2-3 acid sensitive species in any year to the community, or roughly one-third of their richness in adjacent circumneutral streams. Third, frequent occurrences (> 3 occasions) by acid-sensitive taxa in limed streams were barely more common in limed streams than in acid streams (Fig. 3c). Lastly, significant changes in frequency of occurrence were confined to just two taxa. Interestingly, one of those two, *Baetis rhodani*, also rapidly colonized limed Norwegian sites studied by Fjellheim & Raddum (1995). Members of this genus were also among the few taxa responding positively to lime additions in the southern Appalachians (Eggleton, Morgan & Pennington. 1996). *Baetis rhodani* is among the clearest indicators of acid conditions in the UK, and was one of the earliest species to respond to lime additions at Llyn Brianne (e.g. Weatherley & Ormerod 1987; 1992).

Also of interest, *B. rhodani* is one the species known from experimental evidence to be highly sensitive to acid episodes (Ormerod *et al.*, 1987).

Large and contiguous areas of the British uplands have been affected by acid deposition, so that geographical isolation might restrict biological recovery even in instances where streams recover chemically. In turn, poor dispersal from the nearest sources of acid-sensitive taxa has been suggested as a factor possibly limiting the colonisation of limed streams (Weatherley, 1988; Weatherley & Ormerod, 1992; Rundle, Weatherley & Ormerod, 1995). This possibility is being tested directly using methods developed elsewhere (Petersen *et al.* 1999; Z. Masters *et al.* unpubl. data). However 14 out of the 18 recognised acid-sensitive taxa at Llyn Brianne have colonised limed streams at some stage during the 10 years after treatment. Those acid-sensitive species that did not occur at least once in limed streams - such as *Glossosoma conformis* Nebois, *Paraleptophlebia submarginata* (Stephens) and *Philopotamus montanus* (Don.) - were scarce even in circumneutral streams. While all of the dynamics involved in dispersal and recolonisation of streams are far from clearly understood, one inference from our data is that the persistence of acid-sensitive taxa in limed streams presents greater problems than pure dispersal.

With food quality sufficient in the limed streams suitable to allow the survival of acid-sensitive species, and habitat physiography not atypical for them (Z. Masters *et al.* unpubl. data), one of the major questions from this work is about why persistence has been so limited. One possibility apparent from this study is that acid episodes continued to affect the fauna in limed streams even after treatment. Our evidence in this respect is two fold. Firstly, five of the 10 years monitored post-liming had pH minima that fell below pH 5.5. This is sufficient at least for short-term lethal and sub-lethal effects on some acid-sensitive species (Ormerod *et al.* 1987). Secondly, the abundance and richness of acid-sensitive taxa in the limed streams at Llyn Brianne was lowest in years when pH minima were also low and pH ranges were highest. Thus, while post-liming pH minima were not as low as those pre-liming, continued acid episodes explain significant variation among the acid-sensitive invertebrate fauna. Previous work at Llyn Brianne suggests that not only the severity of episodes affect acid-sensitive taxa, but also their frequency, and possibly their duration (Ormerod et al. 1987; Merrett *et al.* 1991; Rundle, Weatherley & Ormerod, 1995).

Further experimental work involving transplantations is now in progress at Llyn Brianne to assess survival under base-flow and storm-flow conditions in limed streams.

The scale and nature of background variation within invertebrate communities can affect the power of experimental manipulations to detect effects, but the consequences of such variation for assessing the effects of liming has never been considered closely (e.g. Gunn *et al*. 1988; Weatherley & Ormerod, 1992; Rundle, Weatherley & Ormerod, 1995). Recent work at Llyn Brianne has shown marked inter-annual variation in invertebrate persistence and stability occur in all stream types and are linked in turn to cyclical climatic variations (D. Bradley & S. J. Ormerod unpubl. data). Periods of low persistence and stability occurred across all the study sites in 1987-1993, and closely followed lime additions in 1987-88. In the early post-liming years, these patterns might well have confounded the experimental outcomes. However, these results over 10 years show that liming did not substantially change invertebrate communities in limed streams even in apparently stable periods when invertebrate persistence was high from 1993-94 onwards. These are additional arguments for extending the time-periods over which the outcomes of large-scale ecological experiments are assessed – not only incorporating sufficient pre-treatment years, but also incorporating long-term environmental cycles.

In total, these data reveal that, at the community level, hysteresis in recovery has been a real feature in the biological response to de-acidification at Llyn Brianne. However, rather than reflecting problems of dispersal, lags appear to be explained more by continued problems associated with acid episodes.

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Table 3. Sources of variation in the taxon-richness and abundance of acid-sensitive taxa in streams at Llyn Brianne between 1985 and 1998 assessed using repeated measures ANOVAs (after Paine, 1996). Conventions as in Table 2 except * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$. (See Fig. 2 for data.)

Source of variation	d.f.	Taxon richness	Abundance
Treatments	1,64	0.0	0.5
Replicates within treatment	4,64	4.4**	6.0***
Time	1,64	4.9*	3.4
Time x treatment interaction	1,64	6.3*	4.5*

Figure 2. a) Mean abundance and b) mean taxon richness ± SE of eighteen acid-sensitive macroinvertebrate taxa between 1985-98 in replicate groups of limed, acid reference and circumneutral streams at Llyn Brianne, mid Wales. Dashed line represents missing data for 1991. Arrows indicate lime additions.

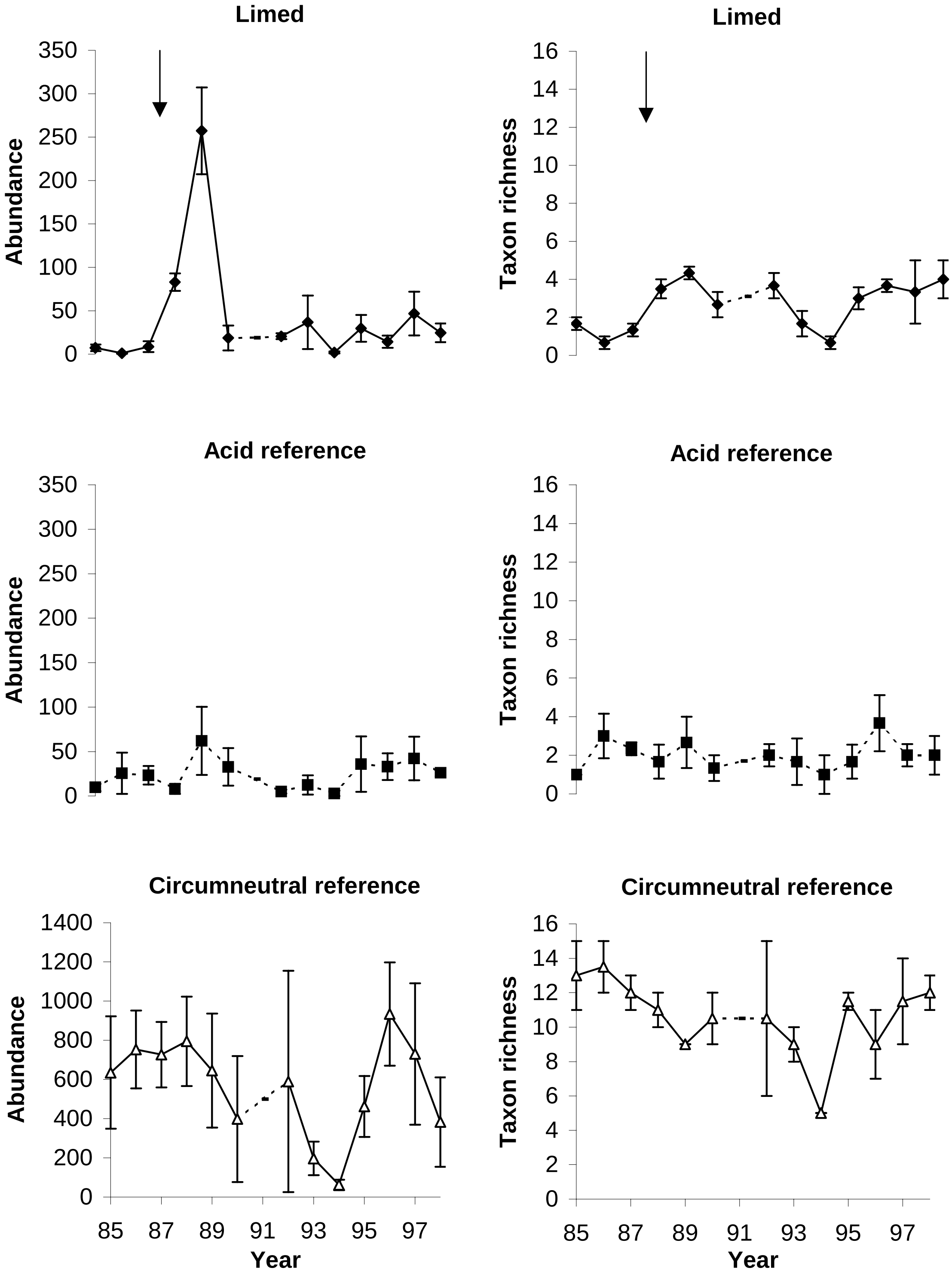


Figure 3. The cumulative percentage of 18 acid-sensitive taxa that have occurred on at least 1 (a), 2 (b) and 3 (c) occasions in circumneutral (▶), limed (♦) and acid reference (■)streams with progression through the post-liming years at Llyn Brianne, mid Wales.

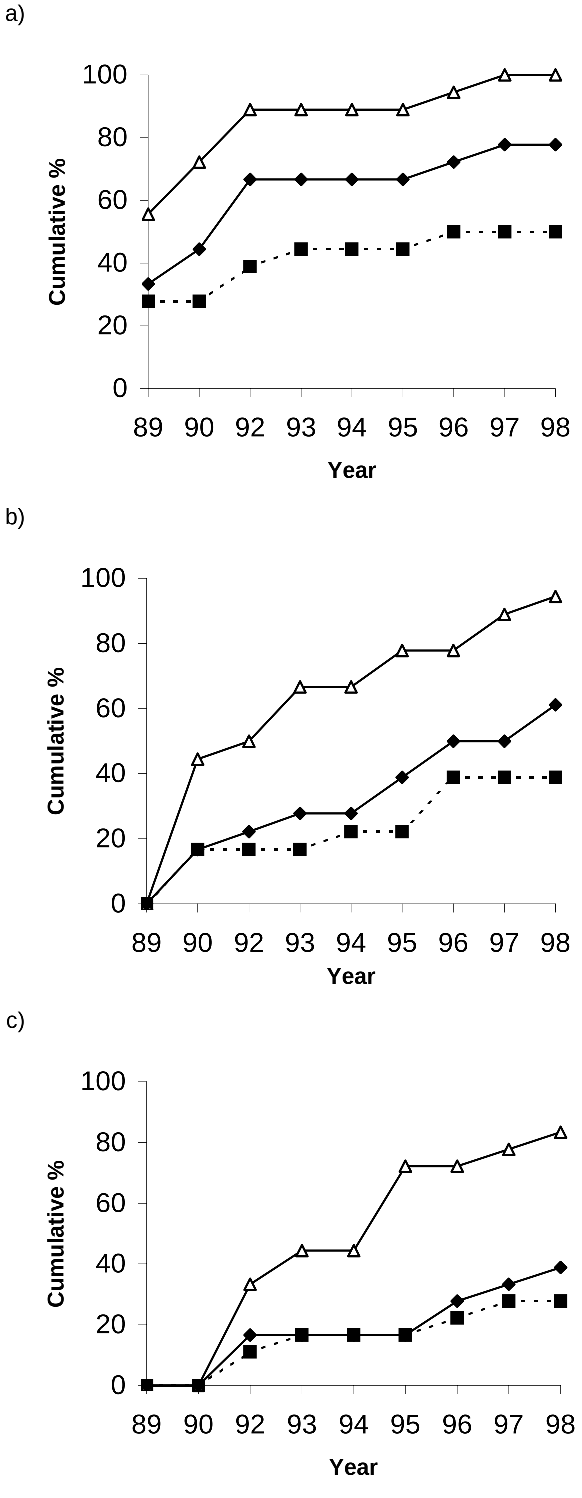
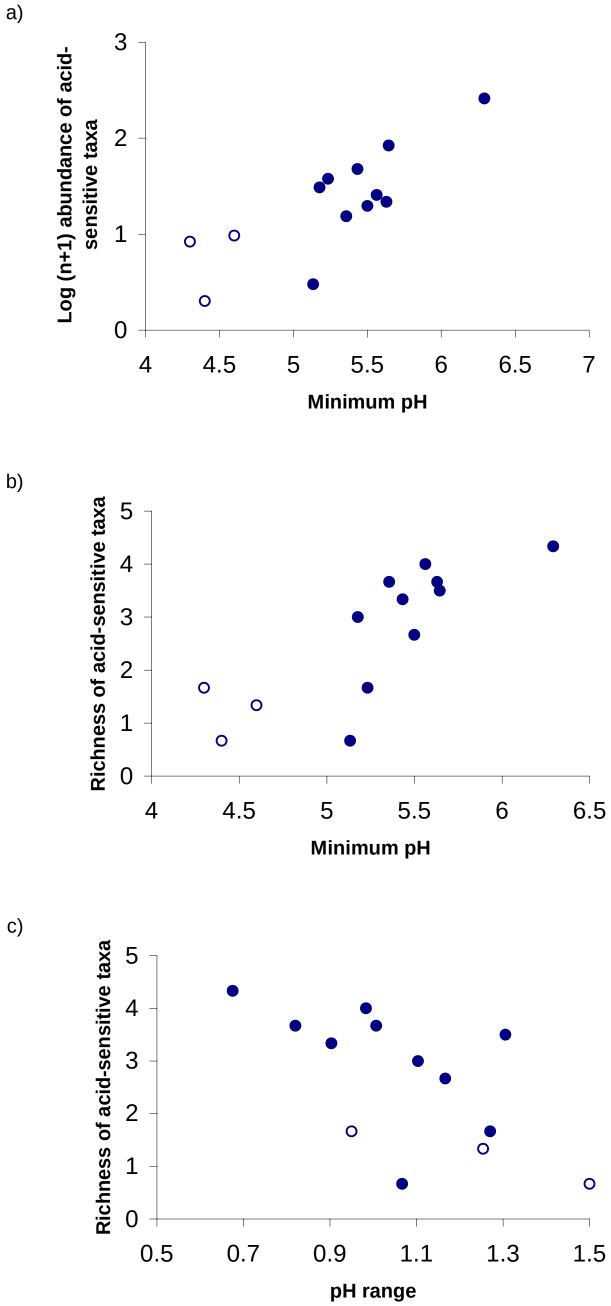


Figure 4. Relationships between the abundance (log 10 numbers of individuals per sample) or richness of acid-sensitive taxa and winter pH in limed streams for pre- (O) and post-liming (•) years at Llyn Brianne, mid-Wales. a) The abundance of acid-sensitive invertebrates and minimum winter pH in limed streams (all years $r_s = 0.76$, $P < 0.01$, n = 13; post-liming years only $r_s = 0.54$, *N.S.*, n = 10); b) the taxon richness of acid-sensitive invertebrates and minimum winter pH (all years $r_s = 0.83$, $P < 0.001$, n = 13; post-liming years only $r_s = 0.74$, $P < 0.05$ n = 10); c) the taxon richness of acid-sensitive invertebrates and winter pH range (all years $r_s = -0.60$, $P < 0.05$, n = 13; post-liming years only y = $r_s = -0.65$, $P < 0.05$, n = 10). All richness and abundance values are means across limed streams.



Task 2.7.2: Evaluating variations in invertebrate communities in upland Welsh streams: persistence tracks the North Atlantic Oscillation

D.C. Bradley & S.J. Ormerod

Summary

1. This work began as an evaluation of apparently stochastic variations through time in invertebrate communities in upland Welsh streams. Large variations between years in the persistence (= constancy) of invertebrate communities was apparent during the Welsh Acid Waters Survey, and was seen as a factor that might confound or obscure other trends. As a result of further work, these apparently stochastic variations are now understandable.

2. The North Atlantic Oscillation (NAO) affects marine ecosystems, lakes and some terrestrial ecosystems around the Atlantic rim, but evidence for effects on rivers is scarce. For 14 years, we sampled riverine macroinvertebrates in 8 independent streams from the Llyn Brianne experimental catchments in central Wales. We assessed whether year-to-year persistence in rank abundance and species composition tracked the NAO.

3. Persistence was quasi-cyclical and varied synchronously across all streams irrespective of chemistry and catchment land use. Invertebrate communities in all stream types or habitats switched significantly from being highly persistent during negative phases of the NAO (winter index <1= cold, dry winters) to unstable during positive phases (>1= mild, wet winters). These effects occurred in both rare and common taxa.

4. Acid episodes could not explain low persistence in positive NAO years because variations in acid-base status were not linked to the NAO. Moreover, fluctuations in persistence were apparent even in well-buffered streams.

5. Discharge in adjacent gauged catchments increased in positive NAO years by 15-18% but neither flow variability nor flow maxima were higher. Nor were variations in invertebrate persistence at Llyn Brianne directly correlated with discharge pattern. Discharge variations alone were therefore insufficient to explain links between persistence and the NAO, but we cannot exclude subtle effects due either to flow or temperature.

6. These data illustrate how the persistence of invertebrate communities at Llyn Brianne is not entirely stochastic, instead varying with environmental fluctuations. Positive phases

of the NAO are accompanied by ecological instability in the Llyn Brianne streams, although the exact mechanisms are currently unclear. The effects of the NAO might confound or obscure other long-term change in rivers such as recovery from acidification or the effects of global warming unless accounted for in monitoring programmes. These patterns, if increase in the amplitude of the NAO continues, might cause further instability in aquatic habitats.

Introduction

The influences of medium-range climatic cycles on marine and continental ecosystems are increasingly well described (Dayton *et al*. 1992; Harrison 2000; Holmgren *et al*. 2001). By contrast with the El Niño Southern Oscillation (ENSO), however, there is less evidence for ecological patterns that reflect its Atlantic analogue - the North Atlantic Oscillation (NAO). In terrestrial ecosystems, the NAO has been linked to the phenology of birds or plants (Forchhammer, Post & Stenseth 1998; Post & Stenseth 1999; Przybylo, Sheldon & Merila 2000), the survival and demography of large mammals (Milner, Elston & Albon 1999) and the effects of northern snow cover on predator-prey interactions (Post *et al*. 1999a; 1999b). In marine systems, planktonic communities reflect year-to-year pattern in temperature, nutrient upwelling or fluctuating salinity driven by the NAO (Fromentin & Planque 1996; Nehring 1998; Lindahl *et al*. 1998; Planque & Taylor 1998; Belgrano, Lindahl, & Hernroth 1999; Hagberg & Tunberg 2000; Hanninen, Vuorinen & Hjelt 2000). In lakes, the NAO affects plankton through temperature, wind-induced mixing, ice cover and altered food web dynamics (George & Harris 1985; Straile & Geller 1998; George & Hewitt 1999; George 2000; Straile 2000). Parallel and synchronous changes across large areas show that such effects are probably widespread (Straile & Adrian 2000). So far, however, and in contrast to lakes, few studies have assessed whether the NAO affects rivers and river organisms (Elliott, Hurley & Maberly, 2000).

As the alternation of differences in atmospheric sea-level pressure (SLP) between the Azores and Stykkisholmur (Iceland), the NAO drives complex weather patterns over a cycle that is now roughly decadal and of increased intensity since the mid 1970s (Hurrell 1995; Hurrell & van Loon 1997). Hurrell’s (1995) NAO index describes these pressure fluctuations through time, negative values being accompanied by cold, dry and calm winters in NW Europe, whereas positive values are accompanied by milder winters, strong westerly winds and rainfall up to 30 percent above annual average (Hurrell 1995). With variations in precipitation and temperature so marked in the NAO, knock-on effects on river ecosystems might be expected. First, variations in river temperature closely follow those of air temperature (Mohseni & Stefan 1999). Second, for some regions rainfall variations in the NAO influence annual river discharge (Cullen & de Menocal 2000; Hanninen, Vuorinen & Hjelt 2000). Variations in discharge, in turn, can drive ecologically important changes in rivers such as floods, droughts and the dilution or mobilization of important ions (Reynolds, Emmett & Woods 1992; House & Warwick 1998; Jarvie, Whitton & Neal 1998; Bishop, Laudon & Kohler 2000). In the case of the ENSO, effects like these influence river organisms (Puckridge, Walker & Costelloe 2000; Mol *et al*. 2000).

In this part of the report, we assess year to year changes among aquatic macroinvertebrates over 14 years in 8 independent streams in western Britain in relation to the NAO. Following previous studies at the same sites and elsewhere, we focus on measures of persistence as a measure of stability or environmental constancy (Townsend, Hildrew & Schofield 1987; Weatherley & Ormerod 1990a; Holomuzki & Biggs 2000). We defined persistence as a characteristic of the whole community revealed by the relative constancy in the rank abundance pattern and composition of assemblages through time (Holling 1973; Connell & Sousa, 1983). Elsewhere in the report, we have assessed long-term trends at the same sites in invertebrate abundance and richness.

STUDY AREA

All the streams were in the experimental catchment area around Llyn Brianne reservoir, mid-Wales (52° 8'N 3° 45'W), previously described by Stoner, Gee & Wade (1984) and Weatherley & Ormerod (1987). The streams are all second or third order, with catchments at altitudes of 215-410m and of 15-264 ha, consisting either of upland sheep-pasture or plantations of sitka spruce (*Picea sitchensis* Carriere) and lodgepole pine (*Pinus contorta* Douglas; Rutt, Weatherley & Ormerod 1989; Table 1). The underlying Ordovician and Silurian shales, mudstones and grits are base-poor with low buffering capacity. Overlying brown podzolic soils, stagnopodzols and peats provide limited buffering so that runoff is soft (mean total hardness 3.9-18.8 mg $CaCO_3$ l^{-1}) and in some cases acidified (mean pH 4.6-6.9; Rutt, Weatherley & Ormerod 1989). The streams form groups classifiable as acid conifer (LI1, LI2, LI8), acid moorland (LI5, CI1, CI4) and circumneutral moorland (LI6, LI7) between which faunal communities differ substantially (Weatherley & Ormerod 1987; Table 1). Typically, the circumneutral moorland streams are species rich, and characterized by ephemeropterans, trichopterans and plecopterans whereas acid forest streams are species poor and dominated by acid-tolerant plecopterans. Acid moorland streams are intermediate between these extremes. The catchments of three further study streams (LI4, CI2 and CI5) were artificially limed over a decade ago, but these are not considered further here (Weatherley & Ormerod 1990a; Bradley & Ormerod *in press*).

The climate at Llyn Brianne is temperate, with mean stream temperatures rarely outside the range of 0-20 °C, and mean annual rainfall c. 1900 mm (Weatherley & Ormerod 1990a, b). Stream substrata vary between gravel (> 2-16 mm) and bedrock, with bryophytes the only submerged macroflora; in marginal areas, *Juncus* spp. and *Sphagnum* spp. form the most abundant vegetation (Rutt, Weatherley & Ormerod 1989; Weatherley & Ormerod 1990a).

Methods

MACROINVERTEBRATE SAMPLING

All streams were kick-sampled for invertebrates each April from 1985 to 1998, excepting 1991 (a year without funding), using a standard net (0.9mm mesh; 230 x 255mm frame). At each site a two-minute sample was taken from mid-channel riffles, and a one-minute sample taken from stream margins by a combination of kicking and sweeping upstream in marginal vegetation (Rutt, Weatherley & Ormerod 1989). This sampling strategy was derived following extensive quantitative sampling throughout several years and reflects communities for the entire winter-spring periods; paired samples in margins and riffles record in excess of 75-85% of the species present in any stream at a given time (Weatherley, Rutt & Ormerod 1989 and unpublished data). Samples were preserved in 70% alcohol on site prior to sorting and identification in the laboratory to the lowest feasible taxonomic level. This involved species except for Oligochaeta, Chironomidae, Simuliidae, Tipulidae, Dixidae, and the early instars of other groups, which were identified to the generic or family level. When there was any doubt about the correct identification of a species we combined all possible species at the generic or family level for the purposes of analysis.

PHYSICOCHEMICAL VARIABLES

Aluminum (mg/l), calcium (mg/l), total hardness (mg/l) and pH were monitored weekly or fortnightly from 6 January 1984 to 24 November 1998 as measures of acid-base status. Weatherley & Ormerod (1987) detail the analytical procedures involved. The data provided mean, maximum, minimum, and the range for each determinand in each stream, either for 14 whole years, for summer (April to September), or for winter (October to March). We made this seasonal separation as the climatic influence of the NAO is highly seasonal (Hurrell 1995). Annual values were calculated from April to the end of March the following year thereby describing conditions antecedent to invertebrate sampling in early April. Continuous hydrological data were not available for the Llyn Brianne catchments, but measures of discharge (cumecs) were made continuously in two directly adjacent streams, the Afon Hafren and Afon Gwy, at the Centre for Ecology and Hydrology Plynlimon experimental site. These streams are second and third order, at similar altitudes to the Llyn Brianne streams and are less than 40km directly to the north. Inter-calibration of sites across Wales indicates that generally similar temporal patterns of discharge occur in streams throughout this region (Stevens *et al.* 1997). Therefore, it can be expected that the Hafren and Gwy will show similar year-to-year discharge regimes to the Llyn Brianne streams. These data were arranged seasonally exactly as the chemical data.

Data for the North Atlantic Oscillation (NAO) index were obtained on-line (http://www.cgd.ucar.edu:80/cas/climind/nao.html source: Hurrell 1995). The NAO index is calculated from the difference of normalised sea level pressures (SLP) between the Azores and Iceland (Hurrell 1995; Belgrano, Lindahl, & Hernroth 1999). The NAO indices were arranged as yearly means and winter (December-March) means because the NAO influences precipitation and temperature largely during the winter (Hurrell 1995; Fromentin & Planque 1996; Forchhammer, Post & Stenseth 1998; Lindahl *et al.* 1998; Planque & Taylor 1998; Belgrano, Lindahl, & Hernroth 1999).

DATA ANALYSIS

Following standard methods (Townsend Hildrew & Schofield. 1987; Weatherley & Ormerod 1990a), invertebrate persistence was measured within streams in two ways. First, to reflect patterns in relative abundance between pairs of successive years, we determined

Spearman's rank correlation coefficients between the abundances of all species in the two communities being assessed. Values range from –1 indicating low persistence to +1 indicating high persistence. Second, persistence in species composition was calculated as the Jaccard's coefficient of similarity (J; see Hellawell 1978):

$$J\,(AB) = j/(a+b-j)$$

where a = number of taxa in community A (first year in any pair), b = number of taxa in community B (the second year in any pair) and j = number of taxa found in both communities. Values range from 0 (no similarity, low persistence) to 1 (identical similarity, high persistence). In both cases (Jaccard and Spearman's ranks), values were determined for the total stream community and also for riffle (R) and margin (M) samples separately. We calculated persistence for the 15 most abundant taxa overall, and for the remaining 'rare' taxa in each community to assess whether the latter were particularly sensitive to change. In addition to pairwise comparisons between successive years within streams, we determined persistence measurements for year pairs of gradually larger and larger intervals to assess any long-term drift (i.e. year pairings of 1, 2, 3 and up to 14 years apart), but none was found.

Measures of persistence were related using Pearson product-moment correlation to the NAO, to chemical variables and to discharge in the antecedant winter (see Townsend, Hildrew & Schofield 1987; Weatherley & Ormerod 1987). We also compared measures of persistence between years with positive (> 1) and negative (< 1) NAO winter indices using one-way Analysis of Variance (ANOVA). Prior to these analyses, chemical variables and taxon abundances were log-transformed and Spearman's rank correlation coefficients were *z*-transformed to normalise variances (Weatherley & Ormerod 1990a).

Results

PATTERNS IN PERSISTENCE

Taxon abundance, taxon richness and measures of persistence varied from year-to-year in all streams. Mean total invertebrate abundance in any year ranged from 64 (± 18 S.E.

individuals per sample) to 822 ± 99, while mean richness across streams ranged from 8 ± 1 to 26 ± 2. Average persistence in rank abundance of total faunal community for each stream between successive years ranged from 0.70 ± 0.10, indicating high persistence, to 0.02 ± 0.40, indicating low persistence. Similarly, Jaccard's coefficients ranged from 0.74 ± 0.08, indicating similar taxonomic composition between years, to 0.27 ± 0.08, indicating low similarity.

These variations were not random, but instead appeared to be quasi-cyclical (Fig. 1). Variations occurred synchronously across all streams so that persistence in rank abundance (all $r > 0.7$, $P < 0.01$) and Jaccard's coefficients (all $r > 0.9$, $P < 0.01$) were significantly inter-correlated between the three stream groups (acid forest, acid moorland, circumneutral moorland). Variations also occurred synchronously for abundant and rare taxa (Fig. 2), and for both stream habitats sampled (Fig. 3). Together, these data illustrate that large-magnitude variations in persistence were a whole-stream and whole-community feature of the long-term trends at Llyn Brianne over the study period, and characterised all stream types.

RELATIONSHIPS WITH THE NAO

The NAO characteristically fluctuates between extreme positive and negative values (Hurrell & van Loon 1997; George & Hewitt 1999), and such fluctuations were apparent during the 14 years of this study. Moreover, concordance between the phases of the NAO and patterns of invertebrate persistence were striking (Fig. 4). For invertebrate communities in all stream types, all habitats, and of either rare or common occurrence, index values were significantly higher following negative phases of the winter NAO (<1) than following positive NAO values (>1) (Table 2). In the case of rare taxa in all streams, and all taxa in circumneutral streams, stability and persistence declined linearly with increasing values of the NAO winter index (Table 2).

Direct measures of total invertebrate abundance or taxon richness were unrelated to the NAO. However, four species declined significantly with the NAO winter index (*Nemurella picteti* Klapalek, $r = -0.793$, $P < 0.01$; *Elmis aenea* Muller $r = -0.575$, $P < 0.05$; *Hydropsyche siltalai* Dohler $r = -0.830$, $P < 0.01$; *Paraleptophlebia submarginata* Stephens $r = -0.944$, $P < 0.001$) whereas *Chloroperla tripunctata* Scopoli increased ($r = 0.834$, $P = 0.001$).

Possible mechanisms through which the NAO might affect invertebrate stability at Llyn Brianne were unclear from the data collected. The annual average discharge of the Hafren (r= 0.645, $P < 0.05$) and Gwy (r= 0.56, $P < 0.05$), and the average winter discharge of the Hafren (r= 0.57, $P < 0.05$) increased during positive phases of the NAO winter index by 15-18%. However, discharge ranges, discharge variability (as coefficients of variation) and discharge maxima were not significantly different in positive NAO years from other years. More directly, there were no significant correlations between discharge in the Gwy or Hafren and any measure of persistence in the Llyn Brianne streams. Nor were there any consistent relationships between any measures of acid-base status (pH, calcium or aluminium concentrations) and either the NAO or measures of persistence.

Discussion

Several studies have shown that environmental factors act across catchments to influence the persistence of river invertebrates (Townsend, Hildrew & Schofield 1987; McElravy, Lamberti & Resh 1989). Weatherley & Ormerod (1990a) observed such supra-catchment effects at the sites in this study from 6 years' of data, but at the time no explanation was available. During the WAWS (Stevens *et al.* 1997), these fluctuations were re-evaluated, and thought to be stochastic. This longer run of 14 years' data shows not only that synchronous fluctuations in persistence have continued, but also that they are consistent across contrasting catchments, taxa and stream habitats in the Lynn Brianne experimental area. On the basis of correlative evidence, the North Atlantic Oscillation provides a region-wide effect of the type required to explain the observed pattern: low year-to-year persistence in rank abundance and composition in all catchments was associated with positive phases of the NAO winter index while high persistence was associated with negative phases. Although this study is not yet long-enough to link stream invertebrate communities unequivocally to the NAO through several cycles (cf George & Harris 1985; Straile & Geller 1998; George & Hewitt 1999; George 2000; Straile 2000; Straile & Adrian 2000) these data are the first to show that a link might exist. So far, evidence that the ENSO affects river organisms is from fish rather than invertebrates, and implicates droughts rather than floods (Puckridge, Walker & Costelloe 2000; Mol *et al.* 2000). At these Welsh sites, persistence among invertebrates was lower in wet phases of the NAO than in dry phases.

Fluctuations in persistence at Llyn Brianne involved two different ecological effects. Spearman's rank coefficients showed that relative abundances across species were more constant during cooler, drier phases of the NAO, but dissimilar during warmer and wetter phases. In turn, Jaccard's coefficients showed that high and low persistence in rank abundance were accompanied respectively by high and low constancy in species composition. These effects might well be linked - for example if shifts in relative abundance were sufficiently large to reduce some species to undectable levels. We can discount sampling error as an explanation for these trends, which in other long-term studies can explain apparently significant community variation between census years (Arnott *et al.* 1999). First, sampling error would be randomly distributed across samples, years and streams, rather than being responsible for highly synchronous variations across streams and habitats. Second, sampling error among aquatic invertebrates is usually greatest among rare taxa and complex habitats, where sampling is most difficult (Parsons & Norris 1996; Cao, Larsen & Thorne 2001). In our case, these effects were apparent because communities from structurally complex marginal habitats and rare species had the lowest overall persistence (Weatherley & Ormerod 1990a). Nevertheless, year-to-year variation in persistence among rare taxa and margins exactly echoed persistence in common taxa and riffles. We conclude that changes in persistence through time at Llyn Brianne have been real.

Few studies have assessed persistence among river invertebrates, particularly over timescales similar to this study. Although none have linked patterns with climatic cycles, those available provide valuable insight into the relationship between persistence and environmental variation. In general, persistence is greatest where environmental conditions are relatively constant (Robinson, Minshall & Royer 2000) and where taxa are adapted to the prevailing environmental regime (Miller & Golladay 1996). For example, in the United Kingdom, Townsend, Hildrew & Schofield (1987) showed that persistence was greatest where streams had low discharge, constantly low summer temperatures, and pH regimes that were acid and stable. By contrast, persistence is least where environmental conditions fluctuate or are characterized by pulse, press or ramped disturbances (Meffe & Minkley 1987; Lake 2000). Those disturbances known to influence persistence include changes in catchment character – for example the replacement of semi-natural forest by agricultural development (Brewin, Buckton & Ormerod 2000) - or particular catchment-scale events such as pesticide use (Hutchens, Chung & Wallace 1998) and fires (Richards & Minshall 1992). Changes in flow conditions also affect persistence, for example where regimes shift from long-term stable to short-term unstable due to freezing conditions or floods (Matthaei, Uehlinger & Frutiger 1997; Bradt *et al*. 1999). Flow effects like this occur at a variety of scales from the whole stream down to the individual patch (Death 1996; Matthaei & Tonwsend 2000). On all of this evidence, variations in persistence at Llyn Brianne between different climatic phases of the NAO would be consistent with a link between persistence and environmental variability.

With the NAO influencing western European rainfall, and discharge variation in turn affecting the persistence of invertebrates, it might be expected that the effects of flow would be central to our results. No discharge data were available directly from the study sites, although average discharge in rivers just 40 km away was indeed larger in positive NAO years by up to 18%. However, extreme flows were no larger, and discharge was no more variable in positive NAO years than in negative NAO years. Thus, on our evidence low persistence cannot have reflected the effects of pronounced floods (Lake 2000). More important, there were no direct correlations between discharge pattern and indices of persistence in the Llyn Brianne streams. Therefore, direct discharge effects cannot alone be sufficient to explain varying invertebrate persistence. We cannot discount subtle effects since the influences of flow on aquatic invertebrates are many and varied (Hart & Finelli 1999; Lancaster 1999; Holomuzki & Biggs 2000; Doisy & Rabeni 2001). Those species which increased or decreased at Llyn Brianne under different phases of the NAO provide clear examples: local density and net-spinning activity in the caddis *Hydropsyche siltalai* reflects current velocity (Stazner & Bretchko 1998); *Paraleoptophlebia submarginata* is a marginal specialist characteristically occurring in areas of low hydraulic stress (Mobes-Hansen & Waringer 1998); *Elmis aenea* belongs to a group with well characterized preferences on current velocity (Dietrich & Waringer 1999), and *Nemurella picteti* shows clear micro-distributional responses to changing flow conditions (Lancaster & Hildrew 1993). We suggest that further assessment of discharge and flow pattern might well explain how invertebrate communities respond to the NAO.

Variations in discharge not only have direct effects on stream organisms, but also indirect effects through changes in stream chemistry. As Townsend, Hildrew & Schofield (1987) noted, persistence is greater in stable chemical environments. At soft-water sites like those in this study, acid episodes are potentially important responses to flow that result from increased base-cation dilution, and from increased titration effects due to anions mobilised from catchment soils (Bishop, Laudon & Kohler 2000). There is clear experimental evidence that acid episodes in some Llyn Brianne streams affect stream invertebrates (Ormerod *et al.* 1987) and might still offset biological recovery from acidification (Bradley & Ormerod *in press*). However, several lines of evidence show that fluctuations in invertebrate persistence due to the NAO were not related to episodicity. First, some of the most pronounced variations in persistence occurred in circumneutral moorland streams - where buffering is greatest, and pH never falls below 6-6.5. Acid episodes therefore do not offer the region-wide effect necessary to explain our data. Second, no measure of acid-base status across years in any stream type was linked to the NAO. Third, measures of persistence were not correlated with either pH, aluminium or calcium concentration. Other aspects of stream chemistry vary with discharge in ways that might be important ecologically, for example because interactions between droughts, floods and temperature affect the mobilization of nitrogen and phosphorus from catchment soils (Reynolds, Emmett & Woods 1992; House & Warwick 1998; Jarvie, Whitton & Neal 1998). At present we have no data on whether such effects track the NAO, nor on what their influence on stream productivity might be.

In addition to rainfall, NAO cycles are also reflected in temperature variations due to air movements from different sources, and because cloud cover or atmospheric dust affect radiation budgets (Moulin *et al.* 1997, Forchhammer, Post & Stenseth 1998). Such temperature variations affect upland British rivers enough to affect the emergence times of salmonid fish (Elliott, Hurley & Maberly 2000). Thermal regimes, in turn, affect invertebrate persistence (Townsend, Hildrew & Schofield 1987). At Llyn Brianne, variations in stream temperatures between months, years and catchments reflect air temperature and insolation, with variations sufficient to affect the emergence periods of some stream insects (Weatherley & Ormerod 1990b). Effects are subtle, however, and there is so far no evidence that they affect communities. Among the species responding to the NAO at Llyn Brianne, some such as *Chloroperla tripunctata* have clear thermal tolerance, but in this instance increased abundance during warm phases would be contrary to expectation in this cold water species (Elliott 1988). Nevertheless, increased winter temperatures affect the timing of insect lifecycles and oviposition success of adults (Chen & Folt 1996), so that warm phases of the NAO could translate into population performance and hence low measures of persistence between years. As with examination of flow responses to the NAO, we suggest that temperature effects should figure in the search for processes linking variations in river invertebrate communities to the NAO.

OVERVIEW

These results provide further evidence that ecosystems in northern and western Europe are affected by fluctuations in the NAO and its associated climatic effects. In addition to lakes and stream fishes, our data show that the NAO probably affects stream invertebrates. The effects occurred in replicate streams of contrasting chemistry and catchment land-use and might therefore be widespread.

Pronounced variations in invertebrate persistence that follow that NAO have both fundamental and applied importance. In fundamental terms, these data not only confirm previous ideas that persistence in invertebrate communities reflects environmental variability, but show also that persistence can vary through time within the same river system: constancy is not a fixed property of a given location. In applied terms, our data confirm Weatherley & Ormerod’s (1990a) view that variations in aquatic communities risk confounding or obscuring the effects of other long-term trends such as recovery from acidification, eutrophication or the effects of climatic change (Chen & Folt 1996; Lancaster *et al*. 1996; Lawlor *et al*. 1998). We recommend that researchers designing long-term monitoring programmes consider such effects carefully.

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Table 1 Catchment characteristics and chemistry for the study streams at Llyn Brianne. Chemical determinands are long-term annual means from 1985-1998 (adapted from Rutt, Weatherley & Ormerod 1989).

Site code	Catchment land use	Catchment area (ha)	Mean pH	Total Hardness (mg $CaCO_3$ l^{-1})	Filterable Aluminium (mg l^{-1})
LI1	Acid stream in conifer forest (c. 40 years old)	264	4.9	7.0	0.40
LI2	Acid stream in conifer forest (c. 40 years old)	194	4.9	6.9	0.43
LI8	Acid stream in conifer forest (c. 25 years old)	112	5.4	7.9	0.23
LI5	Acid stream in moorland	66	6.0	8.5	0.04
CI1	Acid stream in moorland	15	5.2	3.9	0.10
CI4	Acid stream in moorland	71	5.5	5.4	0.12
LI6	Circumneutral stream in moorland	82	6.9	15.7	0.05
LI7	Circumneutral stream in moorland	72	6.9	18.8	0.04

Table 2 Variations between values of persistence or stability among invertebrates at Llyn Brianne in years between 1984 and 1998 charactersied by either positive (NAO > 1) or negative (NAO < 1) values of the NAO winter index.

a) Jaccard Coefficient

	NAO < 1	NAO > 1	$F_{1,9}$
All streams:			
All taxa	0.64	0.38	25.18***
Abundant taxa	0.89	0.63	22.46***
Rara taxa	0.38	0.14	23.69***
Riffles only	0.62	0.36	23.42***
Margins only	0.54	0.34	12.37**
Circumneutral streams	0.64	0.38	34.42***
Acid moorland streams	0.57	0.35	10.88**
Acid forest streams	0.64	0.34	19.64**

b) Spearman’s rank

	NAO < 1	NAO > 1	$F_{1,9}$
All streams:			
All taxa	0.61	0.24	26.16***
Abundant taxa	0.56	0.34	12.37**
Rara taxa	0.02	-0.42	20.73***
Riffles only	0.60	0.23	28.73***
Margins only	0.40	-0.04	27.24**
Circumneutral streams	0.75	0.31	38.50***
Acid moorland streams	0.58	0.10	11.89**
Acid forest streams	0.68	0.15	15.17**

Table 3 Pearson product moment correlations between persistence, stability and the NAO winter index for all streams for individual stream types at Llyn Brianne between 1984 and 1998 (*P= <0.05).

	Persistence (all taxa)	Stability (all taxa)	Persistence (rare taxa)	Stability (rare taxa)
All streams	-0.56	-0.56	-0.64*	-0.62*
Acid forest streams	-0.58	-0.53	-0.64*	-0.59
Acid moorland streams	-0.52	-0.49	-0.39	-0.30
Circumneutral moorland streams	-0.63*	-0.63*	-0.61*	-0.66*

Figure 1.Year-to-year trends in community persistence in streams at the Llyn Brianne experimental catchments between 1985-1998. The values are means for each stream group (see Table 1) and show a) Spearman's rank correlation on rank abundance across species (r) and b) similarity in taxonomic composition (J). The dashed lines represent missing data interpolated for 1991.

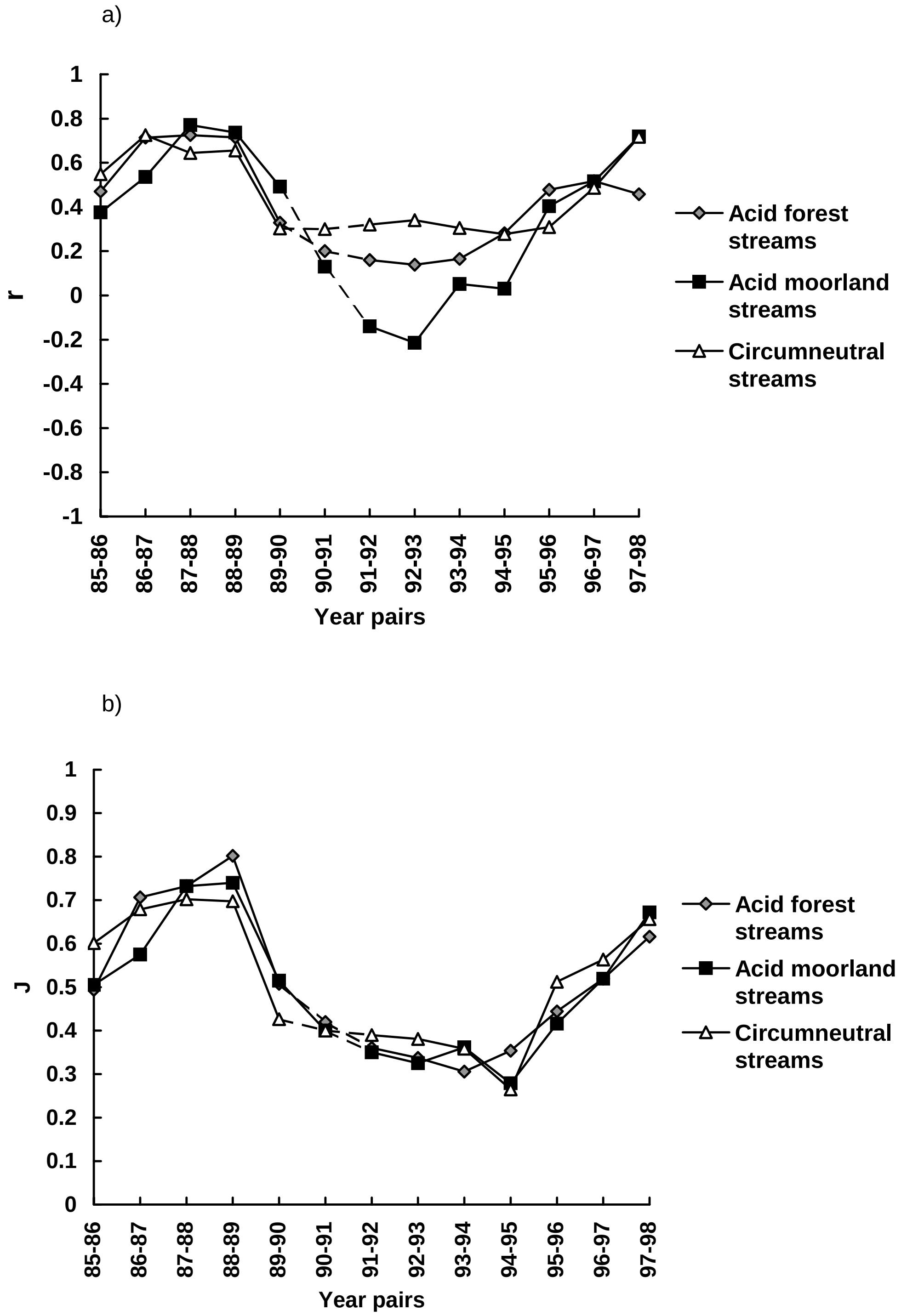
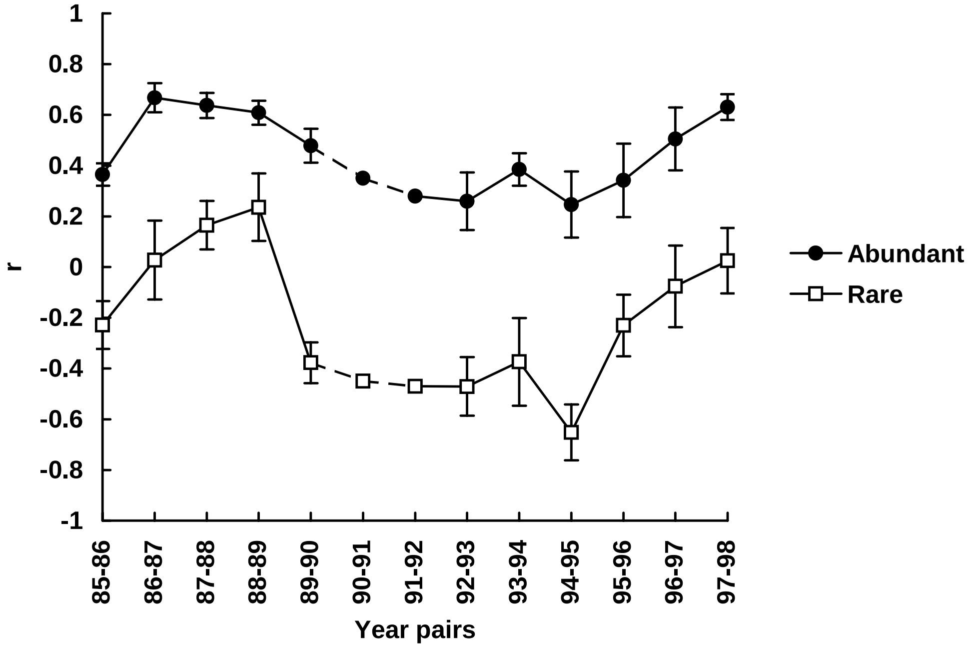


Figure 2. Year-to-year trends in community persistence in streams at the Llyn Brianne experimental catchments between 1985-1998 for abundant taxa (the 15 most abundant species overall) and rare taxa (the remainder). The values are means for all streams ± SE. Other conventions as in Fig. 1.

a)



b)

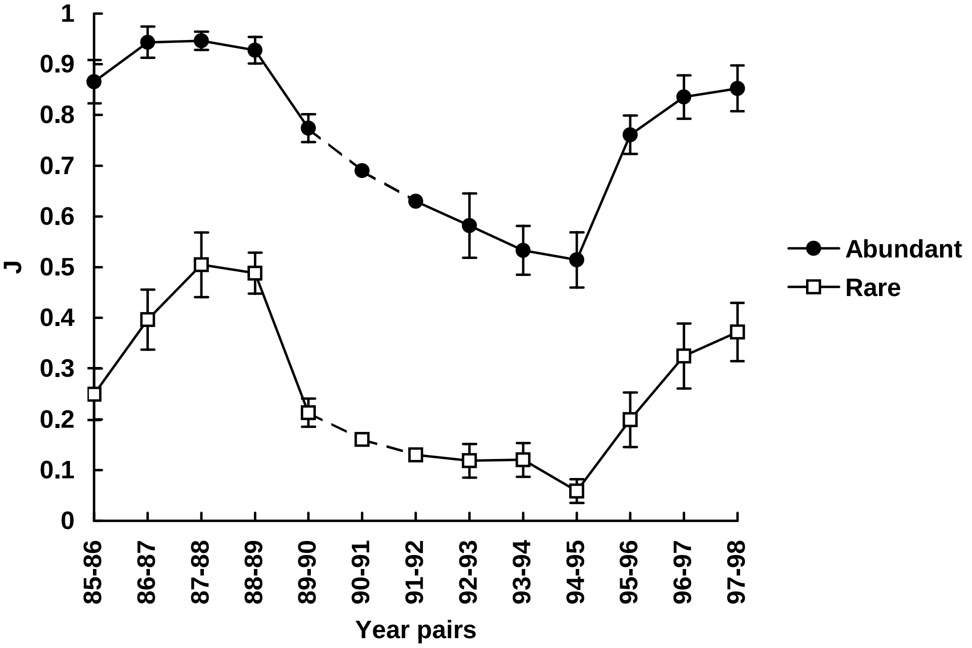


Figure 3. Year-to-year trends in community persistence in streams at the Llyn Brianne experimental catchments between 1985-1998 for riffles and marginal habitats. The values are means for all streams ± SE. Other conventions as in Fig. 1.

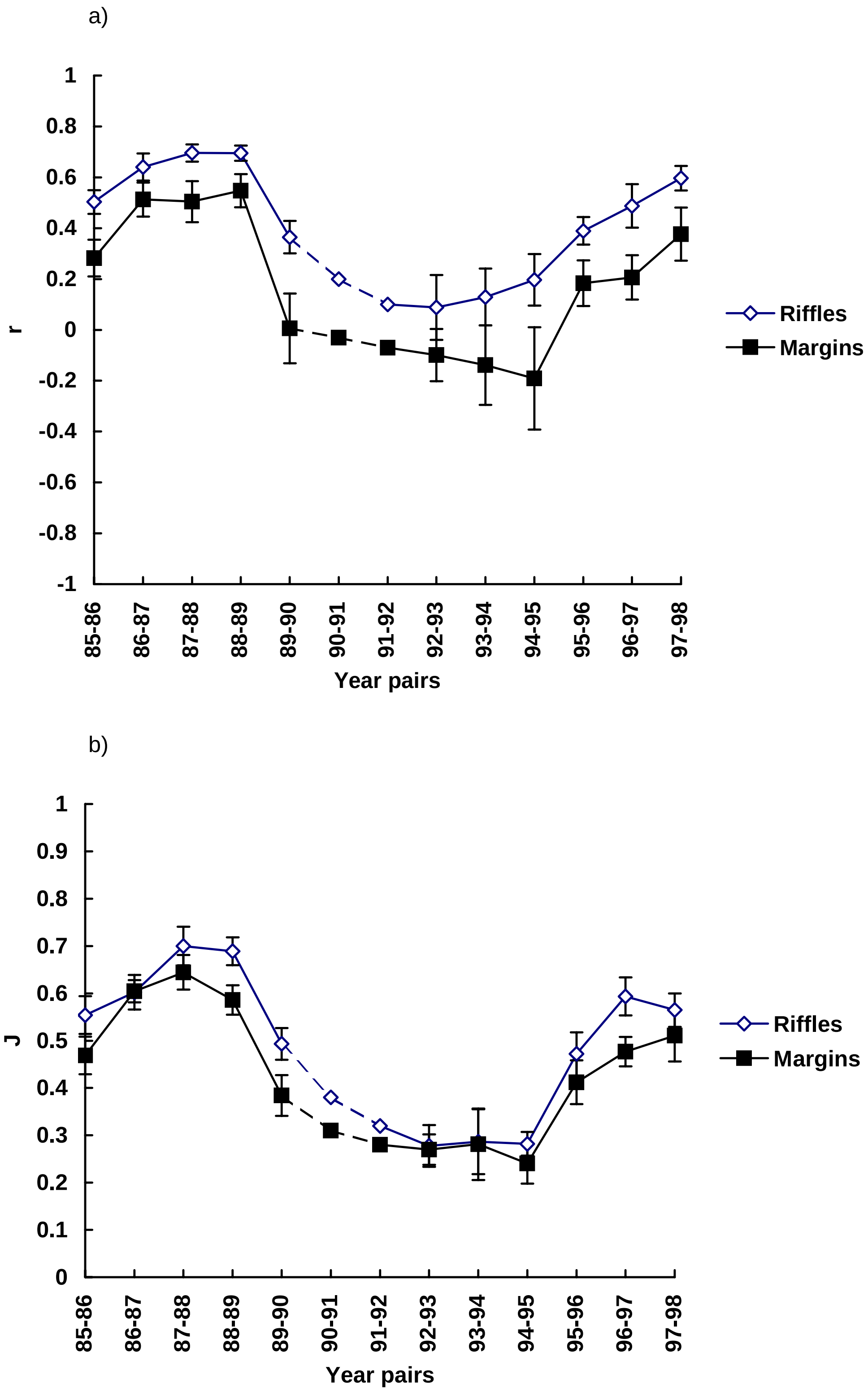
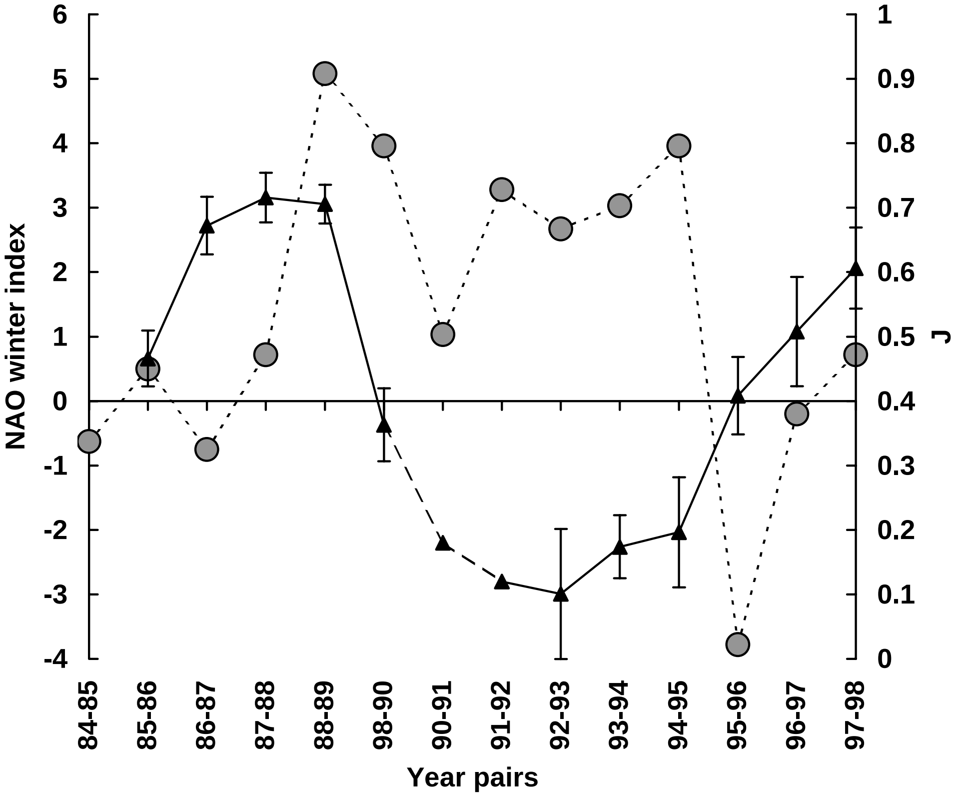


Figure 4. Year-to-year trends in community persistence in streams at the Llyn Brianne experimental catchments between 1985-1998 as shown by Spearman's rank correlation on rank abundance across species (r ▲). The values are means for all streams ± SE and the dashed lines represent missing data interpolated for 1991. The Figure also shows the NAO winter index (● after Hurell 1995).



Task 2.7.3: Parameterising and modelling the ecological effects of acid episodes

D.C. Bradley & S.J. Ormerod

Summary

1. Episodic fluctuations in pH and associated stream chemistry occur in acid-sensitive surface waters and are widely assumed to be important biologically. However, assessing and modelling their ecological effects has presented considerable difficulty since standard measures of episodicity are often correlated with and subsumed within the measurement of mean chemistry.
2. Using regional data from 70 independent streams in Wales, we examined i) whether modal values for pH and Al concentration better described base-flow chemistry from which episodic departures could then be assessed; ii) whether these episodic departures reflected biological status; iii) whether regression models for predicting invertebrate abundance, taxon richness and trout density were improved significantly by measures that separated episodic departures from modal chemistry and iv) whether model performance improved when applied to independent test sites.
3. Chemical data revealed three site group respectively characterised by high modal pH (>6) with pronounced episodes; high model pH (> 6) but no episodes; and chronic acidity (modal pH <6). Invertebrate abundances, taxon richness and trout densities did not differ significantly between episodic and chronically acidic sites. By contrast, there were clear differences between episodic and stable sites despite similarly high modal pH, particularly among acid-sensitive invertebrates.
4. Indices of pH and Al episodicity significantly improved regression models that predicted biological status over models based on modal values alone. Models based on modal pH and mode – minimum pH accounted for the most variation in the faunal communities. At test sites, invertebrate abundance, taxon richness and trout density predicted by models using episodic chemistry generally correlated with actual biological status, but there was evidence of systematic over-prediction at acid sites and under-prediction at sites with higher pH.
5. These data show that measures of episodicity that reflect departures from modal chemistry reflect what appear to be the biological effects of acid episodes. Nevertheless, models of complex interactions between variable stream chemistry and stream biota presents further challenges.

Introduction

Lotic freshwaters are characteristically dynamic and organisms inhabiting these environments experience disturbances on ranging temporal scales. In the long-term, press disturbances describe changes over timescales of years, but organisms may also be susceptible to discrete, short-term or pulsed disturbances (O'Connor & Lake, 1994). Many forms of pollution are typically episodic, and there has been an increased realisation that ecotoxicological responses to brief pulses of pollutants can differ from responses to constant conditions with the same mean chemistry (Thurston *et al*. 1981; Siddens *et al*. 1986; Seager & Maltby, 1989).

Episodes of low pH are well known in acid-sensitive, soft water streams where pH and associated water chemistry can fluctuate on timescales of hours (Ormerod & Jenkins, 1994) and over 2 or more pH units during high stream flows (Weatherley & Ormerod, 1991). The causal mechanisms driving these events are well documented and may be due to natural processes such as base cation dilution during increased runoff (Laudon, Westling & Bishop, 2000), or anthropogenically induced long-term acidification, conditioning catchments for the episodic release of acid (Davies *et al*. 1992; Laudon, Westling & Bishop, 2000). The ecological effects, however, remain poorly understood, but acid episodes are likely to have detrimental effects on sensitive biota, especially in streams that are not chronically acidified during base-flow conditions. Recent indications suggest that episodes may continue to occur and limit the persistence of acid-sensitive invertebrates in streams that have sustained a high mean pH following liming of their catchments (Bradley & Ormerod *in press*).

Identifying the ecological effects of acid episodes using field data is difficult. Continuous and intensive sampling programmes of stream chemistry are required in order to capture short-term and unpredictable depressions of pH; these are costly and logistically difficult (Van Sickle, Wigington & Church, 1997). Brewin *et al*. (1996) suggested that detection errors of minimum pH were up to 0.7 pH units for bimonthly samples compared to weekly samples. Sites with differing base-flow pH must be adequately represented, as episodes occur in both chronically acidified and non-acidified sites, but the ranges of pH experienced, and presumably biotic responses, differ between these site types (Davies *et al*. 1992).

Separating the effects of episodic from chronic acidification on stream fauna is of particular importance in modelling studies. This has been demonstrated successfully experimentally (Merrett *et al*. 1991), but not so from field data where pH and aluminium variability were closely related to mean conditions (Weatherley & Ormerod, 1991). In a previous modelling exercise, Weatherley & Ormerod (1991) reported that predictive models for stream biota based on mean pH and aluminium were improved negligibly with the inclusion of measures of pH or aluminium variability. Rather than close the case on the issue of episodicity, these studies inspire the need to search for a new parameterisation of stream chemistry, away from using mean values to describe average conditions (to which extreme values contribute and thus correlate) (Ormerod & Jenkins, 1994), towards measures which adequately describe extreme chemistry as separate from base-flow conditions.

We therefore offer a new parameterisation of stream chemistry proposing the use of modal values to describe average conditions. We investigate relationships between modal pH and episodic pH (mode – minimum pH) for describing and detecting the episodicity of sites, and allow the development of multivariate models incorporating measures relating to episodic pH and aluminium for predicting the biota of acid-sensitive streams.

Methods

Study area and data collection

Between October 1983 and September 1984 weekly samples of water chemistry were collected by the Welsh Water Authority from 102 sites located in recognised acid-sensitive upland regions of Wales (see Ormerod, Donald & Brown, 1989 for methods). This set remains one of the few complete runs of intensively sampled chemical data collected from streams representative of acid-sensitive soft waters. Slow weathering rates and low buffering capacity of bedrock results in largely base-poor soils in these stream catchments, the main vegetation consists of moorland grasses and coniferous plantations (Ormerod, Donald & Brown, 1989). Consequently, surface water runoff is generally low in alkalinity and often chronically acidic or susceptible to acid episodes (Weatherley & Ormerod, 1991).

Macroinvertebrates were sampled from 95 of these sites in April 1984 by taking a standardized 2-minute kick-sample from a riffle area in each stream (net aperture = 0.23 x 0.25m; mesh size = 0.9mm) (Wade, Ormerod & Gee, 1989; Stevens, Ormerod & Reynolds, 1997). All the animals from each sample were removed and identified to the lowest feasible taxonomic level (to species except Oligochaeta, Diptera and a few difficult genera of Ephemeroptera and Trichoptera). Enumeration was to abundance categories of 1, 10, 100, 1000 individuals due to time and cost constraints.

Salmonid fish were sampled from 57 of the survey sites between July and October 1984. Quantitative fish population assessments were performed using standard catch-depletion methods (Bohlin *et al*, 1989), using 50Hz pulsed D.C. electrofishing equipment (Electrocatch WFC4), powered b a portable electric generator (Honda EX650). A 50m reach of each stream was enclosed by stop nets and fished over three runs using one anode in an upstream direction. After each run fish were anaesthetised in benzocaine solution (25ppm), identified, weighed, measured for fork length, and after recovery, returned to the stream at least 50m downstream of the sampled reach. Brown trout only are included in this analysis because the number of sites where salmon were found was further reduced, as many sites were inaccessible to these migratory fish. In addition, thirty-two of the chemically monitored sites were excluded because they had incomplete runs of data leaving 70 sites with complete chemical and biological data for this analysis.

Data Analysis

A total of 19 chemical determinands were measured from each stream water sample (Stevens, Ormerod & Reynolds, 1997), but here analysis is restricted to only two, which are widely considered to be the most biologically important in terms of episodes, pH and aluminium (McCahon & Pascoe, 1991; Davies *et al*. 1992). Winter values (October 1983-March 1984) of pH and aluminium only were used throughout as this represented the impact period on invertebrates immediately prior to their sampling. Log abundances and taxon richness of all invertebrates were determined for each site, but we also wished to separate the most acid-sensitive invertebrate taxa for analysis. Acid-sensitive taxa were defined as those with pH optima >6.2, derived from presence/absence weighted average models of all streams (n = 37 taxa) (Brewin & Ormerod *unpubl. data*). Abundances of acid-sensitive invertebrates were normalised by log (n+1) transformations prior to analysis.

For each stream, modal pH was used to describe average conditions and mode – minimum pH was calculated to measure episodicity. Relationships between modal and mode – minimum pH were investigated on a scatter plot and from frequency distributions of mode – minimum pH in three categories of modal pH: <5, 5.1-6 and >6. From the scatter plot the episodicity of sites could be determined and sites were classified as either chronically acidified (pH <6), not chronically acidified but susceptible to acid episodes (pH >6, mode – minimum pH >1) and not chronically or episodically acidified, i.e. buffered sites (pH >6, mode – minimum pH <1). Differences among the stream invertebrate communities and trout densities between these site classes were investigated using one-way Analysis of Variance (ANOVA).

To consider alternative chemical parameters that might provide better measures of episodicity we also calculated minimum pH, standard deviation pH, maximum aluminium concentration and maximum – mode aluminium concentration. The proportion of samples with pH <5.5 in each stream was calculated to indicate the frequency of acid episodes as multiple episodes may cause cumulative toxic responses among some invertebrates (Merrett *et al*, 1991). All measures of aluminium concentration were log (n+1) transformed for normality before analysis.

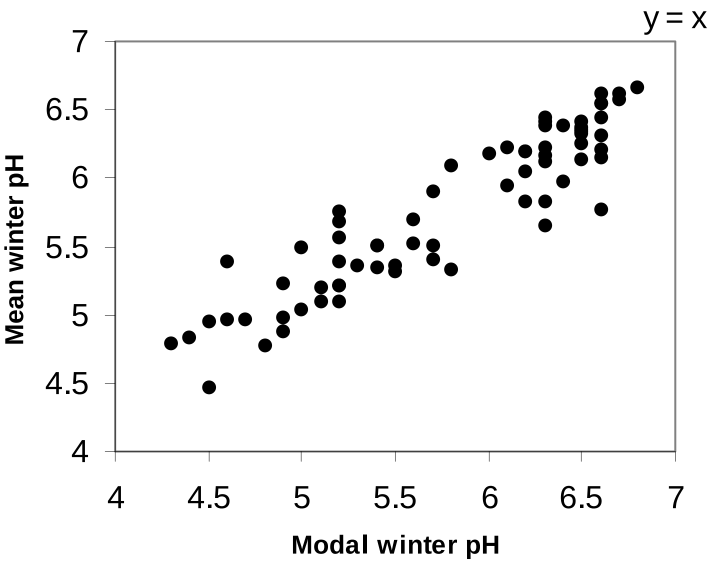
These alternatives and pH mode – minimum were separately incorporated into multiple regressions based only on modal pH to compare which parameters most significantly improved model fit. This procedure was repeated for models based on modal aluminium concentrations for comparison with pH models. The model, which gave the greatest prediction success of the stream fauna, was calibrated on 53 (75%) of the study sites and used to predict invertebrate abundances, taxon richness and trout densities in the remaining 17 test sites. A further validation of the model was performed by testing its prediction success of the fauna of 17 sites with a modal pH >6, hence sites that were not chronically acidified, but may experience episodic depressions of pH and elevations in aluminium concentrations (Davies *et al*. 1992).

Results

Modal winter pH was highly correlated with mean winter pH suggesting that modal pH is an adequate alternative measure of average conditions (Fig. 1). However, the relationship between modal and mean pH at the study sites indicated that in chronically acidified sites (model pH <5.5), model pH tended to underestimate the associated mean pH, whereas, for sites with model pH >5.5, model pH tended to overestimate mean pH (Fig. 1). This was probably an artefact of the generally increased range of episodic pH decline, measured by the difference between modal and minimum pH values, experienced at sites with higher modal pH (Fig. 2). For these sites, say with modal pH above 5.5, the mean will be disproportionately influenced by episodic values, and would not be an accurate measure of base-flow averages.

The relationship between modal pH and episodic pH range was not straight-forwardly linear, as there was much scatter for sites with modal pH >6, indicating that some of these sites with high base-flow pH were more susceptible to acid episodes than others which appeared more highly buffered (Fig. 3). On this basis, the 70 study sites were divided into three groups: Group 1- sites with modal pH <6, described as chronically acidic; Group 2- sites with modal pH >6 and mode – minimum values >1, described as episodic; and Group 3- sites with pH >6 and mode – minimum values <1, described as not episodic or buffered (Fig. 3). Differences among the faunal communities between these site groups were significant. Abundance and taxon richness were significantly higher in the non-episodic, Group 3 sites ($F_{2,69}$ = 11.36, P = <0.001 and $F_{2,69}$ = 24.61, P = <0.001 respectively), but not significantly different between site groups 1 and 2 (Tukey range tests P = <0.05). Likewise, trout densities were significantly higher among Group 3 sites than among site groups 1 and 2 ($F_{2,56}$ = 15.48, P = <0.001) (Tukey range tests P = <0.05). Taxon richness of acid-sensitive taxa were significantly higher among Group 3 sites than among site groups 1 and 2, between which there were no significant differences ($F_{2,69}$ = 30.85, P = <0.001) (Tukey range tests P = <0.05). Abundance of acid-sensitive taxa were significantly higher among Group 3 sites, but among Group 2 sites abundances were also significantly higher than among Group 1 sites ($F_{2,69}$ = 40.00, P = <0.001) (Tukey range tests P = <0.05).

Figure 1. The relationship between mean and modal pH at 70 regional Welsh survey stream sites.

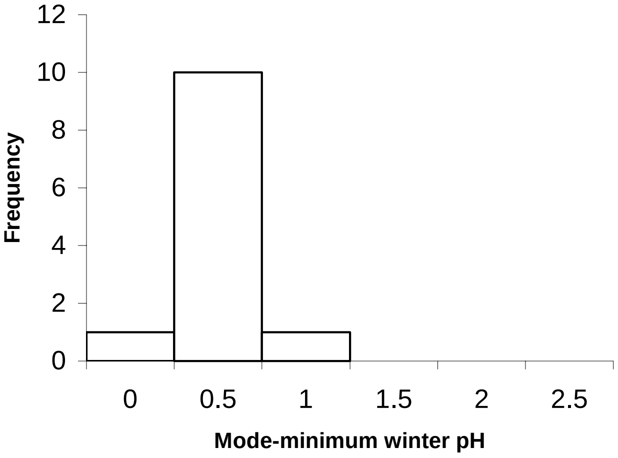


The incorporation of other chemical parameters describing episodicity to regression models relating stream fauna to modal pH or modal aluminium concentration generally improved model fit (Table 1). However, strong collinearity was observed between some episodic measures (minimum pH, relative frequency of pH <5.5, maximum aluminium concentration and standard deviation of aluminium concentration) and modal values. These relationships were not considered further. Models based on modal pH gave a substantially better fit than equivalent models based on modal aluminium concentrations for all measures of stream fauna (Table 1). Incorporating mode – minimum pH more significantly improved regressions based on modal pH for accounting for variation among invertebrate communities than alternative measures, especially for the abundance of acid-sensitive taxa (Table 1).

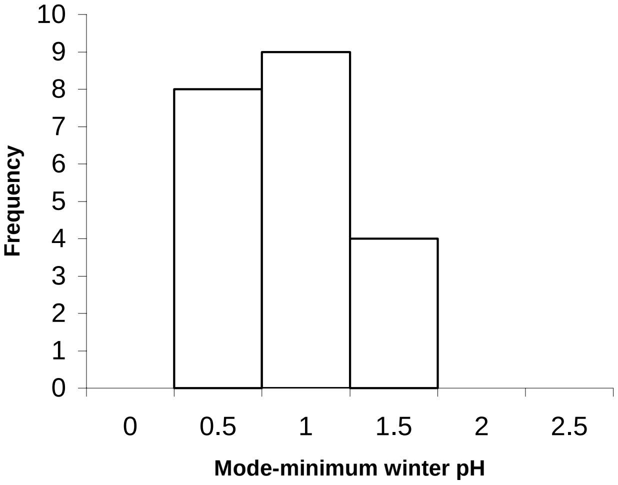
Although standard deviation of pH gave greater improvement than mode – minimum pH for accounting for variations in trout density, and likewise maximum – mode aluminium concentration for invertebrate abundance, these differences were negligible (Table 1).

Figure 2. Frequency distributions of mode – minimum pH values for a) highly acidic sites b) moderately acidic sites and c) not chronically acidic sites.

a) Modal pH <5



b) Modal pH 5.1-6



c) Modal pH >6

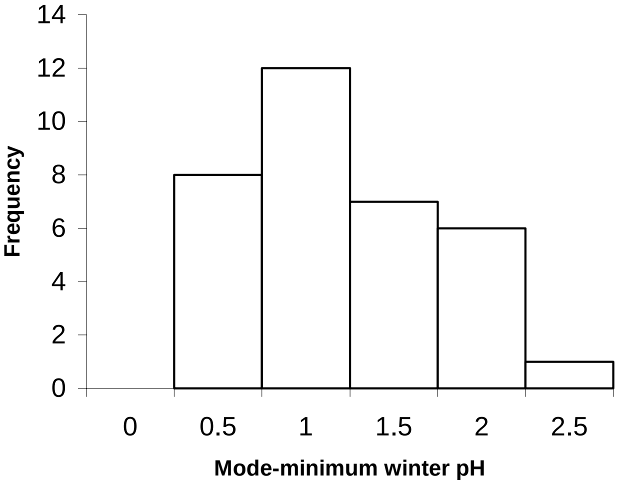
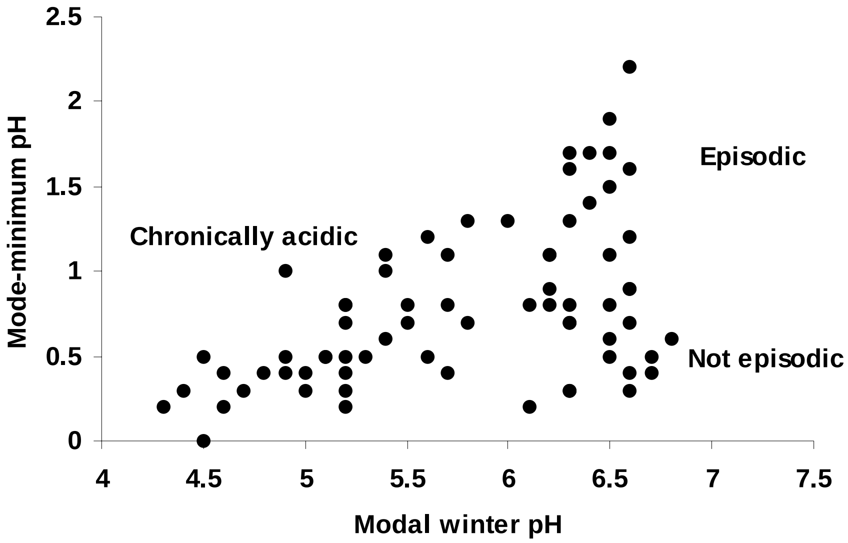


Figure 3. Scatter plot of modal pH versus mode – minimum pH showing divisions of site groupings



Multiple regression relationships between stream fauna, modal pH and episodic pH range developed from two sets of calibration sites are presented in Tables 2 and 3. Although the predicted fauna from the models in Table 2 were significantly related to the actual fauna observed in test sites, prediction success was clearly not good because high invertebrate abundances and taxon richness were under-predicted by the models and trout density generally over-predicted (Fig. 4). Prediction of the fauna in test sites with modal pH >6 from the models in table 3 was even less successful. Predicted and observed invertebrate abundances were not significantly related, also taxon richness, abundance of acid-sensitive taxa, taxon richness of acid-sensitive taxa and trout density were all over-predicted at low values and under-predicted at high values (Fig. 5).

Discussion

The results presented here provide further evidence to suggest that the use of mean values to describe average conditions are not appropriate in studies involving the effects of pulsed changes in stream chemistry. Relationships between mean pH and minimum pH have indicated that streams of intermediate acidity showed the largest ranges in pH (Weatherley & Ormerod, 1991), whereas, the chemical parameters used in this study suggested that the largest pH ranges occurred in the most well-buffered sites. For these sites that are not chronically acidified (pH >6) and often experience the widest pH fluctuations, mean values will be disproportionately influenced by episodic extremes, underestimating base-flow pH. Brewin *et al.* (1996) advised caution in the use of means, as the detection of chemical extremes is less effective at lower sampling frequencies. Therefore, the relationship between means and extreme values may be highly dependent on sampling frequency. Although reliable measures of mode will require many samples, these parameters will not be directly affected by the capture of chemical extremes at higher sampling frequencies.

In contrast to a previous study (Weatherley & Ormerod, 1991), we present evidence that stream biota, particularly the most acid-sensitive invertebrates and trout, are affected by episodic rather chronic conditions in some sites. Invertebrate abundance, taxon richness and trout density at sites with modal pH >6, which suffered episodic depressions of over 1 pH unit were more similar to chronically acidified sites than to sites with similar base-flow chemistry that were not episodic. However, we cannot report on differences in taxonomic composition among invertebrate faunas between these site groups. This would be useful, for example as some taxa may be unable to tolerate fluctuating acidity in sites that are not chronically acidified, but may be tolerant of lower base-flow pH in sites that are less episodic (see Seager & Maltby, 1989).

Regression models relating modal pH to invertebrate abundances, taxon richness and trout density were significantly improved with the addition of measures relating to pH variability (cf Weatherley & Ormerod, 1991). However, validation of these models with two sets of test sites did not give good prediction of their faunas. This highlights the difficulty in modelling complex interactions between dynamic stream chemistry and biology using simple indices. The measure of episodicity used (mode – minimum pH) described only the severity of potentially damaging pH depressions; it gave no indication of the frequency or duration of episodes at each site. Experimental studies have indicated that successive episodes may have a cumulative effect on biota (Merrett *et al*. 1991) and many invertebrate species can tolerate isolated periods of acidity for up to a few days (Sutcliffe & Hildrew, 1989; McCahon *et al*. 1989).

Overall, we conclude that using modal pH to describe base-flow conditions allowed detection of acid episodes in the study sites. Despite the poor performance of models, which included measures of episodic pH in predicting the stream fauna in test sites, invertebrate communities and trout were found to respond to episodic rather than chronic pH in some sites. Short of continually logging stream chemistry over long periods to capture the duration and frequency of episodes, we suggest that models using simple measures of episodicity may be improved by calibration and testing on more sites given adequate chemical and biological data.

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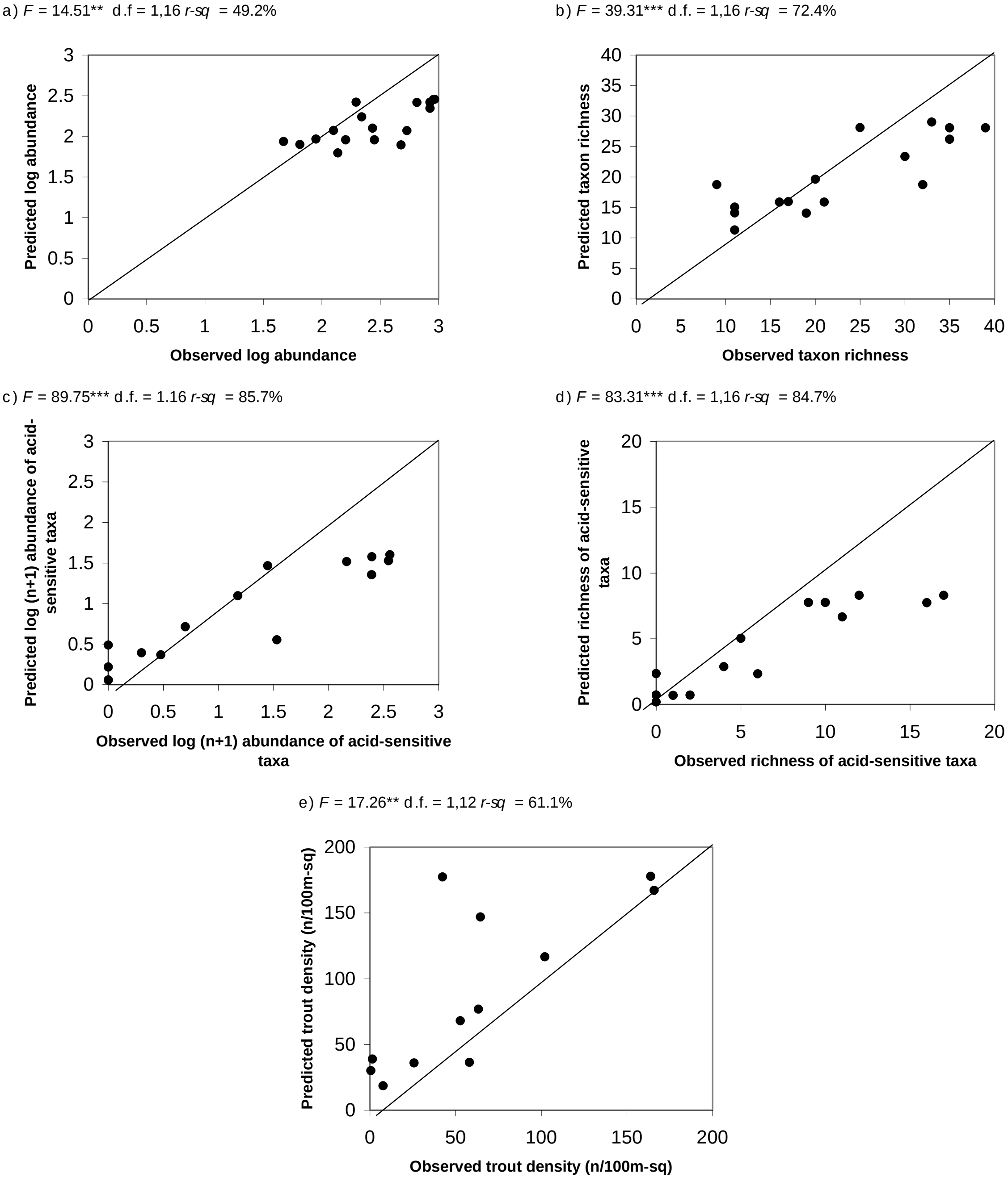
Figure 4. Predicted versus observed stream fauna in 17 test sites. Vertical distance from point to y = x line indicates error in estimating stream fauna from modal pH and mode – minimum pH. * = *P* <0.05; ** = *P* = <0.01; * = *P* = <0.001.**



a) *F* = 14.51** d.f = 1,16 *r-sq* = 49.2%

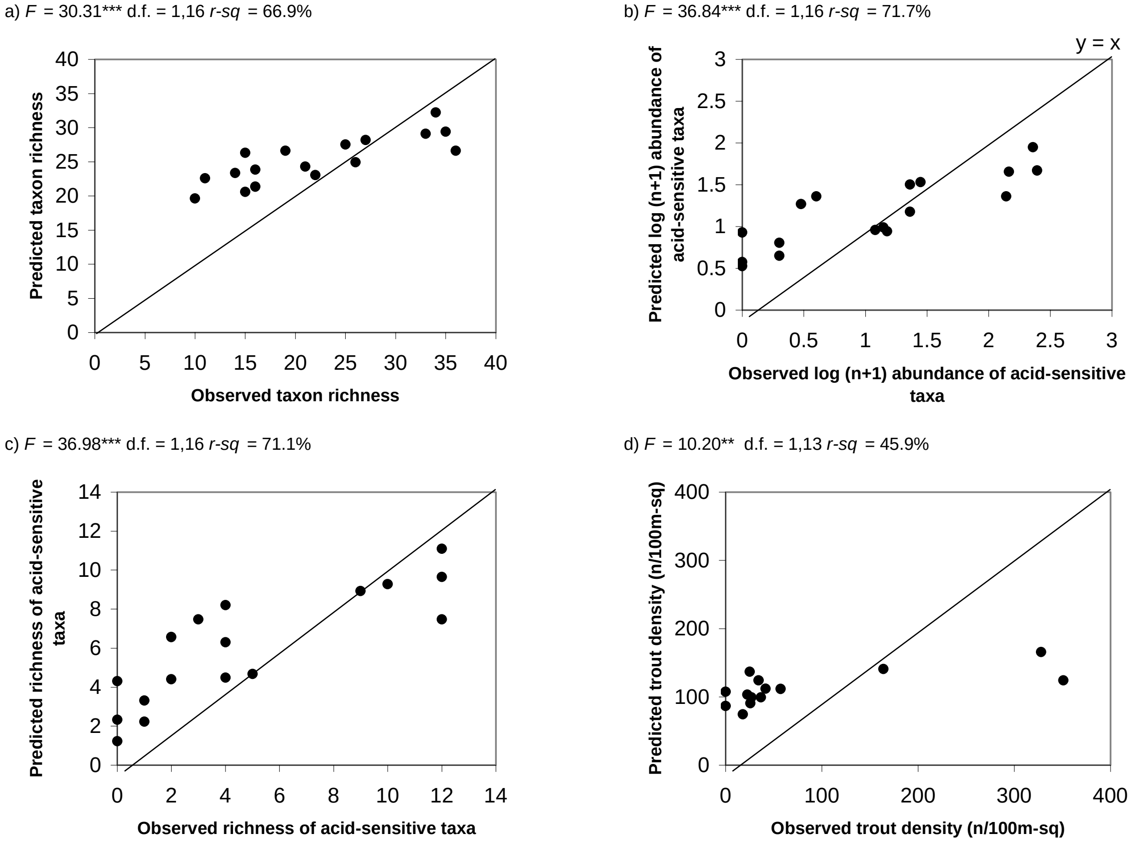
b) *F* = 39.31*** d.f. = 1,16 *r-sq* = 72.4%

c) *F* = 89.75*** d.f. = 1.16 *r-sq* = 85.7%

d) *F* = 83.31*** d.f. = 1,16 *r-sq* = 84.7%

e) *F* = 17.26** d.f. = 1,12 *r-sq* = 61.1%

Figure 5. Predicted versus observed stream fauna in 17 test sites with modal pH >6. Vertical distance from point to y = x line indicates error in estimating stream fauna from modal pH and mode – minimum pH. * = *P* <0.05; ** = *P* = <0.01; * = *P* = <0.001.**



a) *F* = 30.31*** d.f. = 1,16 *r-sq* = 66.9%

b) *F* = 36.84*** d.f. = 1,16 *r-sq* = 71.7%

c) *F* = 36.98*** d.f. = 1,16 *r-sq* = 71.1%

d) *F* = 10.20** d.f. = 1,13 *r-sq* = 45.9%

Table 1 Regression relationships between stream biota, a) modal pH b) modal aluminium concentrations and measures of episodicity. * = *P* <0.05; ** = *P* = <0.01; * = *P* = <0.001.**

a)

	Modal pH		Model 1 Modal pH	Mode-min pH		Model 2 Modal pH	SD pH		Model 3 Modal pH	Max-mode Al	
	r^2	*F*	*t*	*t*	r^2	*t*	*t*	r^2	*t*	*t*	r^2
Log abundance	14.7	11.68***	4.33***	-2.50*	22.0	3.00**	-1.54 NS	17.6	2.58*	-2.58*	**22.4**
Log (n+1) abundance of acid-sensitive taxa	40.7	46.73***	10.35***	-6.05***	**61.7**	6.59***	-5.10***	57.3	5.95***	-2.80**	46.9
Taxon richness	32.4	32.58***	8.05***	-4.75***	**49.4**	5.18***	-3.55***	43.1	4.86***	-2.48*	38.1
Richness of acid-sensitive taxa	32.4	32.52***	9.10***	-6.00***	**56.0**	5.27***	-4.85***	49.9	4.87***	-2.34*	37.5
Trout density (n/100m^2)	20.7	14.35***	5.41***	-3.58***	35.9	3.63***	-3.67***	**36.5**	2.78**	-2.27*	27.6

b)

	Modal Al		Model 1 Modal Al	Mode-min pH		Model 2 Modal Al	SD pH		Model 3 Modal Al	Max-mode Al	
	r^2	*F*	*t*	*t*	r^2	*t*	*t*	r^2	*t*	*t*	r^2
Log abundance	6.7	4.87*	-2.31*	-0.77 NS	7.5	-1.85 NS	-1.82 NS	11.1	-1.05 NS	-2.73**	**16.0**
Log (n+1) abundance of acid-sensitive taxa	22.5	19.75***	-4.71***	-1.57 NS	25.3	-4.11***	-5.05***	**43.9**	-3.23**	-2.64*	29.8
Taxon richness	19.9	16.85***	-4.34***	-1.45 NS	22.3	-3.67***	-3.73***	**33.7**	-2.98**	-2.37*	26.0
Richness of acid-sensitive taxa	18.3	15.26***	-4.31***	-2.03*	23.1	-3.49***	-4.93***	**40.1**	-2.81**	-2.28*	24.2
Trout density (n/100m^2)	12.9	8.18**	-3.12**	-1.47 NS	16.3	-2.61*	-3.61**	**29.8**	-1.46 NS	-2.24*	20.3

Table 2 Multiple regression relationships between stream biota, modal pH and episodic pH (mode – minimum pH) derived from a calibration set of 53 sites to test with 17 sites. Conventions as in Table 1.

Biological response	Regression equation	r^2	**F**
Log abundance =	0.375 + 0.347modal pH – 0.355mode-min pH	16.1	4.80*
Log (n+1) abundance of acid-sensitive taxa =	- 4.40 + 0.99modal pH – 0.864mode- min pH	53.3	28.56***
Taxon richness =	- 26.9 + 9.32modal pH – 9.37mode-min pH	40.9	17.29 ***
Richness of acid-sensitive taxa =	- 24.1 + 5.40modal pH – 5.42mode-min pH	46.5	21.69***
Trout density =	- 415 + 98.9modal pH – 101mode-min pH	35.5	11.27***

Table 3 Multiple regression relationships between stream biota, modal pH and episodic pH (mode – minimum pH) derived from a calibration set of 53 sites to test with 17 sites with modal pH >6. Conventions as in Table 1.

Biological response	Regression equation	r^2	**F**
Log abundance =	0.230 + 0.352modal pH – 0.063mode-min pH	20.4	6.42**
Log (n+1) abundance of acid-sensitive taxa =	- 4.90 + 1.08modal pH – 0.773mode-min pH	58.6	35.32***
Taxon richness =	- 36.7 + 10.9 modal pH – 6.21mode-min pH	49.6	24.58***
Richness of acid-sensitive taxa =	- 28.6 + 6.33modal pH – 5.43mode-min pH	52.8	27.93***
Trout density =	- 370 + 83.1modal pH – 41.7mode-min pH	37.9	12.19***

Task 2.9: Run the MAGIC 7 model for different afforestation strategies

C. Evans, R.Ferrier, R. Helliwell, and A. Jenkins

2.9 Run the MAGIC 7 model for different afforestation strategies

MAGIC 7 simulations have been carried out under different forestry scenarios for the Galloway dataset. Thirty six of the sixty catchments in this region are forested ranging from 10-100% forest cover. Future afforestation policy is expected to have a significant effect on acidification status of soils and surface waters in this region. Two forestry scenarios have been implemented in MAGIC 7 simulations, namely: Scenario 1 (SC1) 'remove all forest as it reaches 50 years of age and allow land to revert to moorland vegetation', and Scenario 2 (SC2) 'remove forest as it matures but immediately replant a second rotation forest'. These land use scenarios combined with the agreed sulphur reductions (REF deposition scenario – HARM modelled deposition predicted for 2010 prior to the further reductions of the Gothenburg Protocol) cause the response of soil and surface water quality to differ markedly within the region.

MAGIC 7 predictions under the 'best case' forestry scenario (Scenario 1) indicate a significant recovery in the surface water Acid Neutralising Capacity (ANC) compared to the ANC under the more realistic scenario which assumes forest rotation at 50 years i.e. planting a second rotation forest (Figure 2.9.1). The model predicts an increase in ANC at all moorland sites, whereas, despite the emission reductions, several forested sites undergo a further deterioration in ANC, albeit relatively small (Table 2.9.1). Recent evidence suggests that for Scottish fisheries, sites with mean surface water ANC ≤0 µeql^{-1} are currently almost fishless. In this respect Table 2.9.2 shows the number of sites in the Galloway region with ANC ≤ 0 µeql^{-1} and ANC ≤50 µeql^{-1} (the critical limit for protecting salmonid fishery status) for land use scenario 1 and 2. The predictions indicate that plantation forestry should be avoided in areas that are sensitive to acidification.

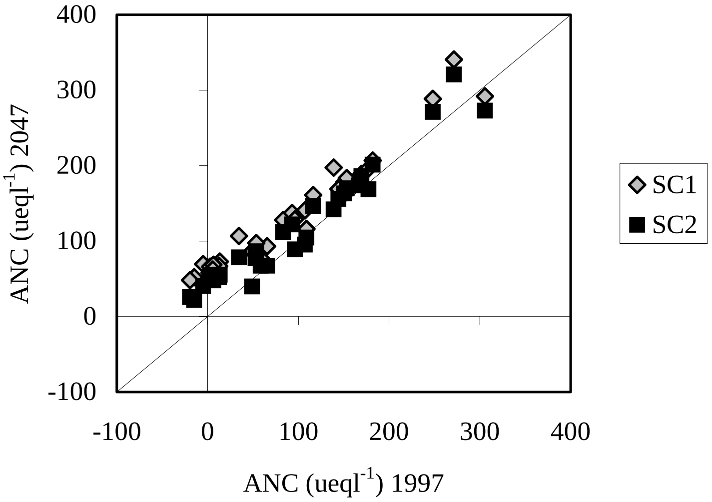
Table 2.9.1. The number of sites where ANC is predicted to deteriorate from 1997-2047.

		No. of sites	Mean ANC (µeql^{-1})
REF	SC1	3	20
	SC2	21	-6

Table 2.9.2. Number of sites with ANC ≤50 µeql^{-1} and ≤0 µeql^{-1} modelled with the REF deposition scenario and alternative land use scenarios for 1997 and 2047.

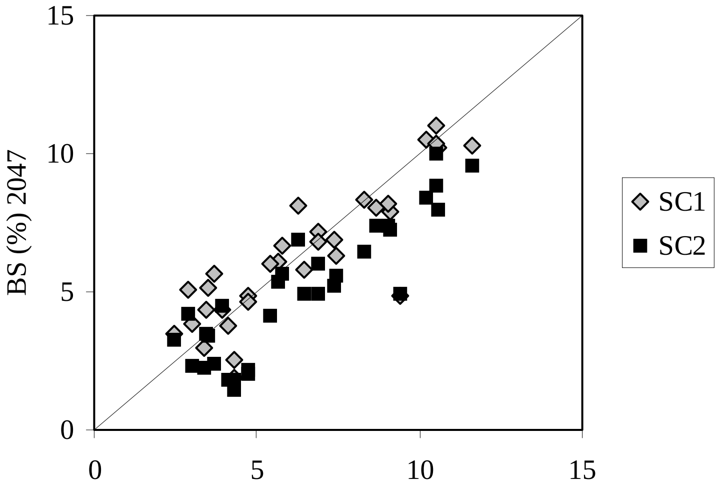
		REF scenario	
		ANC ≤50 µeql^{-1}	ANC ≤0 µeql^{-1}
	1997	13	4
SC1	2047	11	2
SC2	2047	14	5

Figure 2.9.1. Change in surface water ANC under alternative forest management strategies



Despite emission reductions, soil base saturation (BS) is predicted to acidify from 1997 to 2047 for the majority of catchments in the region (Figure 2.9.2). This implies that sulphur deposition is still greater than base cation supply from weathering. Changing land use at these forested sites complicates the future recovery of the base cation status of the soil. Clearly, a second rotation forest (SC2) exerts an additional stress on the soil base cation pool and so tends to exacerbate soil acidification in 2047.

Figure 2.9.2. Change in soil base saturation under alternative forest management strategies



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Task 2.10: Organize a workshop with environmental economists to ensure cost-benefit modellers are well informed of the relevant science output

A. Vipond

2.10 Organize a workshop with environmental economists to ensure cost-benefit modellers are well informed of the relevant science output

A workshop was organised by DETR to scope out methods for economic assessment of ecosystems, and a presentation was made on the scientific information on impacts of acidification on freshwaters by scientists under this contract. Rick Battarbee, Alan Jenkins and Don Monteith attended the workshop. Separate reports of this workshop have been produced outside this contract.