

*Calculation and Mapping of Critical Thresholds in Europe:
Status Report 1999*

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Preface

This report is the fifth in a biennial series prepared by the Coordination Center for Effects (CCE) to document progress made in calculating and mapping critical loads in Europe. The CCE, as part of the Mapping Programme under the UN/ECE Working Group on Effects (WGE), collects critical load data from individual countries and synthesizes them into European maps and data bases. These data bases, together with scientific advice on critical threshold methodologies, are provided to the integrated assessment modeling groups under the UN/ECE Working Group on Strategies (WGS). Via this route the effects-related work has a direct impact on the preparation of new protocols to the 1979 Convention on Long-range Transboundary Air Pollution. In particular, the critical loads data presented in this report, which have been formally approved by the WGE in August 1998, serve as input to the current negotiations of a "multi-pollutant, multi-effect" protocol.

The work of the CCE is carried out in close collaboration with an extensive network of national scientific institutions (National Focal Centers) throughout Europe. At present, 24 National Focal Centers have provided critical loads data to the CCE, four more since the publication of the last Status Report in 1997. From modest beginnings in the early 1990s, we have reached a state where most of Europe is covered by national critical loads data. In addition to submitting data, National Focal Centers also participate in annual CCE Mapping Workshops at which data and methodologies are reviewed.

As critical load and exceedance calculations become ever more complex, the issue of data transparency has become increasingly important over the last two years. Thus a mechanism has been set up by the WGE which gives parties the possibility to obtain critical loads data for work under the LRTAP Convention. In this context, it should also be noted that the CCE has made all data used in integrated assessment under the WGS available to National Focal Centers on CCE's anonymous ftp server.

In addition, they were provided with a software tool (the "CCE Viewer") which allows the user to quickly display and map the entire European critical loads data base. Data transparency also became more pressing after the European Commission decided to use the European critical loads data in the formulation of an EU Acidification Strategy.

This report consists of three parts. Part I describes the present (1998) state of the critical loads data base used in UN/ECE negotiations. Chapter 1 gives an overview of the European critical loads and levels in the form of maps, and summarizes the methodology to calculate exceedances and their reductions by means of gap closures. The chapter is a stand-alone summary of the current state-of-the-art, and is designed to be understood also by the non-technical reader. Chapter 2 reports and analyzes in detail the critical loads and auxiliary data submitted by the National Focal Centers, and allows comparisons between countries. Chapter 3 explains the technical details of the so-called "accumulated exceedance" concept, which has been adopted in the integrated assessment of deposition reductions. Part II consists of two contributions: the first reports on UK help-in-kind to the Mapping Programme, and the second describes independent research on the uncertainties of exceedance calculations due to variations in deposition. Part III, the bulk of this report, consists of reports by the 24 National Focal Centers. They document the input data used to calculate national critical loads. Some of them also describe ongoing research carried out in the context of critical loads and levels. Finally, three appendices describe map projections, computer codes for exceedance calculations and conversion formulae for depositions and concentration units.

We hope that the 1999 CCE Status Report gives a fair overview of the accomplishments with respect to European critical load calculations and mapping, but does not give the impression that nothing remains to be done.

The Editors
Bilthoven, March 1999

1. Critical Loads and their Exceedances in Europe: An Overview

M. Posch, P.A.M. de Smet and J.-P. Hettelingh

Introduction

As part of the Mapping Programme under the UN/ECE Working Group on Effects (WGE), the Coordination Center for Effects (CCE) collects critical load data from National Focal Centers (NFCs) and synthesizes them into European maps and data bases. The CCE also carries out exceedance calculations and assists in the development of the critical loads/levels methodology. Thus the purpose of this chapter is twofold:

- (1) to summarize the definitions and concepts of critical loads and their exceedances in a non-technical manner with special emphasis on their use in European integrated assessment modeling carried out under the Long-range Transboundary Air Pollution (LRTAP) Convention and
- (2) to present maps of critical loads and levels which are used in the current protocol negotiations.

1.1 Critical loads

For the work under the LRTAP Convention a critical load has been 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” (Nilsson and Grennfelt 1988). The first critical loads to be calculated were for acidity (Hettelingh *et al.* 1991), and in the negotiations for the 1994 Sulphur Protocol a so-called “sulfur fraction” was used to derive a critical deposition of sulfur from the acidity critical load (Downing *et al.* 1993, Hettelingh *et al.* 1995). In the preparations for the negotiations for a “multi-pollutant, multi-effect” protocol, nitrogen became the focus, and thus critical loads of N had to be defined as well. This led to a revision of the Mapping Manual (UBA 1996), which now distinguishes the critical loads described below. The maximum critical load of sulfur:

$$CL_{max}(S) = BC_{dep}^* - CI_{dep}^* + BC_w - BC_u - ANC_{le(crit)} \quad (1.1)$$

equals the net input of (seasalt-corrected) base cations minus a critical leaching of acid neutralization capacity. As long as the deposition of N stays below the minimum critical load of nitrogen, i.e.

$$N_{dep} \leq N_i + N_u = CL_{min}(N) \quad (1.2)$$

all deposited N is consumed by sinks of N (immobilization and uptake), and only in this case is $CL_{max}(S)$ equivalent to a critical load of acidity. The maximum critical load for nitrogen acidity (in the case of no S deposition) is given by (UBA 1996):

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S)/(1 - f_{de}) \quad (1.3)$$

which not only takes into account the N sinks summarized in Equation 1.2, but considers also deposition-dependent denitrification. Both sulfur and nitrogen contribute to acidification, but one equivalent of S contributes, in general, more to excess acidity than one equivalent of N. Therefore, no unique acidity critical load can be defined, but the combinations of N_{dep} and S_{dep} not causing “harmful effects” lie on the so-called *critical load function* of the ecosystem defined by the three critical loads from Equations 1.1–1.3. An example of such a trapezoid-shaped function is depicted in Figure 1-1.

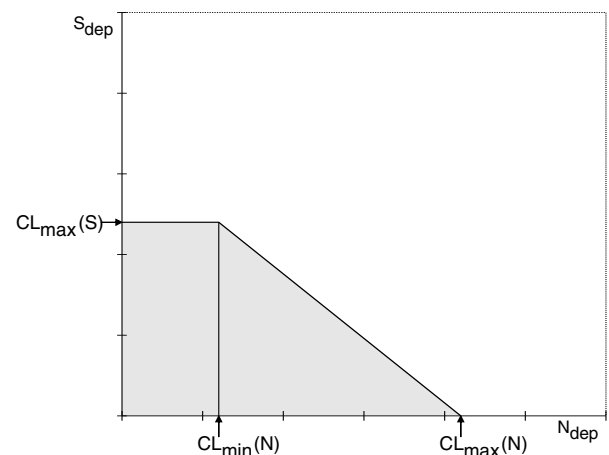


Figure 1-1. Example of a critical load function for S and acidifying N defined by the $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$. Every point of the grey-shaded area below the critical load function represents depositions of N and S which do not lead to the exceedance of critical loads.

Excess nitrogen deposition contributes not only to acidification, but can also lead to the eutrophication of soils and surface waters. Thus a critical load of nutrient nitrogen has been defined (UBA 1996):

$$CL_{nut}(N) = CL_{min}(N) + N_{le(ac)} / (1 - f_{de}) \quad (1.4)$$

which accounts for the nitrogen sinks and allows for an acceptable leaching of N.

It is the four critical loads defined in Equations 1.1–1.4 which Parties to the LRTAP Convention were asked to submit to the CCE and which were used to prepare maps and data bases. In the European integrated assessment modeling effort, one deposition value for nitrogen and sulfur, respectively, is given for each 150×150km² EMEP grid cell. In a single grid cell, however, many (up to 100,000 in some cases) critical loads for various ecosystems, mostly forest soils, have been calculated. These critical loads are sorted according to magnitude, taking into account the area of the ecosystem they represent, and the so-called cumulative distribution function (CDF) is constructed (see Posch *et al.* 1995 for a description of the methodology). From this CDF, percentiles (or other statistics) are calculated which can be directly compared with deposition values. Since no unique critical load of acidity can be defined, the concept of cumulative distribution has been generalized for critical load functions, and instead of simple percentiles so-called *ecosystem protection isolines* are calculated, which – for given depositions of S and N – allow the determination of the ecosystem area protected in a grid cell (see Posch *et al.* 1995, 1997).

1.2 Critical load exceedance and gap closure concepts

If only one pollutant contributes to an effect, e.g. nitrogen to eutrophication or sulfur to acidification (as assumed before 1994), a unique critical load (CL) can be calculated and compared with deposition (Dep), and the difference has been termed the exceedance of the critical load ($Ex = Dep - CL$). In the case of two pollutants no unique exceedance exists, as is illustrated in Figure 1-2. But for a given deposition of N and S an exceedance has been *defined* as the sum of the N and S deposition reductions required to achieve non-exceedance by taking the shortest path to the critical load function (see Figure 1-2). Within a grid cell, these exceedances are multiplied by the respective ecosystem area and summed to yield the so-called *accumulated exceedance* (AE) for that grid cell. In addition, the *average accumulated exceedance* (AAE) is defined by dividing the AE by the total ecosystem area of the grid cell, and which has thus the dimension of a deposition (see Chapter 3 for a detailed derivation).

When comparing present or feasible future deposition scenarios with European critical loads it appeared that non-exceedance could not be reached everywhere. Thus it was decided by integrated assessment modelers to use uniform percentage reductions of the excess deposition (so-called *gap closures*) to define reduction scenarios. In the following we summarize the different gap closure methods used and illustrate them for the case of a single pollutant.

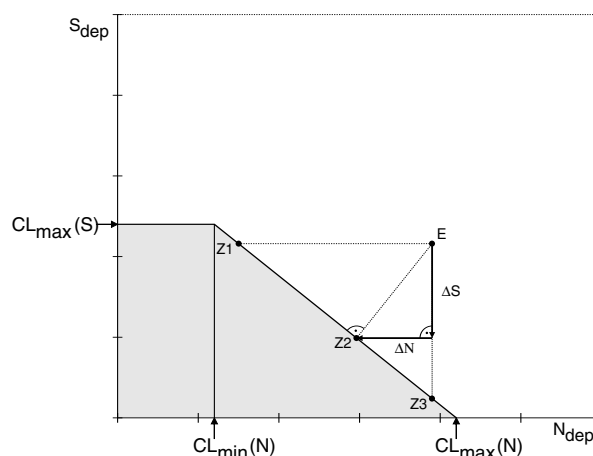


Figure 1-2. Critical load function for S and acidifying N. It shows that no unique exceedance exists: Let the point E denote the current deposition of N and S. Reducing N_{dep} substantially one reaches point Z1 and thus non-exceedance without reducing S_{dep} , but non-exceedance can also be achieved by reducing S_{dep} only (by a smaller amount) until reaching Z3. However, an exceedance has been *defined* as the sum of N_{dep} and S_{dep} reductions ($\Delta N + \Delta S$) which are needed to reach the critical load function on the shortest path (point Z2).

In the 1994 Sulphur Protocol, only sulfur was considered as acidifying pollutant (N deposition was fixed; it determined, together with N uptake and immobilization, the sulfur fraction). Furthermore, taking into account the uncertainties in the CL calculations, it was decided to use the 5th percentile of the critical load CDF in a grid cell as the only value representing the ecosystem sensitivity of that cell. And *the* exceedance was simply the difference between the (current) S deposition and that 5th percentile critical load. This is illustrated in Figure 1-3(a): Critical loads and deposition are plotted along the horizontal axis and the (relative) ecosystem area along the vertical axis. The thick solid and the thick broken lines are two examples of critical load CDFs (which have the same 5th percentile critical load, indicated by “CL”). “D0” indicates the (present) deposition, which is higher than the CLs for 85% of the ecosystem area. The difference between “D0” and “CL” is the exceedance in that grid cell. It was decided to reduce the exceedance everywhere by a fixed percentage, i.e. to “close the gap” between (present) deposition and (5th percentile) critical load. In Figure 1-3(a), a *deposition gap closure* of 60% is shown as an example. As can be seen, a fixed deposition gap closure can result in very different improvements in ecosystem protection percentages (55% vs. 22%), depending on the shape of the critical load CDF.

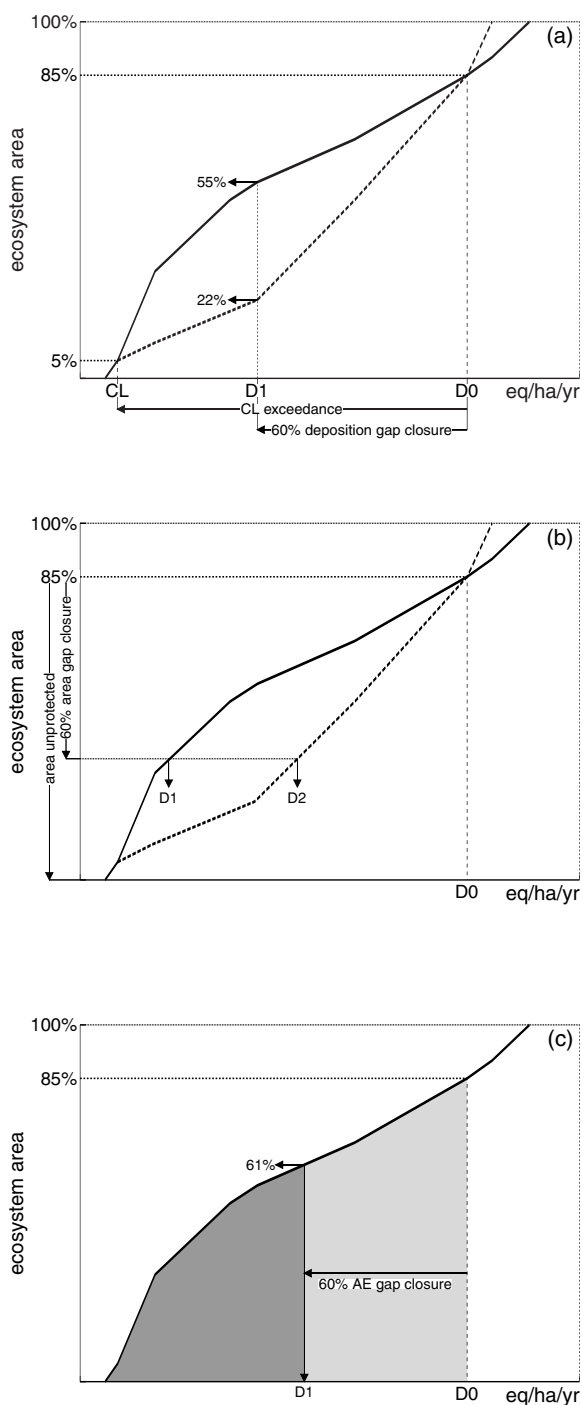


Figure 1-3. Cumulative distribution function (CDF; thick solid line) of critical loads and different methods of gap closure: (a) deposition gap closure, (b) ecosystem gap closure, and (c) accumulated exceedance (AE) gap closure. The thick dashed line in (a) and (b) depict another CDF, illustrating how different ecosystem protection follows from the same deposition gap closure (a), or how different deposition reductions are required to achieve the same protection level (b).

In order to take into account all critical loads within a grid cell (and not only the 5th percentile), it was suggested to use an *ecosystem area gap closure* instead of the deposition gap closure. This is illustrated in Figure 1-2b: for a given deposition “D0” to a grid cell the ecosystem area unprotected, i.e. with deposition exceeding the critical loads, can be read from the vertical axis. After agreeing to a certain (percent) reduction of the unprotected area (e.g. 60%), it is easy to compute for a given CDF the required deposition reduction (“D1” and “D2” in Figure 1-3(b)). Another important reason to use the ecosystem area gap closure is that it can be easily generalized to two (or more) pollutants, which is not the case for a deposition-based exceedance. This generalization became necessary in the preparation for the “multi-pollutant, multi-effect” protocol in the case of acidity critical loads, as both N and S contribute to acidification. Critical load values have been replaced by critical load functions and percentiles replaced by ecosystem protection isolines (see above). However, the use of the area gap closure becomes problematic if only a few critical load values or functions are given for a grid cell. In such a case the CDF becomes highly discontinuous, and small changes in deposition may result in either no increase in the protected area at all or large jumps in the area protected.

To remedy the problem with the area gap closure caused by discontinuous CDFs, the accumulated exceedance (AE) concept has been introduced (see above and Chapter 3). In the case of one pollutant, the AE is given as the area under the CDF of the critical loads (the entire grey-shaded area in Fig.1-3(c)). Deposition reductions are now negotiated in terms of an *AE (or AAE) gap closure*, also illustrated in Fig.1-3(c): a 60% AE gap closure is achieved by a deposition “D1” which reduces the total grey area by 60%, resulting in the dark grey area; also the corresponding protection percentage (61%) can be easily derived. The greatest advantage of the AE and AAE is that it varies smoothly as deposition is varied, even for highly discontinuous CDFs, thus facilitating optimization calculations in integrated assessment.

The advantages and disadvantages of the three gap closure methods described above are summarized in the following table:

	Advantages	Disadvantages
Deposition gap closure (used for the 1994 UN/ECE Sulphur Protocol)	<ul style="list-style-type: none"> • Easy to use even for discontinuous CDFs (e.g. grid cells with only one CL). 	<ul style="list-style-type: none"> • Takes only one CL value (e.g. 5th percentile) into account. • May result in no increase of protected area. • Difficult to define for two pollutants. • Difficult (or even impossible) to define a gap closure for discontinuous CDFs (e.g. grid cells with only one CL).
Ecosystem area gap closure (used for the EU Acidification Strategy)	<ul style="list-style-type: none"> • In line with the goals of CL use (maximum ecosystem protection). • Easy to apply to any number of pollutants. 	<ul style="list-style-type: none"> • AE stretches the limits of the critical load definition.* • Exceedance definition not unique for 2 or more pollutants.
Accumulated Exceedance (AE) gap closure (used for the UN/ECE multi-pollutant, multi-effects protocol)	<ul style="list-style-type: none"> • AE (and AAE) is a smooth and convex function of deposition even for discontinuous CDFs. 	

* It assumes a linear damage function. In the future, however, this feature could become an advantage.

1.3 Maps of critical loads/levels and their exceedance

In this section we present European maps of critical loads and levels, as well as their exceedances, which are used in the current protocol negotiations. It should be noted however, that the maps presented here represent only a small fraction of the total critical loads data held at the CCE. The integrated assessment modelers under the LRTAP Convention have been provided a data base containing all the necessary information (such as percentiles and protection isolines) for linking optimization models to environmental effects. The transfer matrices used to calculate the deposition of S and N, and thus exceedances, were provided by the EMEP Meteorological Synthesizing Centre-West (MSC-W) at the Norwegian Meteorological Institute (EMEP 1998). In the following, maps are presented and discussed which illustrate the quantities and concepts summarized in the previous two sections.

Figures 1-4 and 1-5 are maps of the 5th percentiles of the maximum critical load of sulfur, $CL_{max}(S)$, the minimum critical load of acidifying nitrogen, $CL_{min}(N)$, the maximum critical load of acidifying nitrogen, $CL_{max}(N)$, and the critical load of nutrient nitrogen, $CL_{nut}(N)$. They show that maximum critical loads are lowest in the northwest and highest in the southeast. The low values of $CL_{min}(N)$, as compared to $CL_{nut}(N)$, in the south (Italy, Hungary, Croatia) indicate low values of nitrogen uptake and immobilization, but relatively high values for N leaching and denitrification. The maps on the right display the numbers (in eq ha⁻¹ yr⁻¹) underlying the color classes on the left-hand side. The blue grid squares in the right-hand maps indicate data submitted by National Focal Centers (see Chapter 2). Critical loads in the white grids have been computed from the European background data base held at the CCE (see chapter 6 in Posch *et al.* 1997). The maps in Figures 1-4 and 1-5 also comprise the information on

critical loads provided in printed form to the Working Group on Effects in 1998 (UN/ECE 1998a).

Figure 1-6 shows snapshots of the temporal development (1960–2010) of the exceedance of the 5th percentile maximum critical load of sulfur, $CL_{max}(S)$, earlier called “critical acid deposition”. The exceedance is calculated due to sulfur deposition alone, implicitly assuming that nitrogen does not contribute to acidification. Although this is probably true *at present* in many countries as most of the deposited N is still immobilized in the soil or taken up by vegetation, the long-term sustainable maximum deposition for N not to contribute to acidification is given by $CL_{min}(N)$. However, the main purpose of Figure 1-6 is to illustrate the change in the acidity critical load exceedance over time. As can be seen from the maps, the size of area and magnitude of exceedance peaked around 1980, with a decline afterwards to a situation in 1995 which is better than in 1960. Further improvements can be expected when the Current emission Reduction Plans (the so-called CRP scenario, UN/ECE 1998b) is implemented, which includes all reduction measures already legislated by member countries (*inter alia* the 1994 Sulphur Protocol). However, further emission reductions are needed to reach the goal that deposition of S and N does not exceed critical loads of acidity over all of Europe.

As mentioned in the previous section, a unique exceedance does not exist when considering both sulfur and nitrogen, but for a given deposition of S and N one can always determine whether there is non-exceedance or not. The two maps at the top of Figure 1-7 show the percent of ecosystem area protected from acidifying deposition of S and N in 1990 and 2010. In 1990 less than 10% of the ecosystem area is protected in large parts of central and western Europe as well as on the Kola peninsula. Under the CRP scenario, the situation improves almost everywhere, but still far from reaching complete protection.

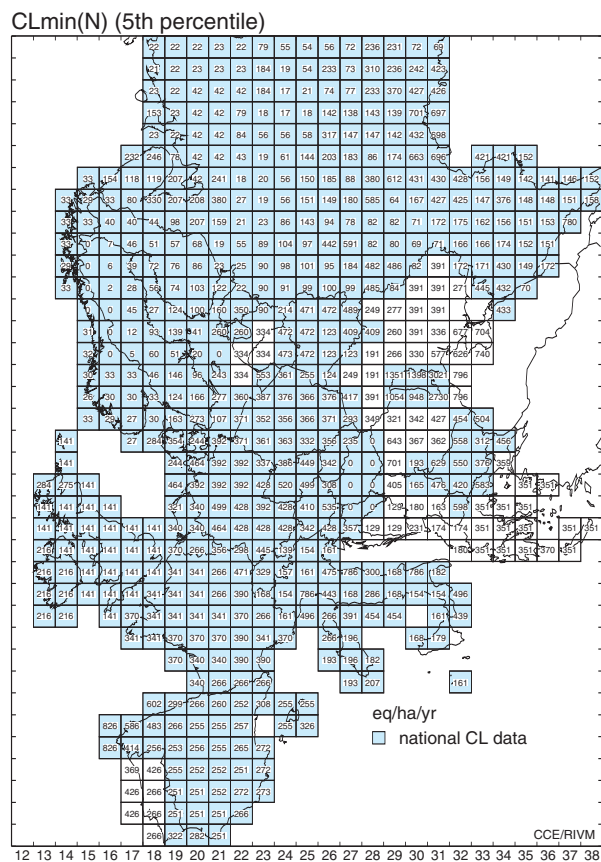
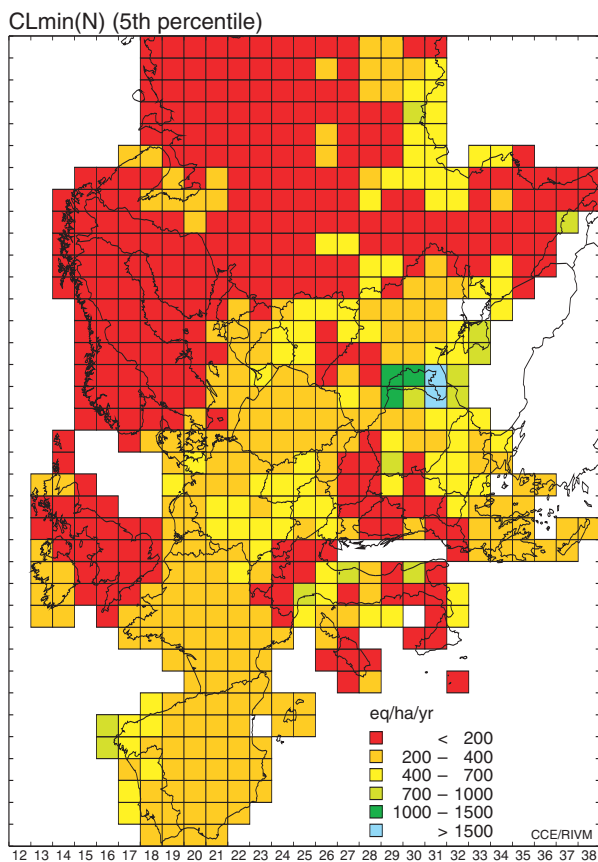
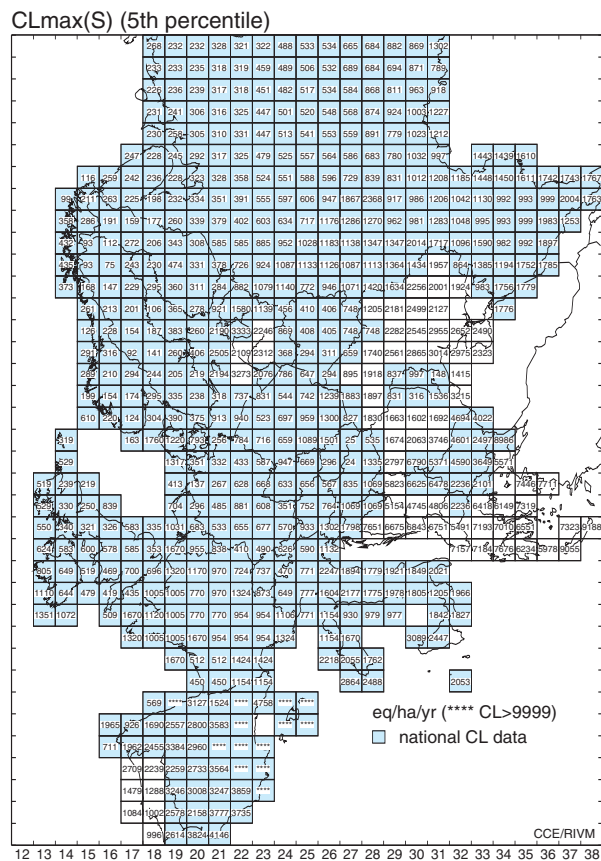
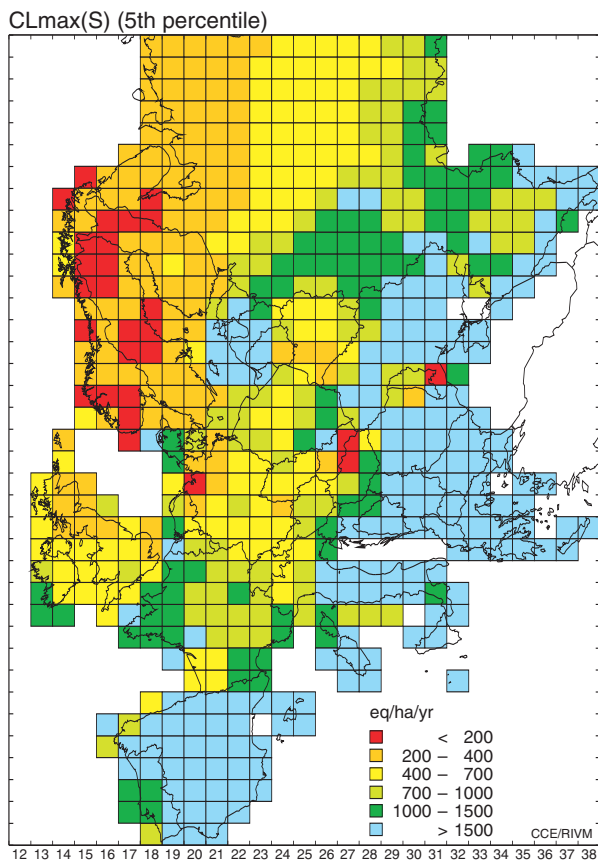
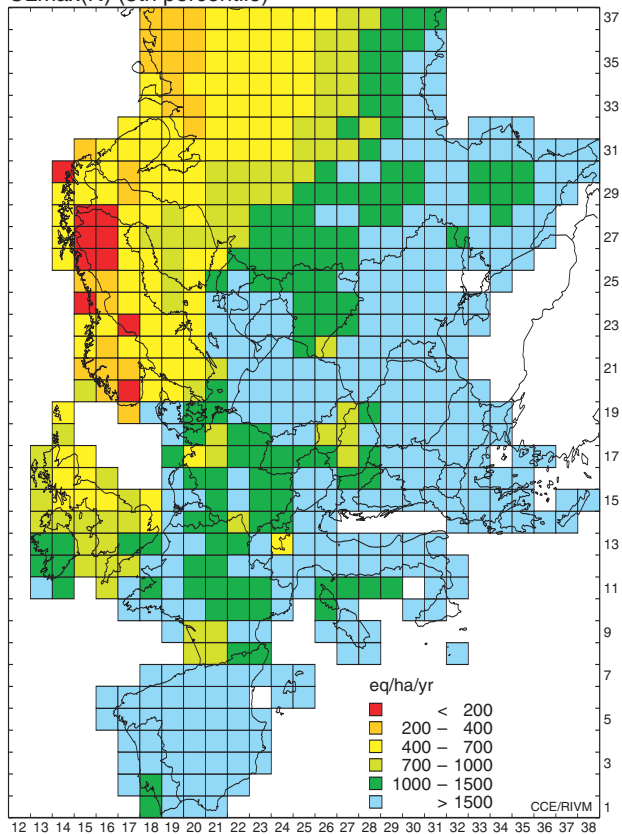
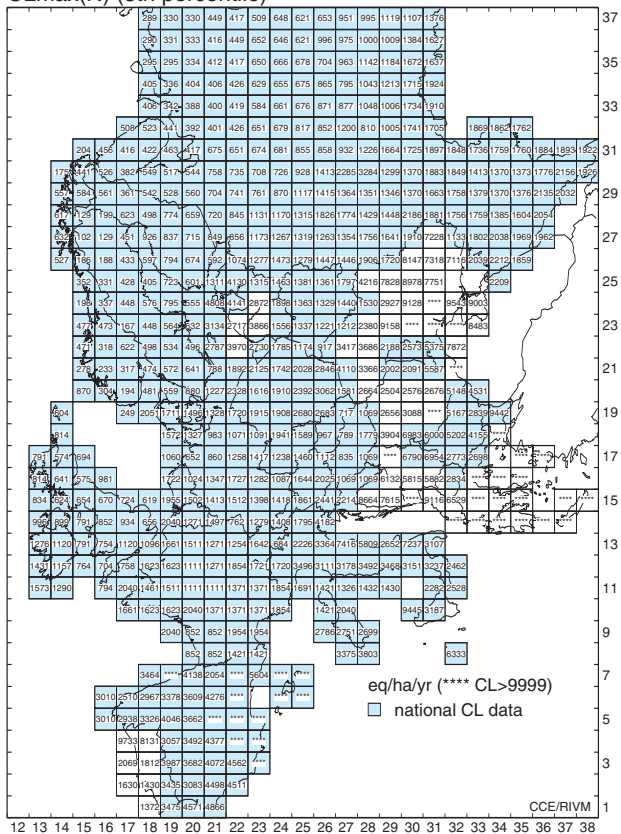


Figure 1-4. The 5th percentiles of the maximum critical loads of sulfur, $CL_{max}(S)$, and of the minimum critical loads of acidifying nitrogen, $CL_{min}(N)$. The maps on the right display the numbers (in $\text{eq ha}^{-1} \text{yr}^{-1}$) underlying the color classes on the left-hand side. The blue grid squares in the right-hand maps indicate data from National Focal Centers.

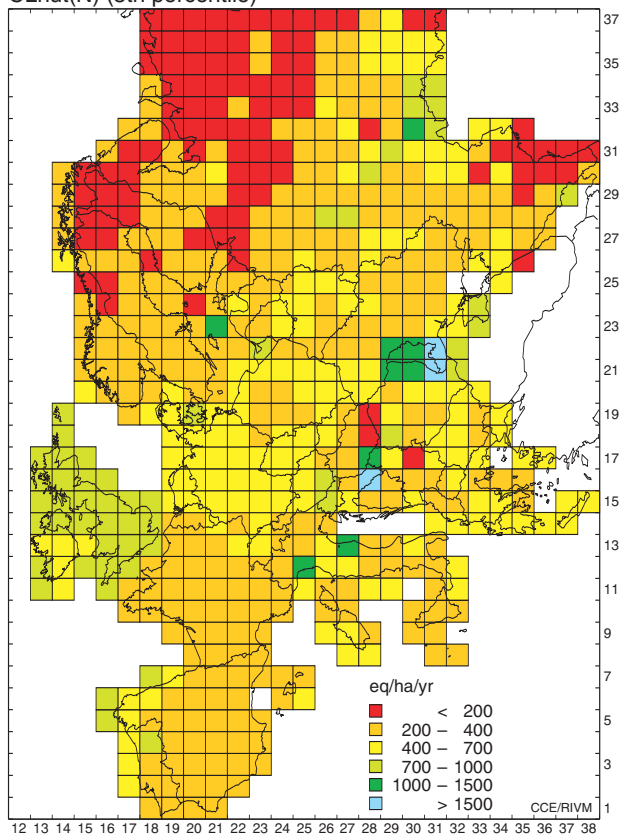
CL_{max}(N) (5th percentile)



CL_{max}(N) (5th percentile)



CL_{nut}(N) (5th percentile)



CL_{nut}(N) (5th percentile)

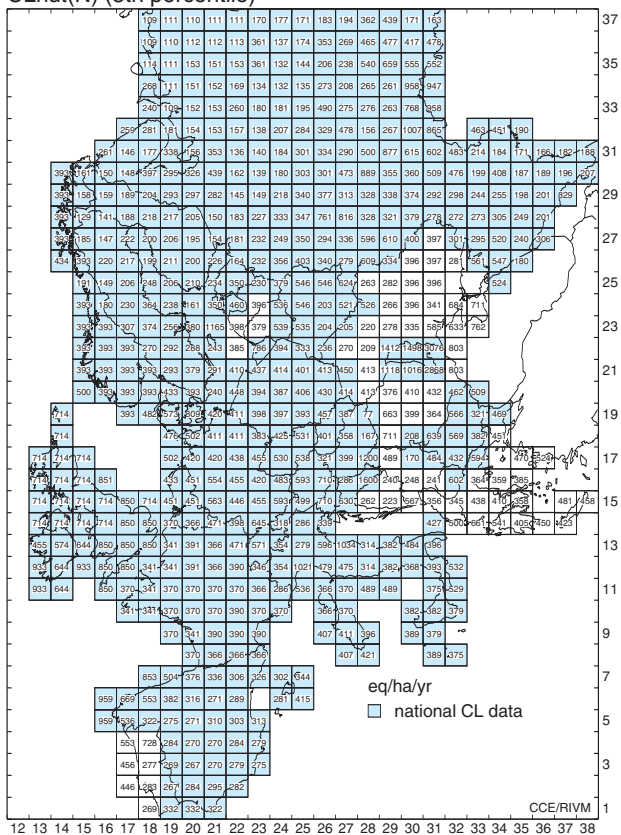


Figure 1-5. The 5th percentiles of the maximum critical loads of acidifying nitrogen, $CL_{max}(N)$, and of the nutrient loads of acidifying nitrogen, $CL_{nut}(N)$. The maps on the right display the numbers (in $\text{eq ha}^{-1} \text{yr}^{-1}$) underlying the color classes on the left-hand side. The blue grid squares in the right-hand maps indicate data from National Focal Centers.

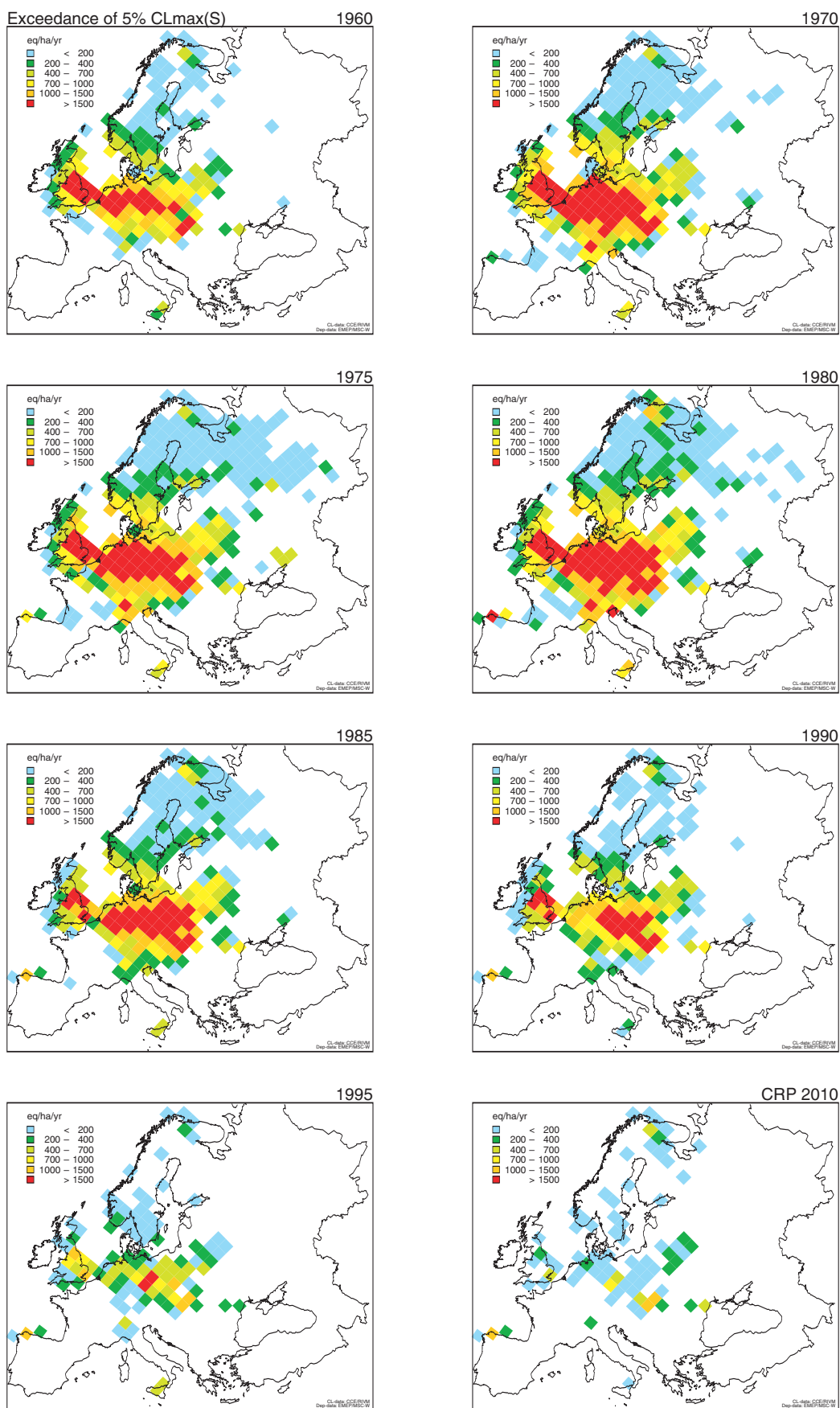


Figure 1-6. Temporal development (1960–2010) of the exceedance of the 5th percentile maximum critical load of sulfur (“acidity critical load”). White areas indicate non-exceedance or lack of data (e.g. Turkey). Sulfur deposition data were provided by the EMEP/MSC-W (EMEP 1998).

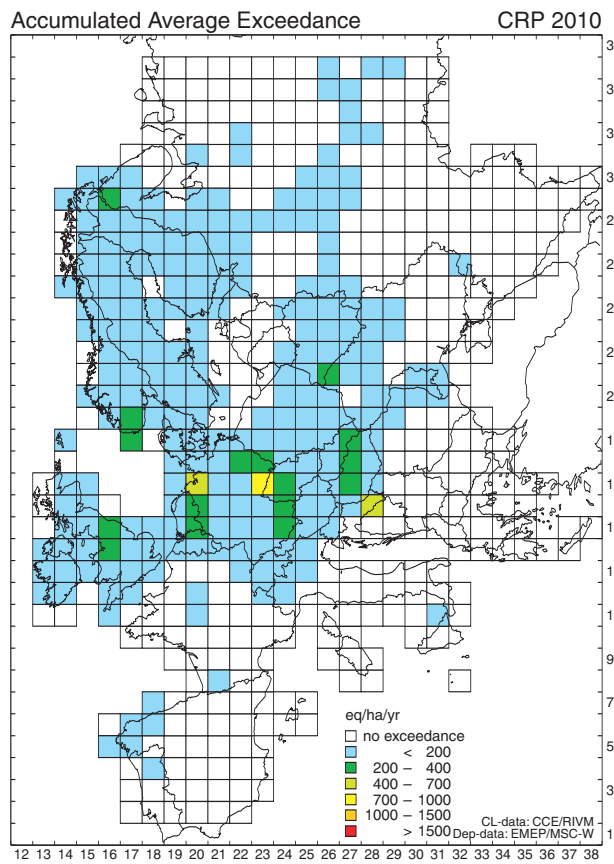
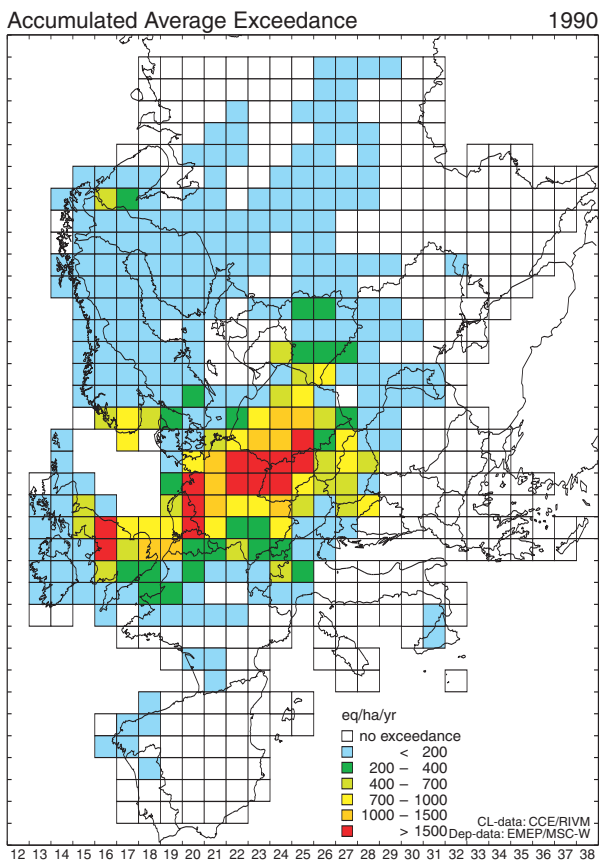
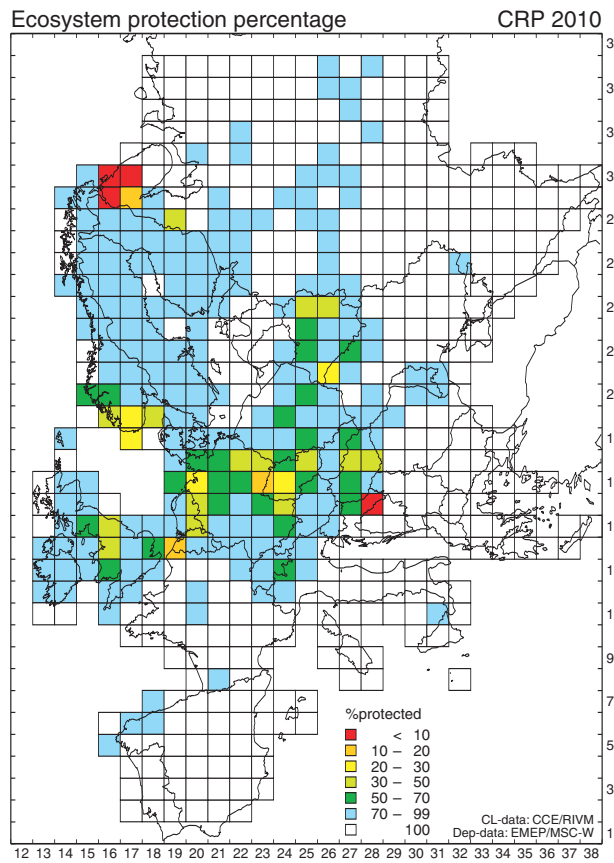
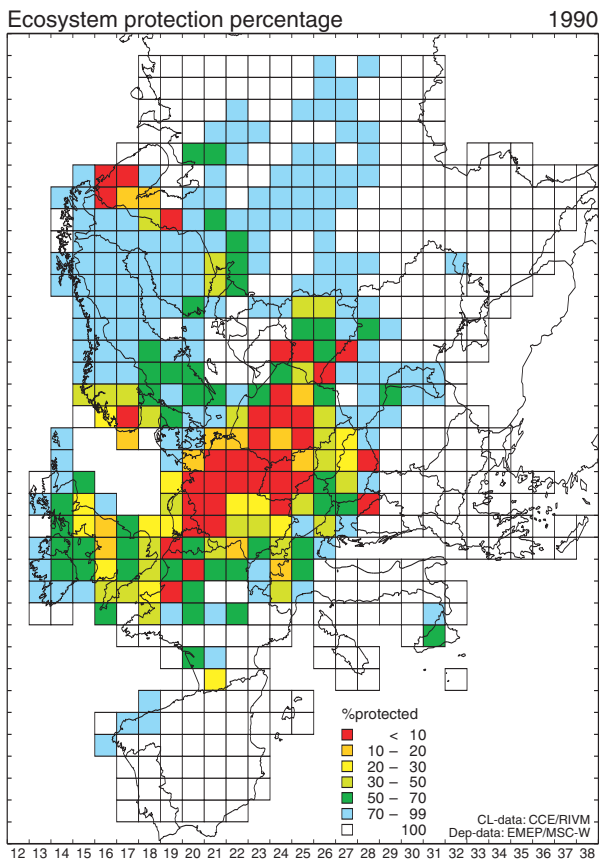


Figure 1-7. Top: The percentage of ecosystem area protected (i.e. non-exceedance of critical loads) from acidifying deposition of sulfur and nitrogen in 1990 (left) and in the year 2010 according to current emission reduction plans (right). Bottom: The accumulated average exceedance (AAE) of the acidity critical loads by sulfur and nitrogen deposition in 1990 (left) and 2010 (right). Sulfur and nitrogen deposition average data were provided by the EMEP/MSC-W (EMEP 1998).

To be able to compare deposition of S and N with the acidity critical load function, an exceedance quantity has been defined (see previous sections). This average accumulated exceedance (AAE) is the amount of excess acidity averaged over the total ecosystem area in a grid square. The two maps at the bottom of Figure 1-7 show the AAE for 1990 and 2010 (CRP scenario). In 1990 the highest excess acidity occurs in central Europe, the pattern roughly matching with the ecosystem protection percentages for the same year. Under the CRP scenario in 2010, excess acidity is reduced nearly everywhere, with a peak remaining in the “Black Triangle” of Germany, Poland and the Czech Republic.

Nitrogen not only contributes to acidification and eutrophication, but is also a precursor for the formation of tropospheric ozone. High levels of ground-level ozone concentration have adverse effects on forests and cause yield reductions in crops. Therefore, critical levels for forests and crops have been derived (Kärenlampi and Skärby 1996). They are based on the “AOT” concept, i.e. the accumulated exposure over a threshold of 40 ppb (called AOT40) during daylight hours in the growing season (May-July for crops and semi-natural vegetation).

The critical level for crops is 3000 ppb·hours (shaded blue in Figure 1-8) independent of the location (so-called Level I critical level). In Figure 1-8 the AOT40 for crops is shown for eight different years between 1985 and 1996. The modeled 6-hourly ozone concentrations have been provided by the EMEP/MSC-W (Simpson *et al.* 1997). The eight maps illustrate that ozone concentrations vary strongly from year to year, and thus a five-year average is recommended for integrated assessment purposes (UBA 1996). The size of the grid squares in Figure 1-8 corresponds to the percentage of arable land in that 150×150km² EMEP grid cell, thus indicating the potential stock-at-risk. However, more work is needed to refine both the land use data and the critical levels (site-dependent Level II critical levels) before an economic evaluation of ozone impacts can be attempted in earnest.

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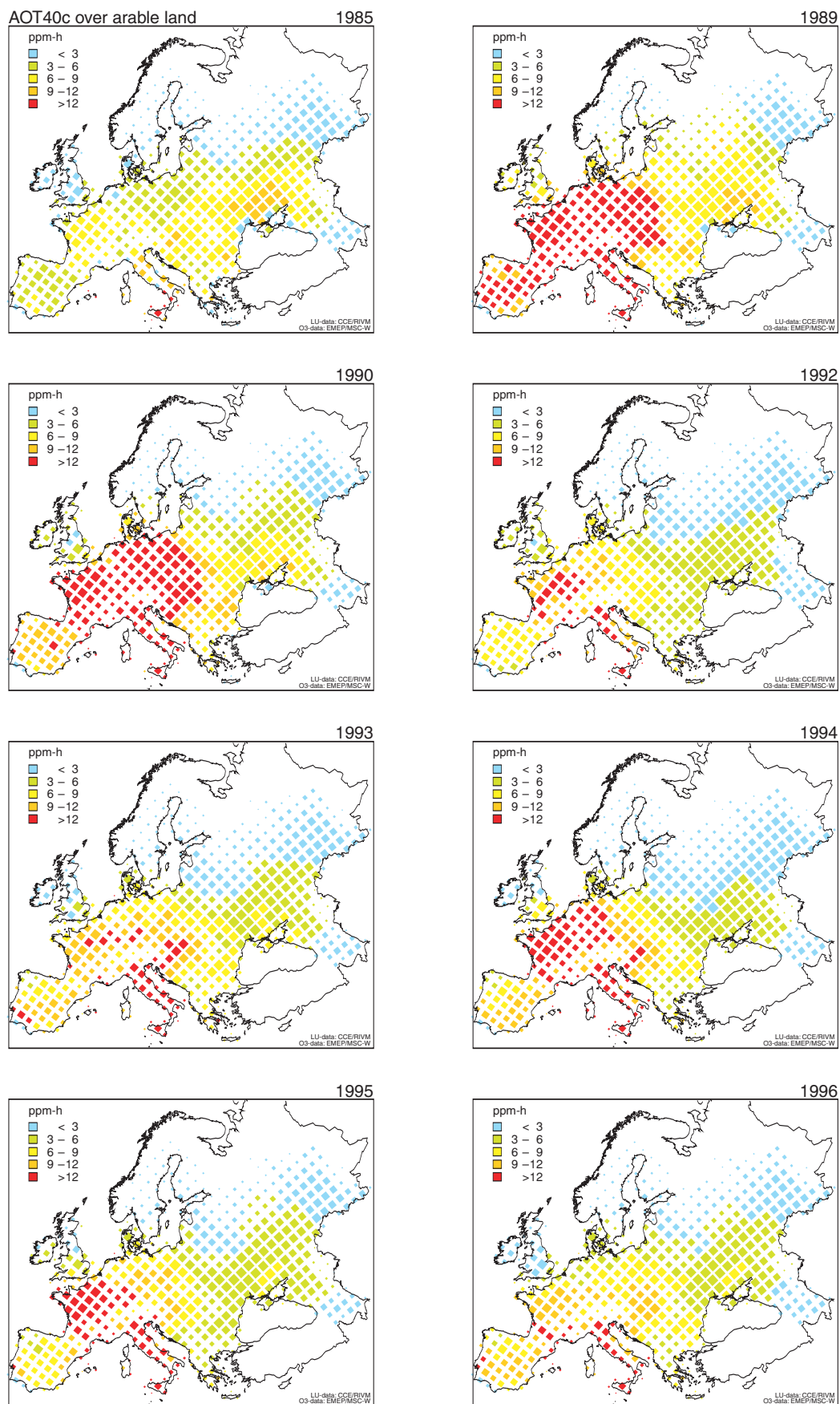


Figure 1-8. The accumulated exposure to ground-level ozone concentrations over a threshold of 40 ppb (AOT40 for crops) in eight years between 1985 and 1996. The blue-shaded grids indicate areas where the critical AOT40 level of 3000 ppb-hours is not exceeded. The size of a grid square corresponds to the percentage of arable land in that 150x150km² EMEP grid cell. Ozone concentration data were provided by the EMEP/MSC-W (Simpson *et al.* 1997).

2. Summary of National Data

P.A.M. de Smet and M. Posch

Introduction

At the request of the UN/ECE Working Group of Effects (WGE), the Coordination Center for Effects (CCE) periodically asks countries to submit up-to-date national critical loads data, so that the integrated assessment modeling groups participating in LRTAP Convention activities can work with the latest data. Such a request gives National Focal Centers (NFCs) the opportunity to submit their latest results. This chapter describes briefly the process and summarizes the results of the 1998 data update cycle. It lists the countries that contributed national data and provides an overview of the ecosystems selected as receptors and density (resolution) of the national data. The cumulative distributions of the critical loads are compared and the input parameters needed to compute the critical loads for forest soils are analyzed in detail.

2.1 Overview of national contributions

The following timetable illustrates the 1998 update of national critical load data:

29.9.1997	CCE issues a call for updated data to all NFCs, as requested by the Working Group on Effects (WGE), with a deadline for submission of 15 January 1998.
14.3.1998	Preliminary European critical load data sets are made available to Task Force on Integrated Assessment Modelling (TFIAM).
9.4.1998	CCE sends the updated data base to NFCs for verification and comments.
23.4.1998	CCE provides the updated data sets to the Task Force on Mapping (TFM).
11.5.1998	New data and maps are presented at CCE workshop in Kristiansand, Norway.
15.5.1998	TFM adopts the new data set, with the understanding that 5 countries will submit minor modifications before 15 June 1998.
7.7.1998	CCE provides TFIAM, NFCs and WGE with final data sets.
26.8.1998	WGE adopts data sets and maps as final version for update round of the year 1998.
4.9.1998	Working Group on Strategies (WGS) announces a "data freeze" for all input data used in the preparations and negotiations of the "multi-pollutant, multi-effect" protocol.

The number of countries that submitted data in 1998 has increased to 24 (listed in Table 2-1). These national contributions have been adopted by the WGE in August 1998. Countries that contributed revised data are Austria, Belgium, Finland, France, Germany, Ireland, Italy, Poland, Russian Federation, Sweden and the United Kingdom. Four countries contributed national data for the first time: Belarus, Bulgaria, Republic of Moldova and Slovakia. The revision of Belgian data included a first-time contribution for the Wallonian part of the country, while the Flemish data remained unchanged from 1997. The revision of the Russian data consisted in a simple area correction of its 1996 data. No changes were submitted for the Netherlands and Spain (which continue to use 1996 data) or for Croatia, Czech Republic, Denmark, Estonia, Hungary, Norway and Switzerland (which submitted data in 1997). Most of the European mapping domain is now covered by national contributions. Further details on most countries' activities can be found in Part III of this report, as well as the CCE Status Report 1997.

At previous mapping workshops, the CCE and NFCs have agreed to the following rules concerning the application of critical loads data for areas that are not covered by national contributions:

- (i) For all grid cells that do not cover a country that contributes national data, critical loads are computed using the European background data base held at the CCE. (See Posch *et al.* 1997, Chapter 6 for a description of this data base).
- (ii) For grid cells that cover parts of one or more countries which have submitted national data, calculations are based on the national critical loads data for this cell only, no matter how small the area or how few data are supplied. For these grid cells, no background data are included.

2.2 Scope of national contributions

National Focal Centers have selected a variety of ecosystem types as receptors for calculating and mapping critical loads. For most ecosystem types (e.g. forests), critical loads are calculated for both acidity and eutrophication. Other receptor types (e.g. streams and lakes) have only critical loads for acidity, on the assumption that eutrophication does not occur in these ecosystems. For some receptors, like most semi-natural vegetation, only critical loads for nutrient nitrogen are computed.

Table 2-1. National data version used at present (1999) and most recent year of adoption by the WGE.

Country	Code	Year of adoption by the WGE			Remark
		1996	1997	1998	
Austria	AT			✓	Update of earlier contribution.
Belgium	BE		✓	(✓)	First contribution for Wallonian part of country.
Belarus	BY			✓	First national contribution.
Bulgaria	BG			✓	First national contribution.
Croatia	HR		✓		Only EMEP 50×50 km ² grid cells (79,43) and (80,43).
Czech Republic	CZ		✓		
Denmark	DK		✓		
Estonia	EE		✓		
Finland	FI			✓	Update of earlier contribution.
France	FR			✓	Update of earlier contribution.
Germany	DE			✓	Update of earlier contribution.
Hungary	HU		✓		
Ireland	IE			✓	Update of earlier contribution.
Italy	IT			✓	Update of earlier contribution.
Netherlands	NL	✓			
Norway	NO		✓		
Poland	PL			✓	Update of earlier contribution.
Republic of Moldova	MD			✓	First national contribution.
Russian Federation	RU	✓		(✓)	Correction of area from 1996 version only.
Slovakia	SK			✓	First national contribution.
Spain	ES	✓			
Sweden	SE			✓	Update of earlier contribution.
Switzerland	CH		✓		
United Kingdom	UK			✓	Update of earlier contribution.
Totals:	24	3	8	13	

Table 2-2 shows by country the ecosystem types, number of records, their total (summed) area in km² and their percentage of the country area. Figure 2-1 shows the distribution of ecosystem types for which critical loads have been calculated, and their areas as a percentage of total country area. The diversity of ecosystem types selected by the countries as being sensitive to acidification and/or eutrophication has been reduced into a more limited set of types for presentation reasons. The histogram in Figure 2-1 shows that most countries have concentrated on mapping critical loads for forest soils, while some countries (e.g. Finland, Norway and Sweden) have also mapped surface waters as an important receptor. Norway and Switzerland have significant areas of (semi-)natural vegetation selected as a receptor. Ireland and the United Kingdom have considerable areas with critical loads for heathland, while grasslands represent substantial areas in Austria, Belarus, France, Italy, the Republic of Moldova, and the United Kingdom.

Table 2-3 provides details on the number, area coverage, and the density of ecosystems for which NFCs have

submitted critical loads of acidity and/or nutrient nitrogen. National data provided for acidity critical loads are summarized in columns A through D. Column A gives the number of ecosystems for which acidity critical loads ($CL_{max}(S)$, $CL_{min}(N)$, and $CL_{max}(N)$) have been calculated. Columns B and C show the total area of these ecosystems and the percentage of the country covered by these ecosystems, respectively. The average size of an ecosystem is given in Column D ($D=B/A$). Similar information for $CL_{nut}(N)$ is provided in Columns E through H. Columns I through L provide information on ecosystems for which both acidity and nutrient critical loads have been submitted. Columns M through P provide information for those ecosystems for which critical loads of acidity and/or nutrient nitrogen have been calculated (col. $M=A+E-I$). The wide range in the number and density of ecosystems among countries can be seen from the table. For most countries, critical loads of acidity and nutrient nitrogen are computed on the same set of ecosystems; thus the number and area of ecosystems are the same for both types of critical loads.

Table 2-2. Type and number of ecosystems for which critical loads data are provided by National Focal Centers.

Country	Ecosystem type	CCE code	No. of eco-systems	Area		Remarks
				km ²	% of country	
Austria	Forest	f	6,604	49,918	59.54	
	Oligotrophic bog	p	205	1,536	1.83	Only $CL_{nut}(N)$.
	Alpine grassland	g	1,092	8,236	9.82	Only $CL_{nut}(N)$.
Belgium	Coniferous forest	c	835	2,642	8.66	Flanders: 652 ecosystems, including 75 mixed forests.
	Deciduous forest	d	1,201	4,154	13.61	
	Mixed forest	m	490	225	0.74	Wallonia: 1880 ecosystems, including 415 mixed forests.
	Lake	w	6	3	0.01	
Belarus	Coniferous forest	c	234	19,398	9.34	
	Deciduous forest	d	79	1,258	0.61	
	Grassland	g	242	29,630	14.27	
Bulgaria	Coniferous forest	c	29	7,579	6.83	
	Deciduous forest	d	55	41,897	37.75	
Croatia	Coniferous forest	c	18	1,438	2.54	Two EMEP 50×50 km ² grid cells.
	Deciduous forest	d	16	1,261	2.23	
Czech Republic	Forest	f	29,418	26,568	33.69	
Denmark	Coniferous forest	c	6,496	2,336	5.42	Spruce and pine species.
	Deciduous forest	d	3,261	813	1.89	Beech and oak species.
	Grass	g	9,027	747	1.73	Only acidity CLs.
Estonia	Coniferous forest	c	99	13,380	29.58	Spruce and pine species.
	Deciduous forest	d	26	3,200	7.08	
	Bog	p	15	2,330	5.15	
Finland	Coniferous forest	c	2,049	148,941	44.05	Spruce and pine species.
	Deciduous forest	d	1034	16,104	4.76	
	Lake	w	1450	107,816	31.88	
France	Coniferous forest	c	28	20,856	3.83	The original data base with detailed ecosystem types has been reclassified into these 4 groups.
	Deciduous forest	d	83	75,432	13.87	
	Mixed forest	m	302	131,757	24.22	
	Grassland (agricultural)	g	178	89,658	16.48	
Germany	Coniferous forest	c	227,506	56,877	15.93	
	Deciduous forest	d	91,957	22,989	6.44	
	Mixed forest	m	90,892	22,723	6.36	
Hungary	Unspecified forest	f	7	1,022	1.10	
	Coniferous forest	c	5	43	0.05	
	Deciduous forest	d	8	557	0.60	
	Grassland/reed/marsh	g	12	889	0.96	
	Heath	h	4	13	0.01	
	Bog	p	4	52	0.06	
	Lake	w	2	271	0.29	
Ireland	Coniferous forest	c	10,078	2,445	3.48	
	Deciduous forest	d	8,951	1,808	2.57	
	Natural grassland	g	7,539	2,044	2.91	
	Moors and heathland	h	7,304	2,605	3.71	
	Fresh waters	w	175	175	0.25	
Italy	Coniferous forest	c	63	17,225	5.72	10 with only $CL_{nut}(N)$.
	Deciduous forest	d	165	60,577	20.11	56 with only $CL_{nut}(N)$.
	Mediterranean forest	m	110	14,109	4.68	35 with only $CL_{nut}(N)$.
	Tundra	h	46	4,709	1.56	
	Acid grassland	g	118	23,235	7.71	25 with only $CL_{nut}(N)$.
Netherlands	Coniferous forest	c	52,949	1,926	4.60	12 species regrouped into coniferous and deciduous forest types
	Deciduous forest	d	74,320	1,270	3.03	
Norway	Forest	f	720	40,522	12.51	
	Lake/stream	w	2,305	180,709	55.80	Only acidity CLs.
	Semi-natural vegetation	h	1,610	99,420	30.70	Only $CL_{nut}(N)$.

Table 2-2 (continued). Type and number of ecosystems for which critical loads data are provided by National Focal Centers.

Country	Ecosystem type	CCE code	No. of eco-systems	Area		Remarks
				km ²	% of country	
Poland	Coniferous forest	c	1,957	86,736	27.74	
	Deciduous forest	d	1,957	86,736	27.74	
Republic of Moldova	Coniferous forest	c	15	53	0.16	
	Deciduous forest	d	32	260	0.77	
	Grassland	g	94	11,672	34.64	
Russian Federation	Coniferous forest	c	4,916	1,141,036	22.42	
	Deciduous forest	d	2,967	171,549	3.37	
	Other	o	6,333	2,204,554	43.31	
Slovakia	Coniferous forest	c	112,440	7,028	14.33	15 species regrouped into coniferous and deciduous forest types.
	Deciduous forest	d	208,451	13,028	26.57	
Spain	Coniferous forest	c	2,237	55,925	11.24	
	Deciduous forest	d	744	18,600	3.74	
	Mixed forest	m	428	10,700	2.15	
Sweden	Forest	f	1,883	188,056	41.79	27 with only $CL_{nut}(N)$.
	Lake	w	2,378	203,125	45.14	Only acidity CLs.
Switzerland	Forest	f	8,467	8,467	20.51	717 only acidity CLs, 29 only $CL_{nut}(N)$.
	Alpine lakes	w	495	495	1.20	431 with only acidity CLs.
	Semi-natural ecosystems	h	14,975	14,975	36.27	11,559 with only $CL_{nut}(N)$.
United Kingdom	Coniferous forest	c	29,309	7,378	3.05	6 with only acidity CLs.
	Deciduous forest	d	69,747	10,331	4.27	31 with only acidity CLs.
	Acid grassland	g	138,535	54,578	22.58	
	Calcareous grassland	g	24,976	10,164	4.20	
	Heathland	h	56,393	9,919	4.10	
	Freshwater catchments	w	1,445	3,449	1.43	Only acidity CLs.

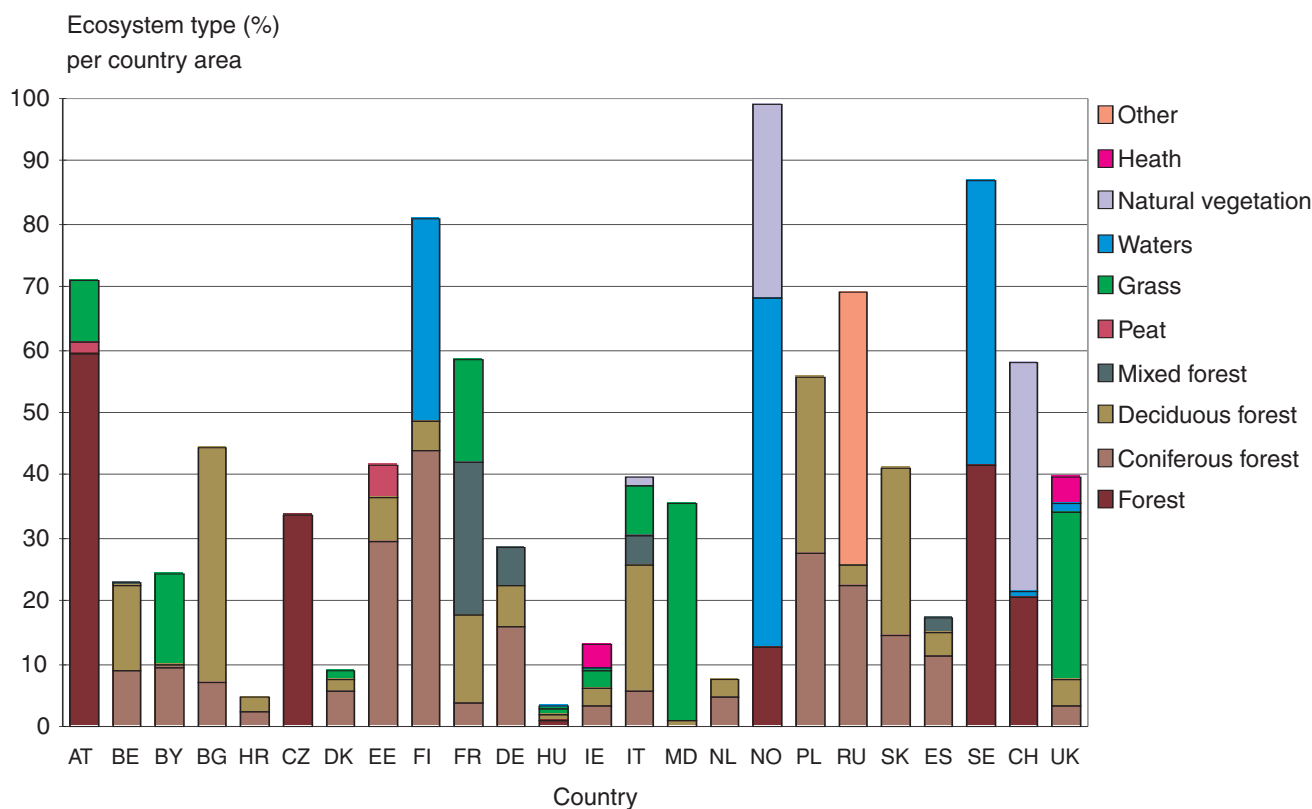


Figure 2-1. The national distribution of ecosystem types and their areas as percentage of the total country area.

Table 2-3. Number of critical loads per national contribution.

Country	Area ¹ (km ²)	A	B	C	D
		Acidity Critical Loads:			
		No. of ecosystems	Area (km ²)	Ecosystem cover (%)	Average ecosystem area (km ²)
Austria	83,845	6,604	49,918	59.5	7.56
Belgium	30,518	2,532	7,024	23.0	2.77
Belarus	207,595	555	50,286	24.2	90.61
Bulgaria	110,994	84	49,476	44.6	589.00
Croatia ²	56,538	34	2,698	4.8	79.36
Czech Republic	78,864	29,418	26,568	33.7	0.90
Denmark	43,094	18,784	3,895	9.0	0.21
Estonia	45,227	140	18,910	41.8	135.07
Finland	338,144	4,533	272,861	80.7	60.19
France	543,965	591	317,703	58.4	537.57
Germany	357,022	410,355	102,589	28.7	0.25
Hungary	93,030	42	2,847	3.1	67.77
Ireland	70,285	34,047	9,077	12.9	0.27
Italy	301,302	376	105,599	35.0	280.85
Netherlands	41,865	127,269	3,196	7.6	0.03
Norway	323,877	3,025	221,231	68.3	73.13
Poland	312,685	3,914	173,472	55.5	44.32
Rep. of Moldova	33,700	141	11,985	35.6	85.00
Russian Federation ³	5,090,400	14,216	3,517,140	69.1	247.41
Slovakia	49,036	320,891	20,056	40.9	0.06
Spain	497,509	3,409	85,225	17.1	25.00
Sweden	449,964	4,234	387,871	86.2	91.61
Switzerland	41,285	12,349	12,349	29.9	1.00
United Kingdom	241,752	320,405	95,818	39.6	0.30
Totals:	9,442,496	1,317,948	5,547,794		

Country	Area ¹ (km ²)	E	F	G	H
		Critical Loads of Nutrient Nitrogen:			
		No. of ecosystems	Area (km ²)	Ecosystem cover (%)	Average ecosystem area (km ²)
Austria	83,845	7,901	59,690	71.2	7.55
Belgium	30,518	2,532	7,024	23.0	2.77
Belarus	207,595	555	50,286	24.2	90.61
Bulgaria	110,994	84	49,476	44.6	589.00
Croatia ²	56,538	34	2,698	4.8	79.36
Czech Republic	78,864	29,418	26,568	33.7	0.90
Denmark	43,094	9,757	3,149	7.3	0.32
Estonia	45,227	140	18,910	41.8	135.07
Finland	338,144	3,083	165,045	48.8	53.53
France	543,965	591	317,703	58.4	537.57
Germany	357,022	410,355	102,589	28.7	0.25
Hungary	93,030	42	2,847	3.1	67.77
Ireland	70,285	34,047	9,077	12.9	0.27
Italy	301,302	502	119,854	39.8	238.75
Netherlands	41,865	127,269	3,196	7.6	0.03
Norway	323,877	2,330	139,942	43.2	60.06
Poland	312,685	3,914	173,472	55.5	44.32
Rep. of Moldova	33,700	141	11,985	35.6	85.00
Russian Federation ³	5,090,400	14,216	3,517,140	69.1	247.41
Slovakia	49,036	320,891	20,056	40.9	0.06
Spain	497,509	3,409	85,225	17.1	25.00
Sweden	449,964	1,883	188,056	41.8	99.87
Switzerland	41,285	22,789	22,789	55.2	1.00
United Kingdom	241,752	318,923	92,363	38.2	0.29
Totals:	9,442,496	1,314,806	5,189,137		

1. Source: Der Fischer Weltatlas '98, Fischer Verlag, Frankfurt.
3. Country area within EMEP domain.

2. Two EMEP 50x50 km² grid cells only.

Table 2-3 (continued). Number of critical loads per national contribution.

Country	Area ¹ (km ²)	I	J	K	L
		<i>Both acidity and nutrient nitrogen CLs:</i>			
		No. of ecosystems	Area (km ²)	Ecosystem cover (%)	Average ecosystem area (km ²)
Austria	83,845	6,604	49,918	59.5	7.56
Belgium	30,518	2,532	7,024	23.0	2.77
Belarus	207,595	555	50,286	24.2	90.61
Bulgaria	110,994	84	49,476	44.6	589.00
Croatia ²	56,538	34	2,698	4.8	79.36
Czech Republic	78,864	29,418	26,568	33.7	0.90
Denmark	43,094	9,757	3,149	7.3	0.32
Estonia	45,227	140	18,910	41.8	135.07
Finland	338,144	3,083	165,045	48.8	53.53
France	543,965	591	317,703	58.4	537.57
Germany	357,022	410,355	102,589	28.7	0.25
Hungary	93,030	42	2,847	3.1	67.77
Ireland	70,285	34,047	9,077	12.9	0.27
Italy	301,302	376	105,599	35.0	280.85
Netherlands	41,865	127,269	3,196	7.6	0.03
Norway	323,877	720	40,522	12.5	56.28
Poland	312,685	3,914	173,472	55.5	44.32
Rep. of Moldova	33,700	141	11,985	35.6	85.00
Russian Federation ³	5,090,400	14,216	3,517,140	69.1	247.41
Slovakia	49,036	320,891	20,056	40.9	0.06
Spain	497,509	3,409	85,225	17.1	25.00
Sweden	449,964	1,856	184,746	41.1	99.54
Switzerland	41,285	11,201	11,201	27.1	1.00
United Kingdom	241,752	318,923	92,363	38.2	0.29
Totals:	9,442,496	1,300,158	5,050,793		

Country	Area ¹ (km ²)	M	N	O	P
		<i>Acidity and/or nutrient nitrogen CLs:</i>			
		No. of ecosystems	Area (km ²)	Ecosystem cover (%)	Average ecosystem area (km ²)
Austria	83,845	7,901	59,690	71.2	7.55
Belgium	30,518	2,532	7,024	23.0	2.77
Belarus	207,595	555	50,286	24.2	90.61
Bulgaria	110,994	84	49,476	44.6	589.00
Croatia ²	56,538	34	2,698	4.8	79.36
Czech Republic	78,864	29,418	26,568	33.7	0.90
Denmark	43,094	18,784	3,895	9.0	0.21
Estonia	45,227	140	18,910	41.8	135.07
Finland	338,144	4,533	272,861	80.7	60.19
France	543,965	591	317,703	58.4	537.57
Germany	357,022	410,355	102,589	28.7	0.25
Hungary	93,030	42	2,847	3.1	67.77
Ireland	70,285	34,047	9,077	12.9	0.27
Italy	301,302	502	119,854	39.8	238.75
Netherlands	41,865	127,269	3,196	7.6	0.03
Norway	323,877	4,635	320,651	99.0	69.18
Poland	312,685	3,914	173,472	55.5	44.32
Rep. of Moldova	33,700	141	11,985	35.6	85.00
Russian Federation ³	5,090,400	14,216	3,517,140	69.1	247.41
Slovakia	49,036	320,891	20,056	40.9	0.06
Spain	497,509	3,409	85,225	17.1	25.00
Sweden	449,964	4,261	391,181	86.9	91.80
Switzerland	41,285	23,937	23,937	58.0	1.00
United Kingdom	241,752	320,405	95,818	39.6	0.30
Totals:	9,442,496	1,332,596	5,686,138		

1. Source: Der Fischer Weltatlas '98, Fischer Verlag, Frankfurt.

2. Two EMEP 50x50 km² grid cells only.

3. Country area within EMEP domain.

2.3 Discussion of national contributions

The density of critical loads mapped varies greatly among countries. Figure 2-2 emphasizes this variation by presenting the total number of ecosystems (black bars) and their total area as a percentage of the country's area (gray bars). The country codes are listed in Table 2-1. For example, the Netherlands computes critical loads for about only 8% (3,196 km²) of its land, but the number of ecosystem points (127,269) is very high compared with other countries. Especially Germany and to a lesser extent Slovakia and the United Kingdom show similar characteristics.

On the other hand, countries like Finland, Sweden and particularly Norway have critical loads mapped for large parts of the country based on a much smaller set of ecosystems. These Fennoscandian countries provide critical loads for both forest soils and surface waters and include most of the country's area. A complication in mapping critical loads for surface waters in these countries is that the extended forests (and natural vegetation in Norway) are overlapping with the catchment areas. All three countries have used distribution ratios for assigning area portions to each type of ecosystem and its critical load. Norway considered the total country area representative for the mapping of critical loads, whereas

Finland and Sweden excluded about 15% of their land area as being built-up or under agricultural use.

Another interesting case, from a geographical point of view, is comparing the mapping methodology for critical loads in the Czech Republic and Slovakia, which formed one nation until recently. Both countries submit critical loads for forest ecosystems only, and the total forest cover is about the same in both countries. Figure 2-2 shows that the Czech Republic has 34% of its area represented by 29,418 records, with an average area per record of 0.90 km². In Slovakia, 41% of the area is represented by 320,691 records, resulting in 0.0625 km² per record. Despite the same forest density the average ecosystem area per ecosystem record is quite different for each country. The methodology of mapping the forest soil critical loads causes the difference: the Czech NFC presents its selected forest areas as irregular polygons, whereas the Slovak NFC mapped its forest areas as fixed grid cells of 250x250 m².

The National Focal Centers are free to choose the resolution for mapping critical loads. They are encouraged to submit data on the same resolution as they use in national applications. In addition, the longitude, latitude and EMEP 50x50 km² grid indices have to be provided. These grid indices are necessary to produce information

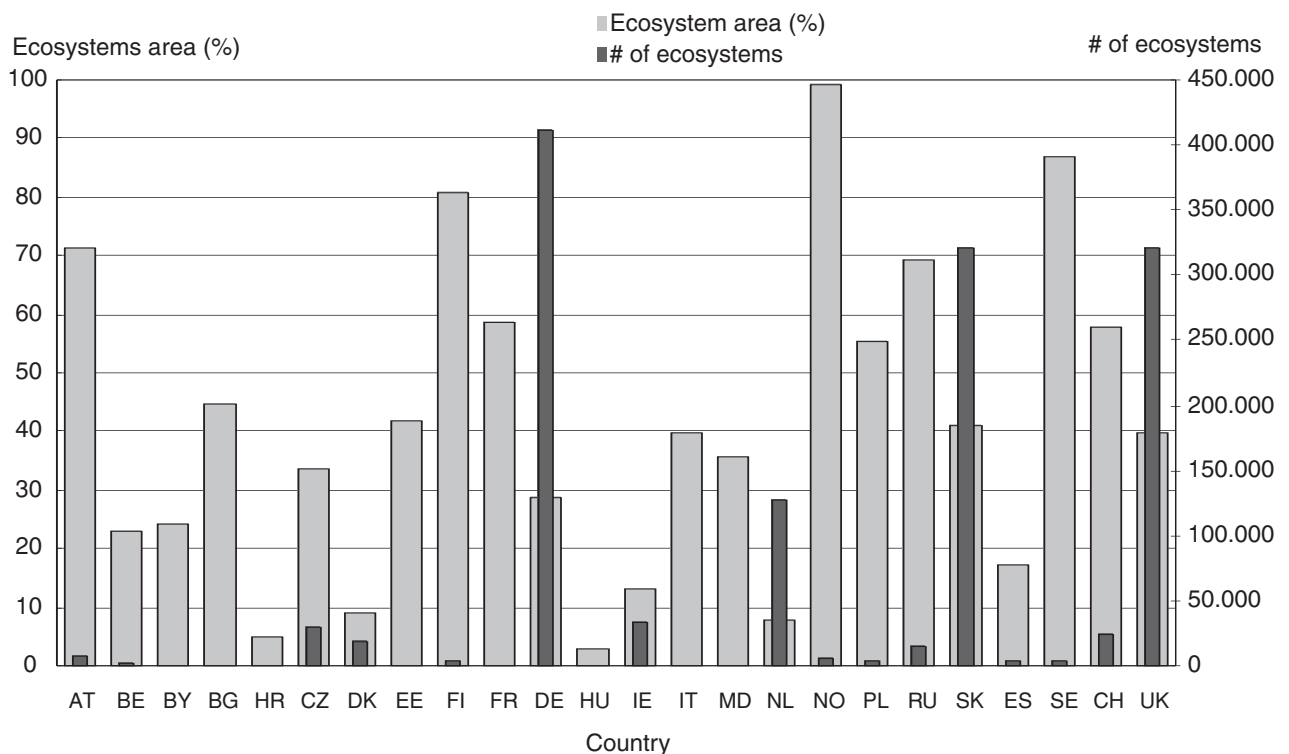


Figure 2-2. Histogram showing the area for which critical loads are provided (percentage of total country area; grey shaded bars) and the number of ecosystems (black bars) per country.

on critical loads relevant for the CRLTAP integrated assessment modeling. While many countries follow this advice, some NFCs aggregate their national data quite substantially. Thus the CCE received only a small number of mapped critical loads per 50×50km² grid cell, often with a large (average) ecosystem area (see Table 2-3). This type of data aggregation significantly reduces the information within a grid cell, and can lead to difficulties when the data are used in integrated assessment modeling. Bulgaria and Italy provided only one record of critical loads per ecosystem type in each grid cell, while Hungary supplied only a single critical load record for each grid cell.

2.4 Comparison of national critical load distributions

While the maps presented in Chapter 1 give an impression of the spatial distribution of a few (low) percentiles of the critical loads over Europe, Figure 2-3 highlights the main characteristics of the distributions of the four critical load quantities provided by the NFCs. For each country, seven percentiles (0, 5, 25, 50, 75, 95, and 100%) are shown, and their relative position gives a fairly good picture of each distribution as a whole.

The distributions of $CL_{max}(S)$ vary considerably between countries. In several countries, the minima are at or near zero (AT, BE, CH, DE, DK, FI, NO, SE, SK and UK), while in some countries the 5th percentile is also quite low (FI, NO, SE and SK). The median of the distributions varies considerably, from less than 1000 eq ha⁻¹ yr⁻¹ (in FI, NO, SE and UK) to about 2000 eq ha⁻¹ yr⁻¹ in most countries and more than 10,000 eq ha⁻¹ yr⁻¹ in Spain. An exception are the Bulgarian critical loads for acidity, which are all above 2000 eq ha⁻¹ yr⁻¹, indicating that there is no acidification problem even at present-day deposition levels. The Swiss distribution shows that about 50% of the critical loads of acidity are more than 2000 eq ha⁻¹ yr⁻¹, reflecting the calcareous soils in the country. In Spain, $CL_{max}(S)$ values for insensitive (calcareous) soils (more than 50% of the total ecosystem area) have been set to 10,000 eq ha⁻¹ yr⁻¹.

The distributions of $CL_{min}(N)$, which reflect the amount of nitrogen retained/removed by immobilization and biomass harvesting (and denitrification in some countries), are quite narrow in almost all countries with the exception of the Republic of Moldova. In that country the soils in more than half of the total ecosystem area seem to have the ability to retain or remove more than 50 kg N ha⁻¹ in a sustainable manner. In contrast, for Hungary all $CL_{min}(N)$ values provided to the CCE are zero.

The distributions of $CL_{max}(N)$ are similar to those of $CL_{max}(S)$, with $CL_{max}(N) \geq CL_{max}(S) + CL_{min}(N)$. This addition of a term to $CL_{max}(S)$ causes the distributions to shift away from zero.

The distributions of $CL_{nut}(N)$ mostly reflect those of the respective $CL_{min}(N)$, to which a N leaching term is added to obtain the critical load of nutrient N. The largest differences between $CL_{min}(N)$ and $CL_{nut}(N)$ distribution can be observed for Hungary and the United Kingdom, indicating a fairly large “acceptable” N leaching and/or denitrification.

2.5 Cumulative distribution functions of national input data

For critical loads derived from a model, e.g. the Simple Mass Balance (SMB) model, variations and differences within and between countries can be explained by variations in the basic input parameters. Since all NFCs submitted critical loads data for forest soils, and were mostly using the SMB model to derive them, the relevant input parameters are analyzed and compared in this section. In total, 1,055,638 critical load values for forest soils have been submitted by the 24 NFCs, which is about 80% of all critical loads submitted.

Figure 2-4 shows the seasalt-corrected base cation deposition provided by the NFCs. The values shown are the area-weighted means in the 50×50 km² grid cells for which forest critical loads are calculated. Generally, base cation deposition is lower in the northwest of Europe and increases towards the southeast. Exceptions to this trend include Belgium, which reports a considerably higher deposition than its neighbors; and Hungary, where the deposition onto forests is rather lower than in Austria and Slovakia.

Cumulative distribution functions of the weathering and the critical leaching on ANC are shown in Figure 2-5. Weathering rates are dependent on parent material and soil, and thus can be expected to vary widely between and even within countries. Countries with weathering rates exceeding 5000 eq ha⁻¹ yr⁻¹ likely contain calcareous soils. Some countries (CH, DK, SE) report both negative and positive critical ANC leaching values, suggesting that acid deposition should be reduced below net base cation input to allow base cation replenishment in the soil. Note that the SMB model always yields a value for $-ANC_{le(crit)} \geq 0$, indicating that those three countries used other models (e.g. PROFILE) to calculate acidity critical loads. For Italy and Norway, no data on weathering and ANC leaching have been reported to the CCE.

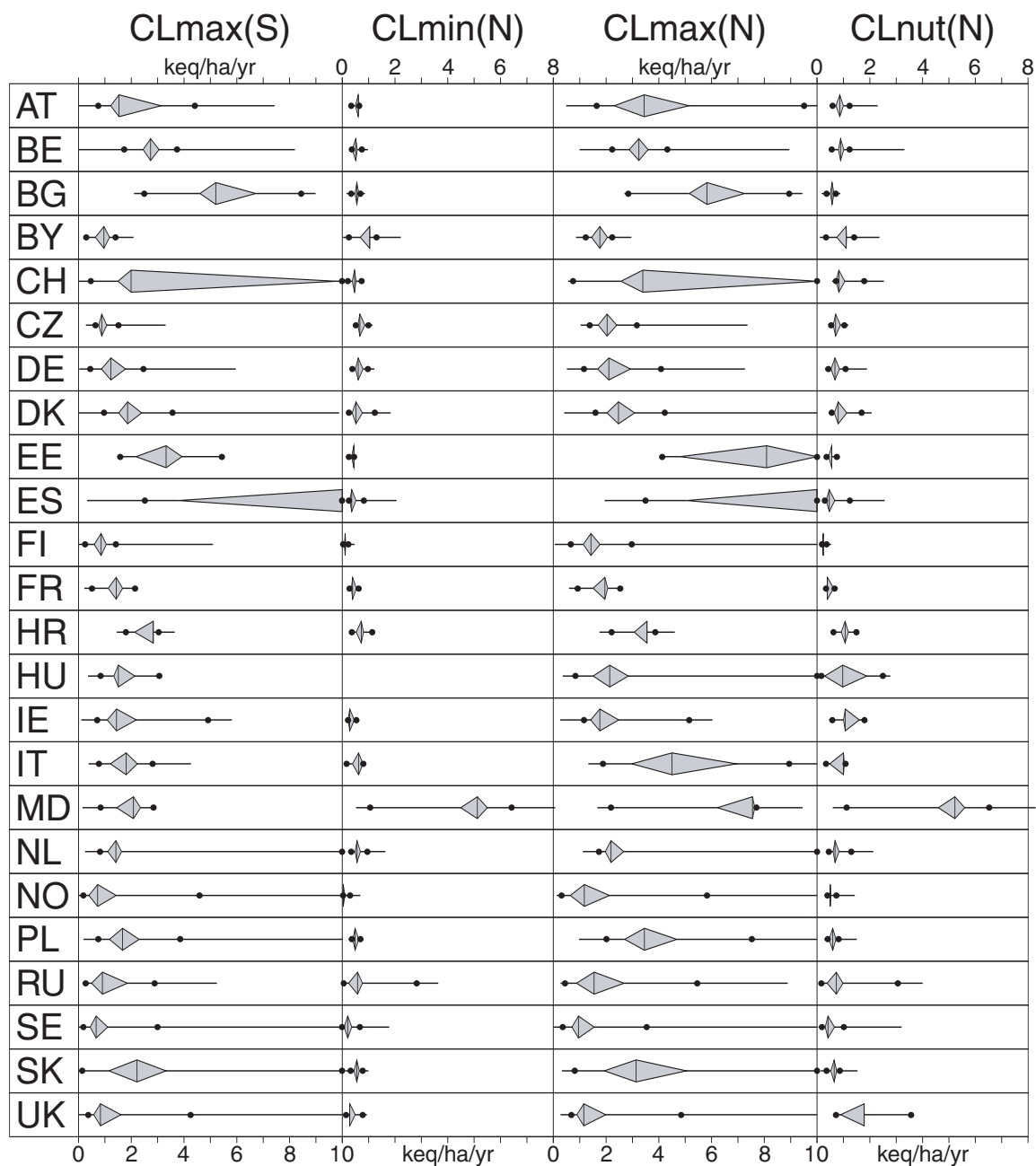


Figure 2-3. The main characteristics of the distributions of the four critical loads as provided by the 24 NFCs. Countries are alphabetically ordered according to their 2-letter code (see Table 2-1 for a listing of codes).

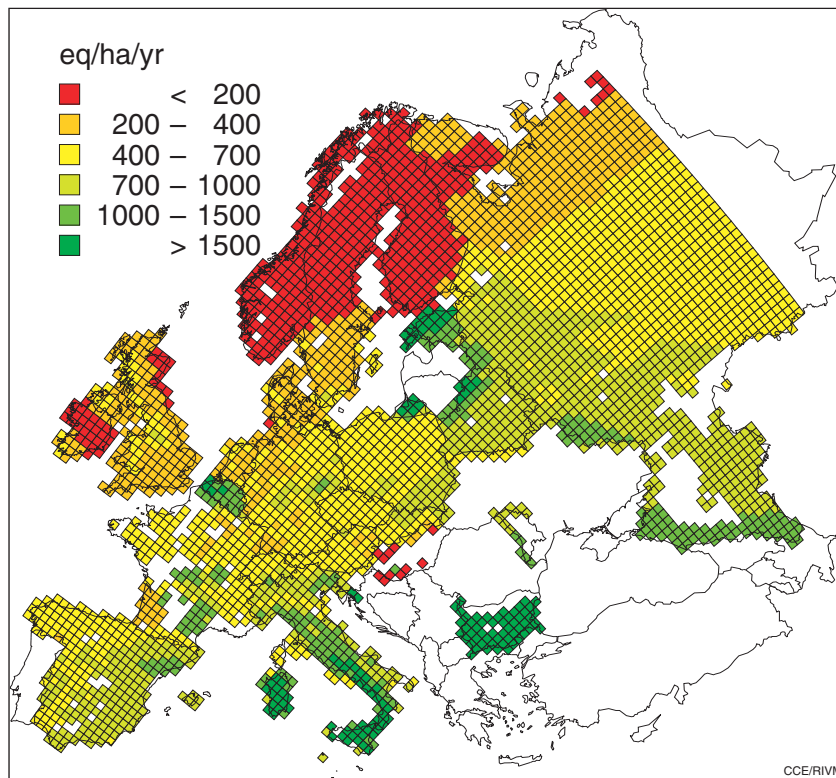


Figure 2-4. Area-weighted mean seasalt-corrected base cation deposition onto forest ecosystems on the 50x50 km² grid as provided by the 24 NFCs.

Figure 2-6 shows the cumulative distributions of nitrogen immobilization and acceptable nitrogen leaching. The Mapping Manual (UBA 1996) recommends values for N immobilization in the range between 0.5 and 1 kg N ha⁻¹ yr⁻¹ (35.7–71.4 eq ha⁻¹ yr⁻¹). These values are based on estimates of the net N immobilization in Swedish forest soils since the last glaciation (Rosén *et al.* 1992). It can be seen from Figure 2-6 that most countries considered (substantially) higher values of N_i as sustainable, i.e. still low enough to avoid N saturation and consequent leaching in the long run. A large variation in the acceptable N leaching can be seen between countries, but also within a few individual countries. This can be mostly explained by the variation in net precipitation (runoff), but might also reflect different criteria used for the acceptable N concentration in the runoff. For Sweden no separate N leaching data have been provided, as they are included in the N immobilization data.

Cumulative distributions of the uptake of base cations (Ca+Mg+K) and nitrogen are shown in Figure 2-7. For calculating critical loads, only the annual average growth uptake (equal to the amount removed by biomass harvest-

ing) should be included, since the large amounts of base cations and N taken up by leaf and needle growth re-enter the soil. Uptake values thus depend not only on the tree species and climatic region, but also on the harvesting practices how much is taken up. For example, strict nature reserves, from which no trees are removed, should be assigned zero uptake values. Uptake values lie in the expected ranges for most countries, although sometimes on the high side, with very high uptake values for the Republic of Moldova. The ratio of base cation to nitrogen uptake for a given tree species should be almost constant, with only minor variations due to climate and site quality. The correlations between base cation and nitrogen uptake (≤ 1600 eq ha⁻¹ yr⁻¹) for each national contribution are shown in Figure 2-8. Comparing this figure with the corresponding one in the 1997 CCE Status Report shows that many countries have improved their national data on uptake values in this respect.

More information on the critical loads and the parameters used to calculate them can be found in the National Focal Center reports contained in Part III of this report.

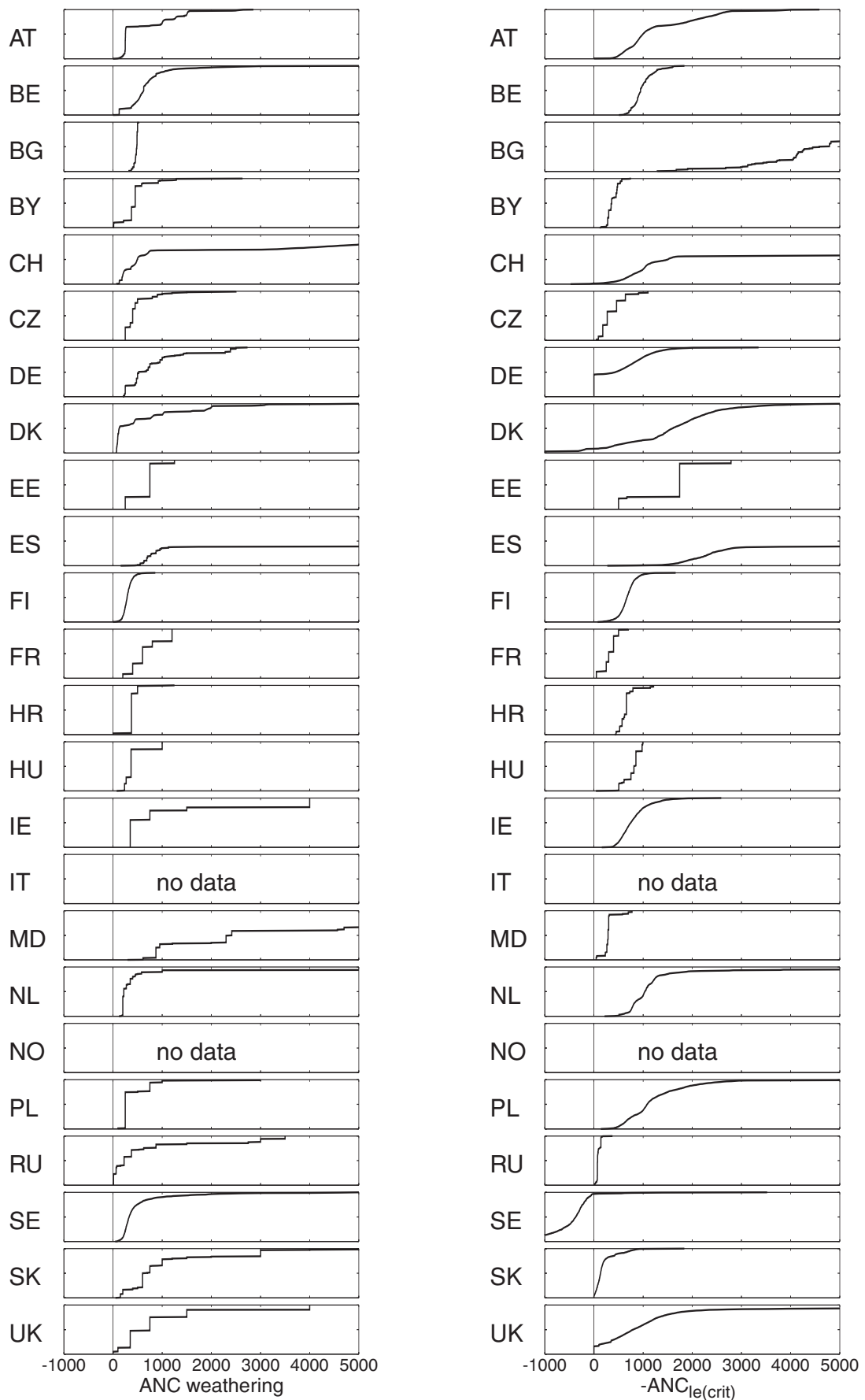


Figure 2-5. Cumulative distribution functions for ANC weathering and critical ANC leaching ($-ANC_{le(crit)}$) in $\text{eq ha}^{-1} \text{yr}^{-1}$ used to calculate critical loads (using primarily SMB) for forest soils from 24 national data contributions.

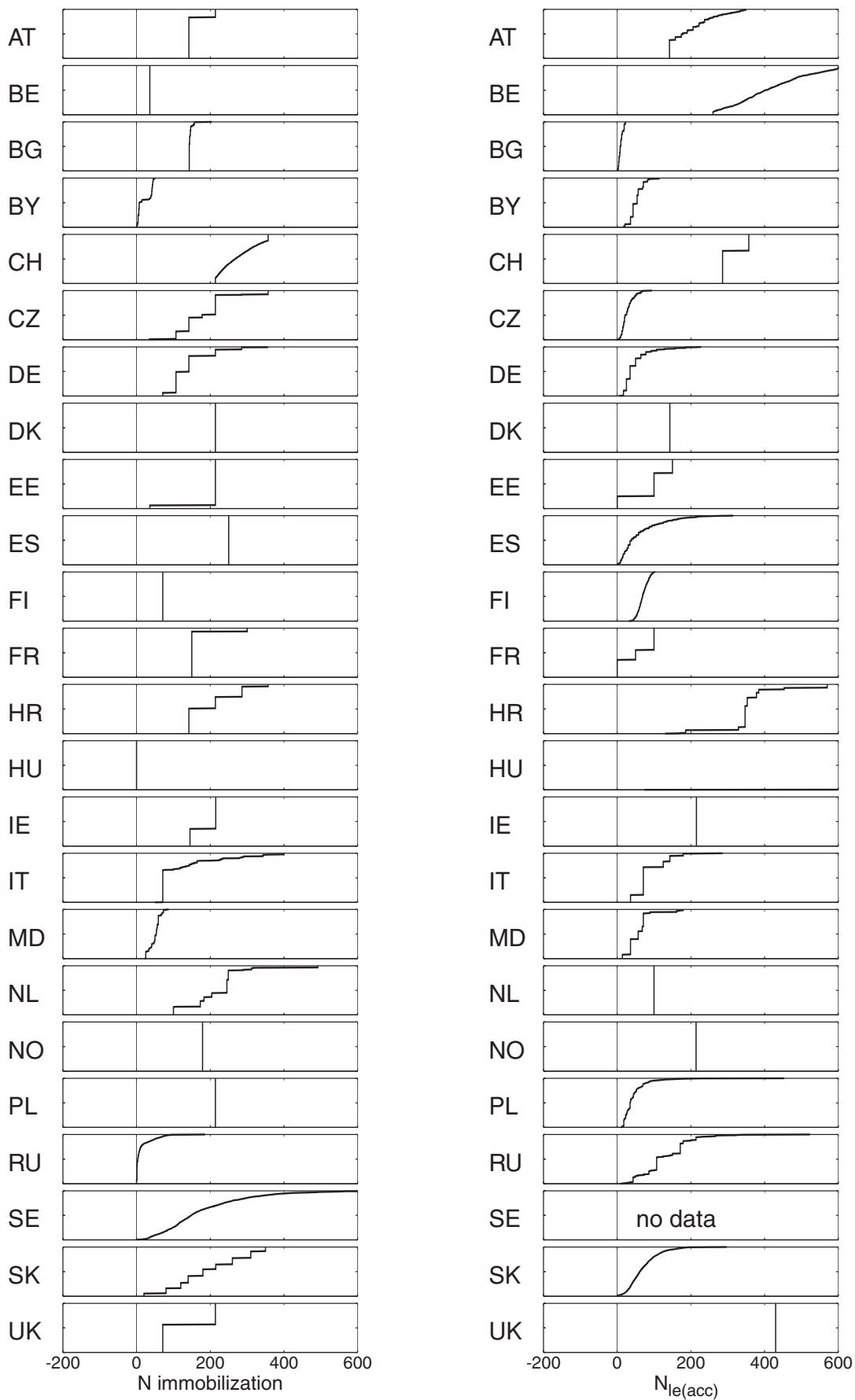


Figure 2-6. Cumulative distribution functions for nitrogen immobilization and acceptable nitrogen leaching ($N_{le(acc)}$) in $eq\ ha^{-1}\ yr^{-1}$ used to calculate critical loads (using primarily SMB) for forest soils from 24 national data contributions.

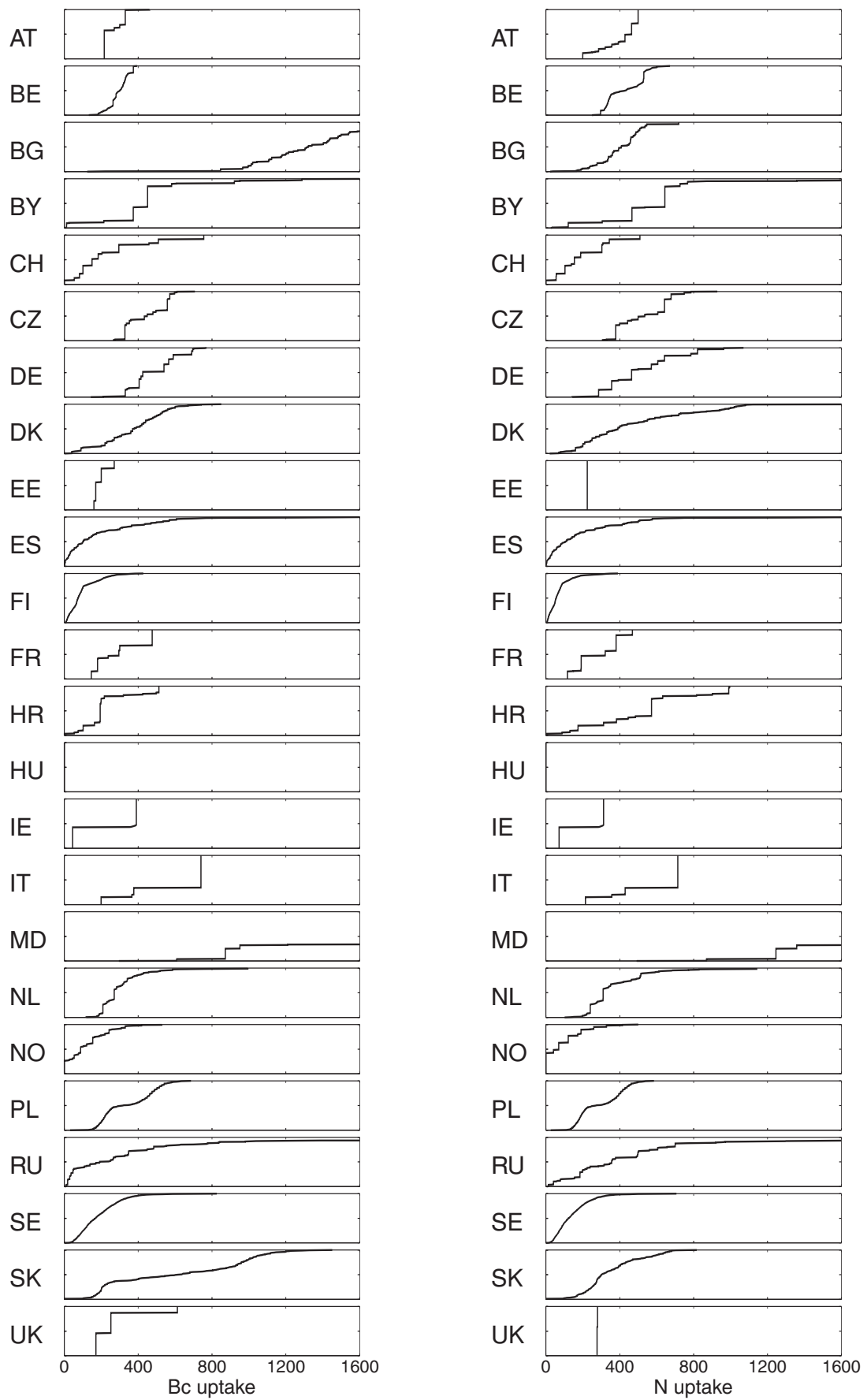


Figure 2-7. Cumulative distribution functions for the base cation (Ca+Mg+K) and nitrogen net uptake in eq ha⁻¹ yr⁻¹ used to calculate critical loads (using primarily SMB) for forest soils from 24 national data contributions.

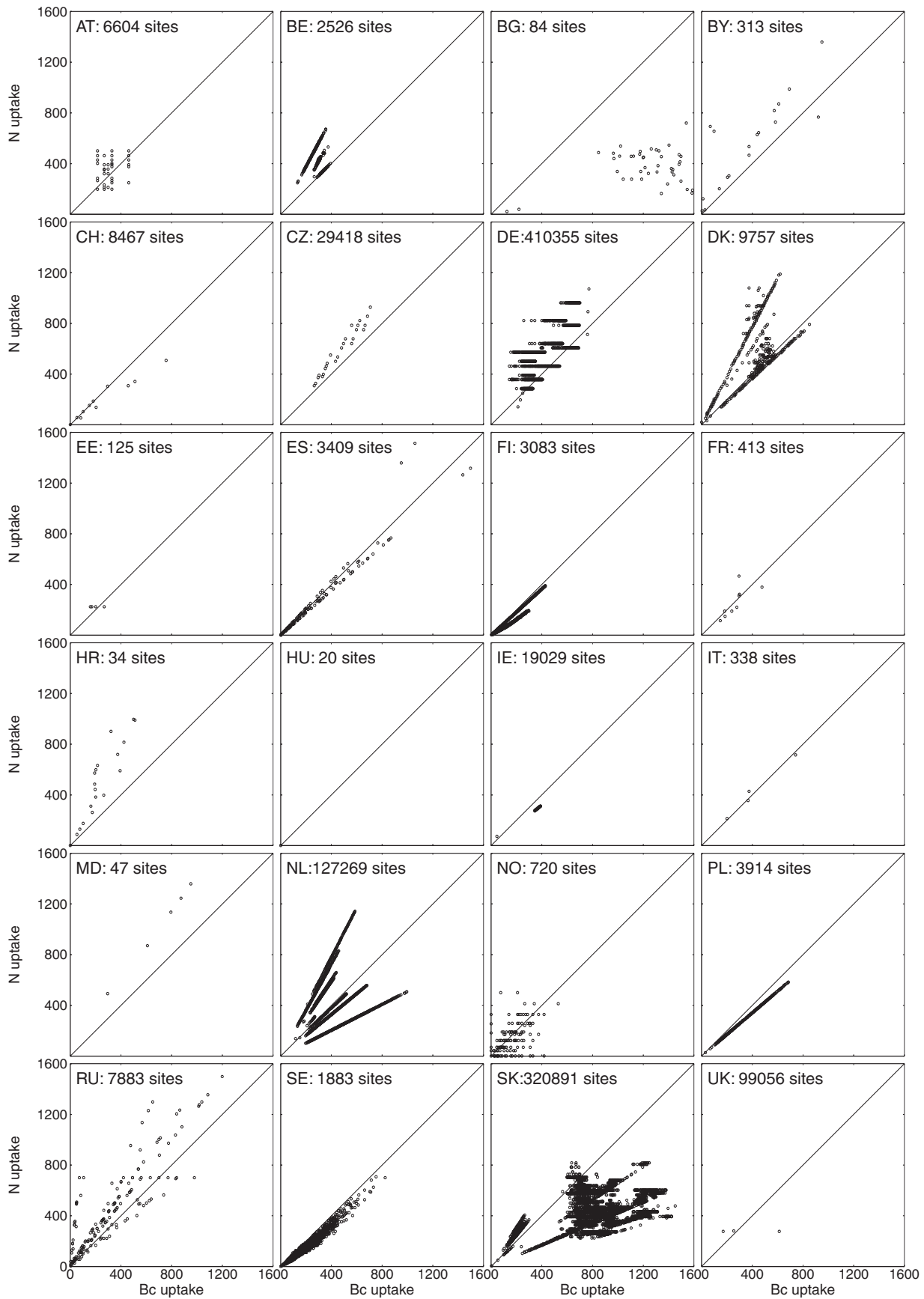


Figure 2-8. Correlation between net base cation (Ca+Mg+K) and nitrogen uptake in eq ha⁻¹ yr⁻¹ used to calculate critical loads (using primarily SMB) for forest soils from 24 national data contributions. (See also Fig. 2-7).

2.6 Concluding remarks

Twenty-four countries contribute now national critical loads data for the work under the LRTAP Convention, and most of Europe is now covered by national data. The data sets approved by the WGE in 1998 and discussed in this report serve as input to the preparations and negotiations on a multi-pollutant, multi-effect protocol, expected to be signed by the end of 1999.

During the 1998 update the CCE noticed a growing understanding at the NFCs of the requirements for mapping critical loads needed within the LRTAP framework. However, the following points should be revisited when preparing for a new update:

Choice of ecosystems: Some countries still submit critical loads for ecosystem types which may have questionable or marginal sensitivity to acidification and/or eutrophication. The Mapping Manual explains that only natural ecosystems unaffected by anthropogenic influences should be mapped. In general, agricultural areas should be excluded from critical load calculations; only grasslands with extensive grazing might qualify as receptors for calculating critical loads.

Cross-border discrepancies: The maps presented in Chapter 1 show still several instances in which two or more countries that each contribute national data have large differences in the calculated values along their common borders. These differences originate in most cases from variations in chosen input values for computing critical loads, and thus the respective NFCs are encouraged to address these discrepancies on a bilateral basis.

Data resolution and aggregation: At present, countries are free to choose the resolution for mapping critical loads. While in general this does not cause problems, a large aggregation leads to a very small number of critical load values per grid cell, and thus to problems in integrated assessment modeling. Thus all countries are encouraged to make full use of the data they have available, and avoid aggregating national data whenever possible.

Input data selection: More care should be taken when selecting parameter values for calculating critical loads. If a parameter cannot be readily determined by field measurements, values or ranges suggested in the Mapping Manual should be used to ensure compatibility of critical loads on a European scale.

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3. Defining an Exceedance Function

M. Posch

Introduction

In case of a single critical load the exceedance is simply defined as the difference between deposition and critical load. The reduction of the exceedances of sulfur (the “gap closure”) was the aim of the 1994 Sulphur Protocol. When considering both sulfur and nitrogen, a unique critical for acidity can no longer be defined and consequently no unique exceedance exists. However, to provide a tool for integrated assessment which allows definition of a gap closure, an exceedance function is defined in this chapter. The exceedance functions for individual ecosystems can be combined to yield the (average) accumulated exceedance for a chosen region (grid cell). And gap closure on this quantity can be used in addition to – or as a replacement for – the gap closure on ecosystem protection percentages (see also Chapter 1).

3.1 The critical load function

Before defining an exceedance function, we recall the derivation of the critical loads of sulfur and nitrogen acidity. For a given N and S deposition, the simplified charge balance of the soil leachate reads (see Chapter 3 in Posch *et al.* 1995 and UBA 1996):

$$S_{dep} + (1 - f_{de})N_{dep} = BC_{dep}^* + Cl_{dep}^* + BC_w - BC_u + (1 - f_{de})(N_i + N_u) - ANC_{le} \quad (3.1)$$

with the different quantities defined in the above references. Equation 3.1 allows calculation of the ANC leaching for any given N_{dep} and S_{dep} . Conversely, fixing ANC_{le} by a critical value, $ANC_{le(crit)}$, confines the deposition of N and S for which Equation 3.1 holds, and every such deposition pair can be called a critical load of N and S. Furthermore, the N sinks in Equation 3.1 cannot balance sulfur deposition, and thus the maximum critical load of sulfur is given by $CL_{max}(S) = BC_{dep}^* - Cl_{dep}^* + BC_w - BC_u - ANC_{le(crit)}$. As long as N_{dep} stays below the so-called minimum critical load of nitrogen, i.e. $N_{dep} \leq CL_{min}(N) = N_i + N_u$, sulfur can be considered alone. Finally, the maximum critical load of nitrogen, in case of $S_{dep} = 0$, is given by $CL_{max}(N) = CL_{min}(N) + CL_{max}(S)/(1 - f_{de})$. These three quantities define the acidity critical load function (see Figure 3-1). It is impossible to define a unique exceedance, i.e. a unique amount of S and N to be reduced to reach non-exceedance. This is illustrated in Figure 3-1: Let the point E1 denote the (current) deposition of N and S. By reducing N_{dep} substantially,

one reaches the point Z1 and thus non-exceedance without reducing S_{dep} ; on the other hand one can reach non-exceedance by only reducing S_{dep} (by a smaller amount) until reaching Z3; finally, with a reduction of both N_{dep} and S_{dep} one can reach non-exceedance as well (e.g. point Z2). In practice, external factors such as the costs of emission reduction measures will determine the path to be followed to reach zero exceedance.

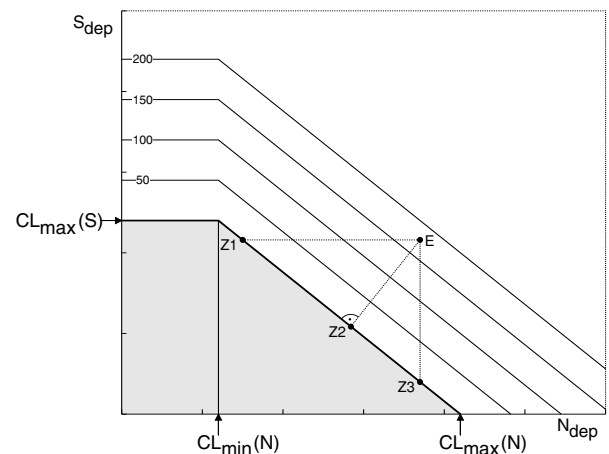


Figure 3-1. Critical load function for S and acidifying N defined by $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ (thick line). The grey-shaded area below the critical load function defines deposition pairs (N_{dep}, S_{dep}) for which there is no exceedance. The thin lines parallel to the critical load function are explained in section 3.4.

3.2 The exceedance function

Intuitively, the reduction required in S and N deposition to reach point Z2 in Figure 3-1, i.e. the “shortest” distance to the critical load function, seems a good measure for exceedance. And we base the definition of an *exceedance function* on this intuition: We define the exceedance for a given pair of depositions (N_{dep}, S_{dep}) and a given critical load function as the sum of the N and S deposition reduction required to reach the critical load function by the “shortest” path. Figure 3-2 depicts the five cases which can arise:

- the deposition falls on or below the critical load function (Region 0). In this case the exceedance is defined as zero (non-exceedance);
- the deposition falls into Region 1 (e.g. point E1). In this case the line perpendicular to the critical load function would yield a negative S_{dep} , and thus every exceedance in this region is defined as the sum of N and S deposition reduction needed to reach point Z1;

functions. The computation of such isolines can be carried out in a similar way as the ecosystem protection isolines (see Chapter 3 and Appendix B in Posch *et al.* 1997, or Chapter 4 and Appendix B in Posch *et al.* 1995). Obviously, protection isolines and (average) exceedance isolines differ from each other, except for the zero isolines which bound the area of non-exceedance. Figure 3-4 displays examples of AAE and protection isolines for the EMEP150 grid cell (21,14).

If a critical load is defined for a single pollutant, e.g. $CL_{nut}(N)$, the exceedance function is given by $Ex = \max\{Dep - CL, 0\}$, and the accumulated exceedance and average accumulated exceedance are given by, respectively:

$$AE(Dep) = \sum_{i=1}^N A_i Ex_i(Dep) \quad (3.5)$$

and

$$AAE(Dep) = AE(Dep) / \sum_{i=1}^N A_i \quad (3.6)$$

A comparison of the different exceedances and gap closure methods can be found in Chapter 1.

3.4 Postscript

Looking at Figure 3-2 and Equation 3.2, the definition of the exceedance function looks quite “complicated”. As an alternative, it has been suggested to define the exceedance function via the charge balance (cf. Equation 3.1):

$$Ex_{le}(N_{dep}, S_{dep}) = S_{dep} + (1 - f_{de})N_{dep} - (1 - f_{de})CL_{min}(N) - CL_{max}(S) \quad (3.7)$$

where we assume for simplicity that $N_{dep} \geq CL_{min}(N)$. This exceedance is obviously zero if (N_{dep}, S_{dep}) lies on the critical load function, negative when it lies below and positive when it lies above. Figure 3-1 depicts isolines of $Ex_{le}(N_{dep}, S_{dep})$ which run parallel to the critical load function. Comparing Figure 3-1 with Figure 3-3 shows that Ex_{le} is, in general, different from the exceedance function defined in section 3.2. This can be easily verified from the equations given in this chapter, but has been done earlier, *inter alia*, in Henriksen *et al.* (1993). The reason for the difference is that one 1 eq of N deposited does not result in 1 eq of acidity leached. Ex_{le} is the excess acidity (or ANC) leached from the soil after all transformations have taken place, whereas Ex defined in section 3.2 is the

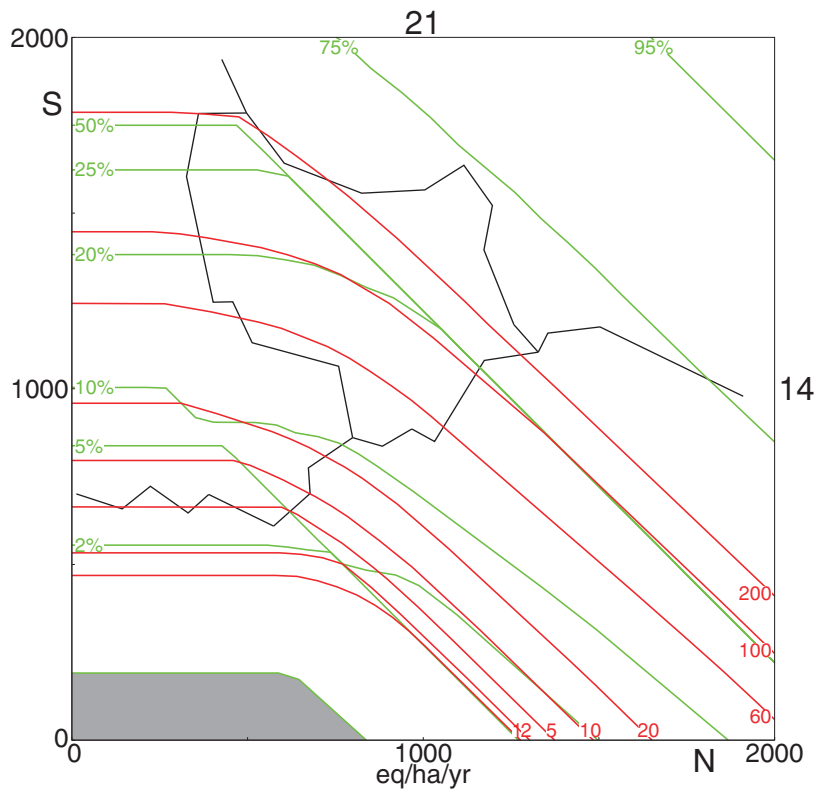


Figure 3-4. Average accumulated exceedance isolines (red) and ecosystem protection isolines (green) for the 7863 critical load functions in EMEP150 grid cell (21,14) which covers Luxembourg (outlined in black). The AAE isolines are labeled in $\text{eq ha}^{-1} \text{ yr}^{-1}$ in the lower right corner, the protection isolines in percent at the left. The grey shaded area is the area of non-exceedance bound by the zero-exceedance isoline.

amount of N and S deposition to be reduced to reach non-exceedance. While the first is important for assessing impacts, the latter is the quantity integrated assessment modelers are interested in.

Acknowledgment

The author would like to thank the many individuals who have offered their suggestions and critiques on the formulation of the above concept. Any shortcomings remain the sole responsibility of the author.

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Part II. Related Research

This part contains two contributions which are of interest to the Effects Programme under the LRTAP Convention. The views expressed in the two papers are those of the authors and do not necessarily reflect the opinions of the editors.

The first paper by J. Hall and colleagues from the UK National Focal Center describes work carried out as “help-in-kind” to the Mapping Programme aiming to contribute to ongoing studies on uncertainties and to consider possible methods for presenting data and results. The paper complements the results presented in Chapter 2 of Part I.

The second paper by K. Kåresen and D. Hirst from the Norwegian Computing Centre is a study on estimating the influence of local variability in deposition on the exceedance of critical loads in Europe, thus contributing constructively to the ongoing debate on the influence of uncertainties on critical load exceedance calculations. Anyone who wants to learn more about their work should contact *david.hirst@nr.no*.

UK Help-in-Kind to the Mapping Programme

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Background

At the 16th session of the UN/ECE Working Group on Effects (August 1997) the UK made an offer of "help in kind" to the Mapping Programme. This was to "contribute to ongoing studies on uncertainties, and to consider possible methods for presenting data and results so as to make them more transparent." As requested by the Working Group on Effects, a proposal of work was drawn up by the UK National Focal Center (NFC) and agreed with members of the Mapping Programme, i.e. the Coordination Center for Effects (CCE) and the Task Force on Mapping (TFM). The proposal aimed to aid the work of the CCE by providing expertise and resources, including Geographic Information Systems (GIS), from the NFC and research groups in the UK. Access to critical loads data for individual countries was gained via correspondence and negotiations with the CCE and the individual NFCs. During this time, initial work was carried out using the UK critical loads data alone. The results of the UK study were presented to the 14th meeting of the Task Force on Mapping in May 1998.

Aims

Four key aims were identified for the study using the European critical loads data:

- (i) to build upon the work carried out by the CCE (Posch et al. 1997) and on work by the UK NFC using UK data only,
- (ii) to improve confidence in critical loads and exceedance maps and data,
- (iii) to explore methods for the presentation and visualization of data and information, and
- (iv) to provide transparency in critical loads data and methods.

Data and questionnaires

Following consultation with the NFCs, 21 out of 24 countries agreed to make their national critical loads data available to the UK NFC for this work under the Convention on Long-range Transboundary Air Pollution.

The data files were supplied to the UK NFC from the CCE; the files were those submitted to, and checked by, the CCE for use in the negotiations for the multi-pollutant, multi-effect protocol.

A database was designed in Microsoft Access for storing, manipulating and cross-referencing the European critical loads data. The database was linked to ArcView GIS to enable direct mapping of ecosystem and critical loads parameter data.

An important aspect of the work was to compare the input values being used for the calculation of critical loads with the recommended values in the Mapping Manual (UBA 1996). To further assist this effort and gain additional information, a questionnaire was sent out to each NFC for each ecosystem considered nationally. This survey gave the minimum and maximum values of each critical loads parameter for each ecosystem, and requested details on the data sources, calculation methods used and the justification for variations from values or ranges recommended in the Mapping Manual. Replies were subsequently received from 12 countries, and some countries additionally provided reports documenting their methods. Completed forms for the UK are included in the UK National Focal Center report contained in Part III of the present report. All replies were entered into the database and examined as described below.

Analysis of data

Ecosystems:

The critical loads data reviewed for the 21 countries cover 10 different ecosystems with individual countries providing data for between one and seven. The most common ecosystems considered are coniferous and deciduous woodland, each being reported for 16 countries (Table 1). It should be noted that for some countries these data are submitted as undistinguished or mixed forest. In most cases both acidity and nutrient nitrogen critical loads were supplied, but for a few ecosystems data for only one of these was calculated. Figure 1 shows the relative areas of different ecosystems in each EMEP 150×150 km² grid square for which critical loads data are reported.

Table 1. Critical loads supplied to the UK NFC by country and ecosystem category.

Country	Ecosystem codes (see list below) showing critical loads (a = acidity, n = nutrient nitrogen) provided by each country										Reply* received
	c	d	f	g	h	m	o	p	v	w	
Austria			a,n	n				n			✓
Belgium	a,n	a,n				a,n				a,n	✓
Bulgaria	a,n	a,n									✓
Belarus	a,n	a,n		a,n							
Czech Republic			a,n								
Denmark	a,n	a,n		a							
Estonia	a,n	a,n						a,n			
Finland	a,n	a,n								a	✓
France	a,n	a,n		a,n		a,n					✓
Germany	a,n	a,n				a,n					✓
Hungary	a,n	a,n	a,n	a,n	a,n			a,n		a,n	
Ireland	a,n	a,n		a,n	a,n					a,n	✓
Netherlands	a,n	a,n									
Norway			a,n						n	a	✓
Poland	a,n	a,n									✓
Russia	a,n	a,n						a,n			
Slovak Republic	a,n	a,n									✓
Spain	a,n	a,n				a,n					
Sweden			a,n							a	✓
Switzerland			a,n						a,n	a,n	✓
United Kingdom	a,n	a,n		a,n	a,n					a	✓
Totals	16a	16a	6a	6a	3a	4a	1a	2a	1a	8a	12
	16n	16n	6n	6n	3n	4n	1n	3n	2n	4n	

* Reply to questionnaire from UK NFC received

Ecosystem key used by the CCE:

- | | | | |
|---|------------------------|---|-------------------------|
| c | coniferous forest | m | mixed forest |
| d | deciduous forest | o | other |
| f | undistinguished forest | p | peatland |
| g | grassland | v | semi-natural vegetation |
| h | heathland | w | water |

In some cases where critical loads were defined at a particular grid square resolution (e.g. 1x1 km²), the ecosystem areas appear to be the same as the grid square. This may be due to large uniform areas of ecosystems or to insufficient data being available on ecosystem areas within the grid square. The latter may lead to overestimation of actual ecosystem areas in the European critical loads database.

The CCE assigns national data for different ecosystems to their own 10 ecosystem categories (Table 1). In some cases this may result in quite different ecosystems being aggregated together. For example, the UK ecosystems of acid grassland and calcareous grassland are both assigned to the CCE grassland category. Although this makes no difference to the way the data are used for European

mapping purposes (and protocol discussions), examination of the data on an ecosystem basis may produce anomalies. While all countries may define coniferous woodland in an identical way, ecosystems such as heathland, peatland and semi-natural vegetation may mean different things in different countries. Since countries were asked to provide data for nationally selected sensitive ecosystems, explicit definitions of ecosystems (from National Focal Centers) and of the CCE categories would highlight anomalies and identify where further clarification is required.

In addition, it is also important that the many ecosystems which may be used for defining nutrient nitrogen critical loads are correctly associated with acidity critical load ecosystems.

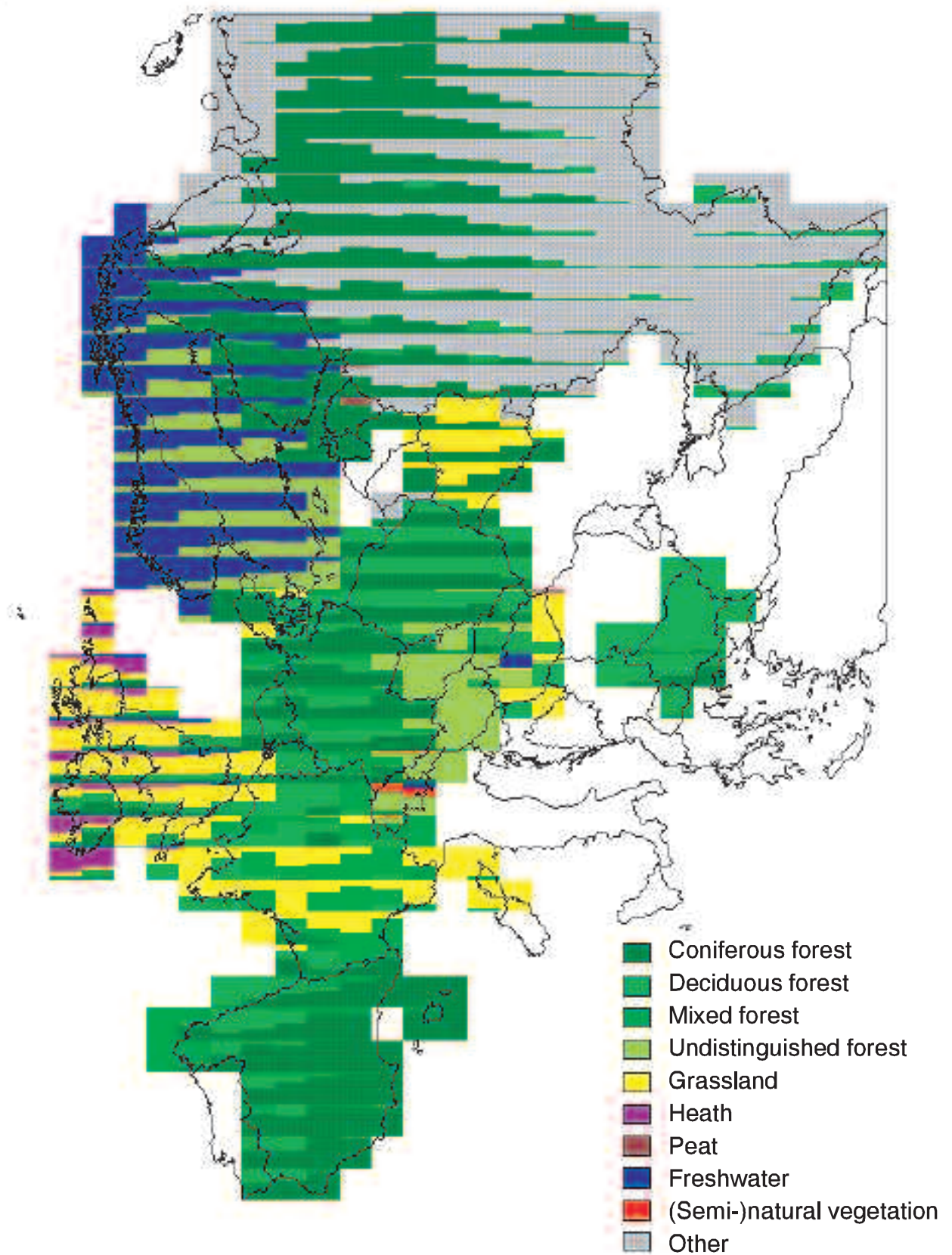


Figure 1. The relative areas of ecosystems in EMEP 150 × 150 km² grid squares for which critical loads are calculated.

Comments on data values:

Values for critical loads parameters submitted by all countries are summarized in Table 2 by ecosystem. For many of these the minimum value recorded is zero; this shows, for example, that some countries set base cation or nitrogen uptake to zero while others do not. Further investigation of some parameters may be required. In addition, there are some very small ecosystem areas recorded (<0.001 km²); however, because they are so small, they will have very little influence on the calculations of percentile critical loads.

Countries responding to the questionnaire provided justification for the data values being used. In most cases the methods and equations used are those recommended in the Mapping Manual (UBA 1996). With respect to particular values for individual critical loads parameters, they are either taken from the Mapping Manual, or were recommended by national experts or literature. The following list describes some of the observations made on the data:

Nitrogen immobilization values were identified by Posch *et al.* (1997) as being greater than the Mapping Manual recommendations of 0.5 to 1 kg N ha⁻¹ yr⁻¹ for long-term sustainability. While the results of this study confirm these findings, the higher values being used are generally supported by national experts or literature (Table 3).

Critical ANC leaching values derived from the simple mass balance (SMB) equation for soil-vegetation ecosystems are a mixture of negative and positive values (Table 2): 14 countries give positive values, 3 negative values and 4 a mixture of both positive and negative values. This requires further investigation. For freshwater ecosystems there can also be valid $ANC_{le(crit)}$ values if a simple mass balance approach is used (e.g. Switzerland). However, it looks as though some critical ANC_{limit} values (e.g. 20 µeq l⁻¹) may have been submitted for $ANC_{le(crit)}$.

Minimum and maximum critical loads for nitrogen ($CL_{min}(N)$ and $CL_{max}(N)$): Some countries include denitrification in these calculations and others do not. The 12 replies from NFCs show that 2 countries include denitrification in the calculation of $CL_{min}(N)$ and four include the denitrification fraction in the calculation of $CL_{max}(N)$; however, the other six countries either do not include denitrification in their calculations, or have not stated explicitly how they calculate $CL_{min}(N)$ and $CL_{max}(N)$.

Chemical criteria used in the calculation of acidity critical loads: Different chemical criteria are used by different countries; for example, in the SMB for soils: base cation to aluminum ratios, calcium to aluminum ratios, critical pH values; and for waters, various ANC limits.

Application of critical loads to specific ecosystems: In

some cases critical loads are applied to one ecosystem, which are actually values calculated for another. For example, where forest or semi-natural vegetation critical loads for nutrient nitrogen have been applied to freshwater ecosystems on the basis that they are present in the same grid square.

Maps:

Critical loads parameters: To visualize the range of input values used in calculating critical loads across Europe, area-weighted mean values have been calculated for some input parameters, by country and ecosystem for each EMEP 150×150 km² grid square. Maps for coniferous woodland ecosystems (the ecosystems mapped by most countries) show the range of values used for base cation uptake and nitrogen uptake (Figure 2). The variation in uptake values may be due to the different amounts of woodland harvested.

Critical load maps: Critical load maps, used under the Convention on Long-range Transboundary Air Pollution for the development of emission abatement protocols, combine critical loads data for all ecosystems. In this way they show percentile critical loads for “sensitive” ecosystems. When differences at country boundaries are observed, they may be thought to be the result of, for example, differences in soils from one country to the next, or countries calculating critical loads for different ecosystems. To explore this issue, 5-percentile maps of $CL_{max}(S)$ were generated for the EMEP area for each ecosystem. The values were calculated on an individual country basis so that cross-border differences are highlighted. For some forest ecosystems (coniferous, deciduous, mixed), cross-boundary differences are apparent (Figure 3). This may be due to countries selecting different chemical criteria in the calculation of acidity critical loads. Table 4 gives the criteria used for different woodland ecosystems for those countries replying to the UK NFC questionnaire.

Exceedance maps: Exceedances of acidity critical loads were calculated to investigate the magnitude of exceedance as well as the areas of ecosystems exceeded. For the former, calculations used 5-percentile acidity critical loads and two EMEP deposition scenarios: 1990 and a 2010 scenario¹. To explore the effects of uncertainty in deposition estimates, exceedances were also calculated using the deposition scenario values ± 30%. The 1990 scenario shows large exceedances across central Europe which significantly decreased in 2010. As expected, decreasing the deposition by 30% decreases the areas exceeded. Conversely, increasing deposition by 30% increases the exceeded area. However, a few countries (or parts of countries) that show

¹ 1990 and 2010 scenarios as used by the Task Force on Integrated Assessment Modelling in 1997. The 2010 scenario is based on emissions for EU countries from the IIASA 3rd Interim Report and emissions for non-EU countries from the December 1996 IIASA report.

Table 2. The minimum and maximum values of critical loads parameters: data for the 21 countries held at the UK NFC.

Minimum and maximum values for critical loads parameters for the following ecosystem categories:																				
Parameter	c	d	f	g	h	m	o	p	v	w										
#CL _{max} (S)	17	41,388	5	40,233	0	48,983	170	5,628	170	5,678	56	28,140	150	5,245	354	2,180	52	29,195	0	106,536
#CL _{min} (N)	0	3,640	0	3,619	0	1,780	0	1,497	0	504	266	1,035	21	3,640	0	350	214	823	0	1,174
#CL _{max} (N)	275	134,640	271	130,647	181	76,805	351	10,172	561	5,894	607	94,115	289	8,885	709	11,338	627	44,298	12	447,194
#CL _{min} (N)	74	4,002	107	4,002	51	3,204	166	3,571	714	2,234	299	1,753	107	4,002	46	499	500	1,786	345	3,310
#BC* _{dep}	0	5,276	0	6,191	12	1,100	54	1,601	70	1,100	0	1,400	114	2,991	95	1,600	14	1,072	12	1,100
#BC _u	0	4,198	0	4,198	0	826	0	1,778	0	45	38	765	6	4,198	0	170	0	755	0	987
#BC _w	0	10,000	0	14,053	0	34,820	0	4,000	266	4,000	127	10,000	10	3,500	73	366	73	13,235	3	20,414
#ANC _{le(crit)}	-3,015	31,135	-12,128	29,828	-3,311	27,580	-2,661	2,472	-2,661	0	-1,840	17,580	5	377	-854	-186	-104	16,217	-958	89,649
#N _u	0	3,498	0	3,498	0	928	0	1,481	0	289	36	892	8	3,498	0	224	0	509	0	27,456
#N _i	0	790	0	1,620	0	1,424	0	215	0	215	35	356	0	185	0	214	214	357	0	360
#N _{le(acc)}	0	1,725	0	1,725	0	2,500	0	2,773	140	2,234	0	698	8	522	0	65	286	357	0	1,730
##Area	0	2,476	0	2,242	0	2,368	0	2,168	0	10	0	2,219	0.02	2,421	0.39	240	1	103	0.01	1,943

Notes:

$$BC^*_{dep} = BC^*_{dep} - Cl^*_{dep} \text{ (i.e. non-marine base cation deposition minus non-marine chloride deposition)}$$

values in eq ha⁻¹ yr⁻¹

values in km²

Ecosystem key used by the CCE:

c	coniferous forest		
d	deciduous forest	m	mixed forest
f	undistinguished forest	o	other
g	grassland	p	peatland
h	heathland	v	semi-natural vegetation
		w	water

Table 3. Nitrogen immobilization values used for forest soils and national justifications for their use.

Country	Nitrogen immobilization values (kg N ha ⁻¹ yr ⁻¹)			Justification given for values
	Coniferous woodland	Deciduous woodland	Undistinguished forest	
Austria	0.5		2.0 – 3.0	Long-term immobilization rates according to literature data.
Belgium	0.5	0.5		Recommended for long-term equilibrium for all soils.
Bulgaria	3.2 – 10.1	0.308 – 10.1		Based on long-term published data. Dependent on soil type.
Finland	1.0	1.0		None given, but in Mapping Manual range.
Germany	1.0 – 5.0	1.0 – 5.0		Dependent on temperature.
Ireland	2.0 – 3.0	2.0 – 3.0		Dependent on soil type.
Norway			2.5	Mapping Manual, pp. 93-94.
Poland	3.0	3.0		Mapping Manual, pp. 93-94.
Slovak Republic	0.28 – 4.9	0.28 – 4.9		Values assigned to annual temperature at forest sites. Based on published data.
Sweden			0.0 – 20.0	Based on “Walse Berg Model” (Walse <i>et al.</i> 1998). Dependent on N deposition, pH, temperature and site characteristics.
Switzerland			3.0 – 5.0	3kg at low altitude (<500m), 5kg at high altitude (>1500m), interpolated in between (FOEFL 1996, p. 29) and Posch <i>et al.</i> 1993.
United Kingdom	1.0 – 3.0	1.0 – 3.0		Based on published data for long-term sustainability. Dependent on soil type.

no exceedance with the original scenario deposition values, still show no exceedance when deposition is increased by 30%, both for the 1990 and 2010 scenarios, reflecting their high critical load values and little likelihood of damage.

To examine the relative areas of ecosystems exceeded under these scenarios, exceedances were calculated using all acidity critical loads data for all ecosystems (i.e. not using a single percentile critical load). This highlights which ecosystems may be protected by a decrease in deposition and which may still be at risk of damage (Figures 4a and 4b). For example, in some EMEP squares in north-east Scotland, acid grassland and heathland ecosystems are the dominant ecosystems exceeded in 1990, whereas coniferous woodlands dominate in 2010.

Conclusions and recommendations for further work

1. Based on the responses to the UK NFC questionnaires, most countries are using the Mapping Manual methods, equations and recommended values.
2. Critical loads (acidity and nutrient nitrogen) are calculated for a wide range of ecosystems across

Europe, with coniferous and deciduous woodland being reported the most often.

3. There may still be some data on ecosystem areas that are grid-based (i.e. the same as the grid size for which critical loads are calculated) rather than “real” areas.
4. Care is needed in the application of critical loads to specific ecosystems to ensure that values are representative of the specific ecosystem. Definitions for both nationally selected ecosystems and the CCE categories would assist this.
5. Nitrogen immobilization values used by most countries are greater than the recommended values of 0.5–1.0 kg N ha⁻¹ yr⁻¹. However, they are generally supported by literature or by expert judgement. The recommendations given in the Mapping Manual may need to be revised in the light of this.
6. Critical ANC leaching values span a wide range of negative and positive values. The reasons for this need further investigation.
7. Some critical ANC limit values (used in freshwater models) may have been submitted as critical ANC leaching values by some countries.
8. Clearer guidance is needed for the calculation of $CL_{min}(N)$ and $CL_{max}(N)$. Some countries include denitrification in the calculation and others do not. The reasons for this are not given in the questionnaire replies and the Mapping Manual is also not clear on this issue.

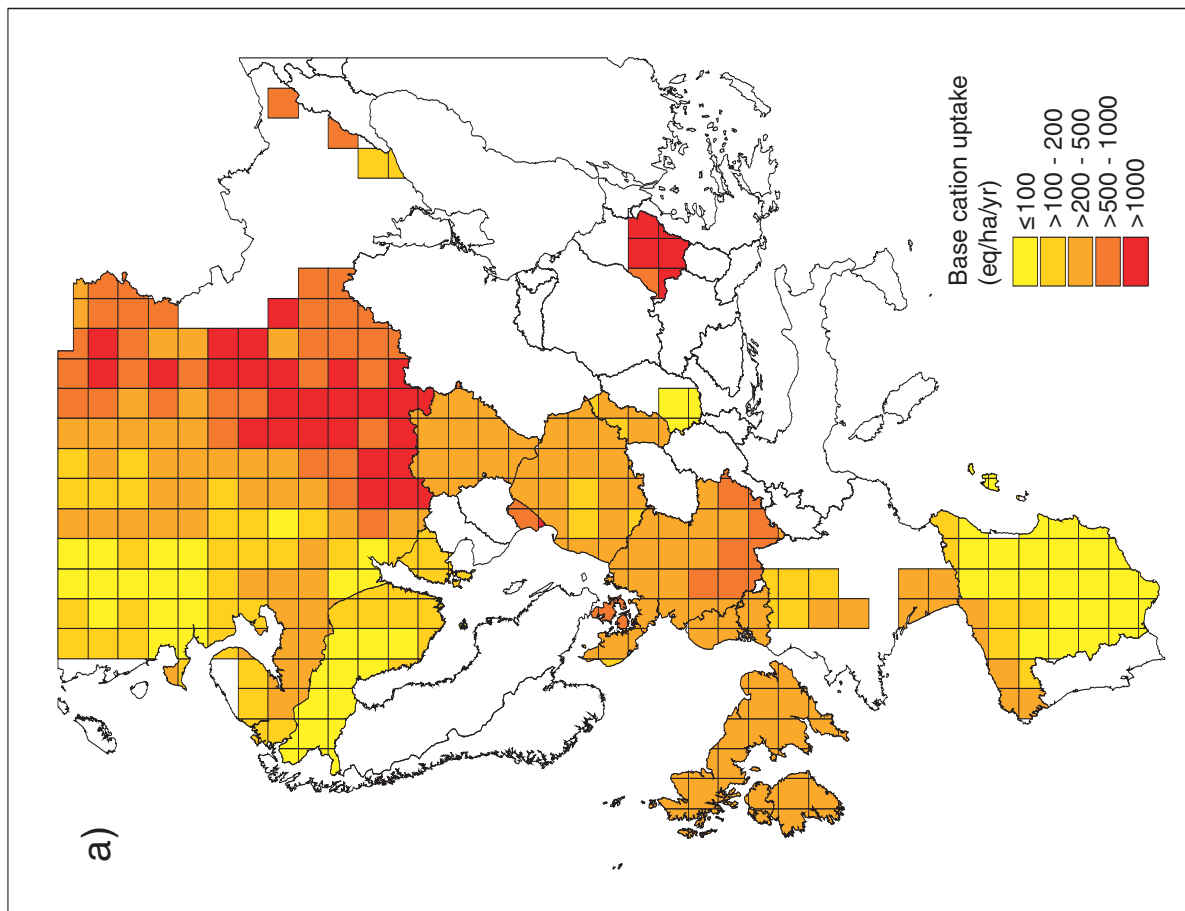
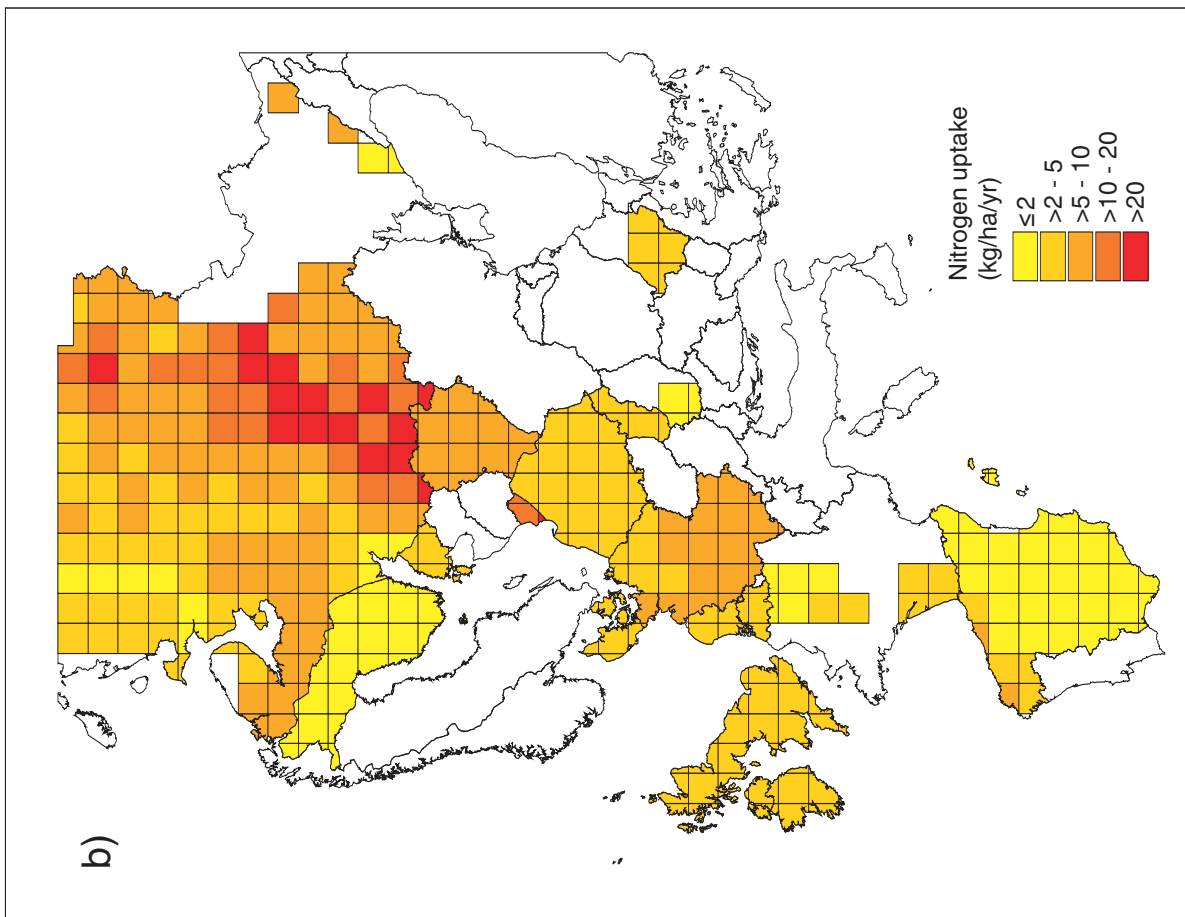


Figure 2. Area-weighted mean uptake values for coniferous woodland (a) base cation uptake, and (b) nitrogen uptake.

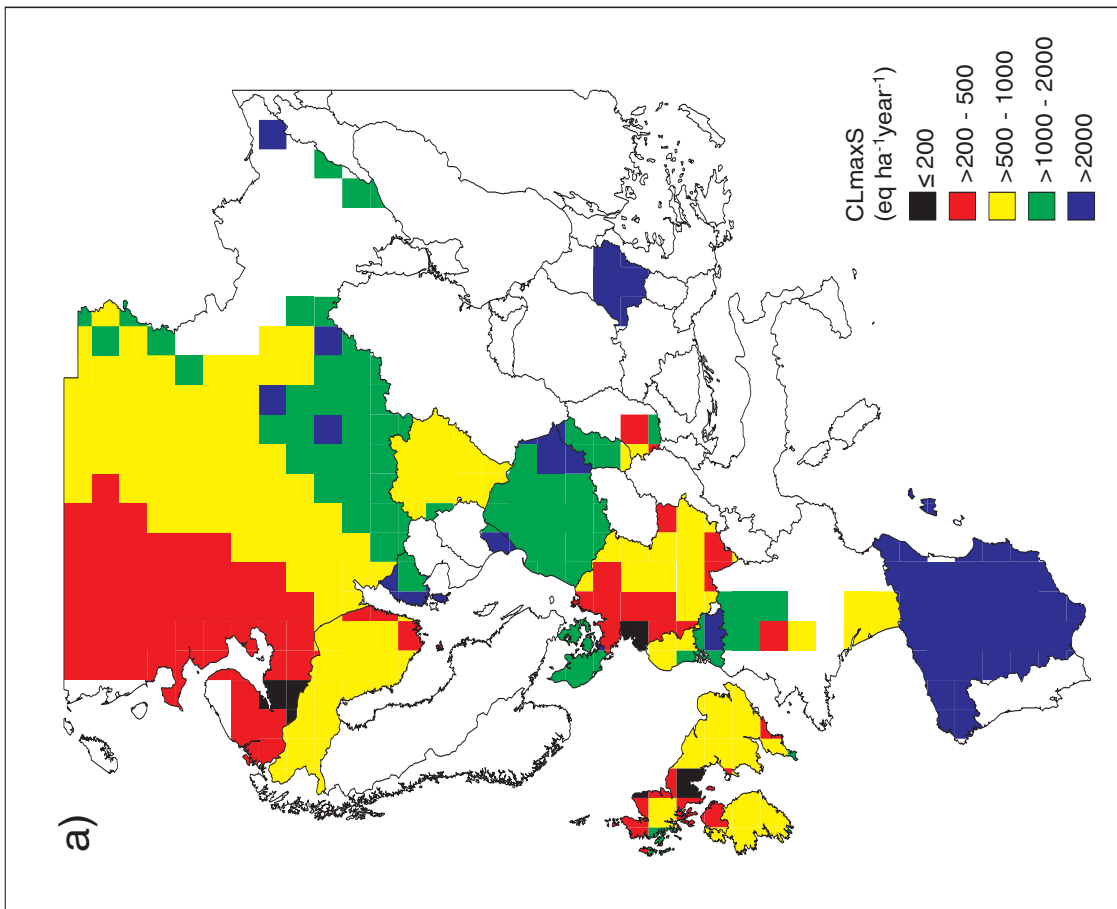
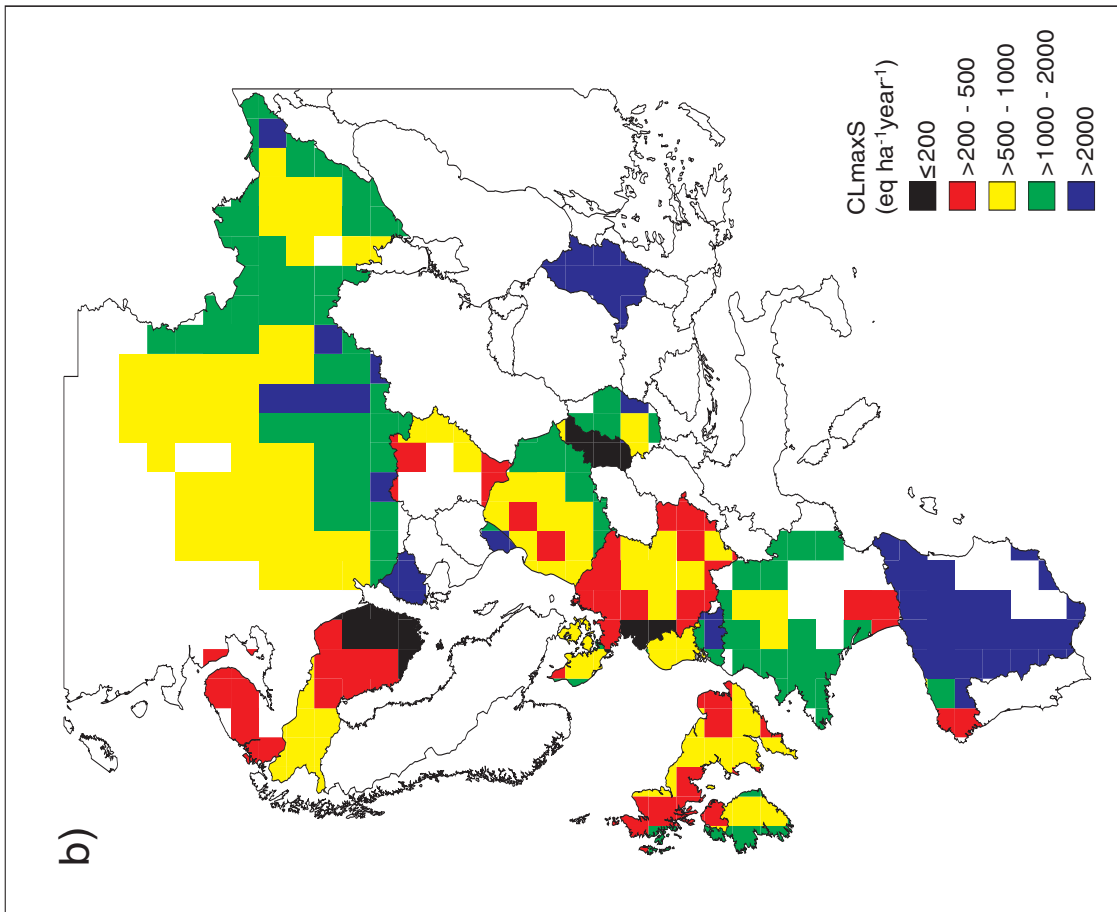


Figure 3. Five-percentile maximum critical loads for sulfur ($CL_{max}(S)$) for (a) coniferous woodland, and (b) deciduous woodland.

Table 4. Criteria used in the calculation of acidity critical loads for forest ecosystems.

Criteria used in calculation of acidity critical loads using the Simple Mass Balance (SMB) equation for:			
Country	Coniferous woodland	Deciduous woodland	Undistinguished forest
Austria			BC:Al
Belgium	Critical pH 4	Critical pH 4	
Bulgaria	Al:Ca = 1.5	Al:Ca = 1.5	
Finland	BC:Al = 1	BC:Al = 1	
Germany	BC:Al	BC:Al	
Ireland	pH 4.2	pH 4.2	
Norway			Ca:Al = 1
Poland	Ca:Al = 1	Ca:Al = 1	
Slovak Republic	BC:Al	BC:Al	
Sweden			BC:Al = 0.7-1.4
Switzerland			BC:Al = 1 and $Al_{\text{leaching}} < Al_{\text{weathering}}$
United Kingdom	Ca:Al = 1	Ca:Al = 1	

9. Maps of 5-percentile critical loads, $CL_{max}(S)$, for single ecosystems, in particular forest ecosystems, highlight some cross-border differences.
10. Ecosystem categories, critical load inputs, critical loads and exceedance data can be presented in a variety of formats that enable complex information to be visualized.

Despite some of the above remarks, scientists and policy makers should have confidence in the European critical loads data. Problem areas identified are generally limited to one or a few countries, or small areas, and with additional work could easily be resolved. The UK NFC will continue this study and examine some of the above issues further. The results will be reported to the Task Force on Mapping and NFCs.

The UK NFC would like to express its thanks to all countries who made their data available for this study and to the NFCs who responded to the questionnaire.

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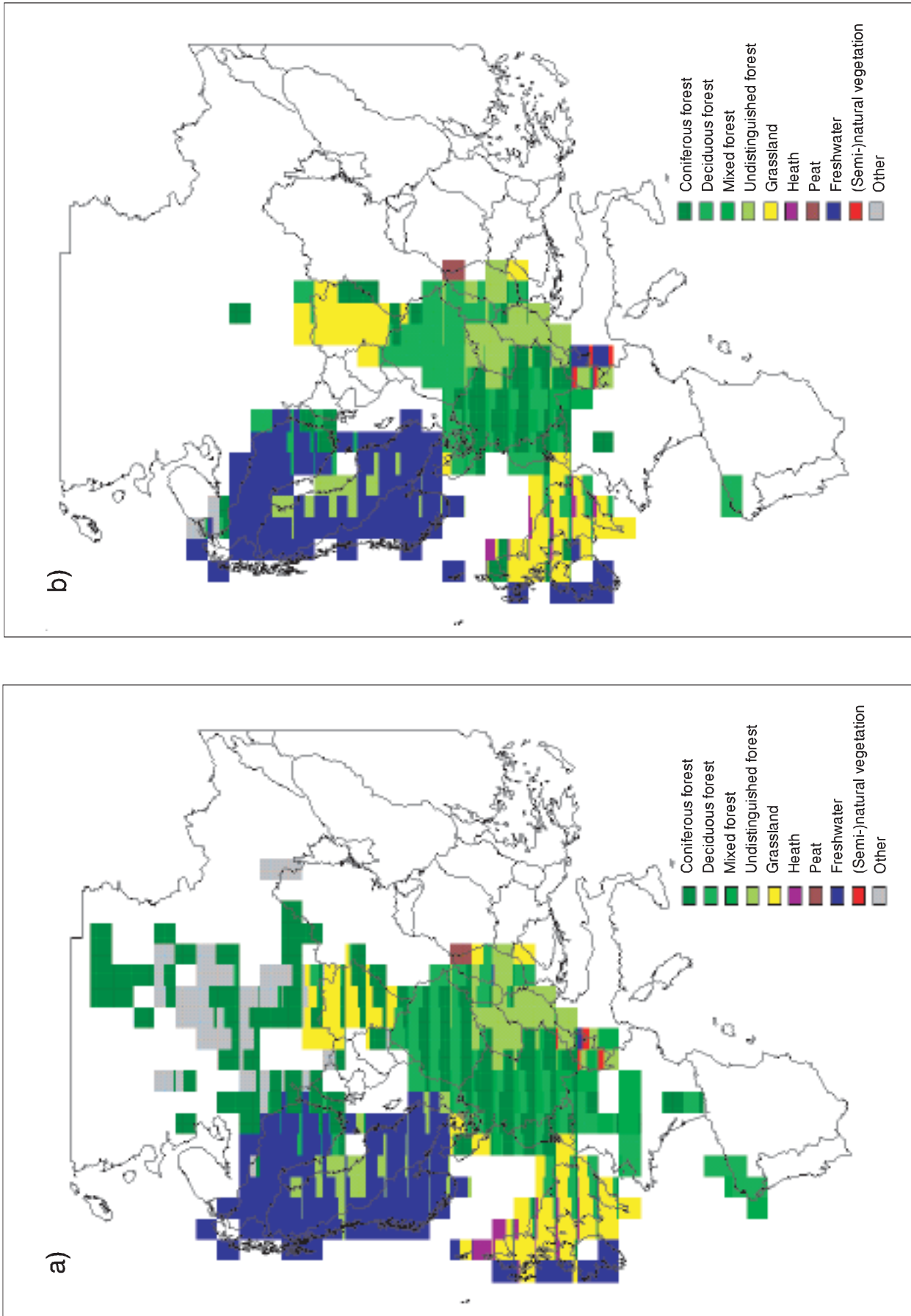


Figure 4. The relative areas of ecosystems in areas where acidity critical loads are exceeded, based on EMEP deposition for (a) 1990, and (b) a 2010 scenario.

Estimating the Exceedance of Critical Loads by Accounting for Local Variability in Deposition

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1. Introduction

When estimating the exceedance of critical loads across Europe it is important to consider not just the mean level of deposition within an EMEP grid square, as estimated by the EMEP model, but also the variability within the square. This is most easily seen when an extreme case is considered: Suppose the mean deposition is just below the lowest critical load within a square. Then the naive estimate of exceedance would be zero. In reality the true deposition is likely to be above the mean in some parts of the square, and thus there is a non-zero probability that the critical load is exceeded in some regions within the square. Therefore a better estimate of the exceedance would be greater than zero. Conversely, if the EMEP model prediction is just above the critical level, the opposite can happen.

In this article we model the deposition of sulfur and nitrogen as stochastic fields, using the deviation between measured and modeled values to estimate distribution within each square. We then use the full distribution to estimate the area of exceedance. This new estimate is considerably larger than the estimate obtained by simply using the EMEP model estimates for each square. The differences are particularly large for acidification, but also important for nutrient nitrogen.

2. Data

Since data from the monitoring stations is only available for wet deposition, we have modeled the annual total of the wet components of both N and S, and assumed that the dry components are exactly equal to the EMEP estimates. This will underestimate the true variability in total deposition, unless there is a large negative correlation between the wet and dry components. We analyzed each year from 1985 to 1995.

The full data on critical loads was not available to us. Instead we used a summary of their distribution for each EMEP 150×150 km² square in the form of critical loads corresponding to fixed percentiles of the total area. For example the deposition which would exceed the critical load over 1% of the total area is given, as it is for 10%, 20% of the area, etc. A total of 29 percentile points was

available for each square. These percentiles did not necessarily correspond to the discrete areas for which critical loads were defined, so we made an approximation as illustrated in Figure 1. The dashed line is a possible exceedance function, which gives the percentage area where the critical load is exceeded for any deposition. This must be an increasing step function, with the number of steps equal to the number of distinct critical loads in the square. The circles are an example of percentages where data are available. The true function must go through these points. We approximated the unknown function by the solid line, which takes the average value of successive known points.

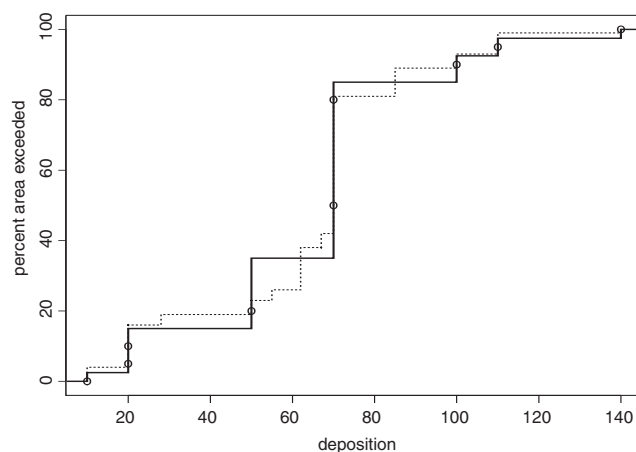


Figure 1. An example of a true critical load function (dashed line) and our approximation (solid line). The circles indicate the points on the function available to us.

3. Modeling the deposition distribution

Our deposition model follows Høst (1996) and Kåresen (1999). Full details are given in Kåresen and Hirst (1999). In outline, the method is as follows:

Let $y(\mathbf{x})$ be the true deposition of a given component at location \mathbf{x} , and let $y^{\text{emep}}(\mathbf{x})$ be the corresponding EMEP model prediction. On a log scale (cf. Section 6.1) we assume that:

$$y(\mathbf{x}) = y^{\text{emep}}(\mathbf{x}) + e(\mathbf{x})$$

where the residual field, $e(\mathbf{x})$, models short-range variability that is not captured by the EMEP model. We model $e(\mathbf{x})$ as a stochastic field that is Gaussian, stationary and has zero mean. The local variability of the field is characterized by an isotropic (circularly symmetric) covariogram (Cressie 1991):

$$c(\|\mathbf{h}\|) = \text{cov}[e(\mathbf{x}), e(\mathbf{x} + \mathbf{h})]$$

Thus $c(d)$ gives the covariance between two stations a distance d apart, and this covariance does not depend on the direction. The covariogram is assumed to follow a three-parameter exponential shape with a nugget term:

$$c(d) = \sigma^2 \exp\{-3d/\alpha\} + \tau^2 \delta(d)$$

where δ is the Dirac delta function

$$\delta(d) = \begin{cases} 1 & \text{if } d = 0 \\ 0 & \text{otherwise} \end{cases}$$

Our covariogram includes a 'nugget' variance τ^2 . This may either be due to measurement error or to fine scale variation with a correlation length negligible compared to the typical distances between the measurement stations. We have assumed the latter, and therefore include the nugget variance in the distribution of the deposition. If it is largely due to measurement error, it should be omitted. The other variance parameter, σ^2 , measures the variability of the smooth part of the field, while α measures the typical correlation range. (Stations farther apart than α have correlation less than 5% of the maximal correlation excluding the nugget.)

4. Predicting the deposition

The distribution assumptions in Section 3 together with the estimated covariogram define the probability distribution of the deposition field. Due to the Gaussian assumption, the conditional distribution of the field given the observations is also Gaussian and its moments are readily deduced. The conditional mean given the observations (the Bayes estimate, also called the kriging estimate in spatial statistics) is

$$\begin{aligned} \mathbf{y}^{\text{krig}}(\mathbf{x}) &= E[\mathbf{y}(\mathbf{x})|\mathbf{y}] \\ &= E\mathbf{y}(\mathbf{x}) + \text{cov}[\mathbf{y}(\mathbf{x}), \mathbf{y}] \mathbf{\Phi} \text{var}^{-1}[\mathbf{y}](\mathbf{y} - E\mathbf{y}) \\ &= \mathbf{y}^{\text{emep}}(\mathbf{x}) + \mathbf{v} \mathbf{\Phi}^{-1} \mathbf{e} \end{aligned}$$

where:

$$\mathbf{v}_i = \text{cov}[e(\mathbf{x}), e(\mathbf{x}_i)] = c(\|\mathbf{x} - \mathbf{x}_i\|)$$

and:

$$\Sigma_{ij} = \text{cov}[e(\mathbf{x}_i), e(\mathbf{x}_j)] = c(\|\mathbf{x}_i - \mathbf{x}_j\|)$$

The conditional variance of the field is

$$\begin{aligned} \text{var}[\mathbf{y}(\mathbf{x})|\mathbf{y}] &= \text{var}[\mathbf{y}(\mathbf{x})] - \text{cov}[\mathbf{y}(\mathbf{x}), \mathbf{y}] \mathbf{\Phi} \text{var}^{-1}[\mathbf{y}] \text{cov}[\mathbf{y}, \mathbf{y}(\mathbf{x})] \\ &= c(0) - \mathbf{v} \mathbf{\Phi}^{-1} \mathbf{v} \end{aligned}$$

All equations above are valid for an arbitrary continuous \mathbf{x} . Because we only have critical load data for the EMEP squares, we compute the kriging estimates for these squares by assuming the distribution of the deposition is constant across the square and equal to the estimate at the midpoint.

5. Estimating critical load exceedance

We estimate the exceedance in three ways:

1. The deposition is assumed to be constant and equal to the EMEP prediction over each square. This does not account for the information from the measurement stations, or the within-square variability.
2. The kriged means are used. This includes the measurement information, but neglects the within-square variability.
3. The full distribution of the deposition estimates is used.

The first method requires only the EMEP predictions and the second only the kriged means. These are then compared with the estimated exceedance functions described in section 2. The third method, however, requires the integration of the probability distribution for the deposition over the intervals defined by the discrete points on the exceedance function for each square.

The process is as follows: Let A be the true area of exceedance within a given EMEP square. Then,

$$E(A) = \sum_k E[I(y > l_k)] a_k \quad (1)$$

where l_k is the critical load for area k within the square, a_k is the area of this area and y is the deposition. I is the indicator function which takes the value 1 if the condition is satisfied, zero otherwise. For acidification the distributions of both S and N must be considered, and therefore both y and l_k are bivariate. Since we have no information on the spatial distribution of the critical loads within the square, we make the assumption that the distribution of y is identical everywhere within the square (in fact we set it equal to the distribution at the midpoint). The first method for estimating $E(A)$ simply inserts the EMEP model prediction for y in Equation 1. The second method takes account of the observations by inserting the kriging mean instead. Since the kriging mean is $E(y)$ this is equivalent to moving the expectation inside the indicator

function. Because the indicator function is strongly non-linear this is theoretically incorrect. For the third method, the expected value of A is calculated as:

$$E(A) = \sum_k \Pr(y > l_k) a_k$$

For nutrient nitrogen this can be calculated by integrating the appropriate Gaussian distributions, but this is harder for acidification since it would be necessary to integrate a bivariate distribution over an irregular area. Therefore we use conditional simulation (Ripley 1981) to generate samples from the conditional deposition field y given the observations. These samples were then used to approximate the expectation $E(A)$. This technique can also be used to calculate standard errors of the estimates. For full details see Kåresen and Hirst (1999).

6. Model discussion

6.1 Unmeasured dry deposition and log transformation

As mentioned in Section 2, the monitoring network measures only the wet deposition of N and S. The stochastic analysis above has therefore only been carried out for the wet deposition. Critical loads, however, are defined in terms of total N and S. Our estimates of the totals were obtained by adding the EMEP predicted dry deposition values to the results from the stochastic analysis of the wet deposition. This is conservative in the sense that we underestimate the total uncertainty of the deposition field. Since dry deposition accounts for approximately 40% of the total, this underestimate could be very important.

Prior to the analysis the wet deposition data were transformed to a log-scale. This transforms the positive deposition data to all of the real line and thereby avoids the possibility of negative estimates. The log transformation also appears to improve the assumptions on variance homogeneity.

6.2 Representativeness of measurements

We have made the assumption that measurement stations are both error-free and randomly located within an EMEP square, i.e. that they have not been systematically located so as to under- or overestimate local deposition. These assumptions may be invalid, but it seems reasonable to include information from monitoring stations to improve the estimation of deposition.

6.3 Covariogram shape

The exponential shape of the covariogram was chosen because it fits the empirical covariograms reasonably well. We attach no particular physical justification to this choice, and it may be of interest to examine the effect of other covariogram shapes. As is typical in spatial statistics,

however, there is not enough data in this study to reliably discriminate between different parametric shapes. We have therefore not investigated this issue further.

As stated in Section 3, the nugget term may be due to measurement error, fine-scale variation, or a combination of the two. The data used in this study allow no discrimination between the two interpretations. Note however, that the variance of the estimated deposition includes variability due to the nugget. If the nugget is interpreted as measurement error it should be omitted since we are interested in true rather than measured deposition. The nugget is not well estimated from the data used in this study since there are very few EMEP squares containing more than one measurement station. A further investigation of this issue should be based on additional information. Such information could be measurements on a fine-scale grid, repeated measurements at the same location, or quality measures for the measurement stations.

7. Results

We have calculated critical load exceedance in 5 different areas of Europe, because levels of deposition and critical loads vary widely across the continent. The five areas are shown in Figure 2. We have also calculated the values for Europe as a whole. The estimates of the percentage areas of exceedance, as calculated by the three different methods, are plotted in Figure 3 for nutrient nitrogen and Figure 4 for acidification, for the years 1985 to 1995. Maps of the estimated percentage area of exceedance for each EMEP square in 1995, for the methods using the EMEP predictions and the expected values, are shown in Figure 5 for nutrient nitrogen and Figure 6 for acidification.

It can be seen that there are some differences between the areas as estimated using the EMEP model and the kriged means, but these are not large. A more interesting comparison is between the kriged estimates and the expected exceedances. The expected exceedances, i.e. those taking variability in deposition into account, tend to be closer to 50%, which in most cases means a substantial increase over the other methods. The estimate for exceedance of acidification critical loads over the whole of Europe is about twice as large as for either of the other methods. The differences are generally smaller for nutrient nitrogen, though still important.

We should emphasize that the methodology in this paper is not designed to look for trends over the 11 years. Figures 3 and 4 are intended only to facilitate comparison of the three methods. Any apparent time trends may well be due to changes in either the monitoring network or in the EMEP model. We have not investigated either of these possibilities.

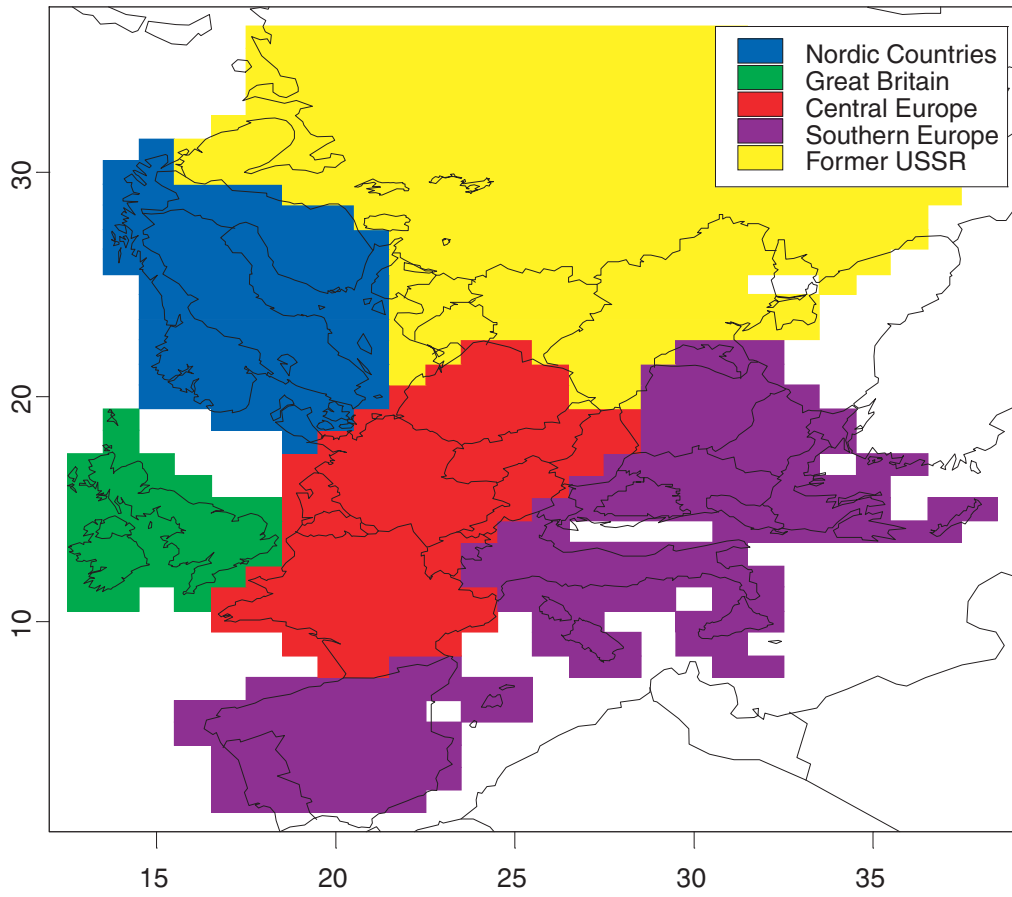


Figure 2. Subregions of Europe used in tables and graphs.

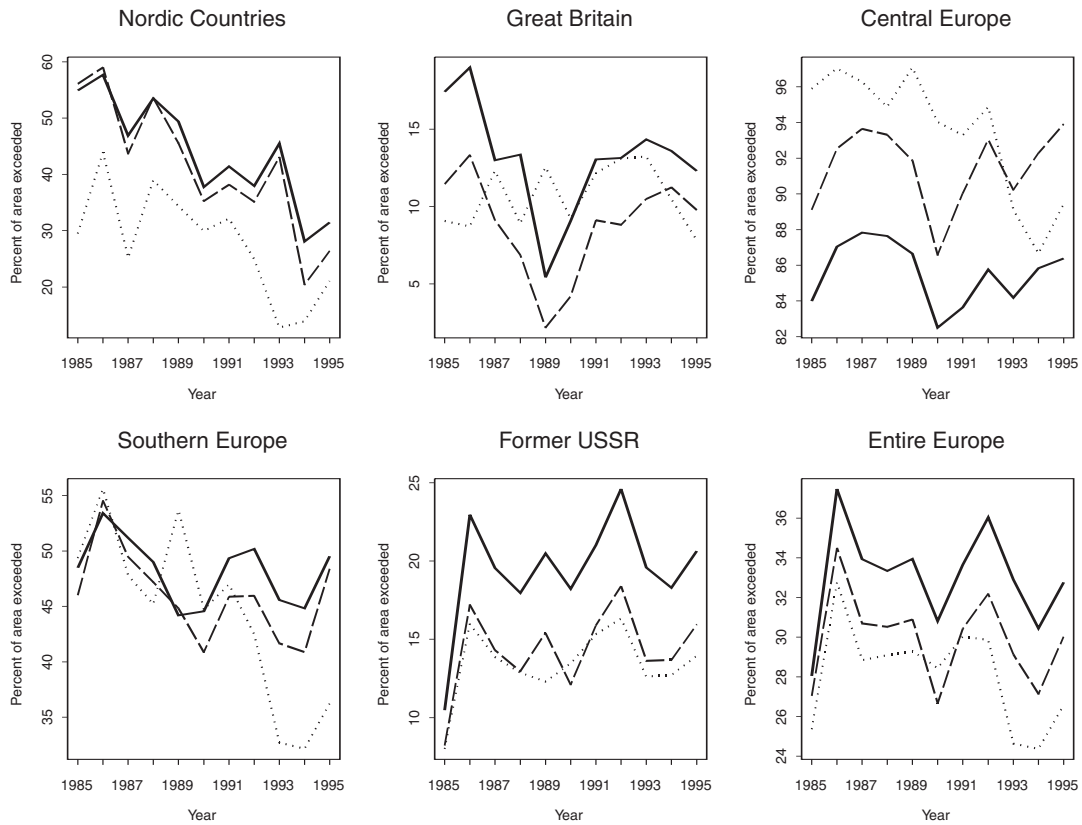


Figure 3. Estimated percentage of area where critical load of nutrient nitrogen is exceeded. Using EMEP-predicted deposition (dotted line), using kriged deposition (dashed line), and expected exceedance (solid line).

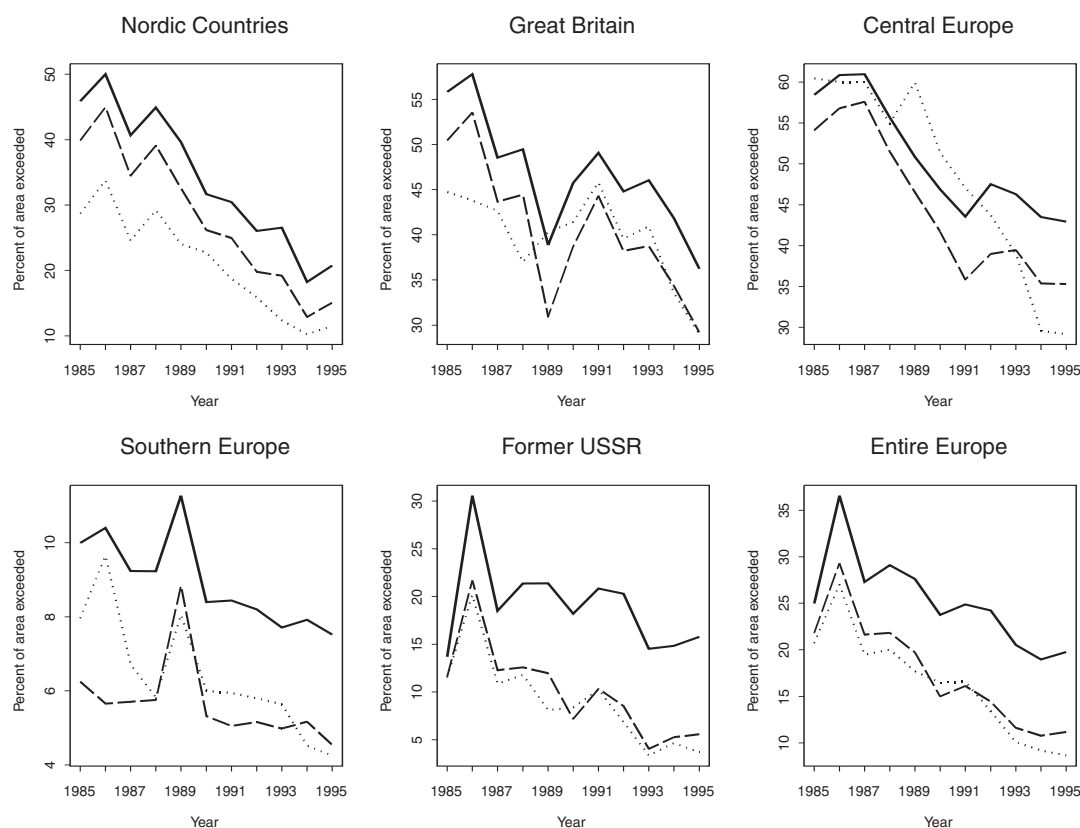


Figure 4. Estimated percentage of area where critical loads of acidification are exceeded. Using EMEP-predicted deposition (dotted line), using kriged deposition (dashed line), and expected exceedance (solid line).

8. Conclusions

Given reliable measurements of deposition from the monitoring stations, it is possible to account for the local (within EMEP square) variability in deposition when estimating critical load exceedance. The differences between the expected exceedances accounting for this variation and estimates obtained only from the kriged or EMEP means are substantial (particularly for acidification) and show how important it is to consider this variability. In general, ignoring this variability will lead to overestimation where exceedance is high, and underestimation where it is low.

Difficult problems remain in estimating the magnitude and spatial distribution of the variability of deposition, but we have developed a methodology which provides a framework for including this variability in the estimation procedure.

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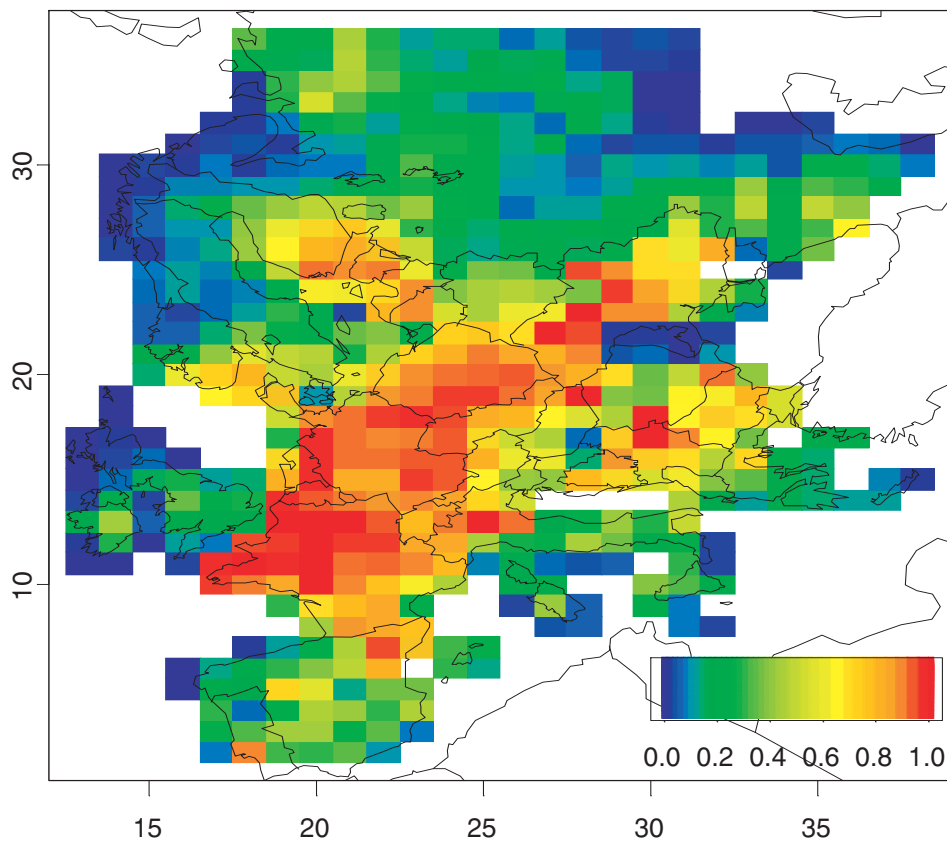
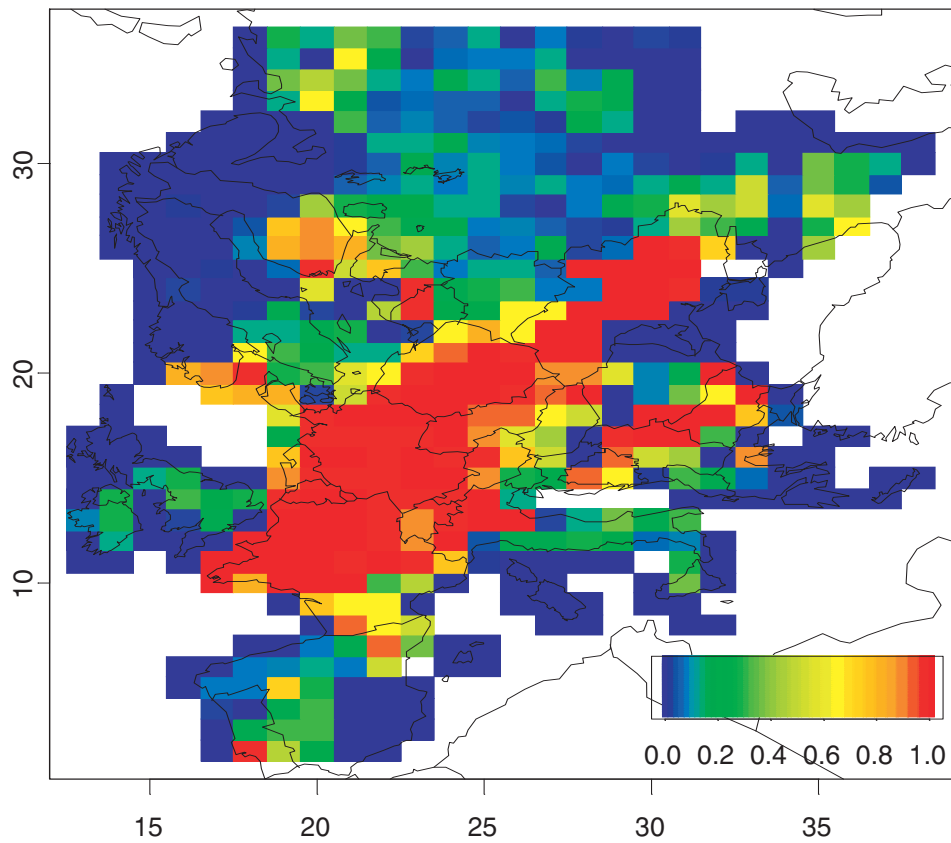


Figure 5. Estimated proportional area of exceedance of nutrient nitrogen critical loads: (1) using EMEP model predictions, and (2) expected value.

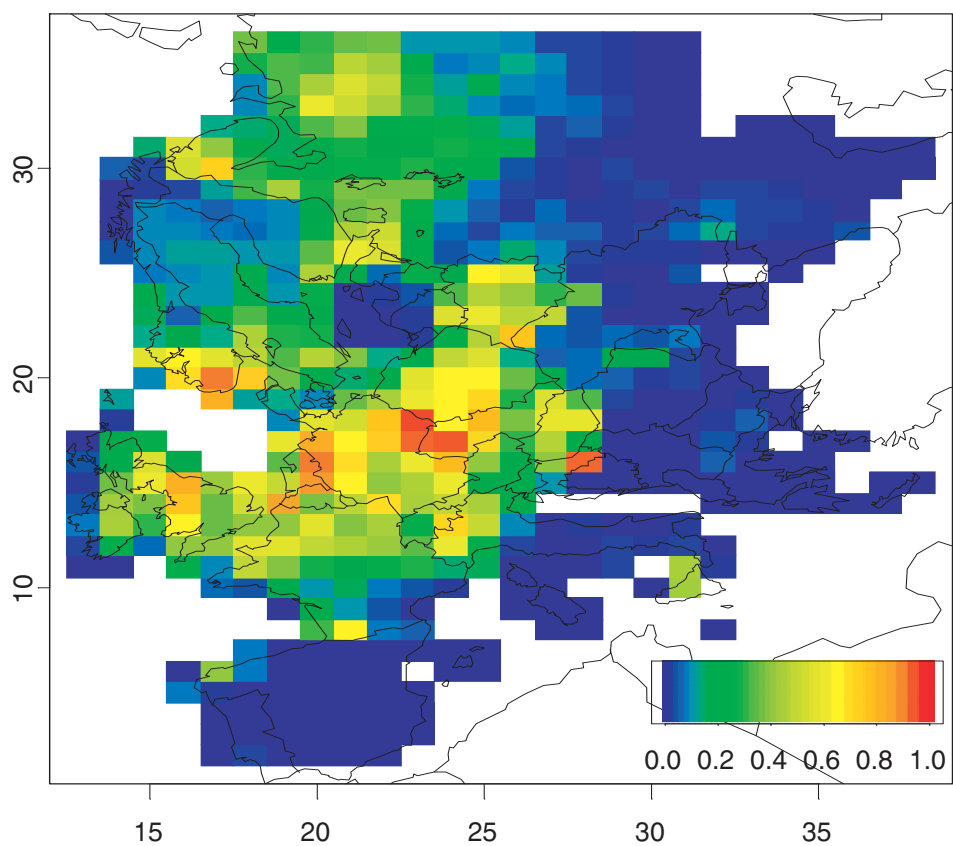
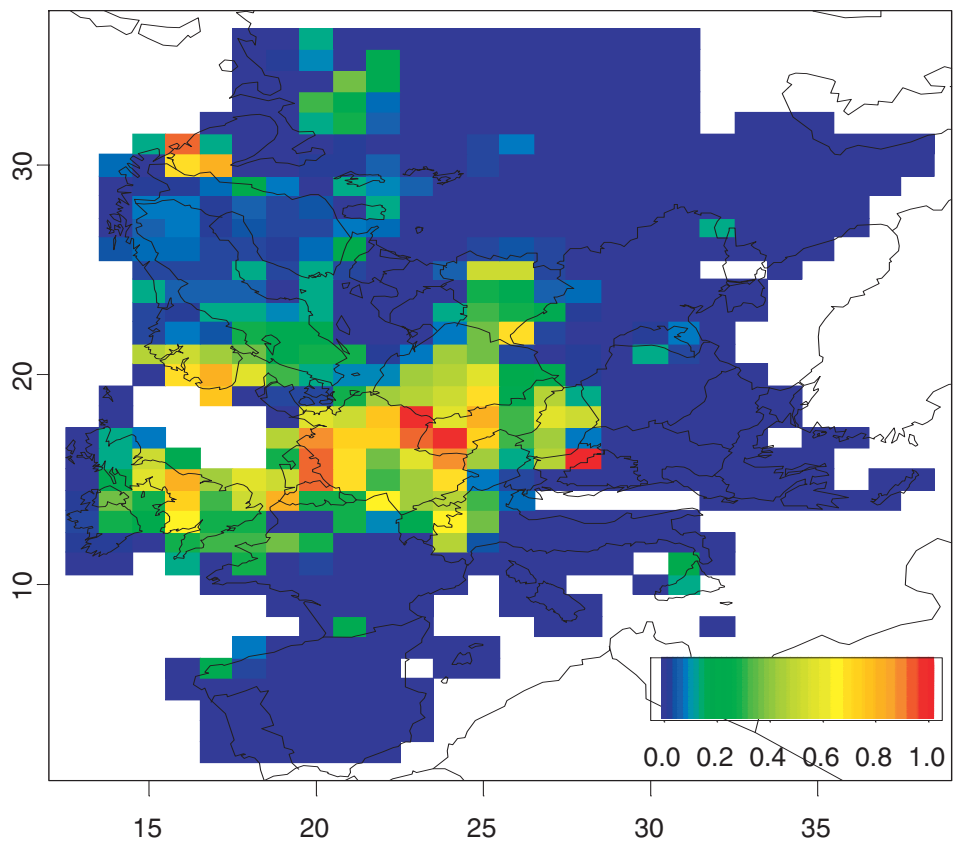


Figure 6. Estimated proportional area of exceedance of acidification critical loads: (1) using EMEP model predictions, and (2) expected value.

Part III. National Focal Center Reports

This part consists of reports on national input data and critical threshold calculations submitted to the Coordination Center for Effects (CCE) by National Focal Centers (NFCs). A total of 24 countries now collaborate in the Mapping Programme by submitting critical loads data and related information to the CCE. Countries which are reporting for the first time on their critical load calculation and mapping activities are Belarus, Bulgaria, the Republic of Moldova and Slovakia; in addition, Belgium now provides data for both Flanders and Wallonia. Nine countries (Austria, Finland, France, Germany, Ireland, Italy, Poland, Sweden and the United Kingdom) submitted updated critical load data, whereas ten countries considered their critical load data bases submitted earlier up-to-date at the last update round in spring 1998 (See also Chapter 2 in Part I).

In addition to describing the data and methods to calculate critical loads, some countries also reported on ongoing and related activities, such as ozone concentration and heavy metals mapping (several countries), mapping of corrosion rates (Germany), assessment of damage to fish populations (Norway) or results of dynamic modeling of soil acidification and recovery (Switzerland). It should be noted that the critical loads reported here are used in the negotiations of a "multi-pollutant, multi-effect" protocol and thus will have a wider impact than earlier versions.

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A. Critical Loads of Acidity

Calculation methods

The Austrian critical loads data for acidity have recently been updated. This update was necessary due to inconsistencies in the old set of data and to reflect new findings concerning the magnitude of some input parameters. The methodology used, however, has not changed and is primarily based on the recommendations in the Mapping Manual (UBA 1996).

Receptors: Forest soils, oligotrophic bogs and alpine grassland.

Method: Steady-state mass balance (SSMB, Chapter 5.3, Mapping Manual), with a grid size of 2.75×2.75 km².

Data sources

A description of the main data sources and the methodology used can be found in Posch *et al.* (1995) and in Knoflacher *et al.* (1995). More recent updates are described in Loibl and Knoflacher (1998) and concern mainly the quantities of:

Base cation uptake: These values were recalculated using information on forest growth, altitude and dominant tree

species. A quality control check was performed using independently published data.

Base cation deposition: These values have been estimated using *inter alia* measurement data from the Austrian deposition monitoring network.

Precipitation surplus: An empirical relationship was used to calculate Q :

$$Q = (P - ((12 - H \cdot 0.005) \cdot P / 100) + (420 - H \cdot 0.005)) \cdot (1 - NK \cdot a)$$

where:

P = precipitation, in mm
 H = altitude, in m
 NK = correction factor for steepness
 a = constant factor

Results

Figure AT-1 shows a map of the maximum critical loads of sulfur, $CL_{max}(S)$, and Figure AT-2 for acidifying nitrogen, $CL_{max}(N)$.

B. Critical Loads of Nutrient Nitrogen

Calculation methods

The critical loads data for nutrient nitrogen are described in Posch *et al.* (1995). The calculations were based on the steady-state mass balance for forest soils and empirical values for oligotrophic bogs and alpine heathland.

Basically, the methodology (steady-state mass balance) and default values (empirical method) described in the Mapping Manual (UBA 1996) were used. Due to previous recommendations (Posch *et al.* 1995) and in contrast to the Mapping Manual, values for N immobilization vary between 2 and 3 kg N ha⁻¹ yr⁻¹.

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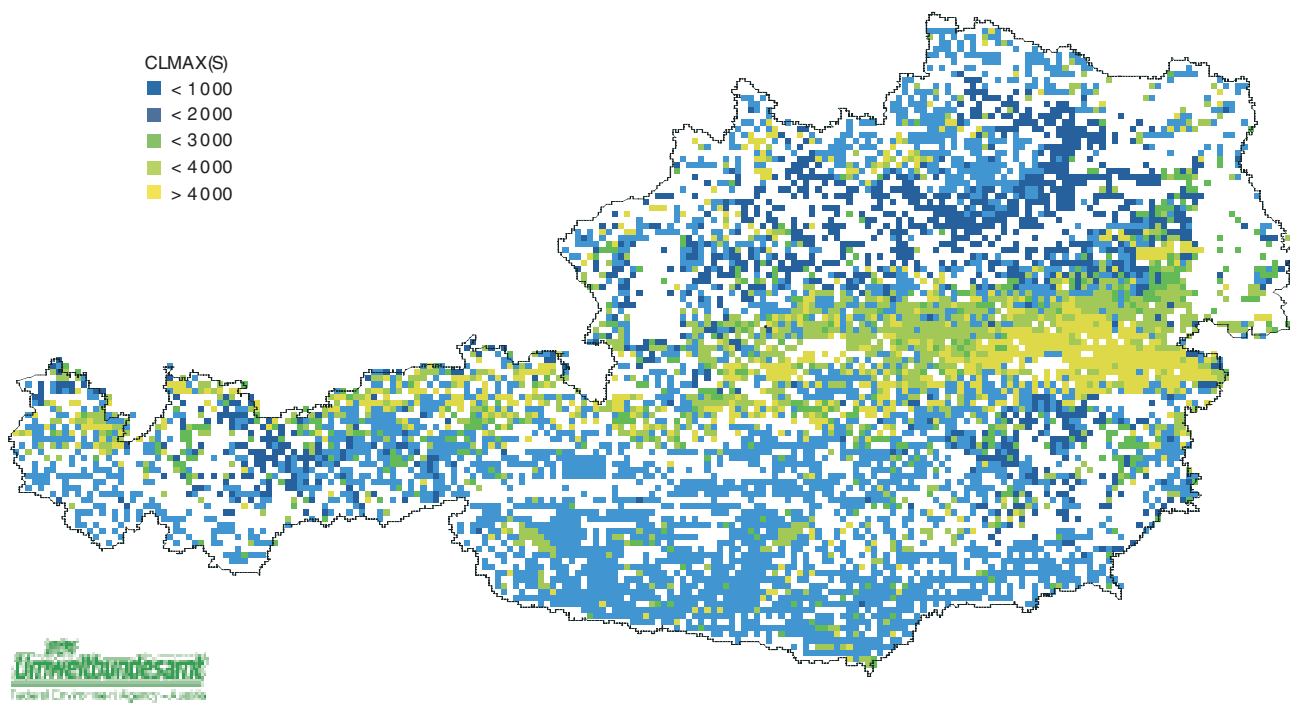


Figure AT-1. $CL_{max}(S)$ for Austria (eq ha⁻¹ yr⁻¹).

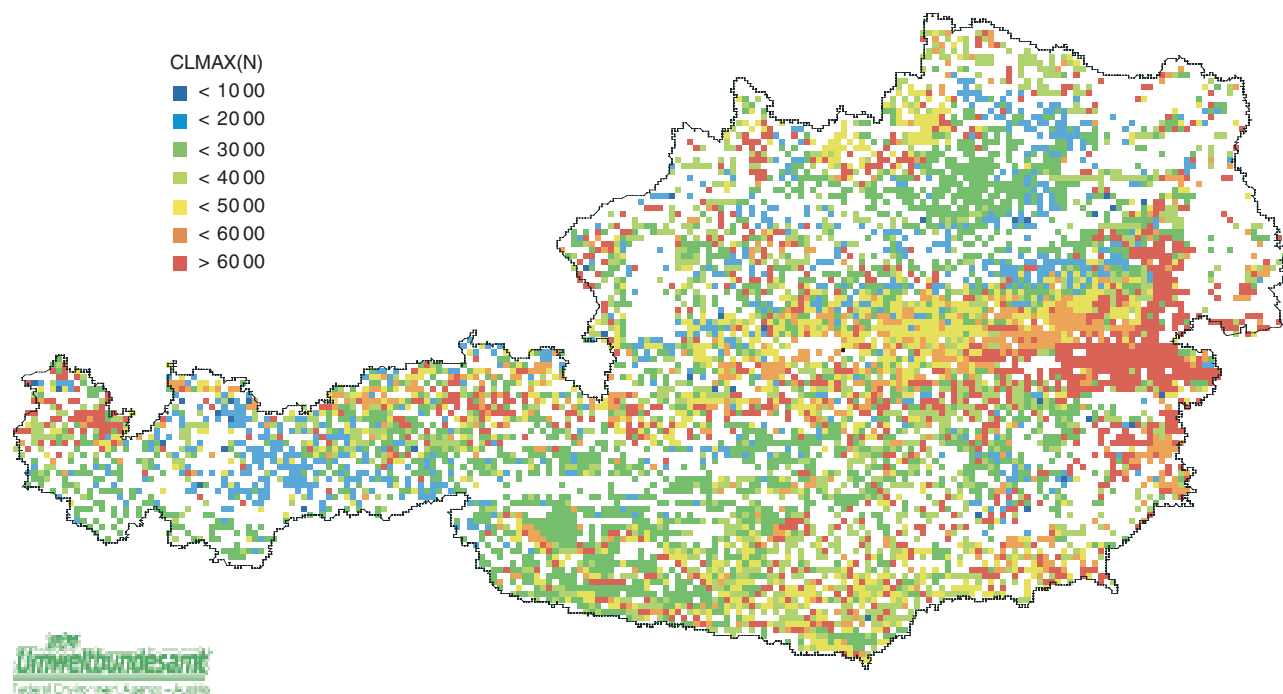


Figure AT-2. $CL_{max}(N)$ for Austria (eq ha⁻¹ yr⁻¹).

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Calculation methods

Critical loads of sulfur, acidifying and nutrient nitrogen were calculated for terrestrial ecosystems in Belarus using modified Steady-State Mass Balance (SSMB) equations. The corresponding algorithms described below were suggested by the CCE Mapping Subcentre (Bashkin 1997).

The parameters used to calculate critical loads include:

ANC_l = acid-neutralizing capacity of soil
 BC_d = base cation deposition
 BC_u = base cation uptake
 BC_w = base cation weathering
 $C:N$ = C:N ratio in the upper soil layer
 C_N = critical content of nitrogen in surface water
 C_b = coefficient of biogeochemical turnover
 C_t = active temperature coefficient (ratio of temperature sum $>5^\circ\text{C}$ to total annual sum)
 D = upper soil layer depth
 K_{gibb} = gibbsite coefficient
 $N:BC$ = ratio of N to BC in plant tissue
 N_{de} = denitrification of soil N
 N_{de}^* = denitrification of deposition N
 N_i = immobilization of soil N
 N_i^* = immobilization of deposition N
 N_l = N leaching
 NMC = nitrogen mineralization capacity of soils (eq ha⁻¹ yr⁻¹)
 N_{td} = total N deposition, wet + dry (NO_x+NH_x)
 N_u = uptake of soil N
 N_u^* = uptake of deposition N
 N_{upt} = annual N uptake
 Q = surface runoff
 W_r = chemical weathering of soil (eq ha⁻¹ yr⁻¹ at 1m depth)

A. The minimum critical load of nitrogen was calculated as:

$$CL_{min}(N) = N_i^* + N_u^*$$

where:

$$N_i^* = \begin{cases} 0.15 \cdot N_{td} \cdot C_t / C_b & \text{if } C : N > 10 \\ 0.25 \cdot N_{td} \cdot C_t / C_b & \text{if } 10 \leq C : N < 14 \\ 0.30 \cdot N_{td} \cdot C_t / C_b & \text{if } 14 \leq C : N < 20 \\ 0.35 \cdot N_{td} \cdot C_t / C_b & \text{if } C : N \geq 20 \end{cases}$$

and:

$$N_u^* = N_{upt} - N_u$$

and the annual N uptake is defined as:

$$N_{upt} = \begin{cases} K \cdot N_{upt} \cdot (1 - 1/C_b) & \text{if } C_b > 1 \\ K \cdot N_{upt} \cdot (1/C_b) & \text{if } C_b \leq 1 \end{cases}$$

The constant $K = 1.2$ for deciduous forests, and 0.8 for coniferous forests.

Uptake of nitrogen from the soil, N_u , is calculated as:

$$N_u = (NMC - N_i - N_{de}) \cdot C_t$$

where:

$$N_i = \begin{cases} 0.15 \cdot NMC / C_b & \text{if } C : N > 10 \\ 0.25 \cdot NMC / C_b & \text{if } 10 \leq C : N < 14 \\ 0.30 \cdot NMC / C_b & \text{if } 14 \leq C : N < 20 \\ 0.35 \cdot NMC / C_b & \text{if } C : N \geq 20 \end{cases}$$

and:

$$N_{de} = \begin{cases} 0.145 \cdot NMC + 0.9 & \text{if } NMC < 10 \\ 0.145 \cdot NMC + 0.605 & \text{if } 10 \leq NMC \leq 60 \\ 0.145 \cdot NMC + 6.477 & \text{if } NMC > 60 \end{cases}$$

B. The critical load of nutrient nitrogen was calculated as:

$$CL_{nut}(N) = CL_{min}(N) + N_l + N_{de}^*$$

where:

$$N_l = Q \cdot C_N$$

$$N_{de}^* = N_{td} \cdot C_t \cdot N_{de} / NMC$$

C. The maximum critical load of sulfur was calculated as:

$$CL_{max}(S) = C_t \cdot (BC_w - ANC_l) + (BC_d - BC_u)$$

where:

$$BC_w = W_r \cdot D$$

$$BC_u = N_u^* \cdot N:BC$$

$$ANC_l = Q \cdot ([H] + [Al])$$

with $[Al] = 0.2$ and $[H] = ([Al]/K_{gibb})^{1/3}$

D. The maximum critical load of nitrogen is calculated as:

$$CL_{max}(N) = CL_{max}(S) + CL_{min}(N)$$

Table BY-1 shows values for some of the parameters used in the above equations.

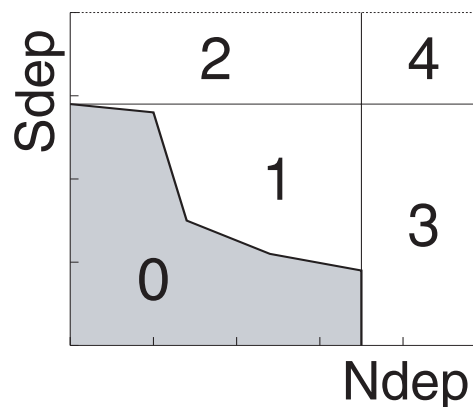


Figure BY-1. Scheme of a protection isoline (95 percentile function).

Exceedance of critical loads

The exceedances of sulfur and nitrogen critical loads were calculated using the “exceedance indifference curve” approach (Posch *et al.* 1997, UBA 1996). This approach is shown schematically in Figure BY-1. The areas in Figure BY-1 are defined in terms of emissions reductions required to achieve critical loads, as follows:

- 0 no exceedance
- 1 voluntary N or S emissions reductions
- 2 mandatory S reductions
- 3 mandatory N reductions
- 4 mandatory N and S reductions.

Data sources

Soil cover: The digitized FAO soil map, 2' × 2' scale (FAO 1989).

Land use: The LuGrid data base (de Smet and Heuvelmans 1997).

N and S deposition: EMEP MSC-W model calculations for 1992 (150×150 km²) and for 1996 (50×50 km²) (EMEP/MSC-W 1998).

Table BY-1. Parameters used to calculate sulfur and nitrogen critical loads for different soil types.

FAO classification	W_r	B	$C:N$	$N:BC$	C	N_{00}	$N:BC$	N_{50t}	K_{gibb}
Dystric Podzoluvisols	250	0.50	14.29	45	7.00	1.00	0.70	25	300
Eutric Podzoluvisols	750	0.50	17.14	55	6.00	1.00	0.80	40	200
Rendzinas	2250	0.50	20.00	90	3.00	1.00	0.80	50	100
Dystric Fluvisols	750	0.50	11.11	50	3.00	1.00	0.80	30	300
Gleyic Luvisols	2250	0.50	16.25	100	0.90	1.00	0.70	80	250
Orthic Luvisols	1750	0.50	18.33	80	2.00	1.00	0.60	45	250
Dystric Histosols	250	0.20	25.00	36	20.00	1.00	0.10	22	9.5
Gelic Histosols	250	0.55	8.75	25	7.00	1.00	0.70	20	9.5
Gleyic Podzols	250	0.50	12.86	20	7.00	1.00	0.70	15	300
Cambic Arenosols	250	0.50	8.00	18	5.00	1.00	0.70	10	1500

Results

Figure BY-2 presents the 5-percentile data for the terrestrial Belarussian ecosystems. These data were used to calculate exceedances with the 1992 and 1996 deposition scenarios.

The comparison of exceedances calculated for the 1992 and 1996 deposition scenarios shows significant changes. Almost all exceedance types (2,3,4) occurred for some terrestrial ecosystems in Belarus in 1992. Due to reductions in sulfur emissions both in Belarus and other European countries, the number of cells which would require mandatory S reductions was decreased, as was the number of cells with mandatory N reductions (Table BY-2).

Acknowledgments

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Table BY-2. Distribution of exceedance types for ecosystems in Belarus in 1992 and 1996.

Year	Number of EMEP 50×50 km ² grid cells	Class 0 No exceedances	Class 1 Voluntary N or S reductions	Class 2 Mandatory S reductions	Class 3 Mandatory N reductions	Class 4 Mandatory N and S reductions
1992	131	14	0	1	81	36
1996	131	93	0	2	36	0

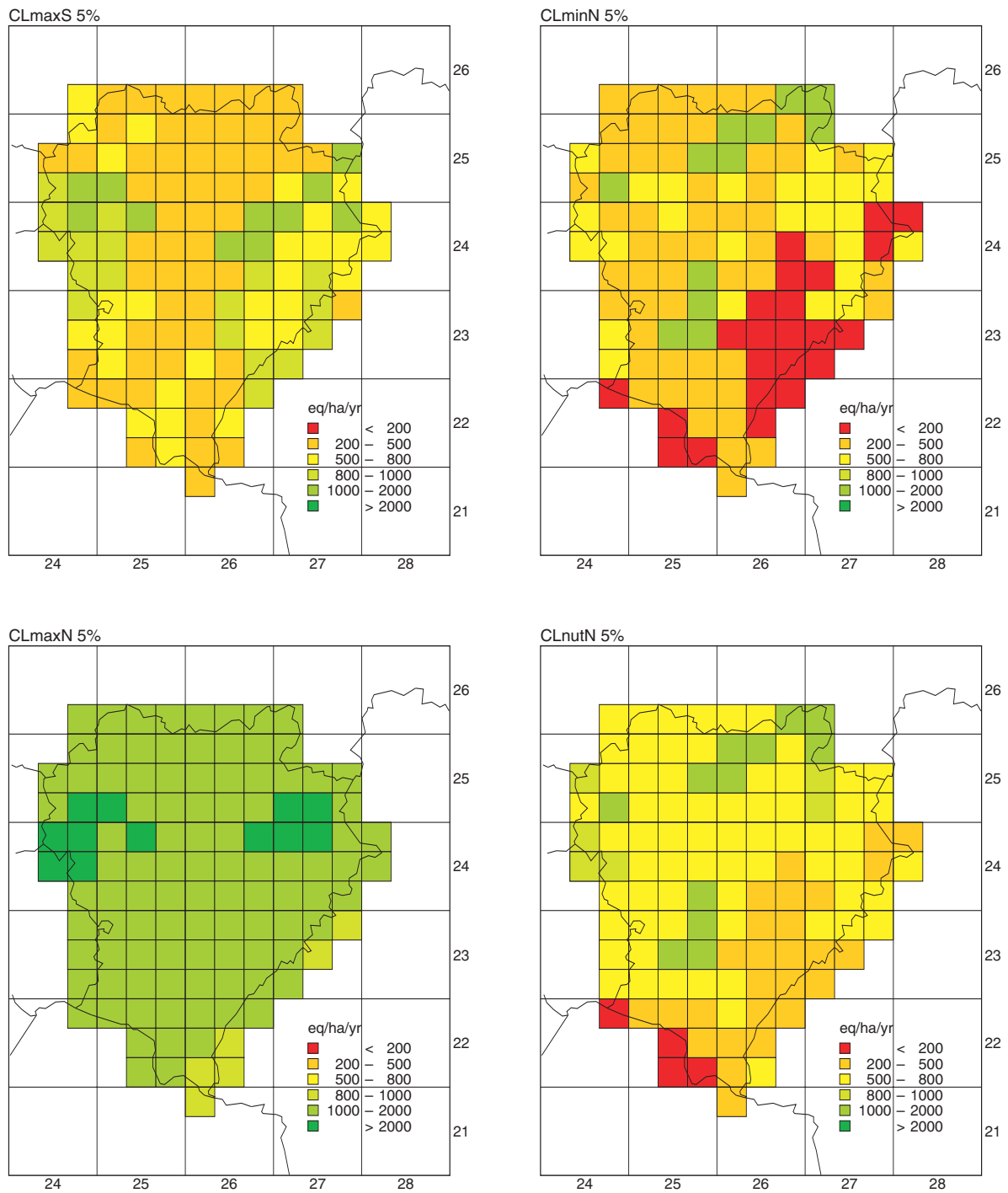


Figure BY-2. Five-percentile maps of: the maximum critical load of sulfur (CLmaxS), minimum critical load of nitrogen (CLminN), maximum critical load of nitrogen (CLmaxN) and critical load of nutrient nitrogen (CLnutN) for Belarus.

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National maps produced

To produce national maps, the contributions of Flanders (northern Belgium) and Wallonia (southern) were combined. The methodologies used to estimate various parameters differed between the two regions as a function of available data. Maps have been produced for coniferous, deciduous and mixed forests in both Wallonia and Flanders, and also for lakes in Wallonia.

Mapping procedure

Digitized maps with a total of 2532 ecosystems (652 in Flanders and 1880 in Wallonia) were overlaid with a 5×5 km² grid to produce the resulting maps. In Wallonia, the critical value given for a grid cell represents the average of the critical values weighted by their respective ecosystem area (forest or lake). As the number of forest ecosystems per grid cell was rather small in Flanders, the lowest critical load value were attributed to the entire cell.

A. Forest Soils

Calculation methods

Critical loads for forest soils were calculated according to the Steady-State Mass Balance (SSMB) method as described in UBA (1996) :

$$\begin{aligned} CL(Ac_{ac}) &= ANC_w - ANC_{le(crit)} \\ CL(Ac_{pot}) &= ANC_w - ANC_{le(crit)} - BC_u + N_i + N_{de} \\ CL_{max}(S) &= CL(Ac_{ac}) + BC_{dep} - BC_u \\ CL_{max}(N) &= N_i + N_u + CL_{max}(S) \\ CL_{nut}(N) &= N_i + N_u + N_{le} + N_{de} \\ ANC_{le(crit)} &= -PS ([Al]_{crit} + [H]_{crit}) \end{aligned}$$

Two criteria were used to determine the critical Al concentration:

The aluminum hydroxide equilibrium (K_{ox}) criterion:

To compute the Al concentration, the H concentration is fixed at 0.1 eq m⁻³, and K_{ox} varies as a function of the soil type. $K_{ox} = 9.5 \text{ m}^6 \text{ eq}^{-2}$ for peat soils (Wallonia), $100 \text{ m}^6 \text{ eq}^{-2}$ for gravelly or loamy soils (Wallonia) and $300 \text{ m}^6 \text{ eq}^{-2}$ for the other soils (Flanders/Wallonia).

The Al/Ca ratio criterion:

$$\begin{aligned} [Al]_{crit} &= (RAl/Ca \cdot BC_{u(crit)}) / PS \\ BC_{u(crit)} &= BC_{dep} + ANC_w - PS \cdot [BC]_{crit} \end{aligned}$$

Table BE-1 summarizes values used for some of the key parameters in the above equations.

Table BE-1. Constants used for critical load calculations.

Parameter	Value	Reference
N_i :		
Flanders	213 eq ha ⁻¹ yr ⁻¹	Posch <i>et al.</i> 1995
Wallonia	36 eq ha ⁻¹ yr ⁻¹	UBA 1996
N_{de} :		
Wallonia	71 eq ha ⁻¹ yr ⁻¹	UBA 1996
Flanders	a fraction of N deposition	
$[H]_{crit}$	0.1 eq m ⁻³	UBA 1996
RAl/Ca	1 eq/eq	Boxman <i>et al.</i> 1988
$[BC]_{crit}$	0.01 eq m ⁻³	

Data sources

Soils:

In *Flanders*, the Flemish soil profile inventory "Aarde-werk" was used to derive information on soil types. All profiles located in forested areas were selected as ecosystems for which critical loads were calculated.

In *Wallonia*, 47 soil types were recognized according to the Walloon map of soil associations from Maréchal and Tavernier. Each ecosystem is characterized by a soil type and a forest type.

Weathering rate: In the absence of specific data, base cation weathering rates (ANC_w) were estimated using the parent material class and the texture class for each soil according to the Mapping Manual (UBA 1996).

Precipitation surplus:

In *Flanders*, the precipitation surplus was calculated as precipitation minus the sum of interception by the forest canopy and evapotranspiration. Data on mean annual precipitation were derived from precipitation data recorded at 5 climatic stations in Flanders over a period of 10 years (1986–1995). The value for each ecosystem was set equal to the value registered in the nearest climatic station. Values for interception fractions were derived from Hootsmans and van Uffelen (1991). Mean annual evapotranspiration was fixed at 320 mm yr⁻¹ (VMM 1996).

In *Wallonia*, the precipitation surplus estimated for each ecological zone corresponds to a fraction of the normal precipitation in this zone. Annual precipitation data were derived from precipitation data registered in 24 climatic stations by the Royal Meteorological Institute. The fraction of precipitation (=0.4) was calculated by computing water balance in 5 catchments located in Wallonia.

Base cation and nitrogen uptake:

In *Wallonia*, nutrient uptakes were calculated using average growth rates based on measurements obtained from 25 Walloon ecological zones and the chemical composition of coniferous and deciduous trees (Duvigneaud *et al.* 1969, Dalhem 1997).

In *Flanders*, the same approach was followed, but in the absence of specific data, data from Dutch literature were used. Growth rates were deduced from yield tables based on soil suitability classes for tree species (de Vries 1990). Nitrogen uptake values range from 245 and 670 eq ha⁻¹ yr⁻¹, while base cation uptake values vary between 130 and 395 eq ha⁻¹ yr⁻¹ as a function of trees species and location.

Base cation deposition:

In *Flanders*, in the absence of recent data, total Ca^{2+} deposition measurements were used. They were carried out from February 1988 to February 1989 in open fields near 10 forest plots. De Vries (1994) stated that in the Netherlands total deposition of Cl^- is in equilibrium with deposition of Mg^{2+} , K^+ and Na^+ , and that BC_{dep} can be approximated by the total Ca^{2+} deposition. For each ecosystem, the value of the nearest plot was taken.

In *Wallonia*, actual throughfall data collected at 5 sites between 1992 and 1996 were used to estimate BC_{dep} . The seasalt contribution to Ca^{2+} , Mg^{2+} and K^+ deposition was estimated using sodium deposition according to the method described in UBA (1996). The mean BC_{dep} data of the 5 sites were extrapolated to all Walloon ecosystems as a function of location and tree species.

Results

The highest $CL(A)$ values were found in calcareous soils under deciduous or coniferous forests. The estimated base cation release rate from mineral weathering processes is high in these areas, and thus provides a high long-term buffering capacity against soil acidification.

In *Flanders*, the lowest critical loads occur in the Campine and the north of Limburg where the ecosystems consist largely of very sensitive coniferous forests on poor sandy soils (VMM 1996).

In *Wallonia*, the more sensitive ecosystems are located in Ardennes and High Ardennes, where coniferous forests grow on sandy-loam or loamy gravelly soils (Siterem 1998).

B. Lakes

Calculation methods

1. The Steady-State Water Chemistry (SSWC) method and the empirical diatom method were used to estimate critical loads of acidity as described in the Mapping Manual (UBA 1996):

$$CL(Ac) = ([\text{BC}]_0 - [\text{ANCl}]_{lim}) Q - BC_{dep} - BC_u$$

$$CL_{(emp)} = [\text{Ca}^{2+}]_0 / 89$$

2. The First-order Acidity Balance (FAB) model was used to estimate critical loads of sulfur and nitrogen (UBA 1996).

Data sources

- The desired ANC threshold which ensures no damage to biological indicators has been established as $20 \mu\text{eq l}^{-1}$ for all lakes, according to UBA (1996).
- The values for runoff (Q) correspond to the flow in the river supplying the lake, and were derived from 1996 monitoring data.
- The base cation and nitrogen uptake by forest ecosystems was estimated using data collected in forest catchments near the lakes.
- Base cation deposition was calculated using 1996 data from the wet deposition monitoring network. Corrections for seasalt have been made using sodium concentrations according to methods described in the Mapping Manual (UBA 1996).
- The original freshwater calcium concentration was calculated using monitoring data collected in 1996, and relationships established in the French Ardennes (Fevrier 1996).
- As site-specific data were unavailable, values for the N_i , s_N , and s_S parameters needed by the FAB model were taken from UBA (1996).

Results

Results for forested ecosystems in Flanders are shown in Figures BE-1 and BE-2. Figures BE-3 through BE-5 show the resulting maps for forests and lakes in Wallonia.

The Gileppe and Eupen lakes are located in the High Ardennes. The catchments of these two lakes are 73% and 79% forested, respectively, while the rest of the area is covered primarily by fens. The critical loads are lower for the Eupen lake as a result of a naturally acidic water leached from the catchment.

The relatively high value of $CL_{nut}(N)$ is mainly due to the chosen value of acceptable N leaching (set equal to $25 \text{ g NO}_3 \text{ m}^{-3}$ according to the European Surface Water Directive 75/440/EEC).

Table BE-2. Critical loads of acidity, sulfur and nitrogen estimated from SSWC, FAB and empirical methods.

Parameters	Values for lakes ($\text{eq ha}^{-1} \text{ yr}^{-1}$)	
	Gileppe	Eupen
$CL(Ac)$ SSWC	1015	598
$CL(Ac)$ empirical	210	-60
$CL_{max}(S)$ FAB	420	20
$CL_{max}(N)$ FAB	1480	990
$CL_{nut}(N)$ FAB	3310	2950

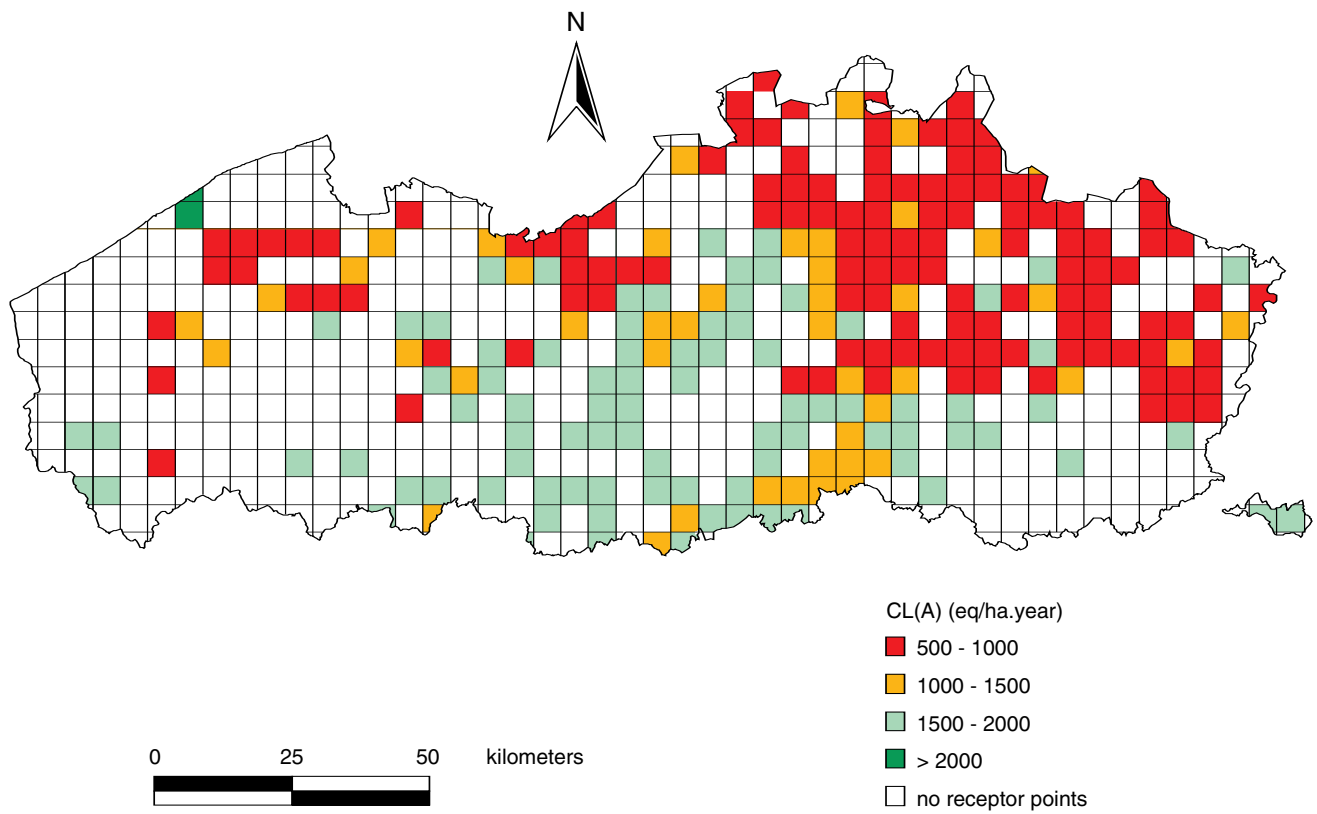


Figure BE-1. Critical loads of actual acidity, $CL(A)$, for forest ecosystems in Flanders.

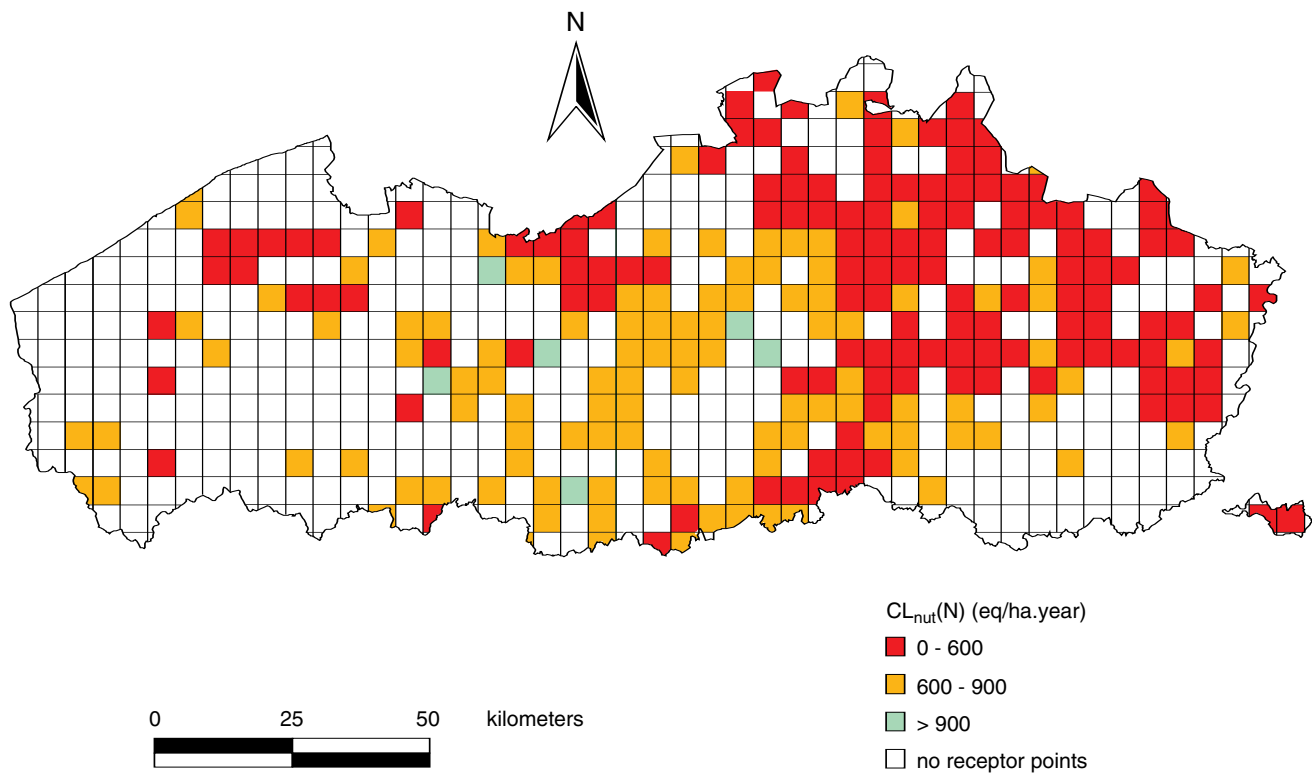


Figure BE-2. Critical loads of nutrient nitrogen, $CL_{nut}(N)$, for forest ecosystems in Flanders.

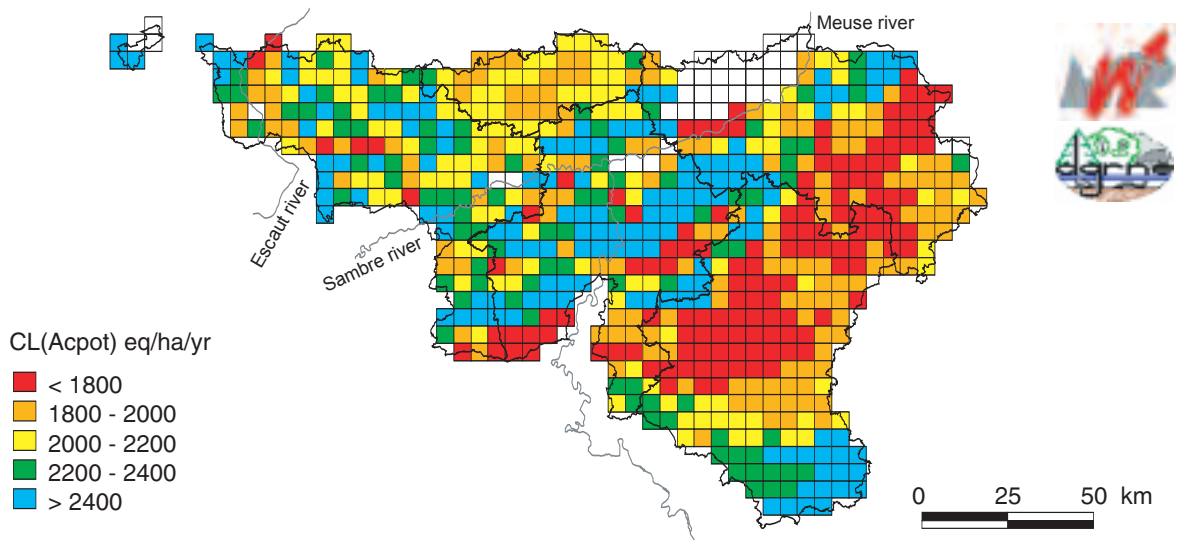


Figure BE-3. Critical loads of potential acidity, $CL(A_{cpot})$, for forest soils and lakes in Wallonia.

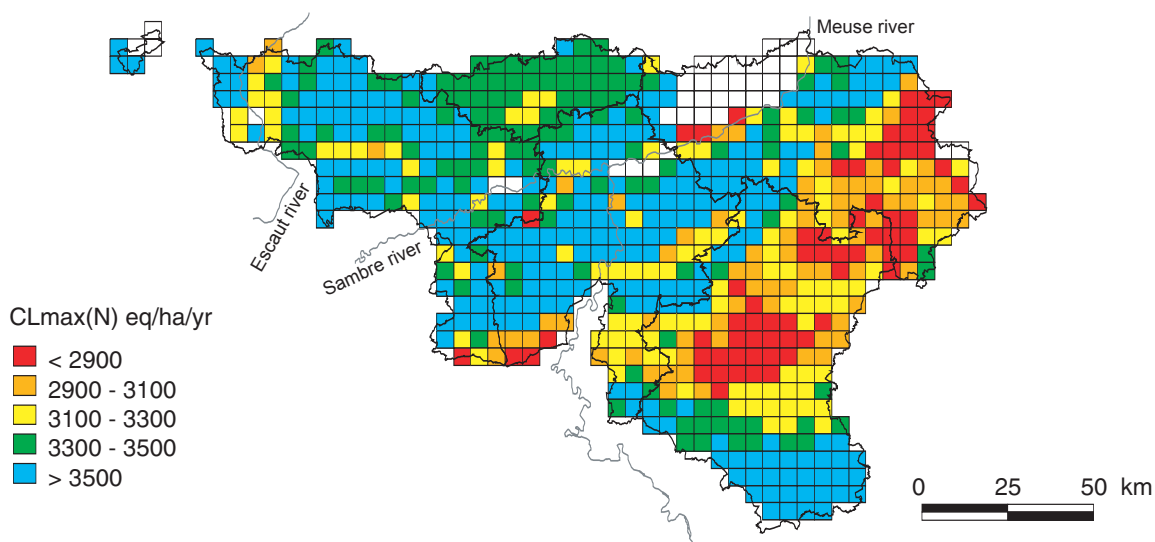


Figure BE-4. Critical loads of nitrogen, $CL_{max}(N)$, for forest soils and lakes in Wallonia.

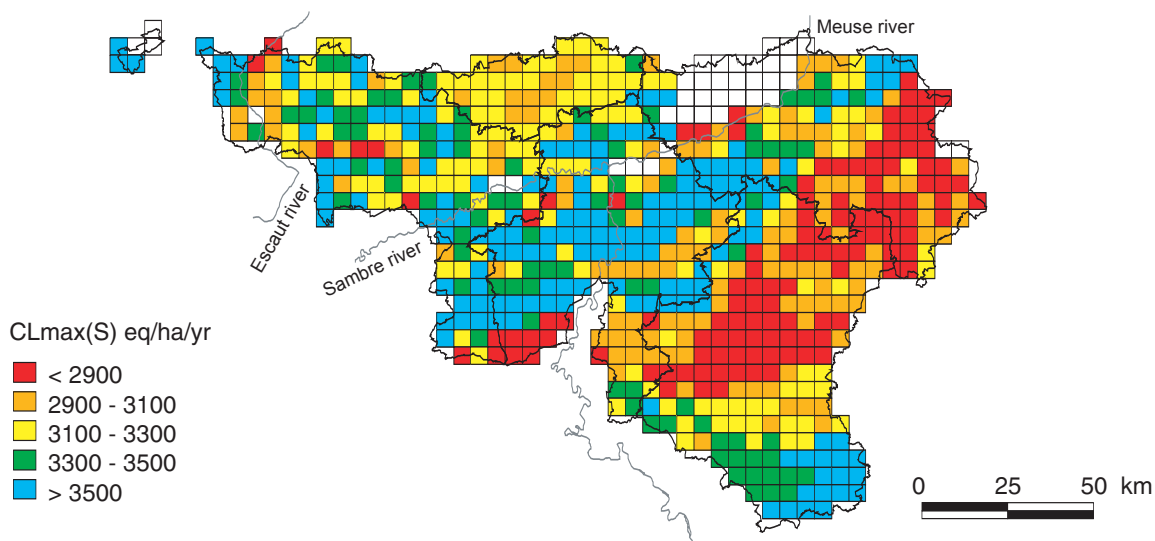


Figure BE-5. Critical loads of sulfur, $CL_{max}(S)$, for forest soils and lakes in Wallonia.

In comparison to soils, both lakes are more sensitive than the forests located in their respective catchments. Over the long term, continued acidic deposition could modify the equilibrium of both oligotrophic lakes (Siterem 1998).

Conclusions

The value of some parameters could vary significantly according to the methods followed by the Flemish and Walloon regions. Moreover, different data selection methods dictated by different quality objectives can introduce additional discrepancies. For the Flemish region, the data sets presently available are too limited to enable the accurate determination of specific critical values. Moreover, as forested areas are relatively few and strongly fragmented, the use of the SSMB method presents difficulties. In Wallonia, monitoring of forests is more intensive due to their economic importance, and variability of the soil types can be adequately addressed.

The environmental conditions in the two regions are quite different with respect to soils and land cover. The computation methods used for both regions rely on the available data sets and are adapted to the prevailing conditions.

Juxtaposition of the differing critical loads calculation methods used shows that the calculated values provide a good initial indication of the spatial variability of the sensitivity of forest or freshwater ecosystems to acidification and eutrophication in Belgium.

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National maps produced

The mapping of critical loads of acidity, sulfur and nitrogen is based on 208 forest soil receptor points, covering both coniferous and deciduous forests. The results are processed for a 45×45 km² subgrid of the EMEP grid net. The evaluation of critical loads involves the following maps:

- Deposition of sulfur, nitrogen and base cations
- Base cation weathering

- Nitrogen and base cation uptake by biomass
- Critical loads of acidity, sulfur and nitrogen for forest soils
- Minimum and maximum critical loads of sulfur for forest soils
- Minimum and maximum critical loads of nitrogen for forest soils
- Critical loads of nutrient nitrogen for forest soils
- Exceedance of critical loads of acidity
- Exceedance of critical loads of sulfur
- Exceedance of critical loads of nitrogen

Calculation methods

1. Critical loads of acidity for forest soils:

The steady-state mass balance model has been applied to forest soils (Hettelingh and de Vries 1992, Downing *et al.* 1993, Party *et al.* 1994, Sverdrup and de Vries 1994, UBA 1996, Posch *et al.* 1997). Critical loads of acidity have been calculated according to the following equations:

$$CL(A) = BC_w + Q \cdot [H]_{crit} + RAl/Ca (BC_{dep} + BC_w - BC_u) \\ = 2.5 BC_w + 0.09 Q + 1.5 BC_{dep} - 1.5 BC_u$$

where:

$CL(A)$ = critical load of acidity

BC_w = weathering of base cations

Q = annual runoff of water under root zone, m³ ha⁻¹ yr⁻¹

$[H^+]_{crit}$ = critical concentration of protons (= 0.09 eq m⁻³ which corresponds to pH 4.0) from Hettelingh and de Vries (1992).

RAl/Ca = critical Al/Ca ratio (= 1.5 eq eq⁻¹) (Rihm 1994)

BC_{dep} = atmospheric deposition of basic cations, eq ha⁻¹ yr⁻¹

BC_u = net growth uptake of basic cations, eq ha⁻¹ yr⁻¹

2. Maximum and minimum critical loads of sulfur and nitrogen:

$$CL_{max}(S) = CL(A) + BC_{dep} - BC_u$$

$$CL_{min}(N) = N_u + N_i$$

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S)$$

where:

N_u = net growth uptake of nitrogen

N_i = nitrogen immobilization

For podsols and histosols, $N_i = 3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (214 eq ha⁻¹ yr⁻¹) and 2 kg ha⁻¹ yr⁻¹ (143 eq ha⁻¹ yr⁻¹) for other soils (UBA 1996).

3. Present loads of sulfur:

$$PL(S) = PL(S-SO_2) + PL(S-SO_4^{2-})$$

where:

$S-SO_2$ = dry sulfur deposition, eq ha⁻¹ yr⁻¹

$S-SO_4^{2-}$ = wet sulfur deposition, eq ha⁻¹ yr⁻¹

4. Present loads of nitrogen:

$$PL(N) = PL(N-NO_2 + N-NH_3) + PL(N-NO_3^- + N-NH_4^+)$$

where:

$N-NO_2$, $N-NH_3^-$ = dry nitrogen deposition, eq ha⁻¹ yr⁻¹

$N-NO_3^-$, $N-NH_4^+$ = wet nitrogen deposition, eq ha⁻¹ yr⁻¹

5. Exceedance of critical load:

$$Ex(A) = PL(S) + PL(N) - BC_{dep} - BC_u - N_u - CL(A)$$

$$Ex(S) = PL(S) - S_f (BC_u - BC_{dep}) - CL(S)$$

$$Ex(N) = PL(N) + (1 - S_f) (BC_u - BC_{dep}) - CL(N)$$

where:

$$S_f = PL(S) / (PL(S) + PL(N) - N_u - N_i)$$

The exceedance maps for nitrogen and sulfur are based upon critical load calculations and average annual atmospheric deposition from the period of 1981–1996.

6. Critical load of nutrient nitrogen:

$$CL_{nut}(N) = N_u + N_i + N_{le(crit)}$$

$$N_{le(crit)} = Q \cdot [N]_{crit}$$

where:

$N_{le(crit)}$ = leaching of nitrogen at critical load, eq ha⁻¹ yr⁻¹

$[N]_{crit}$ = concentration of nitrogen in the soil solution at critical load (for coniferous = 0.0143 eq m⁻³, for deciduous = 0.0215 eq m⁻³) from Posch *et al.* (1995).

7. Critical leaching of alkalinity:

$$ANC_{le(crit)} = Al_{le(crit)} + H_{le(crit)}$$

$$Al_{le(crit)} = R Al/BC (BC_{dep} + BC_w - BC_u)$$

$$H_{le(crit)} = Q \cdot [H]_{crit}$$

where:

$ANC_{le(crit)}$ = critical leaching of alkalinity, eq ha⁻¹ yr⁻¹

$Al_{le(crit)}$ = Al_3^+ critical leaching, eq ha⁻¹ yr⁻¹

$H_{le(crit)}$ = H^+ critical leaching, eq ha⁻¹ yr⁻¹

Data sources

National monitoring data:

- Critical loads have been calculated for all major tree species in grid cells of 16×16 km². A total of 208 forest soil profiles have measured values.

- Runoff of water under the root zone has been measured in grid cells of 10×10 km² for the entire country.
- A network of 38 measurement stations of atmospheric deposition by precipitation and 107 measurement points of air pollutants concentrations have been used for base cations, sulfur and nitrogen deposition (Ignatova 1994, 1995)
- Nitrogen and base cations net uptake rates are obtained by multiplying the element contents of the stems (N, Ca, K, Mg and Na) with annual harvesting rates (Ignatova *et al.* 1997).

National maps:

- Soil type information on the FAO soil map of Bulgaria
- Geological map of Bulgaria 1:500,000
- Vegetation map of Bulgaria 1:500,000

Calculated data:

In the absence of more specific data on the production of basic cations through mineral weathering for most of study regions, weathering rates were calculated according to the dominant parent material obtained from the lithology map of Bulgaria and the texture class taken from the FAO soil map for Europe, according to the clay contents of Bulgarian forest soils (UBA 1996).

In contrast to the previous evaluation of critical loads, where the weathering rates vary between 250–1000 eq ha⁻¹ yr⁻¹ (Rihm 1994), these calculations are based on the assessment of weathering rates between 250–2750 eq ha⁻¹ yr⁻¹ derived from soil types and texture classes according to the clay content in Bulgarian soils (UBA 1996).

Results, comments and conclusions

All the data necessary to evaluate critical loads have been prepared in Excel tables and mapped for the EMEP 50×50km² grid network. Values for each parameter and the resulting critical loads are stored for each forest type (coniferous and deciduous forests) in separate records, and averaged for each EMEP 50×50km² grid cell when the forest is a mixture of both tree types, in accordance to the area fractions of the tree species.

Calculation of critical loads of acidity resulted in the highest values for calcareous soils under deciduous and coniferous forests. The lowest critical loads occur on poor sandy soils. Base cation deposition and estimated base cation release rate from mineral weathering processes are high in the study areas, and thus provides a high long-term buffering capacity against soil acidification.

Calculated values for $CL_{max}(S)$ range between 2101 and 7331 eq ha⁻¹ yr⁻¹ for coniferous, and between 2101 and 8986 eq ha⁻¹ yr⁻¹ for deciduous forests. The values for $CL_{max}(N)$ are similar (between 2698 and 7975 eq ha⁻¹ yr⁻¹ for coniferous forests, and between 2698 and 9442 eq ha⁻¹ yr⁻¹ for deciduous ones). On the contrary, critical load values for nutrient nitrogen, $CL_{nut}(N)$ are lower and range between 383 and 876 eq ha⁻¹ yr⁻¹ for coniferous, and between 170 and 876 eq ha⁻¹ yr⁻¹ for deciduous forests. The lowest critical loads are calculated for $CL_{min}(N)$ (between 376 and 862 eq ha⁻¹ yr⁻¹ for coniferous, and between 165 and 862 eq ha⁻¹ yr⁻¹ for deciduous forests).

Figure BG-1 shows the average values for $CL_{max}(S)$, $CL_{max}(N)$, $CL_{min}(N)$ and $CL_{nut}(N)$ for each EMEP 50×50 km² grid cell, and the frequency distribution of the values is shown in Table BG-1.

The mapped forest ecosystems involve 55 EMEP 50×50 km² grid cells, of which 26 cells are covered by deciduous forests only. In the other 29 cells, the forest is a composite of both deciduous and coniferous tree types.

It can be concluded that the calculated values for acidity, sulfur and nitrogen give a good initial indication of the spatial variability of ecosystem sensitivity to acidification in Bulgaria. In addition, a manual for calculating and mapping critical loads of acidity, sulfur and nitrogen was published (in Bulgarian) to facilitate further assessment and mapping activities (Ignatova *et al.* 1998).

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Table BG-1. Distribution of critical load values in Bulgaria (in percent).

Range (eq ha ⁻¹ yr ⁻¹)	$CL(A)$	$CL_{max}(S)$	$CL_{min}(N)$	$CL_{max}(N)$	$CL_{nut}(N)$
< 200	0	0	0	0	0
200–500	0	0	52.73	0	45.45
500–1000	0	0	43.64	0	50.91
1000–2000	0	0	3.63	0	3.64
> 2000	100	100	0	100	0

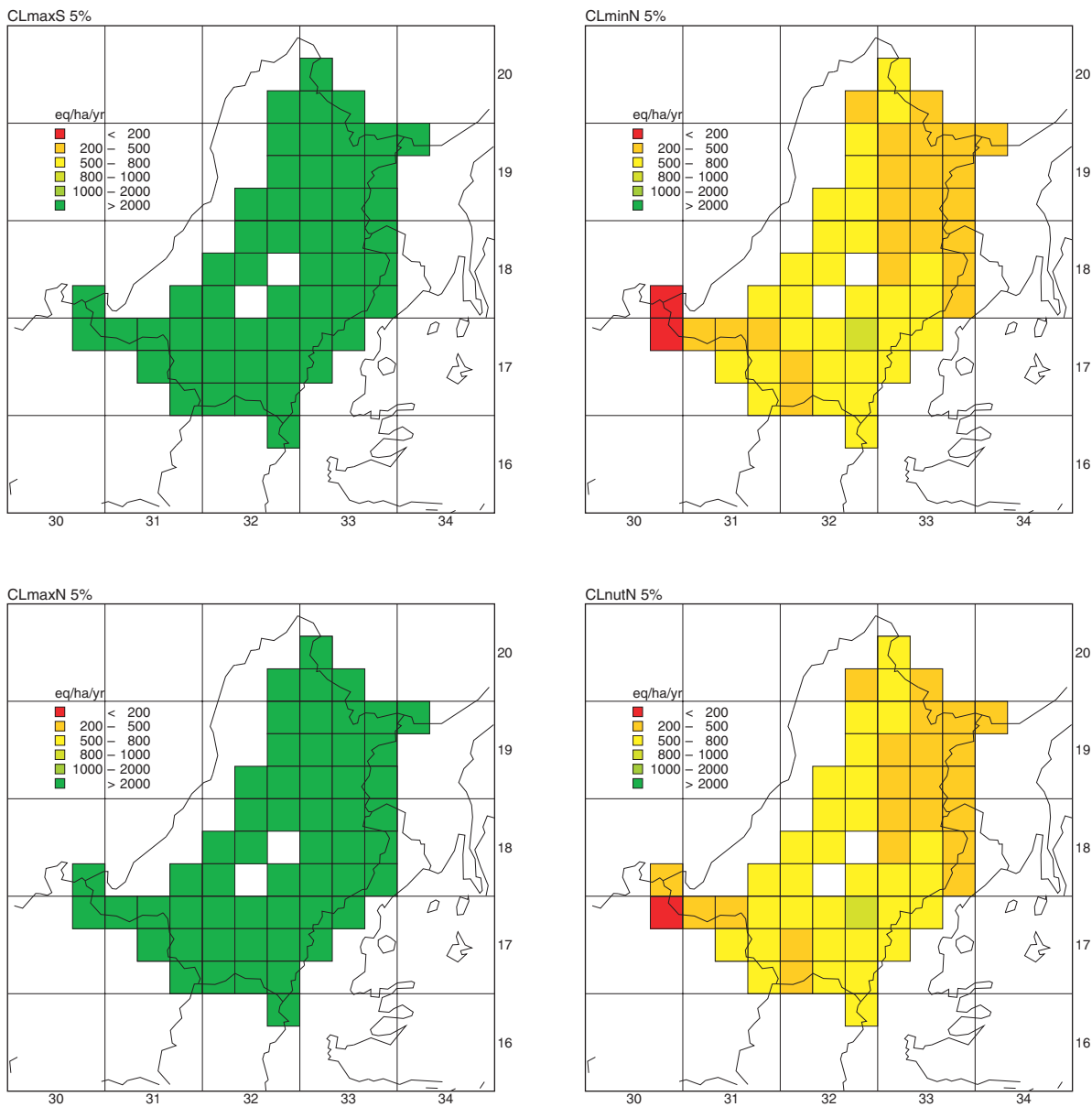


Figure BG-1. Five-percentile maps of: the maximum critical load of sulfur (CLmaxS), minimum critical load of nitrogen (CLminN), maximum critical load of nitrogen (CLmaxN) and critical load of nutrient nitrogen (CLnutN) for Bulgaria.

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National maps produced

Computation and mapping of critical loads have been started in the Republic of Croatia with two 50×50 km² EMEP grid cells (79,43 and 80,43) located within the 150×150 km² grid cell (27,15). The most complex soil-

vegetation relationships in Croatia can be found in this coastal mountain region (Gorski Kotar), which also has the most valuable coniferous forests.

In the second phase, two more 50×50 km² grid cells (80,45) and (80,46) have been mapped, in an inland region (the northwest part of Croatia) dominated by deciduous forests (beech and oak). Note that these 2 grid cells are not included in the CCE critical load data base. Combinations of soil-forest types for both regions were defined as square polygons. The four EMEP grid cells analyzed cover 17.8% of the Croatian territory. (See Figure HR-1.)

Calculation methods

The application of the Steady-State Mass Balance (SSMB) method for critical load mapping of both the Gorski Kotar (GK) and Northwest part of Croatia (NWPC) regions is very complex due to the large site variety and numerous combinations of parent rock, soil and vegetation.

In the GK region, 24 different soil-vegetation combinations were identified. The NWPC region contains some 26 soil-vegetation combinations, of which 3 are identical to those in the GK region. Data are based on 218 forest profiles for the GK region and 213 profiles for the NWPC region. About one hundred profiles have been identified by field soil sampling for the purpose of critical load mapping (Pernar 1997, 1998), on the basis of representative points selected to extend the existing soil data base (Martinović *et al.* 1998).

In the following, comments on some of the main variables of the SSMB model are given.

Weathering ($ANC_w = BC_w$): ANC_w values have been calculated according to the Mapping Vademecum (Hettelingh and de Vries 1992, pp. 34–37). For calcareous soils (class 4), the highest weathering rate of category six has been assumed.

Critical alkalinity leaching is calculated as:

$$Alk_{le(crit)} = -Q \cdot ([Al]_{crit} + [H]_{crit})$$

using the following values (from de Vries 1991):

	[Al] _{crit} (mol _c m ⁻³)	[H] _{crit} (mol _c m ⁻³)
pH > 4.0	0.2	0.1
pH < 4.0	0.4	0.2

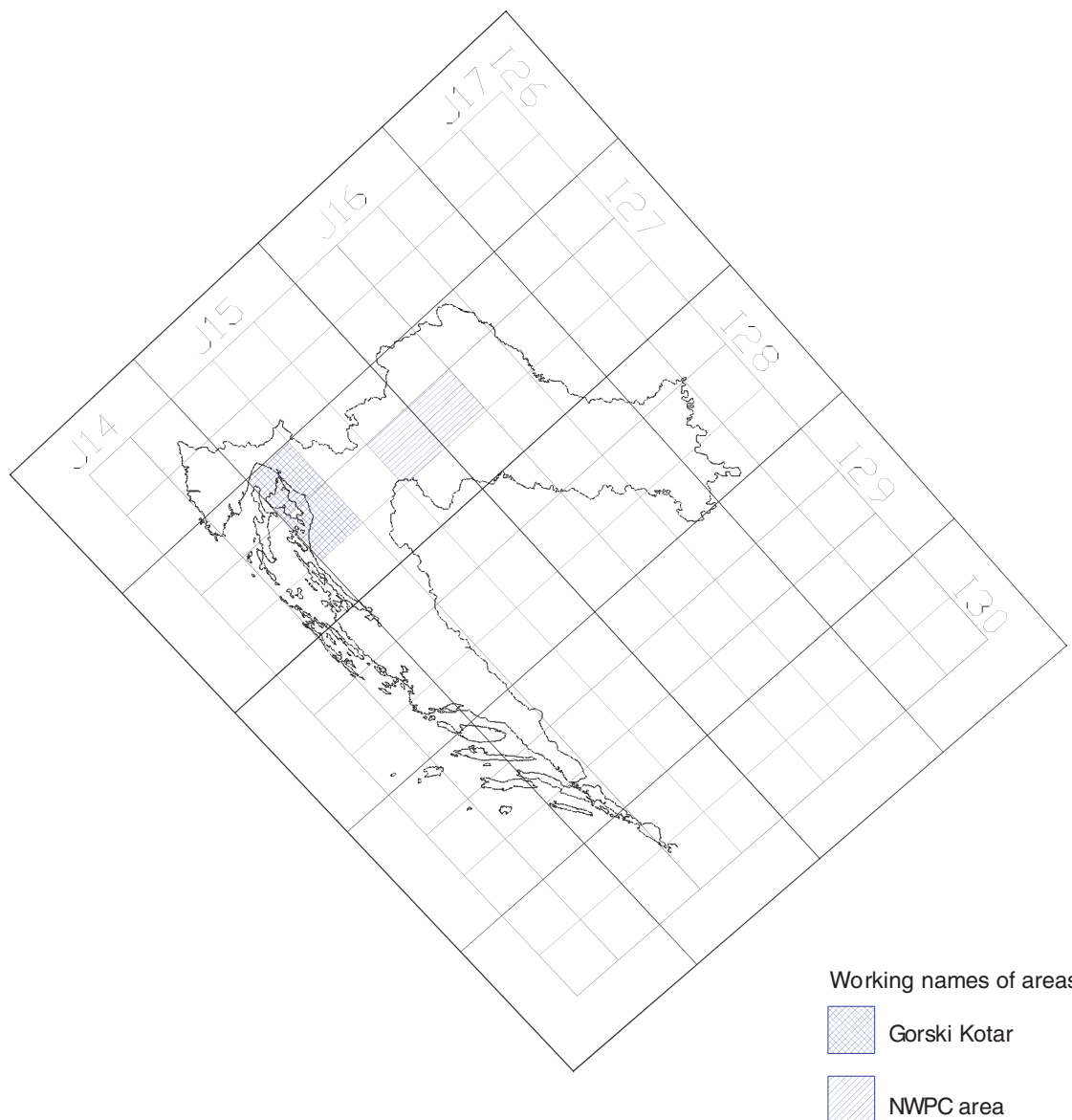


Figure HR-1. Location of the four EMEP 50x50 km² grid cells for which critical loads have been calculated.

Interception: The mean interception has been defined as:

$$I = a \cdot p$$

where:

p = precipitation

Values for a from de Vries (1991): pine 0.25, spruce 0.45, fir 0.40 (species composition: fir 60%, beech 40%, thus $a = 0.34$), beech 0.25 and oak 0.15.

Precipitation has been determined on the basis of thirty years of climate data from five weather stations and associated with different forest vegetation types, according to Bertović (1994).

Base cation uptake (BC_u): Annual volume increment (in m³ ha⁻¹) and harvesting were taken from normally managed forests. Mean values of volume density (in kg m⁻³) and Ca, Mg, K and Na contents were taken from de Vries (1991). For a few receptors, BC_u and N_u were determined to be zero as these receptors are treated as virgin forests (no utilization).

Critical nitrogen leaching: $N_{le(acc)} = Q \cdot [N]_{crit}$
 $[N]_{crit}$ has been defined within the ranges from Posch *et al.* (1993):

Species	$[N]_{crit}$ (mg N l ⁻¹)
spruce and fir	0.15
beech and fir	0.35
oak	0.35

Nitrogen immobilization, N_i : The range of N immobilization (2–5 kg N ha⁻¹ yr⁻¹), from Posch *et al.* (1993) was assigned to receptors on the basis of the total N content in the A soil layer:

N content	$N_{i(crit)}$ (kg N ha ⁻¹ yr ⁻¹)
< 0.40	2
0.40–0.50	3
0.50–0.60	5
> 0.60	5

Denitrification, N_{de} : was defined as:

$$N_{de} = \begin{cases} f_{de} \cdot (N_{dep} - N_u - N_i) & \text{if } N_{dep} > N_u + N_i \\ 0 & \text{otherwise} \end{cases}$$

Values for the denitrification factor f_{de} has been assigned according to Posch *et al.* 1993, in the range of 0.3–0.5.

Base cation deposition, BC_{dep} : Bulk deposition data for base cation deposition were extrapolated from 9 monitoring stations (Vidić 1997, 1998). One of these stations is located in the GK region; the other eight stations (including one EMEP station) are in the NWPC region. Bulk deposition includes only wet deposition (and a very small part of dry deposition). It is assumed that bulk deposition is equal to total deposition, since no other data are currently available. The Mapping Manual (UBA 1996) suggests not using a filtering factor.

Results

Critical loads of sulfur, $CL_{max}(S)$, range between 1447–3649 eq ha⁻¹ yr⁻¹ for the GK region and 946–2854 eq ha⁻¹ yr⁻¹ in the NWPC region. For the GK region, the pentile values for $CL_{max}(S)$ were considerably higher than those calculated using the European background data base (EU-DB) used earlier to calculate critical loads for Croatia (Posch *et al.* 1995), which can only partly be explained by the NFC assumption that $BC_w = BC_{total}$. Critical loads of nutrient nitrogen, $CL_{nut}(N)$, in the GK region are between 531–1794 eq ha⁻¹ yr⁻¹, or approximately the same as the

values from the EU-DB. In the NWPC region, $CL_{nut}(N)$ ranges from 1085–1814 eq ha⁻¹ yr⁻¹, about three times higher than EU-DB values. This could be explained mostly by the higher N_u values that the NFC used in the national calculations. It should be noted that the comparison given above is approximated, because data submitted by NFC comprise only a part of 150×150 km² grid cell for which EU-DB values were available.

Data sources

- Receptor map 1:100,000 (Lindić 1998a). Mapping units were defined by the sequence of soil-vegetation forest types.
- Forest vegetation data: Based on vegetation maps of forest ecosystems (Forestry Institute Jastrebarsko, Lindić 1998b) and other related literature (Pelzer 1982, 1989; Rauš and Vukelić 1994; Trinajstić *et al.* 1992).
- Soil data: Soil data base of Croatia (Martinović *et al.* 1998).
- Precipitation: data on climatic zones of forest vegetation (Bertović 1994).
- Base cation (BC_{dep}) and chlorine (Cl⁻) deposition: Meteorological and Hydrological Service of Croatia, one station from Gorski Kotar (for the years 1981–1994) and eight stations from the NWPC region (years 1995–1996).
- Base cation (BC_u) and nitrogen (N_u) uptake by harvesting: local data on normal wood volume increment and harvest, the average timber quantity in the last 20 years.
- Drainage water (Q): Data on measuring or main receptors, $Q = (P - I) \cdot 0.15$.

Comments and conclusions

Calculation and mapping of critical loads started in the Republic of Croatia with the submission of national data to the CCE in 1997. At present, four 50×50 km² EMEP grid cells have been mapped. These grids were selected on the basis of priority area selection (higher vulnerability, pollution load, forest damage and economic interest). Critical loads mapping considered as a very important task since Croatia is 43% covered by forest, its per capita emission is the lowest in Europe, and its import of transboundary pollution is much higher than its export. Mapping is planned to be conducted for the entire Croatian area. In addition, the grid cells already mapped are planned to be further analyzed by more complex models (e.g., SMART and PROFILE).

Comments on national conditions related to SMB method:

In the SMB method application, national data for the following variables are used:

- net growth and harvest
- volume increase of wood harvest from the mapped forest area
- drainage water
- precipitation by bio-climate regions
- deposition (BC_{dep} and N_{de}).

The other input data are taken from the literature and other instructions as well. The application of the SMB method indicates some national ecological characteristics that should be taken into account:

1. In functional relations by which the f_{de} value is determined, the soil drainage capacity should be included due to its wide value range in Croatia. (This is especially true for clay soils: *terra rossa* is well-drained as compared to clay soils of lake sediments that have poor drainage).
2. Functional relations that determine BC input from parent material weathering require further elaboration of the values obtained.
3. Data on total base cation deposition (wet + dry + cloudwater/fog) are not available, as is very often the case in other national contributions. Sulfur and nitrogen deposition data from bulk deposition are higher than total deposition calculated by EMEP. Common recommendations on the above issues are welcome, as well as reconsideration of possible use of some filtering factor.
4. Consistency in using symbols is recommended in all documents, including NFC contributions.
5. Seasalt value correction methods, and the development of criteria for implementation, need further explanation (e.g. mass proportion of different substances, geographical position of the mapped area, or other aspects).
6. More than one million hectares of Croatian territory consists of hard and pure limestone parent rock from the Mesozoic era. After decomposition, there remains 0.1% to 0.5% of kerolium that is non-calcareous and contains very resistant primary minerals. Methods to determine weathering rates in this case are needed.

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Calculation methods

The critical loads data for the Czech Republic have not been changed since 1997. See the CCE Status Report 1997.

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National maps produced

- Critical load, and exceedance of the critical load of acidity for forest soils and extensively managed, permanent grasslands calculated with PROFILE, and for grasslands with the SSMB model.
- Critical load and exceedance of the critical load of nutrient nitrogen for production forests calculated with PROFILE.
- National deposition maps of NH_x on a 5×5^2 km grid.
- National deposition map of NO_x on a 30×30^2 km grid.
- National deposition map of SO_x on a 20×20^2 km grid.

Calculation methods

Critical loads of acidity and N eutrophication: The PROFILE model has been used to calculate the critical load of acidity and of nitrogen eutrophication, and the values of BC_w , N_w , BC_w , and $ANC_{le(crit)}$. From this calculation, the values of $CL_{max}(S)$, $CL_{min}(N)$, and $CL_{max}(N)$ have been derived. To calculate the critical load for grasslands, the weathering rate for 11 mineralogy classes were calculated at 1000 points with the PROFILE model (Warfvinge and Sverdrup 1992). The calculation of critical loads for grasslands were performed with the SSMB model (UBA 1996). The last major update of the critical load calculations was made in December 1996.

The total number of calculations and the calculated critical loads for the different vegetation types are summarized in Table DK-1.

Table DK-1. Calculated critical loads of acidification and N eutrophication for different ecosystems. All values are given in $\text{keq ha}^{-1} \text{yr}^{-1}$ as the range between the 5 and the 95 percentile.

	No. of calculations	CL(A)	CL _{nut} (N)
beech	2825	0.9 – 2.7	1.2 – 1.9
oak	448	0.8 – 2.2	1.2 – 2.0
spruce	5480	1.4 – 4.1	0.6 – 1.1
pine	1035	1.4 – 2.4	0.5 – 0.7
grass	18178	0.9 – 2.4	—

A BC/AI ratio of 1 was used as the chemical criterion for both forest soils and grasslands. For the calculation of critical loads of nutrient nitrogen, a critical N leaching, $N_{le(crit)}$ of $2 \text{ kg N ha}^{-1} \text{yr}^{-1}$ and an immobilization, $N_{imm(crit)}$ of $3 \text{ kg N ha}^{-1} \text{yr}^{-1}$ were applied. For the model calculations, the root zone has been stratified in a 5-cm thick A/E horizon, and a soil-dependent B and C horizon. A total root depth of 50 cm was applied for spruce and pine, 70 cm for beech, 90 cm for oak, and 25 cm for grasslands.

National deposition maps: A new national calculation of NH_x deposition on a $5 \times 5 \text{ km}^2$ grid has been performed as part of the technical background for the work with a national action plan for the abatement of ammonia emissions from agriculture. The calculation has been based on 1996 emissions. The spatial distribution of emissions has, however, been based on county-level statistics from 1989. As part of the Danish Nationwide Background Monitoring Programme, deposition calculations of both NO_x and NH_x to the Danish sea and land area have been performed on a $30 \times 30 \text{ km}^2$ national grid on a yearly basis. The latest reporting of data from this programme has been in 1998, using 1997 data. The latest national calculation of SO_x deposition was made in 1996 for the year 1990.

Data sources

The main sources of data have not been changed since the CCE Status Report 1997 (Posch *et al.* 1997). The sources and resolution of data are shown in Table DK-2.

Table DK-2. Sources of data.

Parameter	Resolution	Source
soil mineralogy	60 points	DLD, literature
soil texture	1:500,000	DLD
geological origin	1:500,000	DLD
crop yields	county	DSO
forest production	1:500,000	DLD, DSO
ecosystem cover	25 ha	NERI
deposition (S, N)	5×5, 20×20	NERI
meteorology	1:1,000,000	DMI

DLD: National Institute of Soil Science, Dept. of Land Data

DSO: Danish Statistical Office

NERI: National Environmental Research Institute

DMI: Danish Meteorological Institute

Comments and conclusions

The main focus of the Danish NFC in the past two years has been on:

- Evaluation and updates of data for Denmark in the RAINS model used in preparation for the negotiations on the coming UN/ECE multi-pollutant, multi-effect protocol and the EU acidification strategy.
- An analysis of the RAINS model and the sensitivity of calculated emission ceilings to changes and uncertainty in energy and agricultural scenarios, deposition targets, and cost curves (Bak and Tybirk 1998).
- A first calculation of critical loads for lead and cadmium for Danish soils (Bak and Jensen 1998).
- Further work on methods and data for the calculation of critical loads of nutrient nitrogen for sensitive, natural or seminatural terrestrial ecosystems, primarily raised bogs and heathlands.
- Estimation of the uncertainties in calculated critical load exceedances with special emphasis on the influence of local scale variation in NH_x deposition.
- Preparation of an international workshop on critical loads, "Critical Loads Copenhagen," to be held in Copenhagen from 21-25 November 1999.

As indicated, only minor progress has been made in the availability of data for calculating critical loads and exceedances. Maps of $CL(A)$ and $CL_{nut}(N)$ are displayed in Figure DK-1.

In 1996 the results from a Danish monitoring program on heavy metals were reported. The monitoring program was initiated in 1992 and includes the heavy metals Pb, Cd, Ni,

Zn, Cu, Cr, Hg and As monitored at 393 sampling sites covering Danish arable land and nature areas. On the basis of data and findings from this project, a first calculation of critical loads for lead and cadmium was performed.

When adopting a new action plan for the protection of the aquatic environment in 1998, the Danish Parliament called the government to prepare a national action plan for the abatement of ammonia emissions from agriculture before summer 1999. As part of this work, reports on sources and status of ammonia emissions, effects on nature and environment, technical measures and potential of abatement, and on abatement costs, have been prepared, using 1996 as a base year. A key element of the work has been an evaluation of the effects on Danish environment and nature of national reductions in ammonia emissions on 50% and 100%. The effect has been evaluated using the protected ecosystem area as a measure.

Scale dependency: The calculation of critical load exceedances implies in general the combination of critical load data at a high spatial resolution with deposition data at a much coarser spatial resolution. Usually the exceedance is calculated as the difference between a deposition value and a percentile of the critical load values calculated within each grid. The total uncertainty of the calculated exceedance can thus be divided into:

- (i) the uncertainty of the calculated critical loads,
- (ii) the uncertainty of the calculated deposition for the grid, and
- (iii) the spatial variation of the deposition within the grid.

In addition, there are systematic differences between deposition velocities to different types of ecosystems which will introduce an error when using average deposition values for a grid to evaluate critical load exceedances for different types of ecosystems. Especially for coniferous forests and forest edges, deposition velocities are much higher than for open land.

The influence of uncertainty, within-grid variation and differences in deposition velocities have been taken into account in the new Danish assessments of critical load exceedances. The results are significantly higher exceedances than shown by calculations using average deposition values for individual grids. A comparison between the new Danish assessments of critical load exceedances and an estimate using average deposition values for individual grids is shown in Table DK-3.

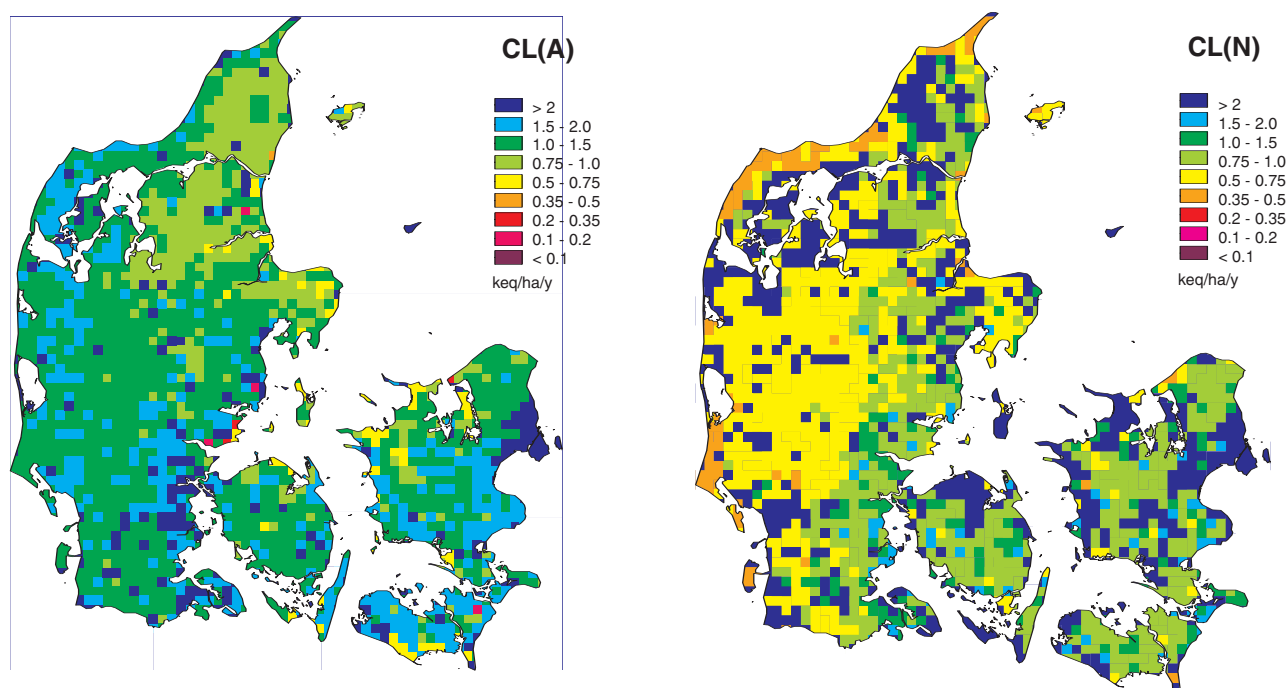


Figure DK-1. Critical loads of acidity and nutrient nitrogen, 5 percentile.

Table DK-3. A comparison between a new Danish assessments of critical load exceedances and an estimate using average deposition values for individual grids. The new assessment includes the influence of uncertainty, within grid variation in deposition, and differences in deposition velocities between different ecosystem types, including forest edges. Median values are presented in the table.

		Area with exceedance of the critical load (%)			
		oak	beech	spruce	pine
New assessment	acidification	38	30	44	47
	eutrophication	51	40	81	81
Assessment using average deposition	acidification	4	2	0	0
	eutrophication	24	7	90	98

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National maps produced

Critical loads of nitrogen and sulfur (minimum, maximum and nutrient) for forest soils and oligotrophic peatlands (raised bogs).

Calculation methods

Two methods have been used to calculate critical loads: the empirical method for raised bogs, and steady-state mass balance calculations for forested areas (UBA 1996). Compared to the critical loads reported in the Status Report 1997, a change was made in using the nitrogen uptake and nitrogen immobilization rates. Earlier, nitrogen uptake had been considered the same for all forested areas, equal to the average value. Using different nitrogen uptake values, the uptake was increased for most commercial forest areas on good soils and lowered on peatlands as there is no harvesting export from those areas. This has resulted in a slight increase of the values of $CL_{max}(N)$. Comparison of the critical loads to actual deposition (Roots and Talkop 1998) does not show much difference from the earlier situation.

Results

Five-percentile maps of $CL_{nut}(N)$, $CL_{min}(N)$, $CL_{max}(N)$ and $CL_{max}(S)$ are shown in Figure EE-1.

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Figure EE-1. Five-percentile $CL_{min}(N)$, $CL_{min}(N)$, $CL_{max}(N)$ and $CL_{max}(S)$ in $\text{eq ha}^{-1} \text{yr}^{-1}$.

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National maps produced

The critical loads of sulfur $CL(S|N_{dep95})$ and nitrogen $CL(N|S_{dep95})$ for forest soils and lakes were calculated using the modeled deposition of sulfur and nitrogen in 1995 (Syri *et al.* 1998). The 5th percentile of each grid cell is shown in Figure FI-1.

The cumulative ozone exposure index, AOT40 (accumulated exposure over a threshold of 40 ppb), was evaluated by the Finnish Meteorological Institute (FMI) for eight background monitoring stations. These results are shown in Figure FI-2 for agricultural crops and forest trees, calculated following the definitions adopted at the UN/ECE workshop in Kuopio (Kärenlampi and Skärby 1996). The AOT40 values are scaled for missing data. The data sets with capture poorer than 80% have been rejected.

Calculation methods

The calculation of critical loads for Finnish forest soils and lakes follows the methodology of the UN/ECE mapping manual (UBA 1996) and described more fully in Posch *et al.* (1997). Critical loads of acidity of N and S for surface waters and forest soils are derived from the acidity balance for the sum of N and S deposition (Posch *et al.* 1997).

$$N_{dep} + S_{dep} = fN_u + (1-r) \cdot (N_i + N_{de}) + rN_{ret} + rS_{ret} + BC_{le} - ANC_{le} \quad (1)$$

where the base cation (BC) leaching is given by

$$BC_{le} = BC_{dep} + (1-r)BC_w - fBC_u \quad (2)$$

where f is the fraction of forested land in the catchment area, r is the lake:catchment area ratio, N_u and BC_u are the net growth uptake of N and BC, N_i is the immobilization of N in soils, N_{de} is N denitrified in soils, N_{ret} and S_{ret} are the in-lake retention of N and S, BC_{dep} is the base cation deposition, BC_w is the base cation weathering, and ANC_{le} is the alkalinity leaching. For lake catchments the term $(1-r)$ limits the influence of N_i , N_{de} and BC_w to the terrestrial area, and f limits the uptake to the forested area only. For forest soils one has to set $f=1$ and $r=0$.

Inserting the deposition-dependent expressions for soil denitrification and in-lake N and S retention into Eq. 1 one obtains

$$a_N N_{dep} + a_S S_{dep} = b_1 N_u + b_2 N_i + BC_{le} - ANC_{le} \quad (3)$$

where the dimensionless constants a_N , a_S , b_1 and b_2 are all smaller than one and depend on ecosystem properties only: denitrification fraction (f_{de}), net mass transfer coefficients for S and N (s_S and s_N), and runoff (Q). For soils, BC_{le} at critical load is computed from Eq. 2. For lakes the net base cation leaching at critical load is computed from water quality data

$$BC_{le(crit)} - ANC_{le(crit)} = Q ([BC]^*_0 - [ANC]_{limit}) \quad (4)$$

where $ANC_{le(crit)}$ is ANC_{le} at critical load, $Q [BC]^*_0$ is the pre-acidification leaching of base cations from the catchment area, and $Q [ANC]_{limit}$ is the critical alkalinity leaching. By prescribing a maximum acceptable leaching of N, the critical load of nutrient nitrogen can be computed for soils, using the mass balance:

$$CL_{nut}(N) = N_u + N_i + N_{le(acc)} / (1-f_{de}) \quad (5)$$

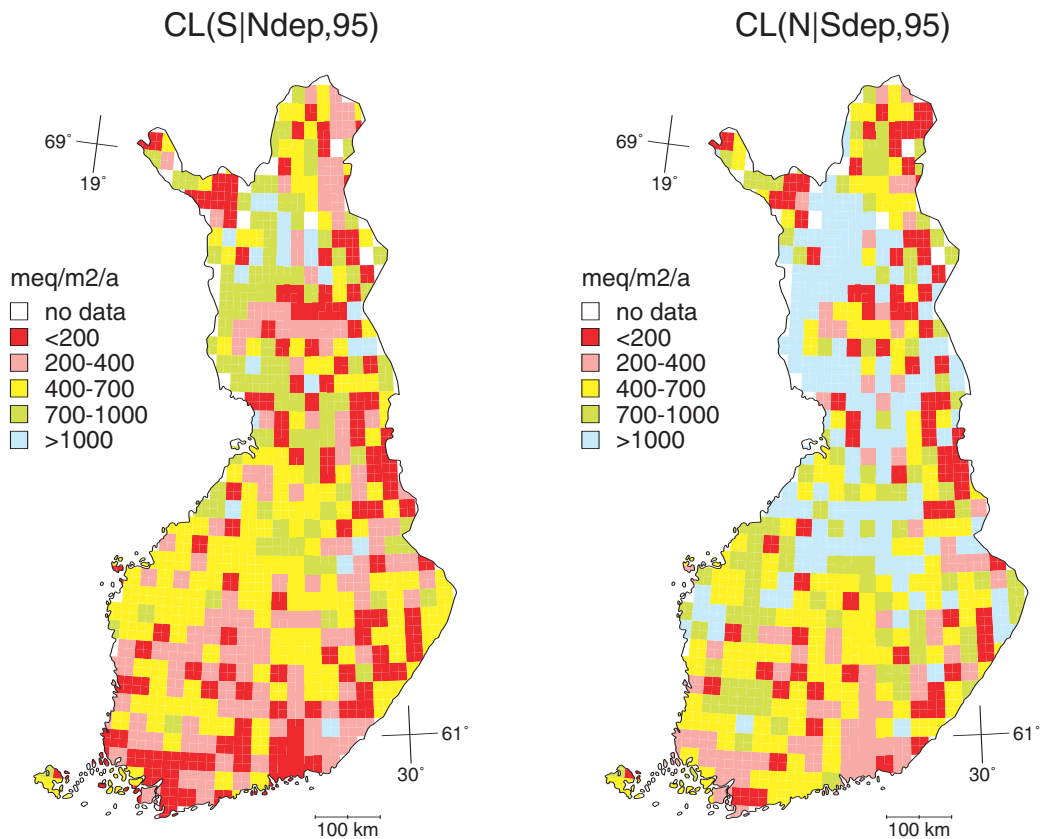


Figure FI-1. The 5th percentile critical loads of sulfur, $CL(S|N_{dep95})$ and nitrogen, $CL(N|S_{dep95})$ for forest soils and lakes using the modeled deposition of sulfur and nitrogen in 1995 (Syri *et al.* 1998).

Input data

Deposition: Sulfur and nitrogen deposition are calculated with the DAIQUIRI (Deposition, Air Quality and Integrated Regional Information) model, employing long-range and mesoscale transfer matrices from EMEP/MSC-W and the Finnish Meteorological Institute (Syri *et al.* 1998). Base cation deposition (Ca, Mg, K, Na) is interpolated from the data from 1993–1995 of a nationwide network of stations measuring monthly bulk deposition (Järvinen and Vänni 1990), and seasalt correction is made where necessary, using Na as a tracer.

Base cation weathering: The historical long-term base average BC_w ($= Ca_w + Mg_w$ in Finnish calculations) was estimated by applying the results of the field studies of Olsson *et al.* (1993) and using the effective temperature sum (ETS) and the total element content of Ca and Mg in the C-horizon as input data. Values of ETS were given by Ojansuu and Henttonen (1983). Total analysis data on the < 2.0 mm fraction of till required by the method were obtained for 1057 plots from the Geological Survey of Finland (Johansson and Tarvainen 1997). The method employed gave weathering rates comparable to those obtained from an input-output budget, the PROFILE

model and the direct use of Zr-depletion method at one Finnish site (Starr *et al.* 1998).

Nutrient uptake: N_u and BC_u refer to the long-term average net uptake of N, Ca, Mg and K in the stem and bark biomass removed from forest via harvesting. They are estimated from annual average potential forest growth, calculated for each tree species based on ETS, and the element contents (Olsson *et al.* 1993, Rosén, pers. comm.) in biomass. The limiting concentration, $[BC]_{min}$, below which trees can no longer extract nutrients from soil solution, is set to a precautionary value of 2 meq l^{-1} (UBA 1996).

Denitrification and nitrogen immobilization: N_{de} is assumed to be proportional to the net incoming N ($N_{de} = f_{de} (N_{dep} - N_i - N_u)$), and the denitrification fraction ($f_{de} = 0.1 + 0.7 f_{peat}$) is related to the soil type by linearly interpolating between a low value of 0.1 for podzolic mineral soils and a value of 0.8 for peat soils (Posch *et al.* 1997). For N_{ir} , including N_{fix} , a constant value of 1.0 kg N $ha^{-1} yr^{-1}$ as a long-term average was used for Finnish forest soils, representing the upper limit of the range of values recommended (UBA 1996).

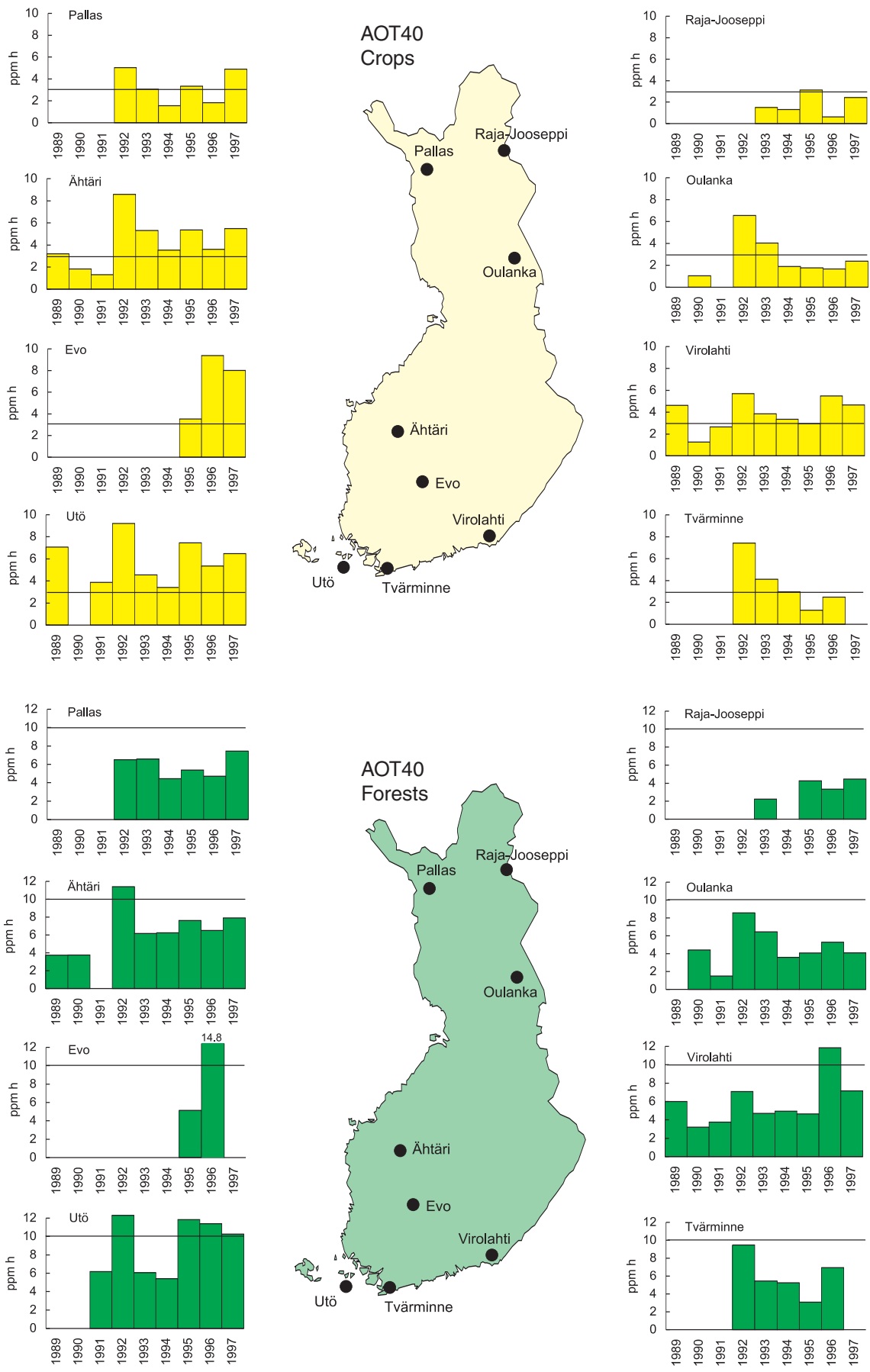


Figure FI-2. The measured yearly AOT40 (accumulated exposure over a threshold of 40 ppb) values, displayed in ppm-h, for agricultural crops and forests, as defined at the UN/ECE workshop in Kuopio 1996 (Kärenlampi and Skärby 1996). Values of zero indicate that the station was not in operation or that the capture was less than 80%. The critical levels (3 ppm-h for crops and 10 ppm-h for forests) are indicated.

Leaching of alkalinity and nitrogen: $ANC_{le(crit)}$ is calculated by adding the critical aluminum leaching, obtained from the molar Al/BC ratio of 1.0, to the hydrogen leaching, calculated from a gibbsite equilibrium ($K_{gibb} = 10^{8.3}$). Acceptable nitrogen leaching is derived with runoff using the concentration criterion 0.3 mg N l^{-1} (Downing *et al.* 1993). The runoff values needed for converting concentrations to fluxes were obtained from a digitized runoff map for 1961–1975 (Leppäjärvi 1987).

Lake-specific parameters: Values for the retention of sulfur S_{ret} , and nitrogen N_{ret} , were computed from kinetic equations (Kelly *et al.* 1987) using the net mass transfer coefficients $s_S = 0.5 \text{ m yr}^{-1}$, and $s_N = 5.0 \text{ m yr}^{-1}$, which were taken from retention model calibrations in North America (Baker and Brezonik 1988, Dillon and Molot 1990). $[BC]^*_0$ was estimated using the so-called F-factor, which relates the change over time in the leaching of base cations to long-term changes in inputs of strong acid anions in a lake, estimated as a function of the present base cation concentration. $[SO_4^{2-}]^*_0$ was estimated from the relationship between present sulfate and base cation concentrations from 251 lakes located in northern Fennoscandia receiving very low acidic deposition (Henriksen *et al.* 1993). An $[ANC]_{limit}$ value of 20 meq l^{-1} was selected as the chemical criterion based on results of a fish status survey conducted in Norway (Lien *et al.* 1996). The data for lakes were mostly obtained from a national statistically based lake survey of 970 lakes conducted in 1987 and 480 additional lakes surveyed in 1987–1989 by the Lapland Water and Environment District (Kämäri *et al.* 1991, Posch *et al.* 1997). The spatial distribution of the lake data set reflects the actual lake density in different regions. Both lake and catchment areas, as well as the forest fraction, were measured from topographic maps.

Tropospheric ozone

According to the UN/ECE recommendations adopted at the Kuopio workshop, the exposure index is calculated for daylight hours, defined as those hours with a clear sky global radiation of 50 W m^{-2} or greater. There is no unique procedure for estimating this radiation quantity, and different methods may result in a different number of daylight hours for the AOT40 calculations. FMI has used the sun elevation angle as the actual criterion in the calculations. A limit of 3 degrees was determined on the basis of local measurements of global radiation. It was observed that this approach leads to a significantly longer daylight period than the theoretical algorithm employed by the Chemical Coordinating Centre of EMEP, for instance. The corresponding AOT40 values are typically 10–30% higher for the longer daylight period, depending on the location. The problem is avoided in the proposed EU

directive, in which the daily time window is defined by fixed hours (8.00–20.00 CET). This definition also results in lower AOT40 values than the sun elevation condition used at FMI.

As a related activity, FMI has continued studies on ozone deposition to different terrestrial ecosystems in northern Europe (Tuovinen *et al.* 1998, 1999). FMI has also participated in the development of improved critical levels (Manninen *et al.* 1998) and mapping methods (Tuovinen 1999). The latter also includes a new parametrization module for ozone deposition in Europe. This deposition module is being applied to the critical levels assessment work (Emberson *et al.* 1999) and will be incorporated in the photochemical transport model of EMEP (Simpson *et al.* 1999).

Comments and conclusions

The total ecosystem area mapped for critical loads is $273,000 \text{ km}^2$ including forest soil and inland surface waters. The land use classification is based on satellite images. The two ecosystem types are initially valued equally at the national level. Forest soils are weighted according to the relative area of tree species and all lakes are considered equally important. Therefore, the weight for each calculation point (as km^2 in the data base) does reflect the actual ecosystem area at that location (e.g. water surface or catchment area of a specific lake), but rather its subjective importance.

There is a consistent geographical distribution in the ozone exposure across Finland (Fig FI-2). The observed concentrations are, however, also markedly influenced by local site characteristics, such as altitude and surrounding vegetation. The AOT40 indices vary strongly from year to year as a result of differences in weather patterns. The critical level for forests, 10 ppm-hours as a 5-year mean, is not exceeded at the Finnish monitoring sites, whereas that for crops, 3 ppm-hours , the mean value is exceeded over large areas.

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National maps produced

- Soil weathering rate
- Critical loads of acidity for soils (2)
- Map of the critical load difference according to mineralogical reconstitution

Mapping procedures

The main steps of critical load mapping of acidity have been as follows:

1. The main soil formations were combined with the main corresponding bedrocks.
2. 12 reference sites from representative catchments of the most significant bedrock types distributed throughout France were selected. Weathering rates have been calculated using the PROFILE model on the basis of chemical data and reconstructed mineralogy according to two normative methods:
 - UPPSALA model (Olsson and Melkerud 1990, Melkerud *et al.* 1992), and
 - Barth's mesonorm (Barth 1960, adapted by Stussi 1997).
3. An empirical model was set up based on calculated weathering rates versus granulometry and soil base cations (Ca+Mg+K). The model was applied to the soils from 100 reference sites belonging to the French permanent network for the monitoring of forest ecosystems (RENECOFOR).
4. The weathering rates (pentile) have been mapped according to normative method (2), as shown in Figure FR-1.
5. The drainage spatialization was derived from residual rainfall map (BRGM 1983) weighted by water storage capacity according to the root depth of the different main tree species.
6. Critical loads of acidity were calculated on the 12 reference soils using the BC/Al ratio as a critical limit indicator. A relationship was established between the weathering rates and critical loads and extrapolated to the 100 other sites where base cation data are lacking.
7. Critical loads of acidity for soils were mapped using Al/BC criterion (R_{crit}) (Figure FR-2) and Al criterion (Figure FR-3).
8. To assess the influence of the mineralogical reconstitution method, critical loads were also mapped using normative method (1) and the BC/Al criterion. The differences in critical load values from Figure FR-2 is shown in Figure FR-4.

Calculation methods

The Steady-State Mass Balance (SSMB) model was applied to soils. Critical loads have been calculated according to the following equation:

$$CL (Ac_{act}) = BC_w + Q \cdot [H^+]_{crit} + R_{crit} \cdot (BC_d + BC_w - BC_u)$$

where $[H^+]_{crit}$ is critical hydrogen concentration in

drainage water ($= 25 \mu\text{eq l}^{-1}$, which corresponds to $\text{pH} = 4.6$ adapted for French forest soils, Party 1999); R_{crit} is the critical aluminum/base cation ratio ($= 1.2 \text{ mol}_c \text{ mol}_c^{-1}$ calibrated at the national scale, Party 1999); BC_w is the weathering of base cations; BC_d is base cation deposition; BC_u is uptake of base cations ($\text{mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$) and Q is annual runoff ($\text{m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$).

The exceedances were calculated for 100 plots of the RENECOFOR network as follows :

$$\text{Ex}(1) = H_p^+ - Cl_{ac} \quad (2)$$

$$\text{Ex}(2) = S_d + N_d - BC_d - Cl_{ac} \quad (3)$$

No exceedance map is presented due to the lack of reliable deposition data.

Results and discussion

Whatever the mineralogical method used, weathering rates and critical loads of acidity exceed $2.0 \text{ keq ha}^{-1} \text{ yr}^{-1}$ for about 40% of the French territory (Fig. FR-1 to FR-3). Weathering rates lower than $0.5 \text{ keq ha}^{-1} \text{ yr}^{-1}$ cover about 25% of the area, whereas corresponding critical loads are of 5% according to the BC/Al (Figure FR-2) criterion and near 0% according to the Al criterion (Figure FR-3).

The areas with the lowest critical load values ($\leq 0.5 \text{ keq ha}^{-1} \text{ yr}^{-1}$) correspond to: the granites and sandstones of the Vosges mountains (NE of France); the acid granites of the Central Massif; the granites and schists of the Vendée, the Bretagne and the Normandie (W part of France); the acid schists of the Ardennes (N of France); the sands of the Landes (SW of France) and the flint sands of the Loire and the Seine valley. These latter appear to be the most sensitive, with critical loads values $\leq 0.2 \text{ keq ha}^{-1} \text{ yr}^{-1}$.

Figure FR-4 shows the differences in critical load values depending on the mineralogical model used. It indicates that 95% of the area result in similar values ($\pm 0.2 \text{ keq ha}^{-1} \text{ yr}^{-1}$). However, for the critical areas which represent about 20% of the forested areas (i.e. 5% of the country area) where critical load values are $\leq 1.0 \text{ keq ha}^{-1} \text{ yr}^{-1}$, the mineralogical normative method (1) leads to critical loads values higher than those derived from mineralogical normative method (2). That means that the critical load maps (Fig. FR-2 and FR-3) take into account the most sensitive areas of French forest ecosystems. The remaining 5% for which critical value differences are important between the two methods only concern areas where critical loads are $\geq 1.0 \text{ keq ha}^{-1} \text{ yr}^{-1}$, and are mainly in between 0.5 and $1.0 \text{ keq ha}^{-1} \text{ yr}^{-1}$. These areas cover lixiviated hydromorphic soils (luvisol and podzoluvisol) on silts from flint formations and acid alluvial deposits.

Among the 100 plots of the RENECOFOR network,

according to Eq. 2, only one plot (near an industrialized site in Normandy) exceed acid critical load values, whereas when Eq. 3 is used, 18 plots show exceedances (Figure FR-5). For 18 other plots exceedances are close to zero. These sites correspond to the sensitive regions mentioned above and are mainly covered by Scots pine, Norway spruce and to a lesser proportion by sessile oak, maritime pine and beech.

The differences between Figures FR-1, FR-2 and FR-3 show that attention must be paid on the universal use of admissible critical alkalinity leaching and about the importance of the retained criteria (R_{crit} , Al). It would be needed particularly to check these criteria according to the vegetation cover.

Data sources

- Map of French soils (1:1,000,000)
- Geological map (1:1,000,000)
- Residual rainfall map (1:1,500,000)
- Map of land use (1:2,500,000)
- Soil and deposition data from the 100 plots of the RENECOFOR network.

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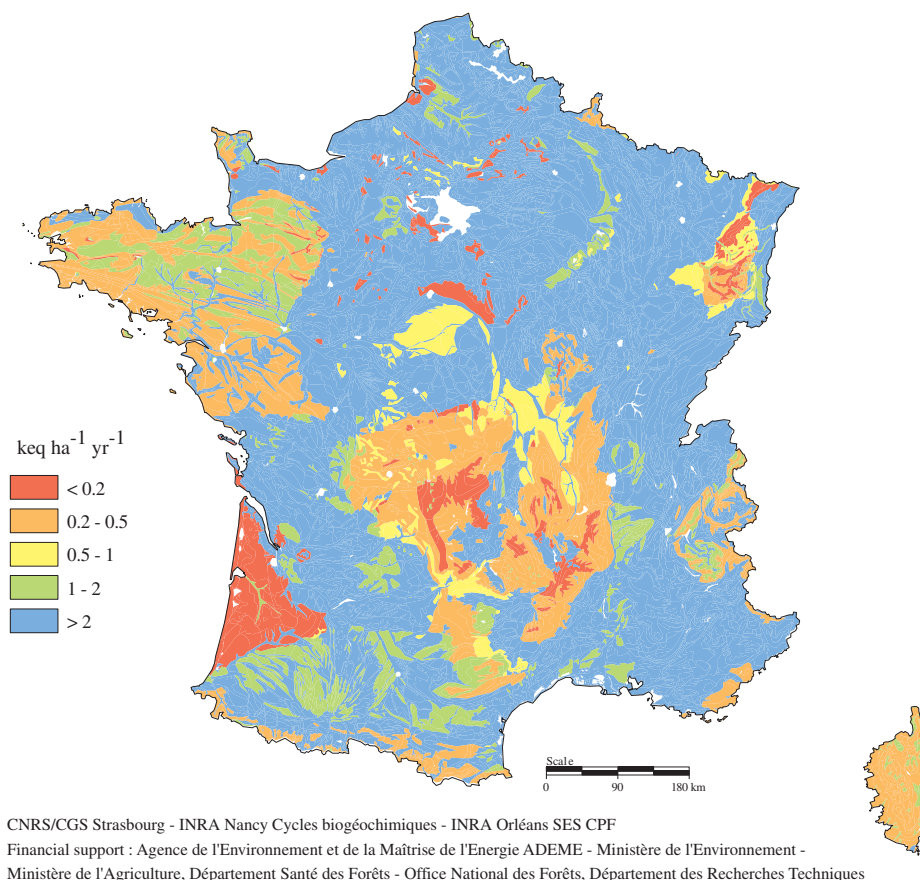


Figure FR-1. Weathering rates calculated using PROFILE after reconstructing mineralogy using a normative method (Barth 1960, adapted by Stussi 1997).

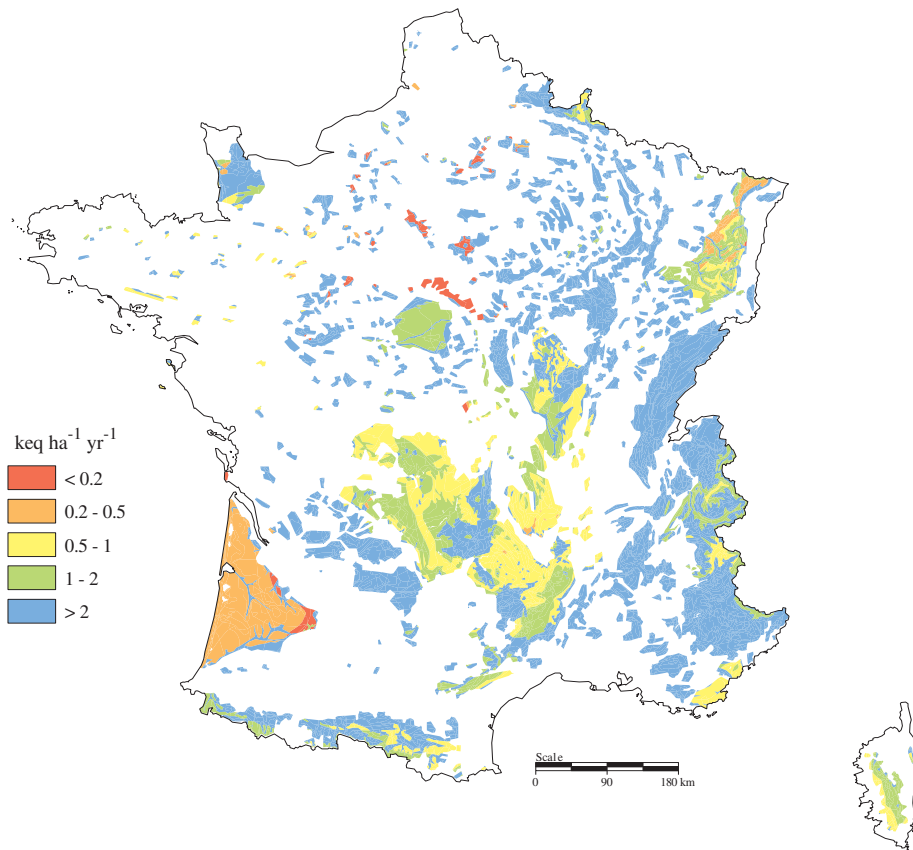


Figure FR-2. Critical load of acidity to forest soils in France using the BC/AI criterion.

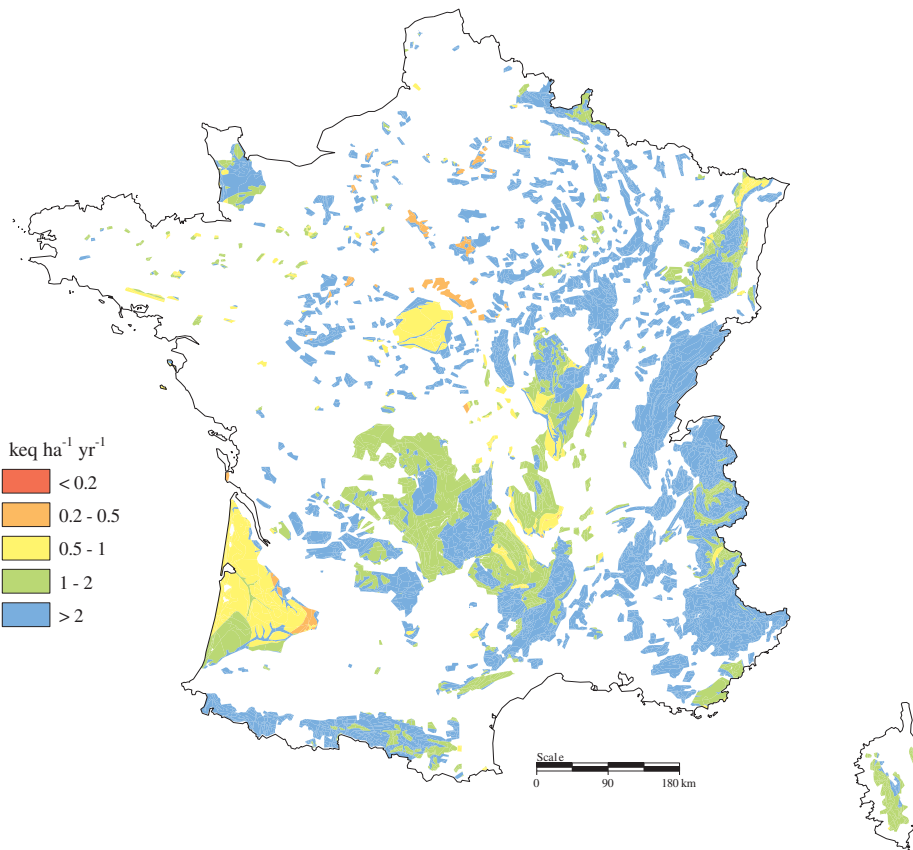


Figure FR-3. Critical load of acidity to forest soils in France using the AI criterion.

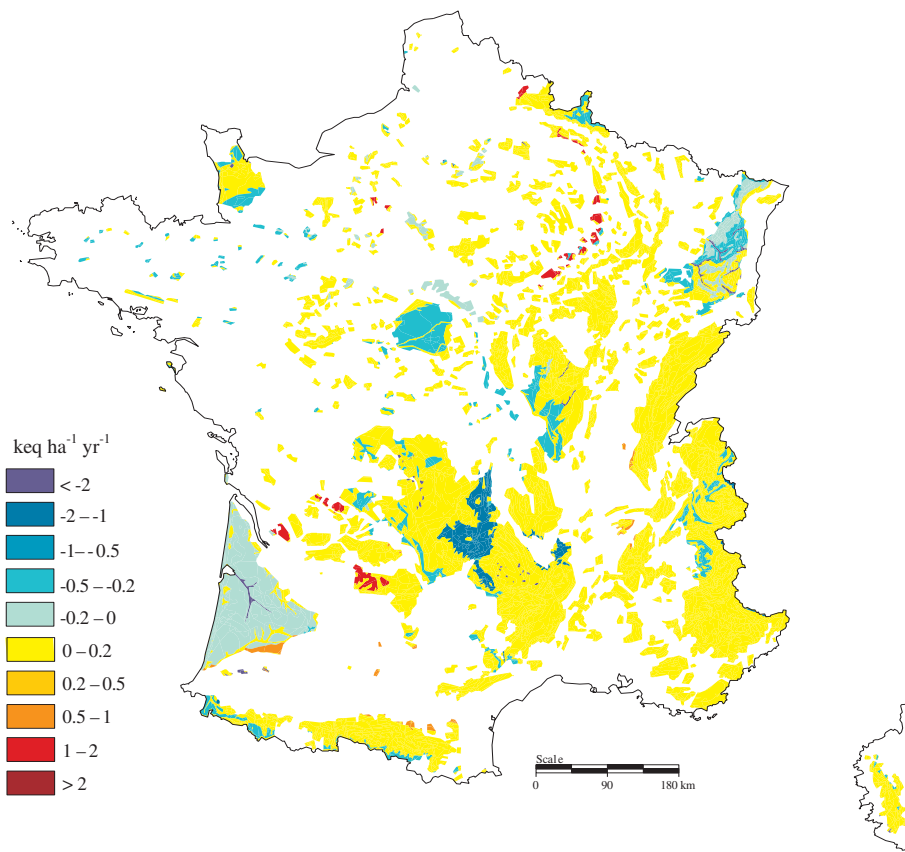


Figure FR-4. Difference between the critical loads of acidity for soils based on two reconstructed mineralogies (Barth 1960 adapted by Stussi 1997, and the UPPSALA model).

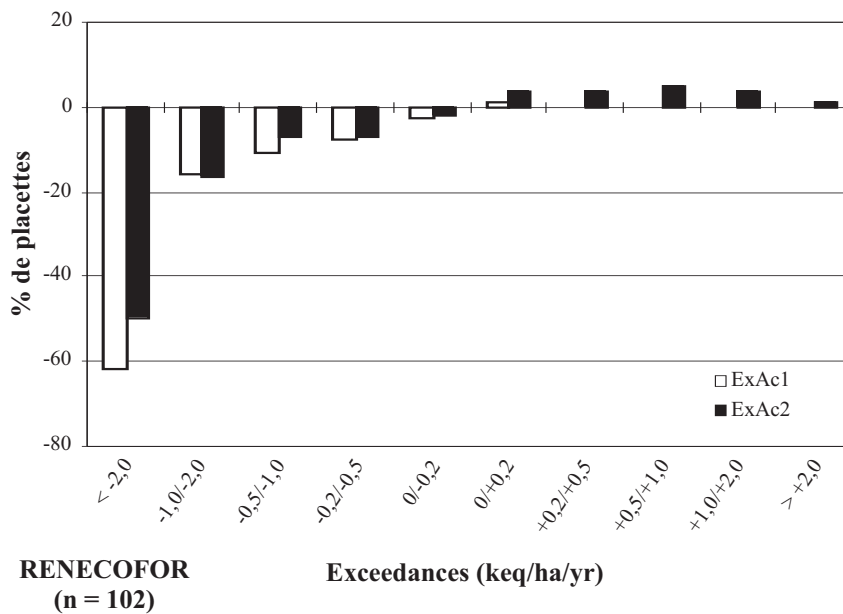


Figure FR-5. Distribution of acid input exceedances calculated for the soils of 100 plots of the RENECOFOR network.

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National maps produced

All of the maps below are prepared on the Lambert Conformal Conic projection, with a grid size of 1×1 km².

Critical loads:

- Critical loads of sulfur and nitrogen, $CL(S+N)$
- Critical loads of nutrient nitrogen, $CL_{nut}(N)$

Deposition:

- Total (dry and wet) deposition of: SO_x^* , NO_y , NH_x , Na, base cations* (Ca, Mg, K), potential acidity ($SO_x^* + NO_y + NH_x + Cl^*$), net potential acidity ($SO_x^* + NO_y + NH_x - BC^* + Cl^*$) in 1987–89 and 1993–95 (* = seasalt corrected)

Critical load exceedances:

- Exceedance of critical loads of sulfur and nitrogen, $Ex(S+N)$, in 1987–89 and 1993–95.
- Exceedance of critical loads of nutrient nitrogen, $Ex_{nut}(N)$, in 1987–89 and 1993–95.

Ozone AOT40:

- AOT40 maps for forests in 1991–95.
- AOT40 maps for crops in 1991–95.

Ozone critical level exceedances:

- Exceedance of critical levels (AOT40) of ozone in 1991–95.

Actual corrosion rates for materials:

- Actual corrosion rates for bronze, copper, weathering steel, zinc, aluminum, tin, nickel, glass, limestone, sandstone, steel panels with silicon alkyd and coil-coated steel with alkyd melamine in 1993–95.

Exceedances of acceptable corrosion rates:

- Exceedance of acceptable corrosion rates for weathering steel, zinc, aluminum, copper, bronze, limestone and sandstone in 1993–95.

Economic costs of corrosion damage:

- Economic costs of corrosion damage for galvanized steel, zinc, aluminum, natural stones, paint coatings and plaster in 1993–95.

Detailed information and corresponding literature can be found at the Web site www.oekodata.com.

Calculation methods

In general the critical loads are calculated for the CCE data set in accordance to the methods described in the Mapping Manual (UBA 1996). Data for $CL_{max}(S)$, $CL_{min}(N)$, $CL_{max}(N)$ and $CL_{nut}(N)$ have been submitted as terms of the critical loads function. The calculation of $CL(S+N)$ differs slightly from the Mapping Manual method, considering the base saturation of soils with the aim to protect soils with a better supply of base cations.

Critical loads of sulfur and nitrogen:

To calculate critical loads of sulfur and nitrogen for forest soils equation 5.16 in the Mapping Manual was used. The calculation method of this approach is valid for acid forest soils. To take into account base soils as they occur in Germany, the base saturation was integrated into the estimation of critical loads. For all soil units with a base saturation > 30% the critical ANC leaching is set to zero. In sensitivity studies it turned out that the results of this classification are relatively robust concerning the choice of 30% as a cut-off value. Since soils with high base saturation tend to have high weathering rates and high ANC leaching values, their critical loads decrease by using this method. Without this assumption, the base saturation of all soils would decrease to values near 5% within a few decades. Since the aim of the critical loads approach is to protect all ecosystems against acidification, we feel that it is justified to preserve also those ecosystems adapted to a high base saturation of their soils. In this case the critical load is determined by the weathering of base cations.

Critical loads of nutrient nitrogen:

The methods of calculating critical loads of nutrient nitrogen are described in detail in the Mapping Manual (Equation 5.21).

Total deposition:

Total deposition was mapped by combining interpolated wet deposition measurements and inferential modeling of dry deposition. (See Figure DE-1.)

Exceedances:

The exceedance of critical loads by deposition in 1987–89 and 1993–95 is shown in Figure DE-2 for sulfur and nitrogen and in Figure DE-3 for nutrient nitrogen.

Ozone AOT40 values and exceedance of critical levels of ozone:

Ozone AOT40 maps have been compiled by calculating the AOT40 for forests and crops at each rural measurement site. The results were interpolated for forests and crops separately. By intersecting both maps with the distribution of forested and agricultural land the AOT40 values have been related to the corresponding receptor (see Figure DE-4).

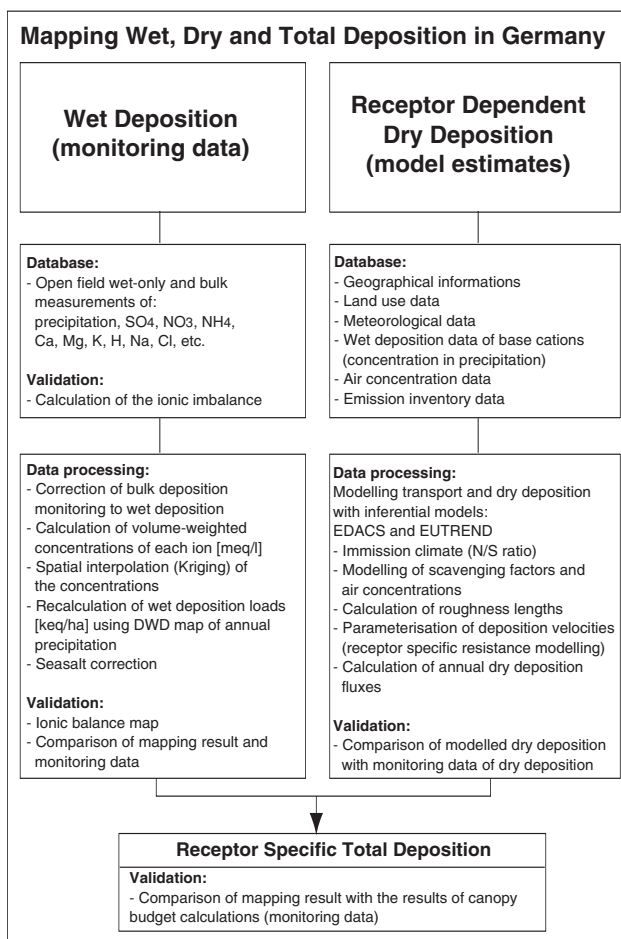


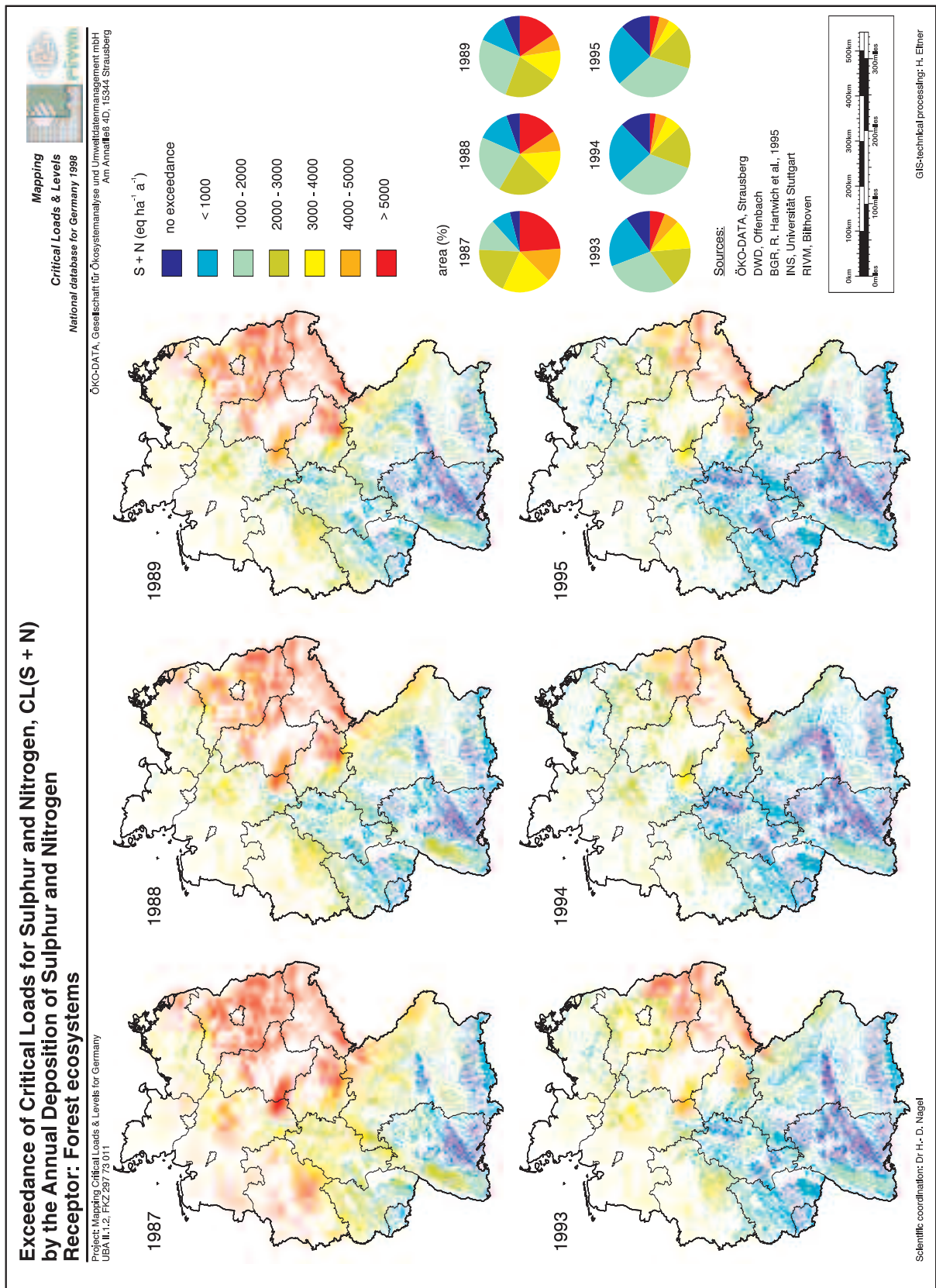
Figure DE-1. Deposition mapping process.

Mapping acceptable levels/loads for effects of air pollutants on materials:

Actual corrosion rates of 12 different materials have been calculated and mapped using the unified dose-response-functions (UBA 1999) for bronze, copper, weathering steel, zinc, aluminum, tin, nickel, glass, limestone, sandstone, steel panels with silicon alkyd and coil coated steel with alkyd melamine (see Figure DE-5). Corrosion attack can be described as yearly mass loss (g m^{-2}), but for some materials other units are more appropriate. For paint coatings, ASTM standards are used. Exceedances of acceptable corrosion rates are mapped. Finally, the economic costs of corrosion damage to materials in Germany are assessed.

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**Exceedance of Critical Loads for Nutrient Nitrogen, $CL_{nut}(N)$
by the Annual Deposition of Nitrogen
Receptor: Forest ecosystems**

Project: Mapping Critical Loads & Levels for Germany
UBA II.1.2, FKZ 297 73 011

**Mapping
Critical Loads & Levels
National database for Germany 1998**

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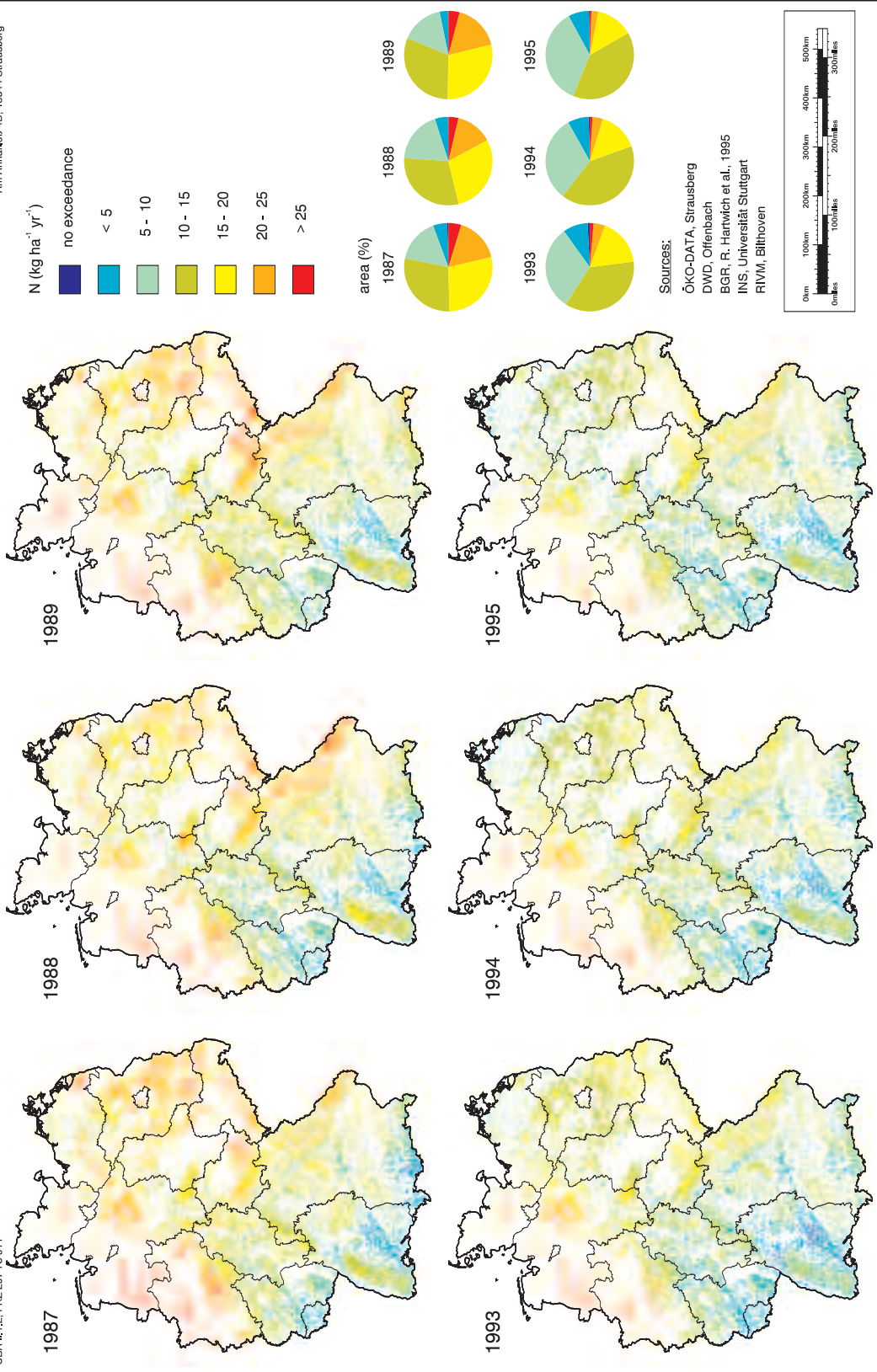


Figure DE-3. Exceedance of critical loads of nutrient nitrogen, $CL_{nut}(N)$.

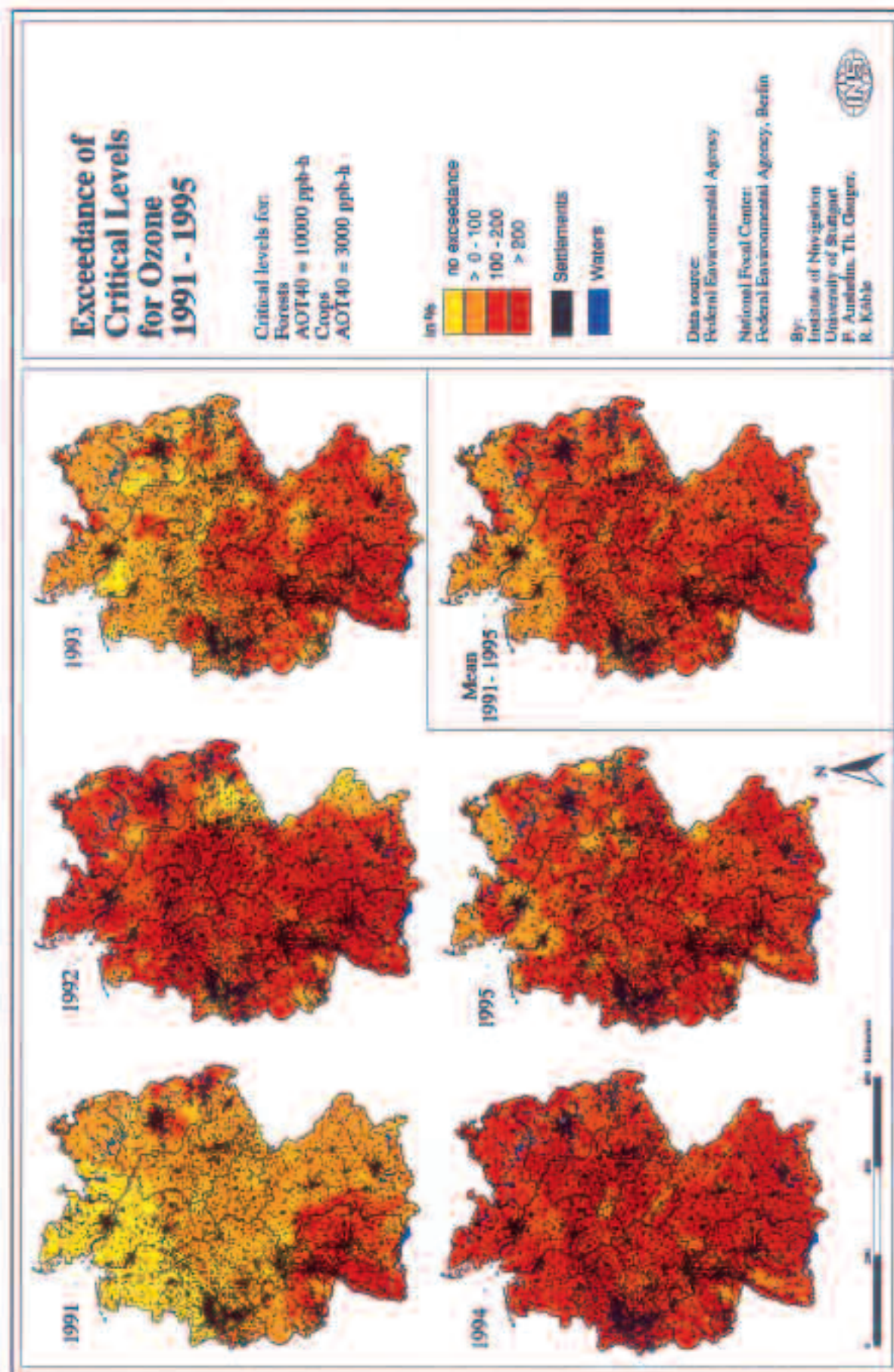


Figure DE-4. Exceedance of critical levels for ozone, 1991–1995.

Actual Corrosion Rates

Calculated with unified ICP-Materials dose-response-functions

Data source:
Federal Environmental Agency, Berlin

National Focal Center:
Federal Environmental Agency, Berlin

By:
Institute of Navigation
University of Stuttgart
F. Anshelm, Th. Gauger,
R. Köble

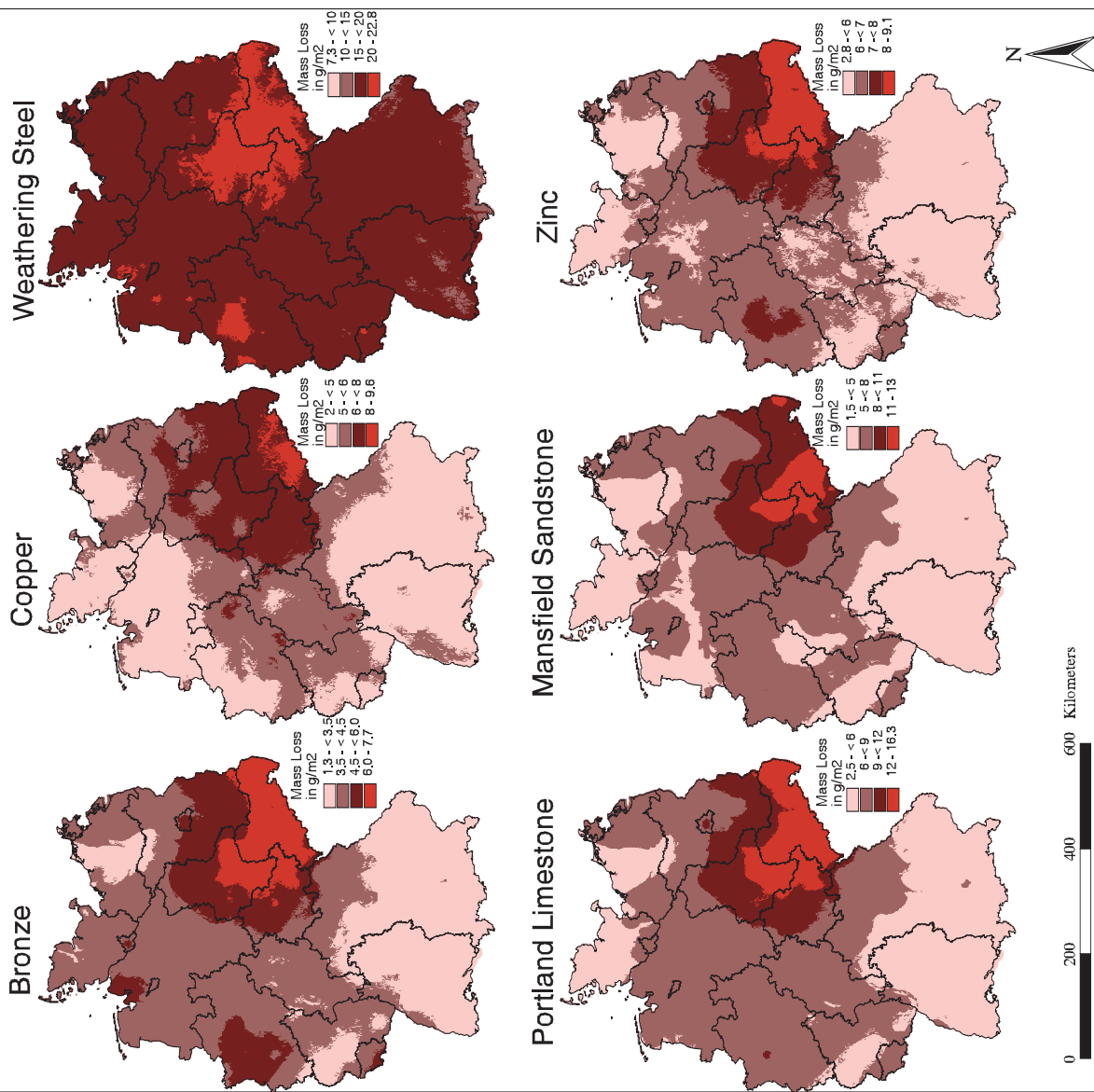


Figure DE-5. Actual corrosion rates.

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National maps produced

Important basic information required to calculate and map critical thresholds has been collected to describe the physical and chemical parameters of genetic soil types, ecosystems, and effects of agricultural activities. Most of these data were transformed into GIS, and maps produced on the following topics.

Soil:

- Soil vulnerability maps of Hungarian soils, nitrate leaching, (1:1,000,000), RISSAC, 1997.
- Soil vulnerability maps of Hungarian soils, acidification, (1:500,000), RISSAC, 1997.
- Solute concentration of selected ions at Hungarian background concentrations: Lead (1:1,000,000), RISSAC-NFC-OTAB, 1997
- Solute concentration of selected ions at Hungarian background concentrations: Cadmium (1:1,000,000), RISSAC-NFC-OTAB, 1997
- Heavy metal contamination of Hungarian soils: Nickel (1:1,000,000), based on AIIR, RISSAC, 1997.

Critical thresholds:

- Maximum critical load of nitrogen, 50×50 km² grid, RISSAC-NFC, 1997.

- Maximum critical load of sulfur, 50×50 km² grid, RISSAC-NFC, 1997.
- Critical loads of nutrient nitrogen, 50×50 km² grid, RISSAC-NFC, 1997.

Calculation methods

The steady-state mass balance approach has been applied (UBA 1996), combined with an empirical method. Figure HU-1 shows the soil vulnerability to acidification based on the Hungarian AGROTOPO data base.

No consensus concerning base cation deposition and base saturation could be achieved. The same problems occur in the case of both land use data and derived data such as nitrogen immobilization and uptake of base cations and nitrogen. Expert estimation was used to review available European data from countries with similar climate and soil. Figure HU-2 shows the soil vulnerability to nitrate leaching.

Due to the lack of new data, no new maps were provided to the CCE in time in 1998. Figures HU-3 and HU-4 show $CL_{max}(N)$ and $CL_{max}(S)$ based on the data presently used under the LRTAP Convention. National exceedance deposition maps have not yet been prepared, as the data are still not available for the Hungarian NFC in a proper format.

Regarding heavy metal mapping activities, the first maps of background concentration and solute concentration of selected ions have been produced (Figures HU-5 and HU-6).

Data sources

Although many activities occurred recently to improve environmental information management, the NFC's access to national data did not improve due to two main reasons:

- Data are not available or there are too fragmented (i.e., on a micro-regional scale)
- Data are owned by special institutes, and thus the information flow through the Ministry is not proper.

Data sources providing information in support of the LRTAP Convention protocols include:

Soils: Digital soil data of RISSAC and Hungarian Geological Institute (MÁFI). Maps include:

- Genetic soil types of Hungary (1:100,000), RISSAC, 1997

- Genetic soil types for Pest County (1:50,000), RISSAC, 1998.
- Soil Degradation Regions of Hungary (1:500,000), RISSAC, 1997.

Ecosystem and forest data: The Forest Management Institute (on a 16×16 km² monitoring network), CORINE Biotopes by the PHARE-supported HAS project; ecosystem threshold capacity calculated or moderated by the Research Institute for Ecology and Botany of the Hungarian Academy of Sciences.

Land cover and ecosystem maps:

- CORINE Biotope Map for Hungary, (1:100,000), MEP-Phare–Research Institute for Ecology and Botany of Hungarian Academy of Sciences, 1997-98.
- CORINE Landcover for Hungary, MEP-Phare Remote Sensing Centre, 1997.
- Biotopes for B-A-Z County, (1:50,000), EISD, 1998-99.

Land use data: CORINE land cover, the PHARE-supported MEP project, coordinator IRS-FÖMI.

Deposition data: The Institute for Environmental protection calculates 50×50 km² deposition data, but this data has not yet been integrated into the NFC information system due to the lack of proper cooperation.

Comments and conclusions

As the National Focal Center had no financial capacity to set up new integrated projects, it has tried to incorporate other projects' results, mainly in the field of mapping changes of heavy metal contamination for cadmium, nickel, arsenic and lead.

It was also decided to gather information on a 1×1 km² grid resolution for ecosystem and soil data. As a first step, three polluted areas will be evaluated and maps calculated for the northern part of the Lake Balaton region, the Budapest agglomeration and the Miskolc industrial area. Further research and model specification will take place for critical base cation uptake, critical nitrogen uptake, and nitrogen immobilization rates.

Ecologists are involved to specify acidification sensitivity in areas where:

- relatively large portions of the soils are not susceptible (in the Great Hungarian Plain) because forest and grassland damages occur there due to a variety of environmental pressures.
- very vulnerable protected species could be found outside of protected areas to achieve a preventive policy target.

Acknowledgements

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Vulnerability maps of Hungarian soils

Acidification

spatial resolution: 625 ha

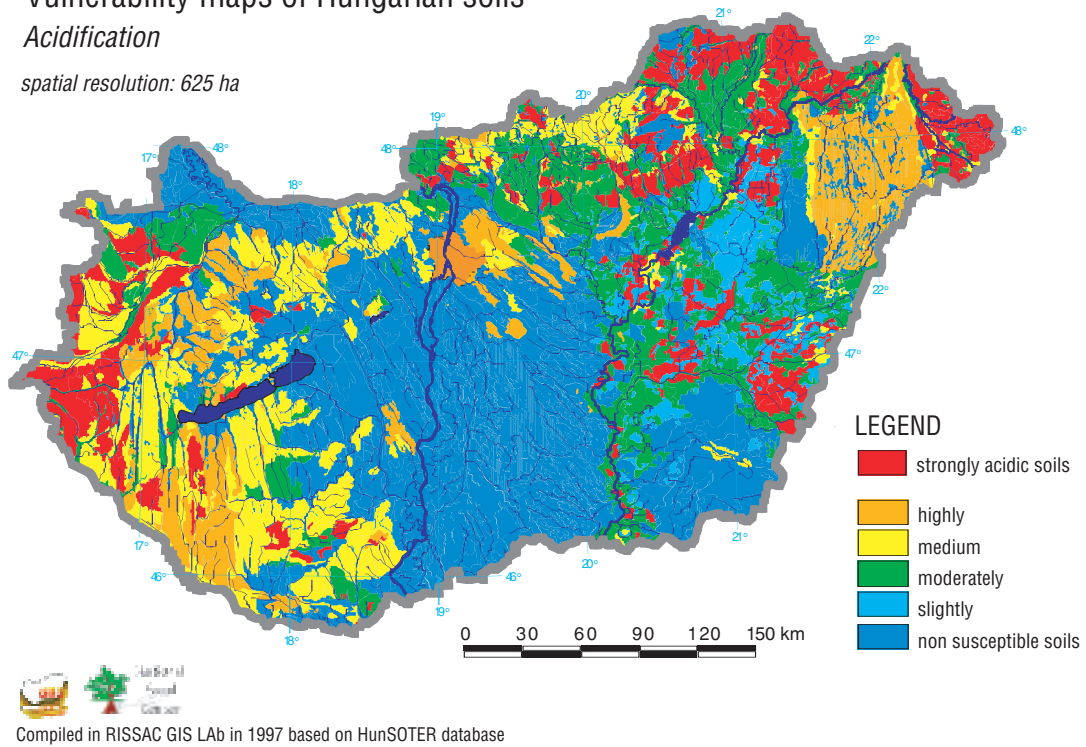


Figure HU-1. Vulnerability of Hungarian soils to acidification.

Vulnerability maps of Hungarian soils

Nitrate leaching

spatial resolution: 175 ha

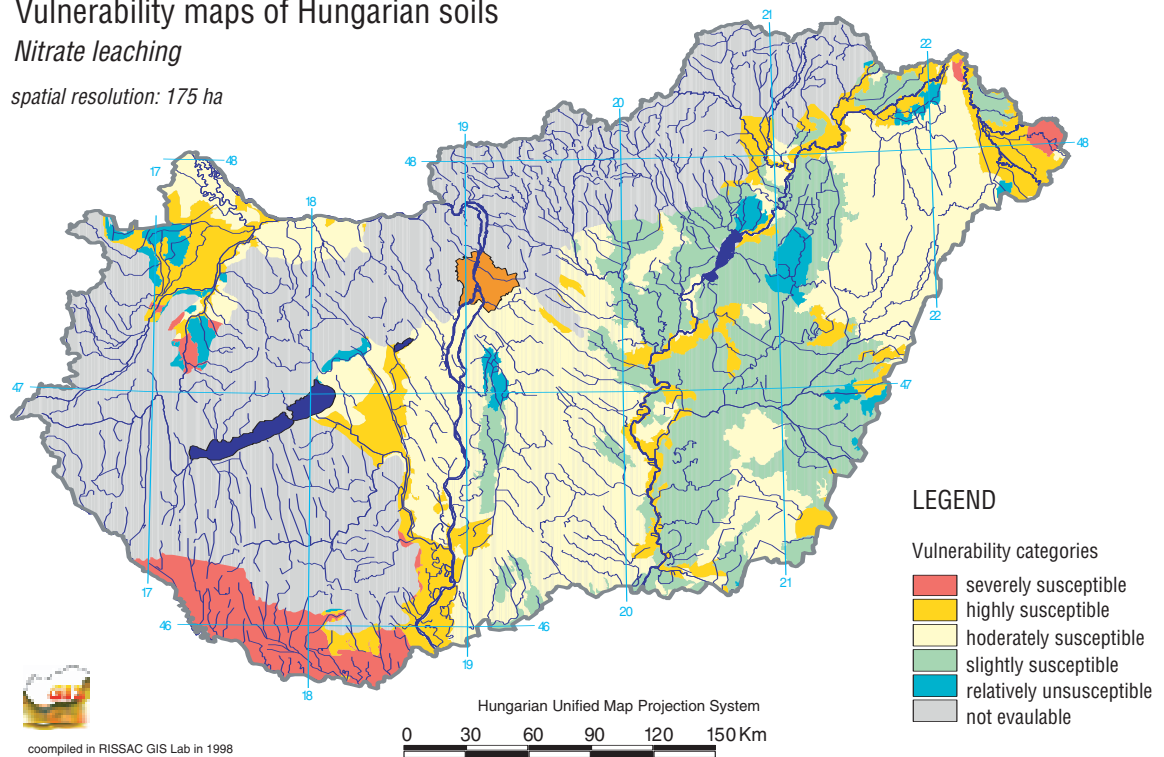


Figure HU-2. Vulnerability of Hungarian soils to nitrate leaching.

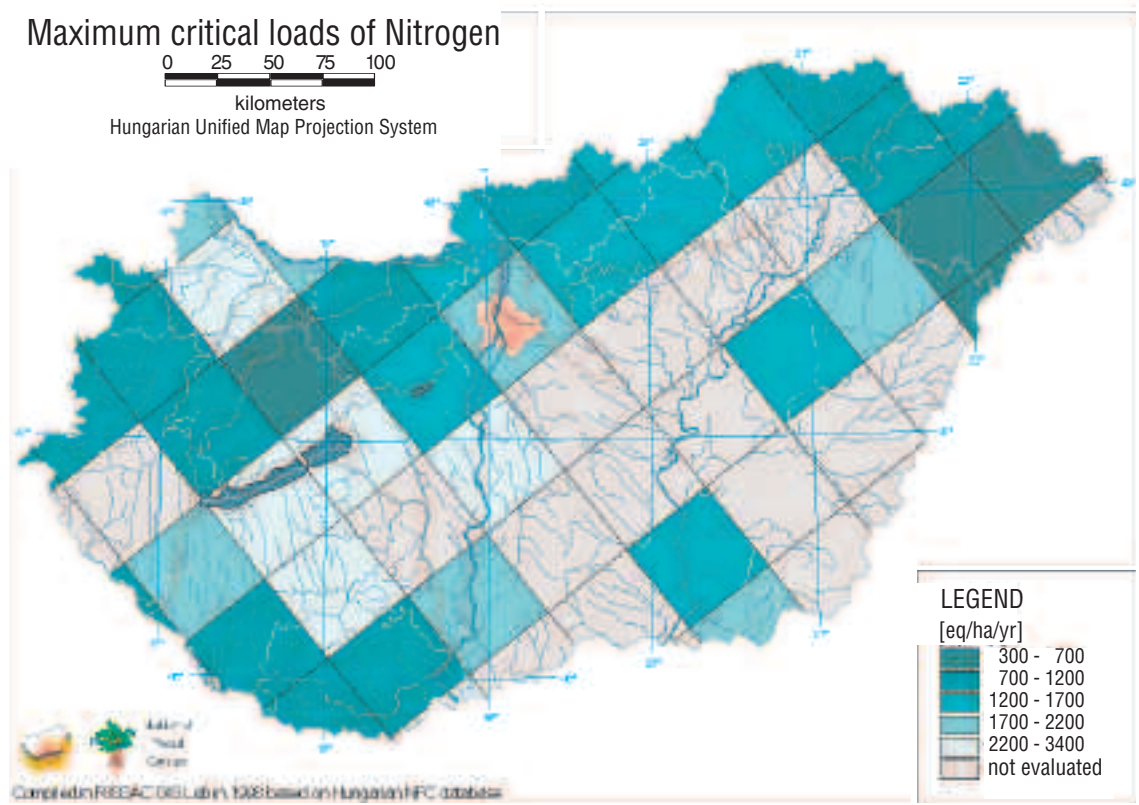


Figure HU-3. Maximum critical loads of nitrogen.

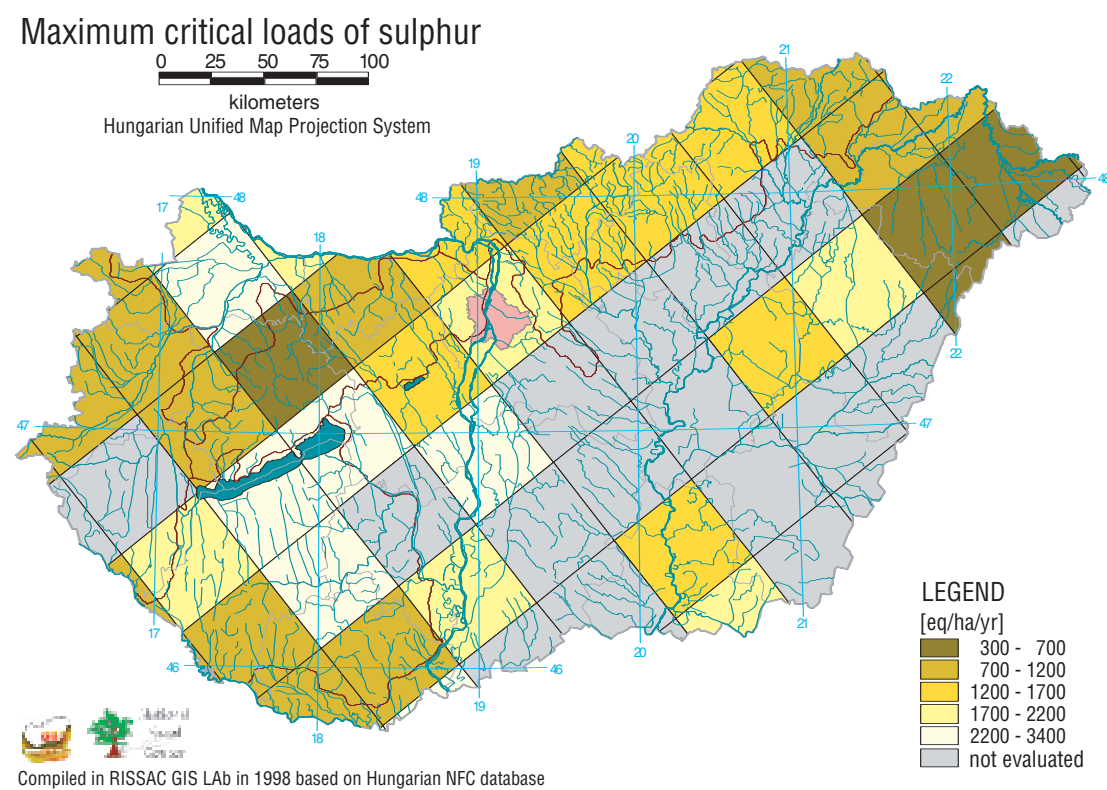


Figure HU-4. Maximum critical loads of sulfur.

Solute concentration of selected ions at Hungarian background concentration in 1:100,000 resolution

Lead

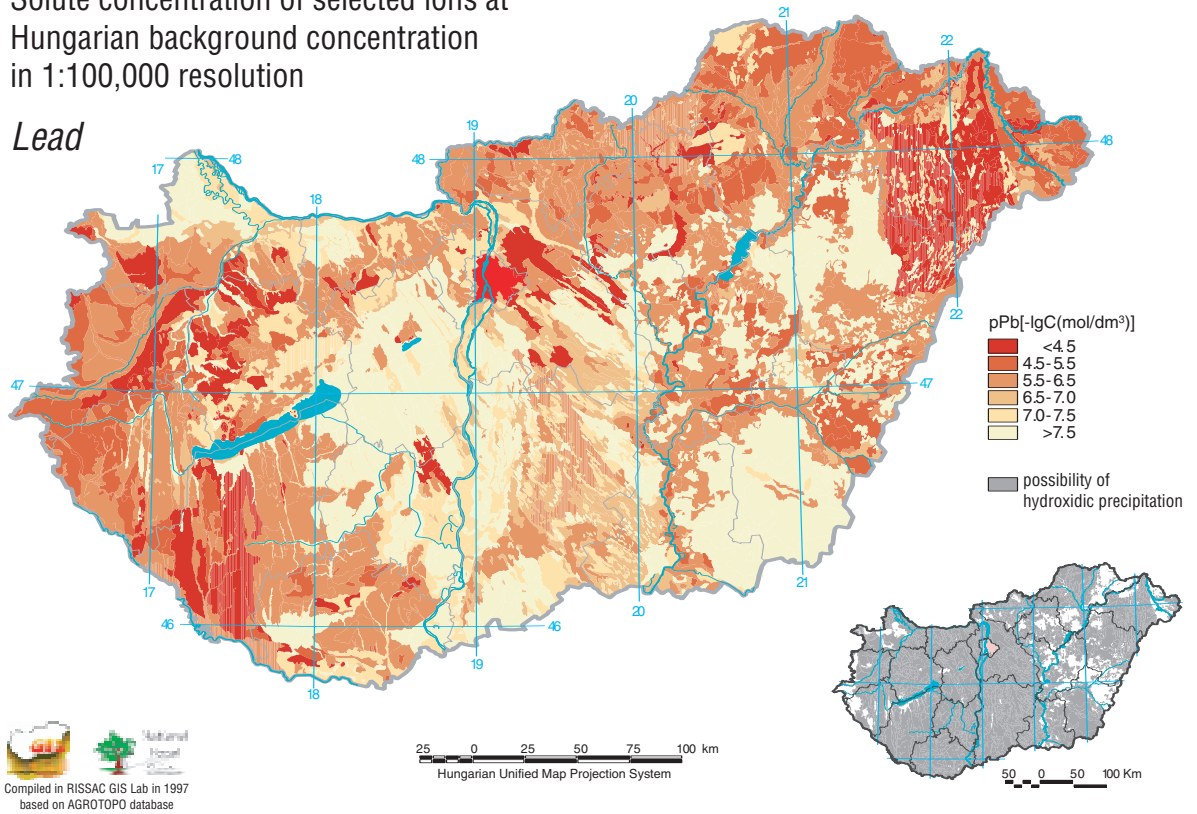


Figure HU-5. Background concentrations of lead in soil solution.

Solute concentration of selected ions at Hungarian background concentration in 1:100,000 resolution

Cadmium

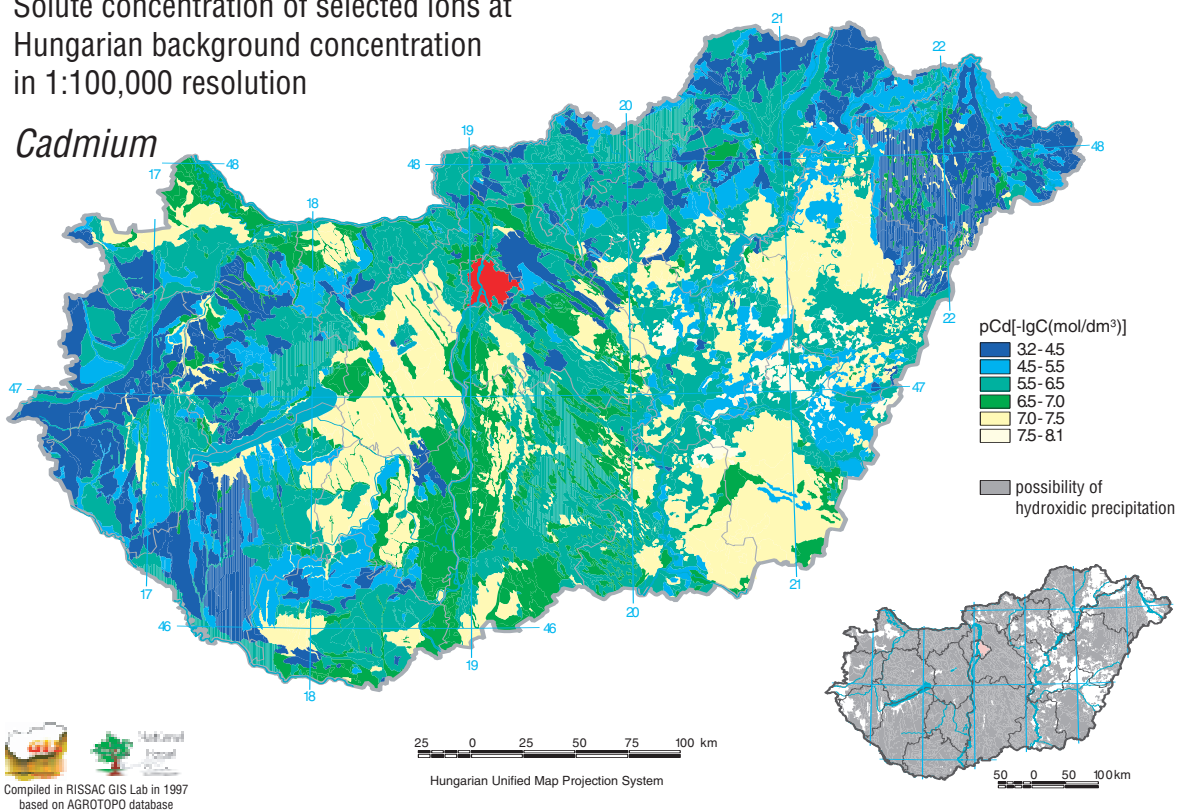


Figure HU-6. Background concentrations of cadmium in soil solution.

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National maps produced

Projection: Irish national grid, 1×1 km².

Receptor ecosystems: Coniferous forest, deciduous forest, moor and heathland, natural grassland and freshwater lakes. The CORINE land cover map for Ireland is used to define the distribution of the receptor ecosystems (Ordnance Survey of Ireland 1993). A data base is held which describes the percentage of each land cover type in every 1 km² (range 0.01–100%).

- Critical loads of acidity for soils.
- Maximum critical loads of sulfur.
- Minimum critical loads of nitrogen.
- Maximum critical loads of acidifying nitrogen.
- Critical loads of nutrient nitrogen (empirical and mass balance).
- Deposition of sulfur, nitrogen and base cations.
- Concentration and critical levels of ozone.

Calculation methods

The maximum critical loads of sulfur, minimum critical loads of nitrogen and maximum critical loads of acidifying nitrogen were calculated using:

$$CL_{max}(S) = CL(A) + BC_{dep} - BC_u$$

where:

$$CL(A) = ANC_w + ANC_{le(crit)}$$

The acid neutralizing capacity due to weathering (ANC_w) is based on a Skokloster classification (Nilsson and Grennfelt 1988, Hornung *et al.* 1995a) of the general soil map of Ireland (Gardiner and Radford 1980). By assigning a Skokloster critical load range to the principal soil of each association on the general soil map of Ireland, a map of ANC_w has been produced (Figure IE-1a). The critical ANC leaching, $ANC_{le(crit)}$, is calculated as described in Hettelingh *et al.* (1991), page 35. A pH of 4.2 was selected as the H⁺ concentration limit and subsequently used to estimate the Al³⁺ critical limit via the gibbsite relationship. The H⁺ critical limit of pH = 4.2 is based on work by Ulrich (1987). The critical loads for organic soils were estimated according to Cresser *et al.* (1993), as the Skokloster classification only considers mineral soils. Critical loads for peats are defined in terms of the acid deposition loads which would cause a specified pH reduction compared with pristine conditions (Cresser *et al.* 1993).

$$CL_{min}(N) = N_u + N_i$$

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S)$$

The empirical approach (UBA 1996) is used to calculate critical loads of nutrient nitrogen for natural grasslands (1790 eq ha⁻¹ yr⁻¹), moors and heathlands (1215 eq ha⁻¹ yr⁻¹) and freshwater lakes (715 eq ha⁻¹ yr⁻¹). For coniferous and deciduous forest ecosystems, the critical loads of nutrient nitrogen was calculated as the minimum of the mass balance approach and empirical approach, where the mass balance was estimated as:

$$CL_{nut}(N) = N_u + N_i + N_{le} / (1 - f_{de})$$

where f_{de} values were based on soil wetness: dry soils (0.3), moderate soils (0.5), gleys and peaty podsols (0.7) and peats (0.8).

The empirical values for coniferous and deciduous forests are set at 1790 and 1215 eq ha⁻¹ yr⁻¹ respectively. Results from the empirical and mass balance approach were merged into one data set by selecting the minimum value of both methods on a cell by cell basis for the 1×1 km² mapping grid (Figure IE-1b). Exceedances of nutrient nitrogen are calculated as present loads minus critical loads (Figure IE-1c).

Data sources

Soils: 1:575,000 general soil map of Ireland and the accompanying Soil Survey Bulletin (Gardiner and Radford 1980).

Land cover: 1:100,000 CORINE land cover project, Ireland (Ordnance Survey of Ireland 1993).

Precipitation: Interpolation (kriging) of long-term average annual precipitation volume for approximately 600 sites between the period 1951–1980 (Fitzgerald 1984).

Precipitation surplus: Estimated as rainfall minus evapotranspiration and surface runoff. Evapotranspiration is estimated from interpolation (kriging) of long-term average annual evapotranspiration volume, 1951–1980. Surface runoff is inferred from soil permeability classes derived from the general soil map of Ireland (Gardiner and Radford 1980).

Deposition: Combination of rainfall with interpolated (kriging) average annual bulk precipitation chemistry concentrations for approximately 20 sites between the period 1985–1994. The minimum sampling period is not less than 3 years. Total base cation deposition was estimated using a filter factor of 2 to scale from bulk deposition to total deposition to forests and 1.5 for total deposition to moors and heathlands.

Weathering rate: Estimated using Skokloster classification ranges assigned to the principal soil of each soil association on the general soil map of Ireland (Gardiner and Radford 1980). The midpoint of each of the five classes is used to define soil weathering, except for the final (non-sensitive) class, set at 4000 eq ha⁻¹ yr⁻¹.

Uptake: Base cation uptake for coniferous ecosystems is estimated as the minimum of ($BC_{available}$, BC_u), where $BC_{available}$ is the available base cation flux estimated according to:

$$BC_{available} = (BC_w + BC_{dep} - BC_{le})$$

BC_{le} is set equal to 0.02 eq m⁻³. BC_u is calculated using a yield class of 16 m³ ha⁻¹ yr⁻¹, a wood density of 390 kg m⁻³ and stem concentrations of Ca²⁺ = 0.056%, Mg²⁺ = 0.021% and K⁺ = 0.0665%. It was assumed that all coniferous trees are Sitka spruce, the yield class is the average yield class for Sitka spruce in Ireland (COFORD 1996) and the stem concentrations are for Sitka spruce in Wales (Emmett and Reynolds 1996). For deciduous forests, natural grasslands and moors and heathlands a BC uptake of 45 eq ha⁻¹ yr⁻¹ was selected to account for uptake by grazing. Nitrogen uptake for coniferous ecosystems is estimated using the same method as for base cations with a stem concentrations of N = 0.05%. For deciduous forests, natural

grasslands and moors and heathlands, an N uptake of 71 eq ha⁻¹ yr⁻¹ was selected to account for uptake by grazing.

Nitrogen immobilization: According to previous mapping guidelines (Downing *et al.* 1993) the nitrogen immobilization at critical load can be approximated by the long-term, natural immobilization of 2–5 kg N ha⁻¹ yr⁻¹, which is assumed to be net immobilization, including fixation. Hornung *et al.* (1995b) suggest 1–3 kg N ha⁻¹ yr⁻¹ depending on warm–cold climate for coniferous and deciduous forests and 0.5–2 for acid grassland. The following values were used:

$N_i = 3$ kg N ha⁻¹ yr⁻¹ for organic and podzolic soils, and $N_i = 2$ kg N ha⁻¹ yr⁻¹ for all other soils.

Immobilization classes were defined using the general soil map of Ireland (Gardiner and Radford 1980).

Acceptable nitrogen leaching: Downing *et al.* (1993) suggest an acceptable leaching of 2–4 kg N ha⁻¹ yr⁻¹ for coniferous and 4–5 for deciduous forests. Hornung *et al.* (1995b) suggest 1–4 kg N ha⁻¹ yr⁻¹ depending on low–high water surplus for coniferous and deciduous forests and 1–3 for acid grassland. The following values were used: $N_{le(acc)} = 3$ kg N ha⁻¹ yr⁻¹ for coniferous forests, and $N_{le(acc)} = 4$ kg N ha⁻¹ yr⁻¹ for deciduous forests.

Ozone: Concentration maps are derived from interpolation (kriging) of annual average monitoring site data (7 sites; 1995–1997). The AOT40 for crops is the Accumulated exposure Over a Threshold of 40 ppb. The critical level for crops is expressed as a maximum accepted cumulative dose over the AOT40.

For crops the AOT40 is calculated for a three-month period (May–July) during daylight hours, when crops are assumed to be most sensitive to ozone, Figure IE-1(d). Daylight hours are defined as those hours with a clear-sky global radiation of 50 W m⁻² or greater. The critical level is set at 3000 ppb-h during this three-month period (UBA 1996). For forests it is more difficult to demonstrate linear relationships between yield decline and cumulative exposure. The critical level is set at 10,000 ppb-h during a six-month period from April–September, also during daylight hours (UBA 1996).

Comments and conclusions

Since the Irish critical load mapping program began in 1996 considerable advances have been made in the application of the critical load concept to Ireland. Currently critical loads have been mapped for five receptor ecosystems (coniferous forests, deciduous forests, moors and heathlands, natural grasslands and freshwater lakes) representing 12.9% (9077 km²) of the land area of the Republic of Ireland.

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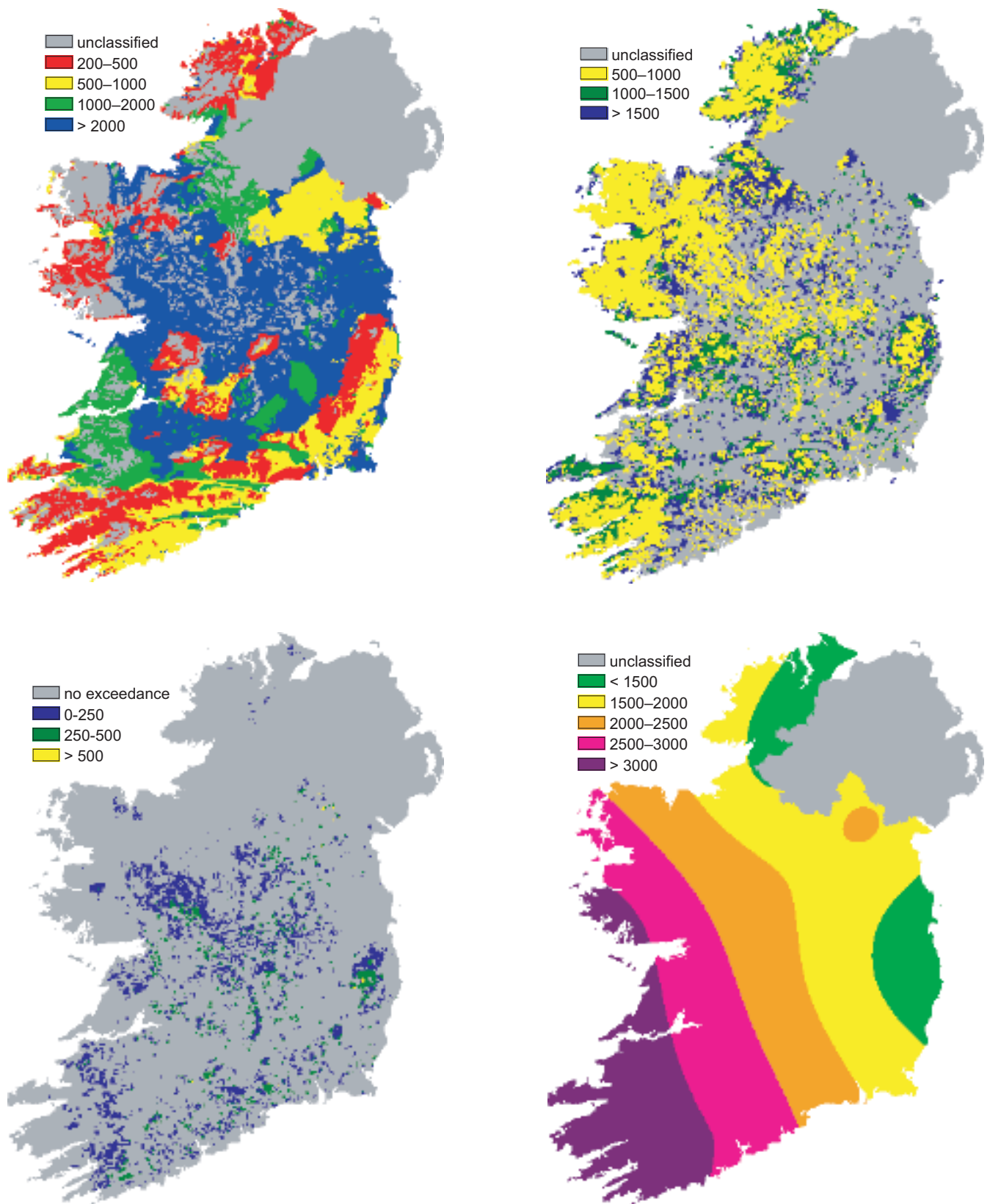


Figure IE-1. (a) top left: Acid Neutralizing Capacity due to weathering, ANC_w , according to the Skokloster classification ($\text{eq ha}^{-1} \text{yr}^{-1}$). Note: The Skokloster classification only considers mineral soils.
 (b) top right: Critical loads of nutrient nitrogen for peat bogs, moors and heathlands, and coniferous and deciduous forests ($\text{eq ha}^{-1} \text{yr}^{-1}$). Note: The minimum of the empirical and mass balance approaches was selected to represent the critical load for coniferous and deciduous forests.
 (c) bottom left: Exceedance of critical loads of nutrient nitrogen, estimated as the present loads minus critical loads ($\text{eq ha}^{-1} \text{yr}^{-1}$).
 (d) bottom right: Average May–July AOT40 for crops 1995–1997 (ppb-h). The critical level is set at 3000 ppb-h, exposures above this level exceed critical load.
Note: “Unclassified” refers to grids with no critical load estimates and regions outside the Republic of Ireland.

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National maps produced

- Critical load of nutrient nitrogen for forest soils
- Critical load of total acidity

The following changes have been incorporated since the 1997 Status Report:

- All maps have been produced using the Steady-State Mass Balance (SSMB) method.
- All maps have been produced on a 1×1 km² grid resolution.

Calculation methods

Critical loads of nutrient nitrogen and total acidity for forest soils have been mapped according to Eqs. 5.21 and 5.22, respectively, of UBA (1996):

$$CL_{nut}(N) = N_i + N_{fire} + N_{vol} - N_{fix} + N_u + \frac{N_{le}}{1 - f_{de}}$$

$$\begin{aligned} CL(S+N) &= CL(S) + (1-f_{de}) CL(N) \\ &= BC_{dep} - Cl_{dep} + BC_w - BC_u + (1-f_{de}) \\ &\quad \cdot (N_i + N_{fire} + N_{vol} - N_{fix} + N_u) - Alk_{le} \end{aligned}$$

Data sources

Land use: The "Map of Italian vegetation" 1:1,000,000 (Ministry of Environment 1992), including 54 different vegetation types. The vegetation types have been classified with the help of Italian experts to categorize ecosystems into seven types (Hornung *et al.* 1995, Ch. 4, Table 1): tundra, boreal forest, temperate coniferous, temperate deciduous, Mediterranean forest, acid grassland and other (moors, mountain shrubs, basophilous grasslands, zonal vegetation). Urban areas, agricultural lands and, for the time being, surface waters have been left out.

N_i : The nitrogen immobilization in the soil organic matter has been determined on the basis of climate: the warmer the climate is, the less N_i will be, and vice versa. Data for N_i have been assigned to each ecosystem into the grid cell by overlaying the "Map of Italian vegetation" with the "Bioclimatic map of Italy" 1:2,000,000 (Tomaselli *et al.* 1972).

N_{fire} : Data on nitrogen losses due to fires have been assessed on the basis of the provincial burned surface area (divided into wooded and non-wooded) for the years 1982–1993 carried out by the Italian Ministry of Agriculture and Forests. It was assumed that data refer only to areas below 1500 m altitude. Data for N_{fire} have been calculated for each ecosystem in the grid cell by combining data on vegetation type, altitude and burned surface area for each province.

N_{fix} : The nitrogen input by biological fixation has been determined on the basis of climate moisture: the dryer the climate is, the less N_{fix} will be, and vice versa. Data for N_{fix} have been calculated for each ecosystem in the grid cell by overlaying the "Map of Italian annual average precipitation for the years 1921–1950," 1:1,000,000 (Ministry of Public Works).

N_{vol} : The nitrogen losses to the atmosphere via ammonia volatilization seem to be significant just for some moist calcareous soils (UBA 1996) which occur only in the presence of temperate coniferous forests (Hornung *et al.* 1995, Ch. 4, Table 1); for all other cases they are considered negligible. Calcareous soil data have been derived on the basis of the 1:1,000,000 "Soil map of the European Communities" (Tavernier and Louis 1992), which includes 312 different soil types, 66 of which are present in Italy.

Soil moisture can be correlated to local average rain trends. Moist soils have been defined as those which receive more than 1500 mm precipitation per year. The greater the precipitation is, the moister the soil will be, and thus the greater the N loss by volatilization. Precipitation data have been derived on the basis of the "Map of Italian annual average precipitation for the years 1921–1950", 1:1,000,000 (Ministry of Public Works). Data for N_{vol} have been calculated for each ecosystem in the grid cell by combining data on vegetation type, precipitation and soil type.

N_{le} : The level of total annual nitrogen leaching from the rooting zone has been determined on the basis of infiltrated water: there will be less N_{le} when the infiltration is less and vice versa. Infiltrated water data can be determined by the following equation describing the hydrogeological cycle:

$$I = P - E - R$$

where:

I = infiltrated water, in mm yr⁻¹

P = precipitation, in mm yr⁻¹

E = evapotranspiration, in mm yr⁻¹

R = surface runoff, in mm yr⁻¹

The actual evapotranspiration (E) has been calculated using the Turc formula:

$$E = \frac{P}{\sqrt{0.9 + \frac{P^2}{L^2}}}$$

where:

$$L = 300 + 25 T + 0.05 T^3$$

T = annual mean temperature, in °C

Temperature data have been derived on the basis of the "Map of annual average real temperature for the years 1926–1955", 1:1,000,000 (Ministry of Public Works).

Precipitation, vegetation and texture class for soil type influence the surface runoff (R). Texture class data have been derived on the basis of the 1:1,000,000 "Soil map of the European Communities" (Tavernier and Louis 1992).

These values were changed on the basis of the indications of Mapping Vademecum (Hettelingh and de Vries 1992). Data for R have been calculated for each ecosystem into the grid cell by using the following equations (Benini 1990):

for wood:

$$R = \begin{cases} P \cdot 0.13 & \text{for texture class 1} \\ P \cdot 0.21 & \text{for texture class 2 or 4} \\ P \cdot 0.36 & \text{for texture class 3} \\ P \cdot 0.28 & \text{for texture class 5 or 6} \end{cases}$$

and for grassland:

$$R = \begin{cases} P \cdot 0.16 & \text{for texture class 1} \\ P \cdot 0.36 & \text{for texture class 2 or 4} \\ P \cdot 0.62 & \text{for texture class 3} \\ P \cdot 0.41 & \text{for texture class 5 or 6} \end{cases}$$

Data for N_{le} have been calculated for each ecosystem into the grid cell by combining data on vegetation type, precipitation and infiltration.

f_{de} : The denitrification fraction values have been related to the soil type on the basis of data in UBA 1996: $f_{de}=0.5$ for sandy soils with gleyic features; $f_{de}=0.7$ for clay soils; $f_{de}=0.6$ for slimy-sandy soils with gleyic features (this value is not present in the Mapping Manual, but it was introduced to address some Italian characteristics); and $f_{de}=0.1$ for all other soils. The soil classification has been taken from the 1:1,000,000 "Soil map of the European Communities" (Tavernier and Louis 1992).

N_u and BC_u : Data on net removal of nitrogen and base cations (Ca, Mg, and K) in vegetation have been assessed on the basis of provincial wood utilization statistics for coniferous and deciduous forests for the years 1989–1993, carried out by the Italian National Institute of Statistics (Istat). Dividing these values by provincial surface area covered by coniferous or deciduous forests, the provincial utilization (in m³ ha⁻¹ yr⁻¹) was obtained. As many Italian forests are young (National Forest Inventory 1985), harvesting rates will increase in the future. For this reason the provincial utilization data have been increased on the basis of a sustainable criterion (obtaining the so-called harvesting rates). Data for N_u and BC_u have been calculated multiplying the harvesting rates with nitrogen and base cations contents of the stems and branches, considering also wood density and ratio of branches to stems (Hettelingh *et al.* 1991).

BC_{dep} : Wet non-marine deposition data for base cations (Ca, Mg, and K) have been derived from 39 ENEL monitoring station for the period 1988–1992, using a kriging method to obtain values for each grid cell. Since data for dry deposition is not available, the total base

cation deposition, including a dry deposition fraction, has been estimated as follows (Downing *et al.* 1993, Appendix II):

$$BC_{dep} = \begin{cases} 2 \cdot BC_{wet} & \text{if } BC_{wet} < 250 \text{ eq ha}^{-1}\text{yr}^{-1} \\ 250 + BC_{wet} & \text{otherwise} \end{cases}$$

Cl_{dep} : Wet deposition data for chloride (and Na) have been derived from 39 ENEL monitoring station for the period 1988–1992, using a kriging method to obtain values for each grid cell. The total chloride deposition has been estimated by applying a sea-salt correction on wet chloride deposition (UBA 1996):

$$Cl_{dep} = Cl_{wet} - Na_{wet} \cdot (Cl_{sw} / Na_{sw})$$

where $Cl_{sw} / Na_{sw} = 1.164$.

BC_w : The weathering rate of base cations (Ca, Mg, K) has been estimated for the main root zone. The Mapping Vademecum (Hettelingh and de Vries 1992) proposes to estimate the weathering rates from soil type information on the FAO-UNESCO soil map of Europe. It was decided to adapt this approach to the 1:1,000,000 "Soil map of the European Communities" (Tavernier and Louis 1992). But the weathering rate thus obtained assumes a soil temperature of 283 Kelvin, and a main root zone of 50 cm (FOEFL 1994). In order to reflect different conditions present in Italy, two modifications have been incorporated into the calculations:

- the weathering rate decreases as main root zone becomes thinner.
- the weathering rate decreases as soil temperature decreases.

The main root zone decreases significantly as altitude increases. Thus the following altitude dependence formula was used to calculate depths of the main root zone (FOEFL 1994):

main root zone = 50 cm, < 1600 m a.s.l.

main root zone = 35 cm, between 1600 and 2000 m a.s.l.

main root zone = 20 cm, > 2000 m a.s.l.

Soil temperature has been calculated as a function of altitude and climate zone applying the following formula:

$$K = \begin{cases} T + 273 & \leq 300 \text{ m a.s.l.} \\ 283.5 - 0.003 \cdot (H - 600) & 300 < H \leq 1700 \text{ m a.s.l.} \\ 280.1 & > 1700 \text{ m a.s.l.} \end{cases}$$

where:

K = soil temperature at a depth of 0.2 m, in K

T = air temperature, in °C

H = altitude, in m

Data for BC_w have been calculated for each ecosystem into the grid cell by applying the following formula (FOEFL 1994):

$$BC_w(D, K) = BCW \cdot \frac{D}{50} \cdot 10^{-3800 \left(\frac{1}{K} - \frac{1}{283} \right)}$$

where:

$BC_w(D, K)$ = weathering rate corrected by soil type, depths of the main root zone and soil temperature

BCW = weathering rate corrected by soil type only

D = depths of the main root zone, in cm

K = soil temperature, in K

Alk_{le} : The alkalinity leaching has been calculated by the following (UBA 1996):

$$Alk_{le} = -I ([Al] + [H])$$

where:

I = infiltrated water, in $\text{m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$

$[Al]$ = aluminum concentration, in eq m^{-3}

$[H]$ = hydrogen concentration, in eq m^{-3} , obtained by the relationship:

$$[H] = \left(\frac{[Al]}{K_{gibb}} \right)^{\frac{1}{3}}$$

where K_{gibb} is an equilibrium constant dependent on soil organic matter percentage.

Comments and conclusions

A map of $CL_{nut}(N)$ on a $1 \times 1 \text{ km}^2$ grid is shown in Figure IT-1.

Presently work is being carried out on:

- Calculation and mapping critical loads of nutrient nitrogen for marine ecosystems. A case study on the Venice Lagoon has been developed (Sarti 1998). The Steady-State Mass Balance method and a $1 \times 1 \text{ km}^2$ resolution were used. Some results are shown in Figure IT-2.
- Calculation of critical loads for 55 alpine lakes in the Canton of Ticino. Chemical characteristics of each lake have been studied, and 90% of them show high sensibility to acidification and in particular 46% of them has an alkalinity value < 20 meq l^{-1} (Boggero *et al.* 1998).
- Calculation and mapping of critical loads of heavy metals (Barilli 1999).

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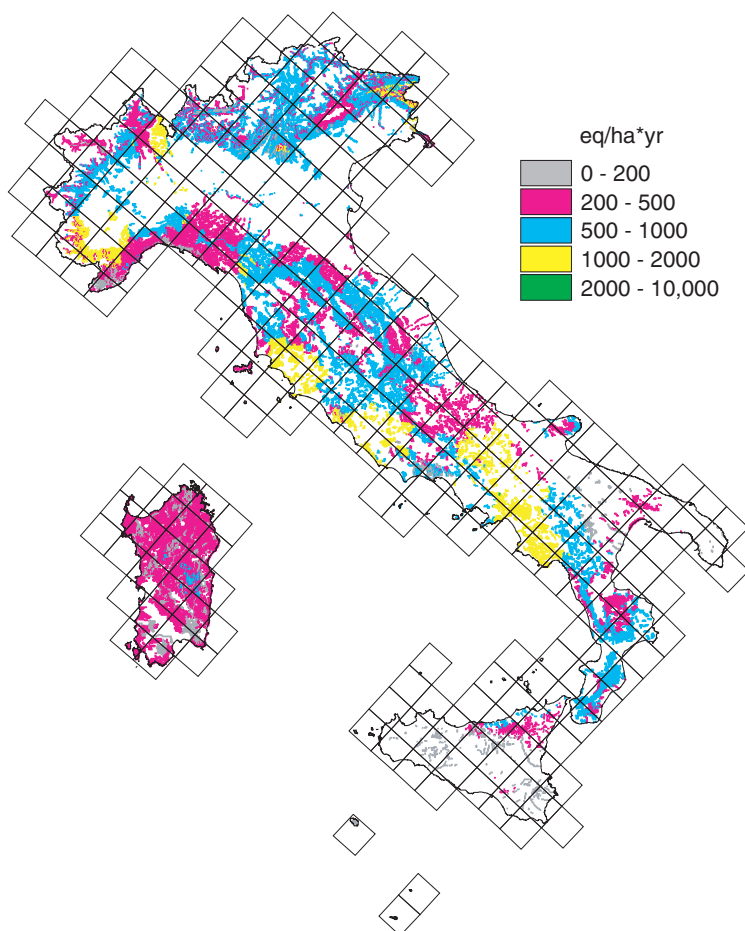


Figure IT-1. Critical load of nutrient nitrogen ($\text{eq ha}^{-1} \text{yr}^{-1}$) on a $1 \times 1 \text{ km}^2$ grid resolution.

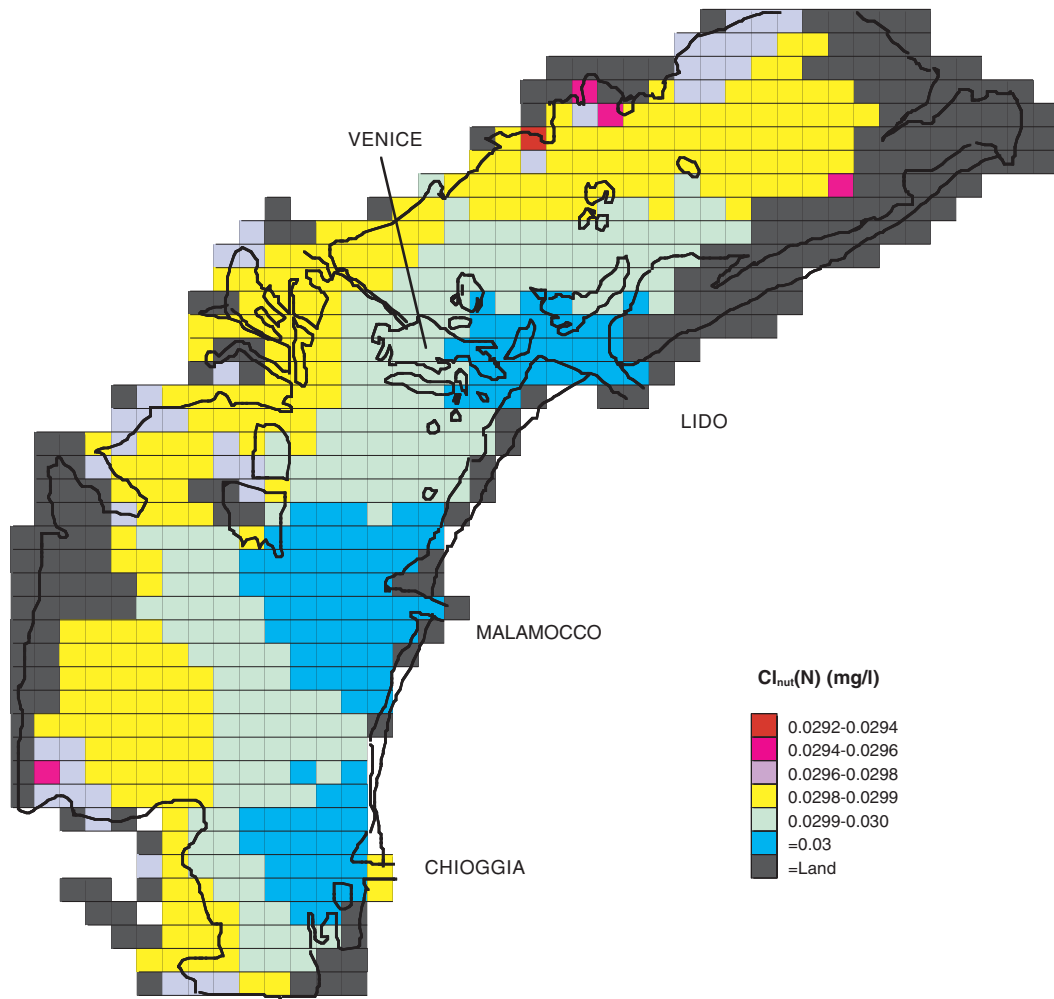


Figure IT-2. Venice Lagoon: Critical load of nutrient nitrogen on a 1x1 km² resolution.

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National maps produced

Since publication of the CCE Status Report 1997, no new maps have been submitted to the CCE. However, critical loads are being updated in 1999 as part of an evaluation of Dutch acid rain abatement strategies. The methods and the data used for these critical load calculations differ slightly from those described in the 1997 CCE Status Report (de Vries 1997). These changes are described in the sections below. It is intended to send the results of these calculations to the CCE after a thorough sensitivity analyses.

Calculation methods

In the 1997 CCE Status Report, the critical N loads were limited to impacts on the forest understory, while allowing only a very low N leaching flux ($100 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$) from the system. At present, the critical loads of nitrogen are calculated for three different aspects: (i) the impacts on biodiversity of the forest understory (vegetation changes), (ii) contamination of groundwater by nitrate and (iii) forest growth. In all three cases, the nitrate leaching flux has been calculated differently.

- In the case of vegetation changes, a constant low flux of $100 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ has been used (see also CCE Status Report 1997).

- With respect to groundwater contamination, the nitrate leaching flux has been calculated by multiplying the precipitation excess with the target value of nitrate in groundwater in the Netherlands, which is $0.4 \text{ mol}_c \text{ m}^{-3}$ or 25 mg l^{-1} .
- In case of forest growth, a relationship has been derived between the optimal N content in forest in relation to growth and the nitrate concentration in soil water, using both literature and empirical data at 150 forest stands in the Netherlands. The optimal N content has been chosen such that adverse effects of elevated N contents, such as increased sensitivity to frost, drought and disease are small. As an alternative, the nitrate leaching flux has been estimated from the amount of N mineralized and deposited on the forest during the dormant winter season. Ultimately, the minimum value of both estimates was used to derive a critical load.

As with nitrogen, calculation of the critical leaching flux (in this case, of acidity) has changed when calculating critical acid loads. In the 1997 version of the critical load calculations for acidity, two criteria were used to calculate $Al_{le(crit)}$: a criterion for the molar $Al/(Ca+Mg+K)$ ratio in the root zone and a negligible depletion of Al hydroxides. The first criterion aims to prevent root damage to plants, while the second criterion is meant to avoid degradation of soil quality. These two criteria protect soils with a low base saturation in which excessive acid deposition mainly leads to the release of Al. However, a considerable decrease in base saturation may take place in soils with a higher base saturation, such as loess and clay soils.

To avoid a decrease in base saturation in those soils with a high base saturation, a third criterion was added in which the critical acidity leaching is equal to the present acidity leaching. The present acidity leaching is calculated from the actual pH of the soils considered and the gibbsite equilibrium. Using this approach, the calculated critical acid loads for loess and clay soils are not much higher than for the sandy soils, since the weathering rates for the above-mentioned soils were calculated at higher pH values (see below).

In addition, separate critical loads of acidity are calculated to avoid the contamination of groundwater by aluminum. As with N, this was done by multiplying the precipitation excess with the target value of Al in groundwater in the Netherlands, set at $0.02 \text{ mol}_c \text{ m}^{-3}$. Furthermore, the weathering rate was changed in this application, allowing for the release of base cations in the subsoil until the depth of ground water extraction (32 m on average).

Data sources

Application of the SMB model:

Compared to 1997, the application methodology of the Simple Mass Balance (SMB) model has been changed. SMB now calculates critical loads at a 250×250 m² grid scale, but the number of soil-vegetation combinations distinguished has been slightly reduced. The number of tree species and soil types and the number of grid cells is now equal to the application of the dynamic soil acidification model SMART (Kros *et al.* 1995). This implies that SMB distinguishes only three groups of tree species: deciduous, pine and spruce. Soil types are now differentiated in 2 non-calcareous sandy soils, one calcareous sandy soil, three loess soils, four non-calcareous clay soils, one calcareous clay soil and five peat soils. All these soil types are, however, further subdivided in five hydrological classes depending on the height and the seasonal fluctuations of the water table. This was done by using a 1:50,000 digitized soil map, instead of a 1:250,000 soil map used in the previous application. Moreover, parameter values used are adapted in such a way that comparable values are used for the same processes in SMB and SMART.

Information on the area and distribution of each specific forest-soil combination in a 250×250m² grid cell was derived from an overlay of the 1:50,000 soil map mentioned above, and a detailed vegetation map based on satellite observations (LGN, Thunissen *et al.* 1992).

Weathering rates:

In the 1997 calculations, the weathering rates for loess, clay and peat soils were derived from the literature. In the 1999 version, weathering rates for these soils are calculated from pedotransfer functions relating the weathering rates to the silt and clay contents of the soils (van der Salm *et al.* 1998, van der Salm 1999). These pedotransfer functions are based on laboratory experiments for loess and clay soils, and separate transfer functions are used for loess and clay soils. The weathering rates predicted with the pedotransfer functions were validated against field weathering rates for loess soils. Weathering rates for clay soils could not be validated due to the lack of measurements of field weathering rates for clay soils. Weathering rates for peat soils were estimated using the pedotransfer functions for clay soils and the clay content of the peat soils.

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National maps produced

Receptors mapped include surface waters, forest soils, and vegetation.

The data base and methods used to calculate critical loads was described in detail in the CCE Status Report 1997 (Posch *et al.* 1997), and have not changed since.

Ongoing Work

A. Binding grid cells in Norway: An evaluation

Information has been requested from the National Focal Centers (NFC) on the background of ecosystems in EMEP grid cells which turn out to be crucial in the optimization of abatement strategies used for WGS negotiations (“binding grid cells”). Such information has been requested for 25 grids in Europe. Of these, six are located wholly or partly in Norway: grids (16,20); (16,21); (16,30); (17,19); (17,20) and (18,21) (Figure NO-1). We have analyzed the reliability of the data for the binding grid cells, especially the data quality for the Birkenes grid (17,20) (Henriksen 1998).

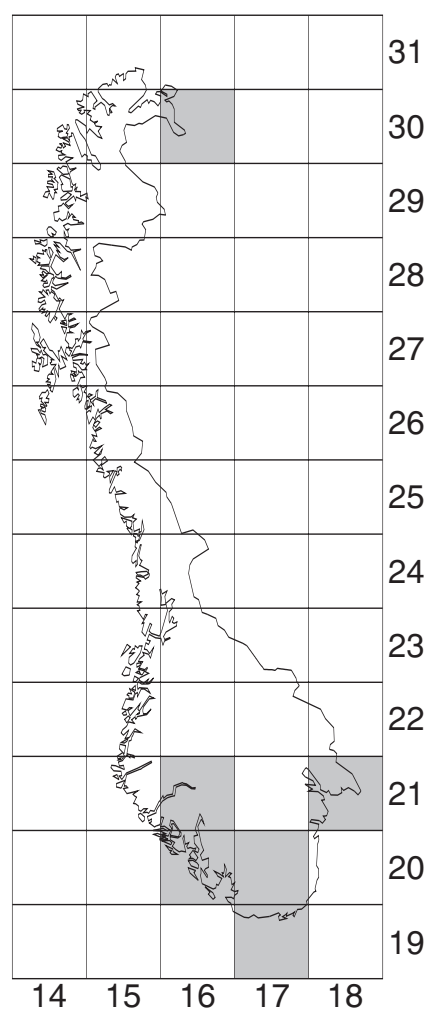


Figure NO-1. EMEP grid cells covering Norway. The “binding” grid cells are shaded.

Characteristics of ecosystems in binding grid cells:

Decline in fish populations in Norway was recorded as early as in the 1910s–1920s, but the link between the fish decline and the increasing acidity of precipitation was established as late as around 1970. Since then acidification of freshwaters has been considered as Norway's most serious environmental problem. The Birkenes grid (17,20) covers the most acidified areas of Norway, containing highly sensitive waters and receiving large amounts of acidic deposition.

Since surface water is the most sensitive ecosystem in Norway (see Figure NO-2), most work has been focused on this ecosystem. Therefore, national maps and calculations for exceeded areas and analyses for effects of future deposition scenarios for Norway are based on the surface water data base.

The CCE calculations for Norway, however, also include soil data provided by the NFC. Low-percentile critical load values for each EMEP grid for both lakes and soils, and for lakes alone, are very similar. This implies that the soil data does not influence critical load distributions significantly in the low range. This is to be expected, as the critical loads of soils are higher than those for surface water for 93% of the grids (Figure NO-2).

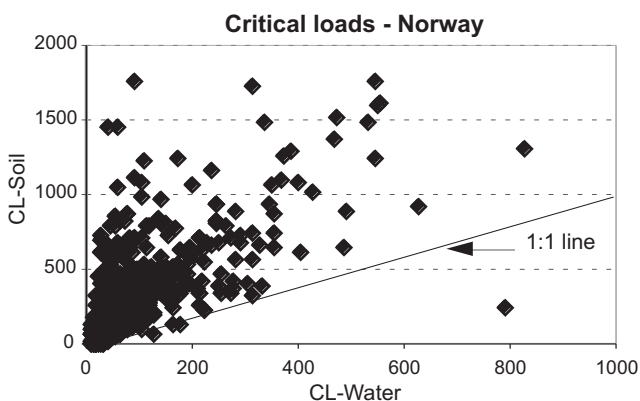


Figure NO-2. Comparison of critical loads for surface water and for forest soils in "NIVA grids" (n=720) where both ecosystems are present.

Each EMEP grid contains a number of "NIVA grids" (each 1° longitude × 0.5° latitude grid cell divided into 16 sub-grids) depending of the size of the grid. For each NIVA grid cell, we have assessed representative water chemistry by selecting a lake located in that grid (see Posch *et al.* 1997). Thus, a number of lakes corresponding to the number of NIVA grid cells within an EMEP cell has been selected to represent that EMEP grid cell. These lakes were selected largely from national lake surveys

carried out in 1986 and 1995 (Henriksen *et al.* 1987, 1998), representing approximately 2000 lakes all over Norway. For all EMEP grid cells there are more lakes available than NIVA cells, and for the Birkenes grid (17,20) we have available a total of 499 lakes (11% of the lake population > 0.04 km² in the grid) to cover 112 NIVA grid cells.

We have calculated percentile values for the CCE data base and our lake data base for critical loads of acidity and compared them (Figure NO-3). For the lower percentiles, the lake data base shows lower values than the data submitted to the CCE. Of special interest is the Birkenes grid (17,20) which covers the area of Norway most subject to acidification, and which had the earliest reliable records of fish kills and fish decline. With present (1994) levels of sulfur deposition and present nitrogen leaching, the critical load of acidity is exceeded in 92% of the grids, whereas with full nitrogen leaching (as predicted by the FAB model), 98% of the area would be exceeded. Since this grid cell represents an area with the best documentation in Norway (and probably also in Europe) with respect to fish damage and water chemistry (11% of the lakes analyzed), we consider the critical load data for this grid to be the most reliable in Norway. The lake percentiles are generally a bit lower than given by the data base submitted to the CCE, indicating that the lakes selected for the CCE data base slightly overestimate the critical load distribution in the grid, i.e. the critical loads are in fact lower than those reported to the CCE.

Conclusions

- Freshwater is the ecosystem most sensitive to acidic deposition in Norway.
- The soil critical loads do not influence the critical load percentile distributions significantly in the low range.
- The critical loads in Norwegian grid cells are well-documented, especially in the most heavily affected areas in southern Norway.
- When the critical load is not exceeded there is only a small probability that the fish population will be damaged, but when the critical load is exceeded the chance of damage increases with the amount of exceedance.
- At critical load there is about 20% risk of damage to fish (brown trout, Arctic char and perch).
- There is no justification to exclude any data from the Norwegian critical load data base.

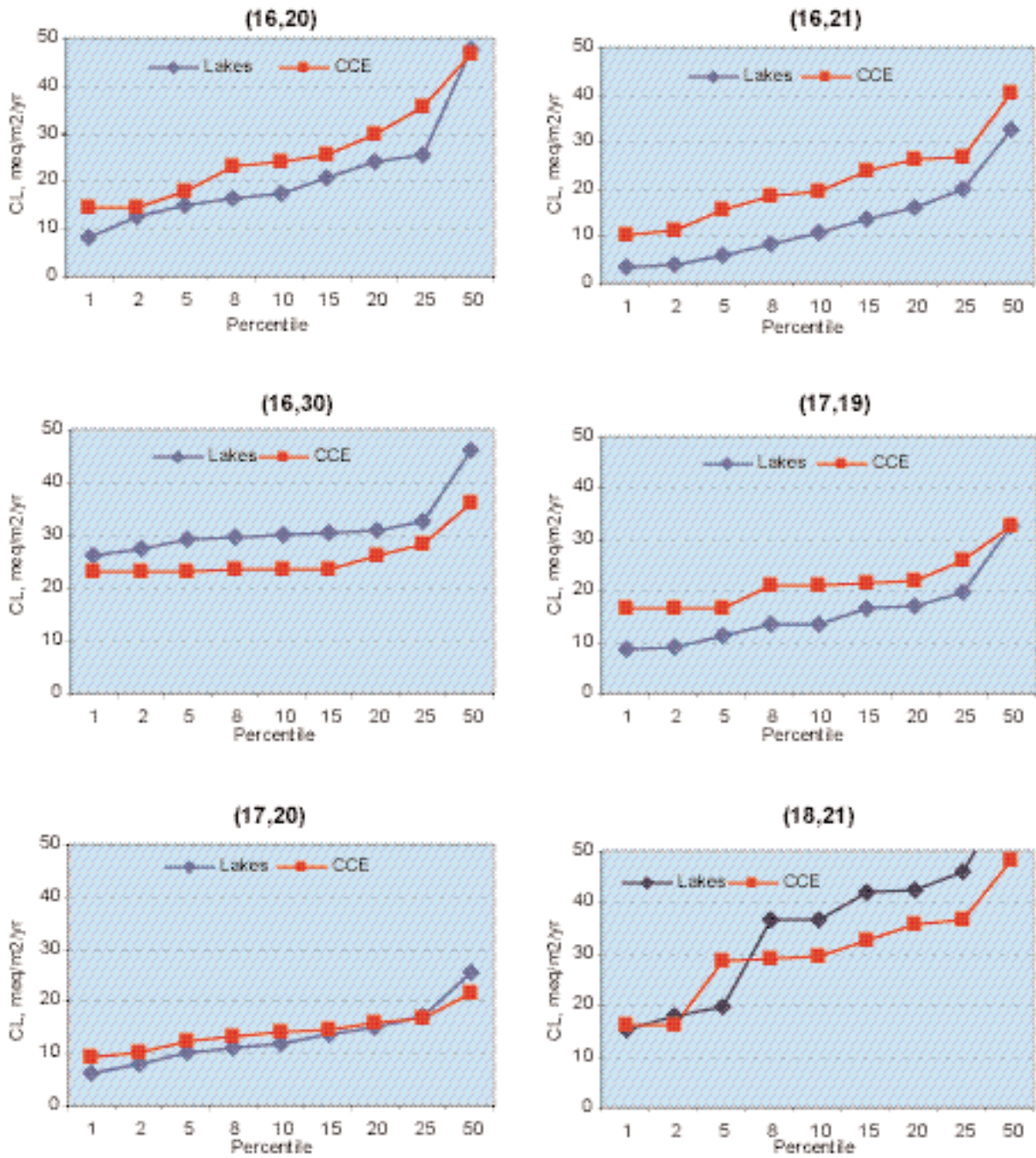


Figure NO-3. Percentiles for critical loads of acidity (CLA) for 6 binding EMEP grid cells based on the Norwegian critical load data base for lakes submitted to the CCE and for the lake data base (Lakes) based on the national lake surveys in 1986 and in 1995. The lakes data base was used for the critical load assessments for Norway.

B. Critical load exceedance and damage to fish populations

Data for water chemistry and fish status from the "1000 Lake Survey" carried out in Norway in 1986 (Henriksen *et al.* 1988, 1989) was used to derive a dose/response function for the probability of damage to fish populations as a function of the critical load exceedance. This function was further compared with a corresponding function derived from independent data bases for fish status (Hesthagen *et al.* 1999) and critical load exceedances within separate grid cells (see Posch *et al.* 1997) throughout Norway.

The 1000 Lake model:

A logistic regression model (see Henriksen *et al.* 1999) made it possible to calculate the probability for a fish population to be classified in one of three fish status classes: 1 (unaffected), 2 (reduced) and 3 (extinct). The regression model generally fitted the data well, and the

correct class was predicted for nearly 70% of the 679 lakes included in the analysis.

There is a rather close relationship between the probability of damage to fish populations and the degree of critical load exceedance (Figure NO-4). When critical loads are not exceeded, the probability of damage to fish populations is low, whereas when the critical load is exceeded, the probability of fish damage increases with increasing exceedance. At critical load the probability of damage to fish populations is about 20%. The regression analyses also indicate that it is difficult to predict damaged fish populations because the profiles for classes 1 and 3 are located closely together. One reason for this is that a damaged population is not in a stable condition, and will disappear with time if the deposition is not sufficiently reduced. It is also more difficult to obtain reliable information through the interview method than for the other two conditions. It is, however, more easy to predict whether a lake's fish population is damaged or not at a given critical load exceedance.

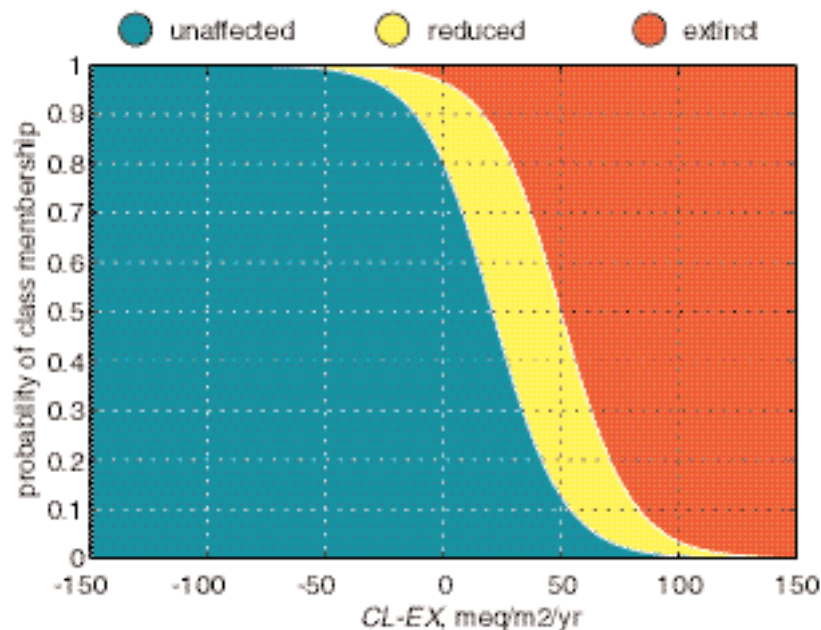


Figure NO-4. Probability of acidification damage to fish populations in lakes as a function of the exceedance of the critical load of acidity (CL-EX). At any point on the exceedance axis, the probability associated with each category of fish status is the height of the area corresponding to that class.

The grid cell model

The dose/response function presented in Figure NO-4 has been derived on the basis of data for water chemistry and fish status for the same set of lakes. The fish status data base (Hesthagen *et al.*1998) and the critical load data base (see Posch *et al.*1997) are both based on the same grid cell system. For each grid there is information about chemistry and fish status for a number of lakes and a mean value was used for each observation. The chemistry and fish status data were not from the same lakes, although for some lakes they could coincide. The fish status and water chemistry information was, however, considered to be representative for lakes in the grid area and similar calculations as for the "1000 Lake Survey" data were carried out.

Applying a model based on the three fish status categories ("unaffected", "reduced", and "extinct") did have lower prediction ability than the model based on the "1000 Lake Survey" data. The model was not able to predict the correct status for any of the populations in the class "reduced". This is not surprising since the critical load data base is based on one lake in the grid, while the fish status information is based on a number of lakes. We therefore joined the classes "reduced" and "extinct" to the class "damaged", and a new regression model was calculated. This simplified version operates with only two categories, "undamaged" and "damaged", to explain

critical load exceedance. This model predicted 94% of the undamaged populations correct, compared with 56% for damaged populations. In order to compare the results from this simplified model with the model based on the "1000 Lake Survey" (Figure NO-4), we re-coded the data from this survey to the categories "undamaged" and "damaged". A logistic regression of fish status versus critical load exceedance predicted 83% of the undamaged populations correctly, whereas the corresponding figure for the damaged populations was 85%. The lake survey curve is located to the left of, and is steeper than, the grid-based model (Figure NO-5). The less steep slope for the grid-based model may partly be due to the fact that the lake survey data are based on chemistry and fish status for the same lakes, whereas the water chemistry for the grid data are based on one lake in a grid. This will introduce an error in the critical load exceedance values. The statistical effect of such an error is that the regression curve will show a lower slope. The greater uncertainty in the response on critical load exceedance for the grid-based model may be due to the heterogeneous environmental conditions within the grids. As the species composition of the two data bases was comparable, it is therefore unlikely that there should have been any significant differences in the acidification sensitivity of the fish societies. Thus, the above-mentioned bias in the results from the interview method may have had a greater impact in the large grid based survey, with many more remote and less accessible lakes, than in the smaller "1000 Lake Survey".

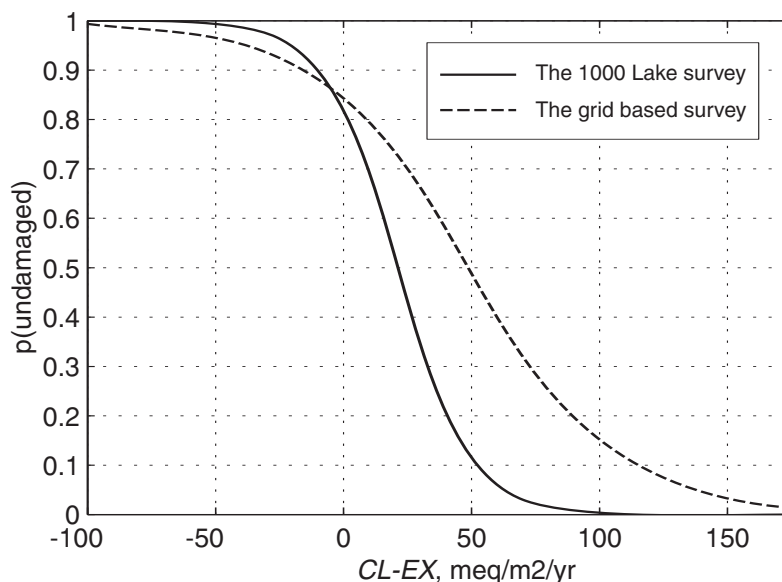


Figure NO-5. The probability (π) for classifying a fish population as undamaged as a function of the lakes critical load exceedance (CL-EX). The curve for the "1000 Lake Survey" is based fish status and critical load exceedance for single lakes. The curve for the grid data base is based on a estimated representative critical load exceedance for a geographic area and the fish status for a selection of lakes located in that area.

Conclusions

The results from these analyses confirm that the critical load concept is a realistic tool for estimating the extent of biological damage to freshwaters due to acidification by long-range transported air pollution. Hence, prognoses based on this method should give reliable estimates of extent of fish damage under future scenarios of acidic deposition.

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Annex: Critical loads for surface waters with catchment-dependent ANC_{limit}

M. Posch and A. Henriksen

In most, especially earlier, applications of the steady-state water chemistry (SSWC) model a constant critical ANC_{limit} had been chosen. Since 1995, however, Norwegian lake critical loads are calculated with a so-called "variable" ANC_{limit} . This has been introduced in Henriksen *et al.* (1995), but here we present a more concise derivation of this catchment-dependent ANC_{limit} .

In the SSWC model the critical load of acidity, CL , is calculated as:

$$CL = Q \cdot ([BC]_0^* - ANC_{limit}) \quad (1)$$

Where Q ($m \text{ yr}^{-1}$) is runoff, $[BC]_0^*$ ($\mu\text{eq l}^{-1}$) the original non-marine base cation concentration and ANC_{limit} ($\mu\text{eq l}^{-1}$) is the ANC concentration above which no damage to fish populations occur. In general, the ANC_{limit} will be a function of the "sensitivity" of the catchment, i.e. a function of the critical load, which expresses this sensitivity numerically with respect to deposition. Thus Eq. 1 reads:

$$CL = Q \cdot ([BC]_0^* - ANC_{limit}(CL)) \quad (2)$$

In order to obtain a value for the critical load of a given catchment the functional dependence of the ANC_{limit} on CL has to be specified. Historically, the ANC_{limit} has been set to a constant value ($20 \mu\text{eq l}^{-1}$ in Norway (Lien *et al.* 1992), thus assuming no variation in the biological reactions of the species to be protected. Inserting the chosen value for the ANC_{limit} immediately yields the critical load. It is more likely, however, that the ANC_{limit} increases with the critical load, since less sensitive ecosystems will have a higher biological variety/diversity

and thus require a higher ANC_{limit} to keep that diversity intact. The simplest functional form with this feature is a linear relationship between ANC_{limit} and CL :

$$ANC_{limit} = k \cdot CL \quad (3)$$

Inserting Eq. 3 into Eq. 2 yields the following implicit equation for calculating CL :

$$CL = Q \cdot ([BC]_0^* - k \cdot CL) \quad (4)$$

or, solving for CL :

$$CL = Q \cdot [BC]_0^* / (1+k \cdot Q) \quad (5)$$

and thus from Eq. 3 also:

$$ANC_{limit} = k \cdot CL = k \cdot Q \cdot [BC]_0^* / (1+k \cdot Q) \quad (6)$$

As for the constant ANC_{limit} earlier, the proportionality constant k has to be derived from data. If we assume that for a CL of $200 \text{ meq m}^{-2} \text{ yr}^{-1}$ the ANC_{limit} does not exceed 50 meq m^{-3} ($\mu\text{eq l}^{-1}$), as has been assumed in Sweden, we arrive at a k value of $50/200 = 0.25 \text{ yr m}^{-1}$. For CL values above $200 \text{ meq m}^{-2} \text{ yr}^{-1}$ we set the ANC_{limit} to a constant value of 50 meq m^{-3} ($\mu\text{eq l}^{-1}$). This means that Eq. 3 should actually be written as:

$$ANC_{limit} = \min\{ k \cdot CL, 50 \} \quad (7)$$

The value of k is derived from the experience in Nordic countries and reflects thus the geology, deposition history and biological diversity (fish species) of that region. For other regions different k values could be more appropriate.

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National maps produced

Receptors mapped: Forest ecosystems represented by 1461 forest monitoring plots.

- Maximum critical loads of sulfur.
- Minimum critical loads of nitrogen.
- Maximum critical loads of nitrogen.
- Critical loads of nutrient nitrogen.
- Protection percentage against acidification due to S and N deposition in 1996.
- Reduction requirements for S and N deposition from 1996 to achieve a 5% protection level.

Calculation methods

Maximum critical loads of sulfur, minimum critical loads of nitrogen, maximum critical loads of nitrogen and critical loads of nutrient nitrogen for 1461 forest plots have been calculated using the Steady-State Mass Balance (SSMB) method outlined in the Mapping Manual (UBA 1996). For high-elevation areas, the relevant modification of the SSMB was applied as described in UBA Vienna (1993).

Data sources

Soil data: Base cation weathering for 148 forest soil profiles was estimated using the PROFILE model (Sverdrup and Warfvinge 1995). For 35 sites evenly spread throughout Poland, soil mineralogy was measured by the State Institute of Geology in Warsaw (Stepniewski 1998). Total element analyses was determined for 148 sites and texture was measured by the Bouyoucos-Casagrande aerometric method (Wawrzoniak *et al.* 1996). The UPPSALA model was used to estimate mineralogy from total analyses. Several other parameters such as dissolved organic carbon, CO₂ pressure, and gibbsite coefficients were taken as default values recommended by the PROFILE developers. For the remaining forest plots, the dominant soil types have been identified from the Polish Soil Atlas 1:300,000, and adequate values of base cation weathering were attributed to them according to procedures described in the Mapping Vademecum (Hettelingh and de Vries 1992).

Meteorological data: Data for precipitation, runoff and average annual temperature were obtained from the Hydrological Atlas of Poland published by the Institute of Meteorology and Water Management (Stachy 1986).

Uptake: Uptake of base cations and nitrogen was calculated as the minimum of growth-limited uptake and nutrient-limited uptake. Forest growth rates related to seven major tree species were obtained from the data bank of the Forest Management and Geodesy Office. Elements content in stems and branches was provided by the Polish Academy of Science's Institute of Dendrology (Fober 1986).

Critical nitrogen leaching: The limiting concentrations of nitrogen for coniferous (0.0143 eq m⁻³) and deciduous (0.02 eq m⁻³) trees suggested in the CCE Status Report 1993 and the estimated precipitation surplus (m³ ha⁻¹ yr⁻¹) were used.

Immobilization of nitrogen in soils: A temperature-dependent long-term immobilization factor was applied, ranging from 71 to 214 eq ha⁻¹ yr⁻¹.

Denitrification: Denitrification fractions were related to the soil type by linearly interpolating between a low value of 0.1 for podzolic mineral soils and a value of 0.8 for peat soils (de Vries *et al.* 1993).

Deposition data: Data on deposition of sulfur and oxidized and reduced nitrogen for 1996 was used to calculate the exceedance of critical loads of acidity. These data were taken from EMEP MSC-W Report 1/98. Base cation deposition data were provided by RIVM (Draaijers *et al.* 1995)

Comments and conclusions

The Level II forest monitoring program which started to operate in Poland in 1995 has allowed a significant refinement of the input data, particularly with regards to soil data and base cation weathering rates. The resulting critical load calculations are summarized in Figures PL-1 through PL-4. Figure PL-5 presents a map of the percentage of area protected against acidification due to S and N deposition in 1996, while Figure PL-6 displays the reduction requirements of the same deposition to achieve a 5% protection level.

In December 1998 a trilateral meeting of the Czech, German and Polish NFCs took place in Prague. The aim of the meeting was to review and revise methods, input data and the results of critical loads calculations to avoid unjustifiable differences in border areas. Through the exchange of data and experience, an acceptable harmonization has been achieved.

A computer program, SONOX, has been developed to reflect the connections between pollutants emission, atmospheric transport and deposition and the ecological impacts. The program focuses on national air pollution problems with regard to transboundary fluxes. The program allows one to simulate the effects of acid deposition to forest soils resulting from actual and assumed emission scenarios of sulfur and nitrogen. The program is equipped with a GIS module which generates maps of emissions, deposition, critical loads and their exceedances. The attached maps of deposition reduction requirements and protection levels resulting from emission scenario for the year 1996 have been produced using this software.

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eq/ha/year

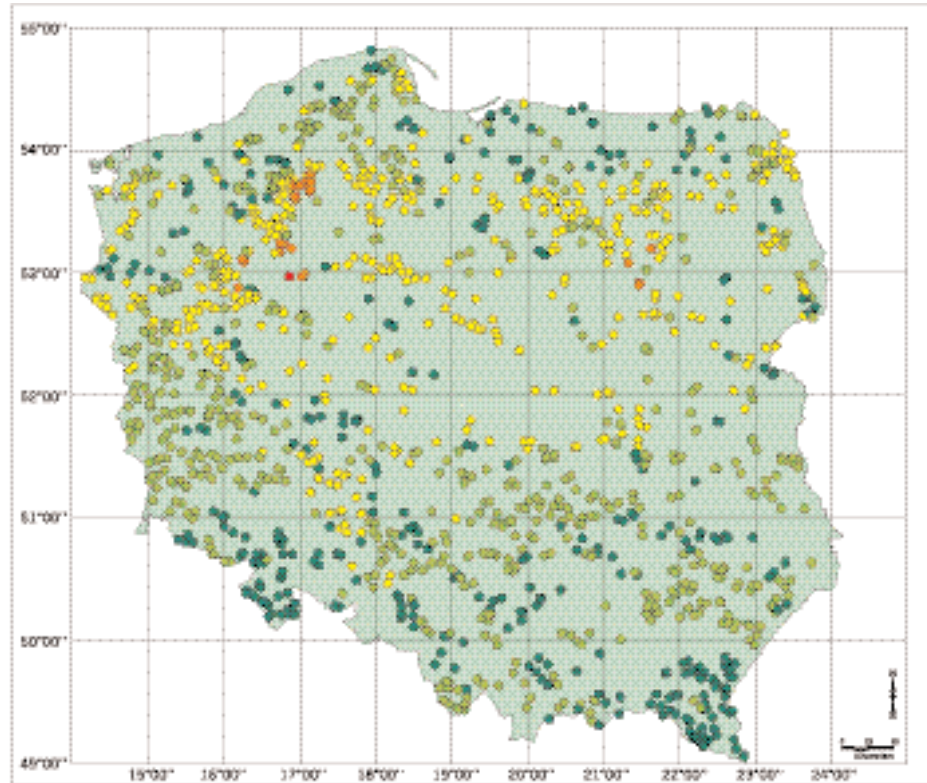
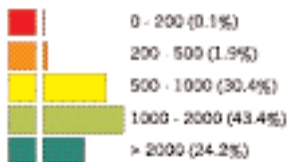


Figure PL-1. Maximum critical loads of sulfur.

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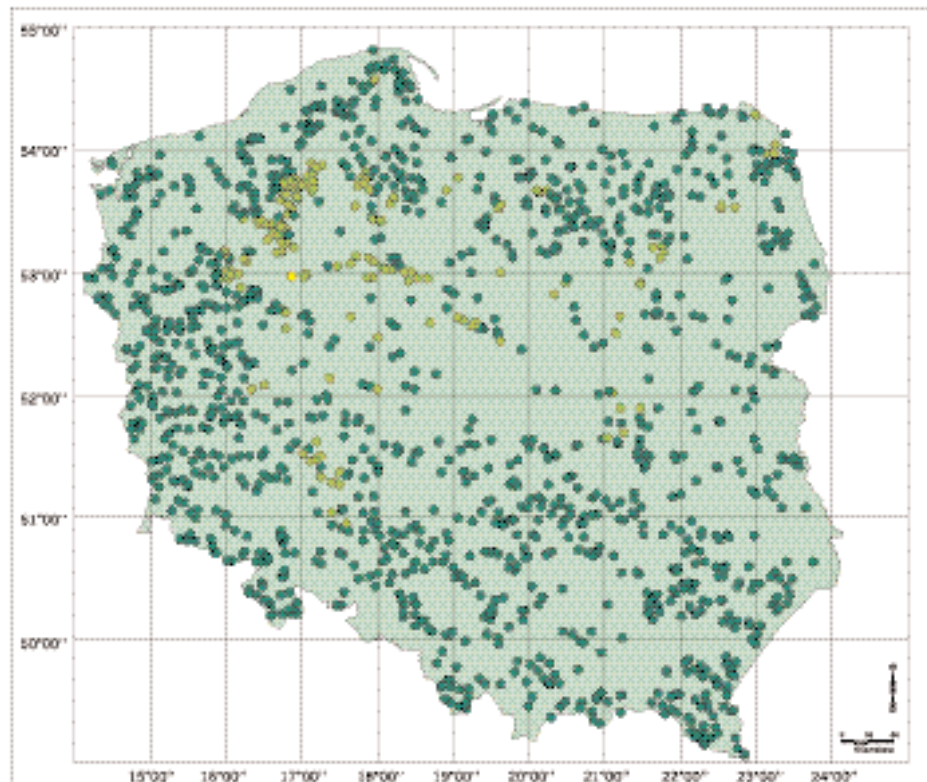


Figure PL-2. Maximum critical loads of nitrogen.

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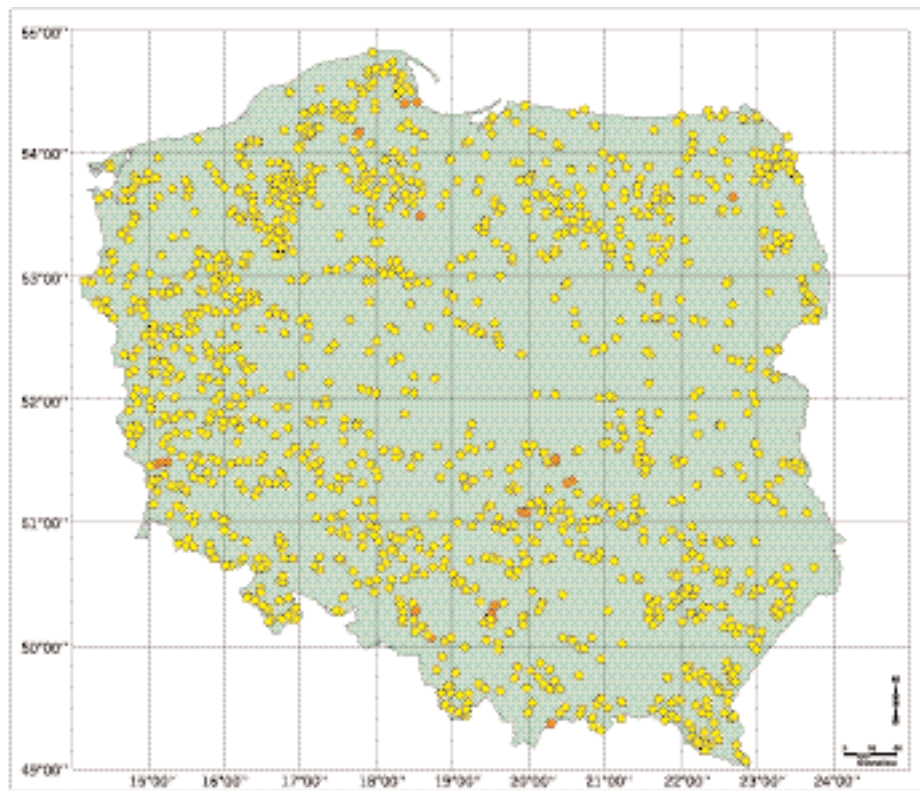


Figure PL-3. Minimum critical loads of nitrogen.

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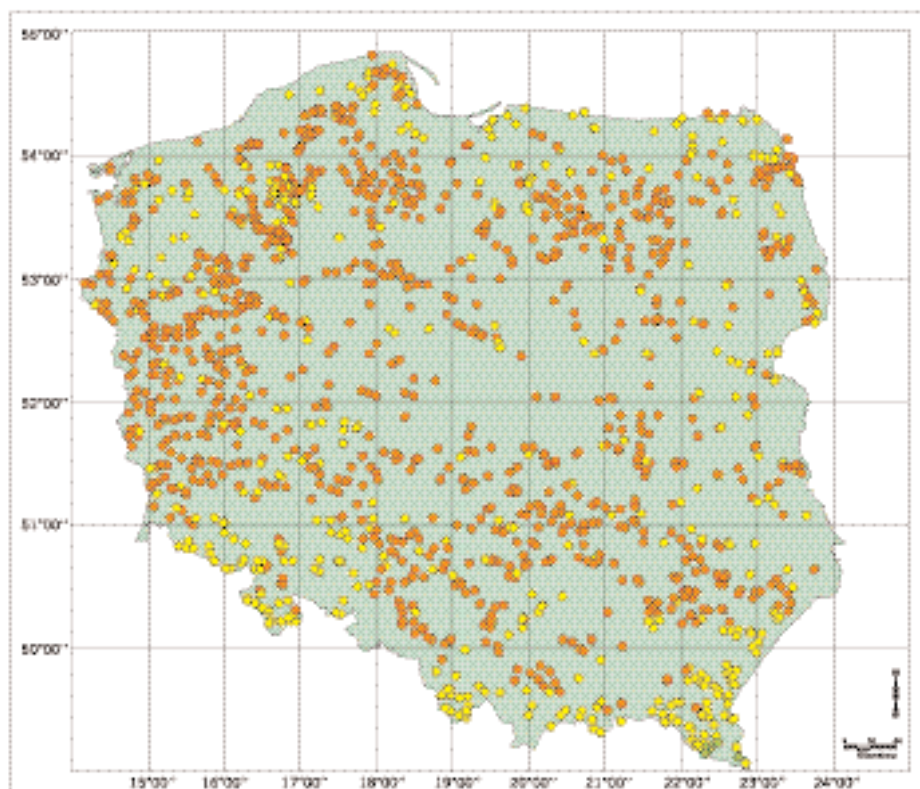
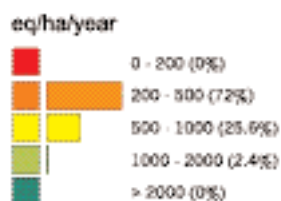


Figure PL-4. Critical loads of nutrient nitrogen.

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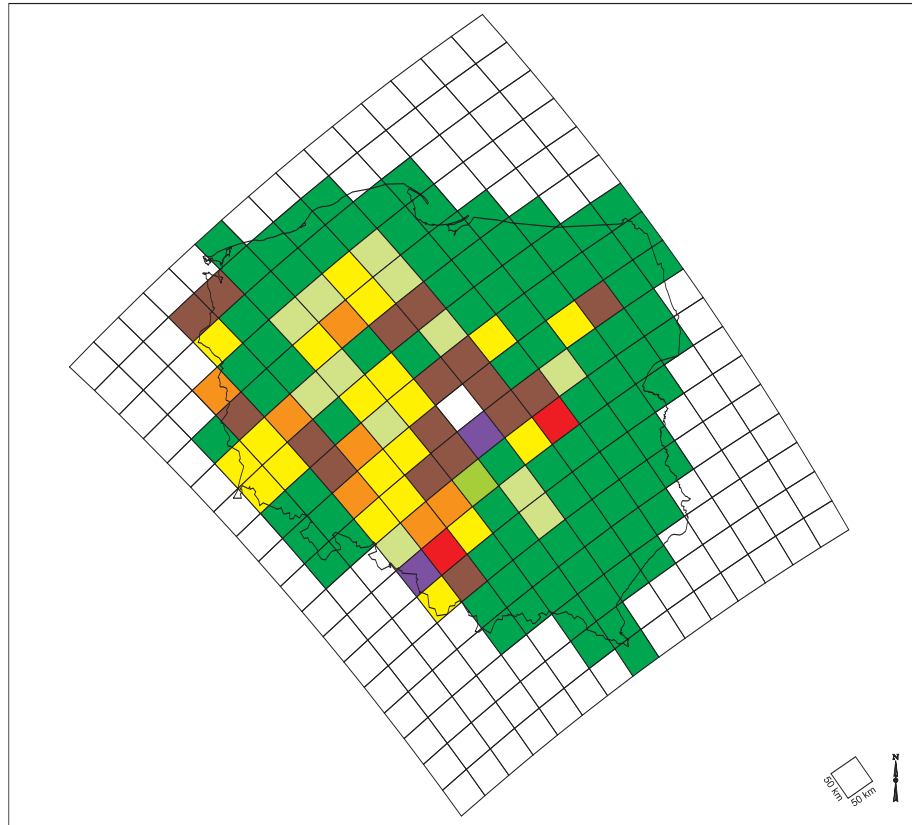
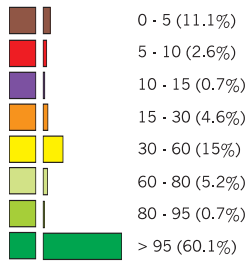


Figure PL-5. Percentage of area protected against acidification due to S and N deposition in 1996.

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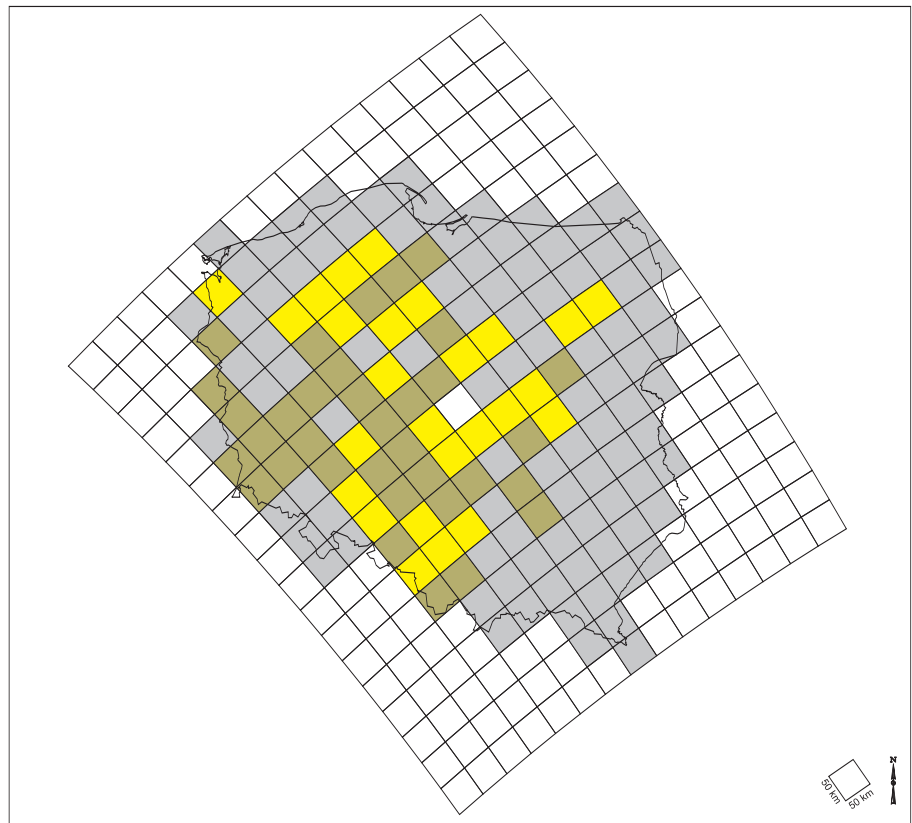
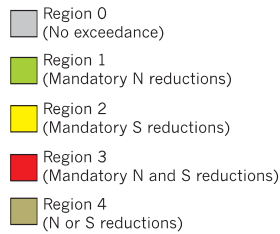
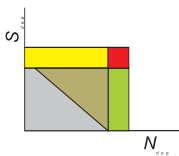


Figure PL-6. Reduction requirements for S and N deposition to achieve a 95% protection level.

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Calculation methods

Critical loads of sulfur, acidifying and nutrient nitrogen were calculated for terrestrial ecosystems in the Republic of Moldova using modified Steady-State Mass Balance (SSMB) equations. The corresponding algorithms described below were suggested by the CCE Mapping Subcentre (Bashkin 1997).

The parameters used to calculate critical loads include:

- ANC_l = acid-neutralizing capacity of soil
- BC_d = base cation deposition
- BC_u = base cation uptake
- BC_w = base cation weathering
- $C:N$ = C:N ratio in the upper soil layer
- C_N = critical content of nitrogen in surface water
- C_b = coefficient of biogeochemical turnover
- C_t = active temperature coefficient (ratio of temperature sum $>5^\circ\text{C}$ to total annual sum)
- D = upper soil layer depth
- K_{gibb} = gibbsite coefficient
- $N:BC$ = ratio of N to BC in plant tissue

- N_{de} = denitrification of soil N
- N_{de}^* = denitrification of deposition N
- N_i = immobilization of soil N
- N_i^* = immobilization of deposition N
- N_l = N leaching
- NMC = nitrogen mineralization capacity of soils (eq $\text{ha}^{-1} \text{yr}^{-1}$)
- N_{td} = total N deposition, wet + dry ($\text{NO}_x + \text{NH}_x$)
- N_u = uptake of soil N
- N_u^* = uptake of deposition N
- N_{upt} = annual N uptake
- Q = surface runoff
- W_r = chemical weathering of soil (eq $\text{ha}^{-1} \text{yr}^{-1}$ at 1m depth)

A. The minimum critical load of nitrogen was calculated as:

$$CL_{min}(N) = N_i^* + N_u^*$$

where:

$$N_i^* = \begin{cases} 0.15 \cdot N_{td} \cdot C_t / C_b & \text{if } C:N > 10 \\ 0.25 \cdot N_{td} \cdot C_t / C_b & \text{if } 10 \leq C:N < 14 \\ 0.30 \cdot N_{td} \cdot C_t / C_b & \text{if } 14 \leq C:N < 20 \\ 0.35 \cdot N_{td} \cdot C_t / C_b & \text{if } C:N \geq 20 \end{cases}$$

and:

$$N_u^* = N_{upt} - N_u$$

and the annual N uptake is defined as:

$$N_{upt} = \begin{cases} K \cdot N_{upr} \cdot (1 - 1/C_b) & \text{if } C_b > 1 \\ K \cdot N_{upr} \cdot (1/C_b) & \text{if } C_b \leq 1 \end{cases}$$

The constant $K = 1.2$ for deciduous forests, and 0.8 for coniferous forests.

Uptake of nitrogen from the soil, N_u , is calculated as:

$$N_u = (NMC - N_i - N_{de}) \cdot C_t$$

where:

$$N_i = \begin{cases} 0.15 \cdot NMC / C_b & \text{if } C:N > 10 \\ 0.25 \cdot NMC / C_b & \text{if } 10 \leq C:N < 14 \\ 0.30 \cdot NMC / C_b & \text{if } 14 \leq C:N < 20 \\ 0.35 \cdot NMC / C_b & \text{if } C:N \geq 20 \end{cases}$$

and:

$$N_{de} = \begin{cases} 0.145 \cdot NMC + 0.9 & \text{if } NMC < 10 \\ 0.145 \cdot NMC + 0.605 & \text{if } 10 \leq NMC \leq 60 \\ 0.145 \cdot NMC + 6.477 & \text{if } NMC > 60 \end{cases}$$

B. The critical load of nutrient nitrogen was calculated as:

$$CL_{nut}(N) = CL_{min}(N) + N_l + N_{de}^*$$

where:

$$N_l = Q \cdot C_N$$

$$N_{de}^* = N_{td} \cdot C_t \cdot N_{de} / NMC$$

C. The maximum critical load of sulfur was calculated as:

$$CL_{max}(S) = C_t \cdot (BC_w - ANC_l) + (BC_d - BC_u)$$

where:

$$BC_w = W_r \cdot D$$

$$BC_u = N_u^* \cdot N:BC$$

$$ANC_l = Q \cdot ([H] + [Al])$$

with $[Al] = 0.2$ and $[H] = ([Al]/K_{gibb})^{1/3}$

D. The maximum critical load of nitrogen is calculated as:

$$CL_{max}(N) = CL_{max}(S) + CL_{min}(N)$$

Table MD-1 below shows values for some of the parameters used in the above equations.

Exceedances of critical loads

The exceedances of sulfur and nitrogen critical loads were calculated using the "exceedance indifference curve" approach (Posch *et al.* 1997, UBA 1996). This approach is shown schematically in Figure MD-1. The areas in Figure MD-1 are defined in terms of emissions reductions required to achieve critical loads, as follows:

- 0 no exceedance
- 1 voluntary N or S emissions reductions
- 2 mandatory S reductions
- 3 mandatory N reductions
- 4 mandatory N and S reductions.

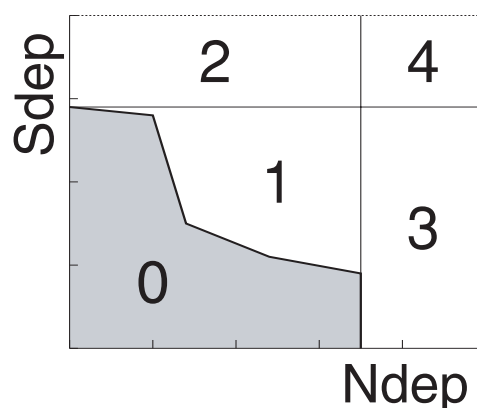


Figure MD-1. Scheme of a protection isoline (95 percentile function).

Data sources

Soil cover: The digitized FAO soil map, 2' x 2' scale (FAO 1989).

Land use: The LuGrid data base (de Smet and Heuvelmans 1997).

N and S deposition: EMEP MSC-W model calculations for 1992 (150x150 km²) and for 1996 (50x50 km²) (EMEP/MSC-W 1998).

Results

Figure MD-2 presents the 5-percentile critical load data for Moldovan terrestrial ecosystems. These data were applied for the calculation of exceedances with 1992 and 1996 deposition scenarios. A comparison of exceedances calculated for the 1992 and 1996 deposition scenarios shows some differences. Only two exceedances types (0,3) were indicated for terrestrial ecosystems in the Republic of Moldova in 1992. Due to reductions in sulfur emissions both in the Republic of Moldova and other European countries, the number of cells with mandatory N reductions was decreased (Table MD-2).

Table MD-1. Parameters used to calculate sulfur and nitrogen critical loads for different soil types.

FAO classification	W_r	D	$C:N$	NMC	C_b	N_{crit}	$N:BC$	N_{upt}	K_{gibb}
Eutric Cambisols	2750	0.75	13.50	110	1.50	1.00	1.00	85	100
Haplic Chernozems	3000	1.00	12.80	150	0.90	1.00	1.20	125	100
Calcic Chernozems	3500	1.00	12.00	120	1.00	1.00	1.20	115	100
Eutric Fluvisols	1750	0.50	13.00	75	2.50	1.00	1.20	50	300
Calcic Kastanozems	3500	1.00	12.80	100	0.95	1.20	1.35	75	150

Acknowledgments

The authors wish to thank the CCE Mapping Subcentre (V. Bashkin, head) for methodological help in critical load calculation and mapping, and the EMEP Meteorological Synthesizing Centre-West (L. Tarrason, S. Tsyro and K. Olendrzynski) for providing sulfur and nitrogen deposition data.

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Table MD-2. Distribution of exceedance types for ecosystems in Moldova in 1992 and 1996.

Year	Number of EMEP 50x50 km ² grid cells	Class 0 No exceedances	Class 1 Voluntary N or S reductions	Class 2 Mandatory S reductions	Class 3 Mandatory N reductions	Class 4 Mandatory N and S reductions
1992	37	26	0	0	11	0
1996	37	32	0	0	5	0

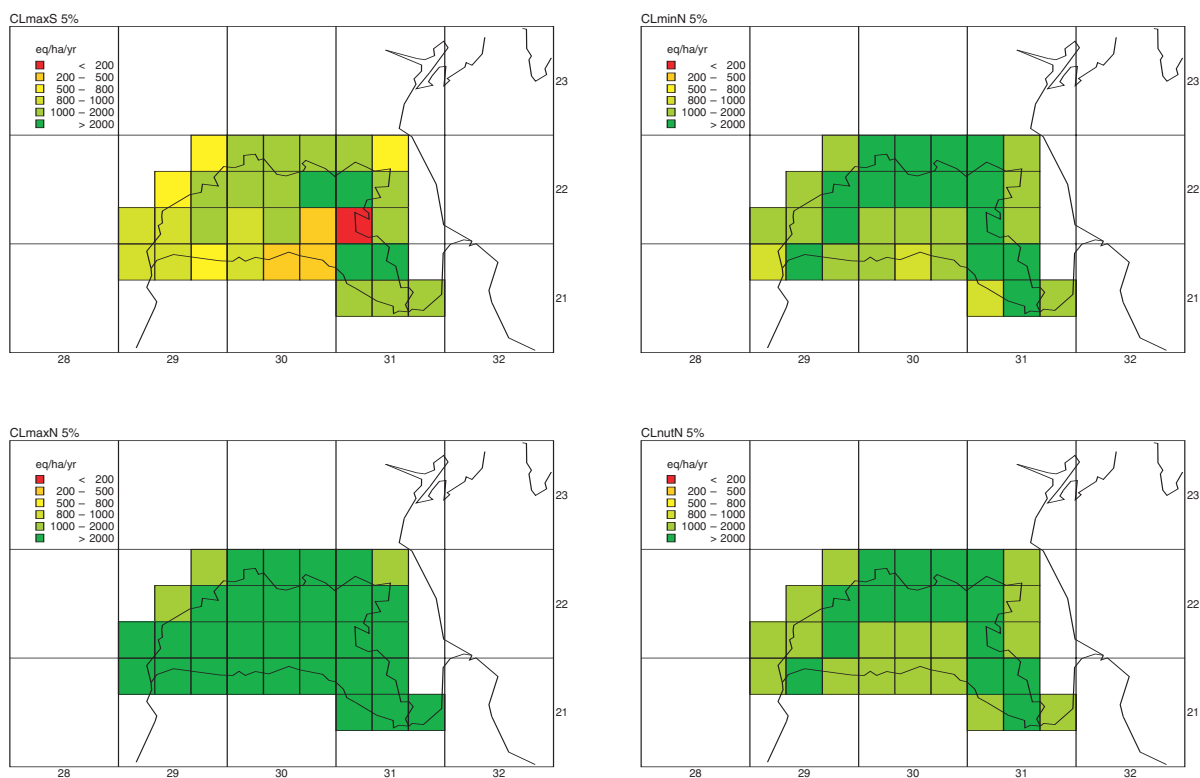


Figure MD-2. Five-percentile maps of: the maximum critical load of sulfur (CLmaxS), minimum critical load of nitrogen (CLminN), maximum critical load of nitrogen (CLmaxN) and critical load of nutrient nitrogen (CLnutN) for Republic of Moldova.

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Calculation methods

Critical loads of sulfur, acidifying and nutrient nitrogen were calculated for terrestrial ecosystems in the European part of the Russian Federation using modified Steady-State Mass Balance (SSMB) equations. The corresponding algorithms are described below.

The parameters used to calculate critical loads include:

ANC_l = acid-neutralizing capacity of soil
 BC_d = base cation deposition
 BC_u = base cation uptake
 BC_w = base cation weathering

$C:N$ = C:N ratio in the upper soil layer
 C_N = critical content of nitrogen in surface water
 C_b = coefficient of biogeochemical turnover
 C_t = active temperature coefficient (ratio of temperature sum $>5^\circ\text{C}$ to total annual sum)
 D = upper soil layer depth
 K_{gibb} = gibbsite coefficient
 $N:BC$ = ratio of N to BC in plant tissue
 N_{de} = denitrification of soil N
 N_{de}^* = denitrification of deposition N
 N_i = immobilization of soil N
 N_i^* = immobilization of deposition N
 N_l = N leaching
 NMC = nitrogen mineralization capacity of soils (eq $\text{ha}^{-1} \text{yr}^{-1}$)
 N_{td} = total N deposition, wet + dry ($\text{NO}_x + \text{NH}_3$)
 N_u = uptake of soil N
 N_u^* = uptake of deposition N
 N_{upt} = annual N uptake
 Q = surface runoff
 W_r = chemical weathering of soil (eq $\text{ha}^{-1} \text{yr}^{-1}$ at 1m depth)

A. The minimum critical load of nitrogen was calculated as:

$$CL_{min}(N) = N_i^* + N_u^*$$

where:

$$N_i^* = \begin{cases} 0.15 \cdot N_{td} \cdot C_t / C_b & \text{if } C:N > 10 \\ 0.25 \cdot N_{td} \cdot C_t / C_b & \text{if } 10 \leq C:N < 14 \\ 0.30 \cdot N_{td} \cdot C_t / C_b & \text{if } 14 \leq C:N < 20 \\ 0.35 \cdot N_{td} \cdot C_t / C_b & \text{if } C:N \geq 20 \end{cases}$$

and:

$$N_u^* = N_{upt} - N_u$$

and the annual N uptake is defined as:

$$N_{upt} = \begin{cases} K \cdot N_{upt} \cdot (1 - 1/C_b) & \text{if } C_b > 1 \\ K \cdot N_{upt} \cdot (1/C_b) & \text{if } C_b \leq 1 \end{cases}$$

The constant $K = 1.2$ for deciduous forests, and 0.8 for coniferous forests.

Uptake of nitrogen from the soil, N_u , is calculated as:

$$N_u = (NMC - N_i - N_{de}) \cdot C_t$$

where:

$$N_i = \begin{cases} 0.15 \cdot NMC / C_b & \text{if } C : N > 10 \\ 0.25 \cdot NMC / C_b & \text{if } 10 \leq C : N < 14 \\ 0.30 \cdot NMC / C_b & \text{if } 14 \leq C : N < 20 \\ 0.35 \cdot NMC / C_b & \text{if } C : N \geq 20 \end{cases}$$

and:

$$N_{de} = \begin{cases} 0.145 \cdot NMC + 0.9 & \text{if } NMC < 10 \\ 0.145 \cdot NMC + 0.605 & \text{if } 10 \leq NMC \leq 60 \\ 0.145 \cdot NMC + 6.477 & \text{if } NMC > 60 \end{cases}$$

B. The critical load of nutrient nitrogen was calculated as:

$$CL_{nut}(N) = CL_{min}(N) + N_I + N_{de}^*$$

where:

$$N_I = Q \cdot C_N$$

$$N_{de}^* = N_{td} \cdot C_t \cdot N_{de} / NMC$$

C. The maximum critical load of sulfur was calculated as:

$$CL_{max}(S) = C_t \cdot (BC_w - ANC_l) + (BC_d - BC_u)$$

where:

$$BC_w = W_r \cdot D$$

$$BC_u = N_u^* \cdot N : BC$$

$$ANC_l = Q \cdot ([H] + [Al])$$

with $[Al] = 0.2$ and $[H] = ([Al] / K_{gibb})^{1/3}$

D. The maximum critical load of nitrogen is calculated as:

$$CL_{max}(N) = CL_{max}(S) + CL_{min}(N)$$

Table RU-1 shows values for some of the parameters used in the above equations.

Exceedances of critical loads

The exceedances of sulfur and nitrogen critical loads were calculated using the "exceedance indifference curve" approach (Posch *et al.* 1997, UBA 1996). This approach is shown schematically in Figure RU-1. The areas in Figure RU-1 are defined in terms of emissions reductions required to achieve critical loads, as follows:

- 0 no exceedance
- 1 voluntary N or S emissions reductions
- 2 mandatory S reductions
- 3 mandatory N reductions
- 4 mandatory N and S reductions.

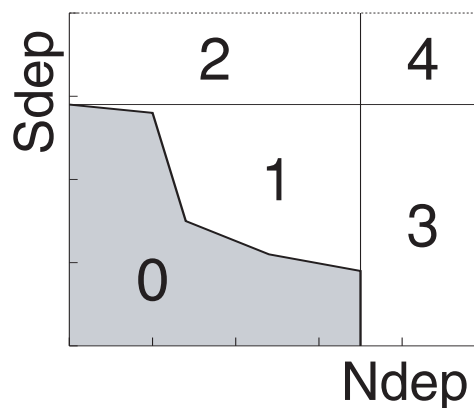


Figure RU-1. Scheme of a protection isoline (95 percentile function).

Data sources

Data sources consist of regionalized geological, soil, geochemical, geobotanic, hydrological, landscape, hydrochemical, and other data. For each elemental taxon, the main links of biogeochemical cycles of S, N and base cations were quantitatively parameterized on a 2'x2' grid resolution using available case studies and monitoring results (Bashkin 1997, Bashkin *et al.* 1997, 1998).

Soil cover: The digitized FAO soil map (FAO 1989).

Land use: The LuGrid data base (de Smet and Heuvelmans 1997).

N and S deposition: EMEP MSC-W model calculations for 1992 (150x150 km²) and for 1996 (50x50 km²) (EMEP/MSW-W 1998).

Results

Since the Russian Federation has not changed the data used for critical loads calculations, the maps of critical loads of S and N published in CCE Status Report 1997 (Bashkin *et al.* 1997) were used for the calculation of exceedances with 1992 and 1996 deposition levels.

A comparison of exceedances calculated for the 1992 and 1996 deposition (Figures RU-2 and RU-3) shows remarkable changes. All exceedances types (0,1,2,3,4) apply to some terrestrial ecosystems in the European part of Russia in 1992. By 1996, however, due to reductions in sulfur emissions both in the Russian Federation and other European countries, the number of cells with mandatory S reductions was sharply decreased. The numbers of cells with mandatory N reduction was also decreased (Table RU-2).

Table RU-1. Parameters used to calculate sulfur and nitrogen critical loads for various soil types.

No.	FAO classification	W_r	D	$C:N$	NMC	C_b	N_{crit}	$N:BC$	N_{upt}	K_{gibb}
5	Orthic Acrisols	250	0.50	9.00	65	1.50	1.00	0.95	65	300
8	Chromic Cambisols	1750	0.50	16.67	85	2.00	1.00	0.60	80	250
10	Eutric Cambisols	2750	0.75	13.50	110	1.50	1.00	1.00	85	100
19	Haplic Chernozems	3000	1.00	12.80	150	0.90	1.00	1.20	125	100
20	Calcic Chernozems	3500	1.00	12.00	120	1.00	1.00	1.20	115	100
21	Luvic Chernozems	2750	1.00	12.00	110	1.20	1.00	0.80	100	100
22	Podzoluvisols	750	0.50	16.67	50	5.00	1.00	0.70	35	300
23	Dystric Podzoluvisols	250	0.50	14.29	45	7.00	1.00	0.70	25	300
24	Eutric Podzoluvisols	750	0.50	17.14	55	6.00	1.00	0.80	40	200
26	Rendzinas	2250	0.50	20.00	90	3.00	1.00	0.80	50	100
39	Mollic Gleysols	1750	0.50	17.50	40	6.00	1.00	0.70	30	300
47	Lithosols	250	0.50	8.33	15	7.00	1.00	0.80	10	1500
50	Dystric Fluvisols	750	0.50	11.11	50	3.00	1.00	0.80	30	300
51	Eutric Fluvisols	1750	0.50	13.00	75	2.50	1.00	1.20	50	300
54	Haplic Kastanozems	3500	1.00	12.27	105	0.80	1.20	1.40	75	150
55	Calcic Kastanozems	3500	1.00	12.80	100	0.95	1.20	1.35	75	150
56	Luvic Kastanozems	3000	1.00	13.89	100	1.10	1.20	1.10	75	150
61	Gleyic Luvisols	2250	0.50	16.25	100	0.90	1.00	0.70	80	250
63	Orthic Luvisols	1750	0.50	18.33	80	2.00	1.00	0.60	45	250
68	Orthic Greyzems	750	0.50	24.00	30	20.00	1.00	0.10	20	200
74	Dystric Histosols	250	0.20	25.00	36	20.00	1.00	0.10	22	9.5
76	Gelic Histosols	250	0.55	8.75	25	7.00	1.00	0.70	20	9.5
79	Gleyic Podzols	250	0.50	12.86	20	7.00	1.00	0.70	15	300
82	Orthic Podzols	125	0.50	9.00	18	6.00	1.00	0.70	12	300
86	Cambic Arenosols	250	0.50	8.00	18	5.00	1.00	0.70	10	1500
93	Gelic Regosols	3000	1.00	15.00	45	1.30	1.50	1.30	70	2000
97	Orthic Solonetz	3000	1.00	16.00	35	0.70	1.60	1.10	50	200
11	Haplic Xerosols	3000	1.00	16.67	30	0.80	1.50	1.10	50	950
11	Luvic Xerosols	3500	1.00	18.75	25	0.80	2.00	0.90	30	950
12	Gleyic Solonchaks	3500	1.00	18.75	22	0.80	2.00	0.90	30	300
12	Orthic Solonchaks	3500	1.00	20.00	23	0.82	2.00	0.95	30	300

Table RU-2. Distribution of exceedance types for ecosystems in the European part of the Russian Federation in 1992 and 1996.

Year	Number of EMEP 50x50 km ² grid cells	Class 0	Class 1	Class 2	Class 3	Class 4
		No exceedances	Voluntary N or S reductions	Mandatory S reductions	Mandatory N reductions	Mandatory N and S reductions
1992	1573	697	7	61	787	21
1996	1573	1388	0	5	180	0

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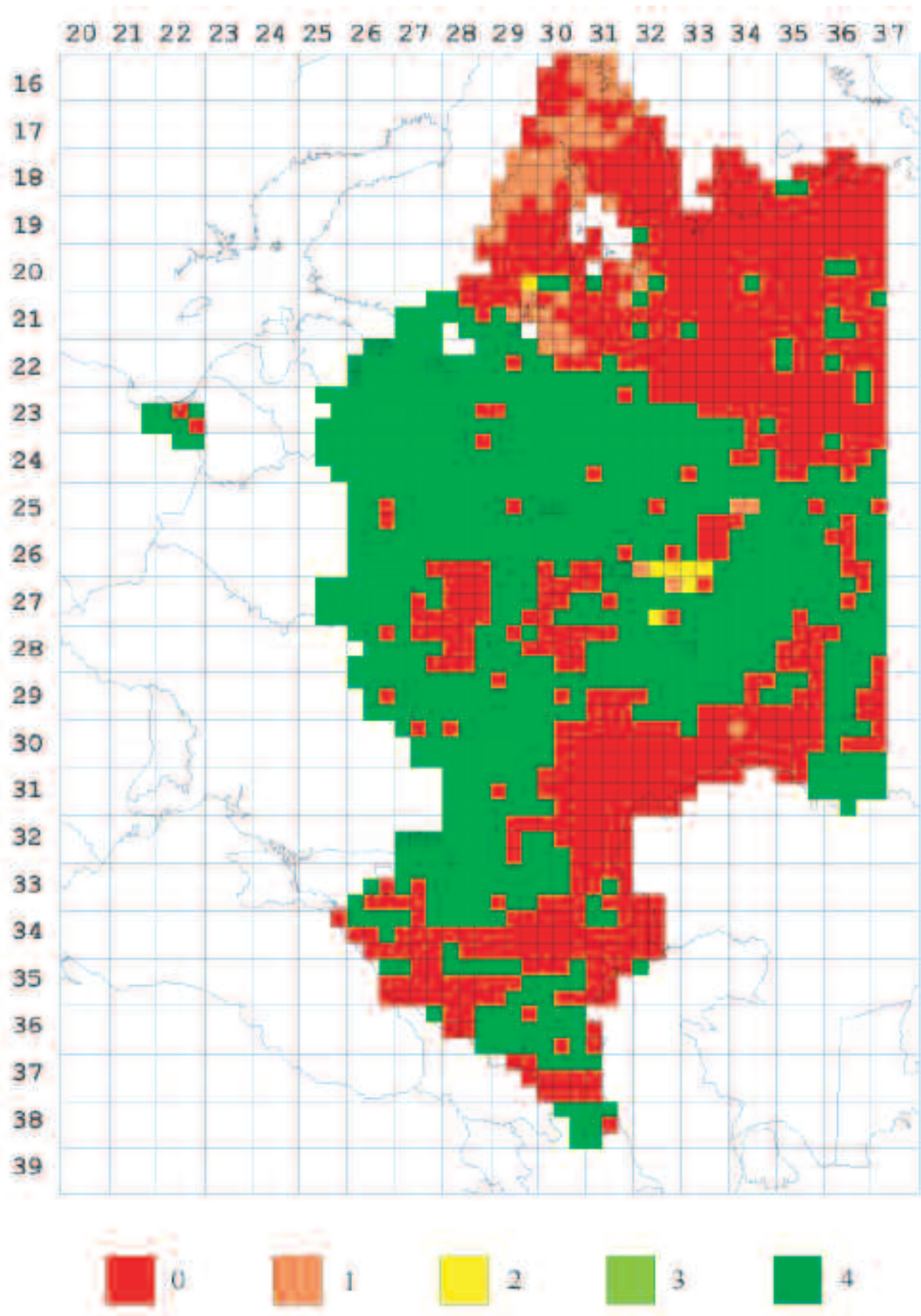


Figure RU-2. Exceedances of critical loads for terrestrial ecosystems in the European part of Russia, at 1992 deposition levels.

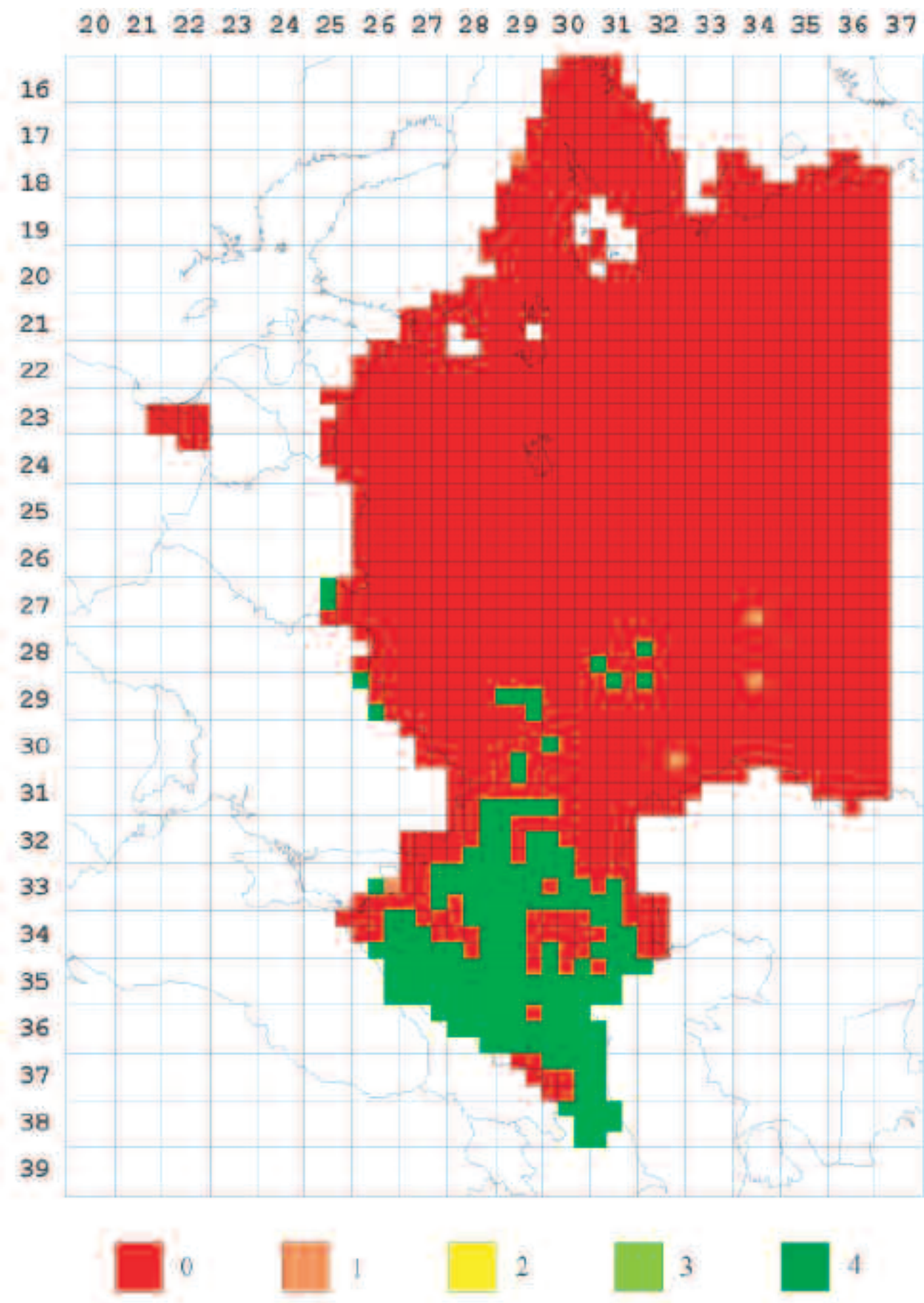


Figure RU-3. Exceedances of critical loads for terrestrial ecosystems in the European part of Russia, at 1996 deposition levels.

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National maps produced

Critical load maps for forest soils:

- Maximum critical loads of sulfur, $CL_{max}(S)$.
- Minimum critical loads of nitrogen, $CL_{min}(N)$.
- Critical loads of nutrient nitrogen, $CL_{nut}(N)$.

Grid size: 250x250 m²

Grid use: Forest distribution data base

Ozone critical level maps:

- AOT40 map for forest ecosystems averaged over 1992–1996.
- AOT40 map for crops (agriculture land) averaged over 1992–1996.

Grid size: 10x10 km²

Grid use: Interpolation of measurements

Calculation methods

Critical loads for forest soils:

The critical loads of sulfur and nitrogen for forest soils were calculated by using the steady-state mass balance method according to the Mapping Manual (UBA 1996):

$$CL_{max}(S) = BC_{dep} - Cl_{dep} + BC_w - BC_u \\ + 1.5 \cdot (BC_{dep} + BC_w - BC_u) / (BC/Al)_{crit} \\ + Q^{2/3} \cdot (1.5 \cdot (BC_{dep} + BC_w - BC_u) / \\ ((BC/Al)_{crit} \cdot K_{gibb})^{1/3}$$

$$CL_{min}(N) = N_i + N_u$$

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) / (1 - f_{de})$$

$$CL_{nut}(N) = N_i + N_u + N_{le(acc)} / (1 - f_{de})$$

All symbols and acronyms used in this report fully correspond with the Mapping Manual.

Critical levels for surface ozone:

AOT40 is defined as accumulated exposure of ozone over threshold of 40 ppb during daylight hours over the period May–July for crops and over the period April–September for forests. Critical ozone AOT40 levels have been established at 3,000 ppb.h for crops and 10,000 ppb.h for forests (UBA 1996).

Data sources

Forest soils

Critical loads were calculated for all major combinations of tree species (15) and soil types (62). Base-level digitized data included tree species composition, altitude, soil types, forest site types, precipitation and forest evapotranspiration. Other input data were derived through one or more of the base levels by using GIS methods. In a basic cell (250x250 m²) only 1 (prevailing) tree species was taken as representative. A critical BC/Al ratio, varying between 0.6 and 6, was considered as a function of tree species (Sverdrup and Warfvinge 1993). The gibbsite coefficient was assumed to be 200 m⁶ eq⁻² for the upper tree line and dwarfed pines, 300 for mountain spruce forest and 500 for mountain mixed forest and thermophilic deciduous forest.

Weathering: Mineral weathering rates of base cations have been related to the soil type and corresponding thickness of the root zone (10–70 cm). Values applied ranged from 50 and 7,000 eq ha⁻¹ yr⁻¹ (Hettelingh and de Vries 1992, Appendix 3; de Vries *et al.* 1997).

Uptake: Net growth uptake of base cations and nitrogen were calculated from average volume increment of wood and bark, multiplying by the concentration of N and BC in wood and bark (Bublinec 1992). Average volume increment was obtained from the national forest inventory data base. Values of average volume increment are based on individual trees according to the forest site classification and forest regions of Slovakia (Vladovic *et al.* 1994, Žihlavník and Brezina 1998).

Precipitation surplus: Precipitation surplus (Q) from the bottom of the root zone was calculated as the difference between precipitation amount and evapotranspiration from the forest. Precipitation data from the period 1951–1980 and calculated forest evapotranspiration data (Tomlain 1991) were interpolated into grid cells.

Nitrogen immobilization: The long-term natural immobilization of nitrogen, N_{i} , was estimated as a function of altitude. Values ranging between 20 and 350 eq ha⁻¹ yr⁻¹ (above 1000 m a.s.l.) were applied.

Denitrification: A constant denitrification fraction has been related to soil type: $f_{de} = 0.1$ for loess soils and those without gleyic features, $f_{de} = 0.5$ for sandy soils with gleyic features, $f_{de} = 0.7$ for clay soils and $f_{de} = 0.8$ for peat soils.

Critical N leaching: The acceptable leaching of nitrogen $N_{le(acc)}$ was calculated from critical concentration of nitrogen in soil solution multiplying by precipitation surplus:

$$N_{le(acc)} = Q \cdot [N]_{crit}$$

Critical N concentration values used were 0.143 eq m⁻³ for coniferous forest and 0.0215 eq m⁻³ for deciduous forest.

Deposition: The concentration of the main ions in rain water was relatively stable over Slovakia in the first half of the 1990s. Only sulfate decreased considerably (approximately 30% between 1990 and 1995). Therefore, wet deposition of N, Cl and BC for each grid cell might be

estimated on the basis of area average (1990–1995) concentrations in rain water (data from 7 regional monitoring stations were used), multiplied by the long-term precipitation amount (interpolated from precipitation maps).

Sulfur deposition was estimated for 1990 and 1995. Total deposition was calculated from wet deposition multiplied by an enrichment factor to reflect the effects of dry deposition and cloud deposition. The enrichment factors below were derived as a function of tree species and grid cell altitude, taking into account published data from through-fall measurements (Ulrich 1983, Skvarenina 1994):

S_{dep}	$N-NO_3 dep$	$N-NH_4 dep$	Cl_{dep}	BC_{dep}
1.5 – 3	1.2 – 2	1.2 – 2	1	1.2 – 1.5

Surface ozone:

AOT40 values for forests and crops were calculated using a special elevation-dependent function derived from surface ozone monitoring data. Because of the small area of Slovakia and a limited number of monitoring stations, the horizontal gradient of average ozone concentrations across Slovakia was neglected. Measurements from 11 suburban and regional monitoring stations were taken into account. No significant trends in surface ozone over Slovakia has been observed in the first half of the 1990s.

Results

Critical loads of acidity and sulfur for forest soils, surface and ground waters for the Slovak Republic have been calculated for the first time within the framework of the Norwegian/Slovak project, “Mapping critical levels/loads for Slovakia”. Results of the project were summarized in Závodský *et al.* (1996). It was documented that forest soil is the most sensitive ecosystem in Slovakia. Forests cover 41% of the territory.

The present submission describes the progress made in implementing the critical load approach within the Slovak Republic. The improved data set based on the national forest inventory system and on the national meteorological, hydrological and air pollution data was used.

The values of $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{nut}(N)$ for forest ecosystems over Slovakia are summarized in Table SK-1 and illustrated in Figure SK-1.

Table SK-1. Percentile values of critical loads (in eq ha⁻¹ yr⁻¹) for nitrogen and sulfur for Slovak forests.

	$CL_{min}(N)$			$CL_{nut}(N)$			$CL_{max}(S)$		
	5%	50%	95%	5%	50%	95%	5%	50%	95%
All ecosystems	328	571	831	394	668	899	1337	5248	14084
Spruce ecosystems	426	549	646	492	649	770	2190	3741	9530
Beech ecosystems	326	559	707	409	664	858	1294	8994	13350
Oak ecosystems	642	779	892	676	835	965	1116	7379	22538

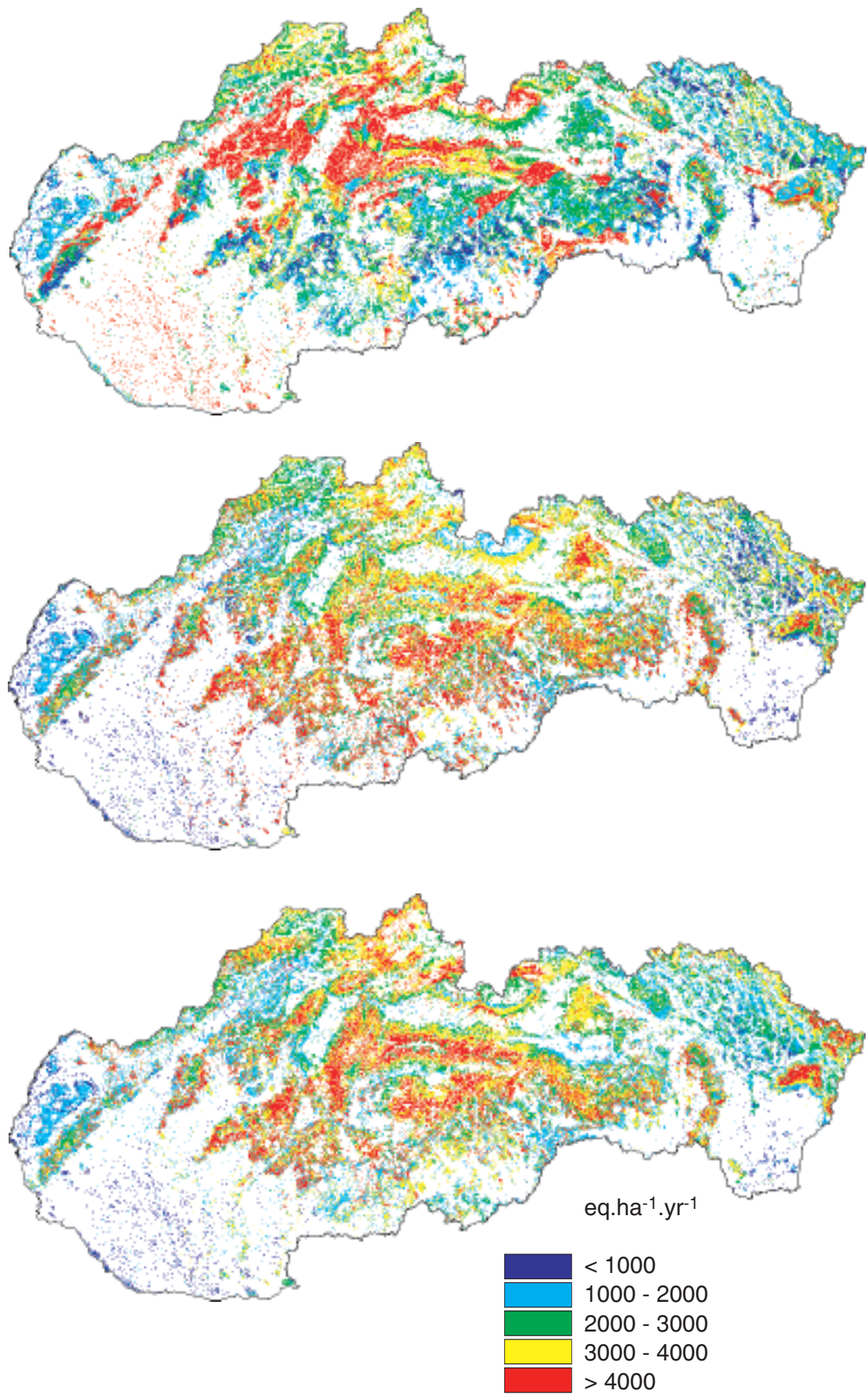


Figure SK-1. Critical loads of $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{nut}(N)$ for forest soils in Slovakia.

Figure SK-2 shows a histogram of critical load exceedances. In 1990, critical loads of acidity were exceeded at 50% of the total forested area, while in 1995 it was 31%. The decrease corresponds well to sulfate trends in precipitation.

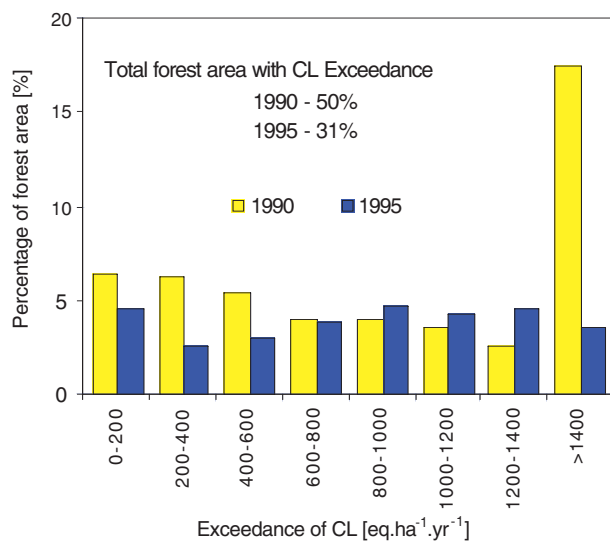


Figure SK-2. Critical load exceedances for forest ecosystems in Slovakia in 1990 and 1995.

Exceedance maps for 1990 and 1995 are presented in Figure SK-3. Calculated data of critical loads of nutrient nitrogen document the exceedances for the whole territory of the Slovak forests. The calculations of critical loads of nutrient nitrogen are based on the critical leaching data in Posch *et al.* (1995), but these data seem to be low for the conditions of Slovak forests. Data from nitrogen balance studies in Slovakia are needed.

In Figure SK-4 the AOT40 maps for forests and agricultural crops (average values for the period 1992–1996) are presented. For surface ozone, considerable critical level exceedances are observed.

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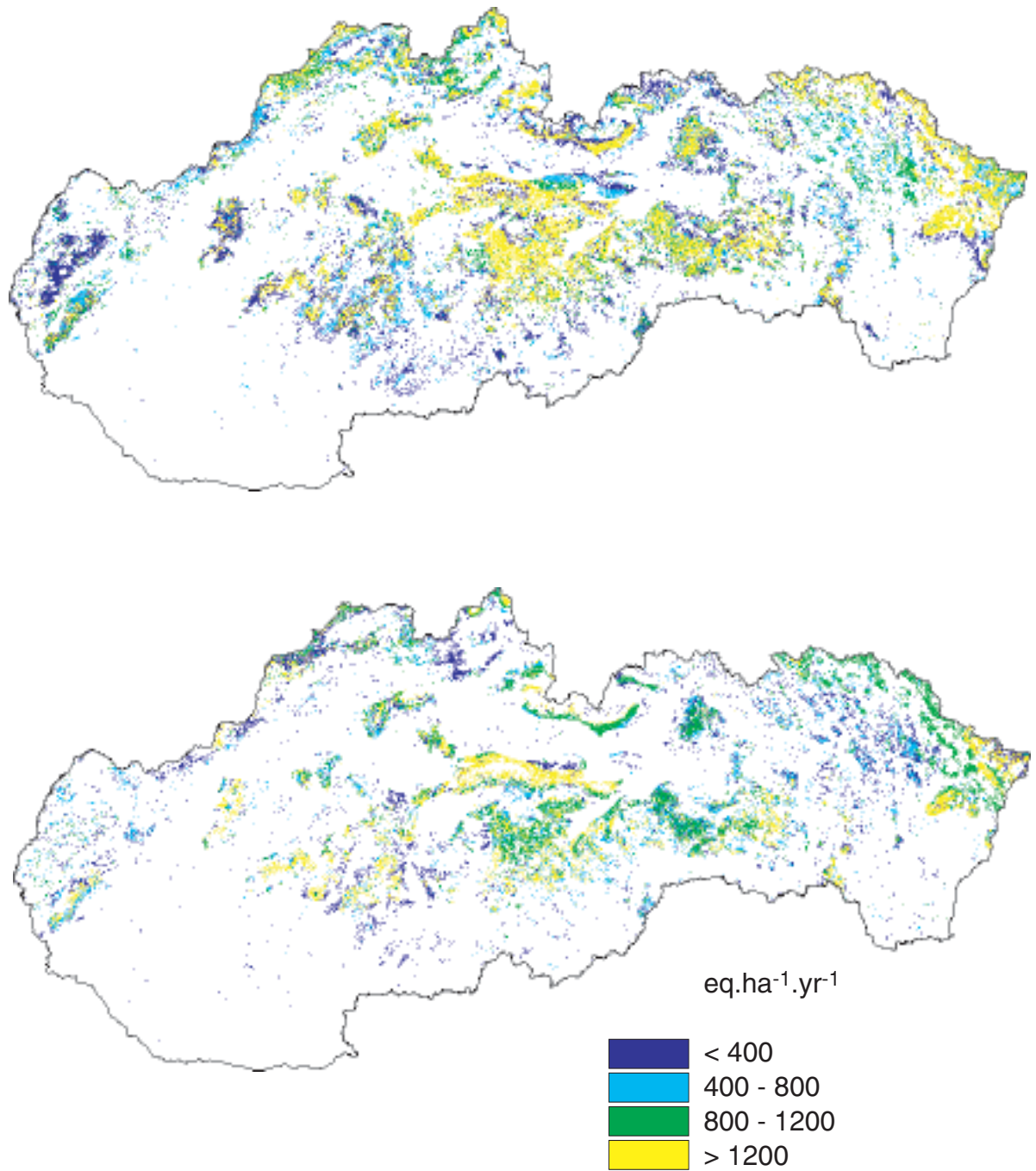
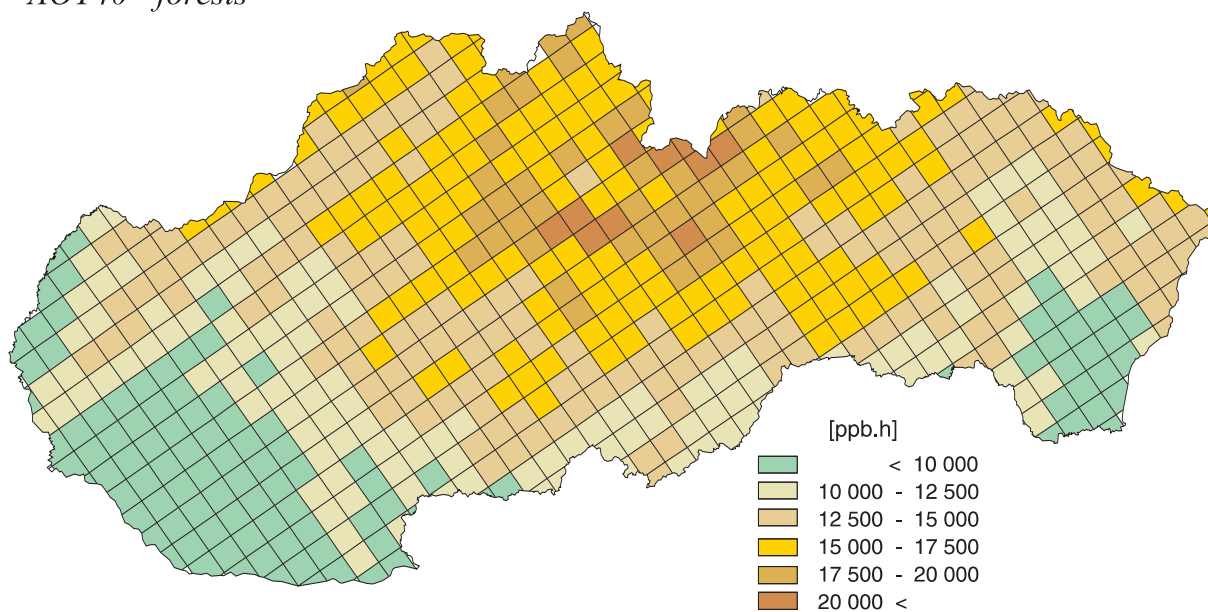


Figure SK-3. Exceedance of critical loads for forest soils in 1990 and 1995.

AOT40 - forests



AOT40 - crops

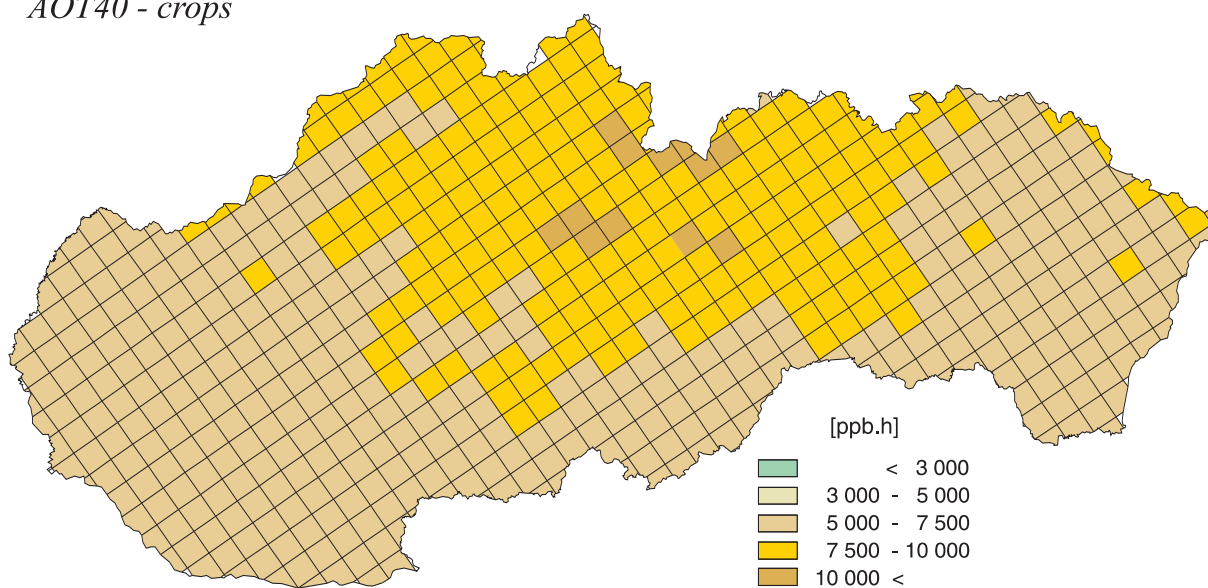


Figure SK-4. AOT40 for forests and agricultural crops in Slovakia in 1992–1996.

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Calculation methods

The critical loads data for Spain have not been changed since 1996. See the CCE Status Report 1997.

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Calculation methods

Forest ecosystems:

Critical loads of acidity for forest ecosystems were calculated using the steady-state mass balance approach, implemented in the PROFILE model. In the model the soil profile is divided into four layers using input data for the thickness of each soil layer (O, A/E, B, C). A critical base cation to Al molar ratio of one in the soil solution was used as the chemical criterion in each soil horizon and used to determine the critical ANC leaching. The critical load functions ($CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$) were calculated according to the Mapping Manual (UBA 1996) with constant N sinks.

The critical load of nutrient nitrogen for forest soils, $CL_{nut}(N)$, was calculated using the steady-state mass balance approach according to the equation:

$$CL(N) = N_u + N_i + N_{de} + N_{le(acc)}$$

where:

N_u = long-term net N uptake by the forest

N_i = N immobilization

N_{de} = denitrification

$N_{le(acc)}$ = acceptable total N leaching

The long-term uptake of N was calculated as the net uptake in forest biomass balanced by the supply of base cations and phosphorus from weathering and deposition. This criterion is introduced to avoid long-term nutrient imbalances in forest trees. The supply of different cations from weathering was calculated using the PROFILE model.

N immobilization was determined by a semi-empirical approach. Immobilization + leaching is assumed to be linearly related to N deposition and set to a maximum of 12 kg N ha⁻¹ yr⁻¹ and a mean of 8 kg N ha⁻¹ yr⁻¹ for southern Sweden at present deposition. This is based on results from N mass balance studies performed in a range of Swedish coniferous forests (Nilsson *et al.* 1998). The immobilization rate for each site was then scaled down from present level to the one pertaining at critical N deposition using an iterative procedure. Denitrification was calculated using the Sverdrup-Ineson equation as given in the Mapping Manual (UBA 1996).

Freshwaters:

Maximum critical loads of sulfur and acidifying nitrogen for freshwater ecosystems, as well as minimum critical loads of N, were calculated using the first-order acidity balance (FAB) model as described in Henriksen *et al.* (1993) and Posch (1995). The chemical threshold, ANC_{limit} , was set to 20 meq l⁻¹ in cases where $[BC]_0^* > 25 \text{ meq l}^{-1}$. In other cases, ANC_{limit} was set to $0.75[BC]_0^*$ to allow for naturally low ANC concentrations. The N immobilization was set to a maximum of 2 kg N ha⁻¹ yr⁻¹ (terrestrial) and then weighted to land use types within the catchment. The average denitrification fraction for each catchment was related linearly to the fraction of peatlands in the catchment area ($f_{de} = 0.1 + 0.7 f_{peat}$) as suggested by Posch *et al.* (1997).

Deposition:

Wet deposition of sulfur and nitrogen as well as air concentrations of sulfur and nitrogen compounds were estimated using a model system, MATCH (Langner *et al.* 1996), based on monitoring data to estimate the long-range transport contribution, along with a dispersion model to estimate the local contribution from Swedish emission sources. The spatial resolution of the model system is 20×20 km².

Dry deposition to forest ecosystems was estimated by inferential modeling based on model-calculated air concentration fields multiplied by dry deposition velocities. The velocities were derived from throughfall data for sulfur and from the literature for nitrogen.

Wet deposition of base cations was estimated based on precipitation chemistry data and MATCH model-estimated precipitation amounts. Total deposition of sodium and chloride was derived from throughfall measurements interpolated between monitoring sites. Deposition of potassium, magnesium, and calcium was estimated from wet deposition, using the same ratio between total and wet deposition as observed for sodium and/or chloride.

Deposition was mapped to different types of ecosystems: Norway spruce, Scots pine/deciduous forest and open land/lakes. Land-use weighted deposition was calculated for 50×50 km² NILU grids.

Mapping:

In computing protection isolines within a grid cell, forest and freshwater ecosystems were given equal weight. The weight assigned to each lake or forest site measured within a grid cell was adjusted so that the total weight of lake ecosystems was equal to that of forest ecosystems in that grid cell. This was done by assigning each lake a weight (km²) equal to half the cell ecosystem area divided by the number of lakes in that cell. For the forest sites, the

weights based on the Swedish Forest Inventory were rescaled up to half the cell ecosystem area. To account for cell areas not at risk from acid deposition, 10% of each cell area was subtracted when calculating the cell ecosystem area.

Data sources

Forest ecosystems:

The forest soil data utilized is based on samplings made within the Swedish Forest Inventory between 1983–1987 (Kempe *et al.* 1992). This Inventory consists of a network of stations evenly distributed over Sweden. Soil samples down to ca. 60 cm depth were collected at 1804 sites. All input data were derived according to Warfvinge and Sverdrup (1995).

Freshwaters:

Water chemistry data were taken from the 1995 Swedish Lake Survey (Wilander *et al.* 1998). Lakes strongly influenced by agriculture were excluded. In total, 2378 lakes were included in the calculation, consisting of 1702 unlimed lakes and 676 additional lakes which were corrected for liming by assuming a constant Ca:Mg ratio for nearby lakes and assuming that Mg concentration was not affected by liming. A long-term average (1961–90) of runoff data was used and taken from the Swedish Meteorological and Hydrological Institute (SMHI).

Land use data and the long-term average of nutrient uptake were derived from the Swedish Forest Inventory 1983–92. In cases where there were not sufficient land use data, the area for the estimation was expanded systematically to include at least 9 to 15 surveyed land plots.

Deposition:

Monitoring data was used as input to the modeling and to more direct deposition estimates. The MATCH model system (Langner *et al.* 1996) requires regional air pollution and precipitation data to assess the contributions from long-range transport. The Swedish contribution and local variations in pollution load were calculated in the MATCH system using an Eulerian atmospheric transport model.

Data used for calculating deposition of sulfur, nitrogen and base cations include:

- Wet deposition monitoring data from the national monitoring network: 30 stations for precipitation chemistry data from other Nordic countries, mainly EMEP sites.
- Throughfall monitoring data from regional forests surveys: approximately 100 sites.
- EMEP air chemistry stations: 6 Swedish stations and 10 stations in other Nordic and Baltic countries.

- Air concentrations from approximately 30 sites with passive sampling of SO₂ and NO₂.
- Data bases on land use and meteorology included in the MATCH model system.

Deviations from the manual

All variables are derived according to the Mapping Manual, except for those in Table SE-1.

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Table SE-1. Deviations from the Mapping Manual.

Variable	Range in the Mapping Manual (kg N ha⁻¹ yr⁻¹)	National range (kg N ha⁻¹ yr⁻¹)	Justification
N_i (freshwaters)	2-5	0-2 median = 1.7	Low N deposition and non-forested, mountainous catchments with low N retention; especially in northern Sweden.
N_i (forest soils)	0.5-1.0 (long-term) 3-10 (short-term)	0-12 median = 2.4	Calculated from empirically established relationship between deposition and N immobilization.
$N_{le(acc)}$ (forest soils)	0.5-1.0	0	$N_{le(acc)}$ included in the N_i term.

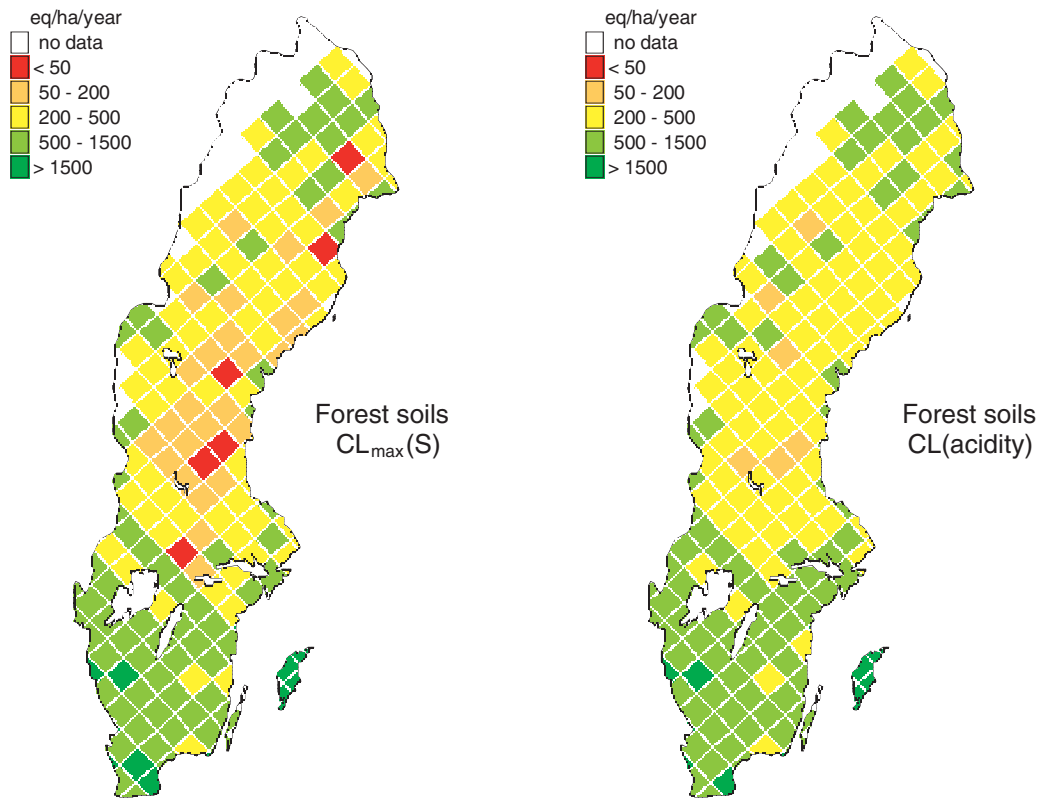


Figure SE-1. The 5th percentile of $CL_{max}(S)$ and $CL(acidity)$ for forest ecosystems.

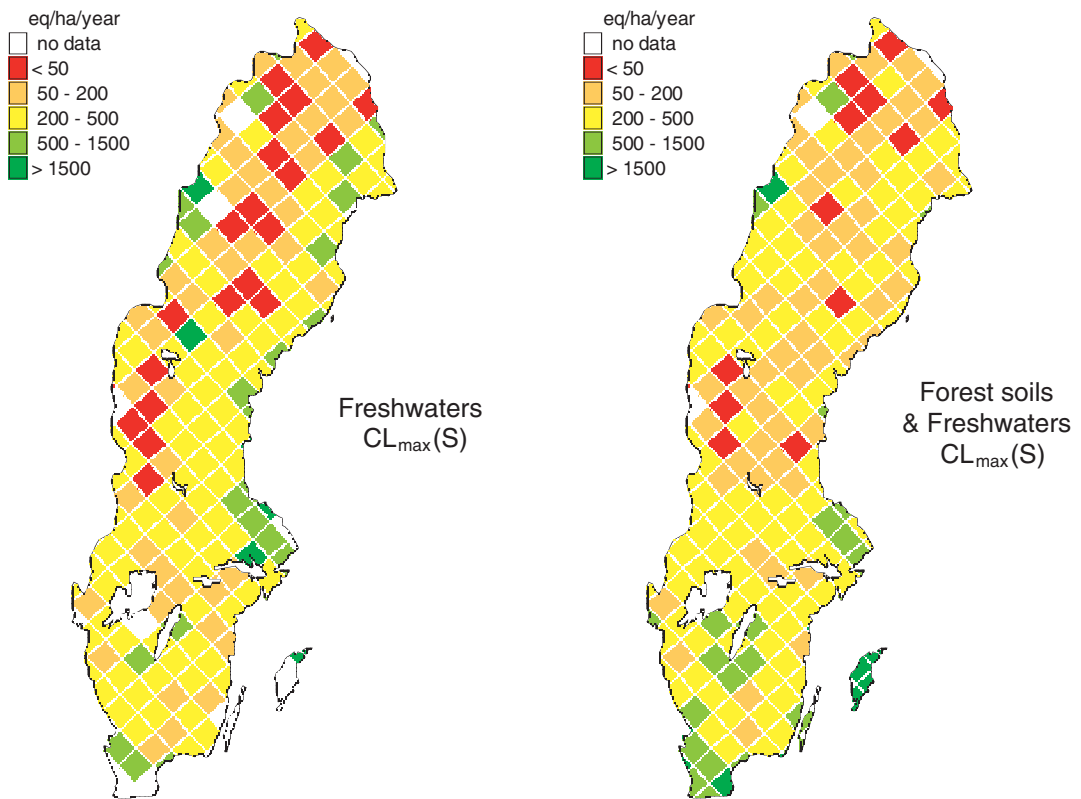


Figure SE-2. The 5th percentile of $CL_{max}(S)$ for freshwaters and forest soils and freshwaters combined.

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Introduction

The Swiss data set used by the CCE has not been changed since December 1996. Therefore, most of the information given in earlier Status Reports and publications is still valid. Concerning methods and data used for calculating critical loads, a few points are clarified in section A. Sections B and C focus on results from the application of the PROFILE and SAFE models. The last section summarizes the legal basis in Switzerland for the protection of sensitive ecosystems.

A. Critical loads: Calculation methods

The methods and data used to calculate critical loads of acidity are described in detail in FOEFL (1994). The SMB method was applied on 11,863 forest sites (Bc/Al criterion and Al-depletion criterion) and on 636 points representing alpine lake catchments (ANC criterion).

The methods and data used to calculate critical loads of nutrient nitrogen are described in detail in FOEFL (1996). The SMB method was applied on 11,863 forest sites. The empirical method was applied on 14,975 sites with sensitive natural or semi-natural ecosystems, such as forests with rich ground flora, species-rich grassland, montane and (sub)alpine grassland, wetlands and alpine heaths.

In most cases, the input parameters were chosen within the ranges recommended in the Mapping Manual. This was *not* the case for nitrogen immobilization, N_i . Values of 4–5 kg N ha⁻¹ yr⁻¹ are used, which are above the recommended range of 0.5–1. The justification for this approach is that: (1) the values also include N losses by fire and erosion, and (2) with the recommended range, critical loads of nutrient nitrogen would become very low in some cases compared with currently known empirical values.

The data set for the CCE was compiled as follows: Critical loads for forests, alpine lakes and semi-natural vegetation are calculated and mapped independently, but with the same spatial resolution of 1×1 km². To calculate critical loads, the following numbers of records (sites) were included, each record representing a raster cell of 1×1 km²: forests, 11,863 records; lakes, 636 records; natural vegetation, 14,975 records; for a total of 27,474 records.

Within one 1×1 km² raster cell more than one ecosystem type can be present. In this case those two (or three) records were aggregated into one record in order to avoid double counting. Thus the data base sent to the CCE was reduced to a total of 23,937 records, which corresponds to a total ecosystem area of 23,937 km². The aggregation was done as follows: If forests and natural vegetation are present in the same 1×1 km² cell, then:

- $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ are calculated for the forest and
- $CL_{nut}(N)$ is the minimum of the SMB-calculated $CL_{nut}(N)$ and the empirical $CL_{nut}(N)$.

For lakes and natural vegetation the procedure is similar.

B. Critical loads: PROFILE model results

A regionalized version of the multi-layer, steady-state soil chemistry model PROFILE (Warfvinge and Sverdrup 1992a) was used to calculate critical loads of acidity for Swiss forest soils at 720 sites of the National Forest Inventory. A base cation to total aluminum molar ratio $Bc/Al \geq 1$ in the soil solution of the tree rooting zone was used as the critical chemical limit. Physico-chemical soil parameters needed by the PROFILE model to calculate the weathering rates were derived from: 1) national surveys such as the National Forest Inventory, covering site-specific information, 2) available point measurements of parameter values and 3) literature sources. Not all parameters were available on a regional scale with sufficient resolution. Input required for the model calculations was therefore derived from the available data sources by means of transfer algorithms including spatial interpolation (SAEFL 1998a).

Critical loads of acidity for Swiss forest soils calculated with the regionalized PROFILE model range from 0.1 to 10.3 $\text{keq ha}^{-1} \text{ yr}^{-1}$ (Figure CH-1). 80% of the sites yield critical loads between 1 and 4 $\text{keq ha}^{-1} \text{ yr}^{-1}$. Weathering rates range below 1 $\text{keq ha}^{-1} \text{ yr}^{-1}$ at 80% of the forest sites considered (Figure CH-2). Critical loads for Swiss forest soils are frequently (approx. 60% of the sites), substantially exceeded by present loads when 1990 is taken as a reference year.

The results obtained from the application of the regionalized PROFILE model were also compared with the output of the Simple Mass Balance SMB (FOEFL 1994). The single-layer SMB has been applied to 11,863 receptor points representing the total forested area of Switzerland. Model-independent input data, in particular deposition and vegetation data, were harmonized prior to the calculation.

PROFILE generally predicts lower critical loads than the SMB (Figure CH-1). However, the effective difference is reduced by considering an aluminum-depletion criterion with the standardized SMB method in addition to the Bc/Al criterion. Still, PROFILE pentile critical loads estimated for the Swiss area in the EMEP grid cells (23,12–14) in the Jura Mountains and Swiss Plateau remain 0.1 to 0.7 $\text{keq ha}^{-1} \text{ yr}^{-1}$ below the standardized SMB predictions. It can be concluded that SMB results have to be considered as conservative estimates for these grid cells with respect to their use for policy decisions on emission reductions in Europe. For the other EMEP grid cells (24,13) and (23,14) in the Alps and Southern Switzerland, PROFILE pentile critical loads are approximately 0.5 $\text{keq ha}^{-1} \text{ yr}^{-1}$ higher than SMB values.

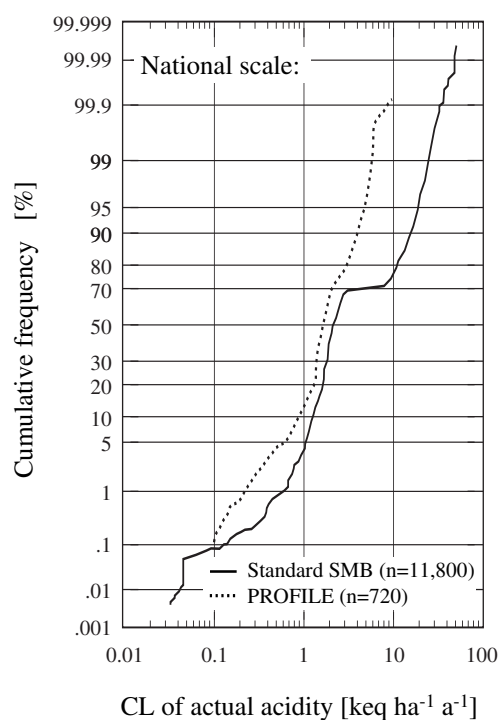


Figure CH-1. Cumulative frequency distribution of PROFILE (dotted line) and SMB (solid line) critical loads of acidity.

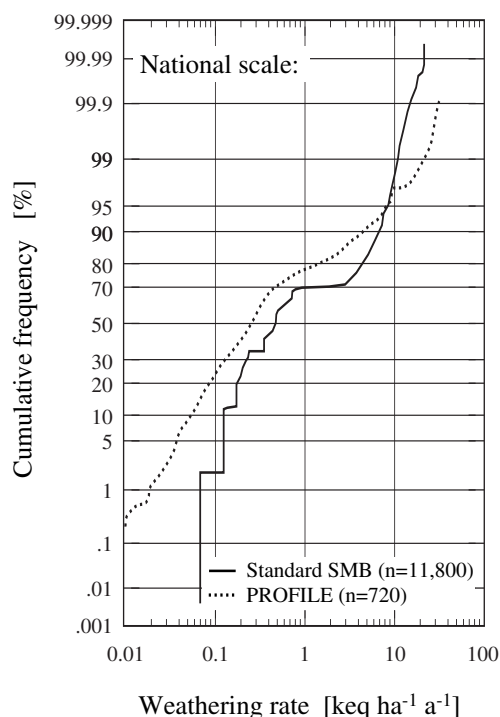


Figure CH-2. Cumulative frequency distribution of PROFILE (dotted line) and SMB (solid line) weathering rates.

An analysis of critical load predictions of the two model approaches suggests several model-dependent sources of prediction discrepancies. 30% of the calculated critical loads of acidity for forest soils differ due to the inherent inability of the SMB to properly account for reactions in the carbonate system. 70% of the critical loads differ due

to discrepancies in the weathering rates used for the calculations. A better agreement of the two weathering rate populations minimizes discrepancies between the two model predictions and improves the quality of the point-to-point agreement. More than 75% of the sites would produce critical loads within $\pm 20\%$ if identical weathering rates were to be used with both calculation methods (Figure CH-3). This remaining difference arises from soil stratification and processes such as aluminum speciation and complexing of aluminum with organic acids, additionally considered in the PROFILE model. For intermediate- to low-weathering forest soils in high-precipitation areas, where hydrogen and aluminum leaching govern the model result, both models estimate comparably distributed critical loads independent of the method used for estimating weathering rates.

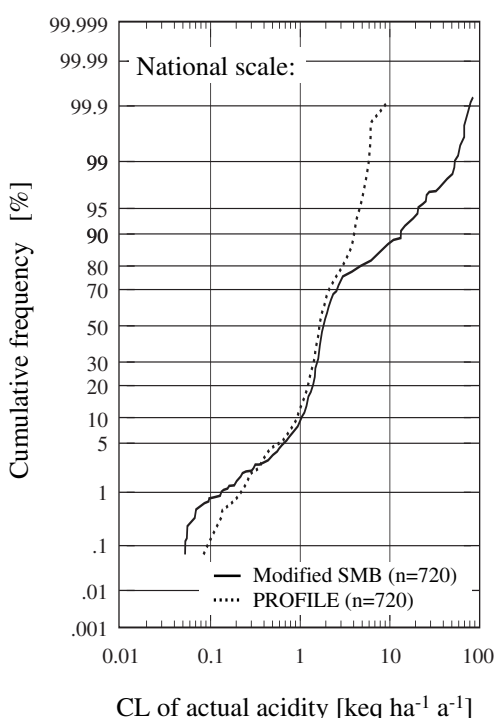


Figure CH-3. Cumulative frequency distribution of critical loads of acidity calculated with SMB (solid line) and PROFILE (dotted line) using weathering rates from PROFILE calculations.

C. Dynamic modeling

The dynamic, process-oriented, multi-layer soil chemistry model SAFE (Warfvinge and Sverdrup 1992b) reconstructs acidification of terrestrial ecosystems by calculating the temporal development of various chemical state variables. SAFE was applied to the rooting zone of ca. 600 evenly distributed forest sites in Switzerland (4x4 km² grid resolution) and a time period between 1850 and 2100 (SAEFL 1998b). The objective was to clarify if and when the chemical status of Swiss forest soils will improve, given several European air pollutant emission reduction scenarios: current reduction plans (CRP), current legislation (CLE) and maximum technically feasible reductions (MTFR). The model application was preceded by a comprehensive revision of the general model approach, particularly of the input data generation necessary for a regional application.

Simulations imply that the present-day chemical status of Swiss forest soils is essentially a result of the last 50 years' acid deposition. Indicative soil parameters such as soil solution pH, total aluminum concentration, acid neutralizing capacity, base cation to total aluminum (Bc/Al) molar ratio and the base saturation, have consistently deteriorated since the beginning of the 1950s, when acid loads started to increase. Modeled pH dropped by an average of 0.6 units in the organic soil layers, and by an average of 1.1, 0.8 and 0.6 units in the model A/E, B and C layers respectively, between the late 1940s and today. Average total aluminum concentration increased by a factor of 10 to 40 in the four soil layers, starting at values below 15 $\mu\text{eq l}^{-1}$ during the same time period. Bc/Al molar ratios decreased by several orders of magnitude (Figure CH-4). 50% of the sites' upper soil layers had Bc/Al molar ratios below 3.7 by the late 1970s, whereas the median value in the lower soil layers falls below 1.9 around the year 2000. Exchangeable base cations are currently markedly depleted at approximately 50% of the sites.

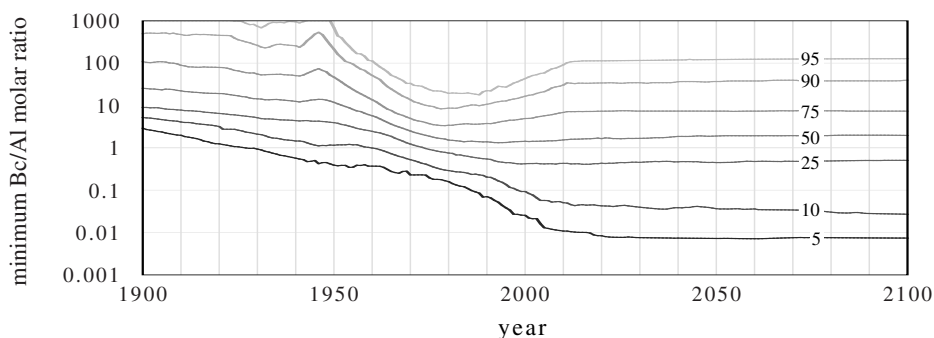


Figure CH-4. Percentile traces of soil solution Bc/Al molar ratio minima as obtained from applying the CLE emission/deposition reduction scenario (national scale, 600 forest sites).

The adverse trend of deteriorating soil chemistry is stopped in the late 1970s in the upper soil layers, and during the first half of the next century in the deeper horizons. CRP and CLE scenarios for sulfur and nitrogen emissions indicate an improvement in upper soil conditions and a halt of acidification of the lower soil horizons in the long-term. The MTFR scenario would substantially improve the chemistry of Swiss forest soils. Changes in the soil parameter values relative to the results obtained from the CLE scenario are orders of magnitude larger. MTFR would additionally lead to partial recovery of the lower soil layers.

To assess the risk potential of acidification, the simulations for selected chemical parameters of the soil solution were compared with currently used critical threshold values. The pH threshold for the second soil layer (4.0), the critical minimum acid neutralizing capacity ($-300 \mu\text{eq l}^{-1}$) and the critical maximum total aluminum

concentration ($200 \mu\text{eq l}^{-1}$) are all violated at a maximum of 50 to 60% of the sites in the late 1970s, shortly after the peak of acid deposition around 1975 (Figure CH-5). These parameters indicate recovery from acidification. Only 17 to 25% of the sites under the CLE scenario and < 10% of the sites under the MTFR scenario continue to have threshold violations by the year 2100. The pH of the third layer (threshold 4.4) and minimum Bc/Al molar ratio (threshold 1.0) on the other hand respond, according to the model predictions, very sluggishly to the assumed acid load decline, with up to 37% (under the CLE scenario) and up to 26% (MTFR) respectively, of the sites still violating the criteria by 2100 (Figure CH-6). The persistence of violations of conventionally used critical chemical parameter values to the end of the simulation period also points to the fact that currently enacted acidifying deposition reductions are not sufficient to attain a long-term sustainable forest ecosystem in Switzerland.

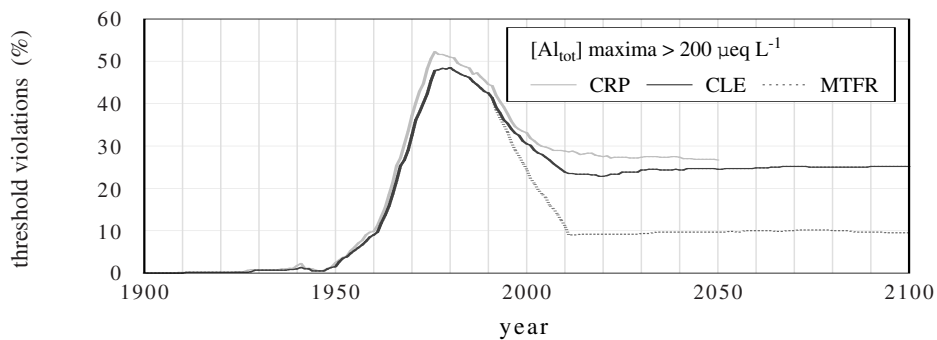


Figure CH-5. Time series of the percentage of sites exceeding the threshold value for $[\text{Al}]_{\text{tot}}$ (national scale, 600 forest sites).

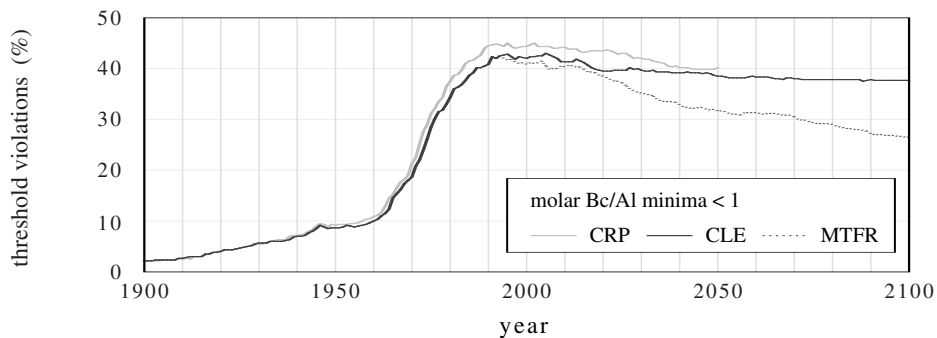


Figure CH-6. Time-series of the percentage of sites with Bc/Al minima below 1 (national scale, 600 forest sites).

D. Legal framework for the protection of sensitive receptors

Switzerland contains many ecosystems which have to be protected according to Swiss legislation (Constitution, Laws and Ordinances; see below) and to international agreements (e.g. Alpine Convention). The ecosystems selected within the mapping procedure with respect to their sensitivity to acidification and/or eutrophication are all identified and designated in Swiss legislation and in international agreements as being worth protecting: forests (coniferous, deciduous, managed, unmanaged, rare woodland communities), dry meadows and pastures, bogs, fens, mire landscapes, alluvial zones, and alpine biotopes in general.

Moreover, Swiss legislation contains criteria according to which impact thresholds for air pollutants have to be set. These criteria are fully effects-oriented and thus in agreement with the definition of critical levels and critical loads used within the UN/ECE Convention on Long-range Transboundary Air Pollution.

Federal Constitution

The Federal Constitution sets the general framework for the protection of human beings and the environment against harmful effects or nuisances, especially by air pollution and noise (Article 24^{septies}). It stipulates that the federal authorities have to enact the necessary legal instruments to ensure this protection.

The Constitution also contains various obligations for the federal authorities in the field of the protection of nature, landscapes and cultural heritage (Art. 24^{sexies}). There, the protection of animals and plants is explicitly mentioned, as well as the protection of bogs and mire landscapes (including fens and raised bogs) due to a national referendum on this issue in 1987. The protection of waters is requested in Art. 24^{bis}, the protection of forests in Art. 24.

Laws

Federal Law relating to the Protection of the Environment:

The purpose of this law is: "... to protect human beings, animals and plants, their biological communities and habitats against harmful effects or nuisances and to maintain the fertility of the soil. Early preventive measures shall be taken in order to limit effects which could become harmful or a nuisance."

Effects under this law are defined in Art. 7: "Effects means air pollution, noise, vibrations, radiation, water pollution or other intrusions in waters, soil pollution, modifications of the genetic material of organisms or modifications of

the natural composition of biological communities, caused by the construction and operation of installations, by the handling of substances, organisms or wastes or by the cultivation of the soil." The law also indicates how effects have to be assessed (Art. 8): "Effects shall be assessed singly, collectively and according to their actions in combination."

Moreover, the law clearly stipulates how impact thresholds for air pollutants must be set (Art. 14): "Impact thresholds for air pollutants shall be set in such a way that, in the light of current scientific knowledge and experience, pollution burden below these levels:

- a. do not endanger human beings, animals and plants, their biological communities and habitats;
- b. do not seriously disturb the well-being of the population;
- c. do not damage buildings;
- d. do not harm soil fertility, vegetation or waters."

In addition, the law specifies that, by setting impact thresholds, not only the average sensitivity of receptor groups shall be taken into account, but particularly the higher sensitivity of groups such as children, the sick, elderly people and pregnant women (Art. 13).

Federal Law relating to the Protection of Nature and Cultural Heritage:

Provisions for the protection of animals, plants and their biotopes are specified in Articles 18–23 of the federal law relating to the protection of nature and cultural heritage.

Art. 18 stipulates the conservation of sufficiently large biotopes to prevent the extinction of indigenous animal and plant species. Protection of the following biotopes is particularly requested: alluvial zones (river banks and lake shores), reedy marshes and bogs, rare woodland communities, hedges (thickets), field copses, dry meadows and pastures and further sites with balancing functions in nature. According to Art. 18a and 23b the Federal Council designates, after consultation with the Cantons, biotopes of national importance, as well as bogs and mire landscapes of particular beauty and of national importance, and determines their boundaries and the specificity of the objects to be protected. The designated biotopes are put together in inventories which are specified in specific ordinances.

Federal Law relating to Forests:

The Federal Law relating to Forests, according to the purpose article (Art. 1), shall *inter alia* "protect forests as a semi-natural community and make sure that forests can fulfil their functions, such as the protective, welfare and economic function."

Ordinances:

Ordinance on Air Pollution Control:

Contains *inter alia* for several air pollutants effects-orientated ambient air quality standards set according to the criteria in Art. 14 of the Federal Law relating to the Protection of the Environment. Air pollution concentrations or depositions are defined to be excessive if one or more ambient air quality standards are exceeded. If for a certain pollutant no specific ambient air quality standard is set, then the pollution burden has to be considered excessive, if:

- a. it endangers human beings, animals and plants, their biological communities and habitats;
- b. it seriously disturbs, on the basis of an investigation, the well-being of a substantial part of the population;
- c. it damages buildings;
- d. it harms soil fertility, vegetation or waters.

Ordinance relating to the Protection of Nature and Cultural Heritage:

The Ordinance relating to the protection of nature and cultural heritage contains provisions with respect to the protection of biodiversity. It contains criteria (indicator plants) to determine the biotopes to be protected (ombrotrophic bogs, transition bogs, fens, alluvial zones, dry meadows and pastures, dry woodland communities and bushes, forests on steep slopes), and it lists the plants and animals to be protected, *inter alia* all amphibians, reptiles, bats, hedgehogs, dragonflies, some butterflies and beetles, and a number of plant species including all orchids.

Ordinance relating to the Federal Inventory of Landscapes and Cultural Heritages:

This Ordinance lists the landscapes and cultural heritages of national importance. The inventory currently contains 153 landscapes of national importance.

Ordinance on the Protection of Alluvial Zones of National Importance:

This Ordinance lists the alluvial zones of national importance to be protected. The inventory associated with it currently contains 169 separate zones.

Ordinance on the Protection of Ombrotrophic Bogs and Transition Bogs of National Importance:

This Ordinance contains a federal inventory of raised bogs and transition bogs. It identifies and designates those areas of national importance which are particularly valuable, beautiful and therefore worth protecting. The inventory currently contains 527 ombrotrophic and transition bogs of national importance.

Ordinance on the Protection of Fens of National Importance:

This Ordinance contains a federal inventory of fens. It identifies and designates those areas of national importance which are particularly valuable, beautiful and therefore worth protecting. The inventory currently contains 1119 fens of national importance.

Ordinance on the Protection of Mire Landscapes of Particular Beauty and National Importance:

This Ordinance contains a federal inventory of mire landscapes. It identifies and designates those areas of national importance which are particularly valuable, beautiful and therefore worth protecting. The inventory currently contains 88 mire landscapes of national importance.

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Calculation methods

The methods for calculating critical loads of acidity and nutrient nitrogen for ecosystems in the UK have been described previously (Hall *et al.* 1998, Posch *et al.* 1997). This information is now also available via the ITE Web site at: www.nmw.ac.uk/ite/monk/critical_loads/nclmp.html.

The ranges of values used in these calculations are given in Table UK-1. For most parameters Mapping Manual guidelines have been followed in the UK. The exceptions to this are:

- (i) using the Simple Mass Balance equation with the ratio of Ca:Al as the chemical criterion.
- (ii) using non-marine chloride deposition estimates for 2010.
- (iii) the values for empirical critical loads of nutrient nitrogen for calcareous grassland.
- (iv) the values for nitrogen immobilization.
- (v) the use of catchment-weighted denitrification values in applying the First-order Acidity Balance (FAB) model, instead of the denitrification fraction.

The justification for applying these values and/or methods in the UK are also given in Table UK-1.

Table UK-1. Summary of UK critical load values and the justification for their use.

Critical loads parameter (and units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification		
$CL_{max}(S)$ (eq/ha/yr)	g	170	4769	$= CL(A) + (BC_{dep}^* - CI_{dep}^*) - BC_u$	Mapping Manual.		
	h	170	4702				
	c	17	13,634				
	d	48	12,132				
	w	0	106,536			$= L_{crit} / (1 - \rho_S)$	
$CL_{min}(N)$ (eq/ha/yr)	g	141	928	$= N_u + N_i$	Mapping Manual. See comments on N_i and N_u .		
	h	361	504				
	c	350	493				
	d	349	492				
	w	15	644			$= fN_u + (1 - r)(N_i + N_{de})$	
$CL_{max}(N)$ (eq/ha/yr)	g	351	5408	$= CL_{max}(S) + CL_{min}(N)$	Mapping Manual.		
	h	561	5206				
	c	367	13,985				
	d	538	12,481				
	w	274	447,194			Mapping Manual. N_{de} values catchment-weighted according to area of different soils present in catchment.	
$CL_{nut}(N)$ (eq/ha/yr)	g	714	3571	Empirical values applied: Acid grassland: 10, 12.5, 25 kg N/ha/yr depending on species present. Calcareous grassland: 50 kg N/ha/yr	Mapping Manual. Empirical values recommended by UK experts (Hall <i>et al.</i> 1998).		
	h	714	1214			Empirical values applied: 10, 15, 17 kg N/ha/yr depending on species present.	
	c	851	1208			Mass balance calculation: $= N_u + N_i + N_{le(acc)} + N_{de}$. N_{de} values between 1–4 kg N/ha/yr depending on soil type.	Mass balance equation as in Mapping Manual. Input values recommended by UK experts (Hall <i>et al.</i> 1998) and related to soil type.
	d	850	1207			Minimum of empirical value (17kg N/ha/yr) or mass balance (where $CL_{nut}(N) = N_u + N_i + N_{le(acc)} + N_{de}$ value applied.	Mapping Manual. Empirical values recommended by UK experts. Input values to mass balance equation recommended by UK experts (Hall <i>et al.</i> 1998) and related to soil type.
	w	–	–			Not calculated.	
$BC_{dep}^* - CI_{dep}^*$ (eq/ha/yr)	g	70	775	BC_{dep}^* = measured mean data 1992–94 for low vegetation. CI_{dep}^* = estimate for 2010	We have measurements of present-day CI_{dep}^* and can model future predictions of CI_{dep}^* . Predicted CI_{dep}^* for 2010 used to match the end date of the 2 nd Sulphur Protocol.		
	h	70	775				
	c	119	1149			BC_{dep}^* = measured mean data 1992–94 for woodland ecosystems. CI_{dep}^* = estimate for 2010	
	d	119	1149				
	w	–	–			Not used.	

CCE Ecosystem Codes:

g = grassland
h = heathland
c = coniferous forest
d = deciduous forest
w = waters

Table UK-1 (continued). Summary of UK critical load values and the justification for their use.

Critical loads parameter (and units)	Eco-system code [#]	Min. value	Max. value	Data sources/ Methods used	Justification
BC_u (eq/ha/yr)	g	0	222	Minimum value: uptake negligible for acid grassland. Maximum value: uptake for calcareous grassland (including removal via sheep).	Based on published data by UK experts.
	h	0	0	No uptake for heathland.	Based on published data. Single value for UK for each of the following: coniferous woodland (all soils), deciduous woodland (Ca-poor soils), deciduous woodland (Ca-rich soils). Regional and species-specific volume increment and concentration in wood to be incorporated in future. NB: Estimates of calcium uptake used in SMB.
	c	252	252	Calculated from: average volume increment \times basic wood density \times concentration in wood, and assuming potential yields achieved. Values based on data for Sitka spruce.	
	d	171	612	Calculated from: average volume increment \times basic wood density \times concentration in wood, and assuming potential yields achieved. Values based on data for oak.	
	w	–	–	Not used.	
BC_w (eq/ha/yr)	g	0	4000	Empirical critical loads of acidity for soils (Skokloster classification). Peat soils set to zero.	Recommended in Mapping Manual. See Hornung <i>et al.</i> 1995. Assigned values checked against application of PROFILE for limited number of sites.
	h	0	4000		
	c	0	4000		
	d	0	4000		
	w	–	–	Not used.	
$ANC_{le(crit)}$ (eq/ha/yr)	g	0	0	Set to zero as only empirical critical loads for grassland ecosystems.	Methods agreed by UK experts (Hall <i>et al.</i> 1998). (SMB only applied to woodland ecosystems in UK).
	h	0	0		
	c	100	9237	Calculated via SMB equation with ratio of Ca:Al = 1 as chemical criterion.	SMB with BC/Al ratio and base cation deposition produced unrealistically high critical loads. Ca:Al ratio recommended in paper by Cronan and Grigel (1995).
	d	0	8235		
	w	–	–	For freshwaters the ANC_{limit} is set at zero $\mu\text{eq/l}$.	Value selected for 50% probability of damage to brown trout populations.
N_u eq/ha/yr	g	70	713	1 kg N/ha/yr for acid grassland, 10 kg N/ha/yr for calcareous grassland.	Based on published data by UK experts
	h	289	289	4 kg N/ha/yr for heathland.	
	c	279	279	Methods as for BC_u .	Based on published data: one value for whole of UK. Regional growth values to be incorporated in future.
	d	277	277		
	w	0	268	Uses N_u value of 279 eq/ha/yr for all coniferous forest multiplied by percentage forest in catchment.	Based on published data: Curtis <i>et al.</i> (1998).

[#] CCE Ecosystem Codes:

g = grassland

h = heathland

c = coniferous forest

d = deciduous forest

w = waters

Table UK-1 (continued). Summary of UK critical load values and the justification for their use.

Critical loads parameter (and units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification
N_i (eq/ha/yr)	g	71	214	Dependent on soil type.	Based on published data for long-term sustainability.
	h	71	214		
	c	71	214		
	d	71	214		
	w	8	214	N_i values catchment-weighted according to area of different soils present in catchment.	
$N_{le(acc)}$ (eq/ha/yr)	g	140	140	2 kg N/ha/yr for acid grassland and calcareous grassland.	Agreed by UK experts.
	h	140	140	2 kg N/ha/yr for heathland.	
	c	430	430	6 kg N/ha/yr for woodland ecosystems.	Values based on data from a limited number of detailed site studies for GB plantations.
	d	430	430		
	w	–	–	Not used.	
Denitrification fraction (f_{de})	g	–	–	Not used.	Use of f_{de} (0.1–0.8) as in Mapping Manual gives N_{de} values up to 25kg N/ha/yr, much too high for UK (Curtis <i>et al.</i> 1998).
	h	–	–		
	c	–	–		
	d	–	–		
	w	–	–	Uses catchment-weighted N_{de} (3–286 eq/ha/yr) values instead of f_{de} .	
Precipitation surplus Q (m)	g	–	–	Not used.	Used in SMB equation for acidity critical loads.
	h	–	–		
	c	0.057	3.876	1km ² runoff data based on 30-year mean rainfall data.	
	d	0.057	3.876		
	w	0.205	4.154	1km ² catchment-weighted runoff based on mean rainfall data for 1992–94.	
K_{gibb} (m ⁶ /eq ²)	g	–	–	Not used.	Value selected as a compromise between values for soils with low organic matter content and mineral soils.
	h	–	–		
	c	950	950	Mapping Manual.	
	d	950	950		
	w	–	–	Not used.	

CCE Ecosystem Codes:

g = grassland

h = heathland

c = coniferous forest

d = deciduous forest

w = waters

Note: “g” represents the grassland ecosystem category used by the CCE. The UK calculate critical loads separately for acid grassland and calcareous grassland, however, the CCE assign them both to their grassland category. The above table provides the information for the two ecosystems combined.

Revisions made to UK critical loads data submitted to the CCE between 1997 and 1998

UK critical loads data for 1998 were submitted to the CCE in January 1998. In addition, new data for the Shetland Islands, in the EMEP 150×150km² square (14,19) were submitted at the beginning of June 1998 to complete the UK data set.

The following information describes the changes made to the UK data sets from the 1997 to 1998 submissions:

- (i) The inclusion of critical loads data for Northern Ireland (none previously submitted) for acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland ecosystems.
- (ii) The inclusion of FAB data for freshwaters in Great Britain (not currently available for Northern Ireland).
- (iii) Nitrogen immobilization values based on soil type alone. We are no longer subtracting nitrogen fixation from nitrogen immobilization, since experts in the UK are not happy with the fixation values.
- (iv) Denitrification values based on soil type alone. We previously modified these values to incorporate functions for soil moisture and temperature; however, UK experts are not satisfied with the resulting values using the available data for these functions.
- (v) ANC_w and $ANC_{le(crit)}$ values were transposed in the data submission in 1997. This was corrected in the data submission for 1998.
- (vi) Base cation deposition: The chloride deposition values used in the chloride correction are estimates of chloride deposition for 2010; this is consistent with the conclusions from discussions in the Task Force on Mapping (December 1996), where the end date of the Second Sulphur Protocol was identified as a suitable year for estimating chloride deposition. The chloride values have been estimated by multiplying the current (1992–94) measured values by the ratio of modeled chloride deposition for 2010 to current modeled chloride deposition. For base cation deposition, current measured non-marine data are used since no estimates of future base cation deposition are available. There are separate values of base cation deposition for woodland and non-woodland terrestrial ecosystems. This is what should also have been submitted in 1997, but it appears that current (1992–94) measured chloride was used instead in the chloride correction.

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Appendix A. The polar stereographic projection (EMEP grid)

To make critical loads useful for pan-European negotiations on emission reductions one has to be able to compare them to deposition estimates. Deposition of sulfur and nitrogen compounds have up to now been reported by EMEP on a 150×150km² grid covering (most of) Europe, but recently depositions have also become available on a 50×50km² subgrid. These grid systems are special cases of the so-called polar stereographic projection. This Appendix describes this projection and how to calculate the area of a grid cell.

The polar stereographic projection:

In the polar stereographic projection each point on the Earth's sphere is projected from the South Pole onto a plane perpendicular to the Earth's axis and intersecting the Earth at a fixed latitude ϕ_0 (see Figure A-1, top). Consequently, the coordinates x and y are obtained from the geographical longitude λ and latitude ϕ (in radians) by the following equations (see Figure A-1, bottom):

$$x = x_p + M \tan\left(\frac{\pi}{4} - \frac{\phi}{2}\right) \sin(\lambda - \lambda_0) \quad (\text{A.1})$$

and

$$y = y_p - M \tan\left(\frac{\pi}{4} - \frac{\phi}{2}\right) \cos(\lambda - \lambda_0) \quad (\text{A.2})$$

where (x_p, y_p) are the coordinates of the North Pole; λ_0 is a rotation angle, i.e. the longitude parallel to the y -axis; and M is the scaling of the x - y coordinates. In the above definition the x -values increase and the y -values decrease when moving towards the equator. For a given M , the unit length (grid size) d in the x - y plane is given by

$$d = \frac{R}{M} (1 + \sin \phi_0) \quad (\text{A.3})$$

where R ($= 6370\text{km}$) is the radius of the Earth. The inverse transformation, i.e. longitude and latitude as function of x and y , is given by

$$\lambda = \lambda_0 + \arctan\left(\frac{x - x_p}{y_p - y}\right) \quad (\text{A.4})$$

and

$$\phi = \frac{\pi}{2} - 2 \arctan(r/M) \quad \text{with} \quad r = \sqrt{(x - x_p)^2 + (y - y_p)^2} \quad (\text{A.5})$$

The \arctan in Eq. A.5 gives the correct longitude for quadrant 4 ($x > x_p$ and $y < y_p$) and quadrant 3 ($x < x_p$ and $y < y_p$); π ($=180^\circ$) has to be added for quadrant 1 ($x > x_p$ and $y > y_p$) and subtracted for quadrant 2 ($x < x_p$ and $y > y_p$). Note that quadrant 4 is the one covering (most of) Europe.

Every stereographic projection is a so-called conformal projection, i.e. an angle on the sphere remains the same in the projection plane, and vice versa. However, the stereographic projection distorts areas (even locally), i.e. it is not an equal-area projection. (In fact, it can be shown that a non-trivial projection cannot be both conformal and equal-area.)

We define a **grid cell** (i, j) as a square in the x - y plane with side length d (see Eq. A.3) and center point as the integral part of x and y , i.e.

$$i = \text{nint}(x) \quad \text{and} \quad j = \text{nint}(y) \quad (\text{A.6})$$

where 'nint' is the nearest integer (rounding function). Consequently, the corners of the grid cell have the coordinates $(i \pm 1/2, j \pm 1/2)$.

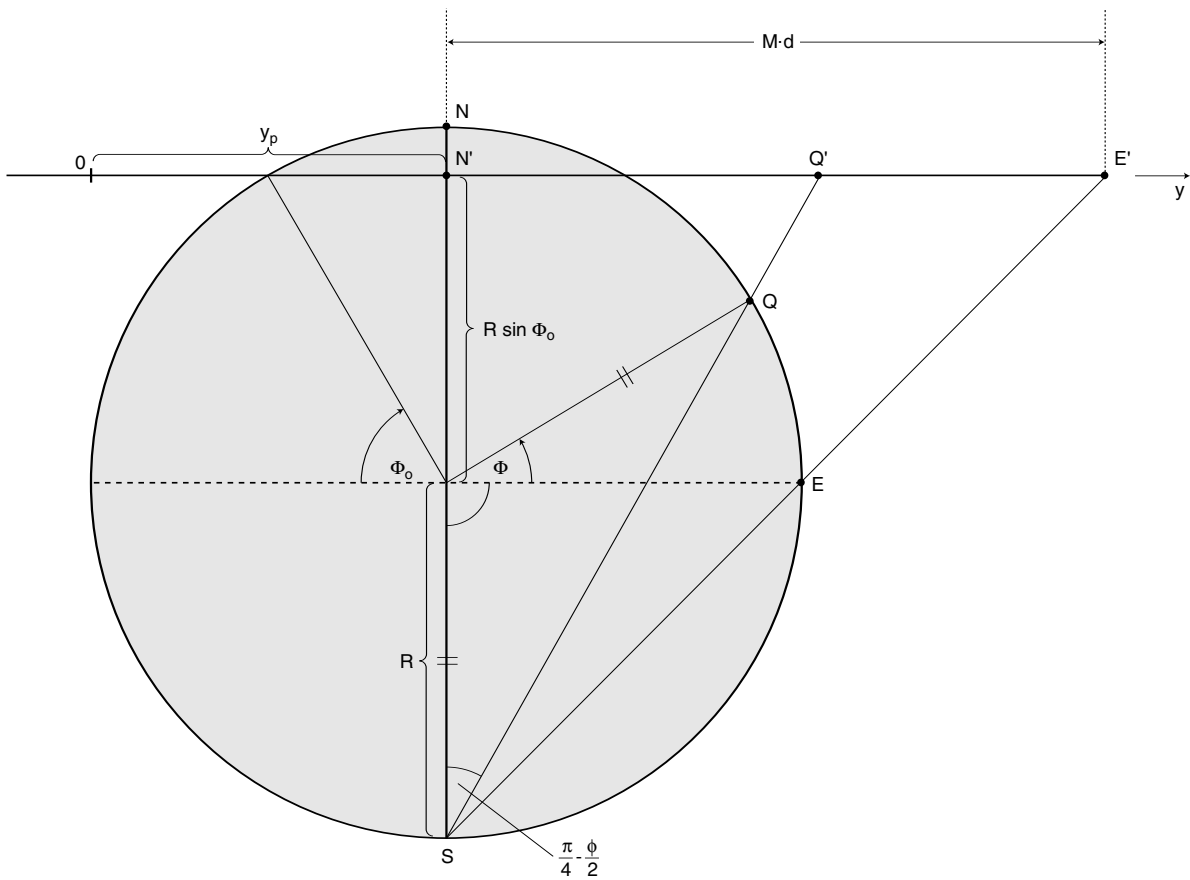
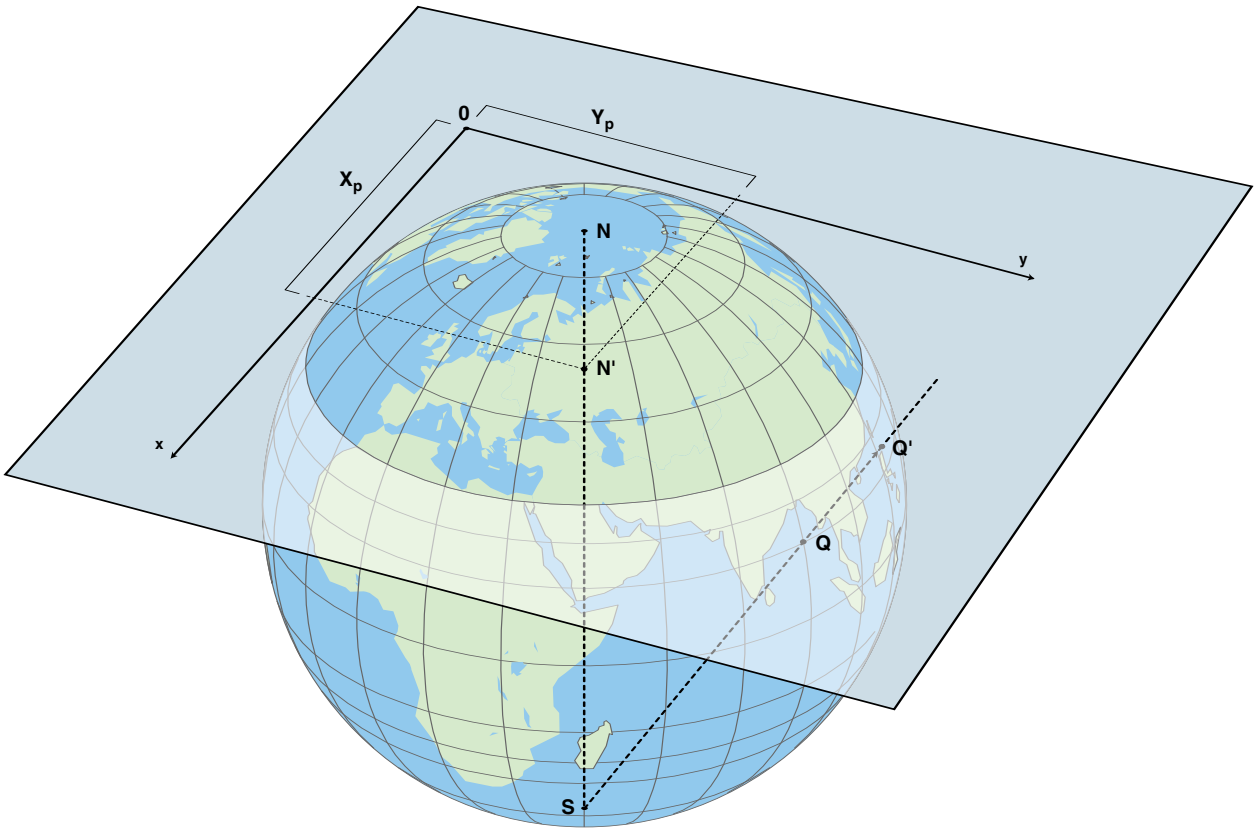


Figure A-1. Polar stereographic projection from the South Pole onto a plane cutting the Earth at a given latitude (top). Geometric relationships in a plane cutting the Earth vertically at a given longitude used to derive the projection equations (bottom).

The 150×150km² grid (EMEP150 grid):

The coordinate system used by EMEP/MSC-W for the Lagrangian long-range transport model is defined by the following parameters:

$$\phi_0 = \frac{\pi}{3} = 60^\circ\text{N}, \quad \lambda_0 = -32^\circ (\text{i.e. } 32^\circ\text{W}), \quad (x_p, y_p) = (3, 37), \quad d = 150\text{km} \quad (\text{A.7})$$

which yields $M=79.2438\dots$

The 50×50km² grid (EMEP50 grid):

Since in the future deposition and concentration fields will become available of a 50×50km² grid, critical loads are currently reported on a grid with the parameters

$$\phi_0 = \frac{\pi}{3} = 60^\circ\text{N}, \quad \lambda_0 = -32^\circ (\text{i.e. } 32^\circ\text{W}), \quad (x_p, y_p) = (8, 110), \quad d = 50\text{km} \quad (\text{A.8})$$

yielding $M=237.7314\dots$

As this is a subdivision of the EMEP150 grid, EMEP50 coordinates p and q are obtained from the EMEP150 coordinates x and y via

$$p = 3x - 1 \quad \text{and} \quad q = 3y - 1 \quad (\text{A.9})$$

An EMEP150 grid cell (i, j) contains $3 \times 3 = 9$ EMEP50 grid cells (m, n) with indices $m=3i-2, 3i-1, 3i$ and $n=3j-2, 3j-1, 3j$. The part of the two EMEP grid systems covering Europe is shown in Figure A-2.

Note: EMEP/MSC-W has expanded its modeling domain and now uses for deposition and concentration calculations a 50×50 km² grid with North Pole coordinates $(x_p, y_p)=(43, 121)$, but otherwise the same parameters as given in Eq. A.9.

To convert a point $(xlon, ylat)$, given in degrees of longitude and latitude, into EMEP150 coordinates $(emepi, emepj)$ the following FORTRAN subroutine can be used:

```
c
      subroutine llemep (xlon, ylat, emepi, emepj)
c
c      Returns for a point (xlon, ylat), where xlon is the longitude and ylat
c      is the latitude in degrees, its EMEP150 coordinates (emepi, emepj).
c
c      real                xlon, ylat, emepi, emepj
c
c      data  xp, yp /3., 37./      ! coordinates of the North Pole
c      data  xlon0 /-32./          ! = lambda_0
c      data  em /79.24387880/      ! = M=(R/d)*(1+sin(pi/3)); R=6370km, d=150km
c      data  pi180 /0.017453293/   ! = pi/180
c      data  pi360 /0.008726646/   ! = pi/360
c
c      tp = tan((90.-ylat)*pi360)
c      rlamp = (xlon-xlon0)*pi180
c      emepi = xp+em*tp*sin(rlamp)
c      emepj = yp-em*tp*cos(rlamp)
c
c
c      return
c
c      end
```

The EMEP50 coordinates can then be obtained with the aid of Eq. A.9.

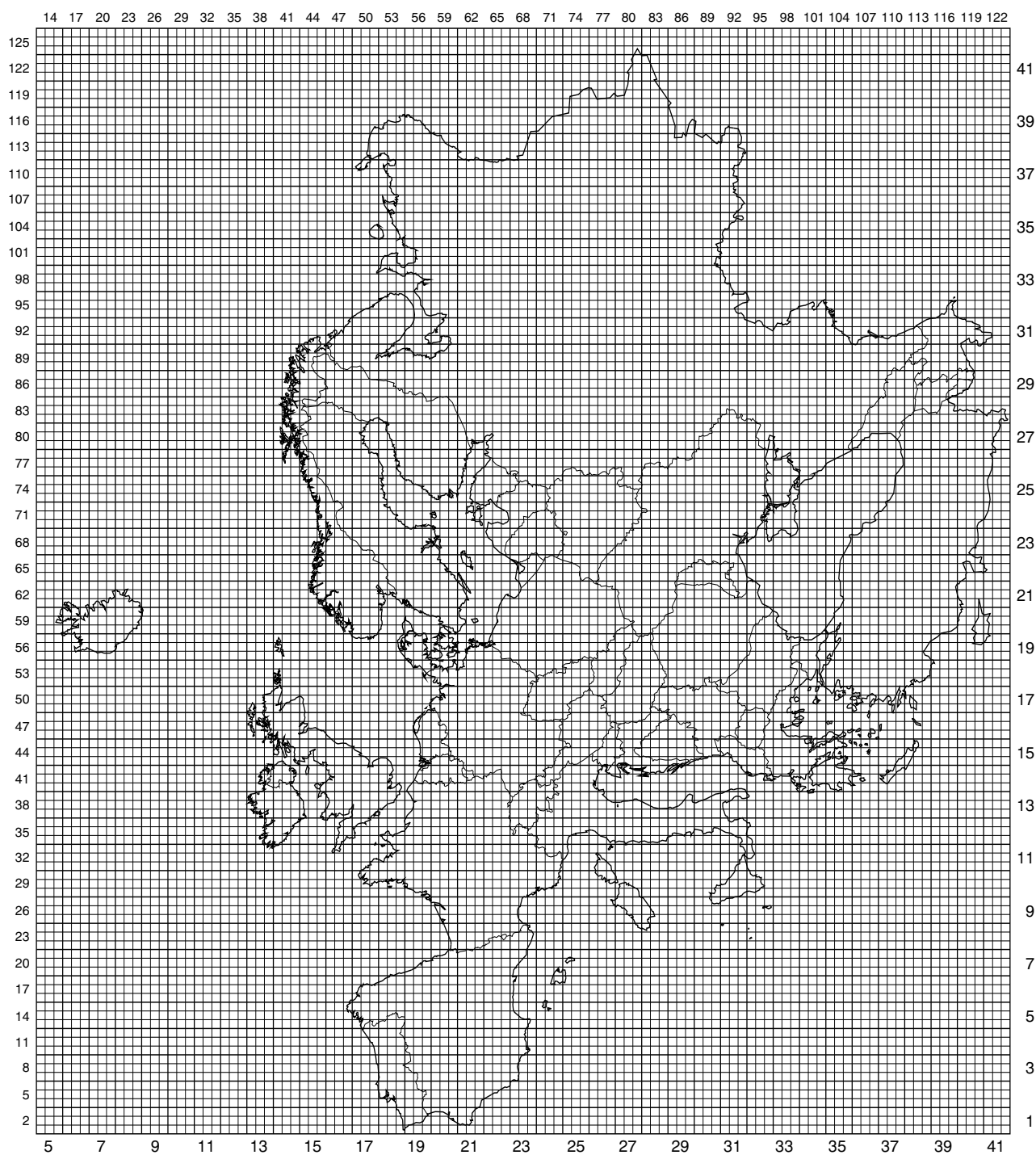


Figure A-2. The EMEP150 grid (thick lines) and the EMEP50 grid (thin lines). The labels at the bottom and right are the EMEP150 grid indices (every second cell), and the labels at the top and left are the EMEP50 grid indices (every third).

Conversely, given the EMEP150 coordinates of a point, its longitude and latitude can be computed with the following subroutine:

```

c
  subroutine emep11 (emepi,emepj,xlon,ylat)
c
c  Returns for a point (emepi,emepj), given in the EMEP150
c  coordinate system, its longitude xlon and latitude ylat in degrees.
c
  data xp, yp /3.,37./      ! coordinates of the North Pole
  data xlon0 /-32./        ! = lambda_0
  data em /79.24387880/    ! = M=(R/d)*(1+sin(pi/3)); R=6370km,d=150km
  data pi180 /57.2957795/  ! = 180/pi
  data pi360 /114.591559/  ! = 360/pi
c
  ex = emepi-xp
  ey = yp-emepj
  if (ex .eq. 0. .and. ey .eq. 0.) then ! North Pole
    xlon = xlon0 ! or whatever
  else
    xlon = xlon0+pi180*atan2(ex,ey)
  endif
  r = sqrt(ex*ex+ey*ey)
  ylat = 90.-pi360*atan(r/em)
                                return
  end

```

To convert the EMEP50 coordinates (p,q) of a point to longitude and latitude, call the above subroutine with $emepi=(p+1)/3$ and $emepj=(q+1)/3$.

The area of an EMEP grid cell:

As mentioned above, the stereographic projection does not preserve areas, e.g. a $150 \times 150 \text{ km}^2$ EMEP grid cell is $22,500 \text{ km}^2$ only in the projection plane, but never on the globe. The area of an EMEP grid cell with lower-left corner (x_1, y_1) and upper-right corner (x_2, y_2) is given by:

$$A(x_1, y_1, x_2, y_2) = 2R^2 \{I(u_2, v_2) - I(u_1, v_2) - I(u_2, v_1) + I(u_1, v_1)\} \quad (\text{A.10})$$

where $u_1 = (x_1 - x_p)/M$, etc.; and $I(u, v)$ is a double integral, which has been evaluated in Appendix A of the 1997 CCE Status Report:

$$I(u, v) = \iint \frac{2dudv}{(1+u^2+v^2)^2} = \frac{v}{\sqrt{1+v^2}} \arctan \frac{u}{\sqrt{1+v^2}} + \frac{u}{\sqrt{1+u^2}} \arctan \frac{v}{\sqrt{1+u^2}} \quad (\text{A.11})$$

These two equations allow the calculations of the area of the EMEP grid cell (i, j) by setting $(x_1, y_1) = (i-1/2, j-1/2)$ and $(x_2, y_2) = (i+1/2, j+1/2)$.

The following FORTRAN functions compute the area of an EMEP grid cell for arbitrary grid indices (i, j) , for the EMEP50 or the EMEP150 grid, depending on the parameter $iopt$:

```

c
  real function aremep (iopt,i,j)
c
c  Returns the area (in km2) of an EMEP grid cell with
c  centerpoint (i,j); iopt=1: EMEP150 grid, iopt=2: EMEP50 grid.
c
  integer          iopt, i, j
  real             dd(2), xp(2), yp(2)
  external         femep
c
  data rearth /6370./      ! radius of the Earth (km)
  data dd(1), dd(2) /150.,50./  ! size of EMEP150/50 grid cell (km)

```

```

data xp(1), yp(1) /3.,37./      ! EMEP150 coordinates of the North Pole
data xp(2), yp(2) /8.,110./    ! EMEP50 coordinates of the North Pole
data drm /1.8660254/           ! = 1+sin(pi/3) = 1+sqrt(3)/2
c
x1 = real(i)-0.5
y1 = real(j)-0.5
emi = dd(iopt)/(rearth*drm) ! = 1/M
u1 = (x1-xp(iopt))*emi
v1 = (y1-yp(iopt))*emi
u2 = u1+emi
v2 = v1+emi
ar0 = 2.*rearth*rearth
aremap = ar0*(femep(u2,v2) - femep(u1,v2) - femep(u2,v1) + femep(u1,v1))
return
end
c
real function femep (u,v)
c
Function used in computing the area of an EMEP grid cell.
c
real u, v
c
ui = 1./sqrt(1.+u*u)
vi = 1./sqrt(1.+v*v)
femep = v*vi*atan(u*vi)+u*ui*atan(v*ui)
return
end

```

The area distortion ratio α , i.e. the ratio between the area of a small rectangle in the EMEP grid and its corresponding area on the globe, is obtained by the following limit operation:

$$\alpha = \lim_{h,k \rightarrow 0} \frac{A(x,y,x+h,y+k)}{hk d^2} = \frac{R^2}{d^2 M^2} \frac{4}{(1+(r/M)^2)^2} \quad (\text{A.12})$$

where R, M, d and r are defined in Eqs. A.1–A.5. Using Eqs A.3 and A.5 and the identities $1/(1+\tan^2 z) = \cos^2 z$ and $2\cos^2(\pi/4 - z/2) = 1 + \sin z$, one arrives at the following expression for the area distortion ratio:

$$\alpha = \left(\frac{1 + \sin \phi}{1 + \sin \phi_0} \right)^2 \quad (\text{A.13})$$

This shows that the distortion ratio depends on the latitude ϕ only, and (small) areas are undistorted, i.e. $\alpha = 1$, only at $\phi = \phi_0 = 60^\circ$ (the approximate latitude of Oslo). In Figure A-3 isolines of the area distortion ratio of the EMEP grid are displayed.

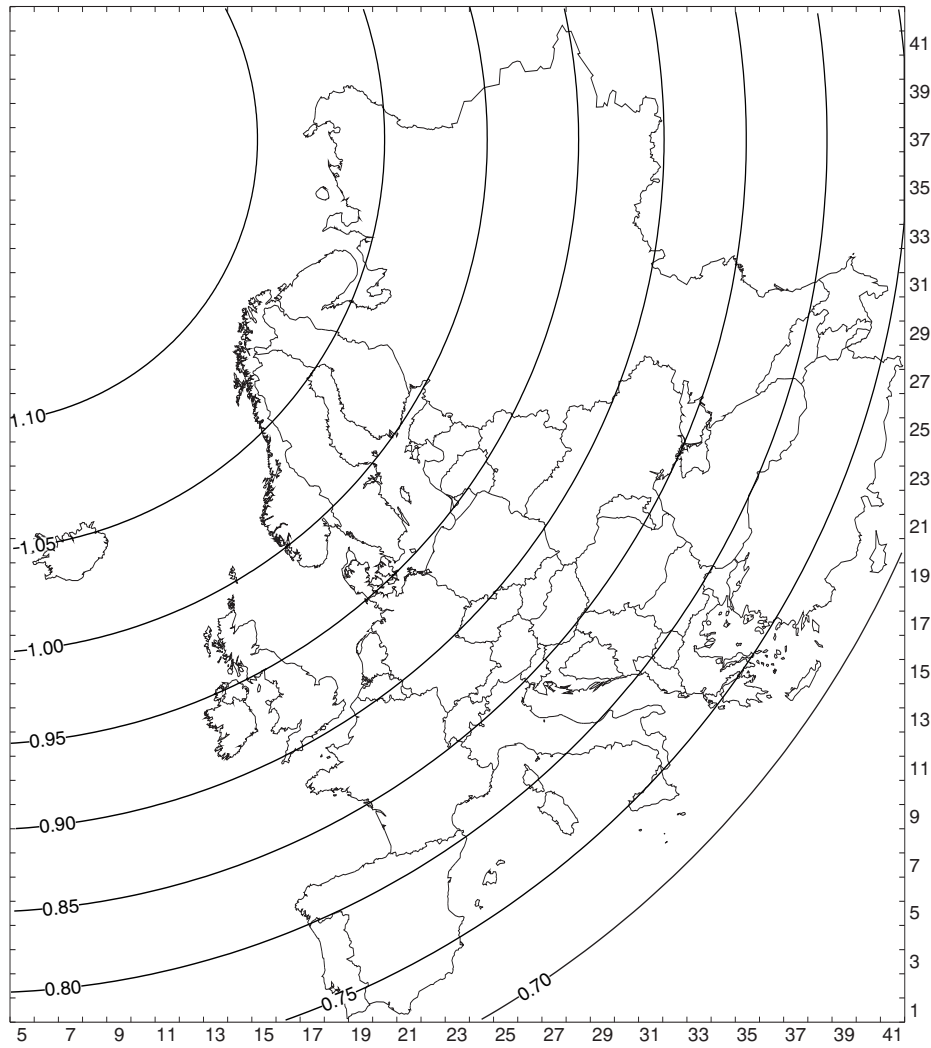


Figure A-3. Isolines of the EMEP50 area distortion ratio of the EMEP grid. Multiply by 22,500 (2,500) to obtain the (approximate) area in km² of an EMEP150 (EMEP50) grid cell on the Earth.

Appendix B. Some FORTRAN routines

In Appendix B of the 1995 and the 1997 CCE Status Reports, FORTRAN subroutines were provided for the computation of certain statistics of distribution functions (e.g. percentiles) and protection isolines. We shall not repeat those subroutines again, but add two new ones useful when dealing with the exceedances defined in Chapter 3 of Part I.

These subroutines are provided on an as-is basis, and no guarantee is given for their correctness. The subroutines contain non-standard features, but they work under Microsoft FORTRAN 5.1. It should not be a problem for the experienced user to convert these subroutines into another programming language.

Computing the exceedance function:

The computation of the exceedance function defined in Chapter 3 of Part I (see in particular Figure 3-2 and Eq. 3.2) requires one first to compute the coordinates of the point Z2 on the critical load function. Let (x_1, y_1) and (x_2, y_2) be two arbitrary points of a straight line g and (x_e, y_e) another point (e.g. E2), then the coordinates (x_0, y_0) of the point obtained by intersecting the line passing through (x_e, y_e) and perpendicular to g are given by:

$$x_0 = (d_1 s + d_2 v) / d^2 \quad \text{and} \quad y_0 = (d_2 s + d_1 v) / d^2 \quad (\text{B.1})$$

with

$$d_1 = x_2 - x_1, \quad d_2 = y_2 - y_1, \quad d^2 = d_1^2 + d_2^2, \quad s = x_e d_1 + y_e d_2, \quad v = x_1 d_2 - y_1 d_1 = x_1 y_2 - y_1 x_2 \quad (\text{B.2})$$

Applying these equations to $(x_1, y_1) = (CL_{min}(N), CL_{max}(S))$, $(x_2, y_2) = (CL_{max}(N), 0)$ and $(x_e, y_e) = (N_{dep}, S_{dep})$ one obtains the point $(x_0, y_0) = (N_0, S_0)$ (e.g. Z2 in Figure 3-2). The final difficulty in computing the $Ex(N_{dep}, S_{dep})$ is to determine into which of the regions (Region 0 through Region 4 in Figure 3-2) a given pair of deposition (N_{dep}, S_{dep}) falls. We do not go into the details of the geometrical considerations, but list a FORTRAN subroutine which does all the calculations and returns ΔN and ΔS as well as the number of the region:

```
c
c      subroutine exceed (CLmaxS, CLminN, CLmaxN, depN, depS, ExN, ExS, ireg)
c
c      Returns the exceedances ExN and ExS (Ex=ExN+ExS) for N and S
c      depositions depN and depS and the critical load function given by
c      CLmaxS, CLminN & CLmaxN.
c      The "region" in which (depN, depS) lies, is returned in ireg.
c
c      integer          ireg
c      real             CLmaxS, CLminN, CLmaxN, depN, depS, ExN, ExS
c
c      ExN = -1.
c      ExS = -1.
c      if (CLmaxS.lt.0. .or. CLminN.lt.0. .or. CLmaxN.lt.0.) return ! error
c      d1 = CLmaxN-CLminN
c      dnn = depN-CLminN
c      dxn = depN-CLmaxN
c      dxs = depS-CLmaxS
c      if (depS .le. CLmaxS .and. depN .le. CLmaxN .and.
c      & CLmaxS*dxn .le. -d1*depS) then ! non-exceedance
c          ireg = 0
c          ExN = 0.
c          ExS = 0.
c      elseif (depN .le. CLminN) then
c          ireg = 4
c          ExN = 0.
c          ExS = dxs
```

```

elseif (dxn*d1 .ge. depS*CLmaxS) then
  ireg = 1
  ExN = dxn
  ExS = depS
elseif (CLmaxS*dxs .ge. d1*dnn) then
  ireg = 3
  ExN = dnn
  ExS = dxs
else
  ireg = 2
  d2 = -CLmaxS
  dd = d1*d1+d2*d2
  s = depN*d1+depS*d2
  v = -CLmaxS*CLmaxN
  x0 = (d1*s+d2*v)/dd
  y0 = (d2*s-d1*v)/dd
  ExN = depN-x0
  ExS = depS-y0
endif

return

end

```

Interpolating between AAE isolines:

The computation of an AAE isoline is done in the same way as protection isolines (see Appendix B of the 1997 CCE Status Report). For given AAE isolines in a grid cell and a given pair of depositions (*depn,deps*) the following program fragment computes the corresponding AAE value by interpolation:

```

.....
do m = 1,maae
  read (1,*) npnt, (xv(n),yv(n),n=1,npnt)
  xv(npnt+1) = 0. ! close polygon by
  yv(npnt+1) = 0. ! adding origin (0,0)
  call inside (depn,deps,xv,yv,1,npnt+1,angle)
  if (abs(angle) .gt. 5.) then ! inside
    if (m .eq. 1) then ! no exceedance
      AAE = 0.
    else ! interpolate
      z = sqrt(deps*deps+depn*depn)
      ang = atan2(deps,depn)
      call isectang (xv,yv,npnt,ang,dist)
      call isectang (xold,yold,npnto,ang,disto)
      AAE = vaae(m) - (vaae(m) - vaae(m-1)) * (dist-z) / (dist-disto)
    endif
    goto 99 ! done for that grid

  else ! store isoline
    npnto = npnt
    do n = 1,npnt
      xold(n) = xv(n)
      yold(n) = yv(n)
    enddo
  endif
enddo ! go and read next isoline
AAE = 99999. ! outside all isolines
99 continue
.....

```

The do-loop runs over *maae* AAE isolines read from a file. The corresponding AAE values are stored in the vector *vaae(m)*, $m=1, \dots, maae$. As soon as two consecutive pre-computed AAE isolines are found so that the given deposition point lies inside one and outside the other (determined with *inside*), the AAE value is estimated by linearly interpolating between the two *vaae*-values using the distances (computed with *isectang*) to the two AAE isolines. The program fragment has to be embedded into loops which run over the desired grid cells and do the necessary writing to an output file. The subroutines *inside* and *isectang* are provided in Appendix B of the 1997 CCE Status Report.

Appendix C. Conversion factors

In this Appendix tables of the most commonly used conversion factors for sulfur and nitrogen deposition as well as for different concentrations are presented.

For convenience we use the term “equivalents” (eq) instead of “moles of charge” (mol_c). If X is an ion with molecular weight *M* and charge *z*, then one has:

$$1 \text{ g X} = \frac{1}{M} \text{ mol X} = \frac{z}{M} \text{ eq X} \quad (\text{C.1})$$

Obviously, moles and equivalents are the same for *z*=1. Conversion factors for sulfur and nitrogen deposition are given in the following tables:

Table C-1. Conversion factors for **sulfur** deposition (g stands for grams of S; *M*=32, *z*=2). For conversion multiply by the factors given in the table.

From:	To:	mg/m ²	g/m ²	kg/ha	mol/m ²	eq/m ²	eq/ha
mg/m ²		1	0.001	0.01	0.00003125	0.0000625	0.625
g/m ²		1000	1	10	0.03125	0.0625	625
kg/ha		100	0.1	1	0.003125	0.00625	62.5
mol/m ²		32000	32	320	1	2	20000
eq/m ²		16000	16	160	0.5	1	10000
eq/ha		1.6	0.0016	0.016	0.00005	0.0001	1

Table C-2. Conversion factors for **nitrogen** deposition (g stands for grams of N; *M*=14, *z*=1). For conversion multiply by the factors given in the table.

From:	To:	mg/m ²	g/m ²	kg/ha	mol/m ²	eq/m ²	eq/ha
mg/m ²		1	0.001	0.01	0.0000714..	0.0000714..	0.71428..
g/m ²		1000	1	10	0.0714..	0.0714..	714.28..
kg/ha		100	0.1	1	0.00714..	0.00714..	71.428..
mol/m ²		14000	14	140	1	1	10000
eq/m ²		14000	14	140	1	1	10000
eq/ha		1.4	0.0014	0.014	0.0001	0.0001	1

Next, we provide conversion factors for concentrations, more specifically between µg/m³ and ppb (parts per billion). One ppb is one particle of a pollutant in one billion (=10⁹) particles of the air-pollutant mixture. How many (and which mass) of them can be found in one m³ depends on the density of the air, i.e. on its temperature and pressure; the conversion formula is

$$1 \text{ ppb} = \frac{V_0}{M} \mu\text{g/m}^3 \quad (\text{C.2})$$

where *M* is the molecular weight and *V*₀=0.022414 m³/mol is the molar volume, i.e. the volume occupied by one mole, at the standard temperature of *T*₀=273.15K (≈0°C) and the standard pressure of *p*₀=101.325 kPa (=1 atm). Assuming ideal gas conditions, the conversion for other temperatures and/or pressures can be accomplished by replacing *V*₀ in Eq. D.2 by

$$V_1 = V_0 \frac{T_1}{T_0} \cdot \frac{p_0}{p_1} \quad (\text{C.3})$$

For example, for *T*₁=298K (≈25°C) and *p*₁=*p*₀ the molar volume *V*₁ is 0.024453 m³/mol.

Table C-3. Conversion factors for concentrations of common pollutants at two different temperatures (1 ppm=1000 ppb).

	From ppb to $\mu\text{g}/\text{m}^3$, multiply by:			From $\mu\text{g}/\text{m}^3$ to ppb, multiply by:	
	<i>M</i>	<i>T=0°C</i>	<i>T=25°C</i>	<i>T=0°C</i>	<i>T=25°C</i>
SO ₂	64	2.855..	2.617..	0.350..	0.382..
NO ₂	46	2.052..	1.881..	0.487..	0.532..
NH ₃	17	0.758..	0.695..	1.318..	1.438..
O ₃	48	2.141..	1.963..	0.467..	0.509..

Converting chemical equilibrium constants:

When dealing with equations of chemical equilibria, the unpleasant task of converting the equilibrium constants to the preferred or required units often arises. Here we give a formula which should cover most of the cases encountered. Let A and B be two chemical compounds which fulfil the following equilibrium equation:

$$[A^{m\pm}]^x = K[B^{n\pm}]^y \quad (\text{C.4})$$

where the square brackets [...] denote concentrations in mol/L (where L stands for liter), implying for the equilibrium constant *K* the units (mol/L)^{*x-y*}. If the concentrations are to be expressed in eq/V, where V is an arbitrary volume unit with 1L=10^{*c*}V, then the equilibrium constant in the new units is given by

$$K' = K \cdot 10^{c(y-x)} \frac{m^x}{n^y} (\text{eq/V})^{x-y} \quad (\text{C.5})$$

Note: To convert to mol/V, set *m=n=1* in the above equation; and to convert to g/V set *m=1/M_A* and *n=1/M_B*, where *M_A* and *M_B* are the molecular weights of A and B, respectively.

Example: The gibbsite equilibrium is given by $[\text{Al}^{3+}] = K[\text{H}^+]^3$, i.e. *m=3, x=1, n=1, y=3* and (e.g.) *K=10⁸(mol/L)⁻²*. If one wants to convert to eq/m³, one has *c = -3*, and thus $K' = 10^8 \cdot 10^{-3 \cdot 2} \cdot 3 = 300 (\text{eq}/\text{m}^3)^{-2}$.