

The Use of Dynamic Models to Set Target Loads

The concept of critical load addresses the level of environmental stress that will not harm ecosystems in a long-term perspective. Methods used to calculate critical loads do not take the time scales of acidification and recovery into account. Therefore, the environmental consequences of different emission reductions can not be fully assessed using only the empirical and steady-state methods to calculate critical loads. Using dynamic chemical models, the change in key chemical parameters in three Nordic ecosystems were predicted for several deposition scenarios. Target loads for S deposition were set, using 3 different dynamic models, under the condition that the chemical criteria should be met by the year 2037. For catchments with shallow soils, the target loads for S in surface water, calculated with all models, were similar. All models indicate that the S deposition at Birkenes in southern Norway must be reduced to $< 0 \text{ meq m}^{-2} \text{ yr}^{-1}$ to meet the chemical target, ANC in run off greater than 20 meq m^{-3} . For a catchment with deep soils, Yli-Knuutila in Finland, the calculated target load ranged from $250 \text{ meq S m}^{-2} \text{ yr}^{-1}$ calculated with a multi-layered model to 1500, determined with a one compartment model. The calculated target loads for soils show similar differences between models, indicating that for soils deeper than the rooting zone, the choice of model largely determines the estimated target load. For acidified sites, the nearer into the future the target year is, the lower the target load. Systems not yet damaged can be protected for the future, if the load is lower than the critical load. The longer the system is to be protected, the lower the target load.

INTRODUCTION

The present state of acidification of soils and surface waters in Fennoscandia 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 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.

The concept of critical load is now widely used as a basis for decisions regarding future emissions of acidifying compounds. 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* (1).

The target load is a political instrument to define the level of deposition that can be tolerated. Target loads may be based on critical-load calculations, and an assessment of the ecological consequences of different degrees of exceedance. The target load is thus an operational value and for a given ecosystem can be set at a level higher, equal or lower than the critical load. In either of these situations, the ecosystem may be damaged or protected depending on

the properties of the system, the current status and the time frame considered. If the acid load is changed to equal the critical load, we expect the ecosystem to approach conditions that correspond to the chemical criteria applied to set the critical load. Critical loads are thus viewed as being intrinsic properties of the environment, whereas target loads are set according to political or management decisions.

Procedures for the determination of critical load for a given ecosystem or region entail the use of models. Three types of models are available: empirical models; steady-state models; and dynamic process-oriented models (2). Much of the mapping work carried out to date is based on steady-state and empirical models (3, 4). 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 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.

Here we illustrate these dynamic aspects of acidification and the implications for

critical loads and target loads. We apply four dynamic process-oriented models of soil and water acidification to data from three catchments in the Nordic area (Fig. 1).

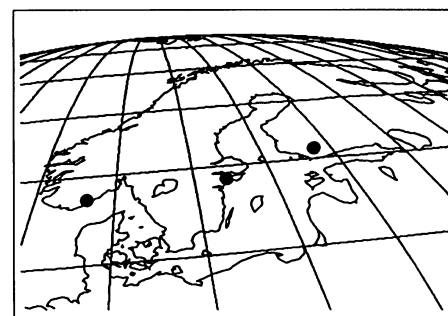


Figure 1. The location of the Nordic sites to which the models have been applied.

The models MAGIC (5, 6), MIDAS (7), SAFE (8) and SMART (9) were all developed for use in Nordic countries. The sites, Birkenes (Norway), Stubbetorp (Sweden), and Yli-Knuutila (Finland) were chosen because of the availability of extensive high-quality sets of data for deposition, soils and surface waters, and because they span a range in present-day acidification status.

We calibrated the models to present-day conditions, and then used the calibrated models to estimate the target loads for sulfur in water and soils. The model predictions were compared and the differences evaluated with respect to the use of these models in determining target loads. This work was conducted as part of the activities supported by the Nordic Council of Ministers Air Group and detailed results have been reported in Wright et al. (10).

METHODS

Models

The models applied are deterministic dynamic models, based on differential equations which describe the change-in-state variables such as soil-solution concentration over time. Compared with steady-state methods, a number of processes that can be omitted under steady-state conditions must be considered in the dynamic calculations. These processes are characterized by the criteria that the pools in the reaction system may arrive at equilibrium with each other. The net flux between the pools will then equal 0. This means that nutrient uptake must be considered in both critical load and target-

load calculations, since the pool of stored nutrient in a managed forest will always change over time. On the other hand, the pools of exchangeable bases and anions may well reach a true steady-state with the surrounding solution, and these processes can therefore be neglected when assessing critical loads.

All models are simplified when compared with the actual ecosystem to be modelled. Nevertheless, dynamic models are quite complex, simply because they consider a large number of chemical processes simultaneously. Most models include several key processes and time scales for the reaction systems (Table 1).

The mixture of processes and time-scales leads to different modeling approaches depending on the intended use and applications of the models. The 4-dynamic models used here all use a similar approach (Fig. 2).

The MAGIC model is an extension of the approach by Reuss and Johnson (11) including all major cations and anions and complexation reactions involving Al, such as hydration and complexation with SO_4^{2-} and F^- . A regionalized version of MAGIC has been successfully used to reproduce the observed distributions of water quality variables derived from regional survey of lakes in southern Norway (12). MAGIC is the most widely distributed and used dynamic model at present.

The MIDAS model focuses on the exchange processes, modelled as a kinetic process involving $Ca^{2+} + Mg^{2+}$ and H^+ and Al^{3+} . The model has been used to explore possible paths of acidification and recovery following decreased acid deposition (7). MIDAS does not predict streamwater chemistry, and was used here only to predict changes in soil base cations, not to calculate target loads.

The SAFE model, and the steady-state version PROFILE, are characterized by the kinetic approach to modeling chemical weathering from geochemical data. In the hydrochemical calculations, $Ca^{2+} + Mg^{2+} + K^+$ are lumped together, and the mass balances are formulated in a way that allows strong acid anions to be neglected.

SMART simulates base saturation and

Table 1. Time scale of chemical processes considered in terrestrial ecosystems. The flux response time reflects the time for a mass flux measured as concentration, water flow or biomass growth rate, to change significantly. The pool response time refers to the time for the entire mass of water, chemical components, water or soil to leave the system to a large extent, under a situation with elevated acid deposition. The soil stratification may change over a rotation period if a change between deciduous and coniferous tree species occurs.

| Process | Flux response time | Pool response time |
|-------------------------------|--------------------|--------------------|
| Soil solution speciation | Instantaneous | Days |
| Hydrological regimes | Days | Seasonal |
| Nitrification/Denitrification | Seasonal | Seasonal |
| Sulfate adsorption | Instantaneous | Decades |
| Cation exchange | Weeks | Many decades |
| Nutrient uptake | Seasonal | Rotation period |
| Chemical weathering | Seasonal | Eons |
| Soil stratification/genesis | Rotation period | Eons |

Figure 2. Basic components of the soil models used.

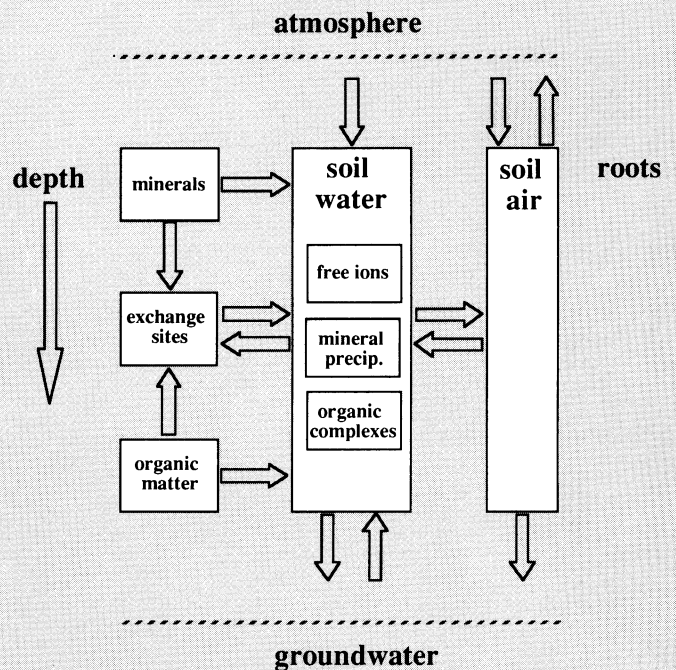


Table 2. Characteristics of the models used to calculate target loads.

| Process | MAGIC | MIDAS | SAFE | SMART |
|---------------------|---------------------------|-----------------------|---------------------|---------------------------|
| Al soil solution | Equilibrium | Equilibrium | Equilibrium | Equilibrium |
| Hydrological | Vertical | Vertical | Vertical/Horizontal | Vertical |
| Nitrification | Input | - | Kinetic | Related to NH_4 |
| Sulfate adsorption | Langmuir | - | - | Langmuir |
| Cation exchange | Gaines Thomas equilibrium | Gaines Thomas kinetic | Gapon mass-transfer | Gaines Thomas equilibrium |
| Nutrient uptake | Input | Input | Input | Input |
| Chemical weathering | Input | Input | Kinetic | Input |
| Soil stratification | One layer | One layer | Multi-layer | One layer |

pH, $\text{Ca}^{2+} + \text{Mg}^{2+}$, Al^{3+} , NH_4^+ , NO_3^- , SO_4^{2-} and HCO_3^- in soil solution. As in the other models, it combines a set of equilibrium equations with mass-balance equations for the different elements.

To facilitate model comparison, simulations were all started in year 1840 on the assumption that the system was stable. Each model was calibrated to fit present-day stream and soil conditions.

In Table 2, the different models used here are characterized with respect to processes included, and model formulations.

Sites

The soils at Birkenes are mainly podzols and brown earths on granitic bedrock with an average depth of 0.4 m, covered by an 80-year old Norway spruce stand with very low productivity. The pool of exchangeable base cations is 17 meq m^{-2} . The annual runoff rate is 1.2 m yr^{-1} , and the pH of the streamwater averages 4.5. The Stubbetorp catchment is covered with a productive forest, and the average soil depth is 0.8 m, and the pool of exchangeable base cations is 10 meq m^{-2} . The runoff from the area, dominated by podzols on gneissic granitic bedrock, amounts to 0.23 m yr^{-1} , and the water is weakly buffered to pH 5.9. Yli-Knuutila is a forested catchment, with an average soil depth of 1.5 m. The soils are podzols and brown earths developed on clays and sands. The present pool of exchangeable bases is extremely large; 275 meq m^{-2} . The topology of the catchment suggests that lateral flowpaths are important. Organic acids strongly affect the buffering of the pH of the runoff to 6.0. The runoff rate is low, 0.17 m yr^{-1} , and the ion strength is 3 times that at Birkenes and Stubbetorp. More details on catchment characteristics are given in Wright et al. (10). Current deposition levels in the regions are given in Löfblad et al. (13).

RESULTS

Comparison of Models

For all 3 sites the calibrated models yield similar estimates for pre-acidification soil and water chemistry in 1840 (Fig. 3). They also produce comparatively similar hindcasts over the period 1840–1988. The models all suggest that the sites have undergone soil and water acidification over the past 140 years, and the models are in broad agreement as to the magnitude and timing of this acidification. Weathering rate is a central parameter determining catchment sensitivity and critical load. Weathering rates are notoriously difficult to determine directly. SAFE calculates weathering rates from soil mineralogy, texture and moisture. Calculated rates for Birkenes, Stubbetorp, and Yli-Knuutila are 13, 68 and 71 meq $\text{m}^{-2} \text{yr}^{-1}$, respectively. These values are comparable to those determined by MAGIC by calibration, i.e. 20, 39 and 54 meq $\text{m}^{-2} \text{yr}^{-1}$, respectively.

Of greater interest, is the ability of the models to predict future acidification of soil and water given various scenarios of future acid deposition. We evaluate 3 scenarios: (i)

Figure 3. Ca + Mg saturation at Birkenes, as calculated with all models. The present value is 13.5%. SAFE calculates Ca + Mg + K saturation, which should be compared with the present-day value 16.2%.

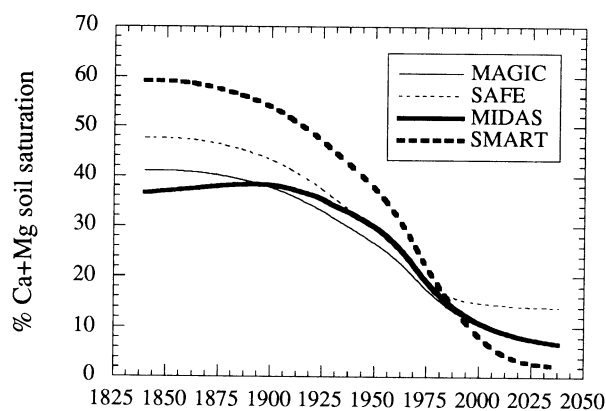
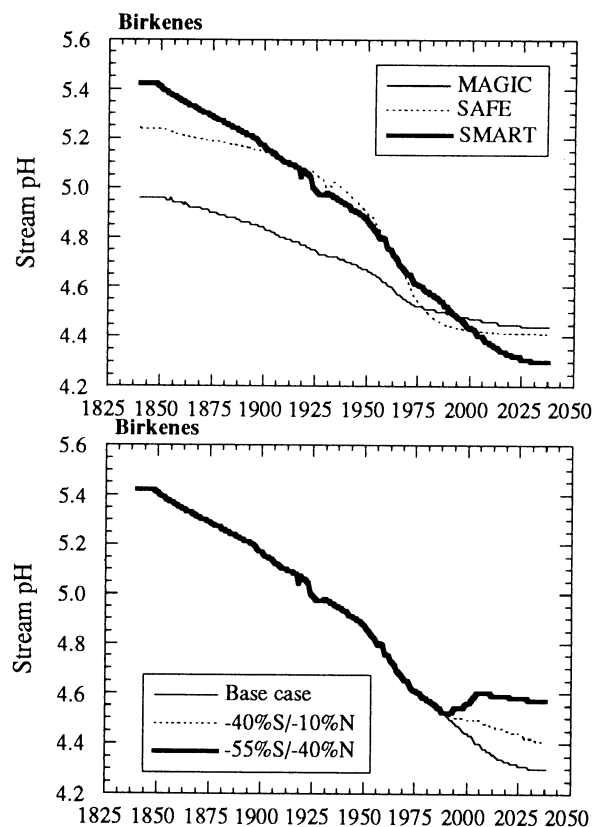


Figure 4. Upper panel shows stream pH at Birkenes, calculated with MAGIC, SAFE and SMART. Present-day volume-weighted pH is 4.5. Bottom panel shows the stream pH calculated for three different deposition scenarios with the SMART model.



a base case scenario with continued deposition at the 1987 level; (ii) reduction of the S deposition by 30% and of N deposition by 10%; and (iii) reductions of 55% S and 40% N. The scenarios were created by applying linear reductions between 1987 and 2005, thereafter keeping the deposition constant at the 2005 level. For the 3 sites, all models predict the lowest base saturation in soil and the lowest pH and alkalinity of streamwater in 2040 for the base case (Fig. 4). All models predict that Birkenes will continue to acidify, unless sulfate deposition is reduced by more than 55%, that Stubbetorp is threatened by acidification at present-day

levels of sulfate deposition, and that Yli-Knuutila can tolerate more than 50 years of acid deposition at current rates without experiencing adverse effects.

Target Loads for Aquatic Ecosystems

The MAGIC, SAFE and SMART models were used to calculate critical and target loads for the 3 sites. For surface water (14) the suggested ANC limits of 0, 20, and 50 meq m^{-3} were used since they are the chemical criteria used in Norway (0 meq m^{-3}), Finland (20 meq m^{-3}) and Sweden (50 meq m^{-3}).

Under these conditions the target load for sulfur at Birkenes for streamwater is about 40–50 meq $\text{SO}_4^{2-} \text{m}^{-2} \text{yr}^{-1}$ for ANC > 0 (as calculated by both MAGIC and SMART) (Table 3). Target loads at Stubbetorp are somewhat higher than at Birkenes, reflecting the thicker soils with higher base saturation. Again the estimates obtained by MAGIC and SMART are similar. At Yli-Knuutila the target load for sulfur calculated by MAGIC and SMART is very high, 5–100 times that of present-day deposition. SAFE gives a somewhat lower target load because it assumes that 25% of the run off comes directly from the 40 cm level in the soil, rather than passing through the entire 1.5 m soil column.

The target loads can be compared with present loads of S which are 161, 53 and 90 meq $\text{SO}_4^{2-} \text{m}^{-2} \text{yr}^{-1}$ for Birkenes, Stubbetorp and Yli-Knuutila, respectively.

Thus, at Birkenes present-day deposition is much greater than the target load, at Stubbetorp about equal to the target load, and at Yli-Knuutila present load is much less than the target load.

Target Loads for Terrestrial Ecosystems

For terrestrial ecosystems the biological indicator is the forest stand, and the criteria used to set critical loads are based on the molar ratio of nutrient cations to Al in solution, the Ca:Al ratio. The concept is based on laboratory data that suggest that elevated concentrations of Al inhibits nutrient uptake (4). The dynamic models calculate Al chemistry from the pH of the soil solution, which in turn is a function of the ANC. In MAGIC and SMART, the criteria is expressed as a the ratio between Ca^{2+} and Al^{3+} , while SAFE uses the ratio between $[\text{Ca}^{2+}] + [\text{Mg}^{2+}] + [\text{K}^+]$ and total inorganic Al. For the one-compartment models, the soil solution criterion is applied to the whole soil column, while SAFE takes the lowest value of the horizons in the root zone.

The future Ca:Al ratio depends on the deposition scenario. Calculations made with the SAFE model suggest that the forest soil threshold value is already exceeded at Stubbetorp (Fig. 5). All models provide similar results regarding the reductions

Figure 5. Trajectories of acidification and response to reduced deposition at the three sites, as calculated with SAFE. A value of one should be achieved by the year 2037.

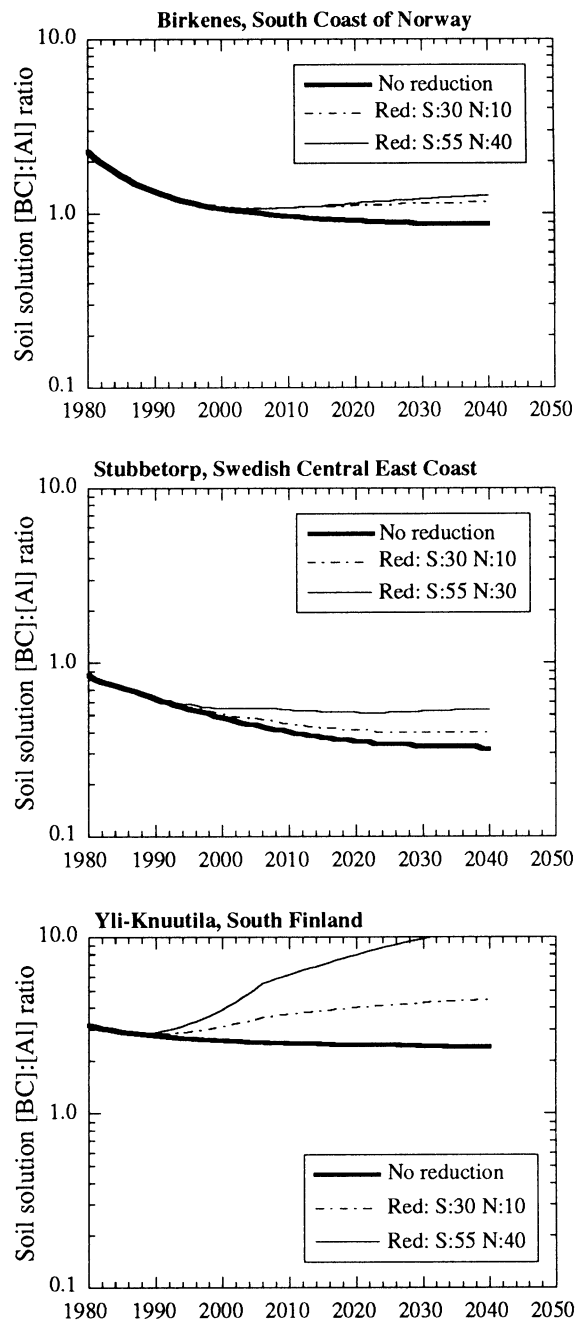


Table 3. Target load and present load for sulfur (meq $\text{SO}_4^{2-} \text{m}^{-2} \text{yr}^{-1}$) calculated by the models under the condition that the criterion is met within 50 years. Criteria are streamwater ANC meq m^{-3} .

| Site/criteria | MAGIC | SAFE | SMART | Present loading |
|---------------------|-------|------|-------|-----------------|
| Birkenes | | | | |
| ANC > 0 | 48 | 13 | 44 | 161 |
| ANC > 20 | < 0 | < 0 | < 0 | |
| ANC > 50 | < 0 | < 0 | < 0 | |
| Stubbetorp | | | | |
| ANC > 0 | 72 | 46 | 78 | 53 |
| ANC > 20 | 60 | 39 | 70 | |
| ANC > 50 | 30 | 26 | 47 | |
| Yli-Knuutila | | | | |
| ANC > 0 | 1720 | 247 | 820 | 90 |
| ANC > 20 | 1570 | 241 | 450 | |
| ANC > 50 | 1300 | 232 | 250 | |

Table 4. Target load of S for forest soils, expressed in meq $\text{m}^{-2} \text{yr}^{-1}$, calculated with 3 different models.

| Site | MAGIC | SAFE | SMART | Present loading |
|--------------|-------|------|-------|-----------------|
| Birkenes | 161 | 145 | 142 | 161 |
| Stubbetorp | 105 | < 0 | 83 | 53 |
| Yli-Knuutila | 6600 | 300 | 5000 | 90 |

BOX

Acidification and recovery follow different paths over time and show a hysteresis effect (see Figure). The rate of response to changes in acid deposition depends on intrinsic factors such as soil depth and mineralogy, local factors such as forestry management and, of course, the history and intensity of acid deposition. Key to the time lags is the pool of exchangeable base cations in the soil. This pool is replenished by inputs from the atmosphere and from chemical weathering of soil minerals, but depleted by net uptake in vegetation and leaching to runoff. Acid deposition generally accelerates the loss of base cations to runoff and results in long-term depletion of the soil pool. Recovery of soils following reduction in acid deposition entails

replenishing this lost pool by inputs from atmospheric deposition and weathering. In many cases, the rate of depletion during acidification is higher than the rate of weathering, thus, recovery can take much longer than acidification. Dynamic process-oriented models, such as those discussed here, are used to quantify these time relationships in soil and water acidification.

Figure text. Paths of acidification and recovery illustrated by results from the MAGIC model applied to Birkenes

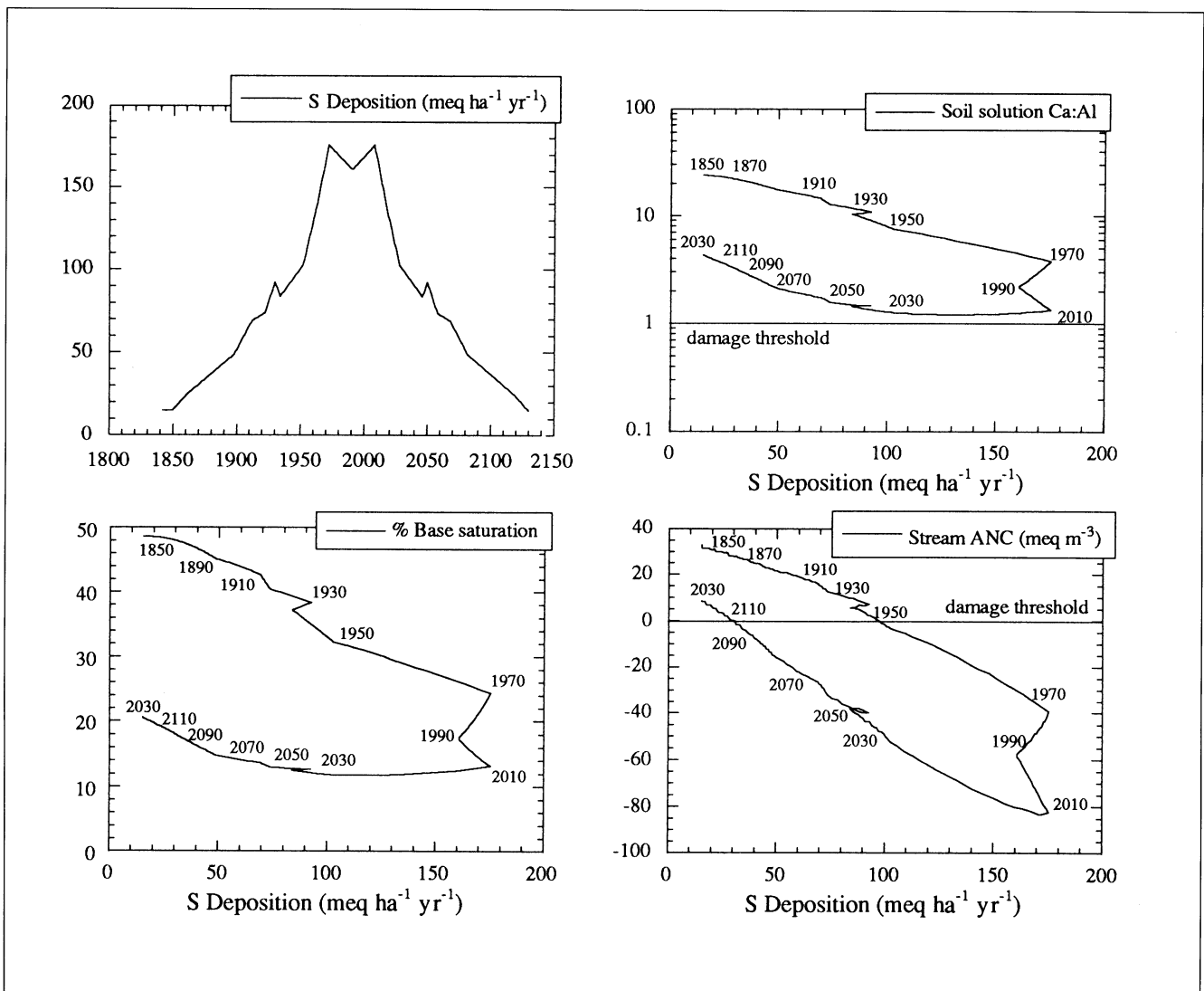
Upper panel: Estimated historical acid deposition from 1859 to 1990, and a hypothetical symmetrical decrease in deposition over the same number of years into the future.

Second panel: MAGIC indicates that

during the past 140 years acid deposition has depleted the base saturation in the soil from about 50% to about 20%. If future deposition declines along an asymmetrical path, base saturation will recover to only 22% 140 years into the future.

Third panel: The soil critical-load criterion Ca/Al declines slowly during 140 years of acidification to levels near the threshold for damage.

Bottom panel: The water critical-load criterion ANC declines from natural levels of about 40 meq m⁻³ to the threshold level of 0 in 1950 (agrees well with historical fish observations). Recovery following symmetrical decline in sulfur deposition is slower.



necessary to reverse the current trend of acidification, suggesting that the S load must be reduced by approximately 50%. Similar predictions were obtained with the MAGIC and SMART models. At Birkenes, SAFE suggests that the chemical criterion is only slightly violated, despite the fact that the system is close to a new steady state of acidification. This is due to the fact that the soil receives high inputs of nutrient cations, while accumulation in forest biomass is very small. This is in contrast to the conditions at Stubbetorp, where the nutrient uptake exceeds the external sources. At Yli-Knuutila, the terrestrial ecosystem does not appear to be threatened at present deposition levels. Here the root zone is buffered by cation exchange, and the deposition of base cations is almost twice the uptake rate.

To set target loads for S, deposition was adjusted so that the calculated Ca:Al ratios were equal to 1 by the year 2037. N deposition was assumed to be balanced by N uptake to the current level. This means that the estimated target loads for S are independent of the N deposition. It was also assumed that S could be reduced without affecting the deposition of base cations.

For Birkenes, the target loads calculated with the different models are very similar, ranging from 142 to 161 meq m⁻² yr⁻¹ (Table 4). This compares with the present S deposition of 161 meq m⁻² yr⁻¹. For the terrestrial ecosystem no reductions would be necessary, although the aquatic ecosystem should certainly benefit from reduced S deposition.

For Stubbetorp and Yli-Knuutila, the models suggest quite different target loads. MAGIC and SMART suggest that the forest soils at Stubbetorp could tolerate twice the S load it receives today, while SAFE suggests that even a total removal of S deposition would not bring the soil condition to acceptable levels. Under such circumstances, liming is necessary to meet the chemical criteria. The differences are even more pronounced for Yli-Knuutila, where MAGIC and SMART suggest target loads in the range 5000 to 6600 meq m⁻² yr⁻¹, as compared with a value of 300 meq m⁻² yr⁻¹ suggested by SAFE. For Stubbetorp, the differences do therefore lead to completely different

conclusions regarding the need for emissions reductions, while the main conclusion regarding Yli-Knuutila is, in practice, the same.

The differences in predictions are consequences of the way soil stratification is modelled. When the soil is modelled as only one compartment, the Ca:Al is taken as an average of the entire soil column. If total thickness of the soil is larger than the rooting zone, this may lead to overestimation of the base-cation concentration in the rooting zone. If the rooting zone is modelled separately, the base cations and alkalinity produced by weathering below the root zone do not influence the conditions experienced by the tree, nor the calculations for the rooting zone. With this approach, the soil column becomes acidified from the top and down. This leads to a more sudden breakthrough in acidification in the soil horizons in the lower part of the rooting zone, which is also reflected in the surface-water concentrations. The system boundaries are thus not comparable in the different modeling approaches.

Time-dependence of Target Loads

The target load depends on the length of time during which the catchment is allowed to adjust to a new loading of sulfur. If the catchment is already acidified, the adjustment time, or the time-lag associated with the target load, can be regarded as the time allowed for recovery. For the not yet acidified catchment, the time lag is the required protection time. For catchments at which present-day deposition has not pushed the stream ANC below the chemical threshold value, the longer the time horizon, the lower the target load. Some sites, such as Yli-Knuutila, may tolerate decades-to-centuries of sulfur deposition at rates well above the inherent long-term steady-state critical load due to the buffering by cation exchange. For acidified catchments, the longer the adjustment time, the higher the target load. This is because when sufficient time is allowed for recovery, the base saturation in the soils can be gradually replenished by weathering and thus the alkalinity of soil solution and

streamwater increases over time. Thus, the target load for a given site is time dependent, and also depends on the history of acid deposition at the site. We illustrate this time dependence of the target load by the MAGIC model as applied here to Birkenes (Fig. 6).

The long-term critical load for sulfur at Birkenes, where the ANC criterion is = 0, as estimated by the steady-state method, is 58 meq SO₄²⁻ m⁻² yr⁻¹. The time-dependence derived from the MAGIC model shows that to obtain ANC > 0 within 10 years the target load is only 12 meq SO₄²⁻ m⁻² yr⁻¹, whereas if one can wait 50 years the target load can be 41 meq SO₄²⁻ m⁻² yr⁻¹ and for 100 years 45 meq SO₄²⁻ m⁻² yr⁻¹. The target load asymptotically approaches the critical load 58 meq SO₄²⁻ m⁻² yr⁻¹.

If, however, the starting point is not the present-day acidified state, but the pre-acidification condition prevailing in, say 1845, the situation is different. But in the pre-acidification condition Birkenes could tolerate much higher loadings, at least for several decades. The MAGIC model shows that to obtain ANC > 0 within 10 years the target load is 270 meq SO₄²⁻ m⁻² yr⁻¹, whereas after 50 years of constant deposition it is 102 meq SO₄²⁻ m⁻² yr⁻¹ and after 100 years 74 meq SO₄²⁻ m⁻² yr⁻¹. The value asymptotically approaches the steady-state value of 58 meq SO₄²⁻ m⁻² yr⁻¹ and at a much faster rate than under the recovery situation (Fig. 6).

DISCUSSION

Comparison Between Target Loads for Different Types of Ecosystems

At all three sites, the target load calculated by MAGIC and SMART for the soil criterion of Ca:Al in soil solution is less stringent than that for the water criterion of alkalinity. The values for soil solution are calculated as an aggregate for the entire soil. Surface horizons of the soil are probably more sensitive and thus have lower target loads than deeper horizons. For forest soils only the uppermost soil—the rooting zone—is of relevance with respect to target loads. At Yli-Knuutila, although the soil as a whole appears well-protected, the uppermost soil horizons may be more sensitive than the streamwater. Here multi-layered acidification models such as SAFE are required to calculate the target load for forest soils.

Comparison Between Target Loads and Critical Loads

While critical loads have been established for forest soil on a regional scale, regional estimates of target loads, based on dynamic models are still lacking. Therefore, reduction targets established by many countries are based on the critical load. It is therefore of interest to evaluate how well critical loads conform with the target loads calculated using dynamic models.

In Table 5 target loads calculated with different models are compared with two estimates of critical loads. The Water Chemistry Method (2) is an empirical method to calculate the critical load of S, while PRO-

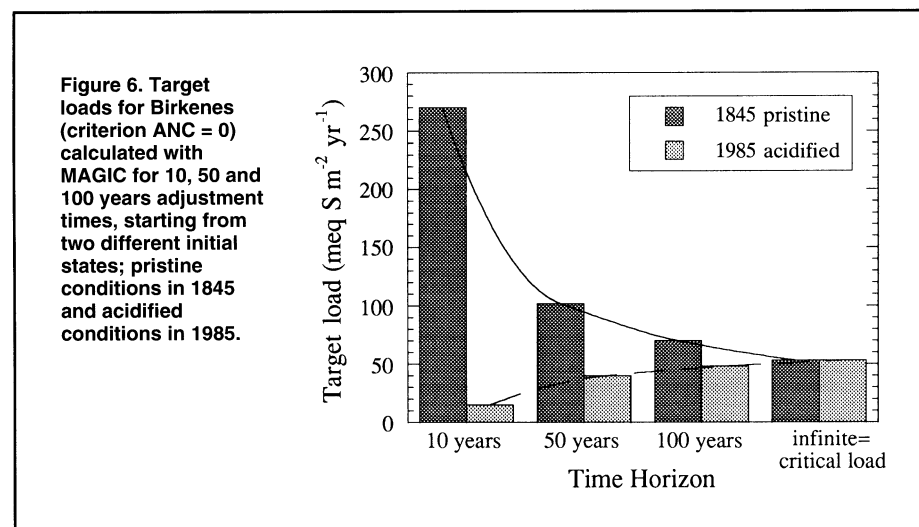


Figure 6. Target loads for Birkenes (criterion ANC = 0) calculated with MAGIC for 10, 50 and 100 years adjustment times, starting from two different initial states; pristine conditions in 1845 and acidified conditions in 1985.

Mathematical modelling indicates that increased damage due to acidification will occur within the next decades if more is not done to reduce emissions of acidifying substances. Will decision-makers react to the ring of the alarm clock and take appropriate measures? A forest stand in eastern Finland. Photo: K. Virtanen, Naturbild.



FILE is the steady-state correspondence to SAFE (3, 4). For the PROFILE model, the critical load of S is defined as:

$$CL_S = BC_W + BC_D^* + N_U - N_D - BC_U - ANC_L$$

where CL_S is the critical load of S; BC_W the weathering rate, BC_D^* is the non-marine base-cation deposition; N_U the N uptake, N_D is the N deposition, BC_U the base-cation uptake; and ANC_L the ANC leaching, calculated as the product between the runoff rate and the ANC criterion for which critical load is defined.

For Birkenes, the critical loads calculated are similar, and proximity of the MAGIC target load to the critical load is an indication that the system will recover to conditions close to steady state by the year 2037, if the load of S is reduced to $48 \text{ meq m}^{-2} \text{ yr}^{-1}$ by 2005. Since the site at present is close to equilibrium with deposition, the target load must be set lower than the critical load to meet the target ANC. This pattern conforms with the SAFE/PROFILE calculations. For Stubbetorp, which is not yet acidified below $ANC = 0$, the MAGIC (and SMART and SAFE) results differ from the Water Chemistry Method. During acidification, the target load would never have to be set to a lower value than the critical load. According to all models, the critical load calculated with the Water Chemistry Method, $183 \text{ meq m}^{-2} \text{ yr}^{-1}$, would lead to damage to the ecosystem. While the dynamic models suggest that the deposition must be reduced or not increased, the Water Chemistry Method allows the S deposition to be increased by almost a factor 3. The PROFILE model suggests a lower critical load than target load. This pattern is repeated for Yli-Knuutila, but in this case the aquatic ecosystem will be well protected

under all circumstances because of the large pool of exchangeable base cations available for buffering. From this we conclude that the Water Chemistry Method is most likely to conform with process-oriented approaches in aquatic systems that are highly acidified, and where base-cation concentrations have dropped back to pristine levels.

To make this comparison for terrestrial ecosystems, the critical load was calculated with PROFILE (15), the steady-state version of SAFE. This makes the target loads and critical loads fully comparable since the models describe identical chemistry. In Table 6, target loads and critical loads for the three sites are compared. For Birkenes, the target load is lower than the critical load. This is because the process of recovery requires that the pool of exchangeable base is partially restored, a process that consumes alkalinity and must be balanced by additionally reduced inputs. The critical load of S for Stubbetorp is approximately 0. This means that the internal load of acidity must be reduced, or alkalinity added to the system by liming. Reducing the N load will have no effect since virtually all N is immobilized in the ecosystem, and since N is

not the limiting nutrient, it does not control the acidity produced by base-cation uptake rate (16). The fundamental problem at Stubbetorp is that nutrient supply to the root zone is too small to sustain the forest stand in a longer perspective, leading to low base-cation to Al ratios in the upper soil horizons. At Yli-Knuutila all models applied suggest that a higher load of S than the present load can be tolerated. Indeed, PROFILE calculations suggest that the critical load is half the target load, and 50% larger than the present deposition rate. The critical load is lower, since the steady-state calculation does not take the buffering by cation exchange into account.

Sources of Uncertainty

Calculations using dynamic models are associated with uncertainty. This is evident from the fact that the different models yield different target loads, even although great care was taken to make the input data compatible. Uncertainties are caused by many factors. From the model comparison presented here, the differences between models are primarily caused by differences in soil

Table 5. Target loads of S surface waters calculated with the dynamic models MAGIC and SAFE, and critical load of S calculated with the empirical Water Chemistry Method (WCM) and the steady-state PROFILE model. Deposition of SO_4^{2-} is expressed in $\text{meq m}^{-2} \text{ yr}^{-1}$. The chemical criteria applied is 0 meq m^{-3} .

| Site | Target load | | Critical load | |
|--------------|-------------|------|---------------|---------|
| | MAGIC | SAFE | WCM | PROFILE |
| Birkenes | 48 | 13 | 52 | 38 |
| Stubbetorp | 72 | 46 | 189 | 22 |
| Yli-Knuutila | 1720 | 247 | ∞ | 77 |

stratification, regarding modeling of soil chemistry and whether horizontal flow paths are considered or not. Although both the one compartment approach and layered-model structures may have their advocates, it should not be forgotten that both approaches represent a one-dimensional view at the catchment scale. Methods to aggregate soil data over catchments in a representative way are therefore important (17).

One consequence of the simplifications in how the soil column itself is modelled, is that short-term variations in surface-water chemistry are not modelled. This is unsatisfactory since pH depressions during spawning periods can cause ecological damage by acidification. In this study, no model with seasonally varying hydrological flowpaths that would model short-term effects was tested. Such effects would have been especially important at Stubbetorp, where high runoff is already connected with low pH-values. The annual average pH or ANC is perhaps not the best measure of damage, if a stream community is suffering from the effects of episodic acidification.

The absence of feedback from the biological components in the terrestrial ecosystem to the soil chemistry is another source of uncertainty. If the forest ecosystem declined, an increased supply of N, nutrient cations and acidity to the soil solution would likely occur. This could affect the soil solution in many ways. The integrated effect of such changes cannot be predicted with the chemical models presented here. The inclusion of interactive growth models is therefore a natural next step in the development of models for predicting long-term changes in forest ecosystems.

Another weak point of the models is the present way of modeling the Al chemistry as a function of the H⁺ concentration alone. This is especially true for the soil calculations where no data are available for the parameters of the Al-equilibrium calculations, and the Al concentration is directly used in the chemical criteria. Simplistic modeling of organic Al complexes is an other source of error. For soils this may actually be the largest source of uncertainty in the process of calculating target loads.

Finally, there are uncertainties connected with data errors. Soil data as well as historical deposition patterns affect the calibration of the models which leads to uncertainties in initial conditions, selectivity coefficients and weathering rates.

CONCLUSIONS

The results presented here show that dynamic soil acidification models provide useful tools for estimating target loads. Several dynamic models give similar results when applied to surface waters under similar conditions. For soils, multi-layer models tend to give lower target loads than one-compartment models. When a target load is estimated, the type of ecosystem (aquatic, terrestrial), the chemical criteria (ANC, Ca:Al ratio) and chemical threshold value (ANC = 0, Ca:Al=1) as well as the time horizon should be defined. While the critical load is

an inherent characteristic, the target load is an operational value that can be set higher or lower than the critical load.

Table 7 summarizes the consequences of applying loads higher or lower than the critical load for different systems. For acidified sites, the nearer into the future the target year is, the lower the target load. Systems not yet damaged can be protected for the future, if the load is lower than the

critical load. The longer the system is to be protected, the lower the target load.

Typically, for time scales longer than 50-100 years the target loads converge to the critical loads for both presently affected and unaffected systems. This is because of the pool response time of the cation-exchange system. Catchments with very large pools of exchangeable cations (e.g. Yli-Knuutila) may represent an exception in this case.

Table 7. Summary of the consequences of setting target loads above or below the critical load. Target year refers to the year when the chemical threshold value should be met.

| | Present-day situation | |
|-----------------------------|-----------------------------|----------------------------|
| | Non-acidified | Acidified |
| Target load > Critical load | Protected until Target year | Not Protected |
| Target load < Critical load | Protected | Protected from Target year |

Table 6. Target load (TL) and critical loads (CLs) of S, expressed in meq m⁻² yr⁻¹, calculated with SAFE.

| Site | TL | CLs |
|--------------|-----|-----|
| Birkenes | 145 | 138 |
| Stubbetorp | < 0 | -5 |
| Yli-Knuutila | 300 | 130 |

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