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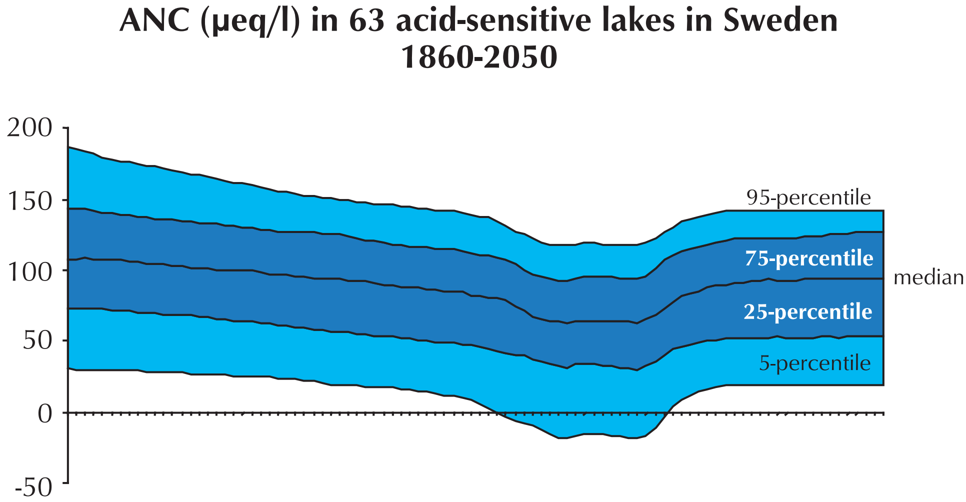
Dynamic modelling of soil and water acidification: display and presentation of results for policy purposes

Acid Rain Research

REPORT 56/03



Dynamic modelling of soil and water acidification: display and presentation of results for policy purposes

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www.ivl.se/om_ivl/partners/nmr/rapporter/Dynamic_modelling.pdf

Preface

This report is the result of a workshop held by at Gårdsjön, Sweden, 23-25 September 2003. The workshop was an activity within the Nordic Council of Ministers, Sea and Air Group project "Dynamic modelling of soil and water acidification: display and presentation of results for policy purposes", NMR project number 03/02. The report is available as pdf file on:
www.ivl.se/om_ivl/partners/nmr/rapporter/Dynamic_modelling.pdf

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Oslo, October 2003

Richard F. Wright

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Chapter 1
Acidification – introduction

What is acidification?

"Acidification" means that water, such as rain, streams, lakes and ground water becomes more and more acid, and hence harmful to plants and animals. When acid rainwater falls on the ground and passes through the soil, the soil will try to counteract or "buffer" the acidity. This buffering is done largely by exchanging the acids in the water with base cations (calcium, magnesium, sodium and potassium) in the soil or by releasing aluminium. Acidification will slowly deplete the pools of base cations in the soil and increase the amount of aluminium in the water leaving the soil and entering the groundwater, streams and lakes.

How do we measure acidification ?

pH – Acidification is caused by protons which are measured by the pH scale going from 1-14. The more protons in the water, the lower the pH value. Neutral water has a pH of 7. When pH falls below 5.5, the fish populations can be damaged and even die, and at pH below 5.0, the lakes and streams become more or less barren of fish.

Base saturation – The soil particles have a given capacity to attach base cations to their surfaces. The fraction of this capacity occupied by base cations is called the base saturation - the more base cations in the soil the higher the base saturation. As acidification progresses, the base cations are lost and replaced by acid cations (hydrogen and aluminium) and the base saturation decreases.

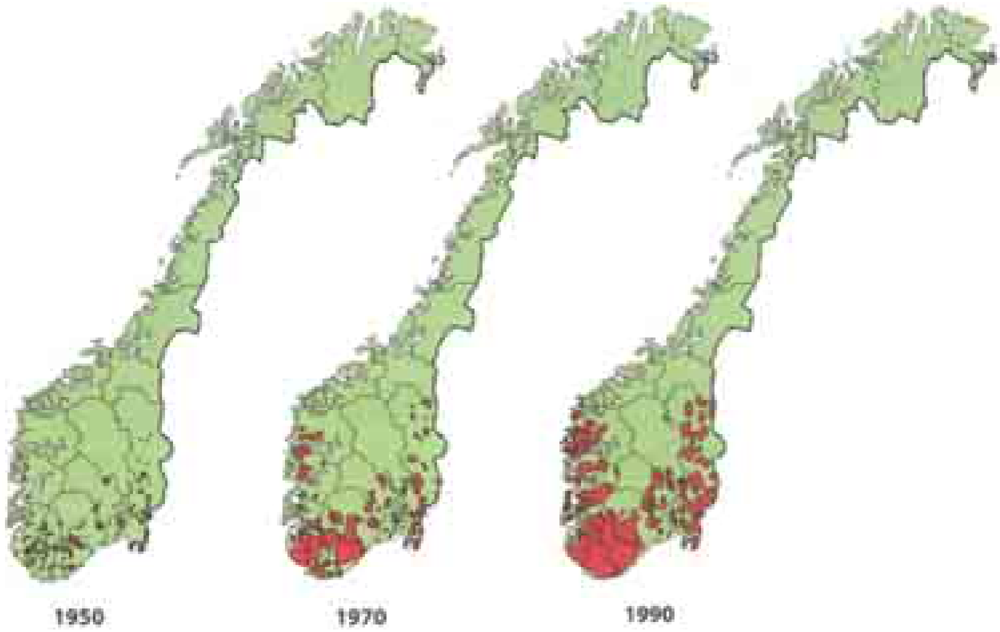
ANC – Acid Neutralising Capacity – is a measure of the ability of the soils and water to buffer the acidity. It expresses the relationship between base cations (SBC) and the strong acid anions (SAA). A low ANC means that the ability to buffer the acidity is small and the lake or soil is sensitive to further acidification. ANC = SBC – SAA. SBC = sum of concentrations of Ca+Mg+Na+K; SAA = sum of concentrations of $Cl+SO_4+NO_3$; units: µeq/l.

BC/Al ratio – Acidification removes base cations and releases aluminium. The ratio between the base cations and aluminium in soil solution will decrease as acidification progresses. Aluminium is toxic to plants and animals. Trees and other plants may be damaged when the BC/Al molar ratio falls below about 1.

What are the effects of acidification?

Acidification will have negative consequences for plants and animals in both terrestrial and aquatic ecosystems. Base cations are important nutrients for plants, and the depletion of the base cation pool in the soil means that there are fewer nutrients available for the plants. When the base cation pool is no longer sufficient to buffer the acid rain water, aluminium is released, which is toxic to roots in the soil and animals such as fish in the streams and lakes. Hence, acidification will damage plant and animal life.

Fish populations in Norwegian lakes and rivers have been dramatically affected in the period 1950-1990 (Hesthagen et al. 1999), and forest dieback became widespread in Czechoslovakia in the 1970s and 1980s.



Areas with damaged and lost fish populations in Norway (SFT, 2000)

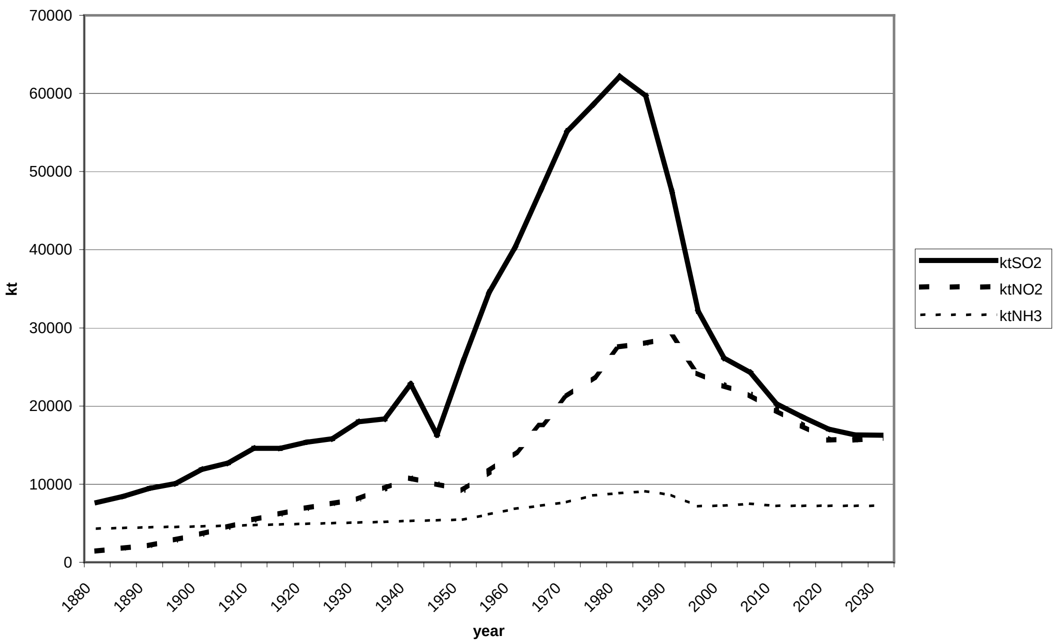
Extensive deforestation in the mountains in Czech Republic was most likely caused by high SO_2 concentrations in the air. Photo: Jiri Cerny,1985.

What causes acidification?

Acidification is caused mainly by the emission of air pollutants –sulphur dioxide and nitrogen oxides as well as ammonia. The oxide gases are formed when fossil fuels (coal, oil and gas) are burnt and are emitted into the atmosphere. In the air, they form sulphuric and nitric acids, which cause the rain to become acidic. Ammonia comes mainly from agricultural activities. Therefore, there is a link between human activities (industry, power production, transport, agriculture) and acidification. Since the acidifying compounds are emitted into the atmosphere, the acid can be transported over large distances – it is a transboundary problem. Emissions of acidifying gases in Europe increased dramatically during the 1900s but began to decline starting in the 1980s, largely as a result of international agreements to reduce transboundary air pollution.

Acidification in the future?

Since the acidification is mainly caused by air pollution with sulphur and nitrogen oxides, reductions in the emissions of these compounds will reduce acidification. Up through the 1990s, the emissions were reduced. The future acidification therefore depends on how much we reduce the emissions.



Historical and future emissions of SO_2, NO_2 and NH_3 in Europe (from Schöpp et al. 2003).

Recovery of damaged ecosystems

Recovery definitions (see Gunn and Sandøy 2003):
- “the goal of recovery is to obtain the biological community which is expected in conditions of minimal anthropogenic impact”
- “Recovery in streams and lakes has taken place if a healthy key species of fish and key species chosen from other components of the community have returned”
- “Biological recovery occurs when a number of key organisms have resumed their role in an ecological system by re-establishing viable populations”.

Environmental goals are to stop acidification and damage to natural ecosystems. This means that emissions of acidifying compounds to the atmosphere must be reduced. When emissions and deposition are reduced, it will take time before the ecosystems recover. The amount of recovery depends on how damaged the ecosystem is and how much the deposition is reduced.

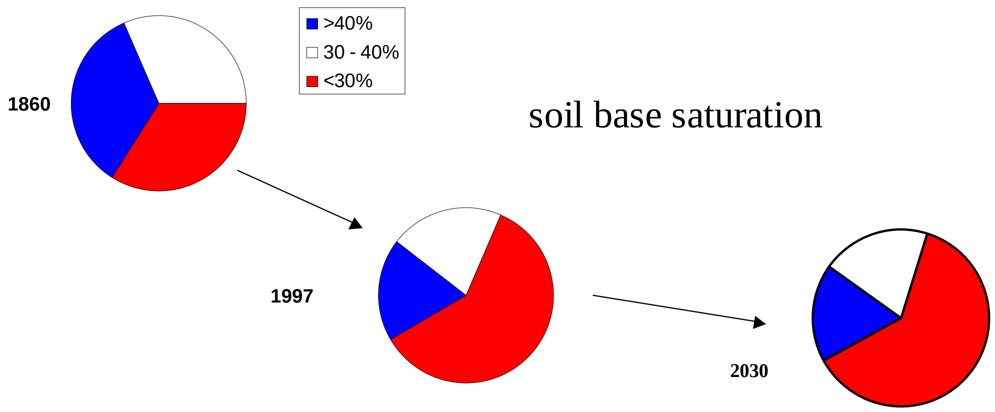
Chapter 2
Time lags

In nature there is often a time delay from the time at which a stress factor is indtroduced to the biological response. Acidification of soils and water was delayed by several decades after the emissions of S and N increased, and recovery from acidification may also lag behind decreases in deposition. Dynamic models are tools to quantify these time lags.

Chemical time lags

Soil base saturation

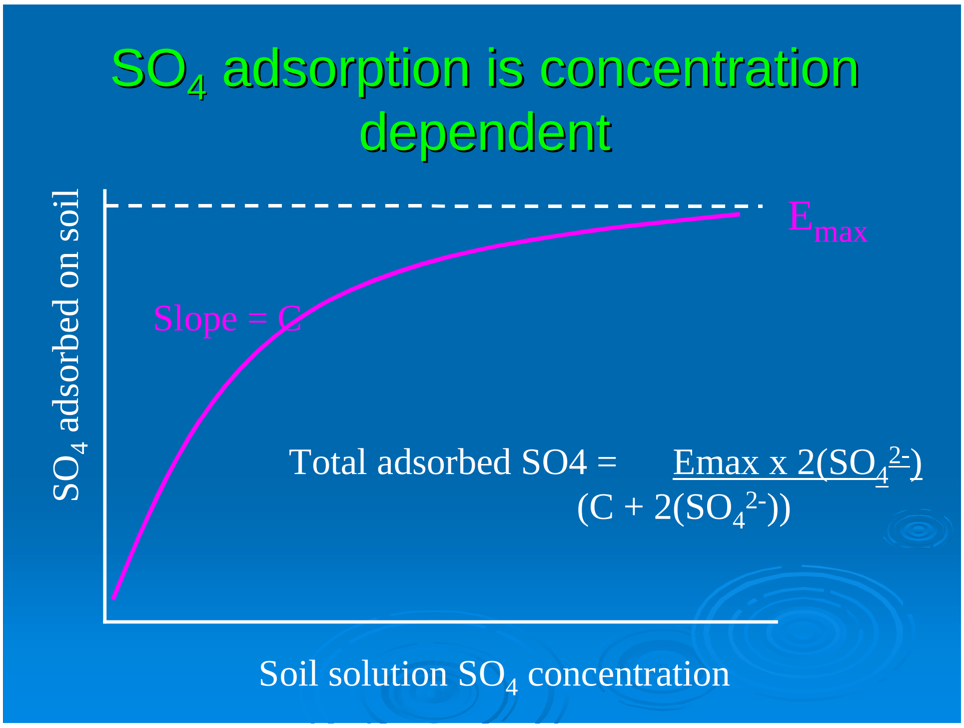
There are chemical time lags, in which chemical changes of surface water or soil is delayed due to geochemical processes. For example the amount of exchangeable base cations Ca, Mg, Na and K in the soil, i.e. soil base saturation, is typically relatively large compared to annual input (from weathering and deposition) and the annual outputs (uptake to biomass and leaching to runoff and to groundwater). Consequently, the base saturation changes very slowly even when the acid deposition changes substantially over only a few years.



Modelled soil base saturation in 143 Swedish lake catchments.

Sulphate adsorption

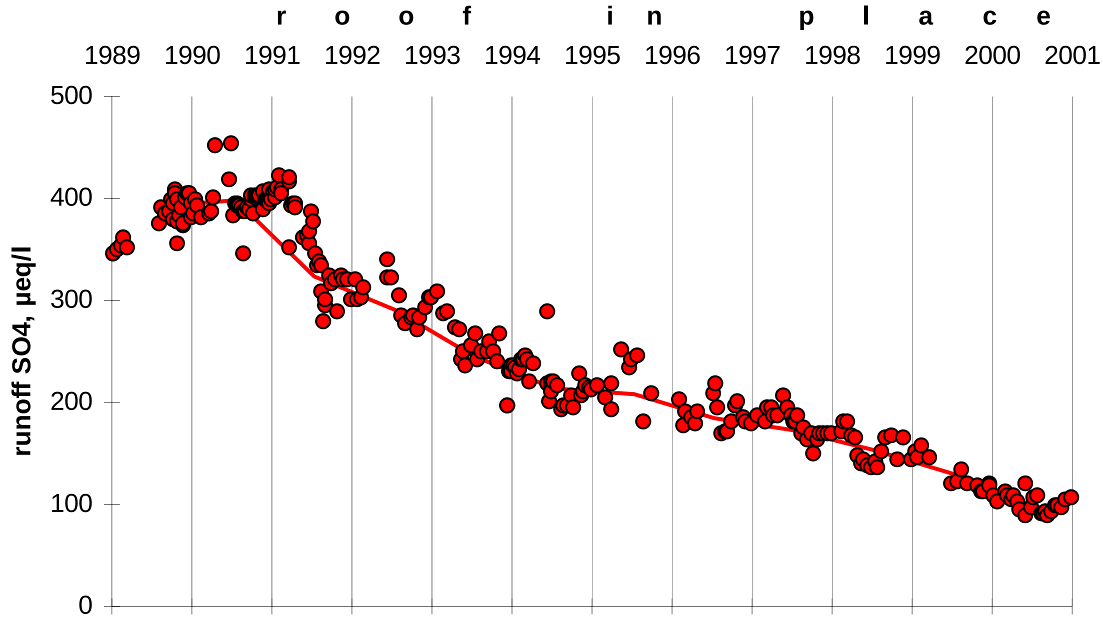
The concentration of sulphate (SO_4) in surface waters may also lag behind changes in deposition of S. This is due to storage of S in the soil, for example, adsorbed on soil particles, or bound as reduced sulphur compounds. The retention and loss is related to the size of the pool and the concentration of SO_4 in soil solution. When the input of S to the soil increases, the increase in the SO_4 in the runoff is delayed, because some of the S input gets adsorbed in the soil. Similarly, when the deposition decreases, the runoff SO_4 will not decrease equally fast since the soil will release stored SO_4.



The figure shows sulphate adsorption as described by the Langmuir isotherm.

An excellent illustration of the delayed response in runoff SO_4 is the roof experiment at Gårdsjön, Sweden (Hultberg and Skeffington 1998). In order to study the recovery process, in 1991 a 6300-m^2 plastic roof was constructed over the small catchment near lake Gårdsjön, Sweden. The roof intercepted the acid deposition and clean rain was sprinkled under the roof. The runoff response was gradual. It took ten years for runoff SO_4 concentration to begin to stabilise at the new much lower level (Moldan 1999).

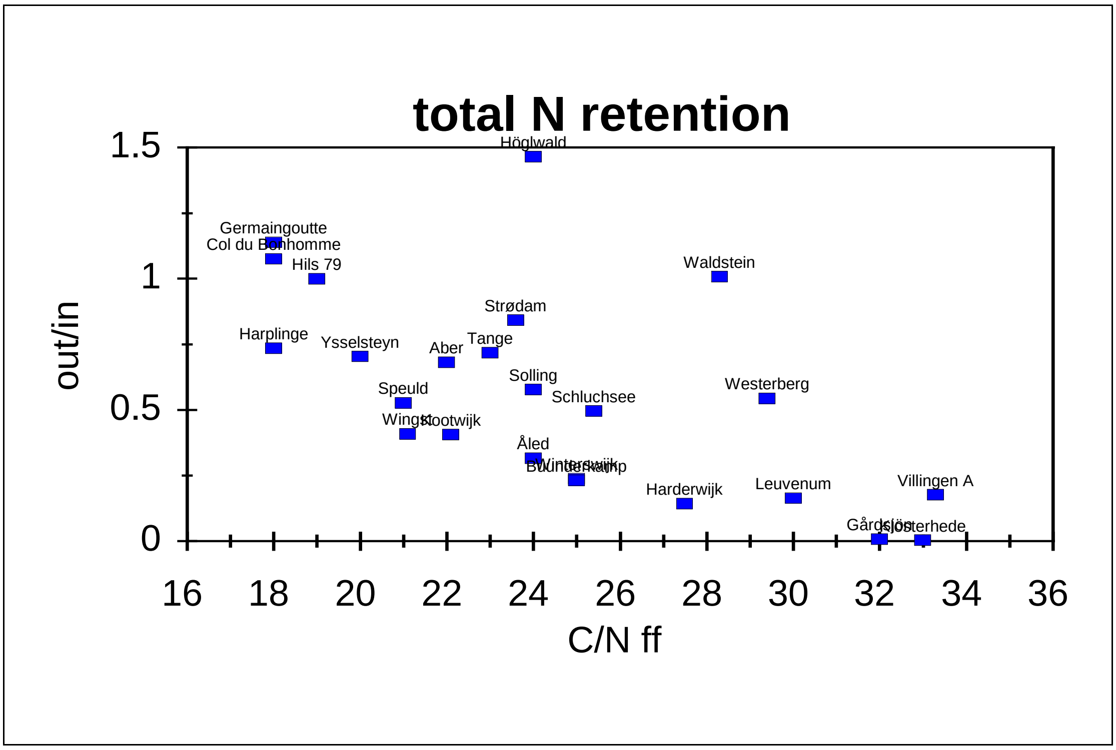
A plastic roof was constructed over the catchment G1 at Gårdsjön, Sweden in 1991.



Runoff sulphate in the Gårdsjön roof experiment started to decline shortly after the ambient precipitation was replaced by the clean water sprinkled under the roof in spring 1991. The decline was, however, gradual, and after the ten years of the experiment the runoff sulphate was still declining (Moldan 1999).

Nitrogen dynamics

The effects of increased nitrogen (N) deposition might also be delayed by several decades. N is typically the growth-limiting nutrient in undisturbed forest ecosystems. An increased input of nitrogen from deposition therefore often stimulates growth and has not much of adverse effect from the acidification point of view. (It might change species composition though.) However chronic elevated deposition of N over long time may eventually shift ecosystems from N limited to N saturated, a condition in which a fraction of the incoming N is no longer retained in the terrestrial ecosystem but it is leached to runoff and to ground waters. This can lead to acidification of soils and waters. The C/N ratio of the uppermost soil layer, the forest floor, provides a good indicator of the N status of the forest, and thus a measure of the potential response to increased N deposition. The more N-poor the forest floor (high C/N), the more N deposition can be retained before the ecosystem begins to leach NO_3 to runoff.



Relationship between fraction of N leached (out/in) to C/N (g/g) in forest floor at forest stands and catchments in Europe (after Gundersen et al. 1998).

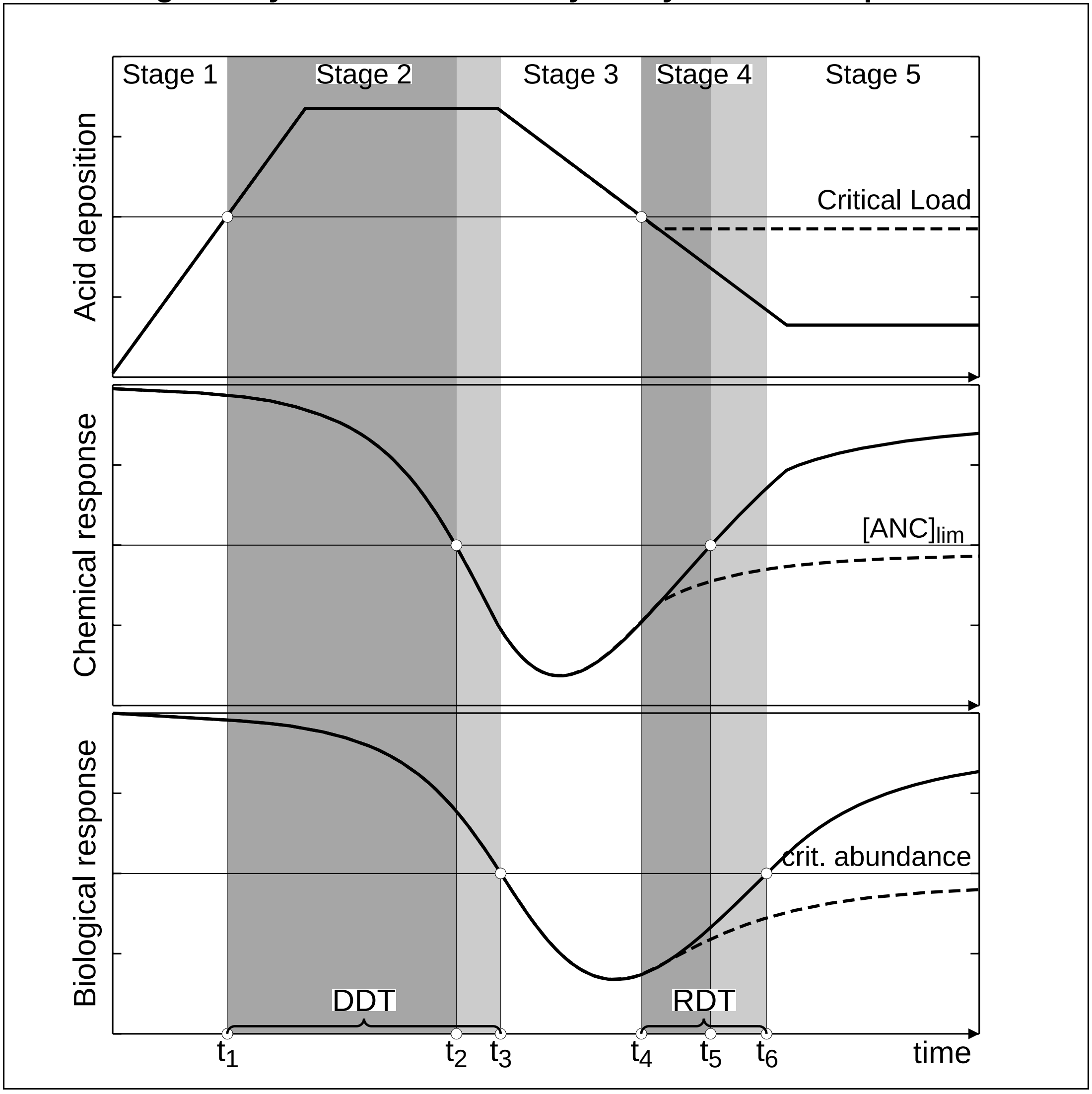
Biological recovery time lags

Recovery of chemical conditions is a precondition to biological recovery. Just as there are delays between changes in acid deposition and surface water chemistry, there are delays between changes in chemistry and the biological response. Because the goal in recovery is to restore good or healthy population of key indicator organisms, the time lag in response is the sum of delays in chemical and biological response. Thus dynamic models for biological response for surface waters are needed. Active research on the subject is ongoing in several countries in North America and Europe (see Gunn and Sandøy 2003).

Once the chemical threshold is reached, an approximate lag times for common, widely distributed species were identified at Workshop on Models for Biological Recovery from Acidification in a Changing Climate, 9-11 September 2002 in Grimstad, Norway (Wright and Lie 2002). The time lags for individual species were identified as follows:

- Algae: 1-2 years
- Macroinvertebrates: 1-3 years in streams (for first appearance of sensitive species; normal population 5-10 years). 1-10 years in lakes
- Zooplankton: 1 year (species with resting stages in sediments) ->10 years (for whole communities).
- Fish: 2-20 years.

The Damage Delay Time and Recovery Delay Time concept

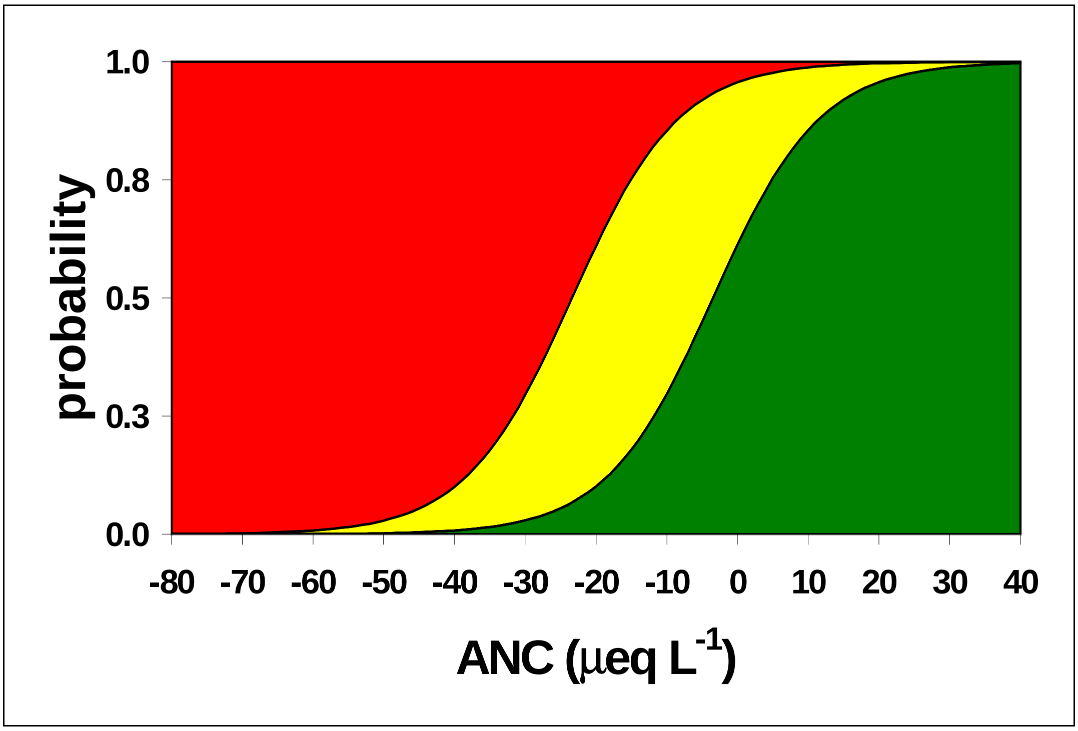


Conceptual patterns of acid deposition effects on a lake chemical variable (ANC) and a corresponding biological response variable (critical abundance of a fish species) during increasing and decreasing deposition are summarised in the figure above. Criterion values for acceptable minimal levels of the chemical and biological variables are indicated as horizontal lines, along with the critical load of deposition that will produce these levels. The delays between the exceedence of the critical load (t_1), the violation of the critical chemical criterion (t_2), and the crossing of the critical biological response (t_3) are indicated in grey shades, highlighting the Damage Delay Time (DDT). Similar delays in chemical and biological recovery during deposition reductions (t_4, t_5, and t_6) define the Recovery Delay Time (RDT) of the system (from Posch et al. 2003c).

Chapter 3
Critical limits: the link between chemistry and biology

Dynamic models predict changes in water and soil chemistry in response to changes in acid deposition. A key chemical variable is the acid neutralising capacity (ANC). Effects on organisms such as fish are currently based on empirical relationships, such as that relating fish population status to ANC in Norwegian lakes.

The diagram shows that if ANC is below about 20 µeq/l there is a 5% probability (risk) that the population will be damaged (yellow or green), and if ANC is below 0 µeq/l there will be a 50% probability (risk) that the population will be damaged. The value of ANC = 20 µeq/l is often used as the “critical limit” for biological damage in determining the critical load (see chapter 4) for freshwater ecosystems.



There are similar dose-response relationships for other species of fish and other groups of organisms, such as aquatic invertebrates and diatoms. These have been developed for various geographic regions in Europe and North America.

At present there are no dynamic models available for biological response. These would predict the time lags (see chapter 2) between changes in chemistry and the biological response.

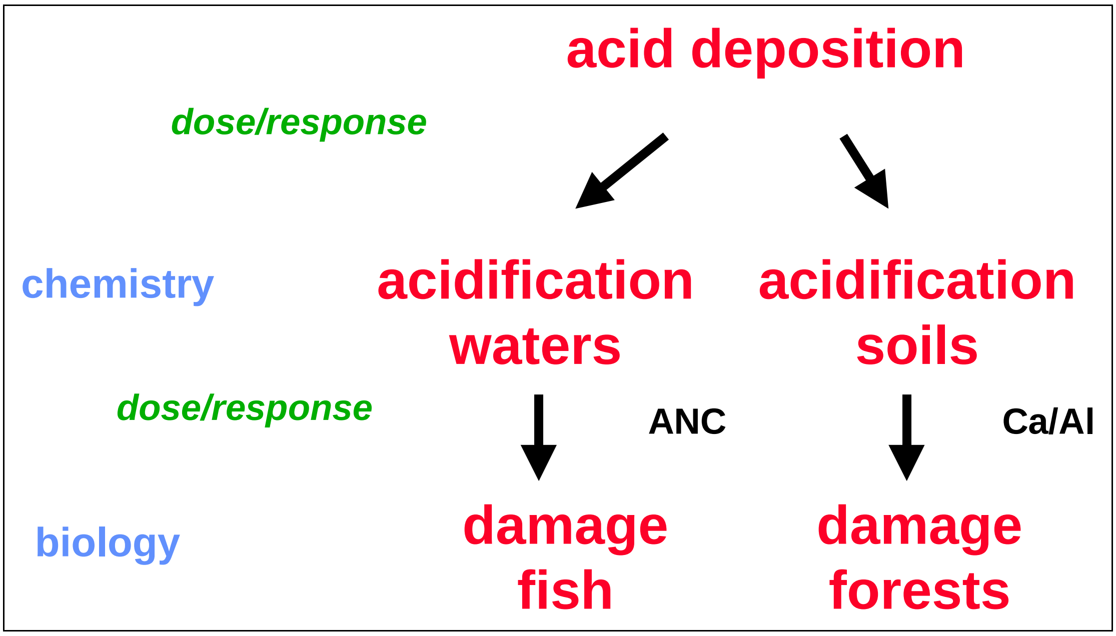
Chapter 4
Critical loads

Critical load is defined as:

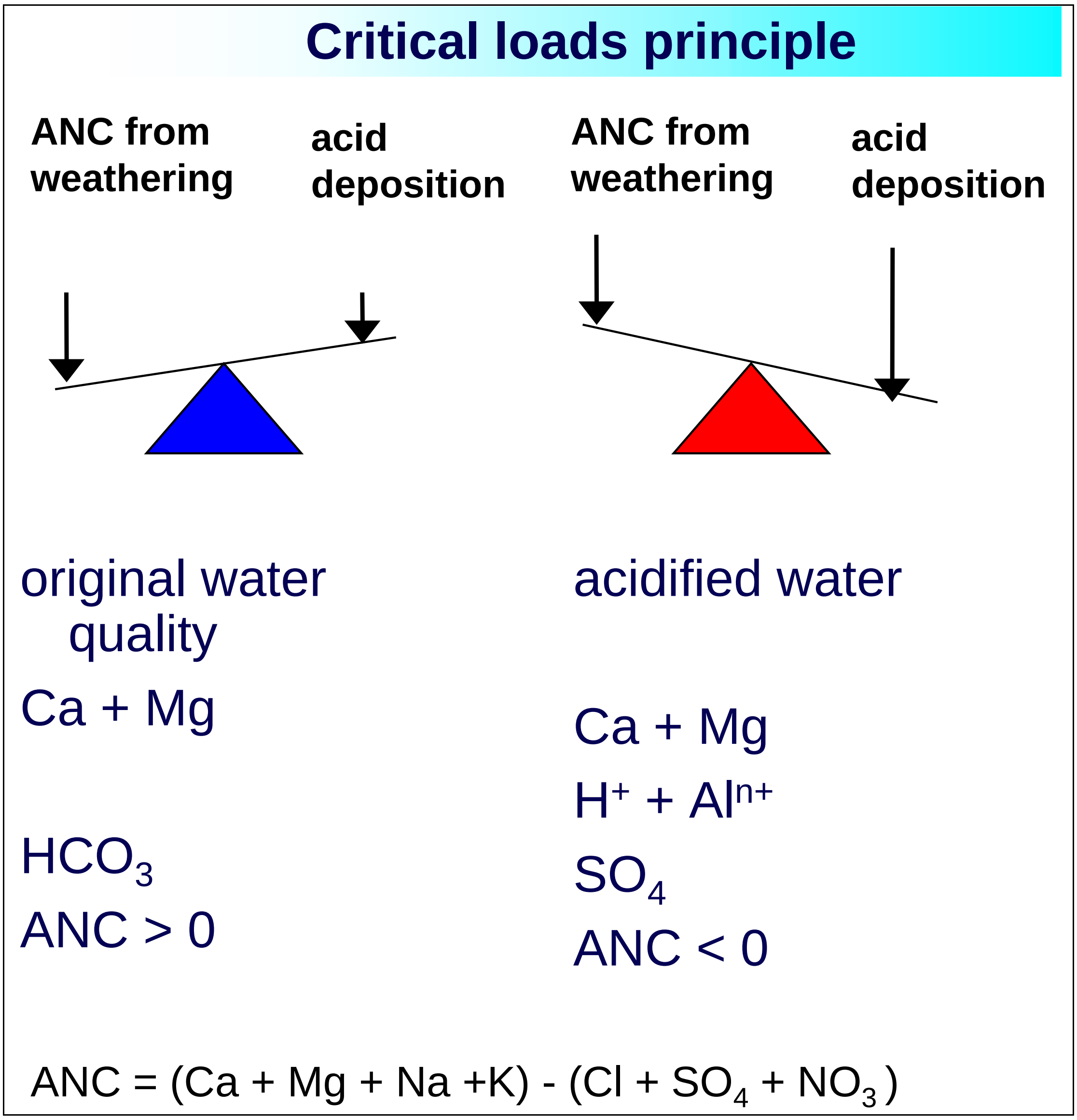
> "A quantitiative estimate of an exposure to one or more pollutants below which significant harmful effects on specified senstive elements of the enviroment do not occur according to present knowledge"

(Nilsson and Grennfelt 1988)

The critical load is the maximum amount of a pollutant that can be deposited on an ecosystem without adverse effects. The concept thus relates a chemical pollutant to a biological impact. In the case of acid deposition the critical load is the maximum deposition of acidity that can be deposited without adverse effect on the ecosystem. Application of critical loads involves identification of key organism (or organisms) to be protected, a "critical limit" for the concentration of, for example, ANC, and a model to relate deposition rate to the concentration of ANC.



The figure illustrates the conceptual links between acid deposition and harmful biological effects. A given dose of acid depostion causes a response in water or soil chemistry. Likewise a given chemical dose cuases a biological response to fish or forests.

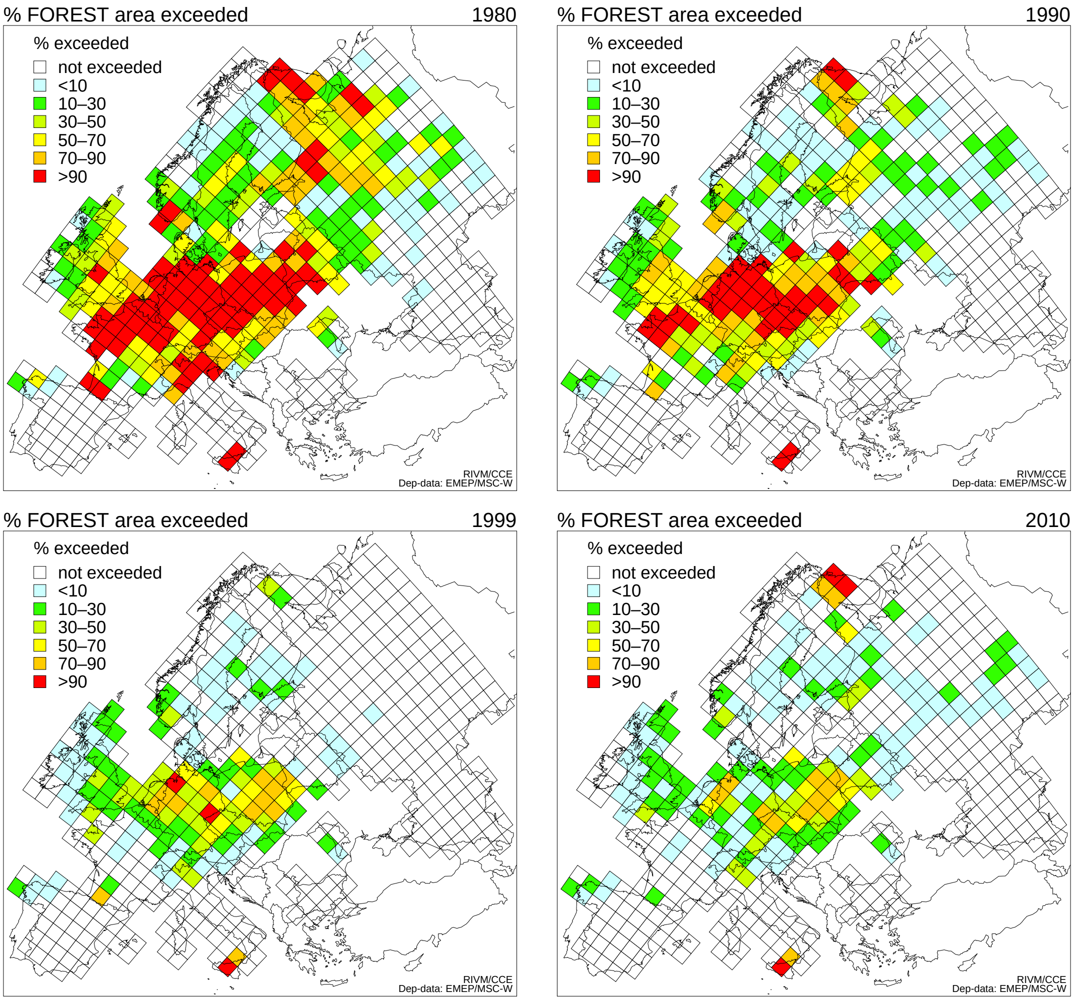


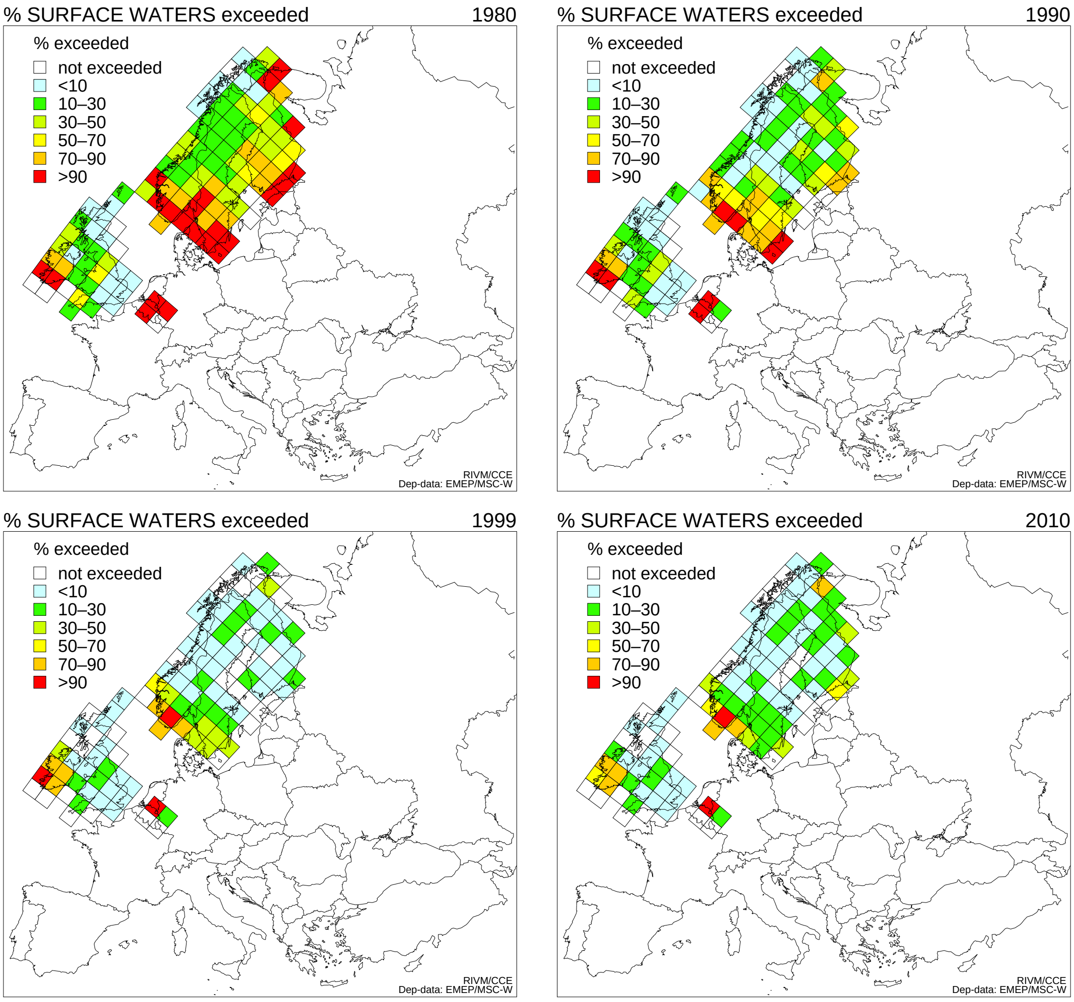
The figure illustrates the basis of the critical load principle. If acid deposition exceeds the weathering rate, the critical load is exceeded and sooner or later water and/or soil acidification will progress to the point at which harmful effects on biological organisms occur. Other factors (in many cases of lesser importance) such as deposition of base cations, uptake by vegetation and retention of nitrogen are also considered in calculation of critical loads.

The critical load concept has been used in the work behind the UN-ECE LTRAP protocols of 1994 (2nd sulphur protocol, Oslo protocol) and the most recent 1999 Gothenburg protocol (multi-pollutant, multi-effect). The work is organised through the Working Group on Effects (WGE), which in turn has established a number of International Co-operative Programmes (ICPs).

Calculation and mapping of critical loads for freshwater, forests, and other ecosystems in Europe has been organised by the ICP Mapping and Modelling (M&M). These critical loads and their exceedences are calculated using various static models, such as the steady-sate water chemistry model (SSWC) and the first-order acid balance model (FAB) (Posch et al. 1997).

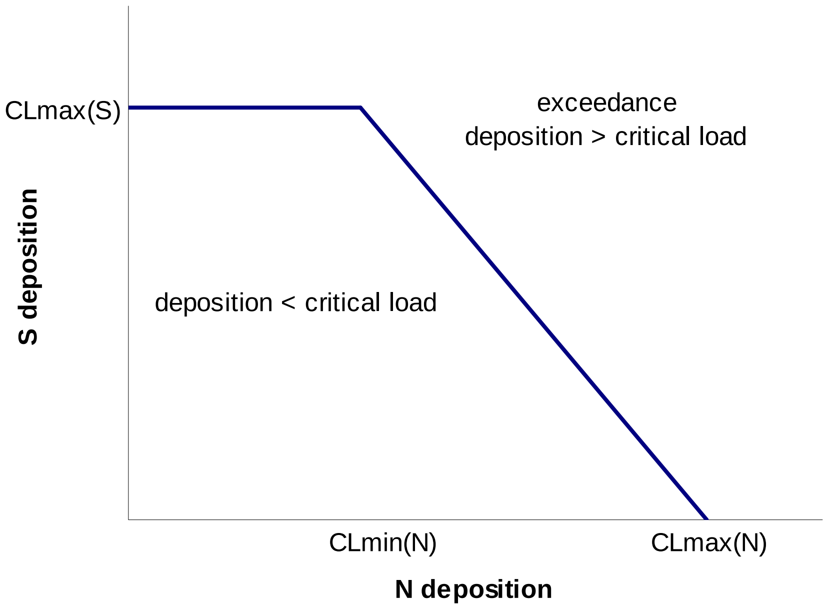
The maps show the calculated exceedences of the critical load of acidity in Europe for forests and surface waters for various years (from Posch et al. 2003a). The deposition of acidity has decreased since the 1970s, and so the area exceeded has also decreased.





Ref: Posch et al. 2003a

Critical load functions



The critical load can be expressed as a load of total acidity or as a function of the deposition of S and N.

Chapter 5
International agreements

Acidifying deposition over Europe has been substantially reduced since its peak in the 1970s primarily as a result of international agreements to reduce the emissions of air pollutants. The shift towards cleaner energy production occurred in response to international agreements and national legislation triggered by concern about the environmental effects of soil and water acidification. The concept of critical loads has been the central tool for assessing the link between effects and deposition, widely used to translate the environmental protection criteria into regional emission reduction requirements.

Protocols to the Convention of Long-range Transboundary Air Pollution (CLRTAP) Status 2002-10-04. Information on the compliance for the protocols can be obtained via 2000 Review of Strategies and Policies for Air Pollution Abatement. Status as of October 2003.(http://www.unece.org/env/lrtap/cov/lrtap_s.htm). Table courtesy of G. Lövblad, IVL.

Protocols to the convention signed	In force	Agreed reduction	Between years	
The first sulphur protocol Helsinki 1985	1987	30%	1980	1993
The nitrogen oxides protocol Sofia 1988	1991	no increase of emission, some countries voluntarily agreed to cut emissions by 30%	1987	1994
The VOC – protocol Geneva, 1991	1997	30%	1984-1990	1999
The second sulphur protocol Oslo, 1994	1998	as a total over Europe; 62% of the emission in 1980*	1980	2000
The protocol on heavy metals, Aarhus 1998	Not yet	Reduce, control and eliminate emissions and use of cadmium, lead and mercury		
The protocol on persistent organic pollutants, Aarhus 1998	Not yet			
The multi-pollutant protocol Gothenburg, 1999	Not yet	as a total over Europe; 75% SO_2, 50% NO_x, 58% VOC, 12% NH_3 in relation to emissions in 1990	1990	2010

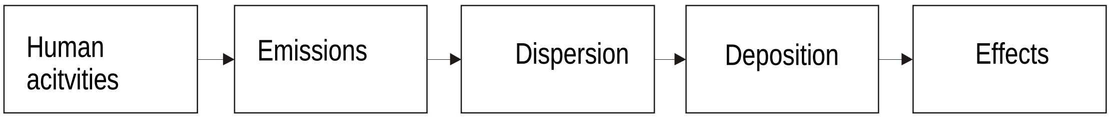
* Effect-based protocol, national ceilings for emissions depending on the exceedance of critical loads in influenced areas

Acidification in Europe is a transboundary problem that cannot be dealt with only on the national scale. Human activities such as the production of energy and its use for transport and industrial processes take place in regions that are sometime even far distant from those in which the chemical and biological responses to deposition of sulphur and nitrogen occur.

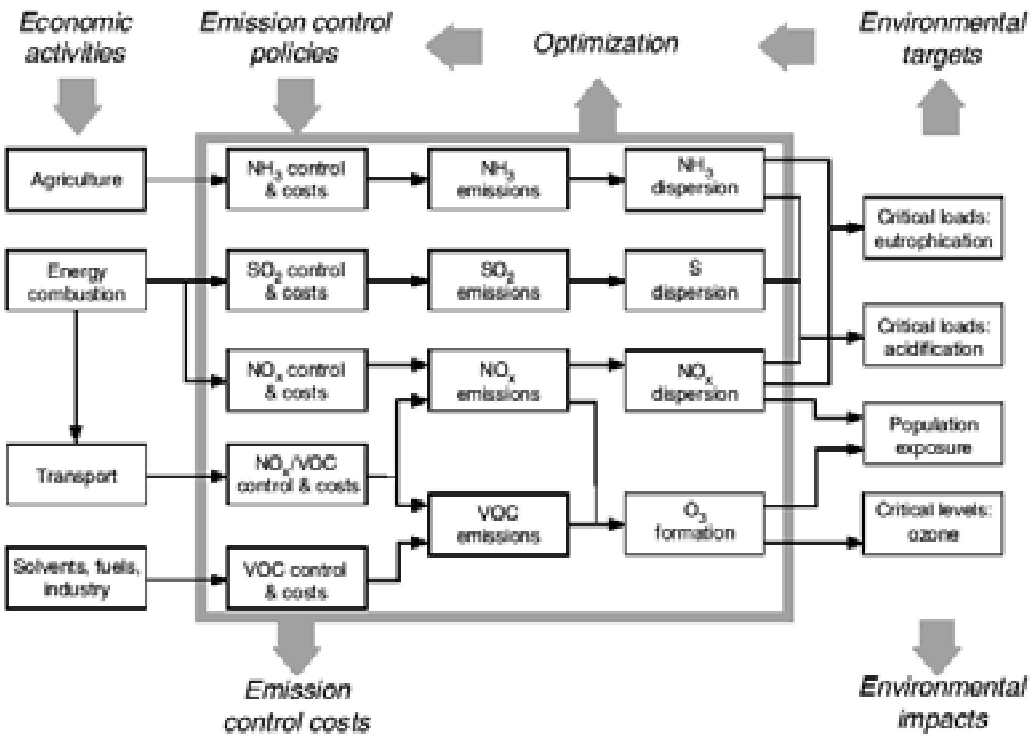
The Convention on Long-range Transboundary Air Pollution (www.unece..org/env/lrtap) was signed in 1979. The introduction of the critical loads concept made it feasible to account for environmental effects in determining emission reduction targets. The second sulphur protocol, signed in 1994, aimed at cost-efficient sulphur reductions with the environmental targets determined in terms of critical loads. The multipollutant protocol, signed in 1999, determines abatement strategies for sulphur and nitrogen emissions derived by employing critical loads to set emission targets.

RAINS model

When the emission reductions under the Gothenburg protocol were set for the different countries, output from a model called RAINS (the Regional Air Pollution Information and Simulation model, www.iiasa.ac.at/rains) was used as an objective scientific base.

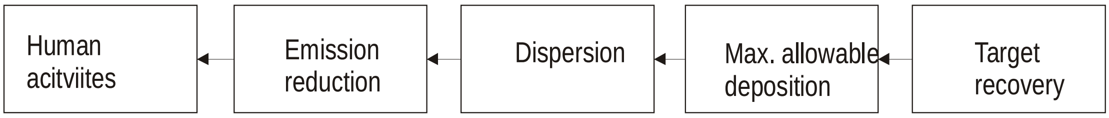


The RAINS is an integrated assessment model. It uses data on human activities (eg. energy consumption in different sectors and countries) and type of fuel and combustion to calculate emissions of pollutant. It then uses so-called transfer matrices to calculate what happens to the pollutants in the atmosphere and where they end up being deposited. The last step is that it uses effect criteria (such as critical loads or in the future target load functions for acidification) to calculate the effects of the pollutants on the ecosystems.



The RAINS model, including the acidification parts as well as eutrophication and effects of particles and ozone.

The question could be reversed and the RAINS model can also be used backwards: If we can accept a certain situation regarding effects on European ecosystems, how much will we have to decrease emissions, where and how much will that cost?

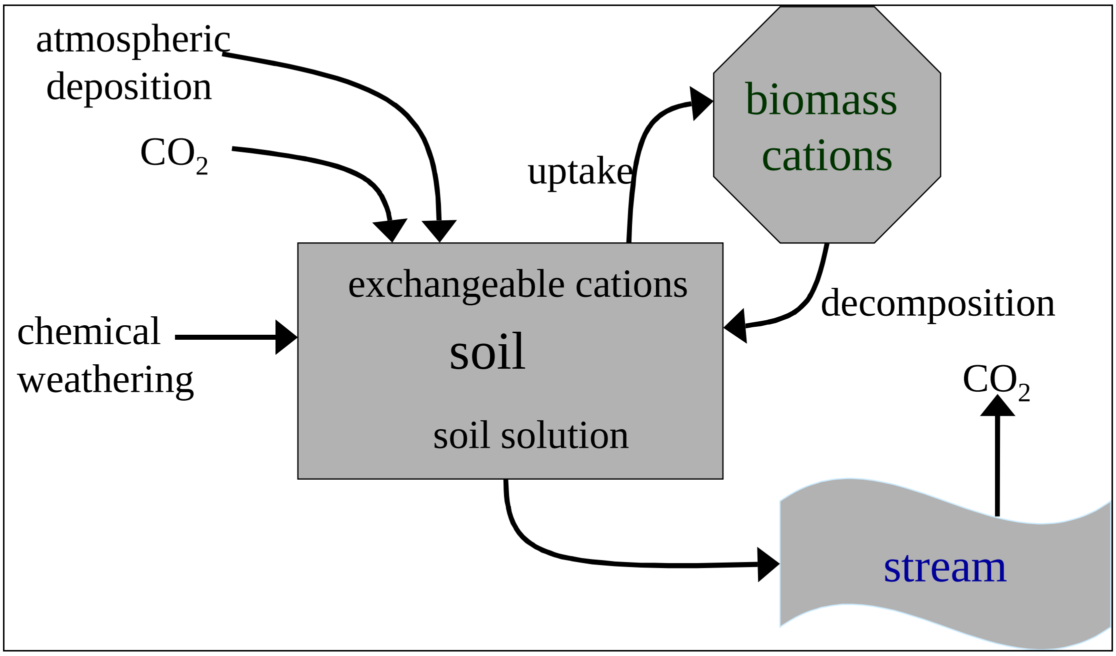


Using an integrated assessment model in this way allows to design the cost-effective strategies to reduce the emissions so that maximum improvement of ecosystems is achieved at the minimum abatement strategy costs.

Chapter 6
Dynamic models

Soil plays a central role in the response of forests and lakes to air pollution. The extensive reservoirs of elements in the mineral and organic components, combined with the large reactive surfaces give rise to long response times of the soil system. Because of the long response time, experiments and monitoring in the field or laboratory give only limited results within a few decades. We need dynamic, process-based models to integrate and interpret theoretical knowledge from soil science and hydrochemistry with results from experiments and monitoring.

Dynamic models that are used for studying the effects of air pollutants on soil and water quality give us information on variables that represent the central reservoirs or pools of elements. The sizes of these pools are altered by fluxes of elements caused by processes operating at various rates. In many dynamic model applications the catchment is represented by one single soil compartment (see chapter 9).



Variables

Concentrations of major anions and cations in soil solution and streamwater (SO_4^{2-}, Cl^-, NO_3^-, Org^-, H^+, Al^{3+}, Ca^{2+}, Mg^{2+}, K^+, Na^+)
Soil base saturation (proportion of base cations of all the exchange sites)

Driving functions

Deposition (S, N, Ca, Mg, K)
Vegetation uptake (Ca, Mg, K, N)
Water flux through the soil

Central processes

Ion exchange
Mineral weathering
Adsorption/desorption
Mineralisation
Dissolution
Complexation

Models

The most well known dynamic models used in acidification studies are MAGIC (Cosby et al. 1985a; Cosby et al. 1985b; Cosby et al. 2001), SAFE (Warfvinge et al. 1993) and SMART (de Vries et al. 1989).

MAGIC has been widely used to predict acidification and recovery, land-use change and climate change. Regional results of MAGIC for water quality in Europe hae recently been summarised by Jenkins et al. 2003a).

The static PROFILE model (Sverdrup and Warfvinge 1992) simulates silicate weathering in different soil horizons and the SAFE model (Warfvinge et al. 1993) uses its results to simulate the dynamics of soil acidification in several layers of soil.

The SMART model (de Vries et al. 1989) has been used to study the regional response of lakes in Finland (Posch et al. 2003b) (see chapter 8).

SAFE, SMART and MAGIC have all been used in conjunction with calculation of criticial loads (chapter 4). Recently a simplified model for soil acidification and calculation of target load functions for soils has been put forward as an alternative; this is the Very Simple Dynamic (VSD) model (Posch et al. 2003c).

Chapter 7
Site specific modelling

Modelling and measurements

We build mathematical models in order to predict what the effects of acid deposition on the ecosystems will be in the future. Models are simplifications of reality and include many interacting processes. A key step in the modelling process is evaluation of model performance, that is, how well does the model describe reality. Methods include comparison of model results with measurements or observations from the real world. Measurements and experiments are essentially important for testing the validity of the models. We here describe four different ways to evaluate the models.

Large scale experiments

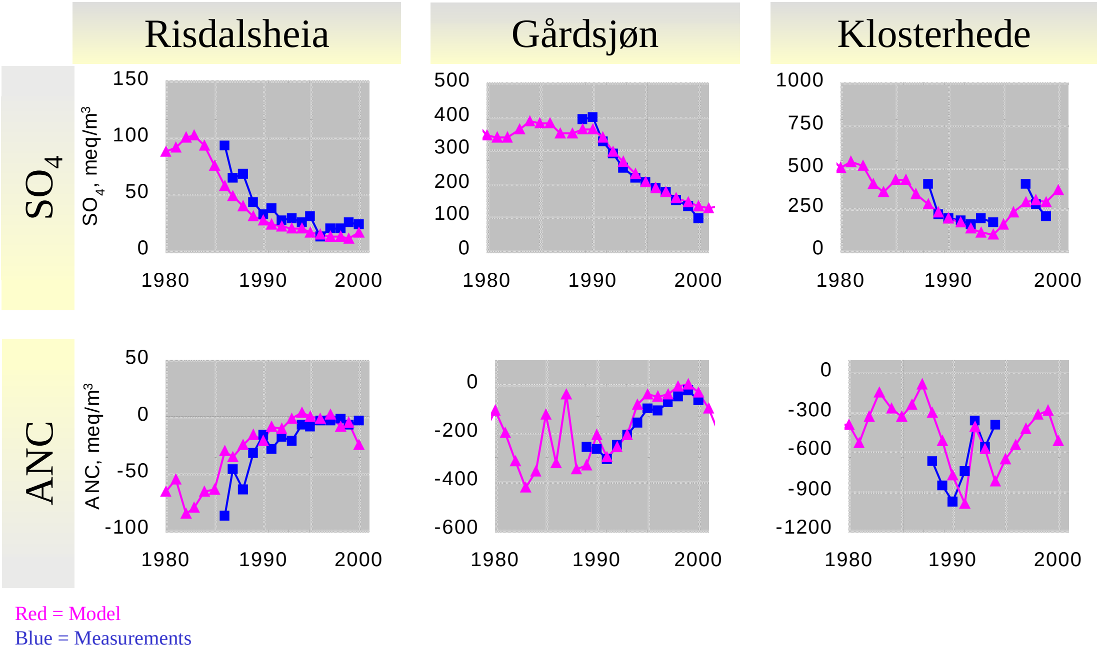
In the 1980s and 1990s three large experiments were started in Norway, Denmark and Sweden. These projects were all built on the same basic concept: an ecosystem was covered by a ”roof” so the acid rain could be removed and replaced with clean and de-acidified rain. The effect on the chemical and biological conditions in ecosystem could then be measured and be used to evaluate to what extent removal of air pollution would help the ecosystems to recover and how fast the recovery would be.

Risdalsheia in Norway – a 1200-m^2 roof removing the acidic input to a complete catchment. Effects on plants, soil and water were studied (Wright et al. 1993).

Klosterhede in Denmark – a 1200-m^2 roof built underneath the forest canopy to remove the acidic input to the soil and replace it with different combinations of clean rain and base cations (Beier et al. 1998).

Gårdsjön in Sweden – a 7000-m^2 roof built underneath the forest canopy of a complete catchment to remove the acidic input to the soil and replace it with clean rain (Hultberg and Skeffington 1998).

Such experiments are powerful tools to test the dynamic models. We can apply the models to the normal situation at the site by matching the model and the chemical conditions at the site. When the model describes the normal situation well, we can let the model predict, what the effect will be of doing the experiment we did with the roof. We can then compare what the model predicts with the measurements done in the experiment and see if the model gives a good/correct description of the results. If the model describes well the effects of the experiment, this is a very strong test of the model's ability to describe effects of changes in the acid input – it will give us confidence that the model predictions for other sites and situations where we do not have experimental data are actually valid. Such a test has been done for the roof covered experimental sites Risdalsheia in Norway, Klosterhede in Denmark and Gårdsjön in Sweden (Beier et al. 1995, Beier et al. 2003).

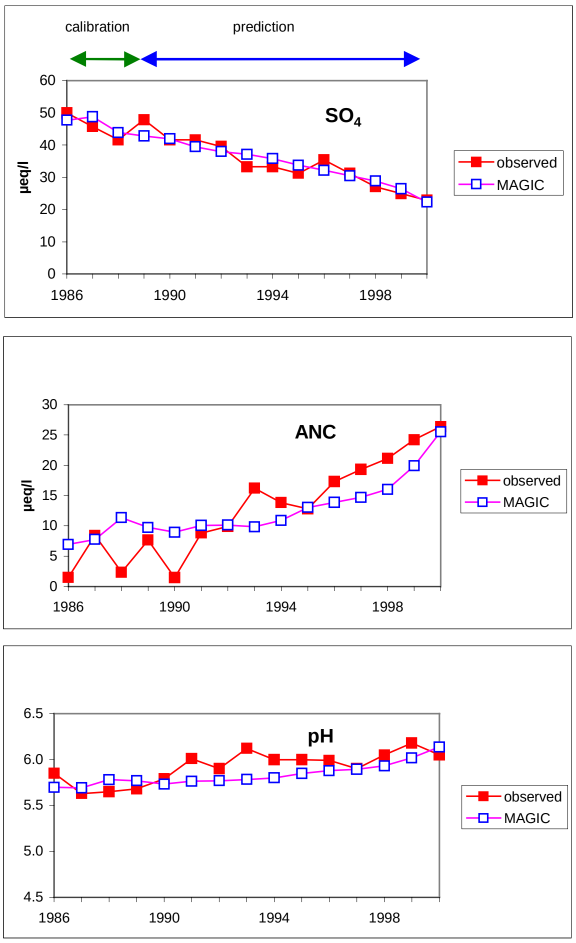


The blue graphs in the figures show the measured changes in SO4 and ANC in the runoff water from the "clean rain" experiments in Risdalsheia, Gårdsjön and Klosterhede. The red graphs show what the MAGIC-model predicts should happen (from Beier et al. 2003).

Long data sets

Long-term measurements of water chemistry under changing acid deposition can be used to evaluate water chemistry simulated by dynamic models. For example, at Stavvatn, a small lake in southern Norway, data collected annually since 1986 show that the lake is recovering from acidification, as acid deposition has decreased. In this example the MAGIC model was calibrated to the first 3 years of the data record, and then the simulated changes for the next 12 years were compared to the measured.

Stavsvatn

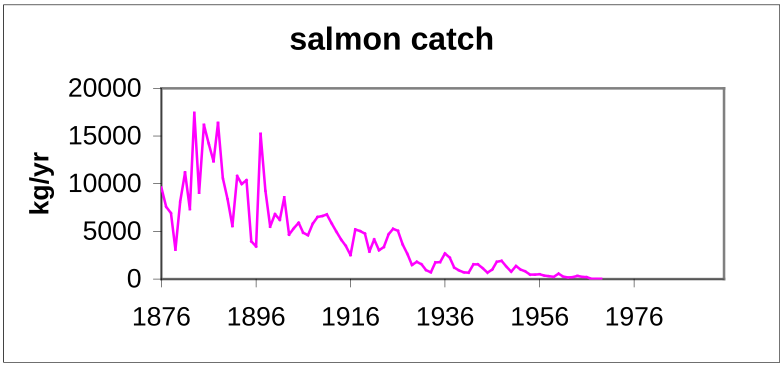


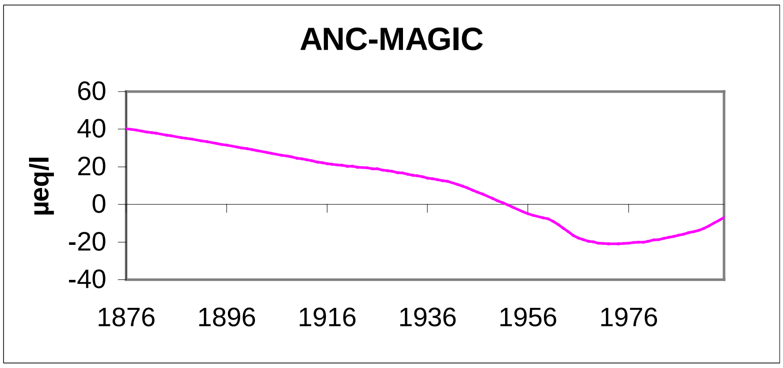
Source: Wathne and Rosseland 1999

Historical biology records

Long-term biological records can also be used to evaluate the output from dynamic models. Here the official salmon catch statistics from the Tovdal River in southern Norway show a dramatic decline during the 1900s with complete extinction of the population in the 1960s. pH measurements show that the river then had pH below 5. The long-term trends in salmon catch agree well with the decline in ANC as simulated by the MAGIC model.

Tovdal





Source: Kroglund et al. 2002.

Paleolimnology

Remains of plants and animals preserved in lake sediments provide a “history book” of life in a lake over time. These paleolimnological records can be used to infer chemical conditions. There are good empirical relationships between assemblages of the algal group diatoms and water chemistry. In this example the diatom assemblages in the sediments were used to infer historical pH of the lakewater. These diatom-modelled pH values can be compared to those from the dynamic model MAGIC. The example is from Round Loch of Glenhead, in the Galloway area of southwest Scotland, an acid sensitive region heavily impacted by acid deposition.

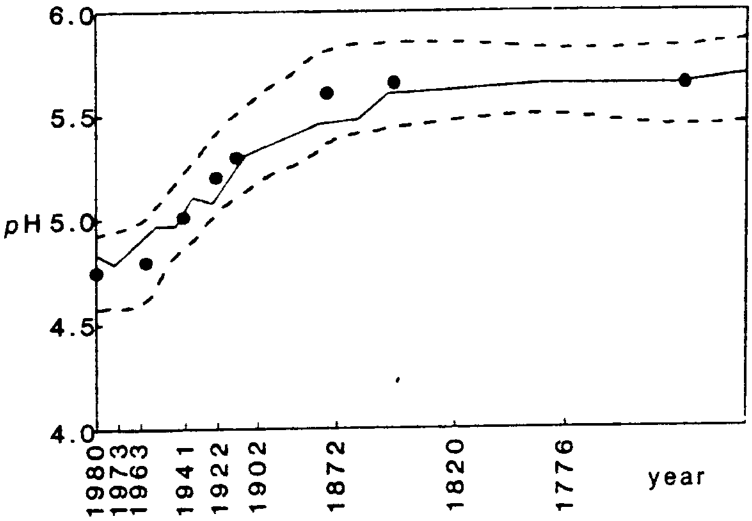


Fig. 1 Round Loch of Glenhead simulation of pH compared with palaeoecological data[9]. Solid line, palaeoecological reconstruction; dashed line, MAGIC reconstruction with 95% confidence bounds; dot, MAGIC reconstruction.

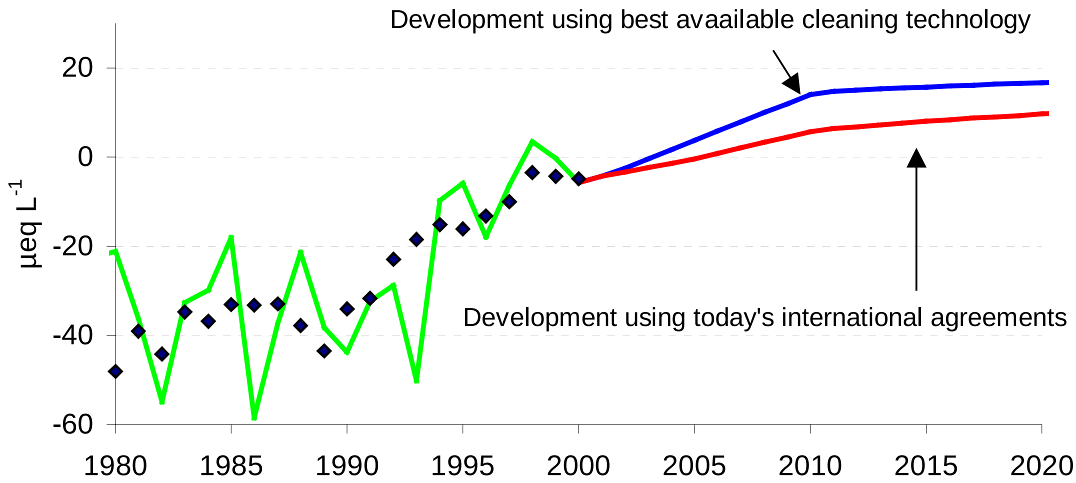
Source: Neal et al. 1988.

Predictions

One of the great advantages of dynamic models is that once calibrated and tested, they can be used as a tool to look into the future. We can use the models to answer questions such as: "What will the effect on recovery of natural ecosystems be if we reduce emissions by 10%, 50% or 100%?" or "how much must emissions be reduced to allow lakes in Norway to recover by 2030?". We do this by running the models into future by applying different scenarios describing the deposition situation in the future and examine the predicted effects. An example of this use of the models are shown below for the Birkenes catchment in southern Norway.

The Storgama catchment, southern Norway.

The figure shows an example on how dynamic models can be used to predict the recovery from acidification in the future. The example shows both modelled (lines) and observed (dots) acid neutralisation capacity (ANC) in a stream in southern Norway (Storgama, Telemark county). With today's international agreements for reduction of acidifying substances, we can expect some further improvement of the water quality. However, even further improvement can be expected if further measures, as installing best available cleaning technology all emission sources, are implemented. From an empirical relationship between ANC and the occurrence of trout in lakes in Norway, we can estimate that with ANC < 0 the lake will be without fish, while when ANC > 20, a healthy population can be expected. The model predictions suggest that the existing international agreements are not sufficient for a healthy trout population in a large part of the lakes in Southern Norway.



Measured and modelled ANC in Storgama stream, southern Norway. In order to show the differences between future scenarios, year-to-year variability in sea salt input and hydrology was smoothed out in the forecast runs which explains the smooth development of ANC after year 2000.

Chapter 8
Regional applications

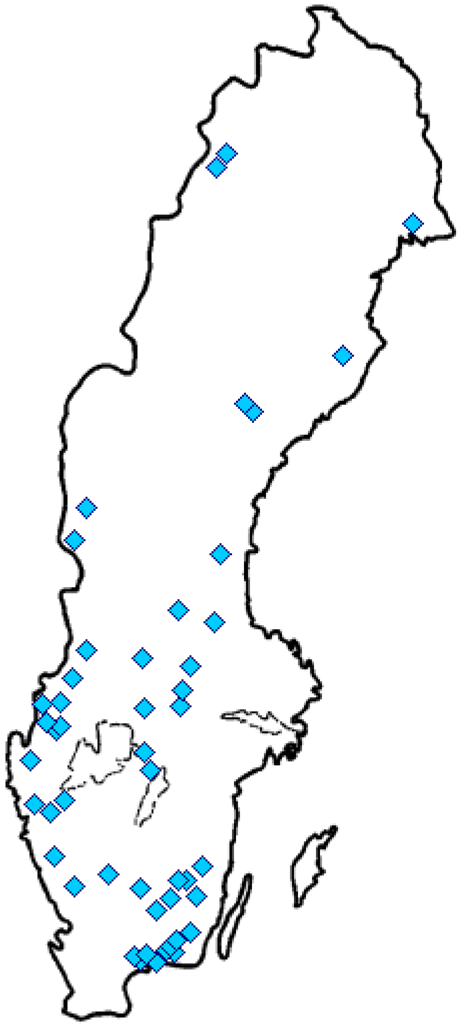
There are two basic approaches by which dynamic models are applied regionally. If the data to run the model are interpolated and expressed for grids that cover the region of interest, the model is applied to these grids and results interpreted as relevant for the grids. That means that the model is applied to data that were interpolated from point measurements or in some other way developed to provide values of necessary parameters (the amount of deposition, surface water quality, type of land use, soil type, mean temperature, precipitation etc) for each grid. Here the disadvantage is that it is more difficult to evaluate model performance, since the model results cannot be compared to observations at any particular site. Because of the non-linearity of most of the modelled processes, the results of modelling with interpolated data need to be interpreted with caution.

In many countries data are available from extensive monitoring of deposition, soils and waters. When the site-specific data are available, the regional model applications can be done by applying a model on multiple sites – typically tens to hundreds of sites – and the results are then interpreted as characteristic for the region. This approach is preferred when data from multiple sites are available. The model is calibrated to each of the site and its performance evaluated by comparing the modelled and observed values. That gives confidence when the model results are extrapolated in time – into the future – or in space – to the areas similar or close to the modelled area.

Modelling Swedish lakes

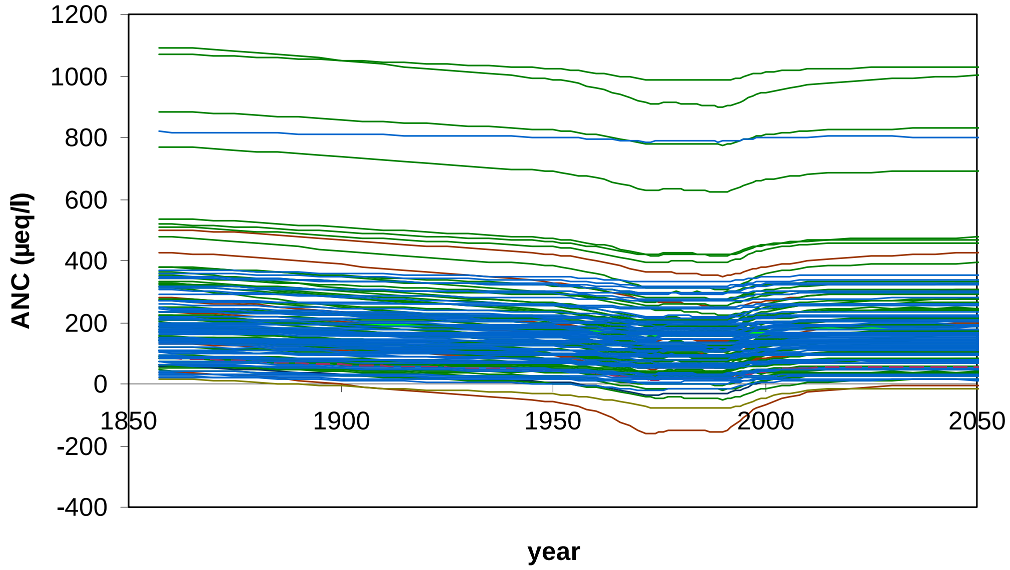
For Sweden the three key data sets needed for regional dynamic modelling are all available on internet; results from soil survey (www.sml.slu.se/sk), lake survey (info1.ma.slu.se/db.html) and deposition (www.smhi.se).

Here is an example from 132 lakes in Sweden modelled using the MAGIC model. It is a site-by-site regional model application based on the data above.



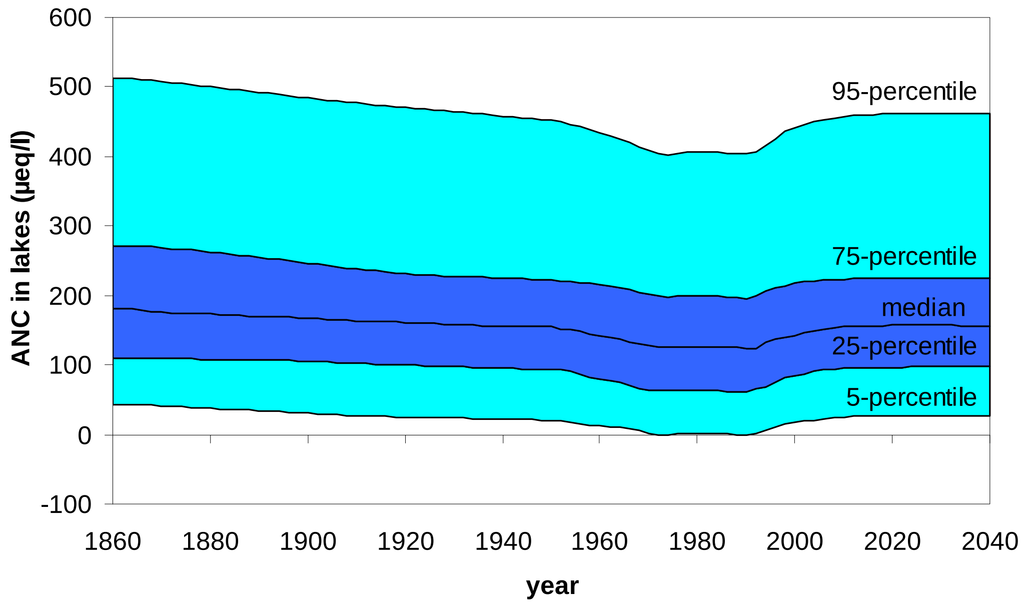
Map of Sweden showing the location of the lakes modelled with MAGIC.

The results of multiple-site model application can be displayed as time series of any chosen parameter modelled by the model, e.g. ANC as shown in the lake water on the figure below.



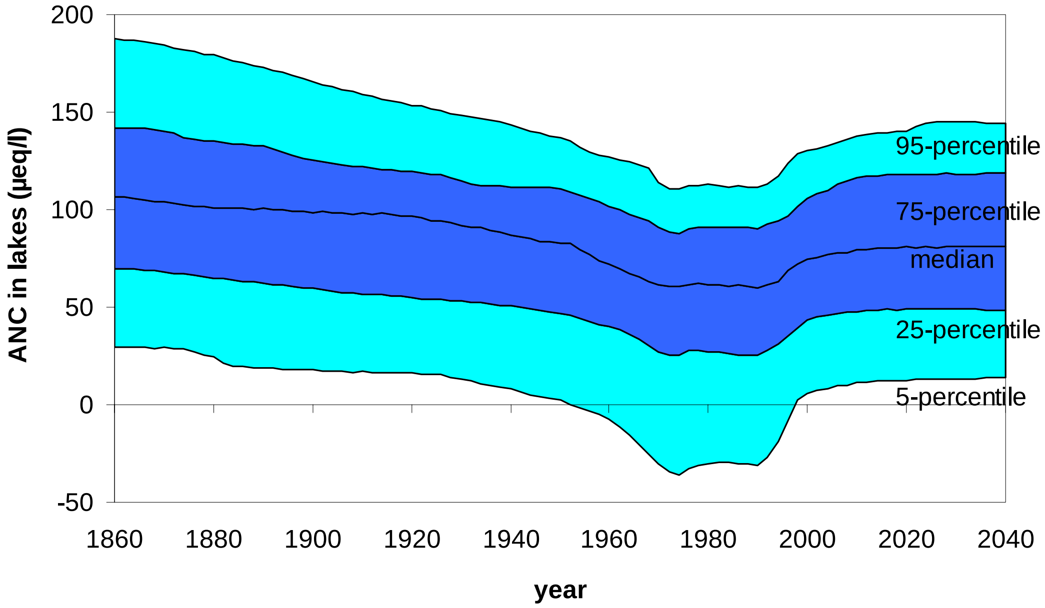
Modelled time series of ANC in 132 lakes in Sweden.

Each curve in the figure above represents one modelled lake. The figure contains a large amount of information, but it is difficult to extract that information by visual inspection. The same data could be summarised by presenting the whole population of the modelled sites as a frequency distribution, such as shown in this example for ANC.

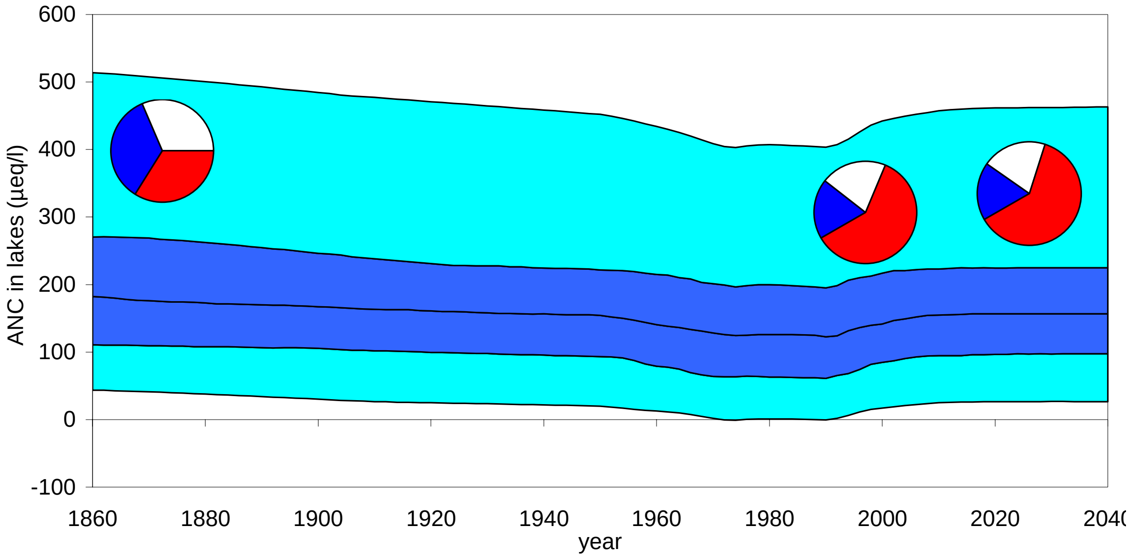


In this "blue chart" the line in the middle represents median of all lakes, the dark blue band encompasses 50% of all lakes, and the light blue part encompasses 90% of all lakes.

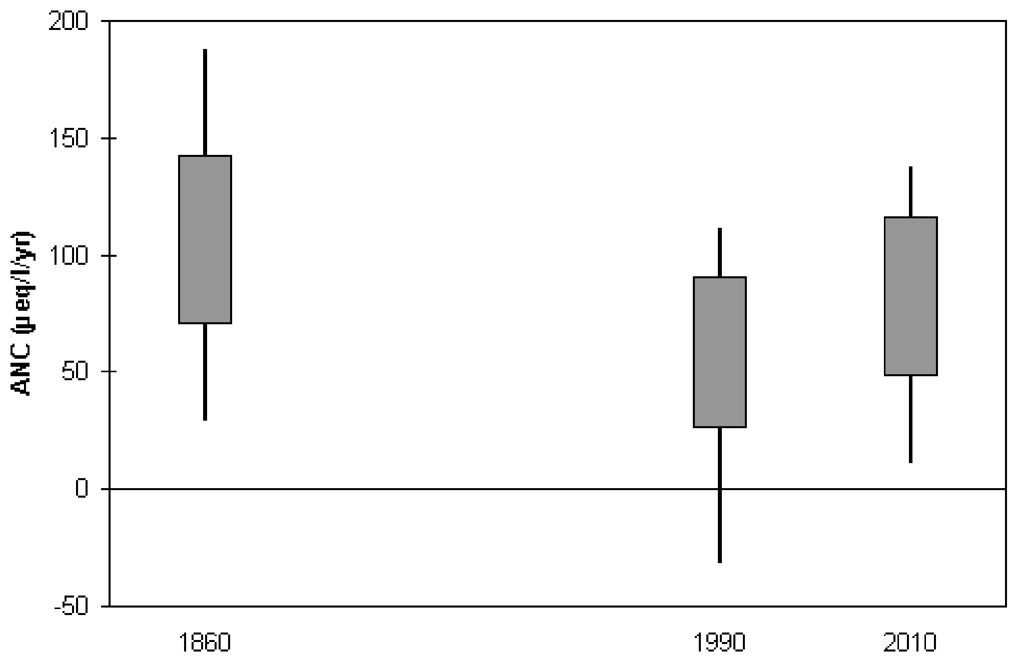
Another useful step in summarising the results of multiple model runs is to stratify the data such that only the relevant part of the modelled sites is presented. In the ANC example above, about half of the lakes are not acidified or acid sensitive according to the criteria used by Swedish EPA. At these lakes ANC never declined to the critical level of ANC considered harmful to fish (20 or 50 µeq/l) or to other biota. These lakes probably never had and never will have a problem with acidification and therefore could be excluded from consideration of how large deposition reduction is needed. Excluding these lakes from the set presented in the diagram gives a picture of acid sensitive lakes only. Such summary picture captures the relevant part of the modelled lakes and illustrates the dynamics of the development of ANC under this one scenario in rather nice and understandable way. The figure below shows the development of ANC over time at the most sensitive lakes; these had low ANC already in the mid-1900s. Under the Gothenburg protocol (see chapter 5) they will recover in the future, but not to historical ANC levels.



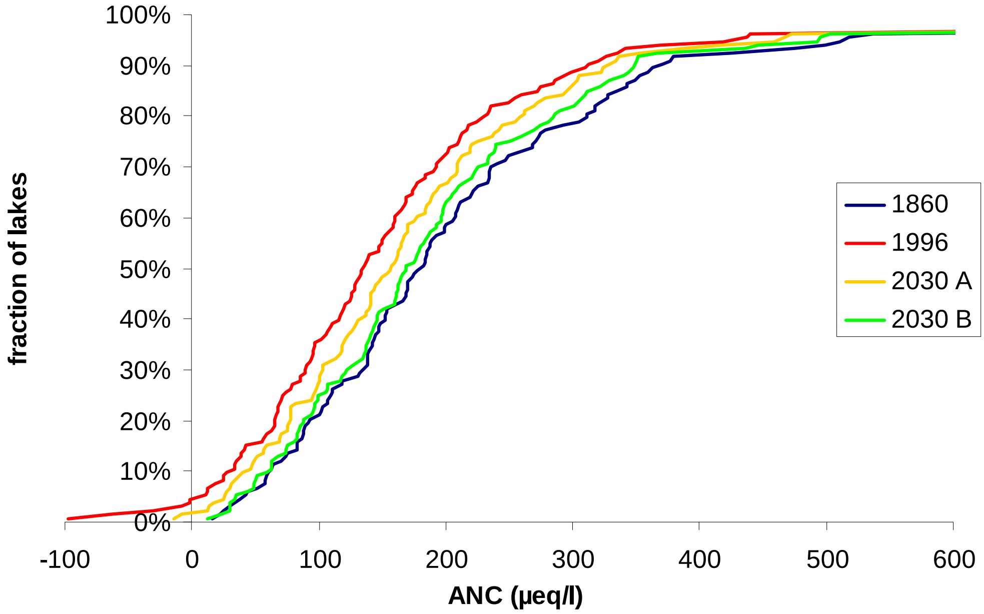
Dynamic acidification models such as MAGIC produce a number of soil and surface water parameters such as concentrations of all major anions and cations including different species of aluminium in the soil, stream and lake water. Furthermore the model calculates changes in stores of base cations, sulphur and nitrogen in the soil. The way of summarising the multiple run results shown above could be complemented by adding information on an additional parameter such as e.g. soil base saturation. Putting information on both waters and soils to the same chart has the advantage that these can be easily compared. The example below shows clearly that the Gothenburg protocol scenario will allow widespread increase in ANC in lake water, but very little replenishment of the soil base saturation.



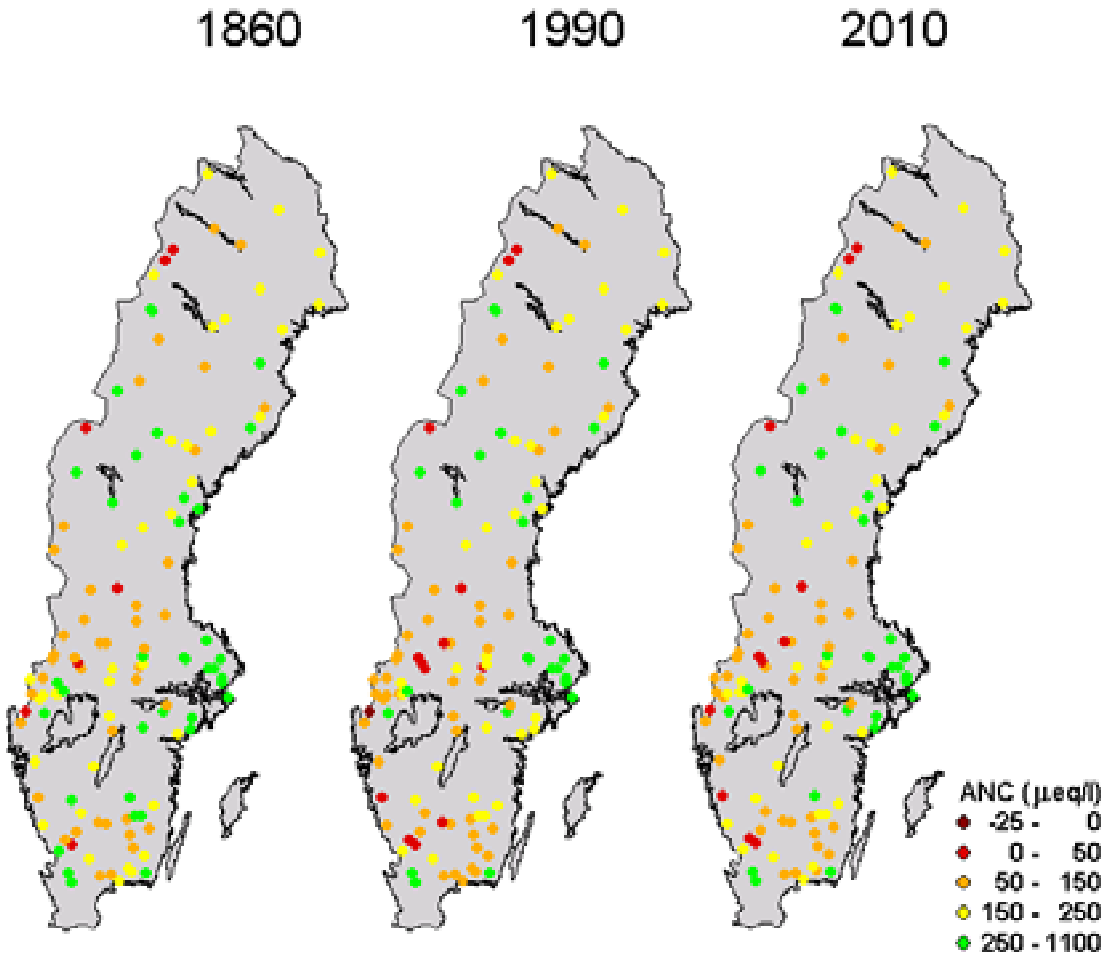
A simpler way of presenting essentially same information as in the continuous blue charts above is a use of box and whisker diagrams which show distribution of any given parameter for a selection of years, for example, 1860, 1990 and 2010 as below. Such a figure clearly shows the shift in the given parameter at the whole population of the sensitive lakes.



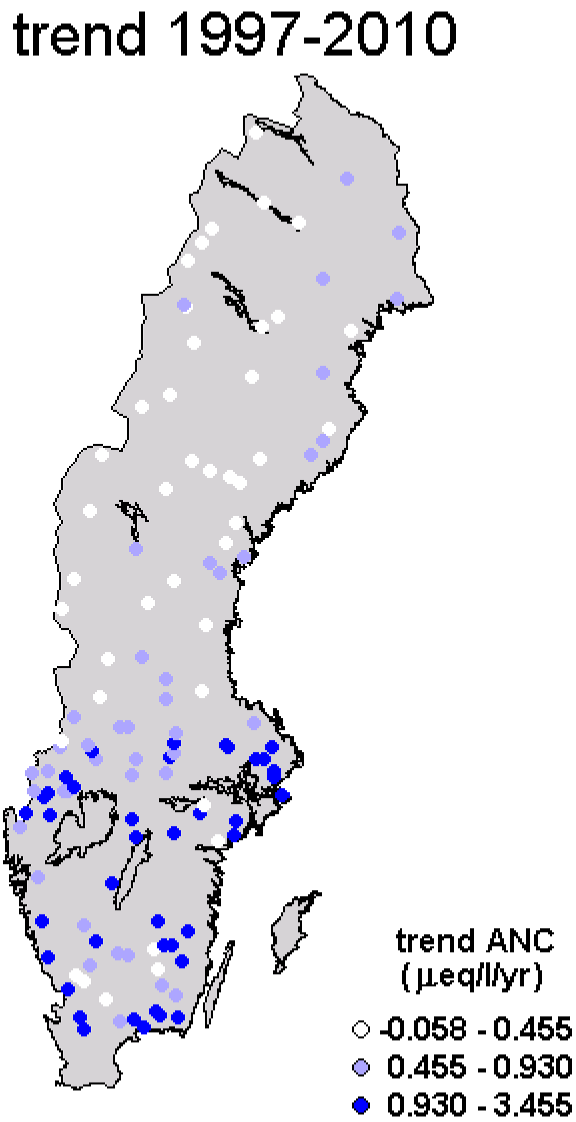
Yet another way of communicating the results for several years at the whole region is to plot cumulative distributions, one for each year. It is easy to compare the different years in which direction and by how much has any given parameter shifted. In the example below the decrease in ANC from 1860 (blue line) to 1996 (red line) could be compared to an increase to the year 2030 under Gothenburg scenario (yellow line) or to the same year 2030 under the maximum achievable reduction scenario (green line). Plots of cumulative distribution, in particular, are efficient ways to illustrate differences in the regional model runs based on different future scenarios.



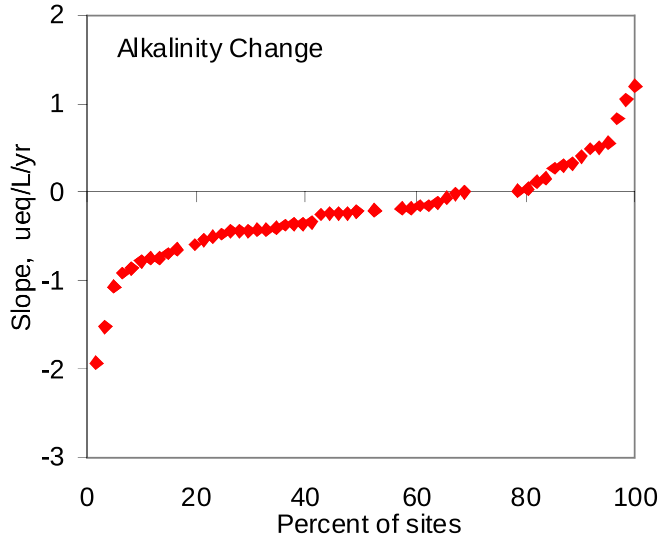
Regional applications of dynamic models can also be summarised in a form of maps. Maps of any given parameter could be shown by assigning different colours to a value intervals (e.g. yellow for ANC 150 – 250 µeq/l, green for ANC above 250 µeq/l as on the map below) and plotting coloured dots at the locations of the modelled sites. Then the maps could be shown as a set of maps one for each year, where the colour of the dots will change from map to map for the modelled lakes, in this example of ANC. Such maps clearly show the geographical distribution of the changes in ANC over time, and also give information on spatial heterogeneity of ANC levels and changes in ANC.



To highlight the trends a similar map could be constructed by plotting the change over time (rather than absolute values at given times) on the map. While such a map does not provide the information on absolute values, it is useful to illustrate the distribution of magnitudes of trends across the region or a country. For instance the information that the lakes with relatively larger increase in ANC are more common in southern Sweden as compared to northern Sweden is not as readily visible from the set of previous three maps as from this map of trends.



Another way to present trends in chemistry over the region is to calculate a trend for each modelled site and then rank these trends from largest negative to largest positive change per year.



Alkalinity change in brook trout streams in the mountains of western Virginia (Jack Cosby pers. comm.)

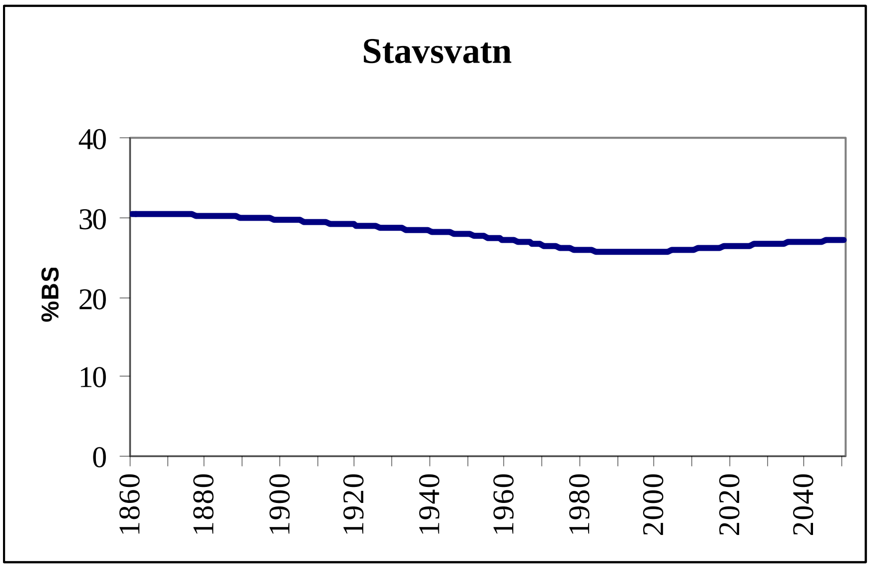
The alkalinity change in the figure above is expressed in yet another way. Each dot represent the slope of the trend in µeq/l/yr for a stream. Presented in this way the diagram reveals that the change in alkalinity ranged from a decrease by approximately 2 µeq/l/yr to an increase of little over 1 µeq/l/yr. Another important point could be made, that although there were both increases and decreases in the ANC, at more than 60 % of modelled sites the trend was a decrease.

An important area of use of dynamic models is to illustrate possible future development under several different scenarios. The presentation techniques are similar to the examples shown above except that instead of showing differences between the years, the charts show differences between scenarios for any given year. This could by done in form of box and whisker plots, cumulative distributions, pie charts or maps, either as sets of maps or one map of differences between scenarios.

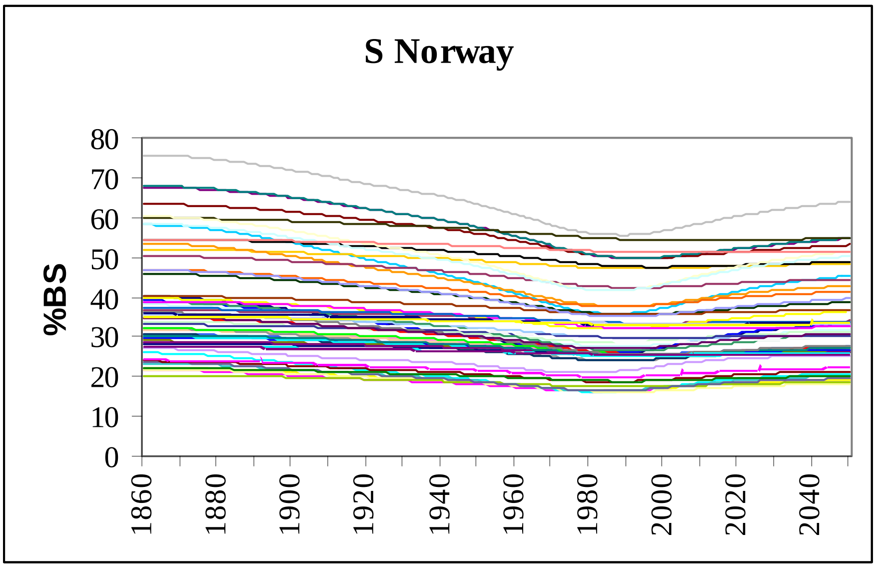
Modelling soil acidification in Norway

Dynamic models such as MAGIC also simulate changes in the pool of exchangeable base cations (% base saturation) in the catchment soil. Again there are many ways in which the results for a region can be displayed.

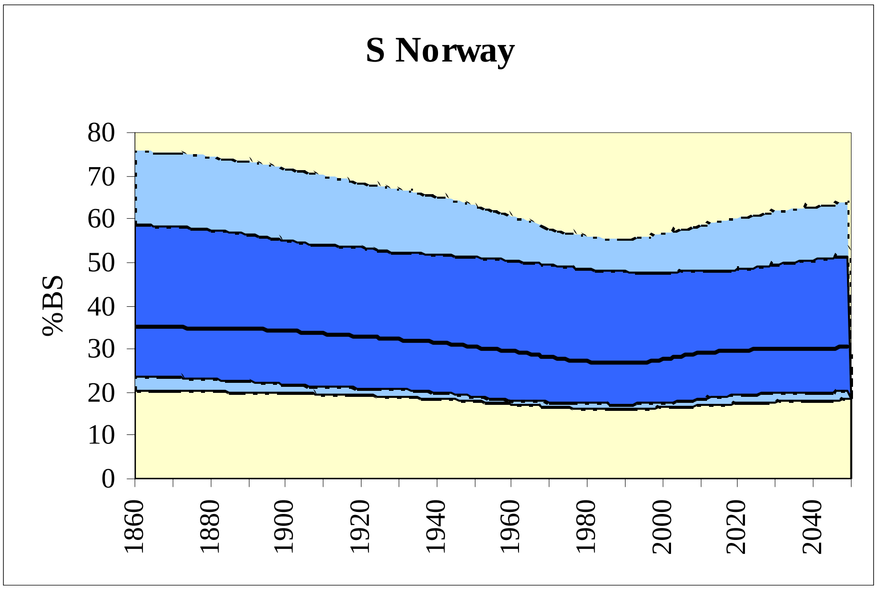
First is the change in %BS for a single catchment, in this case the EMERGE and RECOVER lake Stavsvatn located in Telemark, southern Norway (Wathne and Rosseland 1999). The curve shows a gradual depletion of the %BS during the acidification phase to about 1980, and then a levelling off, and slight recovery (replenishment) in the period 2000-2050.



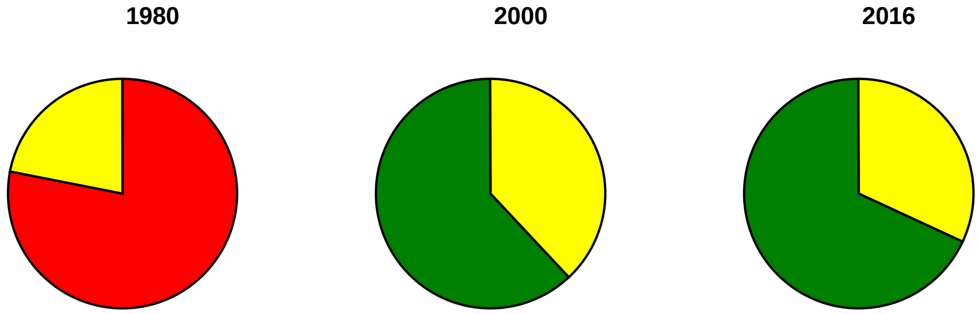
Stavsvatn is but one of 60 lake catchments in the region of S Norway that have been modelled (from Wright and Cosby 2003). The curves for all 60 lake catchments can also be plotted, and show basically the same trends as Stavsvatn, but at different levels of base saturation.



Again these curves can be condensed into the "blue diagram" showing the minimum, 10 %, 50%, 90%, and maximum for the set of 60 sites.

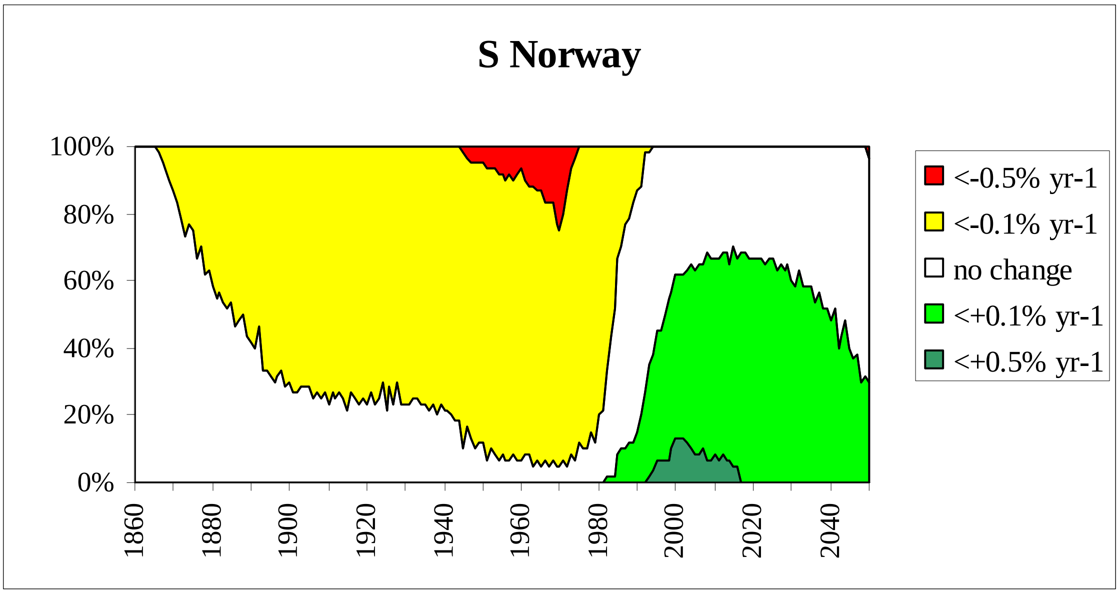


Time cuts through this diagram can also be displayed as pie charts. Here the important feature is not the absolute value of %BS, but the change and rate of change of %BS.



The pies show the % of lake catchments soils falling into one of three categories:
Red = %BS decreasing by more than 1 ‰ of the exchangeable cations store in the soil per year.
Yellow = no change
Green = % BS increasing by more than 1 ‰ per year.

Thus in the year 1980 the base saturation was being depleted at most of the sites (i.e. the soils were acidifying), whereas in the year 2000 and 2016 the %BS are replenished. This can be shown for the entire 200-year period as an area chart:

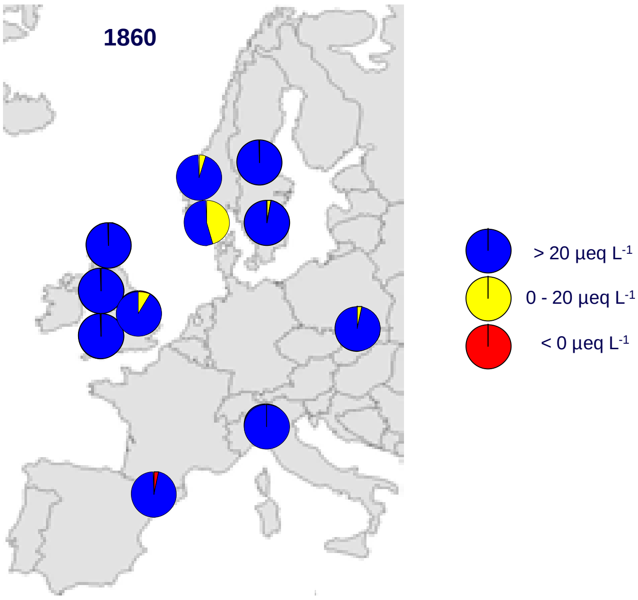


Here is shown the % of sites falling into one of five categories:
Red = %BS decreasing at more than 5 ‰ per year
Yellow = %BS decreasing at more than 1 ‰ per year
White = No change
Light green = % BS increasing at more than 1 ‰ per year
Dark green = %BS increasing at more than 5 ‰ per year

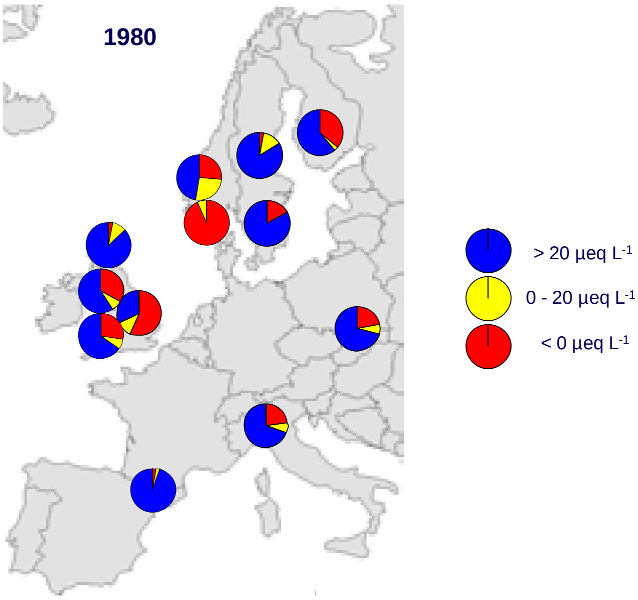
In this example the assymetry is clearly seen. The yellow and red areas are much larger than the green areas. Thus the soils acidify much more to the year 1980 than they recover by the year 2050.

RECOVER:2010 and EMERGE – a regional summary for Europe

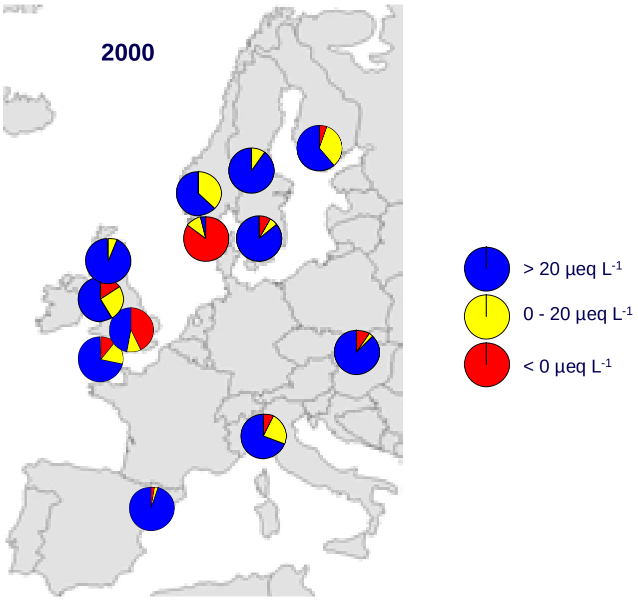
Regional or national modelling efforts are sometimes a part of larger effort done by several research teams in different countries. One such major recent effort was the EU projects RECOVER:2010 and EMERGE, where results of dynamic modelling were summarised for several regions in Europe (Jenkins et al. 2003a). Such a summary provides a European scale overview of the situation in this example of acidification of lakes in all major European lake regions where countries and different years can be compared at a glance; hundreds of sites and model runs lie behind these maps.



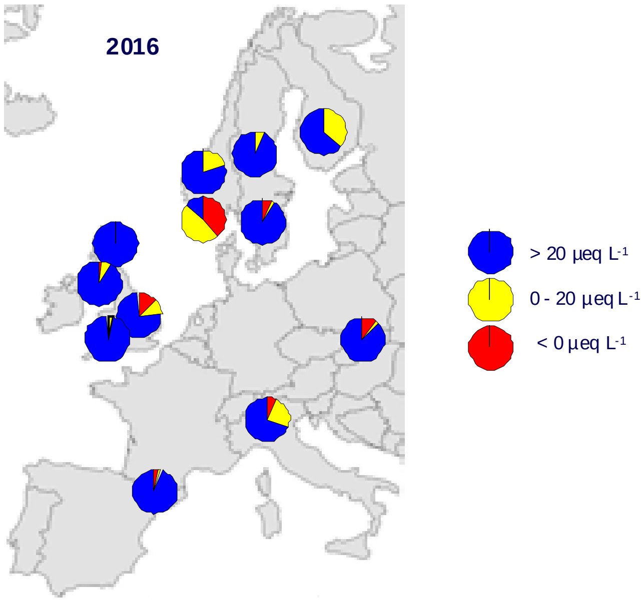
The 1860 model reconstructed surface water ANC concentration for each region expressed in 3 ANC classes. For Finland, the model was initialised in 1960 and no results are available for 1860.



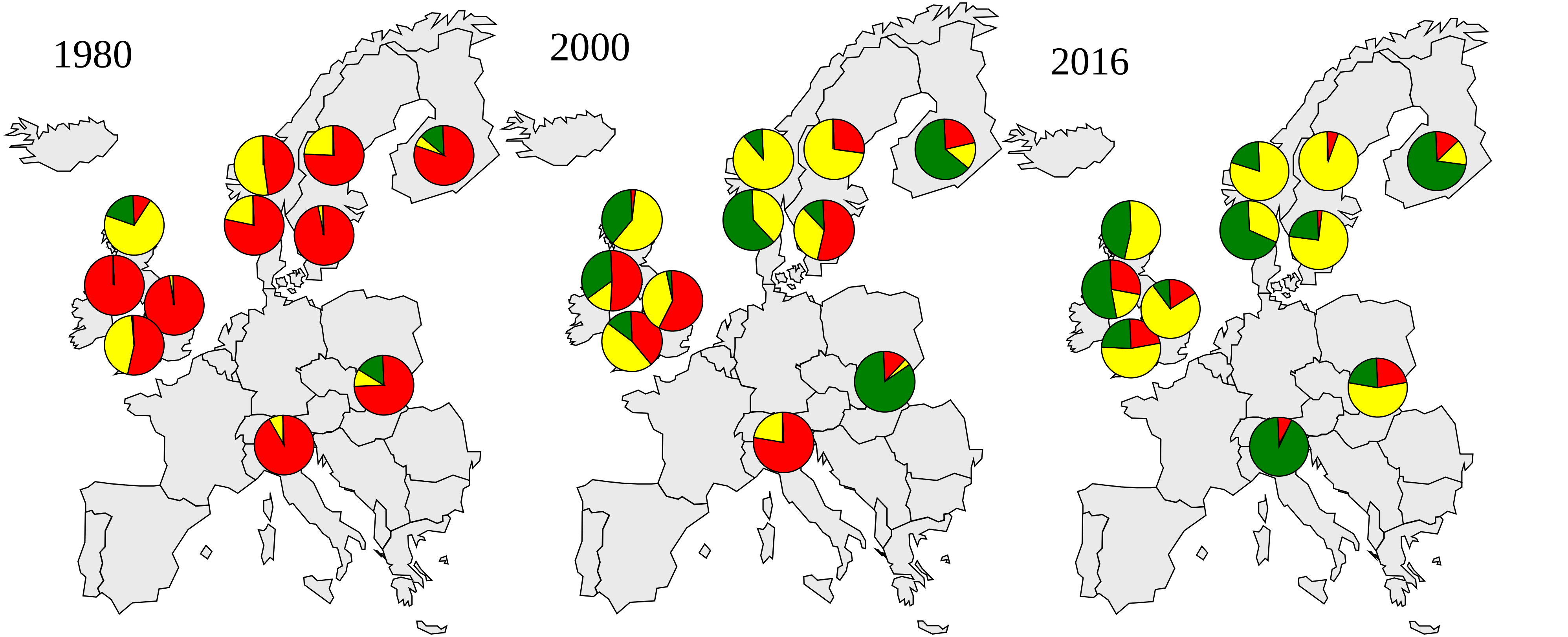
The 1980 model reconstructed surface water ANC concentration for each region expressed in three ANC classes.



The 2000 observed surface water ANC concentration for each region expressed in three ANC classes.



The 2016 predicted surface water ANC concentration for each region expressed in three ANC classes.

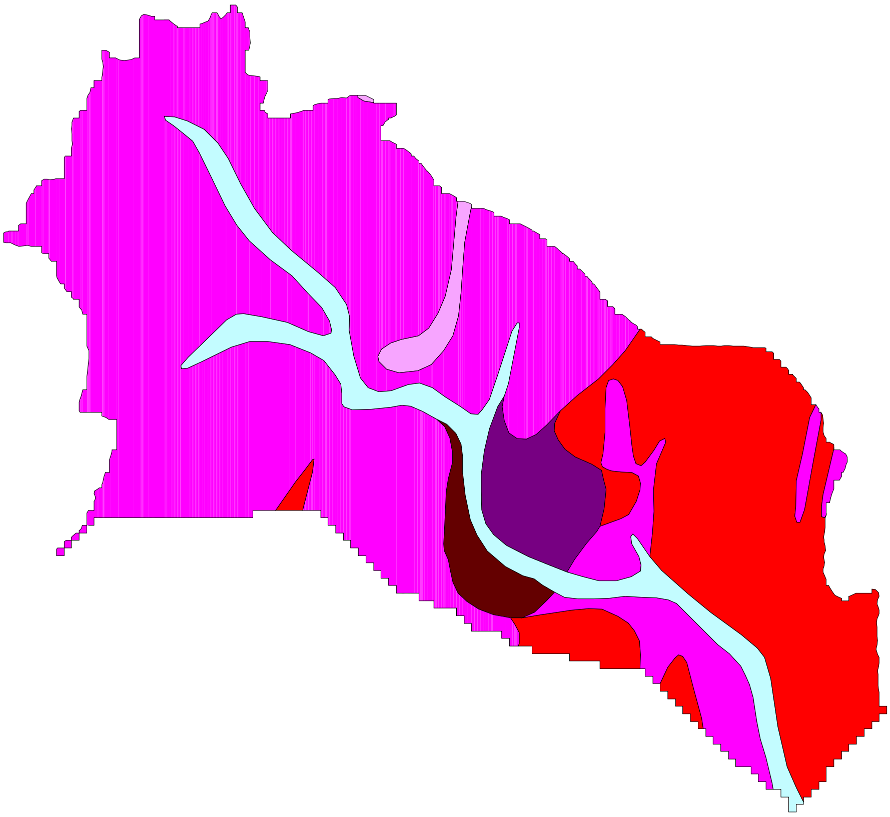


Yearly incremental change in %BS in catchment soils of the lakes and streams in each of 12 regions of Europe. Red denotes depletion of pool of exchangeable bases by more than 1‰ yr-1, yellow denotes no change, and green denotes replenishment by more than 1‰ yr-1.

Chapter 9
Data aggregation

Single site aggregation

When modelling a site meteorological and deposition data, soil characteristics, vegetation characteristics and water chemistry are needed (see chapter 6). The dynamic models use aggregated data, often only one value for each parameter for the modelled site, whereas in reality the parameters will have different values for different parts of the site. Soil parameters for example vary with depth and from point to point within the catchment. For example a catchment might consist of different soil types with very different properties like the Hafren catchment in Wales, below.



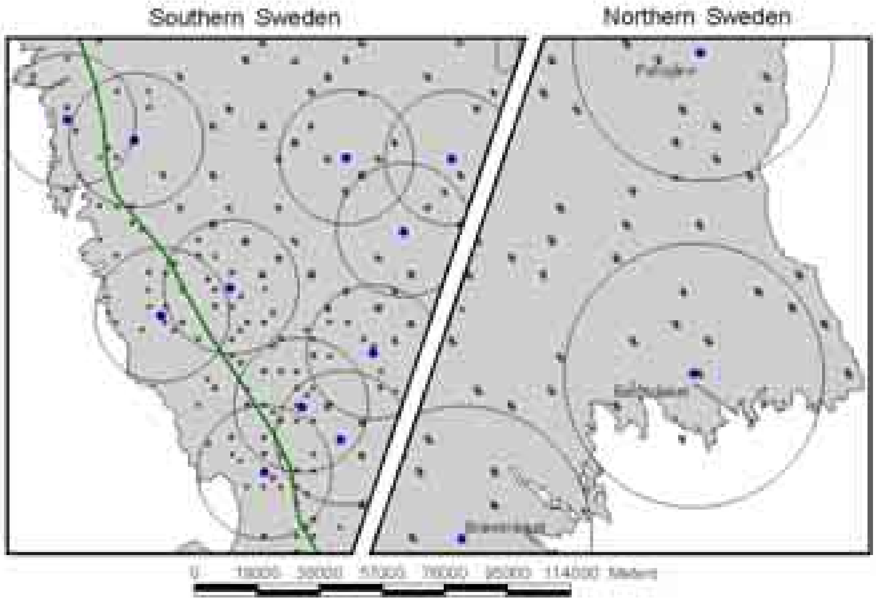
Hafren catchment, Wales, UK: 30 % ironpan stagnopodzol and 70 % peat.

The data for each parameter often must be aggregated into one value that represents the whole site, both vertically and horizontally. The result is soil data that describe the vertically and spatially averaged characteristics of the modelled catchment.

Aggregation for regional application

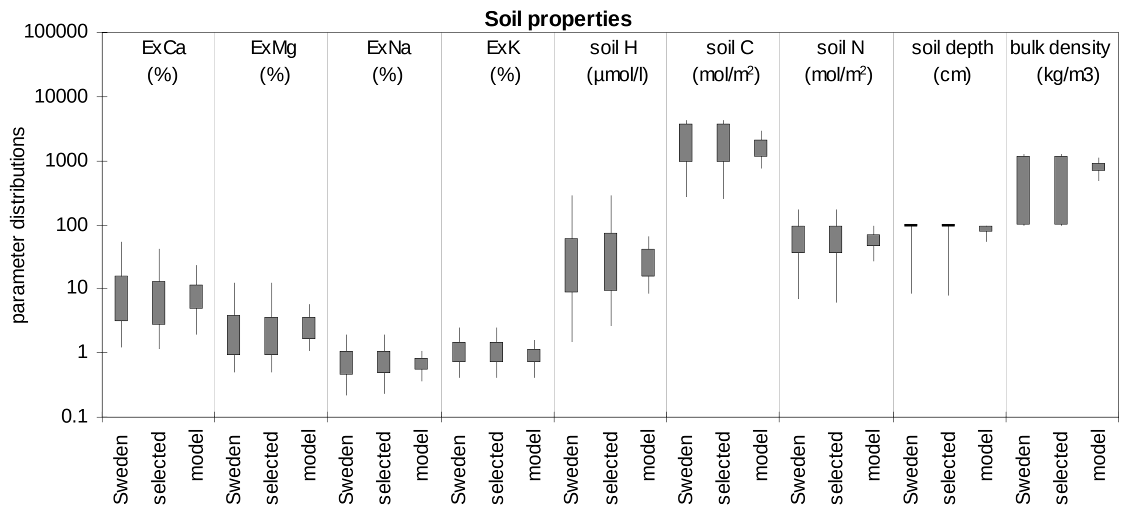
In a regional application of a dynamic model the data quality for the individual sites is usually poorer than when a single site is modelled. The calibration data might come from national measurement programmes, such as the regional application on 143 Swedish lakes, in which lake chemistry from Swedish national lake monitoring programme are used. Lake chemistry has been measured several times per year by the University of Agricultural Sciences, SLU, but data on meteorology, soil characteristics and vegetation in the lake catchments are missing. The data missing for the sites has to be approximated – by using measured data from nearby sites or from sites with similar characteristics, for example the same soil type. In the Swedish regional application, soil data were obtained from 1800 sampling sites from the

National Survey of Forest Soils and Vegetation, where data from the sites close to the lakes were averaged (see figure below).



Values of soil parameters for each lake can be approximated as averages of the values at soil sampling sites near the lakes – those soil sampling sites inside the circles. The criteria for what soil sampling sites are near depend on the spatial heterogeneity of the soils – here northern Sweden has larger circles than southern Sweden.

The distributions of the data should be compared for the original data set used and the data used in the model. The distributions might differ for example if the original data set has all kinds of soils but only sensitive sites are modelled. They might also differ depending on how the data were aggregated and how large the original data set is compared to that for the modelled sites.



Distributions of soil parameters in the Swedish regional application. "Sweden" denotes data from the sites in the National Survey of Forest Soils and Vegetation, "selected" are the subset of those sites near the 143 lakes and "model" are the averaged data used in the modelling (from Moldan et al. 2003).

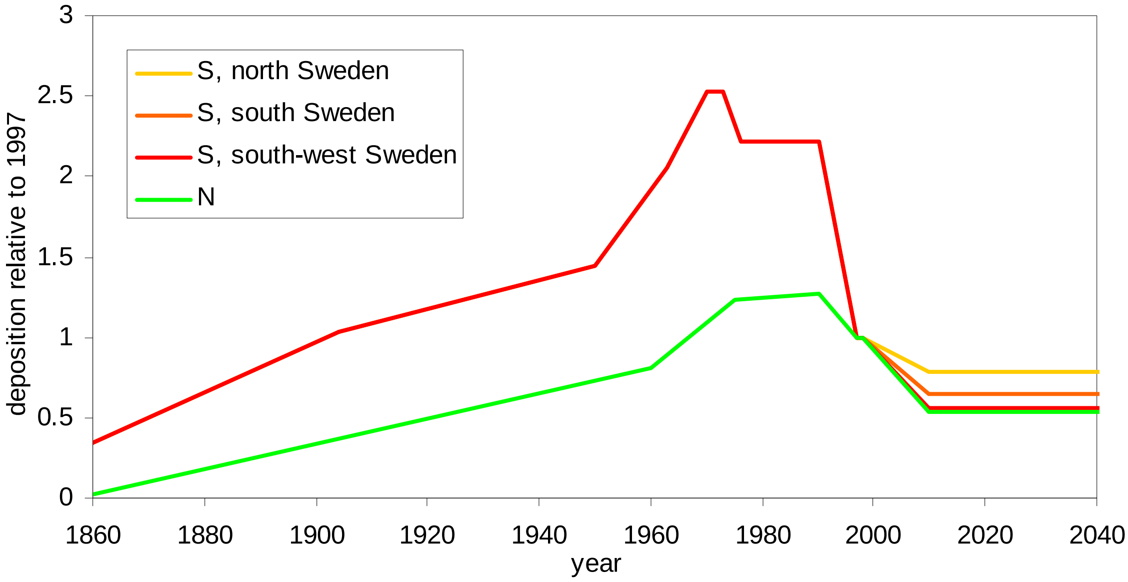
Chapter 10
Scenarios

What scenarios are needed for dynamic modelling?

When using dynamic models, driving variables (see chapter 6) for the calibration year are needed, but also information on the site's history and future, that is scenarios for the past and the future. The scenarios for the past are needed to calibrate the model and to calculate the pre-industrial state of the site. The future scenarios are needed to make predictions for the future state of the site – will it recover and if so how much? The driving variables that change with time are usually deposition and land use.

Construction of scenarios

Historical sequences of deposition come from long series of measurements and reconstructions of deposition using historical data of human activity, for example energy use. With integrated assessment models like RAINS (www.iiasa.ac.at/rains) emissions of pollutants have been calculated from the activity data. Models of atmospheric dispersion, like the EMEP model (www.emep.int), are used to calculate the deposition of acidifying substances for different times and regions from the emission data. Land use history varies from site to site. It is often approximated using information from landowners, or from regional practices in land use.



Sequences for sulphate and nitrogen deposition in Sweden.

Unlike the historical sequences, there are a number of possible future scenarios for each site. For deposition, the effects of emission reductions according to the Gothenburg protocol (www.unece.org/env/lrtap) are often used. Other scenarios that are used are according to current reduction plans and BAT – best available technology. The deposition sequences are calculated from the activities in the same way as for the historical sequences using for example RAINS and the EMEP model.

Future scenarios for land use might come from national forecasts or plans of forestry. They could also be constructed to investigate effects of measures such as forest liming, bio fuel production etc.

Chapter 11
Uncertainties in dynamic modelling

Predictions made using dynamic models have some degree of uncertainty, as is the case with all predictions. In general uncertainties can be grouped into 4 classes (Funtowicz and Ravetz 1990; Saloranta et al. 2003):

- Technical uncertainties (inexactness). These are errors associated with measured data -- model inputs, observations, analytical errors and variability. Technical uncertainty in predictions can be quantified (i.e. precision)
- Methodological uncertainties (unreliability). This type of uncertainty is related to the fact that all models are simplifications of nature and thus cannot perfectly describe nature.
- Epistemological uncertainties (ignorance). “We don’t know what we don’t know”.
- Ethical uncertainty. This is a more abstract form of uncertainty. Alternative value judgements

Sources of “technical uncertainties”

Technical uncertainties are linked to the measurements or the parameters we include when we do dynamic modelling. The measurements or parameter estimates need to be representative for the area (time and space) which may not always be easy. When we measure or estimate a parameter there will always be some errors or uncertainty involved. This becomes even more evident when we need to aggregate several measurements or parameters into one number for a larger area. Examples or technical uncertainty is:

- Parameter estimates may include aggregating several measurements or “qualified guessing” which inherently include some uncertainty.
- Measurement errors are inherent in all types of measurements.
- Natural variability in time and space. Two soil or water samples taken at the same time from the same area will not be exactly similar, and two samples from the same spot will change over time.

Methodological uncertainties

Methodological uncertainty generally relates to the level of scientific understanding. Models are necessarily simplifications of nature and therefore important processes may be excluded or inappropriately described. Also, the measurements we do to evaluate the model results may not reflect the relevant parameter (e.g.: aluminium exists in various forms in solutions, which means that when we measure Al species in solution, it may not be the right species).

Epistemological uncertainties

This type of uncertainty is connected to the limits of our scientific knowledge, “surprises”. This could also be expressed as “We don’t know what we don’t know”. This type of uncertainty is impossible to quantify (by definition), but qualitative assessments may be possible through description of a range of “imaginable surprises”. Examples of epistemological uncertainties could be:

- Exotic species – new plant or animal species invade the ecosystem which then changes to become different from what it used to be.
- Chernobyl – disasters or catastrophic events that may change the ecosystem or the overall conditions completely

Ethical uncertainty

This type of uncertainty relates to policy making, politics and alternative value judgements. The type of consequences we want to include and the way we value things are different. Examples of ethical uncertainties could be:

- Valuation, costs benefit models
- What is “natural” or “good ecological status” ?
- How much ecosystems do we want to protect. Is 95% protected area enough?

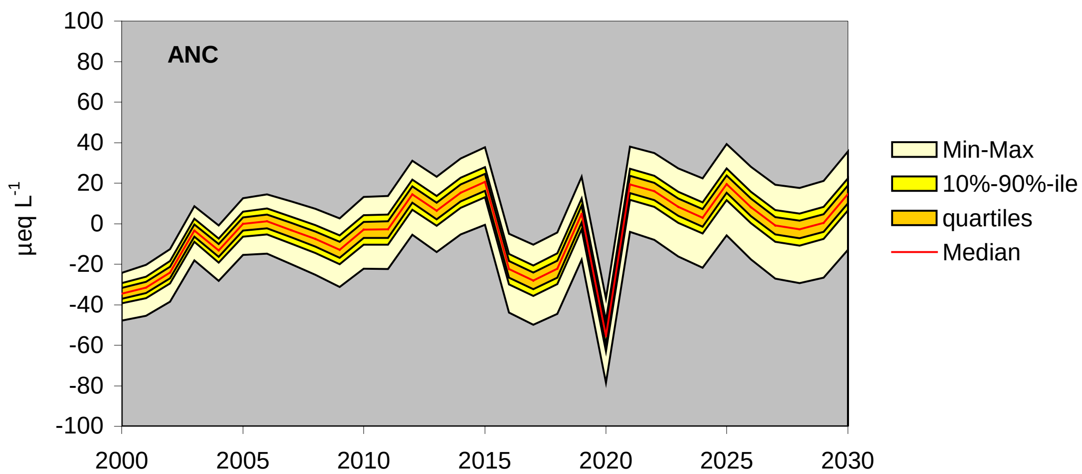
Communication of uncertainty

Because of the uncertainty linked to dynamic modelling and the results they provide, there is an important task in communicating the uncertainties to the people using the results as a basis for actions and decisions. In essence it is the responsibility of the policy makers to decide how certain is good enough.

- Assessment of decision robustness, e.g. “*is the reduction agreement of the protocol good enough even if model results are uncertain?*”
- Presentation of results as risk/probability, e.g. *“We are 95% certain that trout can reproduce in lake x in 2010 if the Gothenburg protocol is implemented”*

When is the water quality good enough? - Is the Gothenburg protocol enough?

We can combine the uncertainties to express probability of reaching a goal. For example:*“High probability for reaching acceptable water quality for trout by 2010 if the Gothenburg protocol is implemented, except in years with extreme sea-salt events”*

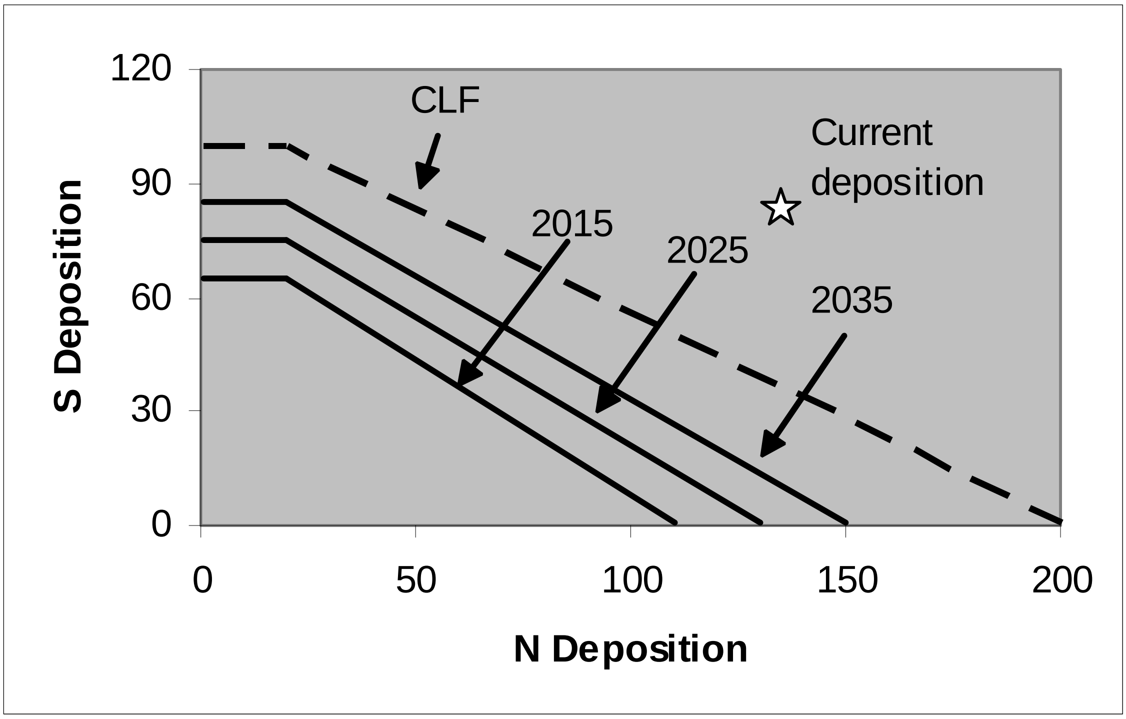


The figure shows predictions of the ANC in the future for the stream at Birkenes, Norway. The estimated uncertainty in the model prediction is included and illustrated in the figure as confidence bands around the median value.

Chapter 12
Target load functions

What is a target load function?

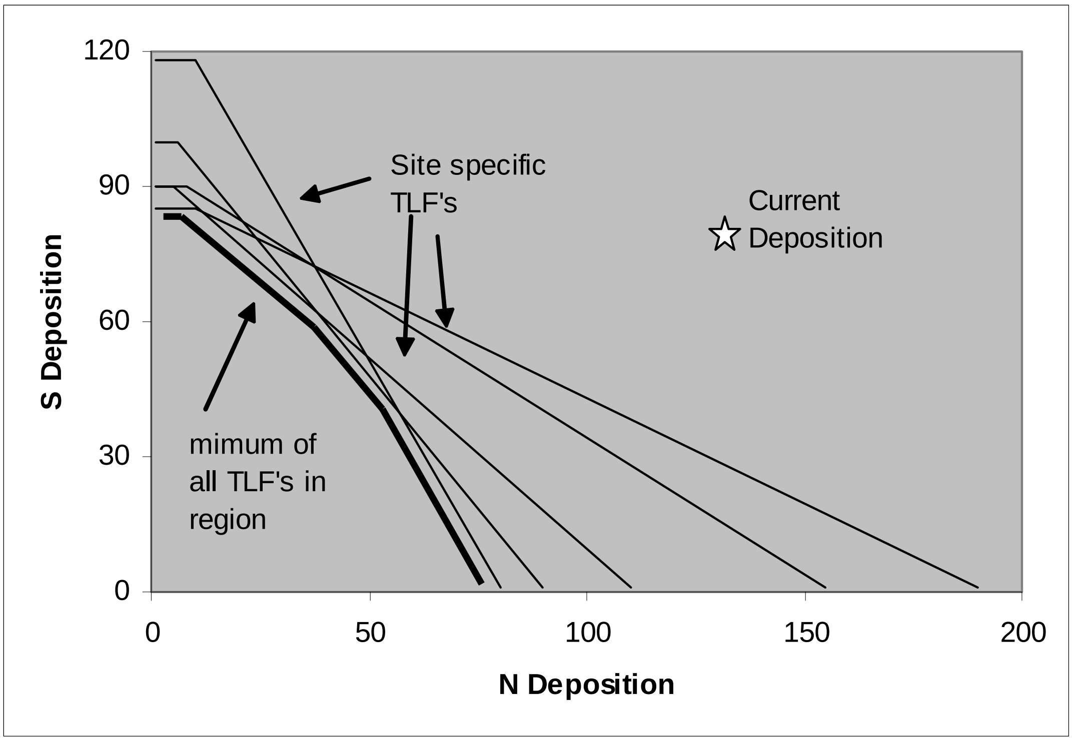
A target load function (TLF) is the dynamic equivalent of the critical load function (see chapter 4) – a function showing the maximum allowable deposition of sulphur and nitrogen. However, critical load shows the maximum allowable load an ecosystem can tolerate in the very long run, whereas the target load function shows the maximum load that allows an acidified ecosystem to recover to a specified year, the target year. Therefore the target load is lower than the critical load (see the figure below). An ecosystem with a deposition exactly at the critical load will never recover. If the deposition is below the critical load but above the target load for a certain year the ecosystem will recover but later than the target year, and if the deposition is less than the target load the ecosystem will recover before the target year. As opposed to the critical load, the TLF depends on the current state of the ecosystem and also on when the decrease in deposition occurs, that is on the accumulated load between now and the target year.



Critical load and target load functions for three years at one ecosystem (Jenkins et al. 2003b).

Regional target loads

For a region with several modelled ecosystems the amount of TLFs (especially if several target years are used) makes the acidification status of the region difficult to interpret.



Target load functions for many ecosystems in a region (Jenkins et al. 2003b).

If the TLFs are to be used as a measure of the region's acidification status (for example in an integrated assessment model like RAINS), they need to be aggregated. This could be done with simple descriptive statistics, where the minimum TLFs or percentiles of ecosystems not recovering to the target year are shown.

Chapter 13
Confounding factors

Reduction of emissions of acidifying compounds in Europe and eastern North America during the past 20 years has led to decreased deposition of sulphur (S) and nitrogen (N). In response, surface waters in many areas are recovering from acidification (Stoddard et al. 1999, Skjelkvåle et al. 2001, Evans et al. 2001). Trends in acidification parameters such as pH and acid neutralising capacity (ANC), however, have seldom been smooth or monotonic, as the chemistry of acidified waters is also affected by variations in climate, deposition of seasalts and other factors independent of acid deposition.

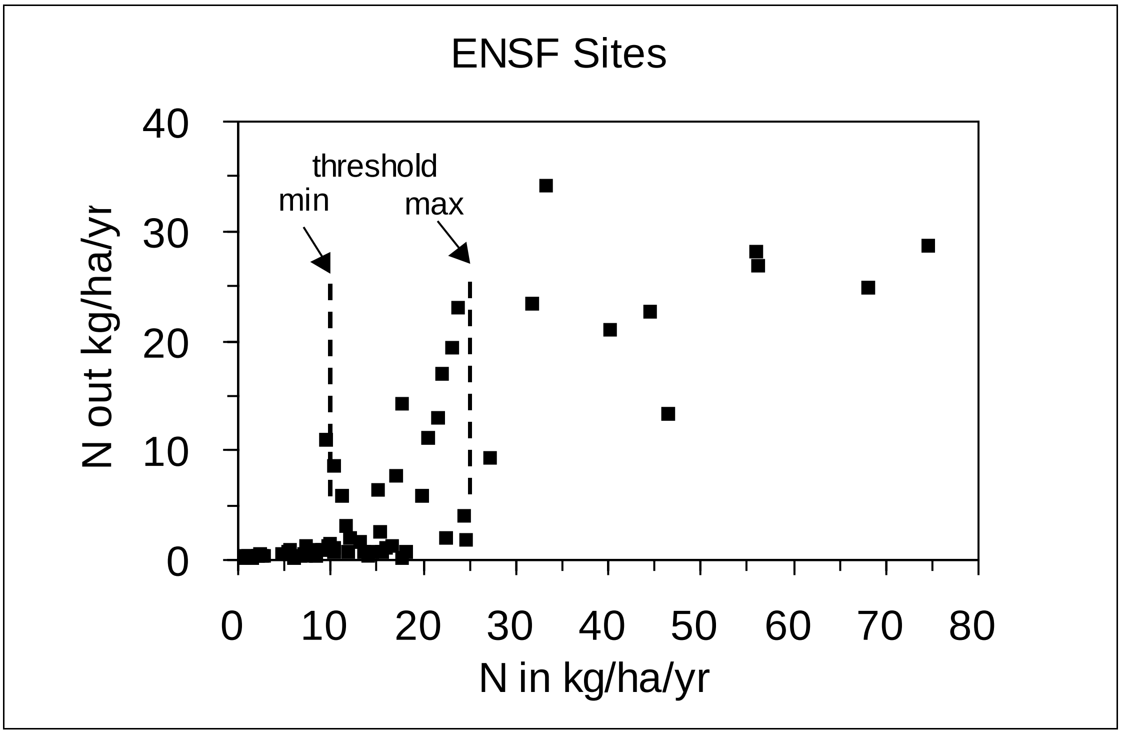
Predictions made with dynamic models make several assumptions with respect to environmental conditions in the future. Predictions are usually based on assumed scenarios for future emissions of S and N and the resulting acid deposition, and also often on scenarios of future land-use practices, such as forest cutting and replanting. There are, however, other environmental factors that may change in the future and that may affect recovery of ecosystems. These “confounding factors” add to the uncertainty in predictions. Nitrogen saturation and global change are two of these confounding factors. Others may include land-use change, changes due to other pollutants such as heavy metals and toxic organic pollutants, as well as shifts in the biological components of the ecosystems caused by, for example, invasion of exotic species.

Nitrogen saturation

During the 1900’s large regions of Europe received elevated deposition of nitrogen (N) compounds. Emissions of oxidised N species from combustion of fossil fuels and emissions of reduced N compounds from agriculture increased dramatically in Europe during the 1900’s to reach peak levels about 1980. Since then emissions have levelled off and decreased slightly in most countries.

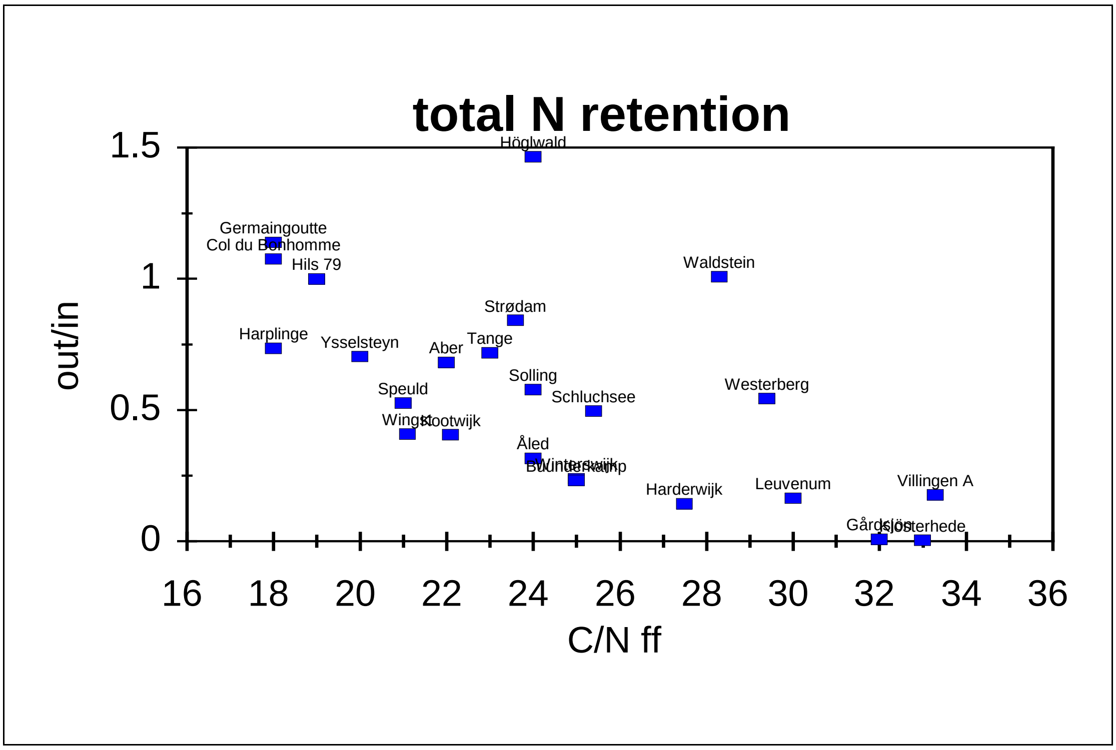
Excess N deposition has long been viewed as a threat to the nutrient balance and health of forests and semi-natural terrestrial ecosystems. In the absence of significant N deposition, N is usually the growth-limiting nutrient in these ecosystems. Chronic excess N deposition can lead to N saturation, defined by Aber et al. 1989 as “the availability of ammonium and nitrate in excess of total combined plant and microbial nutritional demand”. By this definition N saturation is manifest by increased leaching of inorganic N (generally nitrate) below the rooting zone. Inasmuch as nitrate is a strong acid anion, increased leaching of nitrate enhances acidification of soils and surface waters. Increased concentrations of inorganic N in runoff (stream water) thus indicate N saturation of terrestrial ecosystems, under the conditions, of course, that there are no significant sources of N in the catchment (such as fertilisers, municipal and industrial wastewater).

A survey of forest ecosystems in Europe showed that sites with high N deposition leached nitrate, whereas sites with low N deposition did not. The figure indicates two thresholds for N saturation. Below about 10 kgN/ha/yr no site leached NO_3 whereas above about 25 kgN/ha/yr all sites leached NO_3. N deposition levels in Europe in the 1990s were above 25 kgN/ha/yr in eastern UK, southern Scandinavia, central Europe and the southern Alps.



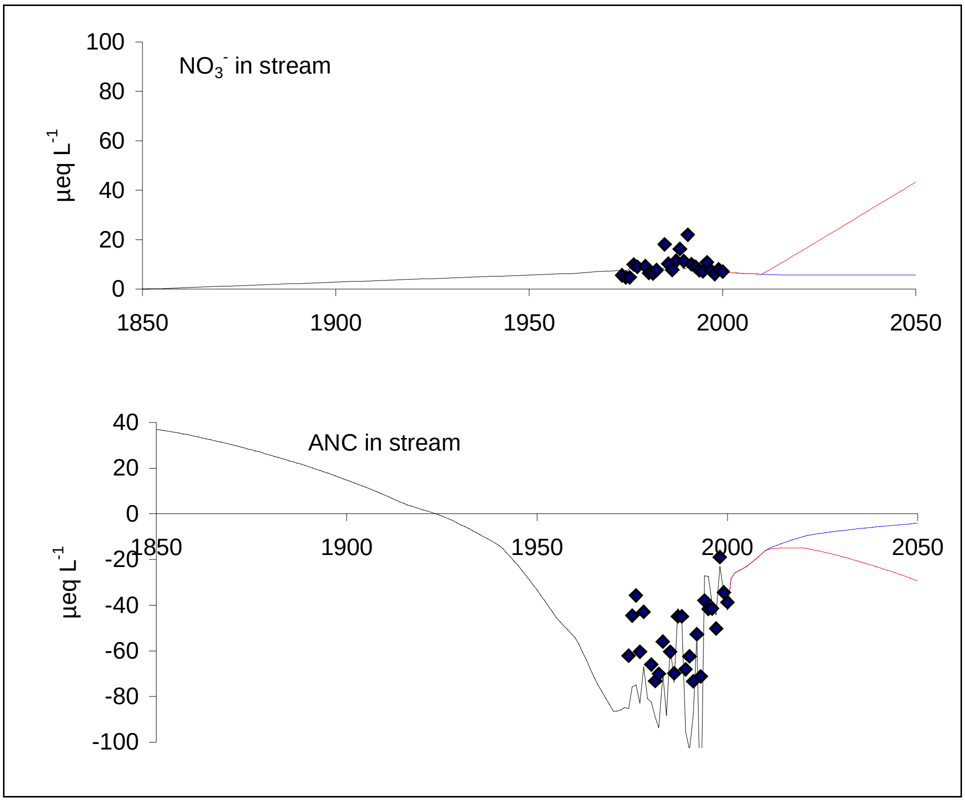
Source: Dise and Wright 1995

The variation in NO_3 leaching from forest stands can be explained in part by the amount of N stored in the uppermost soil layers, the forest floor. Here the empirical data show that sites with low C/N ratios leach a larger fraction of N deposition relative to sites with high C/N ratios.



Relationship between fraction of N leached (out/in) to C/N (g/g) in forest floor at forest stands and catchments in Europe (after Gundersen et al. 1998).

Together the empirical information on these two graphs strongly suggests that over time chronic elevated n deposition results in accumulation of N in the forest floor and a reduced ability of the forest ecosystem to retain incoming N. In other words, chronic N deposition can cause N-limited ecosystems to become N-saturated.

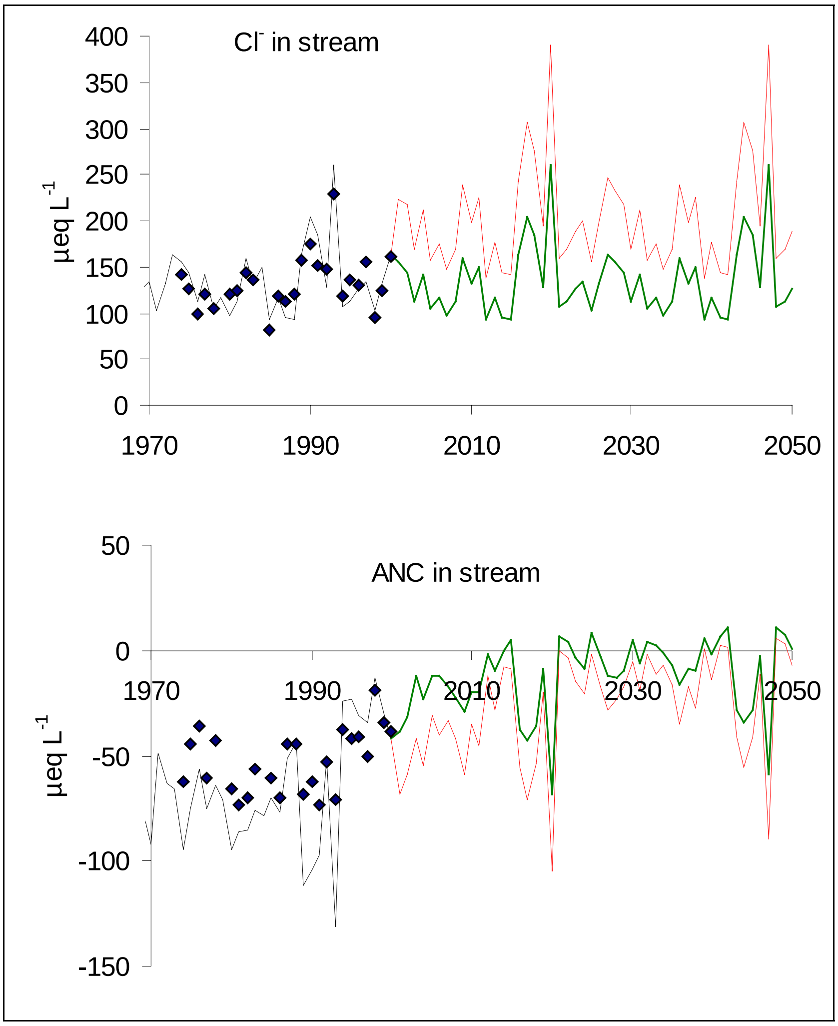


Two scenarios for N retention at Birkenes, a small forested catchment in southernmost Norway showing forecasts for the future: current legislation scenario for future deposition of sulphur and N (blue line = no increased N saturation. red line = increased N saturation. Dots are measurements). If N saturation does not occur, the ANC in streamwater will increase in the future (blue line), but if N saturation occurs, the stream will re-acidify and ANC will decrease. This means that additional measures would be required to achieve recovery at Birkenes (from T. Larssen, NIVA)

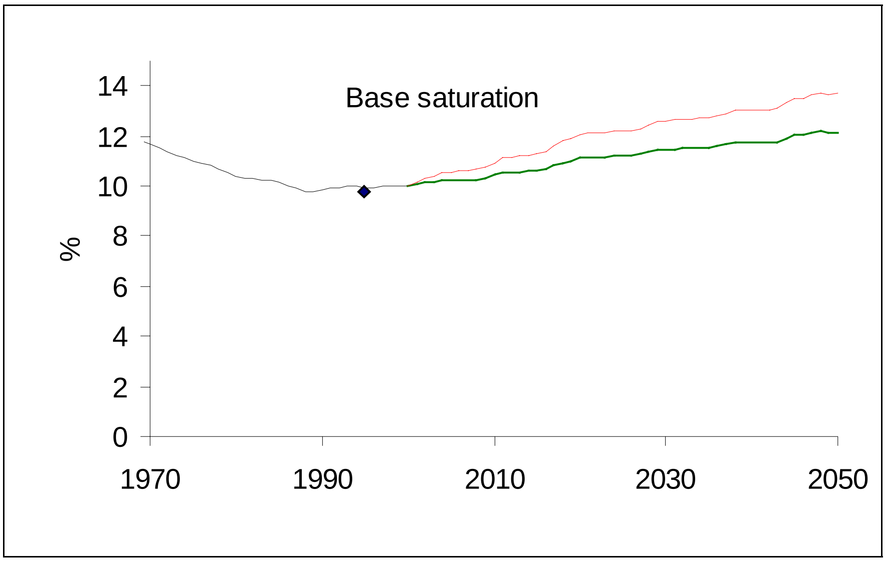
Extreme events

Future climate change may entail more frequent and more severe extreme events, such as storms, floods and droughts. These may influence the recovery of aquatic ecosystems.

Example: The importance of storms at Birkenes, southern Norway.
Birkenes is a small forested catchment in southern Norway. The stream is highly acidified but has begun to recover during the past 10-15 years. Birkenes is located near the south coast, and thus has high inputs of seasalts in deposition. Storm events can cause large seasalt inputs over a short time. Due to cation exchange in the soil, seasalt events cause acidity shocks in the stream.



The effect of increased storm frequency at Birkenes can be seen in this scenario of 50% increased Cl deposition at Birkenes starting in the year 2000. The green line shows the present-day situation in which the seasalt inputs measured during the past 30 years (1973-2000) are simply repeated in the future, whereas the red line shows the situation with a scenario of 50% increase. The ANC curves show that with higher seasalt inputs the episodic drops in ANC will be more severe in the future and thus recovery will be delayed.



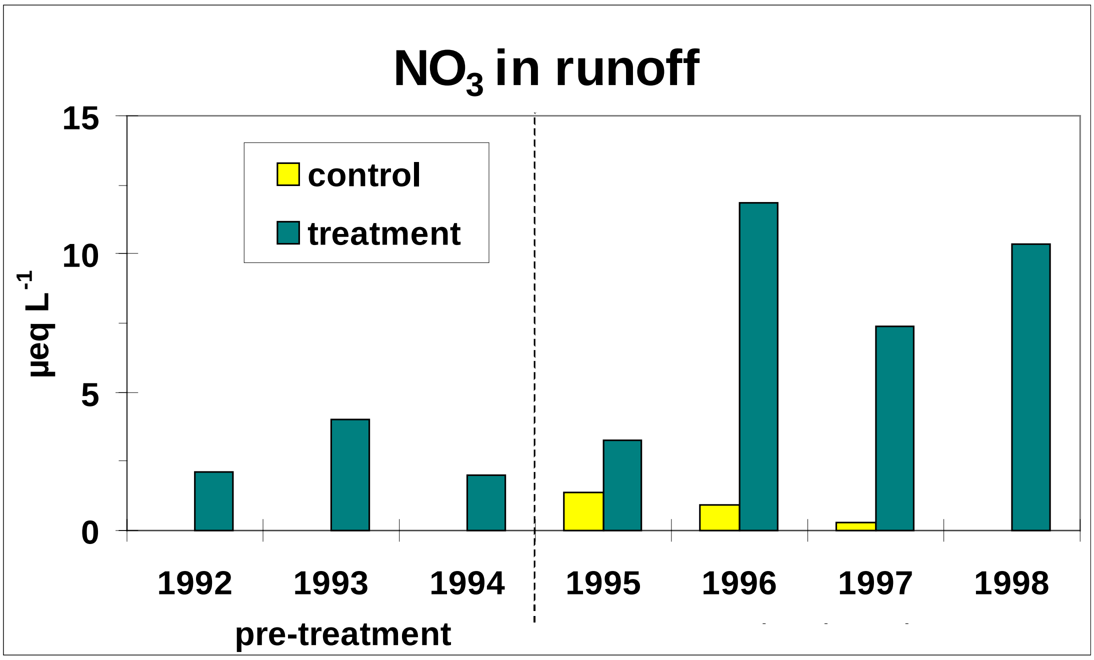
The news is not all bad, however, because over the long term the extra seasalt inputs will speed up the replenishment of the pool of exchangeable base cations in the soil such that the base saturation will increase faster. Thus the recovery of the ANC in streamwater will be slower over the short term, but faster over the long term.

Global warming

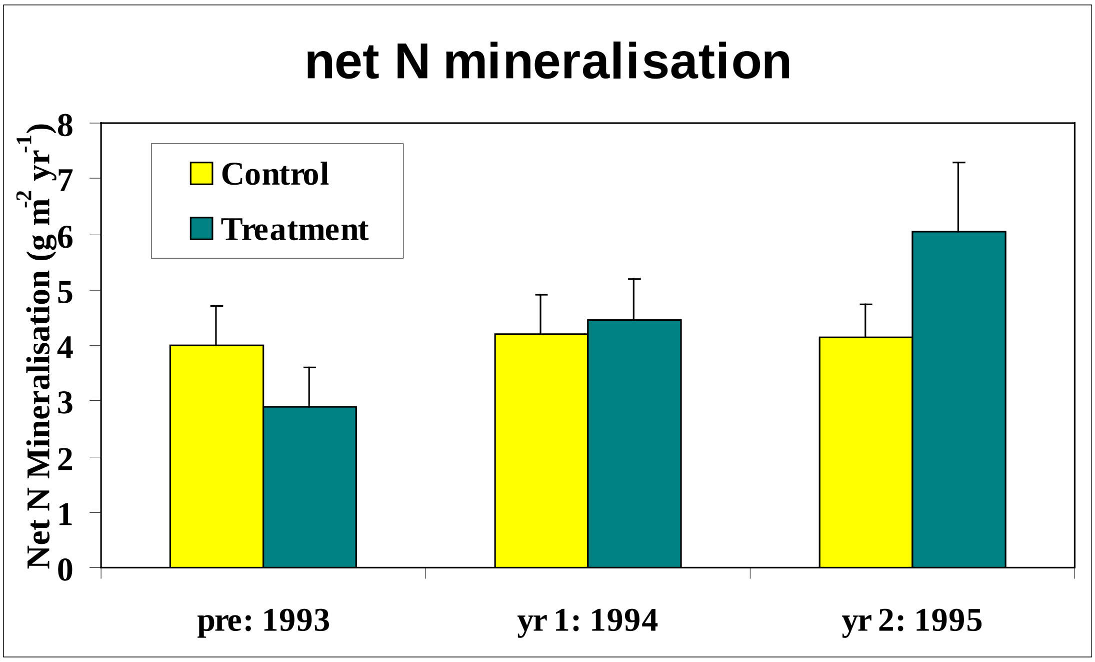
The CLIMEX experiment at Risdalsheia, southern Norway, provides direct information on the potential confounding effects of climate change on the reversibility of acidification. CLIMEX entailed whole-ecosystem warming and elevated levels of CO_2 to a forested catchment.

The greenhouse enclosing KIM catchment, Risdalsheia, southern Norway.

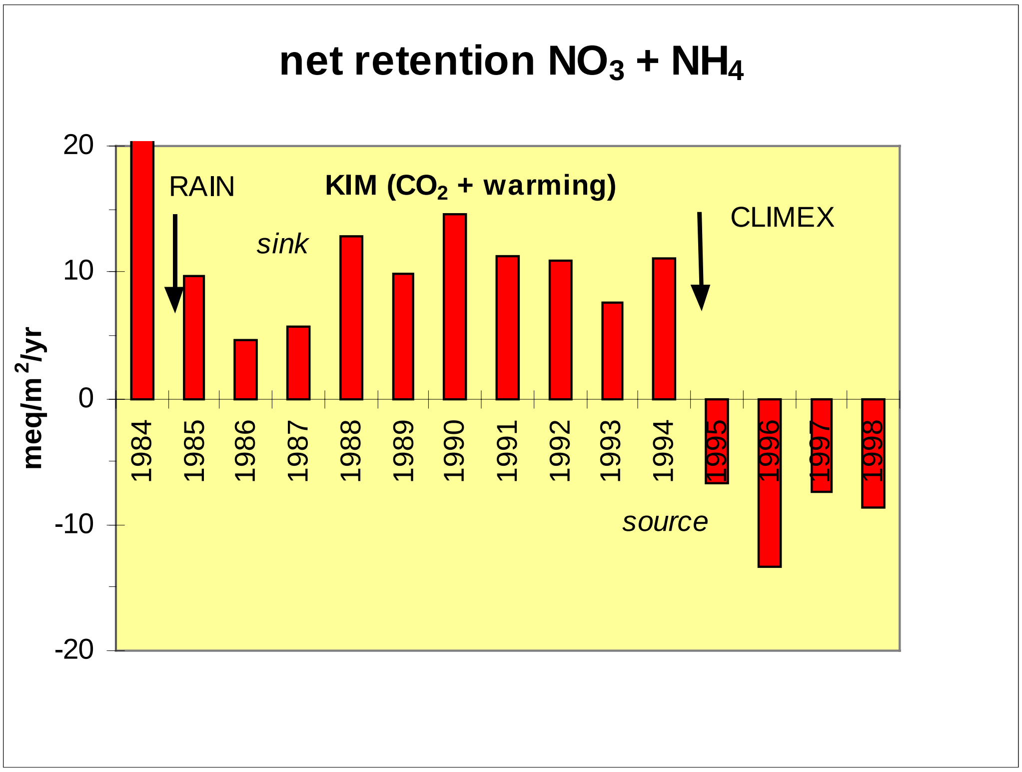
The CLIMEX experiment at Risdalsheia represents four anomalously warm years superimposed upon the 15-year clean rain treatment. At Risdalsheia it is the warm years that gave high concentrations of NO_3 (and to a lesser extent NH_4) in runoff (Wright 1998).



Measurement of N mineralisation in soils at Risdalsheia showed a statistically significant increase of 50% at KIM catchment relative to the reference during the second year of treatment (Verburg and van Breemen 2000, Verburg et al. 1999).



The N input-output budgets show that with the onset of warming and increased CO_2 treatment, the ecosystem switched from a net sink to a net source of inorganic N. During the entire 11-year period prior to the climate change treatment (1984-1994) the ecosystem was a net sink for N, a situation typical for most boreal forests. The ecosystem lost N during all four years of climate change treatment, probably due to the increased decomposition of soil organic matter and release of N to soil solution. After only four years of warming it is difficult to judge whether the increased N release is merely a transient phenomenon.


net retention NO3 + NH4
20
10
0
-10
-20
meq/m2/yr
RAIN
KIM (CO2 + warming)
CLIMEX
sink
source
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
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1995
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1998

Reference List

Aber, J. D., Nadelhoffer, K. J., Steudler, P., and Melillo, J. 1989. Nitrogen saturation in northern forest ecosystems. *Bioscience* **39**: 378-386.

Beier, C., Blanck, K., Bredemeier, M., Lamersdorf, N., Rasmussen, L., and Xu, Y. J. 1998. Field-scale 'clean rain' treatments to two Norway spruce stands within the EXMAN project -- effects on soil solution chemistry, foliar nutrition and tree growth. *For.Ecol.Manage.* **101**: 111-123.

Beier, C., Hultberg, H., Moldan, F., and Wright, R. F. 1995. MAGIC applied to roof experiments (Risdalsheia, N; Gårdsjön, S; Klosterhede, DK) to evaluate the rate of reversibility of acidification following experimentally reduced acid deposition. *Water Air Soil Pollut.* **85**: 1745-1751.

Beier, C., Moldan, F., and Wright, R. F. 2003. Terrestrial ecosystem recovery - Modelling the effects of reduced acidic inputs and increased inputs of sea salts induced by global change . *Ambio* **32**: 275-282.

Cosby, B. J., Ferrier, R. C., Jenkins, A., and Wright, R. F. 2001. Modelling the effects of acid deposition: refinements, adjustments and inclusion of nitrogen dynamics in the MAGIC model. *Hydrol.Earth System Sci.* **5**: 499-518.

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

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

de Vries, W., Posch, M., and Kämäri, J. 1989. Simulation of the long-term soil response to acid deposition in various buffer ranges. *Water Air Soil Pollut.* **48**: 349-390.

Dise, N. B. and Wright, R. F. 1995. Nitrogen leaching from European forests in relation to nitrogen deposition. *For.Ecol.Manage.* **71**: 153-162.

Evans, C. D., Cullen, J., Alewell, C., Kopácek, J., Marchetto, A., Moldan, F., Prechtel, A., Rogora, M., Veselý, J., and Wright, R. F. 2001. Recovery from acidification in European surface waters. *Hydrol.Earth System Sci.* **5**: 283-298.

Funtowicz, S. O. and Ravetz, J. R. 1990. Uncertainty and quality in science for policy. Theory and decision library A: Philosophy and Methodology of the Social Sciences. Kluwer, Dordrecht

Gundersen, P., Callesen, I., and de Vries, W. 1998. Nitrate leaching in forest ecosystems is controlled by forest floor C/N ratio. *Environ.Pollut.* **102**: 403-407.

Gunn, J. M. and Sandøy, S. 2003. Introduction to the Ambio special issue on biological recovery from acidification: Northern lakes recovery study. *Ambio* **32**: 162-164.

Hesthagen, T., Sevaldrud, I., and Berger, H. M. 1999. Assessment of damage to fish populations in Norwegian lakes due to acidification. *Ambio* **28**: 12-17.

Hultberg, H. and Skeffington, R. A. 1998. Experimental Reversal of Acid Rain Effects. The Gårdsjön Roof Project. Wiley and Sons, Chichester, UK

Jenkins, A., Camarero, L., Cosby, B. J., Ferrier, R. C., Forsius, M., Helliwell, R. C., Kopácek, J., Majer, V., Moldan, F., Posch, M., Rogora, M., Schöpp, W., and Wright, R. F. 2003. A modelling assessment of acidification and recovery of European surface waters. *Hydrol.Earth System Sci.* **in press**:

Jenkins, A., Cosby, B. J., Ferrier, R. C., Larssen, T., and Posch, M. 2003. Assessing emission reduction targets with dynamic models: Setting target load functions for use in Integrated Assessment Modelling. *Hydrol.Earth System Sci.* **in press**:

Kroglund, F., Wright, R. F., and Burchart, C. 2002. Acidification and Atlantic salmon: critical limits for Norwegian rivers. Naturens Tålegrenser Fagrapport 111, Norwegian Institute for Water Research, Oslo. 61 pp.

Lien, L., Raddum, G. G., Fjellheim, A., and Henriksen, A. 1996. A critical limit for acid neutralizing capacity in Norwegian surface waters, based on new analyses of fish and invertebrate responses. *Sci.Tot.Environ.* **177**: 173-193.

Moldan, F. 1999. Reversal of Soil and Water Acidification in SW Sweden, Simulating the Recovery Process. Swedish University of Agricultural Sciences, Umeå, Sweden, Doctoral thesis pp.

Moldan, F., Bergkvist, V., and Cosby, B. J. 2003. Swedish lakes: modelling the recovery from acidification in response to the declining atmospheric deposition. *Water Air and Soil Pollution: Focus* **In press**:

Neal, C., Whitehead, P. G., and Jenkins, A. 1988. Reversal of acidification. *Nature* **334**: 109-110.

Nilsson, J. and Grennfelt, P. 1988. Critical loads for Sulphur and Nitrogen. Nordic Council of Ministers Report 1988:15, Nordic Council of Ministers, Copenhagen, Denmark. 418 pp.

Posch, M., Forsius, M., Johansson, M., Vuorenmaa, J., and Kämäri, J. 2003. Modelling the recovery of acid-sensitive Finnish headwater lakes under present emission reduction agreements. *Hydrol.Earth System Sci.* **in press**:

Posch, M., Hettelingh, J.-P., de Smet, P. A. M., and Downing, R. J. 1997. Calculation and Mapping of Critical Thresholds in Europe. Status Report 1997. RIVM, Bilthoven, the Netherlands. 163 pp.

Posch, M., Hettelingh, J.-P., Slootweg, J., and Downing, R. J. 2003. Modelling and Mapping of Critical Thresholds in Europe: CCE Status Report 2003. 259101013/2003, RIVM, Bilthoven. 132 pp.

Posch, M., Hettelingh, J.-P., Slootweg, J., and eds. 2003. Manual for dynamic modelling of soil response to atmospheric deposition. Report 259101012/2003, WGE ICP-M&M Coordination Center for Effects, RIVM, Bilthoven, The Netherlands.

Saloranta, T. M., Kämäri, J., Rekolainen, S., and Malve, O. 2003. Benchmark criteria: A tool for selecting appropriate models in the field of water management. *Environmental Management* **in press**:

Schöpp, W., Posch, M., Mylona, S., and Johansson, M. 2003. Long-term development of acid deposition (1880-2030) in sensitive freshwater regions in Europe. *Hydrol.Earth System Sci.* **in press**:

Skjelkvåle, B. L., Mannio, J., Wilander, A., and Andersen, T. 2001. Recovery from acidification of lakes in Finland, Norway and Sweden 1990-1999. *Hydrol.Earth System Sci.* **5**: 327-338.

Stoddard, J. L., Jeffries, D. S., Lükewille, A., Clair, T. A., Dillon, P. J., Driscoll, C. T., Forsius, M., Johannessen, M., Kahl, J. S., Kellogg, J. H., Kemp, A., Mannio, J., Monteith, D., Murdoch, P. S., Patrick, S., Rebsdorf, A., Skjelkvåle, B. L., Stainton, M. P., Traaen, T. S., van Dam, H., Webster, K. E., Wieting, J., and Wilander, A. 1999. Regional trends in aquatic recovery from acidification in North America and Europe 1980-95. *Nature* **401**: 575-578.

Sverdrup, H. and Warfvinge, P. 1992. PROFILE -- A mechanistic geochemical model for calculation of field weathering rates, p.81-87, In: Kharaka, Y., *7th Water-Rock Interaction Symposium*, A. Balkemaa Publishers, Rotterdam.

Verburg, P. S. J. and van Breemen, N. 2000. Experimental manipulation of a forested catchment: litter decomposition and N mineralization. *For.Ecol.Manage.* **129**: 31-39.

Verburg, P. S. J., van Dam, H., Hefting, M. M., and Tietema, A. 1999. Effects of elevated temperature on microbial transformations of C and N in a boreal forest floor. *Plant and Soil* **208**: 189-197.

Warfvinge, P., Falkengren-Grerup, U., and Sverdrup, H. 1993. Modeling long-term base cation supply to acidified forest stands. *Environ.Pollut.* **80**: 209-220.

Wathne, B. M. and Rosseland, B. O. 1999. MOLAR Final Report. Measuring and modelling the dynamic response of remote mountain lake ecosystems to environmental change: A programme of Mountain Lake Reseach - MOLAR. 4/1999, Norwegian Institute for Water Research,

Wright, R. F. 1998. Effect of increased CO_2 and temperature on runoff chemistry at a forested catchment in southern Norway (CLIMEX project). *Ecosystems* **1**: 216-225.

Wright, R. F. and Cosby, B. J. 2003. Future recovery of acidified lakes in southern Norway predicted by the MAGIC model. *Hydrol.Earth System Sci.* **in press**:

Wright, R. F. and Lie, M. C. 2002. Workshop on Models for Biological Recovery from Acidification in a Changing Climate. Acid Rain Research Report 55/02, Norwegian Institute for Water Research, Oslo. 42 pp.

Wright, R. F., Lotse, E., and Semb, A. 1993. RAIN project: results after 8 years of experimentally reduced acid deposition. *Can.J.Fish.Aquat.Sci.* **50**: 258-268.