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ENCORE: Application of the MAGIC model to catchments in Norway and the UK

A. Jenkins[1] and R.F. Wright[2]

[1] **Institute of Hydrology, Wallingford OXON, OX108BB, UK**

[2] **Norwegian Institute for Water Research, Box 69 Korsvoll, 0808 Oslo, Norway**

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ABSTRACT

This report decribes application of MAGIC, a catchment-scale model of soil and water acidification, to the ENCORE catchments in the UK and Norway. We use the model to examine the catchment response to several scenarios of future environmental change including acid deposition and land-use. We calculate critical loads for sulphur for soil and water at each of the catchments, and use the model to illustrate the dynamic response to target loads. Finally we use these applications to evaluate the robustness of MAGIC as applied to these sites, and to suggest improvements and changes in the model.

Deposition of sulphur is one of the major driving variables within the MAGIC model. We assess the response of catchments to two possible deposition reduction scenarios over a 50-year period. These are; (1) a 55% decrease in sulphur deposition by the year 2006 with deposition held at his lever thereafter, and (2) a business-as-usual scenario whereby we assume no further cuts in sulphur deposition relative to deposition in 1990 are achieved at any site.

At all sites the model successfully simulates present day observed stream and soil chemistry confirming that the model is well suited for application to these types of areas. The predicted response of soils and surface waters to the 2 standard future deposition scenarios are similar at all catchments. All catchments continue to acidify under the worst case scenario (business as usual) and all catchments recover (or begin recovery) under the best case scenario. Exceptions are related to situations with concurrent land-use change such as the forest replant scenario at Hafren and Nant-Y-Bustach, or in the case of nitrogen saturation.

Application of MAGIC to ENCORE catchments will continue and include (1) calibrating MAGIC to other sites, (2) testing of MAGIC at ENCORE catchments that have been the sites of various whole-catchment manipulations, and (3) linking MAGIC to other models applied to data within the ENCORE project.

1. INTRODUCTION

ENCORE (European Network of Catchments Organised for Research on Ecosystems) is a large interdisciplinary project focussing on biological and chemical response to environmental change and the links between terrestrial and aquatic ecosystems (Hornung et al. 1990). ENCORE began in July 1991 and is part of the CEC STEP programme.

In all 29 catchments in 7 European countries are included in ENCORE (Figure 1). The sites span natural gradients in climate, vegetation and soils, as well as the present-day gradients of acid deposition and air pollution. At each of these catchments data are collected to permit calculation of input-output budgets of major chemical species at the catchment scale.

ENCORE catchments

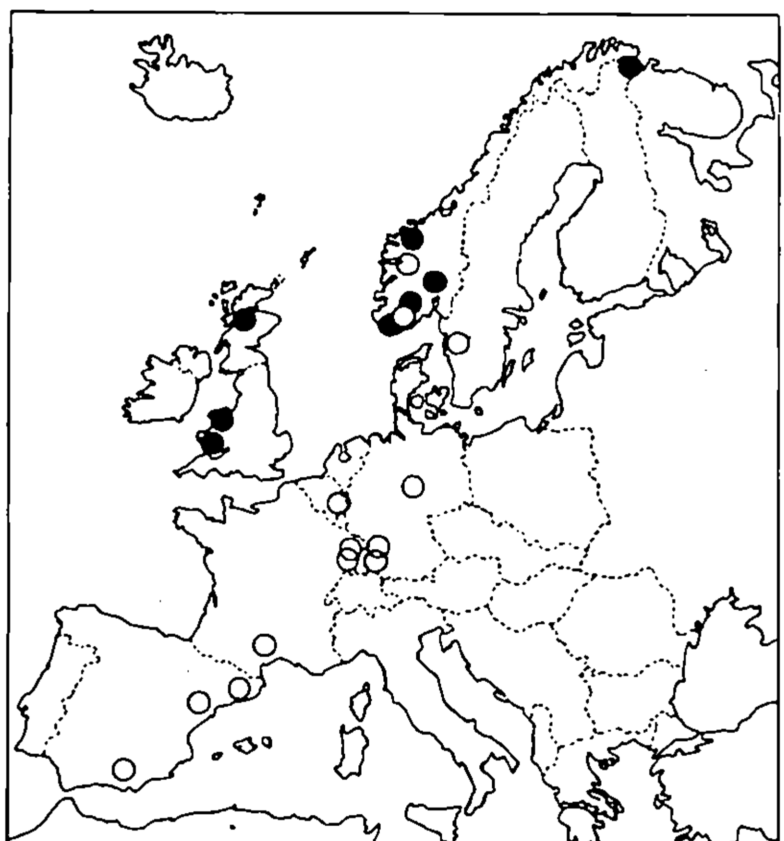

Figure 1. Location of the ENCORE sites. Several sites have more than one catchment. Sites with MAGIC applications included in this report are indicated by solid circles.

Models play a central role in the ENCORE project. Integration and exploitation of the results from catchment studies are achieved by the development of mathematical models. Much of the catchment-based work provides data and information about processes that form the basis for models (Figure 2). Over the past 10 years several catchment-scale models have been developed. These vary in complexity, conceptual basis, and time scale at which they are applied. The models generally fall into one of two categories - "event" and "long-term" models (Whitehead 1990, review in Concise Encyclopedia...Pergamon Press). Event type models are designed to explain the response of catchments at the hourly-to-daily time scale, while long-term models focus on trends over years-to-decades. Both types of models are necessary to account for biological effects of environmental change in terrestrial and aquatic ecosystems.

In the natural sciences the usual approach is to set up an hypothesis and then design an experiment to test the hypothesis. The ecosystem-scale effects of future environmental changes such as acid depostion and land-use, however, are both long-term and multifarious and thus it is difficult to carry out the appropriate experiments. We are thus forced to use mathematical models to simulate future change. To a large extent such models can also be used to test hypotheses.

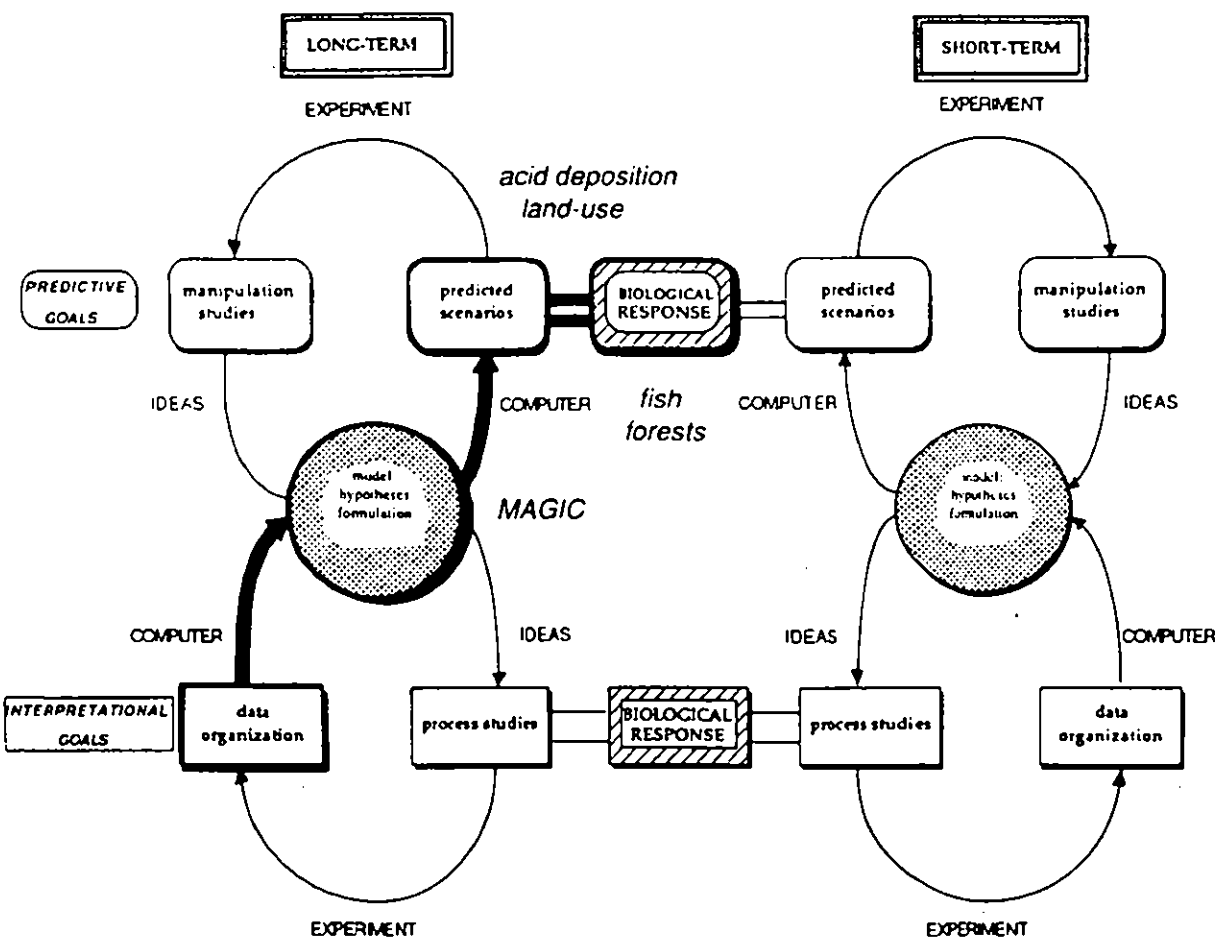

Figure 2. Diagrammatic sketch of the concept of ENCORE. Field measurements, process studies, model development and testing, manipulation experiments, and biological response are linked at the short and long term in small catchments (from Hornung et al. 1990). The highlights denote the links addressed here — data from catchments are used to calibrate the MAGIC model, which in turn generates predicted biological response to given scenarios.

MAGIC (Model for Acidification of Groundwater In Catchments) (Cosby 1985a, 1985b) is one of the more-widely used long-term process-oriented models operating at the catchment scale. MAGIC is an intermediate-complexity process-oriented model for constructing acidification history and predicting future acidification over time periods of decades to centuries MAGIC makes use of lumped parameters on a catchment scale and focusses on chemical changes in the soil caused by atmospheric deposition, vegetation, and leaching to runoff.

In this report we apply MAGIC to the ENCORE catchments in the UK and Norway. We build upon earlier MAGIC calibrations to data from these sites We use the model to examine the catchment response to several scenarios of future environmental change including acid deposition and land-use. We calculate critical loads for sulphur for soil and water at each of the catchments, and use the model to illustrate the dynamic response to target loads. Finally we use these applications to evaluate the robustness of MAGIC as applied to these sites, and to suggest improvements and changes in the model.

2. DESCRIPTION OF THE MAGIC MODEL

The MAGIC model combines a number of key soil chemical processes lumped at the catchment scale to simulate soil and surface water chemistry. MAGIC consists of: 1) soil-soil solution equilibria equations in which the chemical composition of soil solution is assumed to be governed by simultaneous reactions involving sulphate adsorption, cation exchange, dissolution and precipitation of aluminum, and dissolution of inorganic carbon; and 2) mass balance equations in which the fluxes of major ions to and from the soil and surface waters are assumed to be governed by atmospheric inputs, mineral weathering, net uptake in biomass, and loss in runoff (Cosby et al. 1985a, 1985b) (Figure 3).

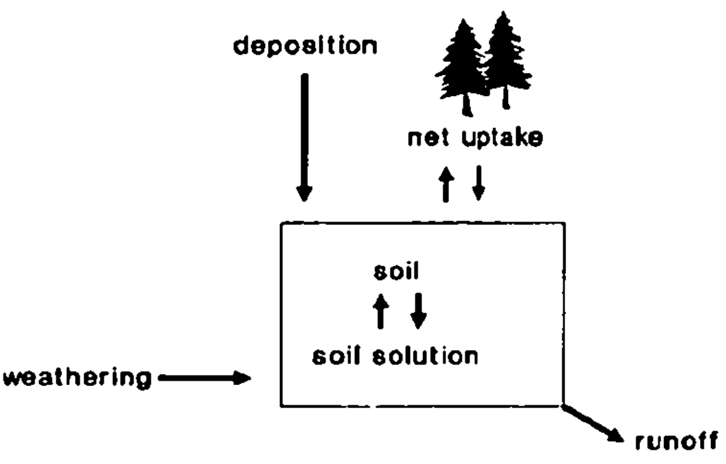

Figure 3. Schematic diagram of MAGIC. The pools of major chemical species in the catchment soil are central in MAGIC. Fluxes into and out of the catchment govern pool sizes; chemical equilibria between soil and soil solution determine the chemical composition of runoff. Terrestrial and aquatic biological response are related to soil and runoff chemistry, respectively.

MAGIC has been extensively used in a variety of applications at sites in both North America and Europe. Application of MAGIC to the whole-catchment experimental manipulations of the RAIN project shows that this intermediate-complexity lumped model predicts the response of water and soil acidification to large and rapid changes in acid deposition (Wright et al. 1990b). These results reinforce other evaluations of MAGIC such as comparison with paleolimnological reconstructions of lake acidification (Jenkins et al. 1990, Neal et al. 1988) and changes in regional lake chemistry in southern Norway (Wright et al. 1991).

In addition several of the assumptions in MAGIC have been tested experimentally (Grieve 1989). MAGIC is one of several dynamic models included in the UN-ECE Handbook on Mapping Critical Loads (Sverdrup et al. 1990).

MAGIC uses a lumped approach in two ways: (1) a myriad of chemical and biological processes active in catchments are aggregated into a few readily-described processes, and (2) the spatial heterogeneity of soil properties within the catchment is lumped to one set of soil parameters.

Whereas standard precipitation and throughfall gauges provide adequate estimates of integrated inputs to catchments and the outputs in runoff are integrated at the weir, corresponding estimates of soil parameters characteristic for an entire catchment are more difficult to obtain. Key soil parameters required by the model include depth, bulk density, porosity, cation exchange capacity (measured at soil pH), and the fraction of exchange sites occupied by Ca, Mg, Na, and K. We estimate these from soils data collected at several points within each catchment as part of routine monitoring programmes. Values were aggregated both spatially and with depth at each catchment to obtain single values for each parameter.

In the applications here we use an optimization procedure to calibrate MAGIC (Jenkins and Cosby 1989). For each catchment we use precipitation data, soil and soil solution data together with estimated acid deposition and net uptake histories to produce a calibrated model. The optimization routine determines the set of initial saturation and weathering rates for each of the 4 base cations for the assumed pre-acidification condition (assumed to be the 1840's). This set of initial values when run forward 140 years in time to the present (1980's) produces the best fit to the present-day measured soil chemistry and runoff chemistry. The calibrated model at each site is then used to predict future soil and water acidification 0-50 years into the future given various deposition and uptake scenarios.

3. SCENARIOS - ASSESSING FUTURE BEHAVIOUR

Once a calibration is completed for a given site, the model can be used in predictive mode to assess the water and soil chemistry changes that may occur in the future in response to prescribed changes in the driving variables through time. This predictive capability provides a powerful tool for the formulation and assessment of different management strategies. The management options considered for this report describe changes in atmospheric deposition of sulphur, changes in land use, focussing on future afforestation policy, and changes in the rate of nitrogen leakage from the terrestrial phase to the aquatic phase of the catchment system. Not all scenarios are utilised at all of the sites covered in this report; rather the scenarios are evaluated at a site only if an impact is likely.

3.1. ACID DEPOSITION

Increased atmospheric deposition of sulphur, above natural background levels, is responsible for the historical acidification of soils and surface waters in Europe and eastern North America. The recognition of this link between acidic deposition and soil and water acidification has prompted the formulation of deposition reduction strategies in an effort to reverse, or at least halt, the acidification process. Deposition of sulphur is one of the major driving variables within the MAGIC model and here we assess the response of catchments to two possible deposition reduction scenarios. These are; (1) a 55% decrease in sulphur deposition by the year 2006, which broadly corresponds to the Large Combustion Plant Directive in the UK and to the proposals for Europe as a whole under the negotiations within the UN-ECE. This scenario represents an optimistic strategy given the rate at which new technology can be implemented to achieve the planned reductions; (2) a business-as-usual scenario whereby we assume no further cuts in sulphur deposition relative to deposition in 1990 are achieved at any site. Clearly, this represents a pessimistic strategy. The actual future deposition at any site may be expected to fall somewhere between the two.

Nitrogen deposition is assumed to remain constant into the future for the worst case scenario and to decrease to 60% of present levels by year 2006 for the best case. This decrease reflects ongoing discussions within UN-ECE extrapolated forward to 2040 at the same rate. We assume that a similar best case is possible for the UK sites, and that there is no change in NH4 deposition through the forecast period in either of the scenarios.

3.2. LAND-USE

The major potential land-use change in upland UK catchments is management strategies associated with commercial plantation forestry. Coniferous trees have been shown to enhance soil and water acidification through three mechanisms; increased ability of the canopy to scavenge acidic compounds, often referred to as occult or filler deposition, uptake of base cations from the soil as the forest grows, and increased water use by the growing forest leading to greater concentration of the chemical throughputs. The effect of these three mechanisms are modelled explicitly within MAGIC by describing historical and future sequences for forest filtering, runoff and uptake (Cosby et al 1990).

Land-use scenarios are carried out at Nant-Y-Bustach (LI 1 Llyn Brianne) and Afon Hafren (Plynlimon) to assess the interaction of the growing forest under different future deposition scenarios. The two deposition strategies outlined in 3.1. are utilised in conjunction with two future forest management scenarios; to replant a felled forest immediately and to leave a felled forest as moorland. In all cases the forest is assumed to follow a 50- year rotation between planting and felling on maturation. Canopy closure is at 15 years of age. Occult deposition and evapotranspiration are assumed to increase linearly from planting to maturity. Base cation uptake is assumed to increase to a maximum at 15 years age and then decrease slowly until the stand is felled (Cosby et al. 1990).

3.3. NITROGEN SATURATION

Observations from surface waters in Norway indicate that during the past 20 years nitrate concentrations have increased although nitrogen deposition has not changed appreciably (Henriksen et al. 1988). This is potentially the result of nitrogen saturation; the terrestrial ecosystems are retaining less and less of the incoming nitrogen. The reasons for this incipient nitrogen saturation are as yet speculative. Whatever the mechanism, the increased nitrate flux represents a potentially serious acidification factor in ecosystems already heavily impacted through decades of sulphur deposition.

Due to the uncertainty regarding the causes of nitrogen saturation, this process is not explicitly modelled within MAGIC. The net retention of ammonium and nitrate inputs to the catchment system is represented by a first-order uptake coefficient which is adjusted to match observed input and output fluxes. By varying this uptake factor, the effect of nitrate saturation on soils and runoff can be simulated. Here we demonstrate the effect of nitrogen saturation for the catchment at Birkenes, Norway. We use the sulphur deposition scenarios given in section 3.1.

4. DETERMINATION OF CRITICAL LOADS

The concept of critical load is now widely used as a basis for decisions regarding future emissions of acidifying gases. The critical load is defined as "A quantitative estimate of the loading of one or more pollutants below which significant harmful effects on specified sensitive elements of the environment are not likely to occur according to present knowledge" (Nilsson and Grennfelt, 1988). The target load is an operational value and for a given ecosystem can be set at a level higher than the critical load (in which case some damage is accepted) or lower than the critical load (in which case a safety margin is provided). Critical loads are thus viewed as being intrinsic properties of the environment, whereas target loads are set according to political or management decisions.

Critical load for soil is of interest because soil acidification affects biological organisms in terrestrial ecosystems (for example, trees) and in aquatic ecosystems (for example, fish). Negative effects in forest ecosystems include shortage of mineral nutrients, nutrient imbalance, and high concentrations of toxic aluminum compounds in soil solution (Nilsson and Grennfelt 1988).

Procedures for determination of critical load for a given ecosystem or region entails the use of models. Two types of models are available: steady-state and empirical models, and dynamic process-oriented models (Sverdrup et al., 1990). Much of the mapping work carried out to date is based on steady-state and empirical models (Henriksen et al. Ambio in prep.). Although these models provide regional estimates of critical loads, they do not take into account the time aspect of acidification and recovery of terrestrial and aquatic ecosystems.

The time-dependent aspects are of central importance in the selection of target loads. Here the question is not only what percentage of the affected or threatened waters, soils and forests are to be protected (or restored) but for what period of time. Clearly if the goal is to restore an acidified lake within 10 years the target load will be lower than if one can wait 50 years.

Criteria for "unacceptable change" are set in relation to effects on terrestrial and aquatic organisms. By itself soil is inanimate and the term "damaged" soil has no meaning. With respect to damage to terrestrial vegetation commonly used criteria include the concentration of inorganic aluminum in soil solution and the molar ratio of calcium to aluminium in soil solution (de Vries 1988, Sverdrup et al. 1990), where the soil solution in rooting depth (0-50 cm) is of primary interest. With respect to aquatic organisms commonly used criteria are that the runoff water should have positive acid neutralizing capacity (ANC) and concentration of labile inorganic aluminum less than 50 µg/l (Nilsson and Grennfelt 1988, Henriksen and Brakke 1988).

Here we illustrate these dynamic aspects of acidification and the implications for critical loads. We use data for both water and soil chemistry at a catchment scale to derive estimates of soil acidification for soil characteristic of the entire catchment. We estimate critical load by means of the dynamic MAGIC model (Cosby et al. 1985a, 1985b) and the static empirical model of Henriksen (1980) and evaluate with respect to criteria for adverse effects to both forests and to surface waters. This procedure is applied to the data from 10 ENCORE catchments in Norway and the UK (Figure 1).

The critical loads for each site are calculated using the MAGIC model under the condition that deposition is suddenly changed to a new level and then held constant for 50 years. MAGIC is run repeatedly with different levels of deposition until the criterion of ANC = 0 µeq/l (fish) or Ca/Al = 1.0 (forest) is met. This deposition is the critical load for sulfur. For all cases it is assumed that the loading and retention of nitrogen compounds are not changed from present-day conditions.

Henriksen (1980) developed an empirical model for predicting water acidification. This model is based on present-day water and precipitation chemistry and is static in that it specifies the water chemistry resulting from a given change in deposition without specifying the time at which this new water chemistry will exist. The model thus does not provide information as to length of time required to achieve steady-state following change in acid deposition. This empirical model has also been extensively used in both North America and Europe. The empirical model is one of the static models included in the UN-ECE handbook (Sverdrup et al. 1990), and is currently being used in conjunction with mapping critical loads for freshwaters in Norway.

5. SITE DESCRIPTIONS, DATA SOURCES AND MODEL OUTPUT

5.1. ALLT A MHARCAIDH

This catchment is an unacidified upland moorland site in the Cairngorm Mountains of Scotland, UK. Total sulphur deposition is relatively low. Rainfall is low and there is less of a sea-salt influence than at the other UK sites considered in this report. The catchment area comprises a Site of Special Scientific Interest within the Cairngorm National Nature Reserve and land-use is confined to deer grazing. The site was identified as a 'transitional' catchment in 1985 for the Surface Waters Acidification Programme and hydrochemical monitoring has continued since that date. The stream is essentially unacidified but is characterised by severe acid pulses associated with rainstorms and snowmelt and as such was chosen to represent a site which might become chronically acidified with continued or higher acidic deposition. Future sampling at this site will continue for at least a further 6 years under the UK Acid Waters Monitoring Network.

MAGIC has previously been applied at this site to assess model sensitivity to soil physical and chemical parameters (Jenkins et al 1988). Sulphate adsorption characteristics were identified as being important with respect to long term acidification at the site. A subsequent calibration was carried out to assess the impact of including two-layer soil representation and flow routing upon the long term predictions (Jenkins and Cosby 1990). The results of this analysis show that long term stream chemistry response is unaffected by this refinement but soil chemistry, particularly the upper layer, may become more acidic in the long term. For this application the model has again been completely re-calibrated using multiple optimisation techniques and most recent chemistry data. A single soil layer methodology was employed although it is clear that at this site at least two layers would improve estimates of soil critical loads and calibration to a monthly or weekly time-step might provide a useful basis for assessing short-term responses.

5.1.1. Physical Characteristics

Area;	9.98 km^2
Vegetation;	Heather and Nardus grassland
Soil Type;	Blanket peats, alpine and peaty podsols
Mean Soil Depth;	0.83 m
Geology;	Biotite granite, glacial drift

5.1.2. Hydrochemical Characteristics

Data Period;	1989
Rainfall;	1066 mm
SO_4* Input Load;	32 meq/m^2/yr
Cl Input Load:	110 meq/m^2/yr
Stream pH;	6.3
Soil Base Saturation;	4.6 %

5.1.3. Data Sources

Input Chemistry;	Ferrier et al. (1990), Warren Springs Laboratory (1990)
Output Chemistry;	Juggins et al. (1990)
Soils Data;	Nolan et al. (1985), Jenkins et al. (1988)

5.1.4. Model Output - Reconstructions and Predictions: Allt A Mharcaidh

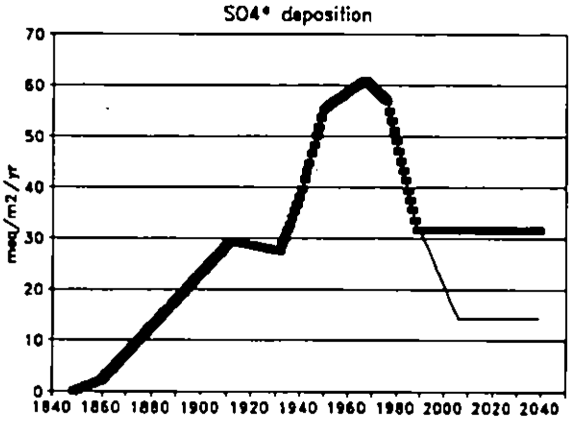

Figure 5.1a. Excess sulphate deposition (meq/ m^2/yr) used for best and worst case scenarios.

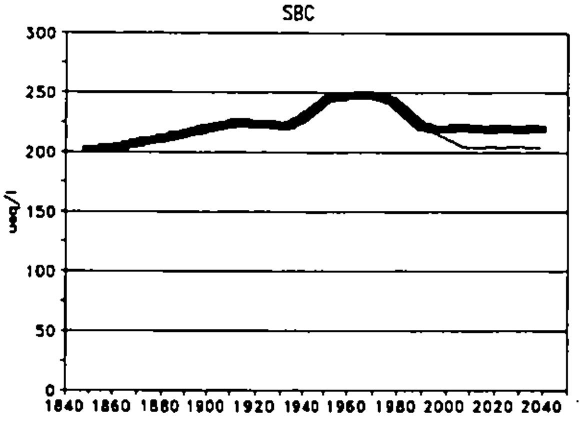

Figure 5.1c. Reconstructed historical streamwater sum of base cations (SBC meq/m^3) and predicted response under each scenario.

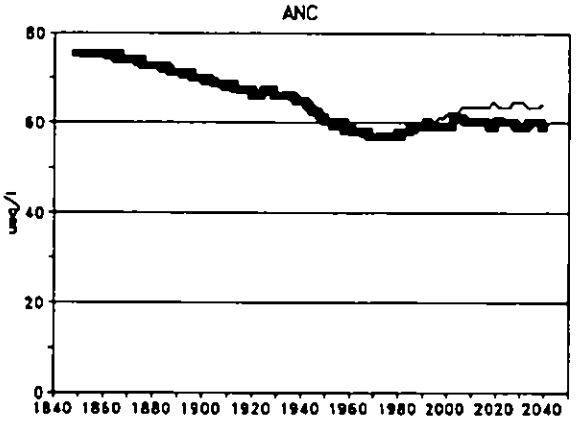

Figure 5.1b. Reconstructed historical streamwater alkalinity (meq/m^3) and predicted response under each scenario.

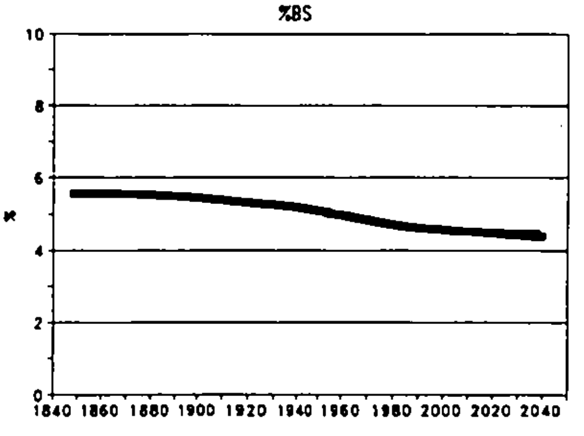

Figure 5.1d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.2. AFON GWY (PLYNLIMON)

This catchment is an acidified upland moorland site in central Wales, UK., receiving a substantial sulphur input, much of which is associated with neutral sea-salt. The catchment supports rough sheep grazing throughout the year and cattle grazing in the lower portion in summer. The stream has been monitored by the Institute of Hydrology since the early 1970's as part of a major study into the water resources impacts of upland afforestation. Chemical sampling commenced in the mid 1980's and will continue for at least a further 6 years within the framework of the UK Acid Waters Monitoring Network.

The MAGIC model has previously been applied to this site in an attempt to assess the hydrochemical impact of afforestation (Whitehead et al. 1988) although using only a crude representation of the forest effects. For this application the model has been completely re- calibrated using multiple optimisation techniques and most recent chemistry data. Problems were encountered in matching observed stream nitrate concentrations since these are appreciably higher than observed nitrate concentrations in wet deposition. The source of the additional nitrate may well be gaseous and dry deposition but may also reflect nitrate generation through soil mineralisation since high concentrations have been found in shallow soil lysimeters (Reynolds et al. 1989). Since the present model utilises only a simple first order uptake to relate nitrate input and output flux, for this application additional nitrate was added to the soil to enable a match with observed stream flux.

This site has recently formed a focus for an attempt to link long and short term hydrochemical responses by linking MAGIC with an end-member mixing model (Robson et al. 1991, Neal et al 1992).

5.2.1. Physical Characteristics

Area;	3.88 km^2
Vegetation;	Nardus, Festuca and Agrostis grassland
Soil Type;	Peat, brown earth, stagnopodsols and stagnogleys
Mean Soil Depth;	0.81 m
Geology;	Mudstones, shales, grits and glacial drift

5.2.2. Hydrochemical Characteristics

Data Period;	1989
Rainfall;	2500 mm
SO_4* Input Load;	129 meq/m^2/yr
Cl Input Load	350 meq/m^2/yr
Stream pH;	4.9
Soil Base Saturation;	21.0 %

5.2.3. Data Sources

Input Chemistry;	Reynolds et al. 1988, 1989
Output Chemistry;	Reynolds et al. 1988, 1989
Soils Data;	Reynolds et al. 1988, 1989

5.2.4. Model Output - Reconstructions and Predictions: Afon Gwy (Plynlimon)

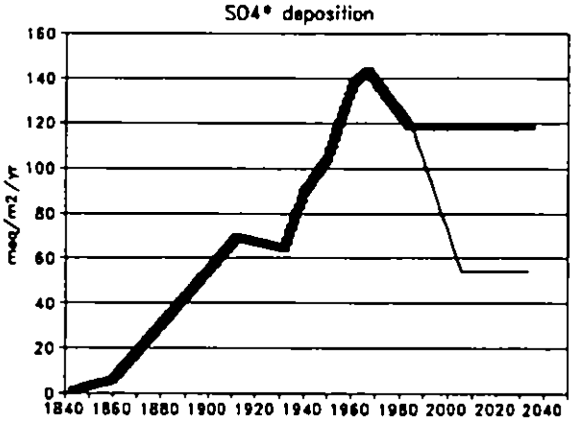

Figure 5.2a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

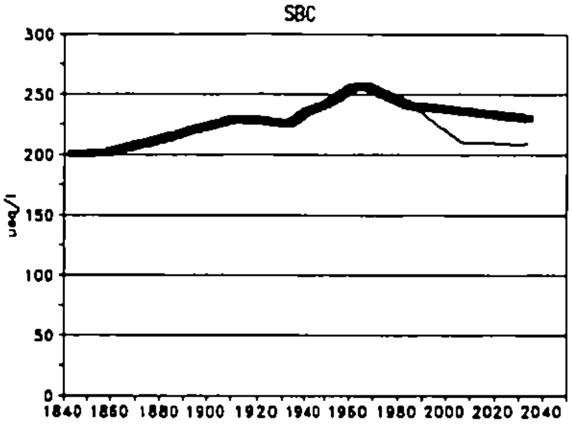

Figure 5.2c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

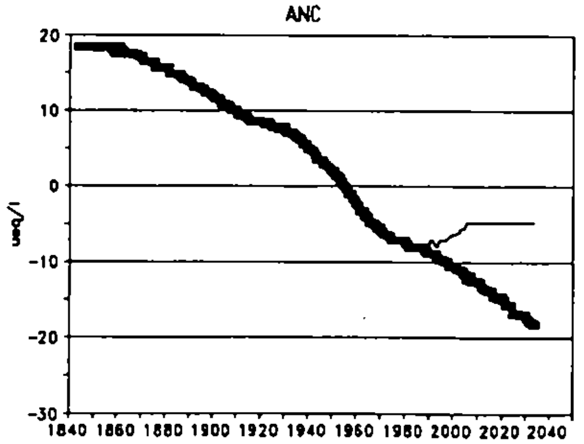

Figure 5.2b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

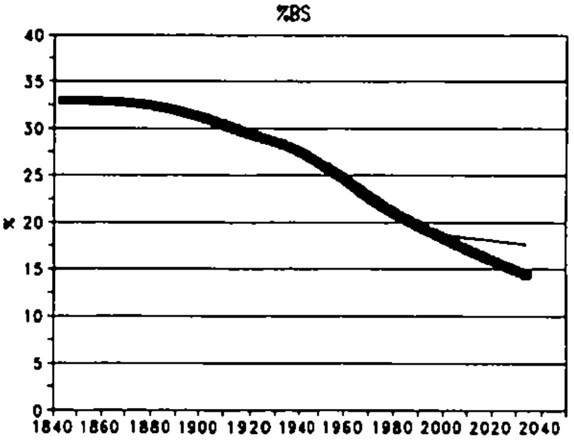

Figure 5.2d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.3. AFON HAFREN (PLYNLIMON)

This site is an acidic forested catchment in upland Wales, UK. The large fluxes of sea-salt derived ions and excess sulphate are due to the high rainfall and the enhancement deposition associated with the forestry. Planting of conifers took place over 50% of the catchment area in two phases; between 1948 and 1950 and between 1963 and 1964. The other fifty percent of the catchment is utilised for rough sheep grazing and prior to afforestation the catchment was entirely exploited as upland sheepwalk. The stream has been monitored by the Institute of Hydrology since the early 1970's as part of a major study into the water resources impacts of upland afforestation. Chemical sampling commenced in the mid 1980's and will continue for at least a further 6 years within the framework of the UK Acid Waters Monitoring Network.

An earlier version of the MAGIC model has been previously applied at this site to assess the impact of upland afforestation (Neal et al. 1986, Whitehead et al. 1988) incorporating only increased occult deposition as the major forestry driven process. For this application the model has been completely recalibrated using the most recent chemistry data, multiple optimisation techniques and explicitly incorporating the ion uptake, pollutant filtering and increased water use of the growing forest. The mid-years of the planting periods were taken to simplify the designation of uptake, hydrology and deposition sequences.

5.3.1. Physical Characteristics

Area;	3.7 km^2
Vegetation;	48% Sitka spruce 52% grassland
Soil Type;	Peat, brown earth, stagnopodsols and stagnogleys
Mean Soil Depth;	1.02 m
Geology;	Mudstones, shales, grits and glacial drift

5.3.2. Hydrochemical Characteristics

Data Period;	1989
Rainfall;	2391 mm
SO_4* Input Load;	126 $meq/m^2/yr$
Cl Input Load	407 $meq/m^2/yr$
Stream pH;	5.0
Soil Base Saturation;	8.3 %

5.3.3. Data Sources

Input Chemistry;	Warren Springs Laboratory (1990)
Output Chemistry;	Juggins et al. (1990)
Soils Data;	Reynolds et al. (1988, 1989)

5.3.4. Model Output - Reconstructions and Predictions: Afon Hafren (Plynlimon)

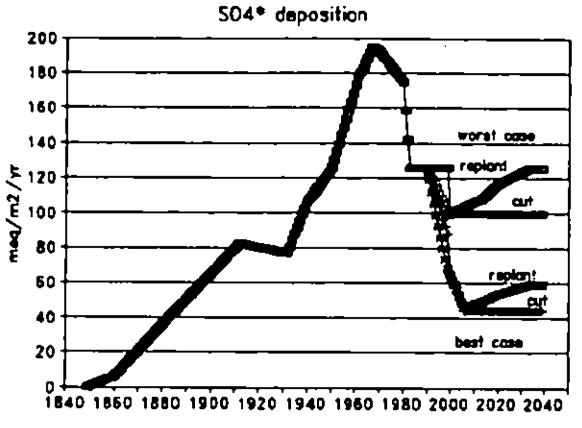

Figure 5.3a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

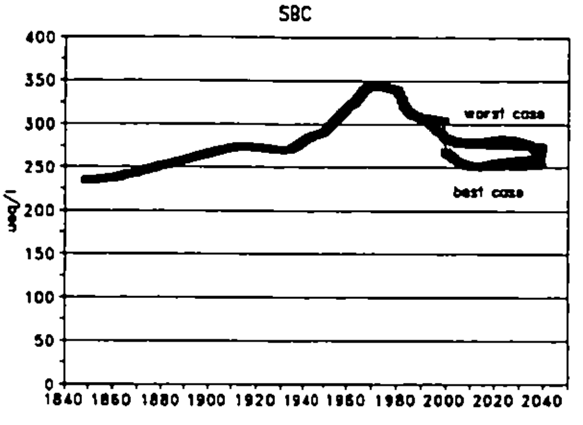

Figure 5.3c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

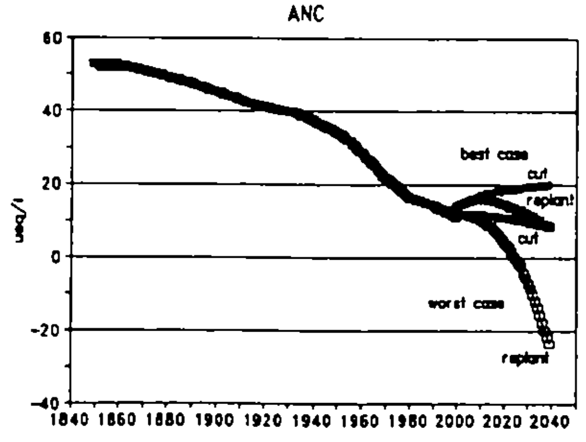

Figure 5.3b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

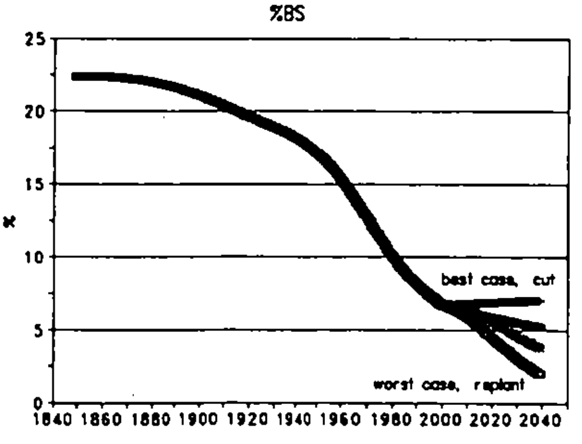

Figure 5.3d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.4. NANT-Y-GRONWEN (CI6, LLYN BRIANNE)

This site is an unacidified moorland catchment in the Cambrian Mountains of mid-Wales. The catchment has a long history of utilisation for rough sheep grazing. Monitoring of the stream commenced in the early 1980's as part of the Llyn Brianne Acid Waters Study within which this site represented the unacidic moorland control. This programme was completed in 1990 and only monthly chemical sampling is currently undertaken.

MAGIC has been previously applied to this site to assess the effect of upland afforestation (Whitehead et al 1988) although increased occult deposition was the only forest process considered. For this application the model has been completely recalibrated using multiple optimisation techniques and most recent chemistry data.

5.4.1. Physical Characteristics

Area;	0.72 km^2
Vegetation;	Grassland
Soil Type;	Peats, gleys and podsols
Mean Soil Depth;	1.0 m
Geology;	Mudstones, shale, grits, greywackes and glacial drift

5.4.2. Hydrochemical Characteristics

Data Period;	1989
Rainfall;	1765 mm
SO_4* Input Load;	114 meq/m^2/yr
Cl Input Load	286 meq/m^2/yr
Stream pH;	5.9
Soil Base Saturation;	20.9 %

5.4.3. Data Sources

Input Chemistry;	Warren Springs Laboratory (1990)
Output Chemistry;	Juggins et al. (1990)
Soils Data;	Reynolds and Norris (1990)

5.4.4. Model Output - Reconstructions and Predictions: Nant-y-Gronwen (CI6, Llyn Brianne)

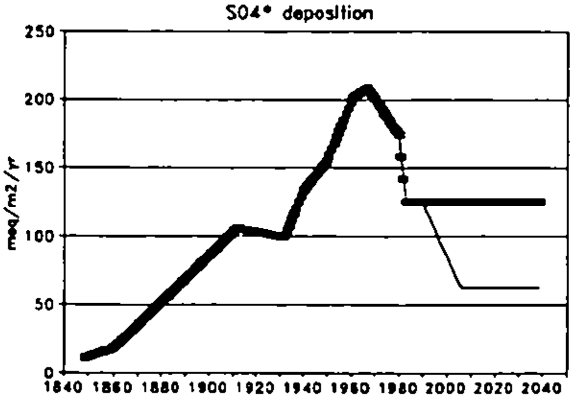

Figure 5.4a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

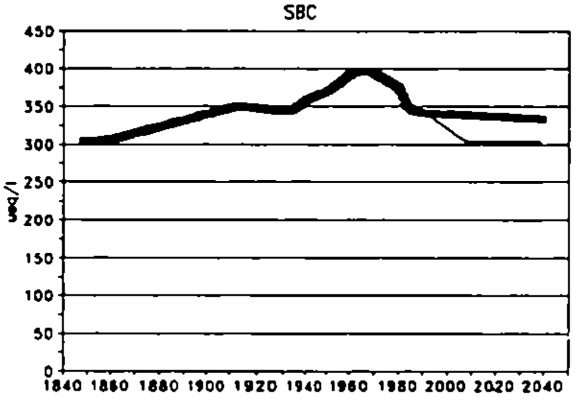

Figure 5.4c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

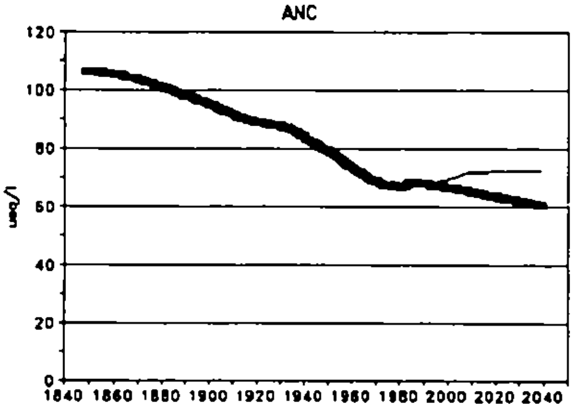

Figure 5.4b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

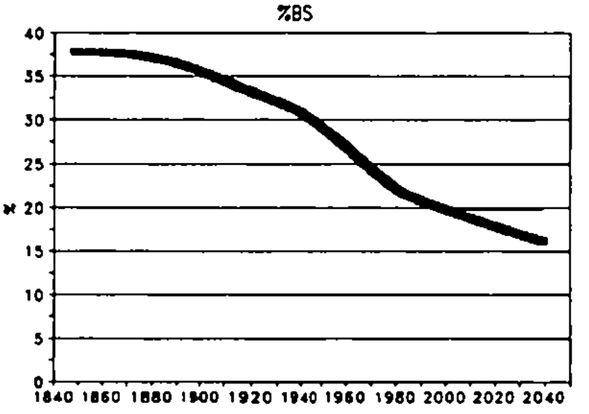

Figure 5.4d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.5. NANT-Y-BUSTACH (PREVIOUSLY REFERRED TO AS LI1, LLYN BRIANNE)

This site is an acidified forested catchment in the Cambrian Mountains of mid-Wales. Conifers were planted between 1961 and 1963 cover the entire catchment area. Prior to afforestation the catchment was under moorland vegetation and land use comprised rough sheep grazing. Monitoring of the stream commenced in the early 1980's as part of the Llyn Brianne Acid Waters Study within which this site represented an acidic mature-forest control. This programme was completed in 1990 and monthly chemical sampling is currently undertaken in conjunction with continuous flow measurement using a steep stream flume.

MAGIC has been previously applied to this site to assess the effect of upland afforestation (Whitehead et al 1988) although increased occult deposition was the only forest process considered. Further analysis of forestry impacts was carried out using the modified model incorporating changes in deposition, hydrology and uptake as the forest grows and utilising multiple optimisation techniques (Waters and Jenkins 1992). The results presented here utilise the calibrated model from this recent application.

5.4.1. Physical Characteristics

Area;	2.55 m^2
Vegetation;	Sitka spruce (1961-63)
Soil Type;	Peats, gleys and podsols
Mean Soil Depth;	1.01 m
Geology;	Mudstones, shale, grits, greywackes and glacial drift

5.4.2. Hydrochemical Characteristics

Data Period;	1989
Rainfall;	1981 mm
SO_4* Input Load;	209 meq/m^2/yr
Cl Input Load	428 meq/m^2/yr
Stream pH;	5.0
Soil Base Saturation;	27.7 %

5.5.3. Data Sources

Input Chemistry;	Warren Springs Laboratory (1990)
Output Chemistry;	Juggins et al. (1990)
Soils Data;	Reynolds and Norris (1990)

5.5.4. Model Output - Reconstructions and Predictions: Nant-y-Bustach (L11, Llyn Brianne)

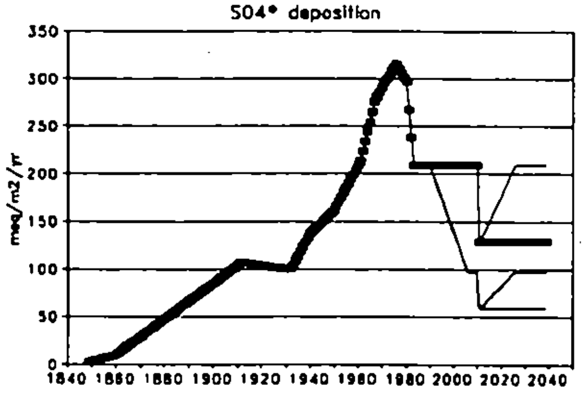

Figure 5.5a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

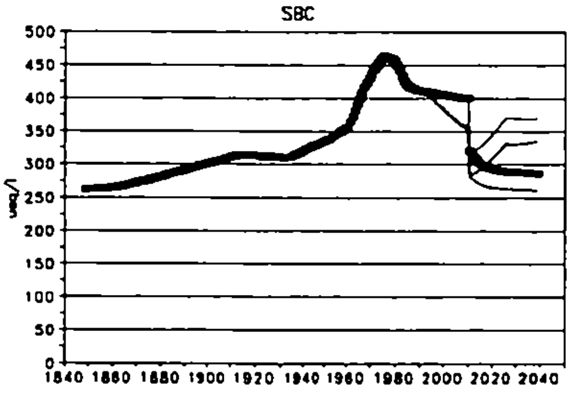

Figure 5.5c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

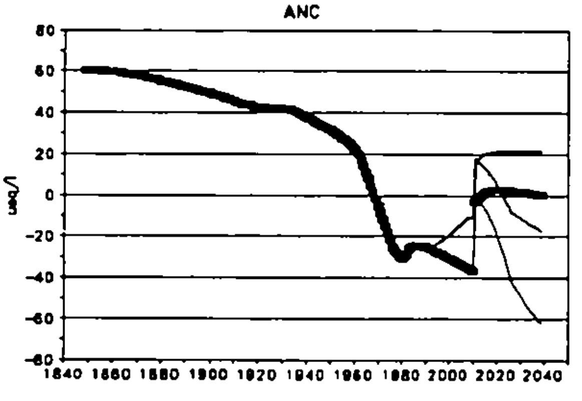

Figure 5.5b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

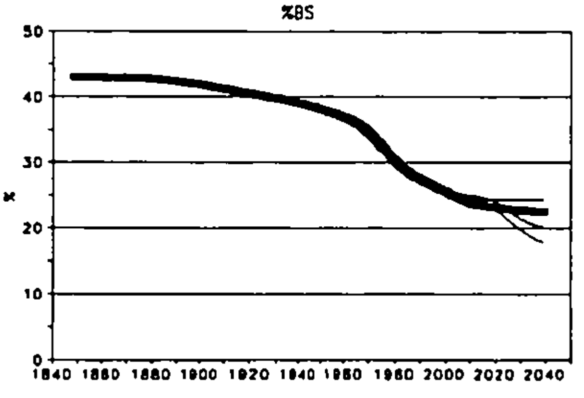

Figure 5.5d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.6. BIRKENES

The Birkenes catchment lies in the region of southern Norway receiving the highest load of acid deposition. Runoff is acidifed with annual volume-weighted pH levels of 4.4 and high concentrations of inorganic aluminum. The stream at Birkenes supported brown trout until about 1950.

Birkenes has been a calibrated catchment and research site since the early 1970's, with continuous monitoring of air, precipitation and runoff. Birkenes has been included in several national and international monitoring networks. It is also the site of several detailed studies of soil chemistry and hydrology.

MAGIC has previously been calibrated to Birkenes and used to estimate critical loads for the catchment (Wright et al. 1990a). We use this calibration in conjunction with sulphur deposition scenarios outlined in section 3.1 and to explore the interaction between target loads for sulphur, nitrogen and nitrogen saturation.

Nitrogen saturation is assumed to double every 12 years until nitrate output equals nitrate input. This follows the observed pattern of doubling over the 12-year period 1974 to 1986 observed in the 1000-lake survey (Henriksen et al. 1988). The retention of ammonium is assumed unchanged at the present level of 95%.

5.6.1. Physical Characteristics

Area;	0.41 km^2
Vegetation;	Mixed spruce, pine and birch forest, >80 years old
Soil Type;	Podsols and brown earths
Mean Soil Depth;	0.40 m
Geology;	Biotite granite, glacial drift

5.6.2. Hydrochemical Characteristics

Data Period;	1973-88
Rainfall;	1471 mm
SO_4* Input Load;	146 $meq/m^2/yr$
Cl Input Load;	146 $meq/m^2/yr$
Stream pH;	4.4
Soil Base Saturation;	17.7%

5.6.3. Data Sources

Input Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Output Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Soils Data;	A.O. Stuanes (NISK) and J.O. Reuss (1989)

5.6.4. Model Output - Reconstructions and Predictions: Birkenes

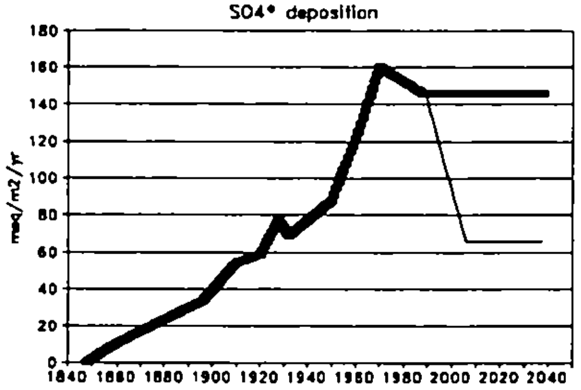

Figure 5.6a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

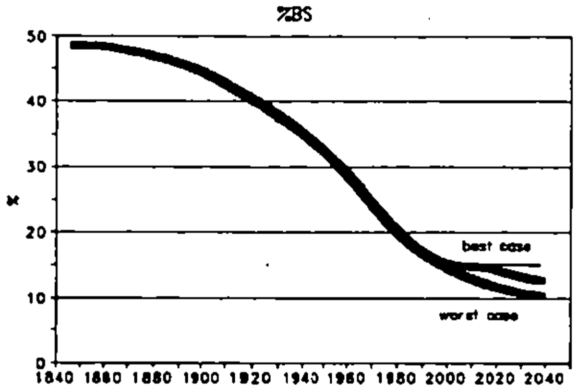

Figure 5.6d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

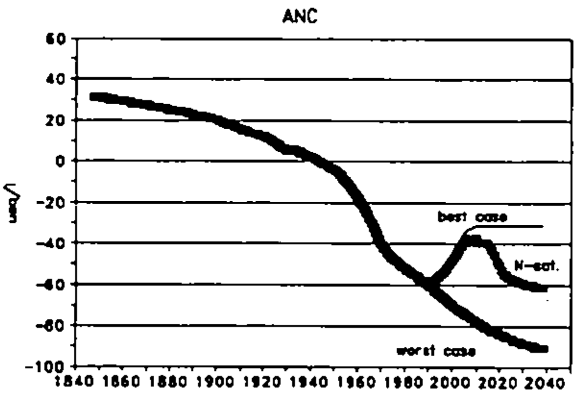

Figure 5.6b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

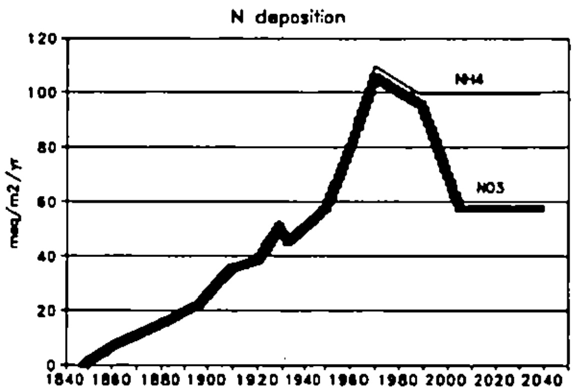

Figure 5.6e.Nitrogen deposition (meq/m2/yr) used for all scenarios.

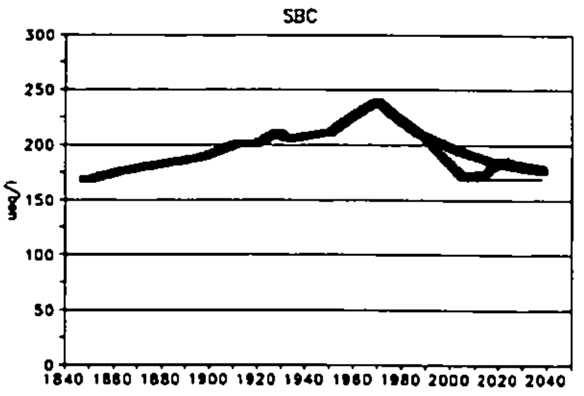

Figure 5.6c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

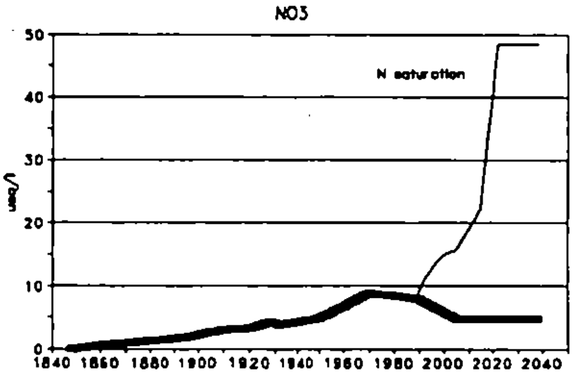

Figure 5.6f. Reconstructed historical streamwater nitrate (meq/m2) and predicted response under constant nitrogen uptake and nitrogen saturation.

5.7. STORGAMA

Storgama is an acidifed catchment recieving moderate amounts (for Norway) of acid deposition. The site is located at about 600 m above sea level in sparse and unproductive forests and moorlands. Brown trout populations in lakes in the regions were lost in the 1950's.

Storgama has been a calibrated research site since 1974 and is included in the Norwegian national environmental monitoring programme since 1980. An adjacent catchment, Tjønnstrond, is the object of a whole-catchment liming experiment carried out in 1983, with subsequent monitoring of runoff chemistry and the restocked fish popualtion.

MAGIC has previously been calibrated to Storgama and used to estimate critical loads for the catchment (Wright et al. 1990a). We use this calibration in conjunction with sulphur deposition scenarios.

5.7.1. Physical Characteristics

Area;	0.60 km^2
Vegetation;	Sparse pine and birch forest, heather
Soil Type;	Podsols and peats
Mean Soil Depth;	0.32 m
Geology;	Granite and gneiss

5.7.2. Hydrochemical Characteristics

Data Period;	1975-88
Rainfall;	1009 mm
SO_4* Input Load;	69 meq/m^2/yr
Cl Input Load	31 meq/m^2/yr
Stream pH;	4.5
Soil Base Saturation;	6.9%

5.7.3. Data Sources

Input Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Output Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Soils Data;	A.O. Stuanes (NISK) and Reuss (1989)

5.7.4. Model Output - Reconstructions and Predictions: Storgama

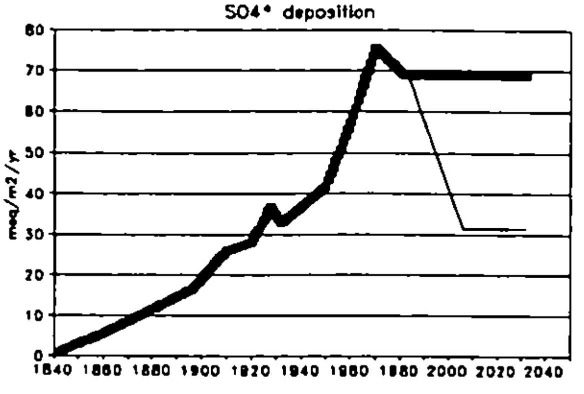

Figure 5.7a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

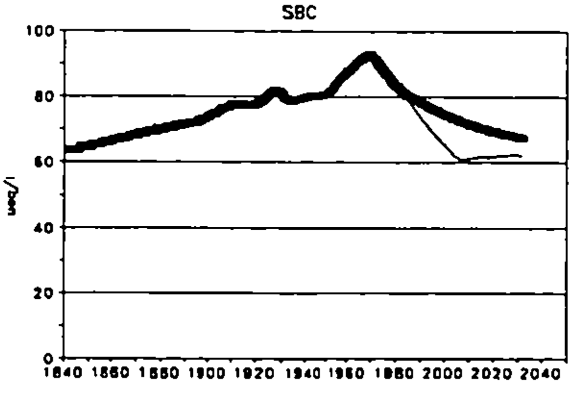

Figure 5.7c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

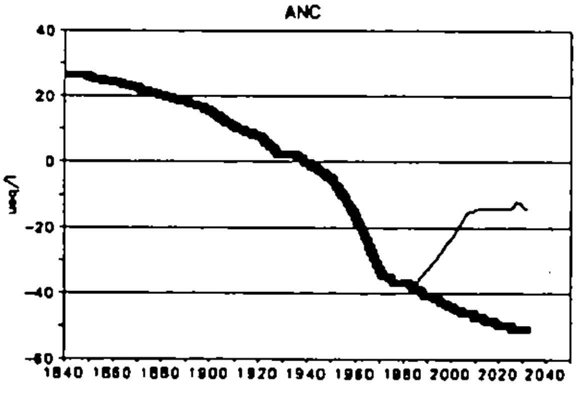

Figure 5.7b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

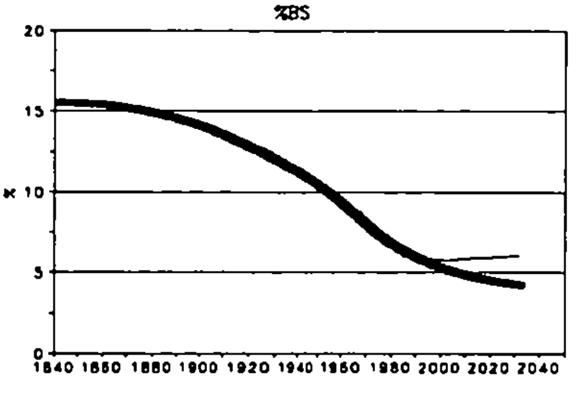

Figure 5.7d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.8. LANGTJERN

Langtjern is an acidifed lake recieving moderate amounts (for Norway) of acid deposition. The lake is typical of inland forest lakes in southern Norway. It is 0.25 km² in area, mean depth of about 3 m and water-retention time of only 2-3 months. Brown trout popuatlion in the lake was lost in the 1960's due to recruitment failure. The lake is marginally acidic (pH 4.7).

Langtjern has been a calibrated research site since 1974 and included in the Norwegian national environmental monitoring programme since 1980. Monitoring points for runoff include the 2 major inflowing streams and the outlet of the lake. Due to the short-water retention time, the chemistry of the outflow water is very similar to that of the inflow water.

MAGIC has previously been calibrated to Langtjern (outlet) and used to estimate critical loads (Wright et al. 1990a). We use this calibration in conjunction with the sulphur deposition scenarios.

5.8.1. Physical Characteristics

Area;	4.56 km²
Vegetation;	Pine and birch forest
Soil Type;	Podsols and peats
Mean Soil Depth;	0.40 m
Geology;	Granite and gneiss

5.8.2. Hydrochemical Characteristics

Data Period;	1975-88
Rainfall;	725 mm
SO_4* Input Load;	44 meq/m²/yr
Cl Input Load	10 meq/m²/yr
Stream pH;	4.6
Soil Base Saturation;	8.9%

5.8.3. Data Sources

Input Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Output Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Soils Data;	A.O. Stuanes (NISK) and Reuss (1989)

5.8.4. Model Output - Reconstructions and Predictions: Langtjern

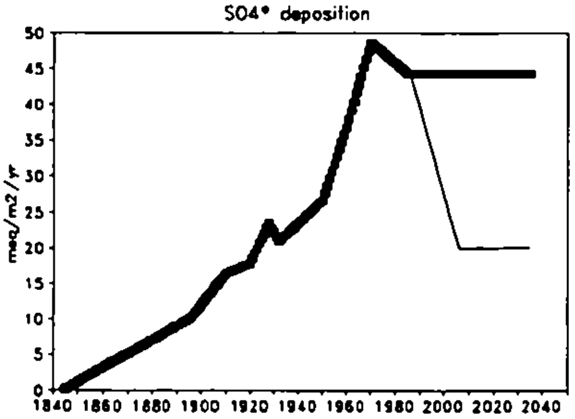

Figure 5.8a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

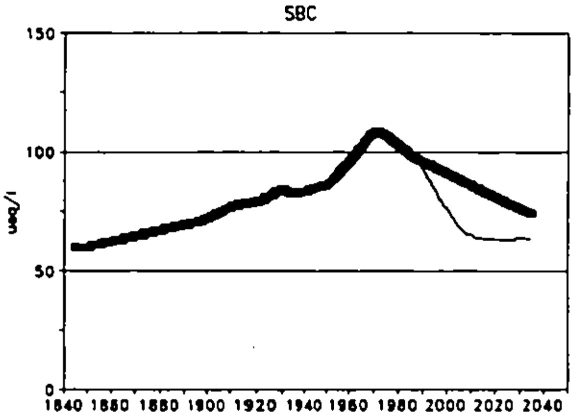

Figure 5.8c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

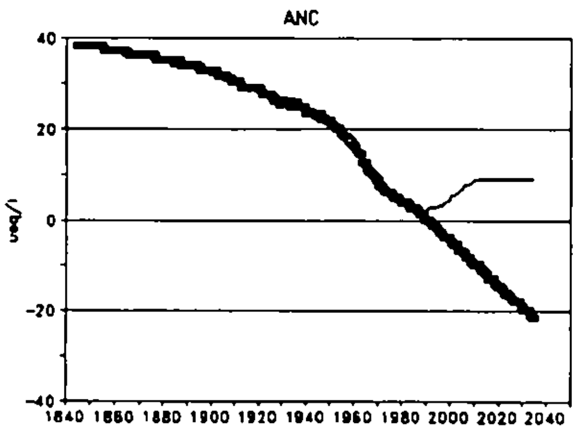

Figure 5.8b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

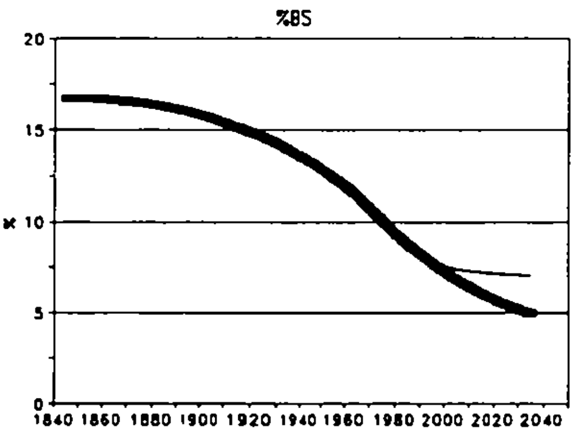

Figure 5.8d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.9. KAARVATN

Kaarvatn is a pristine but acid-sensitive catchment receiving only negligibe amounts (for Norway) of acid deposition. The site is located in the mountians of west-central Norway. Lakes and streams in the area support healthy brown trout populations.

Kaarvatn has been a calibrated reasearch site since 1978 and included in the Norwegain national environmental monitoring programme since 1980.

MAGIC has previously been calibrated to Kaarvatn and used to estimate critical loads for the cathcment (Wright et al. 1990a). We use this calibration in conjunction with sulphur deposition scenarios.

5.9.1. Physical Characteristics

Area;	25 km^2
Vegetation;	Dwarf birch, heather
Soil Type;	Podsols and peats
Mean Soil Depth;	0.29 m
Geology;	Granite and gneiss

5.9.2. Hydrochemical Characteristics

Data Period;	1978-88
Rainfall;	2000 mm
SO_4* Input Load;	15 $meq/m^2/yr$
Cl Input Load	93 $meq/m^2/yr$
Stream pH;	5.6
Soil Base Saturation;	16.3%

5.9.3. Data Sources

Input Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Output Chemistry;	Norwegian State Pollution Control Authority (SFT 1990)
Soils Data;	A.O. Stuanes (NISK) and Reuss (1989)

5.9.4. Model Output - Reconstructions and Predictions: Kaarvatn

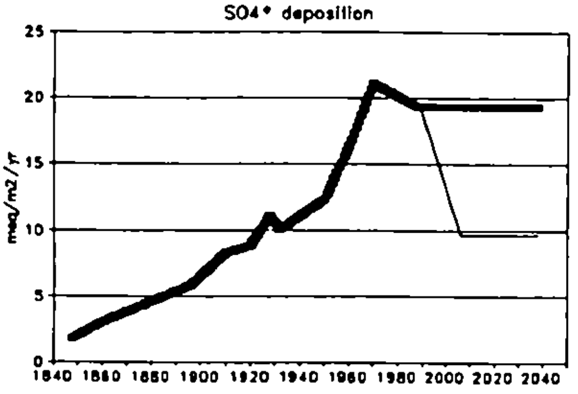

Figure 5.9a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

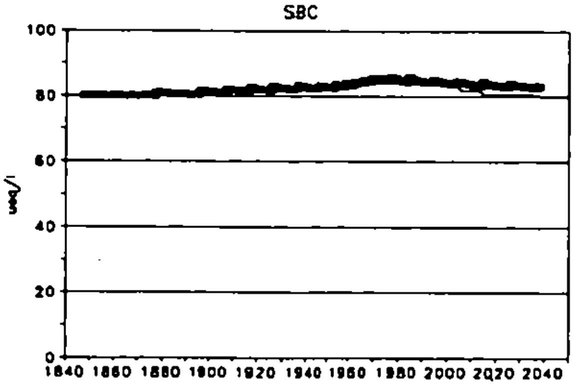

Figure 5.9c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

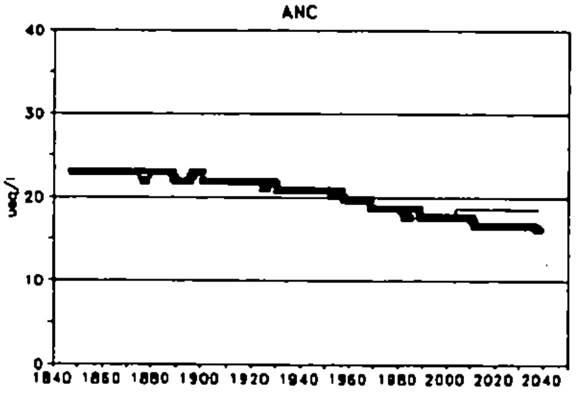

Figure 5.9b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

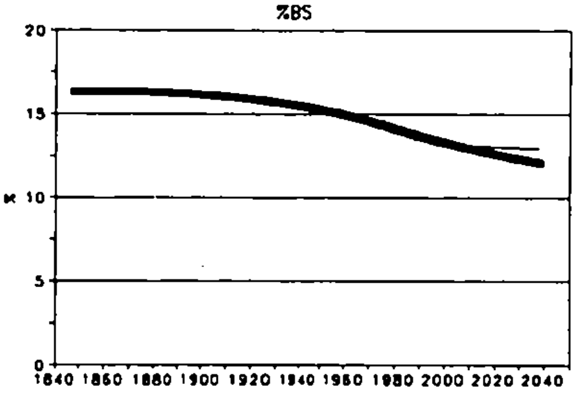

Figure 5.9d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

5.10. DALELVA

Dalelva is an acid-sensitive catchment receiving only moderate amounts (for Norway) of acid deposition. The site is located in northernmost Norway only 30 km from the major copper- nickel smelters in Nikel, Russia. Here emissions of SO_2 began in the late 1930's and peaked in 1980 at about 400 000 tons SO_2/yr. Most of the acid input at Dalelva is thus as dry- depostion of SO_2 gas.

The chemical compostion of runoff at the weir indicates water quality on the verge of becoming acidic. Lakes in the headwaters regions of the Dalelva catchment are already chronically acidic with pH < 5 and have damaged fish popualtions.

Dalelva has been a calibrated research site since 1989 and is included in the Norwegian national environmental monitoring programme.

MAGIC has previously been calibrated to Dalelva and used to estimate critical loads for the catchment (Wright and Traaen 1992). We use this calibration in conjunction with the sulphur deposition scenarios.

5.10.1. Physical Characteristics

Area;	3.2 km^2
Vegetation;	Dwarf birch, heather
Soil Type;	Podsols and peats
Mean Soil Depth;	0.46 m
Geology;	Granite and gneiss

5.10.2. Hydrochemical Characteristics

Data Period;	1987-88
Rainfall;	770 mm
SO_4* Input Load;	55 meq/m^2/yr
Cl Input Load	66 meq/m^2/yr
Stream pH;	5.6
Soil Base Saturation;	21.1%

5.10.3. Data Sources

Input Chemistry;	Norwegian State Pollution Control Authority (SFT 1991)
Output Chemistry;	Norwegian State Pollution Control Authority (SFT 1991)
Soils Data;	A.O. Stuanes (NISK) and Reuss (1989)

5.10.4. Model Output - Reconstructions and Predictions: Dalelva

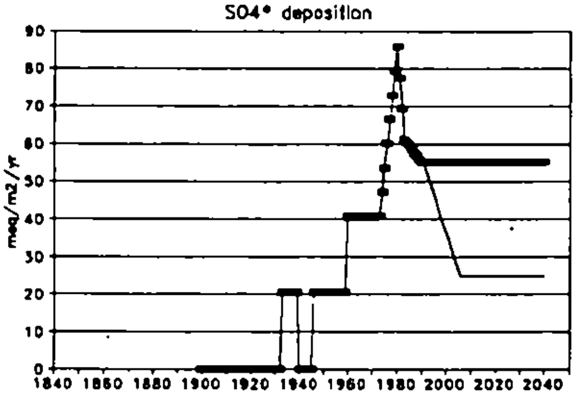

Figure 5.10a. Excess sulphate deposition (meq/m²/yr) used for best and worst case scenarios.

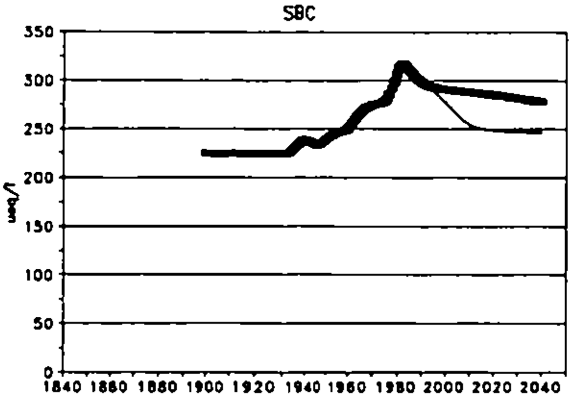

Figure 5.10c. Reconstructed historical streamwater sum of base cations (SBC meq/m³) and predicted response under each scenario.

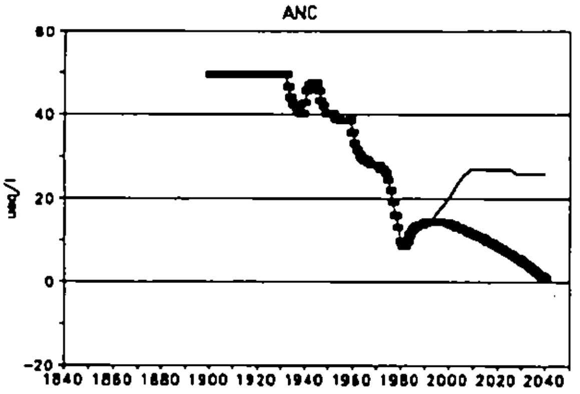

Figure 5.10b. Reconstructed historical streamwater alkalinity (meq/m³) and predicted response under each scenario.

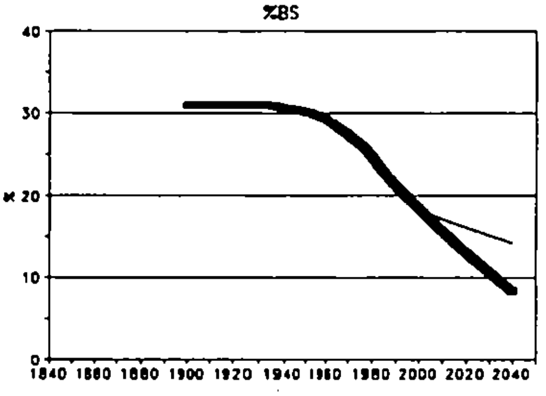

Figure 5.10d. Reconstructed historical soil base saturation (%) and predicted response under each scenario.

6. DISCUSSION

6.1. MODEL RECONSTRUCTIONS

The present state of acidification of soils and surface waters is the result of many decades of acid deposition. The response and rate of response of terrestrial and aquatic ecosystems to these acid inputs depend on many factors including intrinsic properties such as soil thickness and mineralogy, local factors such as forest harvesting, and of course the amount and history of acid deposition. Similarly the degree and rate of recovery of acidified systems following reduction in acid deposition will vary from catchment to catchment.

At all sites MAGIC successfully simulates present-day observed stream and soil chemistry confirming that the model is well suited for application to these types of areas. The sites included represent mainly cool, temperate climates and are characterised by acidic soils overlaying slowly weathering bedrock which consequently has little acid buffering capacity. Excess sulphate deposition covers a wide range from 15 to 209 meq/m^2/yr across all the sites. Chloride deposition also varies considerably between the sites from 10 to 428 meq/m^2/yr, the highest levels being in the UK sites where sea- salts dominate the input flux. Organic acid concentrations vary by a factor of 10 across all of the sites studied (Table 6.1). Mean soil depth at the UK sites is generally about double (c. 1m) the Norwegian sites and in most cases, particularly the Welsh sites, have greater rainfall inputs. This is reflected in the relatively high excess sulphate deposition loads at the UK sites.

Table 6.1. Catchment and soil parameters used in application of MAGIC to ENCORE catchments in the UK and Norway.

Parameter	Units	Gwy	Hafren	Mharcaidh	Nant-y-Gronwen	Nant-y-Bustach	Birk-enes	Stor-gama	Lang-tjern	Kaar-vatn	Dal-elva
catchment area	km2	3.88	3.47	9.98	0.72	2.55	0.41	0.60	4.56	25.00	3.20
precipitation	mm/yr	2500	2391	1066	1765	1981	1470	1010	725	2000	770
runoff	mm/yr	2170	2042	961	1601	1580	1190	970	605	1830	540
soil depth	m	0.81	1.02	0.83	1.00	1.01	0.40	0.32	0.40	0.29	0.46
porosity	ratio	0.45	0.45	0.50	0.45	0.45	0.50	0.50	0.50	0.50	50.00
bulk density	kg/m3	1073	950	924	527	991	936	503	828	764	1179
CEC	meq/kg	73	115	428	109	111	46	121	63	91	27
SO4 ads. half-sat.	meq/m3	151	600	1103	538	400	110	80	100	60	100
SO4 ads. max-capacity	meq/kg	1.00	4.58	2.90	4.48	4.00	0.90	1.00	1.00	6.00	1.00
solubility Al(OH)3	log	7.95	9.02	9.05	9.05	9.06	8.20	8.10	8.20	8.80	8.10
select. coeff. Al-Ca	log	1.49	2.53	6.70	5.27	-1.04	0.50	0.20	1.40	1.10	3.50
select. coeff. Al-Mg	log	1.67	4.88	6.70	5.33	-0.70	0.60	-0.50	1.60	0.70	5.00
select. coeff. Al-Na	log	-1.26	-0.20	2.37	0.39	-2.72	-1.90	-1.70	-2.20	-0.50	-0.20
select. coeff. Al-K	log	-5.97	-3.44	-1.52	-2.13	-5.14	-5.80	-4.30	-6.00	-5.00	-4.00
total organics, solution	mmol/m3	20	13	12	20	75	65	90	60	20	60
organic pK1	-log	4.5	4.5	4.5	4.5	4.5	4.5	4.7	4.5	4.0	4.4
organic pK2	-log	10.3	10.3	10.3	10.3	10.3	8.0	8.0	8.0	8.0	8.0
CO2, soil air	atm	0.0120	0.0066	0.0066	0.0066	0.0150	0.0050	0.0050	0.0050	0.0060	0.0036
soil temperature	oC	9.6	5.0	5.0	5.0	5.0	5.0	5.0	5.0	6.6	1.0
CO2, streamwater	atm	0.0012	0.0007	0.0007	0.0007	0.0007	0.0007	0.0007	0.0007	0.0025	0.0007
stream temperature	oC	9.6	5.0	5.0	5.0	5.0	5.0	5.0	5.0	6.6	2.0

The success of the model across this range of conditions illustrates its robustness and indicates that the major processes included in the model are correctly identified as the major mechanisms controlling catchment chemical response to acidic input.

The effect of anthropogenic sulphate deposition on soil and water acidification is well illustrated by the model reconstructions at the 10 sites. As expected, the sites which receive the highest sulphate load have suffered the greatest loss of soil base saturation and surface water alkalinity and vice versa. This cause and effect relationship is, however, far from linear since the degree of acidification at any site depends crucially upon other catchment physicochemical parameters. These include soil sulphate adsorption capability, soil depth, soil type and flux of base cations from weathering.

The model calibrations yield estimates for cation-exchange selectivity coefficients and base cation weathering rates similar among these 10 sites (Tables 6.1 and Table 6.2). Both these sets of parameters optimisation are obtained by the optimising routine used in the calibrations of MAGIC. The cation-exchange selectivity coefficients vary by several log units among sites; this range is well within that reported from field and laboratory measurements in soils and also are similar to values obtained from MAGIC application to other sites in Europe and North America.

The weathering rates estimated by the MAGIC calibrations are also similar among sites. These sites are all characterised by soils derived mainly from crystalline parent material and are highly-resistent to chemical weathering. Within this narrow range the sites with thicker soils have higher weathering rates.

Table 6.2. Results of MAGIC applications to ENCORE catchments.

Parameter	Units	Gwy	Haf-ren	Mbar-caidh	Gron-wen	Bust-ach	Birk-enes	Stor-gama	Lang-tjern	Kaar-vatn	Dal-elva
Ca saturation, soil 1983/4	%	5.8	3.7	1.6	7.8	11.7	9.1	3.3	3.7	5.9	11.7
Mg saturation, soil 1983/4	%	7.5	0.9	1.2	7.3	8.2	4.0	2.2	1.6	5.3	3.6
Na saturation, soil 1983/4	%	3.7	2.3	0.8	4.1	6.6	1.6	0.7	1.0	0.8	3.3
K saturation, soil 1983/4	%	4.0	1.4	1.1	1.7	1.2	2.8	0.7	2.6	1.8	2.5
total base saturation 1983/4	%	21.0	8.3	4.6	20.9	27.7	17.5	6.9	8.9	13.8	21.1
Ca saturation, soil 1844	%	9.5	8.8	1.9	14.9	18.6	26.3	9.0	8.8	7.2	15.8
Mg saturation, soil 1844	%	12.9	5.8	1.4	14.1	12.9	14.7	4.4	3.3	6.3	7.2
Na saturation, soil 1844	%	6.4	5.2	1.0	6.6	8.9	3.8	1.0	1.9	0.9	5.4
K saturation, soil 1844	%	4.2	2.6	1.2	2.2	2.6	3.8	1.1	2.8	1.9	2.7
total base saturation 1844	%	33.8	22.4	5.6	37.8	43.0	48.6	15.5	16.8	16.3	31.1
Ca weathering	meq/m2/yr	20.9	5.9	22.2	79.8	5.5	12.4	10.4	12.0	23.2	9.3
Mg weathering	meq/m2/yr	3.2	57.2	1.5	41.9	0.0	0.4	0.3	2.3	2.9	10.7
Na weathering	meq/m2/yr	8.7	30.6	38.9	28.9	41.2	5.9	0.0	6.5	8.0	7.8
K weathering	meq/m2/yr	0.0	6.8	4.6	7.1	0.0	1.0	0.0	0.0	2.2	0.9
total weathering	meq/m2/yr	44.7	100.5	67.2	157.7	46.7	19.7	10.7	20.8	36.3	28.7
Ca* deposition	meq/m2/yr	11.0	12.0	4.0	9.0	49.0	16.8	10.0	6.5	7.3	8.1
Mg* deposition	meq/m2/yr	-2.0	-2.0	0.0	0.0	0.0	0.0	0.5	0.9	0.2	3.3
Na* deposition	meq/m2/yr	-19.0	-13.0	-1.0	-1.0	-17.0	2.2	2.0	0.0	0.0	0.0
K* deposition	meq/m2/yr	0.0	2.0	1.0	1.0	9.0	7.6	2.8	3.3	1.6	0.0
total BC* deposition	meq/m2/yr	-10.0	-1.0	4.0	9.0	41.0	26.6	15.3	10.7	9.1	11.4
BC* weathering + dep	meq/m2/yr	34.7	99.5	71.2	166.7	95.7	46.3	26.0	31.5	45.4	40.1
SO4* deposition	meq/m2/yr	129.0	126.0	32.0	114.0	209.0	146.0	69.0	44.0	15.0	55.0
change %BS		-12.8	-14.1	-1	-16.9	-15.3	-31.1	-8.6	-7.9	-2.5	-10

All these MAGIC applications use a one-layer version of the model. This assumes that the catchment soil is uniform both spatially and vertically across the catchment. To achieve a model calibration, soil chemical and physical parameters are weighted to give mean values for the catchment. It is further assumed that all of the rainfall passes through this soil layer, except that which falls directly onto lake surfaces, and there is no flow routing represented in the model structures. Clearly, this is the most simplified representation of the catchment system. Previous modelling work, however, has indicated that the incorporation of additional soil layers and flow routing do not greatly influence the model predictions of streamwater chemistry (Jenkins and Cosby 1990) since the model aggregates soil parameters to the catchment scale and addresses only annual average ionic concentrations. For soils the predicted integrate response in each individual horizon; for deeper soils this may influence estimates for soil critical load. The use of monthly time steps has also been assessed and found to have no appreciable effect on the chemical changes predicted over periods of several years (Wright et al. 1990b). The model makes no attempt to infer short-term event-driven changes in hydrochemistry when a considerably shorter time step and more detailed physical representation of the catchment would be necessary.

6.2. MODEL PREDICTIONS

Comparisons of predictions under standard scenarios

The simulated response of soils and surface waters to the historical build up of sulphur deposition is similar at all catchments (Figure 6.1). Surface waters at all catchments continue to acidify under the worst case scenario (business as usual) and all catchments recover (or begin recovery) under the best case scenario (Figure 6.2). Soil base saturation continues to decline in all cases, albeit more slowely under the best case scenario (Figure 6.3). Exceptions are related to situations with concurrent land-use change such as the replant scenario at Hafren and LI1, or in the case of nitrogen saturation.

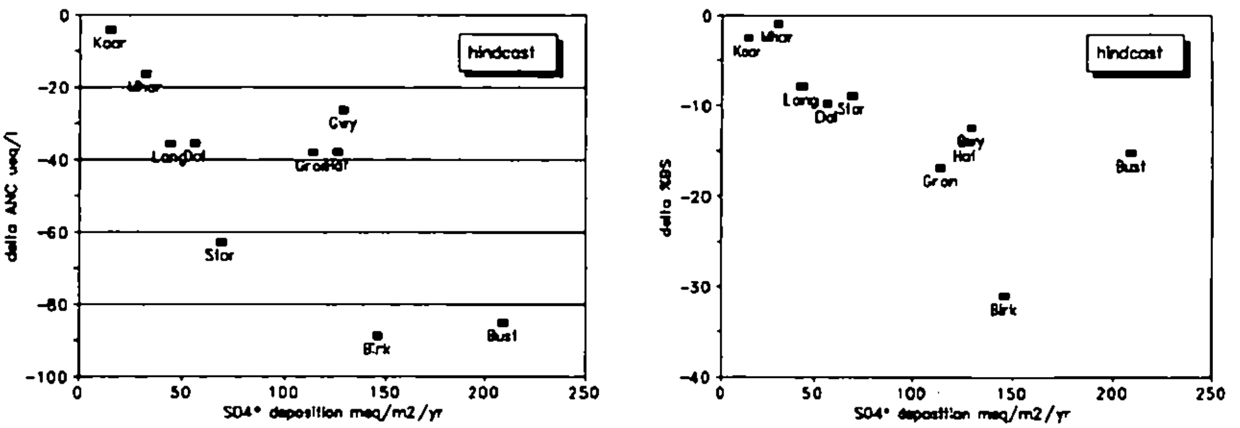

Figure 6.1. Change in streamwater ANC (left panel) and soil % base saturation (right panel) from pre-acidification (about 1840) to the present as reconstructed by MAGIC, and present-day excess-sulphate deposition at 10 ENCORE catchments.

Of the catchments investigated here Birkenes appears to have experienced the largest historical changes in ANC, and the MAGIC predictions indicate that Birkenes will also exhibit the greatest response in the future. By contrast, Mharcaidh with its relatively thick soils and low present-day sulphur deposition, has changed the least and is predicted to change little in the future under the two sulphur deposition scenarios.

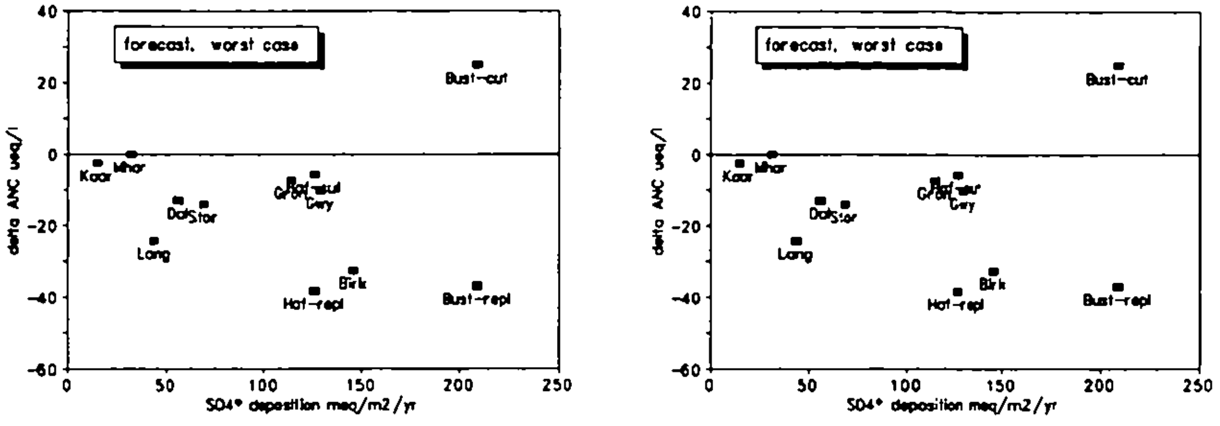

Figure 6.2. Predicted changes in streamwater ANC from present-day to 50 years in the future under woist case (left panel) and t case (night panel) sulphur deposition deposition scenario.

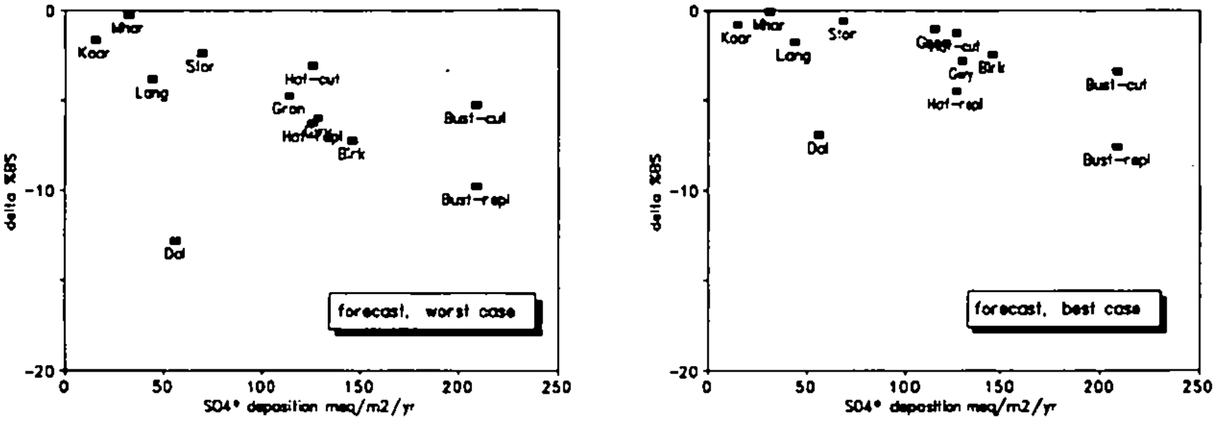

Figure 6.3. Predicted changes in soil % base saturation from pre-acidification (about 1840) to the present as reconstructed by MAGIC, and present-day excess-sulphate deposition at 10 ENCORE catchments.

Afforestation

The combined effect of catchment afforestation and high sulphate deposition has been shown to have an extreme acidifying effect on both soils and surface water. This is borne out by the model reconstructions at the Hafren and Nant-Y-Bustach sites in Wales (figures in section 5.3.4 and 5.5.4) and supports earlier MAGIC applications to forested sites in Scotland (Cosby et al. 1991, Jenkins et al. 1991).

Future response at these sites depends on both the sulphate deposition scenario and the forest management policy. At both sites, the effect of the combined worst/best case sulphate deposition and cut/replant strategies interact such that the business as usual and replanting following forest harvesting scenarios produce a further pronounced decrease in streamwater alkalinity and soil base saturation. On the other hand, the best case sulphate deposition and cut with no replanting scenarios produce a marked recovery in stream alkalinity and a stabilisation of soil base saturation over the 50-year forecast period. It is clear that future sulphate deposition reduction strategies must be formulated with land management strategies in mind.

6.3. CRITICAL LOADS

Across the range of sulphate deposition considered the empirical critical load for water compares favourably with that calculated using MAGIC. (Figure 6.3, Table 6.3). Both methods generally agree on whether each site is currently in exceedence or not. The absolute values at each site, however, show some differences. At Nant-Y-Bustach, Afon Hafren and Afon Gwy the empirical critical load is significantly higher than the MAGIC critical load whilst at Allt a Mharcaidh the opposite is the case.

At Nant-Y-Bustach and Afon Hafren this problem derives from the land use scenario employed in calculating the critical load, since the cut and replant afforestation strategy produces a significantly lower critical load than the cut and leave strategy. This is intuitively correct since the model simulates (Figures in section 5.3.4 and 5.3.5) enhanced acidification with the planting of a second rotation forest at these sites. The optimal critical load, on the other hand, does not take into account changes in land use.

The soils at the Allt a Mharcaidh are capable of buffering acidic input for the 50-year period used to determine the MAGIC critical load. Because the empirical calculation assumes equilibrium conditions, the critical load is lower. The only Norwegian site as yet unaffected by acidic deposition is Kaarvatn where again the MAGIC critical load is higher than the empirical critical load.

At all other sites, the empirical critical load is larger by up to 2 times the MAGIC critical load. This discrepancy is consistent with the problems of considering static and dynamic calculation techniques. The MAGIC model incorporates soil recovery in response to decreased sulphate input. This causes a slower recovery; so over a 50-year period a lower load is required.

Figure 6.3. Critical loads for sulphur calculated by MAGIC using forest criterion (Ca/Al = 1 mol/mol in soil solution) and water criterion (ANC= 0 μeq/l) and by the empirical method (Sverdrup et al. 1990). Also shown are present-day excess-sulphate depositions. Sites are ordered by decreasing SO_4* deposition.

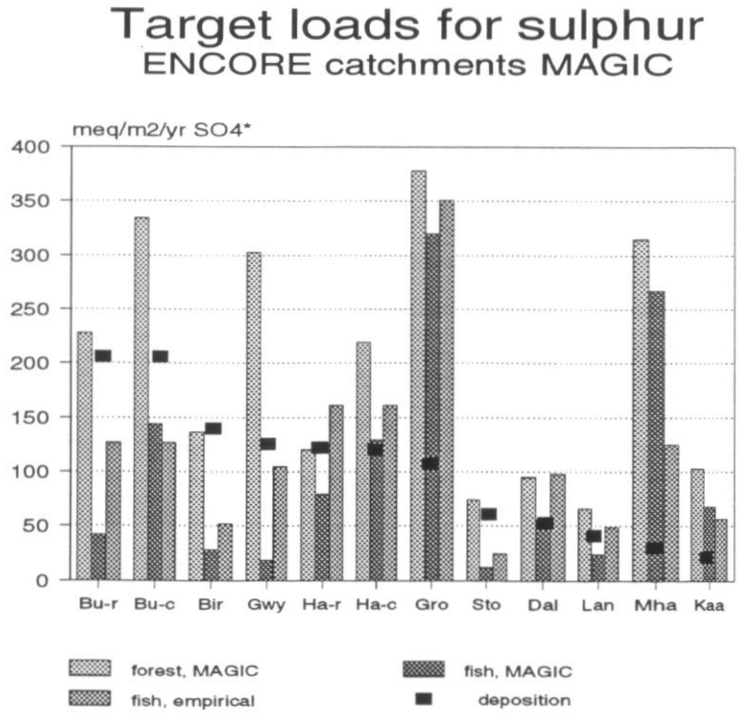

Table 6.3. Calculation of critical load (fish) by the empirical model with variable F-factor. Units: µeq/l. Source for calculation procedure Sverdrup et al. (1990).

		Gwy	Haf-ren	Mhar-caidb	Gron-wen	Bust-ach	Buk-enes	Stor-gama	Lang-tjern	Kaar-vatn	Dal-elva
runoff	m/yr	2.17	2.04	0.96	1.6	1.58	1.19	0.97	0.61	1.83	0.54
SBC 1985	ueq/l	241	308	224	343	413	212	80	98	85	299
Cl 1985	ueq/l	161	199	115	179	271	123	32	17	51	123
SO4 1985	ueq/l	72	77	48	91	152	134	74	75	16	159
alk 1985	ueq/l	-8	15	59	68	-25	-55	-37	5	18	14
SBC* 1985	= SBC -(1.11 x Cl)	62	87	96	144	112	75	44	79	28	162
F-factor	= sin(SBC*/400 x 90)	0.24	0.34	0.37	0.54	0.43	0.29	0.17	0.31	0.11	0.60
SO4*	= SO4 -(1.03 x Cl)	55	57	36	73	124	121	71	73	11	146
critical load condition alk = 0											
requured change alk =	0 - alk 1985	8	-15	-59	-68	25	55	37	-5	-18	-14
F-factor	= change SBC*/change SO4*	0.24	0.34	0.37	0.54	0.43	0.29	0.17	0.31	0.11	0.60
change SO4*	= change alk + change SBC*										
change SO4*	= change alk /(1-F)	11	-23	-94	-147	44	78	45	-7	-20	-35
critical conc. SO4* ueq/l	= SO4* 1985 - change SO4*	45	79	130	219	80	44	26	80	31	181
critical load SO4* meq/m2/yr	= critical conc. SO4* x runoff in m/yr	97	161	125	351	127	52	25	49	57	98
Cl deposition 1985 meq/m2/yr		349	406	110	286	428	146	31	10	93	66

The purpose of determining critical loads is to set goals for future deposition rate of acidifying compounds on a site-by-site basis such that the environment is protected. Critical loads are determined separately for forests, soils, and surface waters and will differ between these three categories for a given area as well from site-to-site depending upon the inherent sensitivity of the natural environment. The critical load for soils is effectively defined with respect to both waters and forests. In practice, then, at a given site the critical load for forests may be greater or lower than the critical load for water. Because the goal is to protect the whole environment the target load thus becomes the lesser of these critical loads.

In other words if the waters are inherently more sensitive than the forests, the critical load for waters will be lower than that for forests and for the environment as a whole the critical load thus becomes the critical load for waters.

For forests only the soil down through the rooting zone is of interest (for coniferous forests commonly 0-50 cm). Here the critical load for forests may be lower than the critical load for waters because runoff derives from the entire soil column including the commonly more alkaline deeper soil water.

Comparison of critical loads calculated using MAGIC shows that the critical load for soil is generally higher than that for water. It must be remembered, however, that the critical load required to protect the top 50 cm of the soil column would be lower than that tolerated by the whole soil depth, assuming soils acidify from the top down. It is possible that the critical load for a thin upper soil layer could be lower than water critical load at some sites.

Influence of N saturation on critical loads: example Birkenes

Because the deposition of both sulphur and nitrogen compounds contribute to acidification, the critical load must be specified in terms of sulfur and nitrogen deposition. The relative effects of sulfur and nitrogen deposition depend on several factors including (1) the type of receptor ecosystem in question (forest, soil or surface water), (2) the sensitivity of the ecosystem to acidification, and (3) the land-use history.

In general the effects of sulphur and nitrogen deposition are additive but not necessarily linear; the critical load for sulphur is thus lower if the ecosystem also receives acid inputs from nitrogen. The response of ecosystems to acidic inputs as either sulphur and nitrogen is dynamic over time. Damage increases with time in ecosystems receiving constant acid deposition; recovery following reductions in deposition may not proceed at the same rate as acidification.

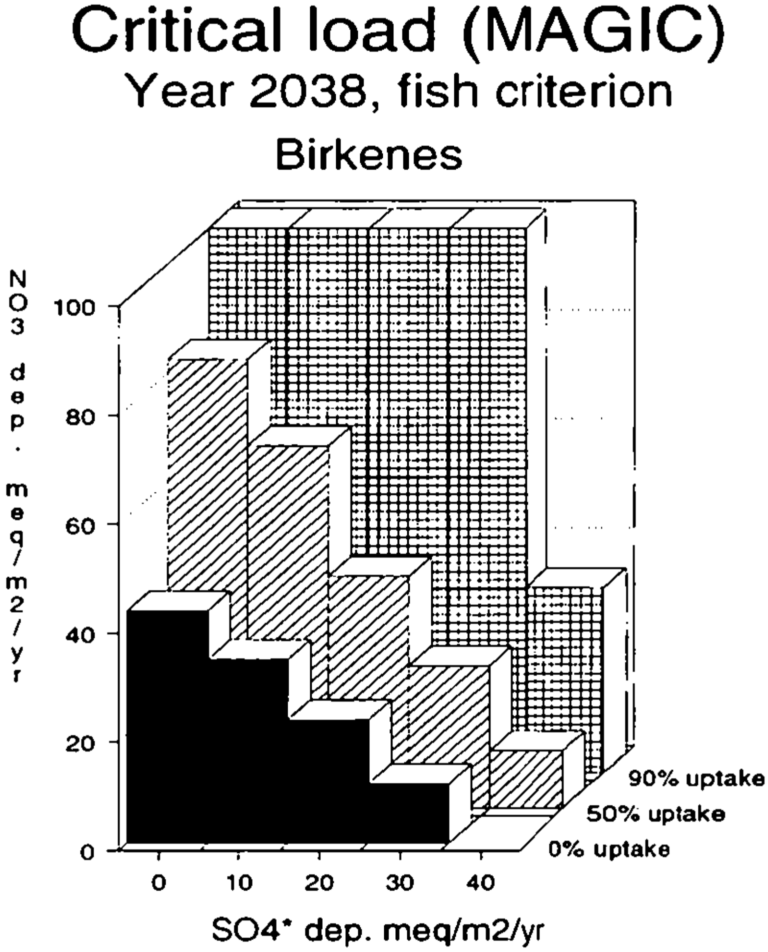

Figure 6.4. Critical load for water at Birkenes calculated by MAGIC, under criterion ANC=0 μeq/l. Shown are critical load for sulphur alone (x-axis), nitrate alone (y-axis) under 3 different scenarios of nitrogen retention in the catchment. Present-day SO_4* deposition is 146 meq/m²/yr; NO_3 deposition is 96 meq/m²/yr; nitrate uptake is 90%.

The ability of catchments to retain incoming nitrogen is a further factor influencing the target load. Indications of incipient nitrogen saturation in Norway, such as the increase in nitrate concentrations in lakes in southernmost Norway over the period 1974 to 1986 (Henriksen et al. 1988), suggest that nitrate may play a larger role in the future. MAGIC provides a tool for the evaluation of the relative roles of sulphur deposition, nitrogen deposition, and nitrogen retention by the catchment. We illustrate these relationships at the Birkenes catchment.

The critical load for sulphur at Birkenes using MAGIC, the fish criterion of ANC = 0, and target year 50 years in the future (year 2038) is 39 meq/m²/yr (Figure 6.4), under the assumptions that nitrate deposition remains constant and that the % nitrogen retained by the catchment also remains constant. If, however, the % nitrate retained decreases from present-day levels of 90% to only 50% in the future, then the catchment will tolerate less sulphur; the target load is now only about 30 meq/m²/yr. And with 0% retention of nitrate, target load for sulphur will be about 0, if nitrate deposition is at present-day levels. Thus all 3 factors must be specified to before critical load for either S or N (or both) can be quantified.

7. FUTURE WORK

We plan to continue work with the MAGIC model applied to ENCORE catchments. We envision a second report dealing with application of MAGIC to other ENCORE catchments such as Svartberget in Sweden, Lange Bramke in Germany, Aubure in France and Prades in Spain. These sites differ in several important respects to the 10 catchments in the UK and Norway included here. For example, the Lange Bramke catchment suffers forest damage, and the Prades catchment is located in a warm and arid climate. Application of MAGIC to these sites will provide further tests of the robustness of the model.

We also plan to apply MAGIC to ENCORE catchments that have been the sites of various whole-catchment manipulations. Here we have the RAIN catchments in Norway (acid addition at Sogndal and acid exclusion by roof at Risdalsheia, respectively), liming experiments at Llyn Brianne in the UK and Tjønnstrand (Storgama) in Norway, and nitrogen addition experiments at Gårdsjön in Sweden.

Finally we look for links to other models applied to data within the ENCORE project. These include short-term hydrological and hydrochemical models such as TOPMODEL (Beven et al.) and EMMA (End-Member Mixing Analysis) (Christophersen et al. 1990). Models of biological effects can also be linked to the MAGIC calibrations and prediction of the type presented here. For aquatic systems biological effects are usually related to changes in water quality. For Norway, for example, fish status is closely related to water quality variables such as ANC, and these relationships can be used to predict future changes in fish status under various scenarios of deposition of sulphur and nitrogen and land-use. These links can lead to ideas for modification of models, new field investigations, and new manipulation experiments within the ENCORE project.

8. REFERENCES

Christophersen, N. N., C., Hopper, R.P., Vogt, R.D. and Andersen S. 1990. "Modelling streamwater chemistry as a mixture of soilwater end-members - a step towards second-generation acidification models." Journal of Hydrology 116:307-320.

Cosby, B. J., Jenkins, A., Milles, J.D., Ferrier, R.C. & Walker, T.A.B. 1990. "Modelling stream acidification in forested catchments: long term reconstruction at two sites in central Scotland." Journal of Hydrology 120:143-162.

Cosby, B. J., G. M. Hornberger, J. N. Galloway and R. F. Wright. 1985. "Modelling the effects of acid deposition: assessment of a lumped-parameter model of soil water and streamwater chemistry." Water Resour. Res. 21:51-63.

Cosby, B. J., R. F. Wright, G. M. Hornberger and J. N. Galloway. 1985. "Modelling the effects of acid deposition: estimation of long-term water quality responses in a small forested catchment." Water Resour. Res. 21:1591-1601.

de Vries, W. 1988. "Critical deposition levels for nitrogen and sulphur on Dutch forest ecosystems." Water Air Soil Pollution 42:221-239.

Ferrier, R. C., Jenkins, A., Miller, J.D., Walker, T.A.B. & Anderson, H.A. 1990. "Assessment of wet depostion mechanisms in an upland Scottish catchment." Journal of Hydrology 113: 285-296.

Grieve, I. C. 1989. "A laboratory test of the soil chemical submodels of two models of catchments acidification." Hydrological Processes 3:339-346.

Henriksen, A. 1980. "Acidification of freshwaters - a large scale titration." P. 68-74. In: Ecological Impact of Acid Precipitation, D. Drabløs and A. Tollan (eds.). 1432 Aas-NLH, Norway: SNSF-project.

Henriksen, A. K., J., Posch, M. and Wilander A. "Critical loads for surface waters in the Nordic countries." Ambio (in press).

Henriksen, A., L. Lien, T. S. Traaen, I. S. Sevaldrud and D. F. Brakke. 1988. "Lake acidification in Norway - Present andpredicted status." Ambio 17:259-266.

Henriksen A. and Brakke, D. F. 1988. "Estimates of critical loads for sulfur to surface waters: examples from Norway and the eastern United States." Environ. Sci. Tech. 22:8-14.

Hornung, M. R., F. and Langan, S.J., Eds. 1990. "A review of small catchment studies in western Europe producing hydrochemical budgets." Air Pollution Research Report, 28, Brussels: Commission of European Communities.

Jenkins, A. F., R.C., Walker, T.A.B. & Whitehead, P.G. 1988. "A Modeling Study of Long Term acidification in an upland Scottish Catchment." Water, Air and Soil Pollution 40:275-291.

Jenkins, A. W., P.G., Cosby, B.J. and Birks, H.J.B. 1990. "Modelling long term acidification: a comparison with diatom reconstructions and the implications for reversibility." Phil. Trans. R. Soc. Land. B 327:435-440.

Jenkins, A. and B. J. Cosby. 1989. "Modelling surface water acidification using one and two soil layers and simple flow routing." In: Regional Acidification Models, J. Kamari, D. F. Brakke, A. Jenkins, S. A. Norton and R. F. Wright (eds.). Berlin: Springer-Verlag.

Juggins, S. W., D., Patrick, S., Jenkins, A. and Beaumont, W.R.C. 1990. The United Kingdom Acid Waters Monitoring Network Data. Report for 1989-1990 (Year 2), ENSIS Ltd., London, UK.

Neal, C. S., C.J., Walls, J. and Dunn, C.S. 1986. "Major, minor and trace element mobility in the acidic upland forested catchment of the upper River Severn, Mid-Wales." Q. J. Geol. Soc. 143:635-648.

Neal, C. R., A. Reynolds, B. and Jenkins, A. 1991. "Prediction of future short-term stream chemistry - a modelling approach." Journal of Hydrology 130:87-103.

Neal, C. W., P.G. & Jenkins, A. 1988. "Reversal of acidification." Nature 334:109-110.

Nilsson, J. and P. Grennfelt. 1988. Critical loads for sulphur and nitrogen, NordicCouncil of Ministers, Copenhagen.

Nolan, A. L., A. and Robertson, J.S. Surface water acidification programme: the soils of the Allt a Mharcaidh catchment. Aberdeen: MLURI.

Reuss, J. O. 1990. Critical loads for soils in Norway. Analyses of soils data from eights Norwegian catchments, Norwegian Institute of Water Research, Oslo.

Reynolds, B. H., M. and Hughes, S. 1989. "Chemistry of streams draining grassland and forest catchments at Plynlimon, mid-Wales." Hydrological Sciences Journal 34 (6):667-685.

Reynolds, B. N., B., Hornung, M., Hughes, S., and Stevens, P.A. 1988. "Impact of afforestation on the soil solution chemistry of stagnopodzols in mid-Wales." Water, Air and Soil Poll. 38: 55-70.

Reynolds, B. &. Norris, D.A. 1990. Llyn Brianne Acid Waters Project: Summary of Catchment Characteristics, ITE, Bangor Research Unit, Bangor, UK.

Robson, A. J., A. and Neal, C. 1991. "Towards predicting future episodic changes in stream chemistry." Journal of Hydrology 125:161-174.

SFT. 1990. Monitoring of long-range transported polluted air and precipitation. Annual Report 1989, State Pollution Control Authority, Oslo.

SFT. 1991. Monitoring of long-range transported polluted air and precipitation. Annual report 1990, State Pollution Control Authority, Oslo.

Sverdrup, H., W. de Vries and A. Henriksen. 1990. Mapping critical loads, Nordic Council of Ministers, Copenhagen.

Warren Spring Laboratory. 1990. Acid deposition in the United Kingdom 1986-1988, Warren Spring Laboratory, Stevenage, UK.

Waters, D. &. Jenkins, A. 1992. "Impacts of afforestation on water quality trends in two catchments in mid-Wales." Environmental Pollution 43:In Press.

Whitehead, P. G., Reynolds, B., Hornung, M., Neal, C., Cosby, B.J. and Paricos, P. 1988. "Modeling long-term stream acidification trends in upland Wales at Plynlimon." Hydrological Processes 2:357-368.

Whitehead, P. G., Birds, S., Hornung, M., Neal, C., Cosby, J. & Paricos, P. 1988. "Stream acidification trends in the Welsh uplands: A modelling study of the Llyn Brianne catchments." Journal of Hydrology 101:191-212.

Wright, R. F., B. J. Cosby, M. B. Flaten and J. O. Reuss. 1990. "Evaluation of an acidification model with data from manipulated catchments in Norway." Nature 343:53-55.

Wright, R. F., B. J. Cosby and G. M. Hornberger. 1991. "A regional model of lake acidification in southernmost Norway." Ambio 20:222-225.

Wright, R. F., A. O. Stuanes, J. O. Reuss and M. B. Flaten. 1990. Critical Loads for Soils in Norway. Preliminary assessment based on data from 9 calibrated catchments, Norwegian Inst. Water Research, Oslo.

Wright, R. F. and T. S. Traaen. 1992. Dalelva, Finnmark, northernmost Norway: prediction of future acidification using the MAGIC model, SFT, Oslo.