METSÄNTUTKIMUSLAITOS
Metsäekologian tutkimusosasto

Critical Loads and Critical Limit Values

edited by

Hannu Raitio and Tuire Kilponen

Proceedings of the Finnish-Swedish Environmental Conference
October 27—28, 1994
Vaasa, Finland

Metsäntutkimuslaitoksen tiedonantoja 513
The Finnish Forest Research Institute. Research Papers 513

Critical Loads and Critical Limit Values

edited by

Hannu Raitio and Tuire Kilponen

Proceedings of the Finnish-Swedish Environmental Conference
October 27–28, 1994
Vaasa, Finland

Metsäntutkimuslaitoksen tiedonantoja 513
The Finnish Forest Research Institute. Research Papers 513

METSÄNTUTKIMUSLAITOS
Kirjasto

Contents

Foreword 4

Sirkka Hautojärvi
Opening of the Conference 6

Markku Luoma
Opening of the Conference 9

Jean-Paul Hettelingh, Peter A.M. de Smet and Robert J. Downing
The critical loads concept and critical loads in Europe 11

Michael Ashmore
Critical levels for forestry in Europe 27

Maximilian Posch
Allocation of critical loads to S and N deposition 36

Wim de Vries, Gert J. Reinds and Maximilian Posch
Assessment of the forested area in Europe at risk under various acid deposition scenarios 48

Staffan Westerlund
The concepts of critical loads and environmental quality standards 66

Kevin Bishop
Critical load models for the regional and local scale: a cautionary scientific perspective 74

Heikki Niininen
Critical load and critical limit values from the standpoint of the energy producer 77

Mats Nebaeus
Critical load and critical limit values from the stand point of the local administration 81

Reino Lammi
Critical loads and critical limit values from the standpoint of the wood and paper industry 87

Kalervo Mäkelä and Matti Perttilä
Critical loads to the Gulf of Bothnia 90

Arne Henriksen, Juha Kämäri, Maximilian Posch, Martin Forsius, Anders Wilander and Tatyana Moiseenko
Critical loads for surface waters in Scandinavia and the Kola region 97

Harald Sverdrup, Per Warfvinge and Kaj Rosén
Critical loads of acidity and nitrogen for Swedish forest ecosystems, and the relationship to soil weathering 109

Alf Björklund
Critical load and special characteristics of the Quark area of Finland 139

Håkan Staaf
Critical loads/levels as a tool for implementing national strategies for pollution control 151

Lennart Mattson
Critical load and critical limit values from the standpoint of the administration 157

Marketta Karhu
Critical loads and critical limit values from the point of view of local government district environmental authorities 162

Hannu Ilvesniemi and Michael Starr
The use of the Ca/Al ratio in the calculation of critical loads for forest soils 165

Remarks of the Moderators

Kaj Rosén
Some aspects on the concept of critical loads
A moderators conclusions 177

Pirjo-Liisa Nurmela
From the viewpoint of a government environmental officer attending a conference dealing with critical loads 181

Kari Kuusiniemi
Critical loads and critical limit values
The legal point of view 188

Foreword

The Council of the Strait of the Gulf of Bothnia (CSGB) is a regional body of co-operation under the jurisdiction of the Nordic Council of Ministers. In Finland, CSGB's functioning concentrates on the region covered by the County of Vaasa and in Sweden on the region covered by the County of Västerbotten and the municipality of Örnsköldsvik. One of the current areas of emphasis for CSGB are various environmental projects. One such project is concerned with the interaction between the state of health of spruce forests and the local conditions in the vicinity of the Strait of the Gulf of Bothnia.

The idea of arranging a Finnish-Swedish environmental conference was the result of an initiative put forward by The Council of the Strait of the Gulf of Bothnia in 1991. In the beginning of 1993 the idea crystallised in connection with the work of "Granskogens hälsotillstånd"-project, a forest project. One reason for choosing critical loads and critical limit values as the subject was that included among the goals of the forest project were to examine more closely the concept of critical loads in regard to the Strait region and to enhanceco-operation between researchers and between researchers and environmental authorities and to promote the dissemination of research results. Thus, the Finnish-Swedish environmental conference partly fulfils the objectives of the forest project.

The Organizing Committee of the conference was appointed in autumn of 1993. In addition having members from the Council of the Strait of the Gulf of Bothnia and the Finnish Forest Research Institute's Parkano Research Station, the Committee has representatives of the County of Vaasa (Finland), Vaasa Water and Environment Authority (Finland), Vaasa University (Finland), Agricultural University of Sweden and the University of Umeå (Sweden), and the County of Västerbotten (Sweden).

Critical load is an internal property of the ecosystem under assessment. It is highly geographically tied, ecosystem-related property. Different regions and different ecosystems differ in sensitivity; i.e. are capable of withstanding different loads. Defining critical loads thus presupposes the availability of precise information on the effect that deposition has on different ecosystems. The precise determining of critical loads and critical limit values helps us to focus emission-reducing actions in places where this is most urgent. Critical loads and critical limit values are, therefore, matters of importance for all concerned – researchers, authorities and people involved in the formulation of legislative measures.

The Organising Committee has selected the following themes for the seminar:

1. Critical loads of the environment and critical limit values from the point of view of aquatic and forest ecosystems.
2. State of current knowledge and research needs.
3. Critical loads and critical limit values from the poits of view of the environmental authorities, the research, the legislation and the industry.
4. Environmental legislation: legislative possibilities for controlling environmental pollution and
5. Critical loads and special characteristics of the Strait of the Gulf of Bothnia.

Altogether twenty presentations of invited speakers and nineteen posters were presented during the two conference days. The Finnish Forest Research Institute has assumed the responsibility for the publishing of the conference proceedings. The editors wish to thank the authors for the cooperation which has enabled us to publish this volume.

The conference is funded by The Council of the Strait of the Gulf of Bothnia, the Academy of Finland, Swedish National Environment Protection Board, Imatran Voima Oy and Wisaforest Oy Ab. Thanks are due to the above-mentioned organizations for their support. The main responsibility for the practical conference arrangements has been borne by the Finnish Forest Research Institute's Parkano Research Station and the Western Finland Institute for Economic Research of the Vaasa University.

Hannu Raitio

Tuire Kilponen

Opening of the Conference

Sirkka Hautojärvi

Secretary General
Ministry of the Environment
P.O. Box 399
FIN-00121 Helsinki, Finland

Ladies and Gentlemen,

It is my pleasure to be able to address to you here today in this Finnish-Swedish Conference where views on critical loads and levels will be exchanged. I am pleased to notice many outstanding speakers on the programme. This gives a promise of a very fruitful Conference here in Vaasa.

Unfortunately Ms. Pietikäinen, our minister of the Environment was unable to be present here herself due to her other obliging commitments. She would have liked to be here today. She asked me to extend her very best wishes to the Conference.

Being aware that there are far better experts in this field present here than I am, I will like to stress in my opening statement four points. Firstly the importance of international cooperation in environmental issues, because only through common efforts we can make progress in the path towards sustainable development. Secondly, we should be ambitious enough to put our good intentions into action. And thirdly, we have to recognize the need for further research and information on issues affecting sustainable development. The level of knowledge is far for being high enough. However, more and more knowledge is being gathered which gives us a reliable basis for action. My last point is that good exchange of knowledge and information is needed which gives me another opportunity to stress the importance of the occasions like the one here in Vaasa.

Transboundary air pollution has long been the most serious environmental problem in Europe. In Finland, acid rain is affecting our lakes and forests. Finland together with 26 other countries signed the second sulphur protocol in Oslo in June this year. The new sulphur protocol marked the start of the second stage of the European agreements on emission reduction. It meant a continuation of the 1985 protocol on a 30 per cent reduction in sulphur emissions, adopted in Helsinki.

It is no doubt we can say that air pollution control has made a progress during the decade after the Executive Body Meeting in Helsinki. This is apparent in both practical efforts to reduce emissions, as well as, in planning. Cooperation in research carried out since the Helsinki sulphur protocol, has made it possible to base the recent protocol on the concept of critical loads. In fact, planning for emission reductions is now based on estimates of nature's tolerance in various parts of Europe, rather than equal reduction percentages for each country. In general, the more you have knowledge, the more specific you can be.

According to the newly signed protocol, the long-term goal of the signatories is to prevent sulphur depositions from exceeding the critical loads calculated for each area by the experts. At present, critical loads are exceeded by a substantial margin. The negotiations operated on the premise that the first step towards this goal would be to reduce the excess in question by at least 60 per cent by the year 2000. The result of the negotiations binds the parties to reduce their emissions in accordance with the reduction percentages recorded in the protocol. The target mentioned above will not be fully reached. Finland, for its part, will attempt to achieve a major reduction in sulphur depositions, and in fact the emission reduction announced by Finland will be 80 per cent of the 1980 amount by the year 2000.

With respect to Finnish ecosystems, the target reduction percentages in the new protocol are not yet satisfactory. Finland expressed some disappointment with the low emission reduction targets during the negotiations, and appealed to many governments on behalf of better achievements. Still we had good reasons to sign the protocol. In many ways the protocol can be regarded as progressive. First of all, basing the protocol on critical loads marks a significant progress. Likewise, inclusion of individual emission targets in a legally binding document must be regarded as a step forward. Thirdly, the fact that an Implementation Committee was established to review the implementation of this protocol and compliance by the parties is an important achievement.

As the last point I would like to pick up an important element of the protocol. It is the possibility to implement emission reduction through joint efforts between countries. The Finnish Ministry of the Environment has granted FIM 185 million (USD 35 million) in support for investment in environmental protection and energy savings since 1991. Support has also been provided for 190 technical aid projects. The fact that the protocol makes it possible to equate such actions with emission reductions at home is important for countries like Finland, who have long used the least expensive means of reducing their own emissions.

Future acidification in Finland depends largely on efforts to reduce emissions in areas near Finland. The Baltic states did not attend the european protocol negotiations. Moreover, only a 38 per cent reduction target has been entered in the

annex to the protocol for Russian Federation. At the Geneva negotiations Russia did not want separate emission reduction figures for its western regions, close to the Finnish boarder, although it did express its readiness to make bilateral agreements with neighbouring countries.

Finland and Russia have in fact undertaken negotiations on a bilateral agreement and the experts on critical loads have already met several times. The first treaty negotiations took place in Helsinki at the beginning of May and the second meeting is expected to be in early December. The negotiators have set a bilateral framework agreement for air pollution control as their goal by the beginning of 1996 at the latest.

Finland and Estonia have signed an agreement concerning cooperation in the area of air pollution control already in July 1993. The sulphur and nitrogen emission of Estonia have reduced by 50 per cent (because they have stopped to export electricity produced in the oil shale fired power plants of Narva). There is a further need for emission reductions due to acid deposition, which exceeds the critical loads in South Eastern Finland and possibly Southern parts of Estonia.

Nordic Working Group on Air and Sea Environment has desided to continue its efforts in catalyzing the scientific and technical work needed to support the further negotiations in Geneve and also the work inside the European Union.

We need sufficient information on issues affecting sustainable development, that is, on the interdependence of nature, human well-being and economic prosperity. By producing this information, science and research can provide a sound foundation for new solutions. In the future, the aims and demands of environmental protection will increasingly be greared towards production based on the principles of sustainable development. The focal point will shift from cleaning up discharges to saving raw materials and energy resources and to preventing environmental damage.

Natural scientists now face a new, more difficult task of preparing the ground for nitrogen protocol of the second stage. Apart from the critical loads of acidic deposition, they should investigate eutrophication and the direct impact of nitrogen dioxide and ozone on vegetation and human health. This is a very complicated multi-component and multi-effect exercice, where also cost-efficiency aspect must be taken into account. Still we must be optimistic that through common efforts and firm determination we can make progress in the path towards sustainable development.

With these words, I wish you all succesfull seminar on critical loads and levels.

Opening of the Conference

Markku Luoma

Department Secretary
Vaasa Provincial State Office
P.O. Box 200
FIN-65101 Vaasa, Finland

Ladies and gentlemen,

Cooperation between the province of Vaasa and Västerbotten goes back a long way. "The Council of the Quark" was founded in 1972. (To those unfamiliar with this concept, the Quark is the narrow part of the Gulf of Bothnia). Initially our cooperation primarily focused on trade, culture and advancement of communication between our two provinces. Our cooperation in the area of environmental conservation began in earnest in 1989 with the delegation of duties between the various regional authorities and officials involved in this work. This gave rise to an efficient network of collaboration in the sphere of environmental protection.

Our first official project was a survey of the current state of the aquatic environment in the region of the Quark.

Our cooperation then progressed from individual projects to official collaboration between our two provinces and their local authorities. A steering committee was set up to coordinate this work, chaired alternately by the Heads of the Environmental Departments at the Provincial State Offices of Vaasa and Västerbotten for a term of two years at a time.

The steering committee formulated a strategy on environmental cooperation, which was then ratified by the Quark Council. Its chief aim is to provide general guidelines for environmental cooperation and to ensure that new projects are launched on a regular basis. The strategy also outlines the need and potential for future funding. Roughly 1–2 million Finnish marks are channelled into cooperative projects every year.

A project set up to establish critical loads on eutrophied lakes in the Quark region was completed in 1993. A three-year project on the impact of airborne loads on forest environments is currently in its final year.

We have considered it useful to hold seminars in conjunction with these projects. These seminars have usually been open to interested parties in the regions of Vaasa and Västerbotten.

The seminar now to be opened. "Critical loads and limits" is based on findings from earlier projects. We have nevertheless decided to broaden the scope of this seminar to enable participants from other regions to attend.

The critical loads concept and critical loads in Europe

Jean-Paul Hettelingh, Peter A.M. de Smet and Robert J. Downing

Coordination Center for Effects
National Institute of Public Health and
Environmental Protection (RIVM)
P.O. Box 1
3720 BA Bilthoven, The Netherlands

Abstract

The sensitivity of ecosystems has been included as a basis for the revision of the second sulfur protocol of the Convention on Long Range Transboundary Air Pollution of the United Nations Economic Commission of Europe. Critical loads, ie, the maximum allowable acid deposition which does not increase the probability of damage to forest soils and surface waters in Europe, have been computed and mapped. The critical load is an indicator of sustainability of ecosystems against acidification in the sense that risk of damage is reduced whenever acid deposition does not exceed the critical load. The reduction of the excess of sulfur deposition over critical loads by 60 % (60 % gap closure) has emerged as a starting point for policy negotiations leading to the second sulfur protocol. The critical load concept has thus lead to a sulfur protocol that protects about 81 %, 86 % and 90 % of terrestrial and surface water systems in 2000, 2005 and 2010 respectively. In addition, the total European area in which sulfur deposition exceeds critical loads by more than 500 eq ha^{-1} yr^{-1} will be reduced from about 19 % in 1980 to about 1 % in 2010.

Introduction

The critical load is defined as "the highest deposition of a compound that will not cause chemical changes leading to harmful effects on ecosystem structure and function" (Nilsson and Grennfelt 1988). In this paper critical loads for acidity and sulfur are emphasized, because of their importance for the recent UN/ECE sulfur reduction agreement. Critical loads for acidity and sulfur have been computed and mapped[1] in Europe (Hettelingh et al. 1991, 1992a, Downing et al. 1993) for forest soils and surface waters. The aim of the critical load approach is to achieve

[1] Critical loads have been corrected for base cation deposition, which buffers the acidifying effect of sulfur, and for base cation uptake, which increases the acidifying effect. The term *critical sulfur deposition*, denoted CD(S), is then used to characterize corrected critical loads.

protection of ecosystems by sufficiently reducing emissions of acidifying compounds such as sulphur dioxide (SO_2), nitrogen oxide (NO_x) and ammonia (NH_3). Emission reductions are considered sufficient when the resulting acidic deposition does not exceed critical loads. Critical loads have been used together with integrated models such as the Regional Acidification Information and Simulation model (RAINS) to assess a variety of emission reduction strategies in relation to abatement costs and environmental protection. Such assessments are used in negotiations to develop protocols on the reduction of sulfur dioxide and nitrogen oxides within the framework of the Convention on the Long-Range Transboundary Air Pollution (LRTAP) of the United Nations Economic Commission for Europe (UN/ECE).

In June 1994, a second protocol for reducing sulfur emissions was signed by 30 countries. The starting point for negotiating national emission reductions was to diminish by 60 % the excess of 1990 sulfur deposition over critical loads for sulfur. This scenario, which was assessed by the RAINS-model, became better known as the "60 % gap closure" scenario. It was the first time that negotiations included environmental effects, by means of the critical load concept, in addition to technical and economic considerations of emission abatements. This paper presents an overview of the critical load concept, the assessment of critical loads in Europe and their usage in the negotiation of the second sulfur protocol. The application of the critical load concept to a multitude of pollutants constitutes the final section of this paper.

Critical loads: calculation and mapping

The critical loads concept

A critical load is an indicator for sustainability of an ecosystem in the sense that it provides a value for maximum allowable load of a pollutant at which risk of damage to an ecosystem is reduced. When critical loads are exceeded, certain physical and chemical properties of an ecosystem may change to an extent where the risk of damage is increased. The chemical changes and effects induced by pollutants vary with pollutants, and the kind of exposure. For example, sulfur dioxide deposition cause one effect, ie, soil and surface water acidification. However, the deposition of nitrogen oxides and ammonia causes acidification as well as eutrophication. The critical load concept has currently been restricted to sulfur, which is the pollutant for which a UN/ECE protocol was first to be revised.

The critical load concept is a methodology according to which critical loads in Europe are used as a criterion to assess whether emission reduction strategies are sufficient, taking national cost of emission reduction into account (Hettelingh et al.

1992a). Figure 1 (dotted line) shows the objective of the iterative procedure is to avoid that sulfur deposition exceeds critical loads at the lowest possible national costs. Since reducing emissions to reach critical loads throughout Europe was found to be infeasible the objective of meeting critical loads was altered during policy negotiations. First an objective by which excess should be reduced (gap closure) is formulated. Then, the required emission reductions at minimum European costs are computed. Finally, the actual excess in different regions in Europe is expressed in terms of the extent to which the national area of ecosystems is protected. This procedure is repeated by changing the objective finally leading to the identification by groups under UN/ECE of the 60 % gap closure being an appropriate starting point of emission reduction negotiations. The 60 % gap closure strategy aims to reduce the 1990 exceedance by 60 % by 2000, which means that parts of Europe will continue to have depositions that exceed critical loads.

Since critical load exceedance may lead to "harmful effects", the question of when these harmful effects will occur becomes relevant. The answer to this question can be investigated by using dynamic soil models, which simulate the temporal changes in the balance of geochemical processes in a soil receiving acid deposition. The procedure of using dynamic models is also part of the critical load concept as illustrated in Figure 1 (solid line). The application of dynamic models is also useful to assess recovery time of an ecosystem once emissions have been sufficiently reduced. The dynamic part of the critical load concept has been initiated (Hettelingh et al. 1992b, Hettelingh and Posch 1993, Bleeker et al. 1994) but has not yet been part of the integrated assessment of emission reduction strategies. However it is expected that the dynamic part of the critical load concept will become increasingly important as indications of target years for establishing required emission reduction vary over Europe. This is especially the case as UN/ECE LRTAP negotiations move towards reductions in nitrogen oxide emissions, which have to be viewed in terms of the total acidifying effect, ie, including the agreed sulfur emission reductions.

Critical load computation

Critical loads have been mapped (Hettelingh et al. 1991, Downing et al. 1993) by the Coordination Center for Effects (CCE). Critical loads were computed by national experts using local data or by the CCE on the basis of existing European databases when national contributions were not available. The computation method is based on the Steady State Mass Balance Method (SSMB) which is extensively described elsewhere (Sverdrup et al. 1990, Hettelingh and de Vries 1991, Hettelingh et al. 1991). SSMB assumes a steady state of chemical compounds in soils and surface waters. The critical load thus becomes an indicator of sustainability of ecosystems against acidification. The ratio of base cations to aluminum, and the

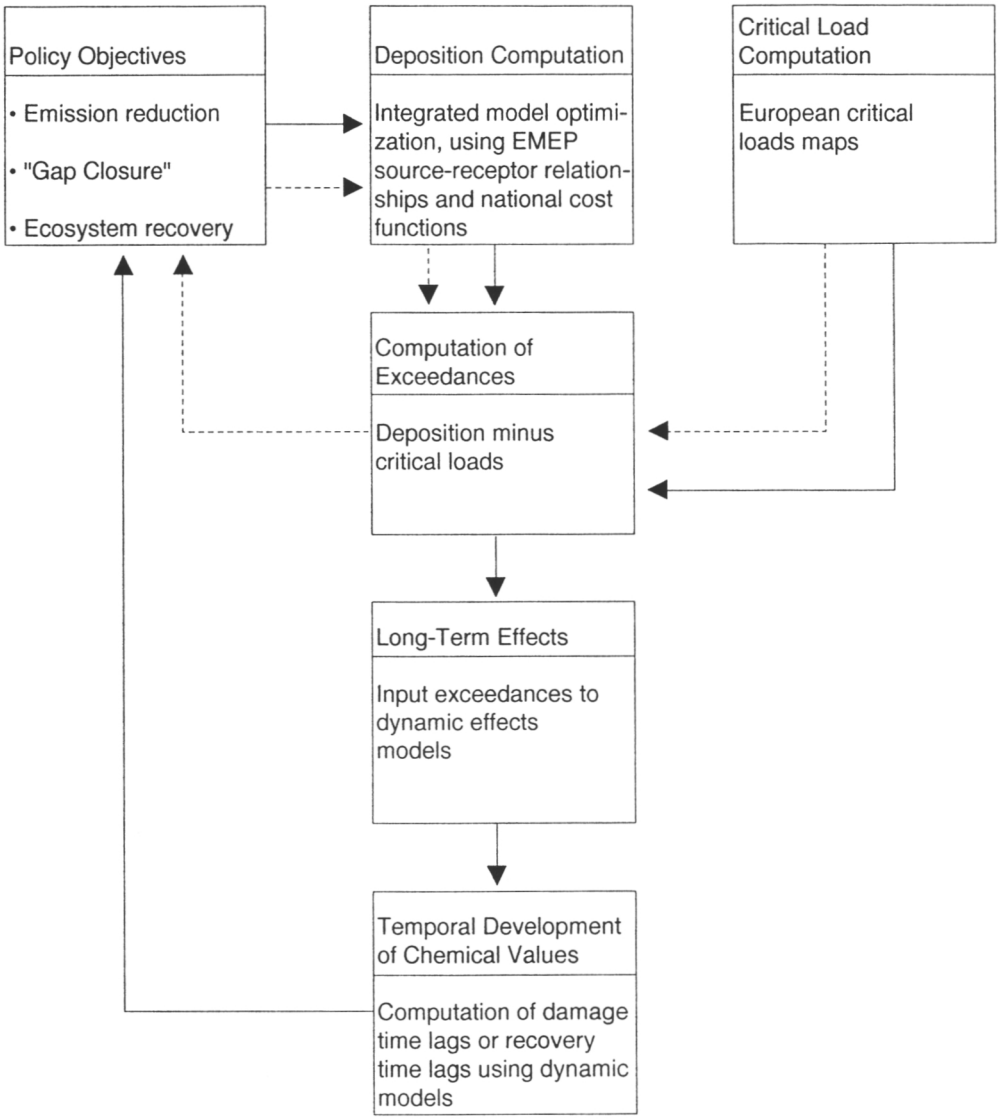


Figure 1. The critical load concept consisting of: (1) iterating required emission reductions to meet critical loads and costs (short circle; dotted line), and (2) computing time horizons before ecosystem damage (continued critical load exceedance) or recovery (whenever exceedance ceases to exist). The computation of time horizons is represented by the solid line (long circle). (Source: Downing et al. 1994, p. 2)

aluminum concentration, are used as indicators for steady state of geochemical processes. By assigning established critical values to these indicators (eg, the concentration of aluminum in soil solution should not exceed 0.2 meq/l and the base cation to aluminum ratio should not be less than one), it is possible to compute the allowable acidification for each ecosystem. The critical load for sulfur was mapped similarly. An extensive overview of critical values for the base cation to aluminum ratio for a large variety of plants and trees can be found in Sverdrup and Warfvinge (1993).

The European map of critical loads for sulfur used in designing the second sulfur protocol is shown in Figure 2.

Figure 2 shows the distribution of critical load ranges over EMEP grid cells. Each range reflects a computed critical load which, when not exceeded by sulfur deposition, results in a protection of about 95 % of the natural forests and/or surface waters. It can be seen that most sensitive ecosystems are located in Northern Europe and scattered areas in Western and Eastern Europe, where a sulfur deposition exceeding the range of 0 to 200 eq ha^{-1} yr^{-1} leads to an increased risk of damage. In Central-West and Eastern Europe large areas have natural systems exposed to risk of damage when sulfur deposition exceeds a range of 200-500 eq ha^{-1} yr^{-1}. Critical loads exceeding 1000 eq ha^{-1} yr^{-1} are found in southern European natural systems that are least sensitive to sulfur deposition in comparison to other parts of Europe.

Other protection levels may be chosen instead of the so-called 5-percentile critical load that decreases the risk of damage to 95 % of the terrestrial and surface water systems in each grid cell. Since a 150x150 km^2 EMEP grid cell may contain many ecosystems of varying sizes, it is possible to compute cumulative distributions of critical loads. The 5-percentile critical load value has been chosen for the policy analysis.

Calculating critical loads of acidity did not provide European policy makers with sufficient tools for the evaluation of required sulfur emission reductions. The reason is that UN/ECE protocols concentrate on distinctive acidifying compounds (sulphur or nitrogen), rather than on acidity as a whole. Therefore it was necessary to develop methods to apportion the critical load of acidity between the acidifying share of sulfur and the acidifying share of nitrogen. From a scientific point it is doubtful whether discrimination between sulfur and nitrogen is justified since an ecosystem is indifferent to whether acid stress is caused by sulfur or by nitrogen.

The critical load for sulfur has been derived from the critical load of acidity by assigning the share of sulfur to the ratio of sulphur to net nitrogen deposition

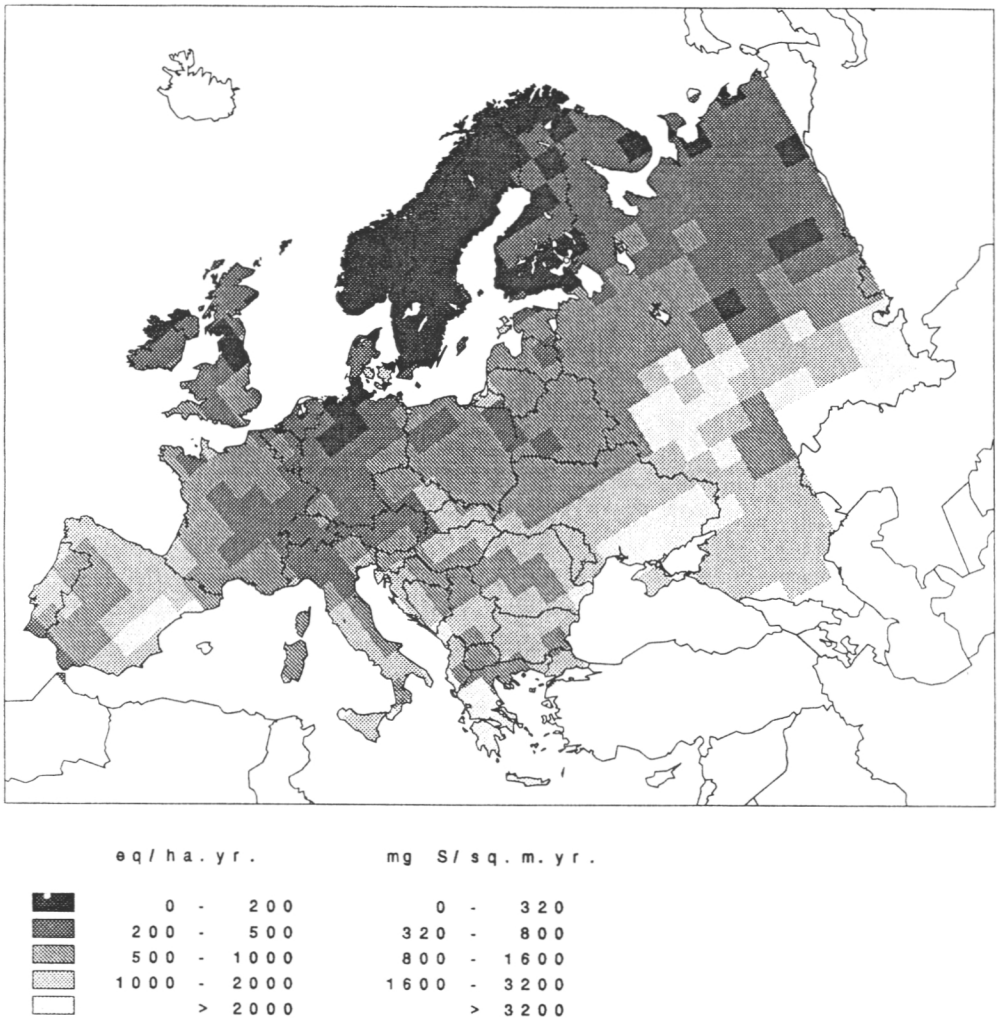


Figure 2. Ranges of sulfur critical loads in Europe after correction for base cation deposition and base cation uptake. (Source:Coordination Center for Effects, RIVM).

(Hettelingh et al. 1991, p. 40). This subdivision was obtained using the following assumptions:

(a) The share of the present sulfur deposition in total acid deposition is used as proxy of the part of the critical load of actual acidity which can be attributed to sulfur.

(b) The share of the present nitrogen deposition in total acid deposition contributes to acidification only when it is not taken up or immobilized by the ecosystem. In other words, nitrogen is acidifying when the ecosystem is unable to use nitrogen as a nutrient.

This fraction holds when the total present load of nitrogen exceeds the nitrogen uptake capacity. The sulfur fraction is assumed to be equal to unity when nitrogen uptake exceeds the total nitrogen load. In that particular case the critical load of acidity becomes equal to the critical load of sulfur. Currently, the use of the sulfur fraction has been improved by a methodology in which the excess of the critical load for acidity and nutrient nitrogen is assessed simultaneously as described in Posch et al. (1993) and elsewhere in these proceedings (Posch 1994).

The RAINS model

The Regional Acidification, INformation and Simulation model (RAINS; see also Alcamo et al. 1989) is used to assess cost-effective emission reduction alternatives. RAINS consists of: (a) an energy/emission module that computes national emissions as function of energy combustion (SO_2, NO_x) and agricultural practices (NH_3); (b) an atmospheric module computing acidic depositions in 150x150 km^2 grid cells using the EMEP transport model[2] (Tuovinen et al. 1994), and (c) an effects module comparing acidic depositions to critical loads produced by CCE. RAINS allows the computation of deposition levels given an energy/emission pattern or for optimization (Amann 1989, 1991). In the optimization mode, minimum cost emission reductions over Europe as a whole can be computed subject to any particular set of deposition levels prescribed over 150x150 km^2 cells covering Europe (EMEP). For the preparation of the second sulfur protocol the RAINS model was used to assess sulfur reduction strategies for the years 2000, 2005 and 2010 in comparison to sulfur emissions in 1990, the base year. The optimization mode was used to find cost-optimal sulfur emission reductions for example with respect to the 60 % gap closure scenario.

Results of the critical load concept

This section summarizes the results of the second sulfur protocol. The sulfur dioxide emission reductions were agreed on a country by country basis, resulting

---

[2] The European Monitoring and Evaluation Programme of the UN/ECE provides governments with information and data on depositions, concentrations, long-range transport and transboundary fluxes of air pollutants.

in a decrease of the European total from about 41 million tonnes in 1990 to 26 million tonnes in 2010. Figure 3 shows sulfur dioxide emission levels from 1980 to 2010. Emissions in 2010 are projected to be about 53 % less than 1980, the base year for the first sulfur protocol.

The result of the emission reductions is a decrease of the areas where sulfur deposition exceeds critical loads. This can be seen by comparing Figures 4, 5, and 6 which show the excess of critical loads in 1980, 1990, and 2010 respectively. Figure 4 shows an excess of more 2000 eq ha^{-1} yr^{-1} covering an area from the east of the United Kingdom, northern France, Germany, Poland, the Czech and Slovak Republics to Hungary and northern Italy. The excess of critical loads is lower in areas in the periphery of Europe. Areas of non-excess are found in the south and east of Europe.

In 1990 (Figure 5) the area of highest excess is reduced to the center of Europe. The area where no excess of sulfur deposition occurs covers larger parts of southern Europe including most of France.

The result of the second sulfur protocol is displayed in Figure 6, which shows the projected excess of sulfur deposition over critical loads for sulfur in 2010. An excess above 2000 eq ha^{-1} yr^{-1} does not occur anywhere. However, still a rather large area of Europe is subject to risk of damage with areas suffering from an excess ranging from 200 to 2000 eq ha^{-1} yr^{-1}.

The levels of ecosystem protection from 1980 to 2010 are shown in Figure 7 and in Table 1. In 1980, slightly more than half (54.0 %) of the terrestrial and surface waters systems in Europe received levels of sulfur deposition below critical loads. By 1990, the results of the first sulfur protocol are clear: now 69 percent of European ecosystems are not exceeded, and the percentage of ecosystems with the greatest exceedance (more than 2000 acid equivalents per hectare per year) has been reduced from 4.7 to 2.2 percent. Analysis of future deposition patterns using the RAINS model shows that 90 percent of ecosystems will be protected by 2010, when the second sulphur protocol is fully implemented.

Although Figures 3 and 4 show that progress has been made with respect to increasing the quality of the environment, it is necessary to remain cautious with the interpretation of the results for a number of reasons. First, these results do not include the effect of nitrogen deposition. The area subject to excess remains extensive when nitrogen is included in the analysis. Secondly, because the magnitude of an excess, however small, does not give any indication about the damage to be expected. Depending on the ability of natural systems to cope with acidity, even a small excess can still cause considerable risk of damage rather soon. The actual risk and the pace by which damage may occur can be forecasted using

dynamic models, which are increasingly being used. Finally, multiple stresses due to the interaction of different pollutants indirectly (through deposition), or directly (through air concentrations) complicates the evaluation of policies geared towards the reduction of one isolated pollutant, such as sulfur.

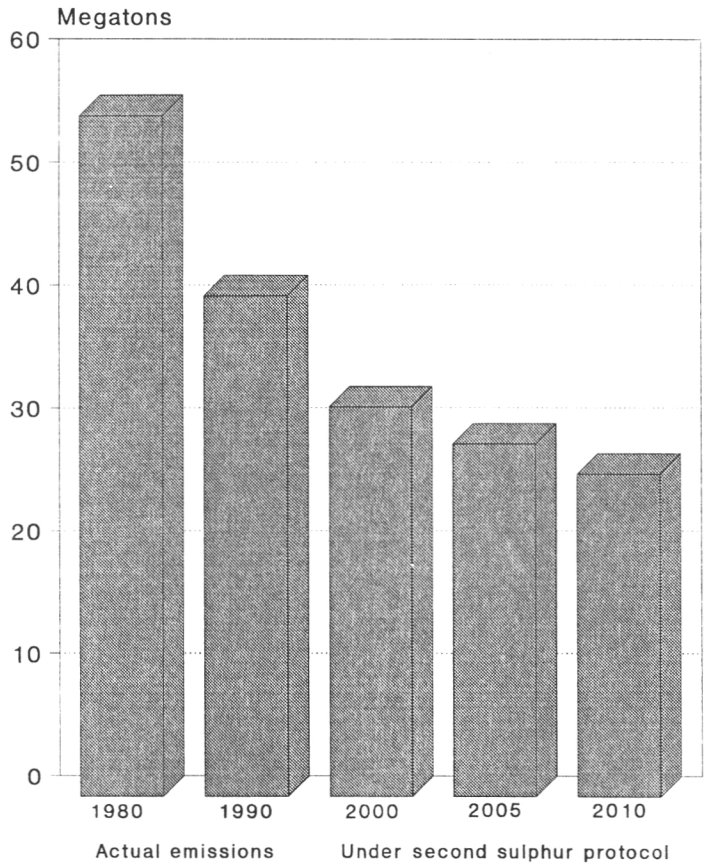


Figure 3. European emissions of sulfur dioxide as reported to UN/ECE for 1980, and 1990 and as projected according to the agreements of the second sulfur protocol.

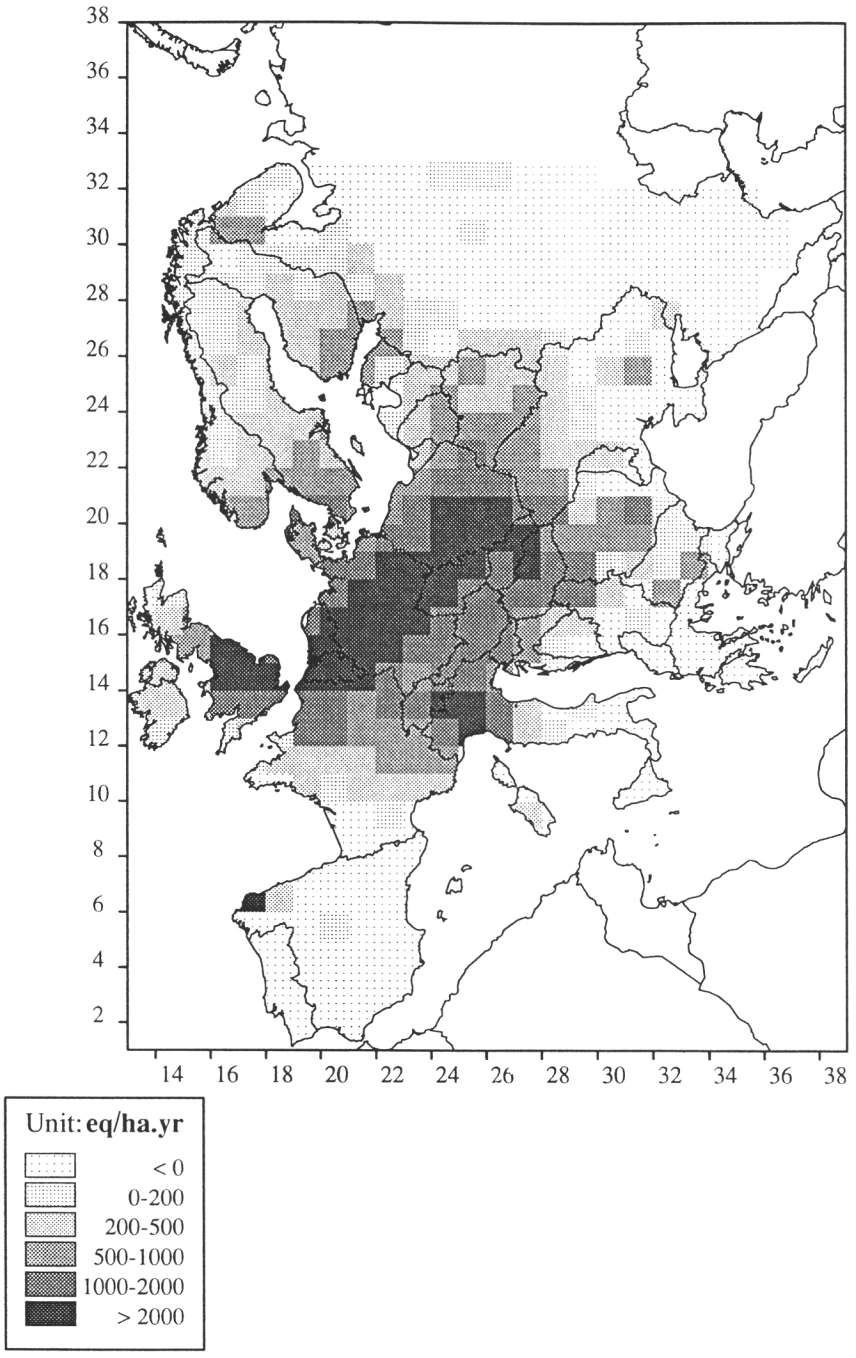


Figure 4. The excess of critical loads by sulfur deposition in 1980. (Source: Coordination Center for Effects, RIVM.)

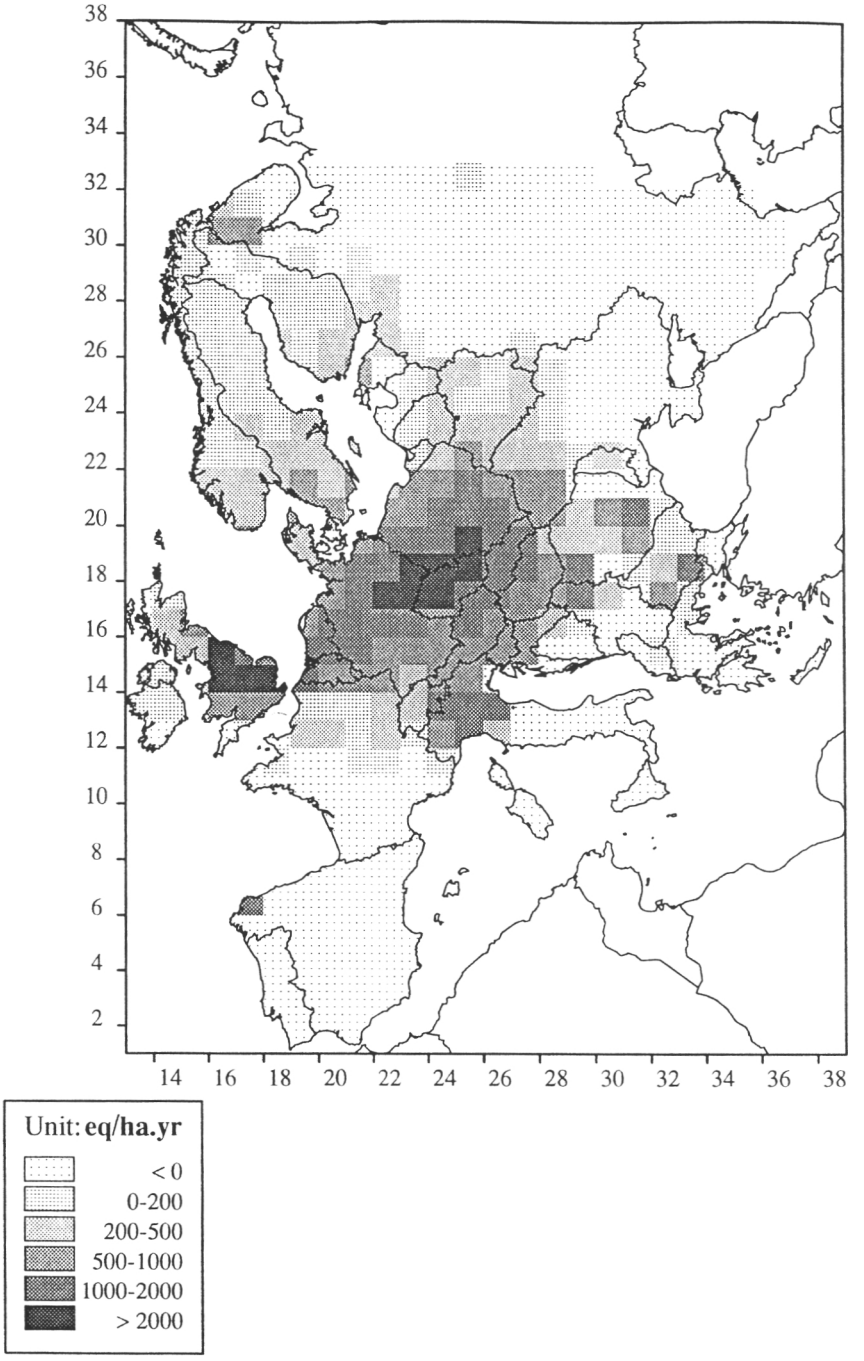


Figure 5. The excess of critical loads by sulfur deposition in 1990. (Source: Coordination Center for Effects, RIVM.)

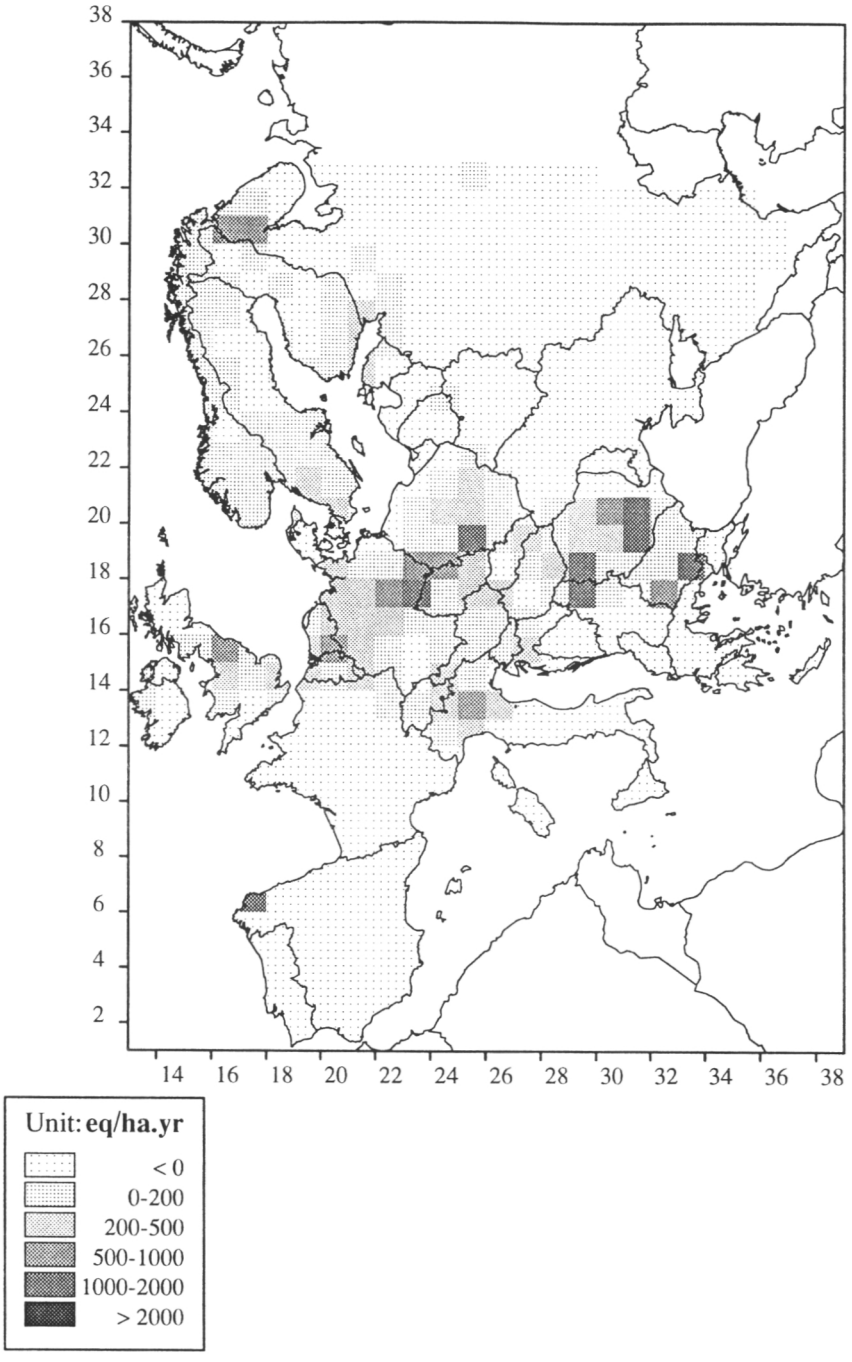


Figure 6. The excess of critical loads by sulfur deposition in 2010 according to the emission reductions agreement of the second sulfur protocol. (Source: Coordination Center for Effects, RIVM.)

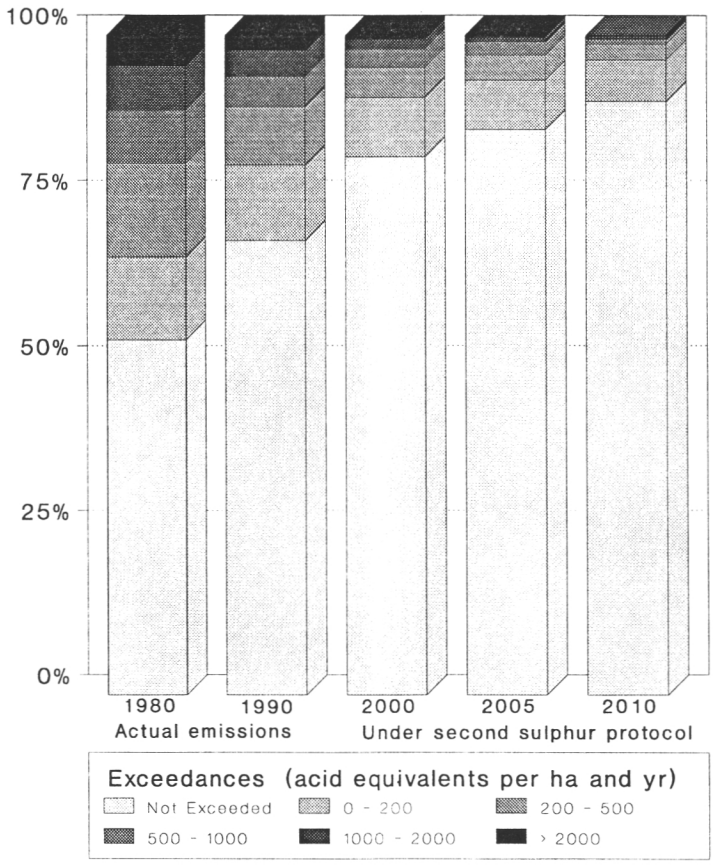


Figure 7. Ranges of the excess of sulfur deposition over critical loads for sulfur. In 1980 about 46 % of terrestrial and surface water ecosystems is subject to risk compared to about 10 % in 2010. The increase in 2000, 2005 and 2010 of protected ecosystems is according to the sulfur emission reductions as agreed in the sulfur protocol. (Source: Coordination Center for Effects, RIVM).

Table 1. Percentage of ecosystem area exceeded by critical load of sulfur.

Exceedance	1980	1990	2000	2005	2010
> 2000	4.7	2.2	0.5	0.2	0.0
1000-2000	6.7	4.1	1.6	0.7	0.5
500-1000	8.0	4.5	2.8	2.0	0.8
200-500	14.0	9.0	4.5	3.8	2.4
0-200	12.5	11.4	9.1	7.6	6.2
< 0	54.0	69.0	81.6	85.7	90.0

Final remarks and outlook

The critical load approach has successfully been applied to support European and national policies to reduce sulfur emissions. The method is now being extended to allow for trade-off between sulfur and nitrogen deposition without exceeding the critical load for acidity and eutrophication (Posch et al. 1993). It can be shown that without additional reduction of NO_x and NH_3 emissions, most ecosystems in Europe are still at risk. The reduction of sulfur emissions alone is not sufficient, as demonstrated using dynamic modeling approaches that forecast the state of acidification in case of continued excess of the critical load of acidity by both sulfur and nitrogen deposition. However, deposition alone is not necessarily the only cause of increased risk for damage. Concentrations of sulfur dioxide, nitrogen oxides and ozone have shown to cause direct damage such as yellowing of tree leaves and decrease of crop yield.

Critical levels, ie, concentrations below which no damage is expected, have been formulated for ozone (Fuhrer et al. 1993), sulfur dioxide and nitrogen oxides (UN ECE, 1993). The result is that the assessment of environmental benefits of emission reductions should include both levels and loads simultaneously. As more pollutants, each causing specific effects, are addressed in developing environmental policies, it is likely that the critical load approach as applied to sulfur will have to be extended. It becomes more appropriate to use the term *critical threshold approach*, which is an iterative procedure of emission reductions of various pollutants, with the objective of not exceeding thresholds which risk ecosystem sustainability.

In trying to establish targets to ensure sustainability of natural environmental systems, the complication of interacting pollutants and various exposure routes does not provide policy makers with simple solutions. For example, nitrogen oxide in the air may lead to an excess of critical levels for ozone through interaction with volatile organic compounds (VOCs). In addition, the reduction of nitrogen oxide emissions with the objective to meet critical loads for acidity and nutrient nitrogen will also have to be paired to policies of reducing VOCs in some parts of Europe if ozone levels are to be kept to acceptable limits. The meaning of a critical threshold approach in this particular case could be that policies are developed with the aim to avoid (a) the excess of the critical load of acidity by sulfur and nitrogen simultaneously, (b) the excess of the critical load of nutrient nitrogen by nitrogen oxides and ammonia, (c) the excess of the critical level of ozone by the interaction of nitrogen oxides with volatile organic compounds, (d) the excess of critical levels of sulfur dioxide and nitrogen oxides. Interactions with other pollutants are likely (eg, acidity with heavy metals) and complexity may increase even more as regional environmental quality is affected by global change (eg, alteration of land cover) in addition to stress caused by pollutants.

To ensure the derivation of mutually consistent critical thresholds, it is important to investigate the synergistic relationships between direct and indirect effects of pollutant concentrations and depositions respectively. Therefore, a reductionist approach towards the establishment of critical thresholds for each pollutant under consideration should be avoided as much as possible.

Continued scientific research is required to extend the methodology currently developed in treating sulfur and nitrogen simultaneously with respect to critical loads of acidity and nutrient nitrogen. This extension has to include ozone and critical levels on the short term to accommodate UN/ECE requirements. Ultimately, science may also consider relationships between acidification levels and leaching of toxics as well as interactions with climate change. Policy making frameworks, such as the LRTAP Convention are increasingly recognizing the importance of not isolating one pollutant as policy target. The preparation of the next UN/ECE protocol on reducing nitrogen, includes investigations of combined effect with ozone and ammonia. The RIVM Coordination Center for Effects can contribute to preparing further modeling and mapping methodologies to accommodate the scientific support of multi-pollutant multi-effect protocols.

References

Alcamo, J., Shaw, R. & Hordijk, L. (eds.) 1990. The RAINS model of acidification: science and strategies in Europe. Kluwer Academic Publishers, Dordrecht, the Netherlands.

Amann, M. 1989. Energy use, emissions, and abatement costs. In: Alcamo et al. 1989.

– 1991. The efficient multinational allocation of emissions reduction measures for reduction of acid deposition: Application Example for Austria. Academic Dissertation, University of Karlsruhe (in German).

Bleeker, A., Posch, M., Forsius, M. & Kämäri, J. 1994. Calibration of the SMART acidification model to selected IM-catchments. In: International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems, Annual Synoptic Report 1994. National Board of Waters and the Environment, Helsinki.

Downing, R.J., Hettelingh, J.-P., de Smet, P.A.M. (eds.) 1993. Calculation and mapping of critical loads in Europe: Status Report 1993. Coordination Center for Effects, National Institute of Public Health and Environmental Protection, Bilthoven. ISBN 90-6960-047-1.

Fuhrer, J. & Achermann, B. (eds.), 1993. Critical Levels for Ozone: a UN-ECE workshop report. Swiss Federal Research Station for Agricultural Chemistry and Environmental Hygiene, No. 16, Liebefeld-Bern, Switzerland.

Hettelingh, J.-P. & Posch, M. 1993. Critical loads and a dynamic assessment of ecosystem recovery, in: J.Grasman and G.van Straten (eds.), Predictability and nonlinear modelling in natural sciences and economics, Kluwer Academic Publishers, Dordrecht, The Netherlands.

– , Downing, R.J. & de Smet, P.A.M. 1992a. The critical load concept for the control of acidification. In: T. Schneider (ed.). Acidification research: evaluation and policy applications. Elsevier Studies in Environmental Science 50, Amsterdam, pp. 161-174.

– , Gardner, R.H. & Hordijk, L. 1992b. A Statistical Approach to the Regional Use of Critical Loads. Environ. Pollut. 77:177-183.

– & de Vries,W. 1991. Mapping Vademecum. Coordination Center for Effects, National Institute of Public Health and Environmental Protection, RIVM Rep. No. 259101002, Bilthoven, The Netherlands.

– , Downing, R.J. and de Smet, P.A.M. (eds.). 1991. Mapping Critical Loads for Europe. Coordination Center for Effects, National Institute of Public Health and Environmental Protection, Rept. 259101001, Bilthoven, The Netherlands.

Nilsson, J. & Grennfelt, P. (eds.). 1988. Critical Loads for Sulphur and Nitrogen, Report from a workshop held at Skokloster, 19-24 March 1988. Nordic Council of Ministers, Miljørapport 1988:15, Copenhagen.

Posch, M., Hettelingh, J.-P., Sverdrup, H.U., Bull, K. & de Vries, W. 1993. Guidelines for the computation and mapping of critical loads and exceedances of sulphur and nitrogen in Europe. In: Downing et al., 1993.

Posch, M. 1994. Allocation of critical acidification load to S and/or N deposition. Proceedings Finnish-Swedish environmental conference: critical load and critical limit values, Vaasa, Finland.

Sverdrup, H., de Vries, W. & Henriksen, A. 1990. Mapping Critical Loads: A Guidance Manual to Criteria, Calculation, Data Collection and Mapping, Nordic Council of Ministers, Miljørapport 1990:14, Copenhagen.

– & Warfvinge, P. 1993. Effect of soil acidification on growth of trees, grasses and herbs as expressed by the (Ca + Mg + K)/Al ratio. University of Lund Report, Lund, Sweden.

Tuovinen, J.-P., Barrett, K. & Styve, H. 1994. Transboundary Acidifying Pollution in Europe: Calculated fields and budgets 1985-93. EMEP MSC-W Report 1/94. Norwegian Meteorological Institute, Oslo.

UN ECE, 1993. Manual on Methodologies and Criteria for Mapping Critical Levels/ Loads and Geographic Areas where they are exceeded. Convention on Long-Range Transboundary Air Pollution, Task Force on Mapping, Geneva. Federal Environmental Agency, Texte 25/93, Berlin.

Critical levels for forestry in Europe

Michael Ashmore

Centre for Environmental Technology
Imperial College of Science Technology and Medicine
48 Prince's Gardens
London SW7 2PE, U.K.

Introduction

Over the past decade, there has been increased interest in the development of critical loads for forest soils to prevent long-term acidification or eutrophication. These critical loads were central to the recent negotiation of a new protocol on the reduction of sulphur emissions within the framework of the UNECE Convention on Long Range Transboundary Air Pollutuion.

Critical loads refer to the total deposition of sulphur, nitrogen or acidity; in contrast, critical levels refer to the direct effects of atmospheric pollutants on vegetation. Critical levels for vegetation were first set at a workshop held at Bad Harzburg in 1988 (UNECE 1988), where the following definition was adopted:–

"CRITICAL LEVELS means the concentrations of pollutants in the atmosphere above which direct adverse effects on receptors such as plants, ecosystems or materials, may occur according to present knowledge."

The direct adverse effects of air pollutants, especially sulphur dioxide, on forestry have long been recognised, and historically these effects have been substantial in many parts of Europe. In addition, there is increasing evidence of direct adverse effects of ozone on forests in several areas of Europe. Indeed, air quality standards to protect forestry were being promulgated, for example for sulphur dioxide by IUFRO, before the concept of critical levels was developed.

Although critical loads and levels have been developed within the context of long-range transboundary air pollutants, several of the pollutants for which critical levels have been formulated are strongly influenced by local sources, and thus their concentrations can be influenced by local pollution control measures. Thus the critical level concept and the values currently set may be of value in the management of local and regional air quality. However, care must be taken to ensure that the critical level values are relevent to the specific local situation.

This paper provides a brief overview of critical levels in the context of forestry. In considers some of the practical difficulties in setting critical level values for forests, and also the issue of what precisely is meant by an adverse effect. It then describes the development of the current critical levels values for forestry and the basis on which they have been set. Finally it briefly considers future research needs, emphasising the need to integrate the assessment of critical levels and loads.

How are critical levels set for forestry ?

The definition of the critical level given above raises a number of interesting questions which are central to defining meaningful values. Firstly, what is meant by the term 'adverse effect'? In managed arable agriculture, there is a clear economic meaning to this term. However, forestry serves a number of important functions, besides its value as an economic crop. This raises questions about how to assess adverse effects, for example, on the conservation value of forests. Often this is related to plant species of the woodland floor, to insect diversity, or to the presence of epiphytic lichens, as much as to the vitality of the trees themselves. Currently, critical levels are based on experiments or observations on the trees, but these other conservation criteria may need further consideration; indeed a lower critical level of SO_2 than that for trees has now been set to protect sensitive epiphytic lichens. Thus the values of critical levels need to be related to a specific criterion used to define the 'adverse effect'.

There are also important scientific issues in defining the term adverse effect. It is well known that air pollutants can have a range of effects on plants, ranging from subtle biochemical changes to visible injury and growth reductions. Although it is generally agreed that effects on growth offer the most robust basis for defining an adverse effect, two particular problems should be identified. Firstly, in the case of nitrogen deposition, for example, the initial effect is often a growth stimulation, i.e. a beneficial rather than an adverse effect. However, continued deposition can lead to nutrient imbalances, or an enhanced sensitivity to secondary stresses. Here, clearly, growth is not an adequate basis on which to define an adverse effect. Secondly, subtle changes in leaf structure, in translocation patterns, in enzyme activities etc. may in many circumstances be insufficient in themselves to lead to adverse effects on growth or tree vitality. However, it is now realised that pollutants may alter the sensitivity of plants to a range of biotic and abiotic stresses, often as a result of precisely these subtle changes in structure or function. Here, whether a particular physiological change is adverse or not depends on the presence of other stress factors. Hence, the definition of 'adverse effect' requires careful scientific judgement and depends on the particlar pollutant and receptor under consideration.

While the definition of an adverse effect raises conceptual difficulties, there are often much more difficult practical problems in defining critical level values. Ideally, one would have well-defined dose-response relationships over a range of concentrations, including those causing no measurable effect (Bull 1991), obtained under realistic conditions for a variety of tree species.

In practice, such relationships come from one of two sources. Firstly, dose-response relationships can be derived from experimental studies; for practical reasons, however, these can only be carried out over relatively short periods on seedlings or young trees, and there are grave difficulties in transposing their results to long-term effects on forest ecosystems.

Secondly, dose-response relationships can be derived from field observations; these can allow long-term effects on mature forests to be evaluated, usually along a transect away from a large pollution source. However, in such situations, there is frequently spatial variation in the levels of not just one, but of several pollutants, and this makes it difficult to use the data to define critical levels for individual pollutants. Furthermore, close to large point sources, the frequency of episodes of very high concentrations is much greater than at other locations with the same long-term mean concentration, and this can modify the dose-response relationships. Confounding of such field effects with climatic and other factors is also possible, while for secondary pollutants such as ozone it is difficult to find appropriate spatial gradients in concentration.

In the case of critical loads for forest soils, the values are derived on the basis of models, or mass-balance equations, which describe the chemical behaviour of the soil system. In contrast, the reliance in setting critical levels on either experimental data or field observations is a serious limitation, since it makes it difficult to extrapolate beyond those species and situations for which actual data are available. Nevertheless, progress has been made in developing mechanistic models of forest response to air pollutants, and these could play an important future role, for example in predicting mature forest responses from data on young trees, or in assessing how air pollutants may interact with other stress factors to infuence forest response (Ashmore et al. 1990).

Besides these problems, there is a lack of data on the response to air pollutants for many species. This is particularly noticeable for Mediterranean species, about which there is a dearth of information (Ashmore et al. 1990), but it is also true of many species of central and northern Europe which are of conservation or aesthetic value, rather than of economic significance. Hence any assessment of critical levels is usually based only on data for a very limited number of species, of which the most sensitive is normally used, since this will be the species showing an adverse effect at the lowest concentration. Ideally, different critical levels might be set for

different species, just as critical loads are set for different soils, but this is difficult on the basis of our limited current knowledge.

Similarly, different critical levels might be set for different soil conditions and different climates since these are known to modify dose-response relationships. In practice this is also difficult, since these factors have not been adequately investigated.

It is vital that all these uncertainties are taken into account when applying the critical levels described below in any policy context. The values have been set 'according to current knowledge' and this knowledge is very limited. It is also important to consider, when applying a critical level in a regional or local context, whether the critical level is appropriate for the particular situation and the particular policy concerns. The critical levels described below should only be taken as general guidelines and not as precise 'no effect' standards.

This discussion clearly indicates the need for further research to provide a sounder foundation for the critical levels set for forestry, and to allow the critical level concept to be developed to allow distinctions to be made between the sensitivity of different forests, based on species composition, soils, climate and other factors. More detailed reviews of these research needs can be found in the background papers for the Egham and Bern workshops at which the current critical levels were set (Ashmore and Wilson 1994, Fuhrer and Achermann 1994).

Values of critical levels

The values of critical levels for forestry first recommended at the Bad Harzburg meeting (UNECE 1988) are summarised in Table 1. These critical levels were reassessed at a further workshop at Egham in 1992 (Ashmore and Wilson 1994); for ozone, an additional workshop was held in Berne in November 1993 (Fuhrer and Achermann 1994) to provide a more substantive basis to the ideas developed at the Egham workshop. The reports of these workshops provide a detailed scientific justification for the critical levels proposed, which are summarised here.

Although short-term critical levels were set at the Bad Harzburg meeting, many of these were removed at the Egham workshop, since it was felt that effects of air pollutants on the growth and vitality of forestry, and indeed other types of vegetation, were largely determined by long-term exposure, rather than isolated incidents of moderate concentrations. Thus this discussion will focus on these long-term critical levels.

Table 1. Comparison of long-term critical levels for forestry proposed at the Bad Harzburg, Egham and Bern workshops.

Pollutant	Bad Harzburg	Egham/Bern
SO_2	20 μg m^{-3} (annual mean)	20 μg m^{-3} (annual or winter mean) 15 μg m^{-3} (annual or winter mean)*
NO_x	30 μg m^{-3} (annual mean as NO_2)	30 μg m^{-3} (annual mean as NO_x)
NH_3	100 μg m^{-3} (monthly mean)	23 μg m^{-3} (monthly mean) 8 μg m^{-3} (annual mean)
Acid Mist	-	1 μg S m^{-3} (sulphate particulate) where cloud cover 10 %
Oxone	25 nl l^{-1} (seasonal 7-hour mean)	10000 nl l^{-1}.h cumulative exposure above 40 nl l^{-1} over 6 months

* This critical level applies where annual accumulated temperature above 5 °C is less than 1000 degree days.

In the case of sulphur dioxide, the critical levels set at Bad Harzburg were based on a considerable body of field observation and experimental data from Europe. The modifications made at Egham were largely based on the known interaction between sulphur dioxide and low temperature. The precise modification of the critical level was derived from a model developed by Makela et al. (1987) based on field observations of Materna in Czecholovakia, showing greater sensitivity to sulphur dioxide at higher elevations. This is also consistent with the observation of effects of sulphur dioxide at low concentrations in Finland. The critical levels were also referenced to winter, as well as annual means, for evergreen species, in view of the known increase in sensitivity over winter.

The Egham workshop also considered effects on lichens and bryophytes, and set a new critical level of 10 μg m^{-3} to protect sensitive epiphytic lichens. Although this critical level is not relevent to forest production, it is significant in terms of protecting the conservation value of forest ecosystems.

The critical levels for sulphur dioxide can be compared with other international guidelines. The WHO have set a value of an annual mean of 30 $\mu g\ m^{-3}$ (WHO 1987), which is higher than any current critical level for forestry, but this guideline is likely to revised shortly. IUFRO in the early 1980s set two annual mean values: 50 $\mu g\ m^{-3}$ to allow full production at most sites, and 25 $\mu g\ m^{-3}$ to allow full production at extreme sites in the mountains or in boreal zones (Wentzel 1983). These values now seem a little high given current knowledge.

For nitrogen oxides and ammonia, the alterations to critical levels made at Egham were not primarily associated with effects on forestry. The critical level for ammonia is derived from data on sensitive heathland species, and it is possible that the appropriate critical level for forest species might be somewhat higher.

A new critical level for direct effects of wet deposition was introduced at the Egham workshop. This was largely based on a review of experimental data showing direct effects of acid mist on a range of physiological parameters, and on growth and visible injury, in tree species, especially conifers (Cape 1993). This important effect on forests, which are frequently exposed to high levels of deposition of polluted cloud and mist, had previously been ignored both in the critical levels and the critical loads context. The critical level, which was based on experimental data demonstrating effects on leaf surfaces, was set at 150 $\mu M\ l^{-1}$ sulphate as an annual mean. However, since long-term data on sulphate concentrations in mist or cloud are not readily available, a procedure based on defining an equivalent particulate sulphate concentration of 1 $\mu g\ S\ m^{-3}$ was developed, with the critical level only applying in areas with frequencies of cloud cover above 10 %. It should be noted that this critical level is based on mists with low base cation contents, and it is not applicable in those parts of Europe (e.g. the Mediterranean region and much of the Alps), where base cation levels in wet deposition are relatively high.

In the case of ozone, the critical levels defined at Bad Harzburg were largely based on American data, since there was very little relevent data for Europe. However, there has been a substanital accumulataion of data on ozone effects on European species over the last decade, and this has provided the basis for the new critical level now set for ozone.

The first important change, at the Egham workshop, was to develop the concept of cumulative exposure above a threshold concentration as the basis for defining the critical level; this is based on the observation that exposure patterns giving the same mean, but a higher frequency of episodes of high concentrations, caused greater impacts on vegetation. The exposure index is calculated by summing the differences between hourly mean concentrations and a threshold concnetration, only for those hours in which the threshold concentration is exceeded.

Analysis of experimental data on tree seedlings, as well as on agricultural crops, at the subsequent Bern workshop confirmed that a threshold concentration of 40 nl l^{-1} was appropriate. Calculating exposure over 6 months, including all 24 hours in the day, the critical level for forests was set at 10000 nl l^{-1}.h above 40 nl.l^{-1}, to prevent a 10 % reduction in growth. The figure was based on experiments on beech, birch and oak seedlings in experimental facilities providing conditions close to those in the field. However, it was emphasised that this was very much a provisional value, given the uncertainty in extrapolating from these short-term experiments to long-term effects on mature forests.

Critical levels and loads for forestry

Since the problem of widespread forest decline began to be identified in Europe 15 years ago, many different hypotheses have been developed to explain its causes. While these causes remain uncertain in many areas, it is now recognised that forest health is often determined by a combination of factors; these may include climatic stresses, the influence of pests and pathogens, management practices and deposition of air pollutants.

Even when considering only the effects of air pollutants, it is clear that the effects may be complex. Thus where pine is growing in the vicinity of large sources of sulphur emissions, there may be high concentrations of sulphur dioxide directly affecting the vitality of needles, episodes of acid mist directly damaging needle surfaces, and high levels of acid deposition to the soil, causing soil acidification and nutrient leaching. These sulphur emissions are often also accompanied by heavy metals, whose impact may depend in turn on soil acidity. Where high regional levels of ozone also occur, the direct effects on needles may be influenced by synergistic effects with sulphur dioxide; ozone is also known to affect root development through reductions in assimilate transport, and these effects may interact with those of acid deposition or heavy metals on the root system and its mycorrhizae.

Despite the increasing sophistication of our understanding of the impact of multiple stresses on forest ecosystems, there has been little attempt to incorporate this understanding into the critical levels/loads approach (Ashmore et al. 1990). In the case of critical levels, the potential importance of interactions between air pollutants was clearly recognised at the Egham and Bern workshops, but it was felt that the available experimental data was inadequate for these interactions to be formally incorporated into the critical levels actually set. Similarly the importance of pollutant interactions with other biotic and abiotic stresses was recognised as important, but only in the case of the interaction between sulphur dioxide and low temperatures was a modified critical level set, based on field observations.

This is clearly an important future issue in developing and refining the critical level concept. However, there has been less recognition of the importance of integrating the critical level and critical load concepts, despite the recognition of the interactions between soil acidification and direct pollutant impacts above ground. This needs to be considered not only in terms of setting critical levels and loads, in a way which incorporates their interaction, but also in terms of mapping the exceedence of existing, separately determined, critical level and load values. For example, a new sulphur protocol has been negotiated based largely on exceedence of critical loads for soils. In much of western Europe the levels of sulphur dioxide found nowadays are below those likely to damage forests but this is not the case in many areas of eastern Europe, where sulphur dioxide concentrations are much higher. By only considering critical loads and ignoring exceedence of critical levels of sulphur dixoide, the procedure used is likely to have underestimated the impact of sulphur deposition on forest health in these regions, even assuming the effects of direct sulphur dioxide damage and of soil acidification were only additive.

Ideally then, the way forward for forests would be to develop a more integrated concept of critical pollutant deposition, incorporating critical loads for sulphur and nitrogen, critical levels for relevent pollutants, and perhaps also the critical loads recently proposed for heavy metals (van den Haut 1994). Initially, a simple mapping approach might assist in identifying those forests in Europe for which the combined impact of multiple pollutant stresses, as indicated by exceedence of their critical level/load, may be important.

However, such an approach has clear limitations because it would not be based on any mechanistic understanding of forest responses to particular pollutant combinations. Thus there is a need to develop scientific tools which would allow the implications of these combined impacts for critical levels and loads to be assessed. One approach might be based around modelling of those parameters (e.g. nutrient cycling) which are central to the response of forest ecosystems to multiple stresses. While such a development will be intellectually challenging, it reflects the essential need to base the combined assessment of critical levels and loads around a proper mechanistic understanding of the response of forest ecosystems, rather than of individual trees.

Acknowledgements

Our work on critical levels is supported by the UK Department of the Environment. I thank Prof. Nigel Bell for his comments on this manuscript.

References

Ashmore, M.R., Bell, J.N.B. & Brown, I.J. 1990. Air pollution and forest ecosystems in the European Community. Air Pollution Research Report 29. Commission of the European Communities, Brussels.

Ashmore, M.R. & Wilson, R.B. (eds.) 1994. Critical Levels of Air Pollutants for Europe. Department of the Environment, London.

Bull, K.R. 1991. The critical loads/levels approach to gaseous pollutant control. Environ. Pollut. 69: 105–123.

Cape, J.N. 1993. Direct damage to vegetation casued by acid rain and polluted cloud: definition of critical levels for forest trees. Environ. Pollut. 82: 167–180.

Fuhrer, J. & Achermann, B. (eds.) 1994. Critical levels for ozone; a UN-ECE Workshop Report. FAC Report 16, Swiss Federal Research Institute for Agricultural Chemistry and Environmental Hygiene, Liebefeld-Bern.

Makela, A., Materna, J. & Schopp, W. 1987. Direct effects of sulfur on forests in Europe – a regional model of risk. Working Paper 87–57, International Institute for Applied Systems Analysis, Laxenburg, Austria.

UNECE 1988. UNECE Critical Levels Workshop Report. Bad Harzburg, FRG, March 1988.

van den Hout, K.D. 1994. The impact of atmospheric non-acidifying pollutants on the quality of European forest soils and the North Sea. VROM, Den Haag.

Wentzel, K.F. 1983. IUFRO studies on maximal SO_2 imisssions standards to protect forests. In: Ulrich, B. & Pankrath, J. (eds.) Effects of Accumulation of Air Pollutants in Forest Ecosystems, pp. 295–302. D. Reidel, Dordrecht.

WHO 1987. Air Quality Guidelines for Europe. World Health Organisation, Copenhagen.

Allocation of critical loads to S and N deposition

Maximilian Posch

Water and Environment Research Institute
P.O.Box 250, FIN-00101 Helsinki, Finland

Abstract

In this paper the historical development of the critical load concept and its use in emission reduction assessments is presented. Special emphasis is put on the methodology for aggregating large numbers of individual critical load values to arrive at a single value for a selected grid. While this has been rather straight-forward for the current method for calculating critical loads – and has been successfully used in negotiating the Second Sulphur Protocol – the newly proposed method for treating sulfur and nitrogen in conjunction also requires new procedures. The unique critical load values are replaced by a critical load function for each ecosystem, and this formulation gives considerable freedom in the allocation of S and N deposition reductions. Although several details still need clarification, the new methodology seems well suited for assisting future negotiations on a multiple pollutant – multiple effect protocol.

Introduction

The signing of the Second Sulphur Protocol in June 1994 constitutes an important step in the history of emission reduction agreements. For the first time ecological concepts have been taken into account in formulating the goals of emission reductions in quantitative terms in a binding international agreement. In this new Protocol the sensitivity of an ecosystem (forest soil or lake) is measured by the so-called 'critical load', i.e. the amount of (sulfur) deposition it can stand without detrimental effects in the long run. The concept of a critical load came up in the early 1980s, and – sponsored by the Nordic Council of Ministers – a series of workshops was organized to clarify the concept, to devise methods for calculating critical loads and to discuss procedures to map them (Nilsson 1986, Nilsson and Grennfelt 1988). In 1988 a 'Task Force on Mapping' was initiated under the UN/ECE Convention on Long Range Transboundary Air Pollution. In a workshop held in Bad Harzburg (Germany) in 1989 the methods for mapping critical levels and loads were documented in a manual (Sverdrup et al. 1990, UBA 1993). In 1990 a 'Coordination Center for Effects' (CCE) was established in Bilthoven (The Netherlands) with the task to synthesize the different national critical load

calculations and to produce standardized European critical load maps which are used within the ECE in the negotiations on emission reduction protocols.

Originally, most of the effort has been directed to produce critical load maps of sulfur in order to assist negotiations for the Second Sulphur Protocol (Hettelingh et al. 1991), but in the near future N critical load maps will be required for the forthcoming negotiations of a new nitrogen protocol. In anticipation of that need workshops were held in Lökeberg (Sweden) in 1992 (Grennfelt and Thörnelöf 1992) and recently in Grange-over-Sands (England) which dealt with both the acidifying and nutrient aspects of nitrogen. The workshop reports and new guidelines published by the CCE (Downing et al. 1993) will serve as background documents for future activities in the field.

In this paper we first shortly present how the critical load of sulfur is calculated for an ecosystem with the present method and how a single critical load value is obtained for a grid cell containing a (large) number of ecosystems. Secondly, we describe the new method of calculating critical loads for both S and N (the critical load function) and ways for using these functions to assess emission reduction requirements. We conclude the paper with a discussion of the methods and an outlook on future developments.

Critical load of sulfur (current method)

Critical loads for ecosystems can be derived in various ways, either by empirical methods or by models with different degrees of complexity. In the case of forest ecosystems not only direct effects play a (poorly understood) role, but also soil-mediated effects, and to date mostly soil chemical criteria (e.g. a critical Al concentration or Al/Ca ratio) have been used to derive critical loads with simple steady-state models. The largest uncertainty in these calculations is the relationship between the critical chemical values and the harmful effects (De Vries 1993).

Steady-state soil models calculate deposition levels which avoid the violation of soil chemical criteria in a steady-state situation. Therefore effects with a finite time scale, such as cation exchange and sulfate adsorption, are not included. The standard model is the so-called steady-state mass balance model, which has been used to produce maps of critical loads of acidity, S (and N) on a European scale. The critical deposition of sulfur, *CD(S)*, has been defined as:[1]

---

[1] We here use the term critical *deposition* of S, since the term critical *load* of S is – for historical reasons – reserved for the quantity $CL(S) = S_f \cdot CL(A)$.

$$CD(S) = S_f \cdot (CL(A) + BC_{dep} - BC_u) \qquad (1)$$

where BC_{dep} and BC_u are the deposition and uptake of base cations, resp., and $CL(A)$ is the critical load of (actual) acidity, the 'ancestor' of all critical loads. It is defined as the difference of the acid neutralizing capacity (ANC) produced by weathering and the ANC consumed by the maximum acceptable alkalinity leaching. The factor S_f in Eq.1 is the so-called sulfur-fraction, defined by

$$S_f = \begin{cases} \dfrac{S_{dep}}{S_{dep} + (N_{dep} - N_u - N_i)} & \text{if } N_{dep} > N_u + N_i \\ 1 & \text{otherwise} \end{cases} \qquad (2)$$

where S_{dep} and N_{dep} are the deposition of S and N, and N_u and N_i are the uptake and immobilization of N, resp.

The above formulation has been used by many European countries to compute critical loads of sulfur. In Finland, input data for 1450 lakes and about 3100 forest soil sites have been collected and critical loads calculated. This large number of critical load values calls for appropriate methods of presenting them; and since not only the values as such are of interest, but also their (approximate) geographical location, maps seem to be the obvious way to display the critical loads. However, it is impossible to print the critical load values for thousands of locations on a map. Consequently, the data have to be grouped into a small number of classes (intervals) and these classes are then plotted as different symbols and/or colors on a map. The advantage of such a display is that the geographical location of the sampling sites can be read from the map; the disadvantages are that data points are overlapping and information is lost by the small number of classes (grey shades or color codes), especially when limited to black and white displays (see, e.g., Posch et al. 1993a).

For presenting spatially varying information given by a large number of point data cumulative distributions have proven useful. The region under consideration is divided into regularly shaped subregions (grids) or irregularly shaped polygons, following some administrative or natural boundaries. All critical load values falling into a subregion (grid) are used to define a cumulative distribution function (cdf). For n values $x_1 \leq .. \leq x_n$ in a given subregion this cdf is defined by:

$$F(x) = \begin{cases} 0 & \text{for } x<x_1 \\ w_1+\ldots+w_k & \text{for } x_k \le x<x_{k+1},\ k=1,\ldots,n-1 \\ 1 & \text{for } x>x_n \end{cases} \quad (3)$$

where w_i is the relative weight (size) of ecosystem i ($w_1+\ldots+w_n=1$). The size of a subregion has to be chosen large enough to make the cdf 'representative' for the critical loads in that subregion, i.e. adding more (randomly sampled) values of the subregion should not significantly change the cdf. Using cdfs, the only information lost is the location of the ecosystem within the subregion. Another advantage of displaying data points as cdfs is that the number of data points used is virtually unlimited. It is also easy to combine data sets for the same subregion, e.g. critical loads for lakes and forest soils, by properly re-scaling the weights.

In Fig. 1 the critical loads of sulfur for lakes are displayed as cdfs in the so-called EMEP-grid, a coordinate system with grid cells of 150km×150km used by EMEP for modeling the long-range transport of sulfur and nitrogen in Europe. The same grid is also used by the CCE for producing harmonized critical load maps for Europe.

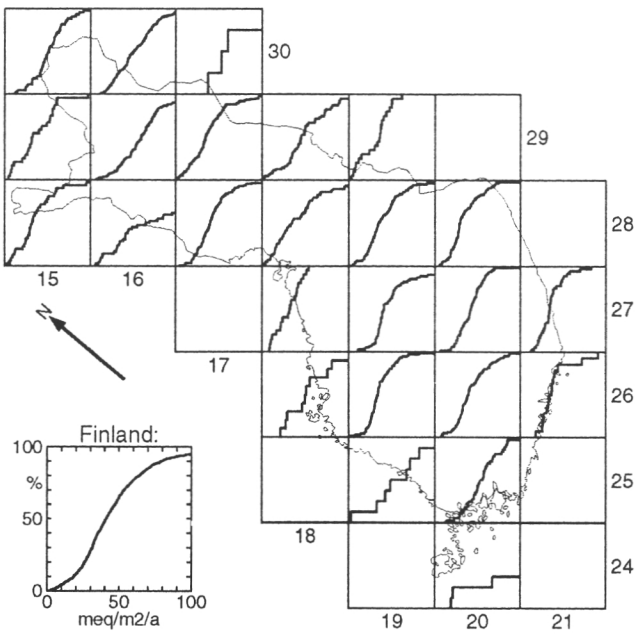


Figure 1. Critical loads of sulfur for lakes, displayed as cumulative distribution functions in each of the EMEP grid cells covering Finland.

Cumulative distribution functions, however, are not easily interpreted by the non-technical user, and therefore a simpler form of display has to be selected. Normally a single statistical descriptor of the distribution is chosen (percentile, mean, etc.) and presented as shaded (colored) grids in different grid systems (small grids for national applications and larger grids in the European context). The mean would be easy to compute, however, it is not a good descriptor for skewed distributions. In addition, the notion of a critical load demands to characterize the lower end of the cdf, i.e. the most sensitive ecosystems. This can be done by selecting a (low) percentile of the cdf. The q-th quantile of a cdf F, denoted by x_q, is the value for which $F(x_q)=q$, i.e. x_q, as a function of q, is the inverse of F; and the p-th percentile is defined as the $p/100$-th quantile. There is no unique way to calculate the percentile of a cdf defined for a finite number of values. We have selected the simplest method, illustrated in Fig. 2; other methods first smooth the cdf before calculating a quantile.

In the UN/ECE mapping exercise the 5th percentile has been chosen to represent the critical load for a grid, i.e. the aim is to protect 95 % of the ecosystems. In Fig. 3 the 5th percentile of the critical loads of sulfur for lakes is displayed in three grid systems: The grid system in Fig. 3a (½°longitude by ¼°latitude) has been chosen for displaying critical loads in Finland, Fig. 3b shows the 5th percentile critical loads in the EMEP-grid used in the European context, and Fig. 3c shows the same critical loads in the so-called NILU-grid, a 3×3-subdivision of the EMEP-grid – a grid which might be used in the future both by EMEP for the deposition modeling and the CCE for mapping critical loads.

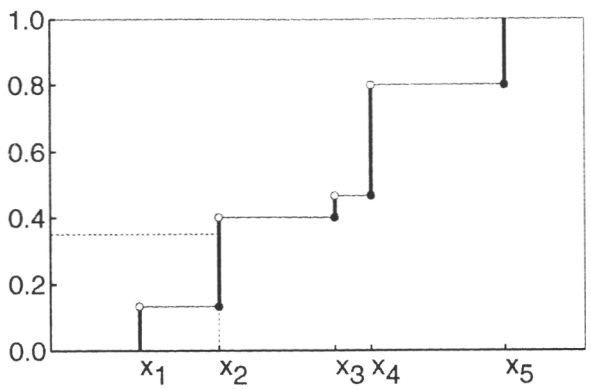


Figure 2. Example of a cdf from 5 values with non-equal weights (thin horizontal lines). The thick vertical lines define the quantile function, and the quantile for q=0.35 is indicated by the dotted line.

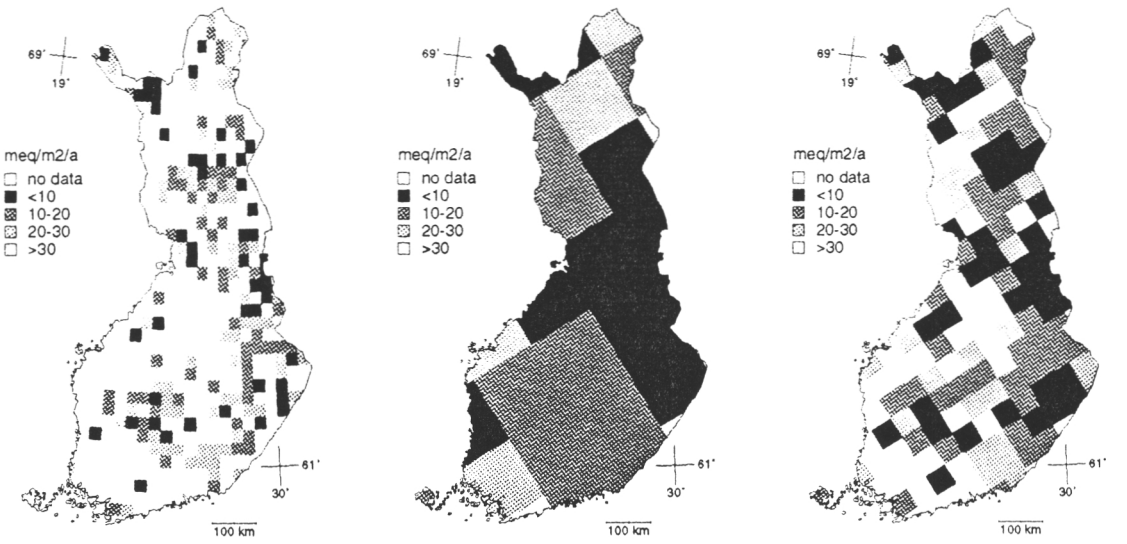


Figure 3. The 5th percentile of the critical loads of sulfur for lakes displayed in three different grid systems covering Finland.

The exceedance of the critical load of sulfur, i.e. the amount by which deposition has to be reduced in order to avoid 'harmful effects' for the ecosystem, is given by

$$Ex(S) = S_{dep} - CD(S) \tag{4}$$

and it is this quantity which enters the models for determining (cost) optimal sulfur emission reduction strategies.

As mentioned in the Introduction, the critical loads (critical depositions) computed by the above model and mapped on a European scale have been successfully used in negotiating the new sulfur protocol. However, progress in the field itself and the need for updating the Sofia Protocol on nitrogen emissions makes it necessary to incorporate N (and possibly pollutants connected with it, such as VOCs) into the critical load formulation, and this will be outlined in the next section.

Critical loads of sulfur and nitrogen (proposed method)

The concept of a critical load rests on the assumption that it is possible to derive a unique number, depending on ecosystem properties alone. However, the sulfur fraction (see Eqs.1, 2) makes the critical load of sulfur dependent on the deposition of sulfur and nitrogen; and consequently *CD(S)* should be re-calculated, whenever the deposition changes (see Posch et al. 1993b and Kämäri et al. 1993 for a discussion). Furthermore, future negotiations will concentrate on nitrogen emission reductions and – although N has been considered in the current method – critical loads of N will have to be derived which take into account not only the acidifying but also the eutrophying effects of N-deposition (and eventually the role of N as a precursor of tropospheric ozone).

One starts from the acidity balance, which can be written as (see Posch et al. 1993c):

$$Ex(A) = \left(S_{dep} + N_{dep} - BC_{dep}\right) + \left(BC_u - N_u - N_i - N_{de}\right) - CL(A) \qquad (5)$$

where now also denitrification, N_{de}, is included. A new aspect arises with N_{de} as denitrification is not a fixed quantity for an ecosystem, but depends on the net input of N. The simplest formulation is given by:

$$N_{de} = f_{de} \cdot \left(N_{dep} - N_u - N_i\right) \qquad (6)$$

where f_{de} $(0 \le f_{de} \le 1)$ is the so-called denitrification fraction of an ecosystem. If the acidity balance (Eq.5) is negative or zero, i.e. $Ex(A) \le 0$, for a given *pair* of deposition (S_{dep}, N_{dep}) we say critical loads are not exceeded. Note, that there are no unique values of S_{dep} and N_{dep} for which the exceedance becomes zero, i.e. unique critical load values cannot be defined. Furthermore, for

$$N_{dep} \le N_u + N_i = CL_{min}(N) \qquad (7a)$$

the maximum allowable deposition for S is given by

$$CL_{max}(S) = BC_{dep} - BC_u + CL(A); \tag{7b}$$

and the maximum 'harmless' deposition of N (for $S_{dep}=0$) is given by

$$CL_{max}(N) = N_u + N_i + \frac{CL_{max}(S)}{1-f_{de}}. \tag{7c}$$

In addition to acidification, also the effects of N deposition on the nutrient status (eutrophication) of an ecosystem should be considered when determining critical loads and exceedances. The exceedance is in this case derived from the mass balance of N and is given by

$$Ex_{nut}(N) = N_{dep} - CL_{nut}(N) = N_{dep} - N_u - N_i - \frac{N_{l,crit}}{1-f_{de}} \tag{8}$$

where $N_{l,crit}$ is a critical nitrogen leaching. Two cases are possible: $CL_{nut}(N) \geq CL_{max}(N)$, in this case the nutrient critical load is not binding, or $CL_{nut}(N) < CL_{max}(N)$, in which case $CL_{nut}(N)$ determines the maximum allowable N deposition. These two cases are illustrated in Fig. 4: The thick line separates the deposition values which cause no 'harmful effects' from those which require reduction. This function is termed the *critical load function* of the ecosystem.

As mentioned above, no unique critical loads of S and N can be defined, and consequently no unique exceedance of a critical load. This is best illustrated by an example (see Fig.4): Let the point E1 denote the current deposition of S and N; reducing N_{dep} substantially, one reaches the point Z1 and therefore non-exceedance without reducing S; on the other hand one can reach non-exceedance by only reducing S_{dep} till reaching Z2; finally, with a smaller reduction of both S_{dep} and N_{dep} one can reach non-exceedance as well (e.g. point Z3). In practice external factors, such as the costs of emission reductions, will determine which path will be followed to reach zero exceedance.

In order to be useful for the analysis of emission reduction strategies, also the new method requires an aggregation procedure for deriving a single critical load function for every grid cell. In the current method the cumulative distribution function of all critical load values within a grid is constructed and then a percentile, i.e. a protection percentage, is chosen as the critical load for that grid. For the newly

proposed method an analoguous procedure can be developed by considering all critical load functions within one grid cell. For every pair of N and S deposition we can then check, whether it lies inside or outside the given critical load functions, thus determining the percentage of protected ecosystems. Connecting those points in the (S_{dep}, N_{dep})-plane, which protect a prescribed percentage p of ecosystems, yields the *(100-p)*-percentile function. This single function, e.g. the 5th percentile function, replaces the hundreds or even thousands of individual critical load functions, in the same way as a percentile was used as a surrogate for the numerous critical load values within a grid in the current method. The calculated percentile function for each grid could then be linked to models which (cost)optimize emission reductions for both sulfur and nitrogen. Both the costs of the individual abatement strategies and the shape of the percentile functions would determine the optimal reduction strategy and thus the allocation of S and N deposition.

If, however, future negotiations are not to involve sulfur deposition – and this is very likely considering that a sulfur protocol has just been signed – the critical load functions can still serve as a valuable tool for determining critical loads of nitrogen for a *fixed* S deposition: By intersecting the critical load function for each ecosystem with the (present or any future) S_{dep} yields critical loads for nitrogen for the given S deposition, $CL(N|S_{dep})$ (see Fig. 5). And these critical nitrogen loads can then be used in the 'traditional way' (see current method), with the additional benefit that they are compatible with the current or anticipated sulfur deposition.

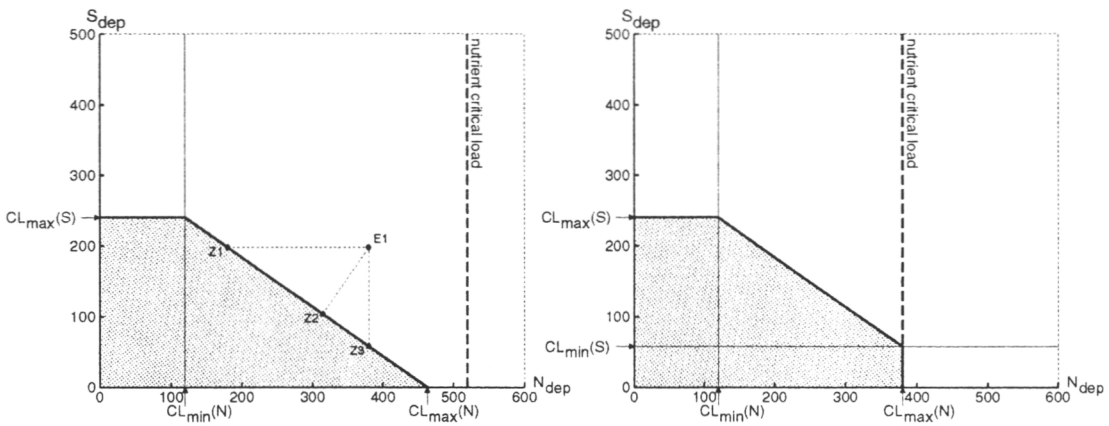


Figure 4. Relationship between sulfur and nitrogen deposition and exceedance of acidity and nutrient nitrogen for a hypothetical ecosystem. The grey areas mark those deposition values causing no exceedance.

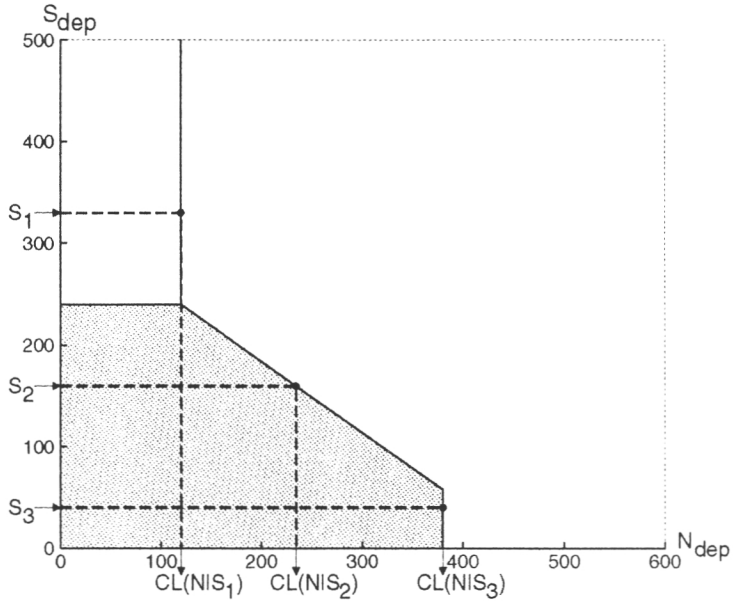


Figure 5. Three examples of how to compute the critical load of nitrogen for given S depositions, $CL(N|S_{dep})$. For $S_{dep}=S_1$, the maximum critical load of S is exceeded and the resulting critical load for N, $CL(N|S_1)$, is given by the minimum critical N load, for the other two deposition values the critical N load is obatined by intersecting the S deposition with the critical load function (dashed lines).

When aggregating the numerous critical loads (resp. critical load functions) of a grid to a single percentile (resp. percentile function) much information is lost, but this is inevitable when dealing with heterogeneous data sets from many countries. On a national level, however, it is easier to make use of the full data set, and in the following we give an example on how one can use the proposed methodology for gaining insight into the relative importance of S and N deposition reductions. In Fig. 6 the percentage of lakes which are exceeded for every possible percent deposition reduction – starting from the present (1990) deposition of S and N – is displayed for the lakes in Finland and Norway. Fig. 6a clearly shows that sulfur reductions (in relative terms) are more effective than reductions in N deposition. For example, 95 % of the lakes can be protected by a 50 % reduction in S deposition together with a 30 % reduction in N deposition. In order to achieve the same protection level with a 40 % S reduction requires a N deposition reduction of 75 %. In Norway, however, S and N deposition reductions are equally effective (in relative terms), as can be inferred from the slope of the isolines (Fig. 6b).

This type of display offers the possibility for a quick assessment, both for comparing *different scenarios* (by replacing the present deposition with the scenario deposition) and for comparing *different regions* with respect to the relative importance of S and N deposition.

Concluding remarks

In this paper the development of the critical load concept has been briefly sketched. While in the past critical loads could be expressed simply as numbers, the newly proposed method characterizes an ecosystem by a function which takes into account both deposition dependent processes and the link between sulfur and nitrogen deposition. Although the original simplicity has been lost (which seems to be inevitable when research advances in a field, and more processes are included in the description), the new method allows the simultaneous evaluation of sulfur and nitrogen emission reduction strategies. The work in this area is far from complete and many details need further clarification, but the proposed methodology is general enough to allow the inclusion of other pollutants and effects and thus provides a useful framework for multiple pollutant – multiple effect assessments.

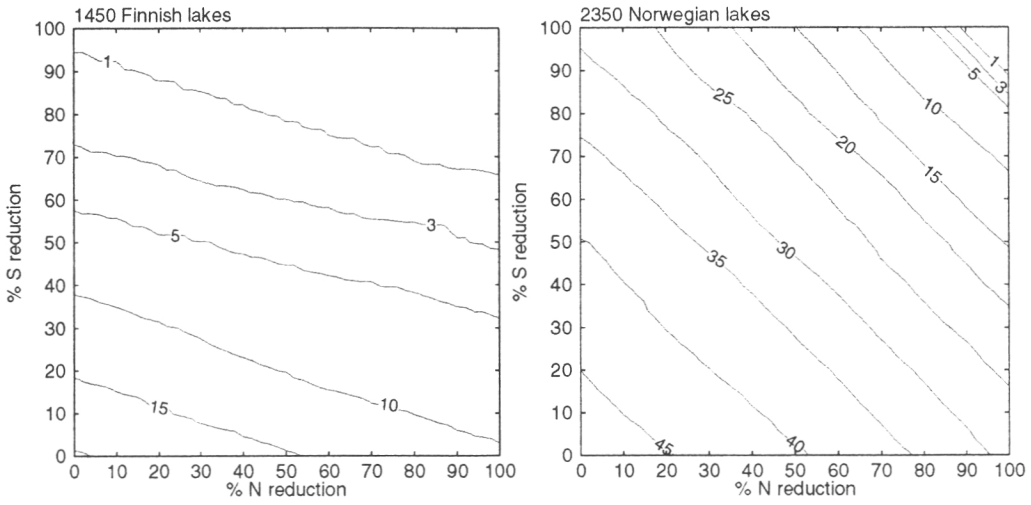


Figure 6. Isolines of the percentage of the exceedance of (a) 1450 Finnish and (b) 2305 Norwegian lakes for every combination of percent reductions from present (1990) S and N deposition.

References

De Vries, W. 1993. Average critical loads for nitrogen and sulfur and its use in acidification abatement policy in the Netherlands. Water Air Soil Pollut. 68: 399-434.

Downing, R.J., Hettelingh, J-P. & de Smet, P.A.M. (eds.). 1993. Calculation and Mapping of Critical Loads in Europe, Status Report 1993, Coordination Center for Effects, National Institute of Public Health and Environmental Protection (RIVM), Bilthoven, The Netherlands.

Grennfelt, P. & Thörnelöf, E. (eds.). 1992. Critical Loads for Nitrogen. Nord 1992:41, Nordic Council of Ministers, Copenhagen, Denmark.

Hettelingh, J-P., Downing, R.J. & de Smet, P.A.M. (eds.). 1991. Mapping Critical Loads for Europe. CCE Technical Report No.1, RIVM, Bilthoven, The Netherlands.

Kämäri, J., Forsius, M. & Posch, M. 1993. Critical loads of sulfur and nitrogen for lakes II: Regional extent and variability in Finland. Water Air Soil Pollut. 66: 77-96.

Nilsson, J. (ed.). 1986. Critical Loads for Nitrogen and Sulphur. Miljörapport 1986:11, Nordic Council of Ministers, Copenhagen, Denmark.

– & Grennfelt, P. (eds.). 1988. Critical Loads for Sulphur and Nitrogen. Nord 1988:97, Nordic Council of Ministers, Copenhagen, Denmark.

Posch, M., Kämäri, J., Johansson, M. & Forsius, M. 1993a. Displaying inter- and intra-regional variability of large-scale survey results. Environmetrics 4: 341-352.

– , Forsius, M. & Kämäri, J. 1993b. Critical loads of sulfur and nitrogen for lakes I: Model description and estimation of uncertainty. Water Air Soil Pollut. 66: 173-192.

– , Hettelingh, J-P., Sverdrup, H.U., Bull, K. & de Vries, W. 1993c. Guidelines for the computation and mapping of critical loads and exceedances of sulphur and nitrogen in Europe. In: Downing, R.J., Hettelingh, J-P. and de Smet, P.A.M. (eds.) Calculation and Mapping of Critical Loads in Europe, Status Report 1993, Coordination Center for Effects, RIVM, Bilthoven, The Netherlands, pp. 25-38.

Sverdrup, H., de Vries, W. & Henriksen, A. 1990. Mapping Critical Loads. Nord 1990:98, Nordic Council of Ministers, Copenhagen, Denmark.

UBA, 1993. Mapping Critical Levels/Loads. Texte 25/93, Umweltbundesamt, Berlin, Germany.

Assessment of the forested area in Europe at risk under various acid deposition scenarios

Wim de Vries[1)], Gert J. Reinds[1)] and Maximilian Posch[2)]

[1)] DLO Winand Staring Centre for Integrated Land Soil and Water Research (SC-DLO), P.O. Box 125 NL-6700 AC Wageningen, The Netherlands
[2)] Water and Environment Research Institute P.O. Box 250, FIN-00101 Helsinki, Finland

Abstract

Critical loads for N, S and total acidity, and amounts by which they are exceeded by present atmospheric loads, were derived for forests in Europe using a steady-state soil model. Furthermore, the chemical response of European forest soils to three emission-deposition scenarios for the years 1960–2050, i.e. official energy pathways (OEP), current reduction plans (CRP) and maximum feasible reductions (MFR), was evaluated with a dynamic soil model. Calculations were made for coniferous and deciduous forests on 80 soil types occurring on the FAO soil map of Europe, using a grid of 1.0° longitude x 0.5° latitude. Results indicated that present acid loads exceed critical values in approximately 45 % of the forested area. The area exceeding critical loads was nearly equal for N (50 %) and S (52 %). However, the maximum exceedances were much higher for S than for N. The area with nitrogen saturated soils, i.e. soils with elevated NO_3 concentrations, will increase in the future for all scenarios. The area with acidified soils, i.e. soils with a high Al concentration and a low pH, was predicted to increase for the OEP and CRP scenario and to decrease for the MFR scenario. Despite uncertainties involved, it is clear that large exceedances in critical N and acid loads occur in Western and Central Europe. This coincides with the area where a decrease in forest vitality has been reported. Furthermore, the forested area at risk is likely to increase when current reduction plans are implemented.

Introduction

The vitality of forests has become a subject of wide public and political discussion since the beginning of the eighties due to the extensive forest damage observed in rural areas of Central Europe (Schütt et al. 1983; Lammel 1984). Since that time, forest vitality characteristics, i.e. needle loss and needle discoloration, are monitored throughout Europe. Acid deposition affects the vitality of forest ecosytems both by direct effects on the forest

canopy and by indirect soil mediated effects on the roots. A recent literature review by Roberts et al. (1989) suggests that direct effects of gaseous air pollutants, such as SO_2, NO_x and O_3, play only a minor role in the declining vitality of forests in Europe. The indirect effects caused by soil acidification and N accumulation are likely to be more important. They concluded that spruce decline in Central Europe mainly results from foliar Mg deficiency. This deficiency is likely to be due to (i) an increased Mg demand due to faster tree growth caused by an increased N input and (ii) a decreased Mg uptake due to damage to roots and mycorrhizae, caused by the mobilization of Al in the soil in response to S and N inputs. Still, the relationship between forest vitality and air pollution is not clear due to a large number of natural stress factors such as unsuitable soil conditions, seasalt deposition, drought, frost, pests and fungal diseases (e.g. Heij et al. 1991). This multi-stress theory implies that critical desposition levels (loads) for N, S and acidity on forests cannot be derived in an empirical way. However, numerous studies, both in the laboratory and in the field, have shown that nutrient imbalances resulting from high concentrations of Al and N (NH_4) as compared to (divalent) base cations (BC) such as Ca and Mg, have a negative influence on root elongation and root uptake (cf. Heij et al. 1991 and De Vries, 1991 for summarizing overviews). Such unfavorable conditions (sometimes called predisposing stress) do increase the vulnerability of forest stands to frost and drought and to insect and pathogen damage (sometimes called triggering stress). In this context, critical values for NO_3 and Al can be used to derive critical deposition loads for N and S indirectly using a steady state model.

In a steady-state model dynamic processes, such as N immobilization and cation exchange, are ignored. However, the response of soil conditions to acid deposition is strongly influenced by these processes. For example, in well-buffered soils, with a large pool of exchangeable base cations, it may take decades or even centuries before measurable changes in the Al/BC ratio occur, even with high atmospheric deposition, whereas it may take less than a decade in very sensitive soils (De Vries et al. 1989). Using a dynamic soil acidification model, areas can be mapped where critical chemical values defined for forest soils (e.g. Al/BC ratio, pH, Al and NO_3 concentration) are exceeded in the course of time. This paper provides an overview of :(i) the variation in critical loads for European forest soils and the amounts by which they are exceeded, using the steady-state model START, and (ii) the change in forested areas in Europe at risk under various deposition scenarios, using the dynamic model SMART (Simulation Model for Acidification's Regional Trends). For given scenarios of S, N and BC deposition, the simulated temporal development of the nitrogen and acidification status of the soil, as expressed by the NO_3 concentration and

C/N ratio (nitrogen status) and the Al concentration and Al/BC ratio (acidification status), is presented. Results are presented in graphs, showing (i) critical loads and their exceedance and (ii) the temporal development of the forested area in Europe where critical values (Table I) are exceeded, and on maps showing the geographical pattern of critical loads, nitrogen saturation and soil acidification. Uncertainties in the predictions are also discussed.

Methods and data

The models START and SMART

START is the steady-state version of the model SMART described in De Vries et al. (1989). Both models calculate the soil solution concentration at the bottom of the rootzone either dynamically or at steady-state, for a given deposition rate. The rooting zone is represented as one homogeneous compartment. The START model is based on the assumption that dynamic processes such as cation exchange, adsorption/desorption of SO_4 and NH_4 and mineralization/immobilization dynamics of N, S and BC are unimportant for the assessment of a long-term critical load. The SMART model, however, includes these processes to assess long-term impacts of deposition scenarios. Both START and SMART consist of a set of mass balance equations, describing the soil input-output relationships, and a set of equations describing the rate-limited and equilibrium soil processes (Table II). All major ions have been included except for Na, K and Cl which were assumed to balance each other in the soil solution.

Table I. Critical chemical values used for various parameters describing the nitrogen and acidification status of a soil.

Criteria	Unit	Critical value[1)]	Effect above/below critical values
NO_3 concentration	$mol_c\ m^{-3}$	0.1	Vegetation changes
C/N ratio	$g\ g^{-1}$	40[2)]	NO_3 leaching
Al concentration	$mol_c\ m^{-3}$	0.2	Damage to root system
Al/BC ratio	$mol\ mol^{-1}$	1.0	Inhibition of BC uptake

[1)] More information on the background of these values is given in De Vries (1991).
[2)] For calcareous soils a value of 20 was used.

Table II. Processes and process descriptions included in START and SMART.

Process	Element	Process description
Rate-limited reactions:		
Growth uptake	BC[1], NH_4, NO_3	Constant growth
N immobilization	NH_4, NO_3	Proportional to N deposition[2]
Nitrification	NH_4, NO_3	Proportional to net NH_4 input
Denitrification	NO_3	Proportional to net NO_3 input
Silicate weathering	Al, BC	Zero order reaction
Equilibrium reactions:		
Dissociation/association[3]	HCO_3	CO_2 equilibrium equation
Carbonate weathering	BC	Carbonate equilibrium equation
Al hydroxide weathering	Al	Gibbsite equilibrium equation
Cation exchange[4]	H[5], Al, BC	Gaines Thomas equations

[1] BC stands for divalent base cations.
[2] In START, a fixed long-term N immobilisation rate is used, whereas SMART calculates N immobilisation dynamically.
[3] Carbonate weathering is not explicitly included in START. For calcareous soils, an extremly high weathering rate is used.
[4] Cation exchange is only included in the SMART model.
[5] Implicitly, H is affected by all processes. This is accounted for by the charge balance.

START and SMART are relatively simple one-layer models since the general lack of input data prevents the application of relatively complex multi-layer models on a European scale. For a more thorough discussion on the aspect of model complexity versus regional applicability we refer to De Vries (1990) and Van Oene and De Vries (1994). Justifications for the various assumptions and simplifications have been given by De Vries et al. (1989) and De Vries (1993). An overview of the basic process descriptions used in the START and SMART model is given in De Vries et al. (1994 a; START) and Posch et al. (1993; SMART).

Calculation of critical acid loads

Acid loads or loads of potential acidity were defined as the sum of S and N loads that are not counteracted by seasalt corrected base cation loads. The

term potential acidity is used here since NH_3 is considered as a potential acid. In the atmosphere, NH_3 may act as a base by the formation of NH_4, but in the soil the transformation of NH_4 to NO_3 causes a net acidification by NH_3. This approach is based on the assumption that NH_4 leaching is negligible. In the following, we will use the term acid load instead of load of potential acidity.

Critical acid loads can be derived with START as described in De Vries et al (1994a) In that study nitrification was assumed to be complete, and a critical acid load was derived as:

$$CL(Ac_{pot}) = BC_{we} - BC_{gu} + N_{gu} + N_{de} + N_{im}(crit) + Ac_{le}(crit) \quad (1)$$

Where *we* stands for weathering *gu* for growth uptake, *de* for denitrification, *im* for immobilisation and *le* for leaching. A critical acid load is thus derived by requiring that it does not exceed the net input of base cations plus the net uptake, reduction and critical immobilization of N plus a critical leaching rate of acidity. This so called steady-state mass balance equation (Sverdrup et al. 1990) has been widely used to calculate critical acid loads on forest soils in Europe on the basis of critical values for the Al concentration and/or Al/BC ratio (Hettelingh et al. 1991).

Independent from acidification, an upper limit is set on the N deposition by the eutrophication aspect according to (Schulze et al. 1989, De Vries 1993):

$$CL(N) = N_{gu} + N_{de} + N_{im}(crit) + NO_{3,le}(crit) \quad (2)$$

where CL(N) is the critical N load and $NO_{3,le}(crit)$ is a critical level of NO_3 leaching related to eutrophication. The model is based on the assumption that any N input above net N uptake by forest growth, denitrification and a critical rate of N immobilization and NO_3-N leaching will finally lead to unacceptable high N contents in soil organic matter, thus causing vegetation changes in forests. The determination of a separate critical N load is important for distinguishing between emission reductions needed for N and S.

Application methodology and input data

Deposition fluxes vary strongly in space. Consequently, uniform deposition areas were defined by a grid net of 1.0° longitude versus 0.5° latitude. A

higher resolution was not considered worthwhile in view of the uncertainty in the calculated atmospheric deposition fluxes. Within each gridcell critical load and dynamic calculations were made for all major combinations of tree species and soil types. A distinction was made between coniferous and deciduous trees to account for differences in dry deposition, transpiration and growth uptake. The spatial variability of the soil was taken into account by distinguishing 80 different soil types according to the FAO-UNESCO Soil Map of the World (FAO, 1981) on the basis of the dominant soil unit, texture class and slope class (De Vries et al. 1994 a, b).

Input data for both START and SMART include system inputs and outputs, i.e. deposition, precipitation, evapotranspiration and growth uptake. These data were derived as a function of location (gridcell) and of forest type, as given in De Vries et al. (1994 a). START only needs data on the present input of SO_4, NO_3, NH_4 and base cations, whereas SMART requires information on emission-deposition scenarios for these ions. Present and future emission and deposition estimates of SO_4, NO_3, NH_4 for the European countries were taken from Alcamo et al (1990) and Lövblad et al. (1992). The consequences for the acidification status of the European forest soils were evaluated for (i) the "Official Energy Pathways" scenario (OEP), based on governments projections for future energy use, (ii) the "Current Reduction Plans" scenario (CRP) which takes into account likely reductions of emission due to proposed abatement measures, and (iii) the "Maximum Feasible Reductions" scenario (MFR), which assesses the impacts of a radical, but technologically feasible, decrease in emissions (mostly in SO_2). The resulting total emissions for Europe in the period 1960−2050 for SO_2, NO_x and NH_3 are shown in Figure 1. The deposition of base cations (sea salt-corrected) was derived from measurements at 65 EMEP stations in Europe (Pedersen et al. 1990). No estimates for future base cation deposition were available and therefore it was assumed constant throughout the modelling period (1960−2050).

Soil data were derived as a function of soil type irrespective of the location. Most soil data were related to readily available soil (and land) characteristics, such as texture and organic matter, using so-called transfer functions. Soil data in both the START and SMART model are the denitrification fraction, the base cation weathering rate, and equilibrium constants for the dissociation of CO_2 and the dissolution of Al hydroxide. Additional soil data in SMART are bulk density, the initial C/N ratio, the carbonate and Al-hydroxide content of the mineral soil, the contents of BC, Al and H at the adsorption complex and cation exchange contants. Detailed information on the various soil data used is given in De Vries et al. (1994b).

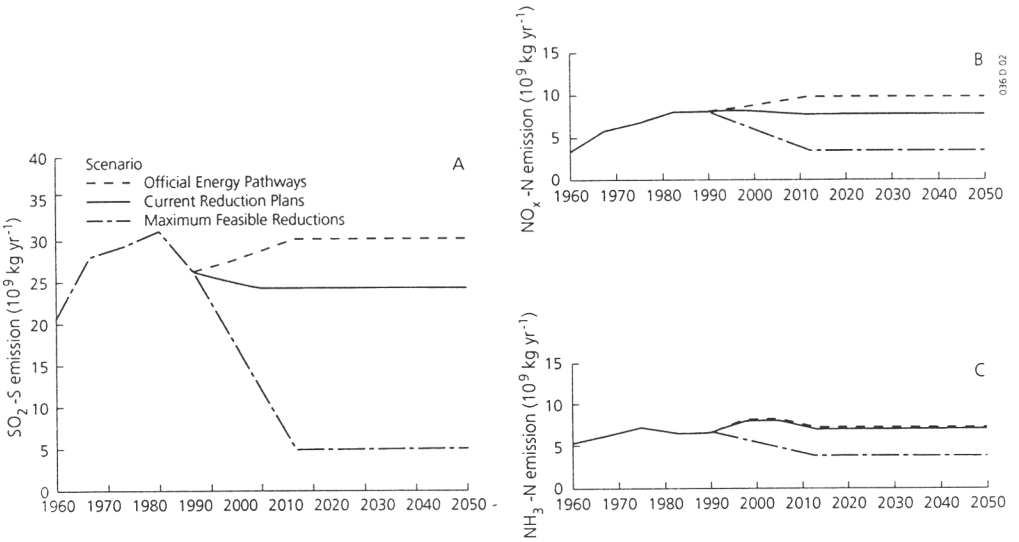


Figure 1. Trends in total emissions of SO_2, NO_x and NH_3 in Europe for the three scenarios used in this study (OEP=Official Energy Pathways; CRP=Current Reduction Plans; MFR=Maximum Feasible Reductions).

Results

Forested area exceeding critical loads for nitrogen and acidity

Results of the excess of present loads over critical loads for N and acidity on coniferous and deciduous forests in Europe are shown in Figure 2 as inverse cumulative frequency distributions, that directly give insight in the forested area above a given excess load. In relative large areas in Europe present loads exceeded critical loads but the difference was mostly less than 3000 mol_c ha^{-1} yr^{-1}. For N, the excess was much higher on coniferous than on deciduous forests (Fig. 2A), due to both higher present loads and lower critical loads. For S the difference was much smaller (not shown) since both present and critical S loads did not deviate that much for both forest types. The forested area exceeding critical loads was nearly equal for N and S, i.e. 50 % for and 52 % for S. For total acidity, the forested area exceeding critical loads equalled 45 % (52 % of all conifers and 33 % of all deciduous forests; Fig. 2B).

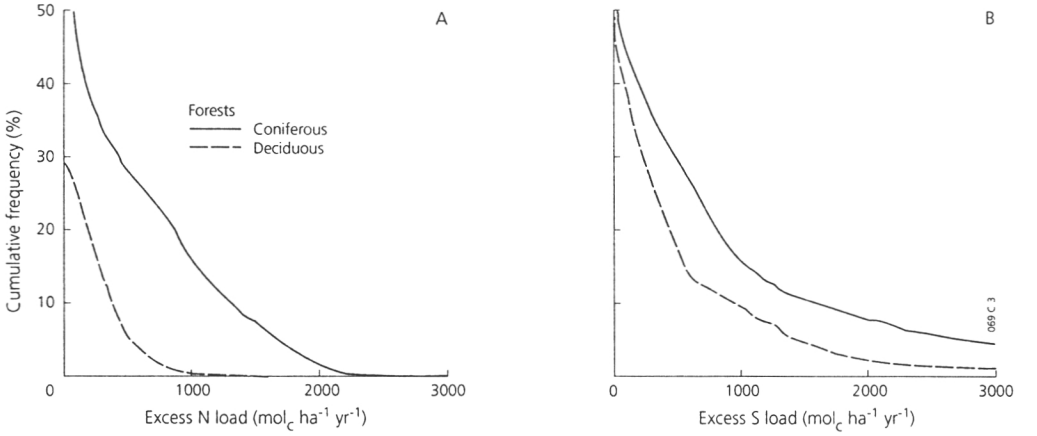


Figure 2. Inverse cumulative frequency distributions of the amount by wich critical loads of nitrogen (A), and acidity (B) are exceeded on coniferous and deciduous forests in Europe.

Maps of the forested area exceeding critical loads for N (Fig. 3A) and acidity (Fig. 3B) showed that an exceedance in critical loads occurred nearly all over Europe except for the Northern part of Norway, Sweden and Finland and large parts of Scotland, Ireland, Portugal, Spain, Italy, Greece and Albania (Fig. 3). Critical acid loads were exceeded in more than 85 % of the forested area in Denmark, the Netherlands, Belgium, Luxembourg, Germany, Poland, Czechoslovakia and Romania. In most Mediterrean countries, the area where critical acid loads are exceeded appeared to be relatively low (less than 15 to 35 %) i.e. in Spain, Portugal, Italy, Yugoslavia and Bulgaria. The same was true for most of the Nordic countries, i.e. Norway, Sweden (except for the southern part) and Finland. Exceedances were almost negligible (less than 5 %) in Albania, Greece and Ireland. Large exceedances (more than 2000 mol$_c$ ha^{-1} yr^{-1}) in critical loads in more than 50 % of the forested area were mainly restricted to Central and Western Europe, i.e. the Netherlands, Belgium, Germany, Poland and the Czech and Slovak Republics. Relative large exceedances in critical loads of N and acidity also occurred in the former USSR and in parts of the UK, Romania and France. Maximum exceedances in critical loads for N and S were close to present loads, i.e. up to 3500 mol$_c$ ha^{-1} yr^{-1} for N in the Netherlands, Belgium and Germany and up to 12000 mol$_c$ ha^{-1} yr^{-1} for S in Czechoslovakia, Poland and Germany (De Vries et al. 1994a).

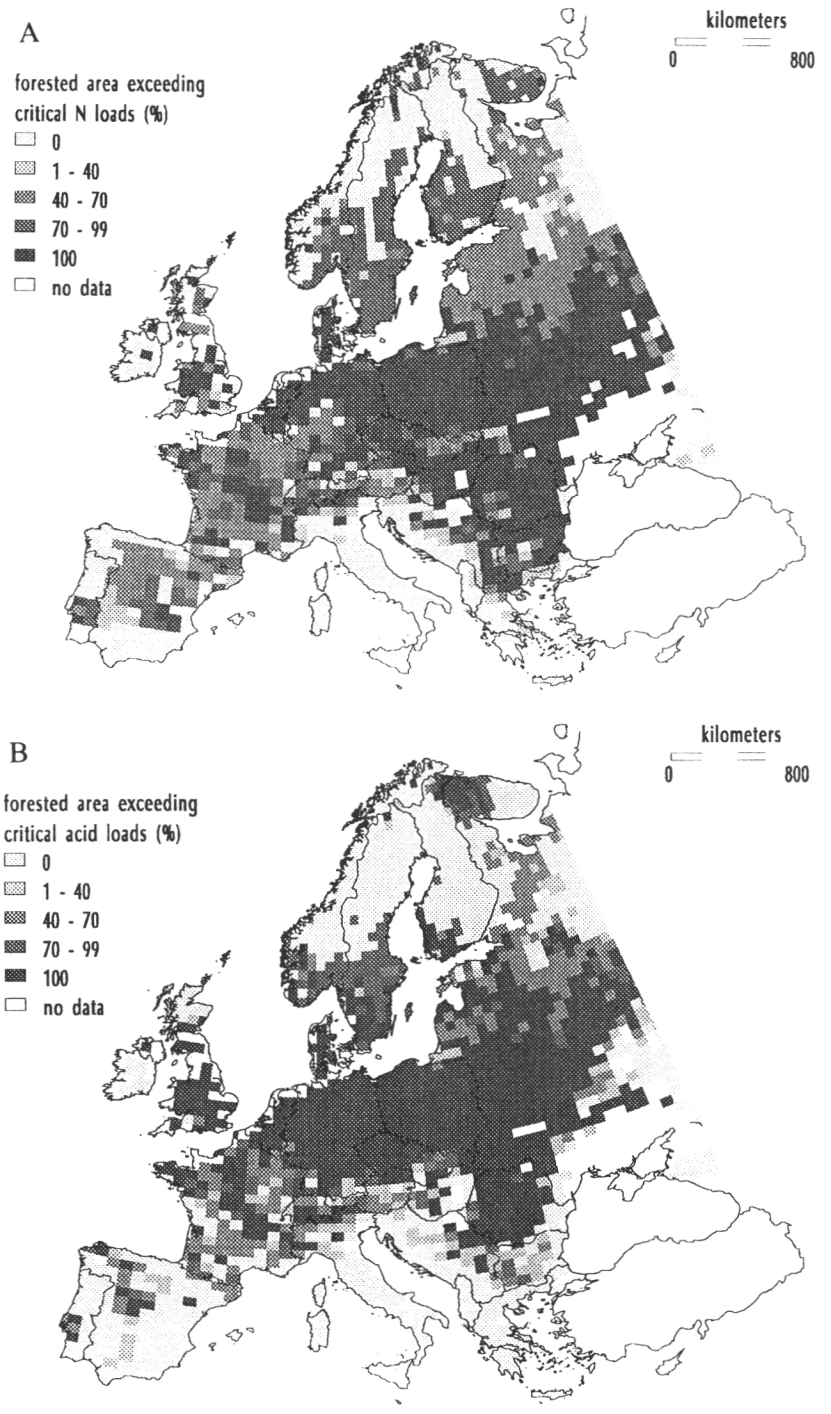


Figure 3. Geographic distribution of the forested area in Europe exceeding critical loads for nitrogen **(A)** and acidity (B).

Trends in parameters indicating nitrogen saturation

The predicted forested area with a C/N ratio below 40 increased continuously between 1960 and 1985 from 23.3 % to 35.8 % (Fig. 4A). After 1985, the increase continued for all scenarios but at a lower rate, especially for the Maximum Feasible Reductions (MFR) scenario. The Official Energy Pathway (OEP) and Current Reduction Plans (CRP) scenarios did not differ significantly in their effects on N saturation. In 2050 the predicted forested area with a C/N ratio below 40 equalled 51.6 % for the OEP scenario and 49.2 % for the CRP scenario compared to a value of 41.3 % for the MFR scenario.

The calculated forested area with a NO_3 concentration above 0.1 $mol_c\ m^{-3}$ (cf Table I) increased from 6.2 % to 16.0 % between 1960 and 1985 (Fig. 4B). A small reduction between 1985 and 2000 was found for the MFR scenario (due to deposition reductions) followed by a small gradual increase after 2010. For both the OEP and CRP scenarios the area still increased after 1985 to more than 25 % in 2050. The NO_3 concentrations at that time were far from steady-state with N deposition levels. Calculations with the START model showed that the forested area in Europe exceeding a NO_3 concentration of 0.1 $mol_c\ m^{-3}$ was as high as 50 % in steady-state situation when using 1985 deposition values. Note that the total N emission for the CRP scenario remains almost the same during the entire simulation period (cf Figure 1).

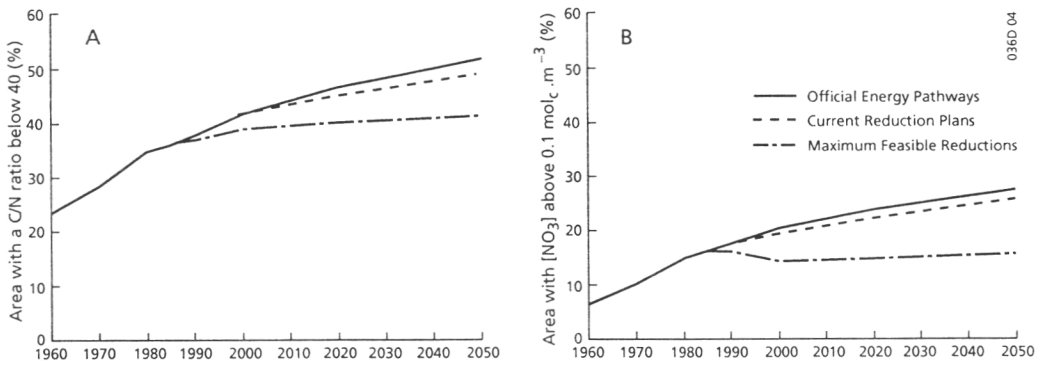


Figure 4. Temporal development of the forested area in Europe (%) with a C/N ratio below 40 (A) and a NO_3 concentration above 0.1 $mol_c\ m^{-3}$ (B) in response to three scenarios.

Trends in parameters indicating soil acidification

Impacts of the different scenarios on the Al concentration and Al/BC ratio are illustrated in Figure 5. Predictions for the area exceeding a critical Al concentration after 1985 showed a steady increase for the OEP scenario, a small reduction between 1985 and 2000 followed by a slight increase for the CRP scenario and a marked decrease, especially between 1985 and 2000, for the MFR scenario (Fig. 5A). The response of the Al/BC ratio was similar except for the CRP scenario. Unlike Al, the area exceeding a critical Al/BC ratio increased after 1985 for this scenario (Fig. 5B). Apparently, the decrease in BC concentration, induced by a decrease in base saturation change in response to ongoing acid deposition, compensated the decrease in Al concentration. As with NO_3, the concentrations in 2050 were not yet at steady state with respect to the deposition level at that time. Even when using the OEP scenario, with higher total S and N emissions in 2050 compared to 1985 (cf Fig. 1), the predicted areas with an Al concentration and Al/BC ratio exceeding critical values in 2050 were 'only' about 28 % and 14 %, respectively. Calculations with the steady-state model START showed that these areas would increase up to 43 % and 30 %, respectively, at constant 1985 atmospheric deposition.

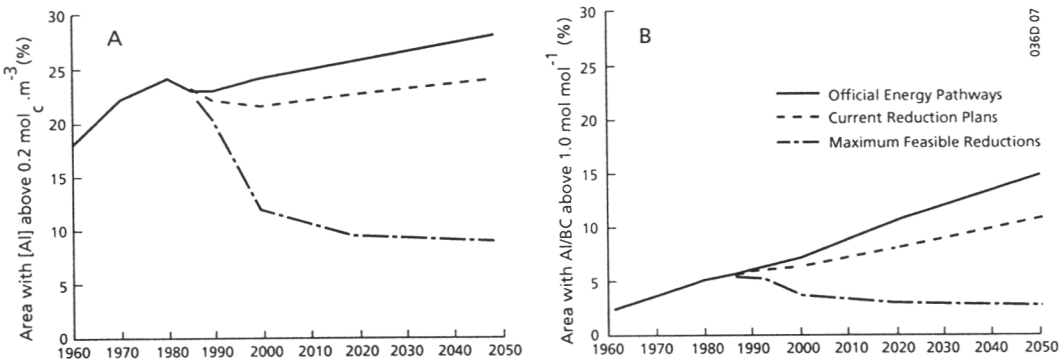


Figure 5. Temporal development of the forested area in Europe (%) with an Al concentration above 0.2 mol$_c$ m^{-3} (A) and a molar Al/BC ratio above 1.0 (B) in response to three scenarios.

Geographical patterns of nitrogen saturation and soil acidification

The predicted geographical patterns of N-saturated and (strongly) acidified forest soils changed with time. However, throughout the simulation period most of these soils were concentrated in the central, western and eastern European countries, i.e. Denmark, the Netherlands, Belgium, Luxembourg, Germany, Poland, former Czechoslovakia, Romania and large parts of the former Soviet Union. The situation for the year 1960, 2000 and 2050, using the CRP scenario, and at steady-state, using 1985 deposition levels, is illustrated with maps of the median predicted Al concentration (Fig. 6). The medians were calculated by weighting the results with the area of each forest soil combination in a grid.

High Al concentrations (> 0.2 mol_c m^{-3}) generally coincided with low values for pH (< 4.5) and base saturation (< 5 %), except for the (podzolic) soils in Scotland and in the Southern part of Sweden and Norway. In these areas the low pH and base saturation are induced by the poor parent material (low weathering rates), whereas the Al concentration is also low due to the relatively low input of SO_4 and NO_3 and partly because of the high precipitation excess (strong dilution), especially in Scotland and in the coastal region of Norway. Very high Al concentrations, exceeding 3.0 mol_c m^{-3}, were mainly limited to the central European countries, i.e. Germany, Poland and Czechoslovakia. High NO_3 concentrations were also predicted in the Netherlands, Belgium and Luxembourg and in large parts of the former Soviet Union. (not shown)

Predictions of the forested area with an Al concentration above 0.2 mol_c m^{-3} in a steady-state situation (using 1985 deposition data) also included large areas in the southern part of Sweden, Norway and Finland (cf Fig. 6D). Since these Nordic countries are densely forested, this explains the large increase in the forested area in Europe above critical values for Al, going from the non-steady-state (SMART) predictions to the steady-state (START) predictions. However, during the simulation period up to 2050 the acidified areas in the Nordic countries hardly increased. The predicted time period to reach steady state was several hundreds and sometimes even thousands of years in areas with a large N immobilization potential and relatively low N inputs. The maps with the area exceeding critical loads for acidity (Fig 3B) are (nearly) similar to those with a steady-state Al concentration above 0.2 mol_c in m^{-3} (Fig. 6D). This implies that the Al concentration criterium is generally limiting the critical acid load. The critical Al/BC ratio of 1.0 (cf Table I) appeared to be less stringent (De Vries 1994 a).

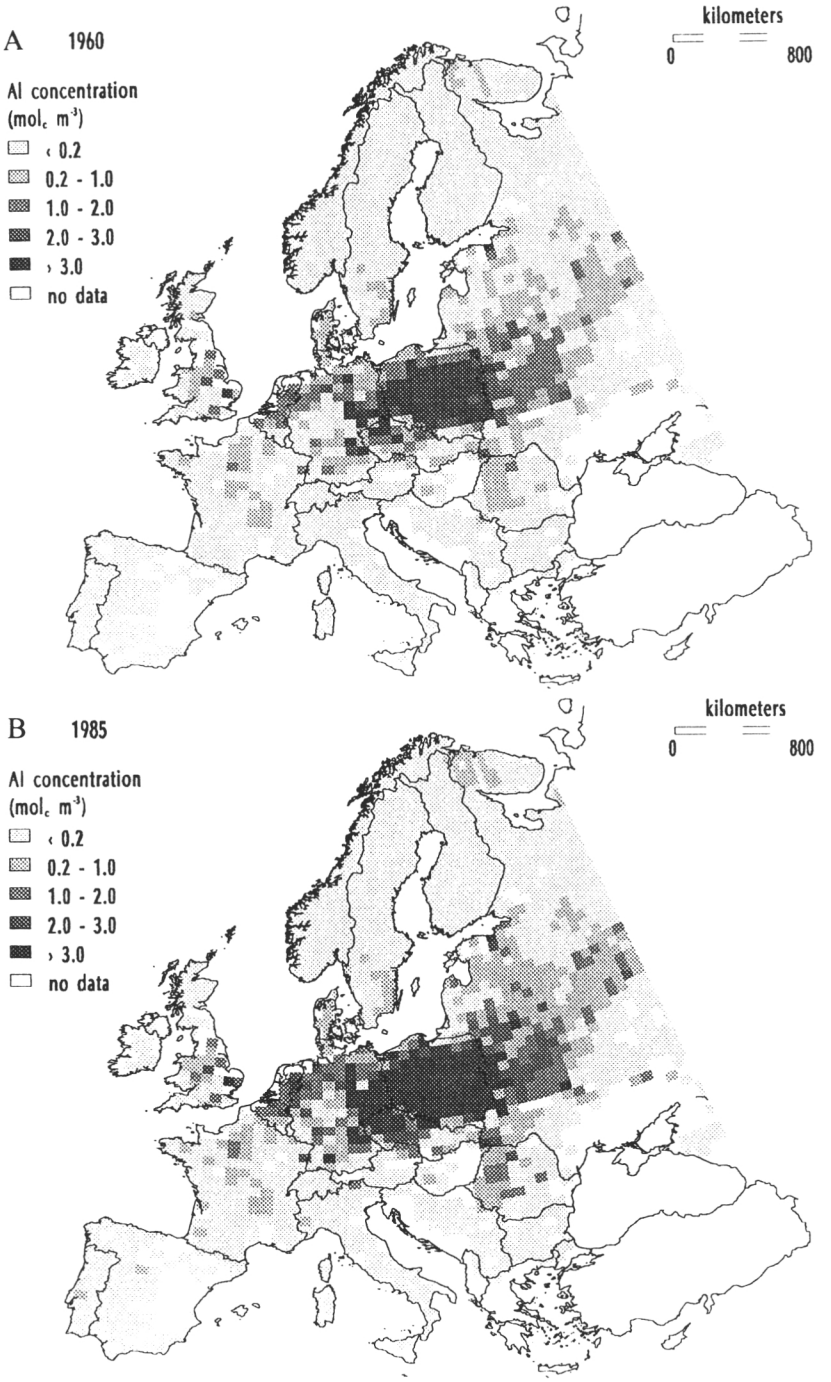
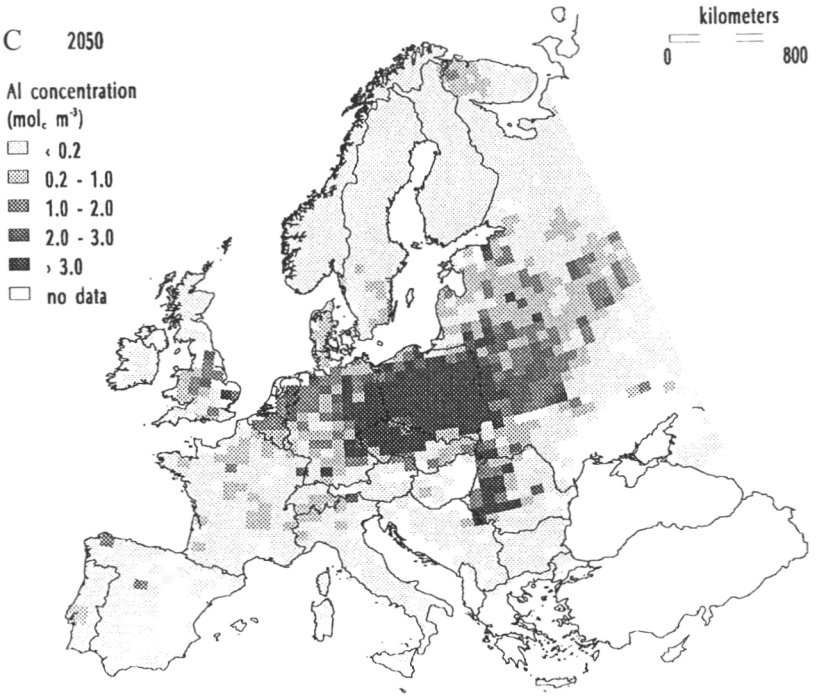
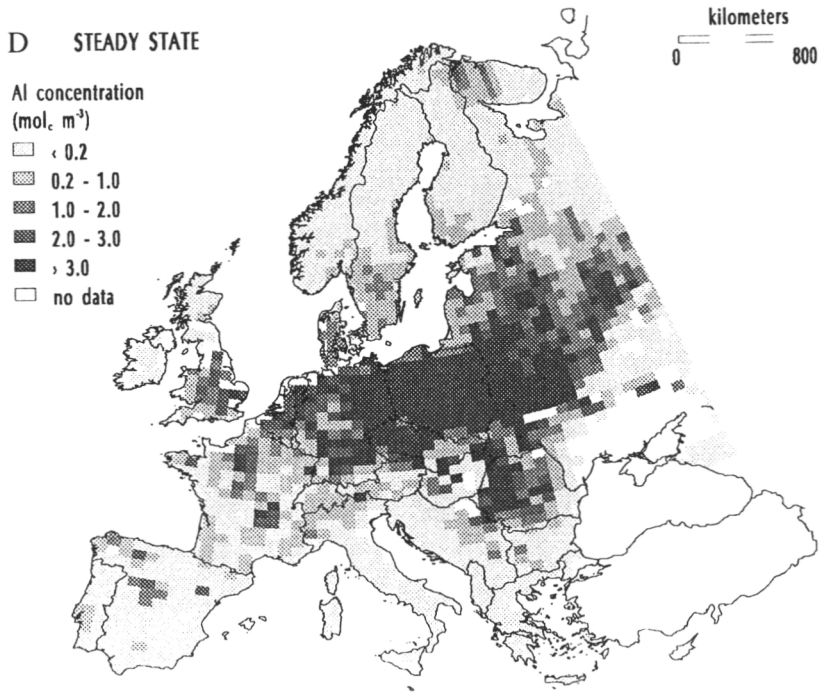


Figure 6. Maps of the predicted median Al concentration for the years 1960 (A), 1985 (B) and 2050 (C) using the CRP scenario, and at steady-state using 1985 deposition data (D).


C 2050
Al concentration
(mol$_c$ m^{-3})
‹ 0.2
0.2 - 1.0
1.0 - 2.0
2.0 - 3.0
› 3.0
no data
kilometers
0
800


D STEADY STATE
Al concentration
(mol$_c$ m^{-3})
‹ 0.2
0.2 - 1.0
1.0 - 2.0
2.0 - 3.0
› 3.0
no data
kilometers
0
800

Discussion and conclusions

Uncertainties in the area exceeding critical loads

The comparison of present loads and critical loads for N and acidity presented in this paper, indicated large exceedances in Western and Central European countries. In these countries the exceedance for N and S was generally more than 2000 mol_c ha^{-1} yr^{-1} up to 3500 mol_c ha^{-1} yr^{-1} for N (Netherlands, Belgium, Germany) and 12000 mol_c ha^{-1} yr^{-1} for S (former Czechoslovakia, Poland and Germany). Large exceedances were also calculated for parts of the UK, France, Romania and the former USSR.

Quite unexpectedly the area where critical N and S loads are exceeded appeared to be nearly equal (50 % vs 52 %, respectively). Note, however, that the calculated critical N loads relate to the occurrence of vegetation changes in forests. Rosén (1990) indeed found a good correlation between the exceedance of critical N loads and the occurrence of vegetation changes in Swedish forests. However, N loads affecting forest vitality, due to imbalances between N and nutrient base cations such as Ca, Mg and K in needles and in the soil (solution), appear to be much higher. Average critical loads derived for Dutch forest soils related to these effects are about 1500 to 3000 mol_c ha^{-1} yr^{-1} (De Vries 1993). Much larger critical N loads and smaller areas exceeding these loads (up to 29 %) were obtained when the critical NO_3 concentration in the soil solution was related to its inherent acidifying effect, affecting forest vitality (De Vries et al 1994 a).

The forested area exceeding critical acid loads was estimated at 45 %. The uncertainty in critical (acid) loads derived above is, however, large. This is mainly determined by the uncertainty in critical chemical values, model structure and model inputs (Sverdrup et al. 1990, De Vries 1994 a). Sensitivity analysis showed that the forested area exceeding critical acid loads is much more influenced by the uncertainty in the critical chemical values for Al and Al/BC, and the soil depth where they are applied uncertainty in model inputs (De Vries et al 1994 a). However, although the estimated uncertainty in the area exceeding critical loads was large (generally about ± 50 % of the average estimated value for N and acidity), it is clear that present loads in Western and Central Europe exceed the critical values in large areas.

For a correct interpretation of critical loads, it should be emphasized that an exceedance does not necessarily cause a visible effect on forests, due to the complex interactions in these ecosystems. However, an exceedance of the critical loads involves a certain risk to the health of forests, which increases

with the magnitude and the duration of the exceedance. Consequently, the calculated large exceedances in Western and Central Europe do not imply the dieback of these forests, but it certainly means that the health of these forests is endangered. This is in accordance with the reports on declining forest vitality in this area.

Trends in the forested area at risk

In this study we predict that the acidity and N-status of European forest soils do not improve when current reduction plans are being implemented. The forested area exceeding critical values for the NO_3 and Al concentration and Al/BC ratio is likely to stay rather constant or to increase slightly. Soil acidification is expected to decrease quite drastically following maximum feasible emission reductions (MFR scenario). The problem of N saturation, appears to be more persistent and likely increases in the future, even for the MFR scenario. N-saturated and acidified forest soils were concentrated in central, western and eastern European countries during the entire simulation period for all scenarios. Very high Al concentrations mainly occurred in central Europe (Germany, Poland and Czechoslovakia) whereas high NO_3 concentrations also showed up in the Netherlands, Belgium, Luxembourg and large parts of the former Soviet Union.

The question about the accuracy of the long-term soil responses estimated by the SMART model cannot be answered satisfactorily. Long-term soil chemistry data, that are needed for a rigorous testing of the model are generally lacking. However, the reasonable comparison between simulated and measured changes in (i) base cation amounts in the rootzone of five Swedish sites between 1949 and 1984 (Van Oene and De Vries 1994) and (ii) soil solution chemistry in a continuously monitored spruce site in Solling, Germany (Van der Salm et al. 1994) show that SMART produces plausible results. Furthermore the general trends described above were predicted irrespective of the uncertainties involved in critical values and input data used. These uncertainties (sometimes strongly) affected the predicted area where critical values are exceeded in a given year, but not so much the trends in time. The results obtained thus indicate that the forested area in Europe under stress will not diminish when current reduction plans are implemented.

References

Alcamo, J., Shaw, R. & Hordijk, L. 1990. The RAINS model of acidification. Science and Strategies in Europe. Dordrecht, The Netherlands, Kluwer Academic Publishers, 402 pp.

De Vries, W. 1991. Methodologies for the assessment and mapping of critical loads and of the impact of abatement strategies on forest soils. DLO Winand Staring Centre, Wageningen, The Netherlands, Report 46, 109 pp.

– 1993. Average critical loads for nitrogen and sulfur and its use in acidification abatement policy in the Netherlands. Water Air and Soil Poll. 68: 399-434.

– , Posch, M. & Kämäri, J. 1989. Simulation of the long-term soil response to acid deposition in various buffer ranges. Water Air and Soil Poll. 48: 349-390.

– , Reinds, G.J. & Posch, M. 1994a. Assessment of critical loads and their exceedance on European forests using a one-layer steady-state model. Water Air and Soil Poll. 72: 357-394.

– , Posch, M., Reinds, G.J. & Kämäri, J. 1994b. Simulation of soil response to acidic deposition scenarios in Europe. Water Air and Soil Poll. 78 (in press.)

Heij, G.J., De Vries, W., Posthumus, A.C. & Mohren, G.M.J. 1991. Effects of air pollution and acid deposition on forests and forest soils. In: Heij, G.J. and Schneider, T. (Eds.). Acidification Research in the Netherlands, Studies in Environmental Science 46, Elsevier Science Publishers, Amsterdam, the Netherlands: 97-137.

Lammel, R. 1984. Endgültige Ergebnisse und bundesweite Kartierung der Waldschadens Erhebung. Allgemeine Forst Zeitschrift 39: 340-344.

Lövblad, G., Amann, M., Andersen, B., Hormand, M., Joffre, S. & Pedersen, U. 1992. Deposition of sulphur and nitrogen in the nordic countries: Present and future. Ambio 21: 339-347.

Pedersen, U., Schaug, J., Skjelmoen, J.E. & Hanssen, J.E. 1990. Data Report 1988, Part 1: Annual Summaries. EMEP/CCC-Report 4/90, Norwegian Institute for Air Research, Lillestrom, Norway.

Posch, M., Reinds, G.J. & De Vries, W. 1993. SMART: A Simulation Model for Acidification's Regional Trends. Model Description and User Manual. Mimeograph Series of the National Board of Waters and the Environment 477, Helsinki, Finland, 43 pp.

Roberts, T.M., Skeffington, R.A. & Blank, L.W. 1989. Causes of type 1 spure decline. Forestry, 62 (3): 179-222.

Rosén, K. 1990. The critical load of nitrogen to Swedish forest ecosystems. Department of Forest Soils. Swedish University of Agricultural Science, Uppsala, Sweden, 15 pp.

Van der Salm, C., Kros, J., Groenenberg, J.E., De Vries, W. & Reinds, G.J. 1994. Validation of soil acidification models with different scale perspectives on an intensively monitored spruce site. In: S. Trudgill (Ed.). Solute modelling in catchment systems (submitted).

Schütt, P., Blaschke, H., Hoque, E., Koch, W., Lang, K.J. & Schuck, H.J. 1983. Erste Ergebnisse einer botanischen Inventur des 'Fichtensterbens'. Forstwiss. Zentralbl. 96: 177-186.

Van Oene, H. & De Vries, W. 1994. Comparison of measured and simulated changes in base cation amounts using a one-layer and a multi-layer soil acidification model. Water Air and Soil Pollut. 72: 41-66.

The concepts of critical loads and environmental quality standards

Staffan Westerlund

Uppsala Universitet
Institutet för miljörätt
Box 512, S-751 20 Uppsala, Sweden

Introduction

Most of people's efforts seem to have been based upon the assumption that Nature shall be conquered, that welfare is equal to changing nature, and that reasonable protective measures shall be based upon what people can afford without to abandon exploitation ideas, not upon what the environment and – for that matter – future generations or present neighbours require. We easily see this in the Swedish and Finnish environmental legislation, especially the respective water acts and the Swedish Environment Protection Act. The modern expression is BAT – best available technology. We can call this the *exploiters' approach.*

However, environment does not refrain from deteorirating however many economic arguments we present in courts, before Franchise Boards and when amending the laws. Environment has a tendency not to care about human economy. Actually, we all know that. It is common sense and common knowledge – we even recognize that there are laws of Nature – but is is not common understanding in the real sense of the word, when it comes to practice.

On the other hand, the concept of safety is widely accepted in many areas, be they safety at sea or safety in elevators. Furthermore, publich health approaches are based upon the understanding that there are limits for human beings, limits beyond which health declines. Setting the limits based upon the ultimate goals actually has a long history. We know it from maximum loads and maximum number of persons in ships, in aircraft and in elevators. Within environmental law, we recognize the approach in laws prescribing air and water quality standards even if we do not have much of that in Finland or Sweden.

A collegue of mine from Germany once delivered a lecture in Brazil concerning acidification and forestry and he began by saying that finally the tortures had become victims. He was referring to the fact that even in Germany forests were affected from acid rain and other kinds of environmental stress. It was, in plain

language, too much for the forests. Described that way, we apply a quite different approach, an *environment* or *receipt* or *effect oriented* approach.

Today, the concept of critical loads and of environmental quality standards are not only recognized but also rather widely accepted, at least on a general level, mostly because the impact from overloading the environment also strikes at the exploiters themselves.

The environment made relevant

We have three terms here to deal with; environmental quality, impact, and load.

I would like to take a fish as the simplified example of what it is all about. There is a movie where a fish did not actually act, but she was there, and in one way or another she was also present and important; even in the name of the movie: A fish called Wanda. A fish could actually be any objective in any environmental discussion. Fish do not act, they simply react to their environment. If the aquatic environment is burdened with pollution, this pollution constitutes an *impact* on the aquatic environment. And the environment, the ecosystem, reacts depending on the *load* and on the time period; this last time aspect also raises issues of concentrations and threshold effects.

If the burden is significant and not only temporary, the pollution keeps coming into the system, and sooner or later there are changes in the system. For example pH drops, the water changes its *quality* and becomes more acid, some organisms who can not take a lower pH die, a new ecological state occurs and maybe Wanda the fish will not be able to reproduce. Or the concentration of nutrient rises, eutrophication increases, a new ecological state occurs and maybe Wanda the fish will leave or die, giving room for other species.

The same picture applies to other systems, including the human organism, and the cell itself. Some strains can be tolerated, but there are limits, beyound which the systems or other objects of the strain maybe perish. They can not take more. They are overloaded; beyond the limits of critical loads.

This is true for the Baltic including the Bothnian areas, is is true for many forests areas, it is possibly true for many cells with respect to toxic and carcinogene substances. It is, in fact, so obvious that we only have to start taking care of the problems. We have to arrange for the environment not to be overloaded with pollution and other kinds of stress.

So, if the environment actually is overloaded, we have to change this situation. If people are actually legally overloading the environment, let us make this illegal. Let us prevent it.

The difficult ecological environment

Coming this far, we must start to discuss how to formulate and implement environmental policy. We have to translate our general understanding of the environment's limited tolerance towards stress into concepts, and into the setting of *action forcing* limits. In doing so, we have to seek for the concept of critical loads and the concept of quality standards. And then we have to apply these concepts, if possible.

As I have already mentioned, the most typical approach has been, so far, the BAT approach. It focuses on the sources of pollution. It is the exploiters approach to environmental protection.

The concept of environmental quality naturally focuses on the objects of the impact from these sources, on the environment itself. It reflects the environment or recipient approach.

One basic problem that we simply can not avoid is that neither the fish, nor the environment in general, respond linearily to impacts. One plus one is, normally, not two in ecological systems. You may be able to add some pollution into the water and see no differences. You add more – no differences as far as you can see, but maybe you would be able to monitor an increased concentration in the water. Then you add just a little more and, whoops! Other species of algae than the normal ones appear and take over. When they die, the content of oxygene in the water decreases considerably and the fish can not take the result – they suffer, leave or even die.

However, the content of nutrients in the water body may increase *even if the load as such is not increased.* For example, the construction of a dam upstream or downstream, or a road bank crossing the mouth of a bay, may change the hydrological conditions in such a way, that the exchange or flow of water decreases and the pollution does not leave the recipient as quickly as before these constructions. This alone may change the water quality of the recipient, resulting in the same effects as increasing the load on the recipient.

Furthermore, pollution load does not only come from point sources. Non point sources are very important here. These include among others agriculture land, forestry land, and traffic. Long range pollution, whether we regard that as

originating from point sources or not, is another significant factor when accounting for pollution load.

Now we easily understand that coping with pollution in order to achieve *environmental* qualities sufficient for a sustainable development is much more complicated than merely using the BAT approach. The fish does not care about the economy of forestry industry, of family economy, of regional polities, of trade balances. She does not care at all. She just reacts to her actual environment and she dies, if the oxygene content is too low, however much we try to urge her to assist us in order to save our so called welfare.

Neither does she care about our efforts to define critical loads, if the application of our definition results in a water quality that makes her react – not act – by dying.

The environmental quality concept

This brings us to the *environmental quality concept*. It does not deal only with defining environmental quality as such, but also with using the environmental quality objectives as something, that one way or another must be set in operation. Our efforts, if applying this concept, should be directed towards actually *achieving* and *maintaining* certain environmental quality standards. And the actual environmental quality, if inferior to the quality objectives, must be made relevant within the policy implementation system. This means that, as long as the prescribed environmental quality is not achieved, the law and all efforts must be directed towards the achievement and maintaining of the intended environmental quality.

This is very far from the BAT approach. Actually, it is quite opposite. But how far is it from the critical load concept ?

The critical load concept

The basic philosophy behind the critical load concept is that beyond the point of critical load, the environment or some of its components reacts unfavourably. That is: the environmental quality, or some of the environmental components, will be too poor.

Defining critical loads therefore seems to be easy, in principle. You calculate, based upon the basic properties of the recipient, how much pollution can be added (= the load) without certain threshold effects occurring. Simple !

Or is it really ? What if the recipient conditions change ? What if the hydrological conditions change because of a dam ? What if the background pollution increases? What if land use is changed in such a way, that more non point source pollution, not accounted for, leaks into the water system ?

The answer is, of course, that all calculations concerning the actual critical must be remade and new critical load figures be issued. This illustrates the limited possible applications of the critical load concept.

Still, it will be important to define the terms *load* and *critical load*, but the better the definitions, the more difficult the practical applications, I should think. Provided that the reason for applying a critical load concept is that certain basic environmental quality objectives really shall be achieved, the critical load figures and nothing but instruments, not objectives. Instruments for achieving and maintaining certain basic environmental quality objectives.

The complexity of implementation issues

Of course, this brings us to the question how to construct the instruments for implementing environmental quality policies. Provided that it has been decided that a certain environmental quality shall be achieved, then the implementation of that objective must be the beacon, by which to navigate, by which to control all actions and activities.

Defining the environmental quality is only a part of this work. Which should be the parameters ? Dissolved oxygene ? Temperature ? Tudbidity ? pH ? Plancton count ? Or the fish herself ?

There is no single answer. There are parameters on several levels. Suppose that we have decided that the fish shall survive and be able to reproduce, then of course the ultimate parameter is exactly that – the fish. However, her envrionment is water and food etc, and in order to secure her environment, we can pick some key parameters and define quality standards for those. That way, we already have quality standards on two or more levels.

But will we really be able to implement our Wanda-the-fish policy merely by defining these quality standards ?

The answer is, of course, negative. Quality standards are not norms explicity telling a specific person or company what to do and what not to do. The quality standard has no receiver, it does not include an order to someone on how to act. There is no person in the other end. Simply put, a water quality standard includes no orders,

but if they did so, the orders would be directed to the water: be clean ! But the water can not act. It simply reacts; but not to orders. We need *agents* in the system, people who *act*. People to whom orders can be directed.

Implementation requires people

How do we include agents, persons, people in this ? How do we set a system *including* and *based on* environmental quality standards into operation ?

Now we arrive at the implementation issues. This is where virtually all problems are to be found. Because, if we really intend to keep Wanda the fish safe and sound, we have to control (directly or indirectly) the collective activities and actions of all people in such a way, that the environmental quality required by Wanda really is there ! In that implementation, we must observe different geological, hydrological and meteorological conditions, the interaction between different environmental factors and components including synergetic effects, the non-linearity of effects in ecosystems, the gaps of knowledge, the fact that there are corrupted people and criminals, political cowards, and not only human saints.

But that is not all. We also have to *assist* all those good peple, who really try to combine living with environmental protection. Assist them by means of standards, by justly distributing the rights or opportunities to benefit from the environment and natural resources, by informing (or, when necessary, ordering) them on how they must act, in order to conform with the over all environmental policy implementation scheme.

All this calls for a variety of components in the implementation system. One component is standards on precaution, possibly based upon the BAT concept. Another is regional implementation plans with legal power. We do not have those – with legal power connected with quality standards – in Finland or Sweden.

We also may want to try to define the total limits of pollution that may be introduced in the ecosystems within the framework of the over all environmental quality police. This is where the critical loads concept may enter the scene. However, it is a weak concept from practical points of view, since the environment does not react uniformly to a specific load and since recipient conditions may vary or change, making critical load figures obsolete.

Total emission reduction approach – super bubbles

Still, there may be some environmental impact issues which are well fit for relying on the critical load concept. I am referring to pollution that in principle equally affects the entire global ecosystem such as adding carbon dioxide or CFC to the atmosphere. However, we must observe that even a climate change will hit different part of the world differently. Therefore, it is mostly from pure practical or pragmatic point of view that the critical load concept is suitable in these cases. Defining environmental quality standards *in these respects* would probably add nothing to the function of a system for the implementation of environmental policies. In stead, some kind of "super bubbles" defining the total amount of emissions in a region and the world seem suitable. You can study examples of that in the EC regulations on CFC and in the Acid Rain part of the US Clean Air Act.

Conclusion

Applying a critical load concept can never be a goal, only an instrument. Behind the notion of critical load are the ideas of environmental quality and sustainable development. Critical load calculations must be adjusted to these ideas and objectives, not to what industry etc can afford economically. Critical load calculations are, if the concept is correctly applied, governed by environmental quality objectives and must observe ecological realities. This means that the application of a critical load concept is only a small part in a very complex and complicated implementation system, the construction of which has barely begun.

Further more, the search for what is actually "critical loads" may make experts focus too much on figures, less on ecological complexities. As long as the Baltic, or the soil, or the streams, are overly polluted, the pollution load must be reduced very much. This is common sense. The search for exact figures for critical loads may obscure common sense. Or, to paraphrase a Swedish author: Some things are so obvious, that it takes an expert not to see them.

One way to avoid the risk of being obscured by the search for the critical load is, of course, always to check with the environment.

Acknowledgements

This paper is based upon a long period of research on environmental law methodology, beginning with two mimeographed publications in 1976 and 1977 (Westerlund: Clean Air Act, Federal Water Pollution Control Act, Miljöskyddslagen... Juridiska Institutionen, Uppsala universitet 1976 and

Westerlund: Det rättsliga genomförandet av luftkvalitetsnormer enligt USA:s Clean Air Act, Juridiska institutionen, Uppsala universitet 1977) and reflected in Westerlund: Miljörättslig grundfrågor (Trondheim 1987). More recent and advanced research on environmental quality standars are now being conducted by Lena Gipperth at the University of Uppsala who also has published two studies (Gippearth: Erfarenheter av USA:s miljökvalitetsnormer, published in Miljörättslig tidskrift 1994:1 and Gippearth: Miljökvalitetsnormer som styrmedel, published in Miljörättslig tidskrift 1994:3). At the University of Uppsala, Birgitta Leyser will focus her research on what I in my paper called super bubbles.

Critical load models for the regional and local scale: a cautionary scientific perspective

Kevin Bishop

Swedish University of Agricultural Sciences
Department of Forest Ecology
S-901 83 Umeå, Sweden

The use of critical loads as the basis for the recent European protocol on sulfur emissions is a historical step in the management of transboundary pollution. For the first time in such an international agreement, the requirements for emission control are related to the deposition that nature can tolerate in different areas. Negotiations are continuing under the auspices of the United Nations Economic Commission for Europe to extend the critical loads approach to international regulation of other transboundary atmospheric pollutants, and eventually even combinations of pollutants. Another potential extension of the critical loads approach would be to narrow the geographical focus from Europe as a whole to a regional or even local scale.

The potential for using critical load models at the regional and local scale was discussed at a meeting of more than 50 acidification researchers sponsored by the Swedish Environmental Protection Agency in the winter of 1993. Perhaps somewhat surprisingly, the response of this group was not positive. This paper seeks to explain, from one researcher's perspective, how it is that some scientists can be less than enthusiastic about such an apparently obvious extension of the generally acclaimed concept of critical loads from the European scale to a regional or local scale.

The central reason for this reluctance is the great demands which the critical loads concept places on the ability of science to model the response of nature to pollution. So far, the only critical load models accepted as a basis for environmental policy are those for sulfur. This pollutant, which has been studied intensively for over a quarter of a century in relation to acidification, has a relatively simple biogeochemical cycle relative to other pollutants. Nonetheless, there are still questions about the reliability of the sulfur critical load models. One area of my own research has to do with the use of paleolimnological data to validate critical load models for sulfur. That research has indicated the need for improvements in the current models. And while sulfur critical load models remain to be satisfactorily validated, it is not yet even clear how critical loads for nitrogen

will be modeled, much less validated. Thus one can hopefully appreciate a degree of scientific skepticism at the prospect of narrowing the geographic scale at which critical load models are applied.

But it is more than just a proliferation of uncertain model predictions that motivates the concern about regionalization of critical load models. Even if the problem of model uncertainties could be solved, one must still ask whether the critical load approach, as adopted for sulfur at the international scale, will be useful in dealing with a range of pollutants at the regional or local scale.

Critical load models define a baseline level of pollution that the more sensitive areas of an ecosystem can tolerate without significant effects. This is useful in international negotiations where it is considered unacceptable for remote (i.e. foreign) pollution sources to effect a nation's environment. Within a country, however, a degree of human influence on the environment is generally accepted as inevitable. Therefore at the regional and local level of environmental planning, it is the trade-off between environmental costs and social benefits that is of central importance. In that situation, what relevance does a critical load level for zero environmental influence have when that is not a feasible goal for local or regional policies ?

Perhaps there is a use for such critical loads as the regional scale, but it is essential to evaluate the usefulness of that critical load before investing the scarce resources available for environmental research and protection into critical load modeling at the regional scale. I would also suggest that to be most useful at the regional and local scale where political compromises between environmental impact and social benefit must be negotiated, two new aspects of critical load models will have to be developed. One is to move from simply defining the level of no significant impact to assessing the degree of impact. It will also be important to not only predict that an impact will occur, but also when. Unfortunately, once the questions of degree and rate of impact are added to critical load models, the elegant simplicity of the critical load concept gives way to the familiar, but confoundingly complex problem of providing the political process with scientific assessments of human impact.

The critical load concept is an excellent principle upon which to base goals for international agreements on controlling transboundary pollution because it relates control efforts to scientific estimates of what nature can tolerate. It is, however, that central role for science in the policy process that gives the scientific community reason to be anxious about the extension of critical loads from the international to the regional scale. It is difficult enough to make defensible critical load estimates at the 100 km^2 grid scale currently being used. A change in geographical scale will make greater demands on the type of information provided by critical load models if they are to be useful when a political balance must be

struck between the social benefits of human activity and the environmental impact generated by that activity.

This paper hopes to have gone some way towards explaining why the initial response of some scientists to applying critical load models at the regional and local level may be less than enthusiastic. Resource managers and politicians should consider the implications of that initial response by asking two questions:

1) Will science be able to provide acceptably reliable critical load models for the range of pollutants that are of interest at the regional and local scale ?

2) Will critical load models provide useful information for the political decisions that have to be made at the regional and local level ?

Critical load and critical limit values from the standpoint of the energy producer

Heikki Niininen

Imatran Voima Oy
Ympäristönsuojeluyksikkö
FIN-01019 IVO, Finland

Critical load concept as a basis for emission reduction programmes

When setting goals for emission reduction programmes both environmental and economic aspects have to be taken into account in a balanced way.

The concept of critical loads is no doubt the most objective way of including the environmental aspects in that balancing process.

In the process of defining critical loads the first phase is, of course, the work of scientists. As a second phase, which nowadays does practically not exist, there should be a profound discussion, in which different parties, among others the industry, could express their views. This is because in the background of any given critical load there are many subjective components i.e. choices and assumptions, and it would be reasonable that all parties of the emission reduction process would be involved in the discussion. This applies even in larger extent to the setting of intermediate goals, so called target loads, the choice of which is purely political in nature.

Emission reduction in energy production

The first Finnish programmes of emission reductions of sulphur and nitrogen oxides, based on the first sulphur and nitrogen protocols and corresponding declarations, have practically been finalized. Nearly all coal fired power plants have been installed with flue gas desulphurization devices, and the newest unit at Meri-Pori also with selective catalytic reduction plant for NO_x emission reduction. In addition, the combustion technologies of most boilers over 50 MW_{fuel} have been modified in order to achieve lower NO_x emission levels.

So far, the costs of SO_2 and NO_x reduction investments in power production are in the range of FIM 3000 – 3600 million. The cost effectiveness of the measured varies mostly from FIM 2000 to 6000 per tonne pollutant removed.

As a result of the emission reduction investments and several advantageous structural changes in energy production the sulphur emissions are at the moment nearly 80 % lower than in 1980, whereas the NO_x emissions show much more moderate decrease (Fig. 1).

Further emission reduction programmes, which will be based on the critical load concept, will imply remarkable additional investment and operational costs. Perhaps even more importantly, the cost effectiveness of further reductions would be very poor, up to over FIM 20000 per tonne pollutant removed. According to the background reports made for the formulation of Sulphur Protocol II the cost per GNP of the Finnish emission reduction programmes is extremely high in comparison with the corresponding figures in other European countries (Fig. 2).

As a consequence, there is a danger that Finnish energy intensive products may loose their competitiveness in the market. Another consequence may also be – as the liberation of the European energy market proceeds now rapidly – that our domestic energy production with strict emission norms will loose its competitiveness against imported energy from countries with lower environmental norms. This again can lead to the increase of overall regional emissions.

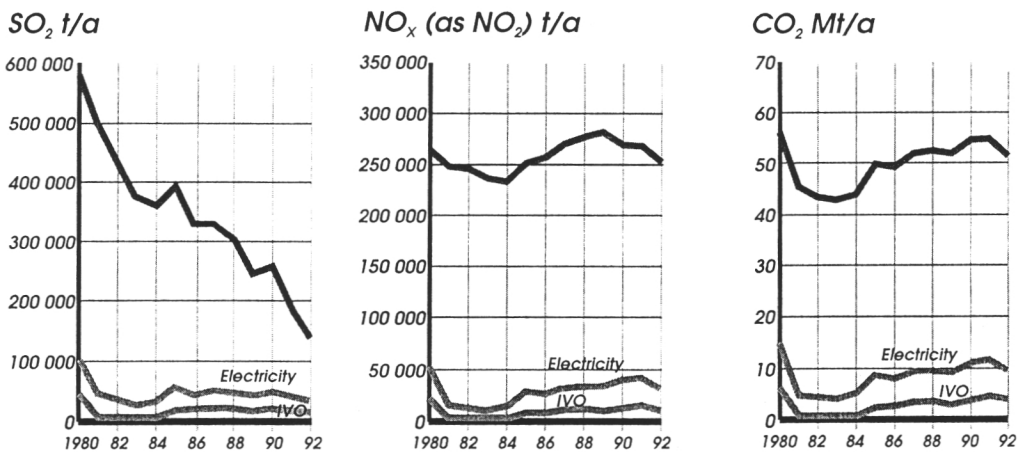
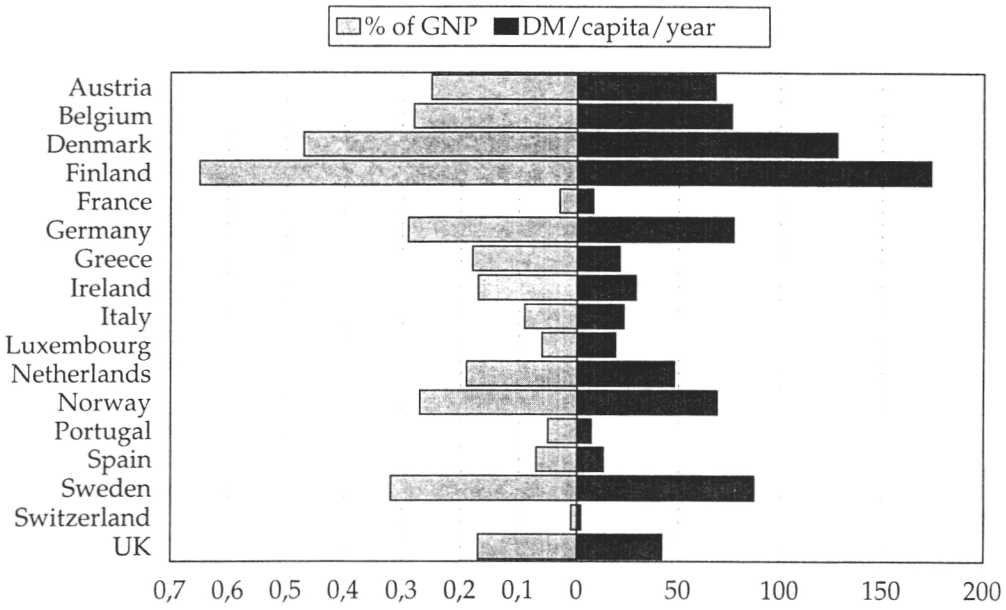


Figure 1. SO_2 -, NO_x - and CO_2 -emissions in Finland.



Source: Closing the Gap between the 1990 Deposition and the Critical Sulphur Deposition Values, IIASA, May 1993

Figure 2. Costs of sulphur emission reduction. 60 % gap closure scenario, year 2000.

Towards a more balanced situation

In order to avoid unfavourable economic and also environmental effects of the emission reduction programmes derived directly from environmental requirements, other elements have to be used and have been used in some extent in the international agreements and multinational emission reduction plans. Such elements are for instance the harmonization of emission requirements and the concept of Joint Implementation.

Sulphur Protocol II (Article 2, Basic Obligations) sets some minimum requirements for emissions of stationary emission sources, but the time scale for the implementation of these requirements is very long.

In the European Union the Large Combustion Plant (LCP) Directive sets common minimum requirement for the emissions of boiler plants, but it covers only new installations. A revision of the LCP Directive should be released by the Commission by July 1 next year. Another process in the EU, which could promote the harmonization of emission requirements is the development Integrated Pollution Prevention and Control (IPC) approach. However, it is still unclear, whether the

coming IPC directive will lead to the formulation of emission requirements on the EU level or on the national level.

About 80 % of the acidic deposition in Finland originates from foreign sources, and a big fraction of that comes from our eastern and southern nearby areas (Fig. 3). Partial solutions towards to a more balanced situation in emission reduction could be the utilization of the concept called Joint Implementation, which has been included in the Sulphur Protocol II. According to the concept the parties of the protocol, two or more countries, may jointly implement the obligations of the protocol. This concept can result in remarkable environmental and economic benefits. For instance, the cost effectiveness of SO_2 removal e.g. in the Baltic power plants is up to 10 times better than that in the further emission reduction in corresponding Finnish facilities.

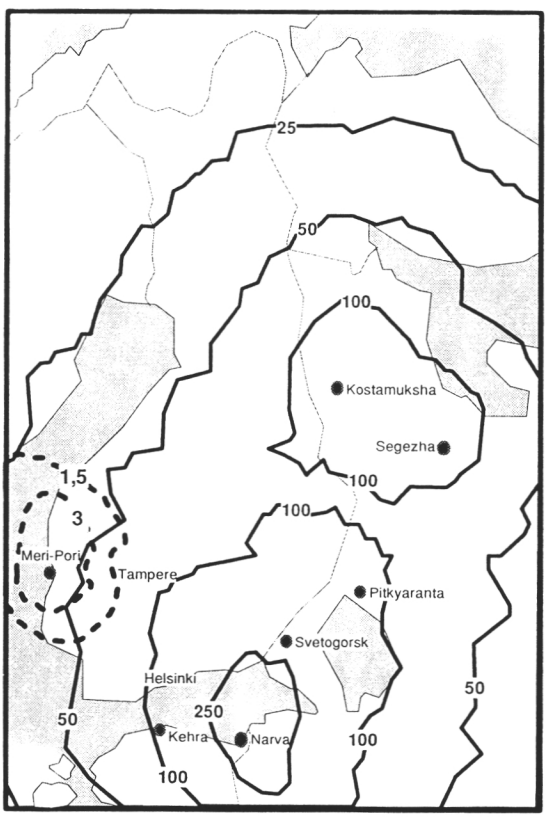


Figure 3. The annual sulphur deposition (mg/m^2) in Finland due to the Meri-Pori power plant and six emission sources in the nearby areas. The total annual sulphur deposition in the southern Finland is 700−900 mg/m^2.

Critical load and critical limit values from the stand point of the local administration

Mats Nebaeus

Umeå kommun, Miljöförvaltningen
Box 1016, S-901 20 Umeå, Sweden

Today there are many expressions and statements of aims, objectives or targets reflecting our hopes and ambitions concerning the environment. These aims are based on knowledge of i.e what nature can sustain and hence a concept of critical load. The gap between aims or critical loads on one hand and what mankind really is doing to nature on the other is sometimes tremendous and we still lack many tools to describe this gap, especially in biological terms or numbers.

The laws that were introduced in 1987 in Sweden enabled more ecological aspects in the planning of the municipalities. Both the use of land and and water were to be described in comprehensive plans in 1990, for all municipalities in Sweden. The statements or aims in these documents are not binding in juridical terms, but they have a moral intention or meaning.

The critical load concept is often used when you are concerned with airpollution and specifically, sulfur and nitrogen. Critical loads play a central role in the work with international agreements about reduced pollution. There is however a need for critical loads or critical limits in a variaty of subjects including acidification (in more detail), eutrophication, metals, longlived organic compounds, the use of land and the environment, and disturbances of the environment and human health.

Critical load can be defined as "the highest load causing long-range negative effects on the most sensitive ecosystems". The concept is based on current knowledge. When interpreting "long-range negative effects" it is important to connect this to the need for preserving the biodiversity and the protection of species that are threatened.

The work at the Public and Environmental Health Office on a local level and the interactions with other governments and certain subjects is described with boxes in figure 1. The results of investigations from different contributors are taken into consideration and are transformed into environmental plans for water- or land in the municipality. In these documents you can find rather concrete measures. The results are also tranformed into a program where the environmental aims or targets of the

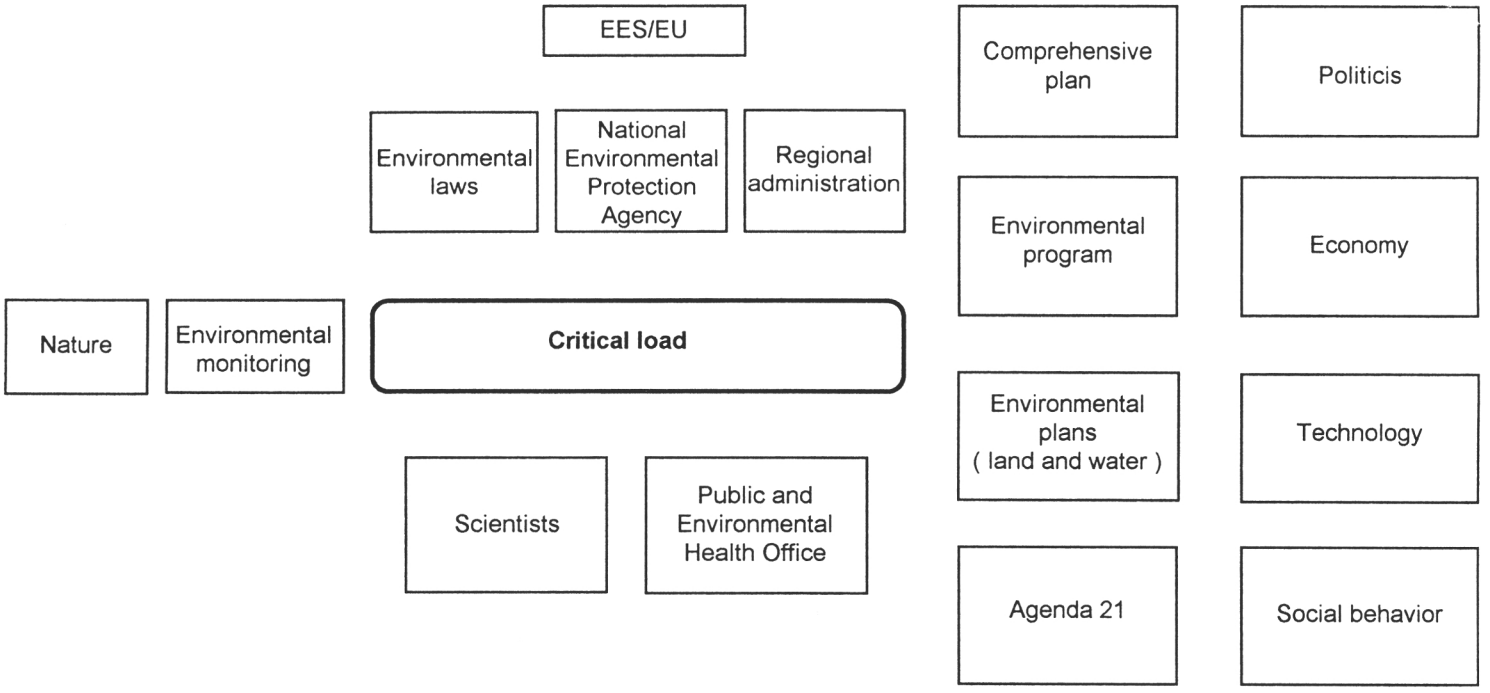


Figure 1. Critical load from the standpoint of the local administration. Explanation in the text.

activities in the municipality of Umeå are presented. In the future there will also be an Agenda 21. Finally the aims or targets and certain measures can be presented to and accepted by the central politicians of the municipality in the comprehensive or master plan according to the laws from 1987. In these programs and plans critical loads, critical limits, aims or targets are very important factors regulating the measures on a local level. It would be desirable to have a differentiated critical load concept that establishes regional and local critical loads in addition to those at the national level. If possible, the critical loads ought to be transformed to guiding principals and some of them to norms that are legally binding according to proposals of a new law, the environmental code.

The requirement for comprehensive plans that was introduced 1987 has resulted in a better understanding of ecological aspects amongst the traditional planners. They eagerly absorbe new knowledge of nature. The critical load/limit concept will probably be incorporated in the planning of the municipality and it is useful when working with Environmental Impact Assessment.

The Swedish Environmental Protection Agency (SEPA) has presented thirteen important subjects or threats to the environment i.e Gases affecting the earths climate, Depletion of the ozone layer, Acidification of soil and water, Eutrophication of seas, lakes and inland waterways, Metal pollution, Organic environmental poisons, Environmental resources and nature conservation. In the work with these subjects critical load critical limit concept could be used.

Umeå is involved in liming projects as are many other municipalities in the northern part of Sweden. In this work, it is of great importance that models or criteria for acidification are adapted to this region. New criteria are necessary, where new knowledge of the effects of atmospheric deposition, geological sources, organic acids and different anthropogenic sources are taken into consideration.

According to Ivarsson and Jansson (1994a) a large part of the atmospheric SO4* depositon is retained in the catchments and there is a great variability in background SO4* on the regional and local level. There are even differences between catchments. The results in the study show that acid conditions in streams in central-northern Sweden, mainly has natural causes. Organic acids are the most important single source followed by sulfate from geological sources while the impact from acid atmospheric deposition is small. The background level is $2-10$ times higher then if the two most recent models for calculation of geological background concentrations in Sweden (Monitor 1991 and Wilander 1994) are used.

In Ivarsson and Jansson (1994b) the River Öre catchment in northern Sweden has been studied. Geological sulfate sources dominated in most parts of the catchment and are particularly high in streams draining earlier marine sediments with high

content of sulfides, and in streams on gneiss bedrock due to frequent sulfide horizons in the gneisses. The contribution from antropogenic sources ranged from a few to 15 %. The retention of the anthropogenic atmospheric SO4* depositon was around 90 % in the upper part of the catchment and 55 % close to the coast.

More knowledge is needed however about the conditions when the snow is smelting during spring period.

There is an obvious need for a better understanding of the links between critical load and biological effects in lakes and streams, thereby improving the precision of critical load concept. This is very important in a long-term perspective and in combination with different land-use practices, for examle forestry. Since running waters are positioned in the immediate interface between terrestrial and aquatic ecosystems, studies of the interplay beteen the two, in terms of chemistry and biology, will provide the earliest possible information on anthropogenic effects.

Biological diversity must come into focus when developing measures for environmental monitoring.

The following aim has been presented by the SEPA: "Biological diversity and genetic variation are to be secured. Flora and fauna are to be preserved so that sustainable populations of species naturally occurring in Sweden are able to survive under natural conditons."

Biodiversity is reduced daily and the velocity of the reduction has accelerated during the last decades. Forestry and agriculture are responsible for the largest lossses of biodiversity on a species level. The protection af biological diversity must be evident aims in all sections of the technosphere. The understanding of the conditions of nature must exist in all human beings in regard to their way of life and work as well as in different social structures.

Some species have already reached the critical limit in Sweden ie otter, freshwater pearl mussle, eagle and in the Baltic, natural salmon.

It is important that systems for monitoring biodiversity are developed and established, not only in order to preserve certain valuable areas, but also to maintain biodiversity in the every day landscape. It would be favourable to find simple and manageable biological measures of biodiversity, not only on species level but also other measures in the landscapes with forest, agriculture lakes, rivers, coasts and sea. Environmental or biological index, or indices used in an environmental profile have been suggested and this could function as a base for presenting critical limit of, for instance forestry. Simple measures of biodiversity are necessery to facilitate

and create understanding of the work with the environment and in the comprehensive planning activities.

Besides using laws, ordinances, environmental aims, critical load and international work, information is a very important part of the work with the environment. Knowlegde of risks and critical load will lead to voluntary involvements by business, different sectors and private persons. This will lead to more political activities which result in better accomplishments than it would have been possible without information. In this context, the work on the local level and Agenda 21 is very important.

SEPA has developed a system of guidelines for quality criteria in lakes and rivers. This system is often used and it needs to be reviewed, especially the acidification criteria. There is also a need for quality criteria for forest and agriculture, groundwater and coast or sea.

It would be preferable if there is an unanimity without too many questions when using terminology and ie biological methods.

A critical load concept will be a very useful tool in the work with the environment and when contributing ecological aspects to the master planning of the municipality. Critical loads can provide a basis for defining aims and timerelated targets. The targets must be relieble on all levels when presented, otherwise there is a risk that they will have the opposite effect than what was intended.

As the new environmental monitoring still is in an initial phase, it will take time to find the crical loads for different subjects or the thirteen threats. Environmental norms will probably be formulated even further into the future.

Positive results have not always been achieved when working with new tools as for example the Planning and Building act, the act for the Management of Natural Resources and Environmental Impact assessment, but it is obvious that the work with critical loads and critical limits will result in raising at least one backgroundlevel, namely the backgroundlevel of knowledge.

References

Ivarsson, H. & Jansson, M. 1994a. Sources of acidity in running waters in central northern Sweden. Water Air Poll. In press.

– & Jansson, M. 1994b. Sources of streamwater sulfate in the River Öre catchment, northern Sweden. Submitted to Appl. Geochem.

Monitor 1991. Acidification and liming of Swedish freshwaters, Swedish Environmental Protection Agency Informs 12, p 144.
Wilander, A. 1994. Water, Air and Soil Pollut. In press.

Critical loads and critical limit values from the standpoint of the wood and paper industry

Reino Lammi

Wisaforest Oy Ab
Box 42
FIN-68601 Pietarsaari, Finland

General

The air emissions of the chemical wood industry arise from pulp digesting and bleaching as well as from producing energy and chemicals. The main sources of emission are recovery boilers, lime kilns, combustion boilers for malodorous gases, the handling of vapours from the bleaching plant, the waste wood digester and the so-called diffuse emissions.

The main air emissions from the pulp mire are:
- carbon dioxide
- sulphur dioxide
- nitrogen oxides
- odorous sulphur compounds (hydrogen sulphide, methylmercaptan, dimethyl sulphide, dimethyl disulphide)
- particles (sodium sulphate, calcium hydroxide, calcium carbonate)
- chlorine dioxide

Sulphur dioxide and nitrogen oxide are acidizing emissions. Malodoroussulphur compounds again are a factor affecting peoples' comfort. Particles emitted cause dirt. Sodium sulphate is a water soluble neutral salt and calcium an acid neutralizer.

As far as energy is concerned the pulp factory is self-sufficient. The paper industry and the mechanical forest industry have to get their energy from outside sources, so the quality of their emissions is depending on the way the energy is produced.

From the beginning of the 1980s, the Finnish forest industry has invested large sums of money into protecting the environment by technical measures and by building purificating systems. Emissions of gaseous sulphur compounds (SO_2 + TRS) have fallen from 52,000 tons in 1980 to 7,200 tons this year, or to little more than a tenth. Today, the share of the sulphurous emissions from the pulp industry

is only 12 % of the total sulphurous emissions in Finland. The nitrogen oxide emissions in Finland are about 250,000 tons annually. The nitrogen oxide process emissions derived from the forest industry are 7,200 tons or about 3 % of the total nitrogen oxide emissions. Correspondingly, the nitrogen oxide emissions from the energy production of the forest industry amount to 12,000 tons annually or about 5 % of the total nitrogen oxide emissions. Both in the forest industry and in the country as a whole, the nitrogen oxide emissions have remained the same for the last fifteen years.

The acidity deposition and critical limit values

Sulphur and nitric emissions cause acidity in the environment. The emissions are transported long distances together with the currents of air from one country to another and are diluted by the air. Every ton of sulphur and nitrogen discharged in the air returns undiminished to the nature. In Fig. 1 the effects of all the nitrogen and ammonia emissions are presented in a diagram.

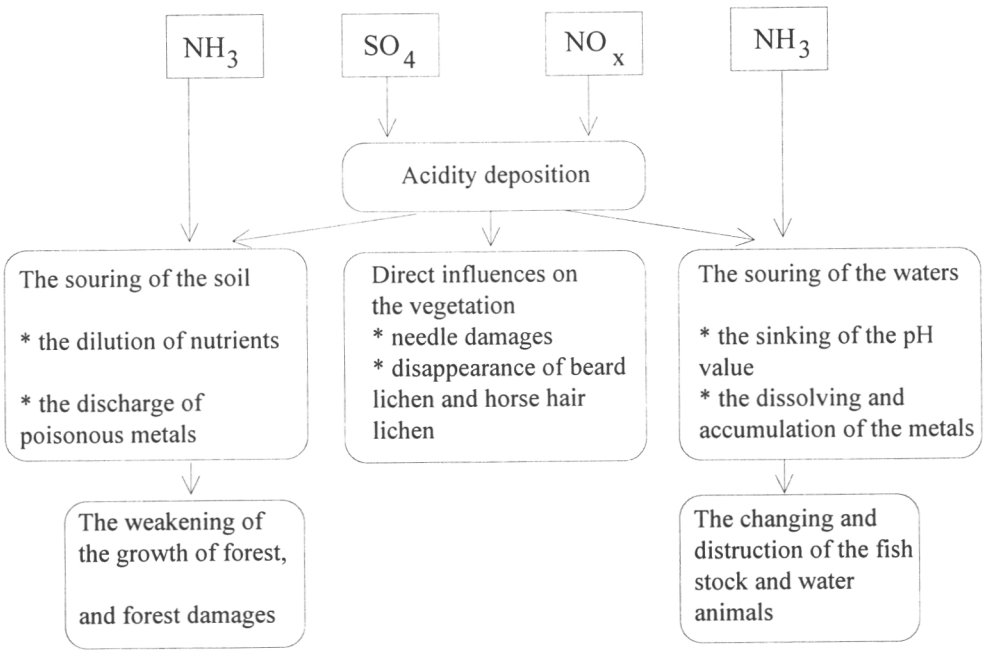


Figure 1. The influences of acidity deposition on the environment.

The values of the critical load limits change radically from one terrain to another – locally and regionally – within the country. The tolerance of the environment depends on climatic conditions and the levels of resistance of the soil.

In Finland there are maps indicating the critical load values. These maps are suitable for an overall review of the situation, but they are perhaps not appropriate when estimating the effect the reduction of the local emissions has had on the depositions and when the limits of the critical load have been exceeded.

It is estimated that the critical stress point of sulphur fallout for the ecological system of the Finnish forest is no more than about half a gram per square metre annually or less. This value is exceeded in Southern Finland and in Eastern Lapland. In Central Finland and in large areas of Northern Finland the sulphur fallout is between 0.3 and 0.5 grams per square metre annually. The best situation is in Western Lapland. The critical value of the nitrogen deposition is difficult to determine, because vegetation is able to bind nitrogen compounds as its biomass at least at the beginning.

It is important to know the critical load values and the real deposit values so that the appropriate measures can be taken. How are these going to be reached in Finland as our own share of the total deposit is only twenty percent ? International co-operation is quite necessary here.

The forest industry's share of our own deposit was just over ten percent, which would be only a few percent of the total deposit in Finland. At the moment the average specific emissions of gaseous sulphur compounds is about 2.5 kg SO_2 per ton of pulp. The present measures are aimed at reducing the sulphurous emissions to under two kilograms sulphur dioxide per ton of pulp.

Critical loads to the Gulf of Bothnia

Kalervo Mäkelä and Matti Perttilä

Finnish Institute of Marine Research
P.O. Box 33
FIN-00931 Helsinki, Finland

Abstract

A joint Finnish-Swedish research programme was carried out in 1991–1994, focusing on the marine environment of the sea area. This programme, called the Year of the Gulf of Bothnia, aims at the description of the state of the Gulf of Bothnia, a listing of its threats, understanding its physical and ecological processes, and at a prediction of its development under different stress conditions.

The projects in the different disciplines of the joint programme contain elements that can be used to evaluate the material balance of the main plankton nutrients in the Gulf of Bothnia. While the monitoring of the chemical and biological conditions has been continued for several decades, the data thus obtained can only reveal changes that have already taken place. The only way to look into the future of any sea area, and also to evaluate the effects of discharges, and estimate what are the critical loads, is through the construction of an ecological model. This has been one of the principal targets of the Year of the Gulf of Bothnia.

By means of an inspection of the effectiveness of the different critical processes controlling the material balance of the nutrients, a long-term forecast was formed for the Gulf of Bothnia under different environmental conditions. In the case that at least half of the Helsinki Commission recommendations of nutrient discharge reductions can be fulfilled, it will take 30–40 years before the nutrient concentrations decrease to the level where they were in the early 1970s. In the case that the nutrient discharges continuously increase, the plankton production would probably increase strongly in the Bothnian Sea, but only moderately in the Bothnian Bay because of the present phosphate deficiency. Silicate concentrations would decrease markedly, and silicate might become gradually the limiting nutrient in the Gulf of Bothnia.

If assessed by the increasing plankton production, the critical level of nutrient discharges into the Gulf of Bothnia has already been achieved. However, the Helsinki Commission recommendations concerning the discharge reductions will be difficult to fulfill, because 20–30 % of the loading comes through the atmosphere, and some 30–40 % from agriculture through rivers. The share of the riverine communities and industries is only some 20 % of the total load.

Introduction

One of the principal aims of environmental monitoring activity is to find out long-term changes in the environment. Water quality results obtained through monitoring only represent the situation at the time of the sampling and in relation to earlier data. However, there is a strong demand from the society to the environmental scientists not only to describe the present-day situation, but also to produce estimations of future environmental conditions. This demand for prognostic evaluations is increased because of the need to assess the cost/effect consequences of environmental measures. The only way to fulfill this task is through a combination of monitoring work, process-oriented research, and ecosystem modelling research. Monitoring strategy should be designed by means of the results of research work so that representative long-term data are obtained, research resources should be directed to provide information on the physical, biological and chemical processes controlling the behaviour of the monitored parameters. Water quality monitoring data and process information should then be combinated into an overall mass balance model which can be used to evaluate the development of the water quality, as well as to indicate where are the needs for additional monitoring and research.

In 1971, Finland and Sweden set up a special marine environmental committee, called the Committee of the Gulf of Bothnia, with the task of coordinating and compiling research and environmental monitoring in the marine area of the Gulf of Bothnia. This Committee set up in 1989 a joint Finnish-Swedish steering group to plan, coordinate and carry out a major study programme with the aim of assessing the state and estimating the future of the Gulf of Bothnia, and of obtaining a better understanding and knowledge of the ecological system of this sea area.

The project "The Year of the Gulf of Bothnia", GBY91, conducted in 1991, represents a unique bilateral effort to describe the basic characteristics and controlling factors of this sea area. In the following, we will go through the organizational aspects of the joint project, and illuminate some of the main findings of the study sectors.

The participation in the project took place through several research institutes and universities on both sides of the Gulf of Bothnia. In Finland, main efforts were given by the Finnish Institute of Marine Research, which also set its research vessel Aranda at the disposal of the project, National Board of Waters and Environment, and several institutes of the University of Turku. In Sweden, the main contributors were the Swedish Meteorological and Hydrological Institute, Swedish National Environment Office, an several institutes at the Stockholm and Umeå universities.

While the both countries had carried out monitoring in the Baltic Sea since the 1960s, using in principle similar methods for sampling and analysis, numerous difficulties have always appeared when interpreting the data. Because of the strong seasonal variability of the monitoring variables in the Baltic Sea, and especially in its northern parts, and because of systematic and random differences between the results from different laboratories, trends are difficult to establish. The station network, while dense in certain parts of the area, has been unevenly distributed, making it difficult to obtain an overall view of the distribution of the nutrients and toxic substances.

The second requirement, estimating the future of the Gulf of Bothnia, made it necessary to include physical, biological and chemical process-oriented studies in the programme. These included a close surveillance of the monthly discharge variations, sea-level, fresh water input, and salinity variations (for the calculation of the water exchange between the basins), sediment and sedimentation studies and several biological studies.

Eutrophication in the Gulf of Bothnia

The eutrophication of the Gulf of Bothnia in relation to the other sea areas in the Baltic Sea can be considered by comparing the late winter nutrient concentrations. At this time of the year, discharges, transport and mixing of the nutrients have led to the maximum nutrient concentrations in the surface water, for use for the plankton production in the spring. The development has been assessed by the Helsinki Commission for the years 1984–1988 (HELCOM 1991). Nitrate concentrations are the highest in Skagerrak, Gulf of Finland and in the Bothnian Bay. The highest phosphate concentrations are found in Skagerrak, Baltic proper and in the Gulf of Finland, but they diminish strongly through the Gulf of Bothnia, reaching the lowest values in the Bothnian Bay. The Bothnian Bay resembles an oligotrophic, phosphorus-limited lake, whereas the other parts of the Baltic Sea are, from the plankton production point of view, either in balance with respect to nitrate and phosphorus, or more nitrate-controlled (Wulff et al. 1994a).

The plankton production consumes up nitrogen and phosphorus nutrients in a constant weight proportion of 7:1. Thus the relation of the concentrations of the inorganic nutrients (nitrate, nitrite and ammonia) to phosphate (the N:P relation) gives an indication which nutrient in the water mass is present in deficient concentrations, and will control the production. In 1987–91 the average winter surface water concentrations in the different basins were:

	N:P
Bothnian Bay	47
Bothnian Sea	9
Baltic proper	4
Gulf of Finland	8
Kattegat	6
Skagerrak	9

The spring production bloom consumes practically all nitrogen in all the Baltic Sea basins, with the exception of the Bothnian Bay.

After the production peak in the spring, organic material slowly settles down to the sea bottom, thus removing nutrients from the biological cycle. However, some nitrogen and phosphorus are released back in the water by the breakdown of the plankton during the sedimentation process. Some of the released nitrate-nitrogen is lost in the process of denitrification, which removes nitrate from the biological cycle in the form of nitrogen gas. Some of the released phosphate is also bound and lost permanently in the form of inorganic complexes, but in general phosphate is more available than nitrate during the summer period in all sub-basins except for the Bothnian Bay. The Bothnian Sea is an intermediate area. Also water exchange transports large amounts of nutrients in the Gulf of Bothnia, but the nitrate, transported from the Bothnian Bay into the Bothnian Sea, is largely balanced by the phosphate, transported from the Baltic proper.

The above controlling mechanism does not apply to the blue-green algae production, that takes place in the later summer in the Gulf of Bothnia, Gulf of Finland and in the Baltic proper. The blue-green algae uses the atmospheric nitrogen as a direct nitrogen source, and its growth is thus totally phosphorus-controlled.

Nutrient discharges

The second Pollution Load compilation of the Helsinki Commission (HELCOM 1993) shows the overwhelming phosphorus and nitrogen contribution from rivers and industries, whereas that from municipalities is almost negligible.

Observed trends

Since the early 1970s, the state of the Baltic Sea has been followed by chemical and biological methods. In most areas of the Baltic Sea, the general observation

has been that of increasing nitrate concentrations. The increase has been rapid in the Bothnian Sea all through the 1970s. However, during the 1980s the situation seems to have stabilized at a relatively high level.

Nutrient distribution in the Gulf of Bothnia

The surface distributions of the nutrients in the Gulf of Bothnia was investigated during the programme of the Year of the Gulf of Bothnia in 1991 (Wulff et al. 1994a).

Phosphate concentrations in the Bothnian Bay are very small all through the year. Only in the Bothnian Sea phosphate is accumulated in the surface layer in appreciable amounts during the winter. Phosphate concentrations decrease rapidly in the surface water in the spring, followed by a stable period of very low concentrations. The recovery takes place slowly in the autumn.

The geographical distribution of the nitrate concentrations shows an opposite tendency as compared to phosphate, with the highest concentration up north and the lowest in the south, at least during the major part of the year. The intermediate role of the Bothnian Sea is evident, even though the seasonal changes are clear in all basins.

The silicate development follows that of nitrate, though being considerably less pronounced.

Description of the model for nutrient cycling

Combining the discharges and removal processes of the substances together with the inflow or outflow of materials due to the water exchange between the Bothnian Bay, Bothnian Sea and the Baltic proper, it is possible to make a calculation to simulate the seasonal and annual variations of the average concentrations (Wulff et al. 1994a), by simulating the Gulf of Bothnia and the Baltic Proper as a series of three consecutive basins, coupled together by water exchange.

Development of eutrophication in the Gulf of Bothnia

While the nutrient discharge loads of the past can be estimated by means of the development of agriculture, industry and communities, we can only sketch the future discharges using a very wide scope of assumptions. One of the political targets of the Helsinki Commission is to reduce the nutrient loading of the Baltic

Sea by 50 %, as compared to the 1987 values, by 1995. This aim is probably extremely difficult to obtain, bearing in mind that the share of discharges directly accessible to reductions (point sources) is only 5–6 % of the total loading in the case of nitrogen, and ca. 15 % in the case of phosphorus. Thus a somewhat weakened assumption of a 25 % discharge reduction has been as a starting point to build one scnerario for the future.

As two more realistic starting points, we have used the discharge estimates of the Swedish Environment Protection Agency (see Wulff et al. 1994b). By means of consideration of the development of the different discharges under controlled and uncontrolled situations, minimum and maximum estimates total nutrient discharges have been reported.

At the 25 % discharge reduction level, the phosphorus concentrations in the Baltic proper and the Bothnian Sea start to decrease slowly. In 2020, the Baltic proper reaches the 1970 level, and the Bothnian Sea reaches the level of the 1980s. The reduction does not affect the Bothnian Bay phosphorus content, which is anyway kept at minimum level by means of the overwhelming nitrogen content. The 25 % discharge reduction would lead to a nitrate level corresponding to some 75 % of the present level. Silicate contents stay unchanged.

It may be a little disappointing to note that even in the drastic and over-optimistic case of 25 % nutrient discharge reductions, it would take a long time for the marine environment to recover from the present state of high productivity. On the other hand, it shows that changes can be made.

If the present level of discharges is maintained, phosphate concentration will rise continuously, whereas the nitrate concentrations will stabilize. The plankton production will increase with the increasing phosphate concentration.

On the basis of the discharge estimates, both phosphorus and nitrogen contents will be increasing. The rise is most pronounced in the Baltic proper, but almost insignificant in the Bothnian Bay, because of the already high nitrogen surplus in the area. As the silica discharges are of natural origin, they will remain at a constant level, and as a consequence the silica contents will start to decrease in all sea areas because of the increasing production.

Concluding remarks

The Gulf of Bothnia is a marine ecosystem in part separated from the rest of the Baltic Sea. Its northern part, the Bothnian Bay is phosphorus-limited, with high nitrogen inflow from the rivers. The southern part, the Bothnian Sea, is roughly in

balance with respect to the nutrients. Because of the high loading from rivers, atmosphere, the Bothnian Bay and the Baltic proper, its coastal areas are eutrophied.

In the case of nutrients, the meaning of the term "critical load" is difficult to define. In a way, the critical level of discharges has been already attained, because the nutrient concentrations have been increasing during the past two decades. On the other hand, the primary production is enhanced by this increase, which then balances off the nutrient surplus. However, if the nutrient concentrations show a clear inbalance, as is the case in the Bothnian Bay, even an increase in the phosphate loading does not increase the phosphate concentration in the water, while even a modest increase in the nitrate loading remains in the water.

Even a drastic 25 % reduction of total nutrient loading would only lead into a slow recovery. By the year 2020 the environmental conditions would roughly correspond to those in the 1970s. A more realistic estimation of the development of the discharge amounts leads into a scneario of further eutrophication, with also the Bothnian Sea turning into a phosphorus-controlled ecosystem.

Because of the minor role of the point source discharges, their further reduction would not contribute to the recovery of the Gulf. Main emphasis should be laid to the reduction of discharges from the drainage area and to the reduction of atmospheric inputs. This would call for international cooperation with the overall aim of reducing the use of nutrients, both nitrogen and phosphorus, in forestry and agriculture.

References

HELCOM. 1991. Second Periodic Assessment of the Baltic Sea. Helsinki Commission.

– 1993. Second Pollution Load Compilation. Helsinki Commission.

Wulff, F., Perttilä, M. & Rahm, L. 1994a. Mass-balance calculations of nutrients and hydrochemical conditions in the Gulf of Bothnia, 1991. Aqua Fennica (in print).

– , Perttilä, M. & Rahm, L. 1994b. Bottniska Vikens tillstånd och framtid / Pohjanlahden tila ja tulevaisuus. Skärgård 1(1994): 67-80.

Critical loads for surface waters in Scandinavia and the Kola region

Arne Henriksen[1)], Juha Kämäri[2)], Maximilian Posch[2)], Martin Forsius[2)], Anders Wilander[3)] and Tatyana Moiseenko[4)]

[1)] Norwegian Institute for Water Research (NIVA)
P.O. Box 173 Kjelsås, 0411 Oslo, Norway
[2)] Water and Environment Research Institute
P.O. Box 25, 00101 Helsinki, Finland
[3)] Department for Environmental Assessment
Swedish University of Agricultural Sciences
P.O. Box 7050, 750 07 Uppsala, Sweden
[4)] Institute of North Industrial Ecology Problems (INEP)
Kola Centre of the Russian Academy of Sciences
14 Fersman Str. 184200 Apatity, Murmansk region, Russia

Abstract

The Nordic countries Finland, Sweden and Norway are particularly affected by acidification of surface waters due to long range transported air pollutants. The Nordic Council of Ministers has supported several projects dealing with critical loads for surface waters in the Fennoscandian countries. On the basis of national regional lake surveys critical loads of acidifying deposition for surface waters have been mapped both nationally and in the EMEP-grid system (150 by 150 km) and its 3 by 3 subdivision (50 by 50 km) (NILU-grids). Frequency distribution curves for calculated critical loads showed a wide variation in sensitivity *within* and *between* the EMEP-grids and the NILU-grids and also between the three countries. Norway appears to have surface waters that are the most sensitive to acidic precipitation in the Nordic countries. Critical load exceedance for sulphur was estimated from available sulphur deposition data. Depending on the chemical criteria used, the exceedance varied from 20 to 70 percent of the area in the Nordic countries. A recent developed model for analyzing the exceedance and the reductions needed for both N and S depositions indicate that practically in the whole of Finland and in the northern parts of Fennoscandia the acidification problem could be solved by reducing S deposition only. In the southern part of Sweden some reductions in N deposition are clearly needed in addition to those for S. In the southern parts of Norway even strong measures to reduce S deposition are not enough, also N deposition has to be reduced considerably. Critical loads and their exceedances have been for 370 lakes sampled in the Kola peninsula. These data they indicate that the surface waters in the Kola North are highly sensitive to acidic deposition. Critical loads of less than 20 $meq \cdot m^{-2} \cdot yr^{-1}$ occur in about 50 percent of the area using the minimum values for NILU-grids. The critical load values fit well with the corresponding values in the border areas with Norway and Finland. Finland, Norway and Sweden are

planning to carry out co-ordinated regional lake surveys in their respective countries in the fall of 1995. The data collected by these surveys will form a new updated basis for assessing critical loads and their exceedances for surface waters in this area. It would be very useful also to incorporate a corresponding survey of the lakes of the Kola peninsula.

Introduction

Critical loads have been widely accepted as a basis for control strategies for regional air pollution. In order to gain more insight in the magnitude and spatial variation of critical loads, the UN ECE Excecutive Body on Long Range Transboundary Air Pollution has set up a Task Force on Mapping Critical Levels/Loads under the Working Group on Effects. The data from individual countries are collected, mapped and reported by the Coordination Center for Effects, located at the National Institute of Public Health and Environmental Protection (RIVM) in Bilthoven, the Netherlands.

The starting point for the scientific work on critical loads of sulphur and nitrogen was established at a workshop organized by the Nordic Council of Ministers in 1986 in Sundvollen, Norway. The workshop report (Nilsson 1986) provided for the first time agreed definitions for critical loads and the first estimates for critical loads of sulphur and nitrogen for forest soils, groundwaters and surface waters. As a follow up of the Sundvollen Workshop, the first workshop held under the auspices of the UN/ECE Convention for Long-Range Transboundary Air Pollution was sponsored and organized by the Nordic Council of Ministers at Skokloster, Sweden, in 1988 (Nilsson and Grennfelt 1988).

As the role of nitrogen in the acidification of soil and surface waters gained increasing attention at the end of the 1980s in both the scientific and political arena, a workshop was organized by the Nordic Council of Ministers and the U.S. Environmental Protection Agency on that topic in Copenhagen in 1988 (Malanchuk and Nilsson 1989). The purpose of that workshop was to review the state of science on the role of nitrogen in the acidification of the environment. The foundation for the actual mapping of critical loads in the ECE countries was laid in the UN/ECE workshop on mapping critical levels and loads held in 1989 in Bad Harzburg, Germany. The mapping methods were documented in a draft manual (ECE 1990). The definitions and methods presented were largely based on the results from the Skokloster workshop. Furthermore, a workshop on critical loads for nitrogen was organized by the Nordic Council of Ministers in Lökeberg, Sweden in April 1992 (Grennfelt and Thörnelöf 1992).

Since 1985 the Nordic Council of Ministers has supported projects dealing with critical loads of air pollutants. Three project have been dealing with critical loads

for surface waters in the Fennoscandian countries. In 1990 a project on inter- and intra-regional variability of critical loads was carried out (Henriksen et al. 1990). In 1992 the variability project was followed by project aiming at deriving critical loads of nitrogen and sulphur and providing the means for assessing target loads on the basis of these critical loads (Henriksen et al. 1993). Further, the working group on Air Pollution in the Northern Fennoscandia ("Luftforurensninger på Nordkalotten") asked the Norwegian Institute for Water Research (NIVA) to organize and carry out a cooperative research project on the status of critical loads of atmospheric sulphur for surface waters in this area, with participation of spesialists from Sweden, Norway and Finland (Henriksen et al. 1994). Under the Environmental Treaty between Russia and Norway, established in 1988, several joint projects between russian and norwegian scientists have been carried out. Critical loads and critical load exceedances have been calculated for the border area between Russia and Norway, and the Institute of the North Industrial Ecology Problems (INEP), Kola Science Centre of the Russian Academy of Sciences, Apatity, Russia has extended the studies to include the whole Kola Peninsula.

This paper will present a summary of the results from the cooperative critical load work carried out in the Fennoscandian countries with support from the Nordic Council of Ministers, and will also include data from the Kola peninsula in the evaluation of the acidification status of surface waters in Scandinavia and the Kola region.

Critical load maps for Fennoscandia and the Kola peninsula

Fennoscandia

Methods and data used for Fennoscandia

Critical loads are based on estimates of weathering rates. For surface waters present water chemistry is used to calculate the integrated weathering rate for a lake catchment. The Steady State Water Chemistry method (SSWC) can be used for this purpose (see Henriksen et al. 1992). National lake surveys to assess the acidification status of surface waters have been carried out in Norway, Sweden and Finland (Henriksen et al. 1988, Bernes 1991, Forsius et al. 1990). These databases have been used for national critical load calculations and mapping, and for a common assessment for the three Nordic countries (Henriksen et al. 1990). In order to determine a critical load, a critical chemical value for a biological indicator has to be set (Henriksen et al. 1990). For surface water this was done by setting a limit for ANC (ANC_{limit}) (ANC = Acid Neutralizing Capacity). It has been agreed to use a value of 20 μeq/l for the Fennoscandian countries (Henriksen et al. 1990).

Different grid systems have been used for mapping deposition and environmental impacts. The ECE-manual used for mapping critical loads of acidity in Europe (45) does not prescribe the map resolution and mapping unit in order to allow each country to work on the scale appropriate for its own data. However, it recommends using the 1° longitude by 0.5° latitude grid (about 50 x 55 km^2 in central Europe), or the EMEP-grid (150 x 150 km^2), or uniform subdivisions of these grid systems. Several institutions use a subdivision of the EMEP-grid into 3 by 3 subgrids (NILU-grids) for deposition calculations. The Nordic maps were made on the basis of this subdivision, since this gave a reasonable areal resolution in relation to the number of lakes sampled.

Critical Load Maps

Local conditions, such as geology, soil composition and thickness, and other factors will determine the critical load for the area studied. These factors influence the cumulative frequency distribution of critical loads for lakes within each Nordic EMEP-grid. There was a wide variation in sensitivity *within* and *between* the EMEP-grids and also between the three countries. Within a grid the critical load varied from zero to more than 100 meq m^{-2} yr^{-1}. Norway appears to be the country with surface waters most sensitive to acidic deposition, especially the southern part of the country. The most sensitive EMEP-grid is the 17/20 grid in southernmost Norway. Here, for about 30 % of the area the critical load for surface waters is zero, and for most of the grid the critical load is below 100 meq m^{-2} yr^{-1}. The highest sulphur deposition in the Nordic region is 80−90 meq m^{-2} yr^{-1}. If the whole of the area received this amount of deposition, the critical load would have been exceeded in 60−70 % of the area. This clearly shows that in this part of the world the surface water is very sensitive to acidic deposition.

Cumulative distributions gives the critical load in a condensed form. It is, however, difficult to communicate to the public with this kind of presentation. Another approach is to select a single value to represent a whole grid and then present different critical load classes with different colors. The question then becomes which value to select from the range of critical loads occurring within a grid square. Henriksen et al. (1990) illustrated the problem by displaying maps with NILU-grids (50 by 50 km), using the minimum, the 25-percentile and the median critical load for sulphur for each NILU-grid. Depending on the percentile chosen, the critical load was less than 50 meq m^{-2} yr^{-1} in 80 % to 25 % of the area covering Norway, Sweden and Finland. This shows the influence of the selected "prerequisites" when preparing one-valued grid maps.

For Europe, 1- and 5-percentile maps have been presented based on EMEP-grids (Downing et al. 1993). These maps displayed the range of critical loads covering

1 % and 5 %, respectively, of the area in each EMEP-grid, and they showed that the whole of the Nordic area falls in the lowest critical load class used for these maps. For the Fennoscandian countries we have chosen to present the results as *5-percentile* NILU-grid maps. Even with the NILU-grid resolution most of the grids located in Norway fall in the lowest critical load class (< 20 $meq \cdot m^{-2} \cdot yr^{-1}$). For Sweden and Finland most of the grids fall in the two lowest critical load classes (< 50 $meq \cdot m^{-2} \cdot yr^{-1}$). In fact, the 5th percentile value for NILU-grids will for most of the grids be the most sensitive lake, because there will mostly be less than 20 lakes located in each NILU-grid.

The magnitude of the critical load exceedance value for a NILU-grid largely depends on which critical load value was selected to represent the grid. Using the minimum critical load value for each grid, the critical load was exceeded in about 70 % of the Nordic area, while using the median gave a critical load exceedance area of only 20 %. The 95th percentile critical load exceedance value was selected to represent each grid. Because the 95 percentile selection in most grids will be the lake with the minimum critical load, the critical load was exceeded in about 70 % of the area in the Nordic countries.

Critical loads for the "Nordkalotten"

As part of the activities of the "Nordkalott project" an assessment of critical loads and critical load exceedances of the surface waters in the "Nordkalott" area (north of the polar circle) was conducted (Henriksen et al. 1994).

The main conclusions of the project were: The acidity of the deposition in the "Nordkalott" area is generally lower than in the southern parts of the Fennoscandian countries. Only in the areas of the smelters on the Kola peninsula the deposition is comparable to those found in the south. Geological and hydrological conditions regulates the critical load value for a given lake or stream, thus both high and low values are found within the area and the levels are very similar to those found for the whole of Fennoscandia. The results demonstrate large spatial variations in critical loads, and sensitive areas are found in scattered areas all over the "Nordkalott" area, especially in the northeast close to the Russian border. The highest exceedances of the critical load are found in these areas. The exceedances are, however, lower than in the most affected areas in Sweden and Norway. An exceedance of the critical loads occurred in about 11 % of the observations in the "Nordkalott" area. For all three countries as much as 35 % of sites are exceeded, and in the most affected areas, in southernmost Norway, almost 100 % of the area is exceeded.

Northern Fennoscandia will respond much faster to reduced deposition than the highly affected areas in the southern parts of Fennoscandia, because the northern areas have been subjected to acidic deposition for a much shorter time than the southern areas. Since there are rather definite plans for reducing emissions from the smelters at the Kola peninsula, the present negative trends in lake water chemistry should turn to positive trends in the near future.

Critical loads for sulphur and nitrogen in Fennoscandia

The third Nordic cooperative project dealt with exceedance of critical loads for lakes in Finland, Norway and Sweden: Reduction requirements for nitrogen and sulphur deposition. In this project two steady-state approaches for assessing exceedance of critical loads, one empirical and one process-oriented were applied to surface water in the three countries. The first approach, a modified Steady State Water chemistry (SSWC) method, enables the calculations of critical loads of acidity and, based on that, present exceedances of incoming total acidity (both sulphur and nitrogen). While the SSWC-method is restricted to the present situation, the second approach, the First-Order Acidity Balance (FAB) model and its extention attempts to account for those processes considered to be relevant for the overall acidity balance of the catchment in a steady-state situation. The FAB model allows the evaluation of the effects of any scenario for S and N deposition.

The leaching of nitrate is today occurring in southernmost and western Norway and in southern Sweden. In the remaining parts of Fennoscandia the leaching is negligible. This leaching contributes to the critical load exceedance and contributes significantly in southernmost Norway.

The FAB model is a tool for analyzing the exceedance and the reductions needed for N and S depositions. The question of which pollutant has to be reduced in order to achieve critical loads is approached by an exceedance function. The results from this analysis show a clear regional pattern in the S and/or N exceedance. Practically in the whole of Finland and in the northern parts of Fennoscandia the acidification problem could be solved by reducing S deposition only. In the southern part of Sweden some reductions in N deposition are clearly needed in addition to those for S. In the southern parts of Norway even strong measures to reduce S deposition are not enough, also N deposition has to be reduced considerably. Reductions in N deposition are necessary in regions where also the present N leaching values are elevated.

The methodology applied allows for the simultaneous analysis of N and S reduction requirements by optimization models, computing cost-effective ways to reduce N and/or S deposition. Alternatively, the two pollutants can be assessed

separately by fixing the one pollutant to a level defined, e.g. to the new sulphur Protocol that was signed in Oslo in June 1994. The reductions required for N deposition can then be evaluated on the basis of the scenario given by this protocol.

Critical loads and critical load exceedance in the Kola North in Russia

Precipitation falling on the Kola North (Russia) and Eastern Finnmark (Norway) is severely polluted by strong acids and heavy metals due to emissions from large nickel smelters located in Russia. Large areas in the region are sensitive to acid precipitation due to the climatological and geological conditions. Extensive chemical analyses of lake water samples collected in the border areas show that extensive surface water acidification has taken place, particularly on the Norwegian side of the border (Traaen et al. 1991). On the Russian side of the border pollution of the lakes by nickel and copper is more severe than acidification, especially in the vicinity of the large smelters. The more distant areas in the Kola North and endangered by acidification if emissions are not reduced.

The critical load maps in the European part of Russia presented to the Task Force on Mapping have been based on calculations from monitoring data for lakes and rivers with large catchments and largely impacted by sewage effluents. INEP has collected water samples from 370 lakes (area $0.2-10$ km^2) in the Kola Peninsula during the years 1990 to 1992. The samples were taken from the outlets of headwater lakes in late autumn and winter by similar methods as in Norway and Finland. Results from the chemical analyses were intercalibrated with norwegian and finnish laboratories.

The calculations of critical loads and critical load exceedance were carried out as in the Fennoscandian countries, using the Steady State Water Chemistry Method (Henriksen et al. 1990). An ANC_{limit} of 20 μeq/l was used in order to obtain comparable results with the three Fennoscandian countries.

Data for sulphur deposition in the Kola Peninsula is at present only available for the near border area with Norway. Here, there is a good correlation between sulphur deposition and non-marine sulphur in the runoff water ($r = 0.80$, $N = 36$). On the basis of this empirical relationship, sulphur deposition was calculated for the whole Kola region and used for the exceedance calculations.

Results

We have compared the results from the russian calculations with those previously carried out for Fennoscandia (Figs. 1 and 2).

Looking separately at the data for the Kola peninsula we find that the surface waters in the Kola North are highly sensitive to acidic deposition. The results show that critical loads of less than 20 $meq \cdot m^{-2} \cdot yr^{-1}$ occur in about 50 percent of the area using the minimum values (Fig. 1). Low median values are found in the northern and eastern parts of the peninsula. The critical load values fit well with the corresponding values in the border areas with Norway and Finland.

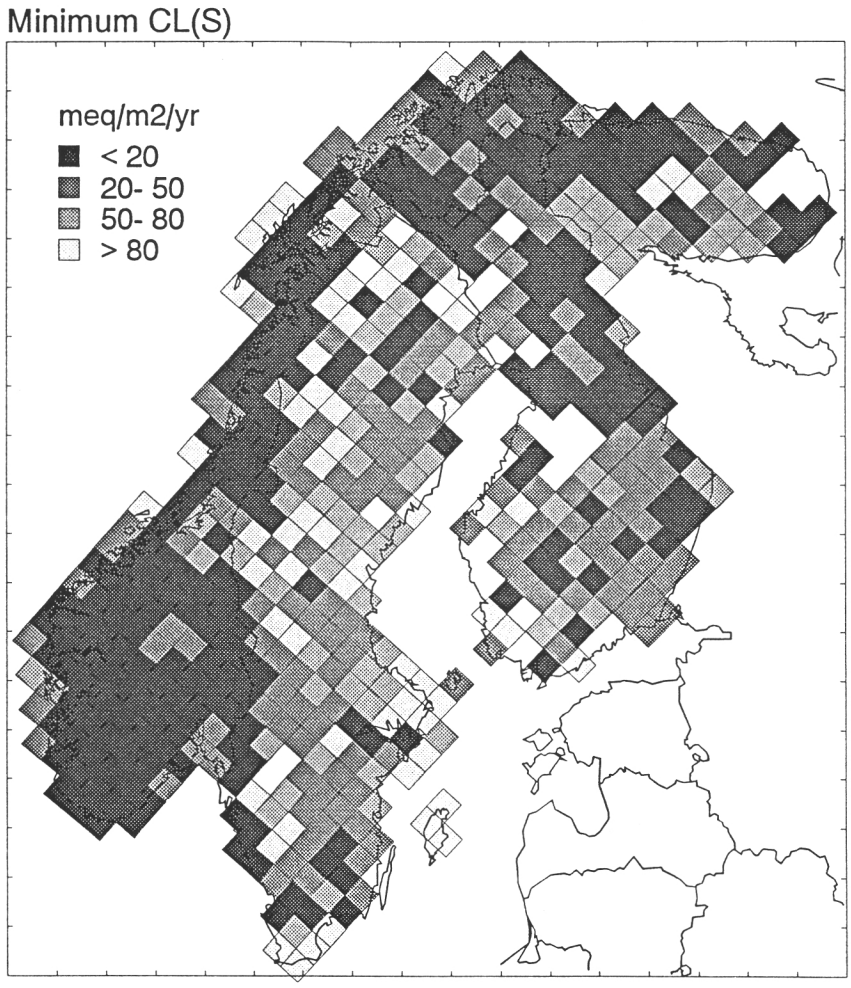


Figure 1. Map of Fennoscandia and the Kola Peninsula showing the minimum critical load value in each NILU-grid (ANC_{limit} = 20 μeg/l).

Sulphur deposition higher than 1.5 g $S \cdot m^{-2} \cdot yr^{-1}$ occurs in more than 30 % of the peninsula (the northwestern and central parts around the metallurgic industries in Nikel and Montegorsk, as well as in the region of the city of Murmansk. In the remote northeastern areas sulphur deposition is in the range 0.5 – 0.7 g $S \cdot m^{-2} \cdot yr^{-1}$, and in the southeastern areas less than 0.5 g $S \cdot m^{-2} \cdot yr^{-1}$. The prevailing directions of the air pollutants are thus to the northeast of the point sources.

The critical load of sulphur is exceeded in about 40 percent of the area (Fig. 2). Highest exceedance is found in the northwest, central and northeastern areas. The critical load was exceeded for almost 50 percent of the lakes sampled.

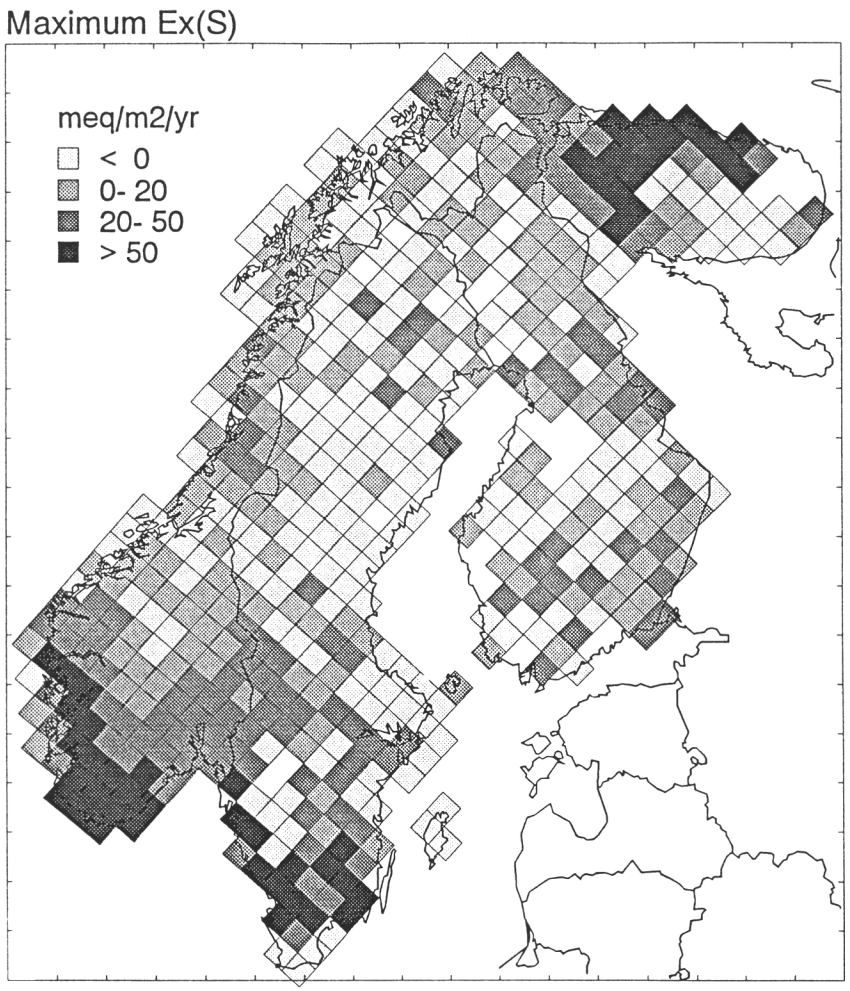


Figure 2. Map of Fennoscandia and the Kola Peninsula showing the maximum critical load exceedance value in each NILU-grid (ANC_{limit} = 20 μeq/l).

Although the results indicate good consistency with the data from the border areas of Norway and Finland, they should be considered as preliminary. They are so far based on a limited number of samples, considering the large area they represent. In order to improve the database it is necessary to improve the deposition data and to increase the number of sampled lakes, particularly in the remote eastern areas on the Kola peninsula.

Conclusions

Large areas in Europe and eastern North America suffer from acidification of surface water due to acid precipitation. Acidification has resulted in fish mortality and other ecological changes. The Nordic countries (Finland, Sweden and Norway) are particularly affected. On the basis of national regional lake surveys in each country we have mapped critical loads of acidifying deposition for surface waters. The critical load of acidity to rivers and lakes can be estimated from present water chemistry. For mapping critical loads and their exceedances the EMEP-grid system (150 by 150 km) and its 3 by 3 subdivision (50 by 50 km) (NILU-grids) is used. Frequency distribution curves for calculated critical loads showed a wide variation in sensitivity *within* and *between* the EMEP-grids and also between the three countries. Norway appears to have surface waters that are the most sensitive to acidic precipitation in the Nordic countries. Critical load exceedance for sulphur was estimated from available sulphur deposition data. Depending on the chemical criteria used, the exceedance varied from 20 to 70 percent of the area in the Nordic countries.

A new model for analyzing the exceedance and the reductions needed for N and S depositions that has been developed indicate that practically in the whole of Finland and in the northern parts of Fennoscandia the acidification problem could be solved by reducing S deposition only. In the southern part of Sweden some reductions in N deposition are clearly needed in addition to those for S. In the southern parts of Norway even strong measures to reduce S deposition are not enough, also N deposition has to be reduced considerably.

The acidity of the deposition in the "Nordkalott" area (north of the polar circle) is generally lower than in the southern parts of the Fennoscandian countries. Only in the areas of the smelters on the Kola peninsula the deposition is comparable to those found in the south. There are large spatial variations in critical loads, and sensitive areas are found in scattered areas all over the "Nordkalott" area, especially in the northeast close to the Russian border. The highest exceedances of the critical load are found in these areas. The exceedances are, however, lower than in the most affected areas in Sweden and Norway. Northern Fennoscandia and also the border areas of Russia will respond much faster to reduced deposition than the

highly affected areas in the southern parts of Fennoscandia, because the northern areas have been subjected to acidic deposition for a much shorter time than the southern areas. Since there are rather definite plans for reducing emissions from the smelters at the Kola peninsula, the present negative trends in lake water chemistry should turn to positive trends in the near future.

The border areas between Norway, Finland and Russia are today heavily polluted. Although some reductions in emissions from the smelters have occurred, only the planned rebuilding of "Pechenganikel" smelter will give significant reduction in emissions of sulphur and heavy metals resulting in reduced adverse effects on the ecosystems in the area. It is thus important to continue environmental research and monitoring in the border areas. Model calculations using the MAGIC-model on data for river Dalelva in Eastern Finnmark (heavily polluted by the smelter emissions) indicate that the river will become croninally acid within 50 years if present sulphur deposition continues. On the other hand, a 95 % reduction in sulphur deposition today will result in pristine water chemistry within a period of 10 to 20 years (Wright and Traaen 1992). A follow up in this area will give us an unique opportunity to study the reversibility of acidification at several ecosystem levels.

Preliminary data from the Kola peninsula indicate that here, like in the Fennoscandian countries, also are large areas sensitive to acidification. Due to heavy sulphur deposition in many areas the critical loads are also exceeded in large areas. Finland, Norway and Sweden are planning to carry out co-ordinated regional lake surveys in their respective countries in the fall of 1995. The data collected by these surveys will form a new updated basis for assessing critical loads and their exceedances for this area. It would be very useful also incorporate a corresponding detailed survey of the lakes of the Kola peninsula.

References

Bernes, C. 1991. Acidification and liming of Swedish freshwaters. Monitor 12. Swedish Environmental Protection Agency. 144 p.

Downing, R.J., Hettelingh, J.-P. & de Smet, P.A.M. (eds) 1993. Calculations and Mapping of Critical Loads in Europe: Status Report 1993. RIVM Report No. 259101003, RIVM, Bilthoven, The Netherlands.

Forsius, M., Kämäri, J., Kortelainen, P., Mannio, J., Verta, M. & Kinnunen, K. 1990. Statistical lake survey in Finland: Regional estimates of lake acidification. In: Kauppi, P., Anttila, P. and Kenttämies, K. (eds). Acidification in Finland. Springer-Verlag, Heidelberg, p. 759-780.

Henriksen, A., Forsius, M., Kämäri, J., Posch, M. & Wilander, A. 1993. Exceedance of critical loads for lakes in Finland, Norway and Sweden:

Reduction requirements for Nitrogen and Sulphur deposition. Acid Rain Research Report 32/1993. Norwegian Institute for Water Research (NIVA) Oslo, Norway. 46 p.

– , Kämäri, J., Posch, M., Lövblad, G., Forsius, M. & Wilander, A. 1990. Critical loads to surface waters in Fennoscandia. Miljørapport 1990:124. Nordic Council of Ministers, Copenhagen, Denmark.

– , Lien, L., Traaen, T.S., Sevaldrud, I.S. & Brakke, D. 1988. Lake acidification in Norway – present and predicted chemical status. Ambio 17: 259-266.

– , Wilander, A., Kämäri, J., Posch, M., Forsius, M., Baxendale, H. & Tarvainen, T. 1994. Critical loads for acidification of surface waters in the Northern Fennoscandia (Nordkalotten). Report nr. 33. The North Calotte publication series. 23 p.

Kämäri, J., Forsius, M., Kortelainen, P., Mannio, J. & Verta, M. 1991. Finnish lake survey: present status of lake acidification. Ambio 20: 23-27.

Malanchuk, J.L. & Nilsson, J. (eds) 1989. The role of nitrogen in the acidification of soils and surface waters, Nord 1989:92, Nordic Council of Ministers, Copenhagen, Denmark.

Nilsson, J. (Ed) 1986. Critical loads for nitrogen and sulphur, Miljørapport 1986:11, Nordic Council of Ministers, Copenhagen, Denmark.

– & Grennfelt, P. (eds) 1988. Critical loads for sulphur and nitrogen. Miljørapport 1988:15, Nordic Council of Ministers, Copenhagen, Denmark.

Traaen, T.S., Moiseenko, T., Dauvalte, V., Rognerud, S., Henriksen, A. & Kudraseva, L. 1991. Acidification of surface waters, Nickel and Copper in lake sediments in the Russian-Norwegian border areas. Progress report for 1989 – 1990. Working group for water and environmental problems under the Norwegian-Soviet environmental protection commission. Norwegian Institute for Water Research (NIVA), Oslo. 20 p.

Wright, R.F. & Traaen, T.S. 1992. Dalelva, Finnmark, nothernmost Norway: prediction of future acidification using the MAGIC model. Statlig program for forurensningsovervåking, Rapport 486/92. NIVA-report O-89187, Norwegian Institute for Water Research, Oslo, 17 p.

Critical loads of acidity and nitrogen for Swedish forest ecosystems, and the relationship to soil weathering

Harald Sverdrup[1)], Per Warfvinge[1)] and Kaj Rosén[2)]

[1)] Department of Chemical Engineering II, Lund Institute of Technology
Box 124, Chemical Center of Lund University, S-221 00 Lund, Sweden
[2)] Department of Forest Soils, Swedish Agricultural University
Box 7007, S-750 07 Uppsala, Sweden

Abstract

The critical loads of acidity and nitrogen has been mapped for Swedish forest soils, using data from the Swedish Forest Inventory for 1804 sites. Important for the critical load of acidity and nitrogen is the weathering rate of minerals. Special care was taken to calculate the weathering rate for Swedish soils from measured soil properties such as mineralogy and texture. The calculated weathering rates have been verified against field observations with good success. The results show that Swedish forest soils are sensitive to acid deposition, mainly because of very low soil weathering rates. The deposition of nitrogen to Swedish forests exceed the critical load, a situation that will lead to nutrient imbalances if not mitigated. A deposition reduction of 75 % of the acidity deposition in relation to 1988 deposition level, is required in order to protect 95 % of the forest resource in Sweden from effects of soil acidification. A reduction of 50 % of the nitrogen deposition is required to protect the forest from developing increasing nutrient imbalances. Forest soils in Sweden are at present still undergoing further acidification, and a reduction of the deposition below the critical load is required to achieve rapid and sufficient recovery. The mapping work was carried out by using the PROFILE model.

Introduction

During the course of 1989, 34 nations in Europe agreed to have critical loads in their countries mapped for the negotiations on a sulphur protocol on emission reductions. In Europe the work is coordinated by the Task Force on Mapping (TFM) through a coordination centre located at the Federal Dutch Institute for Health and Environment (RIVM) in Bilthoven, Netherland, on behalf of the United Nations Economic Commission on Europe (UN-ECE). The critical loads maps will be used as an input to the Working Group on Abatement Strategies (WGAS) and the Task Force on Integrated Assessment Modelling (TFIAM), for optimizing the abatement strategies for whole Europe (Hettelingh et al. 1993). For the mapping work a manual has been developed as a guideline (Sverdrup et al. 1990, Hettelingh

et al. 1993). Critical loads are determined for a number of different receptors, such as forest, grassland, lakes, streams or groundwater.

Definition

The critical load was defined by the International workshop on critical loads held at Skokloster, Sweden, in 1988, as:

The maximum amount of sulphur and nitrogen deposition that will not cause long term damage to ecosystem structure and function.

In the critical loads calculation, the linking of ecological response to deposition level is the central principle (Sverdrup et al. 1990, Hettelingh et al. 1991). Without ecosystem, no critical load.

Receptors

The present forest critical loads map applies to all types of forests, comprising Swedens forest area of 267.000 km^2. For the forest assessment each calculation point was area weighted in relation to the absolute local density of sites in relation to receptor density. For critical loads of acidity, trees were taken as the indicator organism, for nitrogen trees and 4 different types of ground vegetation has been suggested.

Methods

Basic principle

Thus for each ecosystem, an indicator organism is chosen. A chemical limit is found for that indicator organism, and the chemical limit is entered into a chemical mass balance equation including all sinks and sources of acidity in the system. The chemical limit is aplied to the solution concentration in the system. Thus ecosystem become connected through concentrations via mass balances to acid deposition.

Tools

At present the critical load for Sweden is calculated by two methods, the Simple Mass Balance model (SMB) (Sverdrup and de Vries 1994) and the PROFILE

model (Warfvinge and Sverdrup 1990). The one-layer SMB model has been employed by all European countries, including Sweden. PROFILE also employs a mass balance approach to critical load, but contains a fuller description of the system and allow for chemical feedback on biological processes. The two approaches are compatible. A four-layer SMB model hybrid (Short-PROFILE) is used for Germany, Austria, Great Britain, Russia and Poland. The full integrated PROFILE model is at present used by Sweden, Denmark, Norway, Switzerland and for extended regions of China, Russia, Great Britain and Germany. PROFILE is being tested and used for research purposes in universities and research institutes in more than 35 countries world wide. The tools are available for Apple MacIntosh and IBM PC compatibles, free of cost.

Critical load of acidity

The critical load of actual acidity represents the total load of acidity input the system can tolerate regardless of where it originates. The critical load is calculated as the adjusted deposition which will balance all inputs of acidity against all sinks of acidity, thus permitting no net depletion of the base saturation. Beside weathering, depletion of the base saturation will be among the most substantial sources of ANC for neutralizing added acidity.

Rearrangement of a mass balance for acidity can be shown to yield the following expression for the critical load as (Sverdrup and de Vries 1994):

$$CL(A)=W - ANC_L \qquad (1)$$

ANC_L is the leaching of ANC from the system, it is determined by the limits set for Al concentration in the system. In forest ecosystems applying a BC/Al=1, ANC of the leachate from the root zone will be negative, and the critical load will be larger than the weathering rate. The weathering of the mineral matrix will be the major long term source of alkalinity to neutralize acidity in the system, as well as the major supply of base cations to replace removed base cations, and an important factor for determining the chemical status of the soil.

Exceedance is defined as the excess of acid input over the critical load:

$$EX(A) = D + BA - CL(A) \qquad (2)$$

where D is total acidity input from deposition and BA is acidity input due to forest management and tree growth. The total acidity input by the deposition is defined as the sum of all anions minus base cations.

There is an European consensus to use BC/Al molar ratio ≥1.0 as a limit in the critical load calculation for forests, since this appears to be best supported by different types of experiments and observations (Sverdrup et al. 1990, de Vries 1989, de Vries and Kros 1992, Hettelingh et al. 1991, 1993, Sverdrup and de Vries 1994). In the calculation, the BC/Al-limit is used to determine the maximum allowable Al concentration from available base cations.

$$ANC_L = f(BC/Al) \tag{3}$$

The PROFILE model was used to calculate this connection.

Simultaneously, the stability criterium is applied; no Al leaching in excess of what is produced by weathering is permitted. If an excess Al is leached, soil stability can change, affecting soil erosion and soil stability.

Other suggested limits are total Al ≤4.0 mg/l or inorganic labile Al ≤2.0 mg/l. Alternatively, pH in the E-layer ≥pH 4.0 and in the B-layer ≥pH 4.4.

Critical load of nutrient N

The critical N concentration leads to a critical leaching of N and makes a mass balance approach possible for calculating critical loads.

$$N_{dep} = N_u + N_{im} + N_{de} + N_l \tag{4}$$

where:

N_{dep}	=	atmospheric N deposition
N_u	=	N uptake by forest
N_{im}	=	N immobilization
N_{de}	=	denitrification
N_l	=	Critical leaching of N

To find the critical load of nutrient N; N_{dep}=CL(N), let:

$$CL(N) = N_u + N_{im} + N_{de} + N_l \tag{5}$$

Several important processes have feedbacks linked to available N and available base cations. These are (1) uptake (2) denitrification and (3) immobilization. Uptake can be present or the critical uptake as calculated using the nutrient limiting method.

The PROFILE model

Critical loads for forest ecosystems in Sweden were calculated using the steady state mass balance model, implemented in the PROFILE model. Units used are keq/ha yr if not otherwise specified. The soil profile is divided into 4 layers, using input data for the thickness of each soil layer (O, A/E, B, C). The criteria were applied in the 0−50 cm of the soil for coniferous trees, assumed to be the typical rooting depth for this type of forest in Sweden.

Processes included

PROFILE are based on a conceptual model of a forest soil. It containes the following chemical subsystems:

- Deposition, leaching and accumulation of dissolved chemical components
- Chemical weathering reactions of soil minerals with the soil solution
- The net result of soil reactions with N-compounds; nitrification, denitrification
- Internal cycling of N and base cations in the canopy, such as uptake, canopy exchange, litterfall, immobilization and mineralization
- Net uptake of base cations and nitrogen, chemical feedback on uptake and amount nutrients removed by tree harvest
- Solution equilibrium reactions involving the carbonate system, CO_2, speciation and complexing involving Al and organic acids

These processes only represent a selection of chemical reactions in the soil. Among processes that have not been included are sulphate adsorbtion, a series of reactions that may change the CEC of the soil matrix. More or less, all processes included in the model have been subject to necessary simplification in some respect. In the PROFILE model, all soil processes communicate via the soil solution. The temperature dependence is considered for nitrification, denitrification, immobilization, weathering and solution equilibria. The PROFILE model has been set in an automated framework reading input data from a file containing the regional database.

The weathering rate sub-model

The outstanding difference between PROFILE and other comparable soil chemistry models is that the weathering rate is calculated from independent geophysical properties of the soil system. This reduces the degrees of freedom in the model and reduces the need for calibration.

Several chemical reactions between soil minerals and constituents in the soil solution contribute to the base cation release rate, and the total chemical weathering rate will be the sum of the weathering rates of all individual elementary chemical reactions with each mineral. For most minerals several reactions have been identified experimentally, the present model formulation include:

- The reaction with H^+, product inhibition by Al and cations of the parent mineral
- The reaction with H_2O, product inhibition by Al and cations of the parent mineral
- Reactions with dissociated organic acid ligands, surface saturation by organic ligand
- The reaction with dissolved CO_2 molecules

The reactions have been listed in order of quantitative importance under field condition. The reaction orders and the reaction rate coefficients were determined experimentally for most common primary and secondary minerals (Sverdrup 1990). The rate coefficients themselves are functions of the base cation and aluminium concentrations, as discussed in Sverdrup (1990).

The rate of dissolution is inhibited by the reaction products due to complexing on active dissolution sites. Thus Al, and the base cations Ca, Mg, Na and K are important weathering rate inhibitors. The presence of organic acids affect weathering in two ways.

The reaction between the mineral and organic acid is of little importance for weathering under field conditions, contrary to general hearsay among geochemists. This is especially true for those minerals with a high Madelung site energy for the base cation in T4 and S4 sites. The reaction with organic acids is, however, important for low energy mafic minerals such as nesosilicates, pyroxenes and amphiboles. In Swedish soils, mafic minerals account for a significant part of the weathering rate, but only in the deeper part of the soil profile where organic acids occur in low concentrations. Secondly, organic acids may enhance weathering of minerals with a strong Al product inhibition, by complexing Al and base cations. This effect is sometimes cancelled by the fact that organic acids occur in the top layers of the soil, where the mineral content is low. The quantitatively most important chemical reactions are hydrolysis with water and reaction with the H^+-ion.

The activity of exposed mineral surfaces is than unity in unsaturated soils where the availability of moisture is limited. Weathering will only take place on wetted surfaces. All surfaces that participate in chemical reactions must be wetted, but wetness alone is insufficient; there must also be sufficient soil solution present for

the weathering process to interact with other soil processes. Thus mineral surface reactivity is assumed to be proportional to soil moisture saturation.

It is important to test model performance in calculating the weathering rate from geochemical soil properties, since the critical load is so closely connected to the weathering rate. The PROFILE model was sent out to researchers in Europe and America with a request to test the model on field measurements. The returned calculations and independent measurements were used to construct the diagram using 29 sites determinations. The correlation to the 1:1 line is r^2=0.9983 (Sverdrup and Warfvinge 1988, Sverdrup et al. 1993, Jönsson et al. 1993). The model calculates weathering rates with an accuracy better than ±10 %. The comparison has been shown in Fig. 1. Further comparisons can be found in Clemmerud and Frogner (1994), Langan (1994) and Jacks (1994). The calculated weathering rate values include the specific reaction coefficients for the reaction with organic acids as determined in laboratory experiments.

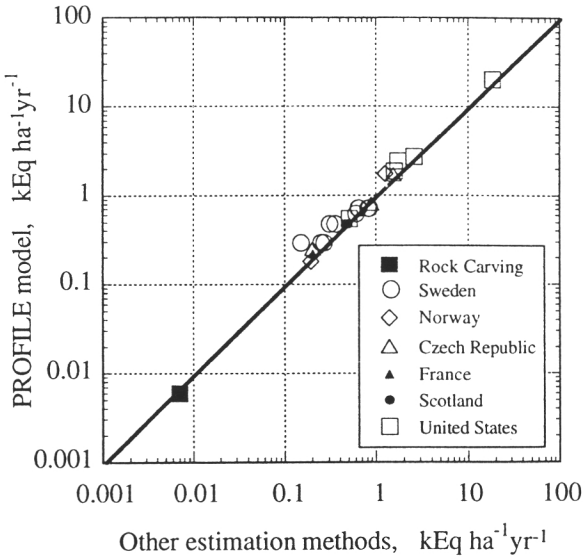


Figure 1. The diagram show a test of the PROFILE model against field estimates of the weathering rate. The PROFILE model was sent out to researchers in Europe and America with a request to test the model against field measurements. The returned calculated weathering rate values and corresponding independent field weathering measurements were used to construct the diagram. The correlation coefficient to the 1:1 line is r^2=0.998 in the range of weathering values from 0.007 keq/ha yr to 18.3 keq/ha yr. This indicate that the field weathering can be predicted within +/−10 %.

The uptake sub-model

Baseline uptake

Uptake in the model is based on input from the Swedish Forest Inventory. Uptake was calculated from growth as calculated with the HUGIN model. Average growth over one rotation period was calculated, and converted to uptake using present element composition in stemwood. This uptake is however calculated without any consideration to nutrient availability or soil chemistry feedback on plants.

Critical nutrient uptake

The long-term uptake of N is defined as the N uptake that can be balanced by a long-term supply of base cations, in the effect of *Liebig's law*; "Growth is limited by the nutrient in least supply". This amount is referred to as the critical uptake N_{crit}. The calculations are thus not based on the present uptake rate, since the present growth may be enhanced by artificial or time-limited supply of nutrients, mobilized from exchange sites by acid deposition. The critical uptake is calculated from mass balances for the nutrient cations Mg, K and Ca separatly. If weathering as separate ions are not available, then the total base cation weathering and a BC/N ratio can be used. The production of different cations from weathering can be calculated with the PROFILE model, from total analysis or estimated from soil type, parent material, temperature, texture and soil wetness. BC weathering rates are available in the SMB input data files.

Taking deposition and weathering as the sources of these nutrients, and uptake and leaching as the sinks the mass balance for a species *i* becomes:

$$U_{crit,\,BC} = D_{BC} + W_{BC} - Q \cdot [BC]_{lim} \qquad (6)$$

where

$U_{crit,\,BC}$	=	critical uptake
D_{BC}	=	atmospheric deposition
W_{BC}	=	weathering base cation
Q	=	water flux from the rootzone
$[BC]lim$	=	limiting concentration for uptake

The limiting concentration is the level when the trees no longer can extract a nutrient from the solution. In the calculations, the limiting concentration for Ca and Mg has been set to 5 meq m^{-3}, and to 0 for K.

From the critical base cation uptake, maximum N uptake is calculated using the nutrient to N limiting ratio. The ratio determines the maximum N that can be taken up without inducing P or base cation deficiency:

$$N_{crit} = \min\left(\frac{U_{crit,Ca}}{Ca:N}, \frac{U_{crit,K}}{K/N}, \frac{U_{crit,Mg}}{Mg/N}\right) \quad (7)$$

where

N_{crit} = critical N uptake
BC/N = nutrient stress limit

P has been ignored at present due to input data limitations. The critical ratios have been listed in Tab. 1. The critical uptake is limited by present deposition of N, no growth is allowed beyond that value even if more weathering is available.

Chemical feedback

The effect of soil acidity on tree uptake of N and base cations, is necessary to include in the calculations. Omission of these feedbacks lead to unrealistic results. This feedback has been implemented using empirical correlations between spruce seedling growth in laboratory experiments and observed bulk soil solution chemistry (Sverdrup and Warfvinge 1993):

$$f = \frac{[BC]^n}{[BC]^n + k \cdot ([Al^{3+}] + p \cdot [H^+])^m} \quad (8)$$

where k=0.35, n=m=p=1, the value is derived from laboratory experiments (Sverdrup and Warfvinge 1993). BC is defined as Ca + Mg + K, Al and BC complexed with organic ligands are excluded from the response expression. The reponse of spruce seedling growth to soil acidity in laboratory experiments have

Table 1. Approximate deficiency nutrient ratios to be applied in the critical loads calculations in order to prevent long term nutrient imbalances.

Tree species	Ca/N	Mg/N	K/N	BC/N
Norway spruce	0.4	0.13	0.17	0.8
Scots pine	0.5	0.17	0.12	0.8
Beech	0.4	0.2	0.2	0.8

been shown in Fig. 2. Tests with different parameters has caused us to express soil acidity through the ratio between base cations required by uptake (BC) and undesired cations such (Al^{3+}, H^{+}). More information on why the ratio is preferred before soil solution concentrations or other parameter combinations is found in the review and synthesis by Sverdrup and Warfvinge (1993).

In PROFILE, uptake reduction is calculated for each soil horizon, and the plant allowed to reallocate uptake from one layer to another, if it cannot get what is required in one particular horizon. Uptake is stopped if the base cation soil solution concentration falls below 15 μeq/l. This way the plant is allowed to optimize uptake in the PROFILE model.

In the PROFILE model, complexation is calculated using a simplified version of the Tipping model, exploiting the Oliver equation (Warfvinge and Sverdrup 1990). Thus, high DOC concentrations in the upper soil layers will be able to prevent a strong soil acidity effect on uptake since a large part of the available Al is complexed. This difference between soil layers is considered in the PROFILE model.

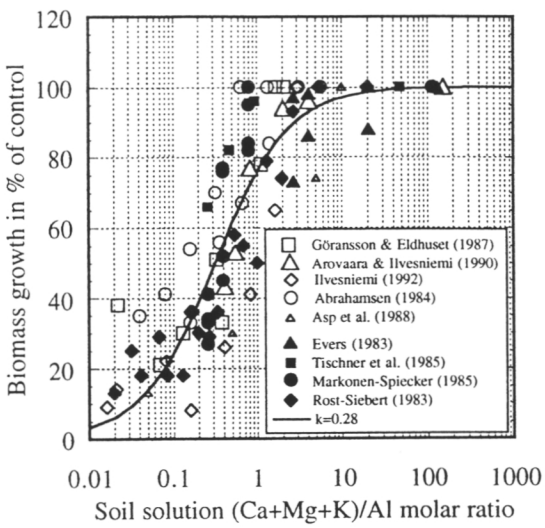


Figure 2. Reponse of spruce seedling growth to soil acidity in laboratory experiments. Soil acidity has been through the ratio between base cations required by uptake and undesired cations such as Al^{3+} and H^{+}. More information is found in Sverdrup and Warfvinge (1993).

Nitrogen reactions

Immobilization, nitrification, denitrification occur both in the calculation of critical load for acidity and the critical load for nutrient nitrogen.

Immobilization

Long term immobilization in forest ecosystems has been estimated using present content divided by soil age. Such estimates ignores the occasional purging of the system when forest fires occur, and also long term effects of denitrification and fixation. Immobilization as estimated from soil age and stored amount has a value of 0.5−1.5 kg N ha yr. This is very much lower than estimates for present immobilization. At present many forest ecosystems accumulate 7−15 kg/ha yr in the forest floor. The assumptions made in this approach are that immobilization is caused by retarded decomposition and microbiological fixation. The rate is given by:

$$N_{im} = k_{im1} \cdot N_{litterfall} + k_{im2} \cdot [N] \qquad (9)$$

The coefficients k_{im1} and k_{im2} depend on soil acidity, soil wetness and temperature in the model. The value of the coefficients have been estimated from field observations. The expressions has been calibrated in two ways for the calculation; (1): Immobilization in northern Sweden was assumed to be 0.5 kg/ha yr at soil pH 5.5 and N deposition 2 kg/ha yr, (2): Immobiliztion was assumed to be 8 kg/ha yr at present at soil pH 4.2 and the deposition 30 kg/ha yr. The immobilization estimate for present is based on a study on denitrification by Sverdrup and Ineson (1994). Retarded immobilization has as process priority over microbiotic fixation.

Nitrification

The expression for nitrification was taken from Sverdrup et al. (1990):

$$N_{nit} = \frac{k_{nit} \cdot [NH_4^+]}{K_{M_{nit}} + [NH_4^+]} \qquad (10)$$

The coefficient k_{nit} depend on soil acidity, soil wetness and temperature, the value was derived from dutch field studies.

Denitrification

The expression for denitrification was taken from Sverdrup and Ineson (1995):

$$N_{de} = \frac{k_{de} \cdot [NO_3]}{K_M + [NO_3]} \tag{11}$$

The coefficient k_{de} depend on soil acicity, soil wetness and temperature. The general expression for modifying rate coefficient is:

$$k = k_0 \cdot f(pH) \cdot g(water) \cdot h(T) \tag{12}$$

All coefficients for denitrification, nitrification and immobilization (k_{de}, k_{nit}, k_{in1}, k_{im2}) are modified this way.

The leaching term, N_l.

At steady-state the, and with a balanced nutrient supply, the N leaching should amount to the natrual leaching from N-limited stands. In the calculations performed for Sweden, initially the basic assumption behind the critical uptake consept was $N_l = 0$. However, not all ecosystems are necessarily dominated by trees.

When nitrogen appear in significant amounts in the soil solution, herbs and plant species composition of the heath and ground vegetation may be changed. Lichen type of vegetation loose to lingonberry (*Vaccinium vitis-idaea / Caluna*) type of vegetation. These in turn loose ground to blueberry type vegetation (*Vaccinium myrtillus*), blueberry vegetation types loose ground to a grass type vegetation, grass vegetation loose to a type dominated by herbs. The maximum permitted leaching is given by the percolation from the bottom of the root zone (i.e. runoff rate) and the critical N soil solution concentration.

$$N_l = Q \cdot [N]_{crit} \tag{13}$$

Im comparison with the nutrient limitation balance approach described above, the difference is that the leaching term no longer is 0, but linked to a critical concentration related to some type of sensitive plant species. Still it must be considered that long term sustainable tree uptake at steady state, cannot be set higher than what is given by the liming nutrient calculation.

Input data sources

All data in this work is based on data and soil samples collected in the Swedish Forest Inventory (Rikskogstaxeringen / Ståndortskarteringen). The inventory data was collected 1983 – 1985. It consists of a network of 27,000 repectively 2,200 stations evenly spread over the complete forest area of Sweden (267.000 km^2). Soil

samples down to the C-layer at approximately 60 cm soil depth were collected. The basic soil parameters for this study were analysed on these samples. The list below represent the input data required for the PROFILE model for each site in the calculations; total deposition of sulfate, nitrate, ammonium and base cations. Precipitation, runoff and mean annual temperature, soil mineralogy of the C-layer, soil texture class for the C-layer, base cation uptake and nitrogen uptake.

Several other parameters such as CO_2 pressure, soil solution dissolved organic carbon, distribution of uptake and evapotranspiration and gibbsite coefficients have less influence and are entered as standard values taken from the literature and stay generally constant between runs and sites. Annual average air temperature, precipitation and runoff was taken from the official statistics of the Swedish Metheorological Institute (SMHI), for each NILU grid (50x50 km). Other input data were derived strictly in accordance with the "Mapping critical loads" by Sverdrup et al. (1990).

Uptake

Basic vegetation data has been measured the 2,200 sites of the "Ståndortskarteringen" within the Swedish Forest Inventory. Each tree species in considered separately, and the total uptake weighted together for each calculation point.

Table 2. The limiting concentrations of nitrogen for inducing vegetation changes. The limits have been based on Swedish forest vegetation survey data as well as a review and partly reevaluation of studies by Tillman and Ellenberg.

Ecological change	Threshold
Coniferous trees → Nutrient imbalances	[N] ≤ 0−0.2 mg N/l
Decideous trees → Nutrient imbalances	[N] ≤ 0.2−0.4 mg N/l
Lichens → Lingon-berry type	[N] ≤ 0.2−0.4 mg N/l
Lingon-berry type → Blueberry type type	[N] ≤ 0.4−0.6 mg N/l
Blueberry → Grass type	[N) ≤ 1−2 mg N/l
Grass → Herb type	[N] ≤ 3−5 mg N/l

Mineralogy

Absolute soil mineralogy was derived for 140 sites by measurement by the Swedish Geological Survey (SGU) at Uppsala and at the Czech Geological Survey in Praha. Total elemental content of the soil after complete dissolution was determined for all 1804 sites. These were analysed using generic wet chemistry methods for Ca, Mg, Na, K, Al, Si, Fe, Ti and trace metals.

The mineralogy was determined by separation of the sample into two density classes and two particle size classes. Each subsample is analysed for total elemental composition using wet chemistry methods, and subsequently analysed for elements using XRF and for minerals using XRD and known standards of mixtures of pure mineral phases. Total analysis imply that the minerals in a soil sample are totally dissolved in a lithiumborate-alkali melt, diluted in water and analysed using standard methods.

Separation of the mineral sample in a liquid with density in the range 2.680 g/cm^3, will generally concentrate easily weathered minerals 30−70 times. The fast weatherable minerals will tend to concentrate in the heavier fraction, and the analysis can then be carried out on the enriched fraction, despite that they may be present in very small amounts in the whole soil sample. The mineralogy determination is generally carried out on the silt and clay fraction of the soil, since all particles larger than 50 μm dissolve to slow to be significant for the weathering rate. These data are used to estimate the mineralogy of the whole soil sample.

The model UPPSALA is a back-calculation model for reconstructing the mineralogy from the total digestion analysis in order to provide input to models like PROFILE from simple survey data. After an initial survey of Swedens soil mineralogy in what was defined as different geological provinces, it was realized that mineralogy for these geological provinces would be correlated to the total analysis chemistry. The mineralogy model is based on an apriori assumption of the stoichiometric composition of the minerals of the particular soil. The minerals have been grouped into assemblies of minerals with similar composition and dissolution rate. For soils of granitic origin or soils derived from rock of secondary granitic origin (sandstones) as well as originating from different types of schists and shales, it is assumed that the following mineralogical groupings are valid; Muscovite is assumed to comprise muscovite, secondary di-octahedral illite, di-octahedral chlorite and vermiculite of secondary weathered type. Chlorite comprise tri-octahedral chlorite, primary illite and tri-octahedral vermicullite of primary type. Hornblende imply all amphiboles, such as hornblende, riebeckite, arfvedsonite, glauconite and tremolite. Plagioclase has for Sweden been assumed to be oligoclase with 80 % albite feldspar component. K-feldspar is assumed to be 10 % albite feldspar component. All phosphorous has been attributed to apatite. The equations

used in the UPPSALA model for calculation of the % weight content of the individual minerals are shown in Tab. 1.

In calchareous soils, eq. V – VII are omitted and eq. IV is replaced by:

$$\text{Calcite} = 1.79 \cdot \text{CaO} - 3.67 \cdot \text{Apatite} - 0.2 \cdot \text{Plagioclase} \quad \text{(I)}$$

Residual Al is used to form clay minerals in a group called vermiculite if they can be matched by a small amount of residual base cations, other residual Al is considered inert. The calculation is checked by calculating the amount quartz and adding all components. Only such sites which lie within the range 95 – 105 % is accepted. If a negative content arises in the calculation, it implies that there is no more adequate ions available, and that mineral is set to zero. For 1804 out of 1913 sites calculated, the non-chalcareous mineralogy was accepted, and an additional 30 sites were estimated to contain calcite, the modified UPPSALA model was applied to them. For these soils, mafic minerals are omitted, since weathering of calcite will dominate.

The mafic minerals in Swedish soils are dominated by hornblende and epidote. Biotite is sparesely occurring, only a few sites have significant amounts. Pyroxenes, garnet and olivines occur only as exceptions. The sum of mafic minerals can be reproduced reasonably well, tha fraction of it being hornblende likewise. The weak point in the UPPSALA model is the expression for epidote. Epidote occur in small amounts only, it is omnipresent in most soils of granitic origin in Scandinavia.

Table 3. The equations of the UPPSALA model used to reconstruct mineralogy from total analysis.

I	K-Feldspar	=	$5.88 \cdot K_2O - 0.588 \cdot Na_2O$
II	Plagioclase	=	$11.1 \cdot Na_2O - 0.22 \cdot$ K-Feldspar
III	Apatite	=	$2.24 \cdot P_2O_5$
IV	Hornblende	=	$6.67 \cdot$ CaO - $3.67 \cdot$ Apatite - $0.2 \cdot$ Plagioclase
V	Muscovite	=	$2.08 \cdot K_2O - 0.208 \cdot Na_2O$
VI	Chlorite	=	$3.85 \cdot$ MgO - $0.39 \cdot$ Hornblende - $0.39 \cdot$ Muscovite
VII	Epidote	=	$0.1 \cdot$ Hornblende + $0.03 \cdot$ Plagioclase - 0.3
VII	Quartz	=	SiO_2 - $0.63 \cdot$ Plagioclase - $0.68 \cdot$ K-Feldspar - $0.38 \cdot$ Muscovite - $0.33 \cdot$ Chlorite - $0.45 \cdot$ Hornblende - $0.42 \cdot$ Epidote

Texture

Texture was measured by granulometry and BET/adsorption analysis on the 124 mineralogy analysis samples, and correlated against field texture classification. The texture for all 1804 sites were read from the correlation using the field classification. The simplest way to calculate the soil texture is from a particle size distribution. The conversion from a standard sieve curve to surface area was carried out using:

$$A_{tot} = (x_{clay} \cdot 8.0 + x_{silt} \cdot 2.2 + x_{sand} \cdot 0.3) \cdot \frac{\rho_{soil}}{1000} \qquad (14)$$

where A has the dimension $10^6 m^3/m^3$ soil, and where we have the condition:

$$x_{clay} + x_{silt} + x_{sand} + x_{coarse} = 1 \qquad (15)$$

and x_{clay} is the fraction of soil particles less than 2 μm, x_{silt} the fraction of 2−60 μm, x_{sand} the fraction of 60−200 μm, and x_{coarse} the fraction of soil matrix coarser than 200 μm. ρ_{soil} is the bulk density of the soil in kg/m^3. The granulometric approach is used, due to difficulties of interferrence from secondary precipitates and organic matter with the BET measurements on whole soil samples. The granulometric relation was determined by measuring BET surface on a series of soil samples divided up into sand, silt and clay fractions. Untreated fractions and such where the precipitations and organic matter had been removed was used. A relation between the field classification of soil texture and laboratory measurements of exposed surface was found:

$$A_{tot} = 0.093 \cdot e^{(0.51 \cdot TX)} \qquad (16)$$

where *TX* is the texture class. A_{tot} is the total surface in $10^6 m^2/m^3$ soil. The relation is based on determinations of surface area and texture class on 100 soil samples.

Deposition

Deposition data were prepared by the Institute of Water and Air Research (IVL) at Göteborg, from the Swedish deposition monitoring network and EMEP data. Modified with filtering factors for different vegetation types. The total deposition is calculated for each calculation point using the filtering factors and the vegetation mixture for that point. The deposition used correspond to 1988 deposition level.

Results

Critical loads of acidity

Critical loads for forest ecosystems is shown in Fig. 3. For Sweden, both the SMB and the PROFILE model was employed, both models show good consistency. The weathering rate came from PROFILE calculations. The average critical load for Sweden is 0.58 keq/ha yr.

Areas where critical loads have been exceeded are shown in Fig. 4, according to PROFILE model. The critical loads in Sweden are low, and hence the area with exceedace large. On the protection level 95 % for conifer ecosystems, nearly 85 % of the country have exceedance. The maps express the critical load in terms of acidity.

The role of forestry for the critical load can be seen in Fig. 5. The map was calculated by setting uptake by the forest to zero. That is equivalent to say that the forest is growing, but all nutrients are cycled. The difference from Fig. 4 is significant. The role of forestry is less an acidifying effect, and more of an effect where net uptake of base cations make the soil more vulnerable to acidification. The role of forestry on soil acidification is therefore mainly an indirect effect. The difference corresponds to a decrease in exceedance of the critical load of approximately 40 %. Forestry makes the soil this much more vulnerable to acid deposition, but it does not acidify the soil correspondingly to 40 % of the deposition. Important is that forestry without acid deposition does not acidify the soil significantly if base cation uptake is balanced by N uptake and base cation uptake remain less than weathering plus base cation deposition.

Without any acid deposition, soil acidity fluxes caused by forest growth will generally not exceed the neutralizing capacity of the soil. It can thus be expected that the forest soils will return to soil pH values significantly higher than today if the acid deposition is reduced below the critical load.

The required reduction in S deposition is shown in Fig. 6. The map show that both S and N deposition must be reduced to have no exceedance in 95 % of the area. It is important to realize that it is a necessity that the acid deposition for a specific site must be below the critical load for that soil to recover to a state better than the critical limit. Reductions in acid deposition which will not bring the deposition below the critical load, will not lead to recovery.

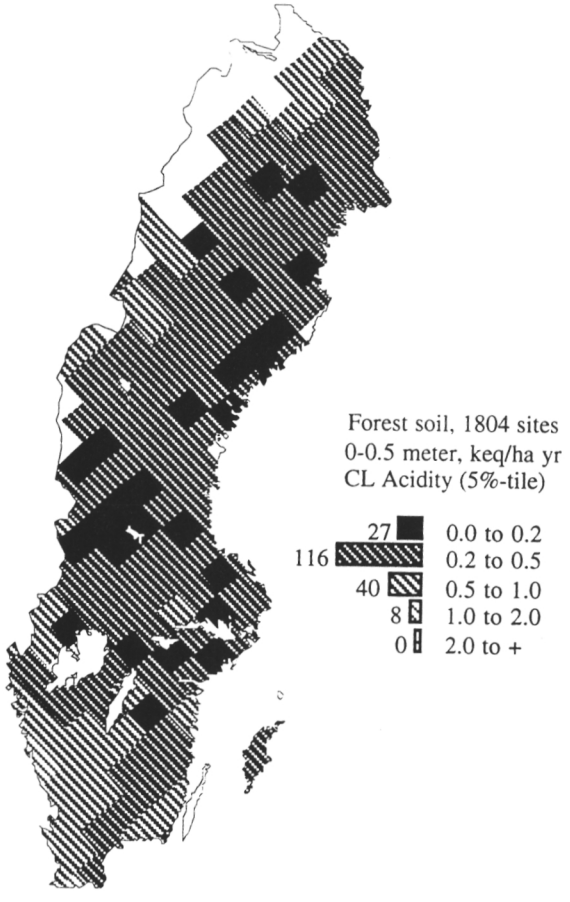


Figure 3. The critical loads for Swedish soils if the objective is to protect 95 % of the area. The number associated with the bars in the legend indicate the number of 50x50 km squares in each category.

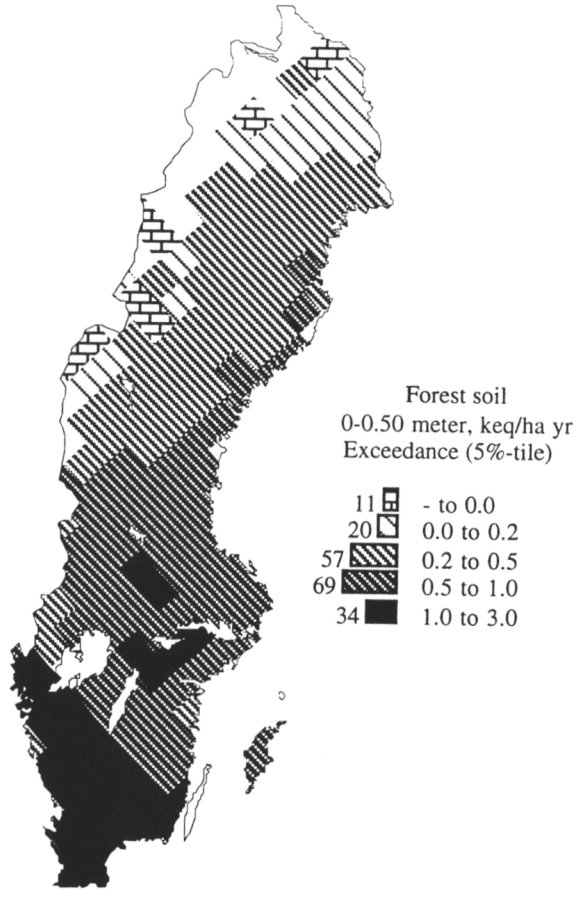


Figure 4. Areas where the critical loads have been exceeded for Swedish soils if the objective is to protect 95 % of the area. 82 % Swedens forested area receives more acidic deposition than the critical load.

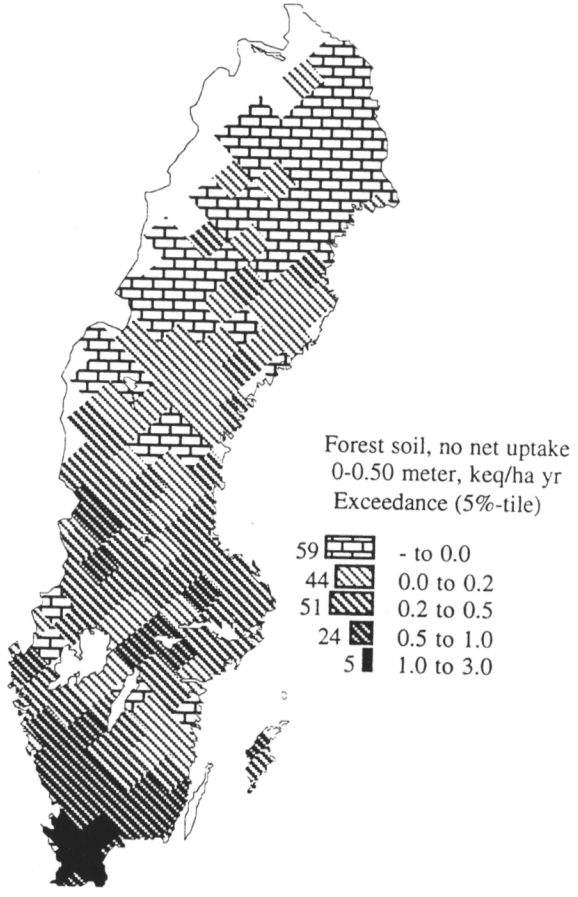


Figure 5. Areas where the critical loads have been exceeded for Swedish soils assuming complete return of all harvested N and base cations.

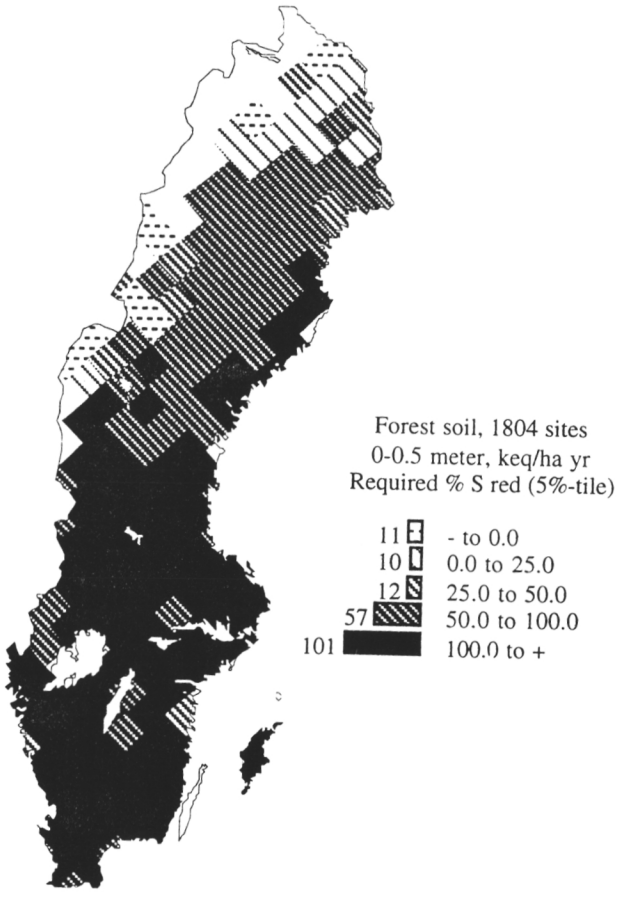


Figure 6. The necessary reduction in S deposition to get the critical load.

Deposition reductions

The assessment indicate that significant reduction in deposition levels will result in a fairly rapid chemical recovery of the soils, and soil acidification seems to be reversible. Whether ecosystem recovery in terms of biology is reversible, is unknown. The precautionary principle would lead us to assume that it is not, until the opposite has been proved.

In order 95 % of the forested area to have no exceedance, acid deposition will have to be reduced by at least 75 % of the 1985 level. This implies an 80 % reduction in S and 30 % reduction in N, but the reductions may for the sake of economic optimization be distributed differently between S and N. It should be made quite clear that abatement of S alone will not be sufficient, S accounts only for approximately 65 % of the deposited acidity in southern Sweden.

Fig. 7 shows the cumulative distribution of fractions of the forest area receiving more acidity deposition than the critical load under different deposition reductions. This corresponds to the fraction of the forest area with BC/Al ratio below 1.0 at steady state. It can be seen that 5 % exceedance is achieved at an acidity reduction corresponding to -85 % S / -30 % N reduction. That would imply that 95 % of the area would get less acid deposition than the critical load.

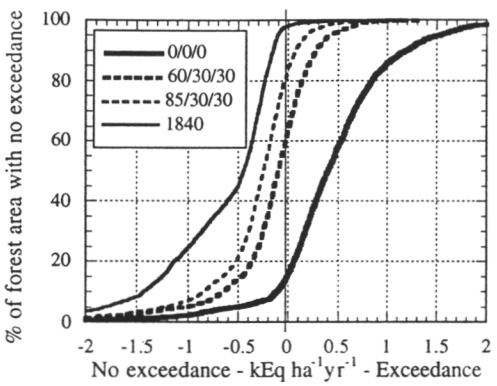


Figure 7. Distribution of forest area receiving more acid deposition than the critical load. This is equal to the forested area with BC/Al ratio below 1 under different deposition scenarios. The diagram is used to determine the required reduction after a protection level has been decided.

Critical load of nutrient N

Fig. 8 and 9 show critical loads for forest trees that have been calculated. It can be seen that the nitrogen deposition over Sweden is far to high from the perspective of long term sustainable forest yield. The present level will induce the forest to grow at a rate above what weathering and base cation deposition can sustain. This may lead the forest into a long term stress situation if this is permitted to continue. A long term permanent nutrient imbalance stress will cause the tolerance to additional stresses due to other partially natural factors to be less, and thus cause the system to have less stability.

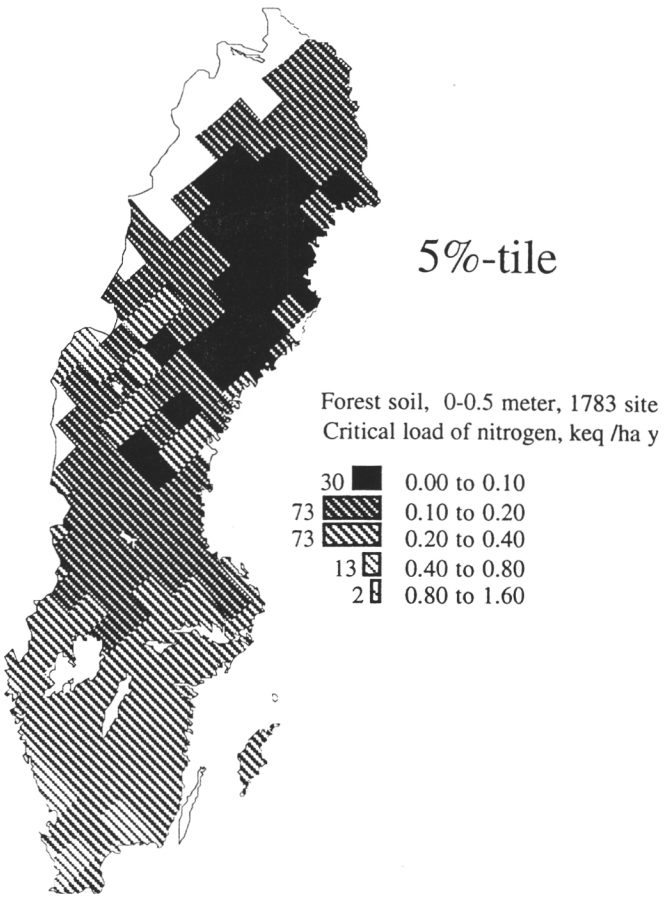


Figure 8. Critical load of nutrient N for foest ecosystems permitting no net N leaching . This map forsee whole tree harvest.

Trees are not the only important species in a forest. Fig. 10 show critical loads of nutrient nitrogen calculated for ground vegetation of the lingonberry type. Similar critical loads for acidity using Lingonberry as indicator could in principle also be calculated. The obtained pattern show remarkable good visual correlation with observed vegetation changes during the last 15 years.

Weathering rates

One important result of the calculations with the PROFILE model is the weathering rate in each point. Values for the net release of Ca, Mg, Na, K and P is obtained. This is useful for estimating the availability of base cations for forest growth and the assessment of long term yield in Swedish forestry. The total weathering rate equals the neutralization rate, available for neutralizing acidity. Fig. 12 show the weathering rate of Mg, the base cation most often appearing to be of limited supply, Fig. 11 show the total weathering rate in the upper 0−50 cm.

The fact that the weathering rate increased down the soil profile also affects the value of the critical load. The weathering rate is low in the upper soil layers, since the content of weatherable minerals is low there. These minerals were lost in the process of podsolization.

The calculated critical loads were correlated back with the weathering rate. The empirical relationship is with W and CL in keq/ha yr:

$$CL(A) = 0.13 + 2.11 \cdot W \qquad (17)$$

The correlation in the plot is $r^2 = 0.87$, making the empirical formula based on weathering only a good predictor of critical load of acidity for conifer forests. The fit is not perfect since weathering is not distributed evenly among different soil horizons.

The average soil weathering rate for the upper 0−50 cm in Swedish soils is 0.28 keq/ha yr. The noncalchareous soils vary in the range 0−4.5 keq/ha yr. This is low average value is caused by a majority of the soils having minerals of granitic origin and the average coarse texture of Swedish forest soils. No silicate soil has a value above 2.5 keq/ha yr.

Fig. 13 show the distribution between soil layers at Gårdsjön, Sweden. The lower graph show the distribution of the weathering rate on the minerals of that soil. It can be seen that plagioclase, epidote, hornblende and apatite account for most of the weathering rate. The role of biotite, often claimed to be important for the soil weathering rate, is less than marginal.

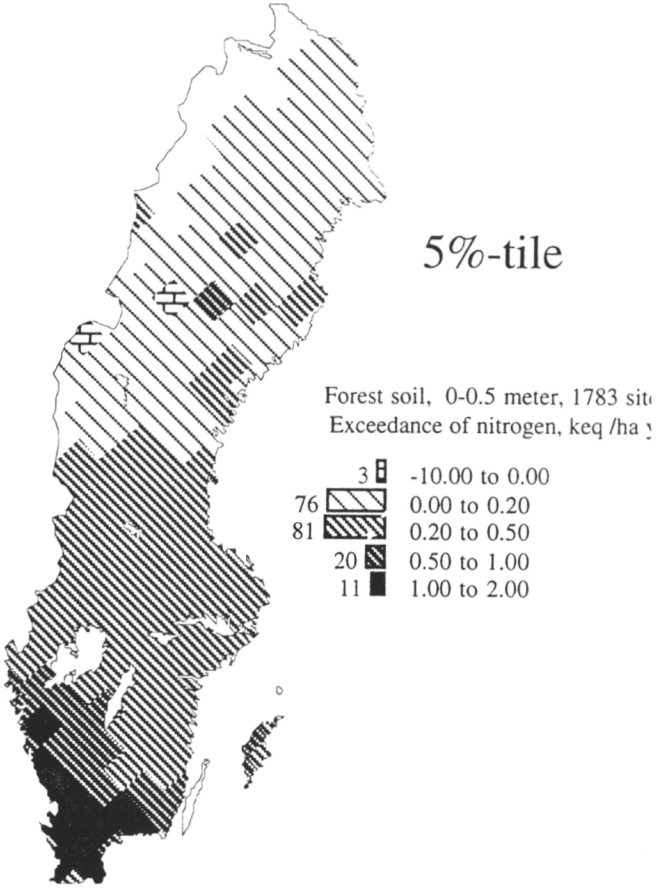


Figure 9. Exceedance of critical load of nutrient N for forest ecosystems permitting no net leaching of N.

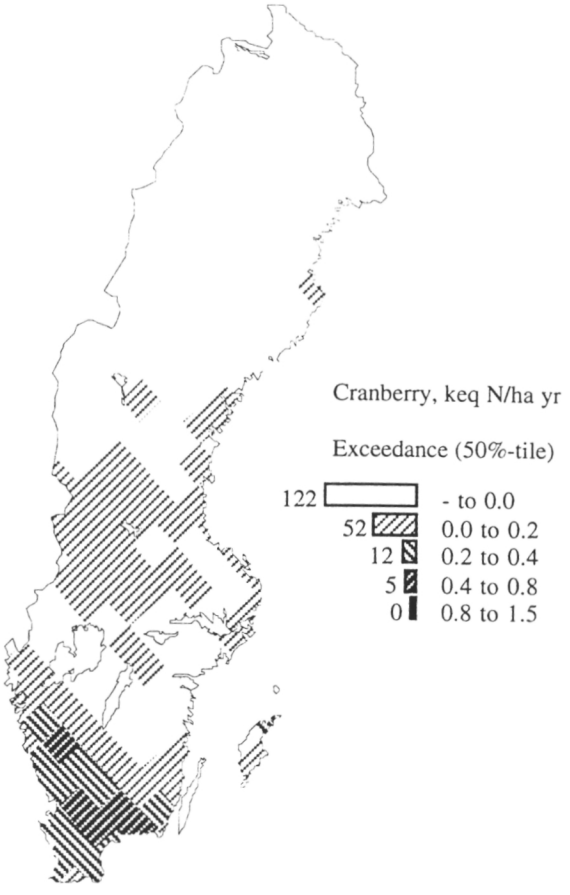


Figure 10. Exceedance of the critical load of nutrient N calculated with respect to ground vegetation of the Lingonberry type. This should not be interpreted as a specific response of lingonberry, but rather as an indication of changes likely to occur in the fraction of the groundsurface regionally covered by such a type of vegetation.

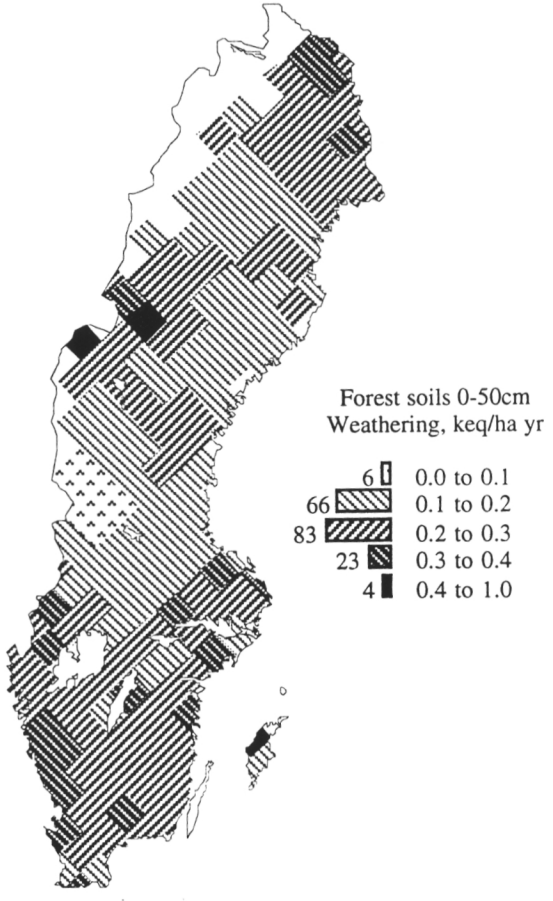


Figure 11. Weathering rate calculated from soil mineralogy, texture and soil chemistry using the PROFILE model. Tests indicate that uncertainty in the weathering rates are approximately ±5 % on the absolute value.

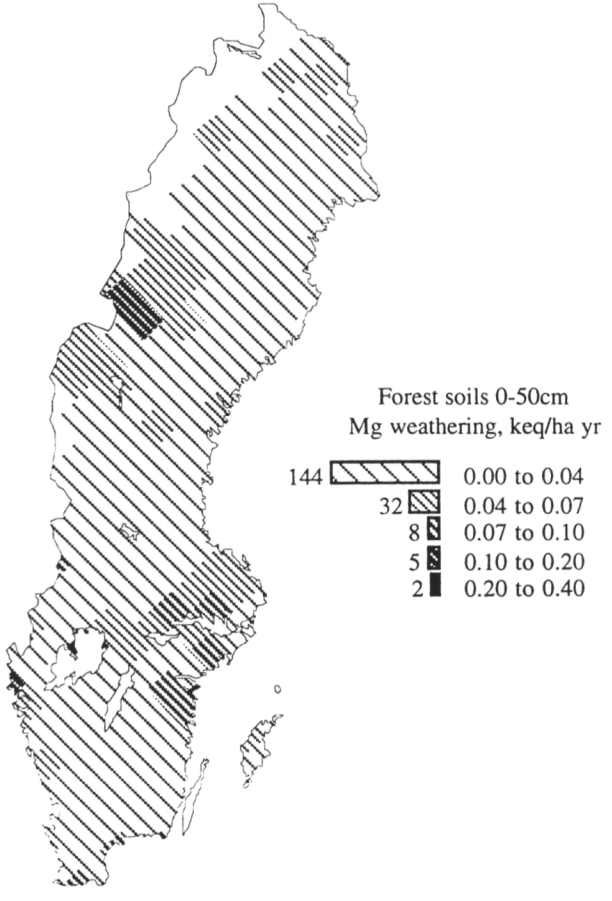


Figure 12. Mg weathering rate calculated from soil mineralogy, texture and soil chemistry using the PROFILE model. Especially the weathering rate of Mg does not follow the pattern of N deposition and present forest yield. This predisposes Swedish forests for high N deposition induced nutrient imbalances.

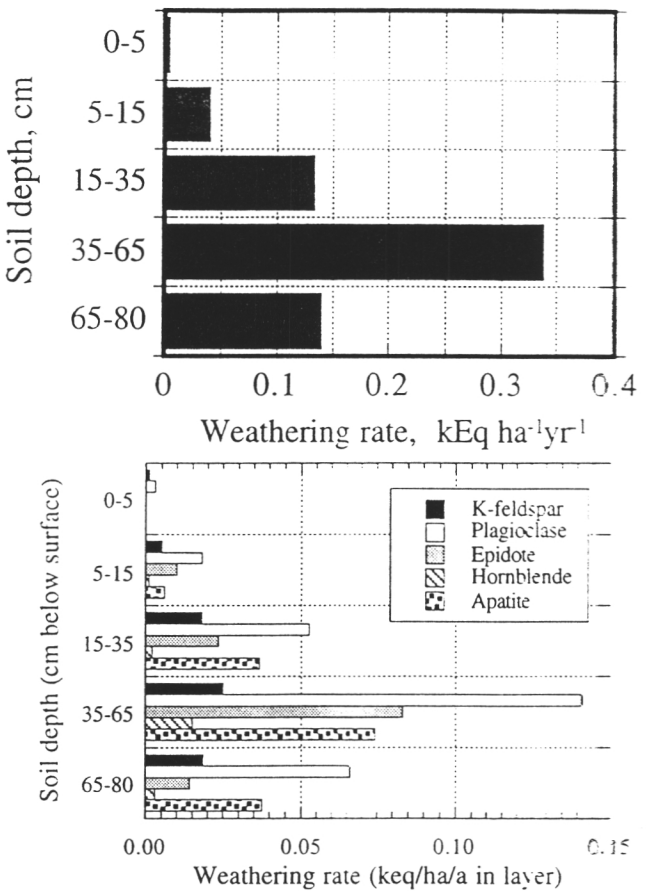


Figure 13. The weathering rate increases down through the soil profile due to a depletion of minerals in the top layers, and an increase in soil bulk density. This makes it important to consider each soil horizon separately in the calculation of critical loads. The above shown example is from the catchment of Lake Gårdsjön (Sverdrup and Warfvinge 1988, 1993).

Field vs. laboratory rates

It has often been claimed that laboratory weathering rates cannot be conciled with observed field rates. This is a very persistent myth, and its is still perpetuated by many researchers. The consistency between laboratory rates and field rates has been shown on several occasions (Sverdrup and Warfvinge 1988, 1993, Warfvinge

and Sverdrup 1991). The PROFILE model utilize weathering rate coefficients derived from laboratory experiments and is capable to reproduce field weathering rates with high accuracy. The reason for this is the fact that a laboratory rate cannot be compared to a field rate at face value since they occur under very different conditions. A laboratory experiment operate fully wetted in solution, under room temperature, in the field wetting of the surfaces is only partial and the temperature is significantly lower. Many experiments were carried out in buffers, under high CO_2-pressure or on freshly ground minerals, all factors that will offset the weathering rate. Finally, weathering is not the only process in the world, and no comparison with field data will be successful, unless the interaction with other soil processes are accounted for. The sum of all these effects have been listed in Tab. 4.

Discussion

Implications for the whole ecosystem

In the calculation of critical loads, trees used for biomass production was used as the indicator plant for the ecosystem. This was done partly because trees were recognized as being among the more sensitive elements in the ecosystem with respect to acidification. But trees were also chosen because they have a monetary value that is recognized by the political partners that will have to work out the conditons and legal aspects of emission reductions. Protection of something that has a defined monetary value and which is connected to jobs and welfare is easier accepted in a political process.

If the empirical relationship observed in laboratory experiments is used to convert predicted BC/Al values to growth changes, unpleasantly large losses in growth potential are predicted. If there is a significantly probability for these predictions to become reality, then acidification pose a most significant threat to the national profitability of Swedish forestry. Both Cronan and Grigal (1995) and Sverdrup and Warfvinge (1993) find the laboratory experiments internally consistent and conclude that the probability for BC/Al larger than 1 to cause adverse effects to tree growth, to be more than 50 %.

The base saturation in a soil kept constant at BC/Al = 1 will have a significantly lower base saturation than what it historically had (Falkengren 1989). Maintaining a BC/Al near or above 1 is probably protecting coniferous trees from growth effects, but there are many other plants in the forest ecosystem which are much more sensitive to Al and pH.

Table 4. The effect of different conditions on the weathering rate in the laboratory, compared to apparent field rates. The differences observed can be fully accounted for, despite the myth that claims the opposite. Important is that all differences are accounted for.

Factor	Effect on weathering rate	Cumulative effect on weathering rate
Buffers	1–10 times increase	1–10
Freshly ground	2–5 times increase	2–50
Temperature	1–15 times increase	2–750
Partial wetting	1–7 times increase	1–5,000
CO_2 overpressure	0.5–15 times increase	1–80,000

Many calciole ground vegetation species will be subject to significant chemical stress even if critical loads for trees are applied as the maximum deposition level. Several endangered species require BC/Al ratios significantly higher than 1 (As an example "Slåttergubbe"; *Arnika montana*, BC/Al-limit=2.5, "Gulsippa"; *Anemone ranunculoides* BC/Al-limit=3, "Lappros"; *Rhododendron lapponica* BC/Al-limit=5; Sverdrup and Warfvinge 1993). Sensitive plant species will be at a disadvantage, and tree and ground vegetation species composition may be significantly shifted over long time. The effect on such plants can be assessed by application of different BC/Al limits and response functions, using the same models and regional database. Such response functions are available for a large number of other plants (Sverdrup and Warfvinge 1993).

The reductions in growth potential will affect both commercially expoited forests as well as forests not cropped at the moment (Virgin forests, young growing commercial forest, national parks). The PROFILE model may also be used to calculate the steady state chemistry in responce to a certain deposition input. Such calculations have been performed, and the effect of certain deposition scenarios on soil chemistry and growth has been assessed (Sverdrup et al. 1995).

Multiple stresses

In order to improve the quality of the assessments shown here several tasks can be identified. Both soil scidity and excess N are known to affect plants adversely. It

is necessary to analyse the available data and separate the effects of soil acidity, excess soil N. The effect of such factors need to be integrated with effects of natural factors such as water availability and temperature. Whenever effects of VOC, ozone and NOX/SOX. How these factors affect not only growth, but also competition and succession needs to be modelled.

Who is to blame ?

The major effect of forestry on exceedance, is not through acidity input, but rather by lowering of BC in the BC/Al response parameter. This makes the role of forestry for soil acidification difficult to quantify in a unique way. The effect can be seen by comparing Fig. 5 with Fig. 4, where net uptake in Fig. 5 is set to zero, assuming virgin forest everywhere (No harvesting at all).

At present, very approximately, exceedance is caused approximately 10−20 % by forestry and landuse, 30−40 % by N deposition and 50−60 % by S deposition. The proportions vary within Sweden. The acidification from N deposition, is to approximately 60 % caused by agricultural activities and 40 % by emissions of oxidized N, mainly from transports.

Weathering in Swedish forest soils can support forest harvests up to 80−85 million m^3yr^{-1} without any adverse effects, provided there is no acid deposition. But there is not enough weathering to support both the amount wood harvested by present forestry and the neutralization of a total acidity deposition equivalent to 400 000 ton of sulpuric acid annually.

In political terms, the issue is who has the right of ownership to the weathering rate of a country. Since the weathering rate is an expression of the mineralogy, it could be considered under the same legal issues as rights to national mineral deposits. In this area, the law and rules are clear for Sweden, and a good legal proof can be made that our neighboring states have taken our mineral deposits into their use without having applied for permission for this with the Swedish government. This could be a way of establishing a legally binding liability with the polluting states, according to the "polluter pays" principle. A legal way to enforce critical loads through international courts, in case the negotiation fail ?

It is stressed that the PROFILE consist of field tested and well established sub-components. The BC/Al ratio used as limit must also be considered to be well established in laboratory experiments.

A sensitivity analysis of the PROFILE model shows that the model can calculate the weathering rate for a given soil within a maximal error range of ±40 %, in field tests, performance was ±10 %, the ANC leaching within ±25 %, and the BC:Al ratio within ±30 % of the best estimate. The analysis was performed by

Monte Carlo simulation, with an assumed error range in each parameter of ±10−100 % depending on parameter type. The error ranges are for the 10'th and 90'th percentiles, that is 80 % of the values fall in this range. The most important factors for weathering rate calculations are soil physical parameters, that is soil density, soil specific area and soil moisture content. For the calculation of the BC/Al ratio, deposition and uptake parameters are important, as well as soil physical constants and soil solution equilibrium constants. As for the calculation of ANC leaching, the most important factors are percolation, deposition and uptake and mineralogy.

The calculation of critical loads have followed a best available approach principle. This implies that the values are in a state of steady change, as knowledge of the involved processes develop and as the values can get revised and updataed. The accurate reader may then notice that the maps will sometimes change somewhat, due to such improvements. It is felt however that the major features are already incorporated in the present maps, and that no drastic changes should be expected.

Conclusions

We can conclude that a minimum deposition reduction over Sweden of 80 % on sulphur deposition and 30 % on the nitrogen deposition is required in order to protect 75 % of the groundwater resources, 95 % of the forest resource and 90 % of the surface water ecosystems.

It must be concluded that the exceedance of the critical load of acidity arise mainly from deposition of S and N, and that the role of forestry is modest, but not insignificant.

Sweden is receiving acid deposition in excess of critical load for more than 80 % of its lake, stream and forest resources. Calculations indicate that the present level of acidic deposition is starting to cause significant economical damage, that will evolve to become a factor that could significantly affect the national economy to the degree where standard of living is affected. Calculations show that significantly more than 60 % reductions in S depositions and 30 % reduction in N deposition are required in order to leave only 11'000 km^2 of forest unprotected and avoid significant economical damage to Swedish forest landowners and the Swedish forest industry.

References

Bonneau, M. 1991. Effects of atmospheric pollution via the soil. In: Landmann, G. (ed.). French research into forest decline – DEFORPA programme, 2nd report. Ecole Nationale du genie rural, des euax et des forets. pp. 87–100.

Hettelingh, J., Downing, R. & de Smet, P. 1991. Mapping Critical Loads for Europe. Coordination Center for Effects, RIVM. CCE Technical Report No. 1, RIVM Report No. 259101001.

Schulze, E-D. 1987. Tree response to acid deposition into the soil, a summary of the COST workshop at Juelich 1985. In: Mathy, P. (ed.). Air pollution and ecosystems. D. Reidel Publishing Company. pp. 225–241.

Sverdrup, H. 1990. The kinetics of chemical weathering. Lund University Press.

– & Warfvinge, P. 1988. Weathering of primary silicate minerals in the natural soil environment in relation to a chemical weathering model. Water, Air and Soil Pollution 38: 387–408.

– & Warfvinge, P. 1993a. Calculating field weathering rates using a mechanistic geochemical model – PROFILE. Journal of Applied Geochemistry 8: 273–283.

– & Warfvinge, P. 1993b. Soil acidification effect on growth of trees, grasses and herbs, expressed by the (ca+mg)/al ratio. Reports in Environmental Engineering and Ecology 1:93: 1–1348. Department of Chemical Engineering II, Box 124, Chemical Center, Institute of Technology, Lund University, 221 00 Lund, Sweden.

– , Warfvinge, P., Frogner, T., Hääöya, A.O., Johansson, M. & Andersen, B. 1992. Critical loads for forest soils in the nordic countries. Ambio 21: 348–355.

– , de Vries, W. & Henriksen, A. 1990. Mapping critical loads. Nordic Council of Ministers, Geneva. Miljörapport 1990:15, Nord 1990:98.

– , Warfvinge, P. & Jönsson, C. 1993. Critical loads of acidity and nitrogen for forest soils, groundwater and first order streams in Sweden. In: Hornung, M. and Skeffington, R. (eds.). Critical loads, Consepts and applications, HMSO. pp. 54–67.

Ulrich, B. 1985. Interaction of indirect and direct effects of air pollutants in forests. In: Troyanowsky, C. (ed.). Air pollution and plants. Gesellschaft Deutsche Chemiker. VCH Verlagsgesellschaft. Weinheim. pp. 149–181.

Warfvinge, P. & Sverdrup, H. 1992. Calculating critical loads of acid deposition with PROFILE – a steady-state soil chemistry model. Water, Air and Soil Pollution 63: 119–143.

Critical load and special characteristics of the Quark area of Finland

Alf Björklund

Department of Geology
Åbo Akademi University
FIN-20500 Åbo, Finland

Abstract

This paper presents the problems of exceedances in the Quark region of western Finland. The quality of waters in rivers and streams is poor in this area, especially in the southern parts, despite low levels of air pollution and insignificant exceedances calculated based on the concept of critical load. The reason for this is a very fast land uplift and extensive land use, which bring large quantities of sulfide-rich soil into the oxidation zone. This lowers the pH and causes leaching of toxic elements into the waters. In this area measures of land use and soil type should be included in the formulas for the calculation of the critical load. The models presently in use are not applicable on local and regional scales in Finland.

Introduction

The concept of critical load is based on the fact, that acidification of the soil will cause mobilization of aluminum and loss of base cations through ion exchange. This increases the aluminum / base cation ratio in the soil solution until a limit, considered critical for the health of the ecosystem, is reached. As a result of the application of the estimation of critical load and the corresponding exceedance, we have seen measures been taken in Europe, which have reduced air pollution more than anticipated only a few years ago. However, the usefulness of the estimation of critical load by means of the methodology applied at present may be of limited use when problems related to acidification are tackled within single countries. The main reason for this is the fact, that acid precipitation is only one and in many areas only a minor source of acidification of soil and water. Sulfur, at present the element of main interest in the estimation of the critical load, occurs in hugh quantities as sulfides in rocks and unconsolidated sediments. Sulfides, which are chemical compounds of reduced sulfur and heavy metals, are relatively stable in the groundwater zone, but above the groundwater table, where free oxygen is available, most of them oxidize readily. In this process, where sulfur oxides are formed in intermediate steps and partly escape into the soil air and into the atmosphere, the ions of heavy metals are liberated and the sulfur is oxidized to sulfate. Soil

formation and associated oxidation thus acidify the soil where sulfides are present. Under exceptual conditions sulfide oxidation may be the dominating source of acidification in areas of sulfide-rich sediments. The present paper describes such an area in the Quark region of Finland.

General description of the area

The area is situated in the Quark region on the coast of the Gulf of Bothnia (Fig. 1). The topography is very subdued with a relief normally less than 10 m. The surface inclination towards the sea is only some 1.5 m/km. The bedrock consists of Vasa granite in the central coastal parts of the area, surrounded by a schist belt (Fig. 1). The Central Finland granite dominates east of the schist belt. n the southern parts the schists are frequently mineralized with sulfides. Black schists enriched in sulfides are also common in the schist belt. The bedrock is covered with a thin relatively continuous till sheet (normally less than 5 m). In valleys and basins extensive postglacial marine and lacustrine fine-grained sediments, mainly clay (Fig. 1) and silt, cover the till and the bedrock. The postglacial marine clays, especially those from the Litorina stage of the Baltic isostatic rebound (land uplift), and

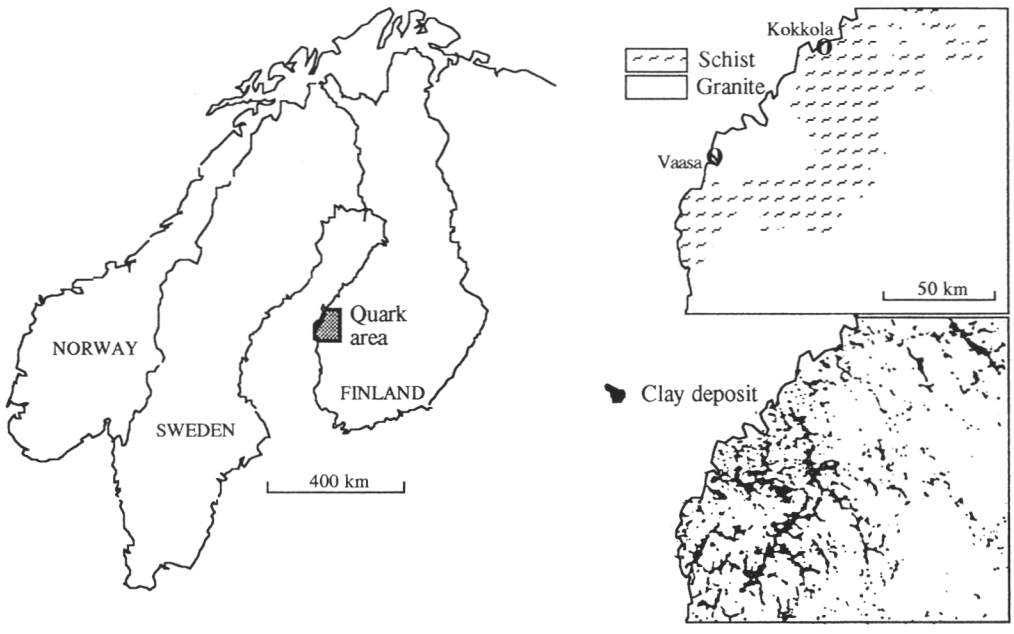


Figure 1. The location of the Quark area described. The bedrock geology (Simplified from Simonen 1980) and clay deposits (Kujansuu and Niemelä 1984) of the area.

lacustrine sediments are rich in sulfide minerals. Glaciofluvial sand and gravel are scarce. A usually thin peat cover overlays till and postglacial sediments in roughly half of the forested areas.

Less than 40 % of the area is cultivated. The major chemical, cellulose, smelter and other industries are situated in the Pietarsaari – Kokkola region in the northern parts, while only three major polluters, a coal-powered station and a chemical industry in Vaasa and a recently built cellulose industry in Kaskinen are located in the southern parts.

Special features of the area

Postglacial land uplift

The Quark region was below the thickest parts (>3 km) of the latest Pleistocene continental ice sheet, which reached its maximum extent some 20000 years ago. At that time the earth´s crust in this area was depressed to some 800 meter below its isostatic equilibrium position (Mörner 1988). Shortly after the retreat of the ice cover, the isostatic rebound (land uplift) was very fast and it has since gradually tapered off. At present the crust is slightly less than 700 m above its deepest position (Mörner 1988) and the uplift is 8 – 9 mm/a in the Quark region. The surface inclination is only some 1.5 m/km, which means that the change of the position of the coast line (sea-water withdrawal) is several meters per year, on an average. The land uplift is faster over the sea to the west of the area, which causes a decreasing inclination of the land surface with time and, consequently, silting of the mouths of the watercourses. These factors are causing water logging and impeded forest growth in large areas.

Geochemistry of till

Almost 100 % of the bedrock is covered by moraine mainly consisting of till. The Geological Survey of Finland has sampled the till deposits at two densities. During the early 1980´s 1057 samples, evenly distributed over the country, were collected in a first stage. The samples were taken from 0.5 – 2 m depth and the <62 μm fraction was analyzed for total and partial (Aqua Regia dissolution) contents of more than 20 elements (Koljonen 1992). In a second stage some 80000 till samples were collected at an average density of one site / 4 km^2. These samples have been analyzed for partial contents (Aqua Regia dissolution) of more than 20 chemical elements (Gustavsson et al. 1994). An obvious feature is a significant difference in the content-variation patterns between total contents and contents of partial extraction for many elements. This is demonstrated by K in Fig. 2. Another feature

of practical importance is a very high correlation between the partially extracted contents of most chemical elements, P and Ca being exceptions.

The maps of the contents show major regional differences in the content levels. Fig. 2 shows that the contents of partially extracted K is strongly elevated in the southern parts of the Quark area. In Fig. 3 the contents of partially extracted Al from the sampling at one site / 4 km^2 confirms the exceptual nature of the geochemistry of till in the Quark area south of Vaasa. All heavy metals and also most other elements analyzed behave in a similar manner. The anomaly pattern does not correlate with the bedrock units of the area. The partially extracted contents of most elements are very high over the southern parts of the schist belt and the Vaasa granite, while they are low over these rock units in the northern parts of the area. From an environmental point of view the partially extracted fraction of elements, which is closely related to the occurrence of sulfide-rich rocks and sediments, is of prime interest because this fraction is more readily mobilized by weathering than the one which is tied up in resistent minerals.

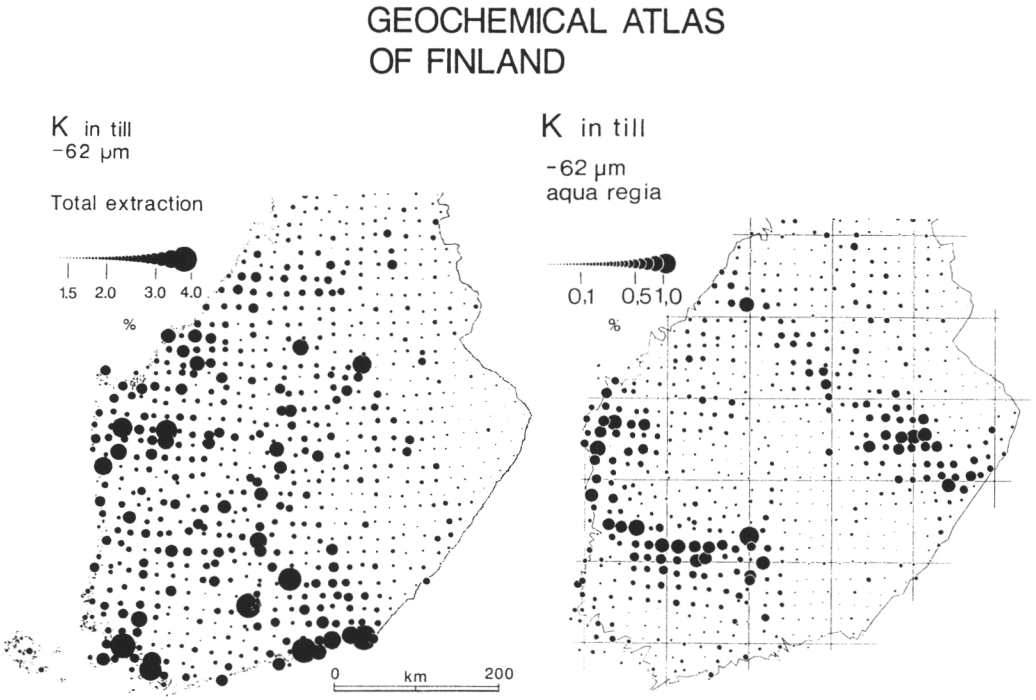


Figure 2. Total and partially extracted (hot Aqua Regia) contents of K in the <62 µm fraction of till. The maps are based on data collected by the Geological Survey of Finland.

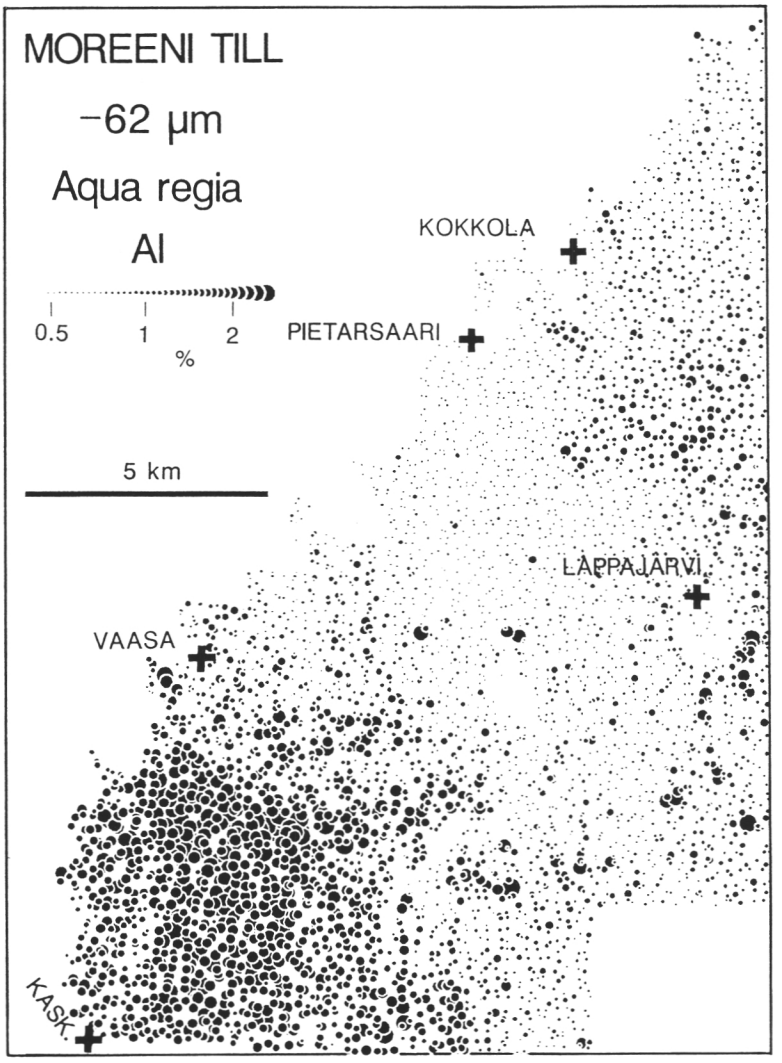


Figure 3. Partially extracted (hot Aqua Regia) contents of Al in the <62 µm fraction of till in the Quark region. The maps are based on data collected by the Geological Survey of Finland.

Sulfate soils

The main part of the sediments used for agriculture consists of marine and lacustrine organic-rich clay and silt deposits. These are frequently strongly enriched in sulfide minerals. Oxidation of the top layer of these has produced sulfate soils,

which have received much attention in the discussion of the acidification of waters. The oxidation of sulfide involves the production of H+. This process is most active in the lower parts of the oxidation zone, where the pH, therefore, may be extremely low. This is demonstrated in Fig 4, which shows that pH values below 3 are common in the lower part of oxidation zone of sulfate soils. The total area of sulfate soils in Finland has been estimated to be some 3360 km^2 (Palko 1994).

Sulfides in the postglacial clays are generally thought to have been formed through reaction between sea-water sulfate and metals enriched by organic matter in the sea-bottom sediments. During the Litorina stage of the Baltic the temperatures and the production of organic matter were higher than at present, which may have produced the higher than normal contents of sulfides in the sediments of this stage. However, if this was the main process of sulfide formation, the contents of elements in the sediments should be evenly distributed along the coast and should be unrelated to the geochemistry of the bedrock and the till deposits of the area. This seems not to be the case. Results of recent mapping of the geochemistry of sulfide clays indicate a regional geochemical pattern of element contents, broadly similar to that in till. Hence, the metal contents are higher in the southern parts of the area also in clays.

Up to the 50´s agricultural land was drained by manual ditching, whereby the groundwater table was close to the surface because of the low relief. As a consequence, deep pipe draining was possible only in restricted areas. In the 50´s and 60´s deep ditching and dredging with modern equipment in rivers and main streams became common, and this resulted in pipe draining of large parts of the farmland and large-scale ditching of forests. Hugh quantities of sulfide-rich material was thus brought into the oxidation zone over a short period of time.

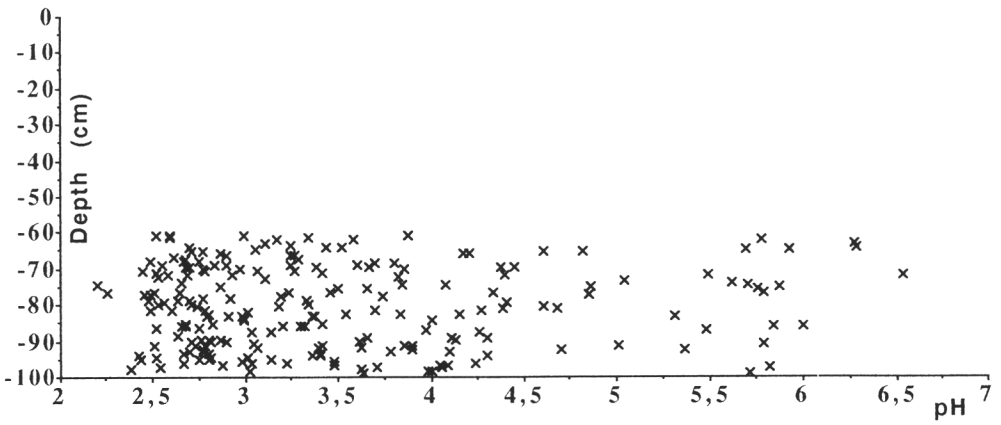


Figure 4. pH values at 60−100 cm depths in sulfate soil (clay / silt) in farmlands in Malax. The data was collected by the Water and Environment District of Vaasa.

Exceedances and ecological damages

In the Quark area the atmospheric deposition is relatively small and local sources of minor air pollution exist mainly on the coast of the northern parts of the area. Accordingly, one would expect the exceedances to be small and the environmental damages to be insignificant. This is, however, not the case. In fact, the water ecology has experienced more severe damages here than in any other region of Finland and the problems are most serious in the southern parts, where also extreme contents of toxic elements in the surface waters indicate high exceedances. Land use in combination with some most exceptual environmental features are now widely accepted as being the source of the strong diffuse load on the waters. The most direct indications that this area has experienced far stronger exceedances than other areas more subjected to acid precipitation is the very high contents of toxic elements and the very low pH in surface waters in the area. An indirect indication is the severe damages to the water ecology in the area.

Geochemistry of the surface waters

Abundant data at the Water and Environment District of Vaasa, collected during the past decades, clearly demonstrate the exceptually low levels of pH and high levels of the concentrations of toxic metals and other elements in the stream and river waters of the Quark region, especially during spring snow melting and high water flow in the fall. In Table 1. examples of the analyses of four waters are listed to show the contrast between areas of different soil geochemistry. Esse å is situated in the northern parts of the area, where metal contents are low in till. Ribäcken, a tributary to Malax å, runs along a cultivated valley of sulfate soils in a forested

Table 1. The geochemistry of some waters in the Quark region, sampled during the autumn high water flow in 1988.

Stream	Size (km^2)	pH	Zn (µg / L)	Al (µg / L)
Esse å	2000	6.7	20	120
Ribäcken	70	4.6	110	3900
Söderfjärden				
– main outlet	20	3.6	1100	58000
– pipe drainage	0.01	3.4	4700	290000

area. The contents of chemical elements in till are among the highest in Finland. The third sample is from the main outlet draining a ca 20 km^2 large meteorite crater, Söderfjärden, covered with sulfide clays. The whole basin is used for agriculture. The fourth sample is water from a pipe draining sulfide-rich clay in Söderfjärden.

The very high concentrations of heavy metals and other chemical elements in waters in the southern parts of the Quark region were also verified in a recent hydrogeochemical mapping program by the Geological Survey of Finland. Waters in some 1200 small headwater streams of forested areas were collected and analyzed by ICP-MS for more than 20 elements. The data has not yet been published. Fig. 5, which is simplified from a colour image of moving median values of Al, shows very high concentrations in the southern parts of the Quark region. Unsmoothed values at single sites are far higher than the highest content levels indicated in the map. Most elements analyzed behave much like Al and many of them (e.g., Zn, Pb, Cd, Tl) are even more focussed on the southern parts of the Quark region.

Forest ditching causes oxidation of organic matter, which has increased the amount of organic substances in the surface waters of the area during the past decades. The very high contents of elements in the waters of this region may partly be a consequence of mobilization by humic substances. In April 1991 TOC (Total Organic Content) in the waters in some main rivers, which enter the gulf of Bothnia, were 9−23 mg / L along the Finnish coast and 3−10 mg / L along the Swedish coast (Pettersson 1992). In the Quark region they are 22−23 mg / L. In such waters many heavy metals may be expected to occur in soluble metallo-organic complexes. Tyler (1985), for example, showed that the variation of Pb contents in pore water of soils is largely dependent on the variation of TOC.

Damages to fish in the area

The harmful effects on the water ecology caused by the oxidation of sulfides in the sediments as a consequence of land uplift and land use have been observed for years in the coastal areas of western Finland. Sudden fish death has occurred from time to time for a long time. The first written record of such events dates back to 1834. The first signs of a general large-scale deterioration could be seen in the 60´s in the form of a decrease of the stock of several fish species in some areas. Two features could be seen in this development; the damages were especially focused to the coastal areas south of Vaasa, where no major industries were located, and those species spawning in river mouths (e.g., burbot, perch and ide) suffered most. Burbot (lota lota) became extinct in the end of the 70´s in large areas south of Vaasa and has not returned since. It became increasingly obvious, that land use and

especially the extensive dredging of watercourses and ditching of farmland and forests carried out during the 60´s and 70´s were the cause of the problems. Hudd et al. (1984) showed, that damages to fish in the Kyrönjoki river in the 1970´s were directly related to human activities in the watercourse. The extremely high contents of elements and low values of pH during spring flooding occur during and after the spawning period of a number of fish species. This has been the main cause of damages to the fish stock because the reproduction of several fish species has been impossible in the mouths of rivers and streams during years of strong spring flooding.

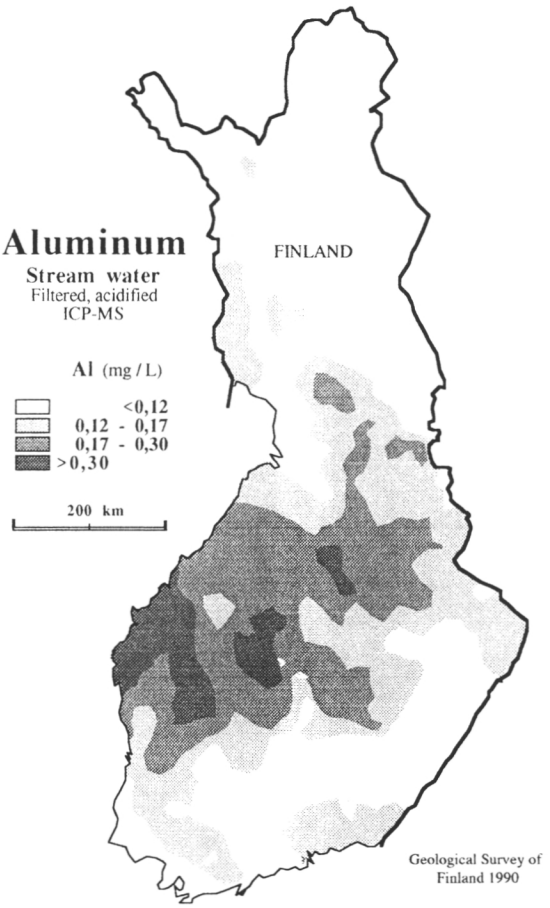


Figure 5. Aluminum contents in 1165 small headwater streams in Finland. With the permission of the Department of Geochemistry, Geological Survey of Finland.

No work has been done, however, to study the regional variation of the problems and its relation to geochemical features of the soils. Therefore, the Department of Geology, Abo Akademi University, carried out a brief study of such a possible relation. The main difficulty in such a study is that there is no record of the development of the state of the stock of fish for the recipients of separate watercourses. Therefore, local fishermen having long experience were interviewed. Four bays / river mouths, Nykarleby, Maxmo, Malax, and Norrnäs, receiving waters from drainage basins of different soil-geochemical environments were selected. For each water area five fishermen were asked to give their estimation of the development of the stock of burbot and perch. They were asked to estimate the success they expected that they would have had during different periods of time since the 1940´s if they would have used the same type of nets. 1945 was used as the year of reference to which the occurrence of the fish species were compared. The results presented in Fig. 6 show that the damage to the two fish species has been far more severe in the southern parts where the soils are very high in sulfides and toxic elements. The results of the interview are in good agreement with the opinion of the author who has followed the development of the stock of fish in this area since the late 1940´s. The results should, however, be used with caution because of the low number of fishermen interviewed and because of the subjective character of this kind of a procedure.

Conclusions

The air pollution and acid precipitation are weak in the Quark region of Finland. Therefore, serious exceedances of the critical load should not occur in this part of the county. Despite this, the southern parts of this area has experienced more damages to the water ecology than any other area in Finland. The reasons for this are well known today. Till and clay / silt sediments are exceptually rich in sulfides and heavy metals here. The fast land uplift of this very low-laying region is bringing hugh quantities of unoxidized sulfides into the oxidation zone over short periods of time. This is enhanced by dredging of rivers and streams, pipe-draining of farmland and ditching in forests. Oxidation of sulfides and peat produces H^+ ions and organic acids, which decrease the pH and mobilize chemical elements. As a consequence of these processes, acidifying compounds and toxic metals are leached from the soils and enter the surface waters, especially during snow melting and high water flow. The heavy load on the waters especially during and after the spawning time has hampered the reproduction of fish to such a degree, that a number of species have disappeared in the southern parts of the Quark region. The exceedances will not be lowered to any major degree by further reduction of air pollution. The solution to the problem in this specific area is major changes in the land-use practices.

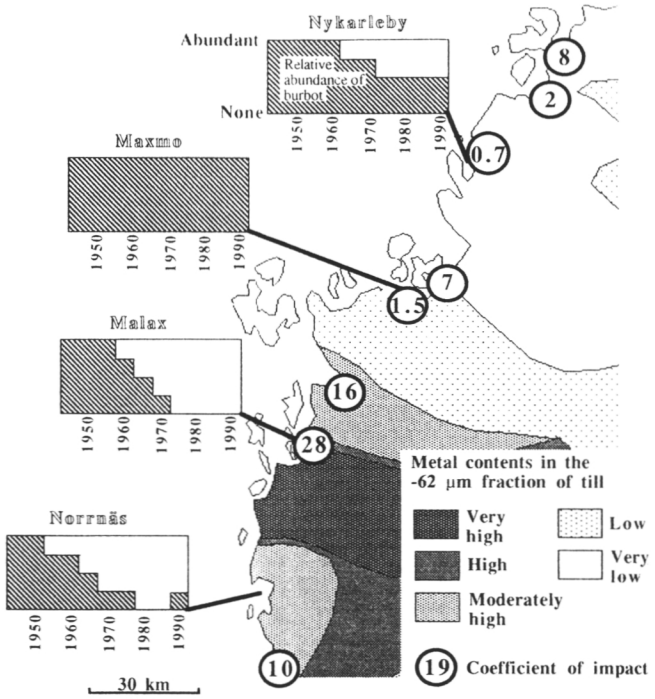


Figure 6. The pattern of the decrease of the occurrence of burbot (lota lota) along the coast of the Quark region based on interviews with five fishermen in each area. The pattern of geochemistry is based on the general occurrence of Cu, Ni, Co, Zn, Al, Mg and K (Aqua Regia dissolution) in the <62 µm fraction of till. The simplification is based on geochemical maps published by the Geological Survey of Finland.

The distribution of Al contents in surface waters shown in Fig. 5 is probably a good measure of the present situation of exceedances at the national scale. This regional pattern of Al contents is very different from the patterns of exceedances estimated on the basis of the concepts of critical load and air pollution. Therefore, it is argued, that the calculations of exceedances must be based on a variety of factors if they are to be applied at a national scale. In the Quark region the land use is a very dominating factor and must be considered in any estimation of exceedances. In this area a measure of the "critical land use" should be applied. The results of a simple approach to this is presented in Fig. 5, where a coefficient of impact of land use on the ecology of the waters in a number of river mouths has been

estimated by the calculation of the ratio between the contents of the sulfide-related metals in till and the size of the drainage basin. This is based on the concept that the impact of land use per unit area is more severe if the contents of such metals are high and less severe if the water is more diluted in the river mouth because of a large drainage basin. These coefficients of impact, presented in Fig. 5, are well correlated with the known ecological problems in the river mouths.

The groundwater table is close to the surface in the forested areas and at the same time the forest soils are locally very rich in sulfides. For this reason, roots of trees are growing in an environment of very low pH and very high contents of Al and toxic heavy metals in such areas. Therefore, negative effects on the health of the forests may be anticipated in the southern parts of the Quark region. In fact, indications of such effects have been seen in recent studies in this area (H. Raitio, pers. comm. 1994).

References

Gustavsson, N., Lampio, E., Nilsson, B., Norblad, G., Ros, F. & Salminen, R. 1994. Geochemical maps of Finland and Sweden. Journal of Geochemical Exploration, 51: 143-160.

Hudd, R., Hildén, M., Urho, M., Axell, M-B. & Jåfs, L-A. 1984. Fiskeriundersökning av Kyro älvs mynnings- och influensområde 1980 – 1982. In swedish. Report no. 243B of the Board of Waters. 277 pp.

Koljonen, T. (Ed.). 1993. The geochemical Atlas of Finland, Part 2: Till. Geological Survey of Finland, Espoo. 218 pp.

Kujansuu, R. & Niemelä, J. (Eds). 1984. Quaternary deposits of Finland. 1 : 1 000 000. Geological Survey of Finland, Espoo.

Mörner, N-A. 1988. Glacial isostasy and long-term crustal movements in Fennoscandia with respect to lithospheric and astenospheric processes and properties. In: N-A. Mörner (Ed.). Present Processes and Properties in the Lithosphere. Proceedings of the Lejondal Symposium on Neotectonics 1988. University of Stockholm. pp 4 – 11.

Palko, J. 1994. Acid sulphate soils and their agricultural and environmental problems in Finland. Ph.D. thesis. University of Oulu, Finland. 78 pp.

Pettersson, C. 1992. Properties of humic substances from groundwater and surface waters. Linköping Studies in Arts and Science No. 79. Linköping University, Sweden. 61 pp.

Simonen, A. 1980. Prequaternary rocks of Finland. 1 : 1 000 000. Geological Survey of Finland, Espoo.

Tyler, G. 1985. Orsakas skogsskador av metallförgifning ? In: Persson, H. (ed.). Vad händer med skogen – skogsdöd på väg. pp. 91-100. Liber förlag, Stockholm.

Critical loads/levels as a tool for implementing national strategies for pollution control

Håkan Staaf

Swedish Environmental Protection Agency
S-171 85 Solna, Sweden

Abstract

In Sweden the critical loads/levels concept has been adopted as a useful tool for international pollution control. This concept has also influenced national policy-making, and for several air pollutants the Parliament has formulated quantitative goals in accordance with critical loads/levels. So far direct use of critical loads/level in the implementation of the environmental goas have been limited, and the concept has not been incorporated into Swedish environmental legislation. However, work is in progress aimed at utilizing the concept in regional planning and pollution control in industry and other sectors.

Background

In Sweden and other Nordic countries the critical levels/loads concept was early adopted as a tool for international emission control. The main reason for this was the urgent need to reduce acidifying emissions in central Europe, which had been identified as the major source of acidification in Scandinavia and Finland. A working group under the Nordic Council of Ministers made the first assessments of critical loads for sulphur and nitrogen in various ecosystems (Nilsson 1986), and in 1994 the first effect-based protocol (sulphur protocol) within the UN-ECE Convention of Transboundary Air Pollution was signed in Oslo.

Policy-making

In Sweden, politicians and policy-makers soon adopted the idea of referring to nature's tolerance limits. The unified environmental proposal launched by the Swedish Government in 1987/88 stated that the ultimate goal for air pollution control should be to reduce pollution to levels "that protect human health and the environment". Similarly, with reference to long-range transboundary pollution the Environmental proposal 1990/91, stated that... "a guideline for future work should be to reduce the deposition of sulphur and nitrogen oxides as well as volatile organic compounds to levels that do not damage the environment or human

health". Basically this policy embraces the concept of critical levels/loads, although this term was not specifically mentioned.

Originally the idea was that scientifically based critical levels/loads for different pollutants would be utilized for setting politically defined target loads. These target levels/loads were to specify the amount of damage caused by a specific pollutant that could accepted by the society. Target loads defined in this way has not been used, however. Instead, long-term goals have been formulated to be in accordance with the regional 95 % critical loads/levels. For instance, the following goals for air pollution were stated in the Environmental bill passed by the Swedish Parliament in 1990/91.

* Sulphur deposition needs to be reduced by 75 % in southwest Sweden and by 50 % in other parts of southern and central Sweden.
* Nitrogen deposition must be reduced by 50 % in southern and western Götaland.
* Background levels of ozone must be reduced by 10–30 %.
* Further reductions may be required in order to protect the most sensitive organisms and ecosystems.

Similar formulations have been used in the background document for the Swedish national budget bills presented during the last few years. In the environmental programme Enviro '93 (Miljö '93) the Swedish Environmental Protection Agency (SNV) has formulated national environmental goals for acidification, eutrophication and ground-level ozone that are in accordance with critical loads/levels (Anonymous 1994b).

Implementation

The critical load concept has obviously been important in developing international control strategies for air pollution. However, for these strategies to be implemented within the countries themselves they must be integrated into national and local anti-pollution programmes. Such work, which has just begun, is faced with a number of difficulties.

Regional planning

Basically, critical loads/levels of pollutants should be considered in the regional or local planning of traffic, industry, power generation, agriculture, hausing etc. Nowadays, the concept is often used in environmental assessment analyses for individual emitters as well as for whole sectors (e.g. Lövblad and Skärby 1992). However, in southern Sweden the deposition of acidifying substances of foreign

origin exceeds the critical loads almost everywhere, and a strict application of the critical loads concept would leave no room for emissions from local sources (Figure 1). Of course, it would not be realistic to stop the establishment of new enterprises in southern Sweden or to move existing ones to areas where the critical loads are not exceeded, i.e. to the northern part of the country. Instead we must develop methods to deal with this situation, which is typical for large areas in the Nordic countries. The overall strategy could include the following steps:

a) Conduct detailed mapping of critical loads/levels for different pollutants and targets throughout the country. Existing national maps showing critical loads for acidity are based on 50x50 km grids. This degree of resolution is probably too low for planning at the local level.

b) Calculate the separate contribution of different sources – international, national and local – to the deposition in each grid. For sulphur this work is in progress at the Swedish Environmental Research Institute.

c) Develop a policy for defining the balance desired between national and international contributions to the deposition of individual grids/regions. Such a policy, which does not yet exist in Sweden, would facilitate long-term planning concerning polluting activities in different sectors. One suggestion is to limit local emisisons to a certain percentage of the critical deposition in the grid. This share might be allowed to vary depending on general level of long-range pollution, the size of the most sensitive ecosystems, and the deposition pattern of the pollutant.

d) Establish regional/local plans for future development within various sectors. Of course, these plans would have to be in accordance with the national strategy and environmental goals.

Industrial pollution control

The Swedish Government has stated its intention to ensure that by the year 2000 emissions from industrial plants and other point sources are down to levels that pose no threat to the environment or to human health. Anti-pollution measures are to be implemented in the form of environmental legislation combined with taxation and other economic instruments. So far the critical loads concept has not been introduced in environmental legislation, which is based on what is technically feasible and economically reasonable.

Emission standards are important when licensing industrial plants. Basically these guidelines reflect the best available technology for a certain type of activity.

However, in setting stricter NOx emission standards for new power plants in southern than in northern Sweden the current pollution levels and the critical loads has been taken into consideration.

Land use – forestry and agriculture

The overall environmental goals for forestry and agriculture in Sweden are to preserve biodiversity, maintain long-term soil productivity and avoid negative impacts on waters. Although, these objectives are certainly in agreement with the idea of critical loads, so far it has been difficult to set quantitive goals for these sectors. Forestry practices influence the soil nutrient status and acidity. However, it has been difficult to distinguish such effects from those of atmospheric deposition.

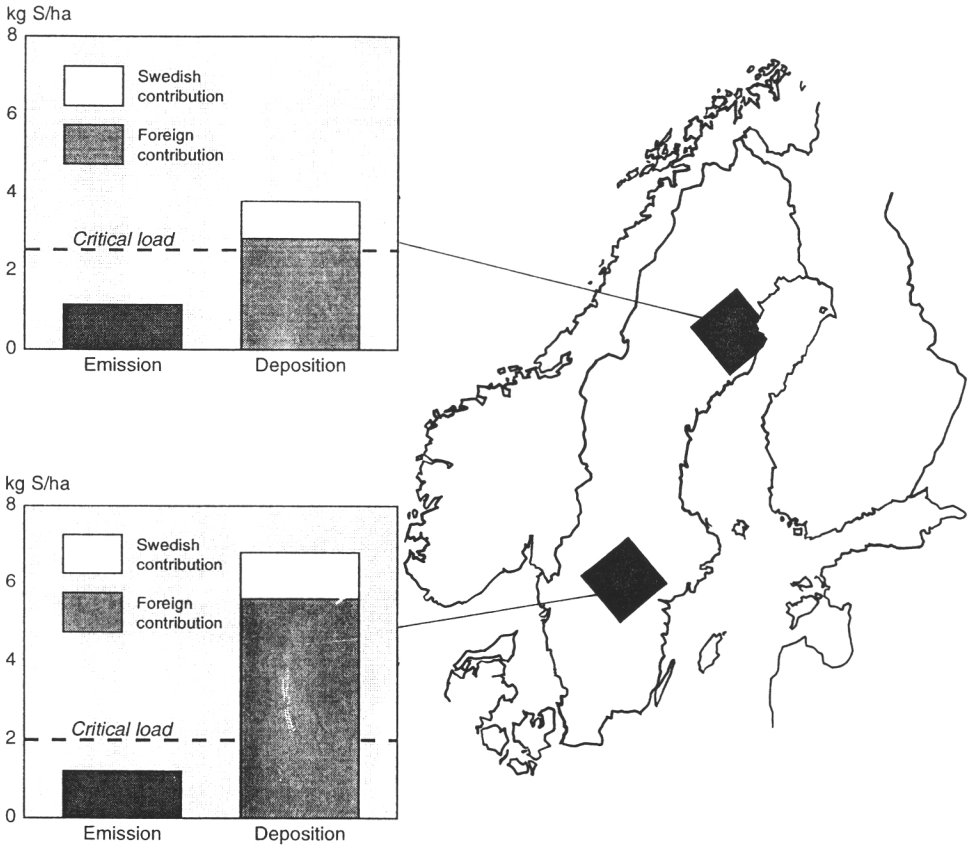


Figure 1. Emission and deposition of sulphur in two EMEP grids in Sweden. The 5 % critical loads for S are also indicated. Data from EMEP for 1985-92 (Sandnes 1993).

A few attempts have been made to consider the critical load concept in these sectors. The first example concerns the use of nitrogen fertilizers in forestry. This practice has been restricted by recommendations issued by the National Board of Forestry and the Swedish Environmental Protection Agency. These recommendations, persuant to the Forestry Act, advice against forest fertilization in southern Sweden. Originally the idea was to avoid further soil and water acidification in the most acidified areas of Sweden, but since forest companies stopped using acidifying fertilizers attenion has turned to the eutropication effects of nitrogen. According to present guidelines forests in southern Sweden should not be fertilized and the total dose over a forest rotation should not exceed 300 kg N in central Sweden and 600 kg N in northern Sweden (Figure 2).

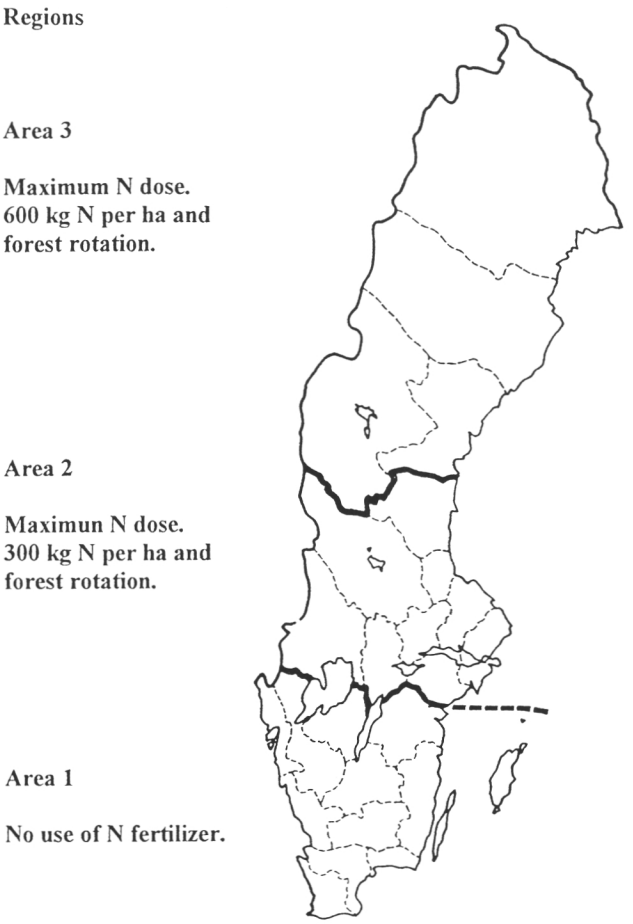


Figure 2. Recommendations for use of nitrogen fertilizer in Swedish Forestry. The recommendations were issued 1990 by the National Board of Forestry and the Swedish Environmental Protection Agency.

Another example relates to the recycling of wood ash, i.e. returning it to the forest after harvesting forest residues for fuel. It has been suggested that recycling of wood ash be manditory in future large-scale bioenergy systems based on forest biomass. Recently the Swedish Environmental Protection Agency issued guidelines for the quality of ash (Anonymous 1994a) applied in large-scale demonstration experiments. These guidelines were designed in part to restrict the supply rate of heavy metals in order to avoid short-term and long-term effects on soil biological systems.

Future development

We can anticipate that in the future the critical loads/levels concept will be more commonly utilized in environmental planning at different levels. Research in this field is given priority in the present programme for acidification research (Bertills et al. 1994). The future will show if critical loads will be directly incorporated in the Swedish legislative system for pollution control or not.

References

Anonymous 1994a. Biobränsle-Aska i kretslopp (Biofuels-Woodash in a recycling system). Swedish Environmental Protection Agency, Solna. SNV Informs. (In Swedish).

– 1994b. Strategy For Sustainable Development. Proposals for a Swedish programme. Swedish Environmental Protection Agency, Solna, Report 4266.

Bertills, U. et al. 1994. Acidifying substances and troposheric ozone – effects and critical loads. Swedish Environmental Protection Agency, Solna. SNV Informs.

Lövblad, G. & Skärby, G. 1992. Regional Miljökonsekvensbeskrivning – En metodstudie för sektorn energiproduktion. IVL Rapport B-1065. (In Swedish).

Nilsson, J. (ed.). 1986. Critical loads for sulphur and nitrogen. Nordic Council of Ministers. Copenhagen. Miljorapport 1986, 11.

Sandnes, H. 1993. Calculated budgets for airborne acidifying components in Europe. The Norwegian Meteorological Institute, Oslo. Technical Report No 109.

Critical load and critical limit values from the standpoint of the administration

Lennart Mattsson

County of Västerbotten
S-901 86 Umeå, Sweden

I have been asked to give you some thoughts about the concept of critical load and critical limit values as an regional administration policy.

First of all a few examples from our daily work with environmental protection where the concept of critical loads and critical limit values can be applied.

The County administration, besides the "National Licensing Board for Environmental Protection (KN)", issues permits to environmentally hazardous activities and also decides the permissable levels of polluting discharges. The discharges are for example nitrogen, phosphourus and heavy metals to water or sulfur dioxides and VOC to the air. The permits are, of course, associated with certain conditions.

Another important task for the County administration is to carry out environmental monitoring and to give proposals on suitable pollution control measures based on these monitoring activities. We also handle certain subsidies for measures against environmental damage. In Västerbotten the acidification of lakes and streams is considered a great problem and subsidies for liming turned over 30 million SEK this year. In this field of action the concept of critical load and critical limit values is applied in a very obvious manner as the acidification of surface water is to a large extent considered to be the result of the atmospheric deposition of sulfur and nitrogen oxides.

Experience proves that the scientifically estimated figures on critical loads have been of little significance in the case of licensing according to the "The Environmental Protection Act". Far more attention is given to the technical and economic viability of measures aimed at reducing emissions. This is true even if the critical load values are not expected to be exceeded.

Anyway we can conclude that the emissions from regional sources have been reduced with 70 % from 1980 to 1993 (Figure 1).

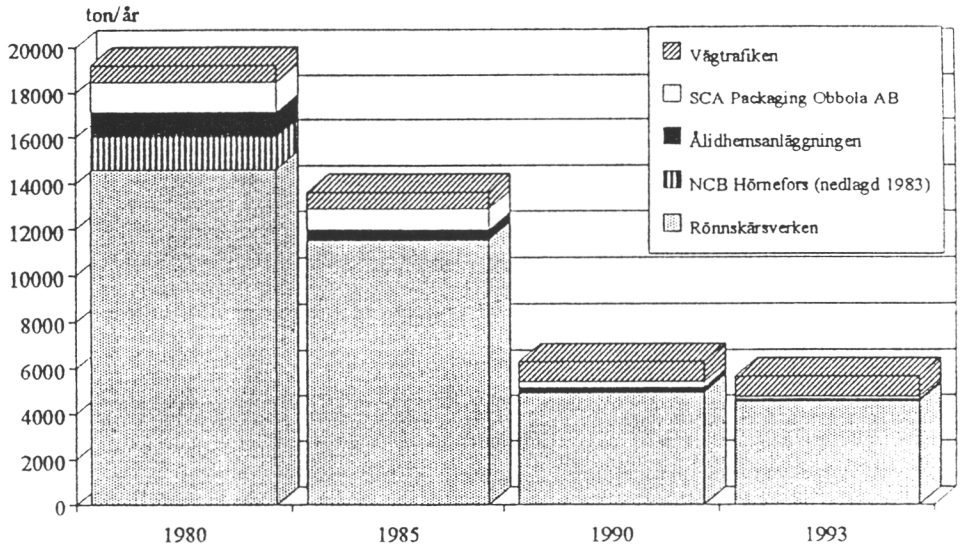


Figure 1. Discharge of sulfur from sources in the county of Västerbotten (ton/year).

As compared with the licensing process, the concept of critical loads is much more relevant when talking about the state of the environment, particularly in the question of acidification.

To be able to use the concept of critical loads as a base for our day-to-day environmental protection work, we need to know both the magnitude of the deposition of acidifying substances and the sensivity of the soil and ecosystems in our county. We can get an idea of the magnitude of the deposition from the calculations presented in, for example, the National Atlas of the Environment, where the diposition is set to 6–9 kg/ha, year for our region. These calculations have been done in a very rough manner and can not be accepted as giving a true picture from our point of view. Recently the Swedish Meteorological and Hydrological Insitute has constructed a data modelling program, "Match-Sweden", which states the deposition in a much more precise manner. In this preliminary map, based on data from 1991, the deposition in the coastal region of our county is estimated to be around 5 kg/ha year. In the rest of the county the deposition is set to 3–5 kg/ha year.

A further way to find out the magnitude of the deposition of acidifying substances from the athmosphere is to measure the amount of sulfur and nitrogen both in precipitaton and as dry deposition, a method that has been designed by the Swedish Environmental Research Institute, "IVL". The County administration operates five

stations, spread over the county, from the coast to the mountain region, where the deposition is measured in this manner (Figure 2). These stations are set up in coooperation with scientists in Finland and Sweden in a "Kvarkenrådet"-project called "Forest Vitality in the Quark Region".

According to these studies, the deposition is highest in the coastal area while the inland and the mountain regions have lower deposition. In the coastal area the deposition of sulfur is more then 5 kg/ha year and thus exceeds the critical load values which have been set to 2.5 kg/ha year for the region. This level is also politically accepted as the national goal for the environment i Northern Sweden.

In the case of the highest mountain areas, there are indications of higher depositions in certain spots. We have, therefore, initiated an investigation that will show how the deposition in high altitudes should be monitored.

To determine regional critical loads, we have to know more about the sensitivity of the soil. The sensitivity will vary greatly since the soil materials have quite different physical and geochemical properties. The County Administration of Västerbotten has, therefor, started a project mapping the sensitivity of soils to acidification. In the first part of the project, available Quaternary deposit maps are generalized and digitized using ARC/INFO. The Quaternary deposits are grouped and ranged with respect to their different capability to neutralize acidic deposition. This will result in a regional sensitivity map which is based on soil texture (Figure 3). The second part of the project includes the use of geochemical data. The sensitivity map, derived from soil texture, will be combined with data on base cation contents of the soil using the GRID technique. In this way the final sensitivity map will take into account both the varying physical and geochemical conditions within the soils. Later on the map will be checked against analyzed soil/groundwater samples. The soil sensitivity map will give us a better understanding of the environmental situation. In turn, this will improve the evaluation of the need for and type of measures, such as the liming of lakes and streams.

Finally I would like to point out that key figures such as critical load values used for illustrating the state of the environment and the effects of acidifying substances are important for both political decicions in general and for describing complicated situations to the public. But when working with the protection of the environment, the knowledge about processes and mechanisms is more important. Hence, the increased understanding of the environment that is the result of the research about critical load and critical limit values is even more important to us then the simple figures that the scientists suggest as critical loads.

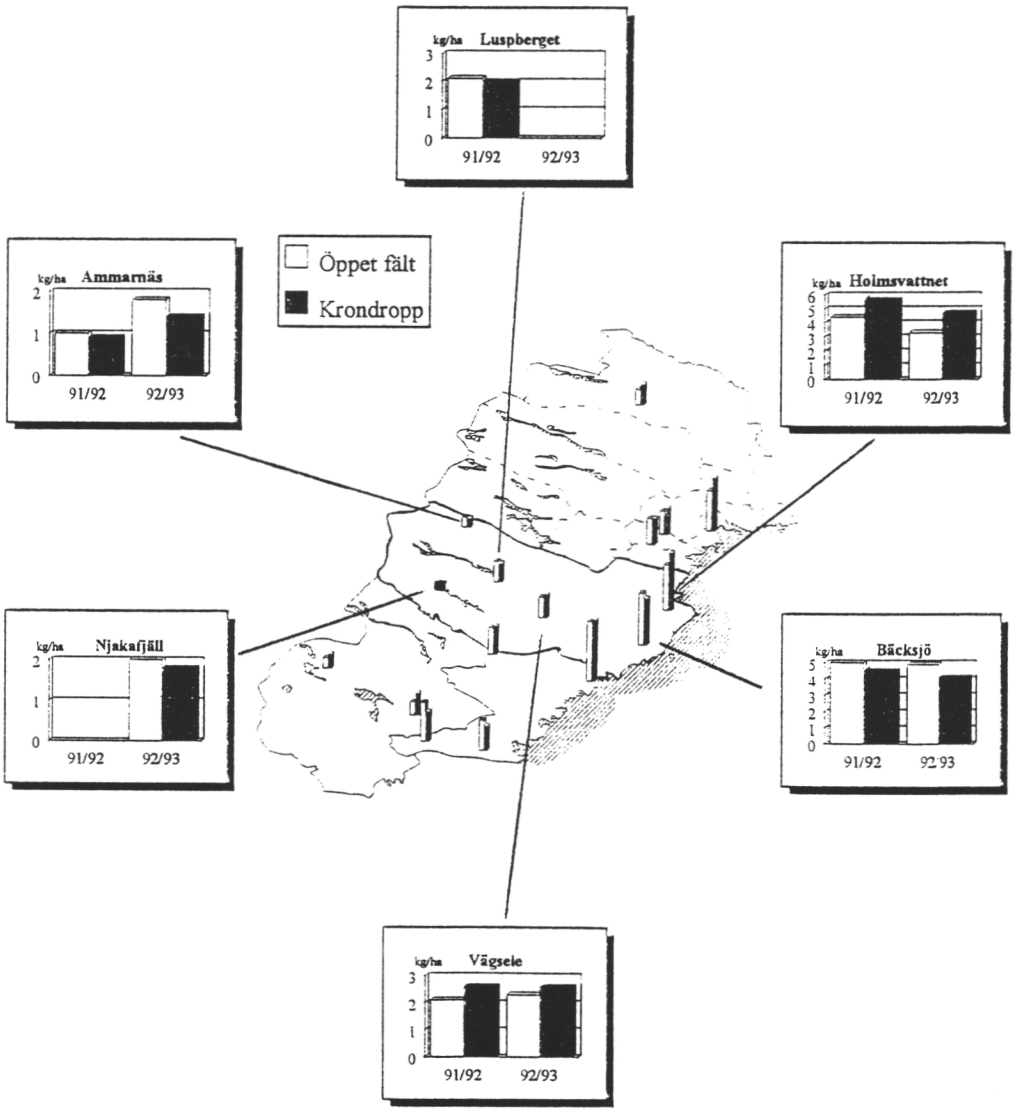


Figure 2. Deposition of sulfur in the county of Västerbotten (kg/ha/year), measured under the hydrological years 91/92 and 92/93 (Öppet fält = wet deposition, Krondropp = dry deposition).

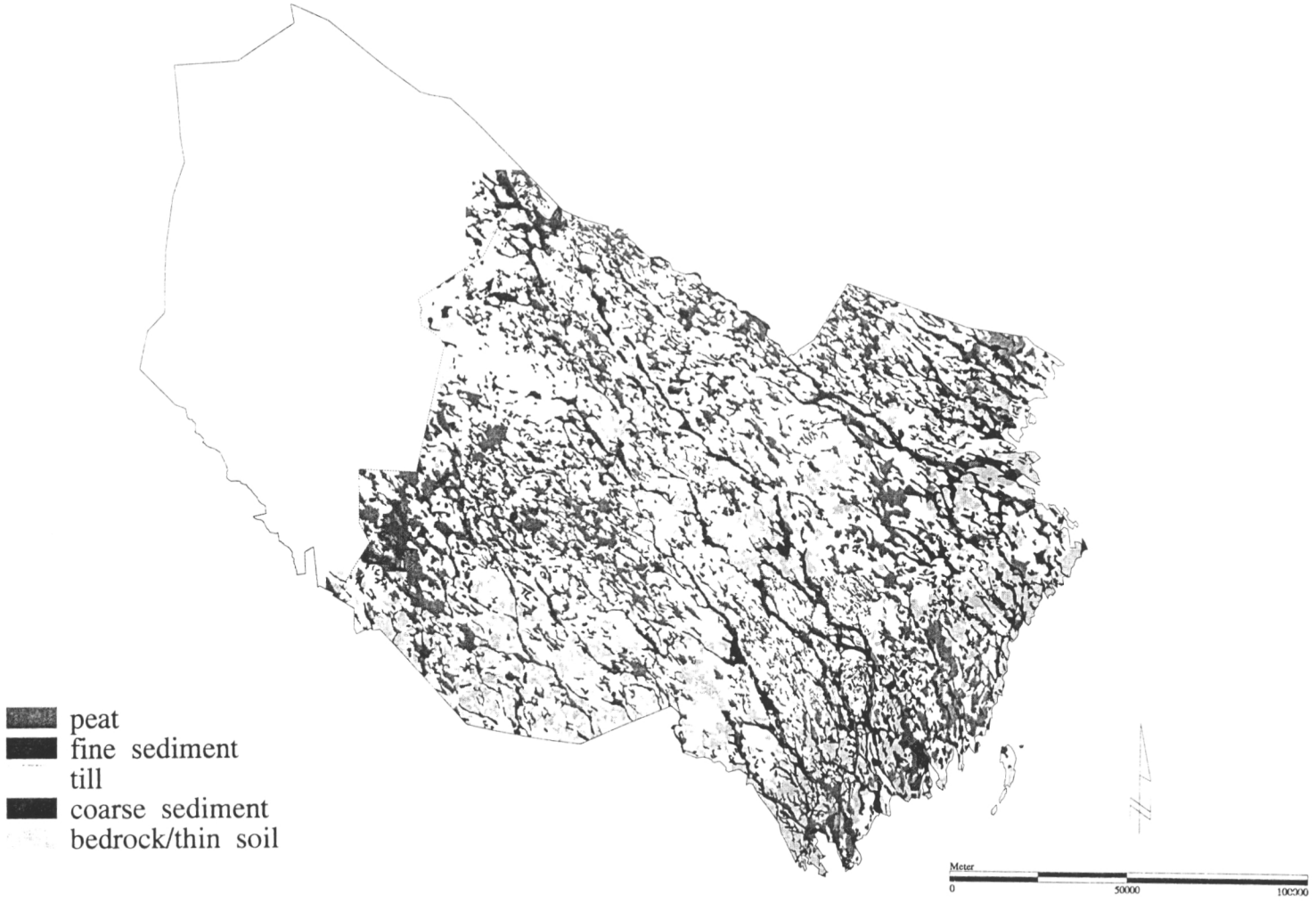


Figure 3. Generalized Queternary map of Västerbotten County below the agricultural (administrative) border.

Critical loads and critical limit values from the point of view of local government district environmental authorities

Marketta Karhu

City of Oulu
Kauppatori
FIN-90100 Oulu, Finland

Critical loads and decision-making

The notion of critical loads has been accepted as the main point of departure in international negotiations on the reduction of emissions, and it is obvious that decisions at the national and regional levels should also be based on environmental tolerance. The emission restrictions as expressed in national targets are usually indicated relative to the volume of production or as a percentage of a certain emission level, which means that less attention is given to environmental tolerance or trends in the actual state of the environment. Decisions today require more information to be available on the environmental effects of various types of emission.

Although it is admittedly difficult to set any critical target values, all goals for environmental protection should be expressed in terms of the loads and concentrations which are critical from the point of view of natural ecosystems. This would involve careful considerations at the local government level particularly as regards the magnitude of permitted emissions and the size of the areas influenced by them, at the same time as other aspects which are already in a fairly good state and thus do not require any further effort are likewise brought to the fore. The target values set for an individual local government district should be included in the targets and operative plans to be approved by its representative bodies.

Critical loading is also a good criterion for taking long-term effects into account. All decision-making should anticipate future trends and be based on estimates of the extent of the long-term detrimental effects likely to be caused by emissions of a given magnitude, for the principle of preventive action forms one aspect of the Finnish environmental legislation.

Decisions concerned with energy supplies and the reduction of emissions require a knowledge of the grounds on which emission threshold values are determined, for which it is also necessary to know the areal distribution of critical loads. A

further aspect to be remembered is that any critical threshold value can only serve as a general guideline and for the present will inevitably involve a number of uncertainty factors. For this reason the environmental decisions taken in particular cases should rely on common sense and a comparison of the various alternatives.

Critical loads and environmental monitoring

Many local government districts today monitor the state of the environment by a variety of means and with high standards of expertise, relying on generally recognised methods. Long time series, comprehensive areal coverage and other corresponding features enable a clear picture to be created of the state of the environment and trends in it.

It is absolutely essential that threshold values should be used as an aid to the interpretation of monitoring results. The data availabe on the dose-response effects of various pollutants in a natural environment are still inadequate, however, so that only a very small number of practicable tolerance-based threshold values are yet available. In the case of air pollution comparison is usually made with the norms laid down by the Council of State, which exceed the tolerance level of the environment, so that more stringent limits are currently being introduced. The guideline and normative threshold value usually employed with respect to acidification is in practice the 0.5 g level of sulphur deposition set for forest areas. It is difficult to interpret the results in an unambiguous manner, however, so that the user of such values should be sufficiently familiar with the related uncertainty factors in order to avoid misinterpretations. This means that, in addition to knowledge of the various critical threshold values, the environmental authorities should also possess extensive expertise and be able to consult outside research workers.

A feature typical of environmental monitoring in Finland today is that all instances, whether local councils or government bodies, implement programmes of their own and publish their results separately, with very little cooperation between them. Does the reason for this lie in excessive safeguarding of their own fields of interest, a lack of comparability between their findings, or an inadequate flow of information ? Whatever the cause may be, considerably more use could be made of the existing monitoring schemes, which would greatly increase the coverage of the national monitoring network.

Although it would not be possible nor even rational to focus on basic research at the local government level, the local authorities could prove to be good cooperation partners as regards practical applications of new ideas, e.g. when moving on from the theoretical examination of critical loads to practical measures. Although there are some uncertainty factors connected with the defination of threshold values, they

should be used as guidelines on the basis of existing knowledge at the earliest possible stage, as it is well-known that by the time adverse changes in natural ecosystems become visible it is extremely difficult to rectity the situation.

Critical loads and official expectations

Most severe environmental problems arise from local actions and the decisions for normalising the situation are often made at the local level. It is likely that even in the future the quality of the environment in a local government district will be determined by local characteristics such as the types of emission and factors affecting the spread of pollutants.

All protective decisions adopted at the local level should be based on the most recent research into the effects of various compounds, but this has so far been hampered by the inadequate flow of information between the parties. This is illustrated by the fact that the authorities in different fields are lamentably badly informed of the threshold values that are critical from the point of view of acidification. There is also a desire at the local government level to prepare for future decisions in good time, e.g. by examing the role of a possibly critical factor with respect to the state of the local environment.

The current trend of relying more on environmental tolerance data when determing critical threshold values can be regarded as a good sign. As far individual threshold values are concerned, the authorities' expectatons apply above all to nitrogen compounds, as the reactions of nitrogen in ecosystems are infinitely more complicated than those of sulphur and it is difficult to find suitable indicators for monitoring them. Sulphur emission problems have already largely been solved in many areas, and even the recently introduced threshold values do not pose any insurmountable difficulties. The definition of critical threshold values has proceeded furthest as regards adicification control, and the crucial problem at the local level is eutrophication in rivers and lakes, the elimination of which is still awaiting information on appropriate threshold values.

The use of the Ca/Al ratio in the calculation of critical loads for forest soils

Hannu Ilvesniemi[1)] and Michael Starr[2)]

[1)] The University of Helsinki, Department of Forest Ecology
P.O. Box 24, FIN-00014 University of Helsinki, Finland
[2)] The Finnish Forest Research Institute, Department of Forest Ecology
P.O. Box 18, FIN-01301 Vantaa, Finland

Abstract

The Ca/Al (Ca+Mg/Al or Ca+Mg+K/Al; collectively, BC/Al) ratio of the soil solution leaching from the soil system has generally been used as the "chemical criterion" in the calculation of critical loads for forest soils. A ratio of 1.0 on a molar basis has been used as the "critical chemical value" – the highest value of the chemical criterionthat does not cause a significant harmful response in the biological indicator (the tree). A steady-state mass balance principle is used to calculate the critical load. The critical load is calculated as the maximum amount of acid deposition that can be added to the system that would maintain a BC/Al molar ratio of 1.0. There is increasing concern, however, about the validity of Ca/Al ratios. While the results of some nutrient solution experiments with seedlings have indicated a critical Ca/Al molar ratio of 1.0, many other studies have found a wide range in seedling sensitivity to Al concentrations and Ca/Al ratios. Neither is there support for a BC/Al molar ratio of 1.0 from field studies with mature trees and soil leachate. The BC/Al value used is known to have a large impact on the numerical value of the critical load. Caution is therefore called for when evaluating the present critical load maps. In this paper we review the case for using BC/Al ratios in the calculation of critical loads and for including Mg and K in the BC component.

The Ca/Al ratio and critical loads

A critical load has been defined as "the highest deposition of a compound that will not cause chemical changes leading to long-term harmful effects on ecosystem structure and function" (Nilsson 1986). The critical load concept makes use of the following elements: a *biological indicator*, a *chemical criterion*, and a *critical chemical value* (Svedrup et al. 1990, Federal Environmental Agency 1993). The biological indicator is the organism used to indicate the status of the receptor ecosystem; the chemical criterion is the parameter which results in harm to the biological indicator, and the critical chemical value is the maximum value of the chemical criterion that does not cause a significant harmful response in the biological indicator.

In the case of critical loads for forest soils, the biological indicator is the tree and the chemical criterion that has most widely been used is the Ca/Al (or Al/Ca) ratio of the soil solution, with a critical value of 1.0 (molar concentrations), i.e. significant harmful effects to forests are assumed to occur if the Ca/Al ratio is <1.0 (or Al/Ca >1.0).

At high enough concentrations, Al^{3+} is known to be toxic to plants, inhibiting root growth and functioning, and ultimately plant growth and performance (Foy 1988, Schaedle et al. 1989, Cronan et al. 1989). In critical load calculations, a critical value for Al (Al^{3+}) concentrations of 0.2 $mol_c\ m^{-3}$ ($\approx$ 2 mg/l) or a molar Ca:Al ratio of 1.0 may be used. However, it is the Ca/Al ratio that has mostly been used (Downing et al. 1993). Presumably, this is because Al affects the uptake metabolism of divalent cations and because there has been some experimental evidence showing that the toxic effects of Al are, at least to some extent, offset in the presence of Ca^{2+} ions (Rost-Siebert 1983). Furthermore, the Ca/Al ratio criteria is unaffected by the water flux, an important parameter used in the steady-state mass balance (SSMB) calculation of critical loads (Hettelingh et al. 1991).

However, there is now an increasing body of evidence indicating that the Ca/Al ratio and the use of the critical value of 1.0, in particular, is not as valid as has been assumed.

Laboratory experiments with seedlings

We re-examined the results from a comprehensive series of pot experiments performed by Arovaara and Ilvesniemi (Arovaara and Ilvesniemi 1990 a, b, Ilvesniemi 1992) in the 1980s. In these experiments, Scots pine and Norway spruce seedlings of different ages and geographical origin were grown in quartz sand. Nutrients (N, Ca, Mg, K, and P) were supplied with the irrigation water in a range of concentrations. The concentrations of Ca, Mg, K and P were either optimal or sub optimal (alone or in combination) with respect to N. Furthermore, for each of these nutrition solution treatments, Al was included at eight different concentrations (0, 185, 370, 740, 1850, 2780, 3700 and 5550 µmol/l). The total dry weight and growth during the experimental period (one or two growing seasons) were measured. All the growth results are expressed as a percentage of the zero aluminium treatment, and ion activities (rather than concentrations) have been used because of the unnaturally high concentrations of nutrients involved. The ion activities were calculated using WAKO, a mathematical model developed by Prenzel (1989) and based on solution chemical equilibria presented by Lindsay (1979).

The growth of the Scots pine seedlings generally decreased with increasing Al concentrations (Fig. 1a), but Al concentrations had a less clear effect on the growth of the Norway spruce seedlings (Fig. 1b). The Norway spruce seedlings used in some experiments showed little or no response to Al, even at very high Al concentrations (50 times higher than found in natural lysimeter soil water), while in other experiments, growth sharply declined with increasing Al concentrations. The seedlings used in the former experiments were two- or three-year-old nursery

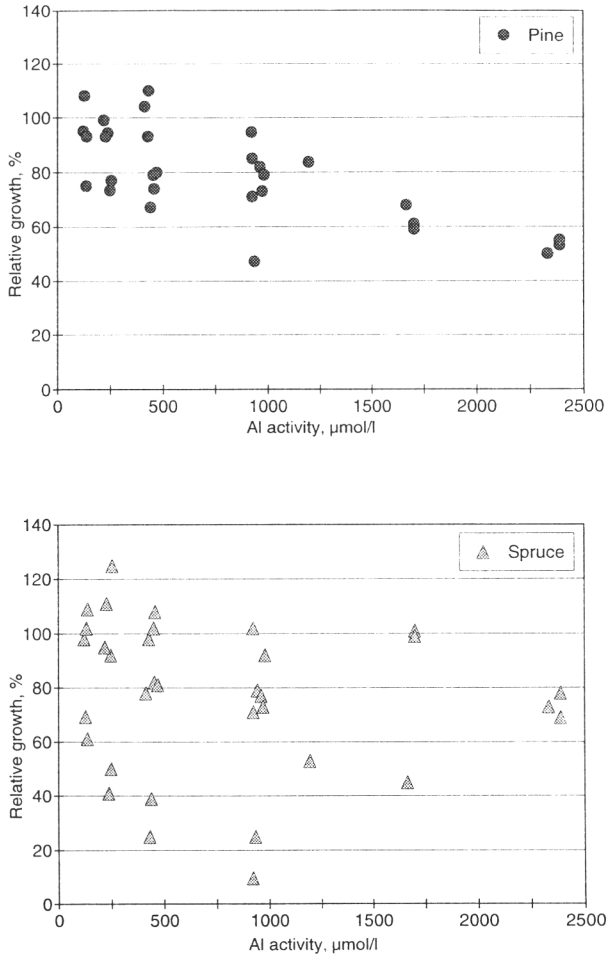


Figure 1. The growth response of Scots pine (upper) and Norway spruce (lower) seedlings in relation to treatment solution aluminium activity. (The results of several separate experiments with different nutrient solution concentrations have been combined in the figure.)

grown seedlings and their lack of growth response over the time of the experiment may be explained as "transplantation shock". The Norway spruce seedlings germinated under treatment conditions did show a growth response in relation to Al concentrations, and at lower concentrations than those having an effect on Scots pine seedlings germinated and grown under similar conditions. The conditions for germination, the age of the seedlings used, and the duration of experiment should all therefore be considered when evaluating the results of such experiments.

Plotting the same growth data against the Ca/Al activity ratio of the treatment solution shows a large range in growth response in treatments having a very low (<0.2) Ca/Al ratio (Fig. 2a). The concentrations of N and P are likely to have a significant influence on the effect of the Ca/Al ratio because these two nutrients mainly determine the growth rate and growth allocation. The growth reducing effect of aluminium therefore depends on the ionic strength and composition of the solution and is poorly described by the Ca/Al ratio. Similar conclusions have been drawn by others (e.g. Eldhuset et al. 1987, Thornton et al. 1987, and Högberg and Jensén 1994).

If Mg or Mg and K are added to the ratio (Figs 2b and c), the form of the relationship does not change, but the scale on the x-axis is increased by an order of magnitude. It is therefore surprising that the same critical value of 1.0 is applied in the calculation of the critical load regardless of whether it is the Ca/Al, Ca+Mg/Al or Ca+Mg+K/Al ratio (referred as base cation (BC) to aluminium ratios) that is used (Sverdrup and Warfvinge 1993). In particular, the inclusion of K in the ratio is questionable since the uptake of K can actually be enhanced by Al (Arovaara and Ilvesniemi 1990 b).

Ca/Al ratios in the field and forest vitality

Field data on soil solution chemistry, particularly that from the rhizosphere as opposed to the low tension bulk soil water or percolating soil water, is extremely difficult to obtain. The data that does exist is sparse (few sites) and shows that soil solution chemistry is highly variable both in space and time.

In Finnish conditions, total Al concentrations measured from lysimeter soil water rarely exceed 50 µmol/l and Ca/Al (using total Al concentrations) ratios are nearly always >1.0 (Derome pers. comm., Lindroos 1992, Starr and Ukonmaanaho 1994). If the more tightly bound soil water is collected with the centrifuge method, total Al concentration values in the range of 100 – 500 µmol/l are obtained (Tanskanen and Nissinen pers. comm.). These concentrations are low compared to those found to give a clear negative growth response in laboratory experiments, such as described above. If we also consider that the amount of Al^{3+} and mononuclear

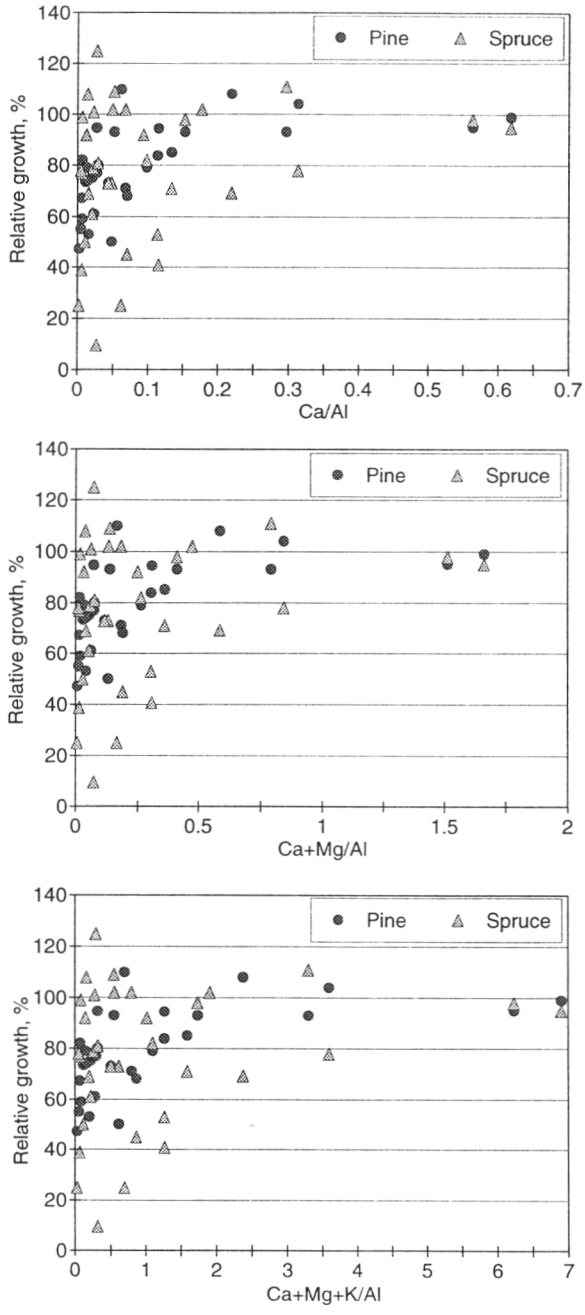


Figure 2. The effect of treatment solution Ca/Al (a), Ca+Mg/Al (b) and Ca+Mg+K/Al (c) activity ratios on the growth of Scots pine and Norway spruce seedlings. (The results of several separate experiments with different nutrient solution concentrations have been combined in the figure.)

hydroxylated Al species (the forms of aluminium involved in root blocking) constitute only a fraction of total Al concentrations in the soil solution of podzols (Nilsson and Bergkvist 1983), these values are effectively smaller still.

In France, Bonneau (1991) found that the Ca/Al molar ratio of soil water did not "reflect the state of the stands very well". Even the Ca/Al ratio of the fine pore water, including that collected from stands in decline, were >1.0. Recent field studies in Sweden (Örlander et al. 1994) have also found no relationship between relative forest growth and BC/Al ratios in soil solution.

A map of the critical load of acidifying deposition for forest soils in the Nordic countries have been produced using the SSMB method (Sverdrup et al. 1992). According to this map, 80 % of the upland forested area of Finland is receiving acid deposition in excess of the calculated critical load. The conclusion is therefore that, if present levels of acid deposition continue, most of the upland forests in Finland should or will eventually show signs of damage. However, annual surveys of defoliation (crown transparency) and discoloration carried since 1985 have not yet observed any reduction in forest vitality that can unequivocally be attributed to acid deposition (Salemaa et al. 1991, Lindgren and Salemaa 1993). A national survey of forest soil properties has also failed to show a relationship between levels of acidification and levels of acid deposition (Tamminen and Starr 1990, Starr and Tamminen 1992).

The conclusion

While the results of some nutrient solution experiments with seedlings have indicated a critical Ca/Al molar ratio of 1.0, many other studies have found a wide range in seedling sensitivity to Al concentrations and Ca/Al ratios. The variation is at least partly related to differences between species, and in the ionic strength and nutrient composition of solutions used. These factors are not taken into account in the critical load calculation. Neither is there support for a BC/Al molar ratio of 1.0 from field studies with mature trees and soil leachate. This may be because it was not the solution in the rhizosphere that was sampled, but studies of the fine pore solution have also found no relationship between forest damage and Ca/Al ratios.

Since the actual critical Ca/Al value used in the critical load calculation has a large impact on the numerical value of the critical load (Sverdrup and de Vries 1994), and in view of the above review of Ca/Al ratios, the use of the Ca/Al ratio criterion and the critical value of 1.0 in the calculation of critical loads should be evaluated.

References

Arovaara, H. & Ilvesniemi, H. 1990a. The effects of soluble inorganic aluminium on the growth and nutrient concentrations of *Pinus sylvestris* and *Picea abies* seedlings. Scandinavian Journal of Forest Research 5: 49-57.

– & Ilvesniemi, H. 1990b. The effects of soluble inorganic aluminium and nutrient imbalances on *Pinus sylvestris* and *Picea abies* seedlings. In: Kauppi, P., Anttila, P. & Kenttämies, K., (eds). Acidification in Finland. Springer-Verlag, Berlin Heidelberg. pp. 715-733.

Bonneau, M. 1991. Effects of atmospheric pollution via the soil. In: Landman, G. (ed). French Research into Forest Decline. DEFORPA Programme 2nd Report. ISBN-2-85710-033-7. ENGREF, 14 Rue Girardet, F-54042 Nancy, France. pp. 87-100.

Cronan, C.S., April, R., Bartlett, R.J., Bloom, B.R., Driscoll, C.T., Gherini, S.A., Henderson, G.S., Joslin, J.D., Kelly, J.M., Newton, R.M., Parnell, R.A., Patterson, H.H., Raynal, D.J., Schaedle, M., Schofield, C.L., Sucoff, E.I., Tepper, H.B. & Thornton, F.C., 1989. Aluminium toxicity in forests exposed to acid deposition:The ALBIOS results. Water, Air, and Soil Pollution 48: 181-192.

Downing, R.J., Hettelingh, J-P. & de Smet, P.A.M. Calculation and mapping of critical loads in Europe. Status Report 1993. Convention on Long-range Transboundary Air Pollution. Coordination Centre for Effects. National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands. 163 pp.

Eldhuset, T., Göransson, A. & Ingestad, T. 1987. Aluminum Toxicity in Forest Tree Seedlings. In: Hutchinson, T. C., Meema, K. M. (Eds). Effects of Atmospheric Pollutants on Forests, Wetlands and Agricultural Ecosystems. NATO ASI Series vol G16. Springer-Verlag, Berlin Heidelberg, pp. 401-409.

Federal Environmental Agency. 1993. Manual on methodologies and criteria for mapping critical levels/loads and geographical areas where they are exceeded. TEXTE 25: 1993. Federal Environmental Agency (Umweltbundesamt), Bismarckplatz 1, W-1000 Berlin 33, Germany.

Foy, C.D., Chaney, R.L. & White, M.C., 1978. The physiology of metal toxicity in plants. Ann. Rev. Plant. Physiol. 29: 511-566.

Hettelingh, J-P., Downing, R.J. & de Smet, P.A.M. 1991. Mapping critical loads for Europe. Coordination Centre for Effects Technical Report No. 1. National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands. ISBN No. 90-6960-011-0.

Högberg, P. & Jensén P. 1994. Aluminium and uptake of base cations by tree roots: A critique of the model proposed by Sverdrup et al. Water, Air, and Soil Pollution 74: 121-125.

Ilvesniemi, H. 1992. The combined effect of mineral nutrient and soluble aluminium on *Pinus sylvestris* and *Picea abies* seedlings. Forest Ecology and Management 51: 227-238.

Lindroos, A-J. 1992. Percolation water quality on the ion-balance monitoring plots of the Lapland Forest Damage Project in the 1990 and 1991 growing seasons. In: Tikkanen, E., Varmola, M. and Katermaa /eds.) Extended Abstracts. Symposium on the State of the Environment and Environmental Monitoring in northern Fennoscandia and the Kola Peninsula. October 6-8, 1992, Rovaniemi, Finland. Artic Centre, University of Lapland. Artic Centre Publications 4: 304-306.

Lindsay, W.L. 1979. Chemical equilibria in soils. John Wiley & Sons. New York. 449 p.

Nilsson, J. 1986. Critical loads for nitrogen and sulphur. Nordic Council of Ministers, Miljørapport 1986:11, Copenhagen.

Nilsson, S.I. & Bergkvist, B. 1983. Aluminium chemistry and acidification processes in a shallow podzol on the Swedish westcoast. Water, Air, and Soil Pollution 20: 311-329.

Prenzel, J. 1990. WAKO - A program for the calculation of complexation equilibria in aqueous solutions. Introduction and example. Reports from the Research Center Forest Ecosystems, University of Göttingen. Ser. B, 16.

Rost-Siebert, K. 1983. Aluminium-Toxizität und Toleranz an Keimppflanzen von Fichte (*Picea abies* Karst.) und Buche (*Fagus silvatica* L.). Allgemeine Forst Zeitschrift 38: 686-689.

Salemaa, M., Jukola-Sulonen, E-L. & Lindgren, M. 1991. Forest condition in Finland, 1986-1990. Silva Fennica 25: 147-175.

Schaedle, M., Thornton, D., Raynal, D.J. & Tepper, H.B., 1989. Responce of tree seedlings to aluminium. Tree Physiology 5: 337-356.

Starr, M. & Tamminen, P. 1992. Suomen metsämaiden happamoituminen (Forest soil acidification in Finland). Geologian tutkimuskeskus - Geological Survey of Finland, Tutkimusraportti - Report of Investigation 115: 7-14.

– & Ukonmaanaho, L. 1994. Metsikkösadannan ja maaveden laatu ympäristön yhdennetty seuranta valuma-alueilla. In: Mälkönen, E. and Sivula, H. (toim.). Suomen metsien kunto. Metsien terveydentilan tutkimusohjelman väliraportti. Metsäntutkimuslaitoksen tiedonantoja 527: 271-284.

Sverdrup, H. & de Vries, W. 1994. Calculating critical loads for acidity with the simple mass balance method. Water, Air, and Soil pollution 72: 143-162.

– , de Vries, W. & Henriksen, A. 1990. Mapping crtical loads: A guidance mannual to criteria, calculation, datya colection and mapping. Nordic Council of Ministers, Miljørapport 1990: 14, Copenhagen.

– & Warfvinge, P. 1993. The effect of soil acidification on the growth of the trees, grass and herbs as expressed by the (Ca+Mg+K)/Al ratio. Lund University, Reports in Ecology and Environmental Engineering. Report 2:1993.

– , Warfvinge, P., Frogner, T., Håøya, A., Johansson & Andersen, B. 1992. Critical loads for forest soils in the Nordic countries. AMBIO 21(5): 348-355.

Tamminen, P. & Starr, M. 1990. A survey of soil properties related to soil acidification in southern Finland. In: Kauppi, P. Anttila, P. and Kenttämies, K., (eds.). Acidification in Finland. Springer-Verlag, Berlin Heidelberg. pp. 237-251.

Thornton, F.C., Schaedle, M. & Raynal, D.J., 1987. Effects of aluminium on red spruce seedlings in solution culture. Env. Expt. Bot. 27: 489-498.

Örlander, G., Westling, O. & Petterson, P. 1994. Markvattens innehol av baskatjoner och aluminium och dess påverkan på tillväxt och kådflöde i kraftigt försurad granskog. Institutet för vatten- och luftvårdsforskning, Rapport B 1155.

Remarks of the Moderators

Some aspects on the concept of critical loads A moderators conclusions

Kaj Rosén

Swedish University of Agricultural Sciences
Department of Forest Soils
P.O. Box 7001, S-750 07 Uppsala, Sweden

The scientific basis

The concept of critical loads was developed in Europe in the middle of the 1980s. It is based on the mass-balance of specific elements or substances in an ecosystem chosen according to the pollutant and ecosystem which is to be studied. The idea is that the pollutant, or element fluxes created by the pollutant, should not disturb or destroy the structure and function of ecosystems in a long-term perspective.

The critical load is normally defined as. *"A quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge"* (Nelson and Greenbelt 1988).

Acceptance and implementation of the concept

As a scientifically based concept, critical loads has the advantage of being built on a neutral and straight forward principle. The logic's behind has also the advantage of being easy to understand, also for non- professionals. This is probably one major reason for the rapid acceptance of the concept among administrators and politicians on the international scene.

Although many aspects of the concept have been widely discussed and criticised, the fundamental scientific basis (i.e. the mass-balance approach) has seldom been questioned. The criticism has more focused on problems when applying critical loads on the ecosystem level. There is a wide spread opinion that lack of data and process knowledge is a major limitation to the use of the critical load concept.

The problem of acceptance is to some extent connected to the scale on which critical loads are applied. When applied on a national or international scale the demand for high resolution is lower and the demand for data often easier to fulfil,

compared to if critical loads are applied on a local scale. When applied locally, lack of data is often limiting or even disqualifying the results from a critical load calculation.

Acceptance also depends on knowledge about the scientific basis as well as the structure and accuracy of the tools (models) used to calculate critical loads. The process oriented, mass balance models used to calculate critical loads are of two major types: steady state models and dynamic models. Both types of models depend on the selection of the target organisms and the chemical criteria set to protect those organisms. At present, the correlation between target organisms and criteria are not always satisfactory validated, which is one major reason for the criticism.

However, it is of great importance to fully understand the nature of the results obtained with the model calculations. The result of the steady state models is a quantitative estimation of the input of a pollutant (e.g. acidity) at which the equilibrium situation in the soil does not result in harmful effects on specific elements of the ecosystem. Steady state models do not say anything about the time it takes to reach this equilibrium. To get an idea about the time it takes to reach a certain chemical situation in the soil, one needs to use dynamic models. Dynamic models are, however, even more complicated than the steady state models and thereby even more difficult to evaluate and validate. They also need more input data. In general the dynamic models, at their present stage of development, seems even more problematic to accept and implement at any scale than the steady state models.

As a major example of implementation on a national and international scale, the concept of critical loads was rapidly adopted by United Nations Economic Commission for Europe (UNEASE), Convention on long-range Transboundary Air Pollution as a base for the multinational negotiations on emission reductions of acidifying substances. Critical load, steady state maps are also produced regionally for some areas, e.g. in Sweden. On a local scale, the results from critical load calculations often fail to prove convincing. The large variation within a landscape or single ecosystem creates a huge demand for data, which seldom or never is available.

What about the future ?

So far the critical load concept has been successfully implemented in connection with the negotiations on European sulphur reductions. The concept has proven to be very effective from a pedagogic viewpoint and reasonably accurate for quantitative estimations of critical loads for acidity on a European scale.

Extensive work on the development of the concept for nitrogen deposition is now going on among the international scientific community. The aim for this work is the negotiations on further reductions on NO_x and ammonium. It seems as if there is a good chance of being successful also in this area.

Principally the concept should be applicable to any kind of pollutant affecting an ecosystem. Work has started on using the concept of critical loads in the area of heavy metal pollution of soils and discussions are going on concerning application of the concept on organic pollutants. It is probably too early to say anything about the prospects for success in these fields on the international scene.

The driving force for the work on critical loads has, so far, been the need for sound and neutral principles on which international negotiations could be founded. The development of the concept has mainly taken place within a relatively small international group of scientists. Although the results often, but not always, have been published in open literature, criticism has been put forward that the work on and development of critical loads has taken place with too little insight from the scientific community. Many experimental scientists are sceptical to the chemical criteria used in the critical load assessment and their relevance to target organisms.

The work on critical loads has so far to a great extent mainly been focused on the large national or international scale and less has been done to implement the concept on a the local scale. There is definitely a need for more information to regional and local decision makers on the possibilities and limitations of the concept.

In parallel to the international work, the regional and local acceptance and implementation of the concept of critical loads has to be focused. Further success depends on continuing and intensified efforts to:

⇒ validate existing or find new criteria which are well correlated to the chosen target organisms.

⇒ improve existing models and explain their advantages and limitations. Especially the latter is of great importance in order to prohibit misuse and misunderstandings.

⇒ support and stimulate data collection on regional and local scale.

⇒ provide information and assistance to local users. This should be done on all levels, i.e. from general information of the concept to data collection, data handling and tangible model exercise.

References

Nilsson, J. & Grennfelt, P. (eds.) 1988. Critical Loads for Sulphur and Nitrogen. Report from a workshop held at Skokloster, 19-24 March 1988. Nordic Council of Ministers. Miljørapport 1988:15. Copenhagen.

From the viewpoint of a government environmental officer attending a conference dealing with critical loads

Pirjo-Liisa Nurmela

County of Vaasa Administration
Department of the Environment
P.O. Box 200
FIN-65101 Vaasa, Finland

Mr Chairman, Ladies and Gentlemen

I have been invited to summarise, from the viewpoint of a government official involved in environmental matters, the papers presented and discussions held at the conference. The heading for my talk is:

From the viewpoint of a government environmental officer attending a conference dealing with critical loads.

'Critical load' as a concept

During the conference, I became aware of the multiformity of the concepts and ways of determining critical loads and critical levels. Indeed, it is important that these problems are brought under discussion. We, as government employees, have to interpret research results as we go about our work of making decisions which affect the quality of the environment – at times far into the future. It can easily happen that the numerical data available is interpreted in a manner that is too straightforward – and which may even lead to the choosing of solutions detrimental for the environment. All those who presented papers at this conference, and treated the problems surrounding the concept of critical loads, did so in a manner that was interesting and many-sided. I shall use professor Harald Sverdrup's ecosystem-oriented manner of approaching the concept of critical loads (Fig. 1). When examining the numerical data telling about critical loads, it is of utmost importance to be aware of the fundamental aspects of the estimate: which part of the ecosystem has been selected to act as the indicator; a tree or lichen species; does the value obtained tell of the critical load caused by one or more chemical factors; has it been possible to assess the joint impact of load levels and concentrations; etc.

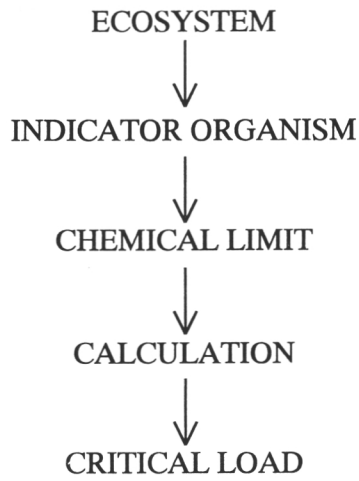


Figure 1. Critical load of an ecosystem.

As for myself, I can tell you that it is the decision maker's need for sufficient background data on the state of the environment on the regional level and of the ability of ecosystems to withstand various impacts that has served to bring government officials to sit around the same table – and come to this conference in Vaasa – with researchers. The dissemination of information has improved and so has our understanding of what is involved.

In the 1980s, on completing my studies in biology at university, the concepts of critical loads and levels had not yet been defined in the scientific fori. Having since then worked in the air-protection sector for nearly ten years, it is only during the past couple of years that I have come across these concepts. The founding of the working group organised by the Nordic Council of Ministers in 1986 in Sundvollen, Norway, is considered to have launched the scientific examination of the concept of critical load with regard to nitrogen and sulphur. A couple of years later, the concept of critical load had a definition. It is understood to refer to the maximum amount of sulphur or nitrogen deposition that does not yet cause long-term damage to the structure or functioning of an ecosystem. The methods used in determining critical loads and levels were documented in 1989. The Coordination Center for Effects was established a year later in Bilthoven, The Netherlands, for the purpose of combining national critical load calculations and produce standardized charts of critical loads in Europe. At first, the interest in defining critical loads and

concentrations was in sulphur. According to the latest estimation model, critical loads can be divided to distinguish separately the effects of sulphur and nitrogen.

We have all observed that the concept of critical load is a young one. A lot of work has been done in recent years to determine the bearing capacity of both terrestrial and aquatic ecosystems – and still there are challenging questions for further research.

Critical loads and concentrations in Europe

The most sensitive ecosystems in Europe from the point of view of critical loads are mainly to be found in the Nordic countries. In the case of Finland, these areas are in the south-western and southern parts of the country, along the west coast and in parts of Lapland. Critical load levels in Finland have been examined in the context of an extensive acidification project (HAPRO), implemented in 1985 – 1990. According to the results obtained, acid deposition exceeds the critical loads of forests and waters almost throughout Finland. In Sweden, it is estimated that 82 % of the forest land receive acid deposition in excess of their critical load levels. Once it is implemented, the Second Sulphur Protocol, signed in Oslo last summer, will mean that in the year 2010 the critical sulphur load will continue to be exceeded in the case of 10 % of Europe's terrestrial and aquatic ecosystems. Another matter to note is that this estimate takes into account the effects of only sulphur deposition on the critical load. The combined effects of sulphur and nitrogen deposition, direct concentration effects, ozone effects, etc., are not included in the assessment. At its maximum, the load will exceed the bearing capacities of ecosystems in Central Europe. According to the currently used calculation models, the overshooting of the critical load levels will be at minimum in the Nordic countries. However, it should be remembered that even slight overshooting can result in serious ecosystem damage. This being the case, the amount by which a critical level is exceeded does not serve as an indication of the scope of the resultant damage. The only thing we can be certain about is that damage will ensue when critical load levels are exceeded.

It is interesting to note that Swedish researchers have made estimates of the potential losses to forestry caused by acid deposition. This is matter of great economic interest to the Nordic countries. Indeed, it gives rise to a question of fundamental interest: Who is legally entitled to make use of the soil's ability to resist acidification (its property of weathering)? Can a nation, in the spirit of the "polluters will pay" principle, demand another nation to pay compensation for the use the other nation has made of the weathering properties of the claimant nation's soil? Another issue at stake simultaneously is that of airborne pollutants not respecting national boundaries.

Critical loads and reducing emissions

The programmes aimed at reducing emissions in Finland have been based on what may be called a percentage policy, although the grounds presented in the 80 %-reduction-in-sulphur programme approved in 1991 do mention additional reductions in emissions being essential for acid deposition to be reduced. At that time, acid deposition was believed to be 2–4 times greater than the critical deposition based on the environment's bearing capacity.

The recently published Second Sulphur Protocol is, in fact, the first emissions-reducing programme to be based on critical loads.

Finland's Air Protection Act came into force in 1982. In accordance with this act, environmental permit officials define production-plant specific atmospheric emissions limits. The Air Protection Act requires that when emissions limits are being set, the decision maker must take into account matters such as technical and economic preconditions for the reduction of emissions. Similarly, the Act requires that the properties of the area affected by the activity must be taken into consideration. This stipulation in the Act has been understood to mean that demanding reductions in emissions could be backed up by referring to the environment's properties, e.g. high natural sulphur concentration of the soil and deposition levels exceeding critical load levels. In practice, however, the implementation of the Act has been characterised by emphasis being laid on the realization of reduction targets expressed as percentages and the technical and economic possibilities for achieving reduced emission levels. This may be explained by the understanding that individual emission sources in relation to the impact area's overall deposition level are of marginal significance. However, the effects on atmospheric concentrations of pollutants have been significant at times. It should be pointed out that even if emissions from one major emission source were to be reduced to zero, it would not necessarily solve the complicated problem of critical load levels within the area affected by the source. On the other hand, our own emissions and the steps taken to reduce them clearly have an effect on the deposition levels. Fig. 2 shows the estimated situation with respect to internal emission sources and the resultant sulphur deposition. The estimate for the year 2000 includes the assumption that emissions have been reduced by 80 % compared to the levels in the year 1980. Furthermore, one must take into consideration the observation that a considerable part of the acid deposition is of transboundary origin, and therefore measures taken to reduce emissions always result in reductions beyond the boundaries of one's own country.

The current Air-Protection Act in Finland was drawn up at a time when the concepts of critical load and level did not exist. This legislation is now under review and it will probably include a requirement for the application of BAT

technology and of environmentally most appropriate practice. Even considering this review, I am of the opinion that the Act will still not be one that is fundamentally ecological in terms of its basic premises.

In accordance with the EEA convention, Finland is implementing the environmental legislation of the European Union, the directives defined within the European Community. In its present form, the EU's environmental legislation sets the minimum level for atmospheric emissions and member countries are required to stay below it. The sensitive ecosystems of the northern regions should be protected through national legislative measures. Emissions standards adequate to protect the structures and functioning of ecosystems in Southern and Central Europe may not

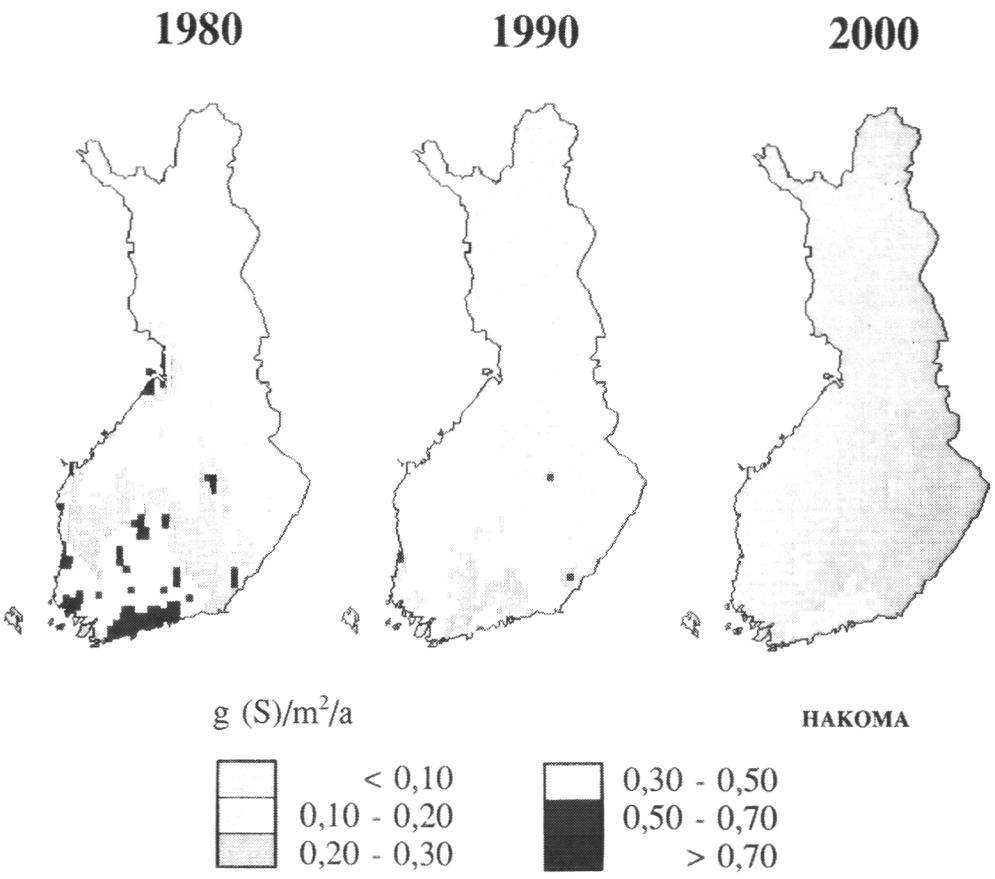


Figure 2. Sulphur deposition originating from domestic emission sources. The estimate for the year 2000 includes the assumption of a reduction of 80 % from the 1980 deposition level.

be sufficient to do the same in the North. This is why we in the Nordic countries, we in Finland, should make special efforts to distinguish between regional differences in critical loads and critical level. Nevertheless, one must realise that if more stringent national emission norms were to result in what may be called 'obstacles to trade', it is unlikely for such norms to be issued within the EU. With Finland, Sweden and Norway most probably joining the EU, we must do everything in our power to try and promote a high level of environmental legislation within the EU. In order that our 'Euro lobbyists' might have an adequate knowledge base for their preparatory work, we must see to it that enough information about the state of the environment is placed at their disposal (not forgetting the state of the regional environment, e.g. variation in critical loads in different parts of the country).

Environmental official's relationship with critical loading

Environmental officials are required to supervise the state of the environment in the region under their jurisdiction. These are also the officials who define plant-specific emission limits. In other words, we need information on both the scope of the load and the bearing capacity of the ecosystems in the area. The said officials should be in possession of information such as the possible long-term effects of various emissions on the critical loads in the area. At present our knowledge of these area-specific critical loads is incomplete.

The discussions held during the conference revealed a concern over the question of will estimates concerning cases of critical loads being exceeded be made use of by directing new load-imposing activities to areas where critical loads are least exceeded or to areas which are relatively clean in regard to loading. This point of view should be borne in mind as something that should never be allowed to happen. The principle applied in locating activities must be that of not worsening the current state of the environment but rather of striving to improve it.

In conclusion

The legislator's relationship with the bearing capacity of ecosystems, mainly one of technical-economic kind, has not, in my opinion, helped to sufficiently direct reduction strategies complying with protection of the environment. Neither has the said relationship complied with the principle of sustainable development. Economic losses caused by critical load excesses (to forestry, for instance) and the difficulty of predicting what the long-term and multi-stress consequences might be have been necessary to make the legislator assume a new view in the matter – the ecosystem view.

A widening of perspective also makes it necessary to see to it that regional research needs and research resources are in balance. We continue to need regional research alongside national projects.

The strategies for reducing emissions should be modified by moving away from pollutant-specific reduction targets to an integrated emission reduction strategy based on critical loads determined from the joint effects of the partial factors. Within the EU, there will be move over to integrated emission reduction and monitoring in accordance with the ongoing formulation of the IPC directive. From the ecosystem point of view, this development calls for a more comprehensive examination of the effects on the environment – and it also calls for a more comprehensive administrative policy of reduction in emissions.

In the above, I have presented what my personal view of the benefits of this conference. It is my belief that differences in experiences is a resource – and thus this, my contribution, will hopefully serve in fulfilling the goal of this conference – that of bringing together experts involved with the subject of critical loads. I also wish to express my appreciation to all those of you who presented papers and took part in the discussions. Thank you.

Critical loads and critical limit values
The legal point of view

Kari Kuusiniemi

Helsinki University of Technology
Institute of Law
Otakaari 1, FIN-02150 Espoo, Finland

1. *The concept of critical load* is defined as the highest deposition of a compound that will not cause chemical changes leading to harmful effects on an ecosystem's structure and function. The concept has gained acceptance in (international) environmental law, even though it refers to the results of biotic environmental sciences.

In the so-called NO_x Protocol (Protocol concerning the Control of Emissions of Nitrogen Oxides or Their Transboundary Fluxes, 31st of October, 1988) to the 1979 Geneva Convention on Long-range Transboundary Air Pollution the concept has been defined as follows:

> Critical load means a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge (Art. 1, point 7).

In the so-called Second sulphur protocol (Protocol to the 1979 Convention on Long-range Transboundary Air Pollution on Further Reduction of Sulphur Emissions, Oslo 1994) critical load is defined similarly. But extremely important is that the Protocol is based on the idea to reduce acidifying effects of sulphur emissions to ensure that depositions of oxidized sulphur compounds do not exceed critical loads for sulphur. But, who is to define which harmful effects are *significant*? It is not the scientist, it is the polcy-maker, naturally assisted by scientists.

The need to continue scientific and technical co-operation to elaborate further the approach based on critical loads and critical levels, including efforts to assess several air pollutants and various effects on the environment, materials and human health, is recognized in the preamble of the Protocol. The Geneva Convention and the NO_x Protocol are, in contrast, based on the principle of best available technology, BAT. However, the basic obligation to ensure that depositions do not exceed the critical threshold, stated in the Art. 2, point 1 of the Oslo Protocol, is

compromised by requisites "as far as possible" and "without entailing excessive costs". Nevertheless, the Oslo Protocol may be called the first effect-based protocol to the Convention.

Critical loads refer to the total deposition of acidifying compounds. The concept of *critical level*, in contrast, means the concentrations of pollutants in the atmosphere above which direct adverse effects on receptors, such as human beings, plants, ecosystems or materials, may occur, according to present knowledge.

2. The goal to keep deposition of pollutants within the limits of critical loads is one criterion for *sustainable development*. If the loads widely exceed the tolerance level of the nature the idea of sustainability has been lost – in the same manner as plant and animal species that cannot bear the loads may even become extinct. Also concentrations exceeding the critical levels may cause irreversible damage on vegetation. The aim at preserving *biodiversity* would suffer.

3. On one hand, *legal regulation* concerning environmental pollution has a long tradition, on the other hand one can say that the principles of pollution control law are still vague and undergoing change. Reasons can be given for both opinions.

Some rules about the interrelationships between neighbouring real estate owners dating back to 2000-year-old principles of Roman Law can still be called the basis of environmental legislation. These rules provided a ban on unreasonable, exceptional harm caused by a land user to his neighbour. Nowadays, rules on environmental pollution often limit emissions into the environment according to certain technological and economical considerations.

Thus, legal environmental regulation can be directed either to *emissions* or to *consequences*. It seems to me that in the 70´s and early 80´s environmental legislation has been concentrated on reducing of emissions of major stationary sources. New techniques have in many plants enabled, e.g., to reach desulphurization rates of over 95 per cent. But, in spite of these encouraging results the quality of the environment has not improved as much, somewhere it has, in all, become worse. The efficiency or the appropriateness of environmental legislation can be evaluated solely on grounds of the merits, i.e. the quality of the environment. Hence, legal rules shall be attached not only to reduction of polluting emissions but also to the resulting state of the environment. Here the ancient rules of neighbourhood law shall be modified so as to set ban on causing remarkable harm to the environment or exceeding the limits of critical loads or critical levels.

4. Environmental legislation includes numerous *vague norms* describing obligations to take precautionary measures to reduce pollution of the environment (e.g. use the best available technology to control the emissions) or undesirable effects of

polluting activities (ban on sanitary nuisance etc.). These flexible legal rules are often concretized by *technical standards*.

Standards about maximum levels of emissions are called *emission limit values* (ELV). The concept of limit value implies that the standard is binding; otherwise one could talk about guidelines. ELV´s are based on progressive technology. The internationally established principle of BAT presupposes that emissions are reduced by using the most effective means, for example as follows:

> The term "Best Available Techniques" (BAT) signifies the latest stage of development of activities, processes and their methods of operation which indicate the practical suitability of particular techniques to form the basis of emission limit values for preventing, or where that is not practicable, minimizing emissions to the environment as a whole, without predetermining any specific technology or other techniques.

From the recipient-orientated point of view it is crucial to define exactly the prohibited adverse effects on the environment. *Environmental Quality Standards* (EQS) point out, e.g., how high sulphur deposition can be tolerated on a certain forest area. EQS objectivate the goals related to the state of the environment.

When preparing EQS, information about critical loads and critical levels should be recognized and taken into account. However, the aspired level (politically defined target load) may, e.g. because of economic reasons, differ from the desirable level. The requirements set in human statutes often deviate from the contents of the laws of the nature. Anyhow, emphasis should be put on harmful environmental *effects* and on the crucial question: how much do human beings and the nature bear?

Methods which would make it possible to assess the effects of combined stresses (pollutants and other biotic and abiotic stresses together) and multiple pollutants on ecosystems should be developed to assist standard-setting (synergistic effects). Actually we would need a great deal of EQS – EQS for various pollutants, for combined existence of various pollutants and for different areas should be differentiated.

Another option is to rely on case-by-case judgments of the authorities who are responsible for issuing environmental permits. This alternative would presuppose that the results of scientific research on critical loads and critical levels is available to the authorities. It is, of course, only applicable to point sources.

In some cases, e.g. to implement directives of the European Union (EU), quality standards are to be directly applicable, *legally binding*. At any rate, quality standards would show the permit authority the estimated tolerance level. The use of legally binding quality standards is not free from problems, even though it is

tempting to think about a citizen´s legal right to a decent quality of the environment. If, say, more than a half of the sulphur deposition comes from abroad it is not realistic to claim that all emissions from local sources should be prohibited, even though the critical load level would be surpassed.

5. The so-called *dual approach* presupposes that the acceptability of a polluting activity, e.g. when deciding the environmental permit application of the plant operator, depends both on the *technical solutions* to minimize emissions and the *impacts* of these emissions and, as well, other emissions facing the environs of the plant. Requirements set on the *reduction of emissions* as well as on *the quality of the environment* have to be fulfilled. The system is based on a *feedback-mechanism*: emissions are reduced in order to reach the level of quality objectives, but even stricter emission standards are set if the acceptable level would be exceeded.

In the proposal for a framework directive on integrated pollution prevention and control (com(93) 423 final, hereinafter the *IPC directive*) measures and procedures to prevent or minimize emissions from industrial installations of the Community, so as to achieve a high level of protection for the environment as a whole, would be provided for. The Directive has, in other words, an integrated approach on the problem of pollution.

As the main principles of the proposed Directive show, dual approach would be introduced.

> "The conditions of the permit shall normally at least include emission limit values for the substances and preparations listed in Annex III likely to be emitted from the installation, - - . - - The emission limit values and equivalent parameters shall, subject to Article 9, be based on the best available techniques and shall at least meet those which have been set at Community level, - - "
>
> "Member States shall take the necessary measures to ensure that emission limit values are set with the aim of ensuring that environmental quality standards will not be breached."
>
> "Where any such standard requires stricter conditions than those achievable by the use of the best available techniques, additional measures shall be required in the permit."

Reduction of emissions by using the best available technology would be a kind of a minimum content of a permit judgment. The level of BAT is described in the emission limit values for various substances. If EQS cannot be fulfilled by using BAT, stricter conditions shall be set to protect the environment.

6. The battle against air pollution must be based on *international co-operation*. Air pollutants do not acknowledge national boundaries, is often repeated. This is also true of water pollution in an international water basin (such as the Baltic Sea). A problem that is international by nature cannot be solved by national measures only.

The above-mentioned Second Sulphur Protocol presupposes that the Parties shall, as a minimum, reduce and maintain their annual sulphur emissions in accordance with the timing and levels specified in annex II. The percentage emission reduction of Finland and Sweden shall be at least 80 in year 2000 (base year 1980). Even if these countries are able to fulfil their obligations some other may be not. For example, countries with their economies in transition (read: former socialist countries) probably do not reach the reduction figures specified in the annex, even though the emission caps are much higher for them. And, as a consequence, sulphur deposition in neighbouring countries may still increase, in spite of powerful domestic measures to implement the obligations of the Protocol.

A new instrument, the role of which might be significant in the future, is called *joint implementation*. A provision about joint implementation is included in the Second Sulphur Protocol (Art. 2, point 7):

> The Parties to this Protocol may, - - , in accordance with rules and conditions which the Executive Body shall elaborate and adopt, decide whether two or more Parties may jointly implement the obligations set out in annex II. - - .

The possibility to implement the emission reductions specified in the annex jointly would provide both environmental and economic benefits. The cost-effectiveness of desulphurization investments may be ten times higher in countries with economies in transition than in highly industrialized countries where the rates of reduction already are remarkably high. At the same time, sulphur deposition in both countries would decrease because the volume of transboundary emissions could be diminished with help of investments on pollution control, partly financed by the industrialized country. This country could, in turn, count to its credit a fair share of the reduction achieved jointly in the host country of the investment.

The Finnish Forest Research Institute, Parkano Research Station. Research Papers.

No. 1 Eero Paavilainen ja Veikko Koskela. Parkanon tutkimusasema 1961-1970. 1972.
No. 2 Eero Paavilainen ja Seppo Kaunisto. Männyn koneellinen istutus Mara-istutuskoneella verrattuna käsinistutukseen avosuon metsityksessä. 1973.
No. 3 Tutkimuspäivän esitykset. 1976.
No. 4 Seppo Kaunisto. Alkkian kenttäkokeet 1961-1975. 1976.
No. 5 Kaarlo Kinnunen. Kylvö- ja istutusajankohdan vaikutus kennotaimien alkukehitykseen. 1977.
No. 6 Kaarlo Kinnunen. Männyn kylvömenetelmien vertailua. 1977.
No. 7 Tutkimuspäivän esitykset. 1978.
No. 8 Tutkimuspäivän esitykset. 1979.
No. 9 Tutkimuspäivän esitykset. 1980.

The Finnish Forest Research Institute. Research Papers.

No. 94 Tutkimuspäivän 1982 esitelmät. 1982.
No. 108 Kaarlo Kinnunen ja Ilkka Laurila. Erilaisten männyntaimien juuriston ja verson alkukehitys karuhkolla moreenimaalla. 1983.
No. 116 Hannu Raitio. Hypoteesi männyntaimien kasvuhäiriöiden synnystä taimitarhoilla ja kivennäismailla. 1983.
No. 137 Metsäntutkimuspäivä Porissa 1983. 1984.
No. 144 Seppo Kaunisto. Alustavia tuloksia kasvuhäiriöisten männyntaimien kehityksestä suonpohjan turpeella. 1984.
No. 177 Seppo Kaunisto. Metsityskokeet Kihniön Aitonevalla. 1985.
No. 184 Metsäntutkimuspäivä Seinäjoella 1984. 1985.
No. 202 Seppo Kaunisto ja Kaarlo Kinnunen. Taimilajin ja taimitarhalla todetun kasvuhäiriön vaikutus männyn taimien alkukehitykseen maastossa. 1985.
No. 215 Kaarlo Kinnunen. Männyn kylvötuppaiden harventamisesta. 1986.
No. 225 Hannu Raitio ja Eero Tikkanen. Nuorten mäntyjen kalsium- ja magnesiumtalouden häiriö kuivalla kankaalla. 1986.
No. 235 Metsäntutkimuspäivä Tampereella 1985. 1986.
No. 236 Seppo Kaunisto, Kaarlo Kinnunen, Sulo Lehtinen, Kalle Nevanranta ja Jorma Tukeva. Alkkian kenttäkokeet 1961-1986. 1986.
No. 270 Metsäntutkimuspäivä Porissa 1986. 1987.
No. 300 Metsäntutkimuspäivä Seinäjoella 1987. 1988.
No. 337 Metsäntutkimuspäivä Tampereella 1988. 1989.
No. 369 Metsäntutkimuspäivä Porissa 1989. 1990.
No. 394 Metsäntutkimuspäivä Nurmossa 1990. 1991.
No. 423 Metsäntutkimuspäivä Tampereella 1991. 1992.
No. 470 Metsäntutkimuspäivä Porissa 1992. 1993.
No. 495 Metsäntutkimuspäivä Seinäjoella 1993. 1994.

Distribution: The Finnish Forest Research Institute, Parkano Research Station
(Kaironniementie 54, FIN-39700 Parkano, tel. +358-33-443 51, fax +358-33-443 5200).

ISBN 951-40-1379-4
ISSN 0358-4283
Parkanon Kirjapaino
Parkano 1994