Modelling and Mapping of Critical Thresholds in Europe:

Status Report 2001

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In addition, the Coordination Center for Effects thanks the following:

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- The UN/ECE Working Group on Effects, the Task Force of the ICP Mapping and the Task Force on Integrated Assessment Modelling for their collaboration and assistance.

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Cover: Typical temporal development of acidifying deposition (top) and critical (soil) variable (bottom). The gray areas highlight delay times in which (non-)exceedance of the critical load is not matched by (non-) violation of the critical limit. Steady-state concepts such as critical loads cannot provide that information; dynamic models are needed to estimate these delay times (for details see Part I, chapter 3).

Errata

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Page 18:
Figure 2-1 is incomplete. The percentages of coniferous forests are not shown. The corrected Figure 2-1 is shown below.

Pages 19/20:
Table 2-4: For 4 countries (Netherlands, Norway, Poland, Rep. of Moldova) the critical loads information was permutated (4 times). The correct rows are shown below (Note that the totals were correct).

Page 26:
Runoff values for Poland were submitted in m^3/ha/yr (instead of mm/yr). Assuming them to be in mm/yr, led to numbers too large by a factor of 10 and thus a wrong map in Figure 2-6a. The corrected Figure 2-6 is shown below.
(Consequently, the penultimate sentence on page 25 is not valid any more).

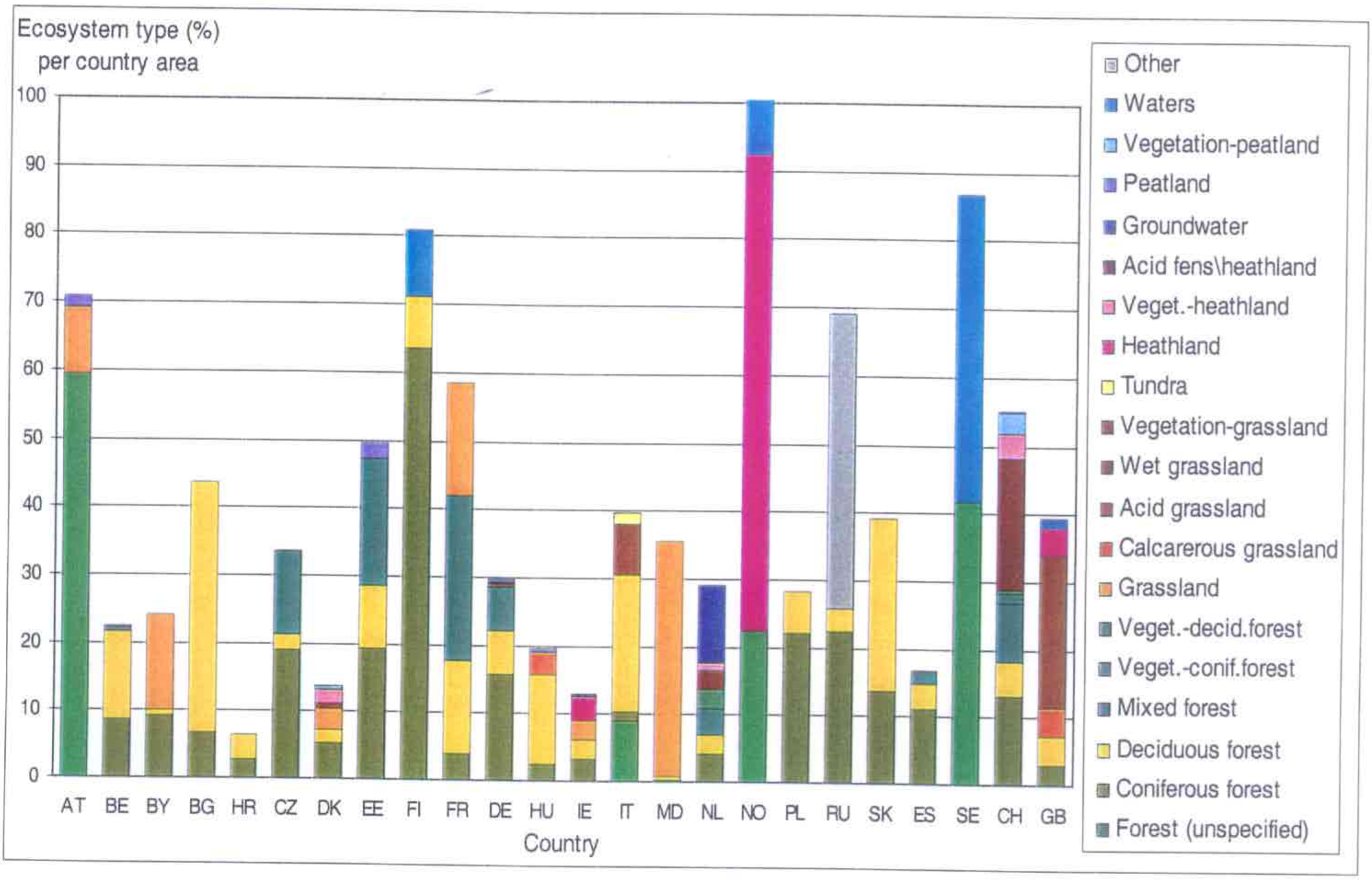
Page 175:
"Collaborating institutions", 2nd column: Replace "Centre for Environment and Health" by "Centre for Ecology and Hydrology".

Corrections to Table 2-4:

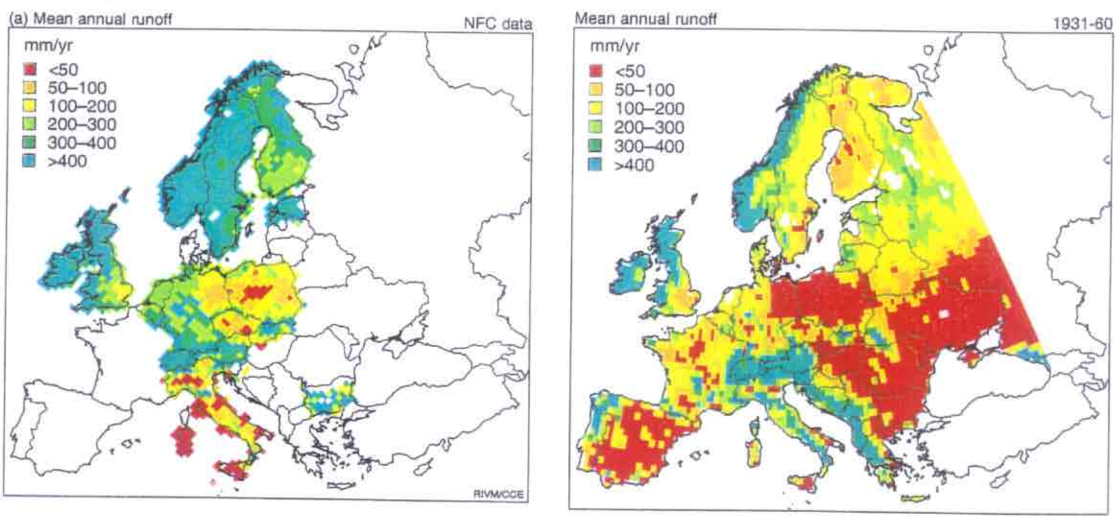
Country	Area[1] (km²)	A Number of ecosystems	B Area (km²)	C Ecosystem cover (%)	D Average ecosystem area (km²)
		Acidity Critical Loads:			
....		...			
Netherlands	41,526	123,143	12,097	29.1	0.1
Norway	323,759	3,025	393,379	121.5	130.0
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
....		...			
		E	F	G	H
		Critical Loads of Nutrient Nitrogen:			
....		...			
Netherlands	41,526	123,434	12,102	29.1	0.1
Norway	323,759	2,330	299,360	92.5	128.5
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
....		...			
		I	J	K	L
		Both acidity *and* nutrient nitrogen CLs:			
....		...			
Netherlands	41,526	123,143	12,097	29.1	0.1
Norway	323,759	720	72,729	22.5	101.0
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
....		...			
		M	N	O	P
		Acidity *and/or* nutrient nitrogen CLs:			
....		...			
Netherlands	41,526	123,434	12,102	29.1	0.1
Norway	323,759	4,635	620,010	191.5	133.8
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
....		...			

1. Source: Der Fischer Weltalmanach 2001, Fischer Verlag, Frankfurt.

Corrected Figure 2-1:



Corrected Figure 2-6:



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Preface

You have before you the sixth Status Report of the Coordination Center for Effects (CCE) of the International Cooperative Programme (ICP) on Mapping under the Working Group on Effects (WGE) of the LRTAP Convention. The earlier reports focused on the mapping of critical loads and their use for the support of negotiations which led to the Gothenburg Protocol in 1999. Since then, the work under the Convention has entered a new phase to prepare for the scientific and technical support of the review and possible revision of protocols, scheduled in about three years from now.

The Convention's Executive Body has expressed the need for more advanced planning of the effects-oriented activities, addressing a number of new objectives. In the near future the mapping of critical loads for the identification of ecosystems at risk of acidification and eutrophication will be extended to include improved knowledge of actual damage and time horizons of recovery. This knowledge is needed to understand the time lags between the phase-in of emission reductions (following the implementation of protocols) and changes in potential effects.

In addition, an increase in activities regarding modelling and mapping methodologies for other pollutants, the temporal and spatial assessment of stock-at-risk and, not least important, uncertainty analysis, are anticipated. These new tasks imply an increase of the modelling capabilities of the ICP Mapping in general, and the CCE in particular. This direction is reflected in the title of this CCE Status Report and its contents.

This report consists of three parts:

Part I describes results of recent activities of the CCE. It includes an analysis of the results of the 2001 "Call for Data" issued by the CCE. The resulting update of the critical loads database is now more tailored to the requirements of producing maps of critical loads for each ecosystem separately, while improving the knowledge of uncertainties (chapter 2). The resulting European maps are described in chapter 1, emphasising the consequences of using maps of critical loads and exceedances of which the resolution has been increased to the 50×50 km^2 grid. Chapter 3 on dynamic modelling sets the stage for this major new task ahead of us. The chapter tries to explain and motivate the use of dynamic models as a logical extension of the work on critical loads, with special emphasis on the linkage to integrated assessment modelling. Finally, chapter 4 summarises the results of a CCE project which assessed the current state of land use/cover mapping by comparing a number of available European maps. The aim is to further the use of land cover information common to all NFCs to enable a more reliable (inter-country) comparison of stock-at-risk, damage and recovery.

Part II contains six contributions covering various subjects of interest to the Mapping Programme. These range from a collaborative project on uncertainties (paper 1), two "contributions-in-kind" by the UK NFC (papers 2 and 3), a contribution by the Netherlands on computing critical load of biodiversity (paper 4) and two contributions on heavy metals, one on critical loads of lead and cadmium on a European scale (paper 5) and the other on mapping atmospheric mercury pollution in Sweden (paper 6).

Part III consists of reports by the National Focal Centres (NFCs). The emphasis has been on the documentation of national critical loads and the input data used to calculate them. These reports were edited for clarity, but have not been reviewed and thus reflect the NFCs' intentions of what to report.

Two appendices describe map projections and conversion formulae for deposition and concentration units.

Finally, if you want to learn more about the CCE, visit the CCE website www.rivm.nl/cce/ from which you can also download earlier Status Reports.

The Editors
Bilthoven, June 2001

1. Current Status of European Critical Load Maps

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

1.1 Introduction

This chapter provides an overview of the critical loads and exceedance maps produced since the 1999 Status Report. The critical loads maps presented in this chapter are the result of the call for critical loads data addressed to National Focal Centres (NFCs) in December 2000, of which 19 responded. In addition to the results being based on improvements of methodologies and databases (see Chapter 2 and Part III), two new requirements have been met.

First, critical load maps now follow the expanded modelling domain of EMEP including 50x50 km^2 (EMEP50) grid cells. Second, each ecosystem is mapped separately. Compliance with the new modelling domain of EMEP ensures compatibility with the computation and mapping of deposition fields. The resolution of deposition and concentration fields have been announced to increase from 150×150 km^2 (EMEP150) to 50×50 km^2 grid cells. The second requirement meets requests by a number of parties to the Convention to have maps of critical loads and exceedances provide information about underlying ecosystems.

In the past, critical load maps were compiled by merging the available critical loads data from National Focal Centres into one single map. At the beginning of the 1990s, critical loads were exceeded both for eutrophication and for acidification in large parts of Europe (see earlier CCE reports), often with large magnitudes. The development of effect-based policies was concerned with scientific and technical support enabling a broad distinction between "ecosystems at risk" and "protected ecosystems" to assess the effectiveness of policy scenarios. In that logic, maps showing the results of policy alternatives in terms of exceedances of *all ecosystems* including the most sensitive ones, were of primary importance. However, due to protocol agreements, exceedances of critical loads are expected to be reduced both spatially as well as in terms of magnitude.

Nowadays, scientific and technical policy support is expected to provide *more detailed information* in addition to the notion of "ecosystem at risk", e.g. addressing *individual ecosystems* and higher map resolutions. This section contains maps of critical loads (a) for individual ecosystems on an EMEP50 scale, and (b) for combined ecosystems on EMEP150 resolution to allow comparison with maps published earlier. Exceedance maps are shown both on an EMEP150 and EMEP50 scale. Possible advantages of using high-resolution critical load and exceedance maps are discussed in this section as well.

1.2 Brief summary of the critical load computation method

The critical loads consists of four basic variables which were asked to be submitted to the CCE by the National Focal Centres, and which in 1999 were used to support the Gothenburg protocol (Hettelingh et al. 2001). These variables are the basis for the maps used in the effect modules of the European integrated assessment modelling effort: (a) the maximum allowable deposition of S, $CL_{max}(S)$, i.e. the highest deposition of S which does not lead to "harmful effects" in the case of zero nitrogen deposition, (b) the minimum critical load of nitrogen, (c) the maximum "harmless" acidifying deposition of N, $CL_{max}(N)$, in the case of zero sulphur deposition, and (d) the critical load of nutrient N, $CL_{nut}(N)$, preventing eutrophication. The equations are summarised as follows (UBA 1996):

$$CL_{max}(S) = BC^*_{dep} - Cl^*_{dep} + BC_w - BC_u - ANC_{le(crit)} \quad (1)$$

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

$$N_{dep} \leq N_i + N_u = CL_{min}(N) \quad (2)$$

all deposited N is consumed by sinks of N (immobilisation 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 a zero deposition of sulphur) is given by:

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) \ / \ (1 - f_{de}) \qquad (3)$$

which not only takes into account the N sinks summarised in Eq. 2, but considers also deposition-dependent denitrification. Both S and N 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 Eqs.1-3. Examples of this function can be found elsewhere (Hettelingh et al. 1995, Posch et al. 1999).

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):

$$CL_{nut}(N) = CL_{min}(N) + N_{le(acc)} \ / \ (1 - f_{de}) \qquad (4)$$

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

1.3 Maps of critical loads for combined ecosystems

This section contains current maps of critical loads on both the EMEP50 and the EMEP150 grid resolution. The latter maps are included to enable comparison with maps published in earlier CCE Status Reports. The maps in the present report are based on updated national contributions from 19 countries. For other countries 1998 data were used, or the European background database of the CCE for countries that have never submitted data.

Figure 1-1 shows 5-percentile maps of $CL_{max}(S)$ and $CL_{nut}(N)$, reflecting values in grid cells at which 95 percent of the ecosystems is protected. In these maps critical loads of different ecosystems have – as was done in the past – been combined into one map on 150×150 km^2 (left hand maps) as well as on a 50×50 km^2 grid cell resolution. A first analysis compares between the two resolutions of these critical load maps. The first conclusion is that a higher resolution (50×50 km^2) enables the identification of areas having low critical loads in comparison to the 150×150 km^2 map. With respect to $CL_{max}(S)$ this is for example illustrated in the southern and middle part of the United Kingdom, the western part of Germany, and in Switzerland (shadings going from orange or yellow to red). Also note the areas in Italy and France for which no data is available, which can only be seen on the higher resolution map. For $CL_{nut}(N)$, this phenomenon is for example clearly visible in the western and eastern parts of the Slovak Republic.

The reason for this phenomenon is that a small ecosystem has a higher probability of becoming dominant in a small grid cell than in larger grid cells, in which many more (or larger areas of) other ecosystems could occur. The opposite may also become true for the same reason, i.e. an area contains less sensitive grid cells when mapped with a higher resolution. This is for example clearly illustrated in the southern part of Switzerland with respect to $CL_{max}(S)$ and in Italy with respect to $CL_{nut}(N)$.

A comparison of the 150×150 km^2 map of recent $CL_{max}(S)$ and $CL_{nut}(N)$ data in comparison to similar maps using 1998 data (see Posch et al. 1999) shows overall minor changes. Lower $CL_{max}(S)$ values can now be found in areas including the northern and south-western part of the UK, and southern Finland. Higher values include areas in Poland, Germany, southern Slovak republic and in the northern part of Hungary. With respect to $CL_{nut}(N)$ lower values include areas in the west of Germany while higher values are found in the southern part of the UK and in Hungary.

Figure 1-2 shows 5-percentile maps of $CL_{max}(N)$ and $CL_{min}(N)$. Relatively low values of the 5-percentile $CL_{max}(N)$ occur mostly in the northern and western regions of Europe. Values of the 5-percentile $CL_{min}(N)$ reflecting the lowest acceptable thresholds of nitrogen uptake and immobilisation, tend to be low everywhere in Europe with the highest values occurring around 700 eq ha^{-1} yr^{-1}.

1.4 Maps of critical loads for individual ecosystems

The use of critical loads of individual ecosystems in integrated assessment modelling enables the optimisation of emission reduction alternatives subject to constraints for each ecosystem individually. This enables even more flexible gap-closure approaches, including different gap-closure targets for each ecosystem.

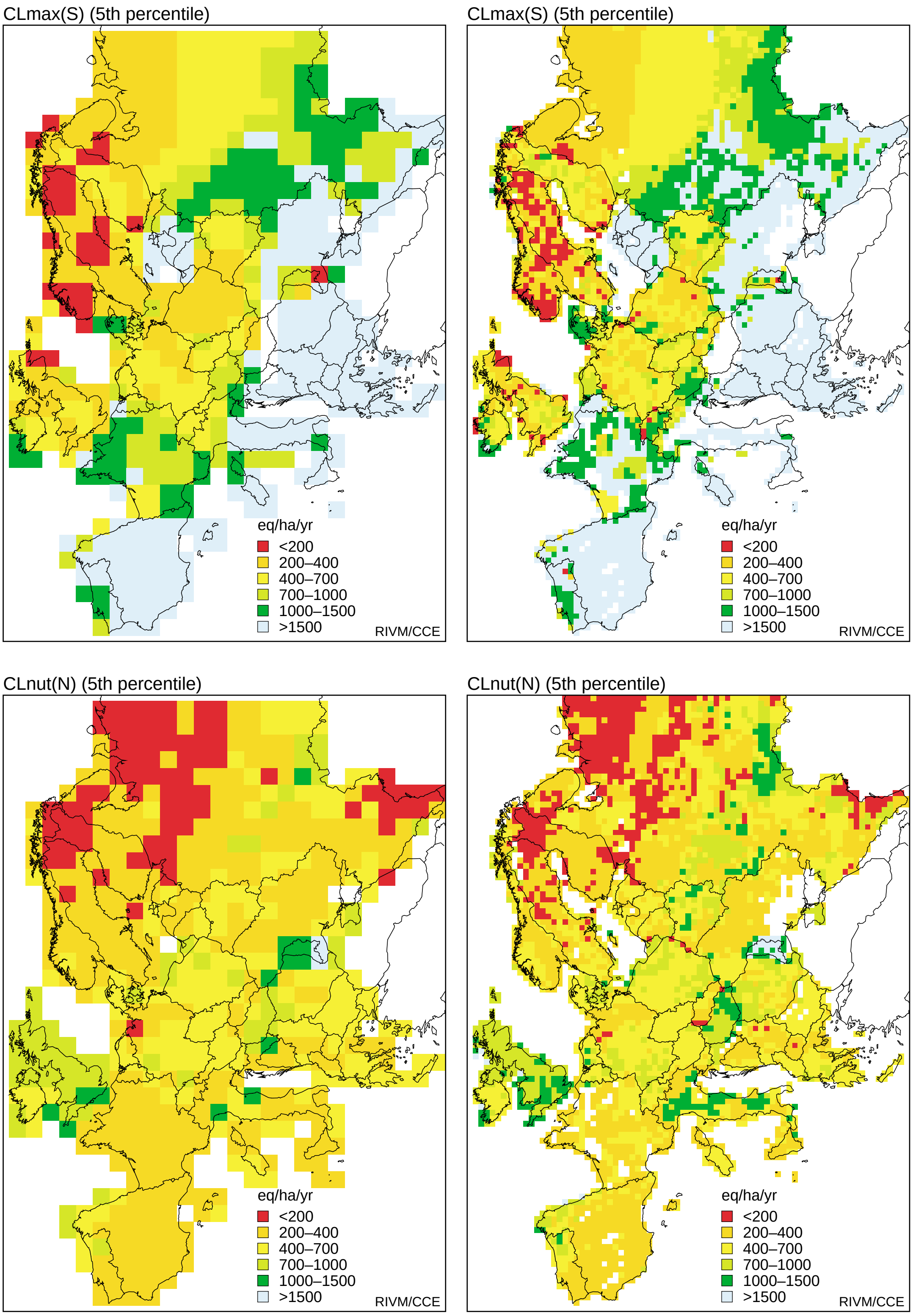

Figure 1-1. The 5th percentiles of the maximum critical loads of sulphur (top), and of the critical loads of nutrient nitrogen (bottom). The maps on the left present these quantities on the EMEP150 grid, while the maps on the right depict the same values on the EMEP50 grid.

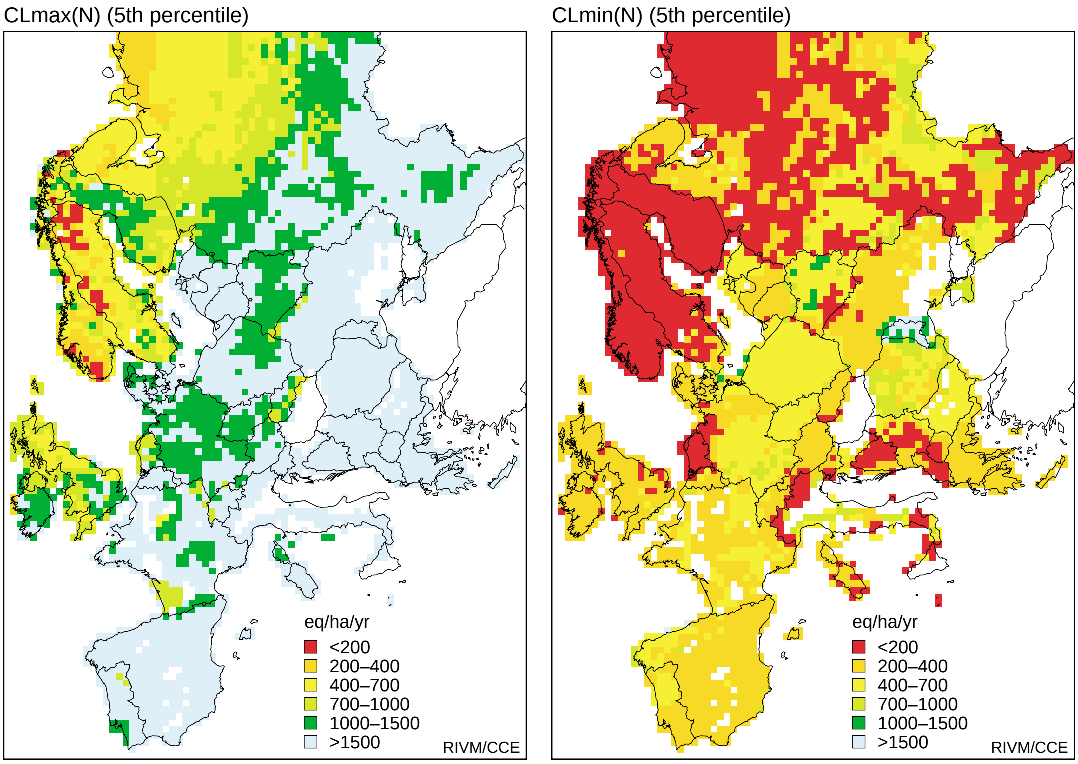

Figure 1-2. The 5th percentiles of the maximum critical loads of nitrogen (left), and of the minimum critical loads of nitrogen (right), on the EMEP50 grid resolution.

The map of critical loads for individual ecosystems is different from the combined map of critical loads shown in Fig. 1-1. First, Europe-wide coverage of critical loads for an ecosystem is only obtained if all NFCs provide data for the same ecosystem. Secondly, the European background data is no longer integrated in the map for countries that have never submitted data. However, European background data could be included in the forest ecosystem maps for the purpose of integrated assessment.

Figure 1-3 shows maps of $CL_{max}(S)$ and $CL_{nut}(N)$ for forests, (semi-)natural vegetation and surface waters on a 50×50 km^2 resolution. Firstly, it is seen that forest ecosystems have been mapped by all the NFCs except for Hungary, which focused on computing $CL_{nut}(N)$.

Critical loads for (semi-)natural vegetation were submitted by 15 NFCs, five of which did not submit $CL_{max}(S)$ values. Finally, for surface waters, six NFCs computed $CL_{max}(S)$ values, while four NFCs provided $CL_{nut}(N)$ data.

The forest and (semi-) natural vegetation $CL_{max}(S)$ (see Fig. 1-3) for Russia do not differ greatly. This can also be seen from the national cumulative distribution functions of $CL_{max}(S)$ and $CL_{nut}(N)$ for each of the mapped ecosystems that are provided in chapter 2 (Figure 2-4).

1.5 Short summary of exceedance computation and mapping methods

When comparing deposition scenarios with critical loads it became apparent that full protection of ecosystems, i.e. non-exceedance, could not be reached everywhere in Europe. Thus integrated assessment modellers proposed to use percentage reductions of the excess depositions, so-called *gap closures*, for the derivation of reduction scenarios.

It was decided to reduce the exceedance everywhere in Europe by a fixed percentage, i.e. to “close the gap” between (present) deposition and the (5-percentile) critical loads. Since the use of a gap closure

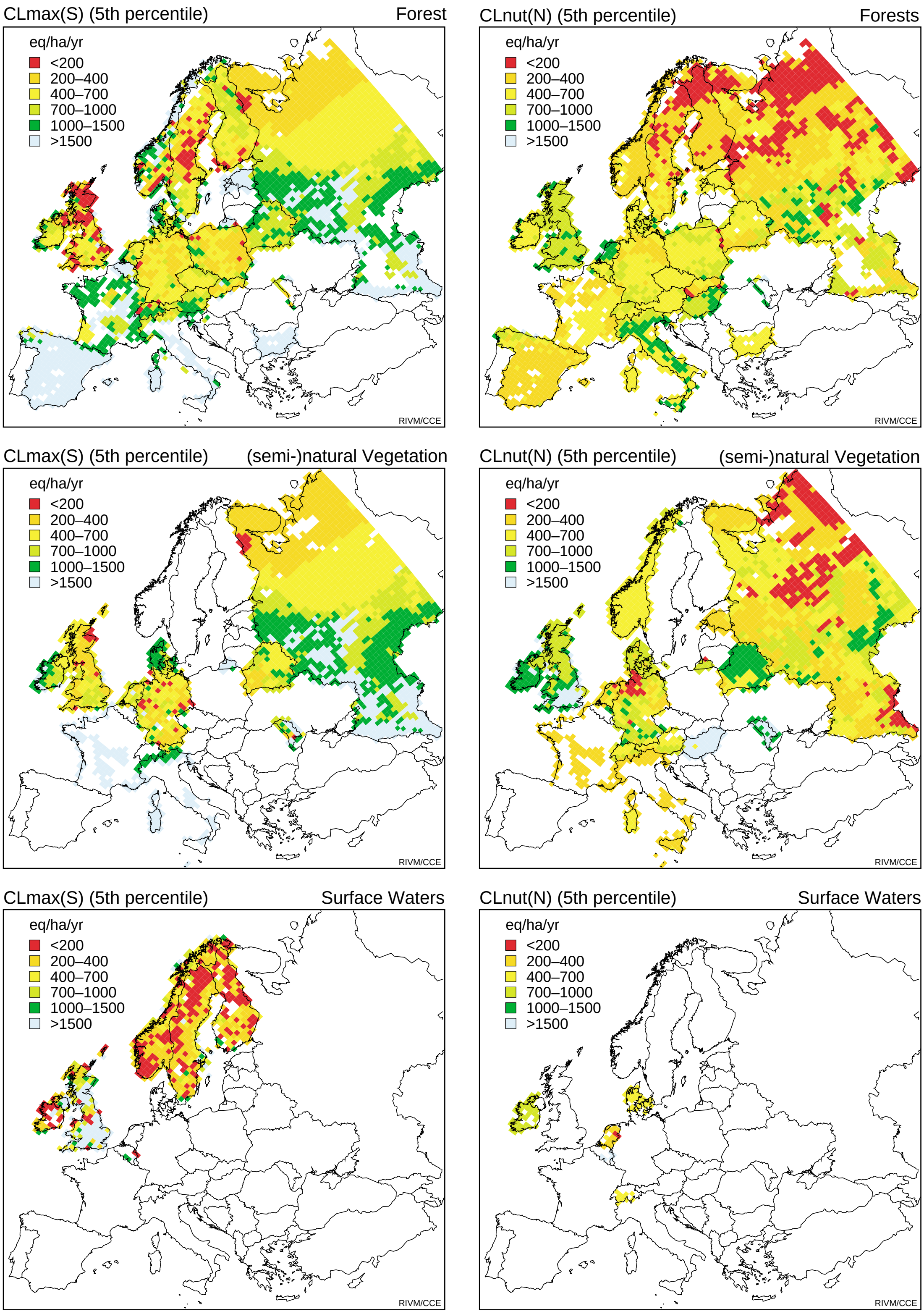

Figure 1-3. The 5th percentiles of the maximum critical loads of sulphur (left), and of the critical loads of nutrient nitrogen (right), on the EMEP50 grid resolution for 3 different ecosystem classes (forests, semi-natural vegetation and surface waters).

implies that not all ecosystems are protected, maps of *ecosystem protection* for a given deposition can be produced, as shown in this section. Ecosystem protection maps provide information about the distribution of the extent of *protected ecosystem areas* in Europe.

However, a disadvantage of using a fixed deposition gap closure is that it can result in very different ecosystem protection percentages depending on the shape of the cumulative distribution function of critical loads. To account for all critical loads within a grid cell (not only the 5th percentile), the use of an *ecosystem area gap closure* was proposed, using all the critical load values of ecosystems in a grid cell. This logic led to the development of the *average accumulated exceedance* (AAE). The AAE is the area-weighted average of all ecosystem exceedances in a grid cell. Maps of AAE provide information about the *magnitude* of the exceedances. (See Posch et al. (1999,2001) for further details.) Both "ecosystem protection" and "average accumulated exceedance" maps are shown in the next section.

1.6 Maps of "Ecosystem protection" and "Average Accumulated Exceedance"

Exceedances have been computed using 150×150 km^2 lagrangian deposition results from EMEP, since the EMEP50 eulerian model results are not yet available for all relevant target years. Figure 1-4 enables a comparison between 1990 and 2010 exceedances of $CL_{nut}(N)$ in two ways. The deposition in 2010 is

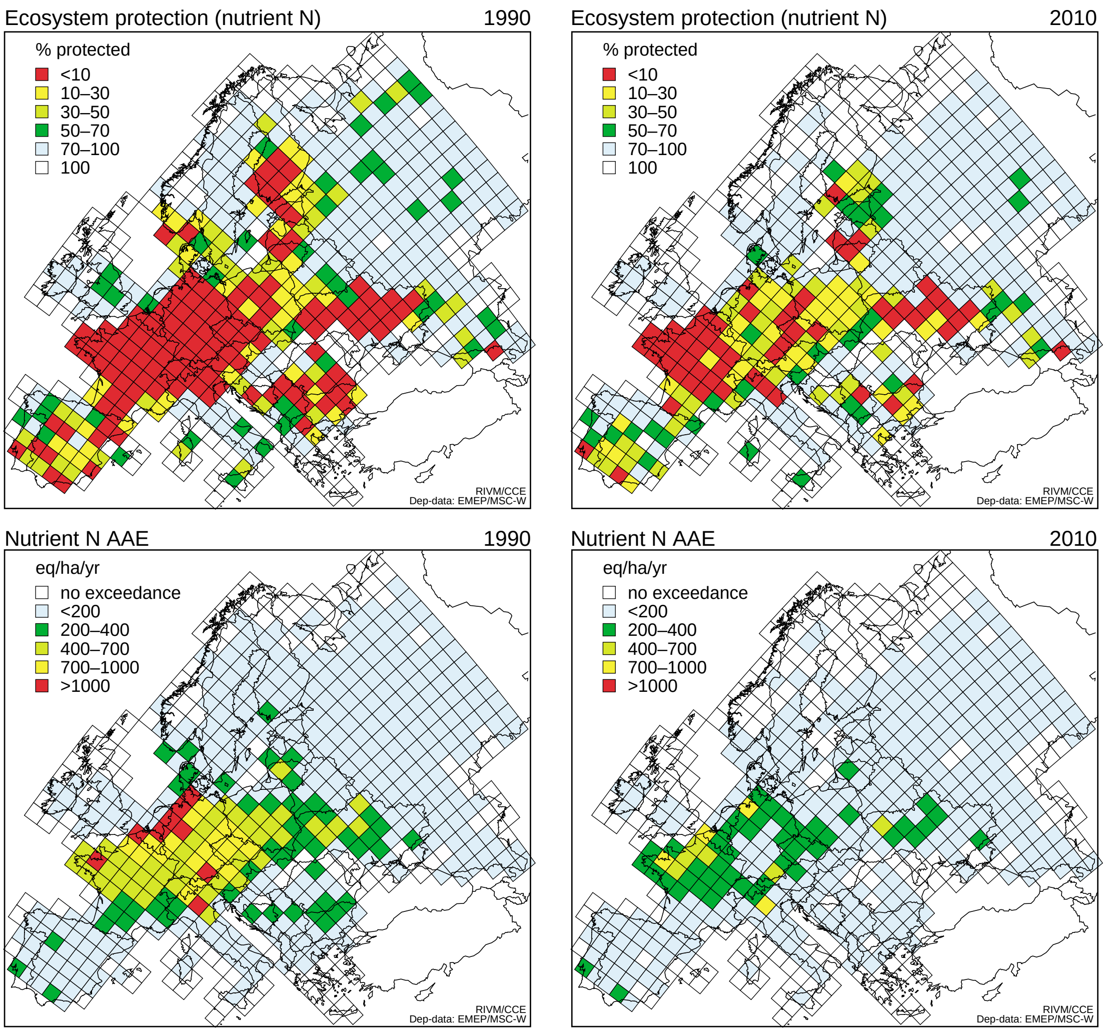

Figure 1-4. *Top:* The percentage of ecosystem area protected (i.e. non-exceedance of nutrient nitrogen critical loads) in 1990 (left) and in 2010, assuming implementation of the Gothenburg Protocol. *Bottom:* The average accumulated exceedance (AAE) of the nutrient nitrogen critical loads in 1990 (left) and 2010 (right). Nitrogen deposition data were provided by EMEP / MSC-W.

computed on the basis of the emissions according to the Gothenburg Protocol. The upper 2 maps show ecosystem protection subject to nitrogen deposition in 1990 and 2010 respectively. Large parts of Europe where less than 10% of the ecosystems were protected in 1990, become more protected in 2010. This can be seen from the size of the red-shaded area in 1990 covering central and western Europe, where in 2010, the protection level increases by up to 70% for a scattering of grid cells.

The lower two maps in Figure 1-4 illustrate the accumulated average exceedance achieved in 1990 and 2010. Peak AAEs in 1990 exceed 1000 eq ha^{-1} yr^{-1} for example in the northwest of Germany, while in 2010 these peaks no longer occur.

Figure 1-5 shows similar phenomena with respect to both sulphur- and nitrogen-based acidity. The exceedance of $CL_{max}(S)$ and $CL_{nut}(N)$ in 1990 results in a large central European region where ecosystem protection does not exceed 10% of the ecosystem area. In 2010, this area is covered mostly by ecosystems of which 70% is protected against the risk of acidification. The 2001 critical loads update did not result in a significantly different ecosystem protection map (1990 depositions) from that published in the 1999 Status Report. However, the magnitude of 1990 AAEs tend to be higher due to the 2001 update.

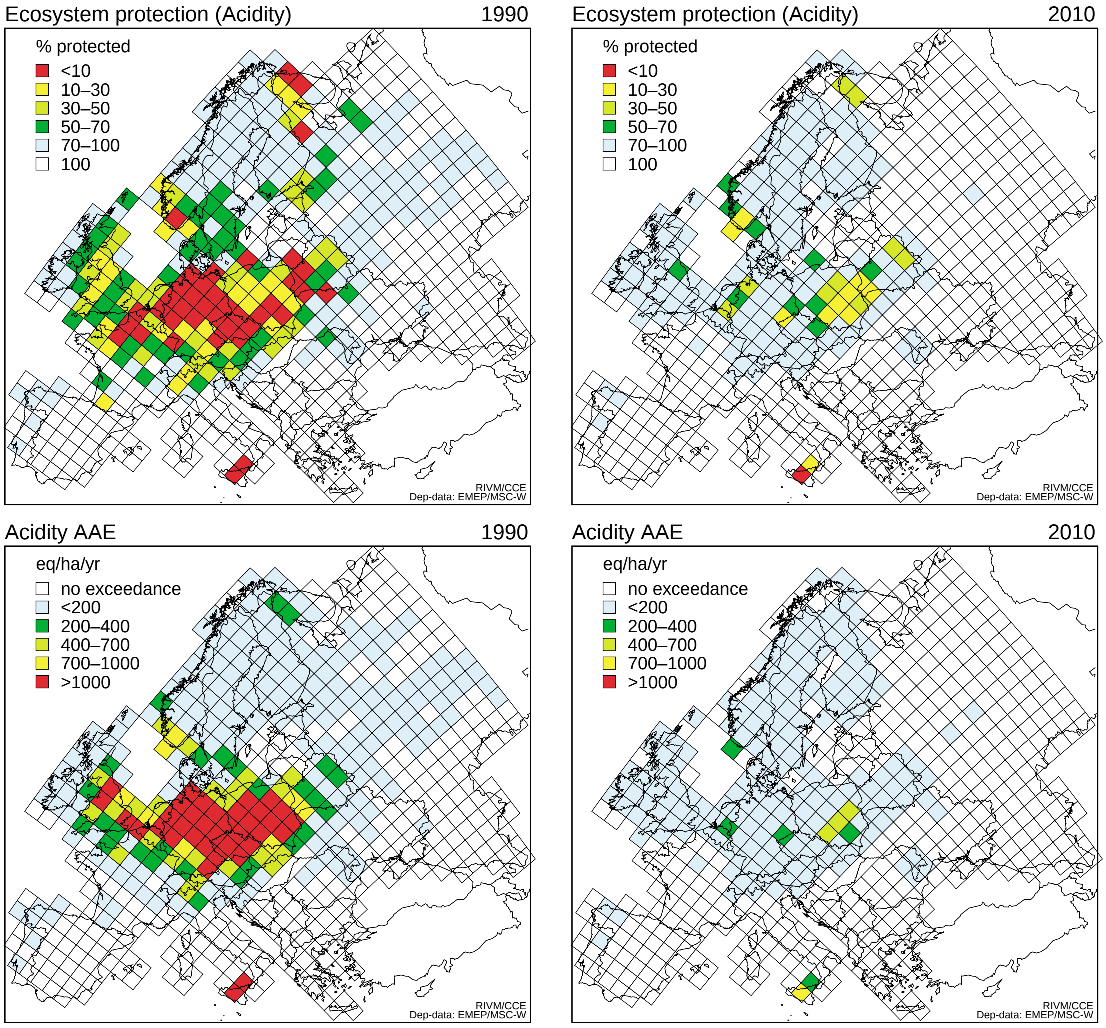

Figure 1-5. *Top:* The percentage of ecosystem area protected (i.e. non-exceedance of acidity critical loads) in 1990 (left) and in 2010, assuming implementation of the Gothenburg Protocol. *Bottom:* The average accumulated exceedance (AAE) of the acidity critical loads in 1990 (left) and 2010 (right). Sulphur and nitrogen deposition data were provided by EMEP / MSC-W.

1.7 *Future high-resolution accumulated average exceedance maps*

The introduction in this Status Report of high-resolution 50×50 km^2 critical load maps raises expectations about the future of exceedance mapping. The use of higher-resolution deposition maps could enable a more precise identification of ecosystems at risk. EMEP is in the process of producing 50×50 km^2 deposition maps based on eulerian modelling to replace the former 150×150 km^2 deposition fields. Preliminary results of the eulerian model are available for 1996 and these have been used to compare eulerian-based average accumulated exceedances to those computed with 150×150km^2 deposition fields for acidity as well as for $CL_{nut}(N)$. The result is shown in Figure 1-6.

Figure 1-6 illustrates that the use of eulerian depositions to compute AAEs for nutrient nitrogen reveals high peaks in the western part of France (top right) as compared to the lagrangian results. Considering the fact that there is no large difference between high- and low-resolution $CL_{nut}(N)$ values (see Figure 1-1) in this area, it is concluded that high-resolution deposition data will improve the identification of areas at risk. The same holds true for the acidity AAE maps (bottom). Also, blank grid cells (those with no data) now appear on the EMEP50 maps which were heretofore not visible on the EMEP150 grid. Therefore, the use of higher-resolution maps will help identify areas where more information on ecosystems is needed.

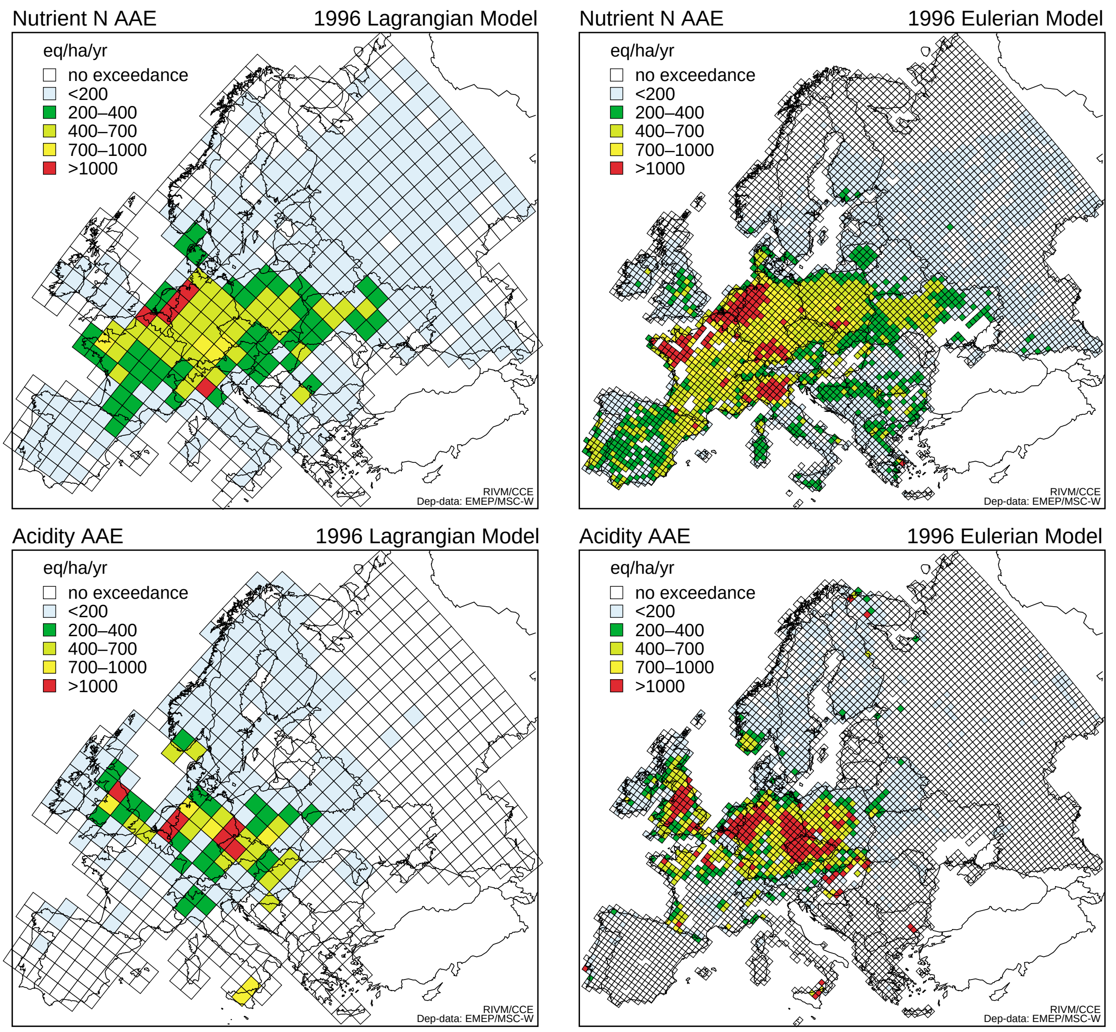

Figure 1-6. The average accumulated exceedance (AAE) of nutrient nitrogen (top) and of acidity (bottom), computed on the EMEP150 grid (left) and the EMEP50 grid (right) for 1996. Deposition data from the lagrangian and eulerian models were provided by EMEP / MSC-W.

In fact, the eulerian-based AAE maps, both for acidity and nutrient nitrogen critical loads, show a larger area covered with exceedances greater than 200 eq ha^{-1} yr^{-1} than identifiable on the EMEP150 resolution. This is also demonstrated in Fig. 1-7 by inspection of cumulative distributions of lagrangian- (solid CDF) and eulerian- (dashed CDF) based AAEs for nutrient nitrogen (top) and acidity (bottom). From the top plot of Fig. 1-7 we can see that about 87% of the lagrangian-based nutrient AAEs have a magnitude of 200 eq ha^{-1} yr^{-1} or less (about 82% in the eulerian case), 92% of 400 eq ha^{-1} yr^{-1} or less (about 87% in the eulerian case), 97% of 600 eq ha^{-1} yr^{-1} or less (90% in the eulerian case), 99% of 800 eq ha^{-1} yr^{-1} or less (about 95% in the eulerian case), and finally about 99% of 1000 eq ha^{-1} yr^{-1} or less (about 97% in the eulerian-based nutrient AAE map). A similar shift to higher exceedance magnitudes holds true when lagrangian-based acidity AAEs are compared to eulerian-based acidity AAEs (see Figure 1-7, bottom CDFs).

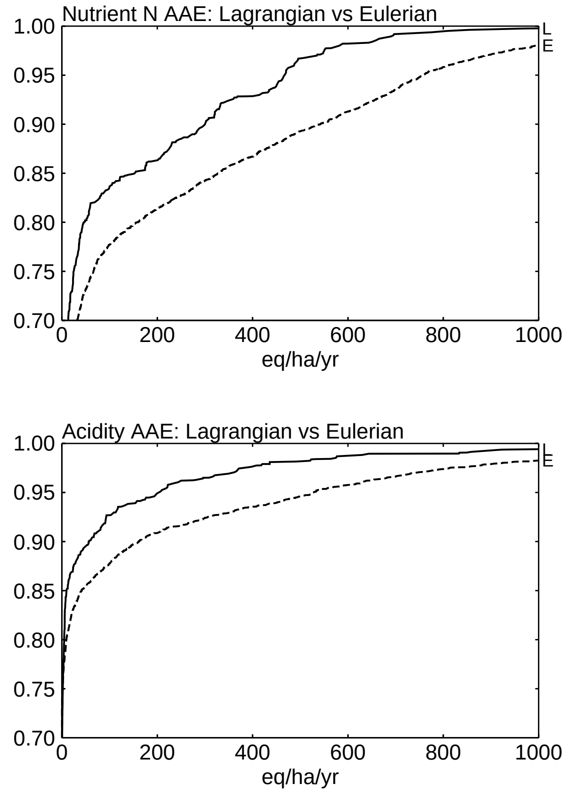

Figure 1-7. Cumulative distribution functions of the average accumulated exceedance (AAE) of nutrient nitrogen (top) and of acidity (bottom). The solid curves reflect the calculations with the EMEP lagrangian deposition model (EMEP150 grid), while the dashed curves show the same quantities using the EMEP eulerian deposition model (EMEP50 grid).

Two main conclusions merge from the above analysis of the European critical load and exceedance maps: (a) nitrogen as a pollutant will need increased attention compared to sulphur, especially considering its multiple effects (acidification, eutrophication and ozone formation), and (b) the use of ecosystem-specific and higher resolution maps (e.g. 50×50km^2) will better allow to pinpoint problem areas.

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2. Summary of National Data

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

Introduction

The European critical loads database prepared in 1998 was the basis for the effect-related work supporting the "Protocol to the 1979 Convention on Long-range Transboundary Air Pollution to Abate Acidification, Eutrophication and Ground-level Ozone". Following the approved 2001 work plan of the ICP Mapping, the Coordination Center for Effects (CCE) in November 2000 issued a call for data, requesting that National Focal Centres (NFCs) submit updates of their critical loads data and accompanying parameters. While the national data sets are described in Part III by the NFCs, this chapter provides an overview and summary of the updates, as well as some inter-country comparison. Comparing the current data set with that of 1998 (see CCE Status Report 1999, Part I, Chap. 2) shows that countries are still making progress in the extension and refinement of their critical loads database.

Overview of national contributions

The following timetable summarises the 2001 update of the national critical load data:

Dec. 1999	CCE contacts all NFCs informing them of the continued contributions of data that are expected from countries, and that they should prepare for a data call to be issued in early 2001. The emphasis of the update will be data consistency checks and verification.
5.4.2000	ICP Mapping meeting; CCE announces that a call for updated critical loads data will be made at the end of 2000.
22.6.2000	CCE sends a reminder letter to NFCs to prepare for an update of critical load data at the end of 2000.
23.8.2000	WGE meeting; CCE announces the call for updating critical loads data by the end of 2000.
30.11.2000	CCE issues a call for data, including analysis of the 1998 data requiring clarifications, instructions for delivering the update, and specifications of the new EMEP grid.
26.2.2001	Deadline for submission of updated critical load data sets.
April / May 2001	Presentation of preliminary analyses of updated data set at the CCE workshop and ICP Mapping Task Force meetings.
July 2001	Feedback to NFCs of the results in European critical load databases.
Aug. 2001	CCE Status Report 2001, summarising the results of the 2001 data update, is presented to Working Group on Effects (WGE).

Reasons for the 2001 update of national critical load data included:

1. To ensure continued contributions of up-to-date national critical load data to the modelling groups participating in the work under the LRTAP Convention.
2. To give NFCs the opportunity to provide the CCE with the improvements made in their national critical load databases.
3. Analyses of previously submitted national critical loads data (see CCE Status Report 1999, Part I) revealed a need for clarification on existing shortcomings of the national submissions used in the work supporting the Gothenburg Protocol.
4. To introduce the new EMEP 50×50km^2 ("EMEP50") coordinates for all NFC databases, thus ensuring compatibility with the modelling domain of the EMEP / MSC-W eulerian atmospheric transport models.
5. To ask NFCs to provide estimates of the uncertainties in their national critical loads data. This information shall be reported to the WGE and the WGSR, and will be used to assess the contribution of critical load uncertainties to the variability of outputs of scenario analysis.

The critical load data and parameters the NFCs were asked to submit to the CCE are shown in Table 2-1. With the database submission documentation had to be included with information on the calculation methods of the critical loads, the derivation of and assumptions on parameter values, deviations from the Mapping Manual (UBA 1996) and an assessment of the uncertainties and reliability of these data.

Table 2-1. Critical loads and parameters requested by the CCE.

Variable	Unit
Longitude	decimal degrees
Latitude	decimal degrees
New EMEP50-i (horizontal) grid index	–
New EMEP50-j (vertical) grid index	–
Ecosystem area	km^2
Maximum critical load of sulphur, $CL_{max}(S)$	eq ha^{-1} yr^{-1}
Minimum critical load of nitrogen, $CL_{min}(N)$	eq ha^{-1} yr^{-1}
Maximum critical load of acidifying nitrogen, $CL_{max}(N)$	eq ha^{-1} yr^{-1}
Critical load of nutrient nitrogen, $CL_{nut}(N)$	eq ha^{-1} yr^{-1}
Base cation deposition, $BC^{*}_{dep}–Cl^{*}_{dep}$	eq ha^{-1} yr^{-1}
Base cation uptake, Bc_u	eq ha^{-1} yr^{-1}
Weathering of base cations, ANC_w or BC_w	eq ha^{-1} yr^{-1}
Runoff, Q [1]	mm yr^{-1}
Gibbsite equilibrium constant, K_{gibb} [1]	m^6 eq^{-2}
Critical leaching of alkalinity, $Al_{le(crit)}+H_{le(crit} = -ANC_{le(crit)}$	eq ha^{-1} yr^{-1}
Nitrogen immobilisation, N_i	eq ha^{-1} yr^{-1}
Nitrogen uptake, N_u	eq ha^{-1} yr^{-1}
Denitrification, f_{de} / N_{de} [1]	– / eq ha^{-1} yr^{-1}
Acceptable nitrogen leaching, $N_{le(acc)}$	eq ha^{-1} yr^{-1}
(Further) ecosystem information	–

1 Parameters requested for the first time to allow more in-depth consistency checks on the critical loads submitted.

Certain parameters (runoff, gibbsite equilibrium constant and denitrification (fraction)) were requested since they are essential for checking the consistency in the calculations of the critical loads according to the Steady-state Mass Balance model.

Twenty-four countries (listed in Table 2-2) contributed critical loads data used in the negotiations of the 1999 Gothenburg Protocol. In the 2001 call for data, there was no country that provided national data for the first time. Nineteen countries contributed revised data: Austria, Belgium, Bulgaria, Croatia, Czech Republic, Denmark, Estonia, Finland, Germany, Hungary, Ireland, Italy, Netherlands, Norway, Poland, Slovakia, Sweden, Switzerland and United Kingdom. Eleven of these countries delivered a revised database before the announced deadline. No updates were received from Belarus, France, Republic of Moldova, Russian Federation and Spain. For these countries their existing databases were adapted by the CCE by changing the EMEP coordinates to the new system and by inserting ‘missing value’ for the newly requested parameters.

Shortcomings and inconsistencies found in the first data submission were reported back to the NFCs with a request to clarify or correct these values. In most cases much iterative communication was necessary to arrive at a final data set and sufficient documentation. This communication process consumed considerably more time than was estimated by the CCE. A shorter response time for the NFCs and a more stringent adherence to the instructions provided would have considerably reduced the time and effort required to incorporate national critical loads data into the European database.

Types, numbers and sizes of ecosystems

National Focal Centres have selected an increased 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 in some countries 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 empirical critical loads for nutrient nitrogen are computed.

Table 2-2. National data contributions to the CCE.

Country	ISO Country Code	2001 update	Remarks
Austria ☺	AT	✓	Parameter / calculation improvements.
Belgium	BE	✓	Parameter / calculation improvements for Wallonia; Gothenburg Protocol data used for Flanders
Belarus	BY		No response from the NFC.
Bulgaria ☺	BG	✓	Parameter / calculation improvements.
Croatia ☺	HR	✓	Parameter / calculation improvements; extension of coverage with 2 more EMEP50 cells: (115,56) and (115,57).
Czech Republic	CZ	✓	Parameter / calculation improvements and identification of types of forests.
Denmark	DK	✓	Parameter / calculation improvements; empirical $CL_{nut}(N)$ values for additional ecosystems.
Estonia	EE	✓	Completely new data set with much higher spatial resolution.
Finland	FI	✓	Parameter / calculation improvements; revised ecosystem area assignment.
France	FR		NFC reported: no update to be expected in time.
Germany ☺	DE	✓	Parameter / calculation improvements.
Hungary ☺	HU	✓	Completely new dataset: empirical $CL_{nut}(N)$ only.
Ireland	IE	✓	Parameter / calculation improvements.
Italy	IT	✓	Parameter / calculation improvements.
Netherlands ☺	NL	✓	Completely new dataset; extension with new ecosystems and introduction of biodiversity criteria.
Norway	NO	✓	Revised ecosystem area assignment.
Poland ☺	PL	✓	Parameter / calculation improvements; higher resolution.
Rep. of Moldova	MD		No response from the NFC.
Russian Federation	RU		NFC reported: no update to be expected.
Slovakia ☺	SK	✓	Parameter / calculation improvements.
Spain	ES		NFC reported: no update to be expected, old data still valid.
Sweden ☺	SE	✓	Parameter / calculation improvements.
Switzerland ☺	CH	✓	Parameter / calculation improvements. Extension of ecosystem identification.
United Kingdom ☺	GB	✓	Parameter / calculation improvements.
Totals:	24	19	

☺ Countries that submitted updated data before the announced deadline (26 February 2001).

Table 2-3 shows by country the distribution of ecosystem types for which critical loads are calculated, including their total (summed) area in km^2 and as a percentage of the country area. Remarks are included when either acidity or nutrient nitrogen critical loads were provided. The diversity of ecosystem types selected by the countries has been reduced in the table into a more limited set of types for presentation reasons. Where possible, details are given of the ecosystem types identified in the original contribution, and their number and summed area.

Figure 2-1 shows the distribution of ecosystem types for which critical loads have been calculated, including their areas as a percentage of the total country area. The diversity of ecosystem types selected by the countries has been aggregated to a more limited set of types for presentation purposes only. Table 2-3 and the histogram in Figure 2-1 show 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 a receptor. Others (e.g. Austria, Denmark, Switzerland and United Kingdom) compute critical loads of grassland covering a large part of the country. Denmark, Germany and United Kingdom distinguish more than one type of grassland. Norway, Ireland, United Kingdom and Switzerland have significant areas of heathland as a receptor. Italy provided critical loads for tundra as a receptor.

Table 2-3. Type and number of ecosystems for which critical loads data were provided by National Focal Centres.

Country	Ecosystem type	CCE code	No. of eco-systems	Area km^2	% of country	Remarks
Austria	Forest	f	6,604	49,710	59.28	
	Alpine grassland	g	1,092	8,236	9.82	Only empirical $CL_{nut}(N)$.
	Oligotrophic bog	p	205	1,536	1.83	Only empirical $CL_{nut}(N)$.
Belgium	Coniferous forest	c	828	2,623	8.59	Flanders: data not updated
	Deciduous forest	d	1,131	3,953	12.95	(652 ecosystem records).
	Mixed forest	m	426	222	0.73	Wallonia: 1,745 ecosystem records.
	Lake	w	12	8	0.03	
Belarus	Coniferous forest	c	234	19,398	9.34	No updated data submitted by NFC.
	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	40,776	36.74	
Croatia	Coniferous forest *	c	21	1,526	2.70	Two points have only $CL_{nut}(N)$.
	Deciduous forest *	d	54	2,063	3.65	
Czech Republic	Coniferous forest	c	16,341	14,966	18.98	
	Deciduous forest	d	2,918	1,893	2.40	
	Mixed forest	m	18,027	9,751	12.36	
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	15,050	1,333	3.09	Only acidity critical loads.
	Raised (ombrotrophic) bogs or fens	vp	1,451	246	0.57	Only empirical $CL_{nut}(N)$; bogs and fens.
	Dry grasslands	vg	4,167	360	0.84	Only empirical $CL_{nut}(N)$.
	Inland dry heathland / coastal heathland	vh	3,025	788	1.83	Only empirical $CL_{nut}(N)$.
	Shallow lakes	w	112	4	0.01	Only empirical $CL_{nut}(N)$.
Estonia	Coniferous forest	c	8,704	8,704	19.25	
	Deciduous forest	d	4,239	4,239	9.37	
	Mixed forests	m	8,507	8,507	18.81	
	Raised bogs	p	961	961	2.12	Only empirical $CL_{nut}(N)$.
Finland	Coniferous forest *	c	2,049	214,860	63.54	Spruce and pine species.
	Deciduous forest	d	1,034	25,543	7.55	
	Lakes	w	1,450	33,231	9.83	Only acidity CLs.
France	Coniferous forest *	c	28	20,856	3.83	No updated data submitted by NFC.
	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	225,869	56,467	15.82	
	Deciduous forest	d	91,084	22,771	6.38	
	Mixed forest	m	90,762	22,691	6.36	
	Natural grassland	vg	7,074	1,769	0.50	
	Acid fens or heathland	af	3,912	978	0.27	
	Wet grassland	wg	1,368	342	0.10	
	Mesotrophic peat bogs	p	3,957	989	0.28	
Hungary	Acidic coniferous forest	c	701	2,389	2.57	Only empirical $CL_{nut}(N)$.
	Deciduous forest *	d	1,458	12,232	13.15	Only empirical $CL_{nut}(N)$.
	Calcareous grassland	cg	951	3,154	3.39	Only empirical $CL_{nut}(N)$.
	Mesotrophic fens or ombrotrophic bogs	p	140	484	0.52	Only empirical $CL_{nut}(N)$.
Ireland	Coniferous forest	c	9,179	2,442	3.48	
	Deciduous forest	d	8,063	1,805	2.57	
	Natural grassland	g	6,887	2,042	2.91	
	Moors and heathland	h	6,803	2,604	3.71	
	Fresh waters	w	175	175	0.25	

* Data for these ecosystem types have been aggregated from more detailed classifications for summary purposes only. See also individual National Focal Centre reports in Part III.

Table 2-3 (continued). Type and number of ecosystems for which critical loads data are provided by National Focal Centres.

Country	Ecosystem type	CCE code	No. of eco-systems	Area km^2	% of country	Remarks
Italy	Boreal / Mediterranean forest	f	151	26,788	8.89	40 with only $CL_{nut}(N)$.
	Temperate con. forest	c	22	4,546	1.51	5 with only $CL_{nut}(N)$.
	Temperate dec. forest	d	165	60,577	20.10	56 with only $CL_{nut}(N)$.
	Tundra	t	46	4,709	1.56	
	Acid grassland	ag	118	23,235	7.71	25 with only $CL_{nut}(N)$.
Netherlands	Coniferous forest	c	8,435	1,779	4.28	Spruce and pine species.
	Deciduous forest	d	9,615	1,125	2.71	
	Vegetation con. forest	vc	28,388	1,690	4.07	Spruce and pine species.
	Vegetation dec. forest	vd	18,129	1,075	2.59	
	Vegetation grass	vg	21,066	1,247	3.00	
	Vegetation heath	vh	6,641	393	0.95	
	Groundwater	gw	30,869	4,788	11.53	
	Moorland ponds	w	291	5	0.01	Only empirical $CL_{nut}(N)$.
Norway	Forests	f	720	72,729	22.46	
	Lakes or streams	w	2,305	320,650	99.04	Only acidity CLs.
	Mountain and heathlands	h	1,610	226,631	70.00	Only empirical $CL_{nut}(N)$.
Poland	Coniferous forest *	c	68,808	68,808	22.01	
	Deciduous forest *	d	19,575	19,575	6.26	
Republic of Moldova	Coniferous forest	c	15	53	0.16	No updated data submitted by NFC.
	Deciduous forest	d	32	260	0.77	
	Grassland	g	94	11,672	34.53	
Russian Federation	Coniferous forest	c	4,916	1,141,037	22.42	No updated data submitted by NFC.
	Deciduous forest	d	2,967	171,549	3.37	
	Other	o	6,333	2,204,554	43.31	
Slovakia	Coniferous forest *	c	112,440	6,746	13.76	15 species aggregated into coniferous and deciduous forest types.
	Deciduous forest *	d	208,451	12,507	25.51	
Spain	Coniferous forest	c	2,237	55,925	11.08	No updated data submitted by NFC.
	Deciduous forest	d	744	18,600	3.68	
	Mixed forest	m	428	10,700	2.12	
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	Coniferous forest	c	340	5,440	13.18	
	Deciduous forest	d	132	2,112	5.12	
	Mixed forest	m	219	3,504	8.49	177 with only $CL_{nut}(N)$.
	Coniferous natural forest, hardly managed	vc	578	578	1.40	Only empirical $CL_{nut}(N)$.
	Deciduous natural forest, hardly managed	vd	262	262	0.63	Only empirical $CL_{nut}(N)$.
	Grass (species rich or alpine grassland)	vg	7,948	7,948	19.25	Only empirical $CL_{nut}(N)$.
	Heath (alpine heath)	vh	1,535	1,535	3.72	Only empirical $CL_{nut}(N)$.
	Peat (raised bogs or mesotrophic fens)	vp	1,321	1,321	3.20	Only empirical $CL_{nut}(N)$.
	Waters (Littorellion)	w	36	36	0.09	Only empirical $CL_{nut}(N)$.
United Kingdom	Coniferous forest	c	29,302	7,376	3.03	
	Deciduous forest	d	69,709	10,325	4.24	
	Acid grassland	ag	138,510	54,578	22.43	
	Calcareous grassland	cg	24,971	10,163	4.18	
	Heathland	h	56,359	9,911	4.07	
	Freshwater catchments	w	1,610	3,717	1.53	Only acidity CLs.

* Data for these ecosystem types have been aggregated from more detailed classifications for summary purposes only. See also individual National Focal Centre reports in Part III.

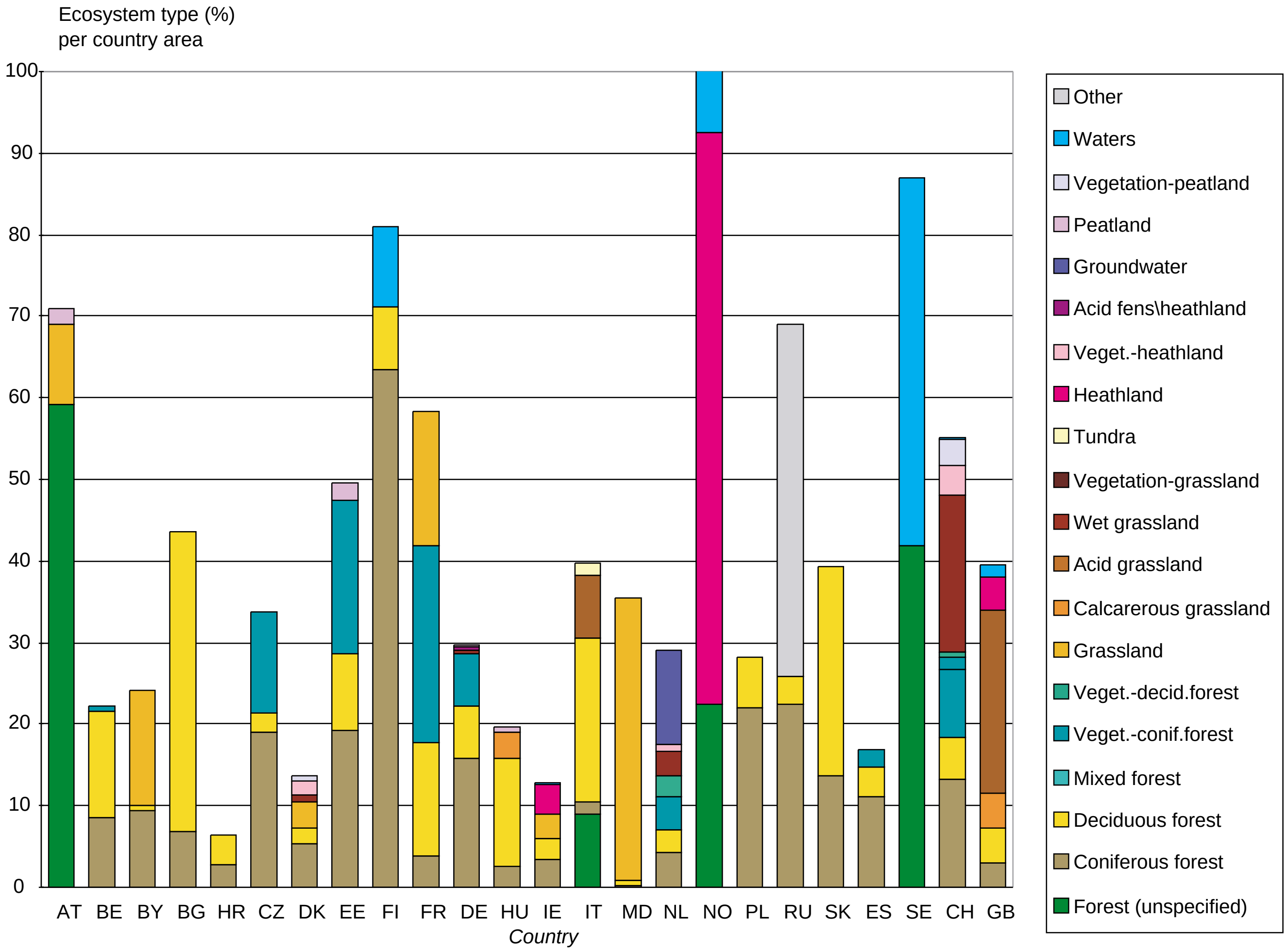

Figure 2-1. Distribution of ecosystem types and their areas as percentage of the total country area in the national critical loads databases.

The increasing number of ecosystem types for which critical loads have been calculated makes it increasingly difficult to decide if equally (or similarly) named ecosystems from different countries mean the same. This confirms the need for a common ecosystem classification to be adopted for future critical load updates. More information on this problem, and a proposal to address these issues, are provided in Part II, paper 2.

Table 2-4 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 summarised 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 (=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 (M=A+E–I). The wide range in both the number and density of ecosystems among countries can be seen from the table. Critical loads of acidity and nutrient nitrogen are computed for the same set of ecosystems for most countries, thus the number and area of ecosystems are the same for both types of critical loads.

Table 2-4. Number of critical loads per national contribution.

		A	B	C	D
		Acidity Critical Loads:			
Country	**Area[1] (km^2)**	**Number of ecosystems**	**Area (km^2)**	**Ecosystem cover (%)**	**Average eco-system area (km^2)**
Austria	83,858	6,604	49,710	59.3	7.5
Belgium	30,528	2,397	6,806	22.3	2.8
Belarus	207,595	555	50,286	24.2	90.6
Bulgaria	110,994	84	48,345	43.6	575.5
Croatia[2]	56,542	71	3,510	6.2	49.4
Czech Republic	78,866	37,286	26,610	33.7	0.7
Denmark	43,094	24,807	4,482	10.4	0.2
Estonia	45,227	21,450	21,450	47.4	1.0
Finland	338,144	4,533	273,634	80.9	60.4
France	543,965	591	317,702	58.4	537.6
Germany	357,022	424,026	106,007	29.7	0.3
Hungary	93,030	0	0	0.0	0
Ireland	70,273	31,107	9,068	12.9	0.3
Italy	301,336	376	105,600	35.0	280.9
Netherlands	41,526	123,143	12,097	29.1	0.1
Norway	323,759	3,025	393,379	121.5	130.0
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
Russian Fed.3	5,090,400	14,216	3,517,142	69.1	247.4
Slovakia	49,034	320,891	19,253	39.3	0.1
Spain	504,750	3,409	85,225	16.9	25.0
Sweden	449,964	4,234	387,871	86.2	91.6
Switzerland	41,285	691	11,056	26.8	16.0
United Kingdom	243,307	320,461	96,070	39.5	0.3
Totals:	9,450,984	1,432,481	5,645,671		

		E	F	G	H
		Critical Loads of Nutrient Nitrogen:			
Country	**Area[1] (km^2)**	**Number of ecosystems**	**Area (km^2)**	**Ecosystem cover (%)**	**Average eco-system area (km^2)**
Austria	83,858	7,901	59,482	70.9	7.5
Belgium	30,528	2,397	6,806	22.3	2.8
Belarus	207,595	555	50,286	24.2	90.6
Bulgaria	110,994	84	48,345	43.6	575.5
Croatia[2]	56,542	75	3,589	6.3	47.9
Czech Republic	78,866	37,286	26,610	33.7	0.7
Denmark	43,094	18,512	4,547	10.6	0.2
Estonia	45,227	22,411	22,411	49.6	1.0
Finland	338,144	3,083	240,403	71.1	78.0
France	543,965	591	317,702	58.4	537.6
Germany	357,022	424,026	106,007	29.7	0.3
Hungary	93,030	3,250	18,259	19.6	5.6
Ireland	70,273	31,107	9,068	12.9	0.3
Italy	301,336	502	119,855	39.8	238.8
Netherlands	41,526	123,434	12,102	29.1	0.1
Norway	323,759	2,330	299,360	92.5	128.5
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
Russian Fed.3	5,090,400	14,216	3,517,142	69.1	247.4
Slovakia	49,034	320,891	19,253	39.3	0.1
Spain	504,750	3,409	85,225	16.9	25.0
Sweden	449,964	1,883	188,056	41.8	99.9
Switzerland	41,285	12,329	22,064	53.4	1.8
United Kingdom	243,307	318,851	92,353	38.0	0.3
Totals:	9,450,984	1,437,647	5,369,293		

1. Source: Der Fischer Weltalmanach '01, Fischer Verlag, Frankfurt. 2. Only four EMEP 50×50 km^2 grid cells were mapped.
3. Country area within EMEP domain.

Table 2-4 (continued). Number of critical loads per national contribution.

Country	Area[1] (km²)	I: *Both* acidity *and* nutrient nitrogen CLs: No. of ecosystems	J: Area (km²)	K: Ecosystem cover (%)	L: Average eco-system area (km²)
Austria	83,858	6,604	49,710	59.3	7.5
Belgium	30,528	2,397	6,806	22.3	2.8
Belarus	207,595	555	50,286	24.2	90.6
Bulgaria	110,994	84	48,345	43.6	575.5
Croatia[2]	56,542	71	3,510	6.2	49.4
Czech Republic	78,866	37,286	26,610	33.7	0.7
Denmark	43,094	9,757	3,149	7.3	0.3
Estonia	45,227	21,450	21,450	47.4	1.0
Finland	338,144	3,083	240,403	71.1	78.0
France	543,965	591	317,702	58.4	537.6
Germany	357,022	424,026	106,007	29.7	0.3
Hungary	93,030	0	0	0.0	0
Ireland	70,273	31,107	9,068	12.9	0.3
Italy	301,336	376	105,600	35.0	280.9
Netherlands	41,526	123,143	12,097	29.1	0.1
Norway	323,759	720	72,729	22.5	101.0
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
Russian Fed.3	5,090,400	14,216	3,517,142	69.1	247.4
Slovakia	49,034	320,891	19,253	39.3	0.1
Spain	504,750	3,409	85,225	16.9	25.0
Sweden	449,964	1,856	184,746	41.1	99.5
Switzerland	41,285	649	10,384	25.2	16.0
United Kingdom	243,307	318,851	92,353	38.0	0.3
Totals:	9,450,984	1,409,646	5,082,943		

Country	Area[1] (km²)	M: Acidity *and/or* nutrient nitrogen CLs: Number of ecosystems	N: Area (km²)	O: Ecosystem cover (%)	P: Average eco-system area (km²)
Austria	83,858	7,901	59,482	70.9	7.5
Belgium	30,528	2,397	6,806	22.3	2.8
Belarus	207,595	555	50,286	24.2	90.6
Bulgaria	110,994	84	48,345	43.6	575.5
Croatia[2]	56,542	75	3,589	6.3	47.9
Czech Republic	78,866	37,286	26,610	33.7	0.7
Denmark	43,094	33,562	5,880	13.6	0.2
Estonia	45,227	22,411	22,411	49.6	1.0
Finland	338,144	4,533	273,634	80.9	60.4
France	543,965	591	317,702	58.4	537.6
Germany	357,022	424,026	106,007	29.7	0.3
Hungary	93,030	3,250	18,259	19.6	5.6
Ireland	70,273	31,107	9,068	12.9	0.3
Italy	301,336	502	119,855	39.8	238.8
Netherlands	41,526	123,434	12,102	29.1	0.1
Norway	323,759	4,635	620,010	191.5	133.8
Poland	312,685	88,383	88,383	28.3	1.0
Rep. of Moldova	33,700	141	11,985	35.5	85.0
Russian Fed.3	5,090,400	14,216	3517,142	69.1	247.4
Slovakia	49,034	320,891	19,253	39.3	0.1
Spain	504,750	3,409	85,225	16.9	25.0
Sweden	449,964	4,261	391,181	86.9	91.8
Switzerland	41,285	12,371	22,736	55.1	1.8
United Kingdom	243,307	320,461	96,070	39.5	0.3
Totals:	9,450,984	1,460,482	5,932,021		

1. Source: Der Fischer Weltalmanach '01, Fischer Verlag, Frankfurt. 2. Only four EMEP 50×50 km² grid cells were mapped.
3. Country area within EMEP domain.

Most data updates (those from AT, BE, BG, HR, CZ, DK, FI, DE, IE, IT, NO, SK, SE, CH, and GB) consist primarily of the inclusion of newly requested parameters, conversion to the new EMEP coordinates, and remedies of previous shortcomings. Three countries (DE, DK, NL) expanded their set of mapped ecosystem types considerably. Germany and Denmark computed critical loads for grasslands, heathlands, fens and peat bogs, and Denmark also submitted new data for shallow lakes. Germany provided acidity and nutrient nitrogen critical loads data for its new ecosystem types, whereas Denmark submitted only empirical nutrient nitrogen critical loads in its new data. The Netherlands introduced a completely revised forest critical load database to remedy minor inconsistencies in the data used in national and in international policy support. The same holds for Germany's database of forest critical loads: the same databases are now used nationally and internationally. In addition, the Netherlands database was expanded with critical loads for ecosystem types computed with new criteria (biodiversity) to set protection levels. Paper 4 in Part II and the Dutch NFC contribution in Part III of this report describe these methods in more detail.

The data updates of CH, CZ, HU and PL include more detailed identification of the ecosystem types compared with their previous versions. Hungary now provides empirical nutrient nitrogen critical loads data only, albeit a much larger number. Bulgaria provided detailed tree species information for every grid cell, but submitted only aggregate critical loads for coniferous and deciduous forest. Croatia extended its mapped area by two more EMEP 50×50 km^2 grid cells: (115, 56) and (115, 57). Estonia and Poland submitted completely revised databases with a grid resolution of 1×1 km^2.

The density of the critical loads data submitted also varies greatly among countries. Fig. 2-2 shows this variation by presenting the total number of ecosystems (black bars) and their total area as a percentage of the country's area (grey bars). The 2-letter country codes are listed in Table 2-2. For example, Germany computes critical loads for about 30% (106,007 km^2) of its country area, but the number of ecosystem points (420,026) is large compared with most other countries.

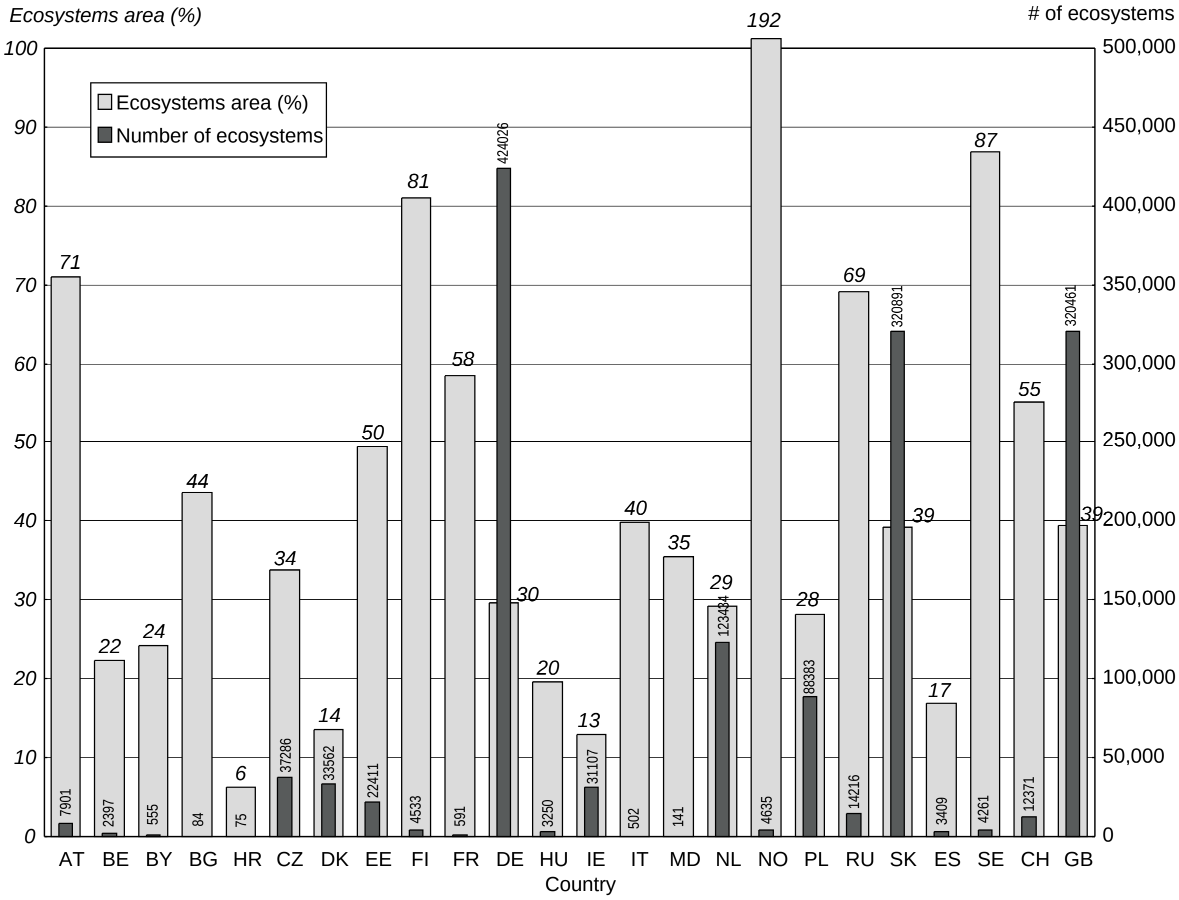

Figure 2-2. Histogram showing the ecosystem area for which critical loads are provided (percentage of total country area; grey shaded bars) and the numbers of ecosystems (black bars) per country.

Norway and Finland both revised their ecosystem area assignment (see NFC reports in Part III for details). Norway assumes that its critical loads for surface waters represent the entire country (all of Norway is covered by catchments). In addition, most of the Norwegian area is covered by the other ecosystems mapped, either forests or (semi-)natural ecosystems. This results in a double-counting of most of the area of Norway.

This problem of multiple critical loads for the same area (e.g. a forest growing in a catchment above an aquifer) needs to be addressed in future updates. Finland used only the lake area to avoid double-counting ecosystem area, and Sweden used the same criterion as before to avoid area double-counting. The Netherlands, confronted with this problem for the first time, extended its mapped area by computing critical loads for areas with (semi-)natural vegetation and groundwater. The groundwater mapped is mainly allocated to those areas where the other ecosystem types are also located. This causes a double-counting of these areas; however, this double-counting applies only to a quite small part of the country and thus does not appear in Figs. 2-1 and 2-2. Poland and Estonia increased their mapping resolution to a 1×1 km^2 grid size, which results in much larger numbers of ecosystems than were presented in the CCE Status Report 1999.

The spatial location of the differences in data density is presented in more detail in Fig. 2-3, which shows the distribution of ecosystem records for critical loads of acidity and / or nutrient nitrogen for each EMEP 50×50 km^2 grid cell that contains critical load data. The maps show only those grid cells for which national critical loads data have been submitted.

Comparison of national critical load data

While the maps in Chapter 1 present only the 5-percentile of the critical loads in the grid cells covering Europe, Figs. 2-4 and 2-5 present the cumulative distribution functions of the four critical load quantities provided by the NFCs. In contrast to earlier presentations, these distributions are shown separately for three groups of ecosystems: forests, surface waters and (semi-)natural vegetation. The last group includes all ecosystems (except groundwater) that do not fall into the other two classes.

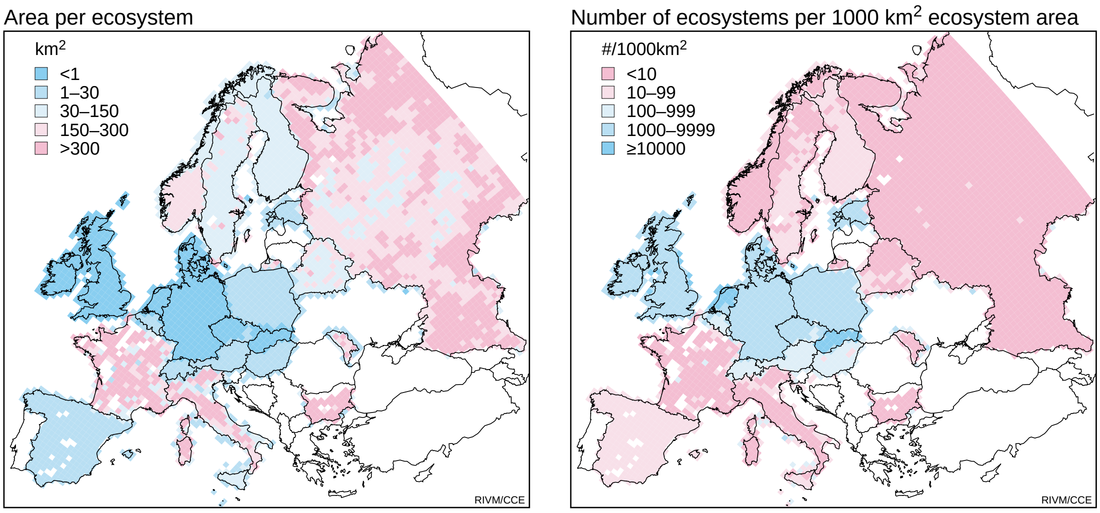

Figure 2-3. The average area (in km^2) for the ecosystem records for which acidity and / or nutrient nitrogen critical loads are submitted (left) and the number of nationally submitted ecosystem records per 1000 km^2 of the total ecosystem area (right) per EMEP 50×50 km^2 grid cell. For the white areas no national contributions are available.

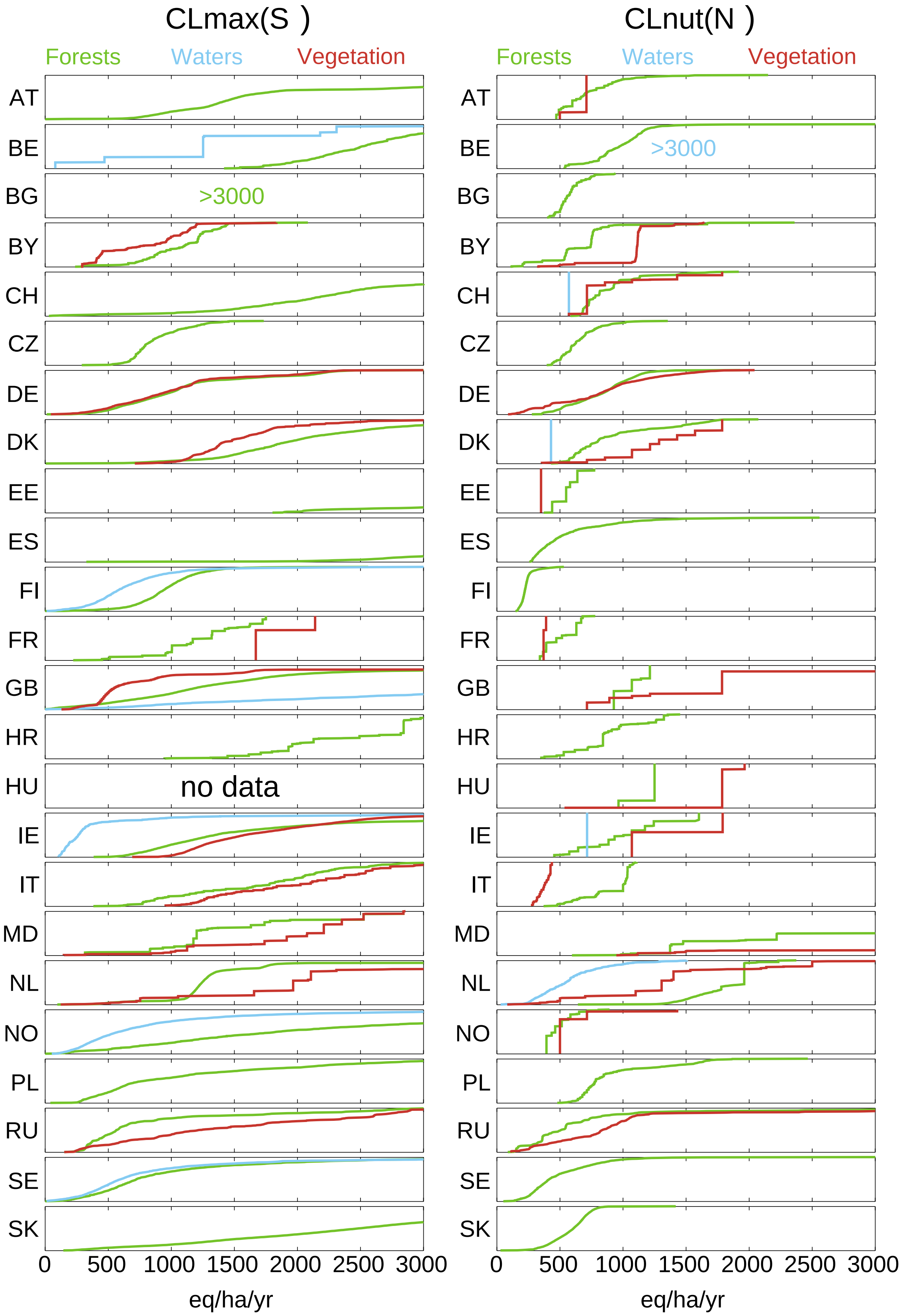

Figure 2-4. Cumulative distribution functions of the $CL_{max}(S)$ (left) and $CL_{nut}(N)$ (right) for three groups of ecosystem types of the 24 countries for which national critical load data are available.

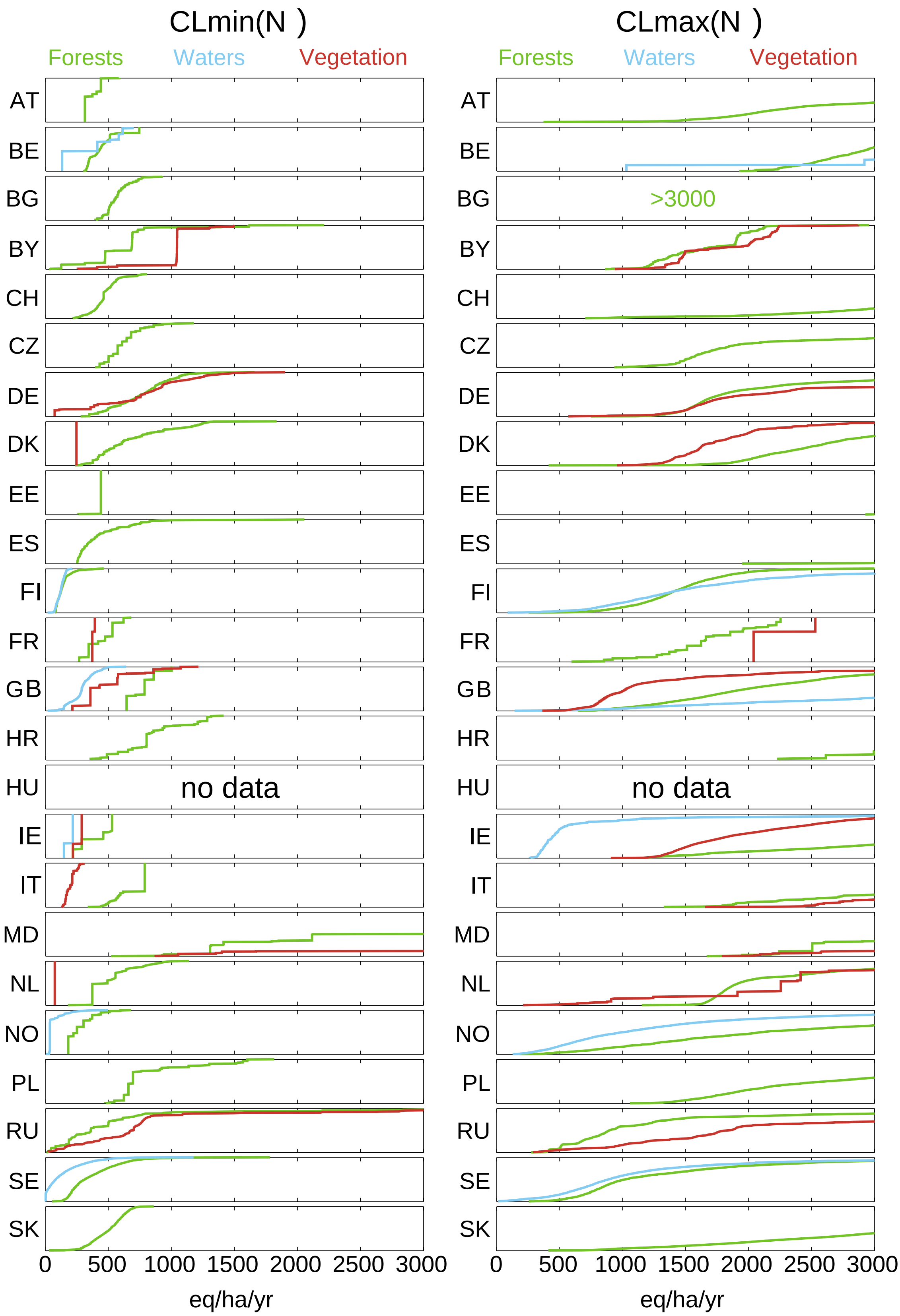

Figure 2-5. Cumulative distribution functions of the $CL_{min}(N)$ (left) and $CL_{max}(N)$ (right) for three groups of ecosystem types of the 24 countries for which national critical load data are available.

The distributions of $CL_{max}(S)$ vary considerably between countries and also between the groups of ecosystems in several countries. In some countries (AT, CH, DE, DK, FI, GB, NO, PL, SE), the lowest critical loads are at or near zero, while for Bulgaria all acidity critical loads are above 3000 eq ha^{-1} yr^{-1}, indicating that there is no acidification problem even at present-day deposition levels. In Austria, Switzerland, Estonia, Spain and Slovakia, a considerable fraction of the ecosystems has critical loads above 3000 eq ha^{-1} yr^{-1}, probably indicating areas with calcareous soils. The distribution of critical loads for forests and (semi-)natural vegetation is almost identical in Germany.

The same holds for the distributions of $CL_{max}(S)$ for forests and surface waters in Sweden, whereas surface waters are more sensitive (i.e. have lower critical loads) than forests in Belgium, Finland, Ireland and Norway, and much less sensitive in the United Kingdom. Acidity critical loads for (semi-) natural vegetation are generally smaller in Belarus, Denmark and United Kingdom, whereas they are less sensitive in France, Ireland, Italy, Moldova, the Netherlands and Russia. Hungary did not provide any critical load data for acidity.

The distributions of the critical load of nutrient nitrogen, $CL_{nut}(N)$, are much less smooth than those for $CL_{max}(S)$, reflecting the fact that many $CL_{nut}(N)$ values rely on empirical data which often consist of a single value for a given ecosystem type. “Extreme” cases are, e.g. Switzerland, Denmark and Ireland, where a single value is assigned to all surface waters. In several countries some $CL_{nut}(N)$ values exceed 3000 eq ha^{-1} yr^{-1}, indicating that a large amount of N can be sustainable leached without causing harmful effects, the extreme case being Belgium, where all nutrient N critical loads for surface waters are greater than 3000 eq ha^{-1} yr^{-1}.

The distributions of $CL_{min}(N)$ (Fig. 2-5) reflect those of $CL_{nut}(N)$ (Fig.2-4). Note that $CL_{min}(N) < CL_{nut}(N)$, if both are calculated with the same model. High values of $CL_{nut}(N)$ indicate that large amounts of N are either taken up by vegetation or are immobilised. The “differences” between $CL_{min}(N)$ and $CL_{nut}(N)$ (if calculated by the same model) indicate the amount of N denitrified (if modelled as fraction of the N deposition) or is allowed to leach without causing “harmful effects”. It should be noted, however, that in some countries $CL_{min}(N)$ values are derived from model calculations, whereas $CL_{nut}(N)$ is taken from tables of empirical data (see Tables 2-2 and 2-3), which makes a direct comparison of these two quantities less meaningful.

The distributions of $CL_{max}(N)$ (Fig. 2-5) largely reflect those of $CL_{max}(S)$ (Fig. 2-4), with $CL_{max}(N) > CL_{max}(S) + CL_{min}(N)$. Thus the relational observations above apply to these quantities as well.

Comparison 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 should be explainable by variations in the basic model inputs. Therefore, NFCs were asked to submit (most) model parameters along with their critical load databases. In this section we compare and analyse the most important of those model parameters.

An important quantity needed to compute the leaching of ANC or N is runoff (or precipitation surplus), which in the case of terrestrial ecosystems is the amount of water percolating through the root zone, most often calculated as the difference between precipitation and the actual amount of evapotranspiration. In Figure 2-6a the mean annual runoff in every EMEP50 grid square is displayed for those countries which submitted information on this quantity. For comparison, Figure 2-6b shows the difference between precipitation and actual evapotranspiration on the 0.5°×0.5° grid averaged over the period 1931–1960. The data are taken from Leemans and Cramer (1991) who based their calculations on monthly meteorological data from 1678 European meteorological stations. Actual evapotranspiration is calculated according to an approach by Prentice et al. (1993). Although one cannot expect perfect agreement between the two maps because of differences in data and calculation methods used, some of the differences are striking. Also the divergence in runoff values at country borders in Fig. 2-6a calls for further investigation and clarification.

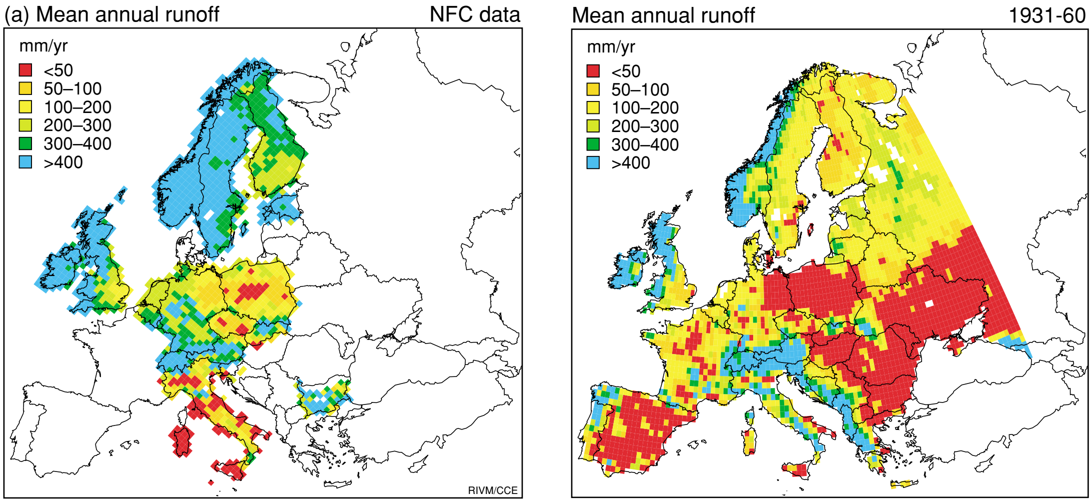

Figure 2-6. Mean annual runoff (precipitation surplus) on the EMEP50 grid derived from NFC data (left) and long-term (1931-1960) average values of precipitation minus evapotranspiration on 0.5°×0.5° grid computed from data by Leemans and Cramer (1991).

All NFCs submitted critical loads for forest ecosystems (see Fig. 2-4) and the total number of records (critical load values) in this category is 1,049,872 or about 72% of all records submitted (see Table 2-4). Therefore we present the following analyses for forest ecosystems only.

Cumulative distribution functions of sea-salt corrected base cation deposition and base cation weathering are shown in Fig. 2-7. These are crucial quantities, since they are the only sources of base cations which can neutralise acidity inputs. The variation in base cation deposition is considerable, with lower deposition in the northern and western countries and higher deposition in the southern (and eastern) countries. The weathering of base cations, calculated or estimated by various methods in different countries (see Part III), show large variations within most countries, but less inter-country variations, since most countries have weathering rates between almost zero and more than 2000 eq ha^{-1} yr^{-1}. Some very high values are most likely computed for calcareous soils.

Fig. 2-8 shows the cumulative distribution functions of the critical ANC leaching and the acceptable N leaching for forest soils. These quantities link the critical chemical value, i.e. the chemical criterion chosen to protect the desired (function of the) ecosystem, with the critical loads of acidity and nutrient N, respectively. The critical ANC leaching covers a wide range of values in most countries, with all values above 1000 eq ha^{-1} yr^{-1} in Bulgaria and Estonia. Italy and Norway did not provide information on ANC leaching, whereas Hungary did not compute critical loads of acidity (see Figs.2-4 and 2.5). The acceptable N leaching is below 300 eq ha^{-1} yr^{-1} in most of the area in most countries. Hungary provided empirical $CL_{nut}(N)$ data and Sweden did not provide separate N leaching data, as they are included in the N immobilisation (see below).

Cumulative distribution functions of N immobilisation and the denitrification fraction are shown in Fig. 2-9. The Mapping Manual (UBA 1996) recommends values for the long-term ("sustainable") net N immobilisation in the range between 0.5 and 1.0 kg N eq ha^{-1} yr^{-1} (35.7–71.4 eq ha^{-1} yr^{-1}). As can be seen from Fig. 2-9 most countries considered (substantially) higher values of N immobilisation as sustainable, i.e. low enough to avoid N saturation and consequent leaching in the long run. The denitrification fraction, i.e. the fraction of the net N input (deposition minus uptake minus immobilisation) that is denitrified, has been requested from NFCs for the first time. This parameter depends strongly on the humidity of the soil, which is often characterised by the soil type (see UBA 1996). Seven countries did not provide any information on this quantity, while four countries compute or estimate denitrification directly (as N_{de}) and not as a fraction of the net input of N. In two countries (Bulgaria and Norway), denitrification is assumed to be zero.

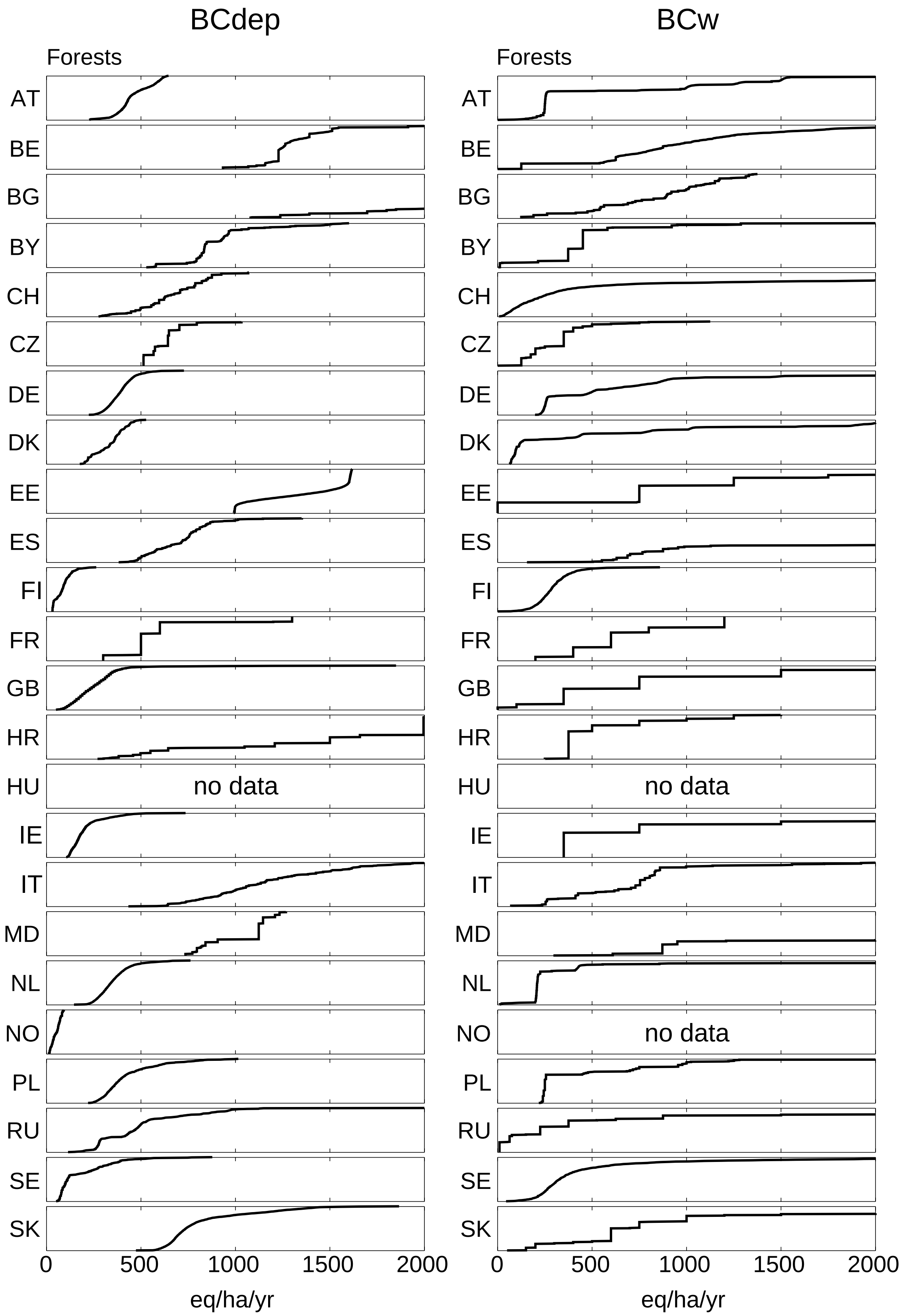

Figure 2-7. Cumulative distribution functions of sea-salt corrected base cation deposition (left) and base cation weathering (right) used to calculate critical loads (using primarily the SMB model) for forest soils from 24 national data sets.

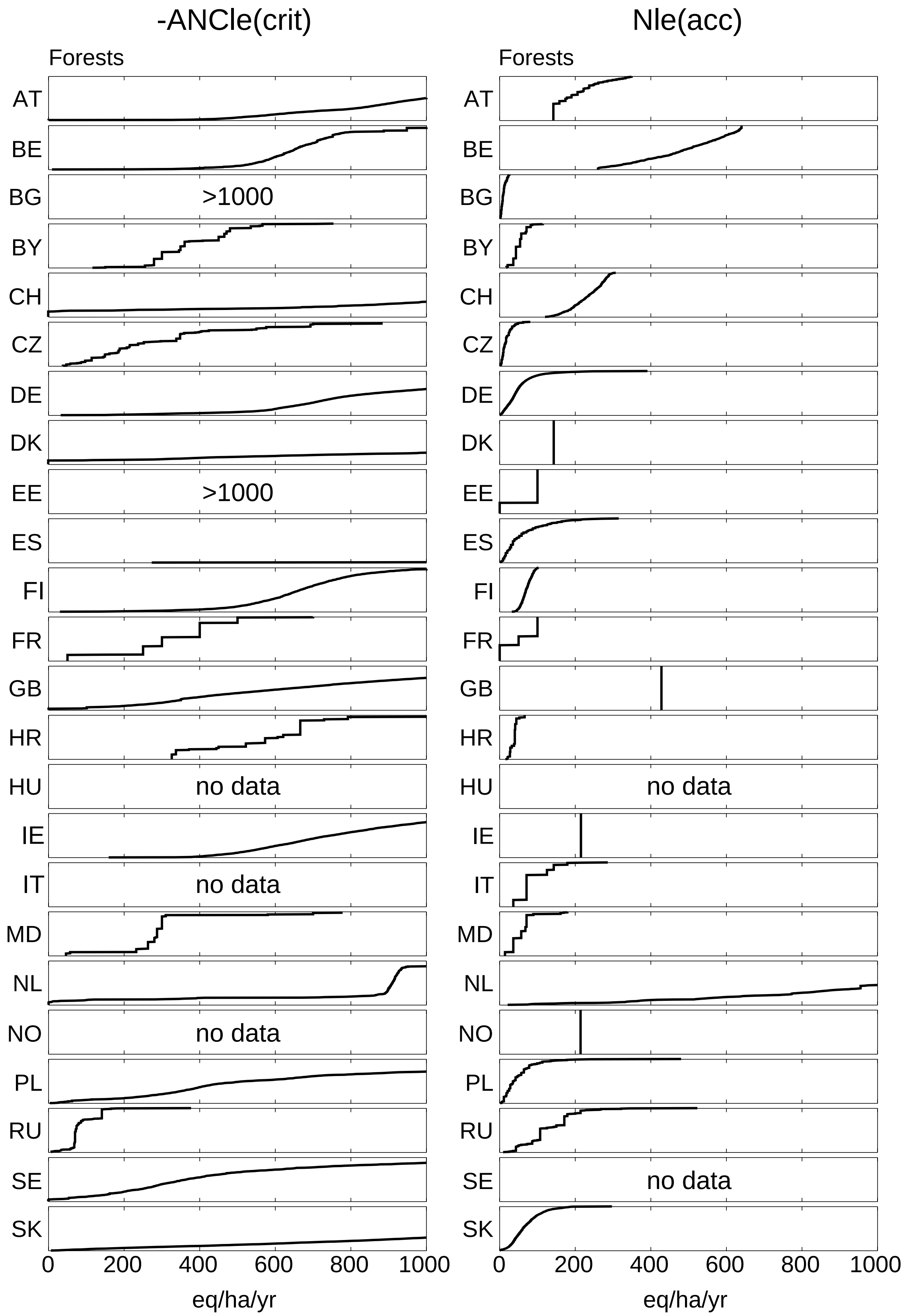

Figure 2-8. Cumulative distribution functions of (negative!) critical ANC leaching (left) and acceptable nitrogen leaching (right) used to calculate critical loads (using primarily the SMB model) for forest soils from 24 national data sets.

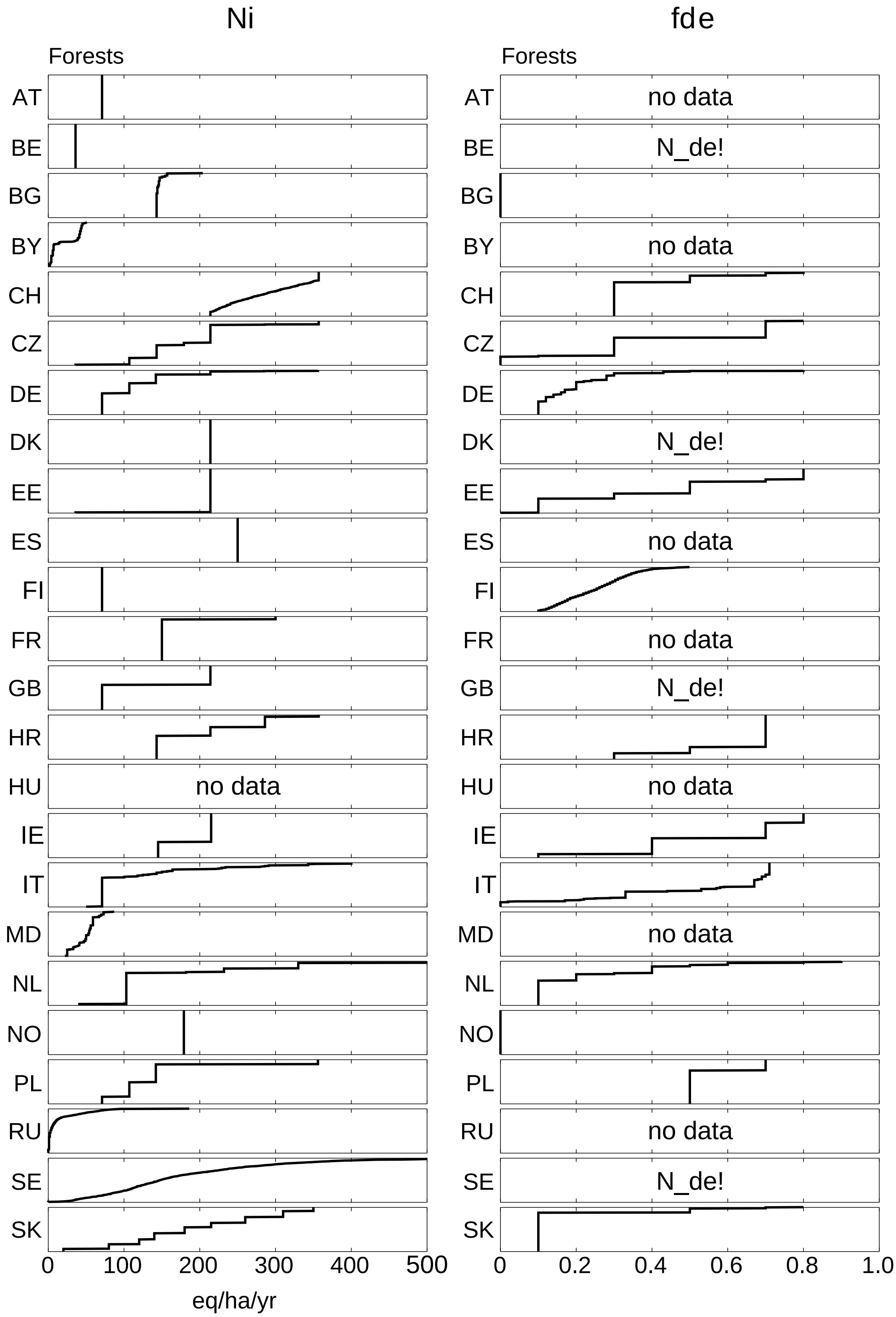

Figure 2-9. Cumulative distribution functions of N immobilisation (left) and denitrification fraction (right) used to calculate critical loads (using primarily the SMB model) for forest soils from 24 national data sets.

Fig. 2-10 shows the correlation between uptake of base cations (Ca+Mg+K) and nitrogen by the forest biomass. These values should only include the annual average amounts of these elements removed by harvesting. Thus they not only depend on tree species and climate, but also on the harvesting practices, e.g. nature reserves from which no trees are removed should be assigned zero uptake values. The uptake values provided by the NFCs lie by and large in the expected ranges. The ratio of base cation to nitrogen uptake for a given tree species should be almost constant with only minor variations to due climate and site quality. Comparing this figure with the corresponding one in the 1999 Status Report shows that some countries have improved their uptake calculations, whereas others (still) use a few values for many thousand forest sites distributed over the whole country.

More information on the critical loads and the assumptions, methods and parameters used to calculate them can be found in the National Focal Centre Reports contained in Part III of this report.

Concluding remarks

Following the most recent call for data by the CCE, 19 out of 24 NFCs submitted revised or updated critical load databases to the CCE, but only 11 of them were received before the announced deadline. The databases of the five countries that did not submit an update were amended by the CCE to fit the requirements (e.g. change of coordinate system). The communication between the CCE and the NFCs to answer questions or resolve data inconsistencies took much more time then anticipated by the CCE, especially considering that, in the absence of new data, only a few changes had been requested to the existing databases. This slow response by some NFCs not only inhibited in-depth reporting of results at the ICP-Mapping Task Force meeting, but also a thorough checking and analysis of the NFC data before the writing of this chapter. Although consistency in the databases and clarity of documentation improved, there is still a need to further harmonise methods and data used.

One area in which harmonisation becomes ever more urgent is the definition of ecosystems among NFCs. Since more emphasis is now put separating critical loads and their exceedances according to ecosystem types, a standardised system of ecosystem classifications used by all NFCs is desirable. This problem has been investigated as a "contribution-in-kind" by the UK NFC and a proposal for a user-friendly system has been put forward (see Part II, paper 2).

Another point for which no solution has yet been agreed upon is the assignment of ecosystem areas in the case of multiple ecosystems "covering" the same area, e.g. a forest growing in a lake catchment above an aquifer, and for which all multiple critical loads are derived. One option is to take the minimum of the critical loads assigned for overlapping areas. Another is to simply count the same area twice (or more); after all the areas are mostly used to "weigh" the influence of an ecosystem in a distribution. Each methods has its benefits and disadvantages, but a uniform solution would be desirable.

With the recent CCE call for data, NFCs were also asked to provide uncertainty estimates for the national critical load parameters. Only a few countries reacted to that call (see Part III). A reason for this could be that no detailed specifications had been provided for the expected quantities. Since the demand for uncertainty analyses will grow when depositions get near to critical loads (see also Part II, paper 1), more emphasis should be put on this activity, both by the CCE and the NFCs.

References

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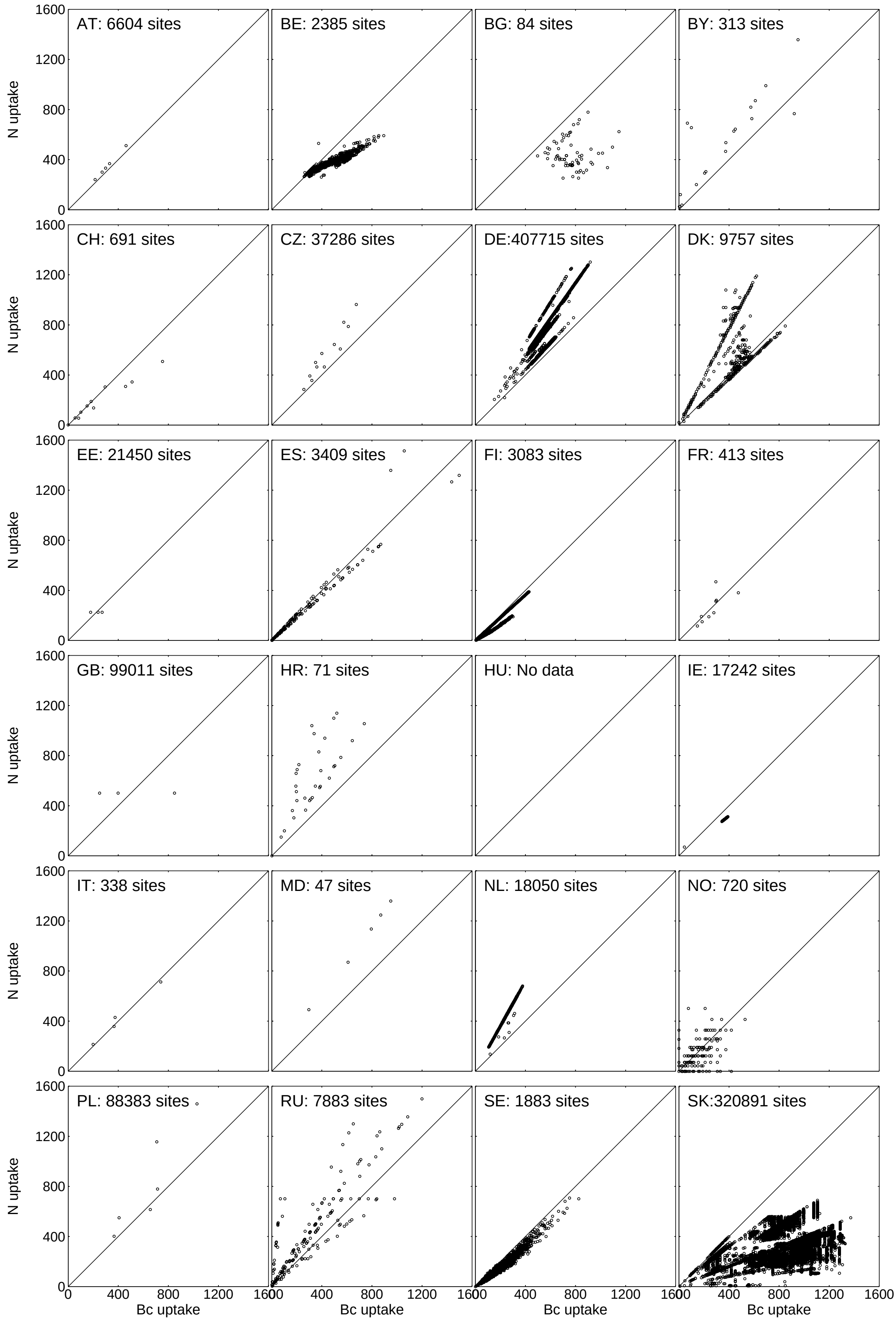

Figure 2-10. Correlation between the base cation (Ca+Mg+K) and nitrogen uptake used to calculate critical loads (using primarily the SMB model) for forest soils from 24 national data sets.

3. From Critical Loads to Dynamic Modelling

M. Posch and J.-P. Hettelingh

3.1 Introduction

The critical load concept has been developed in Europe since the mid-1980s, primarily under the auspices of the 1979 UN/ECE Convention on Long-range Transboundary Air Pollution (LRTAP). European data bases and maps of critical loads have been instrumental in formulating effects-based Protocols to the LRTAP Convention, such as the 1994 Protocol on Further Reduction of Sulphur Emissions and the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone.

Critical loads are based on a steady-state concept: they are the constant depositions an ecosystem can tolerate in the long run, i.e. after it has equilibrated with these depositions. However, many ecosystems are not in equilibrium with present or projected depositions, since there are processes ('buffer mechanisms') at work, which delay the reaching of an equilibrium (steady state) for years, decades or even centuries. By definition, critical loads do not provide any information on these time scales. Therefore, in its 17th session in December 1999, the Executive Body of the Convention "... underlined the importance of ... dynamic modelling of recovery" (UN/ECE 1999).

Dynamic models are not new. For 15 to 20 years scientists have been developing, testing and applying dynamic models to simulate the acidification of soils or surface waters, mostly due to the deposition of sulphur. But it is a relatively new topic for the effects-oriented work under the LRTAP Convention. Earlier work, e.g. under the ICP Integrated Monitoring, applied existing dynamic models at a few sites for which a sufficient amount of input data was available. The new challenge is to develop and apply dynamic model(s) on a European scale and to integrate them as much as possible with the integrated assessment work under the LRTAP Convention, in support of the review and potential revision of protocols.

To advance the knowledge on and use of dynamic models for the effects-related work under the Convention, experts in dynamic modelling met in October 2000 at an UN/ECE workshop in Ystad (Sweden). In one of the recommendations of the workshop the ICP Mapping was "urged" to draft a "Modelling Manual" (UN/ECE 2001). The purpose of such a manual would be analogous to that of the Mapping Manual (UBA 1996): To provide information to NFCs and their collaborating institutions to understand the concepts and data requirements and, ideally, to carry out dynamic modelling themselves, if and when requested by the Working Group on Effects. A first draft of such a Modelling Manual has been presented at the recent meetings of the CCE and the ICP Mapping. It was agreed that a revised and enlarged version with contributions by experts from the NFCs and other ICPs should be prepared.

The main aims of this chapter are to: (a) motivate the use of dynamic models in the effects-oriented work under the LRTAP Convention, (b) highlight the need for consistency between dynamic models and critical loads, and (c) discuss the models possible use in integrated assessment modelling.

3.2 Why dynamic models?

Critical loads assume a steady-state situation, and only two cases can be distinguished when comparing them to deposition at a given site (grid square): (1) deposition is below critical load(s), i.e. does not exceed critical loads, and (2) deposition is greater than critical load(s), i.e. there is critical load exceedance. In the first case there is no (apparent) problem, i.e. no reduction in deposition is deemed necessary. In the second case there is, by definition, an increased risk of damage to the ecosystem, and therefore the deposition should be reduced. A critical load serves its purpose as long as there is exceedance, since it indicates that deposition should be reduced. However, it is often assumed that reducing deposition to (or below) critical loads immediately removes the risk of 'harmful effects', i.e. the chemical parameter (e.g. the Al:Bc ratio), which links the critical load to the effect(s), immediately attains a non-critical ('safe') value. But the reaction of soils to changes in deposition is delayed by (finite) buffers, the most common being the cation exchange capacity (CEC). These buffer mechanisms can delay the attainment of a critical chemical parameter, and it might take decades or even centuries, before an equilibrium (or steady state) is reached. These finite buffers are not included in the critical load formulation, since they *do not* influence the steady state, but *do* influence the time to reach it. Dynamic models are needed to estimate the times involved in attaining a certain soil chemical state in response to deposition scenarios, e.g. the consequences of 'gap closures' in emission reduction negotiations.

For the sake of simplicity and in order to avoid the somewhat vague term 'ecosystem', we talk about (forest) soils, but essentially all considerations hold for lake systems as well, since their water quality is strongly influenced by properties of and processes in catchment soils.

Fig. 3-1 shows the possible development of a soil chemical variable (the Al:Bc ratio) in response to a 'typical' temporal deposition pattern. Five stages can be distinguished:

Stage 1: In this stage deposition was and is below the critical load (CL) and the chemical variable does not violate the criterion. As long as deposition stays below CL, this is the 'ideal' state.

Stage 2: Deposition is above the CL, but the chemical variable is still below the critical value. There is no risk for 'harmful effects' yet, there is a delay before the chemical criterion is violated. Therefore, damage is not visible in this stage despite the exceedance of the CL. We call the time between the first exceedance of the CL and first violation of the chemical criterion the *Damage Delay Time* ($DDT=t_2-t_1$).

Stage 3: The deposition is above CL and the chemical criterion is violated. Measures have to be taken to avoid further deterioration of the ecosystem.

Stage 4: Deposition is below the CL, but the chemical criterion is still violated, and thus recovery is not yet visible. We call the time between the first non-exceedance of the CL and the subsequent non-violation of the chemical criterion the *Damage Recovery Time* ($DRT=t_4-t_3$).

Stage 5: This stage is similar to Stage 1. Deposition is below the CL and the chemical criterion is not violated. However, it may still take a long time before the system, especially with respect to biological indicators, is fully recovered.

In addition to a large number of dynamic model applications to individual sites over the past 15 years, there are several examples of early applications of dynamic models on a (large) regional scale.

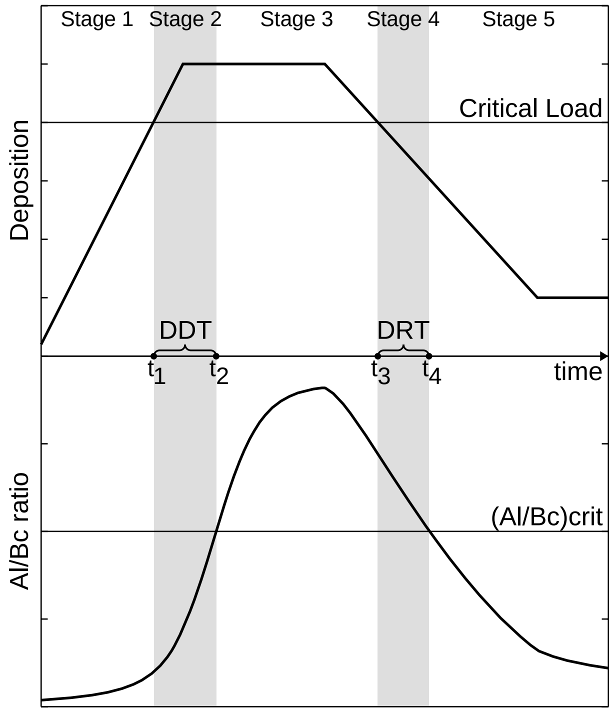

Figure 3-1. 'Typical' temporal (past and future) development of the deposition (top) and a soil chemical variable (Al:Bc ratio). Also depicted are the critical value of that variable and the critical load derived from it. As can be seen, there is a delay between the (non-) exceedance of the critical load and the (non-)violation of the critical chemical criterion: Damage Delay Time (DDT) and Damage Recovery Time (DRT).

Earlier versions of the RAINS model (Alcamo et al. 1990) contained an effects module which simulated soil acidification on a European scale (Kauppi et al. 1986) and lake acidification in the Nordic countries (Kämäri and Posch 1987). Cosby et al. (1989) applied the MAGIC dynamic lake acidification model to regional lake survey data in southern Norway. De Vries et al. (1994) used the SMART model to simulate soil acidification in Europe and Hettelingh and Posch (1994) used the same model to investigate damage delay and recovery times on a European scale.

3.3 Dynamic models and steady state

Steady-state models (critical loads) have been used to negotiate emission reductions in Europe. An emission reduction will be judged successful if non-exceedance of critical loads is attained. To gain insight into the time delay between the attainment of non-exceedance and actual chemical (and biological) recovery, dynamic models are needed. Thus the dynamic models to be used in the assessment of recovery under the LRTAP Convention have to be compatible with the steady-state models used for calculating critical loads. In other words, when critical loads are used as input to the dynamic model, the (chemical) parameter chosen as criterion in the critical load calculation has to attain the critical value (after the dynamic simulation has reached steady state). But this also means that concepts of the dynamic model used have to be a continuation and extension of the concepts employed in deriving the steady-state model. If critical loads are calculated with the Simple Mass Balance (SMB) model, the steady-state version of the dynamic model has to be the SMB; if critical loads are calculated with the PROFILE model, this has to be the steady-state version of the dynamic model used, etc.

Obviously, due to a lack of (additional) data and other resources, it will be impossible to run dynamic models on all sites in Europe for which critical loads are presently calculated (about 1.5 million; see chapter 2). However, the selection of the subset or sub-regions of sites, at which dynamic models are applied in support of integrated assessments, has to be representative enough to allow comparison with results obtained with critical loads.

Dynamic models of acidification are based on the same principles as steady-state models: The charge balance of the ions in the soil solution, mass balances of the various ions, and equilibrium equations. However, whereas in steady-state models only sources and sinks are considered which can be assumed infinite (such as base cation weathering), the inclusion of the finite sources and sinks of major ions into the structure of dynamic models is crucial, since they determine the long-term (slow) changes in soil (solution) chemistry. The three most important processes involving finite buffers and time-dependent sources / sinks are cation exchange, sulphate adsorption and the nitrogen dynamics in the organic soil layer.

Cation exchange is characterised by two quantities: cation exchange capacity (CEC), the total number of exchange sites (a soil property) and base saturation, the fraction of those sites occupied by base cations at any given time. After and increase in acidifying input, cation exchange (initially) delays the decrease in the acid neutralisation capacity (ANC) by releasing base cations from the exchange complex, thus delaying the soil acidification, until a new equilibrium is reached (at a lower base saturation).

On the other hand, cation exchange delays recovery since 'extra' base cations are used to 'replenished' base saturation instead of increasing ANC. Early model formulations of cation exchange reactions in the context of soil acidification can be found in Reuss (1980, 1983) and Reuss and Johnson (1986), and those formulations are still used in many models.

Sulphate adsorption by soils can be an important process for regulating sulphate concentration in the soil solution. Equilibrium between dissolved and adsorbed sulphate in the soil-soilwater system is typically described by a Langmuir isotherm, which is characterised by two numbers: The maximum adsorption capacity and the 'half-saturation constant' determining the speed of the response to changes in sulphate concentration. A description and extensive model experiments can be found in Cosby et al. (1986).

Finite nitrogen sinks: In the calculation of critical loads the terms in the net input of nitrogen are assumed constant over time or, in case of denitrification, a function of the (constant) N deposition. However, it is well known that the amount of N immobilised is in most cases larger than the long-term sustainable ('acceptable') immobilisation rate used in critical load calculations. Observational and experimental evidence (e.g. Gundersen et al. 1998) shows a correlation between the C:N ratio and the amount of N retained in the soil organic layer. This correlation has been used to formulate a simple model of N immobilisation both in the SMART model (De Vries et al. 1994) and recently in the MAGIC7 model (Cosby et al. 2001). In both models the amount of N retained is a function of the prevailing C:N ratio, which in turn is updated by the amount retained. If the C:N ratio falls below a prescribed value, all incoming N is leached.

As mentioned above, finite buffers, such as cation exchange, are not considered in the derivation of critical loads, since they do not influence steady-state situations. However, this does *not* mean that the state of those buffers is not influenced by the steady state! Thus variables characterising those buffers could be selected as chemical criteria for deriving critical loads. For example, in Posch (2000) it is shown how base saturation can be used as a critical chemical variable for deriving critical loads with the SMB model (see also Part II, paper 3).

3.4 Widely-used dynamic models

The equations resulting from the mathematical formulations of the processes mentioned above, or generalisations and variants thereof, together with appropriate solution algorithms and input-output routines have over the past 15 years been packaged into soil acidification models, mostly known by their acronyms. In the following we list a few of these models and shortly describe their main characteristics. The selection is biased towards models which have been (widely) used by others than the authors and which are simple enough to be applied on a (large) regional scale. (See also pp. 116–121 of the Mapping Manual (UBA 1996)).

The MAGIC model:
The MAGIC model (**M**odel of **A**cidification of **G**roundwater **I**n **C**atchments) is (one of) the oldest acidification models and is described in Cosby et al. (1985a,b). Sulphate adsorption is modelled according with a Langmuir isotherm (Cosby et al. 1986). MAGIC considers all four base cations at the cation exchange complex separately. In addition, MAGIC includes the fluoride concentration in the charge balance as well as the complexation of Al with sulphate and fluoride (in the form of equilibrium reactions). In a later version of the model the dissociation of (triprotic) organic acids has been included as well (Cosby et al. 1995). The latest version of the model, MAGIC7, also includes a description of the nitrogen dynamics in the soil (Cosby et al. 2001).

The MAGIC model, as its name implies, has been mostly used to model the ion concentrations in the water of (small) lakes or streams, for which the terrestrial catchment forms the soil compartment where cation exchange and sulphate ad/desorption occur. To obtain lake water concentrations the soil solution is degassed and excess Al precipitated. MAGIC applications are numerous and, in combination with Monte Carlo techniques, it has also been used to simulate changes in lake water chemistry on a regional scale.

The SAFE model:
The SAFE (**S**oil **A**cidification in **F**orest **E**cosystems) model has been developed at the University of Lund (Warfvinge et al. 1993) and a recent description of the model can be found in Alveteg (1998). The main differences between the MAGIC and SMART model are: (a) weathering of base cations is not a model input, but is modelled with the PROFILE (sub) model, using soil mineralogy as input (Warfvinge and Sverdrup 1992); (b) SAFE is a multi-layer model (usually four layers are considered), (c) cation exchange between Al, H and (divalent) base cations is modelled with Gapon exchange reactions, and the exchange between soil matrix and the soil solution is diffusion limited. Recently also sulphate adsorption has been included with an isotherm depending on sulphate concentration *and* pH (Martinson et al. 2000).

The SAFE model has been applied on many sites and more recently also regional applications have been carried out for Sweden and Switzerland (Kurz et al. 1998).

The SMART model:
The SMART model (**S**imulation **M**odel for **A**cidification's **R**egional **T**rends) is similar to the MAGIC model (but simpler) and is described in De Vries et al. (1989) and Posch et al. (1993). It models the exchange of Al, H and divalent base cations (as does SAFE) but describes them with two Gaines-Thomas equations. Sulphate adsorption is modelled as in MAGIC and organic acids can be described as mono-, di- or triprotic. Recently a description of the complexation of aluminium with organic acids has been included.

The SMART model has been developed with regional applications in mind, and an early example of an application to Europe can be found in De Vries et al. (1994).

Other models:
There is no shortage of soil (acidification) models, but most of them are not designed with regional applications in mind and/or are not usable, except by the designer. A comparison of 16 models can be found in a special issue of *Ecological Modelling* (Tiktak and Van Grinsven 1995). These models emphasise either soil chemistry (such as MAGIC, SAFE and SMART) or the interaction with the forest (growth). There are very few truly integrated forest-soil models. One of them is the forest model series ForM-S (Oja et al. 1995), which is implemented not as a "conventional" Fortran code, but is realised in the high-level modelling software STELLA II.

3.5 Input data

The input data needed to run dynamic models will vary with the model, but they can be roughly grouped into the following categories:

Deposition data: Future scenarios of sulphur and nitrogen deposition should be provided by the integrated assessment modellers, based on atmospheric transport modelling by EMEP. Also future base cation and chloride deposition are needed, but at present there are no projections for these elements on a European scale. Thus in most model applications present base cation depositions are assumed also to hold in the future.

Weathering, uptake, immobilisation, denitrification: In principle, these parameters should be the same as those used for critical load calculations. However, instead of using steady-state values, many dynamic models describe these processes as, e.g. a function of actual and projected forest growth. And to do so additional information (e.g. C:N ratios for immobilisation, forest growth rates for nutrient uptake) is needed.

Soil parameters: The most important soil parameters are the cation exchange capacity (CEC) and base saturation as well as parameters describing sulphate adsorption (if applicable), since these parameters determine the long-term behaviour (recovery) of soils. Other parameters include the various exchange and equilibrium constants.

It is important to note that in every application of a dynamic model it is not the actual simulation(s) which are time-consuming, but the preparation of suitable input data (files). Rarely will the required data be available in the form needed by the model. Thus, especially for regional applications, dedicated pre-processor software can be of great help.

3.6 Presentation of model results

For single-site applications the most obvious model output are graphs of the temporal development of the most relevant soil chemical variables, such as base saturation or the concentrations of ions in the soil solution (e.g. [Bc]:[Al] ratio) in response to a given deposition scenario.

In regional (European) applications this kind of information has to be summarised. This can be done in several ways, e.g.:

- Display of the temporal development of selected percentiles of the cumulative distribution of the variable(s) of interest (see, e.g. Kurz et al. 1998).
- Maps displaying the variable of interest at (say) five-year intervals (‘map movies’).

In addition, policy-makers will most likely be interested in the *time to reach a certain (steady) state* for a given deposition scenario. More specifically, answers to the following questions are of interest (Fig. 3-1; see also Warfvinge et al. 1992):

- The present load is greater than the critical load: How long does it take to reach a selected critical value, i.e. when does the risk of damage strongly increase? Information on this *damage delay time* (*DDT*) can be important for the timing of mitigation measures.
- The system is already at risk: For a given deposition at or below the critical load, how long does it take for the system to recover? Again, the knowledge of this *damage recovery time* (*DRT*) can help in the assessment of the effectiveness of mitigation policies.

Answering these questions requires:

- A clear definition of the (delay / recovery) times involved: e.g. if the future deposition equals the critical load, it takes (theoretically!) an infinite time to reach a steady state (asymptotically), although for all practical purposes it comes “close” (but how close is close?) to it within a finite time horizon.
- A way to convey information on DDTs / DRTs: Every soil has its own characteristic delay and recovery times. Thus methods are required to summarise this information on a European scale.

3.7 Dynamic models and integrated assessment

Ultimately, within the framework of the LRTAP Convention, a link has to be established between the dynamic soil models and integrated assessment (models), i.e. between the Working Group on Effects (WGE) and the Task Force on Integrated Assessment Modelling (TFIAM). The following modes of interaction with integrated assessment (IA) models have been identified:

(a) ***Scenario analysis:*** Deposition scenarios from IA models are used as input to dynamic models to analyse their impact on (European) soils, and the results (recovery times, etc.) are reported back.

Presently available dynamic models are perfectly suited to do that. The question is how to summarise the resulting information on a European scale. Also, the 'turn-around time' of such an analysis is bound to be long within the framework of the LRTAP Convention.

(b) Determination of target loads: Dynamic models are used to determine target loads, e.g. the maximum deposition allowed to reach a certain agreed-upon goal (value of a soil variable) within a fixed time horizon. These target loads are communicated to IA modellers to evaluate their feasibility of achievement (in terms of costs or technological abatement options available).

This requires no changes to existing models per se, but some additional work, since dynamic models have to be run backwards, i.e. iterative runs are needed. In addition, since both N and S contribute to acidity, it will not be possible to obtain unique pairs of N and S deposition to reach a given target (compare the critical load function for acidity critical loads!).

(c) Integrated soil module: A dynamic soil model is integrated into the IA models (e.g. RAINS) and used in scenario analyses and optimisation runs.

The presently available models (such as MAGIC, SAFE and SMART) are not easily incorporated into IA models and might be too complex to be used in optimisation runs. Alternatively, a simple dynamic model could be developed and incorporated into IA models, capturing the essential, long-term average features of existing dynamic soil models. This process of model simplification would be in analogy to the requirements that led to the simple ozone model included in RAINS, derived from the complex photo-oxidant model of EMEP.

(d) 'Recovery isolines': Response functions (broadly comparable to protection isolines for critical loads) are derived with existing dynamic models and incorporated into IA models.

These response functions ('recovery isolines' in the form of 'look-up tables') are pre-processed model runs for a large number of plausible future deposition patterns from which the results for every (reasonable) deposition scenario can be obtained by interpolation. A first attempt in this direction has been presented by Alveteg et al. (2000). An example, derived with the SMART model, is shown in Fig. 3-2. It shows the isolines of years ('isochrones') in which [Bc]:[Al] >1 is attained for the first time for a given combination of percent reduction (vertical axis) and implementation year (horizontal axis). The reductions are expressed as percentage of the deposition in 2010 after implementation of the Gothenburg Protocol and the implementation year refers to the full implementation of that additional reduction. For example, a 48% reduction of the 2010 deposition, fully implemented by the year 2030 will result in a (chemical) recovery by the year 2060 (dashed line in Fig. 3-2). Note that for this example site, at which critical loads are still exceeded after implementation of the Gothenburg Protocol, no recovery is possible unless further reductions exceed 32% of the 2010 level.

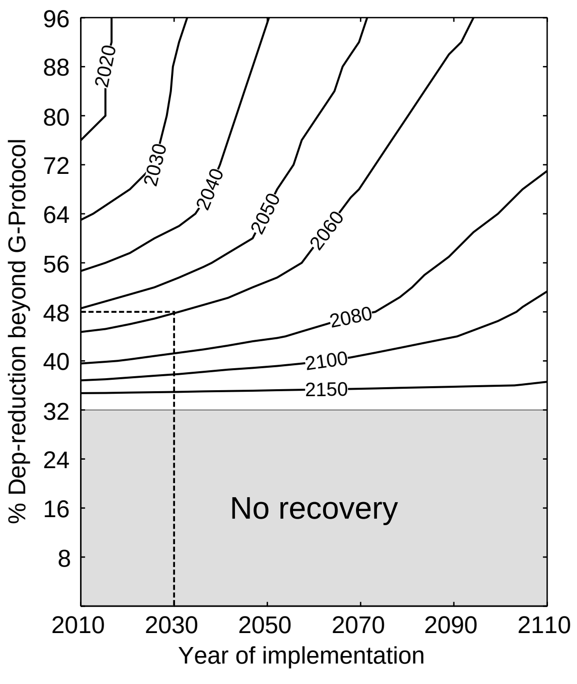

Figure 3-2. Example of 'recovery isolines'. The vertical axis gives the additional reduction in deposition *after* the implementation of the Gothenburg Protocol in 2010 (expressed as a percentage of the 2010 level) and the horizontal axis the year at which these additional reductions are fully implemented. The isolines are labelled with the first year at which [Bc]:[Al] >1 is attained for a given combination of percent reduction and implementation year.

This is only one way how output from dynamic models can be linked with integrated assessment models. Others are conceivable, and collaboration between scientists working with dynamic models and integrated assessment modellers has to be intensified to devise methods for linking (output of) dynamic models on a European scale with integrated assessment models.

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4. Intercomparison of Current European Land Use/Land Cover Databases

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4.1 Introduction

Land use/land cover databases are important to enable improved identification and assessment of ecosystems at risk for work under the LRTAP Convention. At present, all Parties that have submitted critical loads data to the Coordination Center for Effects (CCE) use ecosystem definitions which are based primarily on national or regional practices for environmental and nature conservation mapping purposes. Critical loads data used to support the Gothenburg Protocol (UN/ECE 1999) were assessed for many of these ecosystems which were broadly classified as forest, (semi-)natural vegetation and surface waters. A more detailed common set of definitions for the ecosystems will become more important as the review of protocols under the Convention emphasises the importance of data reliability and assessment of uncertainty. In addition, harmonised ecosystem definitions can contribute to an improved understanding and interpretation of both "stock-at-risk", an important element in the temporal and spatial assessment of exceedances. Common definitions can also increase the compatibility and comparability of data from National Focal Centres (NFCs) which are integrated in the European CCE database.

Therefore, the CCE decided to undertake a survey and comparison of existing European databases, in order to assess their suitability for use in ICP Mapping activities. The study was financed by the Department for Climate and Industry of the Dutch Ministry of Housing, Spatial Planning and the Environment (VROM) and executed in collaboration with the firm Geodan-IT in Amsterdam. This chapter summarises the results of the project. Further details from the intercomparison will be made available to NFCs.

In practice, currently available databases often make no clear distinction when using the terms "land use" and "land cover". For convenience, this paper uses the term "land cover" throughout when referring to these databases.

4.2 Objectives of a common European land cover database

Within the ICP Mapping there is a need for consensus on a common European land cover database to improve the assessment of stock-at-risk both on a European scale as well as comparisons among countries. A common land cover map should enable analyses of:

- which receptors are exceeded by various critical thresholds (critical loads of acidity and eutrophication, critical levels of ozone and critical limits of heavy metals).
- the geographical location of these exceedances.
- dynamic assessments of recovery and damage beyond geochemical assessments to also include temporal horizons on biological changes.
- the ecosystems of countries that have not yet submitted data and do not yet participate in ICP Mapping activities.

An important requirement is that the database can become freely available to work under the Working Group on Effects in general, and the ICP Mapping in particular. The recommended land cover database finally selected as result of this project will be made available to all NFCs.

4.3 Method of work

This database intercomparison project comprised two major phases. The first phase focused on: (a) defining criteria by which to evaluate land cover databases, (b) developing an inventory of available land cover databases and maps, and (c) applying the criteria to assess the databases' usefulness for ICP Mapping work. Phase two of the project compared both statistical and geographical characteristics of the maps with the help of Geographic Information Systems (GIS).

4.3.1 Definition of criteria

In phase 1 of the project, criteria were developed, insofar as possible, to reflect potential ICP Mapping requirements, independent of the structure and characteristics of the databases under consideration. The following points were identified as key criteria by which to gauge the applicability of a database to ICP Mapping work:

- Compatibility with other currently used datasets, such as datasets in longitude / latitude degrees and the EMEP 50×50 km^2 ("EMEP50") grid system.
- A high degree of reliability relative to the described goals of the database.
- The availability of proper land / sea and country borders.
- The appropriateness of the level of detail (distinction of forest, vegetation and agricultural classes and spatial resolution) for both national and Europe-wide purposes.
- The greatest possible degree of European coverage.
- A high update frequency to enable assessment of changes of stock-at-risk over time.
- The ability to process updates for use by the CCE and its network.
- Compatibility with the "CCE Viewer", a geographical data viewer software package distributed to all NFCs.
- A high likelihood of acceptance within the ICP Mapping community.

4.3.2 The inventory of land cover maps

The five land cover databases selected are presented in Table 4-1, including a summary of their characteristics derived from documentation that came with the databases and from the internet. The characteristics listed are: ownership / financier, version and date, availability / copyrights, source dates, spatial coverage, spatial resolution, sources (types, scales, classifications), update frequency, classes, accuracy, and peculiarities.

4.3.3 Application of the criteria to the maps

Table 4-2 depicts the relative strengths of the five land cover databases reviewed vs. the set of pre-defined criteria. The following can be noted from the comparison presented in Table 4-2:

- The *reliability* of data refers to the transparency of the methods used to collect field data, the level of spatial resolution and the way this information is processed to derive the land cover database. CORINE and PELCOM provide the best documentation regarding these aspects. The *appropriateness* refers to the suitability of a database for use by the CCE.
  – IGBP-DIS and Olson are based on the same satellite image analyses. The IGBP Land Cover Legend classification is used to derive the IGBP-DIS database from image analysis, while the Global Ecosystems Legend classification is used for the Olson database. Both classifications are applied on the automatically generated clustering derived with a so-called "unsupervised clustering of monthly NDVI maximum value composites". This clustering focuses on distinguishing agro-ecological zones, which is particularly useful for analysing global climate change-related effects. Both the ISGP-DIS and Olson land cover databases show little variation within Europe.
  – The SEI database is based on a wide range of data sources and expert knowledge. The information in the SEI database consists of polygons that were digitised from large-scale sources. The disadvantage may be that polygons represent large homogeneous areas with a single type of land cover, while in reality the area may consist of more heterogeneous patterns of land cover types. The SEI database was originally developed for land cover studies on a continental scale.
  – The PELCOM database has the advantage that it provides detailed land use information that is relatively well-suited for (national) environmental applications.
  – The CORINE database has the highest resolution, but does not cover the entire European continent. The scanning dates of the satellite images used to derive the CORINE land cover information vary significantly among countries. Many CORINE classes are heterogeneous or developed from compositions based on functional land use.

Table 4-1. The five selected land cover databases with their specifications derived from their documentation and the internet.

Characteristic	**CORINE Land Cover database**	**SEI Land Cover Map of Europe**	**PELCOM (Pan-European Land Cover Monitoring)**	**IGBP-DIS Land Cover Classification (International Geosphere-Biosphere Program - Data and Information System)**	**Olson-Global Ecosystems**
Ownership / Financier	European Environment Agency, Copenhagen, Denmark	The Stockholm Environment Institute, York, UK	Alterra, Wageningen, Netherlands	IGBP-DIS	US Geological Survey - EROS Data Center, 1994
Version / date	V12/99 (CLC-V12-99); June 2000	Version 1; 1999	Version unknown; 1999; ISBN 91-887-14-705	Version 1.2; November 1997	Version 1.2; November 1997
Availability / Copyrights	The EEA agrees to grant to the Customer, who accepts, the non-exclusive and not transferable right to use and process the datasets included in NATLAN 2000. These can be found at: http://natlan.eea.eu.int/termsofuse.htm	Not for free distribution; contact SEI.	Distribution is free, Registration of use is requested. Internet: http://cgi.girs.wageningen-ur.nl/cgi/siteguide/	Freely available, can be downloaded through the Internet: http://www.ngdc.noaa.gov/paleo/igbp-dis and http://edcdaac.usgs.gov/	EROS Data Center. Internet: http://edcdaac.usgs.gov/
Source dates	1985–19897	1970–1996	DLR: 1997 IGBP data: May–October 1995 Lannion data	April 1992–September 1993	April 1992–September 1993
Spatial coverage	EU countries, EU accession states and several eastern European states.	Pan-Europe	Pan-Europe	Global	Global
Spatial resolution	Raster format with 250m cells; Finland, Switzerland and Austria provided as separate databases with projections deviating from the standard.	< 1:2.000.000. 12 Arc/Info export files (vector format).	Erdas Imagine format with 1100m cells.	Erdas Imagine format with 1100m cells.	Erdas Imagine format with 1100m cells.

Table 4-1 (continued). The five selected land cover databases with their specifications derived from their documentation and the internet.

Characteristic	CORINE Land Cover database	SEI Land Cover Map of Europe	PELCOM (Pan-European Land Cover Monitoring)	IGBP-DIS Land Cover Classification (International Geosphere-Biosphere Program - Data and Information System)	Olson-Global Ecosystems
Sources (types, scales, classifications)	Visual interpretation of Landsat / SPOT XS at a scale 1:100,000 with simultaneous consultation of ancillary data (CEC 1993). The project, begun in 1986, is still progressing, leading to large differences in satellite acquisition dates for the various countries in Europe. Ancillary data used for some countries: Topographic maps at the scale of 1:50,000 and at the scale of 1:100,000, (area II) panchromatic aerial photographs at 1:70,000 and 1:32,000 (Berlin region); panchromatic aerial photographs at the scale of 1:70,000, or photographs at 1:100,000 taken with KFA-1000 and MK4 cameras on board the RESURS satellite of the KOSMOS class if there were no panchromatic aerial photographs at the scale 1:70,000.	Source maps: • FAO Land Use Map of Europe, 1:2,500,000 • Land Use of the Former USSR, 1:4,000,000 • ESA Forest Map of Europe, 1:2,000,000 • Forests of the USSR, 1:2,500,000 • CORINE Soil Map of the European Community, 1:1,000,000 • FAO Soil Map of the World, 1:5,000,000.	NOAA / AVHRR images from DLR, the IGBP MVC and from the Lannion database.	Unsupervised clustering of monthly NDVI maximum value composites (MVC) NOAA / AVHRR on a continental basis. DEM data are used to model the ecological factors governing natural vegetation distribution, and are important for identifying Land cover types and stratifying seasonal regions representing two or more disparate vegetation types. Ecological regions data are used to identify regions with disparate land cover types and to stratify seasonal regions representing two or more disparate vegetation types. Both continental and country level ecoregions data are used in this process. Maps and atlases of ecoregions, soils, vegetation, land use, and land cover are used in the interpretation phase of the study and serve as reference data to guide class labelling.	Same sources as IGBP. Seasonal land cover regions Global Ecosystem framework (1994a, 1994b). Olson defined 94 ecosystem classes based on their land cover mosaic, flora properties, climate, and physiognomy. The Global Ecosystems framework provides a mechanism for tailoring data to the unique each continent, while still Providing a means for summarising the data at the global level.

Table 4-1 (continued). The five selected land cover databases with their specifications derived from their documentation and the internet.

Characteristic	CORINE Land Cover database	SEI Land Cover Map of Europe	PELCOM (Pan-European Land Cover Monitoring)	IGBP-DIS Land Cover Classification (International Geosphere-Biosphere Program - Data and Information System)	Olson-Global Ecosystems
Update frequency	An update is planned in the period 2000–2003.	No information on update frequency. "An update of this interim release is planned in the future".	Not planned, new STEMS project is proposed for the commission where the methodology will be improved and better use ofother information sources is planned.	Version 2.0 is recently out containing improvements based on remarks and experiences from evaluation projects.	Version 2.0 is recently out containing improvements based on remarks and experiences from evaluation projects.
Classes	44 level 3 classes, 16 level 2 classes and 7 level 1 classes. (Austria and Switzerland classifications compatible with CORINE level 2 classification only; Finland not fully compatible with CORINE level 2 classification) .	7 main classes, 25 main subclasses, 129 sub-subclasses.	12 classes.	17 general cover types selected based on the requirements of the IGBP core projects.	94 classes.
Accuracy	No specific accuracy information available.	The overall accuracy of the final map is undetermined at present. ESA give the accuracy of the 'Forest Map of Europe' as 82.5%, with a range of 61–100%. The FAO and Eastern European land use map have an indeterminate accuracy. FAO and CORINE soil data have been compiled from systematic soil surveys. For the FAO this was originally compiled at a scale of 1:1,000,000. The soil information included in the database can therefore be considered highly accurate.	Accuracy assessment on basis of 40 interpreted high-resolution images over Europe resulted in an overall accuracy of 69.2%.		

Table 4-1 (continued). The five selected land cover databases with their specifications derived from their documentation and the internet.

Characteristic	CORINE Land Cover database	SEI Land Cover Map of Europe	PELCOM (Pan-European Land Cover Monitoring)	IGBP-DIS Land Cover Classification (International Geosphere-Biosphere Program - Data and Information System)	Olson-Global Ecosystems
Peculiarities	The project started in 1986 and is still progressing, leading to large differences in satellite acquisition dates for the various countries in Europe. Database is still incomplete for pan-European area. Most CORINE classes are heterogeneous, and / or are determined by functional land use and consequently consist of various land cover types. The subjectivity and the dependence on ancillary data for some classes will have major consequences for any updating (Thunnissen and Middelaar 1995, Perdigão and Annoni 1997).	None noted	None noted	From the Pelcom report: “Experiences indicate that the clustering technique does not identify forests satisfactorily (Champeaux et al. 1998a). An additional limitation is that the 1-km database according to the DISCover legend contains complex classes, e.g. cropland / natural vegetation mosaics, which are difficult to apply in environmental studies. In the DISCover data set about 27 per cent of the pan- European land surface is covered by this land cover class. Application of the database in environmental and climate studies for pan-Europe may be limited.”	Same remarks as the IGBP dataset. The classification scheme is the only difference between these datasets.

Table 4-2. Applicability of land cover databases in meeting ICP Mapping criteria, ranked by importance of each criterion as determined during intercomparison exercise.

Criterion *	Rank **	CORINE	PELCOM	SEI	IGBP	Olson
Reliability / appropriateness	1	++	++	+•	••	••
Level of detail in classes	2	++	+•	+++	+	+
Spatial coverage	3	+•	+	++	++	++
Acceptance within ICP Mapping	4	++	+	••	+•	+•
Ability to process new updates	5	••	N / A	N / A	N / A	N / A
Compatibility with the CCE Viewer	6	N / A	N / A	N / A	N / A	N / A
Update frequency	7	+•	+•	+•	+•	+•
Compatibility with other sources	8	N / A	N / A	N / A	N / A	N / A
Land / sea and country borders	9	+++	++	••	++	++

* Scoring scale: +++ good
++ good
+ sufficient
+• reasonable
•• poor
N / A not applicable, or criterion does not sufficiently discriminate among different databases.

** Numbers indicate the ranking of a criterion's importance determined during review of the databases (1=highest, 9=lowest).

- It is important to distinguish a sufficient *level of detail* within forest, vegetation and agricultural land cover types, as the database is to be used to estimate sensitivity of natural receptors to air pollutants. Table 4-3 presents an overview of the number of classes distinguished for these types.

- The *spatial coverage* should preferably focus on Europe rather than other (global) scales. The IGBP-DIS and Olson databases have a world-wide coverage. SEI has a European coverage including the Ural mountains. PELCOM has and European coverage up to 42 degrees East in Russia. CORINE contains coverage for most EU countries as well as several Eastern European countries, e.g. Poland, Bulgaria and Romania.

- Acceptance of a common land cover database within the ICP Mapping depends strongly on the common *support and acknowledgement* by the participating countries of the database. CORINE is used by several countries and is formally embedded in EU-related mechanisms. This formal recognition is lacking for the other databases.

- Implementing a new land cover database update within the ICP Mapping requires the ability to easily *process updates* (e.g. downloading, importing into Arc / Info or other GIS, and converting). In regard to this criterion, the CORINE database consists of different projections, and thus additional processing compared to the other databases.

Table 4-3. The number of classifications of forest and agricultural land cover types.

Database	Number of forest types	Number of agricultural land use types
CORINE	3 classes of forest (level 3)	3 classes "Arable land" 3 classes "Permanent crops" 1 class "Pastures" 4 classes "Heterogeneous agricultural areas"
PELCOM	3 classes of forest	4 classes "Arable land" 1 class "Permanent crops"
SEI	16 classes "needleleaf" 12 classes "broadleaf" 19 classes of "mixed" forest	5 classes "Agriculture" 10 classes "Horticulture"
IGBP	5 classes of forest	1 class "Croplands" 1 class "Cropland / natural vegetation"
Olson	15–20 classes of forest, 4 of which are present in Europe	5 classes

- The capabilities to be *incorporated into the CCE Viewer* appears to imply format conversion procedures which are similar for all databases and therefore not a discriminating criterion.

- The *update frequency* of a database is important particularly for the analysis of temporal trends, which is expected to become more relevant with respect to current and future assessments of stock-at-risk. Most databases examined, with the exception of the CORINE database, lack detailed information on the frequency of updates. CORINE plans an update during 2000–2003, although it is unclear whether this update will extend geographic coverage or will only update existing areas. Updated version of the IGBP and Olson databases have recently become available.

- The criterion *compatibility with other sources* (e.g. for overlay purposes) did not allow sufficient discrimination among databases and was therefore given a low importance.

- *Land/sea and country borders* need to have a sufficient level of detail, since the database is intended to be used on national scales as well. While none of the databases reviewed contain country borders, their geographic accuracy is related mainly to the scale and resolution of the original sources used to compose the land cover database. Thus CORINE contains the most detailed land / sea borders followed by (in decreasing order): PELCOM, IGBP, Olson and SEI.

4.3.4 Recommendations from phase 1

1. The IGBP-DIS and Olson databases are less suitable for use within the Mapping Programme as their potential applications focus more on global rather than national scales. It was recommended not to include these databases in phase 2.

2. The SEI database is composed from mostly large-scale data sources, making it especially suitable for continental rather than national applications. Sources to compose the database originate from a wide range of surveillance dates (1970–1996) requiring semi-quantitative expert judgements. The SEI map is currently used in applications within the ICP Vegetation.

3. The CORINE and PELCOM databases have comparable characteristics with respect to coverage, classification and applicability on national scales. This is also due to the fact that the CORINE database was used as an important reference for the PELCOM database. CORINE is subject to updates driven by the EU, but currently lacks full geographic coverage of Europe.

4.3.5 Intercomparison of selected databases

Phase 2 of the project focused on the intercomparison of the CORINE, PELCOM and SEI databases. To do this, it was necessary to (a) generate a unique classification of land cover types to be applied to each of the databases, (b) choose a common projection format and (c) compute and compare the occurrence of each of the ecosystem classes within Europe, between countries and between EMEP grid cells. This last step also attempted to use GIS overlay techniques.

A common set of land cover types, consisting of 16 classes using the PELCOM classification extended with classes distinguishing between natural ecosystem land cover and agricultural land use, was first defined. Table 4-4 shows the applicability of the 16 classes to each of the databases. The effect of reclassifying the CORINE, PELCOM and SEI databases into the 16 classes is illustrated in Fig. 4-1 for the Netherlands. The original resolution of each of the maps has been kept unchanged, thus illustrating the level of detail provided by the CORINE database.

Next, all three databases were converted into a common projection to allow comparison of land cover classes throughout Europe and within individual countries and EMEP50 grid cells. Overlaying techniques, including “rubbersheeting”, attempted to match land / sea borders. This succeeded only within an interval around the borders ranging from 1–4 km, which is unacceptable as a basis for cross-table analysis by GIS. The cross-tables illustrate for each pair of databases the correlation of occurrence of the same land cover classes at the same allocations in each database. The higher the correlation, the more the contents of the databases agree. The source of the incompatibility of borders is unknown, and could include reasons ranging from the digital treatment of input data to storage mechanics (e.g. polygon versus grid data formats).

Table 4-4. Overview of the regrouping of all land cover classes and their presence in the CORINE, PELCOM and SEI databases.

Description of regrouped classes	CORINE	PELCOM	SEI
1. Urban areas	✓	✓	✓
2. Non-irrigated arable land	✓	✓	✓
3. Irrigated arable land	✓	✓	
4. Permanent crops	✓	✓	✓
5. Pastures	✓	✓	✓
6. Heterogeneous agricultural areas	✓		
7. Broad-leaved forest	✓	✓	✓
8. Coniferous forest	✓	✓	✓
9. Mixed forest	✓	✓	✓
10. Natural grassland	✓		✓
11. Shrubs	✓	✓	✓
12. Open spaces with little or no vegetation	✓	✓	✓
13. Glaciers and perpetual snow	✓	✓	✓
14. Wetlands	✓	✓	✓
15. Water	✓	✓	✓
16. Data gaps	✓	✓	✓

4.3.6 Example intercomparison results for the Netherlands

More detailed findings from the intercomparison of land cover classes within Europe, countries and EMEP50 grid cells will be further summarised and disseminated by the CCE. Results from an application carried out for the Netherlands is included here for illustrative purposes (see Tables 4-5 and 4-6, Fig. 4-2).

4.3.7 Results of the intercomparison exercise

A summary of the results of the database inter-comparison for the whole of Europe includes the following observations:

- CORINE and PELCOM show similar spatial patterns of land use, which is not surprising because CORINE was an important source of information for PELCOM.
- "Mixed forest" occurs in SEI about twice as much as in CORINE and PELCOM.
- The quantity of forested area in Finland is known from the national statistics office: 79% of the country, while the CORINE database reports 45%, PELCOM 55% and SEI 68%. Reasons for these differences are unknown.
- PELCOM and SEI do not distinguish "Hetero-geneous agricultural areas" as a distinct category, while CORINE data include about 14% in this class. This discrepancy can perhaps be attributed to discrepancies in definitions used in the original classification schemes, which could have contributed to higher values of "Non-irrigated arable land" in PELCOM and SEI as compared to CORINE.
- The classification "Natural grassland" is repre-sented in the CORINE and SEI databases, but not that of PELCOM. On the basis of the percentages available for other land cover classes, it is reason-able to assume "Natural grassland" to be part of the "Arable land" classification used in PELCOM.
- Land / sea borders in the SEI database seem to be less accurate than those in CORINE and PELCOM, as evidenced by missing Wadden islands and IJsselmeer polders in the Netherlands.

Comparison for each country and EMEP50 grid cell leads to the general observation that CORINE and PELCOM show similar spatial patterns of land cover in the countries covered by CORINE. This is to be expected from the fact that PELCOM has incorpo-rated CORINE information as appropriate.

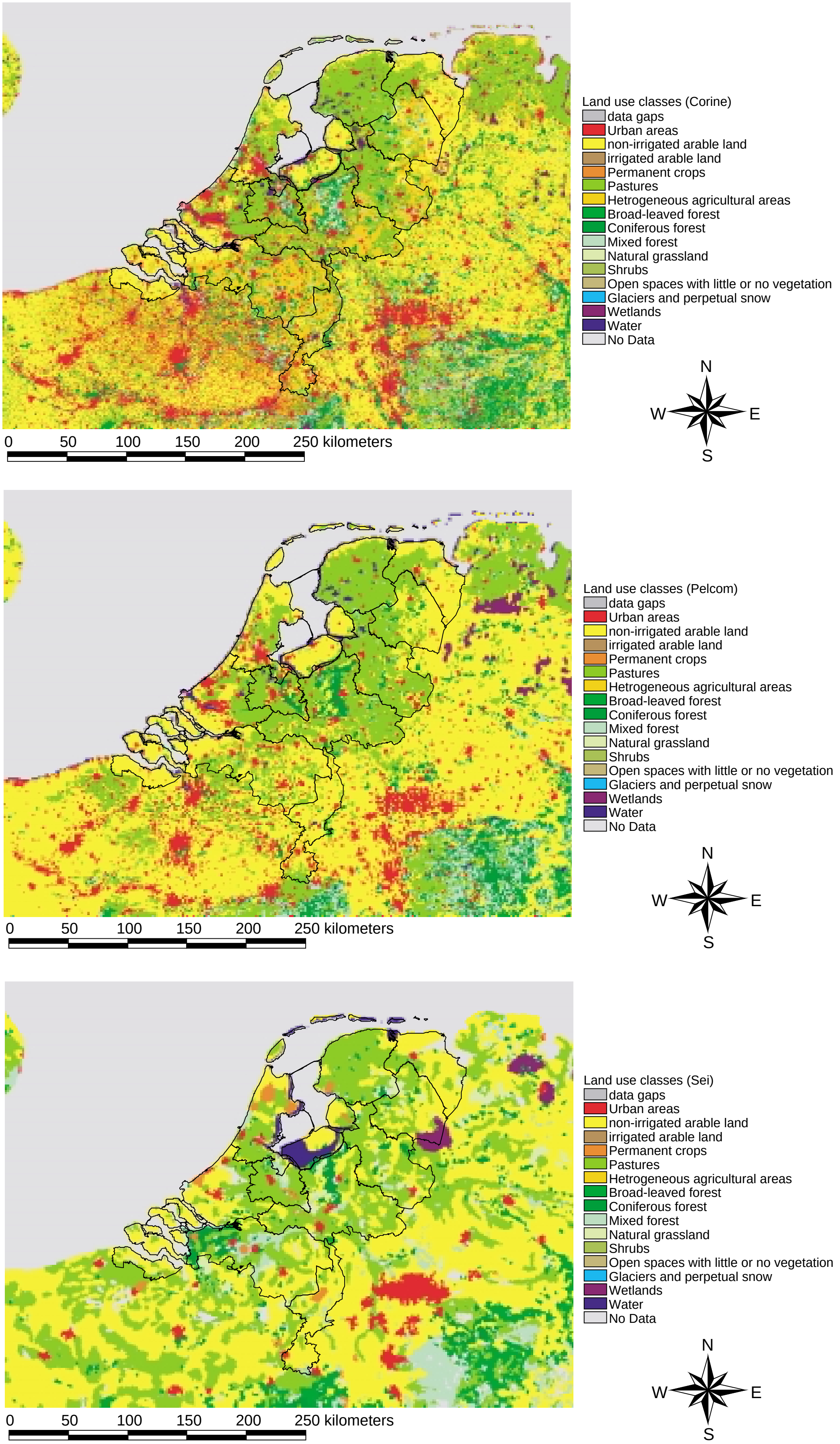

Figure 4-1. Results (for the Netherlands) of reclassification into 16 classes of the CORINE (top), PELCOM (middle) and SEI (bottom) databases in their original resolution.

Table 4-5. The land cover types with their areas (in km^2) and as percentages of the total area of the Netherlands for the CORINE, PELCOM and SEI databases.

		Area (km^2)			Percentage		
Class	**Class description**	**CORINE**	**PELCOM**	**SEI**	**CORINE**	**PELCOM**	**SEI**
1	Urban areas	3,538	2,583	603	10.0	7.3	1.7
2	Non-irrigated arable land	7,754	14,793	12,687	22.0	41.9	35.9
3	Irrigated arable land	0	0	–	0.0	0.0	–
4	Permanent crops	95	0	760	0.3	0.0	2.2
5	Pastures	11,720	14,224	14,048	33.2	40.3	39.8
6	Heterogeneous agricultural areas	6,347	–	–	18.0	–	–
7	Broad-leaved forest	469	74	87	1.3	0.2	0.2
8	Coniferous forest	1,625	1,681	1,995	4.6	4.8	5.7
9	Mixed forest	929	324	1,331	2.6	0.9	3.8
10	Natural grassland	735	–	953	2.1	–	2.7
11	Shrubs	7	0	0	0.0	0.0	0.0
12	Open spaces, little / no vegetation	178	0	0	0.5	0.0	0.0
13	Glaciers and perpetual snow	0	0	0	0.0	0.0	0.0
14	Wetlands	323	12	212	0.9	0.0	0.6
15	Water	1,234	1,640	1,221	3.5	4.6	3.5
16	Data gaps	360	1	1,416	1.0	0.0	4.0
	Totals:	35,314	35,332	35,312	100	100	100

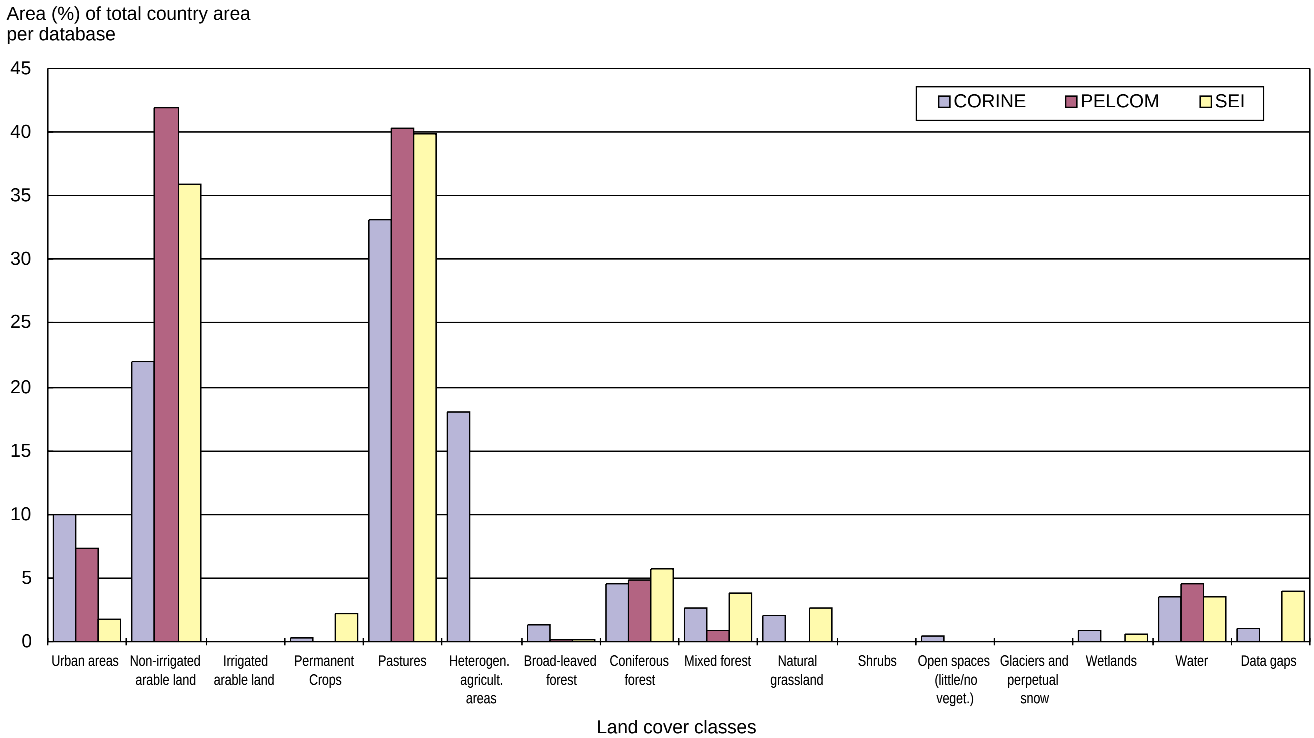

Figure 4-2. Histogram showing land cover types as a percentages of total Netherlands territory in the CORINE, PELCOM and SEI databases (based on data from Table 4-5).

Table 4-6. Comparison of land cover type area as percentage of total country area of the Netherlands verified with FAO statistics for “arable land”, “permanent crops”, “pastures” and “forest”.

Class description	FAO	CORINE	PELCOM	SEI
Arable land	22.0	22.0	41.9	35.9
Permanent crops	0.9	0.3	0.0	2.2
Pastures	25.2	33.2	40.3	39.8
Forest	7.3	8.6	5.9	9.7
Total country area (km^2)	40,840	35,314	35,332	35,312

4.4 General recommendations and follow-up

Based on the above analyses, the most suitable land cover database for CCE purposes is the CORINE database. It has the highest resolution (250-meter grids) which satisfies requirements enabling the use of this database on both national and European scales. The accuracy is high, due to the high-resolution surveillance sources from which it is composed (25-meter SPOT and Landsat imagery). Several NFCs (e.g. Ireland and Spain) already used the database for the critical loads inventories in their countries. The database is also well-documented (Perdigão and Annoni 1997, CORINE website), contains reliable sources, and update activities are overseen by the European Environmental Agency. One important disadvantage is that CORINE does not yet cover the entire European area to which the LRTAP Convention applies.

The SEI map is well-suited for broad-scale European applications. However, its polygon structure based on broad-scale map and data sources may lack sufficient detail for applications on sub-regional and national scales.

The IGBP-DIS and Olson databases mainly serve globally oriented assessments such as climate change. These databases lack sufficient detail for national applications.

In general it should be noted that the comparison of geographical land cover databases turned out to be more difficult than expected. While it is important to ultimately agree on a common land cover database to advance further work regarding stock-at-risk, it is recognised that the selection of such a database is prone to uncertainty. For example, background information lacks the necessary uniformity to enable an error-proof reclassification of all the databases into a common set of 16 classes used in this study. Semi-quantitative judgements are unavoidable when attempting to compare maps for which the land/sea borders cannot reliably be overlaid.

More work will be needed to improve information on land cover. This can best be achieved by encouraging the NFCs to improve a preliminary common database as starting point. We believe that the use of CORINE by the Parties to the Convention now covered by the database, and the use of PELCOM by other parties could be such a starting point. The currently proposed common classification system might serve as a basis for inter-country and inter-regional comparisons of stock at risk. The database can also be used as a background database for NFCs which lack national data. In addition, it can contribute to the further improvement of the verification of the critical loads database.

The CCE will make available both the CORINE and PELCOM databases in a digital format to NFCs, with the following specification:

- 1×1 minute longitude/latitude grid cell.
- both in ASCII format and in CCE Viewer 2.1 compatible format.
- one CORINE file and one PELCOM file per country.
- 16 regrouped classes.
- area percentage per land cover class.

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Part II. Related Research

This part contains contributions which cover various themes from the programme of the Working Group on Effects (WGE) under the LRTAP Convention. The views expressed in these papers are those of the authors and do not necessarily reflect the opinions of the editors.

The first paper, by Suutari et al. presents the results of a collaborative project between the International Institute for Applied Systems Analysis (IIASA) and the Coordination Center for Effects (CCE) to assess the propagation of error in the RAINS modelling chain from emissions to critical load exceedances. The United Kingdom financed the portion of the work performed at IIASA. The aim of the project is to understand and quantify the reliability of assessment of effect-based emission reduction alternatives. Deliverables from the project could include maps displaying the probability (confidence level) of achieving a particular level of ecosystem protection.

The second paper, by Jane Hall from the UK National Focal Centre, reports on a UK contribution-in-kind of the United Kingdom to the ICP Mapping, regarding the harmonisation of ecosystem definitions used in the critical loads work. She advocates the use of the European Nature Information System (EUNIS) habitat classification, and this could be followed up by a future "Call for Data" by the CCE.

The third paper, by Hall et al. summarises conclusions and recommendations of a UN/ECE expert workshop on "Chemical Criteria and Critical Limits", which was organised as a "contribution-in-kind" to the ICP Mapping by the UK NFC in March 2001. Several of the recommendations call for updates or additions to the Mapping Manual.

The fourth paper, by Van Hinsberg and Kros describes the calculation of critical loads for biodiversity for the Netherlands with the SMART2-MOVE model. The approach, properly adapted to local conditions, could be used by other countries to derive critical loads based on more than a single chemical criterion.

The fifth paper, by Reinds et al. uses the methodology developed by the ICP Mapping to calculate and map critical loads of lead and cadmium for European forest soils. The paper is intended as a contribution to the ongoing debate on the methodology for calculating critical loads for heavy metals.

The sixth paper, by Meili describes the mapping of atmospheric mercury pollution of boreal ecosystems in Sweden, with emphasis on the use of a dynamic modelling approach.

1. Uncertainty Analysis of Ecosystem Protection in the Framework of Integrated Assessment Modelling

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Introduction

Integrated assessment models (IAMs) have been used in recent international negotiations to provide quantitative information on emission reductions require to reach ecosystem protection targets, which have been derived from critical loads and levels. In the course of these negotiations many questions concerning the (un)certainty of the computations arose, such as: How good are the IAMs in estimating the percentage of protected ecosystems, i.e. how confident are we that the agreed emission reductions will actually meet the environmental targets? How much further must emissions be reduced in order to achieve a desired confidence level in the ecosystem protection? Should we concentrate on gathering more information to reduce the uncertainty in IAMs, thus avoiding costly emission reductions but achieving the same confidence level in ecosystem protection?

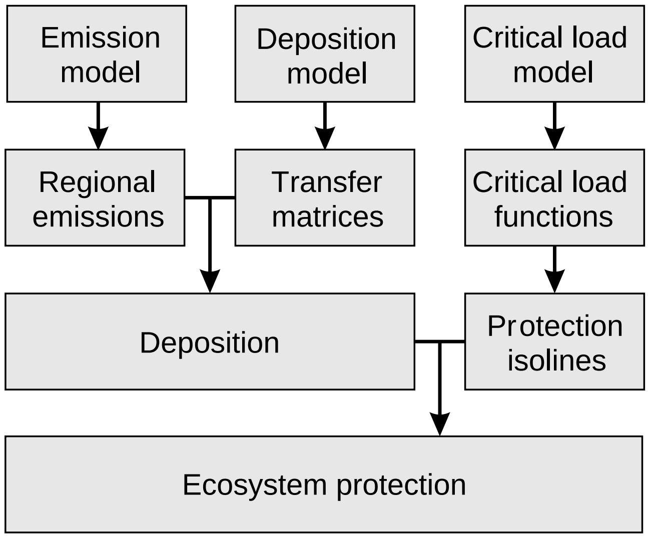

Figure 1. The integrated assessment model combines the results of three independent models (emissions, atmospheric transport and environmental vulnerability) to assess and quantify environmental effects.

IAMs, such as the RAINS model, are by necessity a simplified representation of reality. With respect to acidification, Fig. 1 shows the main compartments, their aggregated output and the link to the final result, the degree of ecosystem protection. The model's simplifications, aggregation (regionalisation) of data, measurement errors and lack of knowledge introduce uncertainties in the model results. In this paper we present a method for addressing the uncertainties involved in the critical loads and for assessing the uncertainty of the protection estimates, given the uncertainties in the critical loads and in the acidifying deposition. We also present some preliminary results of protection uncertainty using two different emission scenarios. Finally, we discuss future work needed to improve the uncertainty estimates of critical loads aggregated to a grid scale.

Methodology

We present a method to describe the uncertainty of ecosystem protection isolines from uncertain critical load input data, as well as a method to compute the uncertainty in ecosystem protection estimates for a given distribution of depositions. The shape and parameters of this distribution, i.e. the uncertainty in the deposition estimates, are *not* derived here. They depend on the combined uncertainties in emissions and atmospheric transport. The uncertainties in the emissions depend, in turn, on the uncertainties in the (economic and technological) activity parameters. Details of the methodology and data of a full IAM uncertainty analysis for the RAINS model are presented in Suutari et al. (2001). Syri et al. (2000) present an earlier analysis on a national level.

Protection isolines from uncertain critical load data: Within a single (EMEP) grid square (typically 150× 150 km^2) there are many ecosystems, the sensitivity of which is characterised by a critical load. Since both sulphur (S) and nitrogen (N) contribute to acidification, there is no single critical load of acidification, but an ecosystem's sensitivity is characterised by the so-called critical load function, a trapezoid-shaped function in the N-S plane (see Posch et al. 1995, Hettelingh et al. 1995, UBA 1996). Since a unique critical load of acidity cannot be defined, it is also impossible to define a unique exceedance. However, it is possible for every pair of deposition values (N_{dep}, S_{dep}) to determine the percentage (area) of ecosystems that is protected (or exceeded). Connecting the points of equal protection (say n%) yields the so-called protection isoline, a unique function of the angle ϕ=arctan(S_{dep}/N_{dep}) in the N-S plane. These protection isolines (or rather polygonal approximations thereof) are computed for a large number of percentage values for every EMEP grid square covering Europe, and this information is used in the IAMs. Detailed definitions and procedures for computing protection isolines can be found in Posch et al. (1997).

The uncertainty ranges of the protection isolines have to be derived from the uncertainties in the critical load functions, each of which is defined by the three quantities $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$. Correlations in the uncertainties of these quantities within a grid cell lead to correlations in the uncertainties between different protection isolines. It is virtually impossible to derive explicit formulae for the uncertainty of protection isolines from the uncertainty of the critical loads, because the calculations are non-linear, including division, sorting and substituting. Therefore, Monte Carlo simulation has been used to estimate the uncertainty parameters of each vertex (x_k,y_k), k=0,…, N, of the polygon describing, say, the n% protection isoline. These uncertainty parameters characterise a one-dimensional distribution along the ray through the origin with angle ϕ_k=arctan(y_k/x_k) (see Fig. 2a). Due to the lack of more detailed information we assume the same type of distribution along the rays, characterised by mean and standard deviation. For an arbitrary angle ϕ we determine the sector k (k=1,…, N), i.e. the angles ϕ_{k-1} and ϕ_k with $\phi_{k-1} \le \phi < \phi_k$, and interpolate (linearly in ϕ) mean and standard deviation, denoting them by $m_k(\phi)$ and $s_k(\phi)$, respectively. Integrating along a ray with respect to the distance from the origin and subtracting from 1 yields for a deposition pair (N_{dep},S_{dep}) lying on that ray the probability to protect n percent of the ecosystems within the grid square (see Fig. 2b). Viewed as a function in f, these cumulative distribution functions form a protection probability surface in the N-S plane for the n% protection isoline (see Fig. 2c). The surface defines the probability P_n to protect n percent of the ecosystems for any given nitrogen and sulphur deposition; in mathematical terms:

$$P_n(N_{dep},S_{dep}) = 1 - G(N_{dep},S_{dep},m_k(\phi),s_k(\phi)) \qquad (1)$$

where:

k = index of sector (of the n% protection isoline) in which (N_{dep},S_{dep}) lies; k=1(1)N

ϕ = arctan(S_{dep}/N_{dep})

$G(.,.,\mu,\sigma)$ = cumulative uncertainty distribution function (mean μ, standard deviation σ)

Ecosystem protection uncertainty:

The main purpose to derive isolines of ecosystem protection is to (quickly) estimate the percentage of ecosystems protected from acidification in a grid square for a given deposition. If the N and S deposition is known precisely, i.e. they have no uncertainties associated with them, Eq. 1 can be directly used to calculate the probability to protect a chosen percentage of ecosystems within the grid square. When there are uncertainties in the deposition as well, the protection probability is obtained by summing the protection probabilities weighted by the probability of the deposition (see Fig. 3). Thus probability to protect n percent of the ecosystems for given depositions is calculated as:

$$P_n | f_D = \int_0^\infty \int_0^\infty P_n(D_S,D_N)\, f_D(D_S,D_N) \;\; \mathrm{d}D_S \mathrm{d}D_N \qquad (2)$$

where f_D is the bivariate distribution of uncertain sulphur and nitrogen deposition.

Finally, to get the overall uncertainty in protection for a given deposition to a grid square one has to calculate the protection probability for all protection percentiles, i.e. evaluate Eq. 2 for all n with $0 \le n \le 100$. In this way one obtains the cumulative distribution function of ecosystem protection (see Fig. 4). For a given deposition scenario this cumulative distribution function allows one to (e.g.) read the confidence levels for ecosystem protection, i.e. the probability of protecting at least n percent of the ecosystems in the grid.

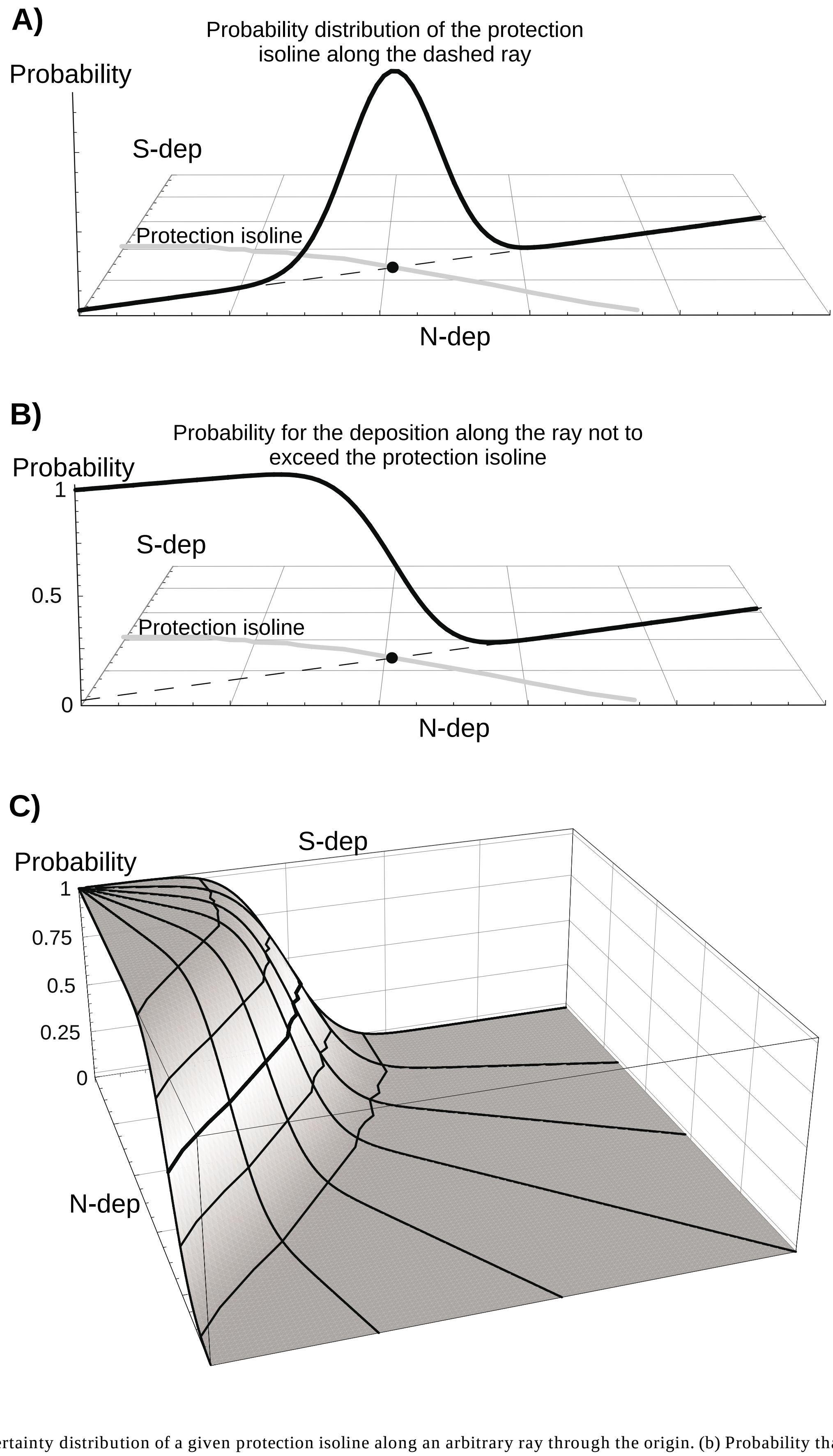

Figure 2. (a) Uncertainty distribution of a given protection isoline along an arbitrary ray through the origin. (b) Probability that the deposition along the ray does protect (= does not exceed) the given percentage of ecosystem area. (c) Protection probability surface for a given protection isoline.

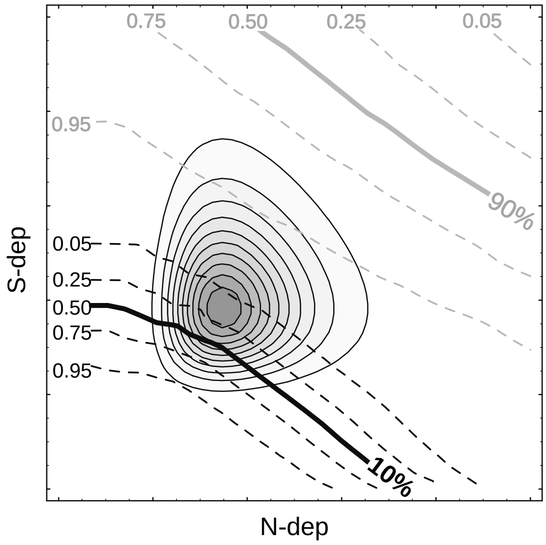

Figure 3. Contours of equal probability of a bivariate log-normal deposition distribution overlaid with the probability contours of two protection isolines (10% and 90% protection).

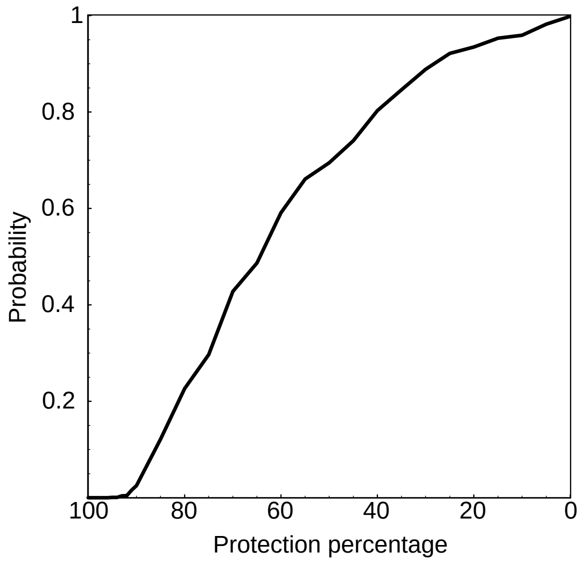

Figure 4. Example of a cumulative distribution of ecosystem protection percentage for a given N and S deposition distribution.

Illustrative example

In this section we present an example of an uncertainty analysis of IAM. The results are to be considered illustrative only since the uncertainty data, especially those for critical loads, are preliminary. Also, some methodological improvements are to be expected in the near future (see "Discussion and conclusions" below). A more detailed analysis, especially of the methods and data leading to the uncertainty estimates of N and S depositions, is presented in Suutari et al. (2001).

The uncertainty ranges of the protection isolines are derived from the uncertainties in the critical load functions defined by the quantities $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$. The uncertainties in these quantities have either to be assessed "directly" (e.g. for empirical critical loads) or derived from the uncertainties in the variables used to calculate them, e.g. the variables and parameters needed in the simple mass balance (SMB) model. Here we use the uncertainty ranges for the SMB variables provided by the German NFC for calculating acidity critical loads (see Part III). We assumed that the uncertainties in the SMB parameters were uniformly distributed around their median values and uncorrelated, with the exception of base cation and nitrogen uptake, for which full correlation was assumed. Using Monte Carlo simulation, we estimated a coefficient of variation (CV) of 0.22 for $CL_{max}(S)$ and 0.12 for $CL_{max}(N)$.

By making the (rather crude) assumptions that (a) the uncertainties of the critical loads given above are representative for the whole of Europe and (b) the uncertainties of the critical loads are fully correlated, these coefficients of variation were used to describe the uncertainty of the protection isolines (with linear interpolation between CV = 0.12 for ϕ=0 and CV = 0.22 for ϕ=90°). This strong correlation assumption results in higher uncertainty estimates for the isolines and thus represents a "worst case" (see "Discussion and conclusions" below).

To compute the uncertainty in the ecosystem protection we need also the uncertainty in the N and S deposition. Since the protection uncertainty is highly dependent both on the magnitude of the expected deposition and the deposition uncertainty, we analyse the protection uncertainty by using two RAINS emission scenarios: the N and S emissions for the years 1990 and 2010, assuming for the latter that the emission reductions of the Gothenburg Protocol are fully implemented. The emission uncertainties and correlations were calculated using preliminary uncertainty estimates for activities, emission per activity unit and removal efficiencies of different abatement technologies for each economic sector and fuel category. The uncertainty in the dispersion of sulphur and nitrogen compounds was estimated from the uncertainty and correlation of the mean transfer coefficients derived from the annual transfer coefficients of the EMEP lagrangian dispersion model for the years 1985–1995. This represents only the uncertainty due to inter-annual variability in the meteorology, but estimates of the uncertainties inherent in the dispersion model are not yet available.

Deposition uncertainties are presented in Fig. 5. Perhaps counter-intuitively, there is more spatial variation of the relative uncertainty in the sulphur deposition than in the nitrogen deposition. There seems to be very little difference between the relative uncertainties of the 1990 and 2010 deposition. The correlation between sulphur and nitrogen depositions was also taken into account, but is not shown here. For this and other details on the deposition data and calculations see Suutari et al. (2001).

Uncertainties in ecosystem protection are computed from the uncertainties of the deposition and the uncertainties of protection isolines by using Eq. 2 for every protection isoline in every grid square covering Europe. The results of this computationally demanding exercise are displayed in Fig. 6 for the two emission scenarios. Fig. 6e,f illustrate the range of uncertainty, expressed as the difference in the percent of ecosystems protected with 5% confidence level (not shown) and 95% confidence level (Fig. 6c,d). This range is highest in grid squares in which deposition levels are close to critical loads, and especially if the critical load values in a grid cell are very similar. This can be seen for 1990 in France and Russia, where the uncertainty ranges are largest because depositions are close to critical loads, and critical load values are few and / or have steep gradients (little dispersion).

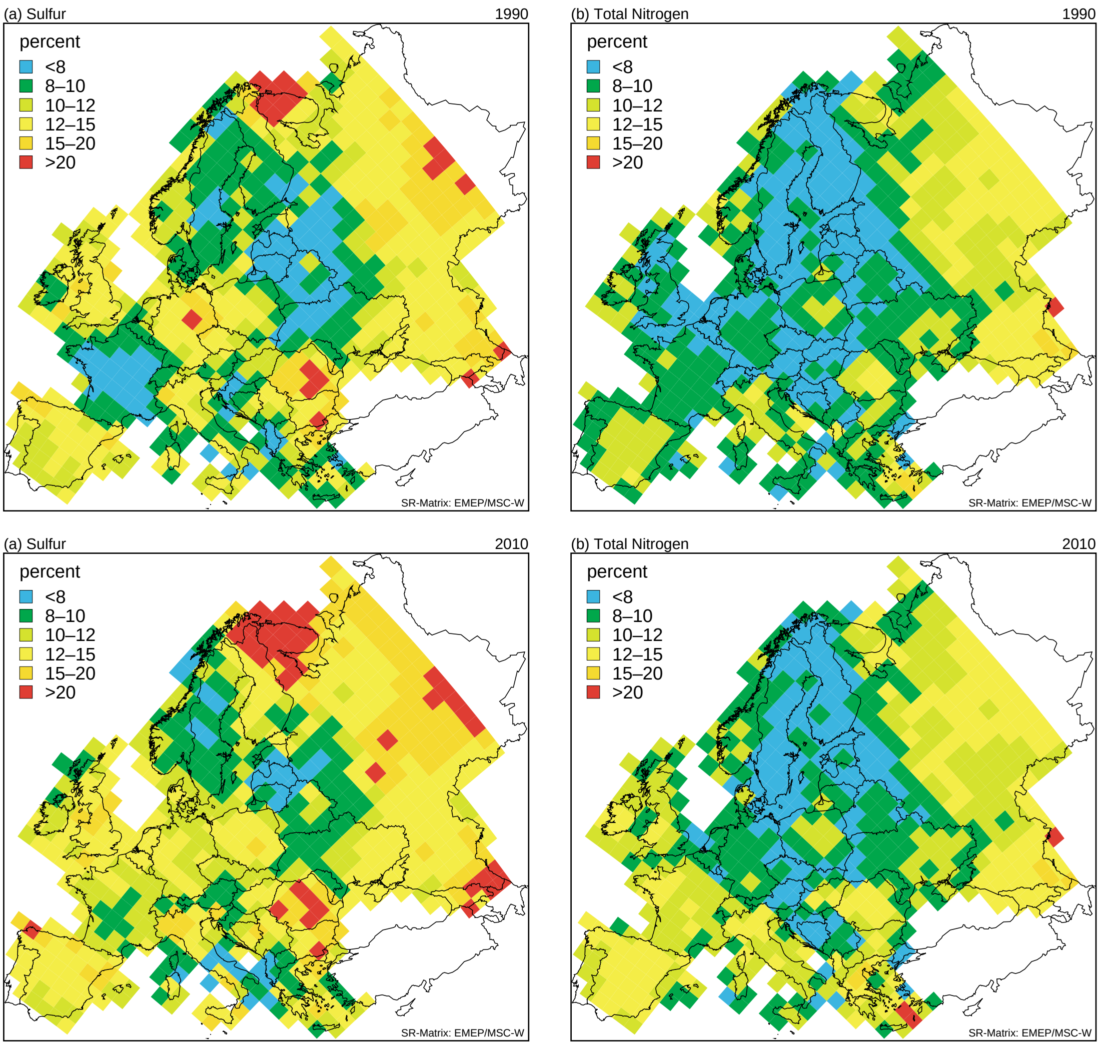

Figure 5. Ranges of relative uncertainties (=confidence interval divided by expected value) of sulphur (a,c) and total nitrogen (b,d) deposition for 1990 (a,b) and 2010 Gothenburg (c,d) emissions.

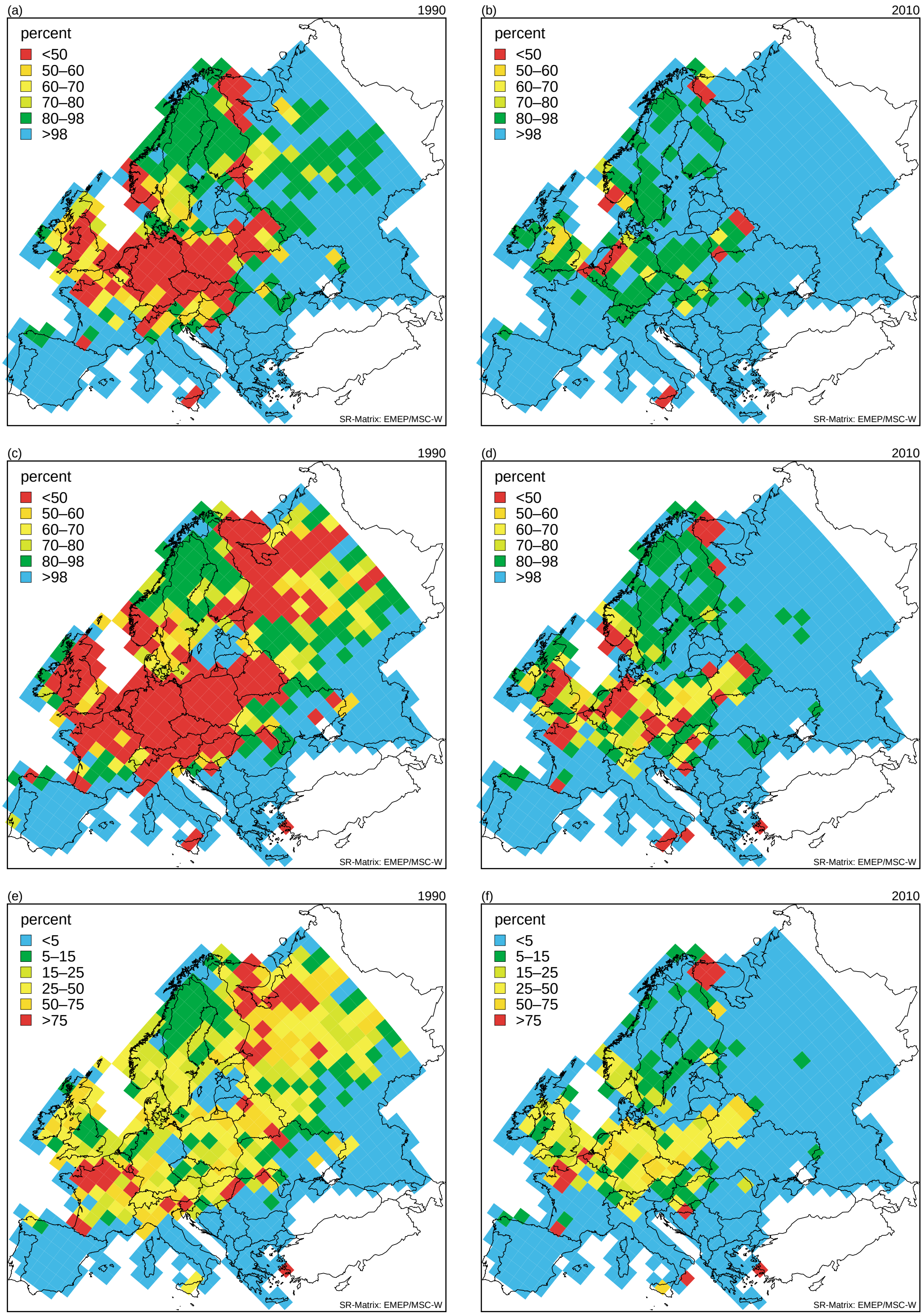

Figure 6. Probabilities (confidence levels) of ecosystem protection percentages (acidity) due to N and S depositions in 1990 (a,c) and 2010 (b,d) (see Fig. 5). Maps (a) and (b) show the deterministic case, i.e. ecosystems are protected with 50% probability (confidence), whereas maps (c) and (d) show the ecosystem protection percentages for the 95% confidence level. Finally, maps (e) and (f) show the uncertainty *ranges* in protection percentages, i.e. the difference between the ecosystem protection percentage at 5% (not shown) and 95% confidence level (see (b) and (d)).

Discussion and conclusions

As shown by this paper and the cited literature, a method to analyse the uncertainties in the protection of ecosystems due to the exceedance of critical loads on a European scale is available. To get the most out of such an analysis, good-quality data characterising the uncertainties is needed, i.e. the analysis should not be questioned due to the “uncertainties in the uncertainties”. An uncertainty analysis as described in this paper does not only provide information on the confidence levels which can be assigned to IAM results, but also indicates where we can most improve the accuracy and precision of the IAM.

There are several areas in which improvements to the methodology are needed and desired, and below we mention two points with respect to deposition uncertainties and discuss at some length the propagation of uncertainties from critical load (functions) to protection isolines.

Concerning estimation of uncertainties in the depositions, consideration of the following two points should lead to an improvement:

- Current deposition estimates represent averages over all land cover types. However, dry deposition is greatly affected by the land cover, and therefore the ecosystems considered (e.g. forests) may receive significantly different (larger) amounts of deposition than the grid square on average. It is thus desirable to obtain separate transfer coefficients for each major land cover type in order to avoid a bias in the ecosystem protection estimates.
- Another error is introduced by the different spatial resolution of deposition and critical load data. Within an EMEP150 grid there can be (and most likely is) a large variation in the actual deposition. If the deposition pattern within the grid were known, we could correct the isoline estimation by shifting the origin for each critical load function by the difference between the grid mean deposition and the deposition at the ecosystem location.

If we do not know the deposition pattern within the grid and / or the ecosystem locations, we have to take into account the in-grid variability of the deposition as an additional source of uncertainty. To do that we need an estimate of the deposition distribution within the grid area. In the Monte Carlo simulations of the isoline uncertainties we could then vary the origin of each critical load function independently according to the in-grid deposition variation.

In the above example the uncertainty in the protection isolines was assumed to be identical to the uncertainty in the critical load function. This is a gross oversimplification, and in the following we shall argue why the properly derived uncertainty in the isolines will in general be smaller. We illustrate the principles for the one-dimensional case.

In Fig. 7a an example of a the cumulative distribution (CDF) of critical loads (CLs; numbers, not functions!) within a grid square is shown. Also shown are their individual uncertainty ranges (± one standard deviation). We obtain the uncertainty range of the CDF by randomly drawing a value from each CL range, sorting them and thus create a new CDF. This is repeated many (several thousand) times. From the

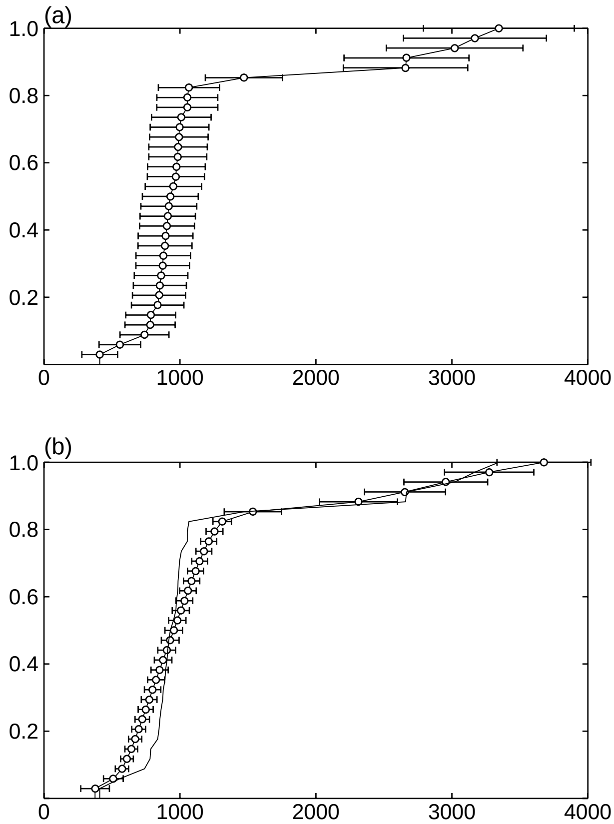

Figure 7. (a) Examples of critical load (CL) values (white circles) displayed as a cumulative distribution function (CDF). The horizontal interval at every CL value indicates its uncertainty range. The thin line connecting the CL values is a guide for the eye only (and the usual way to plot a CDF). (b) CDF of the mean values of the smallest, 2nd smallest, …, largest values of every realisation of randomly and independently selected values from (a) together with their (generally smaller) uncertainty ranges. Also shown is the CDF of (a) as thin line.

many random CDFs the mean and range (standard deviation) is calculated for every value (smallest, second smallest, …, largest) and these define the uncertainty range of the CDF. Fig. 7b was created in that way from the data in Fig. 7a, assuming that the values are independently and uniformly distributed around their respective mean (± 25%). As can be seen, for most parts of the CDF the uncertainties are reduced considerably by this aggregation process, a phenomenon also observed in Barkman (1998).

One has to bear in mind that the data in Fig. 7b do *not* represent individual CLs, but rather the means and uncertainty ranges of the different percentiles of the CDF in Fig. 7a. Any in-grid spatial information is lost, but this does not matter for our application, since we compare the CDF to a single (average) deposition value for the grid. In the example shown in Fig. 7 we assumed statistical independence between the different CLs. With an increasing correlation, the uncertainty bands would become wider, and complete correlation would leave the uncertainty ranges unchanged (i.e. Fig. 7b would be identical to Fig. 7a). The reduction of the uncertainty ranges depends also the degree of overlap between the individual uncertainty ranges. If there were no overlap, there would be no narrowing of the uncertainty band, since the order of the randomly varied CLs would never change (as is the case for some points of the CDF in Fig. 7).

This same reduction in the uncertainty ranges is, of course, also true when dealing with critical load functions and ecosystem protection isolines. The protection isolines are nothing more than the two-dimensional percentiles of the "CDF" of critical load functions. Fig. 8 is example of a set of critical load functions and five resulting protection isolines with their simulated uncertainty bands, assuming a CV=25% for $CL_{max}(S)$, 10% for $CL_{min}(N)$ and 15% for $CL_{max}(N)$. As can be seen these bands are narrower in the 'middle', where the critical load function are closest together.

The method of computing uncertainties of protection isolines just described needs to be applied to the whole of Europe. However, to compute the uncertainty bands for a sufficient number of protection isolines and to evaluate the protection probabilities via Eq. 2 for a single grid square requires already considerable computing resources. Thus, routine applications for Europe demand some further methodological development. But most importantly, reliable grid-specific uncertainty estimates for the

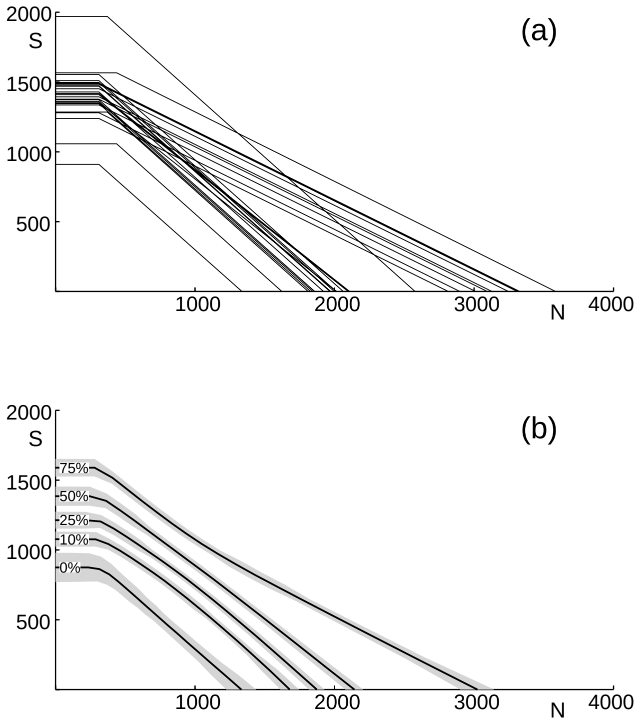

Figure 8. (a) Critical load functions defined by $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$. (b) Some resulting ecosystem protection isolines (0, 10, 25, 50, 75%) and their uncertainty bands, obtained by Monte Carlo simulation.

critical loads are required from every country to make such analyses useful for providing confidence levels that can be used in deciding on future emission reductions.

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2. Harmonisation of Ecosystem Definitions

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Introduction

This study was carried out by the UK National Focal Centre (NFC) as one of its "contribution-in-kind" activities for the International Cooperative Programme (ICP) on Mapping. Countries calculate critical loads for a wide range of ecosystem types that are identified from a number of different data types (e.g. land cover, land use, national inventories or atlases, etc.). However, no information had been collated on the methods and data used to define the ecosystems, so one could only assume that if several countries gave the same name (e.g. heathland) to an ecosystem, that these were similar ecosystems, when this may not have been the case. The UK NFC therefore proposed a study to address this issue.

Aims of the study

The main aims of the study were to:

- Collate information on the definitions, data and methods used to describe and identify those ecosystems selected by individual countries for critical loads work.
- Review and harmonise these ecosystem definitions.
- Classify the ecosystems for future work under the Convention on Long-range Transboundary Air Pollution.
- Report the findings of the work to the CCE and the ICP Mapping.

Approach and methods used

The UK NFC contacted each of the 24 NFCs that had provided critical loads data to the CCE. The following information was requested for each ecosystem for which critical loads had been calculated: ecosystem name, description, land use or land cover classification on which the ecosystem distribution was based (including the classes used), key indicator species representing the ecosystem, and any other information used to define the ecosystem. Replies were received from 14 countries.

Working with the ecosystem names alone, it was possible to aggregate the ecosystems into eight broad classes: forests, coniferous forests, deciduous forests, natural areas, grasslands, heathlands, wetlands and waters (Fig. 1). However, this is a simple approach and does not consider the additional information collated. The types of data used to define the ecosystems can vary from one country to another, some may use land cover, others national inventories or combinations of different data types. This can also vary by ecosystem type. Table 1 lists the various classifications and databases used by NFCs to define their ecosystems.

Table 1. Classifications and databases used by NFCs to classify ecosystems.

- Aerial photography / field observations
- CORINE biotopes classification of Palaearctic habitats
- CORINE Land Cover (levels 3 or 4)
- Lake registers
- PHARE Natural Resources / CORINE Information System
- National land use or land cover maps
- National forestry inventories
- National soil maps
- National species data (indicator species or distributions)
- National survey data
- National vegetation classifications / atlases
- Topographic maps / digital terrain models

Two different approaches were then considered to harmonise and re-classify the ecosystems: (i) by ecosystem type and key indicator species, and (ii) by comparison with other classification schemes. After investigating method (i) it was rejected on the basis that it was:

- subjective as to who did the classification.
- not easy to define new class descriptions.
- not easy to update or extend to include new ecosystems for other countries.
- not related to other classification.
- yet another classification.

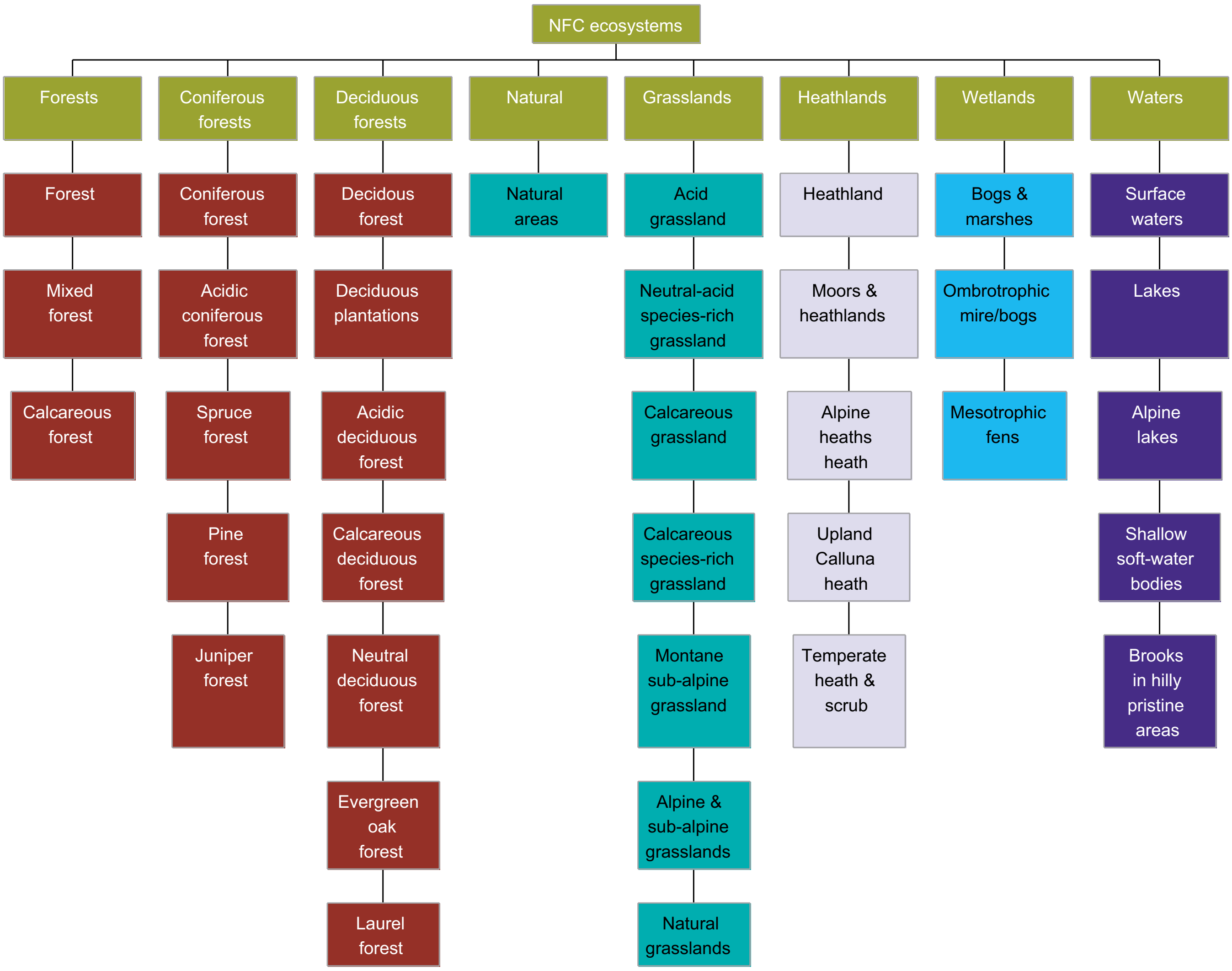

Figure 1. Ecosystem categories defined for the ecosystem names provided by National Focal Centres.

Therefore, method (ii) was explored. The FAO / UNEP Land Cover Classification System (Di Gregorio and Jansen 2000) and the program for the inter-comparison of land classifications prepared for the European Topic Centre on Land Cover (Wyatt et al. 1998) were rejected because they focus on the use of land cover data only, and as seen in Table 1, the sources of information that countries use to define their ecosystems are more complex. The CORINE Biotopes habitat classification was also rejected because of its complexity of classes for different regions across Europe; it was not easy to assign the most appropriate classes to ecosystems.

A simpler classification framework was required. This was found in the form of EUNIS – the European Nature Information System (Davies and Moss 1999). This classification had been developed for the European Environment Agency (EEA), European Topic Centre for Nature Conservation, as a pan-European tool of the EEA. It is a successor to the CORINE habitat classification and uses a common language and a common framework with links to other classifications (e.g. the CORINE Palaearctic classification). The classes can also be cross-matched to those of the CORINE Land Cover Map and the habitats listed in Annex 1 of the EU Habitats Directive. EUNIS is a hierarchical classification with clear criteria for each division; it is applicable at different levels of complexity and is easy for the non-expert to use and apply. The 10 major habitat classes are given in Table 2.

It is not possible to present the full hierarchical structure of the classification here. The EUNIS web site contains a list and descriptions of all the habitat types; the criteria (including criteria diagrams) for the identification of habitats, a very useful glossary of terms, and downloadable files and reports that describe in detail the EUNIS classification and how to use it. The web address for the site is: mrw.wallonie.be / dgrne / sibw / EUNIS / home.html.

Table 2. European Nature Information System (EUNIS) habitat classifications.

Class	Name
A	Marine habitats
B	Coastal habitats
C	Inland surface water habitats
D	Mire, bog and fen habitats
E	Grassland and tall forb habitats
F	Heathland, scrub and tundra habitats
G	Woodland and forest habitats and other wooded land
H	Inland unvegetated or sparsely vegetated habitats
I	Regularly or recently cultivated agricultural, horticultural and domestic habitats
J	Constructed, industrial and other artificial habitats

Results

Using the EUNIS system, the UK NFC were able to classify the ecosystems from the 14 countries into five major habitat classes and, where sufficient information on the ecosystems had been provided, into a further 15 Level 2 categories. In some cases Level 3 categories could be assigned to ecosystems. The full classification of ecosystems by country is not included in this report, but Fig. 2 shows the Levels 1 and 2 classes that could be used. It should be noted that in some cases, ecosystems were assigned to classes that one would not have immediately associated them with, if trying to classify by ecosystem name alone. For example, mesotrophic fens, depending on the actual full description provided by countries, may need to be assigned the "mesic-grassland" class (E2), rather than "base-rich fens" (D4) class.

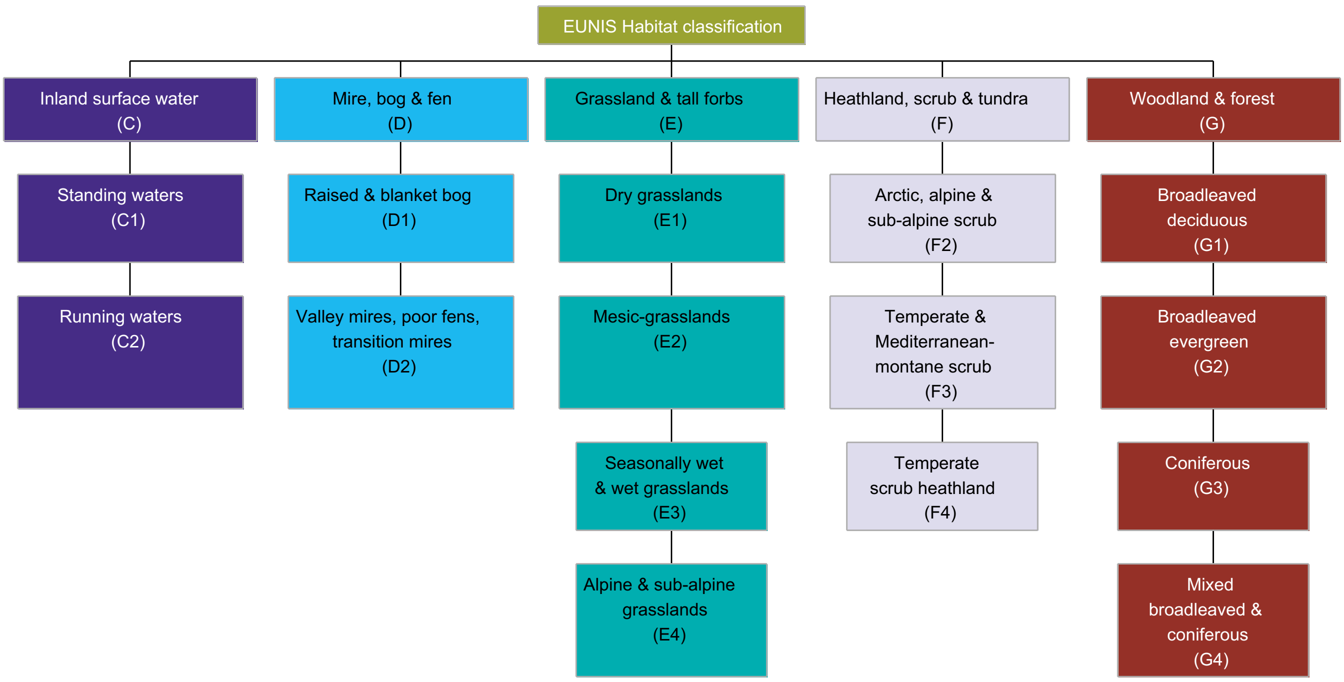

Figure 2. Level 1 and Level 2 EUNIS habitat classes that can be used to describe the ecosystems for which National Focal Centres provide critical loads data.

Conclusions and recommendations

The UK NFC concluded that the EUNIS habitat classification could be used by NFCs as a common framework for recording and classifying European ecosystems for future critical loads work. This was proposed to the Task Force meeting of the ICP Mapping in May 2001 and it was agreed that NFCs should try using the EUNIS classification to assign habitat codes to their ecosystems. Other advantages of using EUNIS are:

1. It is quick and easy to use, one doesn't have to be an expert.
2. The EUNIS classes can be linked to the CORINE Palaearctic classification and cross-matched to classes of the CORINE Land Cover Map.
3. Using it as a framework for classifying ecosystems for critical loads work could provide "added value" to the European critical loads database, because of its links to the habitats in Annex 1 of the EU habitats directive. This could enable the effects of critical loads exceedance to be examined for habitats of particular importance under the directive.

Acknowledgements

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3. UN/ECE Expert Workshop: Chemical Criteria and Critical Limits

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Introduction

The expert workshop on chemical criteria and critical limits was organised by the UK National Focal Centre (NFC) as one of its "contribution-in-kind" activities for the International Cooperative Programme on Mapping, as agreed by the Programme's Task Force meeting in 2000. It was held in York (UK) from 19-21 March 2001 and attended by 33 experts from eight countries.

Aims of the workshop

The key aims of the workshop were to:

- Examine the chemical criteria and critical limits currently used for acidification and eutrophication critical loads models (steady-state and dynamic);
- Consider new or alternative chemical criteria and critical limits;
- Consider what guidance is needed in their application; and
- Draw up conclusions and recommendations to be presented for consideration to the meeting of the Task Force on ICP Mapping (7-9 May 2001 in Bratislava).

Plenary session

The meeting began with a series of plenary presentations on the following topics to provide background information on the issues and promote discussion:

- The validity of the chemical criteria and limits for emission control in Europe, considering the questions: (i) do the criteria and limits provide a fair method of sharing the burden of emission control, and (ii) will they in fact lead to the desired outcomes? (Richard Skeffington, United Kingdom).
- Observations in Swiss forests on acidity effects (Sabine Braun, Switzerland).
- Nitrogen effects on forest ecosystems from the viewpoint of critical loads (Walter Flückiger, Switzerland).
- The relationship between Norway spruce status and soil water base cation:aluminium ratios (BC:Al) in the Czech Republic (Jakub Hruska, Czech Republic).
- The temporal behaviour of BC:Al in soil solution of forest soils (Rock Ouimet, Canada).
- The use of the sodium dominance index for predicting weathering rates and critical loads for soils and waters (Malcolm Cresser, United Kingdom).
- Chemical criterion for salmon: ANC, Al or pH? (Frode Kroglund, Norway).
- Critical ANC – some experiences with its application to Scottish freshwaters (Ron Harriman, United Kingdom).
- Base saturation as a critical limit – the link to dynamic models (Maximilian Posch, Netherlands).
- Limits for soils and freshwaters and the way forward with dynamic modelling (Harald Sverdrup, Sweden).

Discussion groups

Following the plenary presentations and discussions, three discussion groups were set up:

- Acidity – Terrestrial Ecosystems (chaired by Richard Skeffington, UK).
- Acidity – Freshwaters Ecosystems (chaired by Chris Curtis, UK).
- Eutrophication – Terrestrial Ecosystems (chaired by Mike Ashmore, UK).

The discussion groups were asked to consider the following questions:

Criteria and limits:

1. Do they represent a chemical dose to biological response?
2. Are they appropriate, or in need of revision?
3. Are there any new ones that should be considered?

Mapping Manual:
What are the implications of these discussions on criteria and limits, and are revisions to the Mapping Manual necessary?

The full discussions of the groups are not included here; a full report will be produced in 2001 and made available via the UK NFC web site (critloads.ceh.ac.uk). The summary conclusions and recommendations from the expert workshop are given below.

Conclusions and recommendations

The workshop participants agreed upon a number of conclusions and recommendations. These are listed below for each discussion group.

Group 1. Acidity – Terrestrial Ecosystems

This discussion group considered separately each criterion and related conclusions and recommendations for revisions to the Mapping Manual:

(a) Critical molar base cation to aluminium ([BC]:[Al]) ratio:
Conclusions and recommendations:

- The group agreed that it is time to update the review of underpinning scientific experimental data relating [BC]:[Al] to damage.

Revisions to the Mapping Manual:

- Report that the base case of [BC]:[Al] = 1 is ecosystem-specific (coniferous forest).
- Include a table of tree, moorland and grass species with suggested [BC]:[Al] ratios, together with a "reliability rating" for each.
- Tabulate the limitations and advantages of applying [BC]:[Al] on a horizon basis (such as in the PROFILE model) versus a "mixed-tank" single horizon (such as in the Simple Mass Balance equation).
- Report that further research is needed to look at the short-term temporal fluctuations in [BC]:[Al] ratios and how they may be related to adverse effects. For example, are the trees / plants responding to average or extreme [BC]:[Al] ratios? Dynamic modelling needs to consider this.

The gibbsite equilibrium constant (K_{gibb}) was also discussed at this point. It was concluded that the Mapping Manual should:

- Recommend the use of K_{gibb} as a default, with the value set according to the percentage organic matter in the soil as defined in the Mapping Manual.
- Include the option for calculating the critical aluminium concentration ($[Al^{3+}]_{crit}$) by methods other than K_{gibb}. For example, the use of models (e.g. the Windermere Humic Aqueous Model), or if there are good empirical field data to represent the relationship between H^+ and Al^{3+}, these should be used in preference to K_{gibb}.

(b) Critical pH ($[H^+]$):
Conclusions and recommendations:

- Countries should consider the use of critical pH or ΔpH (i.e. acceptable change in pH) as a criterion.
- Existing data on pH dose-response need to be compiled and synthesised and indicator species (i.e. species that may be adversely effected) need to be identified.
- pH should be related to heavy metal dissolution (link this area of work to heavy metal protocols).

Revisions to the Mapping Manual:

- Include a recommendations that critical pH is the most appropriate criterion for organic soils.
- Include a table of plant growth effects versus pH and soil fauna effects versus pH to help in the selection of appropriate limits.

(c) Critical molar base cation to hydrogen ([BC]:[H]) ratio:

- The group agreed that there is a need for a review and further investigation to improve the database on the [BC]:[H] dose-response to biological effects.

(d) Critical aluminium concentration ($[Al^{3+}]$):

- No actions were identified.

(e) Aluminium weathering greater than aluminium leaching ($Al_w \geq Al_{le}$):

- No actions were identified.

(f) *Percentage base saturation (%BS)*:

Conclusions and recommendations:

- The percentage base saturation was introduced as a potential new criterion; it was also considered to be a good criterion for dynamic modelling.
- There is a need to promulgate the relevant equations for use in critical load calculations.
- A protocol has to be agreed on the measurement of %BS, perhaps involving ICP Forests.
- Research is needed to set appropriate %BS limits and to link them to other criteria.

Revisions to the Mapping Manual:

- %BS should be included as a potential new criterion for linking steady-state models to dynamic models.

Group 2. Acidity – Freshwater Ecosystems

Conclusions and recommendations:

- There are relationships for pH and Al with biological response, but the relationship with ANC is best.
- There is a need to define dose-response functions on a regional basis to account for regional differences in, for example, organic acidity or strain-dependent sensitivity of species.
- There is a need to define separate dose-response functions for lakes and streams to account for differences in episodic acidification.
- More work is needed on the definition of chemical targets for biological recovery in dynamic modelling, given hysteresis in both chemical and biological recovery processes which could lead to changing dose-response (ANC / species) relationships.
- There is a need for a forum for data and information exchange between countries, preferably web-based. It was suggested that perhaps ICP Waters could be involved in this development since they already have a web site.

Revisions to the Mapping Manual:

- Recommend the use of ANC as a criterion for lakes and streams (although the limit values may be different).
- Recommend that for international comparisons, there is a need to standardise the indicator (brown trout) and the required probability of damage (using regional dose-response relationships).
- To standardise the definitions for ANC and labile aluminium.
- To include a comment about how countries should treat naturally acid lakes (i.e. ANC ~ 10 $\mu eq\ l^{-1}$) and whether in these situations zero critical loads are acceptable.
- To include a reference to the forum for data exchange, proposed in the conclusions above.

Confidence ratings for criteria used in the calculation of acidity critical loads:

The Acidity Terrestrial Group and the Acidity Freshwaters Group together drew up a list of confidence ratings for different acidity criteria (Table 1).

Table 1. List of confidence ratings for various acidity criteria.

Criterion	Biological Effect	Confidence Rating[1]
pH	–Soil fauna and microorganisms	***
	–Plant growth in organic soils	***
$[Al^{3+}]$	–Surface water and shallow groundwaters	*
[BC]:[Al]	–Tree stability (windthrow)	*
	–Inhibition of root growth	***
[BC]:[H]	–Plant growth in organic soils	*
$Al_w > Al_{le}$	–Soil stability	*
%BS	–Loss of nutrient capital	*
	–Physical stability for vegetation and soils	*
ANC	–Damage to fish species	***

1 Confidence ratings: * low, ** medium, *** high

Group 3. Eutrophication – Terrestrial Ecosystems

This discussion group aimed to review both the current effects criteria, taking into account the recommendations from the Copenhagen Critical Loads Conference (1999), and the empirical critical load values considering whether new evidence had become available since 1995. However, it should be noted that the group's assessment of new evidence was not comprehensive. The group examined the criteria and critical loads on an ecosystem basis.

(a) <u>*Criteria for forests*</u>*:*

- *Soil fauna:* There appeared to be a lack of data on soil fauna, although some data exist for Sweden. It was felt that these data on soil fauna need to be reviewed and considered as a potential indicator or criterion.
- *Pathogens/pests:* Some new data were presented on nitrogen-pathogen interactions and useful chemical criteria identified. The new data suggest that critical loads would need to be lowered. A review of the literature on its potential use as an indicator is needed to understand the interactions between fauna, insect attack and phenolics as driven by nitrogen.
- *Frost/drought:* These criteria need reviewing. However, frost damage was regarded as probably not important. Drought is believed to play a role in damage as a consequence of increased growth and higher water demand.
- *Root/shoot ratio:* Many studies demonstrate changes to this ratio. It was considered useful for pot experiments but would be difficult to apply in the field. The usefulness of the root / shoot ratio as a criterion was therefore questioned.
- *Nutrient imbalances:* There was concern about the use of this criterion with respect to the empirical critical loads approach. It was agreed that nutrient imbalances should be regarded in mass balance models, which may require development. A further review is required.
- *Nitrogen leaching:* A lot of good new data are available. This criterion could be applied to all forest types based on throughfall; a critical throughfall could be used as a useful indicator of effects.
- *Ground flora:* There was a lack of good new data to consider. The ranges of empirical critical loads given in the Mapping Manual to protect from changes in ground flora should be used more explicitly. The possibility of shifting to high / medium / low parts of the range to allow for altitude, base cation availability and temperature should be emphasised. The effects on bryophytes and lichens should probably be considered separately from those for other ground flora.

(b) <u>*Critical loads for wetlands*</u>*:*

- *Ombrotrophic bogs:* While it was agreed that the empirical critical load should remain in the range 5–10 kg N ha^{-1} yr^{-1}, there were new experimental data to support a change in the reliability score given in the Mapping Manual, from “quite reliable” (#) to “reliable” (##).
- *Mesotrophic fens:* There are no new data available at present, but experiments are underway, so a review of new data should be possible in 3 to 4 years.
- *Shallow soft-water bodies:* New modelling data support the existing empirical critical loads range (5–10 kg N ha^{-1} yr^{-1}) given in the Mapping Manual.

(c) <u>*Criteria and critical loads for heathlands*</u>*:*

- *Nitrogen leaching:* This criterion was not thought to be important for heathlands, but should be considered for peat systems. Clarification of some of the nomenclature in the Mapping Manual is needed, for example, the difference between the ecosystems named “upland Calluna heath” and “moor”.
- *Role of management:* The Mapping Manual needs to make the role of management in heathlands more explicit by including (model-derived) estimates of the impact of alternative management regimes on critical loads.
- *Arctic heathlands:* New experimental data for arctic heaths strongly suggest a change in the empirical critical load in the Mapping Manual from 5–15 kg N ha^{-1} yr^{-1} (reliability score: quite reliable (#)) to 5–10 kg N ha^{-1} yr^{-1} (reliability score: reliable (##)). However, a brief review of the data is required to support this proposed change.

(d) <u>*Critical loads for grasslands*</u>*:*

- *Dune grasslands:* It was suggested that dune grasslands be introduced as a new habitat in the Mapping Manual table of empirical critical loads. New data are available to support an empirical critical load in the range 10–20 kg N ha^{-1} yr^{-1} (reliability score: quite reliable (#)).
- *Calcareous grasslands:* New data support changing the empirical critical load in the Mapping Manual from 15–35 kg N ha^{-1} yr^{-1} (reliability score:

quite reliable (#)) to 15–25 kg N ha^{-1} yr^{-1} (reliability score: quite reliable (#)).

- *Montane-subalpine grasslands:* No new data are available at present, so the empirical critical load should not be changed from that given in the Mapping Manual.
- *Neutral-acid grasslands:* Although there is little data available, it was thought that the critical load for this grassland type could be lowered (15–25 kg N ha^{-1} yr^{-1}). However, there are data to support this new value at present.

(e) *General conclusions from the eutrophication-terrestrial ecosystems group:*

- There is a need for a workshop (autumn 2002 was suggested) to revise the empirical critical loads for nutrient nitrogen.
- There is a need to agree the mechanisms to produce the background review documents to prepare for the above workshop.
- There is a need to identify all sections of the Mapping Manual relating to eutrophication and the setting of empirical nutrient nitrogen critical loads, that need revision.
- Use should be made of the experiences of countries that have already applied empirical critical loads.
- Empirical critical loads are important, both for biodiversity and forest sustainability, and more countries should be encouraged to use them.

Summary conclusions from the workshop

- The criteria and limits being used do represent dose-response relationships.
- There are, however, criteria and limits that need revising.
- There are potential new criteria; for example, percentage base saturation.

In addition, the workshop:

- Proposed new empirical nutrient nitrogen critical loads.
- Proposed a workshop (preferably in autumn 2002) to formally review empirical nutrient nitrogen critical loads.

and identified:

- Areas requiring further research and / or review.
- Required revisions to the Mapping Manual.

4. Dynamic Modelling and the Calculation of Critical Loads for Biodiversity

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Introduction

The distribution of many Dutch plant and animal species has decreased over the past decades. Nearly 500 of the 1328 higher plant species have become endangered or even extinct (RIVM 2000). Acidification, eutrophication, desiccation, habitat fragmentation and habitat destruction are considered to be the most important threats.

In response to these threats, the Dutch Ministry of Agriculture, Nature Management and Fisheries has defined nature protection goals. An area of about 750,000 ha has been defined as an “Ecological Network”. Within this ecological network, biodiversity goals for terrestrial ecosystems have been specified using a system of 130 nature conversation target types. Each type is described in terms of target species, target area and desired management strategy. The Ministry is now working with provinces to develop a detailed map locating different nature conservation targets. Although the policy of the Ministry of Housing, Spatial Planning and Environment aims to provide the environmental conditions needed for nature protection, current methods for calculating critical loads cannot be directly applied to the set of nature conservation targets.

This paper describes a method for calculating critical loads for nitrogen and acid deposition based on critical limits for the protection of the plant species of the Dutch nature conservation targets. The critical limits are derived from species-response functions for nitrogen availability and soil acidity (Latour and Staritsky 1995), both important ecological key factors in determining habitat suitability for plant-species occurrence. Critical loads were calculated from these critical limits with the dynamic soil model SMART2.

Until more empirical data become available to enhance empirical critical loads, critical loads can be calculated by applying the present method to a wider range of ecosystems using standardised protection criteria and incorporating regional differences in hydrology and soil characteristics.

Method

The calculation of critical loads consists of two steps (Fig. 1):

1. Critical limits were derived for the different nature conservation targets. The critical limits were based on plant species-specific information on habitat preferences for nitrogen availability and soil pH. The critical limits for the nature targets were defined in terms of the highest tolerable nitrogen input and the lowest tolerable soil pH.
2. The dynamic soil model SMART2 was used to calculate the critical loads at which the above critical limits were not exceeded.

Both steps are described in more detail below.

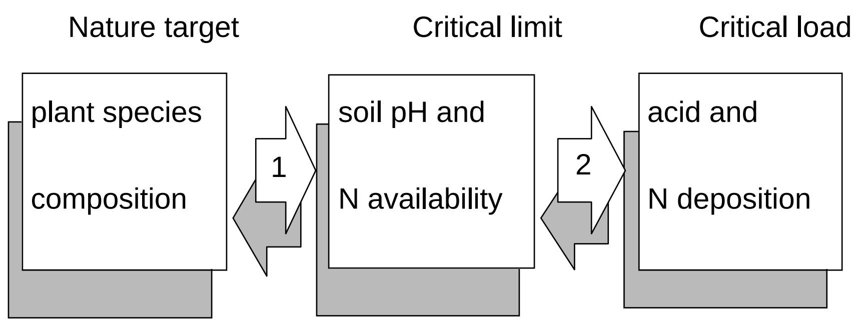

Figure 1. Method for calculating critical loads for nitrogen and acid deposition. In the first step, critical limits for different nature conservation targets are derived from plant-species-specific information on habitat preferences for nitrogen availability and soil pH. In the second step the dynamic soil model SMART2 is used to back-calculate the loads not exceeding the critical limits (= critical loads).

Step 1: The calculation of critical limits
The multiple-stress Model for Vegetation (MOVE) was used to calculate critical limits for the 130 given nature targets (step 1 in Fig. 1) according to the method described by Latour and Staritsky (1995).

MOVE is a species-oriented vegetation model (Latour and Reiling 1993), consisting of a set of regression functions relating the probability of plant species occurrence to abiotic site conditions. The model describes statistically the habitat preferences of plant species in terms of nitrogen availability, soil pH and groundwater level. The model is based on Gaussian logistic regression analyses (Jongman et al. 1987) of data on more than 100,000 vegetation releveés (plots) (Schaminee et al. 1989). The regression functions can be visualised as bell-shaped optimum curves, representing species occurrence along a single environmental gradient (Figure 2). Since MOVE focuses on more than one environmental factor, the real curves are, in fact, multidimensional and bell-shaped.

The species-response curves can be used to determine the range of suitable environments at the species level. Latour et al. (1994) described the use of the 10th and 90th percentiles as measures for risk assessment, analogous to the NOECs (No Observed Effect Concentrations). From this view the 10th and 90th percentiles correspond to sub-optimal environmental conditions with reduced occurrence probabilities due to “limitation” or “intoxication”, respectively. Between these percentiles the environmental conditions are suitable for plant species occurrence. Critical limits for the nature conservation target types could be calculated given the list of species to be protected within a target type and the ranges of suitable environments for species occurrence. These critical limits were calculated as the highest nitrogen availability and lowest soil pH at which 80 per cent of the total number of plant species of a nature target type might be present.

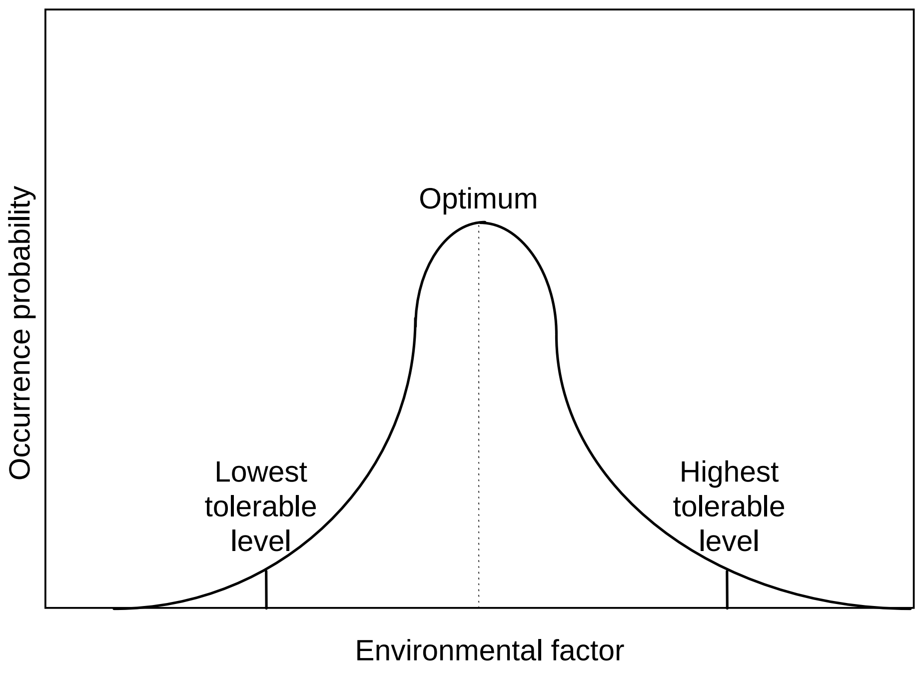

Figure 2. A hypothetical species-response function visualised as a bell-shaped optimum curve, representing species occurrence along a single environmental gradient. The range from the lowest to the highest tolerable level indicates the range of suitable environmental conditions.

Step 2: The calculation of critical loads

In order to calculate critical loads from the specified critical limits, we analysed the relationship between deposition level and abiotic site conditions in reverse (step 2 in Fig. 1). These analyses were made with the dynamic soil model SMART2.

SMART2 is a simple one-compartment soil acidification and nutrient cycling model (Fig. 3; also Kros et al. 1995), which includes all major hydrological and biogeochemical processes in vegetation, litter and mineral soil. SMART2, an extension of the SMART model described by De Vries et al. (1989), consists of a set of mass-balance equations describing the soil input-output relationships and a set of equations describing the rate-limited and equilibrium soil processes. Soil solution chemistry is assumed to be dependent on the net input of elements from the atmosphere (product of deposition and the filtering factor) and groundwater (seepage quality and quantity), canopy interactions (foliar uptake and exudation), geochemical interactions in the soil (CO_2 equilibrium, weathering of carbonates, silicates and / or Al hydroxides, sulphate sorption and cation exchange), and the complete nutrient cycle (litterfall, mineralisation, root uptake, nitrification and denitrification).

In order to relate critical limits to critical deposition levels, simplified relationships between deposition on the one hand, and nitrogen availability and soil pH on the other hand, were first derived from SMART2. These relationships were calculated with regression analyses on a data set of model simulations. The data set was built with a large number of simulation runs (3,841,600), using a wide range of model input data (Table 1). However, various variables were held constant to reduce the number of simulation runs, as will be discussed later. From this data set regression functions were derived for each unique combination of vegetation type, soil type, and groundwater table.

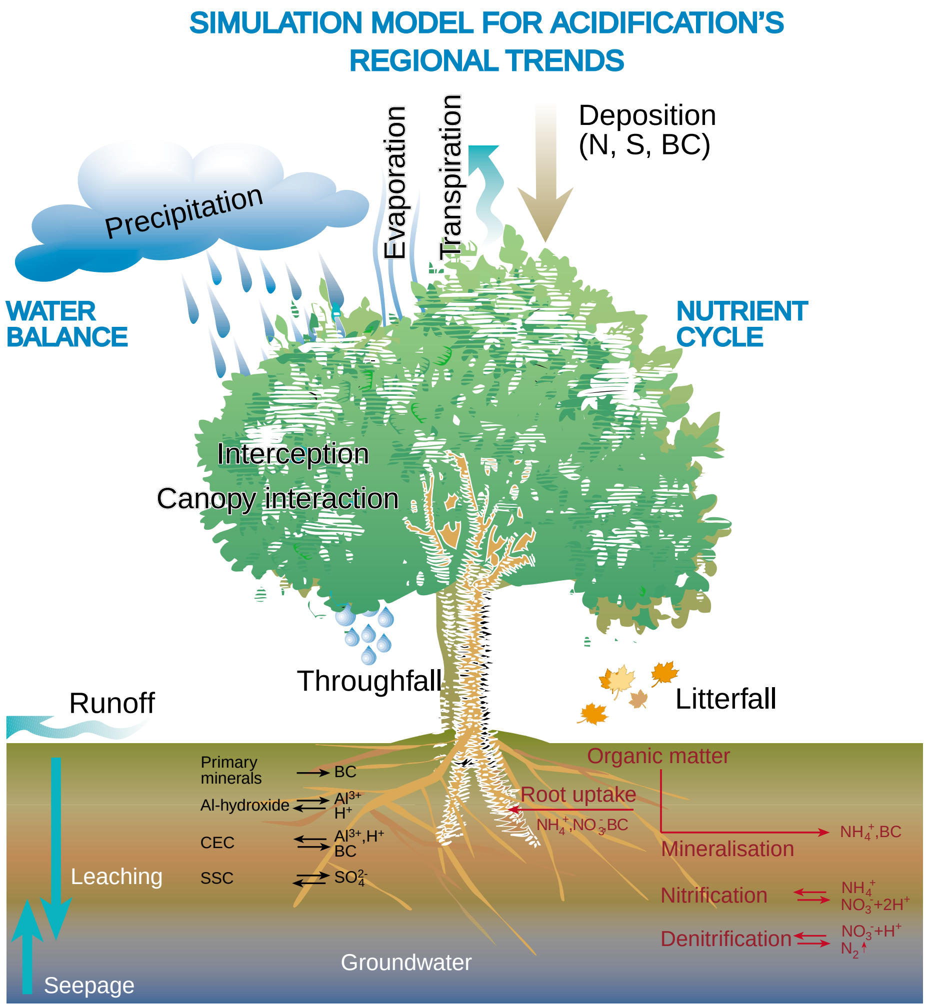

Figure 3. Processes included in the SMART2 model.

Table 1. SMART input parameters used to calculate the relationship between deposition levels, and nitrogen availability and soil pH.

Input	**Variable**	**Value**
Hydrological data	Seepage	Upward seepage flux of 0–1.5 mm d^{-1}
		Constant quality
	Groundwater level	5 water table classes
	Precipitation	Long-term national mean
Deposition	NO_x deposition	250–850 eq ha^{-1} yr^{-1}
	NH_x deposition	250–1880 eq ha^{-1} yr^{-1}
	SO_x deposition	400–1170 eq ha^{-1} yr^{-1}
	BC and Cl^- deposition	Mean level in 1984
Ecology	Soil type	Sand-poor (Carbic Podzols, Arenosols)
		Sand-rich (Gleyic Podzols, Gleysols)
		Sand-calcareous (Arenosols)
		Peat (Histosols)
		Loess (Luvisols)
		Clay (Fluvisols)
		Clay Calcareous (Fluvisols)
	Vegetation type	Spruce forest (60 yr)[1]
		Pine forest (70 yr)
		Deciduous forest (80 yr)
		Heathland (10 yr)
		Grassland (10 yr)
Simulation horizon		10 yr [2]
Number of combinations		3,841,600

1 Values in parentheses refer to the average age of the vegetation linked with the net growth and litterfall, which are important characteristics for the determination of the nitrogen availability.

2 Here, dynamic simulation has been performed with constant deposition (i.e. the critical load) for a period of 10 years. A 100-year time horizon would be better for a fair comparison with SMB results.

3 Parameterisation of soil and vegetation characteristics is described in Kros et al. (1995).

The derived equations (Eqs. 1 and 2) describe the model output (nitrogen availability and soil pH) as a simplified function of model input (deposition levels and seepage) for each vegetation-soil-groundwater table combination. From Eq. 1 it can seen that nitrogen availability is not determined solely by nitrogen deposition but also by seepage, SO_x deposition, soil characteristics and vegetation characteristics. In SMART2, seepage may influence nitrogen availability by input of nitrogen or by influencing, for example, mineralisation fluxes through effects on soil pH. Deposition of SO_x might also indirectly affect the nitrogen cycle by influencing soil pH.

$$\text{N availability} = f\,(\text{NO}_x \text{ deposition, NH}_x \text{ deposition, SO}_x \text{ deposition, seepage}) \qquad (1)$$

$$\text{Soil pH} = f\,(\text{NO}_x \text{ deposition, NH}_x \text{ deposition, SO}_x \text{ deposition, seepage}) \qquad (2)$$

The correlation coefficients were almost all higher than 0.95 and highly significant. Mean absolute differences between the original nitrogen availability and soil pH in the data set and the values which could be calculated with the regression functions, given the same model input, were also small (mean differences are 0.03 and 80 mol ha^{-1} yr^{-1} for soil pH and nitrogen availability, respectively). However, in some cases much larger differences were found.

Given the simplified relationships between deposition, and nutrient availability and soil pH, it was possible to back-calculate under which deposition levels the critical limits were not exceeded. Neither the critical limit for nitrogen availability nor soil pH was assumed to be exceeded under the critical deposition levels. However, in order to solve the above equations, assumptions had to be made for the ratios between the different deposition components. These ratios were set at current (1995) Dutch conditions. National maps on soil type, groundwater tables, vegetation type, seepage and a preliminary map of the nature targets were used as input to derive national critical load maps (see the Dutch NFC report in Part III). However, critical loads could not be calculated for all ecosystem types. Problems occurred mainly in heathlands, probably because SMART2 does not take management practices into account. In cases where no solution could be found, the lowest empirical critical loads from similar ecosystems (Bobbink et al. 1996) were applied.

Results and conclusions

SMART2-MOVE was used to calculate critical loads for biodiversity in terms of the desired plant composition of the Dutch nature conservation targets. The map of critical nitrogen deposition is shown in Figure 4; the critical load map of acid deposition is not shown since both maps show a great degree of resemblance. Results show a high spatial variability in critical loads. Nature targets are particularly vulnerable in the dunes and the central, southern and eastern parts of the Netherlands. Nutrient-poor adapted ecosystems dominate the sandy soils in these areas. The ecosystems on the clay soils in the western part of the Netherlands are less vulnerable to high nitrogen deposition levels than the nature targets of the sandy soils. Similar results have been found using the SMB method (Latour and Staritsky 1995).

Figure 5 illustrates the inverse cumulative frequency distribution of the calculated critical loads for nitrogen, $CL_{nut}(N)$, vs. the distributions of the critical loads for protection of forest growth and groundwater quality (see the Dutch NFC contribution in Part III of this report). From this, it can be concluded that the protection of plant species composition would require lower deposition levels than those necessary to protect groundwater quality and forest growth. Similar conclusions have been drawn in other studies. The calculated critical loads for nitrogen deposition are also in the same range as the empirical critical loads described by Bobbink et al. (1996). The calculated critical loads for acidity are somewhat higher than the critical loads for protection of groundwater quality, which is not surprising, as the latter allows no changes in soil pH.

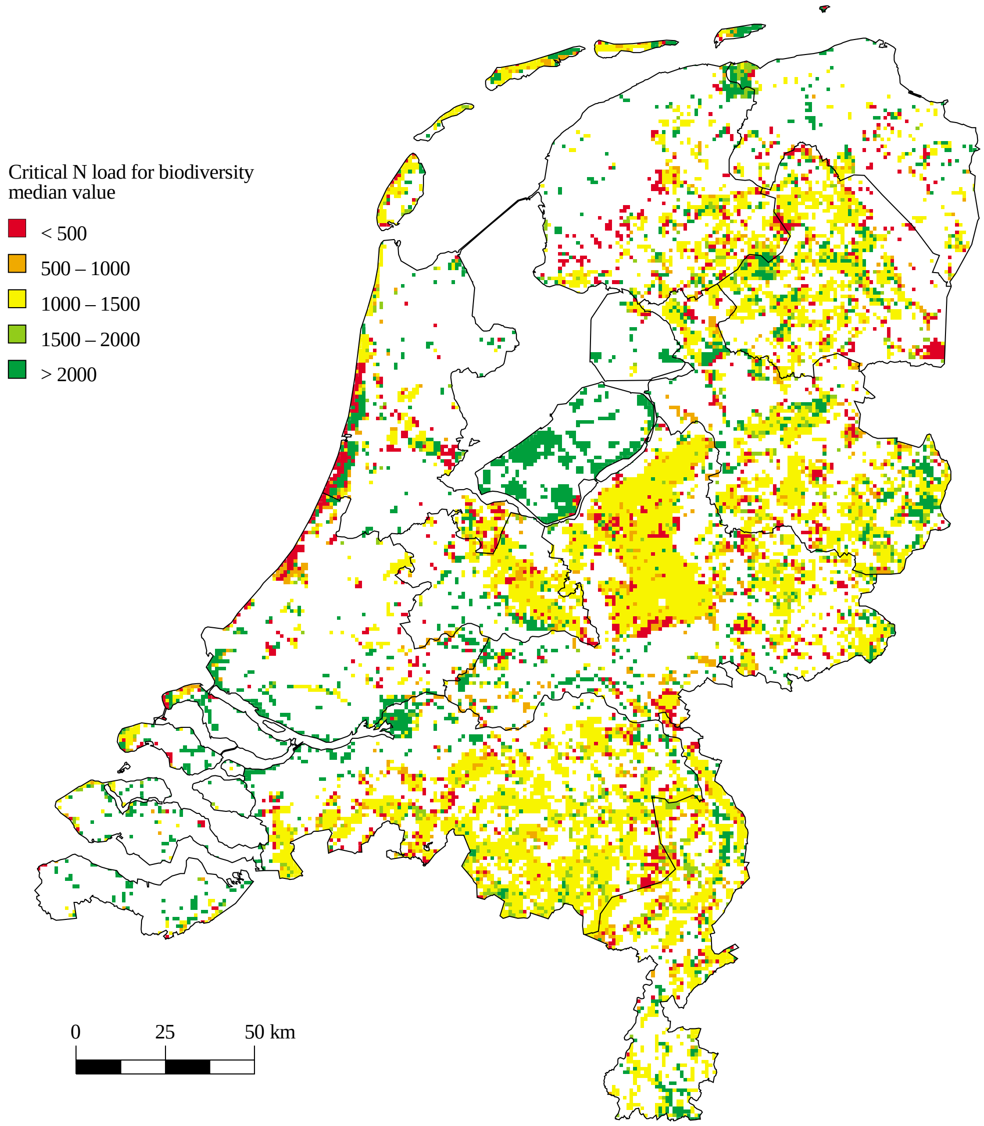

Figure 4. Geographic distribution of the critical nitrogen deposition for protection of plant species composition of Dutch nature conservation targets (median values in 1×1 km^2 grid cells).

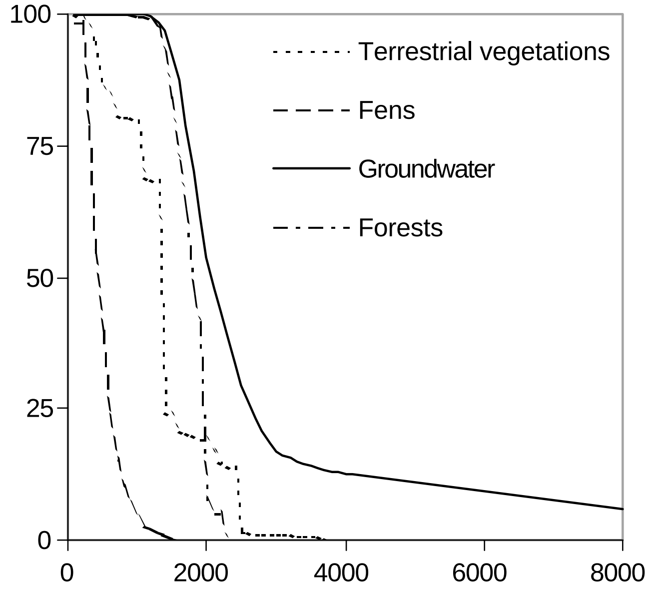

Figure 5. Percentage area at or below a given $CL_{nut}(N)$ value. The percentage protected area values are calculated as the inverse cumulative frequency distribution of the critical loads.

Discussion

The current method links information about species-specific habitat preferences directly to critical limits for ecosystems for which a clear national (and sometimes international) conservation policy has been defined. Earlier attempts to calculate critical loads for the protection of vegetation changes in Dutch forests (Latour and Staritsky 1995) used a standard critical limit of 100 mol ha^{-1} yr^{-1} NO_3 leaching. Below this level no negative consequences on vegetation were expected. However, the link between the leaching flux and changes in species composition, as well as the justification of the critical value itself, have been points of discussion. These issues are less relevant in the present method, as the critical limits refer directly to plant species occurrence and measurements within vegetation releveés.

Moreover, using the current method, we were able to calculate critical limits and critical loads for a wide range of different ecosystems. This was done by incorporating regional information about soil, hydrology and vegetation characteristics, while standardising the protection criteria. In this way the current method can be used to help determine critical loads that are not yet within the database of empirical critical loads.

By using an extensive database of vegetation releveés we were able to calculate significant regression functions for over 900 higher plant species and critical limits for a large set of different plant groups (e.g. nature targets). However, due to the absence of measured abiotic variables in most of the releveés, it was only possible to use indirect estimates of the abiotic conditions. These estimates were calculated from the mean Ellenberg indicator values[1] of plants within the releveés. An additional data set of vegetation releveés in which abiotic conditions were also measured was needed to link the averaged indicator values to abiotic conditions. With the help of this second data set we could determine a significant correlation between the respective estimates for moisture, acidity and nutrient availability with the water level in spring, soil pH and nitrogen availability (Ertsen et al. 1998). However, this extra step introduces additional errors in the calculation of critical limits and critical loads.

In order to link the critical limits to critical loads, relationships between deposition and nitrogen availability, and soil pH, were derived with SMART2. These relationships are simplifications of the dynamic model and the complex reality. Assumptions that had to be made regarding the ratios of deposition components, BC deposition and seepage quality might also influence the calculated critical loads. However, comparison between results of dynamic-effect modelling and calculated exceedance of critical loads showed no large differences in the percentages of protected area (Kros et al. in prep). It should also be noted that the model itself represents a simplification of a complex reality. Processes such as N fixation, NH_4^+ adsorption and uptake, immobilisation and reduction of SO_4^{2-} are not taken into account (Kros et al. 1995). Moreover, SMART is a single-layer soil model, ignoring vertical heterogeneity and seasonal variations. Justifications for the various assumptions and simplifications have been given in De Vries et al. (1989).

[1] Ellenberg attributed indicator values to numerous plant species of Central Europe, characterising the ecological conditions under which these species usually occur (Ellenberg et al. 1991). These indicator values can be used as indirect semi-quantitative estimates of the abiotic site conditions (e.g. Hill and Carey 1997, Ertsen et al. 1998).

Applicability of SMART to other European countries

The method described above might also be useful for calculating critical loads in other European countries. SMART has already been developed and used for modelling on a European scale. The species-response curves are based on a Dutch data set; however, the Ellenberg indicator system is used in many other European countries. It should be noted that the present method could be applied to vegetation classification systems other than the Dutch nature targets. Prerequisites for application of SMART2-MOVE in other countries are a database with national vegetation releveés, and a set of releveés with abiotic measurements for calibrating the Ellenberg indicator values.

An advantage of the current method is that the same models (SMART2-MOVE) might also be used for integrated effect assessment studies. Dynamic effect modelling could provide insight into: (i) the consequences of exceedance of critical loads, (ii) the relationships between acidification and other threats, and (iii) the dynamic aspects of environmental effects. Apart from predicting pH and N availability, SMART2 can also forecast changes in aluminium, base cations, nitrate and sulphate concentrations in the soil solution and solid-phase characteristics depicting the acidification status, i.e. carbonate content, base saturation and readily available Al content.

Moreover, SMART2-MOVE can be used to evaluate the multi-stress effects on flora caused by acidification, eutrophication, desiccation, habitat destruction and pollution by toxic substances (Alkemade et al. 1998). A multi-stress, rather than single-stress, approach is followed, since single-stress factors may have outcomes with confined reality when other stresses interfere. SMART2-MOVE can also be used to rank environmental threats for setting abatement priorities. Effect assessments on animal species and aquatic ecosystems are also being developed.

A disadvantage of the current method is that the calculation procedures differ from the simple mass balance methods. Differences between the simple mass balance models and the dynamic soil model SMART2 make it less straightforward to calculate parameters in SMB terms. The calculated critical load for nitrogen is comparable with the definition of the $CL_{nut}(N)$ (e.g. the critical load of nitrogen as a nutrient). However, the calculation methods differ since SMART2 also takes dynamic soil and vegetation processes into account. Moreover, feedback mechanisms are also an essential part of SMART2.

An important difference between SMART2 and SMB is that the nitrogen availability, which plays a dominant role in the calculation of critical loads for nitrogen, is also affected by changes in soil pH. Changes in soil pH influence the mineralisation flux, which in turn, affects the nitrogen availability. These changes in soil pH can therefore affect the tolerable deposition of nitrogen. Thus, critical acid and nitrogen deposition cannot be calculated separately due to the fact that MOVE is a multi-stress model and levels of tolerable nitrogen availability are dependent on the soil pH (and visa versa).

Future plans

We see the following as being important for the future:

1. The replacement of the SMART2 database output, and the regression functions derived from this database, by an optimisation module by which critical loads can be directly calculated from the dynamic model itself. The advantage of this new method would be that variation in BC deposition and seepage quality can then also be taken into account. Moreover, the different parameters underlying the calculations (nutrient uptake, etc.) can be stored for further analyses.
2. The replacement of the logistic growth curves in SMART2 by a more dynamic vegetation growth module, thus allowing also intensely managed ecosystems to be described.
3. Performing uncertainty analyses in the calculation of critical loads and determining the consequences of the different deposition component ratios.

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5. Critical Loads of Lead and Cadmium for European Forest Soils

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Abstract

Critical loads for lead and cadmium for European forest soils were calculated for situations where: (i) no further accumulation of heavy metals occurs (the "precautionary" or "stand-still" principle approach) and (ii) the concentration of heavy metals is below critical limits in the soil solution (the "effect-based" approach). In the first approach, critical (or rather, "acceptable") limits for the soil solution were derived from present metal concentrations, while the second approach uses the limited available information from the literature. Critical limits for the soil solution were related to the organic layer and the mineral topsoil (0–10 cm), where most microbial activity occurs. Results show that for lead, the critical load calculated using the stand-still principle is lower than that derived by the effect-based approach, whereas for cadmium both approaches give comparable results. This implies that if increases in present metal concentrations are not allowed, effects on soil organisms are, to a large extent, also avoided. It should be stressed that the results from this study are liable to considerable uncertainty. The main sources of uncertainty are the variability in the metal adsorption constant, the initial metal concentrations, the precipitation excess and the critical limit for metal concentrations in the soil solution.

1. Introduction

Concern about deposition of heavy metals (specifically cadmium and lead, but also copper and zinc) on terrestrial ecosystems such as forests is related mainly to impacts on soil organisms and to bio-accumulation in the organic layer (Bringmark and Bringmark 1995, Bringmark et al. 1998, Palmborg et al. 1998). One approach to successful international negotiations on the reduction of atmospheric deposition of pollutants is to determine the maximum atmospheric load that causes no or tolerable damage (the "critical load"). A major advantage of this method is that it can be used to optimise the protection of the environment for a given international investment in pollution control. A major difficulty is the quantification of the relationship between atmospheric emission, deposition and environmental effects. The recent (1998) protocol on heavy metals under the UN/ECE Convention on Long-range Transboundary Air Pollution (CLRTAP) is therefore still based on flat-rate reductions using best available abatement techniques, ignoring differences in susceptibility of receptors to the metal input. Following relevant decisions of the Executive Body of the Convention, the Working Group on Effects is, however, expected to assess environmental effects of heavy metals and to develop critical loads (i.e. long-term acceptable input levels) for these substances.

A study has been carried out to assess critical loads of heavy metals for forest ecosystems on a European scale (Reinds et al. 1995). This study used critical limits for the soil solid phase only, including target values set by the Dutch Ministry of Housing, Spatial Planning and the Environment. Most governments throughout Europe have defined such a set of values. The implicit assumption when setting these target values was that (ecotoxicological) effects are due to metal accumulation in the soil.

One problem with critical metal concentrations, however, is that the official target values for mineral soil in all countries are not ecotoxicologically based. This is due to the fact that the maximum permissible concentrations in laboratory toxicity tests are often lower than background concentrations, which would imply a target value below the background concentration. This apparent inconsistency is due to differences in metal availability in these toxicity tests and in the field (e.g. Klepper and Van de Meent 1997). At present, this inconsistency has not yet been resolved.

In the Netherlands it was thus decided to set the long-term critical limit (target value) equal to the 90 percentile value of the above-mentioned background concentrations in relatively unpolluted areas. A more fundamental problem when using critical limits for the soil is that in most cases, toxic effects on (e.g.)

micro-organisms and soil fauna are mainly due to elevated bioavailable concentrations in soil water (Belfroid 1994, Van Straalen and Bergema 1995) rather than accumulation in the soil.

In a manual for calculating critical loads for heavy metals in terrestrial ecosystems (De Vries and Bakker 1998) it was suggested to use a precautionary policy goal, requiring that present metal concentrations should not increase in the future. The critical load then equals the load that does not lead to further accumulation of metals in the soil, implying that the critical load is determined by the present pollution status. Apart from that, an effect-based approach is relevant, and in this case it is more appropriate to use a critical limit for the soil solution. The mass balance model used to calculate critical loads is also based on this principle. The model calculates the critical load on the basis of an acceptable or critical metal leaching rate, which in turn is defined by a critical metal concentration in soil solution or groundwater.

This paper describes updated approaches, input data and results for critical loads of heavy metals and their exceedances on a European scale. The methods are based on the manual for calculating critical loads for heavy metals in terrestrial ecosystems mentioned above (De Vries and Bakker 1998). This manual was discussed and accepted at a workshop on critical limits and effect-based approaches for heavy metals and persistent organic pollutants (POPs) in Bad Harzburg, Germany in 1997 (UN/ECE 1997).

Critical loads were calculated for a situation where (i) no further accumulation of heavy metals occurs (the "stand-still" principle) and (ii) the concentration of heavy metals is below critical limits in the soil solution (the "effect-based" approach). This approach is in agreement with recommendations made at a 1999 workshop on effect-based approaches for heavy metals in Schwerin, Germany (Gregor et al. 1999). In the first case critical limits for the soil solution are derived from present metal concentrations while the second case used limited available information from the literature. Critical limits for the soil solution were related to the organic layer and the mineral topsoil (0–10 cm), where most microbial activity occurs. Values used (in mg m^{-3}) were 15 for Pb and 2.0 for Cd, based on Tyler (1992), using the second-lowest LOEC (lowest observed effect concentration) from laboratory studies with culture solutions reported by Balsberg-Påhlsson (1989), divided by a safety factor of 10. The minimum critical load from both approaches for both organic topsoil and upper mineral soil was then computed, as the critical load that at the same time protects ecosystems against ecotoxicological effects and does not lead to further metal accumulation in that part of the soil that is most vulnerable to metal pollution.

2. The STRESS model

A summary of the equations used to calculate critical loads is given in Table 1. Explanations of the abbreviations used are given in Annex 1 to this paper. The model consists of a mass balance equation (Eq. 1), a set of rate-limited equations for the major metal fluxes (litterfall, foliar uptake, root uptake, growth uptake and leaching; Eqs. 2–6) and equilibrium equations for the partitioning of metals over the soil solid phase, soil solution and DOC (Eqs. 7–11). The concentration of heavy metals in soils is such that mineral precipitation is negligible, unless strongly reduced conditions occur, such as in swamps and peatlands. These cases, in which mineral precipitation of metal sulphides may occur, are not considered in deriving a critical load, since the following assumptions apply to the mass balance model:

1. the soil system is homogeneously mixed, which implies that both soil properties such as organic matter content and concentrations of the pollutant do not show vertical variation within the observed soil system;
2. the soil is in an oxidised state; and
3. transport of water and heavy metals only takes places in vertical direction (no seepage flow, surface runoff or bypass flow).

The inherent limitations caused by the various assumptions are discussed in De Vries and Bakker (1998) and De Vries et al. (2001).

Litterfall and foliar uptake of heavy metals are both described as a linear function of atmospheric deposition (Eqs. 2 and 3). Growth uptake is based on a relationship with transpiration rate and soil solution concentration according to Eqs. 4a and 4b. Leaching is described as the product of water flux and the total metal concentration in soil solution according to Eq. 5.

Table 1. Process descriptions used in the STRESS model to calculate critical loads of heavy metals.

Mass balance equation:

$$fM_{tl} = -fM_{lf} + fM_{fu} + fM_{ru} - fM_{we} + fM_{le} + \Delta M_s \qquad (1)$$

Flux equations:

$$fM_{lf} = fM_{lf}(nd) + frM_{lf} \cdot fM_{td} \qquad (2)$$

$$fM_{fu} = fM_{fu}(nd) + frM_{fu} \cdot fM_{td} \qquad (3)$$

$$fM_{ru} = fr_{ru} \cdot (fM_{lf} - fM_{fu} + fM_{gu}) \qquad (4a)$$

with:

$$fM_{gu} = pfM_{ru} \cdot E_t \cdot [M]_{tot} \qquad (4b)$$

$$fM_{le} = ((1 - fr_i) \cdot P - E_{se} - fr_{ru} \cdot E_t) \cdot [M]_{tot} \qquad (5)$$

Partitioning equations:

$$[M]_{comp} = K_{p,DOC} \cdot [M]_{unc} \cdot DOC \qquad (6a)$$

with:

$$[M]_{unc} = \frac{(ctM_r / K_f^{\,1/n})}{fm} \qquad (6b)$$

$$[M]_{tot} = [M]_{unc} + [M]_{comp} \qquad (6c)$$

Critical load equations:

$$fM_{tl}(crit) = (fM_{re,d} + F_{re} \cdot [M]_{tot}(crit)) \;/\; fr_{re.d} \qquad (7)$$

with:

$$fM_{re,d} = (-1 + fr_{ru}) \cdot fM_{lf}(nd) - fr_{ru} \cdot fM_{fu}(nd) - fM_{we} \qquad (8)$$

$$F_{re} = fr_{ru} \cdot pfM_{ru} \cdot E_t + F_{le} \qquad (9)$$

$$fr_{re.d} = (1 - frM_{fu}) - fr_{ru} \cdot (frM_{lf} - frM_{fu}) + frM_{lf} \qquad (10)$$

Stand-still principle (steady-state soil solid phase):

$$[M]_{tot}(crit) = \frac{(ctM_r / K_f)^{1/n}}{fm} \cdot (1 + K_{p,DOC} \cdot DOC) \qquad (11)$$

Effect-based approach (steady-state soil solution):

$$[M]_{tot}(crit) = \text{a given threshold for soil solution} \qquad (12)$$

Equilibrium processes that determine the partition of heavy metals between various phases are adsorption to the soil and complexation of the metal with DOC in the soil solution. Adsorption is described by a non-linear relationship between the reactive metal concentration in the soil and the dissolved free heavy metal concentration (Eq. 6b). Complexation by DOC is included by a linear relationship with the free (uncomplexed) metal concentration and the DOC concentration, using a complexation constant (Eq. 6a).

Critical loads are calculated by first combining Eq. 1, neglecting the accumulation term, with Eqs. 2 through 5, thus leading to Eq. 7 for which the various terms (explained in Eqs. 8 through 10) are derived by combining Eqs. 2 through 5. For the critical load based on the stand-still principle (no further metal accumulation) the critical dissolved metal concentration, $[M]_{tot}(crit)$ is related to the present reactive metal concentration in the soil by Eq. 11 that combines Eqs. 2 and 6. If this critical metal concentration is not exceeded, further accumulation is avoided. Effect-based critical loads are also calculated by Eq. 7 but in this situation, $[M]_{tot}(crit)$ is not derived from the soil solid phase but equals a given critical value for the maximum allowable metal concentration in soil water (Eq. 12) that avoids toxic effects.

3. Input data

3.1 Geographic data:

Input data include parameters describing atmospheric deposition, precipitation, evapotranspiration, litterfall, foliar uptake, root uptake, weathering, adsorption and complexation. The input data mentioned above vary as a function of location (receptor area) and receptor (the combination of forest type and soil type) as shown in Table 2.

Table 2. The influence of location, land use and soil type on input data.

Input variable	Location	Forest type	Soil type[1]
Precipitation	x	–	–
Evapotranspiration	x	x	x
Litterfall	x	x	(x)
Foliar uptake	x	x	–
Root uptake	x	x	(x)
Weathering	–	–	x
Adsorption	–	–	x
Complexation	–	–	x

1 Values in brackets imply that soil type may influence the input data, but has not been accounted for in the data used here.

As a basis for the critical load computations, an overlay was made of five maps:

- A map with grid cells of 0.5°×0.5° that serves as the base map for acid deposition, heavy metal deposition and climate data estimates.
- A map with the soil types of Europe, i.e. the EU soil map on a scale 1:1,000,000 for EU countries (except Sweden and Finland) and Central Europe (EC 1985) and the FAO soil map at a scale 1:5,000,000 for the other countries (FAO 1992)
- A map with the forest types in Europe. This map was constructed using detailed NOAA-AVHRR satellite images with a resolution of approximately 1_1 km^2, and distinguishes conifer, broad-leaved and mixed forest based on differences in their reflection (Mücher et al. 2000).
- A map with climate zones for Europe derived from EC and UN/ECE (1996).
- A map with 500-m altitude zones, derived from detailed elevation data from USGS (Row et al. 1995).

The maps with climate zones and altitude zones were used in the procedure to estimate forest growth (described in section 3.4). The resulting map contains about 80,000 different units for which computations were made with the STRESS model.

3.2 Precipitation and evapotranspiration:

To compute the concentration and leaching of compounds in the soil, the annual water fluxes through the soil must be known. These water fluxes were derived from meteorological data available for the 0.5° longitude × 0.5° latitude grid described by Leemans and Cramer (1990), who interpolated selected records of monthly meteorological data from 1678 European meteorological stations for the period 1930–1960. Details of the interpolation procedure are given in Leemans and Cramer (1990).

Actual evapotranspiration was calculated according to a model that is essentially the same as used in the IMAGE global change model (Leemans and Van den Born 1994); it follows the approach by Prentice et al. (1993). Potential evapotranspiration is computed from temperature, sunshine and latitude. Actual evapotranspiration is computed using a reduction function for potential evapotranspiration based on the available water content in the soil described by Federer (1982). Soil water content is in turn estimated using a simple bucket-like model that uses water holding capacity (derived from the available soil texture data) and precipitation data. A full description of this hydrological module is given in Reinds et al. (2001).

3.3 Initial metal concentrations:

Initial metal concentrations are needed to calculate steady-state critical loads based on the stand-still principle because metal concentrations in the soil should stay constant. Initial metal concentrations were estimated from environmental factors such as metal deposition and soil characteristics using regression analysis on available data sets. Soil characteristics were derived from an available soil data set (Reinds et al. 1995), whereas the other explaining variables such as forest type and altitude were derived from the base maps described in section 3.1. Present heavy metal concentrations were derived from Van Mechelen et al. (1997). Table 3 lists the results from the regression analysis.

Table 3. Overview of the predictor variables explaining metal concentrations in the organic layer and mineral topsoil (0–10 cm) and the percentage variance accounted for (R^2_{adj}).

	Pb		Cd	
Predictor variable	**Org**	**Min**	**Org**	**Min**
Metal deposition	x	x	x	
Soil type			x	x
Forest type	x	x	x	
Altitude				
Rainfall				
Acid deposition	x	x	x	
C content in soil	x	x	x	
CEC in soil		x		x
N	1336	465	1270	427
R^2_{adj} (%)	36	48	24	37

The variables forest type, soil type and acid deposition (as an indicator of soil acidity) affect metal concentrations in both organic and mineral layers. Metal deposition has an effect on metal concentrations mainly in the organic layers. Table 3 shows that cation exchange capacity (CEC) is an important explaining variable in the mineral soil (as discussed in Van Mechelen et al. 1997) as well as the carbon content of the soil. This is line with the expectation that the dominant source of metals in the soil organic topsoil will probably be metal deposition, whereas in the mineral soils the parent material and metal binding capacity (through CEC and organic C, affected by pH) also play important roles. It must be stressed that the regression analysis yielded quite low percentages of explained variance, specifically for Cd in the organic layer, which means that the estimates of the initial metal concentrations are very uncertain. These estimates need to be improved in the future, preferably by using measured data obtained with standardised methods. Currently, a data set with present metal concentrations in the humus layer of forest soils is being prepared for the Nordic countries (Ruhling, pers. comm.).

3.4 Metal- and forest type-related data:

Heavy metal and forest type-related data include all data related to metal cycling in the ecosystem, i.e. litterfall, foliar uptake, and root uptake. Data used in the model calculations for coniferous and deciduous forests are summarised in Table 4.

All data have been based on data for total deposition, throughfall, litterfall and growth uptake for four beech sites in Germany (after Bergkvist et al. 1989). Values of fM_{lf}(nd) and frM_{lf} were derived from a linear regression between litterfall and total deposition. The adjusted coefficient of variation, R^2_{adj}, of these relationships varied between 0.73 and 0.96, depending on the metal considered (De Vries and Bakker 1998). Values for fM_{fu}(nd) and fM_{fu} were derived by subtracting throughfall from total deposition and relating the resulting canopy exchange fluxes to the total deposition. Values of R^2_{adj} for this regression relationship were high and varied between 0.77 and 0.98 (De Vries and Bakker 1998). Preference factors were derived from data on heavy metal deposition and growth uptake according to De Vries and Bakker (1998). Forest growth was estimated as a function of climate zone, forest type, altitude zone and stand quality according to the procedure described by Klap et al. (1997).

3.5 Metal- and soil-related data:

Heavy metal and soil-related data include (i) weathering rates, (ii) adsorption constants and (iii) complexation constants of heavy metals with DOC.

3.5.1 Weathering rates:

The simplest method to derive weathering rates of heavy metals is to scale them to the base cation weathering, using the molar ratio of the total metal concentration and the total base cation contents in parent material (Vrubel and Paces 1996) according to:

$$fM_{we} = 5 \cdot 10^{-5} \cdot BC_{we} \cdot \frac{ctM_p}{ctBC_p} \qquad (13)$$

where:

BC_{we} = the weathering rate of base cations ($mol_c\ ha^{-1}\ yr^{-1}$)

fM_{we} = the weathering rate of heavy metal M ($mg\ m^{-2}\ yr^{-1}$)

$ctBC_p$ = total content of base cations in parent material ($mol\ kg^{-1}$)

ctM_p = total concentration of heavy metal in parent material ($mg\ kg^{-1}$)

Base cation weathering rates were assigned to each combination of parent material class (derived from soil type) and texture class, and corrected for the effect of temperature according to a procedure described in De Vries et al. (1994). The ratio of metal concentrations to base cation contents in each major soil type was based on data in parent material (approximately at 1 m depth) of Dutch soils, since the total metal concentration in the topsoil may largely be influenced by accumulation due to (atmospheric) inputs. Due to the unavailability of data on a European scale, use was made of data from Dutch soils.

Table 4. Metal cycling parameters for coniferous and deciduous forests used in the model calculations.

Heavy	FM_{lf}(nd) ($mg\ m^{-2}\ yr^{-1}$)		frM_{lf}		frM_{fu} [1)]		fM_{ru}	
metal	Con	Dec	Con	Dec	Con	Dec	Con	Dec
Pb	0.0	0.0	0.34	0.25	0.36	0.47	0.14	0.13
Cd	0.072	0.094	0.04	0.05	0.35	0.55	0.36	0.13

1. The foliar uptake at negligible deposition was set to 0 for all metals and both forest types.

3.5.2 Adsorption constants:
Estimates of the Freundlich adsorption constant, K_f, for mineral soils were based on relationships with the soil parameters pH, soil organic carbon content (OC), cation exchange capacity (CEC), clay content and the activity of calcium according to:

$$\log K_f = a_0 + a_1 \cdot \mathrm{pH} + a_2 \cdot \log(CEC) + a_3 \cdot \log(\%OC) + a_4 \cdot \log(\%clay) - 0.5 \cdot n \cdot \log(Ca) \quad (14)$$

where:
K_f = the Freundlich adsorption constant for heavy metal M relating total metal concentrations in the soil to uncomplexed metal concentrations (activities) in soil solution ($mol^{1-n}\ m^{3n}\ kg^{-1}$)
CEC = the cation exchange capacity determined at pH 8.2 ($mol_c\ kg^{-1}$)
(Ca) = the Ca activity ($mol\ m^{-3}$)
pH = the equilibrium pH measured in the adsorption experiment

An overview of the coefficients found for the metals and a value for the Freundlich exponent *n* is provided in Table 5. Data are based on results of various adsorption experiments in the literature (Bril 1995).

Table 5. Values for the Freundlich exponent (*n*) and regression coefficients derived by Bril (1995) in the transfer function between the Freundlich adsorption constant K_f and soil properties (Eq. 14).

	Pb	Cd
n	0.55	0.82
a_0	-3.57	-3.15
a_1	0.6	0.50
a_2	0.624	1.00
a_3	0.46	–
a_4	–	-0.24
R^2	0.71	0.96
N *	12	14

* Number of measurements.

For organic layers, K_f values were derived from computed metal activities, based on measurements of metal concentrations and macrochemistry, versus measured metal concentrations in the organic layer of 200 Dutch forest soils using the Freundlich exponent from Table 5. This gave values for K_f of -0.6 and -1.0 for Cd and Pb respectively.

3.5.3 Reactive metal concentrations:
Because the Freundlich equation (Eq. 7a) refers to the reactive metal concentration in the soil, this concentration must be derived from the total metal concentration that is normally determined using an aqua regia extract. The reactive metal concentration ctM_r can be derived from the total metal concentration using a linear relation with soil characteristics (clay content and organic matter content) and $ctM_{aqua\ regia}$ according to:

$$ctM_r = b_0 + b_1 \cdot ctM_{aqua\ regia} + b_2 \cdot \%\ \mathrm{clay} + b_3 \cdot \%\ \mathrm{organic\ matter} \quad (15)$$

Data for the various coefficients are given in Table 6.

Table 6. Value for the regression coefficients in the relationship between reactive metal concentration versus total metal concentration and soil properties.

	Pb	Cd
b_0	3.84	0.0
b_1	0.61	0.65
b_2	-0.38	-0.001
b_3	0.01	0.007
R^2	0.86	0.87
N *	49	49

* Number of measurements.

This relationship was derived from unpublished data on metal concentrations in mineral layers from 49 different soils using EDTA and aqua regia extractions (Bril, pers. comm.). Measurements of aqua regia- and EDTA-extractable metals in organic layers of 11 forest soils in the Netherlands (Groenenberg, unpublished data) show that the immobile fraction of metals in organic layers is negligible; the measured metal concentrations from both methods was about equal. For organic layer the reactive metal concentration was thus assumed to be equal to the total metal concentration.

3.5.4 Complexation constants:
The value of $K_{p,DOC}$ is affected by pH. In this study, this dependence was described as:

$$K_{p,DOC} = K_c M \cdot 10^{-3} \cdot m \cdot \frac{K_a}{K_a + [\mathrm{H}]} \quad (16)$$

where:
K_cM = the complexation constant for heavy metal M with dissociated monovalent organic acid ($mol^{-1}\ l^{-1}$)
m = the concentration of acidic functional groups per kg DOC ($mol_c\ kg^{-1}$ C)
[H] = the proton concentration ($mol\ l^{-1}$)
K_a = the dissociation constant for organic acid ($mol\ l^{-1}$)

The value of *m* was set to 5.5 $mol_c\ kg^{-1}$ DOC for soils in accordance with Henriksen and Seip (1980) and Bril (1996). Values for the dissociation and complexation constants are given in Table 7.

Table 7. Calibrated values of pK_a and log K_cM describing the dissociation and complexation of a reactive monoprotic organic acid with heavy metals (after De Vries and Bakker 1998).

M	pK_a	log K_cM
Pb	9.4	10.5
Cd	4.4	4.1

3.6 Soil data:

To compute critical loads of heavy metals a number of soil characteristics must be known (cf. equations 6a, 14 and 15), including: clay content, organic carbon content, pH, Ca concentration in soil solution and DOC concentration.

Clay content is an attribute of the soil maps: the EC soil map defines five different texture classes, (and the FAO soil map three), each with a range in clay, sand and silt content. The average clay content in the class was used to characterise the soil. The organic matter content, amount of organic layer and present pH values were estimated for each soil type separately from existing databases (De Vries et al. 1993, Van Mechelen et al. 1997). The Ca concentration in the soil water was computed from estimates of Ca deposition, uptake and weathering using the START model as described by Reinds et al. (1995). Bulk density was computed using an equation given by Van Wallenburg (1988) for mineral soils that relates bulk density to clay content and organic carbon content and of Hoekstra and Poelman (1982) for organic soils, that relates bulk density to organic carbon content. DOC concentration was estimated from forest and soil characteristics based on measured DOC concentrations from 150 Dutch forest stands (De Vries et al. 1995).

4. Results

In this section, the computed critical loads for Pb and Cd are described and discussed separately. In order to protect the majority of ecosystems in a grid cell, the 5-percentile critical load is used as an indicator (protecting 95% of the ecosystems in a grid cell). Critical loads for the organic layer and the mineral topsoil (0–10 cm), applying both the stand-still principle and the effect-based approach, are presented in tables to give insight in the most limiting criterion, depending on the soil layer considered (section 4.1). If the ecosystem is to be protected against further metal increase and at the same time against effects on soil fauna, the minimum of both critical loads is a suitable threshold. All soil compartments in turn are protected using the minimum of organic and mineral layer. This minimum of four different critical loads is therefore used to present the geographic distribution of critical loads by maps (section 4.2).

4.1 Ranges in critical loads of heavy metals:

4.1.1 Lead (Pb):

Ranges in critical Pb loads for the organic layer and the mineral topsoil illustrate that critical loads based on the stand-still principle are more stringent than those based on the effect-based approach (Table 8). It should be noted, however, that the "stand-still" critical load is strongly influenced by the adsorption function used. For Pb this adsorption function has a high uncertainty and seems to overestimate the adsorption and thus underestimate critical loads (Groenenberg et al. in prep.), so critical loads for lead derived using this function should be interpreted with care. For areas with a high current Pb deposition such as the Ruhr area, northern France, southern UK and the Benelux region, the initial Pb concentrations are high in the organic layer. In these areas the critical loads for the organic layer are thus often (much) higher than that of the mineral layer (i.e., the initial concentration is less influenced by current deposition). In other areas however, such as Scandinavia the lowest critical loads are calculated for the organic layer due to a low initial concentration in combination with a strong adsorption in this layer.

4.1.2 Cadmium (Cd):

Table 9 lists the ranges in the critical loads for Cd for both the organic layer and the mineral topsoil using each of the two approaches. This table shows that the critical loads are generally lower for the mineral layer than for the organic layer, and that the effect-based approach is generally more stringent than the stand-still principle. Specifically, in central Europe the critical loads based on the effect-based approach are lower than those based on the stand-still principle, because the initial high metal concentrations in acidified soils with weak adsorption lead to high leaching rates in the present situation. Those leaching rates seem to exceed critical limits, and the input should thus be focused on a decrease in soil metal concentrations. The stand-still principle is in those areas not stringent enough.

Table 8. Calculated ranges in critical loads of lead for the organic layer and mineral topsoil (0–10 cm) using both the "stand-still" principle and the effect-based approach.

		Critical Pb load (mg m^{-2} yr^{-1})				
Soil layer	**Approach**	**5%**	**25%**	**50%**	**75%**	**95%**
Organic	Stand-still principle	0.02	0.34	2.4	12.4	180
	Effect-based	3.5	4.4	5.6	8	14.2
Mineral	Stand-still principle	0.1	0.7	1.4	3.1	7.8
	Effect-based	2.6	3.4	4.5	6.4	12.3

Table 9. Calculated ranges in critical loads of cadmium for the organic layer and mineral topsoil (0–10 cm)using both the "stand-still" principle and the effect-based approach.

		Critical Cd load (mg m^{-2} yr^{-1})				
Soil layer	**Approach**	**5%**	**25%**	**50%**	**75%**	**95%**
Organic	Stand-still principle	0.4	0.7	1.0	1.7	3.3
	Effect-based	0.41	0.52	0.65	1.1	1.8
Mineral	Stand-still principle	0.11	0.5	0.9	2.0	4.4
	Effect-based	0.29	0.38	0.50	0.76	1.4

4.2 Geographic variation in critical loads of heavy metals:

4.2.1 Lead (Pb):

Fig. 1 shows the geographical distribution of the 5 percentile minimum critical loads for lead. Lowest values (< 0.25 mg m^{-2} yr^{-1}) are related to the "stand-still" critical load and are thus found in areas with low initial concentrations. As present deposition has a strong influence on the initial Pb concentration, these are regions with low Pb deposition such as Scandinavia and Ireland. Furthermore, low critical loads are found in areas with soils strongly adsorbing Pb.

4.2.1 Cadmium (Cd)

Fig. 2 shows that the minimum critical load for cadmium ranges from less than 0.1 mg m^{-2} yr^{-1} to more than 0.5 mg m^{-2} yr^{-1}. The highest critical loads are found in areas with high precipitation excess (due to a dilution effect) such as the UK and Ireland, southwestern Norway and northwestern Spain. The lowest critical loads are found in regions with low precipitation excess and in regions with relatively low present metal concentrations and strong metal adsorption. The highest adsorption coefficients are associated to soils with a high pH such as calcareous soils (southern Europe) and soils regions not affected by acidification (northern Scandinavia).

5. Discussion and conclusions

5.1 Applicability of the approach and comparison to previous approaches:

To assess critical loads for Cd and Pb and their exceedances, the methods laid out in the critical load manual (De Vries and Bakker 1998) were successfully applied to European forest soils. By overlaying available regional information on the distribution of soils, forest types, climate and altitude in Europe a map was constructed that gives a detailed distribution of relevant receptors in Europe. The improved soil map and soil database and the more detailed forest map are major improvements compared to previous studies (Reinds et al. 1995, Van den Hout et al. 1999). Furthermore, the improved methodology led to quite different results.

In this calculation, present metal concentrations were not allowed to increase (the "stand-still" principle) and toxic effects on soil organisms were avoided. Furthermore, the minimum critical load of both approaches applied to both the organic layer and mineral topsoil (0–10 cm) was used. The previous calculations of critical loads for forest soils on a European scale (Reinds et al. 1995), made use of critical limits for the soil solid phase only, including (i) Maximum Permissible Concentrations (MPC) derived from laboratory experiments with soil

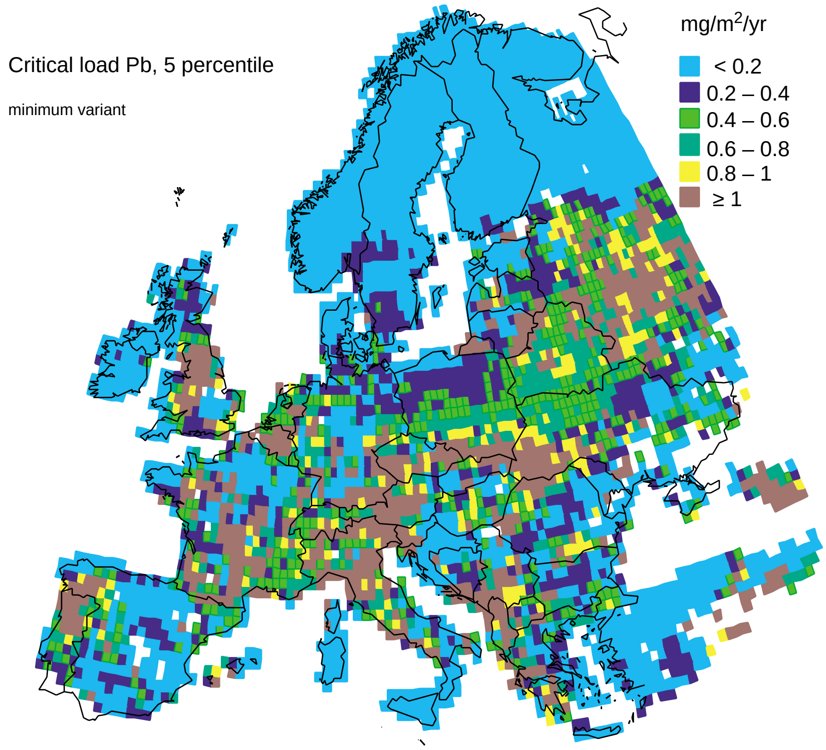

Figure 1. Geographical distribution of the 5-percentile critical Pb loads over Europe.

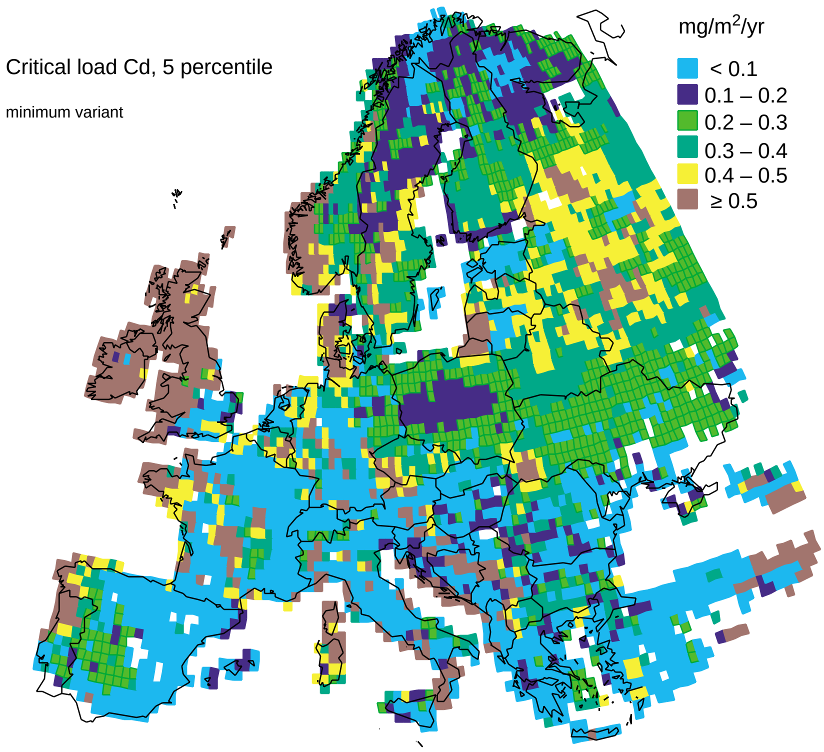

Figure 2. Geographical distribution of the 5-percentile critical Cd loads over Europe.

organisms and plants, (ii) Target Values set by the Dutch Ministry of Housing, Spatial Planning and the Environment (DTV) and (iii) critical concentrations derived for humus layers of Swedish forest soils. The implicit assumption was that (ecotoxicological) effects are due to metal accumulation in the soil.

As stated in the introduction, a more fundamental problem when using critical limits for the soil is that in most cases, toxic effects on (e.g.) micro-organisms and soil fauna are mainly due to elevated bioavailable concentrations in soil water rather than in the soil. In the previous model calculations, critical loads were high for soils with low adsorption rates (e.g. acid sandy soils) since a large part of incoming metals was leached, whereas the reverse was true for soils with high adsorption rates (e.g. calcareous sandy soils). This outdated approach, however, completely ignored the adverse effects of elevated dissolved metal concentrations on soil fauna, vegetation and (metal leaching to) groundwater. A discussion on this topic is further given in De Vries and Bakker (1998). The present pattern of critical loads is more consistent in view of accumulation and leaching.

5.2 Uncertainties in the calculated critical loads:
It should be stressed that the results from this study are uncertain. Main sources of uncertainty for the critical loads calculated by the "stand-still" principle are the adsorption function (through e.g. the uncertainty in estimated pH, especially for Cd), the initial metal concentrations and the complexation constants (Groenenberg et al. 2001). Specifically the uncertainty in present metal concentrations is large. Results of a regression analysis, relating these concentrations to environmental variables such as soil type, climate and heavy metal deposition, were consistent with what could be expected, but the percentage of explained variance was low (< 50 %). The same holds for the soil properties influencing the adsorption constant (content of organic matter and clay and the pH).

In addition, the adsorption function for Pb is quite uncertain. As a consequence, the estimates of initial metal concentrations in soil and in soil solution, and in turn the estimated critical loads, have a high uncertainty, specifically for Pb. Estimates of initial metal concentrations can probably only be improved by using pan-European measurements; thus initiatives in this direction should be encouraged. Furthermore, the transfer functions of adsorption constants against soil properties need further improvement, specifically for the organic layer. Main sources of uncertainty for the critical loads calculated by the effect-based approach are the maximum allowable metal concentration in the soil water and the estimated precipitation excess (Groenenberg et al. 2001).

5.3 Conclusions:
Results show that for lead, the "stand-still" critical load is lower than the effect-based critical load, whereas for cadmium both approaches give comparable results. This means that if an increase in the present metal concentration is not allowed, effects on soil organisms are, to a large extent, also avoided.

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Annex 1. Explanation of symbols in the soil model STRESS

Symbol	**Description**	**Unit**
ctM_{DOC}	Concentration of heavy metal *M* on (complexed with) DOC in soil solution	mg kg^{-1}
ctM_s	Total concentration of heavy metal *M* in the soil	mg kg^{-1}
ctM_r	Reactive concentration of heavy metal *M* in the soil	mg kg^{-1}
DOC	Dissolved organic carbon concentration in the soil solution	kg m^{-3}
E_i	Interception evaporation	m yr^{-1}
E_{se}	Soil evaporation	m yr^{-1}
E_t	Transpiration	m yr^{-1}
f_M	activity coefficient of heavy metal *M*	–
F_{le}	Water flux leaching from the soil	m yr^{-1}
F_{re}	Removal flux	m yr^{-1}
fr_i	Interception evaporation fraction	–
$fr_{re,d}$	Deposition dependent removal fraction	–
fr_{ru}	Root uptake fraction	–
frM_{fu}	Foliar uptake fraction of heavy metal *M*	–
frM_{lf}	Litterfall fraction of heavy metal *M*	–
pfM_{ru}	Preference factor for root uptake of heavy metal *M*	–
K_a	Dissociation constant for organic acid	mol l^{-1}
K_cM	Complexation constant for heavy metal *M* with dissociated monovalent organic acid	mol^{-1} l^{-1}
K_f	Freundlich adsorption constant for heavy metal *M*	mol^{1-n} m^{3n} kg^{-1}
$K_{p,tot}$	Partition coefficient between the total concentration of heavy metal *M* in the soil and the total concentration in the soil solution.	m^3 kg^{-1}
$K_{p,DOC}$	Partition coefficient of heavy metal *M* between dissolved organic carbon and the soil solution	m^3 kg^{-1}
fM_{le}	Flux of heavy metal *M* by leaching	mg m^{-2} yr^{-1}
fM_{lf}	Flux of heavy metal *M* by litterfall	mg m^{-2} yr^{-1}
$fM_{lf}(nd)$	Flux of heavy metal *M* in litterfall at negligible deposition	mg m^{-2} yr^{-1}
fM_{fu}	Flux of heavy metal *M* by foliar uptake	mg m^{-2} yr^{-1}
$fM_{fu}(nd)$	Foliar uptake flux of heavy metal *M* at negligible deposition	mg m^{-2} yr^{-1}
fM_{gu}	Flux of heavy metal *M* by growth uptake	mg m^{-2} yr^{-1}
$fM_{re,d}$	Deposition-dependent removal flux of heavy metal *M*	mg m^{-2} yr^{-1}
fM_{ru}	Flux of heavy metal *M* by root uptake	mg m^{-2} yr^{-1}
fM_{td}	Total deposition of heavy metal *M*	mg m^{-2} yr^{-1}
fM_{tl}	Total load of heavy metal *M*	mg m^{-2} yr^{-1}
$fM_{tl}(crit)$	Critical load of heavy metal *M* on soils	mg m^{-2} yr^{-1}
fM_{we}	Flux of heavy metal *M* by weathering	mg m^{-2} yr^{-1}
$[M]_{unc}$	Dissolved concentration of uncomplexed (free) heavy metal *M* in soil solution	mg m^{-3}
$[M]_{tot}$	Total concentration of heavy metal *M* in soil solution	mg m^{-3}
n	Freundlich exponent	–
ΔM_s	Change per year of pool of metal *M* in the soil	mg m^{-2} yr^{-1}

6. Mapping the Atmospheric Mercury Pollution of Boreal Ecosystems in Sweden

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Background

In Sweden, efforts are currently under way to assess the maximum level of constant atmospheric mercury pollution that causes no or tolerable damage in affected ecosystems ("maximum tolerable concentration" or "maximum permissible load" or "critical load"). Such concepts have previously provided a successful basis for controlling acidifying pollutants in Europe, in particular sulphur (e.g. Henriksen et al. 1992; Posch et al. 1997). Mercury is particularly suitable as a pilot substance for which to develop emission-exposure-effect concepts for various micropollutants, because (e.g.) its biogeochemical cycle is tightly coupled with that of atmospheric and organic sulphur (e.g. Meili 1991, Munthe et al. 2001).

The ecosystems currently the focus of the Swedish work are boreal coniferous forests and softwater lakes, where present mercury levels frequently exceed critical limits. The working strategy includes the following steps:

1. Compile available data on mercury in the atmosphere, precipitation, soils, freshwaters, sediments, and fish.
2. Convert these data to gridded maps.
3. Assess the pathways and transfer dynamics of mercury in terrestrial and aquatic ecosystems of boreal watersheds.
4. Quantify source-receptor relationships for different types of ecosystems.
5. Develop operative dynamic models adapted to the availability of data and applicable in grid systems (GIS).
6. Calculate tolerable atmospheric mercury levels.

This paper presents selected aspects of the current status of this work. Methods to assess the ecosystem dynamics of mercury and to calculate critical levels of atmospheric mercury pollution have been presented earlier (Meili et al. 1999) and have been considered promising within the UN/ECE (Gregor et al. 1999). At present, this approach is being tested at the national level, where comprehensive databases are available. This is done with the aim of developing strategies for boreal regions of Scandinavia that are particularly sensitive to mercury pollution, and at a later stage also for other areas in Europe.

Mercury in the Swedish environment

Long-range atmospheric transport of mercury (Hg) and other micropollutants has caused widespread contamination of soils and lakes even in remote areas, including the boreal forest zone. In Sweden, attention is presently focused on alarmingly high mercury levels in fish, exceeding health advisory guidelines in tens of thousands of lakes (e.g. Lindqvist et al. 1991, Andersson and Lundberg 1995) as well as recent indications of toxic metal effects on the soil microflora (Bringmark and Bringmark 2001a,b; Palmborg et al. 2001). Atmospheric deposition is considered to dominate the mercury input to most soils and lakes in the boreal forest zone (Lindqvist et al. 1991, Fitzgerald et al. 1998), an area of considerable economic interest, both for forestry and sport fishing. Atmospheric mercury deposition has increased 2- to >20-fold over the past centuries due to anthropogenic emissions and subsequent dispersal on local, regional, and global scales. Concern about the health of boreal and other ecosystems has called for concepts on which to base international regulations with respect to atmospheric mercury emissions.

The issue is accentuated by the fact that present mercury levels in most soils and lakes are still far from equilibrium with atmospheric mercury inputs. This is evident not only from mass balance calculations, but also from the delayed response of surface waters to the pollution pulse of the past decades (e.g. Meili et al. 1999). For this reason, mercury concentrations in fish are likely to increase further for several centuries. This increase will be particularly pronounced in humic lakes, where mercury concentrations are highest already, and where most mercury is supplied by soil runoff in a highly bioavailable form, but substantially delayed. In many humic lakes, even natural mercury levels in predatory fish are estimated to reach the common limit of 0.5 mg kg^{-1} fresh weight. In rain-fed clearwater lakes on the other hand, which respond to environmental changes within years or decades, and which today show a similar degree of contamination as the humic lakes, natural mercury levels are estimated to be less than 0.1 mg kg^{-1} (Meili et al. 1999).

The slow ecosystem dynamics need to be accounted for to adequately quantify the relationship between atmospheric deposition and concentrations in soils or fish (e.g. the critical load), even when using simplified models to predict a future steady state based on available data. The data accumulated in Sweden over the past decades provide a suitable way to reconstruct ecosystem dynamics and to test different modelling approaches. A modelling concept has been developed (e.g. Meili et al. 1999) to assess future mercury accumulation in boreal forests and lacustrine fish based on a reconstruction of the past, including a way of minimising the model structure (the number of uncertain parameters and mechanisms) and data requirements (the number of necessary measurements).

In parallel, work has been initiated to produce susceptibility maps for Sweden based on readily available information. The first results presented below show that regionally high mercury levels in fish reflect their high susceptibility to mercury deposition primarily due to emissions from remote sources.

Mapping method

For the regional assessment of the fate of atmospheric mercury input to terrestrial and aquatic ecosystems, available data need to be transformed into a format suitable for spatially distributed modelling. For several reasons, the EMEP grid with a spatial resolution of 50×50 km^2 has been adopted. This grid facilitates the modelling of links between atmospheric input and ecological effects of mercury based on available data, the coordination with European work on critical loads of other substances, and the extension of regional models to the European scale. Sweden is covered by 160 to 230 grid cells, depending on the treatment of national and coastal boundaries. This is an optimal system for testing various modelling approaches, since the grid cells are sufficiently large to find available data for most grid cells, and reasonably few to be manageable, but sufficiently numerous to support statistical evaluations. Once tested models are available, these can be implemented in other systems differing in resolution or other criteria, e.g. the perpendicular grid used for weather forecast modelling, or non-rectangular units such as single watersheds.

Atmospheric emission and deposition of mercury

The atmospheric emission, transport, and deposition of mercury is subject to European modelling work within the framework of EMEP (e.g. Ryaboshapko et al. 1998, 1999; Ilyin et al. 2000). The model used to calculate heavy metal airborne transport and deposition includes basic mechanisms of pollutant transport in and scavenging from the atmosphere, such as the emission of different species, advective transport, turbulent diffusion, dry and wet deposition. The mercury transport model also incorporates a module describing the chemical transformations of mercury in the atmosphere. This eulerian model has a spatial resolution of 50×50 km^2, is operating within the geographical scope of the EMEP region (135×111 cells), and uses a calculation time step of 20 minutes.

The 1995 European emission inventory (Pacyna et al. 2001) shows that Swedish mercury emissions to the atmosphere amount to about 0.9 tons yr^{-1}, of which most are located in the south, especially in the three largest urban areas (Fig. 1). Given as a grid cell mean, the highest emission density (approximately 35 g km^{-2} yr^{-1}) is in Stockholm. The Swedish emission are very small compared to European emissions, which total about 300 tonnes yr^{-1} and have mean densities of over 500 g km^{-2} yr^{-1} in some dozen grid cells (Ilyin et al. 2000, Pacyna et al. 2001).

The model output for 1997–1998, which is partly calibrated by measurements, suggests a total atmospheric mercury deposition over Sweden of about 8 tonnes per year, with a pronounced south-north gradient (Fig. 2a). In combination with current knowledge on the atmospheric mercury cycling, it is estimated that the deposition may be about 3–5 times higher than the natural deposition, about 2–4 times in the north and about 4–8 times higher in the south. For wet deposition alone, this contamination factor is about twice as high. However, Swedish deposition values are low on a European scale, with deposition reaching a total of around 500 tonnes yr^{-1} and exceeding mean densities of 200 g km^{-2} yr^{-1} in some dozen grid cells (Ilyin et al. 2000).

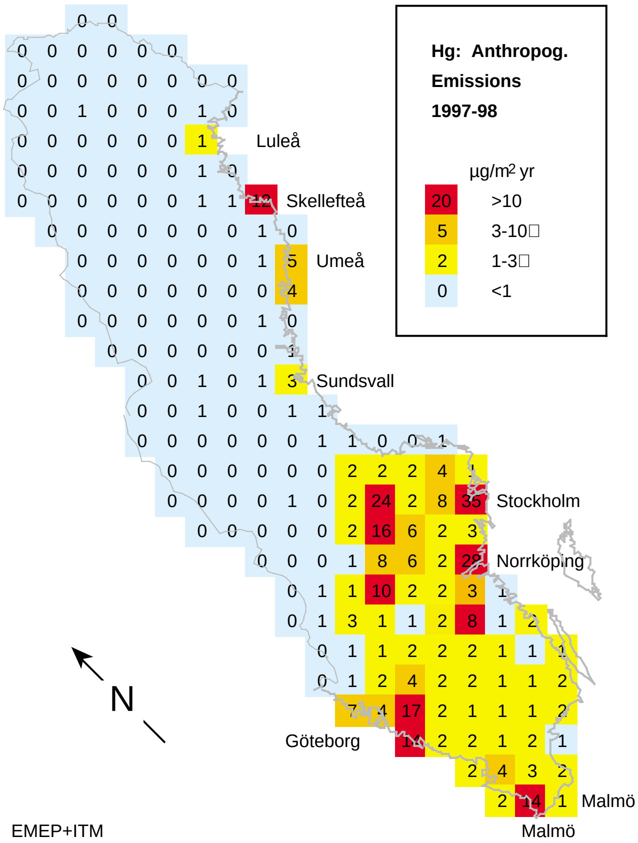

Figure 1. Swedish emissions of mercury to the air during the late 1990s. EMEP data with minor corrections.

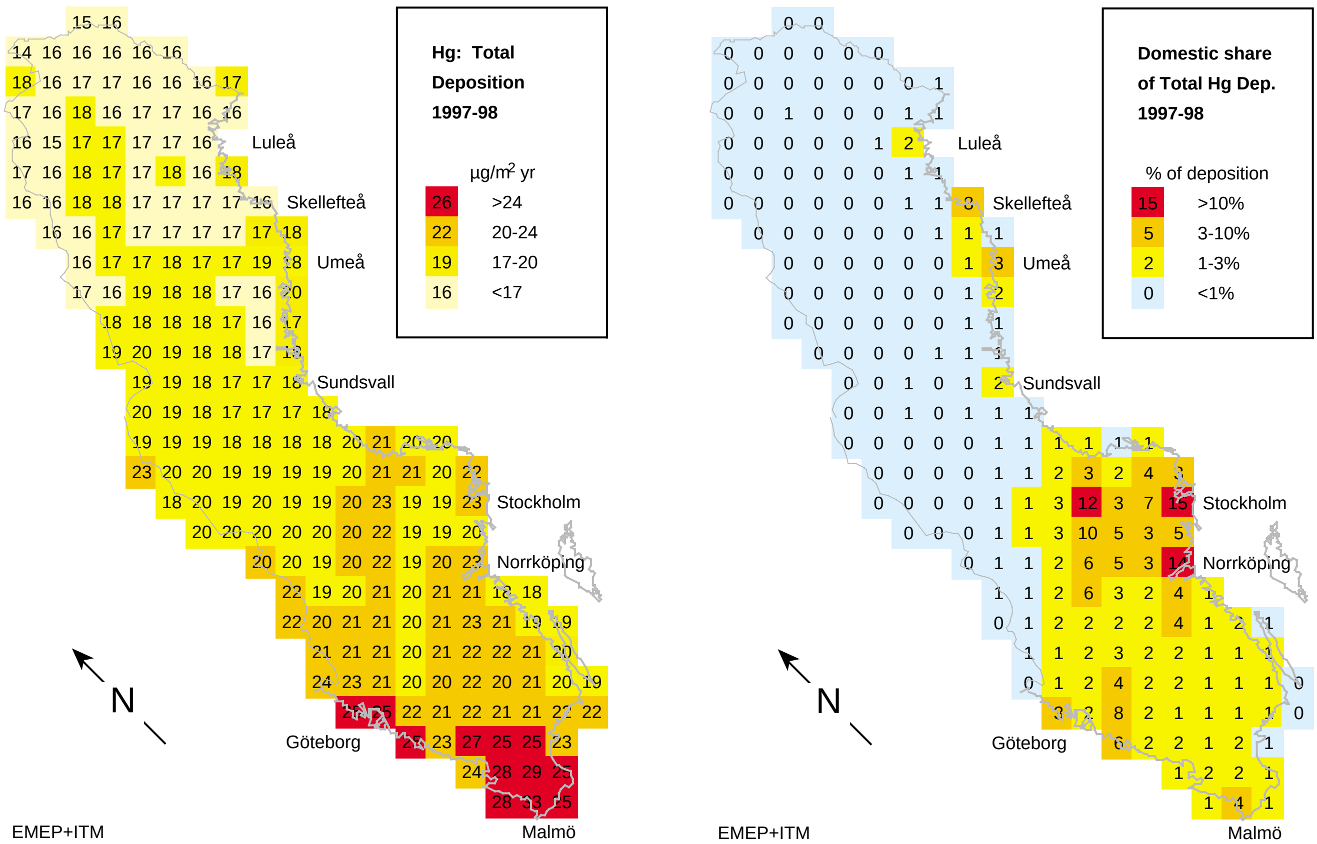

Figure 2. Atmospheric deposition of mercury (dry + wet, left), and the share from domestic emission (right) in different regions of Sweden in 1997–98, based on EMEP model calculations.

Not surprisingly, domestic emissions explain only about 2% of the total deposition in Sweden, although the share is highly variable among regions (Fig. 2b). Given the dominance of foreign sources, mercury deposition in Sweden can essentially only be reduced by reducing emissions elsewhere.

Critical limits of mercury in terrestrial and aquatic ecosystems

Ecotoxicity is proportional to the ratio between dose (e.g. aqueous or tissue concentration) and susceptibility (i.e. toxic effect per dose) at both the individual and ecosystem levels. In the case of mercury in biota, both dose and susceptibility depend on the form of mercury. At each trophic transfer along food chains, the tissue concentrations in animals increase several fold for methylmercury, but decrease for inorganic Hg (e.g. Meili 1997). As a result, the dose of methylmercury increases with the trophic level in food webs, while the dose of inorganic Hg is highest at low trophic levels as represented by microorganisms, in particular detritivores and associated food chains. The susceptibility to methylmercury is particularly high in the central nervous system of developing vertebrate and bird embryos, while the susceptibility to inorganic Hg is known to be considerable at the microbial level. In ecosystems exposed to a given dose of Hg, toxic effects can thus be expected primarily in top predators and at the microbial level.

As a first step in identifying critical limits, the Swedish Environmental Protection Agency (SNV) has suggested the following environmental quality objectives: For the protection of human health and a sustainable management of natural resources the contents of mercury in fish may not exceed 0.5 mg kg^{-1} fresh weight; and for the protection of biological diversity and a sustainable management of natural resources the large-scale accumulation of metals in the humus layer of forest soils must stop (SNV 1996). The critical level for fish is based on weekly intakes given by WHO and used by many other countries. In the USA, a fish tissue level of methylmercury as low as 0.3 mg kg^{-1} fresh weight has recently been confirmed as a recommended limit (US EPA 2001; note that 80–100% of mercury in pike is methylmercury).

A criterion for soils, on the other hand, is still under development. Most Swedish forest soils are covered by an organic humus layer (mor) in which many deposited pollutants are efficiently retained. Since plant root systems and fungi are located in this layer, there is an immediate risk of biological disturbance with potential economic dimensions. Retarded decomposition of organic matter may have direct consequences for the mineralisation of nutrients in forest soils. Indeed, recent findings show weak observational and strong experimental evidence of a reduced respiration in forest soils at mercury concentrations close to those encountered in rural areas of south Sweden (Bringmark and Bringmark 2001a,b; Palmborg et al. 2001). These studies lower the effect levels for mercury (as well as lead) considerably below known values. A tentative critical receptor value is that the mercury concentration in the humus layer (O-horizon) of podzolic forest soils should not exceed 0.5 mg Hg kg^{-1} organic matter, the present mean level in the most contaminated regions of southern Sweden.

In order to implement environmental goals, receptors need to be defined that are spatially and temporally robust and easy to quantify. In boreal forest soils, the organic top layer of podzols (spodosols) provides a fairly uniform matrix that can be used as a surrogate to assess the exposure of soil microflora to mercury and other pollutants. Based on the availability of survey data, a homogenate of the whole mor layer is considered here. Mercury concentrations in biota, on the other hand, show a wide variability. Methylmercury concentrations range over four orders of magnitude in "unpolluted" freshwaters alone, even if normalising to whole-body organic dry weight and disregarding seasonal variations (Meili 1997). Within a given freshwater ecosystem, the variation of methylmercury concentrations among organisms is about 200-fold, and among fish of all species and size classes about 30-fold. To eliminate this source of variation, cross-system comparisons should be based on a system-specific reference value, for example on a single type of fish.

In Sweden, the mean concentration in 1-kg pike (*Esox lucius* L.) has been used since the beginning of mercury monitoring in 1960s, and is now available for several thousand lakes in Sweden (e.g. Lindqvist et al. 1991, Andersson and Lundberg 1995). This lake-specific parameter provides a suitable operative tool as it is related to human consumption habits, easily measured, and spatially as well as temporally rather stable. Moreover, it is a relevant value from an ecotoxicological point of view, since it is closely related to (and only about two-fold lower than) the highest methylmercury concentrations in any type of aquatic organism within the same system (usually the largest individuals of the same species, not considering birds and mammals feeding on lacustrine fish).

Mercury levels in terrestrial and aquatic ecosystems

While the variation of mercury concentrations among biota within ecosystems can be eliminated by standardisation, the variability among ecosystems is a more difficult issue. The variation among lake ecosystems in Sweden alone is considerable, with standardised mean mercury concentrations in the muscle tissue of 1-kg pike (*Esox lucius*) ranging from 0.04–2.6 mg kg^{-1} fresh weight, and typically from 0.1–2.0 mg kg^{-1} fresh weight in properly sampled lakes (at least 5 spring individuals close to 1 kg), even when only considering samples collected and analysed after 1980 in lakes without direct mercury discharge and at some distance from known atmospheric point sources of mercury (Lindqvist et al. 1991). The initial focus is on regional mean values, irrespective of lake type or other sources of variability. As a first mapping step, earlier compilations of data (Andersson and Lundberg 1995) have been complemented with additional data and converted into the EMEP grid format (Fig. 3). More recent data are currently being compiled and evaluated in order to update these databases and also to assess temporal trends. For mercury in fish, standardised concentrations in 1-kg pike were used here, based on over 5000 individual pike data from over 1000 lakes. Compilations show that these standardised values exceed the current guidelines of 0.5 mg kg^{-1} wet weight in about half of Sweden's almost 100,000 lakes, which is also evident for regional mean values within grid cells (Fig. 3).

A national survey of mercury in 356 forest soils in 1983–84 showed concentrations ranging from 0.07-1.0 mg kg^{-1} dry weight (Andersson 1991; see also Alriksson 2001). This country-wide survey was aiming at standardised values for the organic top soil (humus or mor layer, O-horizon, between organic litter and mineral soils) of podzols on sandy till, which is probably the most common soil type in the boreal forest zone. To comply with the binding of mercury to organic matter, concentrations are suitably normalised with respect to the concentration of organic carbon (Meili 1991). In the forested inland of south Sweden, even at some distance from major point sources, this yields regional means close to the critical limit of 0.5 mg kg^{-1} organic matter (Fig. 3).

Susceptibility of lake ecosystems to mercury input

Lakes within the same region can show widely varying fish levels (frequently about five-fold). This illustrates the importance of natural environmental factors at a given atmospheric pollution load. Also regionally, variations in levels are evident (Fig. 3) that appear unrelated to current emission or deposition patterns (Fig. 2). Note in particular the very low fish concentrations in the southernmost grid cell, where deposition is very high, but the biogeochemical setting different.

Regional differences in environmental susceptibility can be conveniently illustrated by comparing receptor / source ratios, although these do not account for slow ecosystem dynamics of mercury. In the case of lakes, suitable ratios are the concentration ratio between fish and precipitation representing the link to atmospheric deposition, and the concentration ratio between fish and soil representing the link to the contamination of runoff waters (Fig. 4). It is evident that these ratios are not constant but rather vary widely among regions. Furthermore, the similarity of the regional pattern for both ratios deserves attention.

Current work aims to elucidate the regional differences in susceptibility by comparison with data on climate, geochemistry, and land cover. Such information can be combined with dynamic models (e.g. Meili et al. 1999) to calculate critical exposure levels, in particular for boreal areas where biotic mercury levels are naturally high, and at a later stage also in other areas.

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The Swedish work for generating operational mercury models has been developed by a working group covering different aspects of the mercury cycle and environmental modelling: Kevin Bishop and Lage Bringmark (Swedish Univ. of Agricultural Sciences), Kjell Johansson (Swedish Environmental Protection Agency), John Munthe (Swedish Environmental Research Institute), Harald Sverdrup (Lund University), and Wim de Vries (Alterra Green World Research, Netherlands).

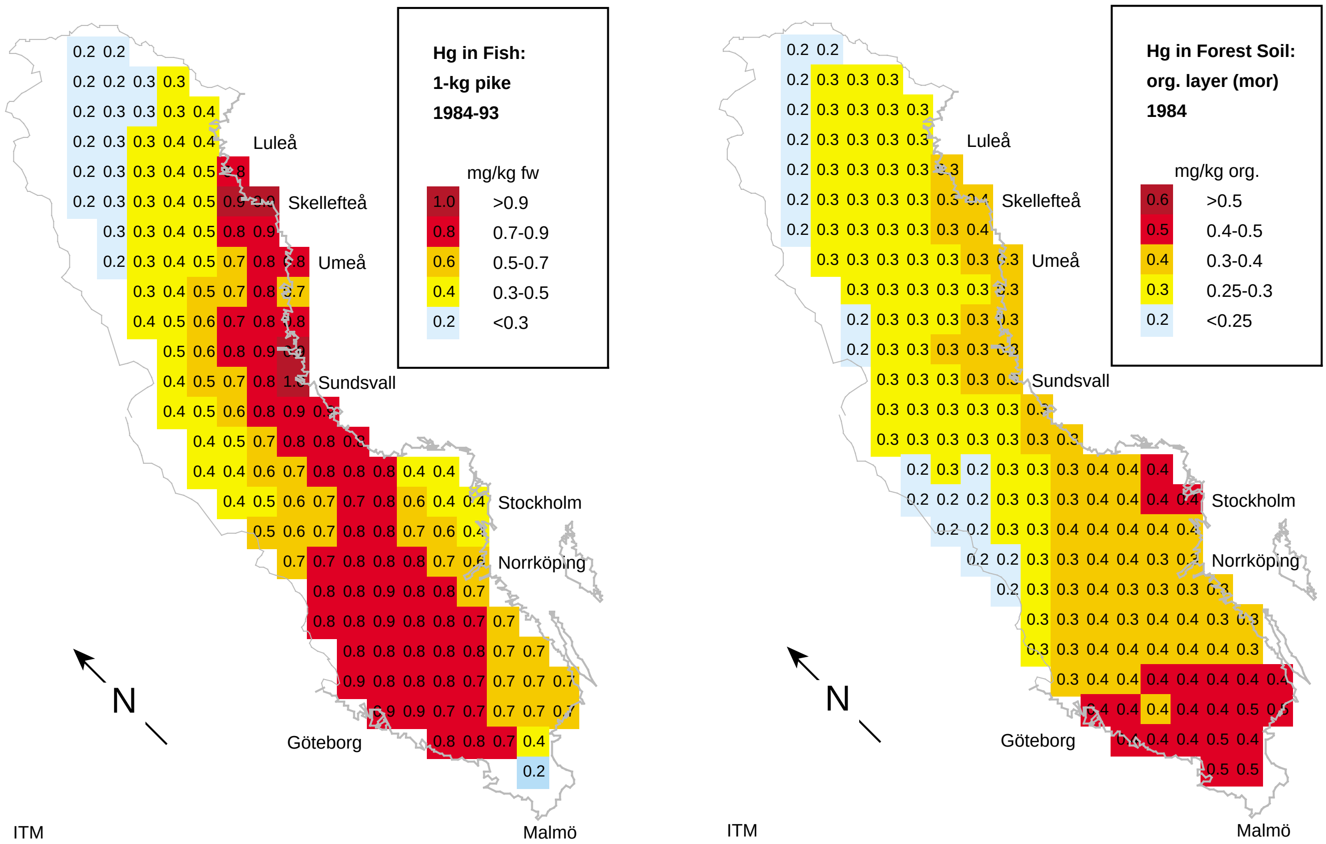

Figure 3. Left: Mercury in freshwater fish; standardised concentrations in muscle tissue of 1-kg pike (*Esox lucius*); regional interpolation based on over 5000 individual pike data from over 1000 lakes. Right: Mercury in the mor layer of forest soils; standardised concentrations expressed per unit of organic matter; regional interpolation based on data from 356 standardised sampling points. Both maps are currently being updated.

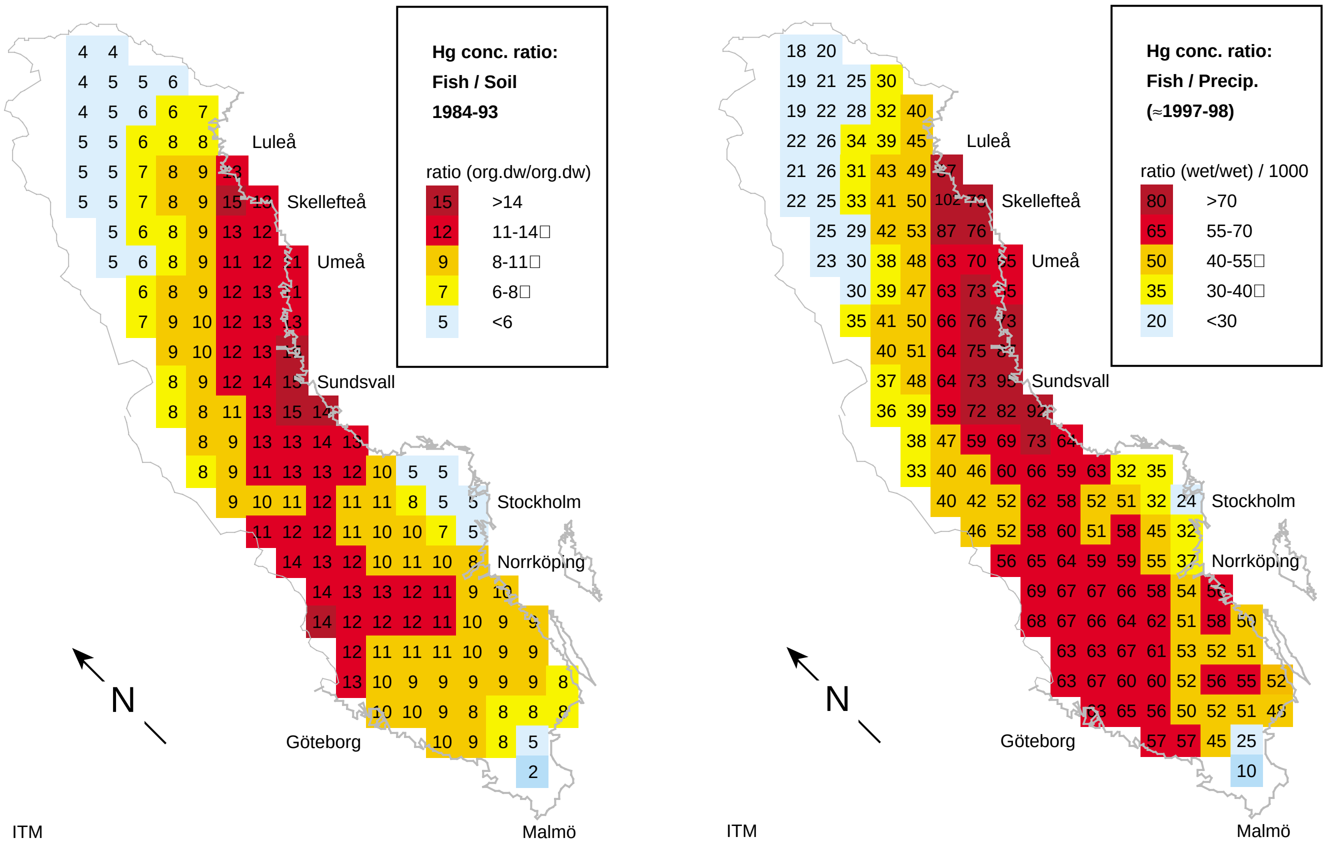

Figure 4. Mercury concentration ratios between fish and soil (left) and between fish and precipitation (right), based on conversions of the data in Fig. 3.

The work for mapping mercury levels and controlling environmental factors is based on a large amount of data that were kindly provided by Tord Andersson (Dept. of Ecology and Environmental Science, Umeå University), Håkan Blomgren (Swedish Environmental Research Institute (IVL), Gothenburg), Kjell Johansson (Dept. of Environmental Assessment, Swedish Univ. of Agriculture), John Munthe (Swedish Environmental Research Institute (IVL), Gothenburg), Josef Pacyna (Norwegian Institute for Air Research, (NILU), Kjeller, Norway), and EMEP (www.msceast.org).

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Part III. National Focal Centre Reports

This part consists of reports on national input data and critical load calculations submitted to the Coordination Center for Effects (CCE) by National Focal Centres (NFCs). A total of 24 countries now collaborate in the ICP Mapping by submitting critical loads data and related information to the CCE. Following the latest call for national critical loads data, 19 countries (Austria, Belgium, Bulgaria, Croatia, Czech Republic, Denmark, Estonia, Finland, Germany, Hungary, Ireland, Italy, Netherlands, Norway, Poland, Slovakia, Sweden, Switzerland and United Kingdom) submitted updates of their critical load databases. Three countries (France, Russian Federation and Spain) confirmed that they still consider the critical load data submitted earlier as valid (see also Chapter 2 in Part I). No reply was received from the NFCs of two countries (Belarus and Republic of Moldova), and their previously submitted data bases were retained unchanged.

In contrast to earlier reports, NFCs were asked to focus in their national contributions on describing their critical load databases and documenting the methods by which critical loads were calculated, especially where they differ from the Mapping Manual. The reports received were edited for clarity and format, but have not been further reviewed.

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Introduction

Critical loads data for Austria have been updated recently. This update was necessary due to inconsistency in the old set of data and to new findings concerning the quantities of some input parameters. The methodology used has not changed and is based mainly on the recommendations in the Mapping Manual (UBA 1996).

Unless otherwise mentioned in the text, a grid size of 2.75×2.75 km^2 was used for all mapping activities.

Receptors

Critical loads of acidity were mapped for one receptor, forest soils. Both coniferous and deciduous forests were included in the mapping. Information on land use was derived from:

- The Austrian Atlas of the Academy of Science, revised by NOAA / AVHRR satellite data.
- Forest data from the Austrian forest inventory.
- Forest ecological areas.

Mapping of soil types was based on:

- The inventory of soil types in Austria based on Fink.
- Geological data from the Hydrogeological Map of Austria (Austrian Academy of Science).

Mapping of critical loads for eutrophication included the following receptors:

- Forest soils: see above.
- Bogs: spatial data derived from the Catalogue of Austrian bogs; see also: www.ubavie.gv.at:/umweltsituation/natur/konventionen/alpen/abis/moore.htm.
- Alpine grassland: spatial data derived from the Austrian Atlas of the Academy of Science, revised by NOAA / AVHRR satellite data.

A. Critical Loads for Acidity

Critical loads for acidity have been calculated using the Simple Mass Balance (SMB) method as described in the Mapping Manual (UBA 1996).

Data sources

BC uptake: Biomass uptake was estimated primarily from data on the element content of dominant tree species for coniferous and deciduous forests and on biomass production. Due to recent results on forest management conditions in alpine regions and their implications on nutrient balance (Herman et al. 1997, Smidt et al. 1997, Knoflacher and Loibl 1998b, Knoflacher et al. 1995, Knoflacher and Gebetsroither 2000, Herman et al. 2001), regional differences in harvesting practices were considered, in particular to calculate uptake rates. Different harvesting rates in protected and unprotected forests (also in relation to geomorphological dynamics) were considered to be average rates in relation to regional management conditions and altitude classes, based on data of the Austrian forest survey 1992–96 (Schieler 2000).

A problem in Austrian forests is the low regeneration rates caused by site-specific discontinuities in management practices. To avoid a transformation of this population dynamic risk factor, average values were used to compensate for the site-specific increases during harvesting periods.

BC deposition: Base cation deposition has been estimated using an empirical interpolation model, starting from recent measurement data from background monitoring sites.

Q (runoff): The spatial distribution of runoff rates is based on a calculation using the following empirical formula:

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

where:

P = precipitation (mm); derived from a digital precipitation map

H = altitude (m); derived from a digital elevation map

NK = slope correction factor

a = constant factor

ANC_{we}: Weathering rates were derived for different soil types from the literature. In order to reflect the very thin organic soil layers in high alpine regions with very little contact to the bedrock material, a very slow weathering rate was assumed for soil types at altitudes above 1500 m a.s.l. The soil types were taken from an Austrian soil map produced by the Austrian Research Centre Seibersdorf.

K_{gibb}: A default value of 300 m^6 eq^{-2} was chosen in agreement with the Mapping Manual.

BC_{le}: Base cation leaching was calculated taking into account site-specific soil types.

B. Critical Loads for Eutrophication

For forest ecosystems, critical loads for nutrient nitrogen were calculated according to the Mapping Manual.

N_u: Nitrogen uptake was estimated from BC uptake taking into consideration typical Austrian BC:N ratios.

N_i: According to the Mapping Manual, a default value of 1 kg N ha^{-1} yr^{-1} was used.

N_{le}: Nitrogen leaching was determined by expert judgement based on soil characteristics and tree type. The values vary between 2–5 kg ha^{-1} yr^{-1}.

f_{de}: Denitrification has been related to the soil type on the basis of data cited in the Mapping Manual.
For bogs and Alpine grasslands, empirical critical loads values were taken, derived from recommendations from international workshops.

Uncertainties in critical load calculations

The uncertainty of the data used to calculate critical loads for Austria has been assessed. In general, all values delivered to the CCE in the official Austrian critical loads data set are "best guesses" of the true values. The uncertainty of the different values were estimated as follows:

a. Base cation deposition (BC_{dep}): Base cation deposition was calculated from monitoring data by applying a multifactorial interpolation model. Related to the spatial scale of the grids, an average variance of ± 30% in relation to the measured data was achieved.
b. Base cation uptake (BC_u): Base cation uptake for forests was calculated considering common harvesting practices and the fact that no harvesting occurs on 15% of the forest area. The average variance is ±20% at the spatial scale of the grids.
c. Weathering of base cations (ANC_w): Weathering of base cations is based on estimation by experts with extensive experience in Austrian soil conditions. A variance of ± 40% is expected at the spatial scale of the grids.
d. Runoff (Q): The accuracy of runoff in mountainous areas is influenced by the accuracy of data on precipitation, evapotranspiration and physical soil characteristics. Precipitation data are based on monitoring data and were calculated by a multifactorial interpolation model. The long-term uncertainty of this data is estimated at ± 30%. A problem for the calculation of Q is the lack of survey data on soil physical characteristics, so a variance of ± 50% is expected at the spatial scale of the grids.
e. Nitrogen immobilisation (N_i): The default value from the Mapping Manual was applied. Because no long-term observation data is available, the uncertainty of this parameter cannot be determined.
f. Nitrogen uptake (N_u): In the calculations, a linear relationship to BC_u was assumed. Therefore, the uncertainty of N_u is believed to be similar to that of BC_u.

g. Denitrification (f_{de}): Data analyses of field investigations in alpine forest ecosystems indicate that the annual denitrification rate is rather independent from nitrogen deposition, and is highly variable from year to year. Values were determined according to different soil types and information derived from literature. Currently it is not possible to define accurately the uncertainty of the data.

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Status of critical loads data

No response was received to the most recent call for data from the CCE. Thus the 1998 critical loads database has been adapted to the new EMEP co-ordinate system by the CCE, and has been included into the European database. For a description of the national data, see the NFC report in the CCE Status Report 1999.

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National maps produced

National maps were generated by combining the contributions of Flanders (northern Belgium) and Wallonia (southern Belgium). The methodologies used to estimate various parameters differed between the two regions depending on the data available. Maps have been produced for coniferous, deciduous and mixed forests in both Wallonia and Flanders, and also for lakes in Wallonia.

Mapping procedure

Digitised maps with a total of 2532 ecosystems (652 in Flanders; 1880 in Wallonia) were overlaid by 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; Eloy 2000, SITEREM 2001). As the number of forest ecosystems per grid cell was rather small in Flanders, the lowest critical load was attributed to the entire cell.

A. Forest Soils

Calculation methods

Critical loads for forest soils were calculated using the Simple Mass Balance (SMB) method as described in the Mapping Manual (UBA 1996):

$CL(A_{ac}) = ANC_w - ANC_{le(crit)}$
$CL(A_{pot}) = ANC_w - ANC_{le(crit)} - BC_u + N_i + N_{de}$
$CL_{max}(S) = CL(A_{ac}) + BC_{dep} - BC_u$
$CL_{max}(N) = N_i + N_u + CL_{max}(S)$
$CL_{nut}(N) = N_i + N_u + N_{le} + N_{de}$

$ANC_{le(crit)} = -PS\ ([Al^{3+}]_{crit} + [H]_{crit})$ for Flanders
$ANC_{le(crit)} = -PS\ (0.2 \text{ eq m}^{-3} - [Al^{3+}]_{estimated})$ for Wallonia

where:
$[Al^{3+}]_{estimated} = K\,[H^+]_{measured\ in\ soil\ solution}$

Two criteria were used to determine the critical Al^{3+} concentration:

1. *The equilibrium* $K = [Al^{3+}]/[H^+]^3$ *criterion (in Wallonia):* The Al^{3+} concentation was estimated by (1) experimental speciation of soil solutions to measure quickly reacting aluminium, Al_{qr} (Clarke et al. 1992) ; and (2) calculation of Al^{3+} concentation from Al_{qr} using the SPECIES speciation software. The K values established for 10 representative Walloon forest soils (Table BE-1) were more relevant than the gibbsite equilibrium constant recommended in the Mapping Manual (UBA 1996). The difference between the estimated Al^{3+} concentrations and the concentrations that cause damage to root systems (0.2 eq Al^{3+} m^{-3}, from De Vries et al. 1994) gives the remaining capacity of the soil to neutralise the acidity.
2. *The Al:Ca ratio criterion (in Flanders):*
   $[Al^{3+}]_{crit} = RAl/Ca \cdot Bc_u\ /PS$
   $Bc_{u(crit)} = BC_{dep} + ANC_w - PS\ [Bc]_{crit}$

Tables BE-1 and BE-2 summarise the values used for some of the parameters.

Table BE–1. Aluminium equilibrium and weathering rates calculated for Walloon soils.

Site	Soil type	*K*	ANC_w (eq ha^{-1} yr^{-1})
Bande (1-2)	Podzol	140	610
Chimay (1)	Cambisol	414	1443
Eupen (1)	Cambisol	2438	2057
Eupen (2)	Cambisol	25	852
Hotton (1)	Cambisol	2736	4366
Louvain-la-Neuve (1)	Luvisol	656	638
Meix-dvt-Virton (1)	Cambisol	2329	467
Ruette (1)	Cambisol	5335	3531
Transinne (1)	Cambisol	3525	560
Willerzie (2)	Cambisol	2553	596

(1) deciduous or (2) coniferous forest

Table BE-2. Constants used in critical loads calculations.

Parameter	Value	Reference
N_i (Flanders)		
(Wallonia)	213 eq ha^{-1} yr^{-1}	Posch et al. 1995
	36 eq ha^{-1} yr^{-1}	UBA 1996
$N_{le\ (acc)}$	260–640 eq ha^{-1} yr^{-1}	
N_{de}	fraction of ($N_{dep} - N_i - N_u$)	
RAl/Ca	1 eq eq^{-1}	Boxman et al. 1988
$[Bc]_{crit}$	0.01 eq m^{-3}	

Data sources

Soils: In *Flanders*, the Flemish inventory of soil profiles “Aardewerk” 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 distinguished according to the soil associations map of the Wallonia (Maréchal and Tavernier 1970). Each ecosystem is characterised by a soil type and a forest type.

Weathering rate: In *Flanders*, in the absence of more specific data, the 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).

In *Wallonia*, the base cation weathering rates (ANC_w) were estimated for 10 different representative soil types (Table BE-1) through leaching experiments. Increasing inputs of acid were added to soil columns and the cumulated outputs of lixiviated base cations were measured. Polynomial functions were used to describe the input-output relationship. To estimate ANC_w, acid inputs were fixed at 900 eq ha^{-1} yr^{-1} in order to keep a long-term balance of base content in soils.

Precipitation surplus: In *Flanders*, the precipitation surplus was calculated as the precipitation minus the sum of interception by the forest canopy and evapotranspiration. Data on mean annual precipitation were derived from precipitation data measured at 5 climate stations in Flanders over a period of 10 years (1986–1995). The value for each ecosystem was set equal to the value measured 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 region corresponded to a fraction of the normal precipitation in the area. Precipitation data were derived from the map of average precipitation measured in Belgium between 1833–1975 by the national weather service (Dupriez and Sneyers 1979). The fraction of precipitation giving rise to the surplus, equal to 0.4, was calculated by establishing the water balance in five forested catchments located in Wallonia.

Net growth uptake of base cations and nitrogen: In *Wallonia*, the net nutrient uptake (equal to the removal in harvested biomass) was calculated using average growth rates measured in 25 Walloon ecological territories and the chemical composition of coniferous and deciduous trees. The chemical composition of the trees appears to be linked to the soil type (acidic or calcareous) (Duvigneaud et al. 1969, Bosman et al. 2001).

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).

The net growth uptake of nitrogen ranges between 245 and 670 eq ha^{-1} yr^{-1}, while base cation uptake values vary between 130 and 855 eq ha^{-1} yr^{-1} depending on the tree species and location in Belgium.

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

In *Wallonia*, actual throughfall data collected at eight sites between 1992 and 2000 were used to estimate BC_{dep}. The marine contribution to Ca^{2+}, Mg^{2+} and K^+ deposition was estimated using sodium deposition according to the method described in UBA (1996). BC_{dep} data from the eight sites were extrapolated to all Walloon ecosystems as a function of the location and the tree species.

Results

In Wallonia, the highest $CL(Ac)$ values were found in calcareous soils under deciduous or coniferous forests. The measured release rate of base cations from soil weathering processes is high in these areas, and thus provides a high long-term buffering capacity against soil acidification.

More sensitive forest ecosystems are met on sandy-loam or loamy gravelly soils. The lowest $CL_{nut}(N)$ values were found in the Ardennes. In this zone, *Picea abies L. Karts.* stands frequently show magnesium deficiency symptoms, which have been exacerbated by atmospheric pollution (Weissen et al. 1990).

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).

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:

$$CL_{(Ac)} = ([BC]_0 - [ANC]_{lim}) \cdot Q$$
$$CL_{(emp)} = [Ca^{2+}]_0 \; / \; 89$$

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

Data sources

- The desired ANC threshold which ensures no damage to biological indicators has been established as 20 µeq l^{-1} for all lakes, according to UBA (1996).
- Runoff (Q) values correspond to the flow in the river which supplies the lake and were derived from 1995 to 1999 monitoring data.
- Base cation and nitrogen uptake by forest ecosystems were estimated using the methodology described above.
- The base cation deposition was calculated using 1999 data supplied by the wet monitoring network. Corrections for the marine contribution have been made on the basis of the sodium concentration according to the method described in UBA (1996).
- The original freshwater calcium concentration was calculated using monitoring data collected from 1995–2000 and relationships established in the French Ardennes (Fevrier 1996).

Results

The Gileppe and Eupen lakes are located in the Belgian High Ardennes. The catchments of these two lakes have 73% and 79% respectively, of their area covered by forest, while the rest of area is covered by fens. The catchment of Ry de Rome lakes is 99% forested, while Nisramont and Eau d'Heure catchment are 40% forested. Bütgenbach and Roberville are 25% forested, while the rest of the area consist of urban or agricultural zones (Marneffe et al. 1997). To estimate exceedances of acidic deposition, the direct contribution to surface waters of urban area and agriculture were subtracted.

The equation of exceedances is defined as:

$$Ex(Ac) = S^*_{dep} + (N_{leach} - N_{urban/agricult}) - CL(Ac)$$

Critical loads are the lowest for Eupen lake as a result of naturally acidic water leached from the catchment, while the highest values, at Eau d'Heure lakes, are due to high calcium concentrations.

Table BE-3 summarises the critical load values for lakes. Three lakes are sensitive and acidic pollutants could modify the equilibrium of these three oligotrophic lakes (SITEREM, 2001).

Table BE-3. Critical loads of acidity and nitrogen (using SSWC, FAB and empirical methods) and exceedances (in eq ha^{-1} yr^{-1}).

	***CL*(*Ac*)**		***CL*$_{max}$(*N*)**	***Ex*(*Ac*)**
Lakes	**SSWC**	**Empir.**	**FAB**	**SSWC**
Eupen	80	74	1030	1517
Gileppe	461	414	2920	1113
Ry de Rome	1224	1104	6880	25
Roberville	2169	1652	13650	-212
Bütgenbach	2270	1898	13720	-598
Nisramont	3008	3921	15610	-1602
Eau d'Heure	11750	21567	28840	-9701

Conclusions

The values of some parameters appear to 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 SMB method presents difficulties. In Wallonia, monitoring of forests is more intensive due to their economic importance, and the variability of the soil types can be addressed adequately.

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

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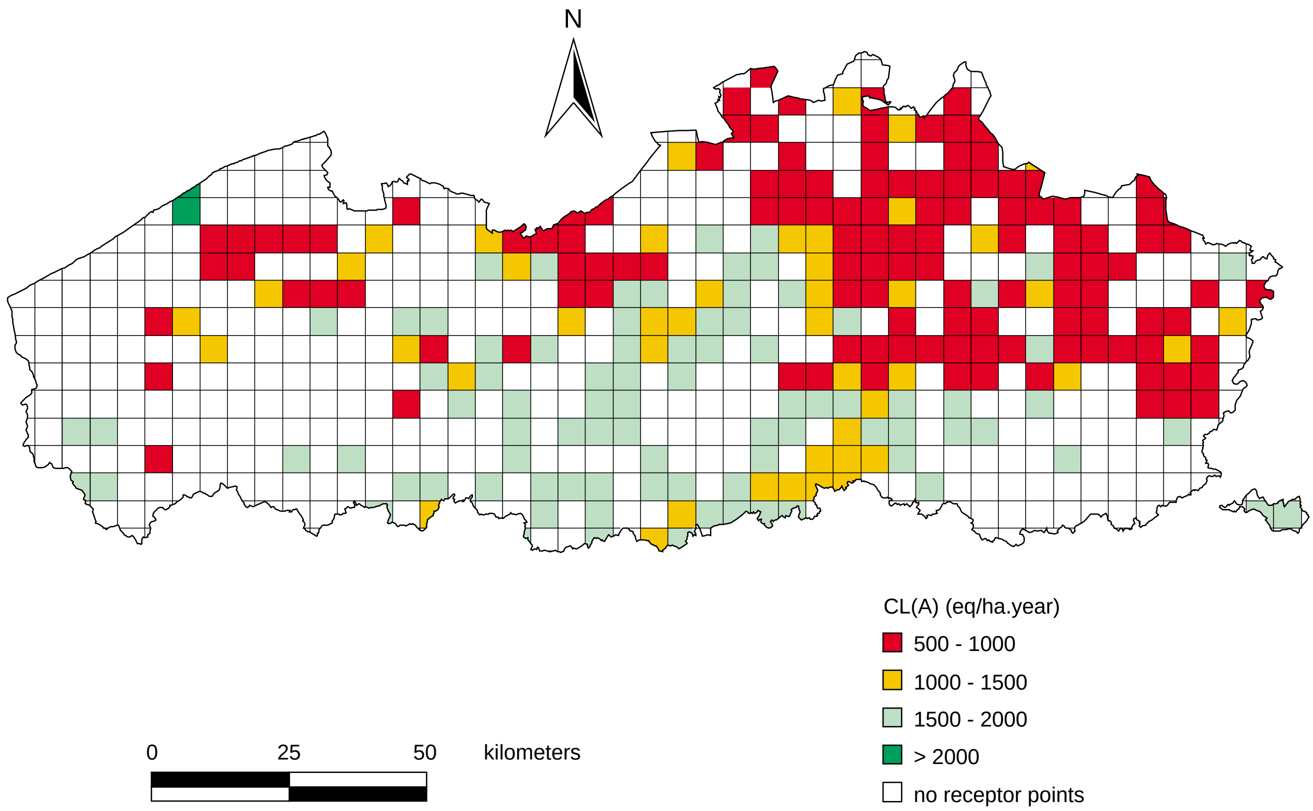

Figure BE-1. Critical loads of actual acidity, *CL(Ac)*, for forest ecosystems in Flanders.

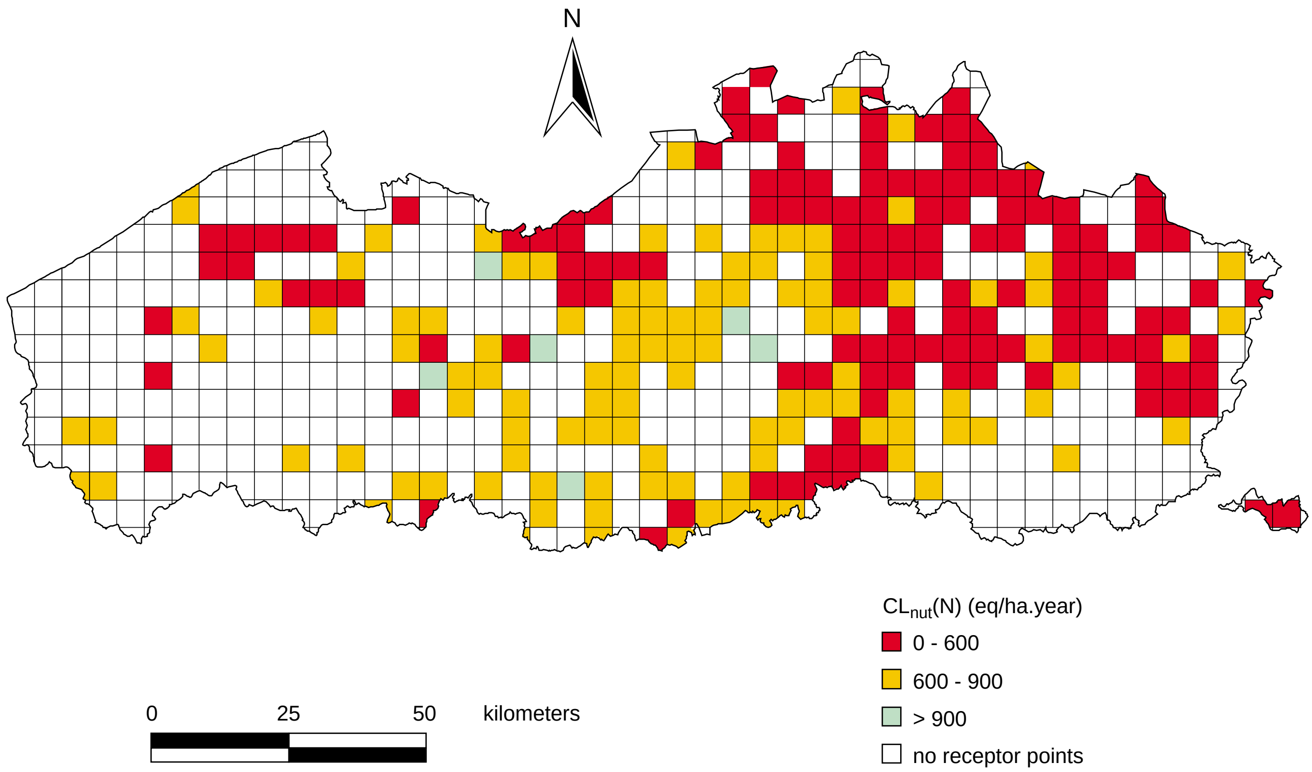

Figure BE-2. Critical loads of nutrient nitrogen, $CL_{nut}(N)$, for forest ecosystems in Flanders.

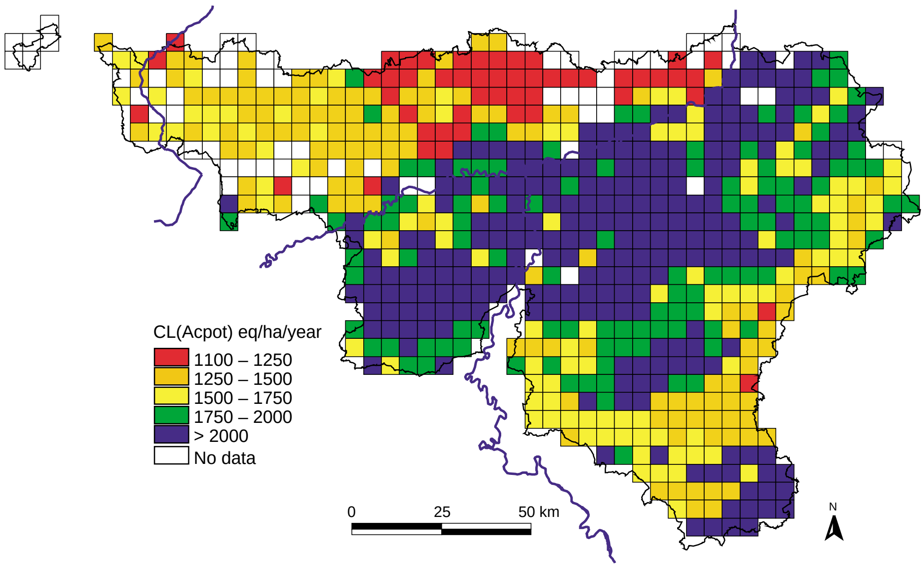

Figure BE-3. Critical loads of potential acidity, $CL(Ac_{pot})$, for forest soils and lakes in Wallonia.

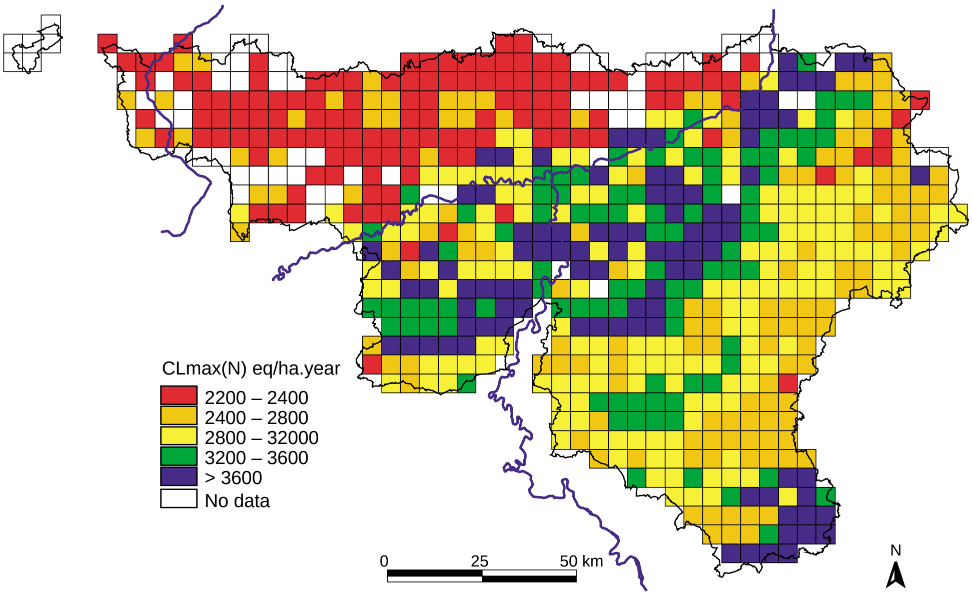

Figure BE-4. Maximum critical loads of nitrogen, $CL_{max}(N)$, for forest soils and lakes in Wallonia.

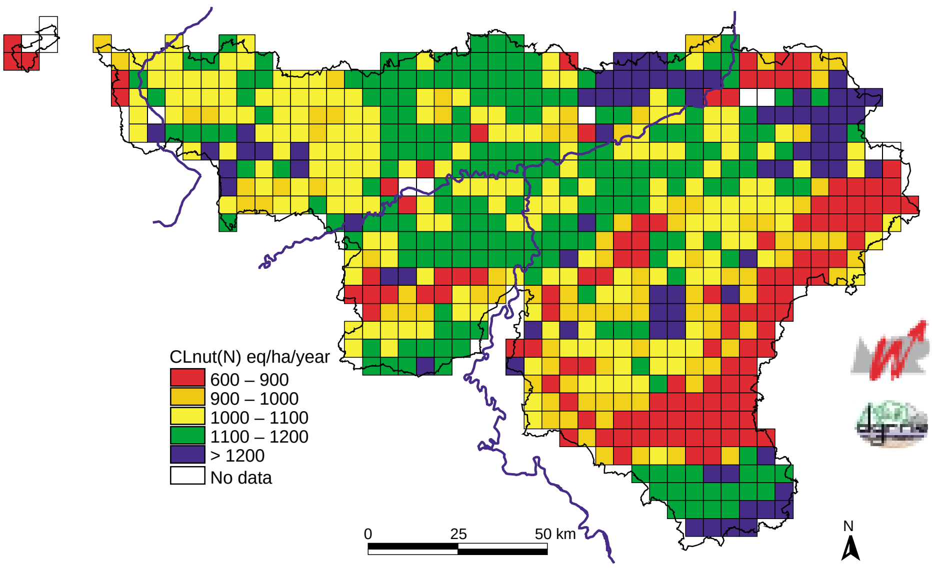

Figure BE-5. Critical loads of nutrient nitrogen, $CL_{nut}(N)$, for forest soils and lakes in Wallonia.

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National maps produced

Critical loads have been calculated and mapped for both deciduous and coniferous forests using the Steady-state Mass Balance method. Critical loads of acidity, sulphur and nitrogen is based on 208 forest soil receptor points, for which results are then processed for the EMEP50 grid. Data submitted to the CCE consists of one critical load record each for deciduous and coniferous forests for every EMEP50 grid cell that contains Bulgarian territory. The following critical loads have been calculated and mapped:

- Deposition of sulphur, base cations and nitrogen.
- Base cation weathering.
- Nitrogen and base cations uptake by the biomass.
- Critical load for acidity, sulphur and nitrogen for coniferous and deciduous forests.
- Maximum critical loads of sulphur for coniferous and deciduous forests.
- Minimum and maximum critical loads of nitrogen for coniferous and deciduous forests.
- Critical loads of nutrient nitrogen for coniferous and deciduous forests.

Calculation methods

1. Critical ANC leaching:

$ANC_{le(crit)} = Al_{le(crit)} + H_{le(crit)}$
$Al_{le(crit)} = R\ \mathrm{Al{:}BC}\ (BC_{dep} + BC_w - BC_u)$
$H_{le(crit)} = Q \cdot [\mathrm{H}]_{crit}$

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}

2. Critical loads of acidity:

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). Critical loads of acidity have been calculated according to the following equations:

$$CL(A) = BC_w + Q \cdot [\mathrm{H}^+]_{crit} + R\mathrm{Al/Ca}\ (BC^*_{dep} + BC_w - BC_u)$$
$$= 2.5\ BC_w + 0.09\ Q + 1.5\ BC_{dep} - 1.5\ BC_u$$

where:
$CL(A)$ = critical load of acidity
BC_w = weathering of base cations
Q = annual runoff of water under root zone, $m^3\ ha^{-1}\ yr^{-1}$
$[H^+]_{crit}$ = critical concentration of protons (= 0.09 eq m^{-3} which corresponds to pH 4.0) (Hettelingh and de Vries 1992).
$R\text{Al}/\text{Ca}$ = critical Al:Ca ratio (= 1.5 eq eq^{-1}) (UBA 1996)
BC^*_{dep} = atmospheric deposition of base cations ($BC_{dep} - Cl_{dep}$), eq $ha^{-1}\ yr^{-1}$
BC_u = net growth uptake of base cations, eq $ha^{-1}\ yr^{-1}$

3. Maximum and minimum critical loads of sulphur 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, eq $ha^{-1}\ yr^{-1}$
N_i = nitrogen immobilisation

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

4. 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 forests = 0.0143 eq m^{-3}; for deciduous = 0.0215 eq m^{-3}; Posch et al. 1995)

Data sources

1. National monitoring data: Critical loads have been calculated for all major tree species (using the soil data base) in 16×16 km^2 grid cells. A total of 208 forest soil profiles have measured values. Runoff of water under the root zone has been measured on a 10×10 km^2 grid for the entire country. A network of 12 stations which measure atmospheric deposition in precipitation and 75 measurement points of air pollutant concentrations have been used for base cations, sulphur and nitrogen deposition (Ignatova 1994, 1995; Ignatova et al. 1997, 1998). Net uptake rates of nitrogen and base cations were obtained by multiplying the element content of the stems (N, Ca, K, Mg and Na) by annual harvesting rates (Ignatova et al. 1997).

2. National maps:

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

3. Other input variables: In the absence of specific data on weathering rates for most study regions, data were calculated according to the dominant parent material (from the lithology map of Bulgaria) and the texture class (from the FAO soil map for Europe), according to the clay content of Bulgarian forest soils (UBA 1996).

In contrast to a previous evaluation of critical loads in which the weathering rate varied between 250 and 1000 eq $ha^{-1}\ yr^{-1}$ (Rihm 1994), these calculations are based on the assessment of weathering rates for the depth of 50 cm (between 125 and 1375 eq $ha^{-1}\ yr^{-1}$) derived from soil types and texture classes according to the clay content in Bulgarian soils (UBA 1996).

The gibbsite equilibrium constant K_{gibb}, was estimated in accordance with the percentage of soil organic matter and soil type using the Mapping Manual (UBA 1996).

Results, comments and conclusions

All the data necessary to calculate critical loads have been stored in Excel tables and mapped on the EMEP 50×50 km^2 grid. Values for each parameter and the resulting critical loads are stored for each forest type (coniferous and deciduous) in separate records for each EMEP 50×50 km^2 grid cell, and also averaged for each EMEP50 grid cell when the forest is a mixture of both tree types, in accordance to the area fractions of the tree species.

Figures BG-1 to BG-4 show individual values for $CL_{max}(S)$, $CL_{max}(N)$, $CL_{min}(N)$ and $CL_{nut}(N)$ for each EMEP50 grid cell for deciduous forests. The frequency distribution of the values for both deciduous and coniferous forests is shown in Table BG-1.

Comparing the average values of the maximum and minimum critical loads for nitrogen, the maximum critical loads for sulphur, and critical loads for nutrient nitrogen for coniferous and deciduous forest types throughout Bulgaria, shown in Figure BG-5 (left), it becomes evident that the differences in the maxium critical loads for sulphur and nitrogen are insignificant. In contrast, the critical loads for nutrient nitrogen are about 30 percent lower for deciduous species compared to those of coniferous forests.

Calculated values for $CL_{max}(S)$ vary between 3053 and 11,598 eq ha^{-1} yr^{-1} for coniferous species, and between 3354 and 13,976 eq ha^{-1} yr^{-1} for deciduous. The values for $CL_{max}(N)$ are similar (between 3974 and 12,520 eq ha^{-1} yr^{-1} for coniferous forests, and between 3992 and 14,779 eq ha^{-1} yr^{-1} for deciduous). On the contrary, critical load values for nutrient nitrogen are lower and range between 581 and 941 eq ha^{-1} yr^{-1} for coniferous, and between 398 and 776 eq ha^{-1} yr^{-1} for deciduous forests. The lowest critical loads are calculated for $CL_{min}(N)$ (between 573 and 926 eq ha^{-1} yr^{-1} for coniferous forests, and between 394 and 768 eq ha^{-1} yr^{-1} for deciduous).

In general, critical loads values throughout the country are higher for coniferous forests than for deciduous ones, due to the lower mean values of most parameters used to calculate critical loads (base cation weathering, deposition and uptake) except nitrogen uptake. There are no significant differences between critical ANC leaching for both coniferous and deciduous forests.

The results obtained throughout the country were also compared with the values for the grid cells in which the forest is represented by both coniferous and deciduous types of ecosystems. The values of maximum critical loads for both sulphur and nitrogen for coniferous forests are lower for the entire country than for grid cells where forests are mixed. On the other hand, deciduous forests are less protected in the grid cells with both coniferous and deciduous ecosystems, because their maximum critical loads for both sulphur and nitrogen are lower than for the total country area.

Another interesting case from an ecological point of view is to compare the critical loads for coniferous and deciduous forests within the same grid cell. A comparison of average maximum critical loads of sulphur and nitrogen for coniferous and deciduous forest ecosystems in the same ecological conditions (Fig. BG-5, right) shows remarkable differences of about 800–1100 eq ha^{-1} yr^{-1} due in particular to the higher critical ANC leaching for coniferous forests than for deciduous.

In almost all grid cells, individual critical loads for coniferous forests are higher than those for deciduous ecosystems. Despite the same geographical parameters, the mean value of $CL_{max}(S)$ for coniferous ecosystems is 8038 eq ha^{-1} yr^{-1}, whereas the average value for deciduous forests in the same grid cells is only 7211 eq ha^{-1} yr^{-1}. Similar differences exist in the maximum critical loads of nitrogen (8827 eq ha^{-1} yr^{-1} for coniferous and 7748 eq ha^{-1} yr^{-1} for deciduous forests). For $CL_{min}(N)$ and $CL_{nut}(N)$, the variability of computed individual data is much smaller, which is reflected in the average values: a $CL_{min}(N)$ of 789 eq ha^{-1} yr^{-1} for coniferous ecosystems, 537 eq ha^{-1} yr^{-1} for deciduous; and an average $CL_{nut}(N)$ of 797 eq ha^{-1} yr^{-1} for coniferous and 549 eq ha^{-1} yr^{-1} for deciduous forests.

Table BG-1. Distribution of critical load values in Bulgaria for deciduous and coniferous forests (in %).

Range	***CL(A)***		$CL_{max}(S)$		$CL_{min}(N)$		$CL_{max}(N)$		$CL_{nut}(N)$	
(eq ha^{-1} yr^{-1})	**Dec**	**Con**	**Dec**	**Con**	**Dec**	**Con**	**Dec**	**Con**	**Dec**	**Con**
< 200	0	0	0	0	0	0	0	0	0	0
200–500	0	0	0	0	32.73	0	0	0	21.82	0
500–1000	0	0	0	0	67.27	100	0	0	78.18	100
1000–2000	0	0	0	0	0	0	0	0	0	0
> 2000	100	100	100	100	0	0	100	100	0	0

It could be concluded that the calculated values for sulphur and nitrogen give a good initial indication of the spatial variability of ecosystem sensitivity to acidification in Bulgaria. More care should be taken when computing critical loads separately for coniferous and deciduous ecosystems, using only parameters determined by field measurements, especially throughfall deposition of base cations and nutrient uptake by biomass.

Taking into consideration that deciduous forest ecosystems occupy two and a half times more area in the country than do coniferous forests, and that critical loads for deciduous forests are much lower than those for coniferous ones at similar geographical and climatic parameters, the deciduous ecosystems cannot be protected enough when using average weighted values, which are higher than the individual critical loads for the deciduous forests in a given EMEP grid cell. It would be more correct to use the lower value of both individual critical loads for the coniferous and deciduous forests, obtained using individual basic parameters for both types of ecosystems in the same EMEP50 grid cell.

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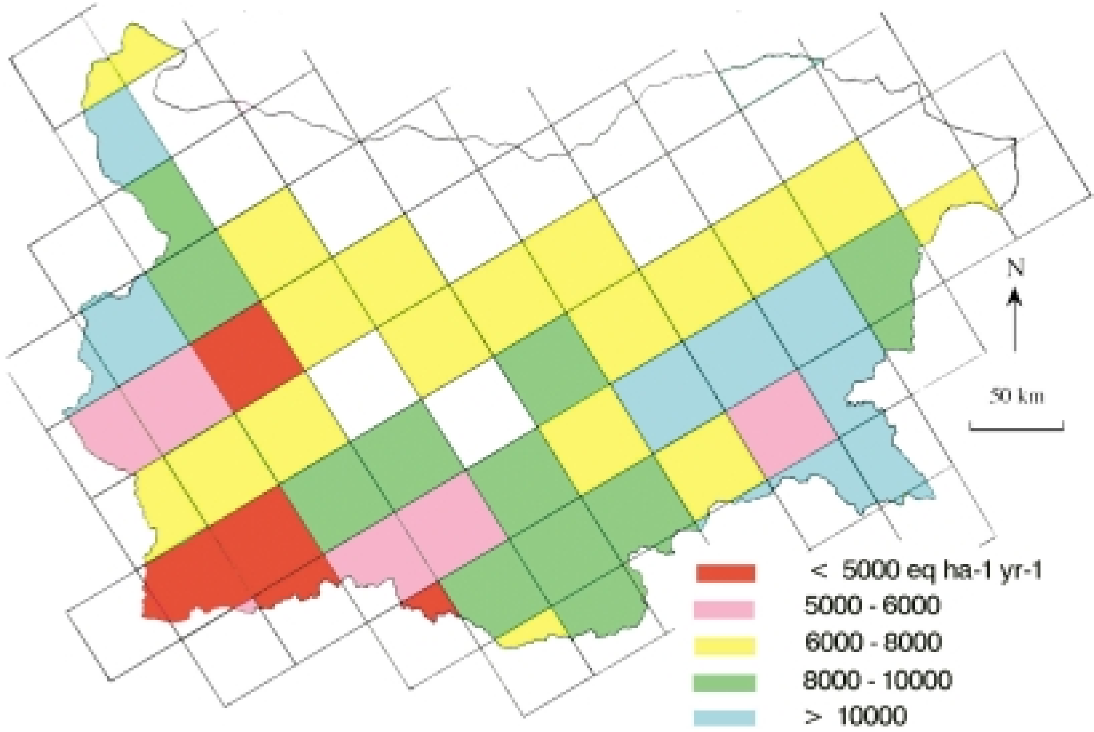

Figure BG-1. Maximum critical loads of sulphur for deciduous forests in Bulgaria.

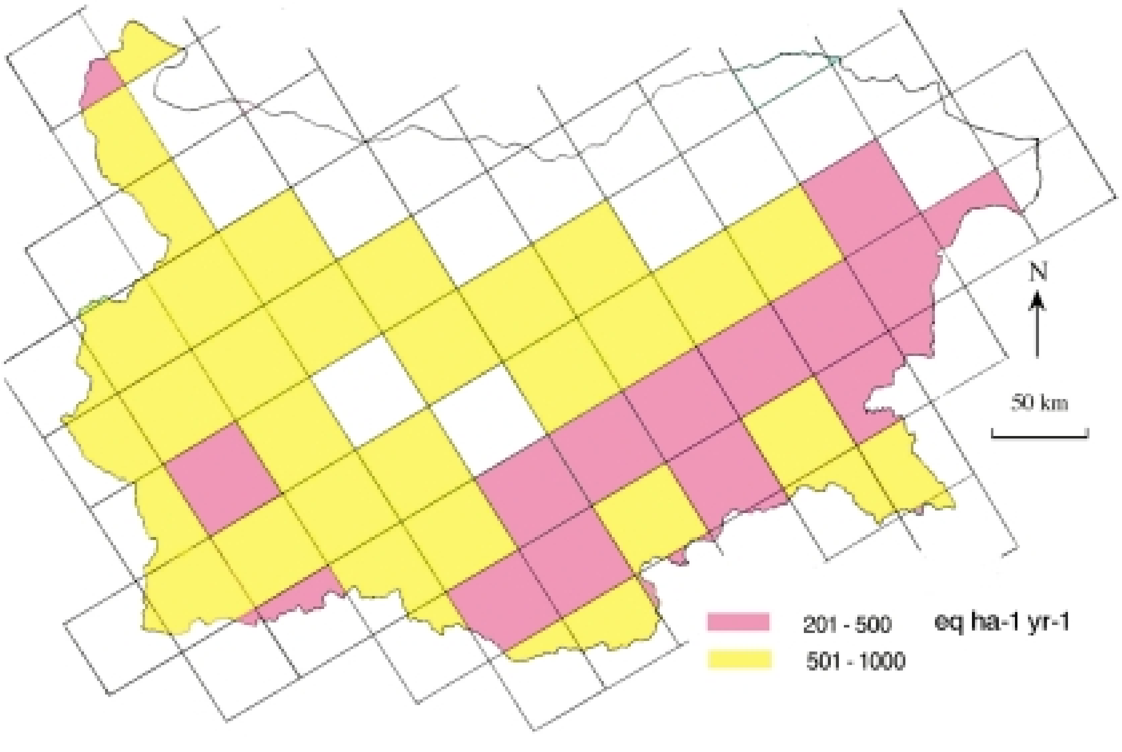

Figure BG-2. Minimum critical loads of nitrogen for deciduous forests in Bulgaria.

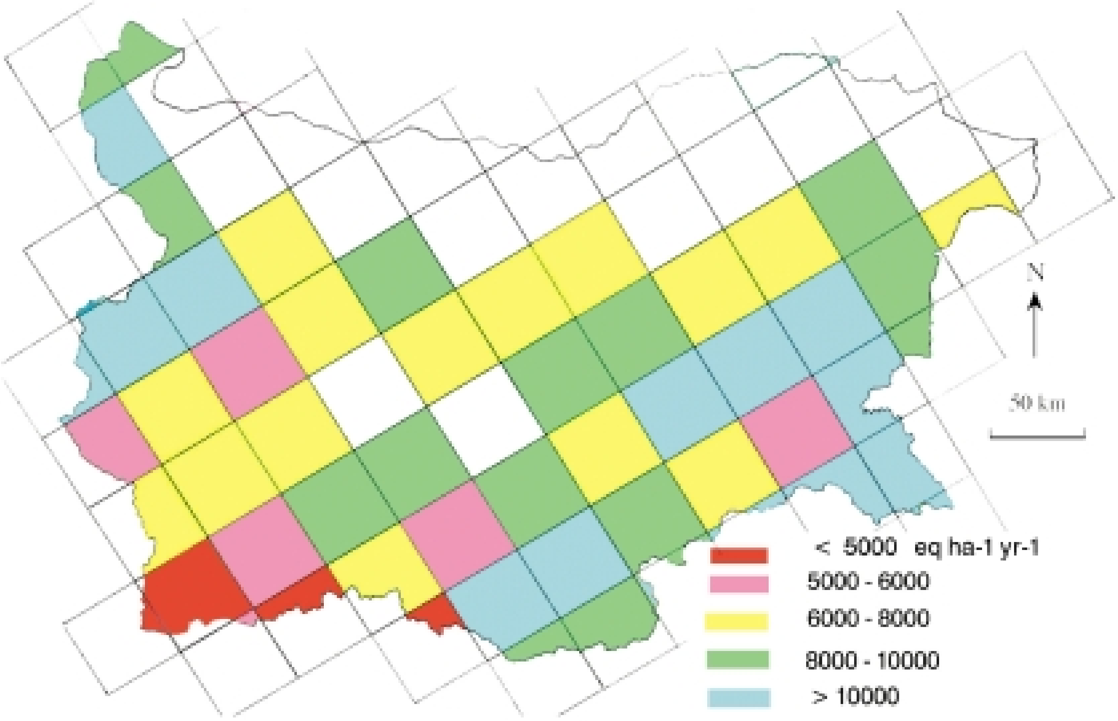

Figure BG-3. Maximum critical loads of nitrogen for deciduous forests in Bulgaria.

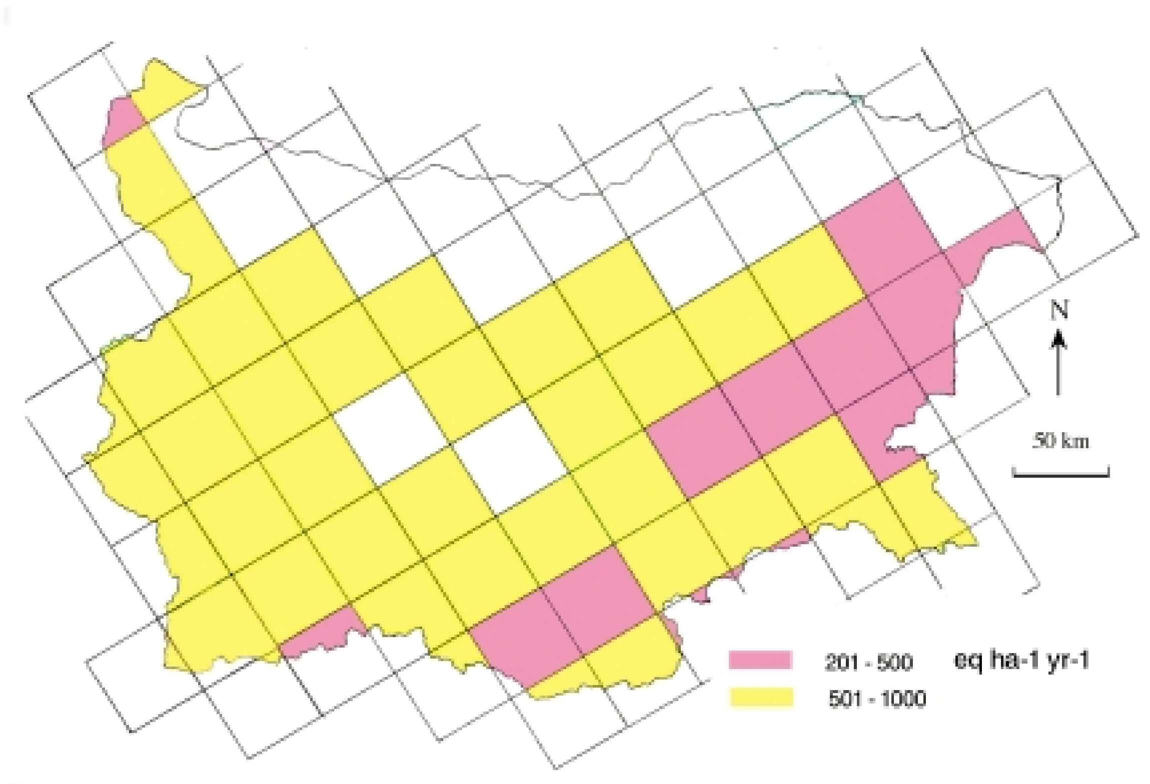

Figure BG-4. Critical loads of nutrient nitrogen for deciduous forests in Bulgaria.

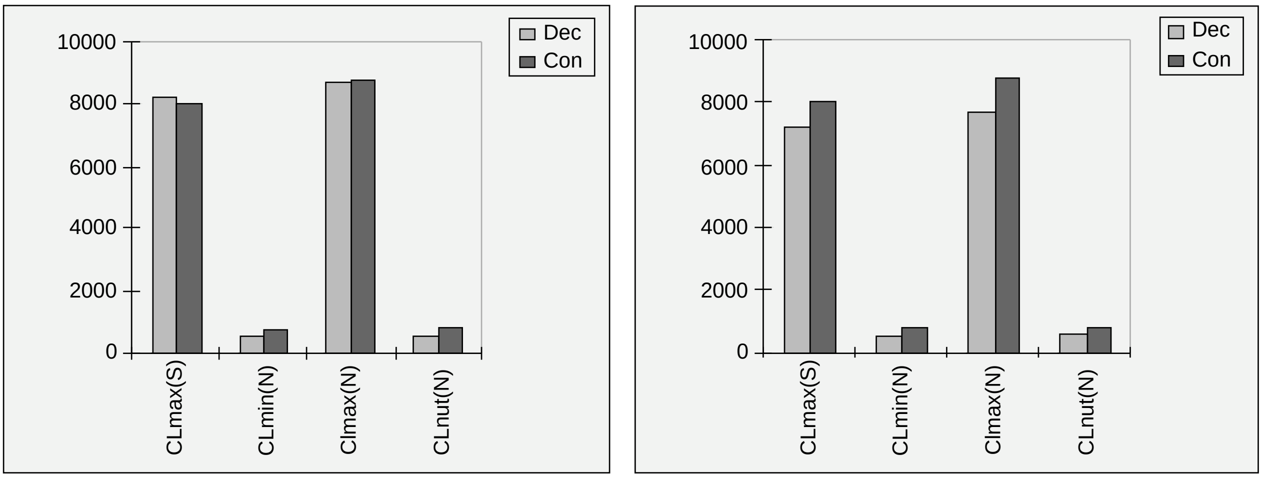

Figure BG-5. Mean values of critical loads for coniferous and deciduous forests for the entire country (left) and for only those grid cells with both coniferous and deciduous forests (right).

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National maps produced

Computation and mapping of critical loads have been started in Croatia with two EMEP50 grid cells (numbers 114,54 and 115,54). 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 EMEP50 grid cells (115,56 and 115,57) in an inland region (the north-western part of Croatia) dominated by deciduous forests (beech and oak), have been mapped. 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 (Fig. 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 select to extend the existing soil data base (Martinović et al. 1998).

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 1981–1994) and eight stations from the NWPC region (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): Measurement data,

$Q = (P - I) \cdot 0.15$.

***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 (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):

pH Range	$[Al]_{crit}$ ($mol_c m^{-3}$)	$[H]_{crit}$ ($mol_c m^{-3}$)
pH>4.0	0.2	0.1
pH<4.0	0.4	0.2

Interception: The mean interception I has been calculated as a function of precipitation P where $I = a \cdot P$. Values for a from De Vries (1991) were used: 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 m^3 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 content 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 utilisation).

Critical acceptable 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})
Pine	0.1
spruce and fir	0.15
beech and fir	0.25
Beech	0.30
Oak	0.35
Ash	0.35
poplar and willow	0.30

Nitrogen immobilisation: The range of N immobilisation (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 (kg N $ha^{-1}yr^{-1}$)
< 0.40	2
0.40–0.50	3
0.50–0.60	5
> 0.60	5

Denitrification: has been defined as:

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

Values for the dentrification factor f_{de} have been assigned according to Posch et al. 1993, in the range of 0.3–0.7 for the GK region and 0.3–0.5 for the NWPC region.

Base cation deposition: Bulk deposition data for base cation deposition were extrapolated from nine 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 to use a filtering factor.

Results

Critical loads of sulphur, $CL_{max}(S)$, 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 earlier, 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 352–1324 eq ha^{-1} yr^{-1}. In the NWPC region, $CL_{nut}(N)$ ranges from 626–1453 eq ha^{-1} yr^{-1} .

Comments and conclusions

Calculation and mapping of critical loads started in Croatia with the submission of national data to the CCE in 1997. At present, four EMEP50 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 is considered a very important task since Croatia is 43% covered by forest, its per capita emissions are the lowest in Europe, and its import of transboundary pollution is much higher than its export. Calculation and mapping of critical loads and levels is scheduled for further investigation in the area of east and central Slavonia region, to complete the data for all of Croatia. According to a calculation of transboundary pollution the lowest percentage of ecosystem area protected (less than 10%) from acidifying deposition of sulphur and nitrogen in the year 2010 is projected for the above area (EMEP 1998).

The following offers some comments on national conditions. In the SSMB 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 guidance. The application of the SSMB 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 the wide range of values in Croatia. (This is especially true for clay soils: *terra rossa* is well-drained 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 often the case in other national contributions.
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. the mass proportion of various substances, geographical position of the mapped area).
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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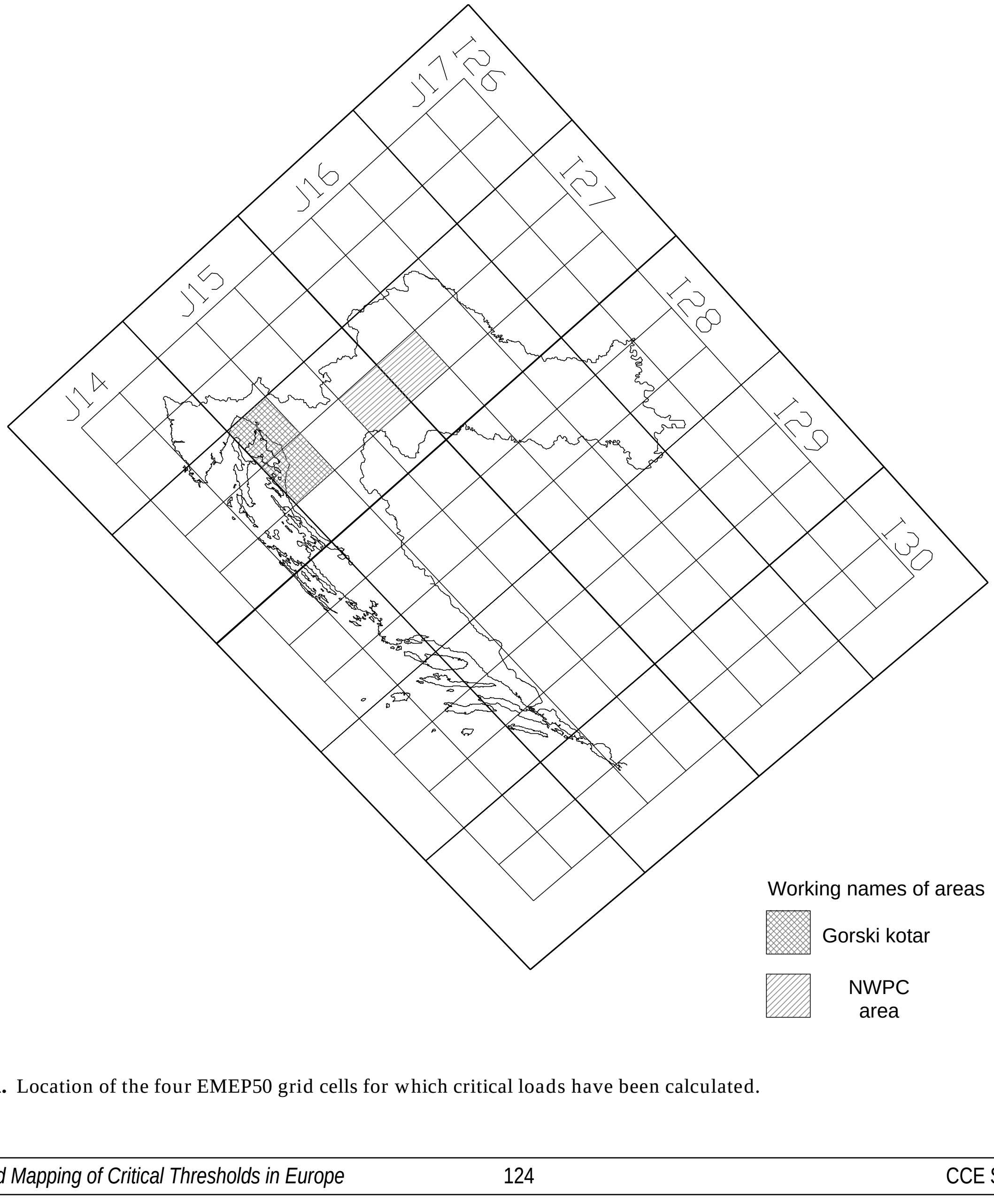

Figure HR-1. Location of the four EMEP50 grid cells for which critical loads have been calculated.

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National maps produced

The national database involves critical loads data and the input variables presented in Table CZ-1. National polygon maps were produced using ArcInfo 7.5 and ArcView 3.2. The following maps were produced:

- Maximum critical loads of sulphur.
- Maximum critical loads of nitrogen.
- Minimum critical loads of nitrogen.
- Critical loads of nutrient nitrogen.

Calculation methods

In the previous database of critical loads no land use / cover classification was applied. A percentage of coniferous (and broadleaf) forests was used to describe forest ecosystems and to determine critical N and BC uptake in the Czech Republic. The usual scale of maps used in the first evaluation of critical loads was 1:1,000,000.

The Czech NFC has now updated the database of critical loads. Land use / cover classifications and an updated hydrology map (averages of specific runoff for the period 1970–1990, scale 1:20,000) and a more detailed map of soil types (1:500,000) have been applied in deriving new critical loads. Three values characterising the critical load function of acidifying S and N have been evaluated: $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$. The critical load preventing eutrophication has been represented as the critical load of nutrient nitrogen: $CL_{nut}(N)$. Land use map classes have been used to describe forest ecosystems. The classes involve deciduous, coniferous and mixed forests including shrubs and / or ground vegetation in forests. In addition the database comprises the assessment of K_{gibb} values for four forest soil layers. The equilibrium constants K_{gibb} have been joined to soil types on the basis of their soil organic matter content derived from chemical analyses of forest soils and types of underlying rocks.

The simple mass balance method summarised in the Mapping Manual (UBA 1996) has been used to calculate critical loads. The calculation of critical loads includes the following equations (a description of symbols used is in Table CZ-1):

$$CL_{max}(S) = BC_w + BC_d - BC_u - ANC_{lec}$$
$$CL_{min}(N) = N_u + N_i$$
$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) \ / \ (1 - f_{de})$$
$$CL_{nut}(N) = N_u + N_i + N_{lec}$$

where:

$$ANC_{lec} = -[H^+]_{le} - [Al^{3+}]_{le}$$
$$[H^+]_{le} = Q \cdot [H^+]_{crit}$$
$$[Al^{3+}]_{le} = Q \cdot [Al^{3+}]_{crit}$$
$$N_{lec} = Q \cdot [N]_{crit}$$

Table CZ-1. Input variables included in the Czech critical load database.

Value	Name	Units
Longitude	Co-ordinate	Decimal degrees
Latitude	Co-ordinate	Decimal degrees
Areakm2	Real area of a polygon	km^2
Forests	Real forest area of a polygon	km^2
EMEP_50	Number i-j of a grid	–
$CL_{max}(S)$	Maximum critical load of S	eq ha^{-1} yr^{-1}
$CL_{min}(N)$	Minimum critical load of N	eq ha^{-1} yr^{-1}
$CL_{max}(N)$	Maximum critical load of N	eq ha^{-1} yr^{-1}
$CL_{nut}(N)$	Critical load of nutrient N	eq ha^{-1} yr^{-1}
BC_d	Base cation deposition	eq ha^{-1} yr^{-1}
BC_u	Base cation uptake	eq ha^{-1} yr^{-1}
BC_w	Weathering rate	eq ha^{-1} yr^{-1}
Q	Specific runoff (draining soils only)	eq ha^{-1} yr^{-1}
K_{gibba}	Gibbsite constant for A_0 layer (ca 0–5 cm)	m^6 eq^{-2}
K_{gibbb}	Gibbsite constant for A layers (ca 5–10 cm)	m^6 eq^{-2}
K_{gibbc}	Gibbsite constant for A / B layer (ca 10–30 cm)	m^6 eq^{-2}
K_{gibbd}	Gibbsite constant for B / C layer (ca 30–50 cm)	m^6 eq^{-2}
ANC_{lec}	Critical ANC leaching	eq ha^{-1} yr^{-1}
N_i	Nitrogen immobilisation	eq ha^{-1} yr^{-1}
N_u	Nitrogen uptake	eq ha^{-1} yr^{-1}
F_{de}	Denitrification	eq ha^{-1} yr^{-1}
N_{lec}	Critical nitrogen leaching	eq ha^{-1} yr^{-1}

Runoff represents the amount of water percolating through the soil profile. Values ranging from 15 to 300 mm yr^{-1} in the Czech Republic represent twenty years of mean values for the period 1970–1990. Critical concentrations of $[H^+]$, $[Al^{3+}]$ and [N] given in Table CZ-2 have been used for calculating critical alkalinity leaching and critical nitrogen leaching.

The critical uptakes of nitrogen and base cations has been derived on the basis of the land use classification map, base cation weathering rates, annual temperature and runoff. Values for N and BC uptake used in the evaluation of critical loads are presented in Table CZ-3. Classes of forest ecosystems can be defined as:

- Broadleaf forests: mainly *Fagus sylvatica, Quercus robur, Quercus petraea, Carpinus betulis.*
- Coniferous forests: mainly *Picea abies, Pinus sylvestris, Larix decidua.*
- Mixed forests: with 50% broadleaf and 50% coniferous forests.
- Shrubs and / or ground vegetation in forests: considered as mixed forests.

Table CZ-2. Parameters used in the calculation of critical loads for various ecosystem types (listed by CORINE class, values in eq ha^{-1} yr^{-1}).

Ecosystem type (CORINE class)	Coniferous (312)	Mixed (313 and 324)	Deciduous (311)
$[N]_{crit}$: Critical N concentration	0.0143	0.02095	0.0276
$[H^+]_{crit}$: Critical proton concentration	0.09	0.09	0.09
$[Al^{3+}]_{crit}$: Critical Al^{+3} concentration	0.2	0.2	0.2

Table CZ-3. Rates (in eq ha^{-1} yr^{-1}) of critical uptake of base cations and nitrogen for coniferous, deciduous and mixed forests for the yield classes.

	Coniferous (312)		Mixed (313 and 324)		Deciduous (311)	
Class	N_u	BC_u	N_u	BC_u	N_u	BC_u
I	607	546	785.5	610.5	964	675
II	464	420	642.5	497.5	821	575
III	357	321	464	360.5	571	400
IV	285	257	392.5	303.5	500	350

Immobilisation rates of nitrogen have been derived from long-term annual temperatures listed in Table CZ-4.

Table CZ-4. Rates of nitrogen immobilisation in forest soils.

Annual temperature (°C)	N immobilisation (kg N ha^{-1} yr^{-1})
5	4.0
6	3.0
7	2.0
8	1.5

The soil type and soil texture determine the rate of base cation weathering (Hettelingh and De Vries 1992). Soil texture characteristics have been used for to derive a denitrification factor f_{de} (Table CZ-5).

Table CZ-5. Denitrification values used for various soil classes.

Clay %	f_{de}
0	0.0
5	0.1
15	0.1
25	0.3
32.5	0.7
52.5	0.7
peat	0.8

The critical loads for sulphur and nitrogen have been compared to actual atmospheric deposition in 1998 and exceedances have been evaluated. Figures CZ-1 and CZ-2 represent critical loads of sulphur of nutrient nitrogen, respectively. Figures CZ-3 and CZ-4 document exceedances of sulphur and nitrogen together and exceedances of nutrient nitrogen only, in 1998.

Data sources

Table CZ-6 below lists various input variables used in the critical load calculations, and their sources.

Comments and conclusions

In comparison to exceedances observed during the first half of the last decade and presented in the CCE Status Report 1997, the absolute values of exceedances has generally decreased throughout the Czech Republic, and the regional number of exceedances has changed significantly. While the highest exceedances were located in the northwest parts of the country at the beginning of the 1990s (i.e. the Kru šné hory and Jizerské hory mountains), the second half of the 1990s typically saw the highest exceedances in the areas of the Orlické hory mountains on the Polish border, and the middle parts of Bohemia and Moravia.

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Table CZ-6. Data sources for critical load calculations.

Map	Scale	Source	Derivation
Soil types	1:500,000	Research Institute of Soil and Water Conservation, Prague	BC_w
Soil texture	1:1,000,000	Czech Geological Institute, Prague	BC_w , f_{de}
Basic runoff	1:200,000	Water Management Institute, Prague	N_{lec} , ANC_{lec}
CORINE map		Czech Ministry of the Environment	N_u , BC_u
EMEP	150×150 km	EMEP programme	BC_d
Map of temperatures	1:1,000,000	Czech Hydrometeorological Institute, Prague	N_i , $f(T)$
Soil carbon content	localities	Forest Management Institute, Brand_s nad Labem	K_{gibb}
Dry deposition	10×10 km	Ecotoxa, Opava	N_d , S_d
Bulk deposition	localities	Central Institute for Supervising and Testing in Agriculture, Brno	N_d , S_d

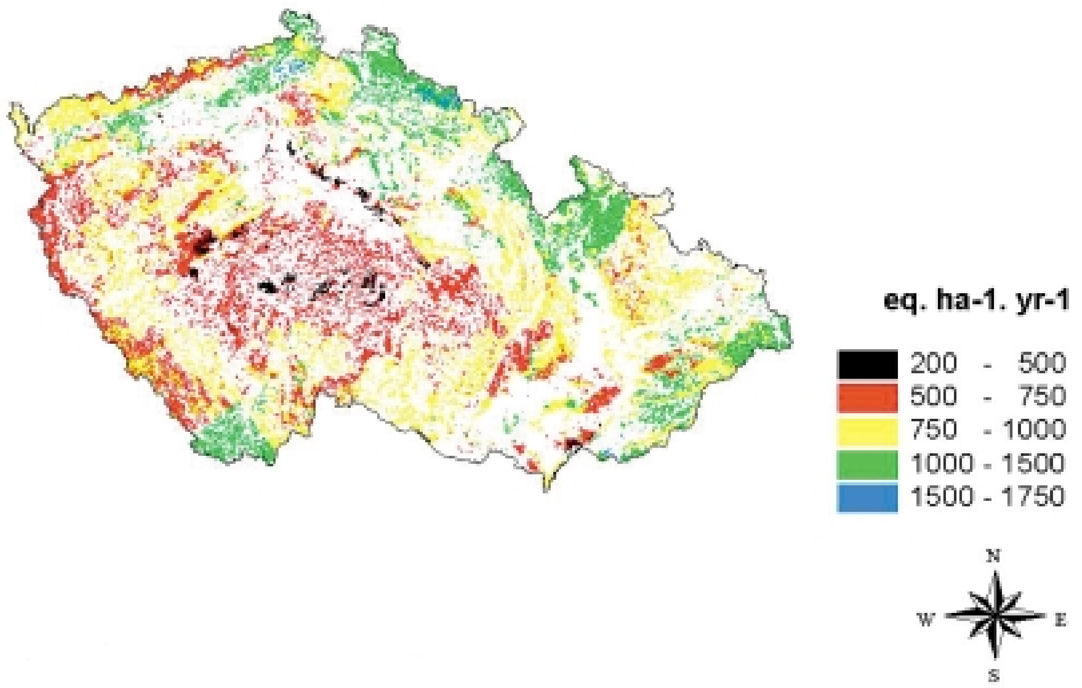

Figure CZ-1. Maximum critical loads of sulphur.

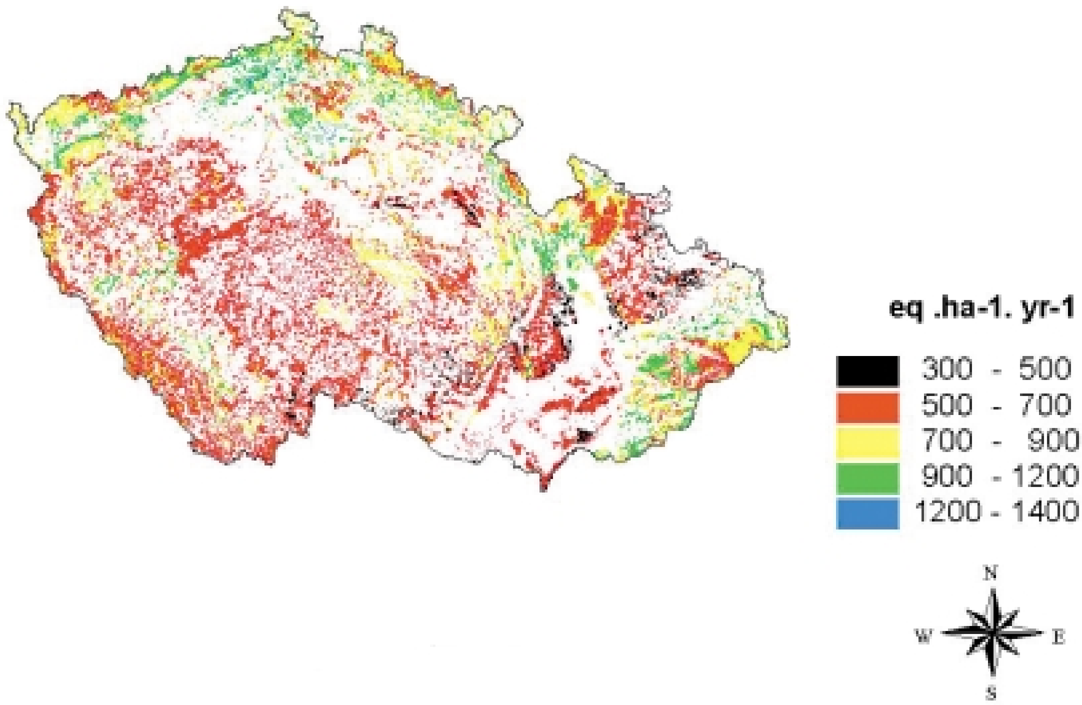

Figure CZ-2. Critical loads of nutrient nitrogen.

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Table DK-1. Calculated critical loads of acidification and N eutrophication for different ecosystems. All values are given in keq ha^{-1} 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	—

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 loads and exceedance of the critical load of nutrient nitrogen for Inland- and coastal heathland, raised bogs, pastures, sensitive meadows, and sensitive (lobelia) lakes.
- Critical load and exceedance of the critical load of nutrient nitrogen for production forests calculated with PROFILE.

Calculation methods

Critical loads of acidity and N eutrophication:
The PROFILE model was used to calculate critical loads for acidity and for nitrogen eutrophication, and for the values of BC_u, N_u, BC_w, and $ANC_{le,crit}$. From these calculations, values for $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ were derived. In calculating critical load for grasslands, the weathering rate for 11 classes of mineralogy 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 data was performed in December 1996. The total number of calculations and the calculated critical loads for the different vegetation types are illustrated in Table DK-1.

A BC:Al ratio of 1 was used as the chemical criteria for both forest soils and grasslands. For calculating critical loads for nutrient nitrogen, a critical N leaching of 2 kg N ha^{-1} yr^{-1} and an immobilisation of 3 kg N ha^{-1} yr^{-1} were applied. For model calculations, the root zone has been stratified in a 5 cm thick A / E horizon, and soil-dependent B and C horizons. 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, respectively.

Empirically based critical loads for N eutrophication: Critical loads of nutrient nitrogen for inland and coastal heathland, raised bogs, pastures, sensitive meadovs and sensitive (lobelia) lakes have been derived on a 5×5 km^2 grid. The basis of the assessment has been a new registration of nature areas protected according to Section 3 of the Danish Nature Protection Act. The quality (and quantity) of available data does not allow critical loads to be assessed on a plot scale, and a distribution function of critical loads has therefore been assessed for each nature type and applied on a 5×5 km^2 grid. The variation in critical loads for each nature type is caused by differences in biotic conditions, management history, conservation status, and administratively set quality targets for the areas.

National deposition maps: Calculation of NH_x deposition (based on 1996 emissions) on a 5×5 km^2 grid has been performed as part of the technical background fora national action plan to abate ammonia emissions from agriculture. The spatial distribution of emissions was, however, based on county-level statistics from 1989. Table DK-2 shows a comparison between this calculation and an

Table DK-2. Background deposition values for NH_X (in kg N ha^{-1} yr^{-1}) from the nationwide calculation on a 5×5 km^2 grid compared to calculations taking ecosystem-dependent differences in deposition velocity and the influence of local sources into account. Edge effects have not been included.

	Background NH_x deposition calculated on a national 5x5 km^2 net			NH_x deposition including ecosystem-dependent differences in deposition velocity and the effect of local sources		
Percentile	**Heath**	**Pasture**	**Forest**	**Heath**	**Pasture**	**Forest**
99	15	15	15	24	29	49
95	13	14	14	19	24	39
75	12	12	12	14	19	30
50	11	11	11	11	15	25
25	9	9	9	9	12	20
5	6	6	6	6	8	14
1	4	5	5	5	6	11

assessment accounting for ecosystem-dependent differences in deposition velocity and the influence of local sources. The higher values of the highest 25 percentile are primarily due to the influence of local ammonia sources, which are not properly reflected on the 5×5 km^2 grid resolution.

As part of the Danish Nationwide Background Monitoring Programme, deposition calculations of SO_2, NO_x and NH_x to the Danish sea and land area are performed annually on a 30×30 km^2 national grid. The latest reporting of data from this programme has been in 2000 with data from 1999.

Data sources

The main sources of data have not been changed since the 1999 status report. The sources and resolution of data are shown in Table DK-3.

Table DK-3. Data sources.

Parameter	Resolution	Source
soil mineralogy literature	60 points	DLD,
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 emphases of the Danish NFC in the past two years has been:

- Further work on methods and data to calculate critical loads for nutrient nitrogen for sensitive, natural or seminatural terrestrial ecosystems, primarily heathlands, meadows, pastures, and raised bogs.
- Estimating the uncertainties in calculated critical load exceedances with special emphasis on the influence of local scale variation in NH_x deposition.
- Further work on dynamic modelling.
- As indicated, only minor progress has been made in producing data to calculate critical loads and exceedances.

When adopting a new action plan for the protection of the aquatic environment in 1998, the Danish Parliament called on 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 the sources and status of ammonia emissions, the effects on nature and environment, technical measures and potential of abatement and on abatement costs, have been prepared using 1996 a base year. The action plan, presented to Parliament in May 2001, aims to reduce ammonia emissions from Danish agriculture by 12%. This reduction goes further than the Danish obligations under the Gothenburg protocol.

Local ammonia sources and the potential of using buffer zones: The potential for using "buffer zones" around sensitive nature areas to protect the areas from the influence of local ammonia sources in agriculture has been explored. Danish nature areas are predominantly small patches spread as a mosaic in the agricultural areas, that occupy ca. 62% of the Danish land area. As a consequence, developing buffer zones around all sensitive or potentially sensitive nature areas will affect a large proportion of the present agricultural land. Table DK-4 lists the affected agricultural area from a general application of 200-m, 500-m, and 1-km buffer zones (perpendicular distance) around all (potentially) sensitive nature areas and forests. The affected area using 1-km buffer zones would be close to 100% of the present agricultural land. This is probably not politically feasible.

Table DK-4. Agricultural area affected by a general application of bufferzones around sensitive nature areas (total area and % of total agricultural area).

Buffer zone (m)	Affected area (km^2)	%
200	14,700	50
500	26,000	88
1000	29,200	99

The potential effectiveness of a limited, targeted use of buffer zones has therefore been investigated.

Environmental benefits are estimated as additional protected area ($Ex(A) < 0$, $Ex(CL_{nut}(N)) < 0$). Buffer zones are only applied where the probability of CL exceedance is larger than 50%, and the application is optimised so the areas in which non-exceedance can be achieved with the least change in agriculture are selected first. Figure DK-1 illustrates the results of this analysis. A targeted applications of buffer zones affecting 100,000 livestock units can (in Denmark) achieve 15% more protected heathlands, 6% more protected pastures, 4% more protected beech forest, and 2% more protected oak forest. The economic efficiency of a targeted use of buffer zones is good compared with the measures taken in the latest international agreements.

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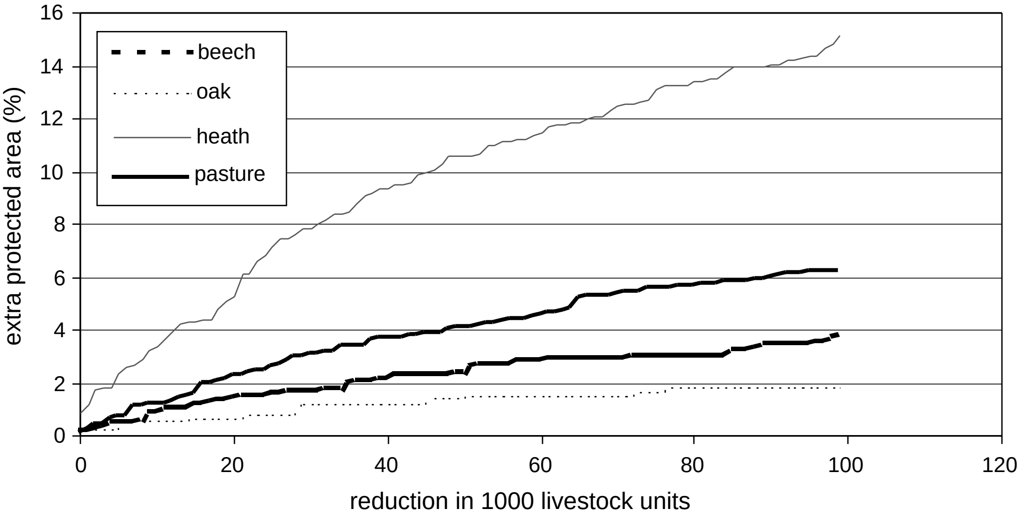

Figure DK-1. Achievable environmental benefits from a targeted, cost-optimal applications of buffer zones around sensitive nature areas.

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Calculation methods

Critical loads calculations were calculated on a 1×1 km^2 grid. Soil data was taken from a digital soil map (scale 1:200,000) using the soil of the grid point as representative for the entire cell. Vegetation data was taken from the CORINE land cover data base for the same grid points for deciduous forests (n=91), coniferous forests (92) and mixed forests (93). Each grid point is thus representative of 1 km^2.

For forest ecosystems (deciduous, coniferous and mixed according to land cover) 21,450 cells were used. Critical loads were calculated using the simple mass balance method. Values for BC_{we}, N_i, $N_{le(acc)}$ and Q were estimated from soil data. Q was assumed to be zero for peatlands and clay; BC_{we} and $N_{le(acc)}$ are zero for peatlands.

For raised bogs (the case in which land cover in the representative point is bog), the empirical method was used to determine $CL_{nut}(N)$. Critical loads for acidification are not applicable, as acidification is not a problem in these ecosystems.

The uncertainty of critical loads has been addressed in published papers (Oja and Kull 2000, Oja et al. 2000, Oja 2000). Dynamic modelling is now being developed in a complex soil-vegetation-atmosphere model RipFor (Mander et al. 1999), derived from the ForSVA model (Arp and Oja 1997, Oja and Arp 1997) for a few sites to allow calculation of critical loads. As of yet, dynamic models have not been used for mapping purposes.

Maps of critical loads for forest ecosystems are presented in Figures EE-1 through EE-4 (Rauk 2000).

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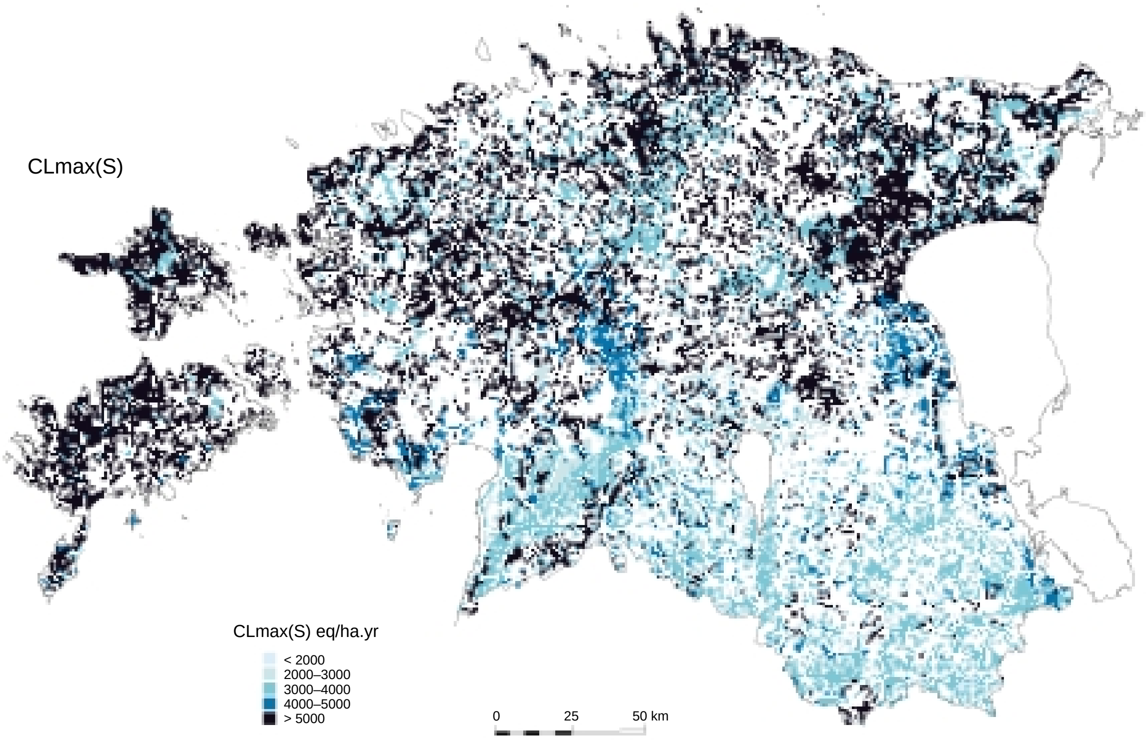

Figure EE-1. Maximum critical loads of sulphur.

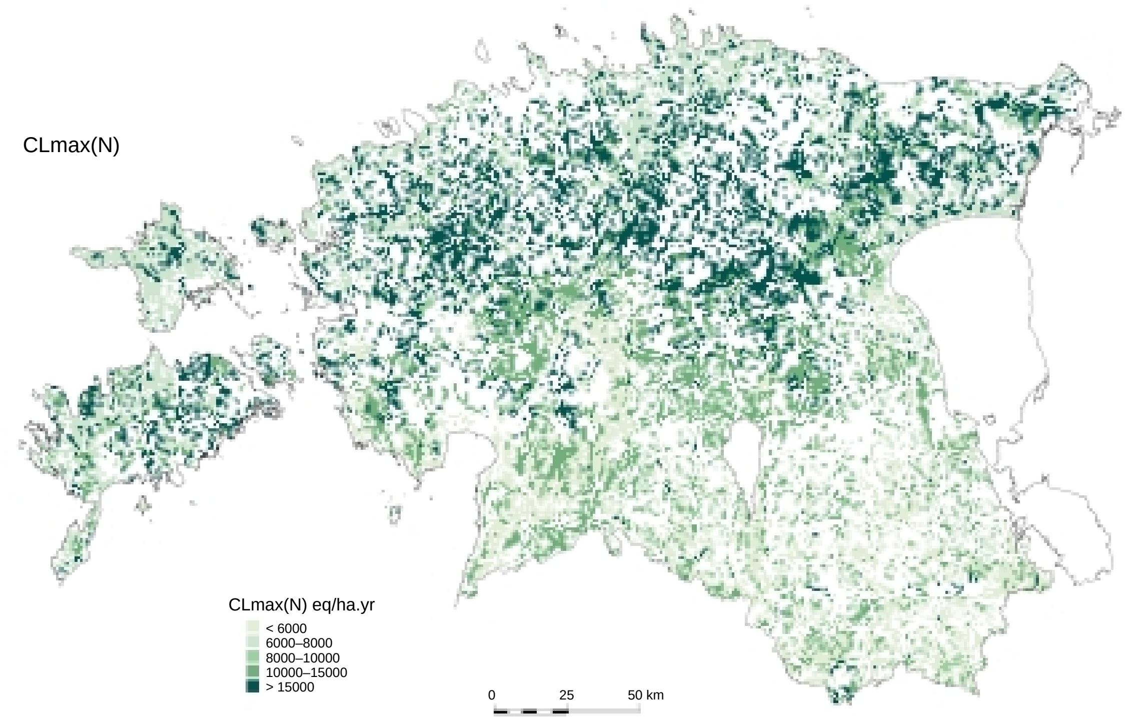

Figure EE-2. Maximum critical loads of nitrogen.

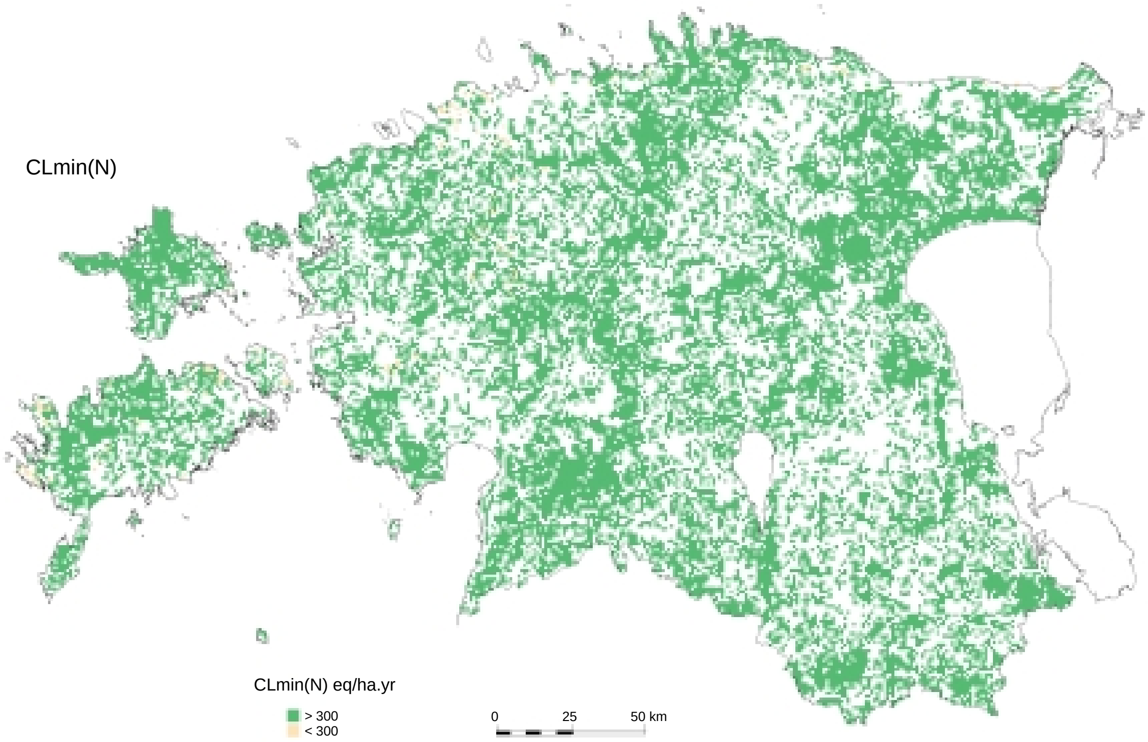

Figure EE-3. Minimum critical loads of nitrogen.

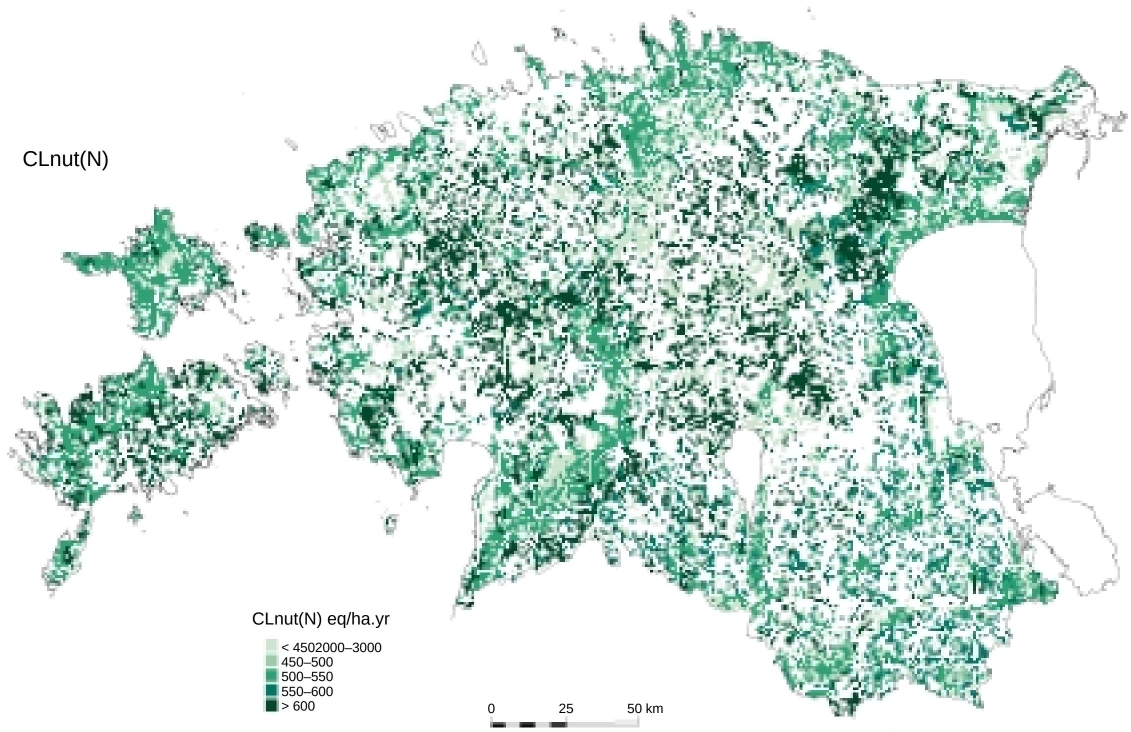

Figure EE-4. Critical loads of nutrient nitrogen.

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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).

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

where the base cation (BC) leaching is given by

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

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 immobilisation 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:

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

where the dimensionless constants a_N, a_S, b_1 and b_2 are all smaller than one and depend on ecosystem properties only: the 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:

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

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:

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

Data sources

Deposition: Base cation (Ca, Mg, K, Na) and chloride deposition were interpolated from the data from the years 1993–95 of a nationwide network of stations measuring monthly bulk deposition (Järvinen and Vänni 1990), and seasalt correction was 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 (Johansson and Tarvainen 1997). 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. The method employed gave weathering rates comparable to those obtained from an input-output budget, the PROFILE model and the direct use of the 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 from tabular national forest inventory data (Kuusela 1977) as a function of tree species and ETS, and the biomass density and element contents (Olsson et al. 1993, Rosén, pers. comm.) in biomass (stem and bark). The limiting concentration, $[BC]_{min}$, below which trees can no longer extract nutrients from soil solution, is set to a precautionary value of 2 μeq l^{-1} (UBA 1996).

Denitrification and nitrogen immobilisation: N_{de} is assumed 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). If peat cover information was lacking for the catchment soil, f_{peat} for lakes was estimated using correlation 0.02472 COD + 0.05105, where COD denotes the chemical oxygen demand, based on observational data from Finnish 1987 lake survey (Henriksen et al. 1993, Posch et al. 1997). For N_i (including N_{fix}) a constant value of 1.0 kg N ha^{-1} a^{-1} as a long-term average was used for Finnish forest soils, representing the upper limit of the range of values recommended (UBA 1996).

Leaching of alkalinity and nitrogen: $ANC_{le(crit)}$ is calculated by adding the critical aluminium leaching, obtained from the molar Al:BC ratio of 1.0, to the hydrogen leaching, calculated from a constant 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). Runoff values needed for converting concentrations to fluxes were obtained from a digitised runoff map for 1961–1975 (Leppäjärvi 1987).

Lake-specific parameters: Values for the retention of sulphur 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 m a^{-1}, and s_N =5.0 m a^{-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 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 μeq 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.

Forest ecosystem areas: Forest soils have been allocated areas using two data sets. First, a satellite image based land use data set was used to estimate total forest area in each EMEP50 grid cell. These data cannot properly separate tree species, which are used in uptake calculations. The seventh national forest inventory data from the Finnish Forest Research Institute was used to assign total forest area to the three tree types (spruce, pine, deciduous) employed. The forest soil area in critical load calculations covers an area of 240,400 km^2.

Lake ecosystem areas: Lakes are reported using both the number of lakes and the surface water area. Critical loads are not calculated for all lakes in Finland due to practical input data availability. The following method allowed critical load calculations for 1450 lakes to representat all Finnish lakes. The Lake Register at the Finnish Environment Institute includes data for all lakes greater than 1 ha in Finland. These lakes were assigned to each EMEP 150×150 km^2

grid cell. The cell-specific total lake number was allocated in the same cell to those lakes, for which the critical loads were calculated. This resulted in one critical load calculation lake to represent on the average 39 lakes (ranging from 1–233) depending on the grid cell. The same method was applied to allocate total lake water area to critical load calculation lakes, and one critical load calculation lake represents on the average 23 km^2 (ranging from 0.2–75 km^2) of lake surface area.

Altogether 55,688 lakes were available in the Lake Register for these calculations. When combining these with those 1450 lakes, for which critical loads are calculated, and excluding those which already are in the Lake Register, we have a total of 56,313 lakes represented in Finland, with a total lake water area of 33,230 km^2.

Uncertainty analyses

Forest soils: An overall uncertainty of 30% was estimated for the forest soil critical loads for acidification due to sulphur and nitrogen in Finland based on the study by Johansson and Janssen (1994). The equations for $CL(S+N)$ were based on Hettelingh et al. (1991) and Kämäri et al. (1992). Symmetric triangular distributions for parameter uncertainties were chosen to portray the distribution of parameter value errors in the study. The uncertainty in the parameter values was expressed in terms of the coefficient of variation (CV), i.e. standard deviation divided by the positive mean. The input parameters included in the analysis were given individual uncertainty ranges from literature values or with best estimates. The correlations between input parameters was not considered. A Monte Carlo approach was chosen for the analysis using standard critical load equations of the time of the study. More details of the method and materials are provided in Johansson and Janssen (1994) and Johansson (1999, 2001).

Lakes: The uncertainty of critical loads for lakes in Finland has been analyzed by Kämäri et al. (1993) and reviewed by Johansson (2001). Several input values were assigned uncertainty ranges. Runoff, nitrogen uptake by forest, mass transfer coefficient for retention of both sulfate and nitrate, and critical acid neutralizing capacity limit were appointed symmetric triangular distributions with a total range of ±20%, ±60%, ±30% and ±100%, respectively. The pre-industrial chemical steady-state of a lake, which is required in the critical load calculation, calls for a F-factor and a scaling factor B to relate long-term changes of base cation leaching to strong acid ion input. B uses the background (pre-acidification) non-marine sulfate concentration, $[SO_4^{2-}]^*_0$, which was estimated with present base cation concentration based on empirical relationship for 61 lakes receiving minor acidic deposition. The mean value of $[SO_4^{2-}]$ from the regression, varying roughly from 15 to 85 $\mu eq\ l^{-1}$, was assumed normally distributed with ±23 $\mu eq\ l^{-1}$ standard deviation, positive and smaller than present $[SO_4^{2-}]^*$. The probability scaling factor B, which enters the calculation of the F-factor, was assigned a separately calculated probability distribution based on diatom analyses from 27 lakes. First, the variation of B due to Gaussian distributed $[SO_4^{2-}]^*_0$ was determined with 1000 simulations per lake. Then, the resulting values were ordered and 90% central interval was selected and rescaled to be the distribution function of the F-factor for further analysis. The more detailed description is given by Kämäri et al. (1993). The output values, the critical loads, were calculated with the Monte Carlo method, and the range mapped was between the 5 and 95 percentiles of the lake-specific uncertainties. The results for critical loads of sulphur and nitrogen, $CL(S+N)$, suggest an uncertainty of ±7 and ±13%, respectively. The standard deviation ±10% can be selected to represent the overall critical load variation for lakes.

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Table FI-1. Uncertainties of input parameters expressed as coefficient of variation (CV) in percent. The uncertainty of the weathering rate (BC_w) was separately calculated by varying the effective temperature sum (ETS) and total soil contents of calcium and magnesium with their estimated uncertainties. Similar calculation was done for the nutrient net uptakes (BC_u and N_u) affected by variations in the ETS, biomass density and nutrient contents in the biomass.

Parameter	Unit	CV in %		
BC^*_0	meq m^{-2} yr^{-1}		30	
$SO_{2,d}$	meq m^{-2} yr^{-1}		20	
$NO_{x,d}$	meq m^{-2} yr^{-1}		30	
$NH_{4,d}$	meq m^{-2} yr^{-1}		35	
filtering factors for acidifying deposition	–		5	
forest area (of each tree species)	km ^{2}		5	
runoff	mm yr^{-1}		5	
total content in bottom soil (Ca and Mg)	mass %		5	
ETS	degree days		10	
BC_w	meq m^{-2} yr^{-1}		24	
biomass density	kg m^{-3}		6	
Nutrient content in stem and bark:		**birch**	**spruce**	**pine**
[Ca]	mass %	28	16	29
[Mg]	mass %	28	16	36
[K]	mass %	28	22	36
[N]	mass %	28	32	29
BC_u	meq m^{-2} yr^{-1}	37	48	40
N_u	meq m^{-2} yr^{-1}	37	55	44

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Status of critical loads data

In response to the most recent call for data, the NFC informed the CCE that no revisions to previous critical loads data were to be submitted. Thus the 1998 critical loads database has been adapted to the new EMEP co-ordinate system by the CCE, and has been included into the European database. For a description of the national data, see the NFC report in the CCE Status Report 1999.

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Calculation methods

The German NFC provided updated national critical load data to the CCE, stressing the necessity to continue the application of the mass balance approach in the future, in compliance with the effect-based protocols as well as a basis for dynamic modelling. About 30% of the area of Germany is represented by forests and other (semi-)natural vegetation for which critical loads and their exceedances have been computed (see Table DE-1).

In general, critical loads are calculated in accordance with the methods described in the Mapping Manual (UBA 1996). The calculation of $CL(S+N)$ differs slightly from the Mapping Manual method, as soils soils with high base saturation are explicitly protected. The German critical load database consists of 424,026 records. A detailed description of the data and the methods by which they were derived is given in Table DE-2.

Table DE-1. Ecosystem types used as receptors for the critical loads approach.

Ecosystem types / Receptors	% receptor area of total area of Germany	% total receptor area
Deciduous forest	6.5	21.6
Coniferous forest	16.0	52.9
Mixed forest	6.4	21.3
Natural grassland	0.5	1.8
Acid fens and heathland	0.3	1.0
Wet grassland	0.1	0.4
Mesotrophic peat bogs	0.3	1.0
Total:	30.1	100

Critical loads of sulphur and nitrogen, $CL(S+N)$: To calculate critical loads of sulphur and nitrogen for forest soils and other (semi-)natural vegetation, various equations from the Mapping Manual (listed in Table DE-2) were used. To protect soils with high base saturation, this term was integrated into the estimation of critical loads. For all soil units with a base saturation >30%, the critical ANC leaching was set to zero. In sensitivity studies it turned out that the result of this classification is relatively robust concerning the 30% as a cut-off value. Since soils with a high base saturation tend to have high weathering rates and high ANC leaching values, their critical loads decrease by using this cut-off value. Without this assumption the base saturation of all soils would decrease to values near 5% within a few decades. Since the the critical loads approach is designed to protect all ecosystems against acidification, it is justified to also preserve those ecosystems adapted to a high base saturation in the soil. For this case the critical load was determined by base cation weathering only.

Table DE-2. National critical load database and calculation methods / approaches.

Parameter	Term	Unit	Description
Critical load for acidity	$CL_{max}(S)$	eq ha^{-1} yr^{-1}	Mapping Manual, Eq. 5.17 (5.36, 5.38, 5.41, 5.42); if base saturation ≥ 30 % , then $ANC_{le(crit)}$ = 0 (UBA 1996)
	$CL_{min}(N)$	eq ha^{-1} yr^{-1}	Manual, Eq. 5.23
	$CL_{max}(N)$	eq ha^{-1} yr^{-1}	Manual, Eq. 5.24
Critical load for nutrient nitrogen	$CL_{nut}(N)$	eq ha^{-1} yr^{-1}	Manual, Eq. 5.21
Uptake of base cations by vegetation	Bc_u	eq ha^{-1} yr^{-1}	See Tables DE-5 and DE-6.
Weathering of base cations	BC_w	eq ha^{-1} yr^{-1}	See Tables DE-3 and DE-4.
Gibbsite equilibrium constant	K_{gibb}	m^6 eq^{-2}	300
Acid neutralisation capacity leaching	$ANC_{le(crit)}$	eq ha^{-1} yr^{-1}	The minimum value of all approaches described in the Manual was taken for the calculation.
Nitrogen immobilisation	N_i	eq ha^{-1} yr^{-1}	See Table DE-7.
Nitrogen uptake by Vegetation	N_u	eq ha^{-1} yr^{-1}	See Tables DE-5 and DE-6.
Denitrification factor	f_{de}	-	See Table DE-8.
Nitrogen leaching	$N_{le(acc)}$	eq ha^{-1} yr^{-1}	$N_{le(acc)} = Q \cdot 10 \cdot [N]_{crit}$; $[N]_{crit}$ = 0.0143 eq m^{-3}

Table DE-3. Assignment of soil units to parent material classes.

Parent material class	Soil units of the General Soil Map of Germany (BUEK 1000)
0 Peat	6, 7
1 Acidic	1, 12, 16, 17, 25, 28, 29, 31, 33, 34, 44, 45, 46, 48, 55, 56, 57, 59, 60, 61, 63, 64, 70, 71
2 Intermediate	4, 5, 15, 18, 22, 26, 32, 40, 41, 42, 43, 47, 58, 62, 65, 67
3 Basic	8, 10, 11, 14, 19, 20, 23, 24, 27, 30, 36, 37, 38, 39, 51, 52, 53, 54, 68
4 Calcareous	3, 9, 13, 21, 35, 49, 50, 66, 69

No receptor areas exist for soil unit 2 (tidal soil).

To calculate base cation weathering, soil units from the General Soil Map of Germany (BUEK 1000) were assigned to parent material classes (Table DE-3). Using data on clay content and coarse fraction for each horizon, the effective clay content was calculated and assigned to a texture class (see Table DE-4).

Table DE-4. Classification of effective clay content to texture classes.

Texture class	Clay content (%)
1	< 10.5
1 / 2	≥ 10.5 to < 20.0
1 / 3	≥ 25.0 to < 30.0
1 / 4	≥ 30.0 to < 37.5
2	≥ 20.0 to < 25.0
2 / 3	≥ 37.5 to < 45.0
2 / 4	≥ 52.5 to < 57.5
3	≥ 45.0 to < 52.5
3 / 4	≥ 57.5 to < 62.5
4	≥ 62.5 to < 70.0
5	≥ 70.0

The respective weathering rates for each horizon (following annex IV of the Mapping Manual) were then identified. Finally, the weighted average of the weathering rates was computed. The critical loads of acidity are shown in Figure DE-1 (left).

Critical loads of nutrient nitrogen, $CL_{nut}(N)$:
The methods of calculating critical loads of nutrient nitrogen are described in detail in the Mapping Manual (Eq. 5.21). For the national approach, net uptake of nitrogen and base cations were calculated by applying a simplified equation for forests and natural ecosystems. The average growth derived from the abiotic site conditions (site type) was multiplied with the element contents (X) given in Table DE-5 (for forests) and Table DE-6 (for other natural vegetation):

$X_u = k_{gr} \cdot ctX$

where:

k_{gr} = average growth (t ha^{-1} a^{-1}) of plants (dry mass), if harvested; otherwise set to zero

ctX = content of X in harvested dry mass (eq kg^{-1})

Table DE-5. Nitrogen and base cation content in wood.

	Contents (eq t^{-1} dry mass)			
Species	**N**	**Ca**	**Mg**	**K**
Pinus sylvestris	137.43	59.13	21.39	20.84
Picea abies	158.85	87.08	26.33	32.1
Fagus sylvatica	240.96	84.83	28.8	33.89
Quercus spec.	260.59	112.77	23.86	47.06
Alnus glut. / Fraxinus exc.	201.69	83.21	33.33	36.32
Betula pendula	201.69	125.25	30.86	26.22
Pinus mugo	146.36	92.81	33.74	29.67
Salix spec.	201.69	89.07	33.33	30.18

Table DE-6. Nitrogen and base cation content of natural vegetation.

	Contents (eq t^{-1} dry mass)			
Natural vegetation type	**N**	**Ca**	**Mg**	**K**
Dystric meadows	360	109.8	41.1	26.2
Heathland	285	109.8	41.1	26.2
Calcareous dry grassland	430	185.3	82.3	33.3
Salt grassland	430	185.3	82.3	33.3
Swamp grassland	500	169.7	82.3	33.3
Flooded grass land on river sites	500	179.6	82.3	33.3
Eutrophic fresh grassland	465	169.7	82.3	33.3

Nitrogen immobilisation rates were ranked according to temperature (see Table DE-7) and the assignment of the denitrification factor to the clay content is shown in Table DE-8. Critical loads for nutrient nitrogen are shown in Figure DE-1 (right).

Table DE-7. Classification of temperature to N_i.

Temperature [°C]	N_i (eq ha^{-1} yr^{-1})
< 5	357
5	286
6	214
7	143
8	107
> 8	71

Table DE-8. Assignment of f_{de} to clay content classes.

Clay content [%]	f_{de}
< 10.5	0.1
≥ 10.5 to < 20.0	0.1
≥ 20.0 to < 25.0	0.2
≥ 25.0 to < 30.0	0.2
≥ 30.0 to < 37.5	0.3
≥ 37.5 to < 45.0	0.3
≥ 45.0 to < 52.5	0.3
≥ 52.5 to < 57.5	0.3
≥ 62.5 to < 70.0	0.5
≥ 70.0	0.5
Histosols	0.8
Podsols	0.1

Uncertainty

An uncertainty analysis for all input parameters and the results of the critical load calculations is available. For further information see www.oekodata.com.

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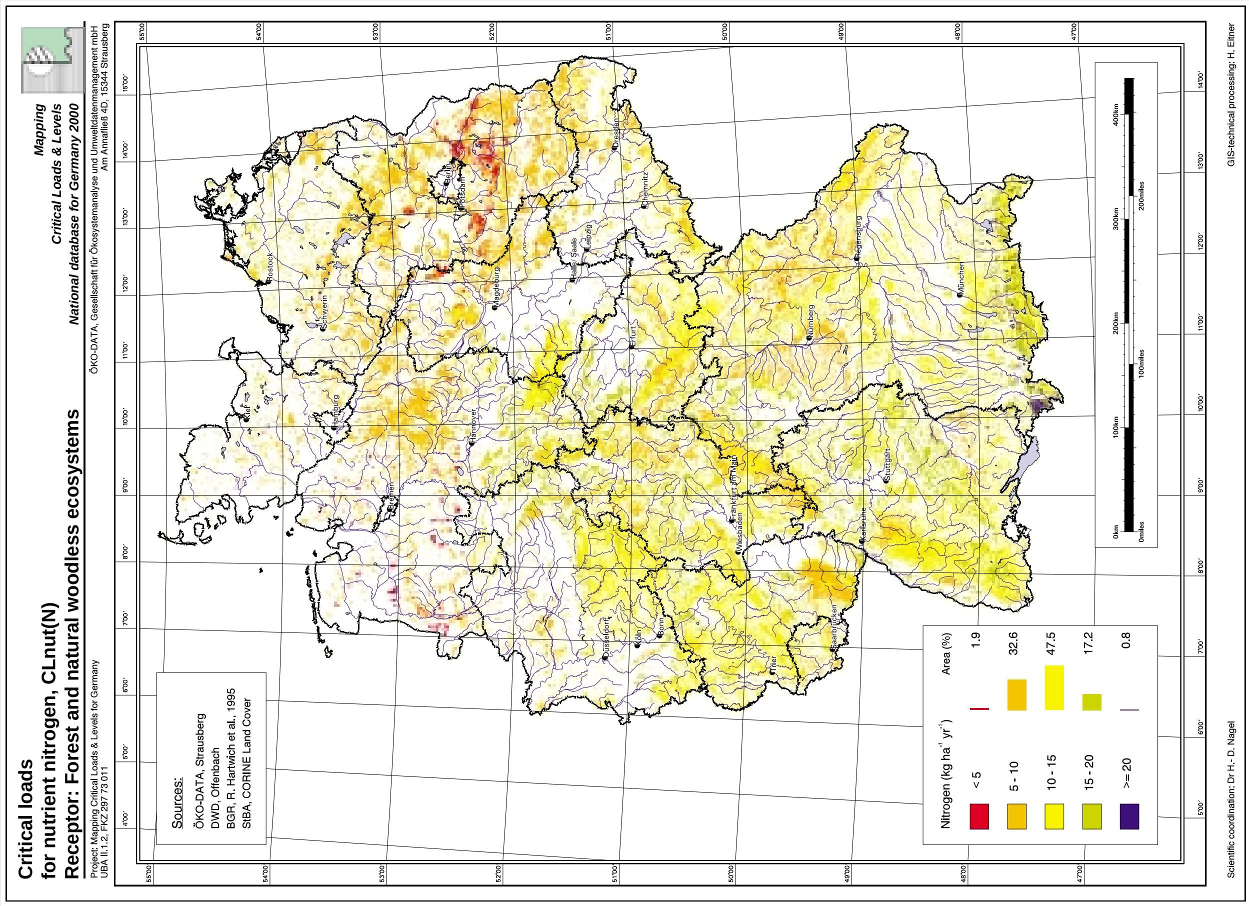
UBA (1996) Manual on Methodologies and Criteria for Mapping Critical Levels / Loads and geographical areas where they are exceeded. UN / ECE Convention on Long-range Transboundary Air Pollution. Federal Environmental Agency (Umweltbundesamt) Texte 71 / 96, Berlin.



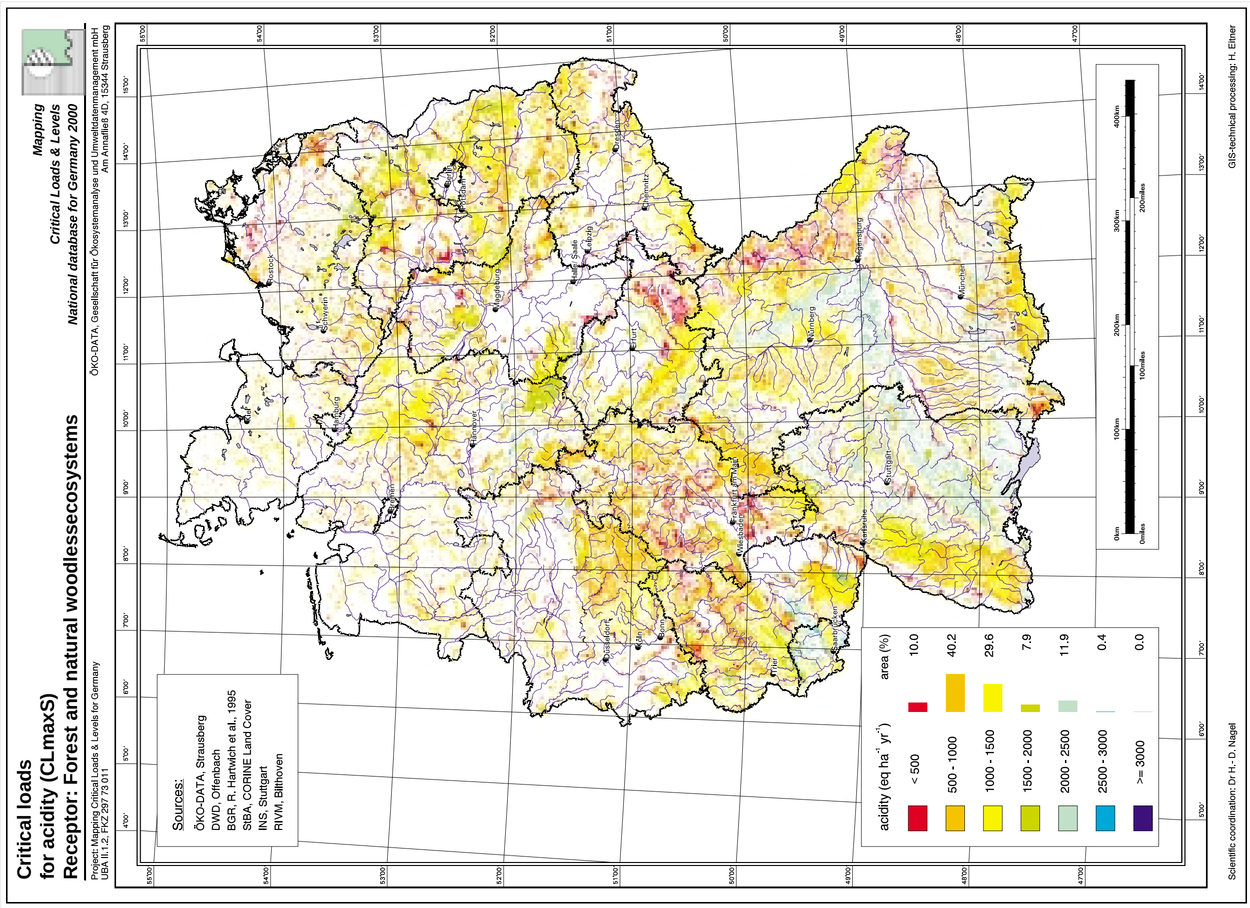


Figure DE-1. Critical loads of acidity (left, in eq ha^{-1} yr^{-1}) and of nutrient nitrogen (right, in kg N ha^{-1} yr^{-1}).

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Calculation methods

Critical loads of nutrient nitrogen for forests were calculated with the SMB model. Critical loads have been calculated and mapped in Hungary for 50×50 km² EMEP grid cells. Critical load for grasslands had been derived from literature data (UBA 1996).

For two ecosystem types – coniferous (spruce) and deciduous (beech) forests – nitrogen uptake has been estimated from growth data. Other parameters (nitrogen immobilisation, acceptable nitrogen leaching) were not measured at the sites. (Table HU-1.)

Table HU-1. Input parameters and critical loads of nutrient nitrogen calculated for two ecosystem types (in eq ha^{-1} yr^{-1}).

	Beech	Spruce
$\boldsymbol{N_u}$	**2999.97**	**1392.8**
N_i	214.28	214.28
f_{de}	0.5	0.5
$N_{le(acc)}$	71.428	142.86
$CL_{nut}(N)$	3249.9	1678.56

For other ecosystem types, critical loads for nutrient nitrogen have been collected from literature (UBA 1996 p. 73; Table HU-2). Ecosystem types have been assessed with the CORINE Land Cover categories (CEC 1992, Ellenberg 1988, Wyatt et al. 1990).

Data for input parameters of the SMB model has been provided by the Hungarian Research Institute for Forestry. Deposition data were collected by the Hungarian Meteorological Institute, maps were produced by the GIS labor of Research Institute for Soil Science and Agricultural Chemistry.

Results

National deposition maps of SO_x and NO_x on a 50x50 km² grid. Exceedances of nutrient nitrogen based on SMB model and literature data.

Comments and conclusions

The main focus of the Hungarian NFC in the past year has been on:

- a first calculation of critical loads of nutrient nitrogen.
- harmonisation of the ecosystem categories used by NFCs and Hungarian databases.
- starting activities on dynamic modeling.
- opening two site measurement project to measure input parameters.

As indicated, only a part of the SMB model has been finalized due to lack of data available in 2000. The uncertainty of the results is unknown, which will be estimated in the future progress. The $CL_{nut}(N)$ values provided for coniferous forest was too high compared to the European average, thus a lower value of 571.2 eq ha^{-1} yr^{-1} has been used for exceedance maps.

Table HU-2. Critical loads of nutrient N for two ecosystem types derived from literature data (UBA 1996, p. 73).

Ecosystem type	CORINE Land Cover category	$CL_{nut}(N)$ (eq ha^{-1} yr^{-1})
Forests:		
–Acidic coniferous forests	3.1.2 Coniferous forest *Definition: Vegetation formation composed principally of trees, including shrub and bush understories, where coniferous species predominate*	964.3
–Calcareous forests	3.1.1 Broad-leaved forest *Vegetation formation composed principally of trees, including shrub and bush understories, where broad-leaved species predominate.*	1247.8
Species-rich grasslands:		
–Calcareous grasslands	3.2.1 Natural grasslands *Low productivity grassland. Often situated in areas of rough uneven ground. Frequently includes rocky areas, briars, and heathland* 3.2.4 Transitional woodland-scrub *Bushy or herbaceous vegetation with scattered trees. Can represent either woodland degradation or forest regeneration/colonisation*.	1782.5
Mesotrophic fens	4.1.1 Inland marshes *Low-lying land usually flooded in winter, and more or less saturated by water all year round*	1960.7
Ombrotrophic bogs	4.1.2 Peat bogs *Peatland consisting mainly of decomposed moss and vegetable matter. May or may not be exploited.*	534.7

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National maps produced

Critical loads for coniferous forests, deciduous forests, moor and heathland, natural grassland and freshwater lakes were calculated on a 1×1 km^2 national grid. The CORINE land cover map for Ireland has been used to define the distribution of the receptor ecosystems (Ordnance Survey of Ireland 1993). A data base describes the percentage of each land cover type in every 1 km^2 (range 0.01–100%).

The following have been calculated and mapped:

- Critical loads of acidity for soils.
- Maximum critical loads of sulphur.
- Minimum critical loads of nitrogen.
- Maximum critical loads of acidifying nitrogen.
- Critical loads of nutrient nitrogen (empirical and mass balance).

Calculation methods

The maximum critical loads of sulphur, 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). A pH of 4.2 was selected as the H^+ concentration limit and subsequently used to estimate the Al^{3+} 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 (1070 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 (0.1), moderate soils (0.4), gleys and peaty podsols (0.7) and peats (0.8). The empirical values for coniferous and deciduous forests are set at 1790 and 1070 eq ha^{-1} yr^{-1} respectively. When the empirical critical load of nutrient nitrogen was selected as minimum, the maximum critical load of acidity nitrogen was calculated as:

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

When $CL_{nut}(N)$ of the simple mass balance approach was the selected minimum, it was calculated as:

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

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 km^2 mapping grid (Figure IE-1b).

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 in 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 for 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 was 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 m^3 ha^{-1} yr^{-1}, a wood density of 390 kg m^{-3} and stem concentrations of Ca^{2+} = 0.056%, Mg^{2+} = 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, N uptake of 71 eq ha^{-1} yr^{-1} was selected to account for uptake by grazing.

Nitrogen immobilisation: According to previous mapping guidelines (Downing et al. 1993), nitrogen immobilisation at critical load can be approximated by the long-term, natural immobilisation of 2–5 kg N ha^{-1} yr^{-1}, which is assumed to be net immobilisation, 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,
N_i = 2 kg N ha^{-1} yr^{-1} for all other soils.

Immobilisation 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 kg 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,

$N_{le(acc)}$ = 4 kg N ha^{-1} yr^{-1} for deciduous forests.

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 km^2) of the land area of Ireland.

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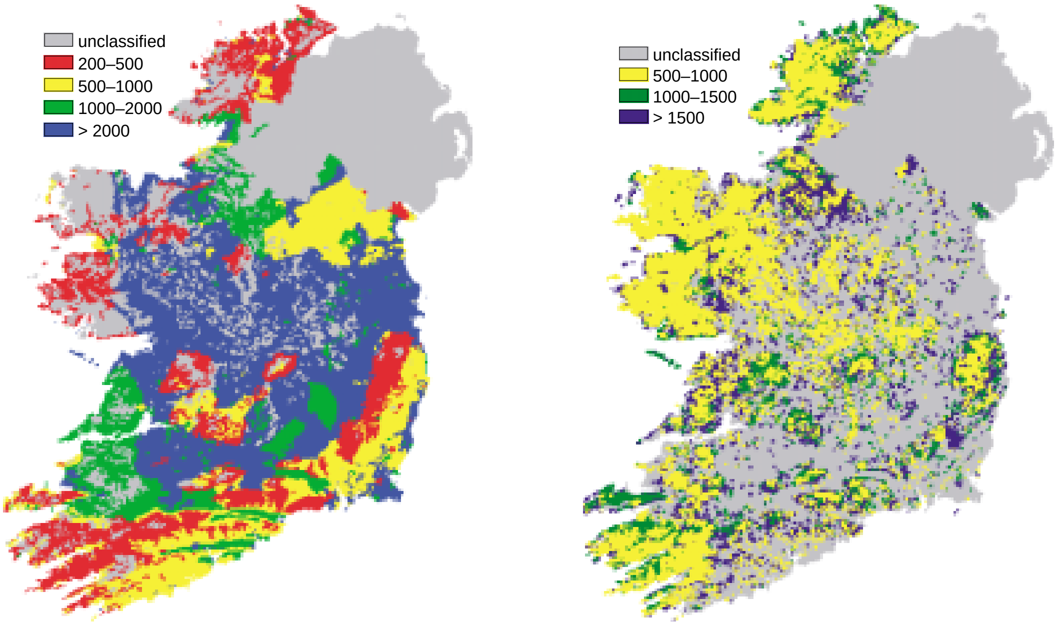

Figure IE-1. (a) left: Acid Neutralizing Capacity due to weathering, ANC_w, according to the Skokloster classification (eq ha^{-1} yr^{-1}). Note: The Skokloster classification only considers mineral soils. (b) right: Critical loads of nutrient nitrogen for peat bogs, moors and heathlands, and coniferous and deciduous forests (eq ha^{-1} yr^{-1}). Note: The minimum of the empirical and mass balance approaches was selected to represent the critical load for coniferous and deciduous forests.
Note: "Unclassified" refers to grids with no critical load estimates and regions outside the Republic of Ireland.

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Status of critical loads data

In response to the CCE call for data in early 2001, the NFC submitted a revised data set of critical loads. However, no national report describing the data was received.

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National maps produced

The quantity and quality of many Dutch ecosystems have declined during the last decades. Drinking water production, wood production and nature conservation, all different ecosystem functions, are threatened. The Ministry of Housing, Spatial Planning and the Environment and the Ministry of Agriculture, Nature Management and Fisheries have defined goals for protection of the different ecosystem functions. Earlier CCE reports (e.g. 1997) presented critical loads maps of (i) acidity related to the protection of both soil quality (no changes in readily available Al) and tree roots (by avoiding elevated Al:BC ratios) and (ii) nutrient nitrogen related to the protection of forest vegetation. In a recent evaluation of Dutch acid rain abatement strategies, critical loads were calculated for:

1. Groundwater quality, for protection against contamination by nitrate (critical N load) and Al (critical acid load).
2. Forests (soils) against nutrient unbalance due to elevated foliar N contents (critical N load) and against root damage due to elevated Al:BC ratios or soil quality deterioration by requiring no changes in pH (or base saturation) and / or readily available Al (critical acid load).
3. Plant species composition in terrestrial ecosystems against eutrophication (critical N load) and acidification (critical acid load).
4. Plant species composition in fens against eutrophication (critical N load).

All these maps have been submitted to the CCE. Integrated critical load maps (based on the minimum critical load in 1×1 km^2 grids) were used to evaluate various policy scenarios of the Dutch acid rain abatement strategies. The methods and results of the individually derived critical loads are discussed below.

Calculation methods

General approach:
The present maps describe critical loads for several protection functions. These critical loads are calculated by using different methods, based on different critical limits. Table NL-1 describes the different protection functions, critical limits and methods involved. Critical loads for protection of groundwater and forest are calculated with the SMB model as described below. Critical loads for protection of plant species composition are calculated with dynamic models, using information on the range of suitable environments for the occurrence of plant species within nature targets as critical limits (Table NL-1). For terrestrial ecosystems, the SMART-MOVE model was used (Van Hinsberg and Kros, Part II of this report). The dynamic model AquAcid (Wortelboer 1998) was used to calculate critical loads for fens. A general description of those models is given later.

Table NL-1. Critical limits and calculation methods used to assess critical loads of nutrient nitrogen and acidity for the distinguished protection criteria.

		Method	
Ecosystem type	**Critical limit**	**Nitrogen**	**Acidity**
Ground water (quality)	Critical nitrate concentration, pH and Al concentration[1]	SMB	SMB
Forests: impacts on roots and soil quality	Negligible depletion of base saturation or Al-hydroxides[2] Critical Al/(Ca+Mg+K) ratio[3]	–	SMB
Forests: impacts on foliage/growth	Optimal nitrogen content in leaves/needles[4]	SMB	–
Terrestrial vegetation (plant species composition)	Ranges in moisture availability, soil pH and nutrient availability[5]	SMART/MOVE + empirical data	SMART/ MOVE + empirical data
Heathland lakes (plant species composition)		AquAcid	–

1 Values are based on target levels for drinking water production in the Netherlands, i.e. a nitrate concentration below 0.4 $mol_c\ m^{-3}$ (the EU target for nitrate in ground water is 0.8 $mol_c\ m^{-3}$), a pH value higher than 5.3 and an Al concentration lower than 0.02 $mol_c\ m^{-3}$.
2 Negligible changes in soil quality parameters are based on the "stand-still" principle.
3 Values used, based on a literature survey on effects of elevated Al:BC ratios on tree roots, vary between 0.8 and 2.0 mol mol^{-1}, depending upon tree species (Sverdrup and Warfvinge 1993).
4 Values used, based on literature information on impacts on wood growth (no further stimulation) and adverse effects on drought stress, frost stress and pest and diseases, vary between 1.8% for conifers and 2.5% for deciduous forest.
5 Ranges of suitable environments for Dutch nature conservation targets, derived from the vegetation model MOVE (Van Hinsberg and Kros, this report).

Simple mass balance (SMB) equations for impacts on groundwater and forests:

The critical loads of nutrient N for the protection of ground water quality and tree growth were calculated with Simple Mass Balance equations, according to:

$$CL(N) = N_{gu} + N_{im(\mathrm{crit})} + NO_{3,le(crit)}\ /\ (1 - f_{de}) \qquad (1)$$

where:
N_{gu} = growth uptake flux of nitrogen
$N_{im(crit)}$ = critical long-term N immobilisation rate
$NO_{3,le(crit)}$ = critical NO_3 leaching flux
f_{de} = denitrification fraction

The critical loads of acidity (the sum of S and N) for the protection of groundwater quality and soil quality were also calculated with Simple Mass Balance equations, according to:

$$CL(S{+}N) = BC^*_{td} + BC_{we} - BC_{gu} + N_{gu} + N_{im(crit)} + N_{de} + Ac_{le(crit)} \qquad (2)$$

Data for CCE work, i.e. $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ were calculated as (see also the CCE Status Report 1993):

$$CL_{max}(S) = BC^*_{td} + BC_{we} - BC_{gu} + Ac_{le(crit)} \qquad (3)$$

$$CL_{min}(N) = N_{gu} + N_{im(crit)} \qquad (4)$$

$$CL_{max}(N) = N_{gu} + N_{im(crit)} + CL_{max}(S)\ /\ (1 - f_{de}) \qquad (5)$$

where:
BC^*_{td} = seasalt-corrected total deposition flux of base cations
BC_{we} = base cation weathering flux
BC_{gu} = growth uptake fluxes of base cations and nitrogen respectively
$N_{im(crit)}$ = critical long-term nitrogen immobilisation rate
$Ac_{le(crit)}$ = critical leaching flux of acidity

The critical NO_3 or acidity leaching fluxes were calculated by multiplying the precipitation excess with a target NO_3 or acidity concentration, depending on the protection criterion involved according to:

$$Ac_{le(crit)} = PS \cdot [\mathrm{Ac}]_{(crit)} \qquad (6)$$

$$NO_{3,le(crit)} = PS \cdot [\mathrm{NO_3}]_{(crit)} \qquad (7)$$

Those criteria are given in Table NL-1. In some cases, the value could be used directly (e.g.. for groundwater quality, a target level of 0.4 $mol_c\ m^{-3}$ was used) but most often had to be derived from other criteria (e.g. for tree growth, this concentration was derived from the critical foliar concentration). More information on the calculation of critical nitrate and acidity leaching fluxes is given below.

Calculation of critical nitrate and acidity leaching fluxes for the protection of groundwater quality:
The calculation of critical leaching fluxes for protecting groundwater quality are based on nationally defined target levels for nitrate and aluminium concentrations (Table NL-1). The critical leaching flux of acidity to protect groundwater quality was calculated by multiplying the precipitation excess by the critical concentration of protons and aluminium (0.02 $mol_c\ m^{-3}$) minus the critical HCO_3 concentration. The critical H concentration was derived from the critical Al concentration using empirical equilibrium relations (no Gibbsite relationship) between Al and H in the soil solution according to:

$$[\mathrm{H}]_{crit} = \left(\frac{[\mathrm{Al}]_{crit}}{K_{emp}} \right)^{1/n} \qquad (8)$$

where $[\mathrm{H}]_{crit}$ and $[\mathrm{Al}]_{crit}$ are the critical H and Al activities (mol l^{-1}), K_{emp} is the empirical equilibrium constant and n is the reaction stoichiometric constant (Van der Salm and De Vries 2001). Values for K_{emp} and n are given below in Table NL-2. The HCO_3 concentration was related to the proton concentration using the bicarbonate equilibrium. (Note that the base cation weathering flux was calculated for an average groundwater depth of 32 meters below the surface).

Table NL-2. Overview of the values of K_{emp} and n used to calculate critical acidity leaching.

Soil type	Vegetation type	Soil depth (cm)	K_{emp}	n
Sand	Forest	60	5.20	2.51
Sand	Grass / Heath	30	2.14	1.88
Clay	All	–	7.88	2.65
Löss	Forest	100	4.55	2.17
Löss	Grass / Heath	30	3.29	1.90
Peat	All	–	-1.06	1.31

The critical NO_3 leaching flux to protect groundwater quality was calculated by multiplying the critical NO_3 concentration of 0.4 $mol_c\ m^{-3}$ by the precipitation surplus (see Eq. 7).

Calculation of critical acidity leaching fluxes for the protection of soil quality and tree roots:
The critical Al leaching flux related to a negligible depletion of Al hydroxides was calculated as:

$$Al_{le(crit)} = fs_{Ca} \cdot Ca_{we} + fs_{Mg} \cdot Mg_{we} + fs_K \cdot K_{we} + fs_{Na} \cdot Na_{we} \qquad (9)$$

where fs_{Ca}, fs_{Mg}, fs_K and fs_{Na} are the stoichiometric ratios of the base cations with respect to Al ($mol_c\ mol_c^{-1}$). For Dutch soils these factors are set to 3, 0.6, 3 and 3 respectively, since K and Na are mainly released by microcline and albite, Ca by anorthite and Mg by chlorite in sandy soils (De Vries and Breeuwsma 1986). The critical Al concentration was calculated by dividing the critical Al leaching flux derived from Eq. 9 by the precipitation surplus. The critical H concentration was calculated from the critical Al concentration using Eq. 8. The critical H leaching flux, $H_{le(crit)}$, was calculated by multiplying the precipitation surplus with the critical H concentration in the soil solution.

The development of empirical relations between Al and H concentrations used data on soil solution concentrations, measured at four different depth in 200 forested sites on sandy soils, 38 on non-calcareous clay soils, 40 on loess soils and 30 peat soils (Leeters et al. 1994, Klap et al. 1998). For these sites Al^{3+} activity was calculated from the total concentration of Al and dissolved organic carbon (DOC) using the speciation program MINEQL+ (Schecher and McAvoy 1994), combined with a triprotic organic acid model in which complexation of Al by DOC is taken into account (Santore et al. 1995). More information on the derivation is given in Van der Salm and De Vries (2001). An overview of the values for K_{emp} and n is given in Table NL-2.

The critical Al leaching fluxes for soil quality are strongly dependent on the base cation weathering rates (Eq. 9). The SMB model thus calculates high critical Al leaching fluxes in soils with high weathering rates and consequently critical H leaching fluxes are also high (Eq. 8). These leaching fluxes are substantially higher than current acidity leaching fluxes at soils with a high base saturation. The calculated critical loads will thus lead to a decrease in base saturation and a decline in pH of these soils, leading to possible adverse effects such as Mg deficiency in the vegetation. To avoid a decrease in base saturation, a second criterion was added which limits the critical acidity leaching to the current acidity leaching. The current acidity leaching is calculated from the actual pH of the soils and the equilibrium relation between Al and H in the soil solution (Eq. 8).

The critical Al leaching flux for root protection was calculated using a criterion for the maximum tolerable Al:BC ratio in the root zone:

$$Al_{le\,crit} = 3 \cdot (Al:BC)_{crit} \cdot (BC_{td} + BC_{we} + BC_{gu}) \qquad (10)$$

where $(Al:BC)_{crit}$ is the acceptable molar Al:BC ratio in the root zone. The critical Al:BC ratios are based on a function given by Sverdrup and Warfvinge (1993) and are strongly dependent on the average BC concentration in the soil solution. To derive critical Al:BC ratios for Dutch soils, average BC concentrations based on measurements at 150 forested sites in the Netherlands were used.

An overview of the critical Al:BC ratios used is given in Table NL-3. More information is given in Van der Salm and De Vries (2001).

Table NL-3. Critical limits used for Al:BC ratios.

	Al:BC	
Tree Species	**Original[1]**	**Dutch sandy soils[2]**
Pine	0.8	3.0
Spruce[3]	2.0	3.9
Deciduous[4]	1.7	4.5
Heather	0.8	5.0[5]
Grassland	1.0	3.3[5]

1 Values given by Sverdrup and Warfvinge (1993), using a BC concentration of 0.05 mmol l^{-1}. Those data were used to calculate the critical loads submitted to the CCE.

2 Values calculated using the median BC concentration (mmol l^{-1}) in the upper part (0–30 cm) of the root zone of Dutch sandy soils.

3 The SMB model considers the tree class Spruce consisting of Norway Spruce and Douglas fir, critical Al:BC limits are taken as the average of the values for Norway spruce and Douglas fir.

4 The SMB model considers the tree class deciduous trees; critical Al:BC limits were taken as the average of the values for oak and beech.

5 This value is based on the range in measured critical Al:BC ratios at a growth reduction of 80%.

Calculation of critical nitrogen leaching fluxes for the protection of wood growth:

The SMB calculations of the critical loads for nitrogen for protection of wood growth are based on the growth uptake of nitrogen, the long-term immobilisation and denitrification and, the critical NO_3 leaching flux. N uptake is calculated from the optimal nitrogen content of stems together with information about the optimal growth rate. The critical nitrogen leaching flux is derived from a relationship between the optimal N content in forest in relation to growth and the nitrate concentration in soil water, using both literature and empirical data for 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 diseases, are small. As an alternative, the nitrate leaching flux has been estimated from the amount of N mineralised and deposited on the forest during the dormant winter season. Ultimately, the minimum value of both estimates was used to derive a critical load. Critical values for the N content in foliage (as used in the submission to the CCE) are 1.8% for pine forest, 1.7% for spruce forest and 2.5% for deciduous forest (Albers et al. 2000).

The dynamic SMART-MOVE model for calculating critical loads for the protection of plant species composition:

Critical loads for the protection of plant species composition of the Dutch nature conservation targets were calculated with the dynamic models SMART2-MOVE (Van Hinsberg and Kros, Part II of this report).

The calculation was made in two steps:

1. In the first step, critical limits for the different nature conservation targets were derived, with the model MOVE (Van Hinsberg and Kros, this report). The critical limits were based on plant species specific information on habitat preferences for nitrogen availability and soil pH.
2. The dynamic soil model SMART2 was then used to calculate the critical loads at which the above critical limits were not exceeded (Van Hinsberg and Kros, this report). It was assumed that under the critical deposition levels, neither the critical limit for nitrogen availability nor soil pH was exceeded. To calculate critical loads, it was assumed that the ratios between NO_x, NH_x, and SO_x deposition were similar to the present (1995) condition. In cases where, under these conditions, no critical loads could be calculated the lowest empirical critical loads from similar ecosystems (Bobbink et al. 1996) were applied. Both steps are described in more detail in Van Hinsberg and Kros (contained in Part II of this report).

The dynamic AquAcid model for calculating critical loads for the protection of plant species composition in heathland lakes:

Critical loads for the protection of plant species composition in fens were calculated with the dynamic model AquAcid. The model describes the growth of vegetation as the competition between two species: *Littorella uniflora*, one of the characteristic species, and *Juncus bulbosus*, a species which can be very abundant in acidified lakes (Wortelboer 1990). The availability of carbon dioxide in water and sediment plays an important role in the competition between these two species, and in the changes in vegetation which have occurred over the past decades, high concentration of carbon dioxide favouring the domination by *Juncus bulbosus*. The simulations were started from a situation in which a sandy sediment was assumed.

Since the lakes are rather small (1–50 ha), the surrounding environment plays an important role in determining the critical loads, the main factors being:

1. High load of organic matter from vegetation and trees on the shores to the lakes at high levels of nitrogen deposition. This gives rise to higher mineralisation levels and higher carbon dioxide concentrations in the water.
2. Accumulation of carbon dioxide in the water by the diminished exchange between the water and atmosphere due to surrounding forest.

Critical loads were estimated from model runs over periods of 30 years at fixed levels of atmospheric deposition. The ratio between sulphur and nitrogen in the deposition applied was varied in accordance with the current ratio and the ratio in background natural deposition estimates. The simulation were started from a situation in which a sandy sediment was assumed. The level of nitrogen deposition was varied and the effect on the vegetation development was recorded. The critical nitrogen deposition level was estimated as the level at which the species *Littorella uniflora* contributed 90% of total vegetation biomass, with exceedence of the critical level occurring at lower relative biomass. Thus the critical level is an indication of the potential of these lakes to sustain a healthy population of the rare isoetid plant species.

Data sources

Vegetation and soil types distinguished: Both SMB and SMART2-MOVE calculate critical loads at a 250×250m^2 grid scale. The specification of vegetation-soil combinations in the 250×250m^2 grids were derived from an overlay of the 1:50,000 soil map and a vegetation map based on both satellite observations (LGN) and several additional detailed vegetation surveys. Five types of vegetation and seven major soil types were distinguished (Van Hinsberg and Kros, this report). Regarding vegetation types, we distinguished three groups of tree species (deciduous forests, pine forests and spruce forests; see Table NL-2), grassland and heathland. Soil types were differentiated among two non-calcareous sandy soils, one calcareous sandy soil, loess soil, non-calcareous clay soil, calcareous clay soil and peat soil. In the SMB calculations the loess, peat, and non-calcareous clay soils were, however, further subdivided into three, five, and four sub-types respectively (Van der Salm 1999). All these soil types were further subdivided in five hydrological classes depending on the height and the seasonal fluctuations of the water table. Parameterisation was held similar for the same processes in both SMB and SMART2 (Kros et al. 1995). Table NL-4 describes the ecosystems for which critical loads were calculated.

Data for all vegetation-soil combinations within each grid cell were derived by using relationships with basic land characteristics such as tree species and soil type, which were available in geographic information systems.

Table NL-4. Ecosystems for which critical loads were calculated.

Ecosystem	Key species
Deciduous forest	e.g. *Quercus spec., Betula spec., Fagus spec.* and species from ground vegetation
Pine forest	e.g. *Pinus sylvestris* and species from ground vegetation
Spruce forest	e.g. *Pseudotsuga menziesii* and species from ground vegetation
Grassland (semi-natural)	Several species depending on moisture status (wet-dry), soil acidity (acid – calcareous), and nutrient availability (nutrient-poor – nutrient-rich)
Heathland (dry, wet and bogs)	Wet heathlands: e.g. *Erica tetralix;* Dry heathlands: e.g. *Caluna vulgaris*
Heathland lakes (fens)	*Littorella uniflora*

Base cation deposition: Bulk deposition data for base cations for a 1×1 km^2 grid were interpolated from 14 monitoring stations for 1993. However, bulk deposition only includes wet deposition (and a very small part of dry deposition). Dry deposition was calculated by multiplying base cation concentrations in the bulk (wet) deposition by a scavenging ratio to estimate air concentrations, which in turn were multiplied by a deposition velocity, depending on meteorology and land use, using the model DEADM (Erisman and Bleeker 1995). An estimate of seasalt inputs of Cl and SO_4 was made by assuming an equivalent Cl:Na and SO_4:Na ratio in both bulk deposition and dry deposition equal to these ratios in seawater, namely 1.165 for Cl and 0.116 for SO_4. Both Cl and seasalt SO_4 were subtracted from the total base cation deposition values to derive seasalt-corrected base cation inputs.

Weathering rates: SMB weathering rates for the distinguished loess, clay, and peat sub soil types were calculated from pedotransfer functions relating weathering rates to the silt and clay contents of the soils (Van der Salm 1999). The pedotransfer functions for loess and clay soil were based on laboratory experiments. Weathering rates for peat soils were estimated using pedotransfer functions for clay soils and the clay content of peat soils.

Uptake: To derive critical acid loads, uptake rates of nitrogen and base cations were calculated based on the concept of nutrient-limited uptake, defined as that uptake that can be balanced by a long-term supply of base cations. This value, referred to as the critical base cation uptake, $BC_{gu(crit)}$, is calculated from mass balances for each base cation (Ca, Mg and K) separately, as total deposition and weathering minus a minimum leaching of BC. We used a minimum leaching of 50 mol_c ha^{-1} yr^{-1} for Ca and Mg and 0 mol_c ha^{-1} yr^{-1} for K. From the critical base cation uptake, the corresponding critical N uptake, $N_{gu(crit)}$, was calculated from the ratio between each cation and nitrogen in the biomass (cf. Posch et al. 1993, Eqs. 4.7 and 4.8).

Nitrogen immobilisation: The critical N immobilisation rate is calculated by accepting a change of 0.2% of nitrogen in organic matter in the upper soil layer (0–30 cm) during one rotation period (100 years). The pool of organic matter (kg ha^{-1}) in this layer is calculated by multiplying the thickness of the soil layer (0.3 m) by the bulk density of the soil layer (kg m^{-3}) and the fraction of organic matter. Bulk density is calculated as a function of organic matter and clay content (cf. van der Salm et al. 1993). Data for the contents of clay and organic matter are based on field surveys of 250 forest soils (150 sandy soils, 40 loess, 30 clay and 30 peat). Immobilisation rates increase with higher organic matter contents, and generally range between 100 and 350 mol_c ha^{-1} yr^{-1}. These values correspond well with the range (2–5 kg ha^{-1} yr^{-1}) mentioned in the Mapping Manual (UBA 1996).

Denitrification: Denitrification fractions were derived for each soil type based on data in Breeuwsma et al. (1991) for agricultural soils. These data were corrected for the more acidic forest soils. Values thus derived varied between 0.1 for well-drained sandy soils to 0.8 for peat soils (De Vries 1996).

Precipitation and evapotranspiration: Precipitation estimates have been derived from 280 weather stations in the Netherlands, using interpolation techniques to obtain values for each grid. Interception fractions, relating interception to precipitation, have been derived from the literature for all tree species considered. Data for evaporation and transpiration have been calculated for all combinations of tree species and soil types with a separate hydrological model (De Vries 1996).

Comments and uncertainties

Simple Mass Balance method: Uncertainties in critical loads calculated with the SMB model are due to both uncertainties in the input data and in the critical limits used. Uncertainties in the given site-specific rates for base cation deposition, weathering, uptake, nitrogen immobilisation, denitrification and precipitation excess have previously been estimated at approximately 50% (De Vries et al. 1994, 2000). In several cases, however, the uncertainties in critical limits, such as critical foliar N concentrations and critical Al:BC ratios (see Table NL-3) are more important. Quantitative information on the impact of changes in those criteria, including a discussion, is given in Albers et al. (2000).

Dynamic modelling: Differences between the SMB method and the dynamic models SMART2-MOVE and AquAcid make it less straightforward to calculate the desired CCE parameters in SMB terms. The critical load for nitrogen as calculated with the dynamic models is comparable with the definition of

$CL_{nut}(N)$; however, this refers only to effects of changes in nutrient availability, whereas the dynamic models also take (in)direct effects of acidification into account. In this respect the critical loads calculated with the dynamical models resemble the empirical critical loads.

Moreover, the calculation methods also differ since additional dynamic processes and feedback mechanisms are taken into account. An important difference between SMB and SMART2-MOVE is that nitrogen availability (which plays a dominant role in the calculation of the critical loads for nitrogen) is also affected by changes in soil pH. In SMART2, changes in soil pH influence the mineralisation flux, which in turn affects nitrogen availability (Kros et al. 1995). Changes in soil pH can therefore affect the tolerable deposition of nitrogen. Thus, the critical acid and nitrogen deposition cannot be calculated separately using dynamic models.

To derive a critical acid load, we assumed a constant N:S ratio of 2:1, equal to the average present ratio in deposition. The critical load of acidity (S+N) thus equalled 1.5 times the critical load of nutrient nitrogen. In order to allow calculations with the results according to the CCE procedures, we assumed that $CL_{max}(S)$ equals $CL(A)$, and calculated $CL_{max}(N)$ according to Eq. 5 assuming a negligiblke N uptake, a low constant N immobilisation of 1 kg ha^{-1} yr^{-1} and a low constant denitrification fraction of 0.1.

For heathland lakes, it must be mentioned that uncertainties in the calculations are quite large, due to the fact that the estimation of the local depositions and the calculations of the expected water chemistry and plant species are not validated yet. The uncertainty in the critical loads for nitrogen is estimated to be about 50%. For terrestrial vegetation, the ranges of empirical critical loads might be applied.

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National maps produced

Critical load maps for surface waters, forest soils and nutrient nitrogen to vegetation have been produced. National maps were last published in Henriksen and Buan (2000). There are no major changes in the data since the CCE Status Reports of 1997 and 1999 (Posch et al. 1997, 1999).

A. Surface waters

Calculation methods

The SSWC method was used to calculate the critical loads of acidity using variable ANC_{limit} (Henriksen et al. 1995). $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ were computed according to the FAB model (Posch et al. 1997).

Grid size: Each 1° longitude × 0.5° latitude grid was divided into 4×4 subgrids, each covering about 12×12 km^2 in southern Norway (with decreasing grid width at higher latitudes). The land area covered by each of the 2315 grids has been calculated.

Data sources

Data from national regional lake surveys and monitoring programs were used. For precipitation a weighted average total deposition value for each EMEP50 grid cell has been calculated from ambient air concentrations and wet deposition, taking land use data (coverage of different receptors) into account (Tørseth and Pedersen 1994). Weighted means for the period 1988–1992 were used. For base cations, the non sea-salt fraction was multiplied by 0.7 to account for reductions in anthropogenic base cation emissions since 1990. Deposition values for each grid cell were estimated from the NILU data base.

Surface water chemistry within a subgrid was estimated from available water chemistry data for rivers and lakes within each grid. The chemistry of the lake that was judged to be the most typical was chosen to represent the grid. If there were wide variations within a subgrid, the most sensitive area was selected if it amounted to more than 25% of the grid's area. Sensitivity was evaluated on the basis of water chemistry topographical and geological maps (1:1,000,000; Norwegian Geological Survey). Mean annual runoff was from runoff maps prepared by Norwegian Water and Energy Works. The database was last revised in 1996.

B. Forest soils

Calculation methods

The dynamic acidification model MAGIC (Cosby et al. 1985) was used to calculate the critical loads of acidity. The criterion used was an Ca:Al ratio of 1.0 in the upper 60 cm of the soil. The FAB model was used to derive CL_{max}(S), CL_{min}(N) and CL_{max}(N).

Grid size: The same grid system as for surface water was used. Of the total 2315 grids, 706 grids are in productive forest (birch, spruce and pine). The remaining area has unproductive forest and critical loads for forest soils have not been calculated.

Data sources: Soil data were based on data from the national forest monitoring plots (9×9 km^2; Norwegian Institute for Land Inventory - NIJOS). Surface water data (described above) was used. Vegetation uptake data was estimated based on forest monitoring data from the Norwegian Institute for Forest Research.

C. Nutrient Nitrogen – Vegetation

Calculation methods

Critical loads have been estimated from empirically-derived relationships between N deposition and vegetation type (Esser and Tomter 1996). The following vegetation types and critical load values have been used:

- Ombrotrophic bog: 5 kg N ha^{-1} yr^{-1}
- Coniferous forest: 7 kg N ha^{-1} yr^{-1}
- Deciduous forest: 10 kg N ha^{-1} yr^{-1}
- Calluna heath: 15 kg N ha^{-1} yr^{-1}
- Others 20 kg N ha^{-1} yr^{-1}

Assignment of areas to the different critical loads: The area assigned to the surface water critical load data for each grid adds up to the total area of Norway (approx. 323,000 km^2). The total area for the grids with forest soils critical loads adds up to the total area of productive forest (72,700 km^2). The total area of the nutrient nitrogen to vegetation critical loads totals approximately 70% of the total area of Norway, i.e. the area of Norway covered by mountains and heathlands.

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Calculation methods

Maximum critical loads of sulphur, minimum critical loads of nitrogen, maximum critical loads of nitrogen and critical loads of nutrient nitrogen have been calculated for forest ecosystems by using the Simple Mass Balance method outlined in the Mapping Manual (UBA 1996), and partly by use of the PROFILE model. In comparison to the calculations reported in the CCE Status Report 1999, (Posch et al. 1999), the following revisions have been made:

- A 1×1 km^2 grid resolution has been applied to a total of 88,383 sub-ecosystems representing almost all forests in Poland.
- The coniferous and deciduous trees specification has been replaced by six dominating tree species.
- Changes to input data are shown in Table PL-1.

Exceedance of critical loads

Use of the SONOX model (Mill 2001) indicated temporal variations in the exceedance of critical loads for sulphur and nitrogen, $CL(S+N)$, and the exceedance of critical load of nutrient nitrogen were calculated and mapped for the period 1980–1998. The results show the considerable progress in the mitigation of ecological effects to forest ecosystems of sulphur and nitrogen emissions in Poland during this period. The observed trend suggests a real prospect to achieve a protection level required by the Gothenburg Protocol obligations for Poland in the coming years.

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Table PL-1. Summary of critical loads input and output data for Polish forest ecosystems.

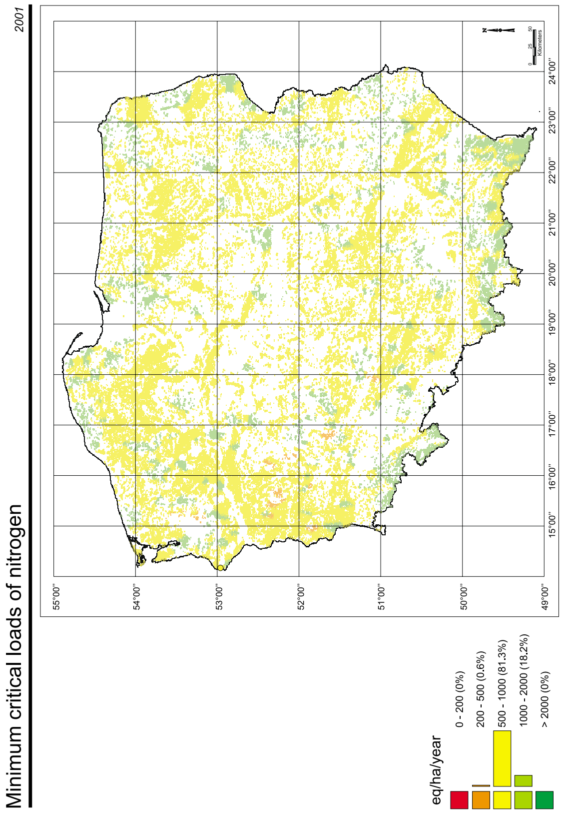
Tree	Min	Mean	Max			
Parameter	**Species ***	**value**	**value**	**value**	**Methods used**	**Data source**
	p	1189	1198	7246		
	s	26	965	5047		
$CL_{max}(S)$	f	894	1823	5545	$CL_{max} = BC_{dep} + BC_w$	Mapping Manual.
(eq ha^{-1} yr^{-1})	o	26	843	8113	$-BC_u - ANC_{le(crit)}$	
	bch	45	1620	8799		
	br	26	1693	7503		
	p	621	678	906		
	s	850	1014	1135		
$CL_{min}(N)$	f	686	826	971	$CL_{min}(N) = N_i + N_u$	Mapping Manual.
(eq ha^{-1} yr^{-1})	o	1530	1590	1815		
	bch	1228	1347	1513		
	br	474	559	759		
	p	727	3469	24775		
	s	972	3365	17744		
$CL_{max}(N)$	f	2510	5461	19177	$CL_{max}(N)=CL_{min}(N)+$	Mapping Manual.
(eq ha^{-1} yr^{-1})	o	1618	3612	28643	$CL_{max}(S) / (1 - f_{de})$	
	bch	1390	5562	30628		
	br	597	4579	25521		
	p	624	777	2271		
	s	878	1285	2194		
$CL_{nut}(N)$	f	743	1117	1570	$CL_{nut}(N)=N_i+N_u$	Mapping Manual.
(eq ha^{-1} yr^{-1})	o	1532	1644	2109	$+N_{le(acc)} / (1 - f_{de})$	
	bch	1232	1499	2465		
	br	479	626	1228		
	p	220	436	1015	Distribution of base	Draaijers et al. 1995.
	s	221	429	965	cation deposition	
BC_{dep}	f	260	418	599	derived from RIVM	
(eq)	o	220	407	1014	model calculations.	
	bch	254	441	831		
	br	220	390	996		
	p	406	406	406	Uptake of base cations	Elements content in
	s	713	713	713	related to six major tree	stems and branches
(eq ha^{-1} yr^{-1})	f	656	656	656	species was calculated	provided by the Polish
BC_u	o	1029	1029	1029	as the minimum of	Academy of Science,
	bch	708	708	708	growth limited uptake	Institute of Dendrology
	br	365	365	365	andnutrient-limited	(Fober 1986); Forest
					uptake.	growth rates obtained
						from databank of the
						Forest Mgmt. and
						Geodesy Office.
						Consultant: Józef
						Zwoliński.
	p	0	445	2557	Calculated according	Soil samples delivered
	s	0	560	2443	to Mapping Manual.	by the Forest Research
BC_w	f	244	589	2443	The PROFILE model	Institute (Wawrzoniak
(eq ha^{-1} yr^{-1})	o	0	546	2557	has also been applied	et al. 2000); Mineral
	bch	0	623	2557	for a limited number	composition analysis
	br	0	502	2500	of sites.	(Stępniewski 1998).
						Consultant: Stanislaw
						Drzymala.

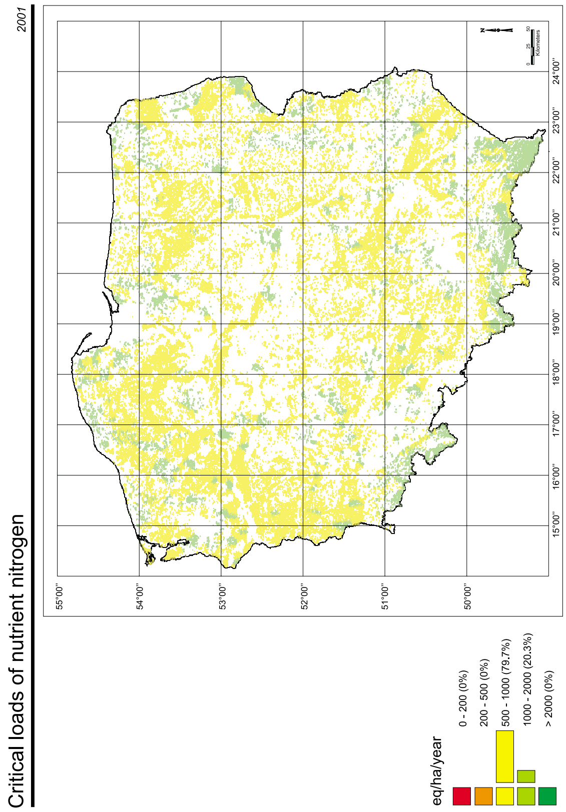
*p = pine o = oak s = spruce bch= beech f = fir br = birch

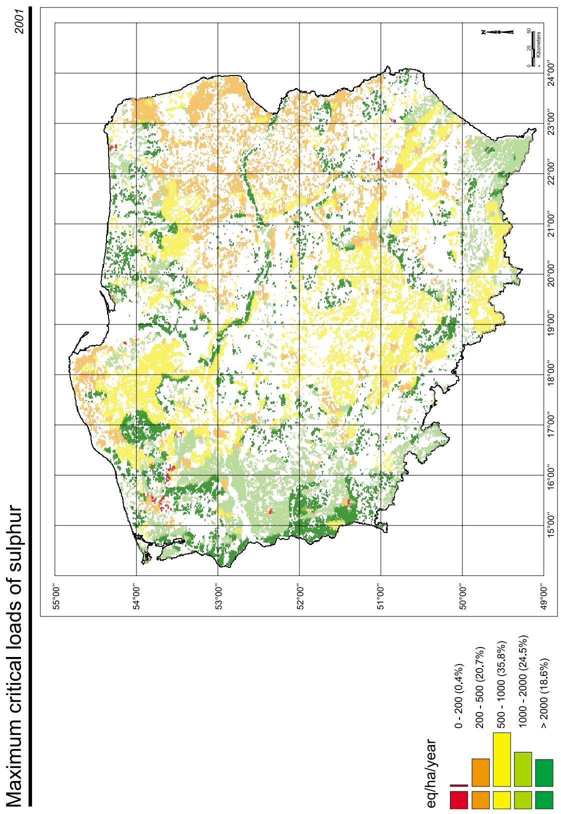
Table PL-1 (continued). Summary of critical loads input and output data for Polish forest ecosystems.

Tree	**Min**	**Mean**	**Max**			
Parameter	**Species ***	**value**	**value**	**value**	**Methods used**	**Data source**
	p	50	1258	13600	Calculated as the	Hydrological Atlas of
	s	300	3285	9000	difference between 30-	Poland (Stachy et al.
Runoff	f	300	3225	6000	year mean	1986).
(m^3 ha^{-1} yr^{-1})	o	50	1077	4200	precipitation *P* and	
	bch	50	2770	13600	evapotranspiration	
	br	100	1351	6700	*EVP*: PS = P – EVP	
					(m^3 ha^{-1} yr^{-1}).	
	p	300	300	300	300 m^6 eq^{-2}, as	Mapping Manual.
	s	300	300	300	recommended in the	
K_{gibb}	f	300	300	300	Mapping Manual.	
(m^6 eq^{-2})	o	300	300	300		
	bch	300	300	300		
	br	300	300	300		
	p	18	729	4129	Calculated via SMB	Mapping Manual.
	s	16	617	3010	equation with Bc:Al	Consultant: Józef
$ANC_{le(crit)}$	f	651	1446	3464	ratios differentiated	Zwoliński.
(eq ha^{-1} yr^{-1})	o	3	500	5824	for six tree species.	
	bch	21	1163	6301		
	br	26	1151	4995		
	p	71	124	356	A temperature-	CCE Status Report
	s	71	235	356	dependent long-term	1995.
N_I	f	71	211	356	immobilisation factor	
(eq ha^{-1} yr^{-1})	o	71	131	356	was applied, ranging	
	bch	71	190	356	from 71 to 356	
	br	71	155	356	eq ha^{-1} yr^{-1}.	
	p	550	550	550	Uptake of nitrogen	Elements content in
	s	779	779	779	related to six major	stems and branches
N_u	f	615	615	615	tree species was	provided by the
(eq ha^{-1} yr^{-1})	o	1459	1459	1459	calculated as the	Polish Academy of
	bch	1157	1157	1157	minimum of growth	Science, Institute of
	br	403	403	403	limited uptake and	Dendrology (Fober
					nutrient-limited uptake.	1986). Forest growth
						rates obtained from
						data bank of the Forest
						Mgmt. and Geodesy
						Office. Consultant:
						Józef Zwoliński.
	p	0.5	0.6	0.7	Values from 0.5 to 0.7	Mapping Manual.
	s	0.5	0.6	0.7	were applied,	
f_{de}	f	0.5	0.6	0.7	depending on soil	
	o	0.5	0.6	0.7	type.	
	bch	0.5	0.6	0.7		
	br	0.5	0.6	0.7		
	p	2	44	480	Limiting N concent-	CCE Status Report
	s	11	116	318	rations for coniferous	1993.
$N_{le(acc)}$	f	11	114	212	(0.0143 eq m^{-3}) and	
(eq ha^{-1} yr^{-1})	o	1	23	88	deciduous	
	bch	2	58	286	(0.02 eq m^{-3}) trees.	
	br	2	28	141		

*p = pine o = oaks = spruce bch = beech f = fir br = birch







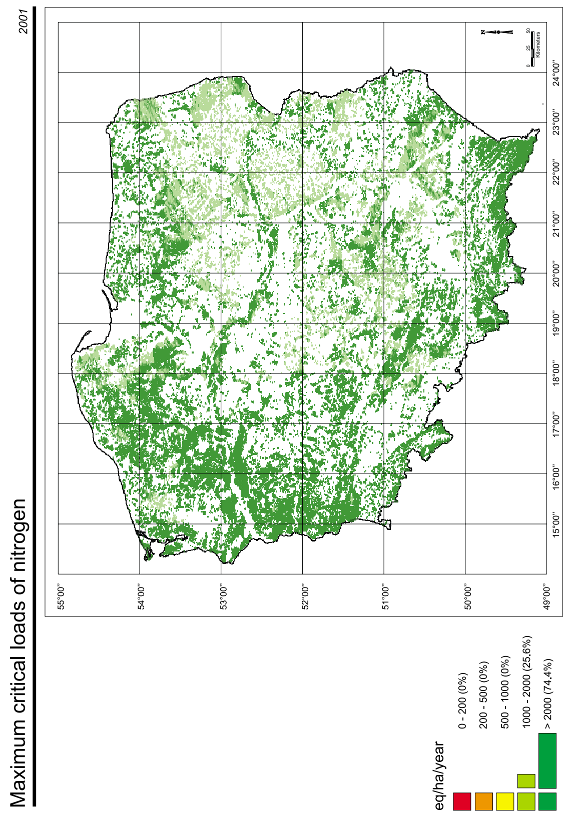


Figure PL-1. Maximum critical loads of sulphur, minimum critical loads of nitrogen, maximum critical loads of nitrogen, and critical loads of

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Status of critical loads data

No response was received to the most recent call for data from the CCE. Thus the 1998 critical loads database has been adapted to the new EMEP co-ordinate system by the CCE, and has been included into the European database. For a description of the national data, see the NFC report in the CCE Status Report 1999.

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Status of critical loads data

In response to the most recent call for data, the NFC informed the CCE that no revisions to previous critical loads data were to be submitted. Thus the 1998 critical loads database has been adapted to the new EMEP co-ordinate system by the CCE, and has been included into the European database. For a description of the national data, see the NFC report in the CCE Status Report 1999.

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Calculation methods

The methods for calculating critical loads of acidity and nutrient nitrogen for forest ecosystems in Slovakia have been described previously (Posch et al. 1999, Mindas et al. 1999).

The ranges of input data and calculated values are given in Table SK-1. For most parameters Mapping Manual guidelines have been followed in Slovakia. The main exceptions to these calculations are:

- using the Simple Mass Balance equation with the ratio of Bc:Al as the chemical criterion for soil acidification (tree species-specific data based on Sverdrup and Warfvinge 1993)
- using Bc deposition values for the period 1990–2000.
- using values for nitrogen immobilisation in relation to mean annual temperature (altitude).
- using $[N]_{crit}$ = 0.0143 eq m^{-3} for coniferous and $[N]_{crit}$ = 0.0215 eq m^{-3} for deciduous forests.

The justification for applying these values and / or methods in Slovak forest ecosystems are also documented in Table SK-1.

Revisions to critical loads data

SK critical loads data for 2001 were submitted to the CCE in April-May 2001. The following information describes the main changes made to the critical load data sets for Slovak forests from the 1999 to 2001 submissions:

- Precipitation surplus (Q) was recalculated for the Slovak lowlands that have low values for precipitation and high amounts of potential evapotranspiration (minimum Q data).
- Nitrogen and base cation uptakes were recalculated according to direct N and BC uptake analyses for 111 forest monitoring plots (from the ICP Forests 16×16^2 km network; direct analyses of N, Ca, Mg, K contents in wood and bark, direct estimation of wood stock and increment).

Table SK-1. Summary of Slovakian critical load values and the justification for their use.

CL parameter	Eco-system	Min Value	Max Value	Data sources / Methods used (same for both forest categories)	Justification (same for both categories)
$CL_{max}(S)$	c	580	17639	$= CL(\mathrm{A}) + BC_{dep} - Bc_{up} + Bc_w$	Mapping manual.
eq ha^{-1} yr^{-1}	d	144	26715		See comments on BC_{dep}, Bc_{up}, Bc_w.
$CL_{min}(N)$	c	44	747	$= N_u + N_i$	Mapping Manual.
eq ha^{-1} yr^{-1}	d	26	858	as in Mapping Manual	See comments on N_{up}.
$CL_{max}(N)$	c	1122	33049	$= CL_{min}(N) + CL_{max}(S)/(1-f_{de})$	Mapping Manual.
eq ha^{-1} yr^{-1}	d	410	46928		
$CL_{nut}(N)$	c	125	1056	$= N_u + N_i + N_{le,acc}/(1-f_{de})$	Mapping Manual.
eq ha^{-1} yr^{-1}	d	28	1418		See comments on N_{up}.
BC_{dep}	c	473	1854	BC_{dep} = measured mean data	Mapping Manual.
eq ha^{-1} yr^{-1}	d	478	1866	1990–2000.	EMEP data + intensive forest monitoring deposition data, canopy budget deposition model.
Bc_{up}	c	7	448	Calculated from average volume.	Mapping Manual.
eq ha^{-1} yr^{-1}	d	45	1372	increment of wood and bark concentration of BC in wood and bark, average volume increment was obtained from the national forest inventory database, values based on individual trees according to forest site classification	Direct analyses of Bc_{up} for 111 forest monitoring sites (ICP Forests).
BC_w	c	50	7000	Weathering data, soil-dependent.	PROFILE model
eq ha^{-1} yr^{-1}	d	50	7000	Type, parent material, soil texture and root zone depth established according to De Vries et al. (1993).	
Q	c	0	1404	Q = Precipitation –	GIS tools.
mm				evapotranspiration from forests	
	d	0	1380	(values obtained from digitizing maps from Slovak Hydro-meteorological Institute)	Evapotranspiration model.
K_{gibb}	c	200	500	Data based on main forest types	Mapping Manual.
m^6 eq^{-2}	d	200	500	Sverdrup and Warfvinge (1993)	
$-ANC_{le}$	c	198	9998	$= Q \cdot ([\mathrm{Al}]_{crit} + [\mathrm{H}]_{crit})$	Mapping Manual.
	d	6	19215		
N_{im}	c	20	350	Values assigned to mean annual	Mapping Manual.
eq ha^{-1} yr^{-1}	d	20	350	temperature (altitude) at forest sites	
N_{up}	c	4	397	Calculated from: average	Mapping Manual.
eq ha^{-1} yr^{-1}	d	6	689	volume increment of wood and bark × concentration of N in wood and bark, average volume increment obtained from national forest inventory database, values based on individual trees according to forest site classification	Direct analyses of N_{up} for 111 forest monitoring sites (ICP Forests).
f_{de}	c	0.1	0.8	f_{de} = 0.1 for loess soils and sandy soils without gleyic features	Mapping Manual.
	d	0.1	0.8	f_{de} = 0.5 for sandy soils f_{de} = 0.7 for clay soils f_{de} = 0.8 for peat soils (De Vries et al. 1993)	
$N_{le,acc}$	c	0	201	$= Q \cdot [\mathrm{N}]_{crit}$	Mapping Manual.
eq ha^{-1} yr^{-1}	d	0	297		

# CCE ecosystem codes:
c = coniferous forests
d = deciduous forests

Results

The present report describes the progress made in implementing the critical load approach within the Slovak Republic. The improved data set based on the national meteorological, hydrological, air pollution, forest inventory and remote sensing data was used. The values of $CL_{min}(N)$, $CL_{nut}(N)$ and $CL_{max}(S)$ for forest ecosystems over Slovakia are summarised in Table SK-2.

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Table SK-2. Percentile values of critical loads (in eq ha^{-1} yr^{-1}) for nitrogen and sulphur for Slovak forests.

	$CL_{min}(N)$			$CL_{nut}(N)$			$CL_{max}(S)$		
	5%	**50%**	**95%**	**5%**	**50%**	**95%**	**5%**	**50%**	**95%**
All forests	239	463	581	317	607	789	435	2487	11039
Coniferous forests	268	454	561	357	611	761	1253	2724	8402
Deciduous forests	228	469	588	300	604	801	356	2178	12142

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Status of critical loads data

In response to the most recent call for data, the NFC informed the CCE that no revisions to previous critical loads data were to be submitted. Thus the 1998 critical loads database has been adapted to the new EMEP co-ordinate system by the CCE, and has been included into the European database. For a description of the national data, see the NFC report in the CCE Status Report 1999.

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Calculation methods

Deposition: Wet deposition of sulphur and nitrogen and air concentrations of sulphur 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 and a dispersion model to estimate the local contribution from Swedish emission sources. The spatial resolution of the model system is 20×20 km^2.

Dry deposition to forest ecosystems was estimated by inferential modelling based on model-calculated air concentration fields multiplied by dry deposition velocities. Velocities were derived from throughfall data for sulphur 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 base cations was estimated using a simple model based mainly on monitoring data on throughfall and wet deposition (Lövblad et al. 2000).

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 km^2 NILU grids.

Forest ecosystems: The critical load of acidity for forest ecosystems was calculated using the Steady-State Mass Balance approach, implemented in the PROFILE 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 $[BC]:[Al]_{crit}$ in the soil solution was used as the chemical criterion in each soil horizon and used to determine the critical ANC leaching. The following $BC:Al_{crit}$ was used; 1.0 for Norway spruce and birch, 0.8 for Scots pine and 0.6 for European beech and oaks. 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:

$$CL_{nut}(N) = N_u + N_{de} + N_{im} + N_{le(acc)}$$

N_u = long-term net uptake by the forest
N_{de} = denitrification
N_{im} = N immobilisation
$N_{le(acc)}$ = 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 was introduced to avoid long-term nutrient imbalances in forest trees. The supply of different cations from weathering was calculated using the PROFILE model.

N immobilisation, including acceptable leaching, was determined by a semi-empirical approach. Immobilisation + leaching was 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 value is based on results from N mass balance studies performed in a range of Swedish coniferous forests (Nilsson et al. 1998). The immobilisation 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 sulphur 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 µeq l^{-1} in cases where $[BC^*_o] > 25$ µeq l^{-1}. In other cases, ANC_{limit} was set to $0.75[BC^*_o]$ to allow for naturally low ANC concentrations. N immobilisation 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 ($f_{de} = 0.1 + 0.7 \cdot f_{peat}$) as suggested in Posch et al. (1997).

Mapping

The area 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 (km^2) 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 scaled up to half the cell ecosystem area. To account for areas not at risk from acid deposition, 10% of each cell area was subtracted when calculating the cell ecosystem area.

Data sources

Deposition: Monitoring data was used as input to the modelling 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 sulphur, 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 SO_2 and NO_2.
- Databases on land use and meteorology included in the MATCH model system.

Forest Ecosystems: The forest soil data 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 in productive forest land in Sweden. For this study, soil samples were collected down to ca 60 cm depth at 1804 sites representing all major forest types in Sweden. The sites were grouped in 11 classes according to the tree species composition. 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 an additional 676 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 from the Swedish Meteorological Institute (SMHI) were used. 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.

Deviations from the manual: All variables were derived according to the Mapping Manual, except for those described 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 immobilisation.
$N_{le(acc)}$ (forest soils)	0.5–1.0	0	$N_{le(acc)}$ included in the N_i term.

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Status of Swiss critical loads data

According to the workplan of the Working Group on Effects for 2001 to 2003, the Swiss critical load data are being reviewed, improved and completed especially with respect to reliability and transparency. As a first step in this process, new data sets were supplied to the CCE in February 2001, replacing the former data from December 1996. The data are provisional, as further revisions and updates are planned within the next two years. In the current data set, critical load values for alpine lake catchments are not included, but revised values will become available during the next year.

Major changes in the current data set were made for ANC_{leach} and BC_w. These parameters are now calculated with the multilayer steady-state PROFILE model, while former versions used the SMB method. As shown in the Swiss contribution to the CCE Status Report 1999, the PROFILE model results seem to be more realistic under certain circumstances such as calcareous soils, but there are also some disadvantages (e.g. a smaller number of sites). Therefore, the new data should be tested and compared with the results from other countries and further improvements could then be achieved.

The data set for forests

Information on forest ecosystems is supplied in 691 records that are related to the sampling points on the 4×4 km sub-grid of the national forest inventory (see Figure CH-1). Therefore, each point represents 16 km^2, for a total forest area of 11,056 km^2.

All records contain values for critical loads of acidity ($CL_{max}(S)$, $CL_{max}(N)$). ANC_{leach} and BC_w are the results of the regional PROFILE model application (SAEFL 1998). The values for $CL_{max}(S)$ are in a similar range as the former SMB results. Relatively high values (>4000 eq ha^{-1} yr^{-1}) can be explained by the presence of calcareous compounds even in the upper soil layers at a considerable number of sites. The unrealistically high values for ANC_{leach} and BC_w (>6000 eq ha^{-1} yr^{-1}) produced by the SMB method for calcareous soils are now very rare.

The differences between PROFILE and SMB are (1) model-inherent and (2) related to different input data. For example, to calculate BC_w using PROFILE, chemistry and mineralogy data from 700 soil samples are also used, while the SMB is primarily a simplified classification of BC_w based on a soil map.

For K_{gibb}, different values have been used for each of the four soil layers modelled in PROFILE: 6.5, 7.5, 8.5 and 9.2 (SAEFL 1998). $CL_{nut}(N)$ for forests is calculated with the SMB method (FOEFL 1996). A constant denitrification factor (f_{de}) is used in accordance with the Mapping Manual (UBA 1996). There are 42 records with no values (-1) for $CL_{nut}(N)$. These sites are supposed to be unmanaged (inaccessible sites, bush forest) and therefore, N_{upt} and BC_{upt} are zero.

The input parameters BC_{dep}, runoff, N_{imm}, N_u, BC_u and f_{de} have been described previously (FOEFL 1994). In principle they have not changed, but the assignment of the f_{de} values was improved. $N_{le(acc)}$ is newly defined as follows: 4 kg N ha^{-1} yr^{-1} in the lowlands (500 m a.s.l) with a linear decrease to 2 kg N at 2000 m a.s.l. With this, the values correspond to the recommendation of the Mapping Manual. Altitude dependence was introduced, because the rate of N cycling and tree growth generally decrease with altitude.

The ecosystem types are coded as proposed in the CCE Status Report 1999 (Posch et al., p. 16): “c” for coniferous forests (>95% coniferous trees), “d” for deciduous forests (< 5% coniferous trees), and “m” for mixed forests (5–95% coniferous trees).

The data set for natural and semi-natural ecosystems

This data set contains only values for $CL_{nut}(N)$ obtained by the empirical method according to the Mapping Manual. The data sources and and the procedure for implementing the empirical method have been described previously (FOEFL 1996).

The file contains 11,680 records representing 1 km^2 each (see Fig. CH-1). The data base is a compilation of various vector and raster data sets. Where spatial overlays with the sites in the national forest inventory occurred, only the SMB value for $CL_{nut}(N)$ was used. Thus there is no double-counting of ecosystem areas.

The ecosystem types are coded as proposed in the CCE Status Report 1999:

c = coniferous forests (natural forest only, poorly managed)
d = deciduous forests (natural forest only, poorly managed)
g = grass (species-rich and alpine grassland)
h = heath (alpine heath)
p = peat (raised bogs and mesotrophic fens)
w = waters (Littorellion)

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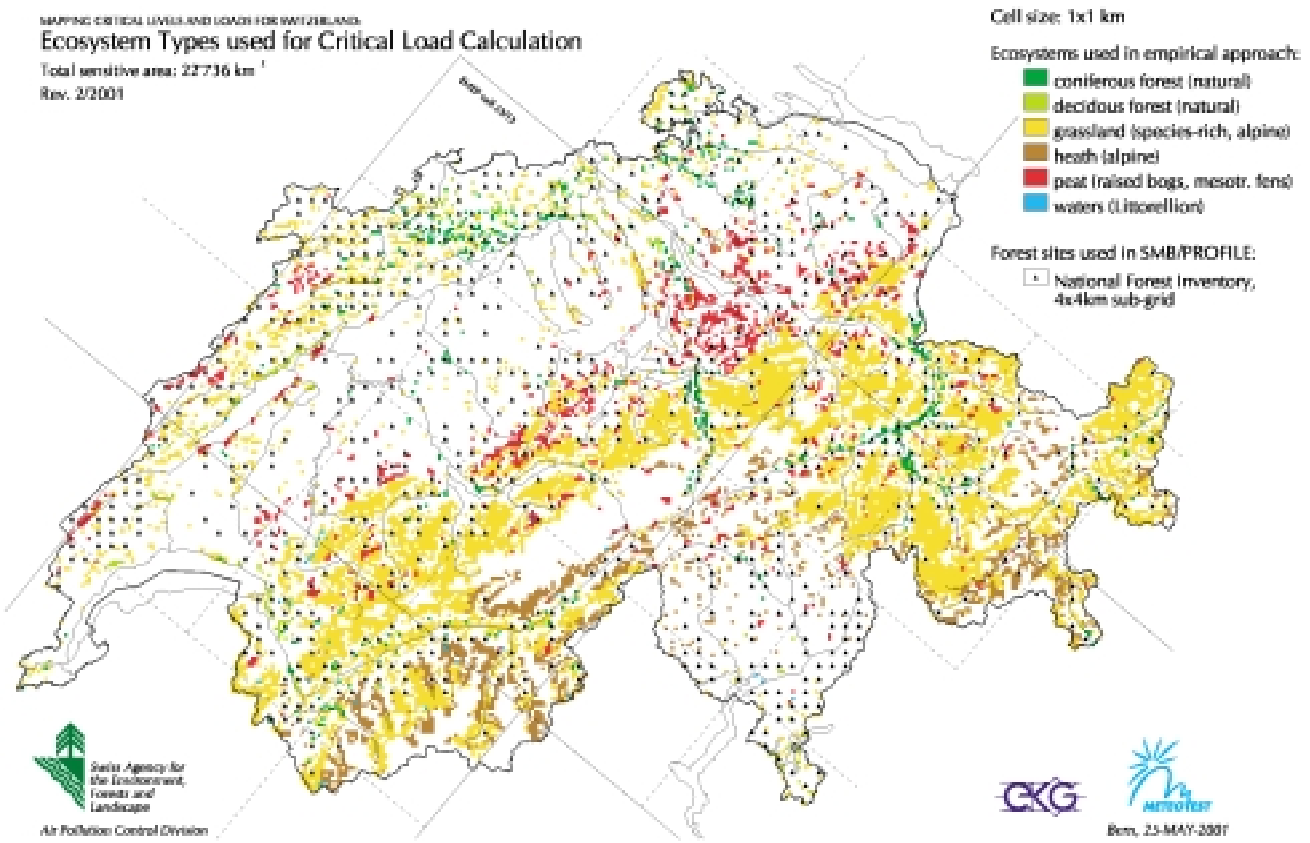

Figure CH-1. Spatial distribution of the different ecosystem types. Data sources: National forest inventory, WSL 1992, Hegg et al. 1993.

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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, UK NFC report in Posch et al. 1997). However, some revisions have been made to the UK critical loads data submitted to the CCE in February 2001. Reports on the methods used in the UK are provided on the UK NFC web site: critloads.ceh.ac.uk.

Revisions made to UK critical loads data

The changes made to the UK calculations are listed below:

The Simple Mass Balance equation for woodland ecosystems:

(i) Criteria: critical molar ratio of Ca:Al = 1 for mineral soils; critical pH 4.0 for organic soils; empirical critical loads for peat soils.
(ii) K_{gibb} values: 950 m^6 eq^{-2} for mineral soils and 9.5 m^6 eq^{-2} for organic soils.
(iii) 1995–97 5km total (marine plus non-marine) calcium deposition, updated from 1992–94 20km data.
(iv) New calcium uptake values for coniferous and deciduous trees.

Maximum critical load for sulphur:

(i) New acidity critical loads for woodland ecosystems.

(ii) 5km non-marine base cation deposition values for 1986–91 as long-term mean values, instead of previous 1992–94 20km data.

(iii) 5km non-marine chloride deposition values for 1986–91 as long-term mean values, instead of previous 20km estimates for 2010.

(iv) New base cation uptake values for woodland ecosystems.

Minimum critical load for nitrogen:

(i) New nitrogen uptake values for woodland ecosystems.

(ii) The inclusion of denitrification into the equation (not previously included).

Maximum critical load for nitrogen:

Changes in values due to changes made to the maximum critical load for sulphur and the minimum critical load for nitrogen.

Maximum critical load for nitrogen:

Changes in values due to changes made to the maximum critical load for sulphur and the minimum critical load for nitrogen.

Critical loads of nutrient nitrogen:

(i) The minimum of empirical or mass balance critical loads applied to both coniferous and deciduous woodland ecosystems (previously only applied to conifers).

(ii) New nitrogen uptake values for mass balance critical loads for woodland ecosystems.

Critical loads for freshwater ecosystems:

(i) The FAB critical loads now applied to 1470 sites in Great Britain and 140 sites in Northern Ireland.

(ii) Runoff data for 1961–1990 for Northern Ireland (1941–1960 data still used for Great Britain).

(iii) Forest areas in Northern Ireland defined from CORINE land cover (areas in Great Britain from the CEH Land Cover Map).

The justification for applying these values and / or methods in the UK are also given in Table UK-1.

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Table UK-1. Summary of UK critical load values and the justification for their use.

Critical loads parameter (units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification
$CL_{max}(S)$ (eq ha^{-1} yr^{-1})	AG	130	5030	$= CL(A) + (BC^*_{dep} - Cl^*_{dep}) - BC_u$	Mapping Manual (UBA 1996).
	CG	598	4798	See BC_{dep}, Cl_{dep} and BC_u comments below.	
	H	130	5010		
	C	10	11732		
	D	4	11108		
	W	0	36900	$= L_{crit} / (1 - \rho_S)$	
$CL_{min}(N)$ (eq ha^{-1} yr^{-1})	AG	213	570	$= N_u + N_i + N_{de}$	Mapping Manual.
	CG	857	1214		
	H	433	790		
	C	643	1000		
	D	643	1000		
	W	15	638	$= fN_u + (1 - r)(N_i + N_{de})$	Mapping Manual. N_{de} values catchment- weighted according to area of different soils present in catchment.
$CL_{max}(N)$ (eq ha^{-1} yr^{-1})	AG	363	5550	$= CL_{max}(S) + CL_{min}(N)$	Mapping Manual.
	CG	1455	5972		
	H	583	5466		
	C	733	12651		
	D	647	11751		
	W	143	201500		
$CL_{nut}(N)$ (eq ha^{-1} yr^{-1})	AG	714	1786	Empirical values applied: Acid grassland: 10, 12.5, 25 kg N ha^{-1} yr^{-1} depending on species present.	Mapping Manual. Empirical values recommended by UK experts (Hall et al. 1998). However, the UK will review these after the UN/ECE workshop in 2002 to review empirical critical loads for nutrient nitrogen.
	CG	3571	3571	Empirical value applied: 50 kg N ha^{-1} yr^{-1}	
	H	714	1214	Empirical values applied: 10, 15, 17 kg N ha^{-1} yr^{-1} depending on species present.	
	C	928	928	Minimum of empirical value (13 kg N ha^{-1} yr^{-1}) or mass balance value (where $CL_{nut}(N) = N_u + N_i + N_{le(acc)} + N_{de}$). N_i and N_{de} values between 1–4 kg N ha^{-1} yr^{-1} depending on soil type. Previously only mass balance used for conifers.	Mass balance equation and empirical value as recommended in Mapping Manual. Input values recommended by UK experts (Hall et al. 1998). Empirical value lower everywhere.
	D	1071	1214	Minimum of empirical value (17 kg N ha^{-1} yr^{-1}) or mass balance value (where $CL_{nut}(N) = N_u + N_i + N_{le(acc)} + N_{de}$). N_i & N_{de} values between 1–4 kg N ha^{-1} yr^{-1} depending on soil type.	Mapping Manual. Empirical values recommended by UK experts. Input values to mass balance equation recommended by UK experts (Hall et al. 1998).
	W	–	–	Not calculated.	

Table UK-1. Summary of UK critical load values and the justification for their use (continued).

Critical loads parameter (units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification
BC^*_{dep} – Cl^*_{dep} (eq ha^{-1} yr^{-1})	AG	0	1150	BC^*_{dep} and Cl^*_{dep} changed to measured mean data 1986–91 for low vegetation.	Mapping Manual.
	CG	0	1150		
	H	0	1150		
	C	0	1850	BC_{dep}* and Cl_{dep}* changed to measured mean data 1986–91 for woodland ecosystems.	
	D	0	1850		
	W	–	–	Not used.	
Bc_u (eq ha^{-1} yr^{-1})	AG	0	0	Set to zero; uptake negligible for acid grassland.	Based on published data by UK experts.
	CG	222	222	Includes removal via sheep.	Based on published data by UK experts.
	H	0	0	No uptake for heathland.	
	C	250	250	New values. Calculated from: average volume increment × basic wood density × concentration in wood and assuming potential yields achieved. Values based on data for Sitka Spruce.	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. These used in CL_{max}(S) calculations only, estimates of calcium uptake only used in SMB for mineral soils.
	D	400	850	New values. Calculated from: average volume increment × basic wood density × concentration in wood and assuming potential yields achieved. Values based on data for Oak. Minimum value for Ca-poor soils and maximum value for Ca-rich soils.	
	W	–	–	Not used.	
ANC_w (eq ha^{-1} yr^{-1})	AG	–	–	SMB not used: empirical critical loads of acidity for soils (Skokloster classification) applied, therefore ANCw not assigned.	Methods agreed by UK experts (Hall et al. 1998). (SMB only applied to woodland ecosystems in UK).
	CG	–	–		
	H	–	–		
	C	0	4000	Set to zero for peat soils.	Recommended in Mapping Manual. See Hornung et al. (1995). Assigned values checked against application of PROFILE for limited number of sites.
	D	0	4000		
	W	–	–	Not used.	

Table UK-1. Summary of UK critical load values and the justification for their use (continued).

Critical loads parameter (units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification
$ANC_{le(crit)}$ (eq ha^{-1} yr^{-1})	AG	–	–	SMB not used: empirical critical loads of acidity for soils applied, therefore $ANC_{le(crit)}$ not calculated.	Methods agreed by UK experts (Hall et al. 1998). (SMB only applied to woodland ecosystems in UK).
	CG	–	–		
	H	–	–		
	C	0.1	7734	Calculated via SMB equation with ratio of Ca:Al = 1 as chemical criterion for mineral soils and critical pH 4.0 for organic soils. Empirical acidity critical loads applied to peat soils.	SMB with BC:Al ratio and base cation deposition produced unrealistically high critical loads. Ca:Al ratio recommended by Cronan and Grigel (1995).
	D	0	7067		
	W	–	–	For freshwaters the ANC_{limit} is set at zero μeq l^{-1}.	Value selected for 50% probability of damage to brown trout populations.
N_u (eq ha^{-1} yr^{-1})	AG	70	70	Equivalent to 1 kg N ha^{-1} yr^{-1}	Based on published data by UK experts.
	CG	714	714	Equivalent to 10 kg N ha^{-1} yr^{-1}.	
	H	290	290	Equivalent to 4 kg N ha^{-1} yr^{-1}	
	C	500	500	New values. Methods as for BC_u.	Based on published data – one value for whole of UK; regional growth values to be incorporated in future.
	D	500	500		
	W	0	279	$= fN_u$. Uses N_u value of 279 eq ha^{-1} yr^{-1} for all coniferous forest multiplied by percentage forest in catchment.	Based on published data (Curtis et al. 1998).
N_i (eq ha^{-1} yr^{-1})	AG	71	214	Dependent on soil type.	Based on published data for long-term sustainability.
	CG	71	214		
	H	71	214		
	C	71	214		
	D	71	214		
	W	7	214	N_i values catchment-weighted according to area of different soils present in catchment.	
$N_{le(acc)}$ (eq ha^{-1} yr^{-1})	AG	–	–	Empirical nutrient nitrogen critical loads used, therefore $N_{le(acc)}$ not assigned.	
	CG	–	–		
	H	–	–		
	C	428	428	Equivalent to 6 kg N ha^{-1} yr^{-1}	Values based on data from a limited number of detailed site studies for GB plantations.
	D	428	428		
	W	–	–	Not used.	

Table UK-1. Summary of UK critical load values and the justification for their use (continued).

Critical loads parameter (units)	Eco-system code#	Min. value	Max. value	Data sources/ Methods used	Justification
N_{de} (eq ha^{-1} yr^{-1})	AG	71	286	Used in $CL_{min}(N)$ only. (Empirical critical loads of nutrient nitrogen used). Values assigned according to soil type.	
	CG	71	286		
	H	71	286		
	C	71	286	Used in $CL_{min}(N)$and mass balance of $CL_{nut}(N)$. Values assigned according to soil type.	
	D	71	286		
	W	7	285	Uses catchment-weighted N_{de} values (based on soil type) instead of f_{de}.	Use of f_{de} (0.1–0.8) as in Mapping Manual gives N_{de} values up to 25kg N / ha / year – much too high for UK (Curtis et al. 1998).
Precipitation s urplus Q (m)	AG	–	–	SMB not used, therefore Q not assigned.	
	CG	–	–		
	H	–	–		
	C	0.057	3.876	1-km runoff data based on 30-year (1941–1970) mean rainfall data.	Used in SMB equation for acidity critical loads.
	D	0.057	3.876		
	W	0.097	3.364	1km catchment-weighted runoff based on mean rainfall data for 1941–70 for GB and 1961–90 for Northern Ireland.	Used in FAB.
K_{gibb} (m^6 eq^{-2})	AG	–	–	SMB not used (empirical acidity critical loads applied).	
	CG	–	–		
	H	–	–		
	C	9.5	950	Minimum value applied to organic soils and maximum value applied to mineral soils.	Mapping Manual.
	D	9.5	950		
	W	–	–	Not used.	

Appendix A. The polar stereographic projection (EMEP grid)

To make critical loads useful for pan-European negotiations on emission reductions, one must be able to compare them to deposition estimates. Deposition of sulphur and nitrogen compounds have traditionally been reported by EMEP on a 150× 150km^2 grid covering (most of) Europe, but in recent years depositions have also become available on a 50×50km^2 subgrid. The EMEP grid systems are special cases of the so-called polar stereographic projection. Here we describe 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):

$$x = x_p + M \tan\left(\frac{\pi}{4} - \frac{\phi}{2}\right)\sin(\lambda - \lambda_0) \tag{A.1}$$

and

$$y = y_p - M \tan\left(\frac{\pi}{4} - \frac{\phi}{2}\right)\cos(\lambda - \lambda_0) \tag{A.2}$$

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

$$d = \frac{R}{M}(1 + \sin\phi_0) \tag{A.3}$$

where R (= 6370 km) is the radius of the Earth. The inverse transformation, i.e. longitude and latitude as function of x and y, is given by

$$\lambda = \lambda_0 + \arctan\left(\frac{x - x_p}{y_p - y}\right) \tag{A.4}$$

and

$$\phi = \frac{\pi}{2} - 2\arctan(r/M) \quad \text{with} \quad r = \sqrt{(x - x_p)^2 + (y - y_p)^2} \tag{A.5}$$

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°) 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.

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.

$$i = \mathrm{nint}(x) \quad \text{and} \quad j = \mathrm{nint}(y) \tag{A.6}$$

where 'nint' is the nearest integer (rounding function). Consequently, the four corners of the grid cell have coordinates ($i \pm 1/2, j \pm 1/2$).

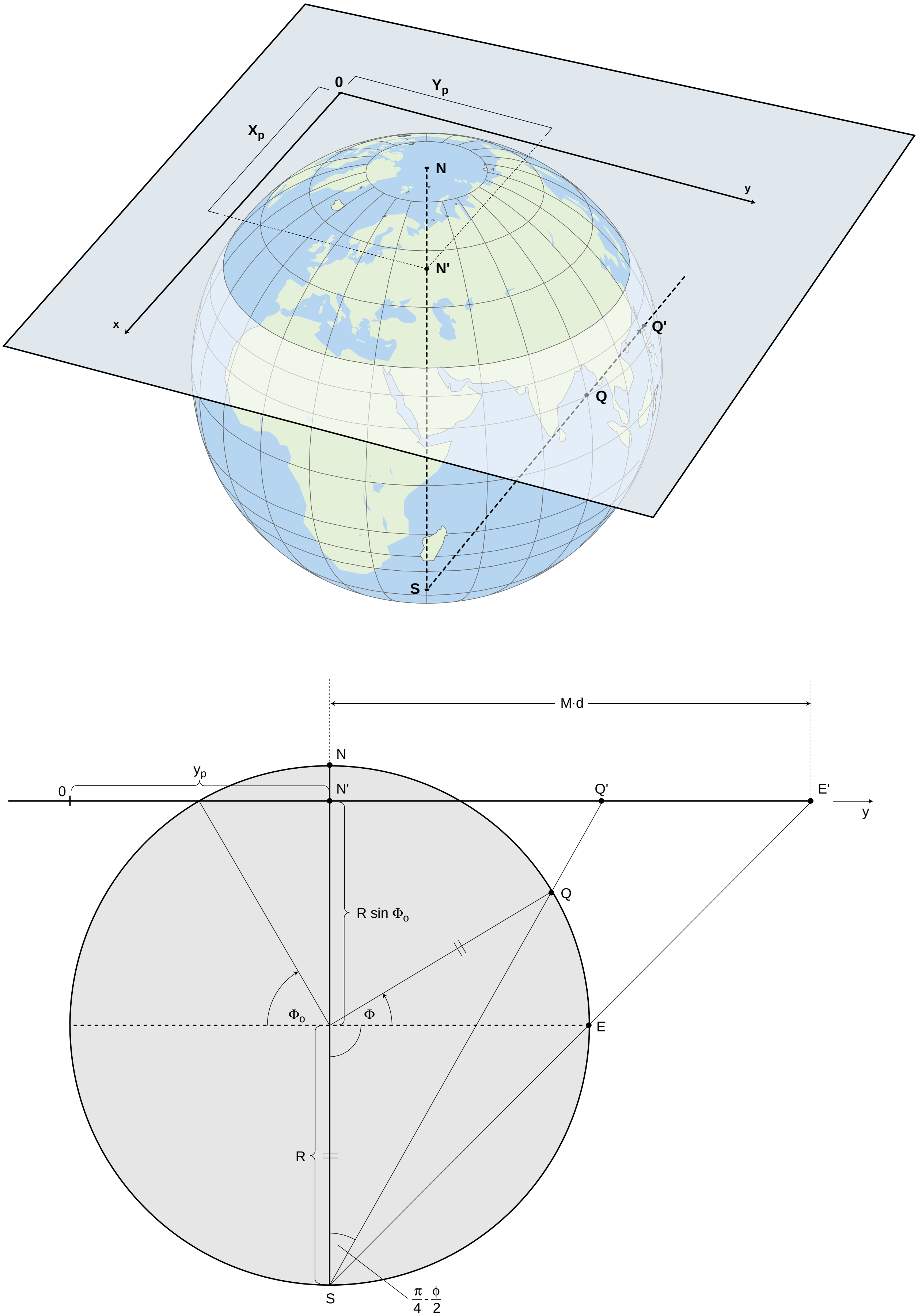

Figure A-1. Polar stereographic projection from the South Pole onto a plane cutting the Earth at a given latitude (top). Geometric relationships in a plane cutting the Earth vertically at a given longitude used to derive the projection equations (bottom).

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 (Saltbones and Dovland 1986):

$$d = 150\text{km}, \quad (x_p, y_p) = (3,37), \quad \phi_0 = \frac{\pi}{3} = 60°\text{N}, \quad \lambda_0 = -32°(\text{i.e. } 32°\text{W}) \tag{A.7}$$

which yields M=79.2438...

The (extended) 50×50km² grid (EMEP50 grid):

The eulerian dispersion model of EMEP / MSC-W produces concentration and deposition fields on a 50×50km² grid with the parameters (Olendrzynski 1999, Tsyro and Støren 1999):

$$d = 50\text{km}, \quad (x_p, y_p) = (43,121), \quad \phi_0 = \frac{\pi}{3} = 60°\text{N}, \quad \lambda_0 = -32°(\text{i.e. } 32°\text{W}) \tag{A.8}$$

yielding M=237.7314...

An EMEP150 grid cell (i, j) contains 3×3=9 EMEP50 grid cells (m,n) with all combinations of the indices m=3i+33,3i+34,3i+35 and n=3j+9,3j+10,3j+11. The part of the two EMEP grid systems covering Europe is shown in Figure A-2.

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,par,emepi,emepj)
c
c     This subroutine computes for a point (xlon,ylat), where xlon is the
c     longitude (<0 west of Greenwich) and ylat is the latitude in degrees,
c     its EMEP coordinates (emepi,emepj) with parameters given in par().
c
c     par(1) ... size of grid cell (km)
c     (par(2),par(3)) = (xp,yp) ... EMEP coordinates of the North Pole
c
      real               xlon, ylat, par(*), emepi, emepj
c
      data  Rearth /6370./       ! radius of spherical Earth (km)
      data  xlon0 /-32./         ! = lambda_0
      data  drm /1.8660254/      ! = 1+sin(pi/3) = 1+sqrt(3)/2
      data  pi180 /0.017453293/  ! = pi/180
      data  pi360 /0.008726646/  ! = pi/360
c
      em = (Rearth/par(1))*drm
      tp = tan((90.-ylat)*pi360)
      rlamp = (xlon-xlon0)*pi180
      emepi = par(2)+em*tp*sin(rlamp)
      emepj = par(3)-em*tp*cos(rlamp)
                                  return
      end
```

EMEP150 coordinates are obtained by calling the above subroutine with *par*(1)=150, *par*(2)=3 and *par*(3)=37; and the new EMEP50 coordinates are obtained with *par*(1)=50, *par*(2)=43 and *par*(3)=121.

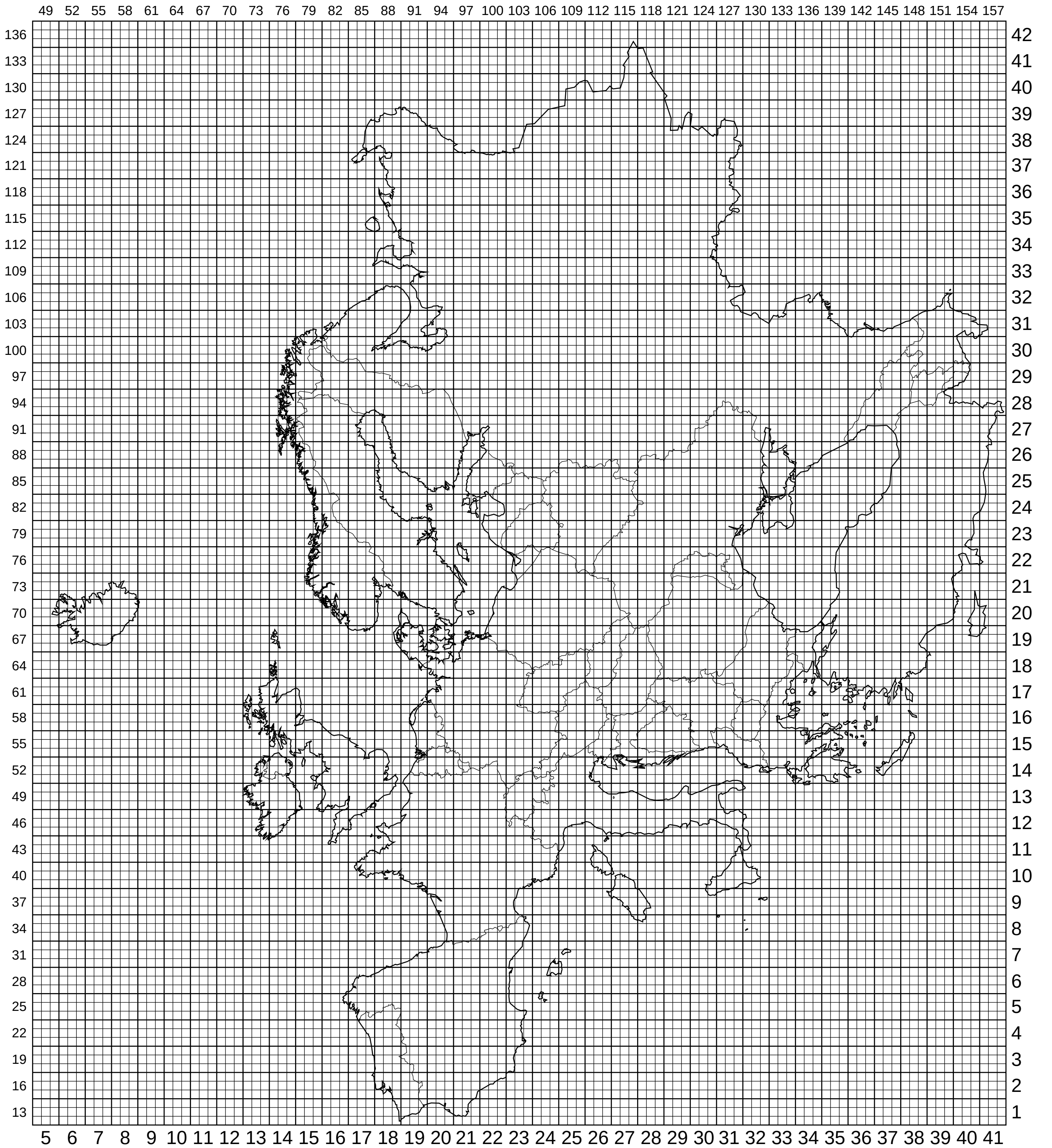

Figure A-2. The EMEP150 grid (thick lines) and EMEP50 grid (thin lines). The labels at the bottom and right are the EMEP150 grid indices; the labels at the top and left are the EMEP50 grid indices (every third).

Conversely, for a given EMEP coordinate system, the EMEP coordinates of a point can be converted into its longitude and latitude with the following subroutine:

```
c
      subroutine  emepll  (emepi,emepj,par,xlon,ylat)
c
c     This subroutine computes for a point (emepi,emepj) in the EMEP
c     coordinate system, defined by the parameters in par(), its
c     longitude xlon and latitude ylat in degrees.
c
c     par(1) ... size of grid cell (km)
c     (par(2),par(3)) = (xp,yp) ... EMEP coordinates of the North Pole
c
      real               emepi, emepj, par(*), xlon, ylat
c
      data  Rearth /6370./      ! radius of spherical Earth (km)
      data  xlon0 /-32./        ! = lambda_0
      data  drm /1.8660254/     ! = 1+sin(pi/3) = 1+sqrt(3)/2
      data  pi180 /57.2957795/  ! = 180/pi
      data  pi360 /114.591559/  ! = 360/pi
c
      emi = par(1)/(Rearth*drm) ! = 1/M
      ex = emepi-par(2)
      ey = par(3)-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*emi)
                                  return
      end
```

The area of an EMEP grid cell:

As mentioned above, the stereographic projection does not preserve areas, e.g. a 50×50km^2 EMEP grid cell is 2,500 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:

$$A(x_1,y_1,x_2,y_2) = 2R^2\{I(u_2,v_2) - I(u_1,v_2) - I(u_2,v_1) + I(u_1,v_1)\} \tag{A.9}$$

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:

$$I(u,v) = \iint \frac{2du dv}{(1+u^2+v^2)^2} = \frac{v}{\sqrt{1+v^2}}\arctan\frac{u}{\sqrt{1+v^2}} + \frac{u}{\sqrt{1+u^2}}\arctan\frac{v}{\sqrt{1+u^2}} \tag{A.10}$$

These two equations allow the calculation 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 in *par*() (see above):

```
c
      real function  aremep  (par,i,j)
c
c     Returns the area (in km2) of an ax-parallel cell with
c     centerpoint (i,j) in the EMEP grid defined by par().
c
c     par(1) ... size of grid cell (km)
c     (par(2),par(3)) = (xp,yp) ... EMEP coordinates of the North Pole
c
      integer            i, j
      real               par(*)
c
      external           femep
c
      data  Rearth /6370./    ! radius of spherical Earth (km)
      data  drm /1.8660254/   ! = 1+sin(pi/3) = 1+sqrt(3)/2
c
      x1 = real(i)-0.5
      y1 = real(j)-0.5
      emi = par(1)/(Rearth*drm) ! = 1/M
      u1 = (x1-par(2))*emi
      v1 = (y1-par(3))*emi
      u2 = u1+emi
      v2 = v1+emi
      ar0 = 2.*Rearth*Rearth
      aremep = ar0*(femep(u2,v2)-femep(u1,v2)-femep(u2,v1)+femep(u1,v1))
                                          return
      end
c
      real function  femep  (u,v)
c
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:

$$\alpha = \lim_{h,k\to 0} \frac{A(x,y,x+h,y+k)}{h\,k\,d^2} = \frac{R^2}{d^2 M^2}\,\frac{4}{\left(1+(r/M)^2\right)^2} \tag{A.11}$$

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:

$$\alpha = \left(\frac{1+\sin\phi}{1+\sin\phi_0}\right)^2 \tag{A.12}$$

This shows that the distortion ratio depends on the latitude ϕ only, and (small) areas are undistorted, i.e. $\alpha = 1$, only at $\phi=\phi_0=60°$ (see also Figure A-3 in the CCE Status Report 1999).

References:

Olendrzynski K (1999) Operational EMEP Eulerian Acid Deposition Model. EMEP/MSC-W Note 4/99, Norwegian Meteorological Institute, Oslo.

Saltbones J, Dovland H (1986) Emissions of sulphur dioxide in Europe in 1980 and 1983. EMEP/CCC Report 1/86, Norwegian Institute for Air Research, Lillestrøm, Norway.

Tsyro S, Støren E (1999) New meteorological model for air pollution transport models. EMEP/MSC-W Note 3/99, Norwegian Meteorological Institute, Oslo.

Appendix B. Conversion factors

In this Appendix tables of the most commonly used conversion factors for sulphur 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:

$$1\text{ g X} = \frac{1}{M}\text{mol X} = \frac{z}{M}\text{eq X} \tag{B.1}$$

Obviously, moles and equivalents are the same for z=1. For depositions one has:

Table B-1. Conversion factors for **sulphur** deposition (g stands for grams of S; M=32, z=2). For conversion multiply by the factors given in the table.

From: To:	mg/m²	g/m²	kg/ha	mol/m²	eq/m²	eq/ha
mg/m²	1	0.001	0.01	0.00003125	0.0000625	0.625
g/m²	1000	1	10	0.03125	0.0625	625
kg/ha	100	0.1	1	0.003125	0.00625	62.5
mol/m²	32000	32	320	1	2	20000
eq/m²	16000	16	160	0.5	1	10000
eq/ha	1.6	0.0016	0.016	0.00005	0.0001	1

Table B-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/m²	g/m²	kg/ha	mol/m²	eq/m²	eq/ha
mg/m²	1	0.001	0.01	0.0000714..	0.0000714..	0.71428..
g/m²	1000	1	10	0.0714..	0.0714..	714.28..
kg/ha	100	0.1	1	0.00714..	0.00714..	71.428..
mol/m²	14000	14	140	1	1	10000
eq/m²	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 $\mu g/m^3$ and ppm (part per million) or ppb (parts per billion). One ppm is one particle of a pollutant in one million particles of the air-pollutant mixture. How many (and which mass) of them can be found in one m^3 depends on the density of the air, i.e. on its temperature and pressure; the conversion formula is

$$1\text{ppm} = 1000\text{ ppb} = \frac{M}{V_0}\ \mu\text{g/m}^3 \tag{B.2}$$

where M is the molecular weight (g / mol) and V_0=0.022414 m^3/ 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. B.2 by

$$V_1 = V_0\ \frac{T_1}{T_0}\cdot\frac{p_0}{p_1} \tag{B.3}$$

For example, for T_1=298K (=25°C) and p_1=p_0 the molar volume V_1 is 0.024453 m^3/mol.

Table B-3. Conversion factors for concentrations of common pollutants at two different temperatures (1 ppm=1000 ppb).

		From ppb to μg/m³, multiply by:		From μg/m³ to ppb, multiply by:	
	M	***T*=0°C**	***T*=25°C**	***T*=0°C**	***T*=25°C**
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:

$$[A^{m\pm}]^x = K[B^{n\pm}]^y \tag{B.4}$$

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^cV, then the equilibrium constant in the new units is given by

$$K' = K \cdot 10^{c(y-x)} \frac{m^x}{n^y} \ (eq/V)^{x-y} \tag{B.5}$$

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 $[Al^{3+}]=K[H^+]^3$, i.e. m=3, x=1, n=1, y=3 and (e.g.) $K=10^8(mol/L)^{-2}$. If one wants to convert to eq/m^3, one has c = –3, and thus $K' = 10^8 \cdot 10^{-3\cdot(3-1)} \cdot 3 = 300\ (eq/m^3)^{-2}$.