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Convention on Long-range Transboundary Air Pollution

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Modelling and Mapping of Critical Thresholds in Europe: CCE Status Report 2003

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- The EMEP Meteorological Synthesizing Centre-West for providing European sulphur and nitrogen deposition data.

- The Working Group on Effects and the Task Force of the ICP on Modelling and Mapping for their collaboration and assistance.

Summary

This Status Report of the Coordination Center for Effects (CCE) informs Parties to the Convention on Long-range Transboundary Air Pollution (CLRTAP) and its network of National Focal Centres of recent updates of the European critical loads database and related work. In 2003 the database was extended with variables needed for dynamic modelling for the first time. This information is necessary to support European air quality policies with analyses of time delays of ecosystem damage or recovery caused by changes, over time, of acidifying deposition. The database was updated and extended following the request of the Working Group on Effects (WGE) at its 21st session (Geneva, 28-30 August 2002).

In response to the CCE call for data in November 2002, 19 countries submitted data on critical loads of acidity and eutrophication, while ten countries also submitted the requested dynamic modelling parameters. Other countries indicated their intentions to prepare data for submission for the next call, planned for autumn 2003. This report also contains national reports on the methods applied to contribute to the European critical loads database. The report includes an analysis of the data to help cross-border consistency checks. A comparison is made with the database used to support negotiations of the 1999 CLRTAP Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the "Gothenburg Protocol") and the 2001 EU National Emission Ceiling Directive. The results described in this report will facilitate the intended next step, the call for data to be used in integrated assessments for the scientific and technical support of the review and revision process of these European agreements, expected in 2004/2005.

This report is being submitted to the 22nd session of the Working Group on Effects (Geneva, 3-5 September 2003).

Samenvatting

Dit Status Rapport van het Coordination Center for Effects (CCE) informeert de Conventie van grensoverschrijdende luchtverontreiniging (CLRTAP) en het netwerk van National Focal Centra over de meest recente Europese database van kritische drempels voor verzuring en vermesting en werk dat hiermee in verband staat. Deze database is in 2003 voor het eerst uitgebreid met gegevens die de gevolgtijdelijke (dynamische) modellering van geochemische processen, vooral in bodems, mogelijk maakt. Deze informatie is nodig om het Europese luchtbeleid te kunnen ondersteunen met kennis van tijdsvertragingen van ecosysteemherstel of -schade als gevolg van veranderingen, in de tijd, van verzurende depositie. De aldus uitgebreide database van kritische drempels en dynamische modellering is door het CCE gemaakt op verzoek van de Working Group on Effects onder de Conventie op haar 21e vergadering (Genève, 28-30 augustus 2002).

In antwoord op het verzoek van november 2002 om databijdragen stuurden 19 landen gegevens in waarvan tien inclusief dynamische modellerings parameters. De onderbouwing van de ingezonden data is voor elk land afzonderlijk in het rapport opgenomen. Daarnaast bevat het rapport een analyse van de grensoverschrijdende consistentie van de nieuwe database. Ook is een vergelijking opgenomen met data die zijn gebruikt bij de ondersteuning van het 1999 CLRTAP Protocol voor de bestrijding van verzuring, vermesting en troposferische ozon (het "Gothenburg protocol") en de EU-richtlijn 2001/81/EG van het Europese Parlement (2001) inzake nationale emissieplafonds voor bepaalde luchtverontreinigende stoffen (NEC directive). De in de rapportage beschreven resultaten zijn belangrijk voor de volgende stap, te weten de ondersteuning van het revisieproces van deze Europese overeenkomsten waarschijnlijk in 2004/2005.

Het rapport wordt op de 22e vergadering van de Working Group on Effects (Genève, 3-5 september 2003) gepresenteerd.

Preface

You have before you the seventh Status Report of the Coordination Center for Effects (CCE) of the International Co-operative Programme on Modelling and Mapping of Critical Levels and Loads and Air Pollution Effects, Risks and Trends (ICP M&M). This ICP is part of the Working Group on Effects (WGE) of the 1979 Convention on Long-range Transboundary Air Pollution (CLRTAP).

This report documents a new phase in the modelling and mapping of critical loads, in which the CCE and National Focal Centres (NFCs) now embarked on dynamic modelling applications. The present Status Report includes the results of the decision taken by the Working Group on Effects at its 21st session, inviting the CCE to issue, in the autumn of 2002, a call for updated critical loads and parameters for dynamic modelling.

The call was conducted to familiarise the network of National Focal Centres with the increasing complexity resulting from the extension of the European critical loads database with dynamic modelling data. In the short term, dynamic modelling of soil acidification can contribute to a better understanding of time delays of recovery in regions where critical loads are no longer exceeded and time delays of damage in regions where critical loads continue to be exceeded. In the longer run dynamic modelling can help improve knowledge on the (biological) effects in Europe of (e.g.) excessive deposition of nitrogen compounds. This can become relevant in the ICP M&M network to help support sustainable multi-source, multi-effect approaches to reduce excess nitrogen inputs, which also affect the carbon cycle in European ecosystems.

Since the introduction to dynamic modelling in its Status Report 2001, the CCE has produced a Dynamic Modelling Manual and a Very Simple Dynamic (VSD) model which were discussed and reviewed in various meetings under the ICP M&M, and posting updates publicly available on the CCE website. A paper version of this manual became recently available as a RIVM report, thus this material is not included again in detail in this Status Report.

This report consists of two parts:

Part I describes results of recent activities of the CCE. Chapter 1 provides a comprehensive summary of European maps of critical loads and contemporary exceedances including a comparison with results based on the 1998 critical loads database. The latter database was used to support of the 1999 CLRTAP Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the "Gothenburg Protocol") and the 2001 EU National Emission Ceiling directive. .Chapter 2 includes a detailed overview of the results of the call for data on critical loads and dynamic modelling issued by the CCE in November 2002. Chapter 3 describes the current status of dynamic modelling with particular attention for the linkage with Integrated Assessment Modelling. This linkage will be important to the call for data intended at the end of 2003, which will aim at results which could be made available to the Task Force on Integrated Assessment Modelling, following the appropriate procedure under the Convention. Finally, Chapter 4 describes the update made to the European background database used to compute and map critical loads and dynamic modelling parameters for countries who have not yet responded to calls for data.

Part II of this report consists of reports by the National Focal Centres. The emphasis has been to document national critical loads and dynamic modelling and the input data used to calculate them. These reports were edited for clarity, but have not been further reviewed and thus reflect the NFCs' intentions of what to report.

Three appendices describe the EMEP grid, sea-salt corrections, and conversion formulae.

Finally, if you want to learn more about the CCE, visit the CCE website www.rivm.nl/cce/ from which you can also download other CCE reports, including the Dynamic Modelling Manual.

Coordination Center for Effects
Netherlands Environmental Assessment Agency (MNP)
National Institute for Public Health and the Environment (RIVM)
June 2003

1. Status of European Critical Loads and Dynamic Modelling

Jean-Paul Hettelingh, Maximilian Posch and Jaap Slootweg

1.1 Introduction

The UNECE Working Group on Effects (WGE) at its 21st session invited the CCE "…to issue, in the autumn of 2002, a call for updated critical loads and parameters for dynamic modelling.…" (EB.AIR/WG.1/2002/2 para. 41g). The maps and graphs presented in this chapter are the result of this call for data.

In comparison to calls for data reported in earlier Status Reports, the purpose of the most recent call was not restricted to updating current national information in the European critical loads database, but also to familiarise the network of National Focal Centres (NFCs) with the increasing complexity entailed by the extension of the European critical loads database with dynamic modelling parameters. An important requirement was to establish consistency between critical loads calculations and dynamic modelling, with the objective that the updated database contains data which can be used to both calculate critical loads and apply dynamic models.

This chapter provides an overview of the critical loads and exceedance maps derived from the data submitted in 2003. In addition, a comparison is made to data used in support of the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the "Gothenburg Protocol") and the 2001 EU National Emission Ceiling Directive. Finally, a map of a key dynamic modelling parameter, the available base cation pool in the soil, is presented to illustrate progress made in extending the European critical loads database. A detailed overview and analysis of national data submissions is provided in Chapter 2.

1.2 Summary of critical load calculation methods

The critical loads database consists of four basic variables which NFCs were asked to provide to the CCE, and which 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 sulphur 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, Posch et al. 2001):

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

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.:

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

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 of nitrogen acidity (in the case of a zero deposition of sulphur) is given by:

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

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 (e.g. Hettelingh et al. 1995).

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):

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

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

1.3 Maps of critical loads for all ecosystems

This section contains maps of critical loads for all ecosystems combined on the 50×50 km^2 EMEP grid. This resolution anticipates the use of critical loads in comparison to depositions computed with EMEP's eulerian model.

The maps in the present report are based on updated national contributions from 19 countries. For other countries, either the most recent available data submission (2001 or earlier) was used, or the CCE's background database for those countries that have never submitted data. This procedure to ensure full European critical load coverage was proposed at the 13th CCE workshop and accepted at the 19th Task Force meeting of the ICP Modelling and Mapping (Estonia, May 2003). However, this procedure does *not* allow using background data for parts of countries that submitted critical loads calculations for only a portion of their countries. This accounts for the blank spots in the critical load maps.

Figure 1-1 shows 5th and 50th percentile (median) maps of $CL_{max}(S)$ and $CL_{nut}(N)$, reflecting values in grid cells at which 95 and 50 percent of the ecosystems are protected from the impacts of sulphur and nitrogen deposition. In these maps critical loads for different ecosystem types have been combined into a single map.

Comparison of the 5th and 50th percentile maps shows that low values (up to 700 eq $ha^{-1}a^{-1}$) of $CL_{max}(S)$ occur in north and central-west Europe (top left map), while the protection of even 50% of the ecosystems requires low deposition in northern Europe in particular. In contrast, the difference between the 5th and 50th percentile of $CL_{nut}(N)$ illustrates the occurrence of low values in areas other than northern Europe, including Spain and southern Italy.

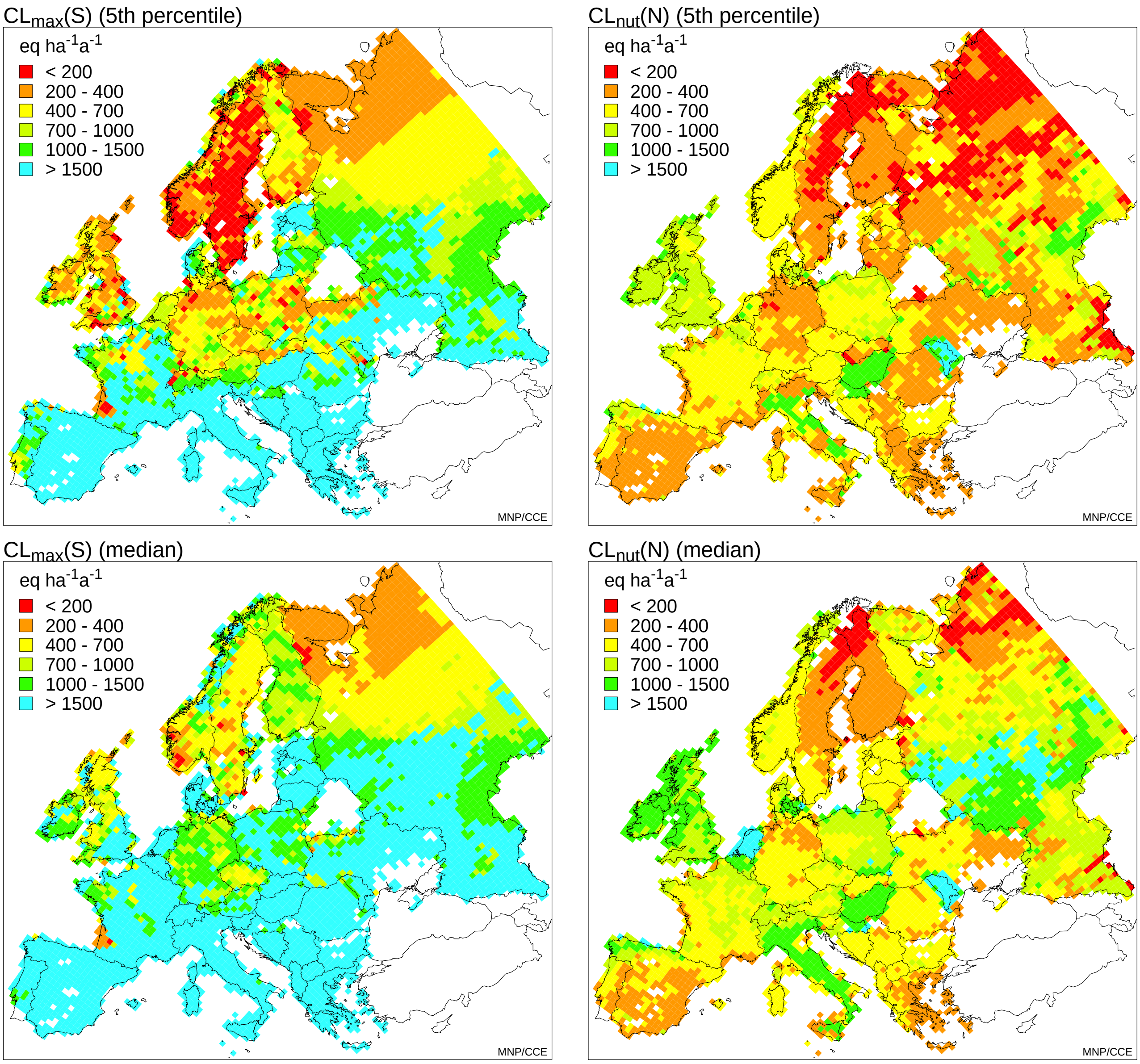

Figure 1-1. The 5th percentiles of the maximum critical loads of sulphur (top left), and of the critical loads of nutrient nitrogen (top right). The 50th percentiles (median) are shown at the bottom left and right, respectively. The maps present these quantities on the EMEP50 grid.

Figure 1-2 shows similar maps for $CL_{max}(N)$ and $CL_{min}(N)$. Relatively low values of the 5th percentile $CL_{max}(N)$, indicating the maximum critical load of nitrogen acidity at zero deposition of sulphur, occur mostly in the northern and western regions of Europe. Values of the 5th percentile $CL_{min}(N)$ reflecting the lowest nitrogen uptake and immobilisation, tend to be low nearly everywhere in Europe.

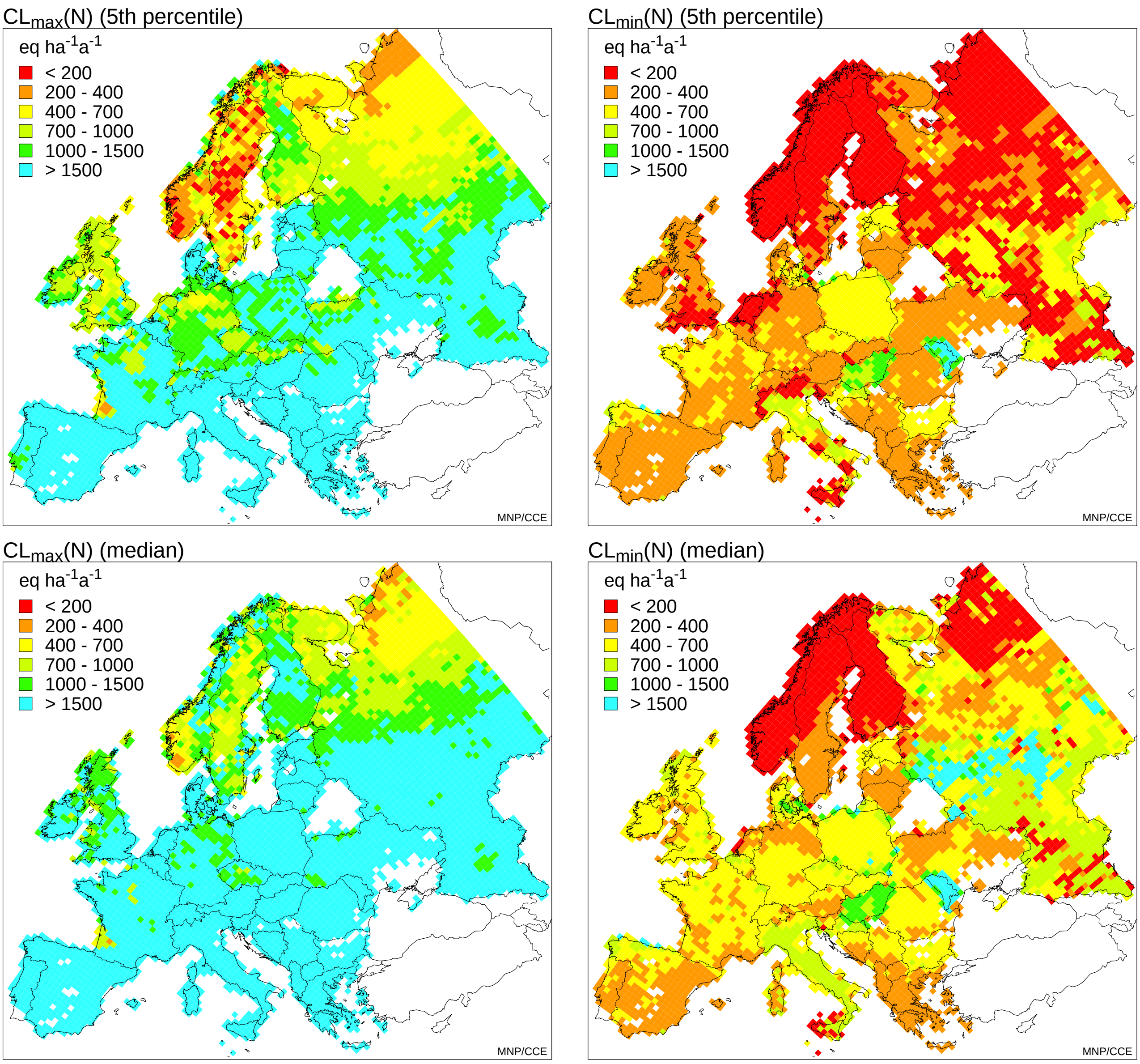

Figure 1-2. The 5th percentiles of the maximum critical loads of nitrogen (top left), and of the minimum critical loads of nitrogen (top right), on the EMEP50 grid resolution. The 50th percentiles are shown at the bottom left and right, respectively.

1.4 Maps of critical loads for individual ecosystem classes

Figure 1-3 shows maps of $CL_{max}(S)$ and $CL_{nut}(N)$ for forests, (semi-)natural vegetation and surface waters on the 50×50 km^2 EMEP grid. Forest ecosystems have been mapped by most NFCs.

Critical loads for (semi-)natural vegetation were submitted by ten NFCs, two of which did not submit acidity critical loads. Finally, for surface waters, six NFCs computed acidity critical loads, while two NFCs provided nutrient N critical loads for surface waters.

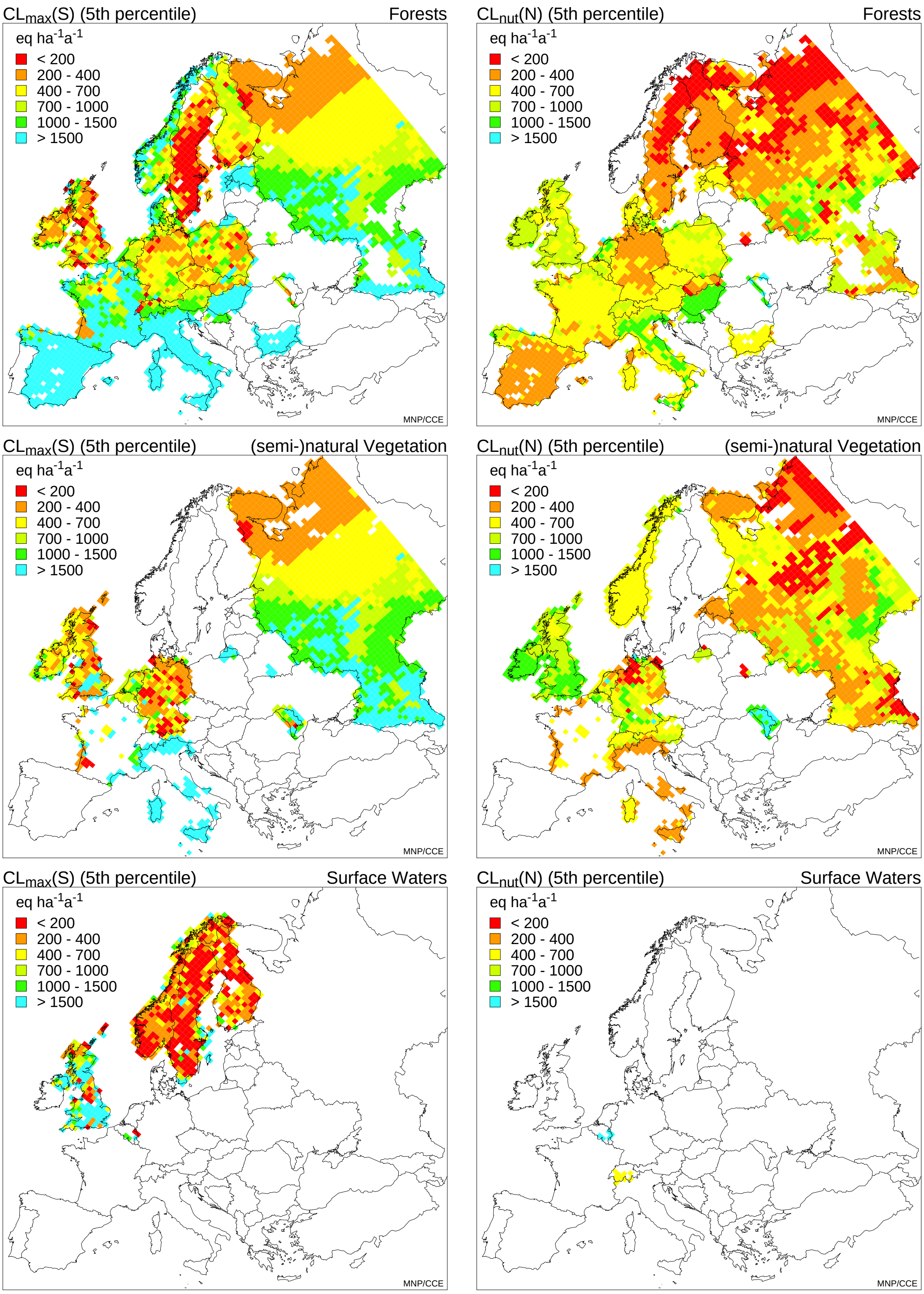

Figure 1-3. The 5th percentiles of the maximum critical load of sulphur (left), and of the critical load of nutrient nitrogen (right) on the EMEP50 grid for three different ecosystem classes (forests, semi-natural vegetation and surface waters).

1.5 *Comparison of the 2003 and 1998 critical load databases*

The deadline is approaching for producing a map of critical loads in 2004 that can be used in the process of reviewing (and possibly revising) the Gothenburg Protocol. In anticipation of this, an interim comparison has been made of 2003 submissions (including the map-filling procedure) to maps used in the support of 1999 Gothenburg Protocol and the 2001 EU National Emission Ceilings Directive. The results are shown in Figure 1-4, which displays the 5th percentile maps of 1998 and 2003 $CL_{max}(S)$ (top) and of $CL_{nut}(N)$ (bottom).

In 2003 the areal coverage of critical loads data has been improved in France, Hungary and in Italy (in particular for $CL_{max}(S)$), but has decreased in Belarus. Markedly lower values of critical loads protecting 95% of the ecosystems against acidification now occur in areas of countries such as the United Kingdom, Sweden, Poland and the Ukraine.

Critical loads protecting 95% of the ecosystems against eutrophication have increased in several countries including in France, Ireland and Norway. The reason for the increase in Norway is the exclusion of $CL_{nut}(N)$ for forest ecosystems in the 2003 submission. A decrease can be seen in areas of e.g. Romania and Greece due to the update of the European background data used in the map-filling procedure (see Chapter 4). Russia, for example, shows no change, since no new data has been provided since the 1998 maps were produced.

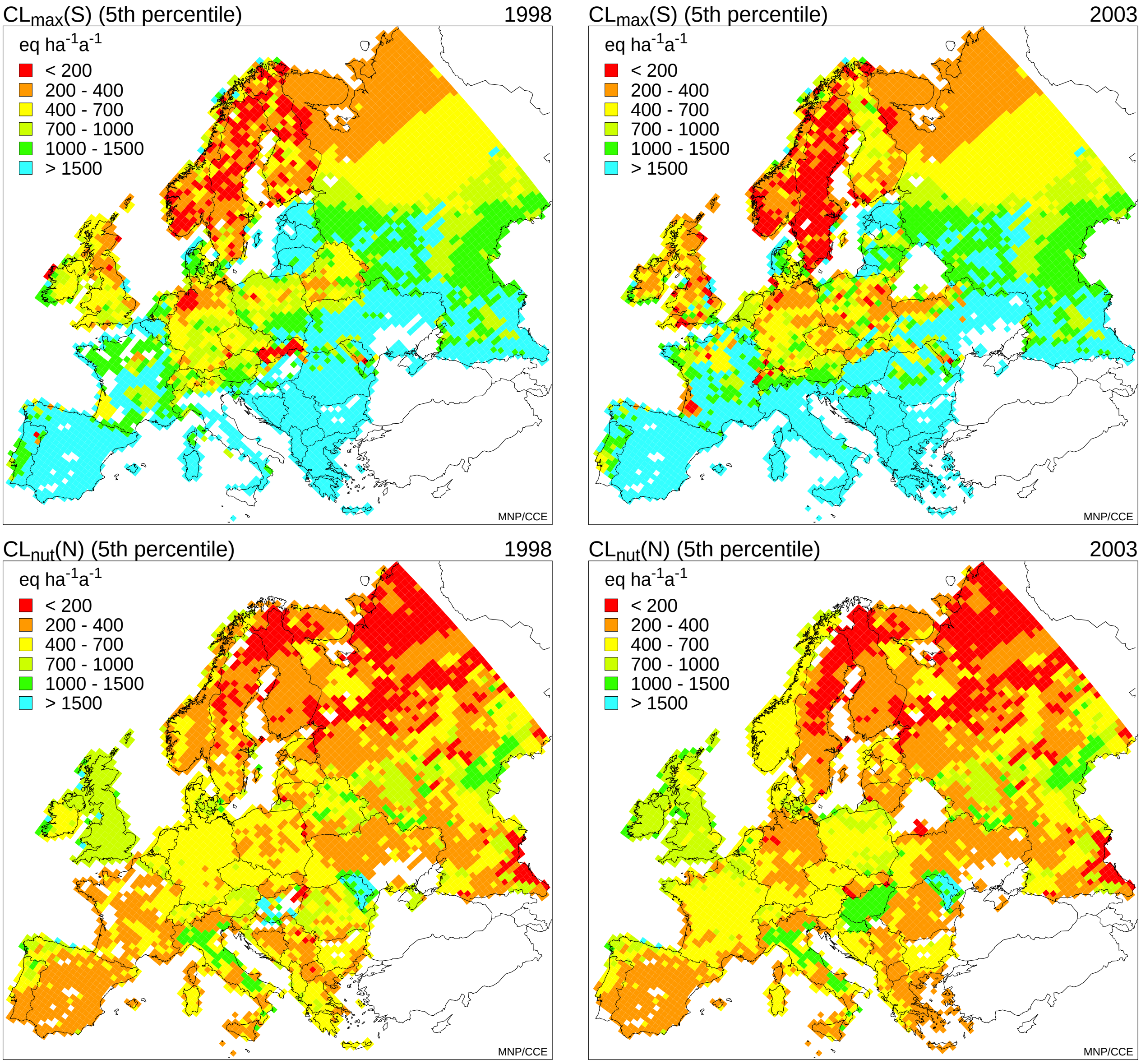

Figure 1-4. The 5th percentiles of the maximum critical loads of sulphur in 1998 (top left) and 2003 (top right), and the critical load of nutrient nitrogen in 1998 (bottom left) and 2003 (bottom right), on the EMEP50 grid resolution. For countries that never submitted data an update of the 1998 background database is used in 2003.

Figure 1-5 provides a more detailed comparison of 1998 and 2003 data. For each country that provided data in 2003, the minimum, 5th, 25th, 50th, 75th, 95th percentiles and the maximum of the critical loads are shown in a “diamond plot”. (See Table 2-2 on p. 13 for a key to the 2-letter country codes used.) Statistics for $CL_{max}(S)$ (left column) range from 0 to 6000 eq ha^{-1} a^{-1}, while $CL_{nut}(N)$ (right) values range from 0 to 3000 eq ha^{-1} a^{-1}. The dark blue and turquoise diamonds reflect 2003 and 1998 statistics respectively. Significant changes in the 2003 distribution of $CL_{max}(S)$ can be noted in Switzerland (due to a smaller range of values), France (broader range due to increased areal coverage), Hungary (minimum now exceeds 8000 eq ha^{-1} a^{-1}) and Italy (median exceeds 8000 eq ha^{-1} a^{-1}). The 5th percentile has now become somewhat lower in Belgium, the Netherlands, Czech Republic, France, United Kingdom, Croatia, Ireland, Poland and Sweden. In general, the distributions (cf. e.g. median values) of $CL_{max}(S)$ have shifted to the left (i.e. become more sensitive) in Belgium, Czech Republic, Germany, United Kingdom, Croatia, Ireland, Poland and Sweden.

For $CL_{nut}(N)$, Figure 1-5 shows the distributions have shifted for Belarus, Switzerland, Czech Republic, Germany, United Kingdom, Sweden and Slovakia. The Netherlands has a notable increase in $CL_{nut}(N)$ due to the introduction of a critical limit based on biodiversity rather than nitrogen leaching. The consequences of these differences in critical load distributions on exceedance calculations is discussed in the next section.

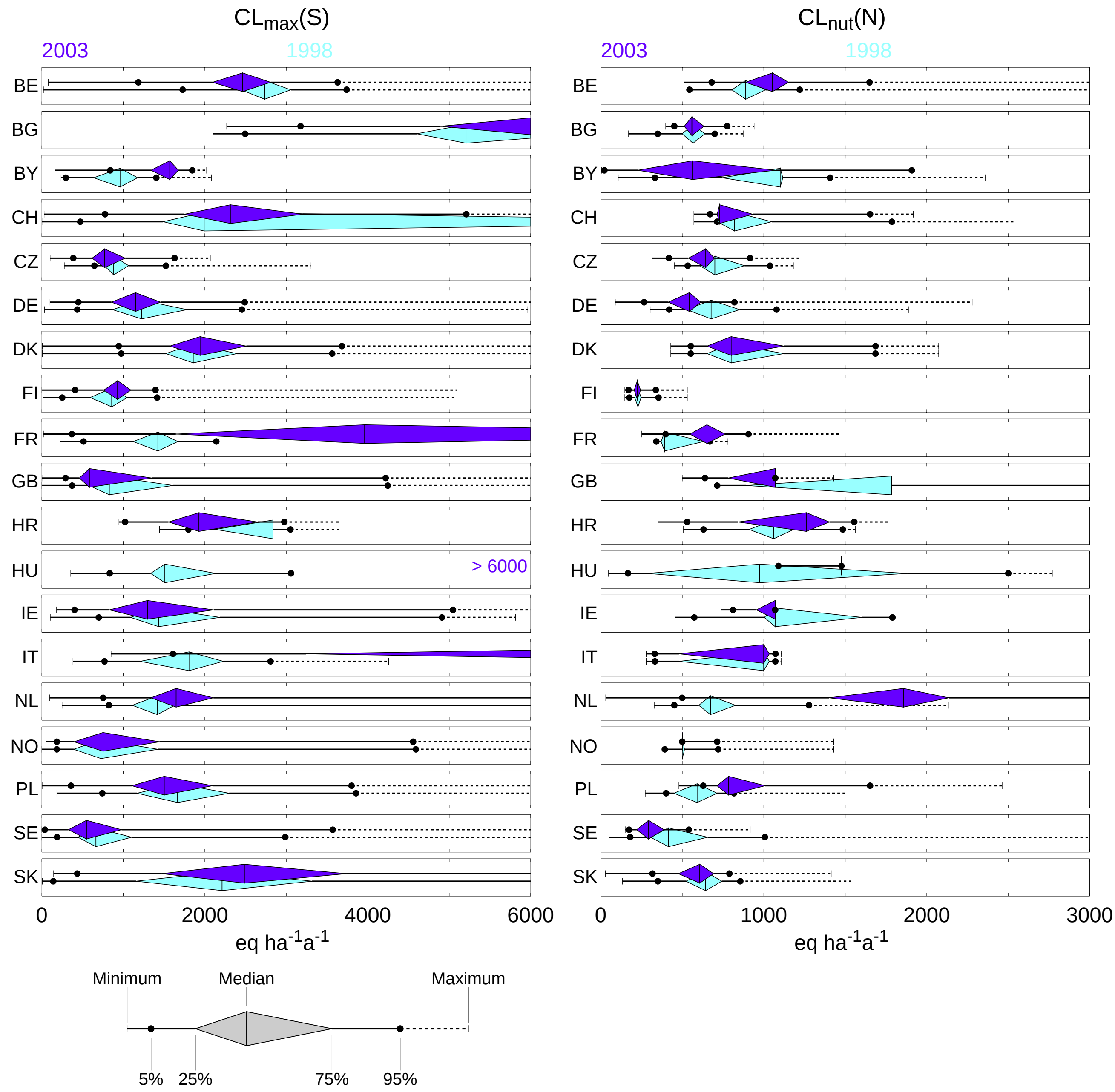

Figure 1-5. Diamond plot of the minimum, 5th, 25th, 50th, 75th, 95th percentiles and maximum critical loads of $CL_{max}(S)$ (left) and $CL_{nut}(N)$ (right) for the national data of 2003 (blue) and 1998 (turquoise).

1.6 Maps of "ecosystem protection" and "average accumulated exceedance"

The analysis of exceedances in the integrated assessment of emission reduction alternatives rests on two main indicators – "ecosystem protection" and the "average accumulated exceedance" (AAE). The AAE is the area-weighted average of *all* ecosystem exceedances in a grid cell, and not only at the exceedance of the most sensitive ecosystem. Maps of AAE provide information about the *magnitude* of the exceedances, whereas maps of ecosystem protection characterise the *extent* of exceedances (see Posch et al. 2001a, 2001b for further details). Both "ecosystem protection" and "average accumulated exceedance" maps are shown below using 1998 and 2003 submissions of critical loads and 2000 deposition data.

Exceedances were computed using deposition data from the langrangian model on the 150×150 km^2 EMEP grid, as EMEP50 eulerian model results are not yet available for all relevant target years at present. Figure 1-6 compares AAE values in 1998 (left) and 2003 (right) of the critical loads of acidity (top) and of ecosystem protection against acidity (bottom). These indicators were calculated using acidic deposition calculated from sulphur and nitrogen oxide emissions in 2000 according to the 1999 CLRTAP Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the "Gothenburg Protocol") and the 2001 EU National Emission Ceiling Directive.

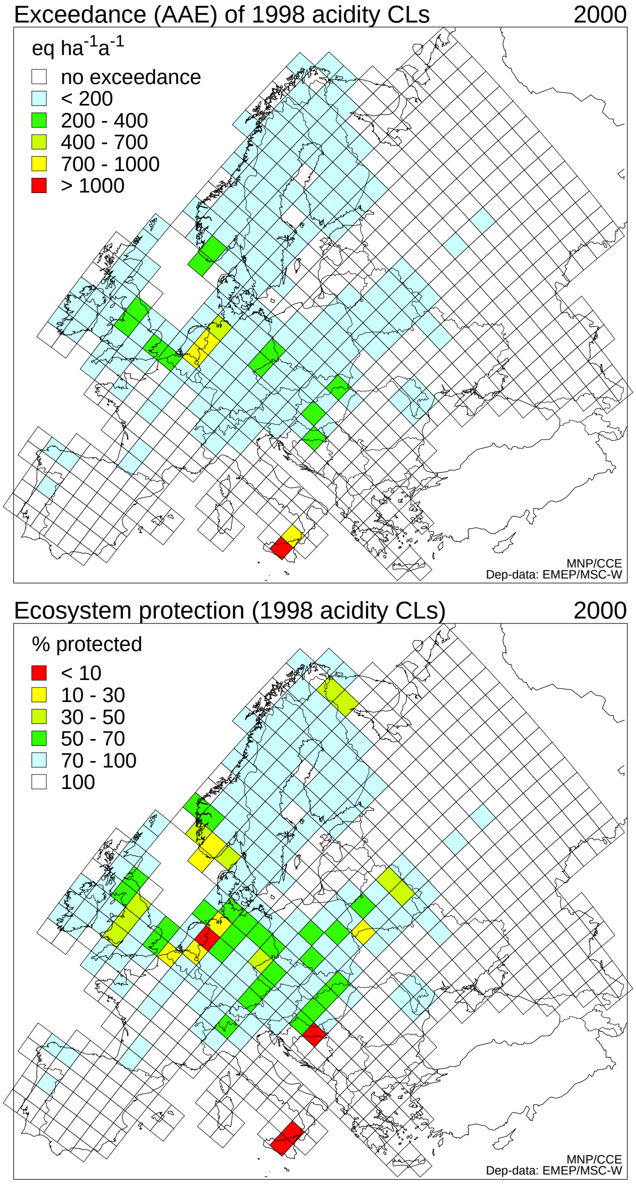

Figure 1-6. Average accumulated exceedance (AAE), computed from critical loads of acidity submitted in 1998 (top left) and 2003 (top right), and ecosystem protection, using 1998 (bottom left) and 2003 (bottom right) acidity critical loads, on the EMEP150 grid of acid deposition in 2000.

Figure 1-6 shows that the AAE covers a larger area at risk using 2003 critical loads data (right) than in 1998 (left), and now also includes central and southern parts of France, northern Poland and western Ukraine. A decrease in the area occurs in Hungary, and in northern Croatia. The distribution and peaks of the AAE change as well, most notably in central Germany and southern Italy. Figure 1-6 (bottom) shows that percentages of protected areas range from 10–30% in Germany (using 2003 critical loads data), where according to 1998 values between 50–70% of the ecosystems were protected. Areas in which less than 10% of the ecosystems are protected occur now only in the border area of Germany and Czech Republic when 2003 critical loads data are used (see Figure 1-6, bottom right).

Figure 1-7 is similar to Figure 1-6 with respect to eutrophication, and shows that the area with positive AAE diminishes particularly in France and Poland. Maxima in the border area of the Netherlands and Germany decrease to ranges that now also occur in western Germany. The number of grid cells with protected ecosystem areas exceeding 10% turn out in central France and in Romania where formerly (1998 critical loads) less areas were protected.

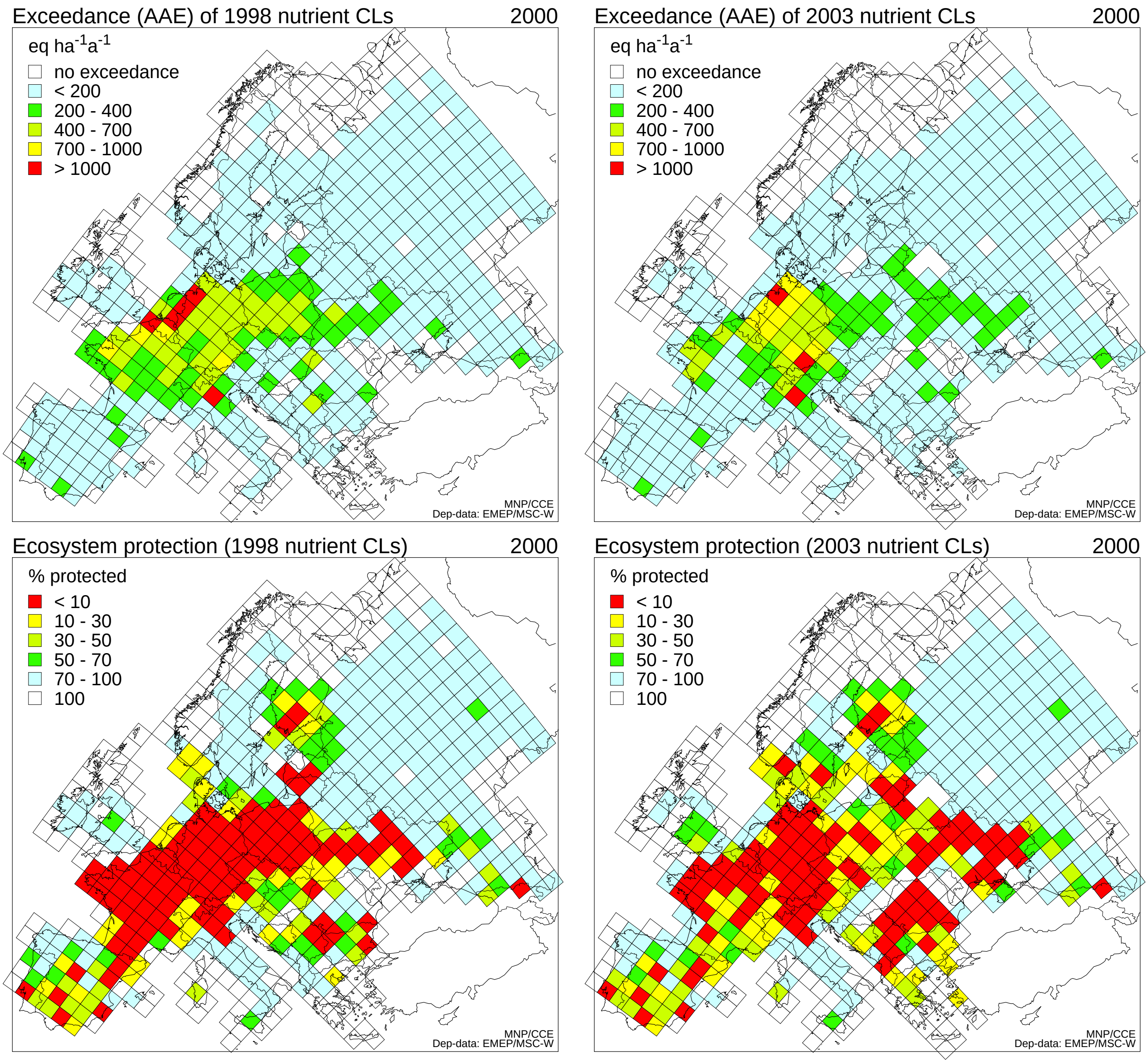

Figure 1-7. Average accumulated exceedance (AAE), computed from critical loads of eutrophication submitted in 1998 (top left) and 2003 (top right), and ecosystem protection using 1998 (bottom left) and 2003 (bottom right) eutrophication critical loads, on the EMEP150 grid of nitrogen deposition in 2000.

1.7 Maps of exchangeable base cations

An important requirement of the CCE's 2002 call for data was to establish consistency between critical load calculations and dynamic modelling, with the objective that the updated database contains data which can be used to both calculate critical loads and apply dynamic models. The call was therefore focused on *minimum* input requirements for the dynamic modelling extension, which are necessary to run any currently available dynamic model in general, and which are *sufficient* for operating the Very Simple Dynamic (VSD) model made available to NFCs by the CCE.

Ten NFCs submitted dynamic modelling variables, while about another ten countries indicated their intent to respond to the CCE call for data planned in autumn 2003. The latter will also include dynamic modelling output variables, i.e. target load functions. Chapter 2 provides a more detailed description of the submission of dynamic modelling variables, while a summary of dynamic modelling methodologies in general and target load functions in particular can be found in Chapter 3.

Figure 1-8 illustrates the 5th percentile (left) and the median, in each grid cell, of the amount of exchangeable base cations in the soils in the 1990s. This amount can be considered as the upper limit of the buffer which is available for the neutralisation of acidic deposition. Cation exchange is a crucial process in all dynamic models. The pool of exchangeable base cations (in any given year) is computed from the soil layer thickness (z), bulk density of soils (ρ), the cation exchange capacity (*CEC*), and the exchangeable base cation fraction (*bsat*). Preferably, these variables should be taken from measurements. In the absence of measurements, the various data can be derived from so-called transfer functions. Derivation approaches including an overview of available dynamic modelling methodologies have been described in the Dynamic Modelling Manual (Posch et al. 2003), which the CCE distributed to all NFCs prior to the 2002 call for data.

The base cation pool has been calculated computed from the dynamic modelling parameters submitted by ten NFCs. This amount can be considered the upper limit of the buffer available for neutralising acidic deposition, and which should not be further depleted (and even replenished) in many areas of Europe to foster recovery from acidification in the nearest possible future. Except for the Netherlands and eastern Bulgaria, soils with a low base cation pool (5th percentile < 20 eq ha^{-1} a^{-1}) occur widely in Denmark, Germany, Poland, Slovakia and Switzerland. Note that in Figure 1-8 the map-filling procedure using the background database has not been applied, in order to highlight national contributions.

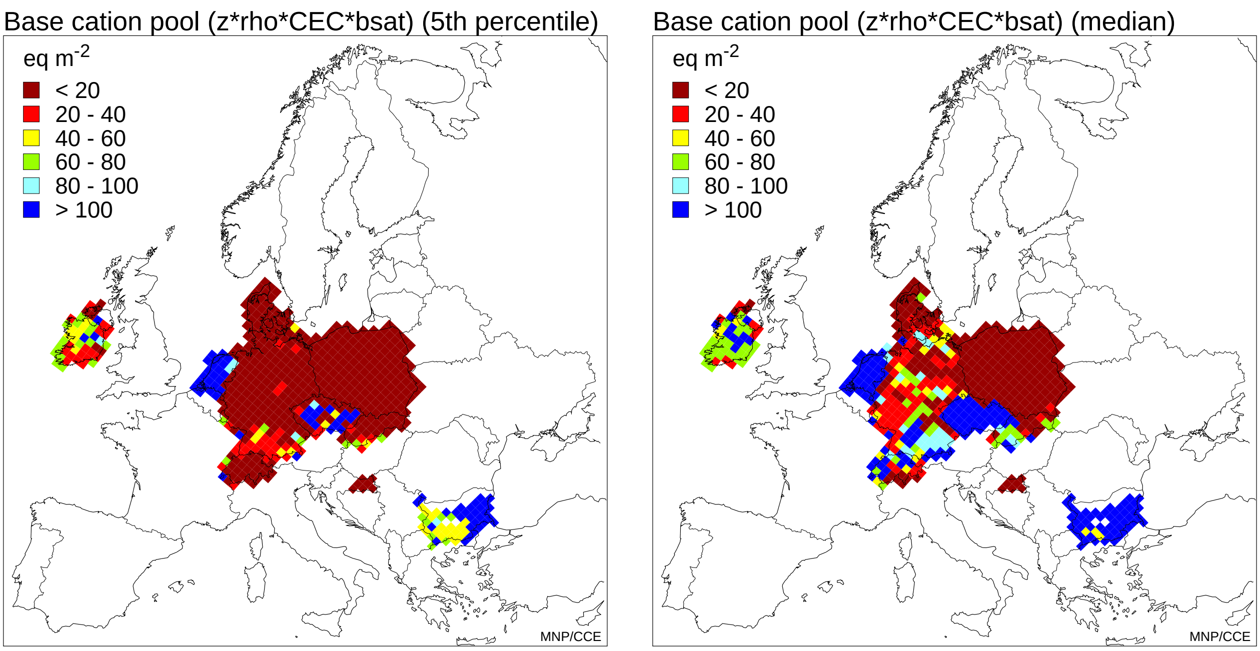

Figure 1-8. The 5th percentile (left) and the median (right), in each grid cell, of the amount of exchangeable base cations in the soils in the 1990s, calculated from data submitted by ten NFCs. This amount can be considered as the upper limit of the buffer which is available for the neutralisation of acidic deposition.

Concluding remarks

This chapter shows that exceedances of acidity critical loads in Europe have decreased markedly by 2000. But also areas where critical loads are no longer exceeded have not necessarily recovered yet. In contrast, exceedances on nutrient N critical loads remain high almost everywhere in Europe. To assess recovery of ecosystems from acidification and eutrophication, dynamic modelling is needed. In the short term, dynamic modelling of soil acidification can contribute to a better understanding of time delays of recovery (in regions where critical loads are no longer exceeded) and damage (where critical loads continue to be exceeded). In the longer run, dynamic modelling can help improve knowledge about the (biological) effects of (e.g.) excessive deposition of nitrogen compounds. This can become relevant in the ICP M&M network to help support sustainable multi-source, multi-effect approaches to reduce excess nitrogen inputs, which also affect the carbon cycle in European ecosystems.

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2. Summary of National Data

Jaap Slootweg, Maximilian Posch and Jean-Paul Hettelingh

2.1 Introduction

The 1998 European critical loads database was used to support the negotiations of the effects-based Gothenburg Protocol of the 1979 Convention on Long-range Transboundary Air Pollution. Since then the scientific community has made progress in supporting the effects-related work in two ways. Firstly, new knowledge has become available that calls for an update of critical loads calculations. Secondly, as dynamic modelling is now also on the agenda of the Working Group on Effects (WGE), input data for such models are needed on a European scale. Consequently, the WGE, at its 21st session, invited the CCE "…to issue, in the autumn of 2002, a call for updated critical loads and parameters for dynamic modelling…." (EB.AIR.WG.1/ 2002/2 para. 41g). The purpose of the call was to:

- step up the NFC preparedness to apply dynamic models in support of the review and possible revision of the Gothenburg protocol,
- provide minimum input requirements for the dynamic modelling extension which are necessary to run dynamic models, and
- ensure consistency between critical loads and dynamic modelling.

This chapter presents the results of the CCE call for data issued in November 2002 with a deadline of 31 March 2003. The chapter includes a comparison between the recently submitted data with the 1998 critical load data that were used for the Gothenburg Protocol.

2.2 Requested variables

Compared to previous calls for data, two groups of variables were added to the most recent call. In addition to the variables considered to be minimum input requirements for dynamic modelling a group of variables was also requested to check other data and results for consistency and/or to derive dynamic modelling parameters from transfer functions. A full list of the variables requested is provided in Table 2-1.

2.3 National responses

The CCE received responses from 19 of the 24 countries that contributed critical loads data used in the negotiations of the Gothenburg Protocol. Of these countries, 18 updated their critical loads calculations, and 10 submitted data for dynamic modelling. Finland explicitly instructed the CCE to use the data previously submitted.

Several countries that did not submit data indicated that they planned to participate in the next CCE call for data, expected in late 2003. Some countries that provided data for dynamic modelling did not do so for all ecosystems. An overview of the national contributions is given in Table 2-2. Data types are as used in MS-Access: "Single" means a (real) number, "Integer" an integer number, and "Text(10)" means a string of maximum 10 characters.

Table 2-1. List of variables requested in the 2002 call for data (including corrections to the original list).

Variable name	Data type	Description (units)
Group 1: Critical load variables		
Lon	Single	Longitude (decimal degrees)
Lat	Single	Latitude (decimal degrees)
I50	Integer	EMEP50 horizontal coordinate
J50	Integer	EMEP50 vertical coordinate
ecoarea	Single	Area of the ecosystem within the EMEP grid (km^2)
CLmaxS	Single	Maximum critical load of sulphur (eq ha^{-1} a^{-1})
CLminN	Single	Minimum critical load of nitrogen (eq ha^{-1} a^{-1})
CLmaxN	Single	Maximum critical load of nitrogen (eq ha^{-1} a^{-1})
CLnutN	Single	Critical load of nutrient nitrogen (eq ha^{-1} a^{-1})
BCdep	Single	Sea-salt corrected base deposition minus sea-salt corrected Cl deposition (eq ha^{-1} a^{-1})
Bcupt	Single	Net growth uptake of plant available base cations (eq ha^{-1} a^{-1})
BCwe	Single	Amount of base cations produced by weathering (eq ha^{-1} a^{-1})
Qle	Single	Amount of water percolating through the root zone (mm a^{-1})
Kgibb	Single	Equilibrium constant for the Al:H relationship (m^6 eq^{-2})
nANCcrit	Single	The positive quantity $Al_{le(crit)} + H_{le(crit)}$ (eq ha^{-1} a^{-1})
Nimm	Single	Acceptable amount of nitrogen immobilised in the soil (eq ha^{-1} a^{-1})
Nupt	Single	Net growth uptake of nitrogen (eq ha^{-1} a^{-1})
Nfde	Single	Amount of nitrogen denitrified, N_{de} (eq ha^{-1} a^{-1}), or the denitrification fraction f_{de} ($0 \leq f_{de} < 1$) (–)
Nleacc	Single	Acceptable nitrogen leaching (eq ha^{-1} a^{-1})
ecocode	Text(10)	EUNIS code
Group 2: Minimum requirements for dynamic modelling		
thick	Single	Depth of the rooting zone (m)
rho	Single	Bulk density of the soil (g cm^{-3})
theta	Single	Volumetric water content at field capacity (m^3 m^{-3})
CEC	Single	Cation exchange capacity (meq kg^{-1})
EBC	Single	Base saturation (–)
yearEBC	Integer	Year in which the base saturation was determined
Cpool	Single	Amount of carbon in the topsoil (g m^{-2})
CNrat0	Single	C:N ratio in the topsoil (g g^{-1})
Group 3: Additional variables for consistency checks and transfer functions input		
soiltype	Text(10)	FAO soil type
clay	Single	Clay content of the mineral soil (%)
sand	Single	Sand content of the mineral soil (%)
Corg	Single	Organic carbon content of the soil (%)
pH	Single	(–)
Prec	Single	Mean annual precipitation (mm a^{-1})
Temp	Single	Mean annual temperature (^oC)
Alt	Single	Altitude above sea level (m)

Table 2-2. Status of data submissions for critical loads and dynamic modelling variables.

Country	Code	Critical loads data	Dynamic modelling data
Austria	AT		
Belarus	BY	x	
Belgium	BE	x (Flanders)	
Bulgaria	BG	x	x
Croatia	HR	x	x
Czech Republic	CZ	x	x
Denmark	DK	x	x
Estonia	EE		
Finland	FI	x (2001 data)	
France	FR	x	
Germany	DE	x	x
Hungary	HU	x	
Ireland	IE	x	x
Italy	IT	x	
Netherlands	NL	x	x
Norway	NO	x	
Poland	PL	x	x
Rep. of Moldova	MD		
Russia	RU		
Slovakia	SK	x	x
Spain	ES		
Sweden	SE	x	
Switzerland	CH	x	x
United Kingdom	GB	x	–
Totals	**24**	**19**	**10**

The EUropean Nature Information System (EUNIS) classification was used to characterise ecosystems. Of the countries that submitted data using this classification, the hierarchic level (number of characters) used varied. At present, only two digits (i.e., equivalent to EUNIS Level 2) are stored in the European critical load database. The non-EUNIS ecosystem codes submitted by some countries have been translated by the CCE into EUNIS codes, based on the description provided. The resulting list of ecosystems is then aggregated into 10 classes that are comparable with previous CCE reports and the CCE land use map. For the cumulative distribution functions of the variables shown later in this chapter, a further aggregation was made to three classes: "forest", "water" and "vegetation".

The list of EUNIS codes, their description and the corresponding aggregated classes are listed in Table 2-3.

Table 2-3. EUNIS (Level 1 and 2) codes used by countries that submitted critical loads data.

Code	EUNIS code description	Land use category	Ecosystem class
A2	Littoral sediments	Other	Vegetation
B1	Coastal dune and sand habitats	Other	Vegetation
B2	Coastal shingle habitats	Other	Vegetation
C	Inland surface water habitats	Water	Water
C1	Surface standing waters	Water	Water
C2	Surface running waters	Water	Water
C3	Littoral zone of inland surface water bodies	Other	Vegetation
D	Mire, bog and fen habitats	Wetlands	Vegetation
D1	Raised and blanket bogs	Wetlands	Vegetation
D2	Valley mires, poor fens and transition mires	Wetlands	Vegetation
D4	Base-rich fens	Wetlands	Vegetation
D5	Sedge and reedbeds, normally without free-standing water	Wetlands	Vegetation
D6	Inland saline and brackish marshes and reedbeds	Wetlands	Vegetation
E	Grassland and tall forb habitats	Grassland	Vegetation
E1	Dry grasslands	Grassland	Vegetation
E2	Mesic grasslands	Grassland	Vegetation
E3	Seasonally wet and wet grasslands	Grassland	Vegetation
E4	Alpine and subalpine grasslands	Grassland	Vegetation
F	Heathland, scrub and tundra habitats	Heathland	Vegetation
F1	Tundra	Heathland	Vegetation
F2	Arctic, alpine and subalpine scrub habitats	Shrub	Vegetation
F4	Temperate shrub heathland	Shrub	Vegetation
F9	Riverine and fen scrubs	Shrub	Vegetation
G	Woodland and forest habitats and other wooded land	Forest	Forest
G1	Broadleaved deciduous woodland	Broadleaved forest	Forest
G2	Broadleaved evergreen woodland	Broadleaved forest	Forest
G3	Coniferous woodland	Coniferous forest	Forest
G4	Mixed deciduous and coniferous woodland	Mixed forest	Forest
Y	– (unspecified)	Other	Vegetation

The following table and histograms show the ecosystem area per country for which data have been submitted. It provides an overview of the resolution that countries have used, and illustrates which ecosystems were deemed relevant for inclusion in the critical load calculations.

Figure 2-1 shows the area covered per ecosystem type (i.e. percentage of total country area) for critical loads of acidity and/or eutrophication, while Figure 2-2 shows the coverage of these ecosystems for which dynamic modelling data have been submitted. The bar charts depict the different ecosystem types from Table 2-3. In Figure 2-1 Norway appears twice, the second bar being for the "water" classification only, which includes the catchment area. Therefore, the total area for water and soil ecosystems is larger than 100%. Two remarks should be made. Part of EUNIS class G4 in the United Kingdom (GB) is unmanaged woodland of either coniferous or broadleaved trees. Though this is not mixed forest, it is classified as such. France used potential vegetation types and submitted the EUNIS codes for these "ecosystems".

Dynamic modelling variables are derived primarily for forests, due partly to the fact that some countries use empirical critical loads for natural vegetation, or only provide critical loads for eutrophication for certain ecosystem types.

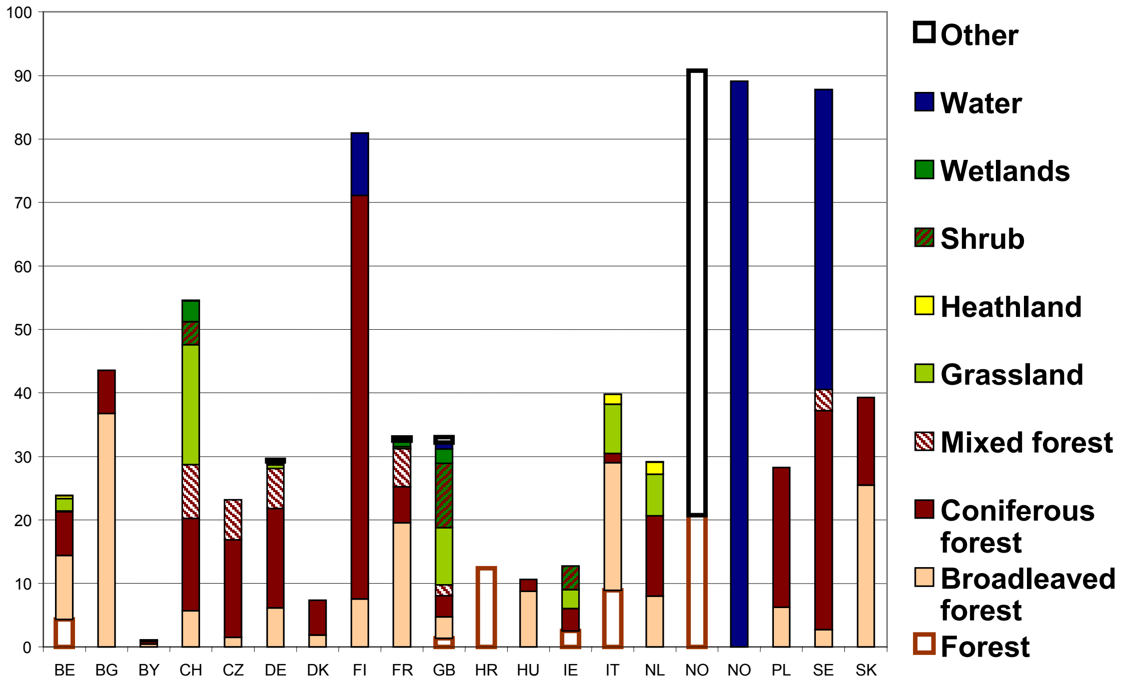

Figure 2-1. National distribution of ecosystem types and their areas (as % of total country area).

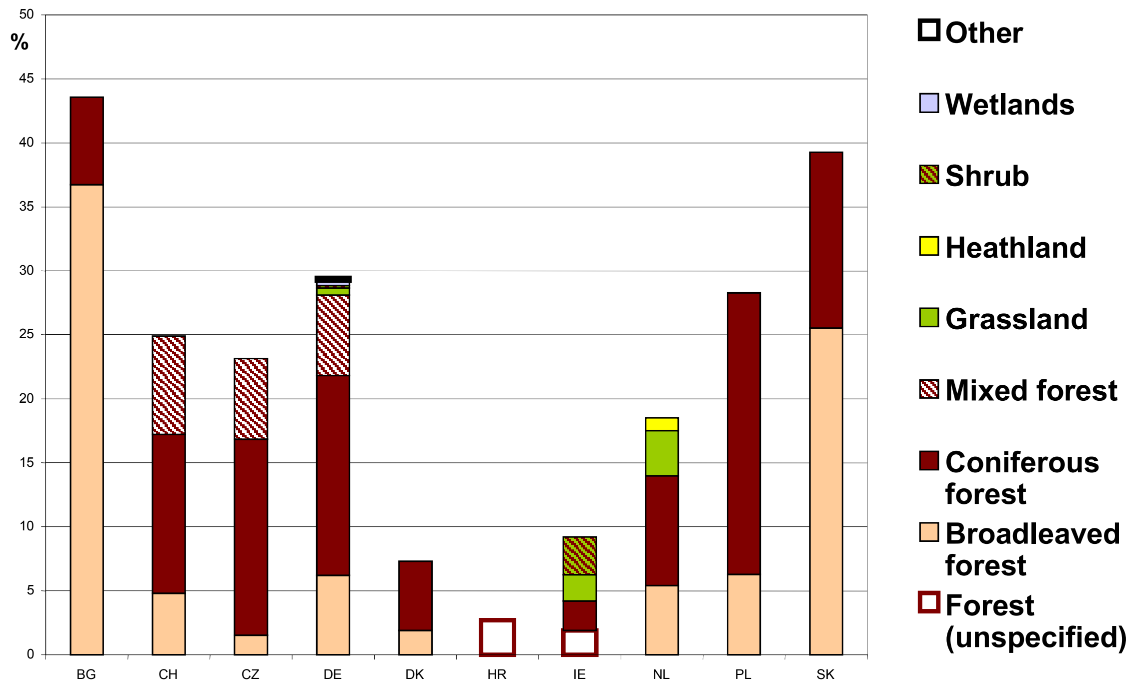

Figure 2-2. National distributions of ecosystems for which dynamic modelling variables have been submitted and their areas (as % of total country area).

Table 2-4 shows the number of ecosystems and their area relative to the total country area for each ecosystem type for all records, but also separately for acidity, eutrophication and dynamic modelling.

Table 2-4. Type and number of ecosystems for which data were provided by National Focal Centres in response to the 2002 call for data.

				All submitted records		Acidification		Eutrophication		Dynamic modelling parameters	
Country	**Code**	**Ecosystem type**	**Ecosystem area (km^2)**	**# of eco-systems**	**% area**	**# of eco-systems**	**% area**	**# of eco-systems**	**% area**	**# of eco-systems**	**% area**
Belarus	BY	Broadleaved forest	977	75	0.47	75	0.47	75	0.47		
		Coniferous forest	792	52	0.38	52	0.38	52	0.38		
		Grassland	62	9	0.03	9	0.03	9	0.03		
		Mixed forest	256	22	0.12	22	0.12	22	0.12		
		Wetlands	193	14	0.09	14	0.09	14	0.09		
Belgium	BE	Broadleaved forest	3,064	766	10.04	766	10.04	766	10.04		
		Coniferous forest	2,107	616	6.90	616	6.90	616	6.90		
		Forest (unspecified)	1,327	1,690	4.35	1,690	4.35	1,690	4.35		
		Grassland	601	482	1.97	482	1.97	482	1.97		
		Heathland	136	79	0.45	79	0.45	79	0.45		
		Mixed forest	39	351	0.13	351	0.13	351	0.13		
		Water	8	12	0.03	12	0.03	12	0.03		
Bulgaria	BG	Broadleaved forest	40,776	55	36.74	55	36.74	55	36.74	55	36.74
		Coniferous forest	7,569	29	6.82	29	6.82	29	6.82	29	6.82
Croatia	HR	Forest (unspecified)	7,009	144	12.40	140	12.26	144	12.40	21	2.72
Czech Republic	CZ	Broadleaved forest	1,195	744	1.51	744	1.51	744	1.51	744	1.51
		Coniferous forest	12,088	4,021	15.33	4,021	15.33	4,021	15.33	4,021	15.33
		Mixed forest	4,990	2,528	6.33	2,528	6.33	2,528	6.33	2,528	6.33
Denmark	DK	Broadleaved forest	813	3,261	1.89	3,261	1.89	3,261	1.89	3,258	1.89
		Coniferous forest	2,336	6,497	5.42	6,497	5.42	6,497	5.42	6,495	5.42
Finland	FI	Broadleaved forest	25,544	1,034	7.55	1,030	7.55	1,034	7.55		
		Coniferous forest	214,860	2,049	63.54	2,049	63.54	2,049	63.54		
		Water	33,231	1,450	9.83	1,450	9.83				
France	FR	Broadleaved forest	106,365	2,698	19.55	2,698	19.55	2,698	19.55		
		Coniferous forest	30,968	482	5.69	482	5.69	482	5.69		
		Grassland	1,576	81	0.29	81	0.29	81	0.29		
		Mixed forest	32,704	659	6.01	659	6.01	659	6.01		
		Other	2,724	154	0.50	154	0.50	154	0.50		
		Wetlands	5,092	66	0.94	66	0.94	66	0.94		
Germany	DE	Broadleaved forest	22,078	88,311	6.18	88,311	6.18	88,311	6.18	88,311	6.18
		Coniferous forest	55,803	223,213	15.63	223,213	15.63	223,213	15.63	223,213	15.63
		Grassland	1,957	7,829	0.55	7,829	0.55	7,829	0.55	7,829	0.55
		Mixed forest	22,442	89,766	6.29	89,766	6.29	89,766	6.29	89,766	6.29
		Other	854	3,414	0.24	3,414	0.24	3,414	0.24	3,414	0.24
		Shrub	693	2,772	0.19	2,772	0.19	2,772	0.19	2,772	0.19
		Wetlands	1,330	5,319	0.37	5,319	0.37	5,319	0.37	5,319	0.37
Hungary	HU	Broadleaved forest	8,119	669	8.73	669	8.73	669	8.73		
		Coniferous forest	1,743	363	1.87	363	1.87	363	1.87		
Ireland	IE	Coniferous forest	2,449	9,195	3.48	9,195	3.48	9,195	3.48	6,422	2.31
		Forest (unspecified)	1,805	8,047	2.57	8,047	2.57	8,047	2.57	6,180	1.91
		Grassland	2,050	6,895	2.92	6,895	2.92	6,895	2.92	4,850	2.02
		Shrub	2,631	6,847	3.74	6,847	3.74	6,847	3.74	5,419	2.95
Italy	IT	Broadleaved forest	60,577	165	20.10	165	20.10	165	20.10		
		Coniferous forest	4,546	22	1.51	22	1.51	22	1.51		
		Forest (unspecified)	26,787	151	8.89	151	8.89	151	8.89		
		Grassland	23,235	118	7.71	118	7.71	118	7.71		
		Heathland	4,709	46	1.56	46	1.56	46	1.56		
Netherlands	NL	Broadleaved forest	3,325	37,359	8.01	37,359	8.01	37,359	8.01	19,230	5.42
		Coniferous forest	5,248	45,258	12.64	45,258	12.64	45,258	12.64	16,870	8.57
		Grassland	2,713	31,738	6.53	31,738	6.53	31,738	6.53	10,672	3.53
		Heathland	811	8,788	1.95	8,788	1.95	8,788	1.95	2,147	1.01
		Wetlands	5	291	0.01			291	0.01		

Table 2-4 (continued). Type and number of ecosystems for which data were provided by National Focal Centres.

Country	Code	Ecosystem type	Ecosystem area (km^2)	All submitted records: # of eco-systems	All submitted records: % area	Acidification: # of eco-systems	Acidification: % area	Eutrophication: # of eco-systems	Eutrophication: % area	Dynamic modelling parameters: # of eco-systems	Dynamic modelling parameters: % area
Norway	NO	Forest (unspecified)	67,124	663	20.73	662	20.70				
		Other	226,631	1,610	70.00			1,610	70.00		
		Water	288,522	2,304	89.12	2,304	89.12				
Poland	PL	Broadleaved forest	19,575	19,575	6.26	19,575	6.26	19,575	6.26	19,575	6.26
		Coniferous forest	68,808	68,808	22.01	68,808	22.01	68,808	22.01	68,808	22.01
Slovakia	SK	Broadleaved forest	12,507	208,452	25.51	208,452	25.51	208,452	25.51	208,452	25.51
		Coniferous forest	6,746	112,439	13.76	112,439	13.76	112,439	13.76	112,439	13.76
Sweden	SE	Broadleaved forest	12,173	136	2.71	128	2.55	136	2.71		
		Coniferous forest	155,050	1,581	34.46	1,492	32.75	1,581	34.46		
		Mixed forest	15,000	146	3.33	144	3.31	146	3.33		
		Water	212,879	2,983	47.31	2,887	45.67				
Switzerland	CH	Broadleaved forest	2,350	370	5.69	132	5.12	370	5.69	124	4.81
		Coniferous forest	6,009	909	14.55	340	13.18	909	14.55	320	12.40
		Grassland	7,777	7,777	18.84			7,777	18.84		
		Mixed forest	3,504	219	8.49	219	8.49	177	6.86	198	7.67
		Shrub	1,512	1,512	3.66			1,512	3.66		
		Water	38	38	0.09			38	0.09		
		Wetlands	1,348	1,348	3.27			1,348	3.27		
United Kingdom	GB	Broadleaved forest	8,362	83,303	3.44	76,383	3.09	83,303	3.44		
		Coniferous forest	7,944	36,606	3.26	36,533	3.26	36,606	3.26		
		Forest (unspecified)	3,285	32,032	1.35			32,032	1.35		
		Grassland	21,897	119,062	9.00	99,509	8.23	119,062	9.00		
		Mixed forest	4,103	38,646	1.69	37,417	1.64				
		Other	2,119	10,299	0.87			10,299	0.87		
		Shrub	24,785	78,985	10.19	78,550	10.14	78,985	10.19		
		Water	2,441	1,161	1.00	1,161	1.00				
		Wetlands	5,506	19,079	2.26	18,682	2.24	19,079	2.26		

Some countries provided critical loads of acidity, eutrophication and dynamic modelling parameters for all ecosystems they submitted. For several reasons this is not true for all countries. For example, dynamic modelling parameters will seldom be available for ecosystems for which (only) empirical critical loads are derived. Also, dynamic models are not necessarily suited for all ecosystems, such as (semi-)natural vegetation. One should bear in mind, however, that differences in the number of ecosystems used for calculating critical loads and dynamic model output may lead to inconsistencies, if the (subset of) ecosystems for which dynamic modelling variables are provided do not cover the entire range of sensitivity to an equal degree (see section 2.6).

2.4 Comparison with 1998 data

This section compares the results of the 2002 call for data to the 1998 database which was used for the Gothenburg Protocol. The data are not quantified statistically, but plotted next to each other for visual comparison. Variables are shown as cumulative distribution functions (cdfs) showing the (area-weighted) distribution normalised for each country. The cdfs are computed separately for three main ecosystems classes: "forest", "water" and "vegetation", as described in Table 2-3. Note that even if two cdfs look similar, the data may differ in different areas (grid cells) of the country. All figures show the 1998 values at the left and the 2003 data at the right.

Chapter 1 includes several maps for the critical loads of acidity and eutrophication, for both the 5th and 50th percentile. The cdfs in this chapter show the entirety of the distribution. The numbers on the right side of each graph indicate the number of ecosystems reported for each class. It is also indicated if all values are outside the range displayed (e.g. by ">3000"). If ecosystem numbers are shown and a corresponding cdf is not visible, it means that it is underneath the cdf displayed.

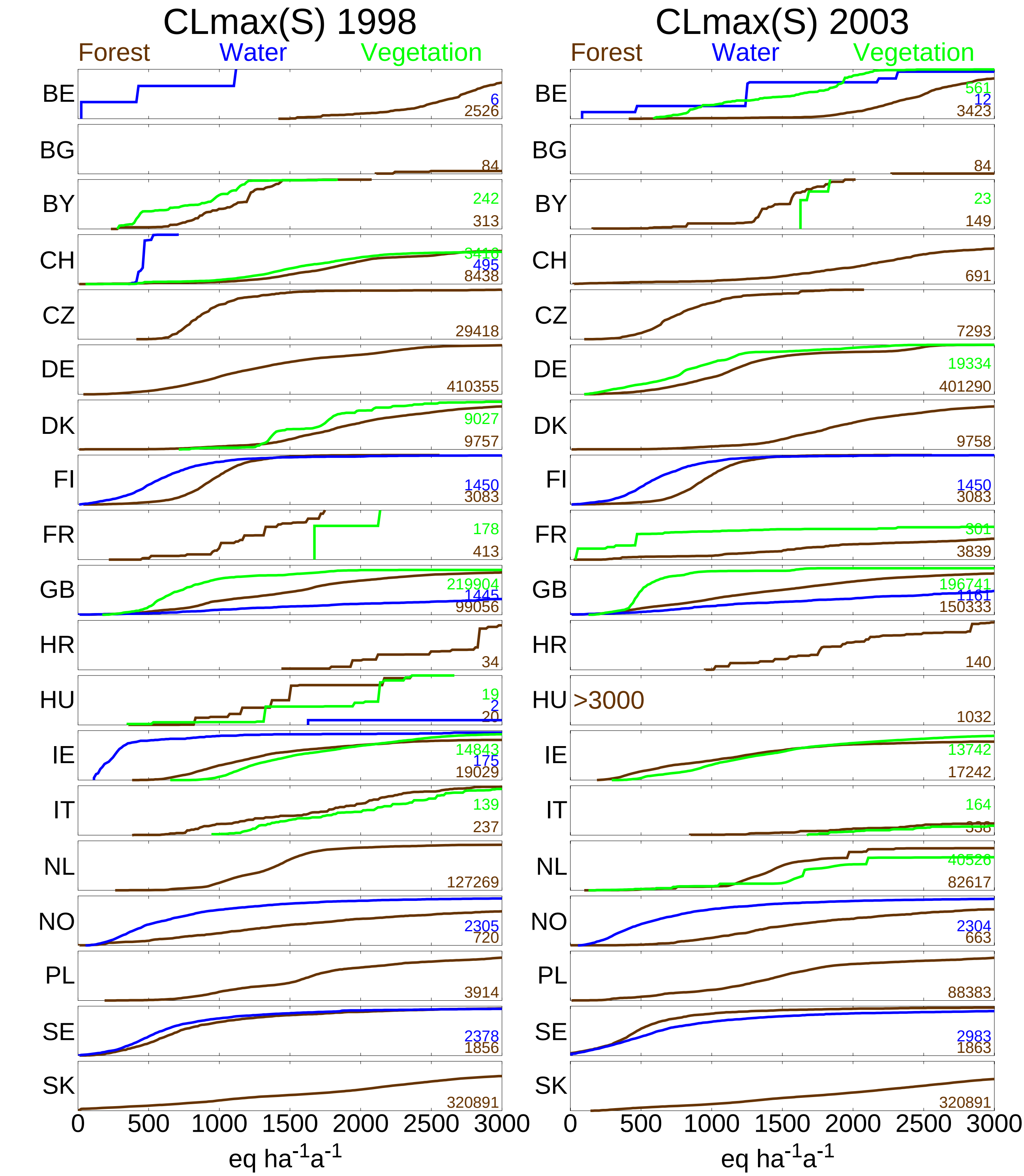

Figure 2-3. Maximum critical load of sulphur from 1998 (left) and 2003 (right).

The cdf plots in Figures 2-3 for $CL_{max}(S)$ and 2-4 for $CL_{nut}(N)$, as well as Figures 1-1 through 1-4 in Chapter 1, indicate that critical load values have changed since 1998. These changes are caused primarily by the expansion of areal coverage or the addition of ecosystem types in the national calculations, or by new insights into underlying variables. This makes it is useful to also compare cdfs for the most important variables:

- Base cation deposition (BCdep) in Figure 2-5.
- Base cation uptake (BCupt) in Figure 2-6.
- Base cation weathering (BCwe) in Figure 2-7.
- Critical leaching of Acid Neutralising Capacity (–ANCle(crit)) in Figure 2-8.
- Nitrogen immobilisation (Nimm) in Figure 2-9.
- Acceptable leaching of N (Nle(acc)) in Figure 2-10.

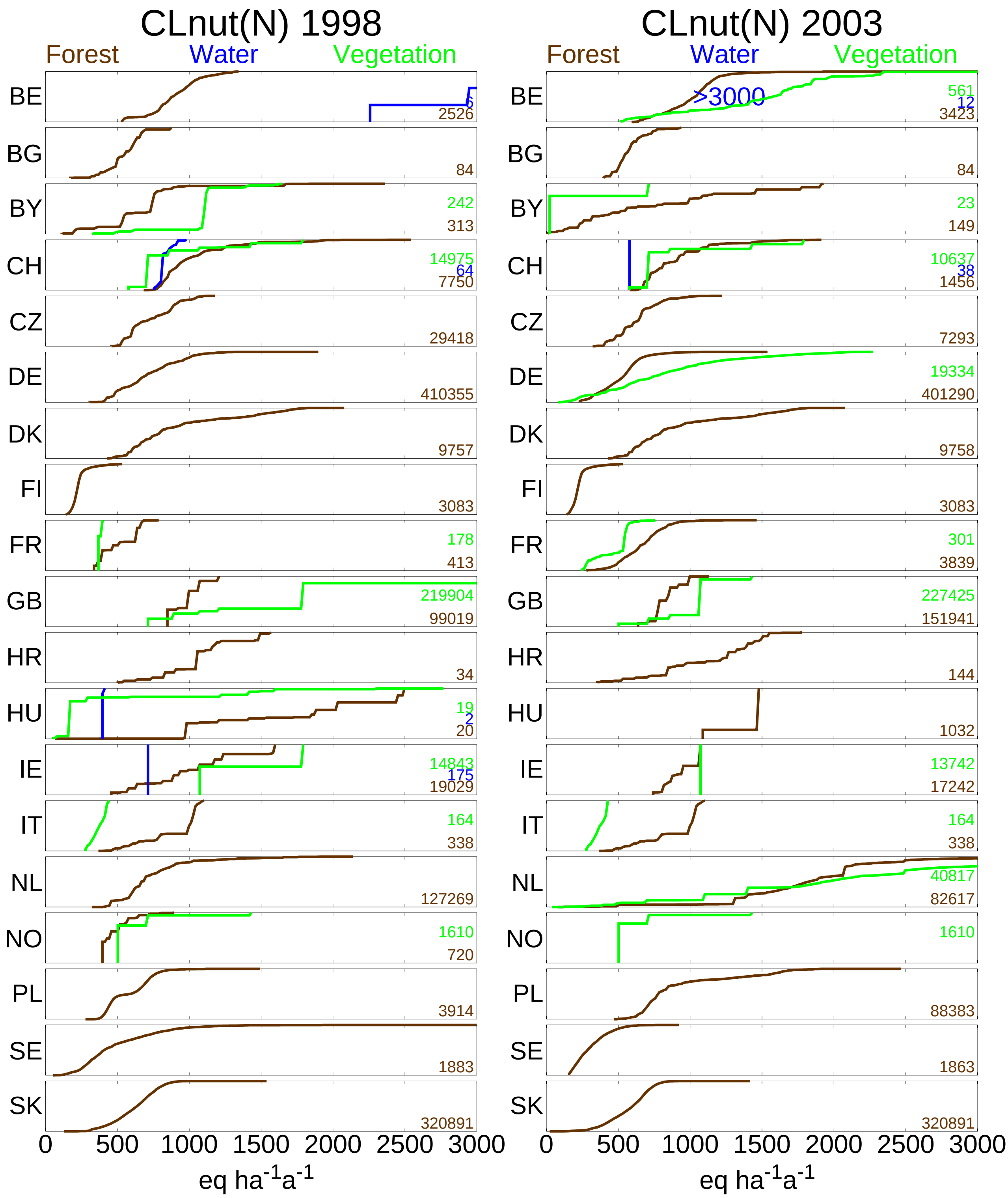

Figure 2-4. The critical load of nutrient nitrogen from 1998 (left) and 2003 (right).

Figure 2-5. Base cation deposition from 1998 (left) and 2003 (right) for the three main ecosystem classes.

Figure 2-5 summarises the base cation deposition data submitted by each country. The following figures do not include data submitted for surface waters, since this variable is not an input to surface water critical load models.

Bulgaria, France, Croatia, the Czech Republic Germany and Poland have updated base cation deposition data noticeably (see also the respective NFC reports in Part II). Note that in this and the following figures, the number of ecosystems can be lower than in Figs. 2-3 and 2.4, since values for the respective variables were sometimes not provided.

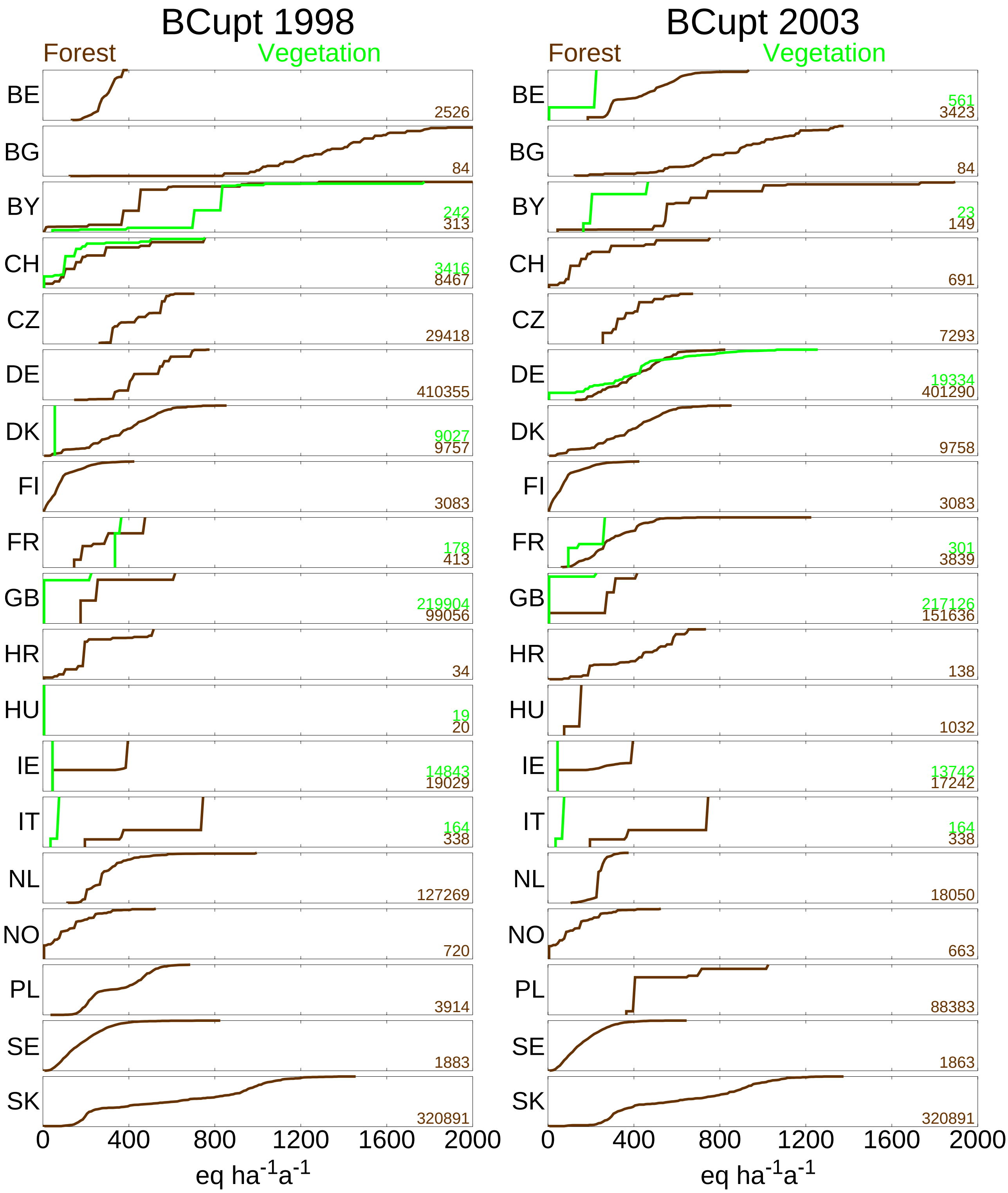

Figure 2-6. Base cation uptake from 1998 (left) and 2003 (right) for the three main ecosystem classes.

Figure 2-6 demonstrates that most countries have uptake values comparable to 1998, except for Belgium, France, Hungary and Poland. The changes in Belarus can be explained by the large change in area coverage of this year's submission.

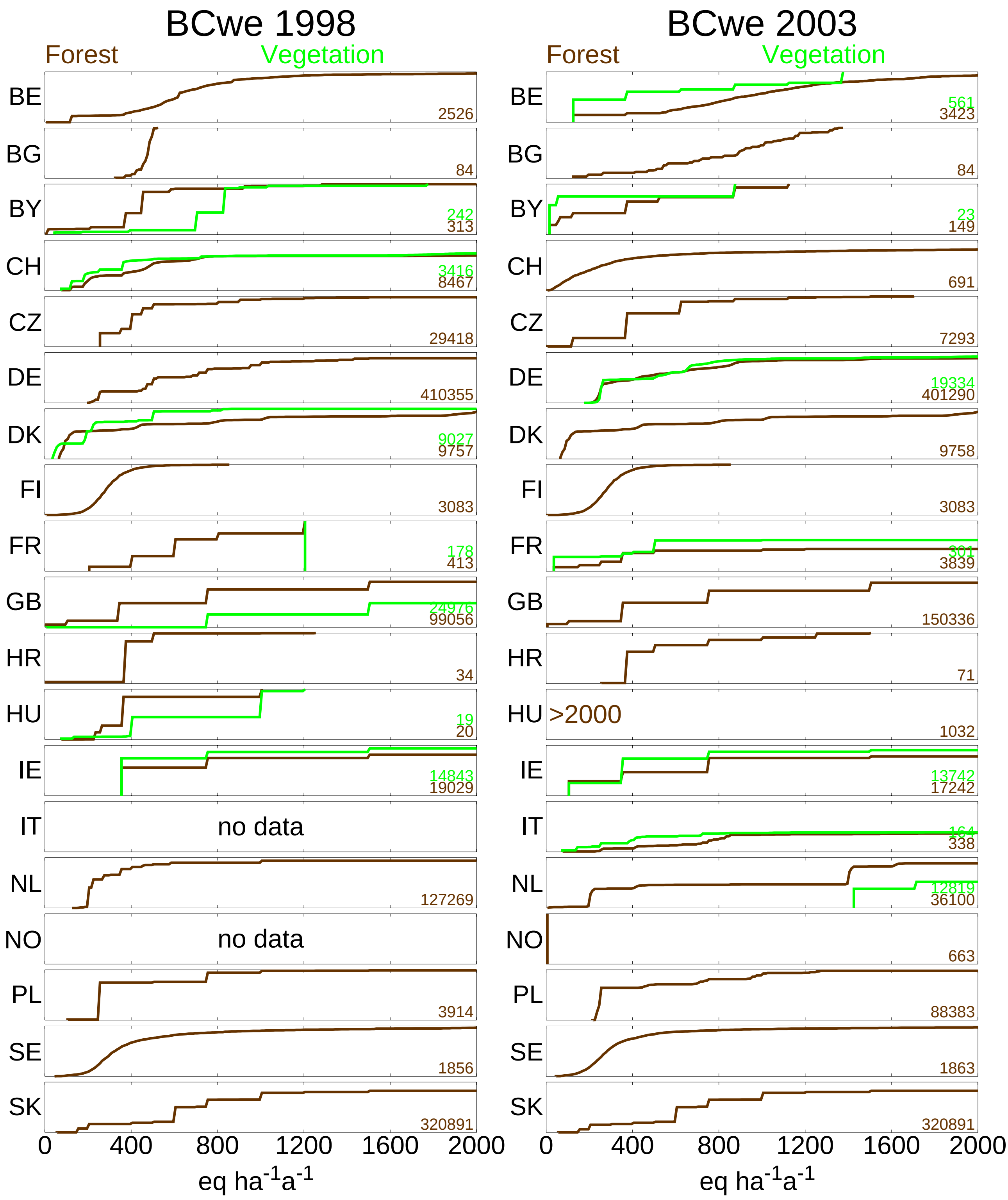

Figure 2-7. Base cation weathering for 1998 (left) and 2003 (right).

Figure 2-7 shows that significant changes in base cation weathering data have occurred in Bulgaria, France, Hungary and Netherlands. (See the respective NFC reports in Part II.)

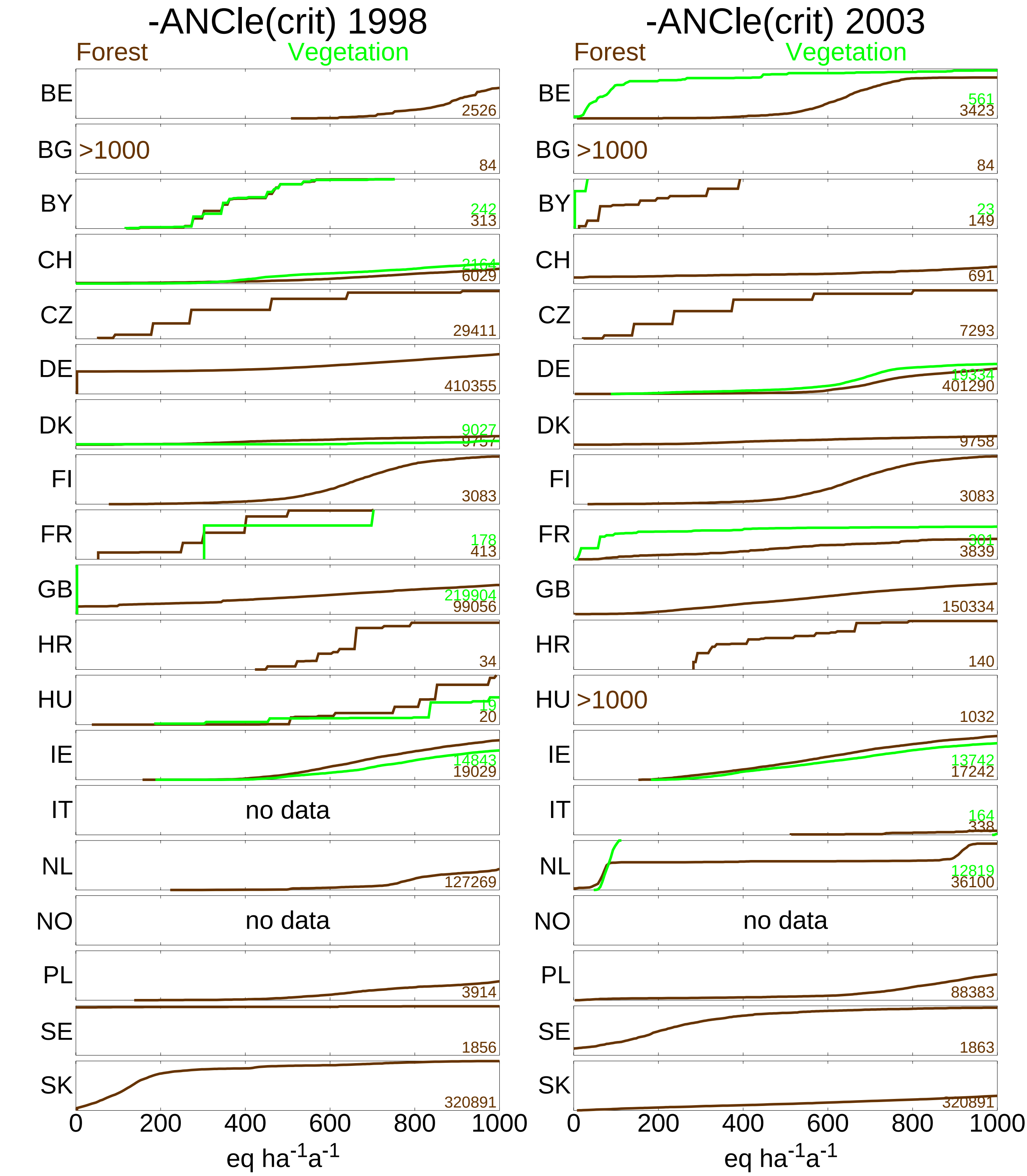

Figure 2-8. Critical leaching of acid neutralising capacity (=$Al_{le(crit)}$+$H_{le(crit)}$) from 1998 (left) and 2003 (right).

In Figure 2-8, note that the scale differs from the previous figures. The absolute changes are relatively small in most countries, except in Belarus (much smaller ecosystem area in 2003), France and the Netherlands (new methodology since 2001, see Posch et al. 2001).

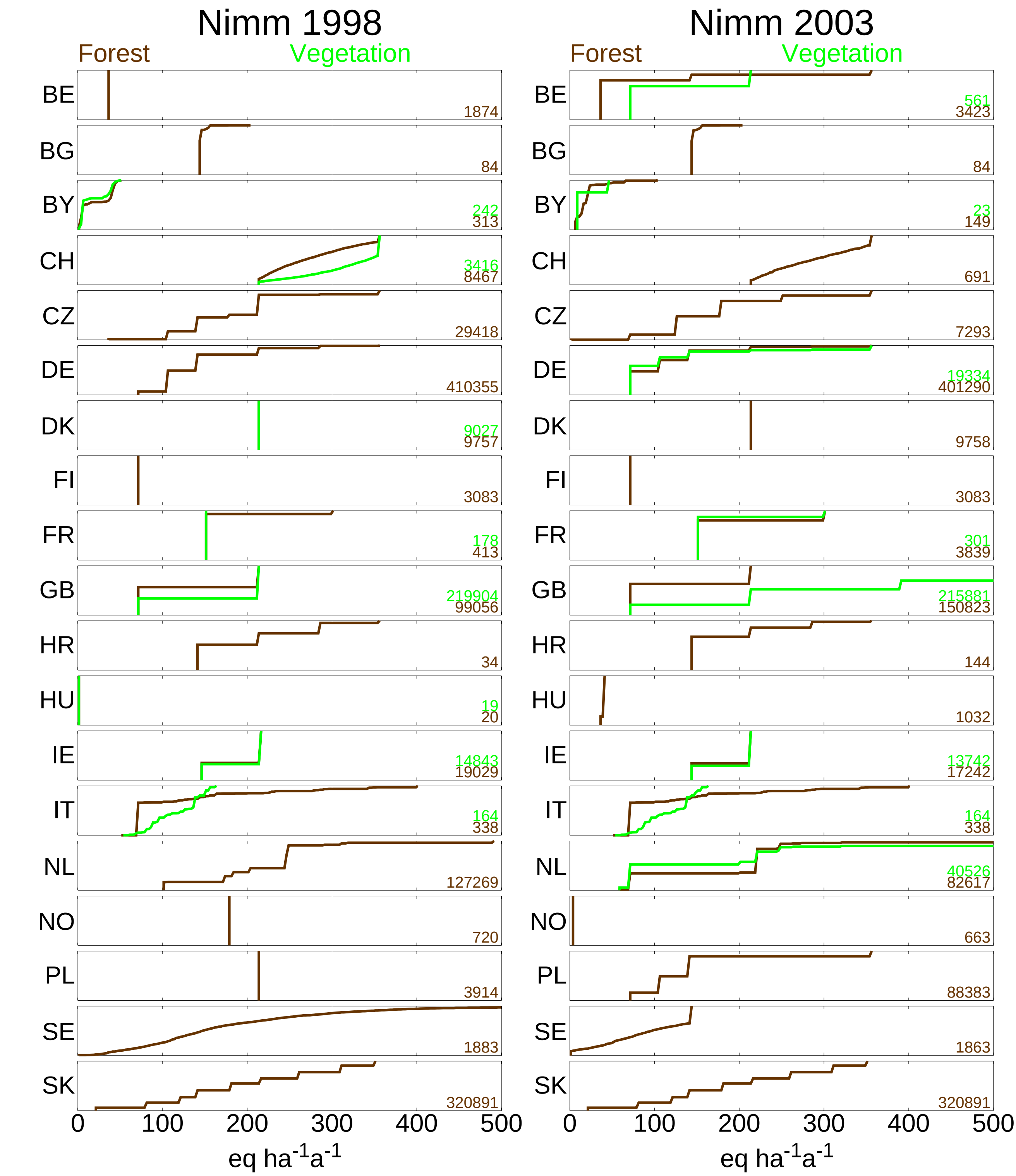

Figure 2-9. Nitrogen immobilisation from 1998 (left) and 2003 (right).

Figure 2-9 shows that most values for N immobilisation are relatively unchanged from 1998. Long-term nitrogen immobilisation is recommended at 0.5–1 kg N ha^{-1} a^{-1} (71–142 eq ha^{-1} a^{-1}) in the Mapping Manual (UBA 1996).

Figure 2-10. Acceptable nitrogen leaching from 1998 (left) and 2003 (right).

In Figure 2-10, a noticeable change can be seen in the Dutch data. Already in 2001 the acceptable nitrogen leaching submitted had been based on groundwater quality (based on strict levels for drinking water) and for nutrient imbalances in forest soils. For terrestrial vegetation and heathland lakes a methodology is used without using the concept of acceptable N leaching. (See the Dutch NFC report in Posch et al. 2001).

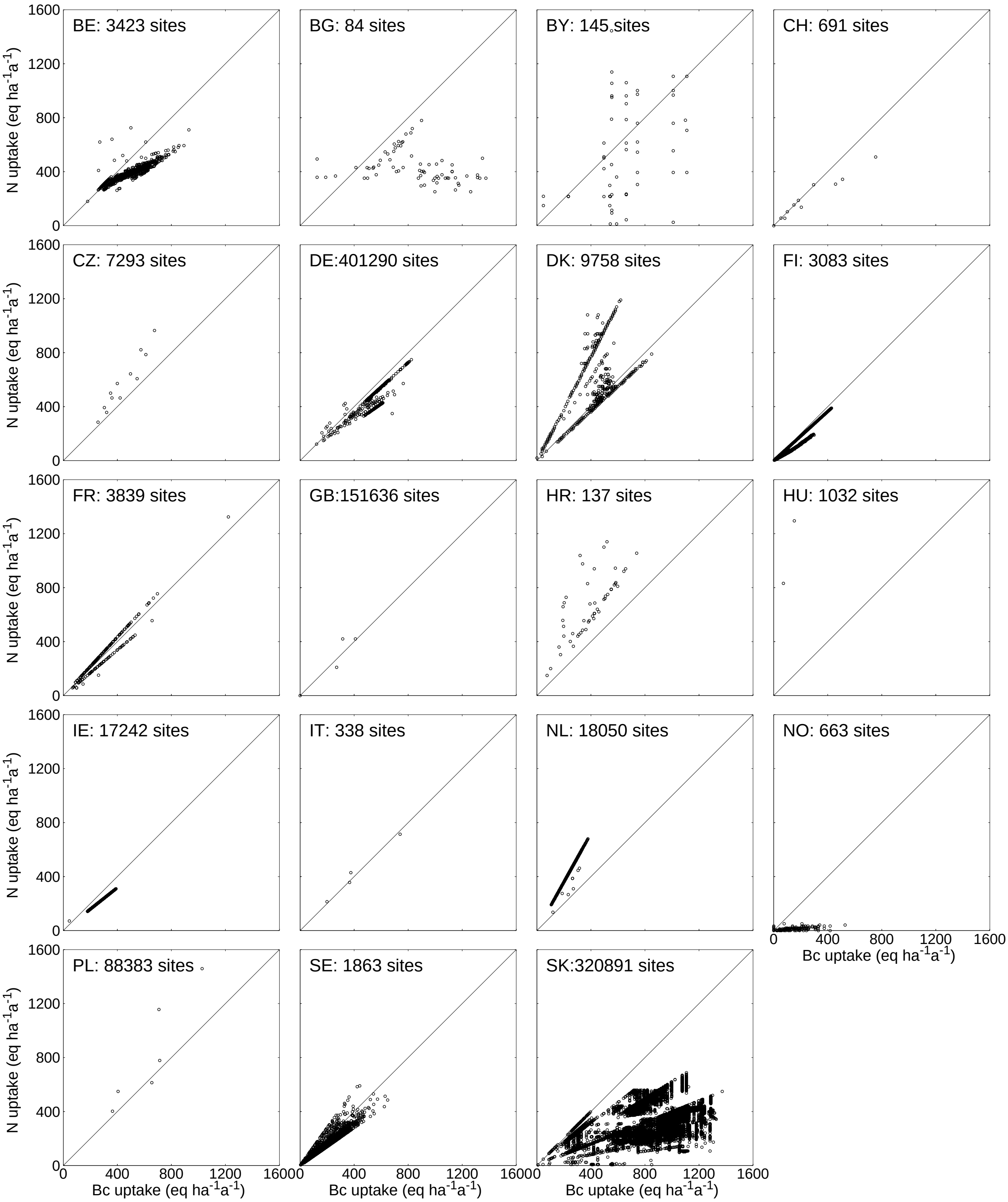

Figure 2-11. Base action uptake versus nitrogen uptake for forest ecosystems.

Figure 2-11 shows the correlation between uptake of base cations (Ca+Mg+K) and nitrogen by forest ecosystems in each country. Note the discrepancy between the number of sites and the number of different uptake values for some countries. The uptake values shown above are assumed to reflect *net* growth uptake, i.e. they should equal the annual average amount of these elements removed by harvesting. Thus they depend not only on tree species and climate, but also on harvesting practices; e.g. nature reserves from which no trees are removed should be assigned zero uptake values (for both variables). The *ratio* of base cation to nitrogen uptake for a given tree species should be relatively constant, with only minor variations due to climate and site quality. For example, it is very unlikely that the uptake of N is very small when the uptake of base cations for the same ecosystem is within expected ranges.

2.5 Data for dynamic modelling

This section discusses the cumulative distributions from the ten countries that submitted variables for dynamic modelling. As noted in the NFC reports in Part II, other countries beside these ten have been working on dynamic modelling in anticipation of the next CCE call for data.

Figure 2-12 shows cumulative distributions of (from left to right): cation exchange capacity (CEC), base saturation (EBC), carbon pool (Cpool), and C:N ratio (CNrat). The product of CEC and base saturation is the (maximum) amount of base cations available to buffer (net) acidity inputs, and its depletion can be considered detrimental for the soil chemical status. The magnitude of CEC determines the speed of recovery when (and if) acidifying deposition becomes low enough. Base saturation characterises the fraction of base cations left at the exchange sites, and a low value indicates a strong depletion of these ions. Base saturations near one (or 100%) indicate calcareous soils.

The carbon pool and the C:N ratio determine the time-dependent N immobilisation in some dynamic models. Low C:N ratios (below 15 g g^{-1}) mean a high saturation of the soil with nitrogen, and thus an increased future leaching of nitrogen is likely.

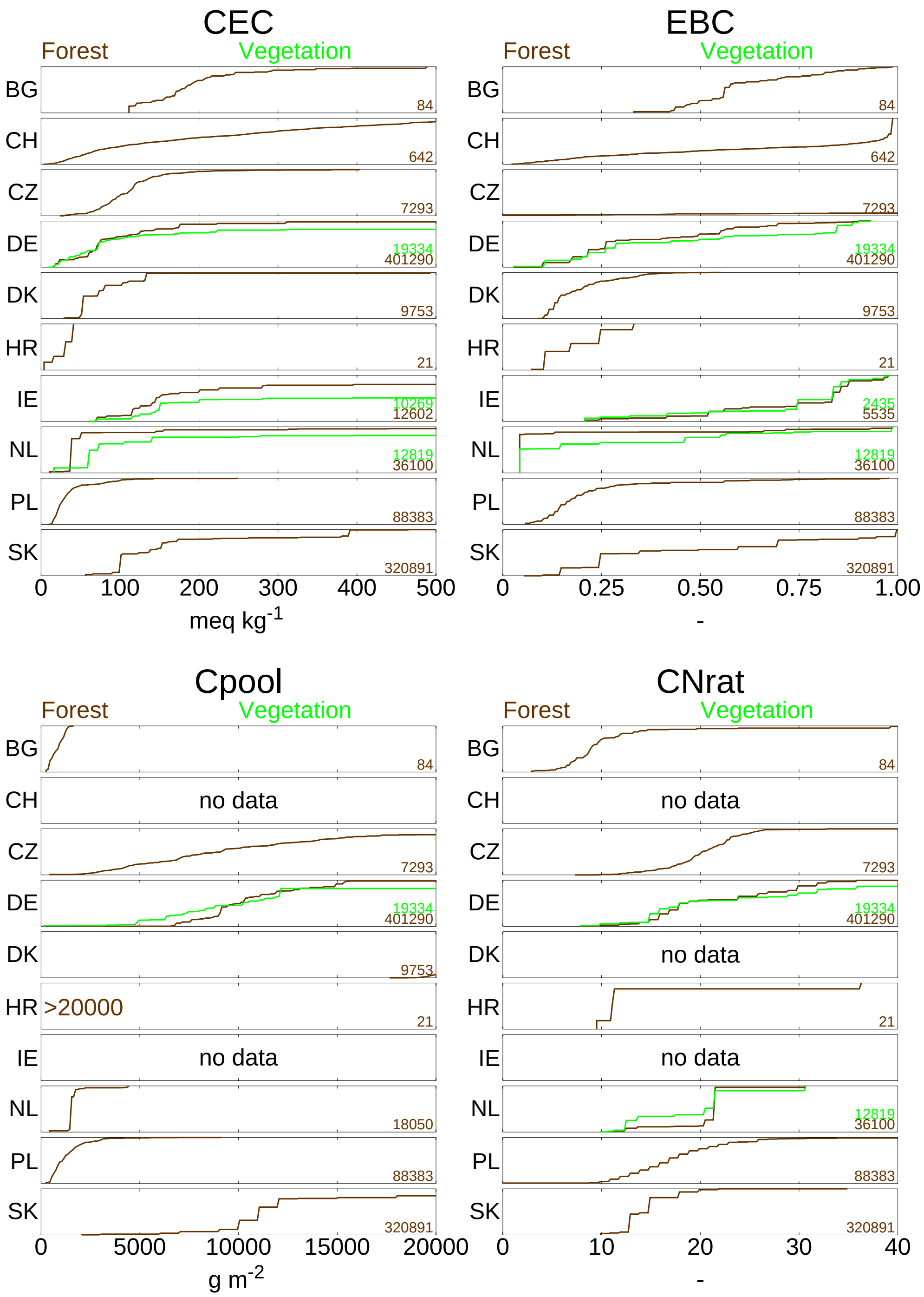

Figure 2-12. Cumulative distributions of cation exchange capacity (CEC), base saturation (EBC), carbon pool (Cpool), and C:N ratio (CNrat).

2.6 Discussion

One of the purposes of the 2002 call for data was to encourage countries to compile and submit data for variables needed for dynamic modelling. An important aspect of this exercise is to address the compatibility of the critical loads and dynamic modelling data. This issue has two major aspects: (a) the steady-state solution of a dynamic model application should coincide with the critical loads for the same site, and (b) the distribution of sensitivity indicators (critical loads) of the ecosystems within an EMEP grid cell should be the same or, at least, very similar.

This does not necessarily mean that dynamic modelling must be carried out at all sites, but at a selection of sites that yield the same distribution (at least for the most sensitive ecosystems). Figure 2-13 compares, as an illustrative example only, the cdfs of $CL_{max}(S)$ for 3 countries for *all* ecosystems in the country for which acidity critical loads have been provided (green) with those ecosystems for which dynamic modelling data have also been submitted (purple). The figure shows that in country A, there is practically no difference between the cdfs, which is not surprising since the number of ecosystems differ by less than 10%. Despite the large difference in numbers of ecosystems, the two cdfs for country B are quite similar, especially in the important lower range. For country C, however, the quite substantial difference in the lower range of the cdfs (e.g. the 10th percentiles differ by several hundred equivalents) requires further consideration regarding the selection of ecosystems for which dynamic modelling is planned. The purpose of this example to highlight one issue that requires attention before the next data submission.

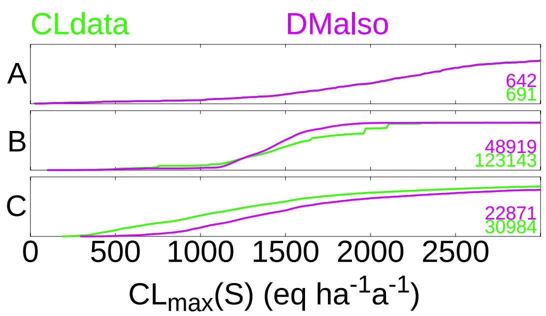

Figure 2-13. Comparison of cumulative distributions of $CL_{max}(S)$ for 3 countries: (a) for *all* ecosystems in the country (green), and (b) for those ecosystems for which also dynamic modelling data have been submitted (purple).

2.7 Concluding remarks

Critical loads data have changed since 1998 for most of the countries. Changes are caused by new insights, added (or discarded) areas/ecosystems or improved methodology.

Ten countries succeeded in submitting a minimal set of variables needed for dynamic modelling. Most other countries reported or indicated activities concerning dynamic modelling. Thus it is reasonable to expect a good response to the next call for data, including results of dynamic modelling calculations.

Until this call, variables used to calculate critical loads for surface waters were not requested in a uniform way. Some countries provided these variables instead of similar soil variables, but this has not been done in a consistent way. Therefore, the CCE will create a separate format for the submission of data for surface water ecosystems at its next call for data.

It takes considerable time and effort to collect, check and process the new or changed parts of submitted data. For example, not all countries have yet managed to implement the EUNIS classification system; also errors in chemical units are easily made. It has been useful for both the CCE and the NFCs to carry out this call in order to be better prepared for the call that will yield data to be used in integrated assessment in 2004.

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Posch M, De Smet PAM, Hettelingh J-P, Downing RJ (eds) (2001) Modelling and mapping of critical thresholds in Europe: Status Report 2001, Coordination Center for Effects, RIVM Rep. 259101010, Bilthoven, Netherlands.

UBA (1996) Manual on methodologies and criteria for mapping critical levels/loads and geographical areas where they are exceeded. UNECE Convention on Long-range Transboundary Air Pollution. Federal Environmental Agency (Umweltbundesamt) Texte 71/96, Berlin.

3. Dynamic Modelling and Target Loads

Maximilian Posch, Jaap Slootweg and Jean-Paul Hettelingh

3.1 Introduction

European databases and maps of critical loads have been instrumental in formulating effects-based protocols to the 1979 Convention on Long-range Transboundary Air Pollution (LRTAP), such as the 1994 Protocol on Further Reduction of Sulphur Emissions and the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the "Gothenburg Protocol"). Critical loads are based on a steady-state concept; they are the constant depositions an ecosystem can tolerate in the long term, i.e. after it has equilibrated with these depositions. However, many ecosystems are not in equilibrium with present or projected depositions, since processes ("buffer mechanisms") are 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. Recognising this, the Executive Body of the Convention "... underlined the importance of ... dynamic modelling of recovery" (UNECE 1999), and subsequently dynamic modelling has become an important part of the medium-term work plan of the ICP on Modelling and Mapping.

Although dynamic models have been used for 15 to 20 years, the new challenge is to develop and apply dynamic model(s) on a European scale and to link them with the integrated assessment work under the LRTAP Convention, to support the review and potential revision of protocols. In order to pool the knowledge of the various ICPs under the Working Group on Effects, a Joint Expert Group on Dynamic Modelling has been established, which has convened several workshops to advance the use of dynamic models for the effects-related work under the Convention. One result of this collaborative work is the preparation of a "Dynamic Modelling Manual" which has been recently finalised (Posch et al. 2003), and which will also become part of the updated Mapping Manual (UBA 2003). This chapter provides a summary of the main issues related to dynamic modelling, with emphasis on the expected outputs.

3.2 Why dynamic modelling?

In the case of critical loads, i.e. in the steady-state situation, only two cases can be distinguished when comparing them to deposition: (1) the deposition is at or below critical load(s), i.e. does not exceed critical loads, and (2) the deposition is greater than critical load(s), i.e. there is critical load exceedance. In the first case there is no (apparent) risk of ecosystem damage, 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. Thus, a critical load serves as a warning 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". This means the chemical criterion (e.g. the Al:Bc ratio[1]) that links the critical load to (biological) effect(s) immediately attains a non-critical ("safe") value and that there is immediate biological recovery as well.

But the reaction of soils, especially in their solid phase, to changes in deposition is delayed by (finite) buffers, the most important 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 a steady state is reached. These finite buffers are not included in the critical load formulation, since they do not influence the steady state, but only the time required to reach it.

Therefore, dynamic models are needed to estimate the time required to attain a certain chemical state in response to deposition scenarios, e.g. the consequences of "gap closures" in emission reduction negotiations. In addition to the delay in chemical recovery, there is likely to be a further delay before the original biological state is reached, i.e. even if the chemical criterion is met (e.g. $Al/Bc < 1$), it will take time before biological recovery is achieved.

Figure 3-1 summarises the possible development of a chemical and biological variable in response to a "typical" temporal deposition pattern. Five stages can be distinguished:

Stage 1: Deposition was, and is, below the critical load (CL), and the chemical and biological variables do not violate their respective criteria. As long as deposition stays below the CL, this is the ideal situation.

Stage 2: Deposition is above the CL, but chemical and/or biological criteria are not violated because there is a time delay before this happens. No damage is likely to occur at this stage, therefore, despite exceedance of the CL. The time between the first exceedance of the CL and the first violation of the biological criterion, i.e. the first occurrence of actual damage, is termed the *Damage Delay Time* ($DDT=t_3-t_1$).

[1] In the Mapping Manual (and elsewhere) the Bc:Al ratio is used. However, this ratio becomes infinite when the Al concentration approaches zero. To avoid this inconvenience, its inverse, the Al:Bc ratio, is used here.

Stage 3: The deposition is above the CL and both the chemical and biological criteria are violated. Measures (e.g. emission reductions) have to be taken to avoid a further deterioration of the ecosystem status.

Stage 4: Deposition is below the CL, but the (chemical and) biological criteria are still violated and thus recovery has not yet occurred. The time between the first non-exceedance of the CL and the subsequent non-violation of both criteria is termed the *Recovery Delay Time* (RDT=$t_6 - t_4$).

Stage 5: Deposition is below the CL and both criteria are no longer violated. This stage is similar to Stage 1; and only at this stage can the ecosystem be considered to have recovered.

Stages 2 and 4 can be subdivided into two sub-stages each: Chemical delay times ($DDT_c=t_2 - t_1$ and $RDT_c = t_5 - t_4$; dark grey in Figure 3-1) and (additional) biological delay times ($DDT_b = t_3 - t_2$ and $RDT_b=t_6 - t_5$; light grey). Very often, due to the lack of operational biological response models, damage and recovery delay times mostly refer to chemical recovery alone; this is used as a surrogate for overall recovery.

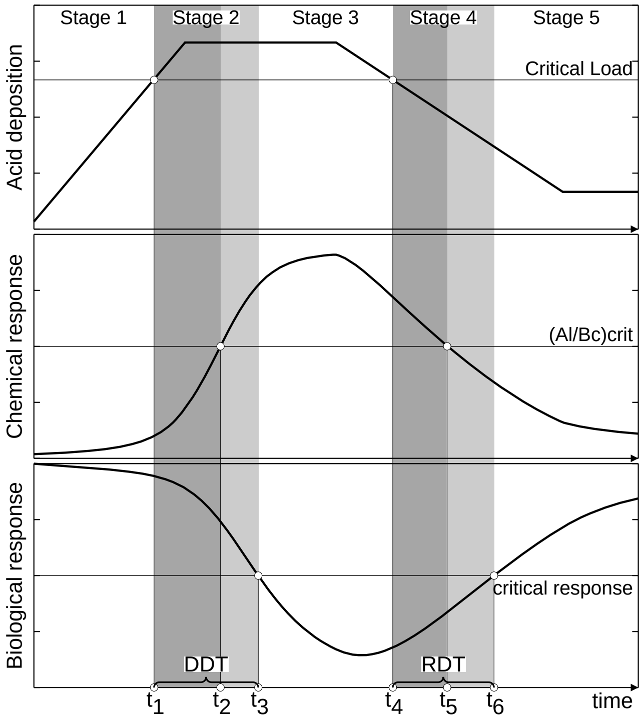


Figure 3-1. "Typical" past and future development of the acid deposition effects on a soil chemical variable (Al:Bc ratio) and the corresponding biological response in comparison to the critical values of those variables and the critical load derived from them. The delay between the (non-)exceedance of the critical load, the (non-)violation of the critical chemical criterion and the crossing of the critical biological response is indicated in grey shades, highlighting the Damage Delay Time (DDT) and the Recovery Delay Time (RDT) of the system.

While for simplicity's sake we refer here (and in the Dynamic Modelling Manual) to (non-calcareous) forest soils, most of the considerations hold also for aquatic ecosystems. More details on the specific issues and dynamic modelling activities concerning surface waters can be found in Jenkins et al. (2002).

As explained above, the non-exceedance of an ecosystem's critical load does not mean that it is immediately in a "good state"; non-exceedance is only a precondition for recovery. Thus not only ecosystems which are exceeded now – or e.g. after the implementation of the Gothenburg Protocol in 2010 – should be investigated by dynamic models, but also ecosystems which have been exceeded in the past.

Figure 3-2 shows those EMEP50 grid cells that will still be exceeded in 2010 (red squares) and those that have been exceeded in the past but are no longer exceeded (yellow squares). This shows that most of Europe, with the exception of southern and south-eastern countries, has ecosystems for which dynamic models are needed to assess their recovery. Note that the exceedance calculations have been carried out with depositions computed with EMEP's lagrangian dispersion model, which underestimates deposition to forested ecosystems. Thus the number of grid cells with some exceedances is likely to be larger than that shown in Figure 3-2.

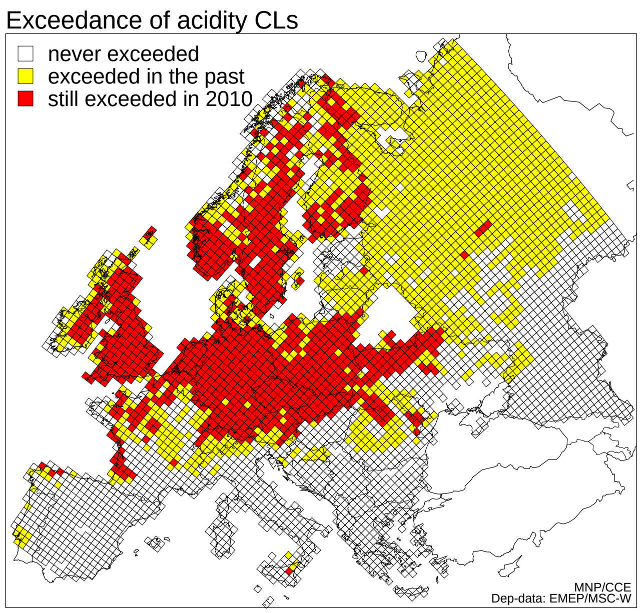


Figure 3-2. Areas in Europe where acidity critical loads within an EMEP50 grid cell will be exceeded in 2010 after implementation of the Gothenburg Protocol (red squares) and areas where they were exceeded in the past (yellow squares).

3.3 Target loads

The most straightforward use of dynamic models is for *scenario analysis*: the future chemical (and biological) status of an ecosystem is evaluated for a prescribed future deposition pattern. This is very simple for selected sites and requires only minor extra effort for large numbers of sites (regional databases). The results of a scenario analysis can then guide stakeholders in their quest for further deposition reductions. This relatively slow process could be accelerated and rationalised if the environmental target could be set and the models would calculate the (optimal) emission scenario. For integrated assessment models (IAMs) this requires optimisation procedures. And to link dynamic models to such an optimisation procedure requires either the full integration of the dynamic model into the IAM or dynamic model output – *response functions* – that can be used in optimisation.

Such response functions encapsulate an ecosystem's temporal behaviour to reach a certain (chemical) state in response to a broad range of (future) deposition patterns; or they characterise the amount of deposition reductions needed to obtain a certain state within a prescribed time.

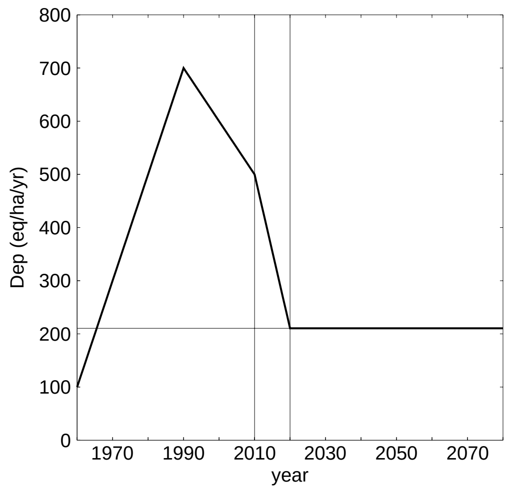
In the first case these response functions are pre-processed model runs for a large number of plausible future deposition patterns from which the results for every reasonable deposition scenario can be obtained by interpolation. An example is provided in Posch et al. (2003) which shows the isolines of years ("recovery isochrones") in which $Al/Bc < 1$ is attained for the first time for a given combination of percent deposition reduction and implementation year.

In the second case – i.e. answering the question "what is the maximum deposition allowed to obtain (and sustain) a desired chemical state (e.g. Al/Bc=1) in a prescribed year?" – so-called *target loads,* or – in the case of two pollutants – *target load functions*, have to be computed. Not least due to their similarity with critical load functions, target load functions are considered by the effects community as a promising way to link dynamic models with integrated assessment modelling.

The computation of target loads is not straightforward. After specifying the target year and the year of implementation of the (yet unknown) target load, the dynamic model has to be run iteratively until the deposition (= target load) is found that causes the soil to reach the desired chemical status in the specified target year. The following examples demonstrate the different cases that can arise when calculating target loads and what can happen when doing such calculation "blindly". For simplicity we consider a single pollutant (target load), but the conclusions hold for target load functions as well.

Figure 3-3 shows an example deposition history (left) and the resulting molar Al:Bc ratio (right) as simulated for three different soils, characterised by their cation exchange capacity (CEC=40, 60 and 80 meq kg^{-1}). In two cases the Al:Bc ratio at 'present' (year 2010) is above the critical value (=1), while for CEC=80 the soil remained below it during the past.

To investigate the future behaviour of these soils, we let the deposition drop to the critical load (which is independent of the CEC) during the "implementation decade" from 2010 to 2020 (marked by two vertical lines in Figure 3-3).



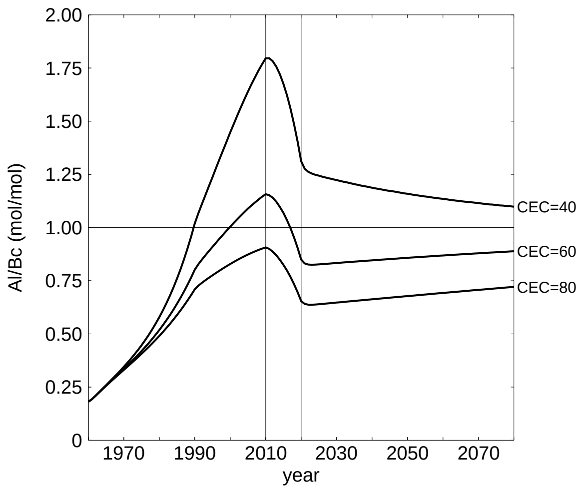


Figure 3-3. Temporal development of acidifying deposition (left) and corresponding molar Al:Bc ratio (right) for three soils varying in CEC. The vertical lines separate 50 years of "history", 10 years (2010–2020) of implementation of emission reductions, and the future. Also shown are the critical load and the critical value $(Al/Bc)_{crit}$=1 as horizontal lines. The deposition drops to the critical load within the implementation period and the Al:Bc ratios (slowly) approach the critical value.

Obviously, for *CEC*=80, the Al:Bc ratio stays below one, whereas for *CEC*=60 it drops below one within the first decade and then slowly rises again towards the critical value. For *CEC*=40, the Al:Bc ratio stays well above the critical value, approaching it asymptotically over time. In all three cases the approach to the critical value is very slow.

Next we look at target load calculations for these three soils. Figure 3-4 shows the results of target load calculations for 40 years, i.e. achieving $(Al/Bc)_{crit}$=1 in the year 2050. For *CEC*=40 meq kg^{-1} the target load is smaller than the critical load, as one would expect. For *CEC*=60 and 80, however, the computed target loads are higher than the critical load. As Figure 3-4 illustrates, this does not make sense: after reaching the critical limit, these two soils deteriorate, and the Al:Bc ratio increases further. Since target loads are supposed to protect an ecosystem also *after* the target year, we stipulate that **w*henever a calculated target load is higher than the critical load, it has to be set equal to the critical load.***

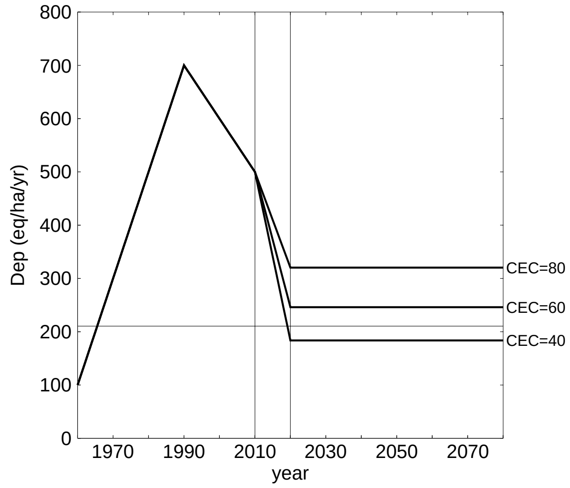
In light of the above considerations we define that ***a target load is the deposition for which a pre-defined chemical or biological status is reached in the target year and maintained (or improved) thereafter.***

The flow chart in Figure 3-5 allows one to determine whether, for a particular ecosystem, target load calculations are necessary or not. The first check at every site is whether the critical load (CL) is exceeded in the reference year (2010 in our case). If the answer is "yes"(as for the soils with CEC=40 and 60 in Figure 3-3), the next step is to run the dynamic model with the deposition equal to the critical load. If in the target year the chemical criterion is no longer violated (e.g. $Al/Bc \leq 1$), the target load equals the critical load (TL=CL).

If, after running the model with the critical load as deposition, the criterion is still violated in the target year, the model has to be run with zero deposition until the specified target year. "Zero" deposition means a deposition small enough as not to contribute to acidification (or eutrophication). In the case of nitrogen this would mean that N_{dep} is set equal to $CL_{min}(N)$, thus avoiding computational problems, e.g. negatively influencing forest growth in case of zero N deposition.

If, after running the model with "zero" deposition, the criterion is still violated in the target year, then the target cannot be met in that year. In such a case recovery can only be achieved in a later year. Otherwise, a target load exists and has to be calculated (see below); its value lies somewhere between zero and the critical load.

If the critical load is not (or no longer) exceeded in 2010 (as for the soil with *CEC*=80 in Figure 3-3), this does *not* mean that the risk of damage to the ecosystem is already averted – it only means that *eventually*, perhaps after a very long time, the chemical criterion will no longer be violated. Only if, in addition, the chemical criterion is not violated in 2010, no further emission reductions are required for that ecosystem. Also, if the model is run with the 2010 deposition until the target year and the criterion is no longer violated in that year, no further emission reductions are required. If the criterion is still violated in the target year, the procedure continues with running the model with "zero" deposition etc. (see Figure 3-5).



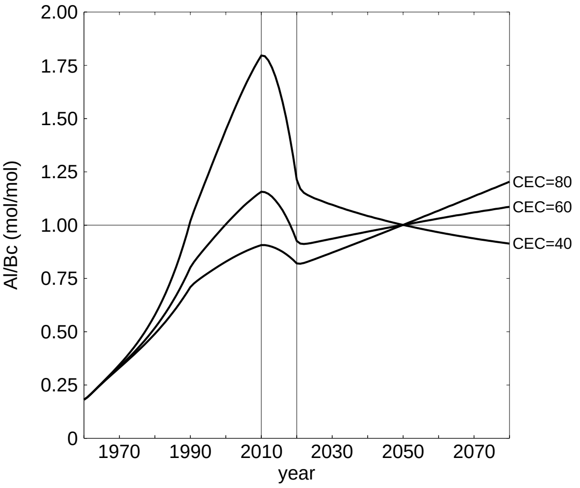


Figure 3-4. Target loads (with 2050 as target year) for three soils (left) and the resulting Al:Bc ratios (right). Note that for *CEC*=60 and 80 the target load is higher than the critical load, even when $(Al/Bc)_{crit}$<1 at present (for *CEC*=80). Clearly, in such cases target load calculations do not make sense.

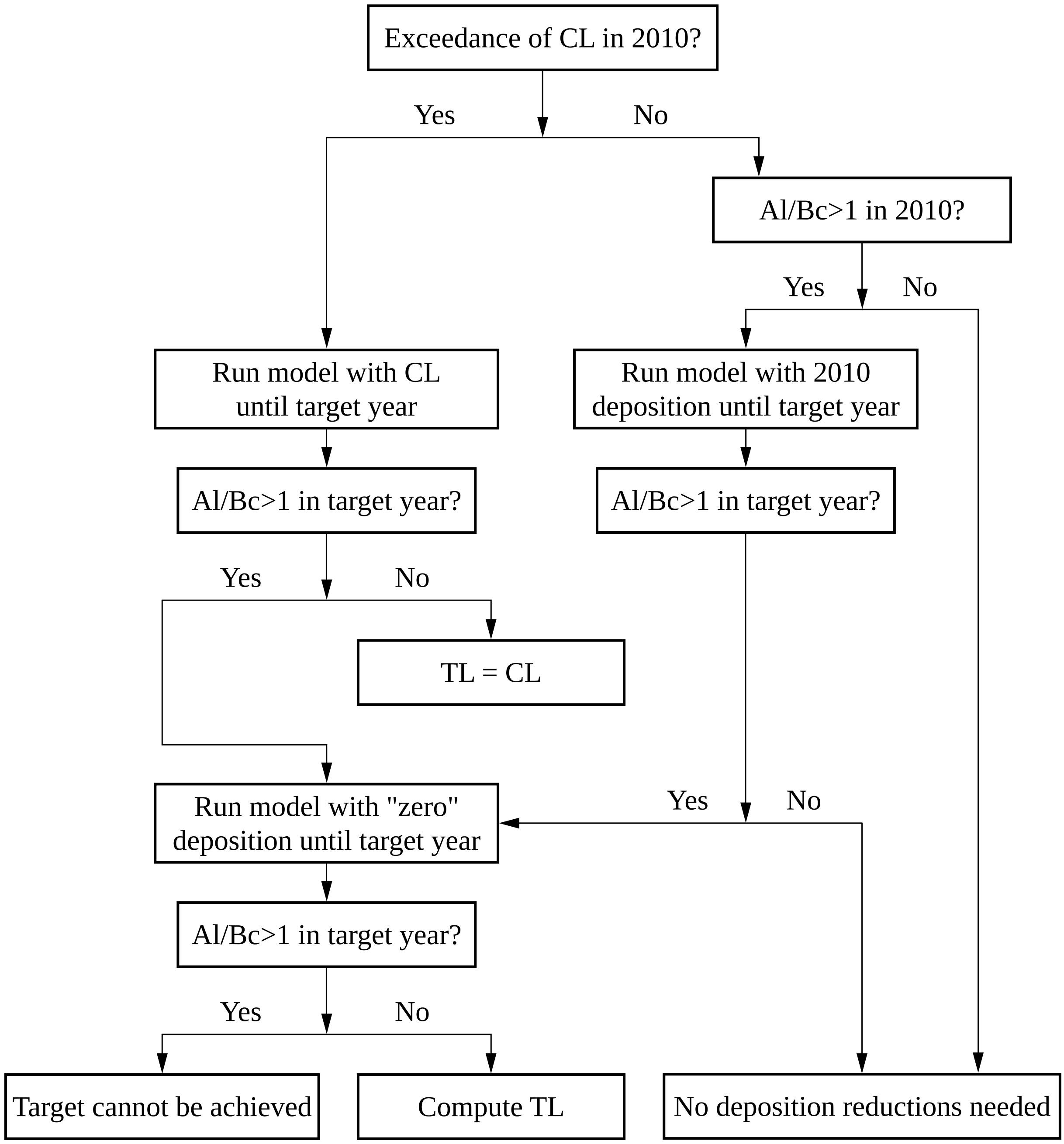


Figure 3-5. Flow chart of the procedure to calculate a target load, avoiding the pitfalls mentioned in the text (e.g. computing a target load that allows violation of the criterion *after* the target year).

In the implementation of the above procedure one could omit the step in which the model is run with the critical load as deposition (in case of exceedance in 2010) and immediately start with target load calculations (if a target load exists), and only afterwards check if this target load is greater than CL (and set it to CL) (see soil with *CEC*=60 in Figure 3-4). However, in view of the fact that TL calculations require iterative model runs, and also to avoid surprises due to round-off errors, it makes good sense to include that intermediate step.

An issue requiring attention in all target load calculations is the assumptions about finite nitrogen buffers. If it is assumed that (e.g.) a soil can immobilise N for the next 50 years to a larger degree than assumed in the critical load calculations, then target loads would always be higher than the critical load. This might cause confusion about the long-term N input, and demands careful attention.

When summarising target load calculations for ecosystems in a grid square (or region) it is important not only to report those sites for which target load functions have been derived, but all three cases (and their areas), i.e. the sites for which (i) no further deposition reductions are necessary, (ii) a target load has been calculated, and (iii) no target load exists (for the given target year). Note that in case (i), the 2010 deposition has necessarily to be below (or equal to) the critical load.

3.4 Activities under the ICP on Modelling and Mapping

Training of NFCs:
An important focus of the ICP M&M is to establish consistency between critical loads and dynamic modelling with the objective that the updated European critical load database will contain data which can be used to run both critical loads and dynamic model assessments. Minimum input data requirements for running the currently available dynamic models were established (see Posch et al. 2003). A Very Simple Dynamic (VSD) model has been developed for which these minimum input requirements are sufficient. The VSD model was tested (and compared with existing models such as SAFE) on data provided by the NFCs of Poland (Mill and Schlama 2002), Switzerland (Kurz and Posch 2002) and the United Kingdom (Evans 2003). A further study comparing VSD with the MAGIC model in Norway is underway. During the CCE workshop in Tartu (Estonia, 19-21 May 2003) a training session was held in which 43 representatives from 20 countries had the opportunity to familiarise themselves further with dynamic modelling inputs, outputs (target load functions) and model calibration.

Software development:
The VSD model developed at the CCE has been made available as an executable programme ("vsd.exe"). The Fortran source code ("vsd.for") has been also made available to allow potential users to check the implementation of the different (chemical) processes.

To enable users to call the VSD model from programs in other computer languages without having to re-code the Fortran code, a dynamic link library (DLL) 'vsd.dll' has been made available. The interface of this DLL consists of a method (function) which solves the basic equations for dynamic modelling for a single year. This enables the user to link the VSD model with a variety of software, even to combine it with their own models.

The DLL is also part of "VSD Studio", a stand-alone graphical user interface that can be downloaded and installed like most commercially available software. "VSD Studio" comes with an extensive Help facility, which explains not only how to use the model, but also the model equations and their derivation. A separate report to document the VSD model is in preparation (Posch and Reinds 2003).

To demonstrate the use of the VSD DLL, two third-generation language sample programmes have been created (in C and Delphi-Pascal). Also a MS-Excel and a MS-Access version have been made with a Visual Basic code calling the DLL. All but the Access version are only suited for calculations for a single site. The Access version has been designed for doing calculations for many sites.

The Excel version had been created with the 2002 call for data in mind. It combines the data of the call with exchange constants and N and S deposition time series to produce the temporal development of base saturation, molar Al:Bc ratio and pH. The same input can be also used to determine target load functions, an example of which is shown in Figure 3-6. This figure shows that in this example the future deposition according to the

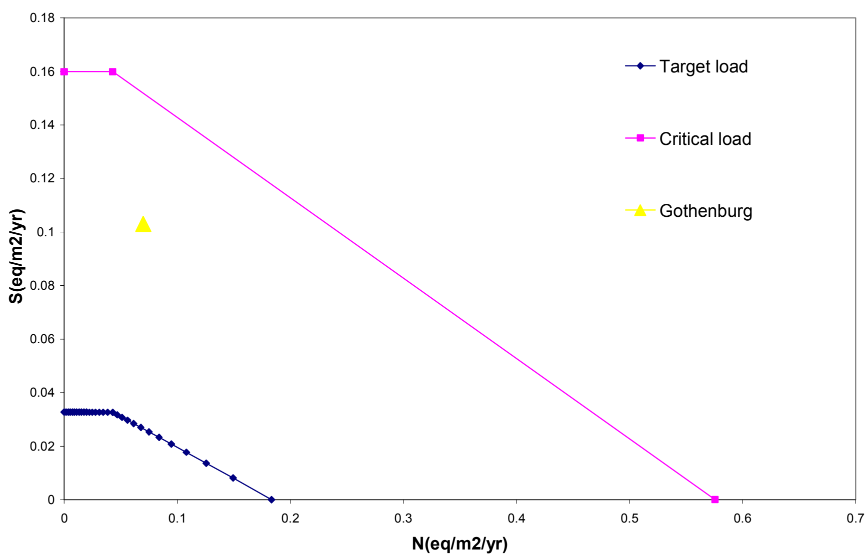


Figure 3-6. The Excel output of a target load function (blue curve), using the VSD-DLL. Also shown are the critical load function (red curve) and the S and N depositions according to the Gothenburg Protocol (yellow triangle).

Gothenburg Protocol does not cause exceedance of critical loads (red curve), i.e. there is ecosystem protection in the long run. However, to achieve recovery by the chosen target year (2040), much lower depositions of sulphur and nitrogen are required (blue curve).

All the software described above is available on the CCE website www.rivm.nl/cce. Updates of the software, following recommendations made at the May 2003 CCE training session and comments from various users will become available before the next call for data.

Acknowledgements

The graphical user interface for the VSD model (the "VSD Studio") has been developed by Gert Jan Reinds (Alterra, Wageningen, Netherlands). The discussions among scientists and modellers within (and outside) the Joint Expert Group (JEG) on Dynamic Modelling have helped shaping the ideas on dynamic modelling in support of the effects-based work under the LRTAP Convention.

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4. The European Background Database

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4.1 Introduction

A primary task of the Coordination Center for Effects (CCE) is to collect and collate national data on critical loads and – more recently – on dynamic modelling, and to provide European maps and other databases to the relevant bodies under the LRTAP Convention, especially for the purpose of integrated assessment. Ideally these data are based on national data submissions, provided by National Focal Centres (NFCs) in response to a CCE call for data. However, if a country does not contribute national data, the CCE uses for those areas values from a European background database (EU-DB) which is maintained and updated by the CCE.

An earlier version of the European background database has been described in an earlier Status Report (De Smet et al. 1997). That EU-DB in turn was to a large extent based on an older database used by De Vries et al. (1993, 1994) for a critical load study on a European scale. Over the last years new databases have become available (e.g. Reinds et al. 2001), and thus an update of the EU-DB was conducted. This update was also required in light of the fact that the database had to be extended to include variables needed for dynamic modelling.

As before, only forests (forest soils) are considered in the European background database. Whereas previously only coniferous and broadleaved (deciduous) forests had been distinguished, now separate variables are also estimated for mixed forests. When deriving data for the EU-DB which are not directly available in existing databases, the recommendations and transfer functions in the Dynamic Modelling Manual (Posch et al. 2003) and in the Mapping Manual (UBA 1996, 2003) are followed as much as possible.

This chapter describes the data sources and procedures used for deriving the variables in the European background database. EU-DB contains the same data as were requested of NFCs in the 2002 call for data.

4.2 Map overlays

Input data for critical load calculations and dynamic modelling include parameters describing climatic variables, base cation deposition and weathering, nutrient uptake, N transformations and cation exchange. These data vary as a function of location and receptor (the combination of forest type and soil type) as shown in Table 4-1.

Table 4-1. The influence of location, forest type or soil type on derived input data.

Input data	Location	Forest type	Soil type
Precipitation surplus	x	–	x
Base cation deposition	x	x	–
Base cation weathering	x	–	x
Nutrient uptake	x	x	–
Denitrification	–	–	x
N immobilisation	x	x	x
Cation exchange capacity	–	–	x
Base saturation	x	x	x

A map with the information required to derive the input data was constructed by overlaying the following base maps:

- A map with EMEP grid cells of 50×50 km^2, in which S and N deposition data are given (see Appendix A).
- A map with soil types at scale 1:1,000,000 for all European countries (Eurosoil 1999); except for Russia, Belarus, Ukraine and Moldova, for which the FAO 1:5,000,000 soil map (FAO 1981) was used.
- A map of forest types, distinguishing coniferous, broadleaved and mixed forests, taken from the PELCOM land cover map of Europe (Mücher et al. 2000). This map is derived from NOAA-AVHRR satellite images with a resolution of approximately 1×1 km^2.
- A map with climate zones for Europe, derived from EC/UNECE (1996).
- A global map of detailed elevation data (on a 30"×30" grid) from NOAA/NGDC (Hastings and Dunbar 1998).

Overlaying these maps, merging polygons within every EMEP50 grid cell differing only in altitude and discarding units smaller than 0.1 km^2 results in about 91,000 different forest-soil combinations.

The soil maps are composed of so-called soil associations, with each polygon on the map representing one association. Every association, in turn, consists of several soil typological units (soil types) that each covers a known percentage of the soil association. The soil typological units on the maps are classified into more than 200 soil types (Eurosoil 1999).

For each soil typological unit, available information on soil texture and drainage classes are used here to derive other input data. Texture classes are defined in Table 4-2.

Table 4-2. Soil texture classes and assigned clay content.

Texture class	Name	Definition	Clay content (%)
1	coarse	clay < 18% and sand ≥ 65%	6
2	medium	clay < 35% and sand ≥ 15%; but clay ≥ 18% if sand ≥ 65%	20
3	medium-fine	clay < 35% and sand < 15%	20
4	fine	35% ≤ clay < 60%	45
5	very fine	clay ≥ 60%	75
9	organic soils	soil types O (Histosols)	5

The drainage classes are derived from the dominant annual soil water regime (Eurosoil 1999, FAO 1981) and are given in Table 4-3:

Table 4-3. Drainage classes and assigned denitrification fraction.

Drainage class	Name	Denitrification fraction (f_{de})
EX	excessive	0
W	well	0.1
MW	moderately well	0.2
I	imperfect	0.4
TP	temporarily poor	0.7
P	poor	0.7
VP	very poor	0.8

Table 4-4 shows the distribution of forest over soil types in Europe for the 10 most common forest-soil types derived from the overlay of the soil- and forest map. Most forests are located on Podzols (FAO soil classes Po, Pl, Pg, Ph; together about 24% of the total forested area in Europe), especially in the Nordic countries, and to a lesser extent on Podzoluvisols (De, Dd, about 15%), Cambisols (Be, Bd and other Cambisols about 17%), Luvisols (Lo, Lg, Lc about 9%) and Lithosols (I, about 5%). Forest soils occur mainly on coarse (texture class 1, 37%) and medium textures (class 2, 48%). Forests on medium fine (class 3, 5.1%) and fine and very fine textures (classes 4 and 5, about 2.5%) are relatively rare. About 10% of European forests are located on peat soils (Od and Oe). Some inaccuracy in these estimates exists, because the soil map consists of soil associations. The map overlay thus gives a forested area for each association, not for each soil type. Forests have been assigned evenly to all soil types within the association, which in reality will not always be the case. However, an earlier study (De Vries et al. 1993) showed that when forest are assigned to "poor" soils in an association first (instead of evenly distributed), this hardly makes a difference in the forested area per soil type on a European scale.

Table 4-4. Area of the ten most common forest-soil combinations in Europe.

Soil type	Area (km^2)	Area (%)
Podzol (P)	734,118.1	24.0
Cambisol (B)	508,448.3	16.6
Podzoluvisol (D)	462,030.7	15.1
Histosol (O)	299,818.9	9.8
Luvisol (L)	262,818.0	8.6
Gleysol (G)	158,037.0	5.2
Lithosol (I)	149,199.4	4.9
Regosol (R)	121,967.5	4.0
Arenosol (Q)	87,806.8	2.9
Rendzina (E)	85,428.0	2.8

4.3 Input data for critical loads and dynamic modelling

Precipitation surplus and soil water content:
To compute the concentration and leaching of compounds in the soil, the annual water flux through the soil has to be known. It was derived from meteorological data available on a 0.5°×0.5° grid described by Leemans and Cramer (1991), who interpolated selected records of monthly meteorological data from 1678 European meteorological stations for the period 1931–1960.

Actual evapotranspiration was calculated according to a model used in the IMAGE global change model (Leemans and van den Born 1994) following the approach by Prentice et al. (1993). Potential evapotranspiration was computed from temperature, sunshine and latitude. Actual evapotranspiration was then 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 complete description of the model can be found in Annex 4 of Reinds et al. (2001). These computations also yield the annual average soil water content θ.

The available water content (AWC) was estimated as a function of soil type and texture class according to Batjes (1996) who provides texture class-dependent AWC values for FAO soil types based on an extensive literature review.

Base cation and chloride deposition:
The bulk deposition (wet deposition and a very small part of dry deposition) of base cations and chloride was derived from 89 to 96 (depending on the ion) EMEP/CCC monitoring stations in Europe (Hjellbrekke et al. 1998) averaged over the years 1991-1995. Grid values were derived by inverse-distance weighted interpolation of the values from the five nearest stations (Reinds et al. 2001). The seasalt-corrected wet base cation deposition is computed by assuming that all chloride originates from seasalt (see Appendix B).

Dry deposition was computed by multiplying the wet (bulk) deposition with a factor f_{dd}. Values of f_{dd} were derived from the ratio of Na in bulk deposition and throughfall by Ivens (1990) for 47 sites in Europe, resulting in median values of 0.6 for deciduous forests and 1.1 for coniferous forests. For mixed forests the average of these values (0.85) was used.

Base cation weathering:
Weathering of base cations was computed as a function of parent material class and texture class and corrected for temperature, as described in De Vries et al. (1993) and Reinds et al. (2001); see also the Mapping Manual (UBA 1996, 2003).

Weathering rate classes as function of texture class (see Table 4-2) and parent material are given in Table 4-5.

Table 4-5. Weathering rate classes as a function of texture and parent material classes.

Parent material	Texture class 1	2	3	4	5
Acidic	1	3	3	6	6
Intermediate	2	4	4	6	6
Basic	2	5	5	6	6
Organic	Oe: class 6. Other organic soils: Class 1.				

Parent material, in turn, was assigned from soil types according to Table 4-6.

The actual weathering rate (eq m^{-2} a^{-1}) for non-calcareous soils of thickness z (in m) is then computed according to:

$$BC_w = 0.05 \cdot (WRc - 0.5) \cdot z \cdot \exp\left(A/281 - A/(273+T)\right)$$

where WRc is the weathering rate class (Table 4-5), T (°C) is the average annual (soil) temperature and A=3600 K. This means that weathering rates (without temperature correction) vary from 0.025 eq m^{-2} a^{-1} (class 1) to 0.275 eq m^{-2} a^{-1} (class 6) for a soil of 1m depth.

Weathering rates of Ca, Mg, K and Na as fractions of BC_w were estimated as a function of clay and silt content (in %) for texture classes 2 to 5 (Van der Salm 1999) and as fixed fractions of total weathering for class 1 (De Vries 1994).

Table 4-6. Conversion between soil type and parent material class.

Parent material	FAO soil type
Organic	O, Od, Oe, Ox
Acidic	Ah, Ao, Ap, B, Ba, Bd, Be, Bf, Bh, Bm, Bx, D, Dd, De, Dg ,Gx, I, Id, Ie, Jd, P, Pf, Pg, Ph, Pl, Po, Pp, Q, Qa, Qc, Qh, Ql, Rd, Rx, U, Ud, Wd
Intermediate	A, Af, Ag, Bv, C, Cg, Ch, Cl, G, Gd, Ge, Gf, Gh, Gi, Gl, Gm, Gs, Gt, H, Hg, Hh, Hl, J, Je, Jm, Jt, L, La, Ld, Lf, Lg, Lh, Lo, Lp, Mo, R, Re, V, Vg, Vp, W, We
Basic	F, T, Th, Tm, To, Tv
Calcareous	Bc, Bg, Bk, Ck, E, Ec, Eh, Eo, Gc, Hc, Ic, Jc, K, Kh, Kk, Kl, Lc, Lk, Lv, Nc, Rc, S, Sg, Sm, So, Uk, Vc, X, Xh, Xk, Xl, Xy, Z, Zg, Zm, Zo

Acidic: Sand (stone), gravel, granite, quartzine, gneiss (schist, shale, greywacke, glacial till).
Intermediate: Gronodiorite, loess, fluvial and marine sediment (schist, shale, greywacke, glacial till).
Basic: Gabbro, basalt, dolomite, volcanic deposits.

Nutrient uptake:
Net uptake of base cations and nitrogen was computed by multiplying the estimated annual average growth of stems and branches with the element contents of base cations and N in these compartments (see also UBA 2003).

Forest growth was estimated according to a procedure described by Klap et al. (1997, see also Reinds et al. 2001), that calculates forest growth from yield tables as a function of forest type, forest age and climate zone (combination of climate and altitude). Since forest age is not known, the average growth of a rotation period from the yield tables was used.

Minimum and maximum contents of N, Ca, Mg and K in stems and branches of coniferous and deciduous forests are derived from Swedish data (Rosén 1990, see also Reinds et al. 2001 and UBA 2003). The average value of coniferous and deciduous forests at the same location was used for mixed forests.

Denitrification and N immobilisation:
The denitrification fraction, f_{de}, was computed as a function of drainage status, which is known for each soil type and is given in Table 4-3.

N immobilisation consists of a (time-independent) constant part, which is the same as used in critical load calculations (1 kg N ha^{-1} a^{-1}) and a time-dependent part, which is computed as a function of the prevailing C:N ratio of the topsoil. This C:N ratio is estimated from a transfer function (Klap et al. 2002) that can also be found in the Dynamic Modelling Manual (Posch et al. 2003). This transfer function computes the C:N ratio as a function of soil texture, forest type, climate variables and the N deposition of the relevant year (1995). The rate of change of the C:N ratio depends on the size of the C pool in the topsoil. This C pool for the top 20 cm is estimated from the organic C content (available for every soil type) and bulk density.

The bulk density ρ of the soil was computed from a transfer function using clay and organic carbon content, derived from data by Hoekstra and Poelman (1982) and Van Wallenburg (1988, see also Posch et al. 2003).

Clay content is an attribute to the soil map (see Table 4-2). The organic carbon content for each soil type was derived from a European database on forest soils (Vanmechelen et al. 1997).

Cation exchange capacity and base saturation:
Cation exchange capacity (CEC) was computed as a function of clay content, organic carbon content and soil pH according to a transfer function by Helling et al. (1964; see also Posch et al. 2003).

Base saturation for the reference year (1995) was estimated from a transfer function derived by Klap et al (2002, see also Posch et al. 2003). This transfer function computes the base saturation as a function of soil texture and forest type as well as the S, N and base cation deposition.

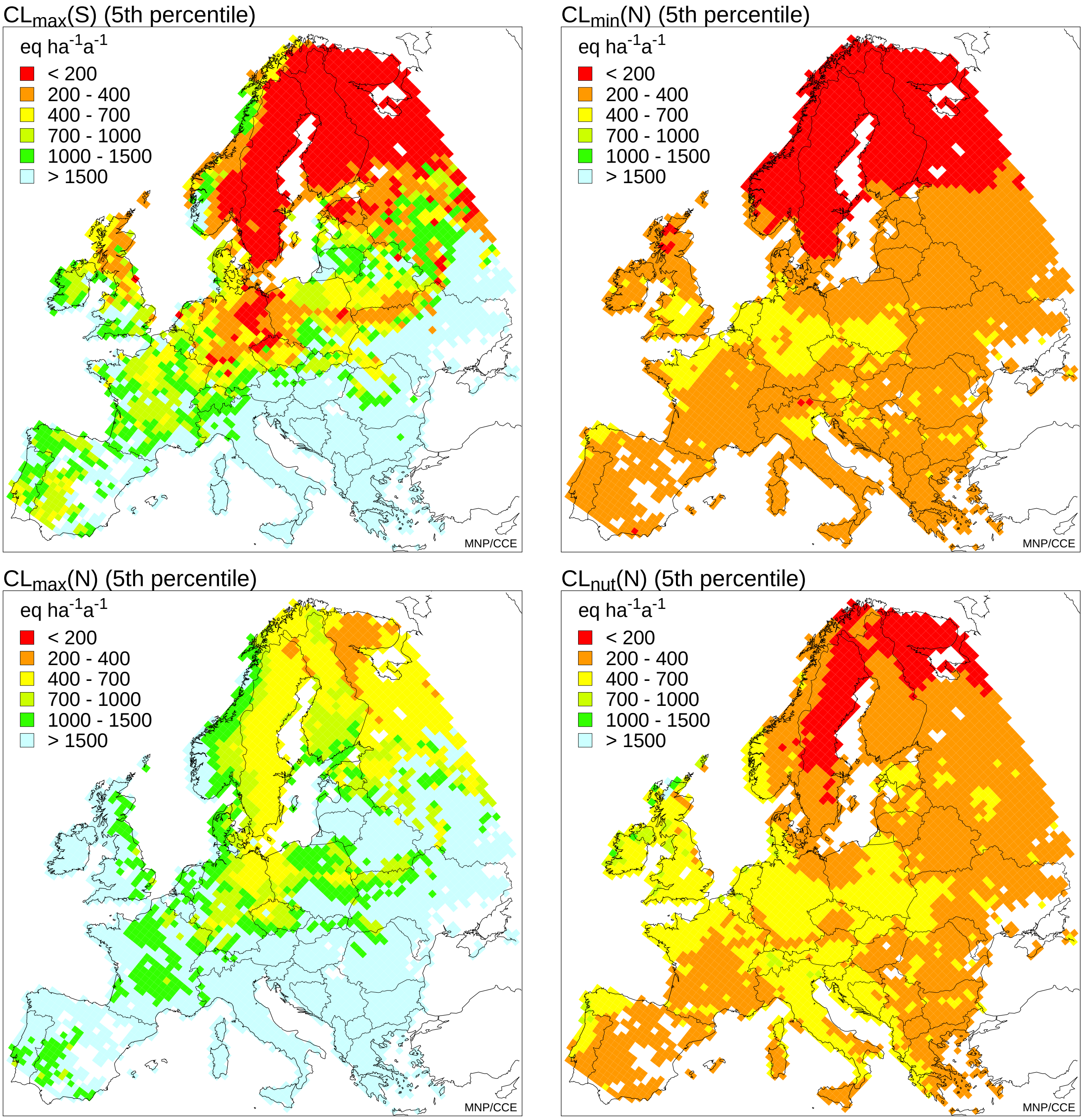


Figure 4-1. 5th percentile of the critical load quantities $CL_{max}(S)$, $CL_{min}(N)$, $CL_{max}(N)$ and $CL_{nut}(N)$ on the EMEP50 grid, computed from the European background database.

4.4 Results

The EU-DB obtained from the data(bases) and transfer functions described above can be used to compute critical loads of S and N acidity and nutrient N. Critical loads have been computed with the Simple Mass Balance (SMB) model, using a critical Bc:Al ratio of 1 mol mol^{-1} for all forests and soils, except for peat soils (Histosols), for which a critical molar Bc:H ratio is used (1 for conifers, 1/3 for deciduous forests and an average value of 2/3 for mixed forests). In Figure 4-1 the 5th percentiles of the three quantities (see Chapter 1) defining the critical load function for S and N acidity ($CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$), as well as the critical load of nutrient N ($CL_{nut}(N)$) are displayed on the EMEP50 grid.

The critical loads thus calculated from the European background database are used for those countries that do not provide national data to the CCE. The maps in Figure 4-1 can be compared with the maps in Chapter 1 to see the similarities and differences between critical loads computed from the EU-DB and national critical loads (see also below).

In addition to the variables needed to compute critical loads, additional variables are needed for dynamic modelling, especially those describing the finite buffers in the soils such as cation exchange capacity (CEC) and base saturation (E_{Bc}, Bc=Ca+Mg+K). Figure 4-2 displays the 5th percentile and the median of the available pool of base cations (for a 0.5 m soil) in each EMEP50 grid cell.

Figure 4-2 should be compared with Figure 1-8 in Chapter 1, which displays the same quantity computed from the data submitted by 10 NFCs (see also below). Results of dynamic modelling on a European scale using EU-DB are not yet available, but in the next section dynamic modelling variables are compared with those provided by NFCs.

4.5 Comparisons with national data

In this section we compare some variables and derived quantities from the European background database with those provided by NFCs from their national databases. In Figures 4-3 and 4-4 two of the most basic variables, mean annual temperature and precipitation are displayed. Comparing the two respective maps shows that there is quite a good agreement between the EU-DB and national climatic variables.

NFCs that have provided data for dynamic modelling cover only a relatively small fraction of the European area. In addition, maps can only show a very limited number of statistical descriptors (percentile, mean, etc.). Therefore, we compare in the following EU-DB data and NFC data with the aid of cumulative distribution functions (cdfs).

In Figure 4-5 the cdfs of the four critical load quantities ($CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ defining the acidity critical load function and the critical load of nutrient N, $CL_{nut}(N)$) from the EU-DB are compared with those for forest soils from the 19 countries that have submitted an update of their national data (see Chapter 2). Refer to Table 2-2 for a listing of 2-digit country codes.

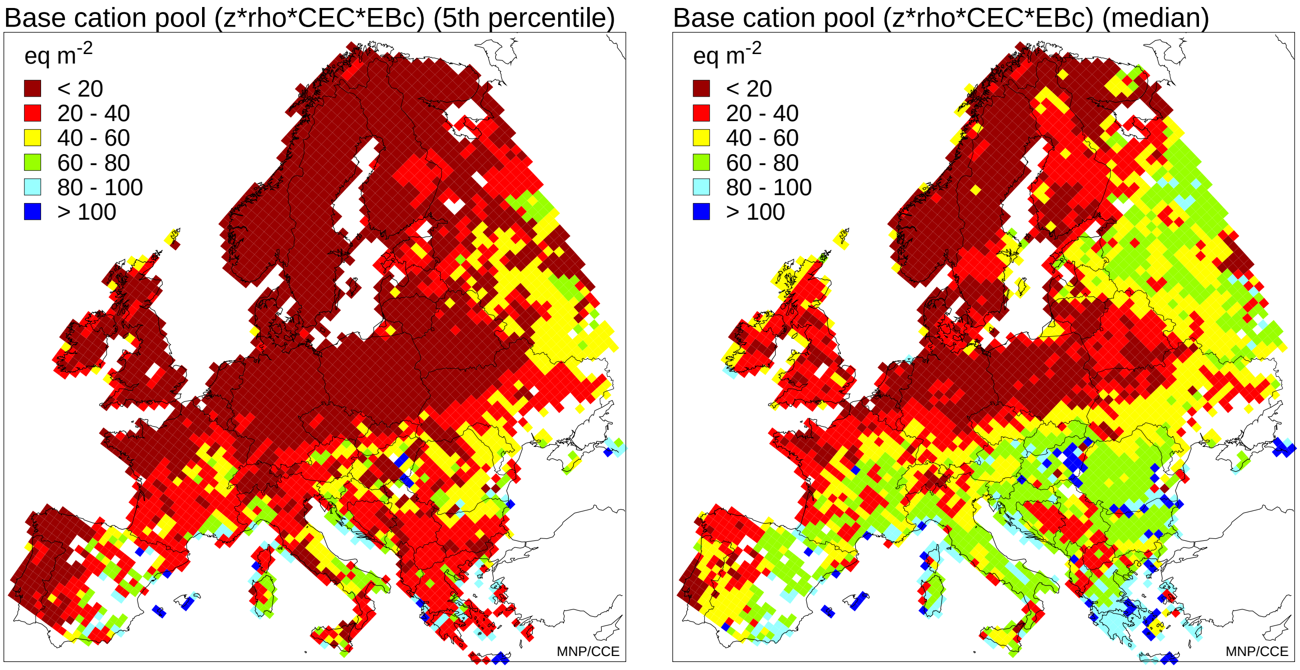
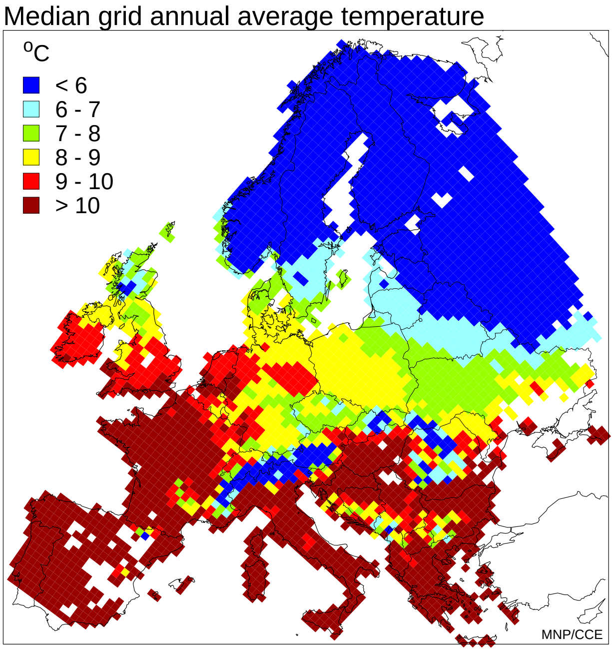


Figure 4-2. 5th and median value of the available pool of base cations ($z{\cdot}\rho{\cdot}CEC{\cdot}E_{Bc}$ with z=0.5m) in each EMEP50 grid cell for reference year (1995).



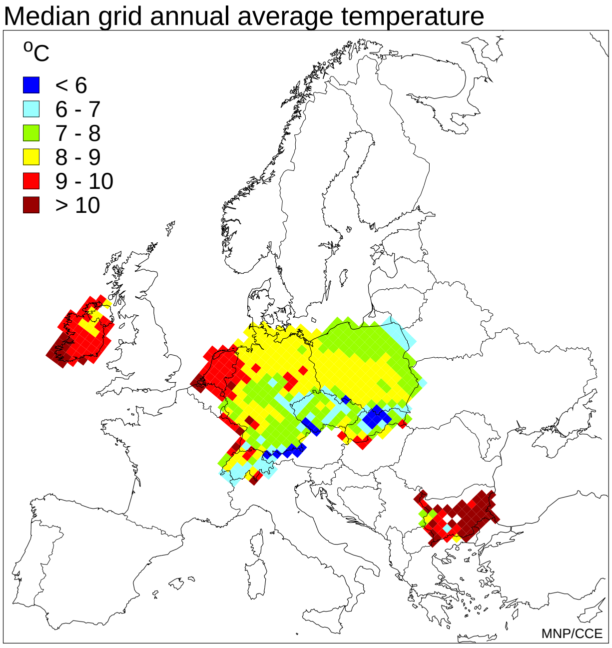
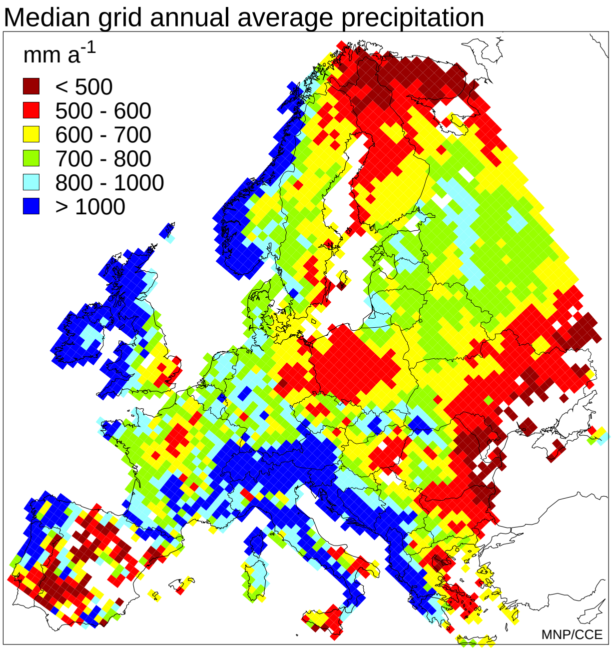


Figure 4-3. Median grid annual average temperature: (left) from the European background database (after Leemans and Cramer 1991), (right) from data provided by 10 NFCs.



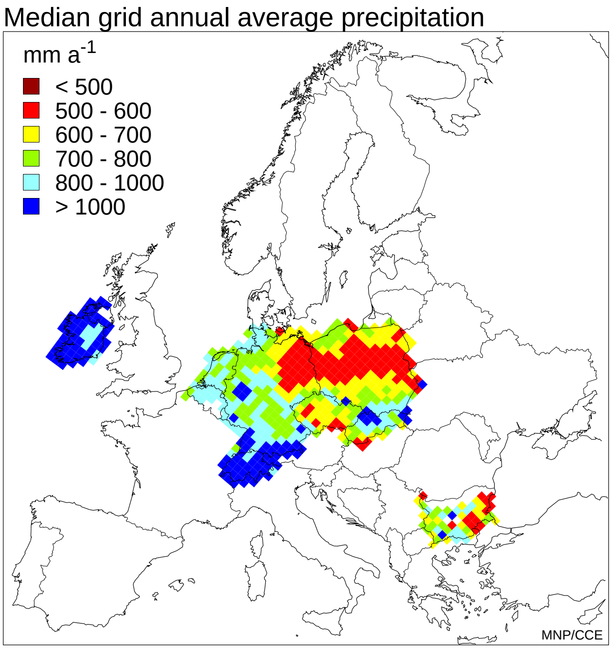


Figure 4-4. Median grid annual average precipitation: (left) from the European background database (after Leemans and Cramer 1991), (right) from data provided by 10 NFCs.

Figure 4-5 shows that in some of the countries, cdfs derived from national data and those from the European background database are remarkably similar, e.g. $CL_{max}(S)$ for the UK, Poland and Sweden, while others, such as $CL_{nut}(N)$ for the Netherlands, differ greatly. The numbers on the right side of each graph indicate the number of ecosystems reported for each class. It is also indicated if all values are outside the range displayed (e.g. by ">3000"). If ecosystem numbers are shown and a corresponding cdf is not visible, it means that it is underneath the cdf displayed.

When making these comparisons two points should be noted. Firstly, similar cdfs do not necessarily mean that similar values are found in the same location, since all spatial information is lost in cdfs. An idea of the spatial distribution of the critical loads is provided by the maps in Figure 4-1 and Chapter 1. Secondly, differences in the variables displayed are not only due to different data, but also due to different *assumptions* made. For example, in the EU-DB the long-term N immobilisation (which influences all N critical loads) is set to 1 kg N h^{-1} a^{-1} for all

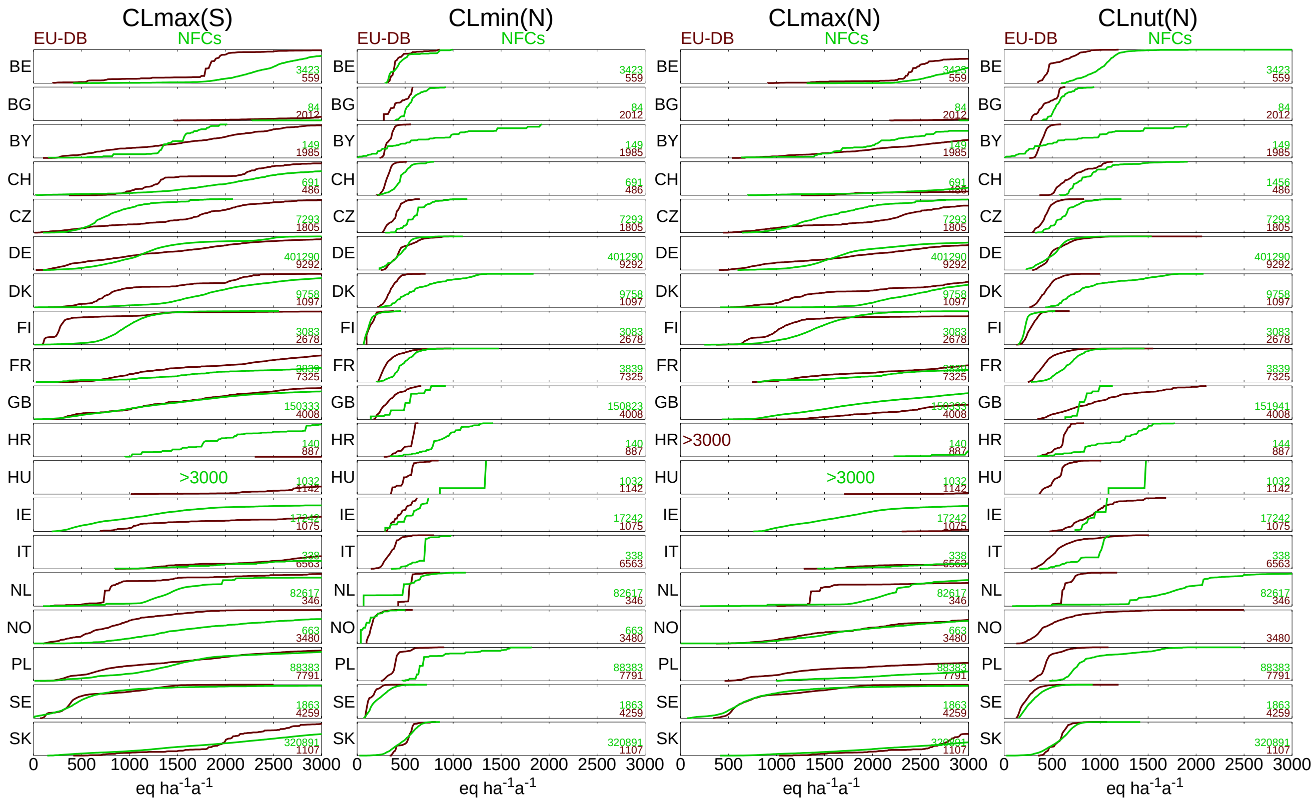


Figure 4-5. Comparison of the cdfs of the 4 critical load quantities from the European background database (EU-DB; brown) and national data for forest soils from 19 NFCs (green).

Europe, whereas several countries use much higher values (see Chapter 2).

In Figure 4-6 the cumulative distribution functions of four key variables needed for dynamic models are displayed for 10 countries. The cdfs of data from the European background database are compared with those from national data for the 10 countries that submitted dynamicmodelling variables in response to the 2002 call for data. The first two variables determine the base cation pool in the soil that counteracts acidification, and the other two the nitrogen pool in the topsoil which influences N accumulation. For some countries the cdfs are fairly similar, whereas in some there are remarkable differences. Some of the large differences in base saturation can be explained by the fact that in EU-DB only non-calcareous soils are considered, whereas, e.g., some of the Swiss soils are calcareous. The generally large differences in the carbon pool in the topsoil can probably be explained by the definition (depth) of the topsoil, and further clarifications are needed to derive quantities that are meaningful for dynamic modelling.

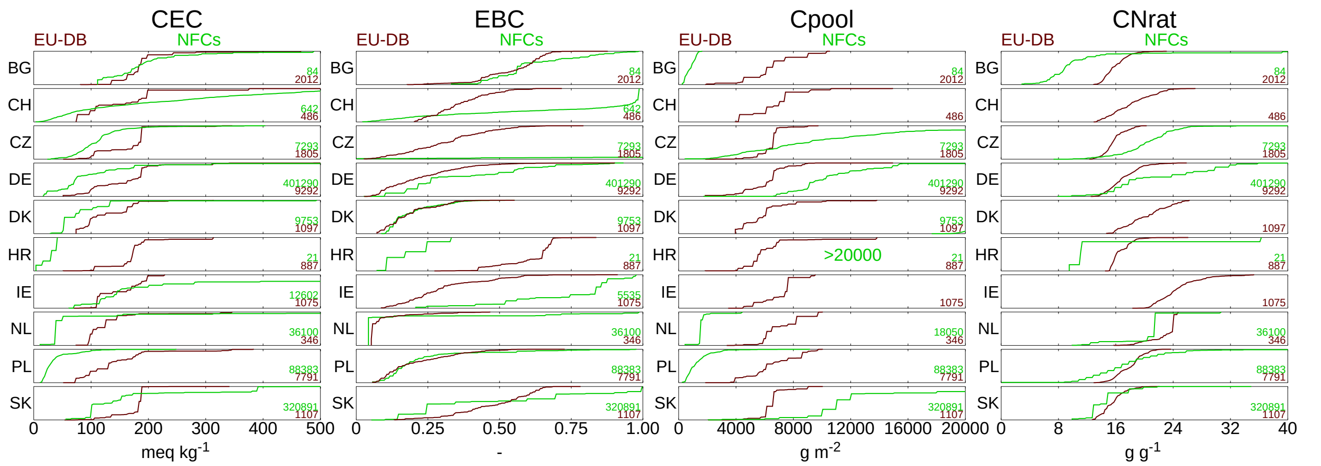


Figure 4-6. EU-DB and NFC cumulative distribution functions of cation exchange capacity (CEC), base saturation (EBC), carbon pool (Cpool) and C:N ratio in the topsoil (CNrat) for 10 countries.

4.6 Concluding remarks

A new European background database (EU-DB) has been created to fill gaps left when countries can not deliver data, as well as for other possible studies on a European scale. EU-DB not only includes the latest available data on a European scale for the calculation of critical loads, but also variables needed for running simple dynamic models. The database, however, is not a final product; it will be checked and updated whenever inconsistencies in the existing data are found or new data become available.

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Part II. National Focal Centre Reports

This part consists of reports on national input data on critical load and dynamic modelling calculations submitted to the Coordination Center for Effects (CCE) by National Focal Centres (NFCs).

A total of 24 countries collaborate with the ICP on Modelling and Mapping by submitting critical loads data and related information to the CCE. Following the call for data made in late 2002 (with the deadline of 31 March 2003), 18 countries (Belarus, Belgium, Bulgaria*, Croatia*, Czech Republic*, Denmark*, France, Germany*, Hungary, Ireland*, Italy, Netherlands*, Norway, Poland*, Slovakia*, Sweden, Switzerland* and United Kingdom) submitted updates of their critical load databases. Finland confirmed that it still considers the critical load data submitted earlier as valid. An analysis of the data submissions is provided in Chapter 2 of Part I. The Republic of Moldova, the Russian Federation and Spain did not submit a response. Other countries who did not submit (dynamic modelling) data indicated that they will provide results in response to the upcoming call for data, to be included in the European database in 2004. Their previously submitted databases were retained unchanged.

NFCs were asked to focus in their national contributions on describing their critical load and dynamic modelling databases and documenting the methods used, and especially include a justification if data or models are applied which are not given in the Mapping Manual or the Dynamic Modelling Manual.

The NFC reports received were edited for clarity and format, but have not been further reviewed.

* Denotes that the country also submitted dynamic modelling data.

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Status of critical loads data

The Austrian NFC supports the extension of the data set for the calculation of critical loads of acidification and eutrophication to include parameters which enable the application of (simple) dynamic models.

However, due to limited resources it was not possible to gather all the necessary data in time. In particular, data sets for soil density and water content at field capacity have to be generated. Therefore, the Austrian NFC plans to initiate an internal project at the Federal Environment Agency to be able to submit a data set for dynamic modelling at the end of 2003, provided that the required data will be available.

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Calculation methods

Critical loads have been calculated in 2002 for the following EMEP grid cells: (113,80), (114,81), (114,79), (113,79), and (112,80). The steady-state mass balance (SSMB) method was used, along with an additional algorithm developed by Prof. Bashkin of the National Focal Centre of Russia.

Data sources

- Base cation deposition (BC_{dep}) is taken from the data of the nearest monitoring station of all considered territory regardless of ecosystem type.
- The permissible concentration of mineral nitrogen in the surface waters ($N_{le(crit)}$) was evaluated in dependence of the maximum permissible concentration for fishing aims (1g l^{-1} N).
- The values of $Al_{le(crit)}$ and $H_{le(crit)}$ were defined according to the Mapping Manual, depending on the pH value of the active soil layer.
- The rest of the values were defined during the calculation. Data from landscape-geochemical explorations in Belarus (1974–2000) have been used in the work.

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Critical loads data and methods

Flanders:
Critical loads of acidity and nutrient nitrogen for ecosystems in Flanders have been calculated using the SMB method as described in the Mapping Manual (UBA 1996). Compared to the results presented in the CCE Status Report 2001, critical loads for forests have been revised (Langouche et al. 2002). For grassland and heather, new critical loads have been calculated (Meykens et al. 2001) and are reported here.

Wallonia:
No new data have been submitted. The data submitted in 2001 have been used to produce the European critical load maps.

Revision of critical loads for forests

The following changes have been made to the critical load calculation for forests in Flanders:

- More receptors for which critical load calculations have been made (from 652 to 1425).
- Adaptation of the criterion for acidification: critical molar ratio of Al/Bc=1 for all soil types. Critical nitrogen leaching was set to 100 eq ha^{-1} a^{-1}.
- Revision of nitrogen- and base cation-related parameters.
- Precipitation surpluses calculated based on data from more weather stations.

Table BE-1 presents the calculation results and describes the methods and data sources used. Table BE-2 shows the links between the ecosystem classification used and the EUNIS code. There is no one-to-one relationship.

Table BE–1. Summary of critical load values for Flanders (Belgium) and justification for their use.

Parameter	Ecosystem code	Min value	Max value	Methods used / Data source	Justification
$CL_{max}(S)$ (eq ha^{-1} a^{-1})	BF	407	13,359	$CL_{max}(S) = BC_{dep} + BC_w - Bc_u - ANC_{le(crit)}$	Mapping Manual (UBA 1996).
	CF	1230	13,904		
	WH	1655	1811		
	DH	1645	2227		
	ACG	1317	3381		
	NAG	605	2156		
	CG	1893	1894		
	AGG	589	2052		
$CL_{min}(N)$ (eq ha^{-1} a^{-1})	BF	553	868	$CL_{min}(N) = N_i + N_u$	Mapping Manual.
	CF	537	997		
	WH	357	500		
	DH	357	500		
	ACG	143	286		
	NAG	786	989		
	CG	786	786		
	AGG	786	929		
$CL_{max}(N)$ (eq ha^{-1} a^{-1})	BF	1311	44,788	$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) / (1 - f_{de})$	Mapping Manual.
	CF	2281	28,344		
	WH	2339	4010		
	DH	2185	4955		
	ACG	2026	7942		
	NAG	1754	7942		
	CG	4572	7098		
	AGG	1453	7587		
$CL_{nut}(N)$ (eq ha^{-1} a^{-1})	BF	664	1386	$CL_{nut}(N) = CL_{min}(N) + N_{le(acc)} / (1 - f_{de})$	Mapping Manual.
	CF	648	1310		
	WH	558	994		
	DH	530	1049		
	ACG	511	1231		
	NAG	1175	3259		
	CG	1304	2339		
	AGG	1175	3269		
BC_{dep} (eq ha^{-1} a^{-1})	BF	920	1200	Monitoring data forest deposition 1994–1999.	
	CF	950	1200		
	WH	1275	1365	Monitoring data from open field (1998-1999) × enhanced deposition factor of 1.5.	Enhancement factor based on Bobbink et al. 1992.
	DH	990	1365		
	ACG	660	910	Monitoring data from open field (1998-1999).	
	NAG	660	910		
	CG	740	740		
	AGG	660	910		
Bc_u (eq ha^{-1} a^{-1})	BF	260	930	Data from Belgian and Dutch literature.	
	CF	180	610		
	WH	0	0	Based on Posch et al. 1999.	
	DH	0	0		
	ACG	0	0		
	NAG	222	222		
	CG	222	222		
	AGG	222	222		
BC_w (eq ha^{-1} a^{-1})	BF	125	5000		Mapping Manual.
	CF	125	5000		
	WH	125	125		
	DH	125	375		
	ACG	125	1375		
	NAG	125	1375		
	CG	1375	1375		
	AGG	125	1375		

Table BE–1 (continued). Summary of critical load values for Flanders (Belgium) and justification for their use.

Parameter	Ecosystem code	Min value	Max value	Methods used / Data source	Justification
Q (mm)	BF	148	378		Mapping Manual.
	CF	106	264		
	WH	90	157		
	DH	90	175		
	ACG	134	298		
	NAG	117	298		
	CG	165	297		
	AGG	134	298		
K_{gibb} (m^6 eq^{-2})	BF	100	1000	Minimum value applied to organic soils; maximum value applied to mineral soils.	Mapping Manual.
	CF	1000	1000		
	WH	950	3000		
	DH	100	3000		
	ACG	950	3000		
	NAG	100	3000		
	CG	100	3000		
	AGG	100	3000		
$ANC_{le(crit)}$ (eq ha^{-1} a^{-1})	BF	262	7699	Criterion Al/Bc=1	Mapping Manual.
	CF	765	8014		
	WH	255	381	pH criterion of 4.3	
	DH	72	742		
	ACG	322	1266		
	NAG	32	131	pH criterion of 4.7	
	CG	0	1	pH criterion of 6.7	
	AGG	17	39	pH criterion of 5.0	
N_i (eq ha^{-1} a^{-1})	BF	143	143	Fixed value, average of broadleaved trees.	Belgian literature.
	CF	357	257		
	WH	71	214	Function of soil type.	Hall et al. 1998.
	DH	71	214		
	ACG	71	214		
	NAG	71	214		
	CG	71	71		
	AGG	71	214		
N_u (eq ha^{-1} a^{-1})	BF	410	725	Function of tree species.	Belgian and Dutch literature.
	CF	180	640		
	WH	286	286	Fixed value based on UK literature.	Posch et al. 1999.
	DH	286	286		
	ACG	71	71		
	NAG	714	714		
	CG	714	714		
	AGG	714	714		
f_{de}	BF	0.1	0.8	Function of soil texture and drainage.	Mapping Manual.
	CF	0.1	0.7		
	WH	0.1	0.5		
	DH	0.1	0.5		
	ACG	0.1	0.7		
	NAG	0.1	0.8		
	CG	0.5	0.7		
	AGG	0.1	0.8		
$N_{le(acc)}$ (eq ha^{-1} a^{-1})	BF	100	100	Constant N mass flux.	De Vries 1996.
	CF	100	100		
	WH	141	247	Constant N concentration of 2.2 mg l^{-1} × precipitation surplus.	
	DH	141	274		
	ACG	211	468		
	NAG	183	468		
	CG	259	468		
	AGG	211	468		

Table BE-2. Ecosystem type relationship with EUNIS vegetation codes.

Ecosystem type	Ecosystem code	Corresponding EUNIS codes
Broad-leaved forest, mixed forest with more than 50% broad-leaved trees	BF	G5.2 G5.5 G4.F G1.C
Coniferous forest, mixed forest with more than 50% coniferous trees	CF	G5.4 G5.5 G4.F G3.F
Inland wet heathland	WH	F4.11 D1.11
Lowland dry heathland	DH	F4.21 F4.22 F4.13 F1.73 F5.31
Acidic grassland	ACG	E1.71 E1.9 B1.47 E3.51
Neutral-acidic grassland	NAG	E3.42 E3.14 E3.41 E3.45 E5.41 E2.1 E2.22
Calcareous grassland	CG	B1.41 E1.26 E2.22
Agricultural grassland	AGG	E2.6 E2.61 E2.62 A2.6 E2.13

Maximum critical load of sulphur:

- New values for base cation deposition and uptake.
- New criterion for $ANC_{le(crit)}$.

Minimum critical load of nitrogen:

- New values for nitrogen immobilisation, dependent on broadleaved/coniferous forest type.
- New values for nitrogen uptake.

Maximum critical load of nitrogen:

- Inclusion of the denitrification into the equation.
- Denitrification is linearly dependent on deposition.

Critical loads of nutrient nitrogen:

- New criterion based on constant low nitrate flux (previously based on constant low concentration).

Mapping procedure

Critical loads were calculated for 1786 locations in Flanders for forest, grassland or heath. Critical loads were assigned to the entire ecosystem area using the technique of Thiessen polygons with GIS software. All ecosystems falling within the Thiessen polygon around a critical load location were assigned the same critical load value. This mapping procedure was repeated for eight ecosystem types (Mensink et al. 2001).

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Calculation methods

The mapping of critical loads of acidity, sulphur and nitrogen is based on 208 coniferous and deciduous forest soil receptor points. The results have been processed in the EMEP 50×50 km^2 grid.

Critical loads of nitrogen as a nutrient, maximum values for the critical loads of sulphur and acidifying nitrogen, and minimum critical loads of nitrogen have been calculated according to the Mapping Manual (UBA 1996) using the steady-state mass balance method as follows:

1. Critical loads of acidity for forest soils:

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

where:

$CL(A)$ = critical load of acidity, eq ha^{-1} a^{-1}
BC_w = weathering of base cations, eq ha^{-1} a^{-1}
Q = annual runoff of water under root zone, m^3 ha^{-1} a^{-1}
$[H]_{crit}$ = critical concentration of protons (= 0.09 eq m^{-3} which corresponds to pH 4.0; Hettelingh and De Vries 1992)
Al: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} a^{-1}
BC_u = net growth uptake of base cations, eq ha^{-1} a^{-1}

2. 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} a^{-1}
N_i = nitrogen immobilisation, eq ha^{-1} a^{-1}

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

3. 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} a^{-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)

4. Critical ANC leaching:

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

$Al_{le(crit)} = \text{Al:BC}\ (BC_{dep} + BC_w - BC_u)$

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

where:

$ANC_{le(crit)}$ = critical leaching of alkalinity, eq $ha^{-1}\ a^{-1}$

$Al_{le(crit)}$ = critical Al leaching, eq $ha^{-1}\ a^{-1}$

$H_{le(crit)}$ = critical H leaching, eq $ha^{-1}\ a^{-1}$

Dynamic modelling

The Very Simple Dynamic (VSD) model has been used for dynamic modelling. This model consists of a set of mass balance equations describing the soil input and output relationships and fluxes, and soil properties. Many of the input data needed to run the VSD model are also required for the steady state mass balance model for calculating critical loads described in detail in the previous status reports on calculating and mapping critical loads of acidity, sulphur and nitrogen (Ignatova et al. 1999, 2001). The most important additional soil data parameters required are the carbon content in the soil, C:N ratio, soil bulk density, clay and sand content, cation exchange capacity and base saturation, as well as soil pH.

Mean annual temperature, annual bulk precipitation and the average altitude above sea level has been derived for each grid cell. Data for volumetric water content at field capacity are not available for all EMEP grids.

Data sources

A) National monitoring data:

- Critical loads have been calculated for all major tree species using a soil database with organic matter content (%), clay content for the fraction 0.01 mm in the soil (%), soil bulk density, cation exchange capacity (CEC), base saturation, C:N ratio and soil pH, in grid cells of 16×16 km^2. A total of 208 values from measured forest soil profiles have been included in the calculations. Data on base saturation from 1996 have been obtained by the Ganev method (Ganev 1990), whereas a value of 0.1 M $BaCl_2$ has been used in 2001 (ISO 11260 and ISO 14254).
- Runoff of water under root zone has been measured in grid cells of 10×10 km^2 for the entire country (Kehayov 1986).
- Data from a network of 12 atmospheric deposition measurement stations have been used for base cation deposition.
- Nitrogen and base cations net uptake rates were obtained by multiplying the element contents of the stems (N, Ca, K, Mg and Na) with annual harvesting rates (Ignatova et al. 1997).
- Data on biomass removal for forests have been derived from the National Forest Survey Agency. The content of base cations and nitrogen in the biomass has been taken from the literature for different harvested parts of the plants (stem and bark of forest trees) (Jorova 1992, Ignatova 2001, De Vries and Bakker 1998, De Vries et al. 2001)

B) National synthetic maps:

- Soil type information on the FAO soil map of Bulgaria.
- Geological map of Bulgaria 1:500,000.
- Vegetation map of Bulgaria 1:500,000.
- Mean annual temperature map 1:500,000.
- Mean annual precipitation map 1:500,000.

C) Calculation data

- In the absence of specific data on the production of base cations through mineral weathering for most study regions, weathering rates have been calculated according to the dominant parent material obtained from the lithology map of Bulgaria and the texture class taken from the FAO soil map for Europe, according to the clay content of Bulgarian forest soils (UBA 1996).
- The gibbsite equilibrium constant, K_{gibb}, for the Al:H relationship ($m^6\ eq^{-2}$) has been estimated according to the soil organic matter and soil type (UBA 1996).
- The resulting database contains separate records of critical load data for deciduous and coniferous forests for each EMEP50 grid cell that contains Bulgarian territory.

Results and comments

All data necessary to run the VSD model and to evaluate critical loads of acidity, sulphur and nitrogen (36 parameters in total) have been prepared in Excel database files and mapped for the EMEP 50×50 km^2 grid using ArcView software.

Values for each parameter and the resulting critical loads are stored for each forest type (coniferous and deciduous forests) in separate records for each EMEP 50×50 km^2 grid cell when the forest is a mixture of both tree types, in accordance with the area fractions of the tree species.

The frequency distribution of the values for both deciduous and coniferous is shown in Table BG-1. All values of critical loads of acidity, as well as maximum critical loads of sulphur and nitrogen are much greater than 2000 eq $ha^{-1}\ a^{-1}$. Compared to previous calculations

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} a^{-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	16.36	0
500–1000	0	0	0	0	67.27	100	0	0	83.64	100
1000–2000	0	0	0	0	0	0	0	0	0	0
> 2000	100	100	100	100	0	0	100	100	0	0

(Ignatova et al. 2001), there is an increase in the percentage of $CL_{nut}(N)$ values between 500 and 1000 eq ha^{-1} a^{-1} for deciduous receptors (83.64%), and a decrease in values between 200 and 500 eq ha^{-1} a^{-1} (16.36%).

Comparing the average values of the $CL_{max}(N)$, $CL_{min}(N)$, $CL_{max}(S)$, $CL_{min}(S)$ and $CL_{nut}(N)$ for coniferous and deciduous tree types throughout Bulgaria, it becomes evident that the differences in the maximum critical loads of sulphur and nitrogen are insignificant. In contrast, minimum critical loads of nutrient nitrogen are about 30 percent lower for the deciduous species than for coniferous ones.

Calculated values for $CL_{max}(S)$ vary between 2303 and 10,652 eq ha^{-1} a^{-1} for coniferous, and between 2269 and 9946 eq ha^{-1} a^{-1} for deciduous forests (Figure BG-1). On the contrary, critical load values for nutrient nitrogen are lower, ranging from 581 to 941 eq ha^{-1} a^{-1} for coniferous, and 398 to 776 eq ha^{-1} a^{-1} for deciduous forests. The lowest critical loads are calculated for $CL_{min}(N)$ (between 573 and 926 eq ha^{-1} a^{-1} for coniferous forests, and from 394 to 768 eq ha^{-1} a^{-1} for deciduous; Figure BG-2).

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

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

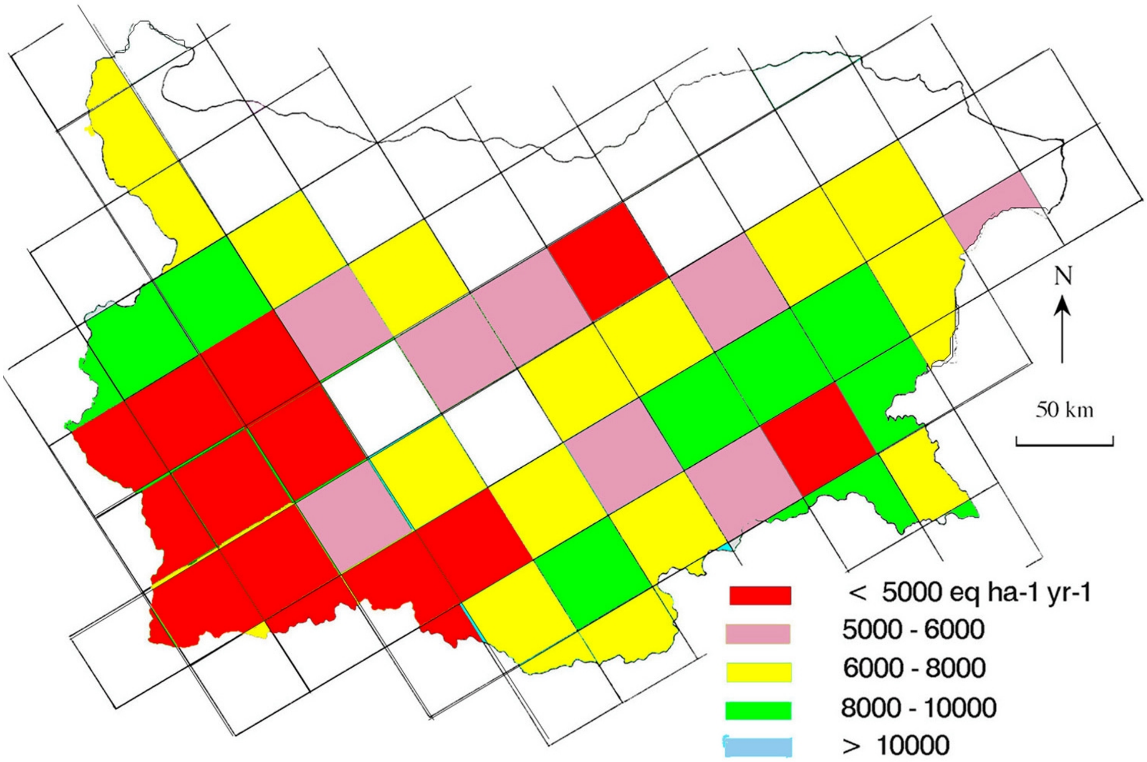
In almost all grid cells, individual critical loads for coniferous ecosystems are higher than those for deciduous. The mean value of maximum critical loads of sulphur for coniferous ecosystems has been calculated as 7096 eq ha^{-1} a^{-1}, whereas average values for deciduous forests in the same grid cells are only 6241 eq ha^{-1} a^{-1}.

For the minimum critical loads of nitrogen as well as the critical loads of nutrient nitrogen the variability of computed individual data is much smaller, which is reflected in the average values (789 eq ha^{-1} a^{-1} for coniferous ecosystems for minimum critical loads of nitrogen with 533 eq ha^{-1} a^{-1} for deciduous ones, and 797 eq ha^{-1} a^{-1} for coniferous for nutrient nitrogen against 549 eq ha^{-1} a^{-1} for deciduous forests).

Regarding cation exchange capacity and soil base saturation, it should be stressed that both parameters are higher for deciduous forests than for the coniferous ones. The average value of cation exchange capacity for coniferous forest ecosystems is about 173 meq kg^{-1} with a base saturation of 57%, whereas for deciduous forests this parameter has an average value of 192 meq kg^{-1} for CEC and 63% for base saturation.

Soil acidity is almost the same for the both types of receptors. The average value of the soil pH for coniferous forests is about 5.1, with a minimum pH of 4.6 and a maximum of 6.5, vs. an average value for deciduous forests of 5.2 (ranging from 4.3 and 7.4).

The coniferous forests studied are situated at an average altitude of 727 m a.s.l. (from 180 m to 1200 m) and the deciduous ones at 525 m (from 142 to 1200 m). The C:N ratio is higher for coniferous forests (average 13.1, with a minimum of 5.0 and maximum of 74) than for deciduous ones (an average of 11, ranging from 2.9 and 74).



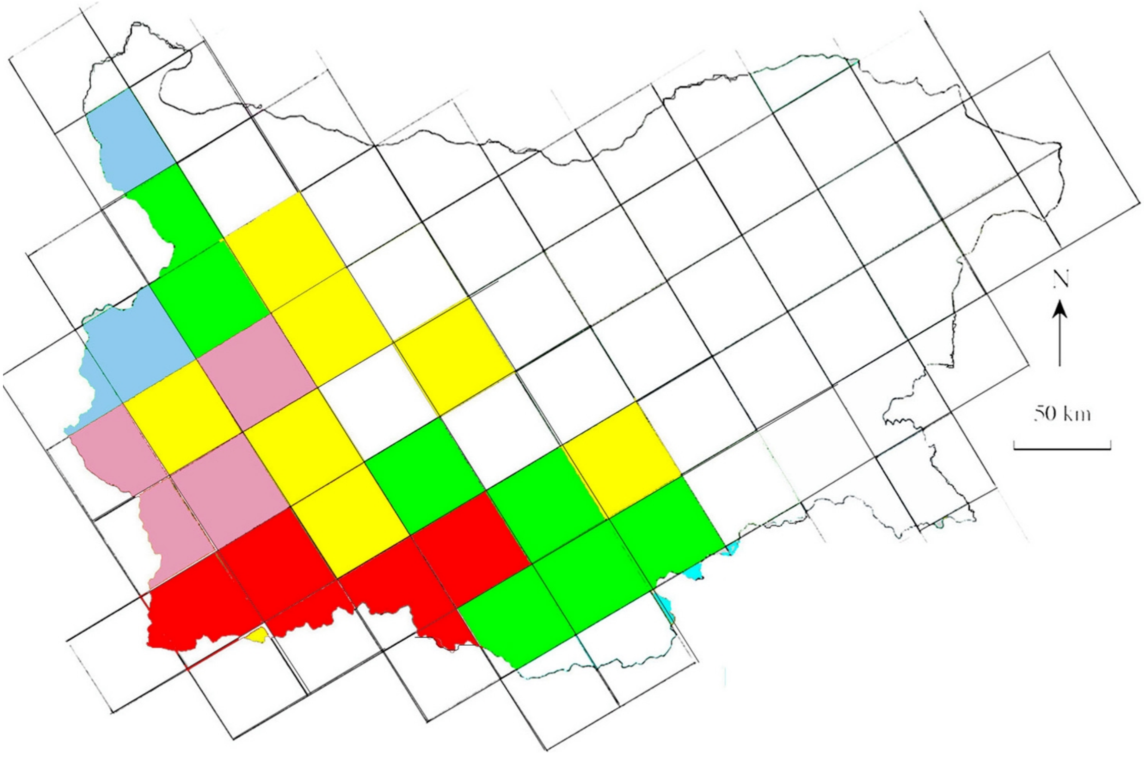


Figure BG-1. Maximum critical loads of sulphur for deciduous (top) and coniferous forests (bottom) in Bulgaria.

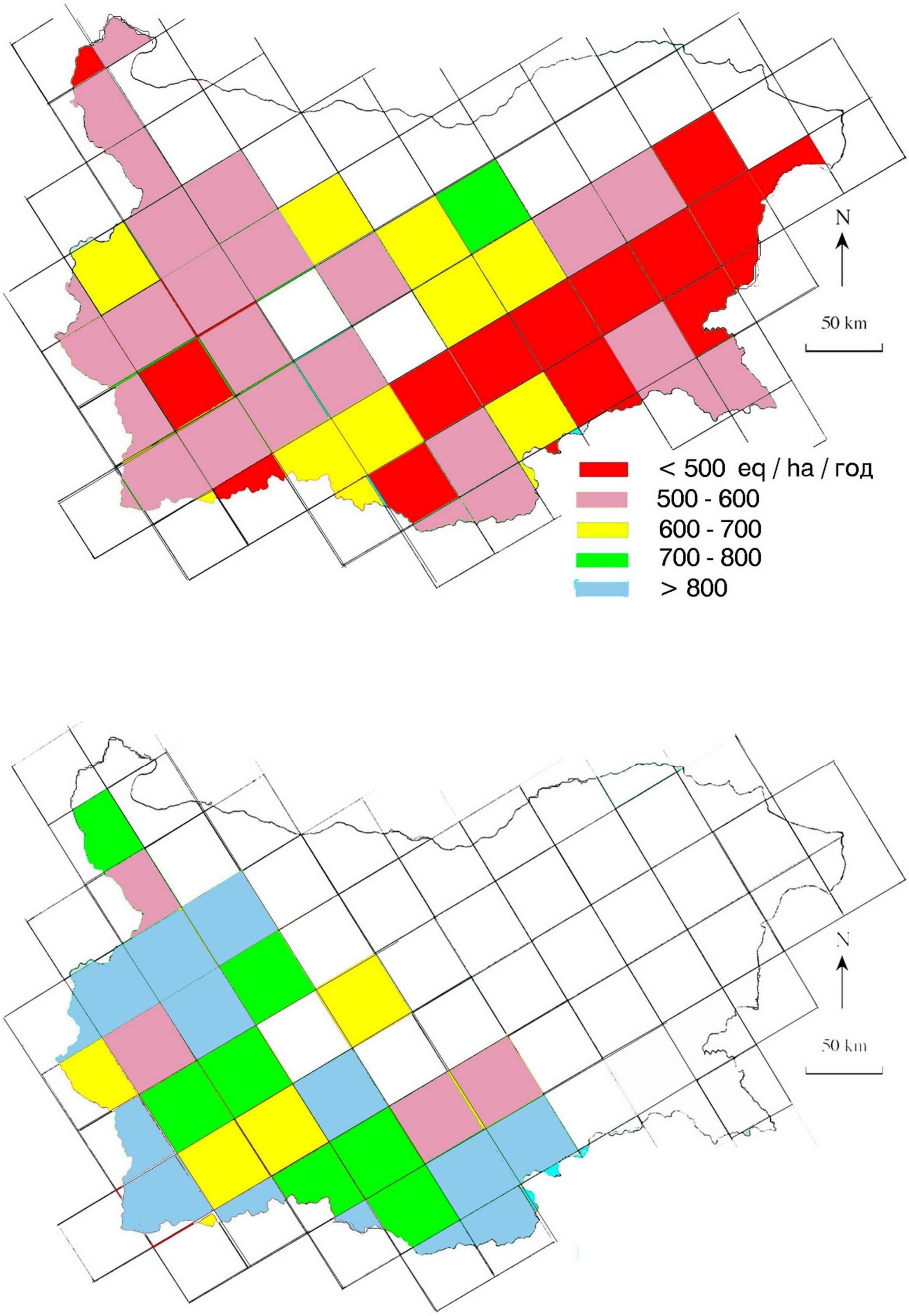


Figure BG-2. Minimum critical loads of nitrogen for deciduous (top) and coniferous forests (bottom) in Bulgaria.

Conclusions

Almost all parameters needed to run the Very Simple Dynamic model for both coniferous and deciduous forests, determined mostly by field measurements, are available for each EMEP grid cell.

The calculated values for critical loads of sulphur and nitrogen give a good initial indication of the spatial variability of ecosystem sensitivity to acidification in Bulgaria.

Considering that deciduous forest ecosystems occupy 2.5 times more area than coniferous in Bulgaria, and that the critical loads for deciduous forests are much lower than those for coniferous ones with similar geographic and climatic conditions, deciduous ecosystems could be used as a biological monitor for atmospheric pollutant reduction.

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National maps produced

Computation and mapping of critical loads have now also been conducted in the eastern part of Croatia (Slavonia region) in the following EMEP50 grid cells: (82, 46), (82,47), (82,48), (82, 49), (83,47), (83,48), (83,49), (84,48), (84,49). (Figure HR-1).

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 the Slavonia region by using the Simple Mass Balance method according to the Mapping Manual (UBA 1996).

In the Slavonia region, 20 soil-vegetation combinations were identified based on numerous forest profiles. About 20 profiles have been identified by field soil sampling for critical load mapping (Pernar 2002), using representative points selected to extend the existing soil databases (Martinović et al. 1998).

Data sources

- Receptor map 1:250,000 (Martinović 2002). 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 1998) and other related literature (Pelzer 1982, 1989; Rauš and Vukelić 1994, Trinajstić et al. 1992).
- Soil data: Soil database of Croatia (Martinović et al. 1998).
- Precipitation: Data on climatic zones of forest vegetation (Bertović 1994).
- Base cation (BC_{dep}) and chlorine (Cl_{dep}) deposition: Meteorological and Hydrological Service of Croatia, four stations (for 1998–2002).
- 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 (***$ANC_w = BC_w$***): BC_w values have been calculated according to the Mapping Vademecum (Hettelingh and De Vries 1992, pp. 34-37). Critical alkalinity leaching is calculated as:

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

using the following values (from De Vries 1991):

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

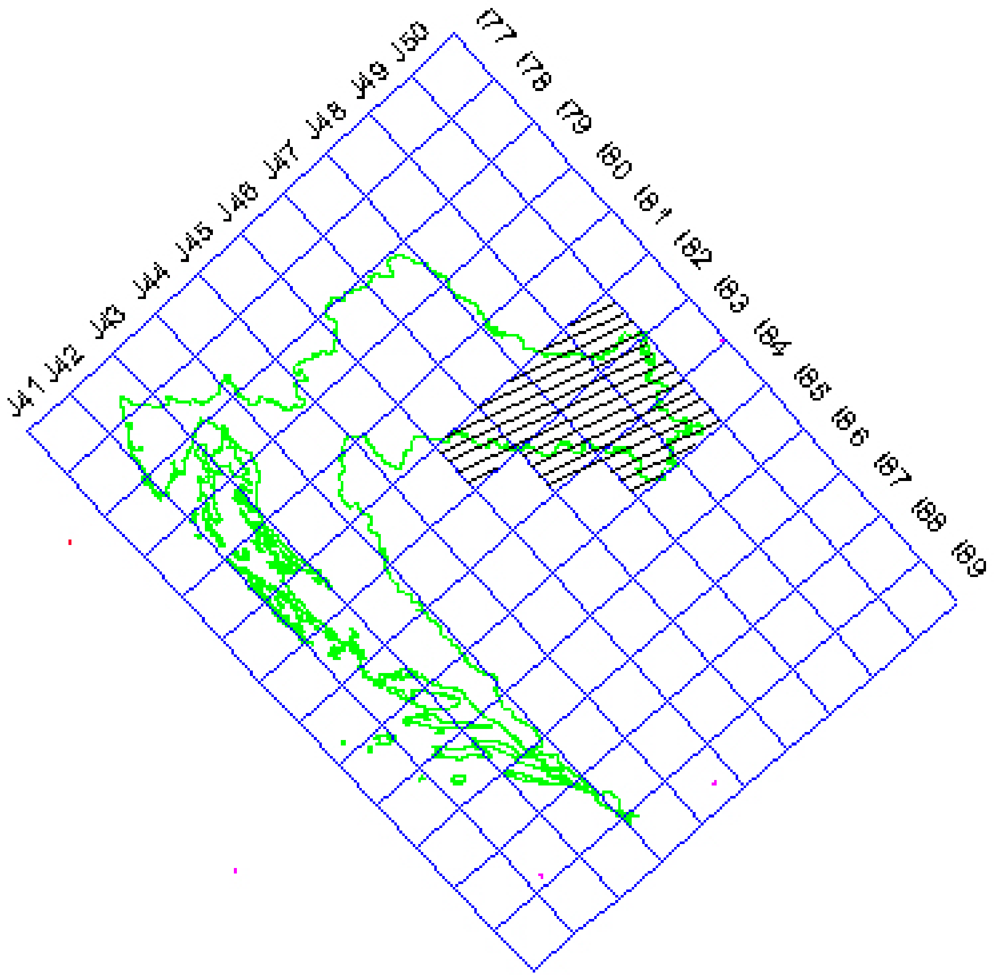

Figure HR-1. Location of the EMEP50 grid cells (Slavonia region) for which critical loads have now been calculated. (See also Croatian NFC report in Posch et al. 2001 for earlier work.)

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) and Martinović (1996) were used: for beech/fir $a = 0.34$, beech 0.25, oak 0.15, poplar and willow 0.15, and ash 0.03.

Precipitation has been determined on the basis of 30 years of climate data from four weather stations and associated with various 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).

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})
Beech and fir	0.25
Oak	0.35
Ash	0.35
Poplar and willow	0.30

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

Denitrification: has been calculated 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 denitrification factor f_{de} have been assigned according to the Mapping Manual (UBA 1996), in the range of 0.1–0.7.

Base cation deposition: Bulk deposition data for base cation deposition were extrapolated from four monitoring stations (Vidić 2002). 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

The following results were calculated for the study area:

Parameter (eq ha^{-1} a^{-1})	Minimum value	Maximum value
$CL_{max}(S)$	1022	2953
$CL_{min}(N)$	544	1088
$CL_{max}(N)$	2997	8650
$CL_{nut}(N)$	1059	1780

Comments on national conditions related to SMB method

The SMB method has been successfully applied in Croatia, using the national data for the following variables:

- net growth and harvesting
- volume increase in wood harvest
- drainage water
- precipitation by bio-climate region
- deposition (BC_{dep} and N_{dep}).

The other input data are taken from the critical load mapping literature. The application of the SMB method indicates some ecological national characteristics that should be taken into account.

Comments on data needed for dynamic modelling

Data for dynamic modelling are based on 20 profiles for five vegetation-soil combinations, identified by field soil sampling for the purpose of critical load mapping (Pernar 2002), on the basis of representative points selected to extend the data in the existing soil database. Note that these data are preliminary and further investigations are needed.

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National data and maps

The database provided to the CCE includes critical loads of sulphur (maximum), nitrogen (maximum and minimum), nutrient nitrogen and related data (Table CZ-1). In addition, the soil parameters require to apply dynamic model are included. The evaluation of critical loads was carried out for forest ecosystems. Three forest ecosystem types have been investigated (listed with EUNIS ecosystem codes and predominant tree types):

- Coniferous forest ecosystems – EUNIS code G3 (*Picea abies*, *Pinus sylvestris*, *Larix decidua*)
- Broadleaved deciduous forest ecosystems – EUNIS code G1 (*Fagus sylvatica*, *Quercus robur*, *Quercus petraea*, *Carpinus betulis*)
- Mixed forest ecosystems – EUNIS code G4

The geographic coordinates for forest ecosystems represent the reference points of polygons in the CORINE map. Data processing includes combining information from GIS layers such as temperatures, precipitation, runoff, base cation depositions, and the actual measured soil data provided by the Forest Management Institute in Brandýs nad Labem. Due to a lack of measured soil characteristics for all forest ecosystems in the Czech Republic, the database represents only 62% of forest ecosystems (of the above-mentioned ecocodes). The following maps were produced:

- Maximum critical loads of sulphur
- Critical loads of nutrient nitrogen
- Critical loads of acidity
- Exceedances of acidity
- Exceedances of nutrient nitrogen.

Calculation methods for critical loads

The Mapping Manual (UBA 1996), the Dynamic Modelling Manual (Posch et al. 2003) and the CCE Status Report 2001 (Posch et al. 2001) are the main methodological sources for the evaluation of critical loads and related data provided in the database. The simple mass balance method summarised in the Mapping Manual 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):

$$CL_{max}(S) = BC_w + BC_{dep} - BC_u - ANC_{le(crit)}$$
$$CL_{min}(N) = N_u + N_i$$
$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) / (1 - f_{de})$$
$$CL_{nut}(N) = N_u + N_i + N_{le}$$
$$CL(Ac)_{pot} = BC_w - BC_u + N_i + N_u + N_{de} - ANC_{crit}$$

where:
$ANC_{le(crit)} = -Q \cdot [H]_{crit} + [Al]_{crit})$
and
$N_{le(acc)} = Q \cdot [N]_{crit}$

The critical values of concentrations for H, Al and N in the soil solution for the ecosystem types investigated are given in Table CZ-2. Runoff, *Q*, represents the amount of water percolating through the soil profile. The annual water fluxes were provided by the Water Management Institute, Prague. The data represent the 20-year average of water basic runoff by hydrogeological regions.

Table CZ-1. Data included in the national database on critical loads of nitrogen and sulphur and in the dynamic modelling.

Value	Name	Unit
Longitude	Co-ordinate	Decimal degrees
Latitude	Co-ordinate	Decimal degrees
I	EMEP50 grid – i index	–
J	EMEP50 grid – j index	–
Ecoarea	Ecosystem area	km^2
CLmaxS	Maximum critical load of S	eq ha^{-1} a^{-1}
CLminN	Minimum critical load of N	eq ha^{-1} a^{-1}
CLmaxN	Maximum critical load of N	eq ha^{-1} a^{-1}
CLnutN	Critical load of nutrient N	eq ha^{-1} a^{-1}
BCdep	Base cation deposition	eq ha^{-1} a^{-1}
BCu	Base cation uptake	eq ha^{-1} a^{-1}
BCw	Base cation weathering rate	eq ha^{-1} a^{-1}
Q	Specific runoff (draining soils only)	eq ha^{-1} a^{-1}
Kgibb	Gibbsite equilibrium constant	m^6 eq^{-2}
–ANClecrit	Critical leaching of acid neutralising capacity	eq ha^{-1} a^{-1}
Ni	Nitrogen immobilisation	eq ha^{-1} a^{-1}
Nu	Nitrogen uptake	eq ha^{-1} a^{-1}
fde	Denitrification fraction	eq ha^{-1} a^{-1}
Nleacc	Acceptable nitrogen leaching	eq ha^{-1} a^{-1}
Ecocode	Ecosystem code	–
thick	Depth of the rooting zone	m
rho	Bulk density of the soil	g cm^{-3}
Theta	Volumetric water content at field capacity	m^3 m^{-3}
CEC	Cation exchange capacity	meq kg^{-1}
EBC	Base saturation	–
yearEBC	Year for which base saturation was calculated	–
Cpool	Initial amount of carbon in the topsoil	g m^{-2}
CNrat0	Initial C:N ratio in the topsoil	–
soiltype	Soil type	–
clay	Clay content of the mineral soil	%
sand	Sand content of the mineral soil	%
Corg	Organic carbon content of the soil	%
pH	Soil pH	–
Prec	Mean annual precipitation	mm a^{-1}
Temp	Mean annual temperature	ºC
Alt	Altitude above sea level	m

Table CZ-2. Critical concentrations used in the calculations of critical loads (in eq m^{-3}).

	EUNIS Ecosystem Code		
Parameter	G3	G4	G1
$[N]_{crit}$	0.0143	0.02095	0.00276
$[H]_{crit}$	0.09	0.09	0.09
$[Al]_{crit}$	0.2	0.2	0.2

Soil type and soil texture determine the rate of base cation weathering, BC_w (UBA 1996). Soil texture characteristics have also been used to derive the denitrification factor f_{de} (Table CZ-3).

Table CZ-3. Denitrification fractions used for various soil classes.

Clay %	f_{de}
< 20	0.1
≥ 20 and < 30	0.2
≥ 30 and < 55	0.3
≥ 55	0.5
Podsols	0.1
Histosols	0.8

Nitrogen immobilisation, N_i, (see Table CZ-4) has been derived from long-term annual temperatures (See German NFC report in Posch et al. 2001).

Table CZ-4. Rates of nitrogen immobilisation in forest soils according to annual mean temperatures.

Annual temperature (ºC)	N_i (eq ha^{-1} a^{-1})
< 5.5	357
5.5	250
6.5	178
7.5	125
≥ 8.5	71

Uptakes of nitrogen, N_u, and base cations, BC_u, have been derived on the basis of land use classification maps, base cation weathering rates, annual temperatures and runoff. The values of N_u and BC_u used in the evaluation of critical loads are presented in Table CZ-5.

Table CZ-5. Rates of base cation and nitrogen uptake for coniferous, deciduous and mixed forests for various yield classes (eq ha^{-1} a^{-1}).

	Coniferous (G3)		Mixed (G4)		Deciduous (G1)	
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

Calculation methods for additional soil parameters

While the evaluation of critical loads requires data available from GIS databases, data requirements for dynamic modelling include additional measured parameters on soil properties and chemical composition. The soil database provided by the Forest Management Institute in Brandýs nad Labem contains about 2500 soil analyses, comprising 41 unique forest areas. The soil parameters required for dynamic modelling (pH, E_{BC}, CNrat0, C_{org}, clay and sand compositions) were taken from this database and converted to a depth of 0.5 m of the soil profile (with a few exceptions).

The data were processed for each forest area separately and joined to the soil map according to the soil type, clay contents and C_{org} composition. The coverage of the soil map was not complete (only 62%) due to the lack of data on some soil types, clay content or C_{org} classes. The other data such as θ, ρ, CEC and C_{pool} were derived using equations described below.

Volumetric water content:

$\theta = 0.04 + 0.023 \cdot \min\{clay/30, 1\}$ for $C_{org} \leq 15\%$

$\theta = 0.75$ for $C_{org} > 15\%$

where θ = soil moisture content at field capacity, clay and C_{org} are the clay content and organic carbon content (in %), respectively.

Soil bulk density: is calculated as:

$\rho = 1/(a_0 + a_1 \cdot C_{org})$ for $C_{org} \leq 15\%$

$\rho = 0.825 - 0.037 \cdot \log(2 \cdot C_{org})$ for $C_{org} > 15\%$

where:
ρ = the bulk density (g m^{-3})
C_{org} = the organic carbon content (%)

and:

$$\rho = (1/z)\sum_{l=1}^{n} z_l \rho_l$$

where ρ = the mean bulk density, and the total thickness (soil depth) z is given as:

$$z = \sum_{l=1}^{n} z_l$$

where z_l is the thickness of the lth soil layer.

The initial amount of carbon in topsoil is calculated as:

$$C_{pool} = 10^4 \cdot \rho_{top} \cdot z_{top} \cdot C_{org,top}$$

where:
$C_{org,top}$ = the organic C content in the topsoil (%),
ρ_{top} = bulk density of the soil layer considered (g cm^{-3})
z_{top} = the thickness of that layer (m)

Cation exchange capacity is calculated as:
$CEC = (0.44 \cdot pH_{KCl} + 3.0) \cdot clay + (5.1 \cdot pH_{KCl} - 5.9) \cdot C_{org}$

where:
CEC = cation exchange capacity (meq kg^{-1}), and *clay* and C_{org} are the clay content and organic carbon content, respectively (%).

Conversion of CEC data measured or derived for each layer of the soil profile to the total depth (0.5 m) was conducted as follows:

$$CEC = (1/zp)\sum_{l=1}^{n} z_l \rho_l CEC_l$$

Results

Critical loads have been calculated using data derived from GIS maps. The types of maps applied, their scales and sources are shown in Table CZ-6. Critical loads of sulphur and nitrogen have been compared to actual atmospheric deposition for the period 1991–2001, and exceedances evaluated. Figures CZ-1 and CZ-2 depict $CL_{max}(S)$ and $CL_{nut}(N)$, respectively. Figure CZ-3 depicts critical loads of acidity.

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

Map	Scale	Source	Updated
Soil type and texture	1:200,000	Agricultural University, Prague (Nemecek et al. 2000)	yes
Soil organic matter	1:200,000	Agricultural University, Prague (Nemecek et al. 2000)	yes
Annual mean runoff	1:200,000	Water Management Institute, Prague (Olmer et al. 1998)	no
CORINE map		Ministry for the Environment of the Czech Republic (1995)	no
Annual mean base cation deposition	2×2 km^2	Czech Hydrometeorological Institute, Prague (Fiala and Livorova 2003)	new
Annual mean temperature	1:500,000	Czech Hydrometeorological Institute, Prague (Kveton et al. 1999)	yes
Annual mean precipitation	1:500,000	Czech Hydrometeorological Institute, Prague (Kveton et al. 1999)	new
Altitude	1:500,000	ArcData (from the map of the Czech Republic, 1996)	new
Soil measured data	localities	Forest Management Inst., Brandýs n.L. (Pokorny et al. 2001)	new
Annual mean atmos-pheric depositions of S and N	2×2 km^2	Czech Hydrometeorological Institute, Prague (Fiala and Livorova, 2002)	yes

Comments and conclusions

Lower values of $CL_{max}(S)$ compared to earlier calculations were mainly the result of using updated BC_{dep} input data. National base cation depositions (mean annual values for the period of 1990–2001) were lower by 500 eq ha^{-1} a^{-1} in some cases compared to the EMEP data used in calculating critical loads published in the CCE Status Report 2001. The values of critical loads of nutrient nitrogen have not changed significantly.

In comparison to the critical loads of sulphur and nitrogen published in the CCE Status Report 2001, the national database at present involves more data based on updated GIS layers as well as measured data on soil properties and chemistry. These parameters enable a better comparison of calculated exceedances of critical loads and actual state of the soil. Values for exceedances of critical loads of acidity have decreased significantly from 1990 to present throughout the entire territory of the Czech Republic. Exceedances of nutrient nitrogen critical loads, meanwhile, have not changed as much.

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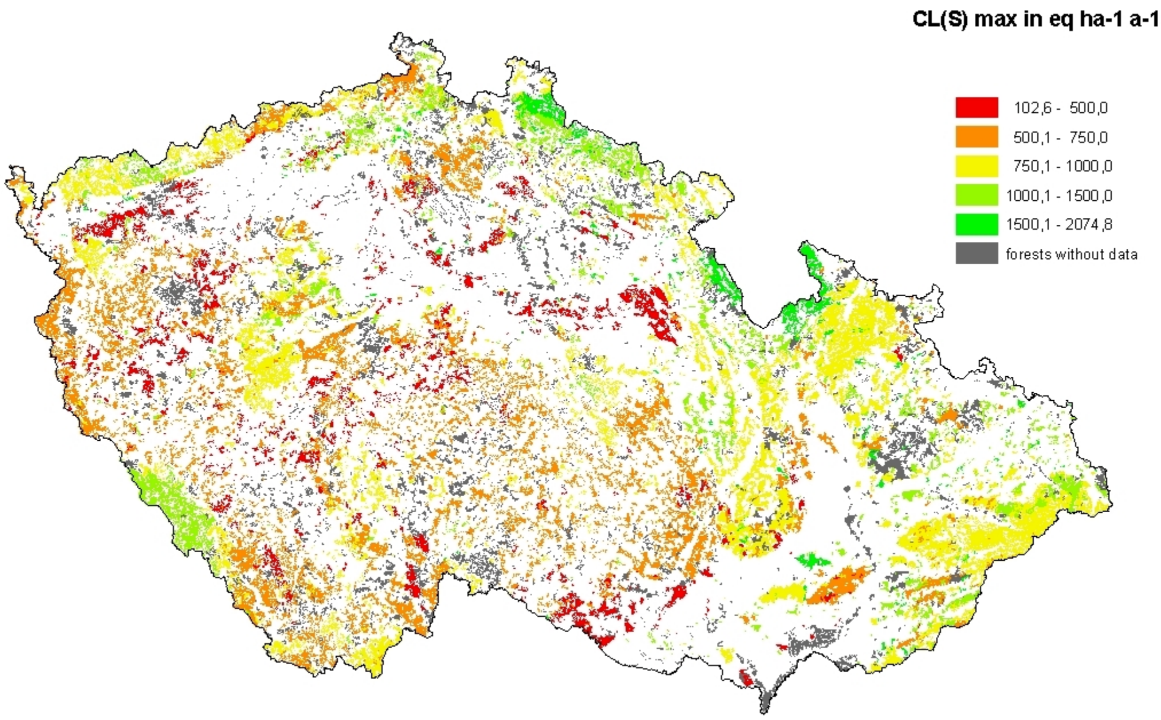

Figure CZ-1. Maximum critical loads of sulphur (eq ha^{-1} a^{-1}).

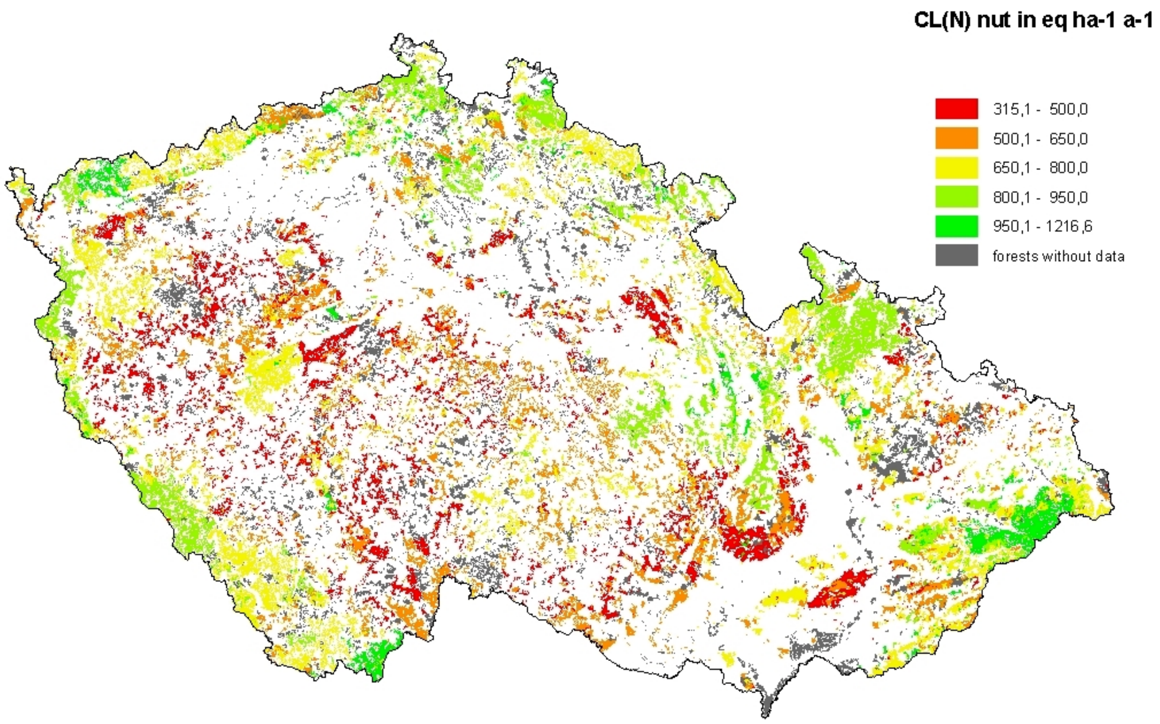

Figure CZ-2. Critical loads of nutrient nitrogen (eq ha^{-1} a^{-1}).

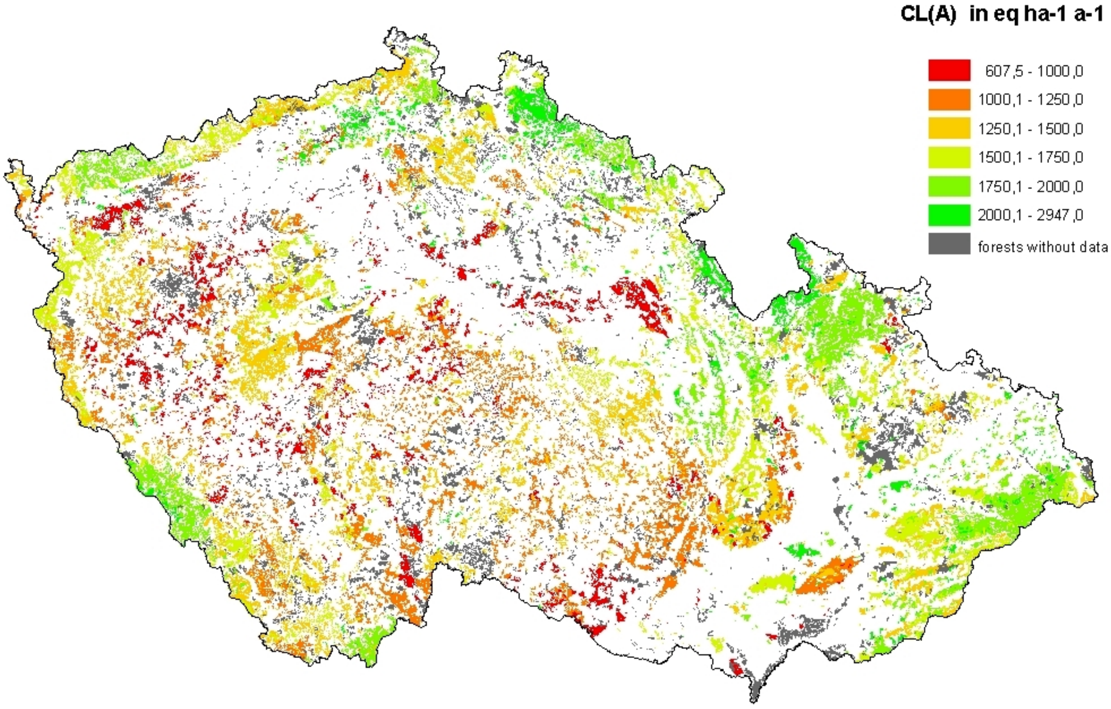

Figure CZ-3. Critical loads of acidity (eq ha^{-1} a^{-1}).

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National maps produced

- Critical load, and exceedance of the critical load of acidity for forest soils and extensively managed, permanent grasslands calculated with PROFILE (Warfvinge and Sverdrup 1992) and for grasslands with the SMB model.
- Critical load and exceedance 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.
- National deposition maps of NH_X on a 30×30 km^2 grid.
- National deposition map of NO_X on a 30×30 km^2 grid.
- National deposition map of SO_X on a 30×30 km^2 grid.

Calculation methods

Critical loads of acidity and N eutrophication:
The PROFILE model has been used to calculate critical loads of acidity and for nitrogen eutrophication, and the values of BC_u, N_u, BC_w, and $ANC_{le(crit)}$. From this calculation, the values of $CL_{min}(S)$, $CL_{max}(S)$, $CL_{min}(N)$, and $CL_{max}(N)$ have been derived. In calculating critical loads for grasslands, the weathering rate for 11 mineralogy classes were calculated at 1000 points with the PROFILE model. The calculation of critical loads for grasslands were performed with the SMB model (UBA 1996). The total number of calculations and the calculated critical loads for the different vegetation types are illustrated in Table DK-1.

Table DK-1. Critical loads of acidification and N eutrophication for various ecosystems. All values are given in keq ha^{-1} a^{-1} as the range between the 5th and 95th percentile.

	No. of calculations	***CL(A)***	***$CL_{nut}(N)$***
Beech	2,825	0.9 – 2.7	1.2 – 1.9
Oak	448	0.8 – 2.2	1.2 – 2.0
Spruce	5,480	1.4 – 4.1	0.6 – 1.1
Pine	1,035	1.4 – 2.4	0.5 – 0.7
Grass	18,178	0.9 – 2.4	—

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

Empirically based critical loads of eutrophication:
Critical loads of nutrient nitrogen for inland and coastal heathland, raised bogs, pastures, sensitive meadows and sensitive (lobelia) lakes have been derived on a 5×5 km^2 national grid. The basis of the assessment has been the registration of nature areas according to section 3 of the Danish Nature Protection Act and the revision of the empirical based critical loads following the 2002 Bern workshop. The quality and quantity of the 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.

Data for dynamic modelling:
In response to the most recent CCE call for data for dynamic modelling, a Danish dataset has been prepared. A full set of parameters for dynamic modelling exist only for a very limited number of research sites which is insufficient to represent the whole country. It has therefore been decided to extend the existing critical load database with a set of additional parameters needed for dynamic modelling of soil acidification. Some of the data already exists, since the PROFILE model has been used for critical load calculations. Data for dynamic modelling of nitrogen processes is not yet available. The extension has been made for all the forest points in the Danish critical load database, i.e. 9788 data points. Before the next call for data, the extension will be made also for grasslands. In addition, a national validation exercise will be conducted, comparing

VSD results based on generalised input data with results obtained with the SAFE model at locations where better input data is available. Table DK-2 summarises the transfer functions used in deriving the data.

Table DK-2. Derivation of additional data for dynamic modelling.

Variable	Source	
Thick	Soil-dependent, national data	
ρ	$1/(0.065 + 0.05 \cdot C_{org}\,\%)$	where $C_{org} < 15$
	0.759	where $C_{org} > 15$
θ	$0.04 \cdot 0.0077 \cdot$ clay %	
CEC	Transfer function from Mapping Manual	
BS	Transfer function from Mapping Manual	
C_{pool}	Thick (m) $\cdot C_{org}\,\% \cdot 200{,}000$	
C:N	Transfer function from Mapping Manual	

National deposition maps:
As part of the Danish Nationwide Background Monitoring Programme, deposition calculations of NH_Y, NO_x and NH_x to Danish sea and land area have been performed on a 30×30 km^2 national grid on a yearly basis. The latest reporting of data from this programme has been in 2002.

Data sources

The main sources of data have not changed since the 1997 Status Report. In addition to the existing data sources, a dataset of 1000 points from the Danish grid net for soil data has been included as a basis for deriving and checking transfer functions. Table DK-3 shows the sources and resolution of input data.

Table DK-3. Data sources.

Parameter	Resolution	Source
Soil mineralogy	60 points	DLD, literature
Soil texture	1:500,000	DLD
Geological origin	1:500,000	DLD
Crop yields	county	DSO
Forest production	1:500,000	DLD, DSO
Ecosystem cover	25 ha	NERI
Deposition (S, N)	5×5, 20×20 km^2	NERI
Meteorology	1:1,000,000	DMI

DLD: National Institute of Soil Science, Dept. of Land Data
DSO: Danish Statistical Office
NERI: National Environmental Research Institute
DMI: Danish Meteorological Institute

Comments and conclusions

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

- Further work on methods and data for the calculation of critical loads of nutrient nitrogen for sensitive, natural or semi-natural terrestrial ecosystems, primarily raised bogs and heathlands.
- Estimation of uncertainties in calculated critical load exceedances, with special emphasis on the influence of local-scale variation in NH_X deposition (Bak and Tybirk 1998).
- Preparation of data needed for dynamic modelling of soil acidification, in response to the 2002 call for data.

As indicated, only minor progress has been made in the availability of data for calculating steady-state critical loads. National deposition maps, now updated annually, are believed to provide a better basis for calculating critical load exceedances. In the exceedance calculations, the 30×30 km^2 deposition fields are downscaled to a 1×1 km^2 resolution for each ecosystem type. The NH_Y deposition values are further modified on the basis of the emission density in a circular neighbourhood with a radius of 2.5 km. Furthermore, local variation in deposition within the 1×1 km^2 grid is taken into account in the exceedance calculations.

Data needed for dynamic modelling has primarily been derived from existing datasets by using transfer functions, both from the draft mapping manual and derived from national data. The usefulness of the dataset has not yet been sufficiently validated. This will be done in preparation for the next call for data.

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Calculation methods

Maps of critical loads of sulphur and nitrogen covering all of Estonia have not been updated since 2001. New deposition data has been collected showing no exceedance of critical loads.

Dynamic modelling is proceeding with further applications of a complex soil-vegetation-atmosphere model RipFor on new sites. The model allows one to calculate critical loads and assess their uncertainties. Pilot studies on application of the VSD dynamic model for assessment of critical loads have been carried out on selected sites.

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Critical loads

The calculation of critical loads for Finnish forest soils and lakes follows the methodology of the Mapping Manual (UBA 1996) and was described in detail by Johansson (1999) and Johansson et al. (2001). Neither the methods nor the input data for the calculation of Finnish critical loads of sulphur and nitrogen have been updated since 2001, thus the Finnish data used in the present report correspond to those reported in 2001.

The soils and lakes in Fennoscandia are sensitive to acidification, partly because minerals weather slowly and have low contents of base cations (Henriksen et al. 1998). Therefore the calculated critical loads are low, compared with values for areas with carbonaceous minerals. Critical loads are exceeded in some areas in Finland, although the deposition load is lower than in central Europe. In Figure FI-1 (from Holmberg and Syri 2000), the difference between the critical load of acidity and the deposition of acidifying sulphur and nitrogen is expressed as a percentage of the deposition. For the year 2010, the modelled deposition corresponds to emissions following the UNECE (1999) protocol. Following this protocol, the calculated reduction requirement is lower than what was estimated for the year 1995.

Current methods for calculating critical loads of acidity for forest soil were reviewed as a result of a project funded by the Nordic Council of Ministers (Holmberg 2000). The consequences of four sets of assumptions concerning the soil model structure, parameter values and the critical loads criterion were explored by comparing the values of the average accumulated exceedance (AAE) calculated for Finland with deposition values for the year 1995. Using a critical limit for the molar ratio of the concentrations of base cations to aluminium in soil solution gave the lowest AAE. Assuming organo-aluminium complexes and leaching of organic anions resulted in AAE = 4 eq ha^{-1} a^{-1}, which was 20% less than the value obtained with the standard approach, assuming gibbsite equilibrium and no leaching of organic anions. The lowest critical load, and the highest AAE (25 eq ha^{-1} a^{-1}), was obtained when the effects-based criterion (critical concentration or critical base saturation) was substituted with one restricting the deterioration of the neutralising capacity of the soil, $ANC_{le(crit)} = 0$.

The test with critical base saturation resulted in critical load values in between these extremes (Holmerg et al. 2001). These tests illustrate the variability of the critical load values for acidity that can be introduced by changing the criterion or by varying the calculation method, without, however, representing the extreme values of critical loads that could be derived. The uncertainties in the critical loads dominate the total uncertainty of integrated assessment modelling of acidification for Finland. Thus further research efforts to reduce uncertainties should focus mainly on decreasing the uncertainty in critical load values by improving ecosystem process descriptions (Syri et al. 2000, Syri 2001).

Dynamic modelling

Finland has not done and does not anticipate any regional implementation of dynamic modelling of target load functions for forest soils. Calculation and reporting of dynamic target load functions for selected sites that are part of the ICP Integrated Monitoring network are planned for 2004. The SMART dynamic acidification model has been used to predict recovery for 40 acid-sensitive Finnish headwater lakes, for which both catchment soil and water quality observations were available (Posch et al. 2003). Target load functions have not yet been calculated for these lakes.

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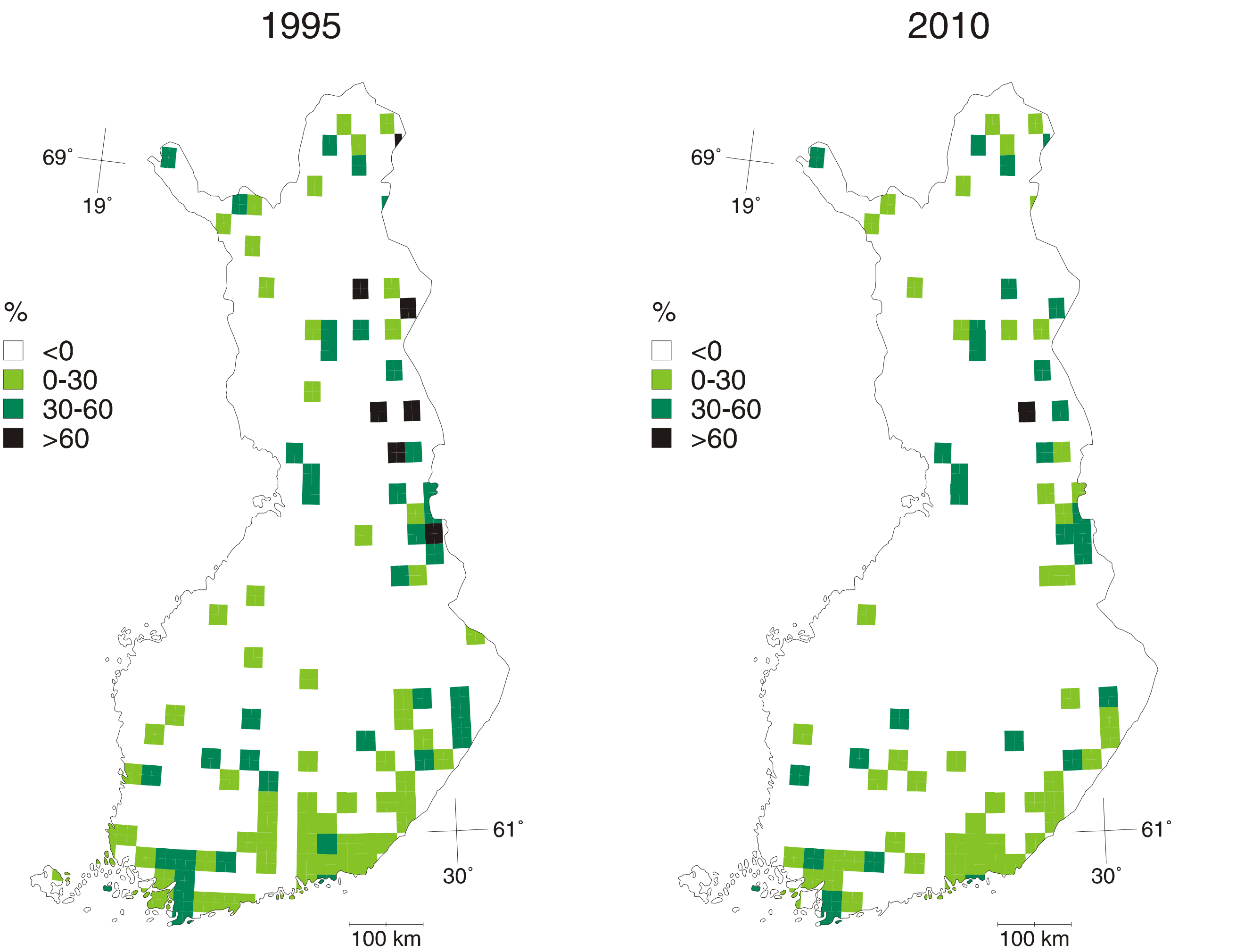

Figure FI-1. The reduction requirements for acidifying deposition, in % of modelled deposition, for the years 1995 and 2010 (from Holmberg and Syri 2000).

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National maps produced

The French ecosystems list was re-created in 2003 to update the previous critical loads data for forest ecosystems (Probst and Party 1999). The main steps in developing the new ecosystem map were the following:

- French soil formations (21 types) are combined with corresponding bedrocks (31 types, INRA 1998). After deleting the combinations that do not occur in French territory, a total of 31 soil/ bedrock combinations remained (Party 1999).
- This intermediate list of soils and parent materials was then overlaid with a potential vegetation map (18 types).

The combination of these three datasets results in a new map of French ecosystems with 5529 objects and 282 different ecosystems. Each ecosystem is characterised by a soil type, a bedrock type and a potential vegetation type.

To calculate critical loads, this ecosystem map was combined with the present land use map. Ecosystems with dominant agricultural land use or those with 50 to 85% grassland were grouped into the "crops" class and were not considered in the data submitted to the CCE. (Applying the steady-state mass balance approach to derive critical loads as described in the following paragraph is irrelevant for crops because of huge anthropogenic contributions other than atmospheric.)

Polygons with more than 85% grassland were coded as "grasslands" and considered as natural grassland. To remove objects which are due to imperfect overlapping during the overlaying of maps, only polygons > 1 km² were considered. The ecosystem code is comprised of a soil and bedrock type (31 classes) and a potential vegetation type (18 classes). After all map combinations, 241 ecosystem types exist in the French forest ecosystems list. In 1999, data were provided for 38 ecosystem types (785 records). In 2003, 241 ecosystem types (4141 records) are represented.

Calculation of critical loads

The Steady State Mass Balance (SSMB) model was applied on the top soil layer as described in Posch et al. (1995). The critical loads of acidity (Eqs. 1, 2.1 and 2.2), sulphur (Eq. 3), acidifying nitrogen (Eqs. 4, 5) and nutrient nitrogen (Eq. 6) were calculated as:

$$CL(AC_{act}) = BC_w + ANC_{le(crit)} \qquad (1)$$

$$ANC_{le(crit)} = Q \cdot [H]_{crit} + Q \cdot (Al/BC)_{crit} \cdot (BC_{dep} + BC_w - BC_u) \quad (2.1)$$

if $BC_u > BC_{dep} + BC_w$ then $ANC_{le} = Q \cdot [H]_{crit}$

$$BC_{dep} = Ca_{dep} + Mg_{dep} + K_{dep} - Cl_{dep} \quad (2.2)$$

(with $Na_{dep} = 0$)

$$CL_{max}(S) = BC_{dep} + BC_w - BC_u + ANC_{le(crit)} \quad (3)$$

$$CL_{min}(N) = N_i + N_u \quad (4)$$

$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S) \quad (5)$$

$$CL_{nut}(N) = N_i + N_u + N_{de} + N_{le} \quad (6)$$

where N_i, N_u, N_{de}, and N_{le} are the immobilised nitrogen flux, nitrogen uptake, denitrified nitrogen and nitrogen leaching (Party and Thomas 2000).

Data sources

$[H]_{crit}$ is the critical hydrogen concentration in drainage water (= 25 μeq l^{-1}, which corresponds to pH = 4.6, adapted to French forest soils, Party 1999). The critical Al:BC ratio, $(Al/BC)_{crit}$, is 1.2 eq eq^{-1} calibrated at the national scale (Party 1999). BC_w has been determined with the PROFILE model (Sverdrup and Warfvinge 1995, Party 1999). Q is the drainage water (BRGM 1983).

BC_{dep} (base cation deposition), N_{dep} (NO_3 and NH_4 deposition) and Cl_{dep} (chloride deposition) were determined using ONF data (French National Forest Office) on a 10×10 km^2 grid (Croise et al. in prep). Only open field deposition data could be used. All relevant data have been sea-salt corrected, assuming Na deposition to originate entirely from sea salt. However, the influence of dry deposition in the forest areas of concern is significant. To mitigate the lack of spatialised throughfall data at the national scale, a coefficient was applied to open field data to derive total deposition (Ulrich et al. 1998).

First the deposition data are corrected for sea salt deposition and then a coefficient is applied to derive throughfall values for each cation X:

$$X^*_{dep} = (X_{dep} - Na \cdot (X/Na)_{sea}) \cdot X_{coef} \quad (7)$$

where X is one of the base cations: Mg, K, Ca or Cl, and $(Mg/Na)_{sea} = 0.12$, $(K/Na)_{sea} = 0.036$, $(Ca/Na)_{sea} = 0.038$ and $(Cl/Na)_{sea} = 1.80$, and X_{coef} is the ratio between open field and throughfall deposition.

$(X/Na)_{sea}$ is the ratio between $[X]$ and $[Na]$ in seawater for a salinity of 35 g kg^{-1}, assuming that all sodium deposition in France comes from the sea, i.e. $Na^*_{dep} = 0$. BC^*_{dep} is then calculated as:

$$BC^*_{dep} = Mg^*_{dep} + K^*_{dep} + Ca^*_{dep} - Cl^*_{dep} \quad (8)$$

BC_u was determined using IFN (Forest National Inventory 2002) productivity data for French forest ecosystems and the estimated base cation and nitrogen concentrations in the biomass (after Posch et al. 1995, modified).

Results

The results of critical loads calculations are presented in Table FR-1, and in Figures FR-1 through FR-4.

Table FR-1. Critical loads data (2003) for French forest ecosystems (minimum, median and maximum values).

Data	Units	Min	Med	Max
$CL_{max}(S)$	eq ha^{-1} a^{-1}	19	12,014	88,115
$CL_{min}(N)$	eq ha^{-1} a^{-1}	207	435	1,474
$CL_{max}(N)$	eq ha^{-1} a^{-1}	342	12,439	88,544
$CL_{nut}(N)$	eq ha^{-1} a^{-1}	251	640	1,463
BC_{dep}	eq ha^{-1} a^{-1}	112	850	2,181
BC_u	eq ha^{-1} a^{-1}	67	263	1,222
BC_w	eq ha^{-1} a^{-1}	30	10,000	30,000
Q_{le}	mm a^{-1}	0	350	1,500
ANC_{crit}	eq ha^{-1} a^{-1}	0	2,185	56,645
N_i	eq ha^{-1} a^{-1}	150	150	300
N_u	eq ha^{-1} a^{-1}	57	285	1,324
N_{de}	eq ha^{-1} a^{-1}	0	121	776
N_{le}	eq ha^{-1} a^{-1}	0	0	100

Comparison between 1998 and 2003 data

The critical loads maps for sulphur and acidifying nitrogen prepared in 1998 and 2003 are rather similar. Nevertheless, there are some differences between the 1998 and 2003 5-percentile maps. Firstly, there is increased precision in ecosystem definitions and the interpolation of these data through the statistical computation of the 5th percentile. Furthermore, input data for biomass cations uptake and atmospheric deposition have been updated (described above) and adapted to French ecosystems.

Comparing 1998 and 2003 $CL_{max}(S)$ maps, it appears that the Landes region, the central part of France, and Brittany are more sensitive, whereas Massif Central is less sensitive. Nevertheless, the Landes region, the Vosges mountains, the central part of France and Brittany are still the most sensitive areas in France. Comparing 1998 and 2003 $CL_{min}(N)$ 5-percentile maps, it appears that all data are in the same range but the data distribution has changed. Therefore, $CL_{max}(N)$ reflects the same changes because it represents the sum of the two previous maps.

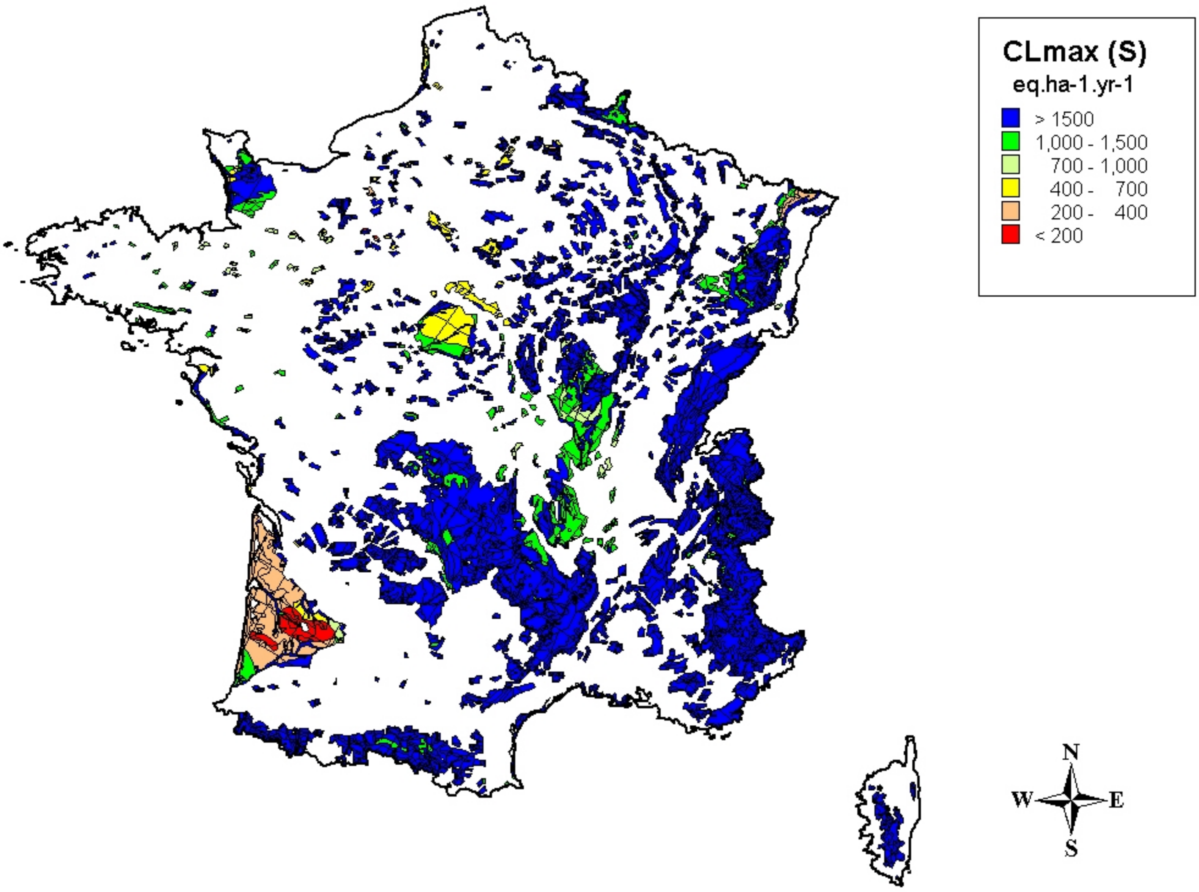


Figure FR-1. Critical loads of sulphur for French forest ecosystems (eq ha^{-1} a^{-1}).

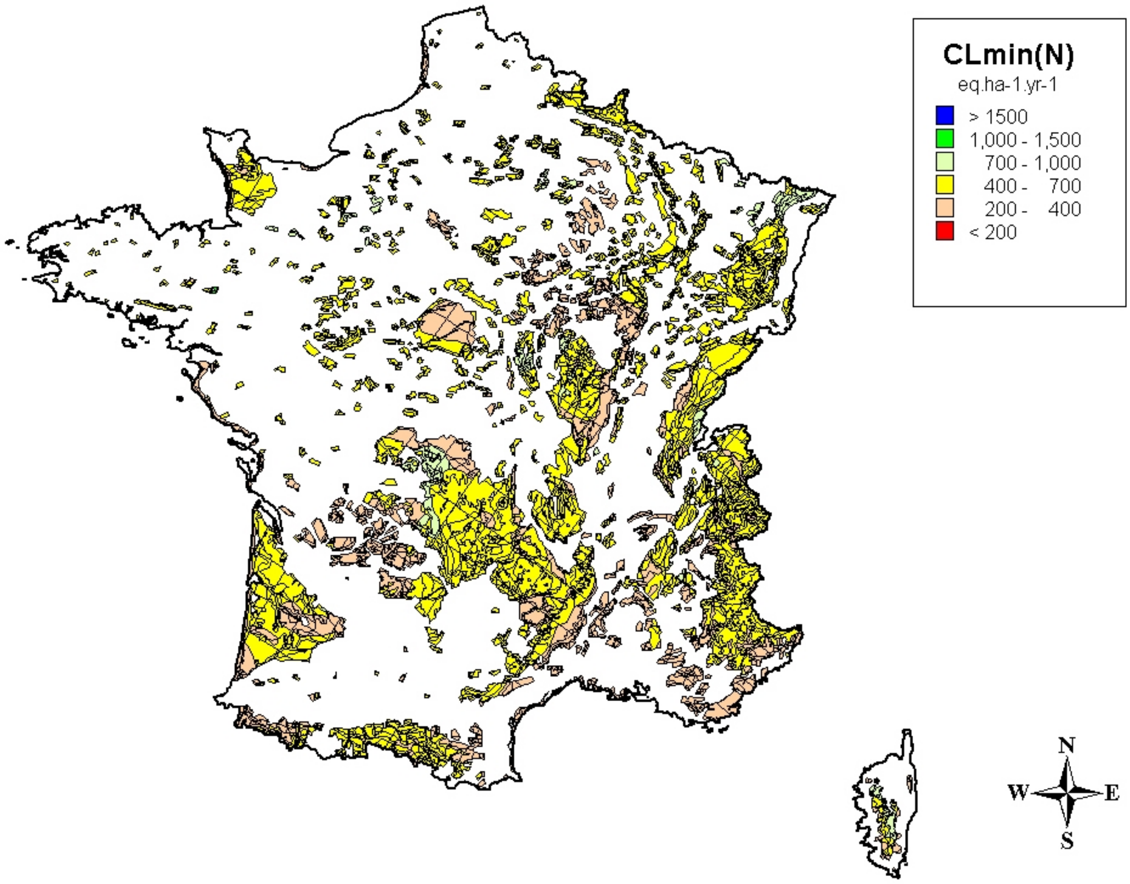


Figure FR-2. Minimum critical loads of acidifying nitrogen for French forest ecosystems (eq ha^{-1} a^{-1}).

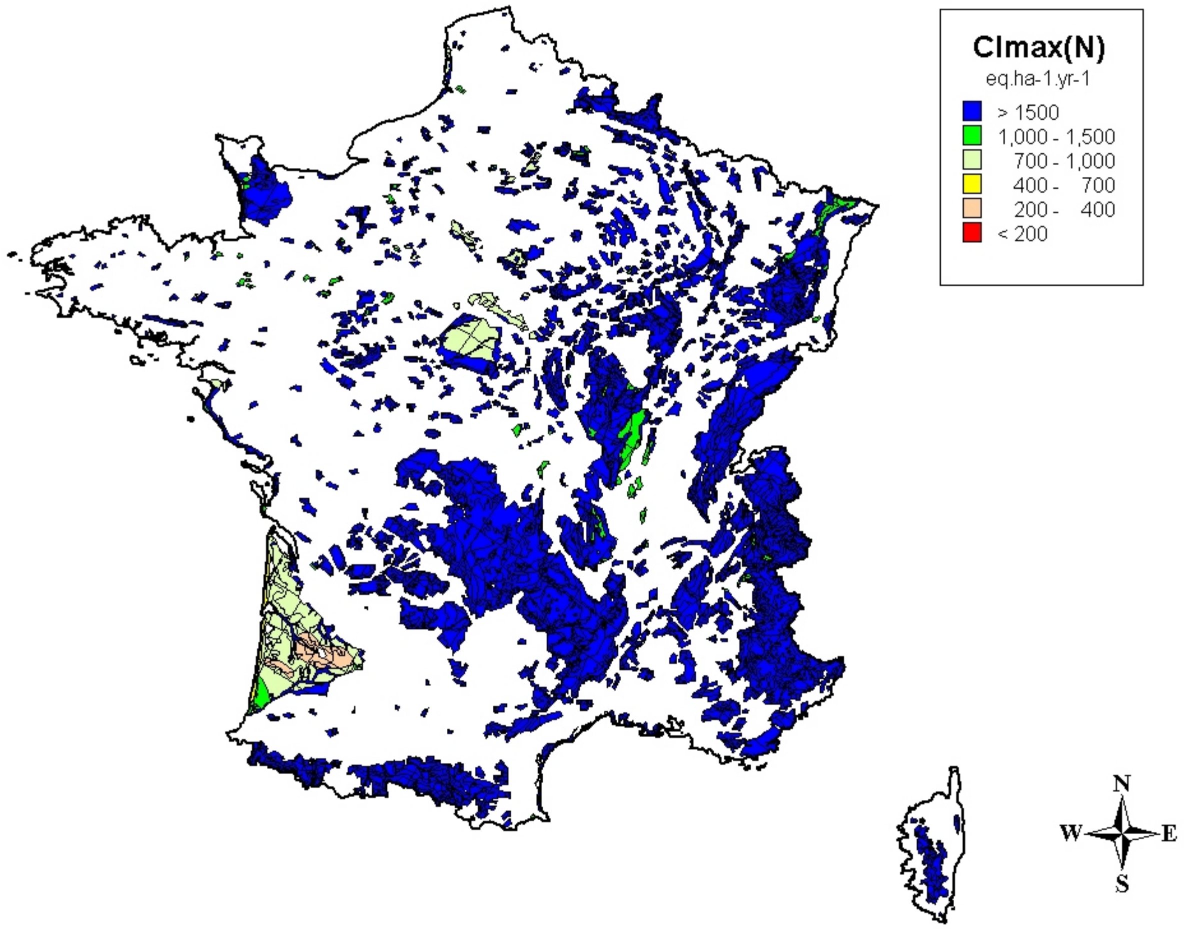


Figure FR-3. Maximum critical loads of acidifying nitrogen for French forest ecosystems (eq ha^{-1} a^{-1}).

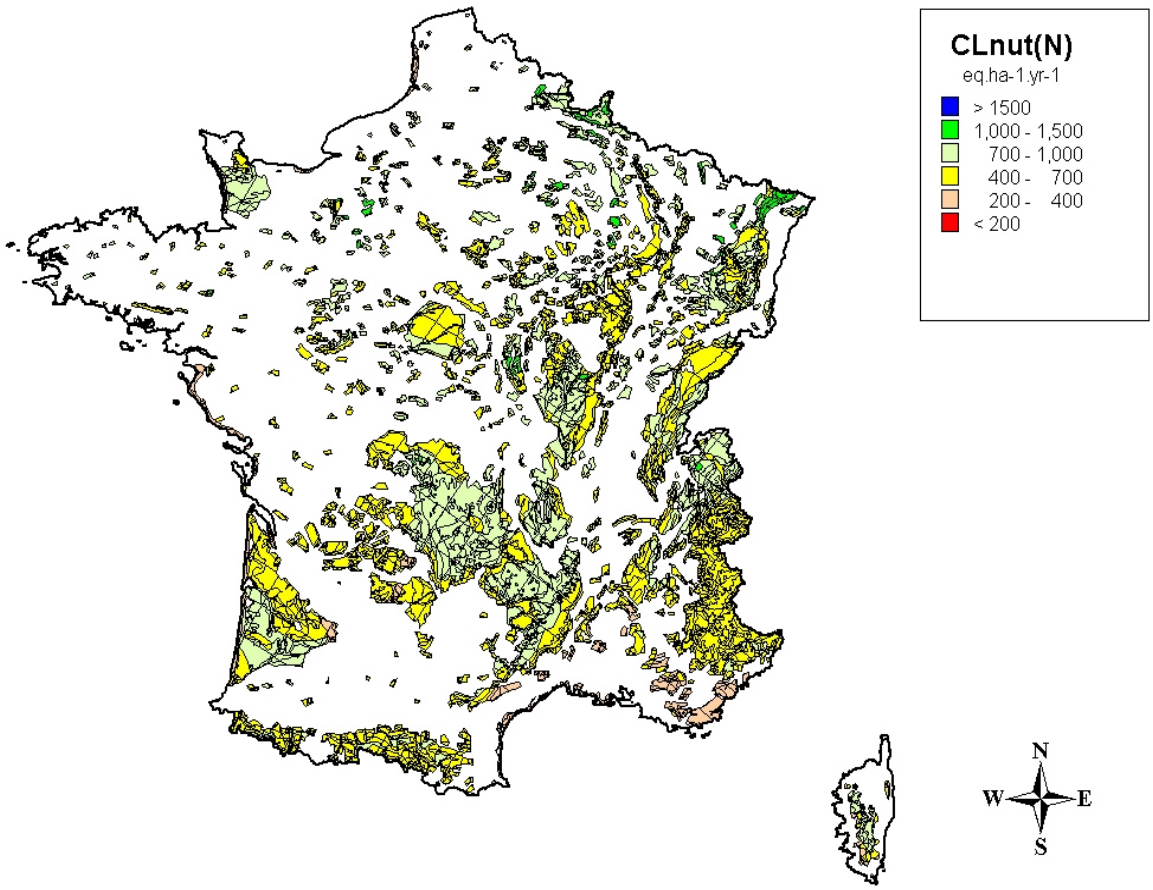


Figure FR-4. Critical loads of nutrient nitrogen for French forest ecosystems (eq ha^{-1} a^{-1}).

One the other hand, there are substantial changes between the maps of 1998 and 2003 for nutrient nitrogen in France. The main reason is that in 1998 critical loads of nutrient N were generally estimated from expert advice, whereas in 2003 they were calculated by most countries according to the methods outlined by the CCE (Posch et al. 1995).

Importance of weathering feedback for critical load dynamic modelling

Methodology: Dynamic models have been developed to simulate the evolution of soil and soil solution parameters in response to acidification/recovery scenarios. The mono-layer VSD model (Very Simple Dynamic Model, Posch et al. 2003) has been compared with the multilayer WiTCh model (Weathering at The Catchment scale, Probst et al. 2002) for adaptation to French soil conditions (see Table FR-2).

These two dynamic models were applied to two French forest ecosystems taken from the critical loads database using the same acidification scenario, as described in Figure FR-5.

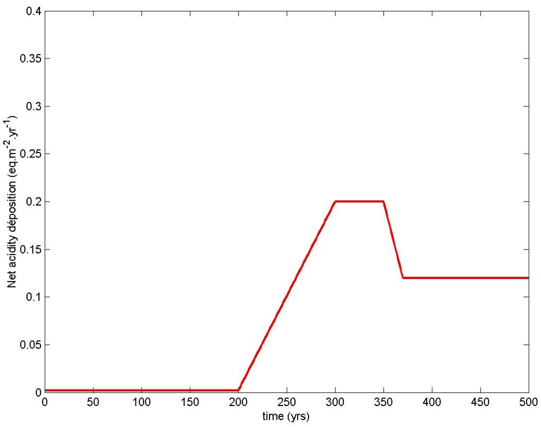


Figure FR-5. Acid deposition scenario (SO_2 only), used as input for the VSD and WiTCh models (eq m^{-2} a^{-1}).

The input parameters for modelling are given in Figure FR-6. The two sites were selected for their different sensibility to weathering:

- *Pinus maritimus,* podzosol on acid sandy material (site: Landes), characterised by a poor weatherable parent material that provides few base cations to the soil.
- *Fagus silvatica,* andosol on a basaltic bedrock (site: Massif Central) characterised by a potentially high weathering rate which provides an important amount of base cations to the soil.

Results and discussion

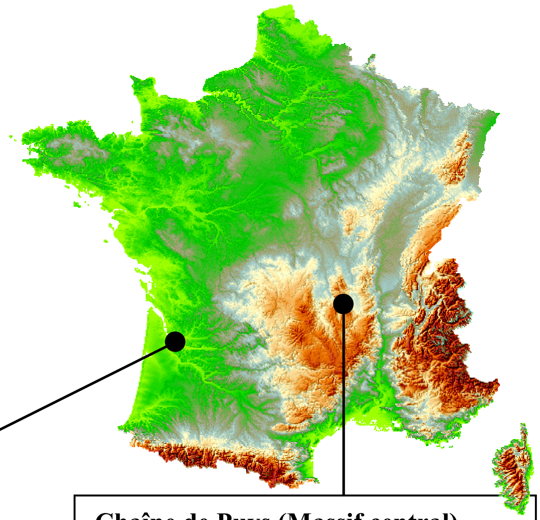
The WiTCh model calculates mineral weathering rates at each time step using kinetic laws, which vary depending on the chemical composition of the soil solution and mineral saturation. Weathering rates increase with the acidification of the soil solution, as base cations are provided to the soil system. This buffering capacity is higher where mineral weatherability is high, like in the Massif Central basaltic bedrocks, which contains a high mass percentage of weatherable minerals like apatite and anorthite.

In the VSD model, there is no feedback between the changes in soil solution properties and the weathering rate which is a constant input data for the model. Therefore, in places where buffering capacity is low, like Landes (Figure FR-7), VSD model outputs are quite similar to the WiTCh model outputs (Figure FR-8). Conversely, at sites where weathering rates are high,), VSD gives similar outputs (Figure FR-9) as for Landes, whereas WiTCh simulates a high buffering effect on pH outputs (Figure FR-10). Soil response to acidification is therefore strongly dependent on substrate lithology and the mineral weathering rate which provides base cations to the system and ensures buffering capacity.

Table FR-2. Comparison between the main processes considered by VSD and WiTCh model.

Parameter	VSD	WiTCh
Exchange equations	Gapon or Gaines-Thomas	Gapon
Weathering rate determination	No mineralogy data: weathering rates are constant input data or can be simulated as a function of pH.	Weathering fluxes determined from the mineralogy and chemical properties of soil solution with kinetic laws
Secondary minerals	–	Precipitation with thermodynamic laws
Carbon	One species, HCO_3	Complete speciation
Nitrogen	Nitrogen budget	–
Soil profile	Monolayer	Multilayer

Considered in all simulations:
- 3 layers (0–10–40–140 cm)
- DOC content (20, 2, 1 mg l^{-1})
- Reactive surface (0.5, 1.0, 1.5· 10^6 m^2 m^{-3})
- Volumetric water content (0.35, 0.30, 0.30)
- Moisture bulk density (800, 1300, 1400 kg m^{-3})
- P_{CO2} (1, 15, 15 present atmospheric level)



Les Landes
Sandy soil
Drainage: 0.4 m a^{-1}
NaCl dep: 1.14 keq ha^{-1}
93% wt Quartz
3% wt Orthose
1.5% wt Anorthite

Chaîne de Puys (Massif central)
Loess soil
Drainage: 0.5 m a^{-1}
33% wt Anorthite
26% wt Albite
< 10% wt Chlorite, Quartz, Orthose

Figure FR-6. Simulation conditions and location of two sites studied.

It appears that a simple model like VSD is not well-suited to French soils where weathering is expected to be high. The French NFC has decided to run the two dynamic models, VSD and WiTCh, on a sufficient number of sites and ecosystems. The behaviour of sites where complex medium weathering rates are known should be carefully checked. Moreover on these sites the role of glacial till on buffering capacity should be also considered with caution (Probst et al. 1995). This will allow one to choose the best appropriate dynamic model for French forest soils before submitting input and output data for dynamic modelling for the next call for data in late 2003. Work is still in progress to run the two models on sites extracted from critical loads database.

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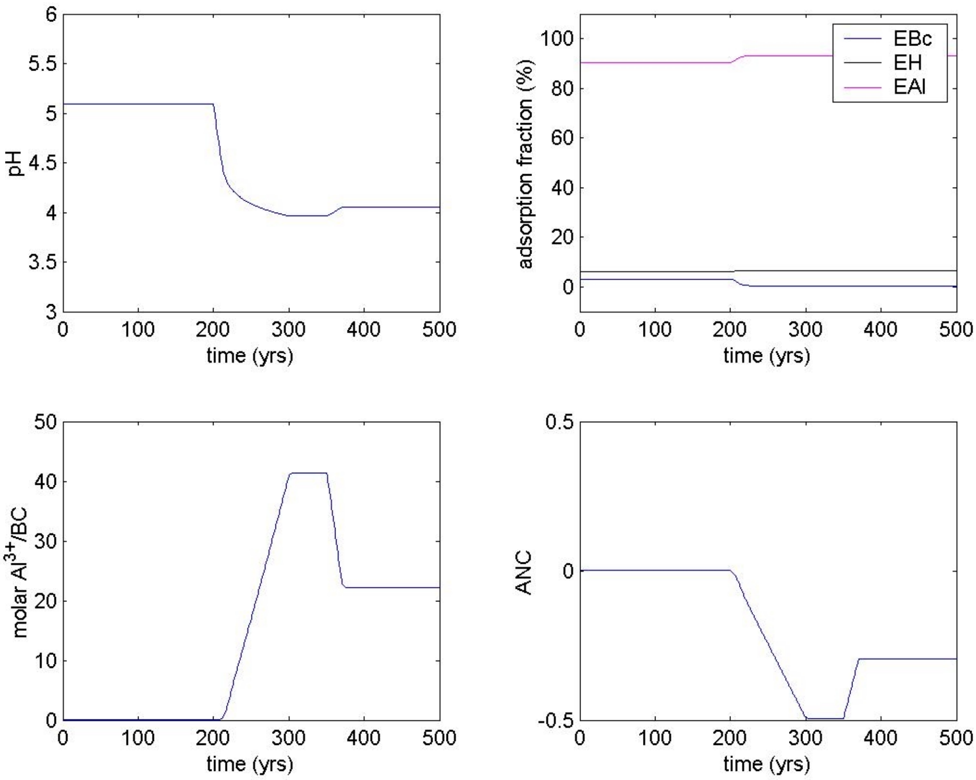

Figure FR-7. VSD model simulation on a single top layer (Landes). (a) pH evolution, (b) adsorption rates on complex, (c) Al:BC ratio and (d) ANC (acid neutralising capacity). EBc = base cation adsorption rate, EH = H adsorption rate, EAl = Al adsorption rate.

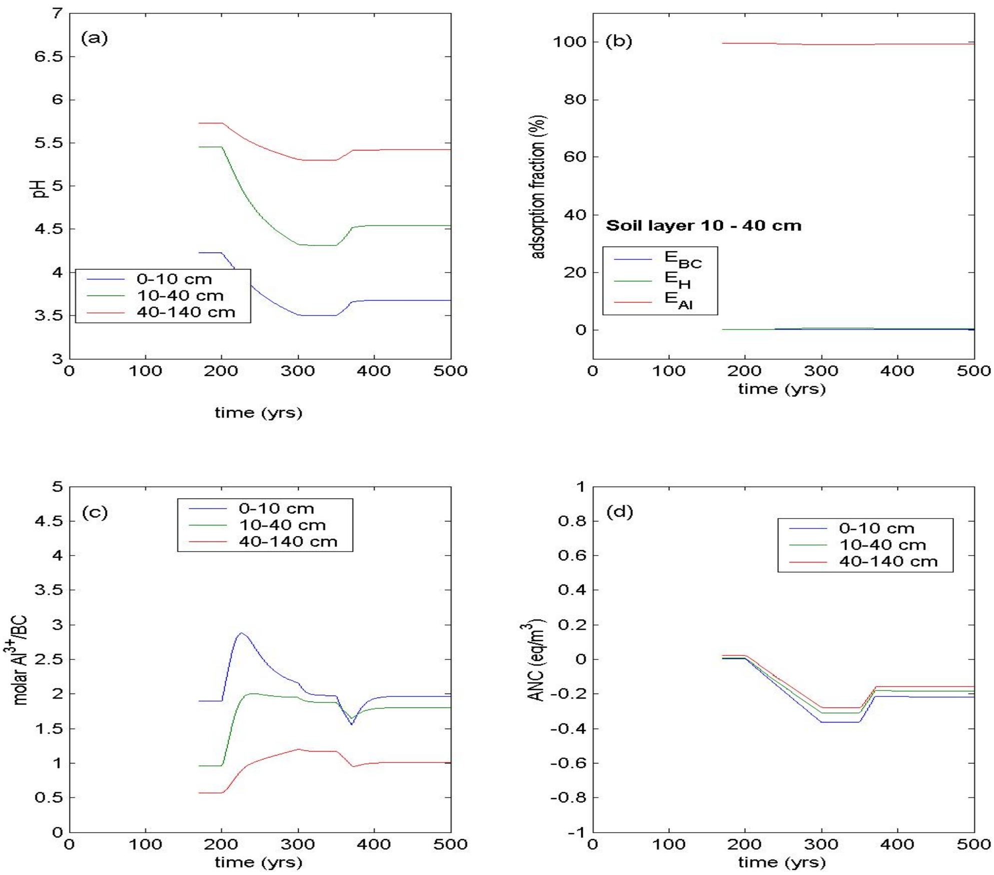

Figure FR-8. WitCH model simulation on 3 layers (Landes). (a) pH evolution, (b) adsorption rates on complex, (c) Al:BC ratio and (d) ANC (acid neutralising capacity).

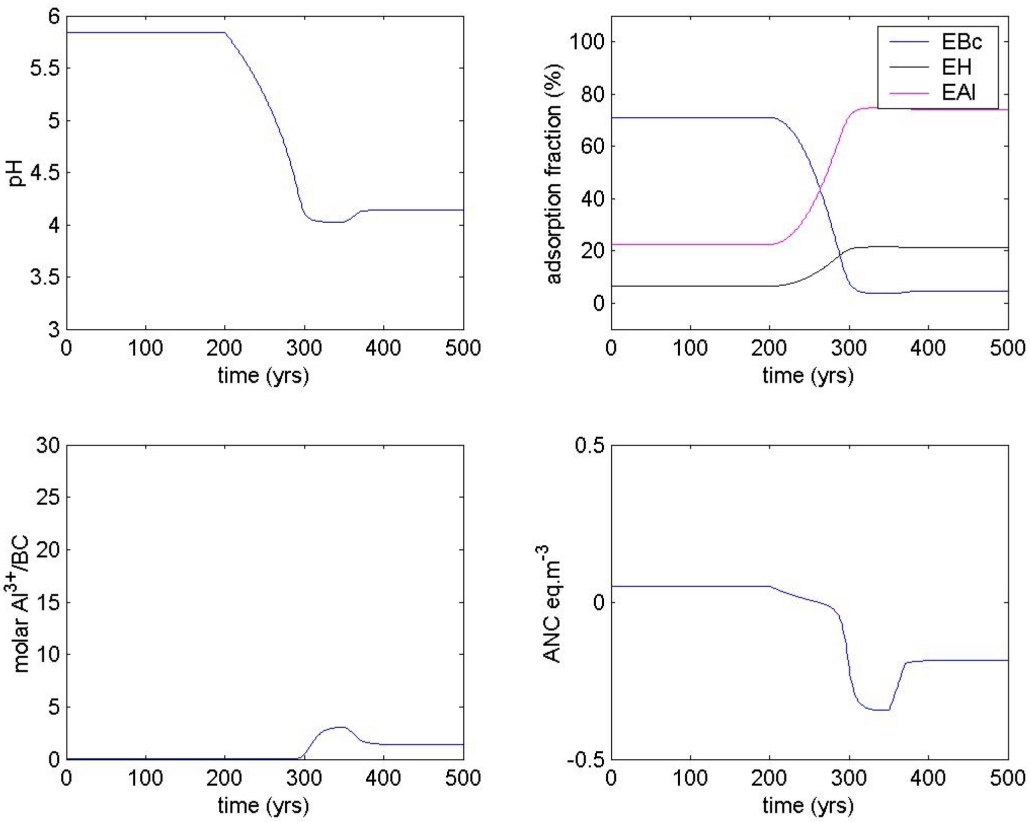

Figure FR-9. VSD model simulation on a single top layer (Massif Central). (a) pH evolution, (b) adsorption rates on complex, (c) Al:BC ratio and (d) ANC (acid neutralising capacity). EBc = base cation adsorption rate, EH = H adsorption rate, EAl = Al adsorption rate.

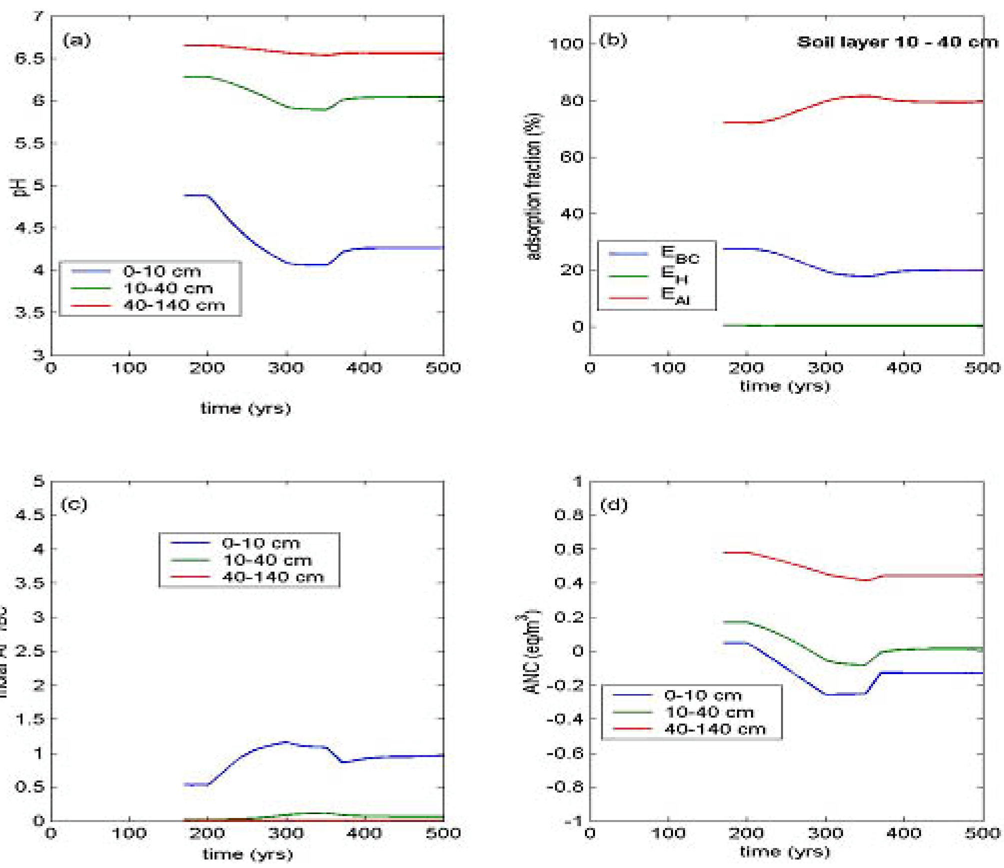

Figure FR-10. WitCH model simulation on 3 layers (Massif Central). (a) pH evolution, (b) adsorption rates on complex, (c) Al:BC ratio and (d) ANC (acid neutralising capacity).

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Calculation methods

Critical loads of acidity and nutrient nitrogen:
The German NFC has provided updated national calculations of critical loads of acidity and nutrient nitrogen (using the steady-state mass balance approach) as well as data required for dynamic modelling.

About 30% of Germany is covered by forests and other (semi-)natural vegetation for which critical loads of acidity and nutrient nitrogen have been computed (see Table DE-1). A full overview of the German receptor area using the EUNIS ecosystem classification code is available upon request from the NFC.

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: soils with high base saturation are explicitly protected (see below). The German critical load database consists of 420,624 records. A detailed description of the data and methods for their derivation 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	6.4	21.2
Coniferous forest	16.1	53.6
Mixed forest	6.4	21.3
Natural grassland	0.5	1.7
Acid fens and heathland	0.3	0.9
Wet grassland	0.1	0.3
Mesotrophic peat bogs	0.3	0.9
Total:	30.0	100

Critical loads of sulphur and nitrogen, CL(S+N):
The calculation of critical loads of sulphur and nitrogen for forest soils and other (semi-)natural vegetation was conducted according to Equation 5.17 (5.36, 5.38, 5.41, 5.42) of the Mapping Manual. For base cation deposition the values for the most recent year (1999) have been used, while for chloride deposition the 3-year mean (1997–1999) was used in order to smooth large variations of this parameter due to meteorological influences (Gauger et al. 2002).

The calculation of the net uptake of base cations and nitrogen (X_u) was conducted by multiplying the average growth rate of plants (or compartments of plants) to be harvested with the element contents (X). The base cation contents in forest biomass (Table DE-3) were revised according to new data and literature studies (Jacobsen et al. 2002). Values for base saturation of soils (Table DE-4) were also derived from literature (Klapp 1960, Schmidt et al. 1998). To describe soil parameters based on the General Soil Map of Germany ("BUEK 1000", see Hartwich et al. 1995), generally the actual rooting zone was considered.

To protect soils with high base saturation, this parameter was integrated into the estimation of critical loads. For all soil units with a base saturation >30% (42.2% of the total receptor area), the critical ANC leaching was set to zero. Without this assumption the base saturation of all soils would decrease to values near 5% within a few decades. Since the aim of the critical loads approach is to protect all ecosystems against acidification it is justified to also preserve those ecosystems adapted to a high base saturation of their soils. In this case the critical load is determined by the weathering of base cations only. As soils with a high base saturation tend to have high weathering rates and high values of ANC leaching, their critical loads decrease by using this cut-off.

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

Parameter	Term	Unit	Description (see UBA 1996)
Critical load of acidity	$CL_{max}(S)$	eq ha^{-1} a^{-1}	Mapping Manual, Eq. 5.17 (5.36, 5.38, 5.41, 5.42); if base saturation ≥ 30%, then $ANC_{le(crit)} = 0$
	$CL_{min}(N)$	eq ha^{-1} a^{-1}	Mapping Manual, Eq. 5.23.
	$CL_{max}(N)$	eq ha^{-1} a^{-1}	Mapping Manual, Eq. 5.24.
Critical load of nutrient nitrogen	$CL_{nut}(N)$	eq ha^{-1} a^{-1}	Mapping Manual, Eq. 5.21.
Uptake of base cations by vegetation	Bc_u	eq ha^{-1} a^{-1}	See Table DE-5 (forests) and NFC report in Posch et al. (2001).
Weathering of base cations	BC_w	eq ha^{-1} a^{-1}	See NFC report in Posch et al. (2001). Weighted mean for actual rooting zone, fitted to 0.5 m soil depth.
Gibbsite equilibrium constant	K_{gibb}	m^6 eq^{-2}	= 300
Acid neutralising capacity leaching	$-ANC_{(crit)}$	eq ha^{-1} a^{-1}	The minimum value of all approaches described in the Mapping Manual was used in the calculations.
Nitrogen immobilisation	N_i	eq ha^{-1} a^{-1}	Temperature-dependent. See NFC report in Posch et al. (2001).
Nitrogen uptake by vegetation	N_u	eq ha^{-1} a^{-1}	See Table DE-5 (forests) and NFC report in Posch et al. (2001).
Denitrification factor	f_{de}	–	Dependent on clay content.
Acceptable N leaching	$N_{le(acc)}$	eq ha^{-1} a^{-1}	$N_{le(acc)} = Q \cdot 10 \cdot [N]_{crit}$; $[N]_{crit} = 0.0143$ eq m^{-3}

Table DE-3. Nitrogen and base cation content in wood.

	Contents (eq t^{-1} dry mass)			
Species	**N**	**Ca**	**Mg**	**K**
Pinus sylvestris [1]	77.82	53.89	19.75	16.62
Picea abies [1]	87.10	70.36	14.81	19.69
Fagus sylvatica [1]	109.96	89.82	21.39	35.81
Quercus spec. [1]	149.93	123.25	14.81	26.86
Alnus glut. / Fraxinus exc. [2]	99.95	84.83	24.69	33.25
Betula pendula [1]	121.37	59.88	16.48	19.18
Pinus mugo [3]	82.1	47.9	18.1	15.6
Salix spec. [2]	99.95	149.7	19.75	25.57

1) Jacobsen et al. (2002).
2) Jacobsen et al. (2002), data for *Pinus nigra v. lar.*
3) De Vries et al. (1990), Kimmins et al. (1985).

Table DE-4. Assignment of 71 legend units of the General Soil Map of Germany (BUEK 1000, Hartwich et al. 1995) to two classes of base saturation of soils.

Base saturation	Legend units of BUEK 1000
<30%	1, 6, 7, 16, 17, 20, 25, 27[1)], 30, 31, 32, 33, 34, 55, 56, 57, 58, 59, 60, 61, 62, 63, 64[1)], 71
>30%	2, 3, 4, 5, 8, 9, 10, 11, 12, 13, 14, 15, 18, 19, 21, 22, 23, 24, 26, 28, 29, 35, 36, 37, 38, 39, 40, 41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, 65, 66, 67, 68, 69, 70

[1)] The legend units 27 and 64 have been assigned to the class > 30% base saturation, if the land use type is non-forested semi-natural ecosystems or deciduous forest.

To calculate base cation weathering, the soil units of the BUEK 1000 were assigned to parent material classes (see tables and further explanations in the German NFC Report in Posch et al. 2001, pp.140-143). The effective clay content was calculated and assigned to a texture class, and the respective weathering rates for each horizon (according to Annex IV of the Mapping Manual) was identified. The weighted average of the weathering rates in the actual rooting zone was then computed.

Critical loads of nutrient nitrogen

Methods to calculate critical loads of nutrient nitrogen, $CL_{nut}(N)$, are described in detail in the Mapping Manual (Eq. 5.21). According to a recent study (Jacobsen et al. 2002), mean nitrogen content of the harvested biomass of forests is considerably lower than values used in the critical load calculations until 2002 (see Table DE-3). Values for element contents in natural non-woodland vegetation have not been changed. Nitrogen immobilisation rates were ranked according to temperature, and the denitrification factor is assigned according to the clay content (see German NFC Report in Posch et al. 2001, pp. 140-143).

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

Derivation of input data for dynamic modelling

The minimum set of data needed for dynamic modelling was delivered to the CCE as requested. An overview of the parameters, including a short description of the principles for their derivation, is provided in Table DE-5. Most data are based on soil properties described for the reference profiles of the units of the General Soil Map of Germany (BUEK 1000).

This applies also to the (soil) data of more general interest. Note that the pH values refer to KCl solution. The FAO soil type is described by short codes on the basis of the Revised FAO Legend (1990), which differ slightly from the codes in FAO 1974. Short English descriptions of the soil types according to FAO (1990) were provided in addition to the data requested. Climate data were provided by German Weather Services. Both the precipitation and temperature are 30-year means (1961–1990). Data on altitude above sea level stem from the German Federal Agency for Cartography and Geodesy.

Maps of $CL_{max}(S)$ and $CL_{nut}(N)$ for Germany are presented in Figures DE-1 and DE-2.

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Table DE-5. Minimum set of data needed for dynamic modelling.

Parameter	Term	Unit	Description [1)]
Depth of rooting zone	*thick*	m	80% of the actual rooting depth, which depends on soil and vegetation type (compared to potential rooting depth, which is a function of soil properties).
Bulk density	ρ	g cm^{-3}	Weighted mean over the depth of rooting zone.
Volumetric water content at field capacity	θ	m^3 m^{-3}	Weighted mean over the depth of rooting zone.
Cation exchange capacity	*CEC*	meq kg^{-1}	$CEC_{pH6.5}$ = 10 (0.5· clay[%] + 0.05· silt[%] + 0.8 · 2 · humus[%]) after AG Boden (1994), with factor 0.8 to calculate the effective CEC of humus at pH 6.5
Base saturation	E_{BC}	–	Typical values for stands without influence of acidifying depositions. The acidifying influence of coniferous needles is considered.
Year for which EBC was determined	$yearE_{BC}$		Reference year is 1960, based on Klapp (1965), Wolff and Riek (1996).
Initial amount of carbon in topsoil	C_{pool}	g m^{-2}	Weighted mean over all layers, for which a C content was described in BUEK 1000 and in Baritz (1996): in woodlands: Oh + Ah (+ Bh); in semi-natural non-woodlands: Ah (+ Bh)
Initial C:N ratio in the topsoil	CNrat0		Reference year is ca. 1960, after Wolff and Riek (1996), Konopatzki and Kirschner (1997).

1) all parameters describing soil properties are based on the description of reference profiles of the General Soil Map of Germany (BUEK 1000).

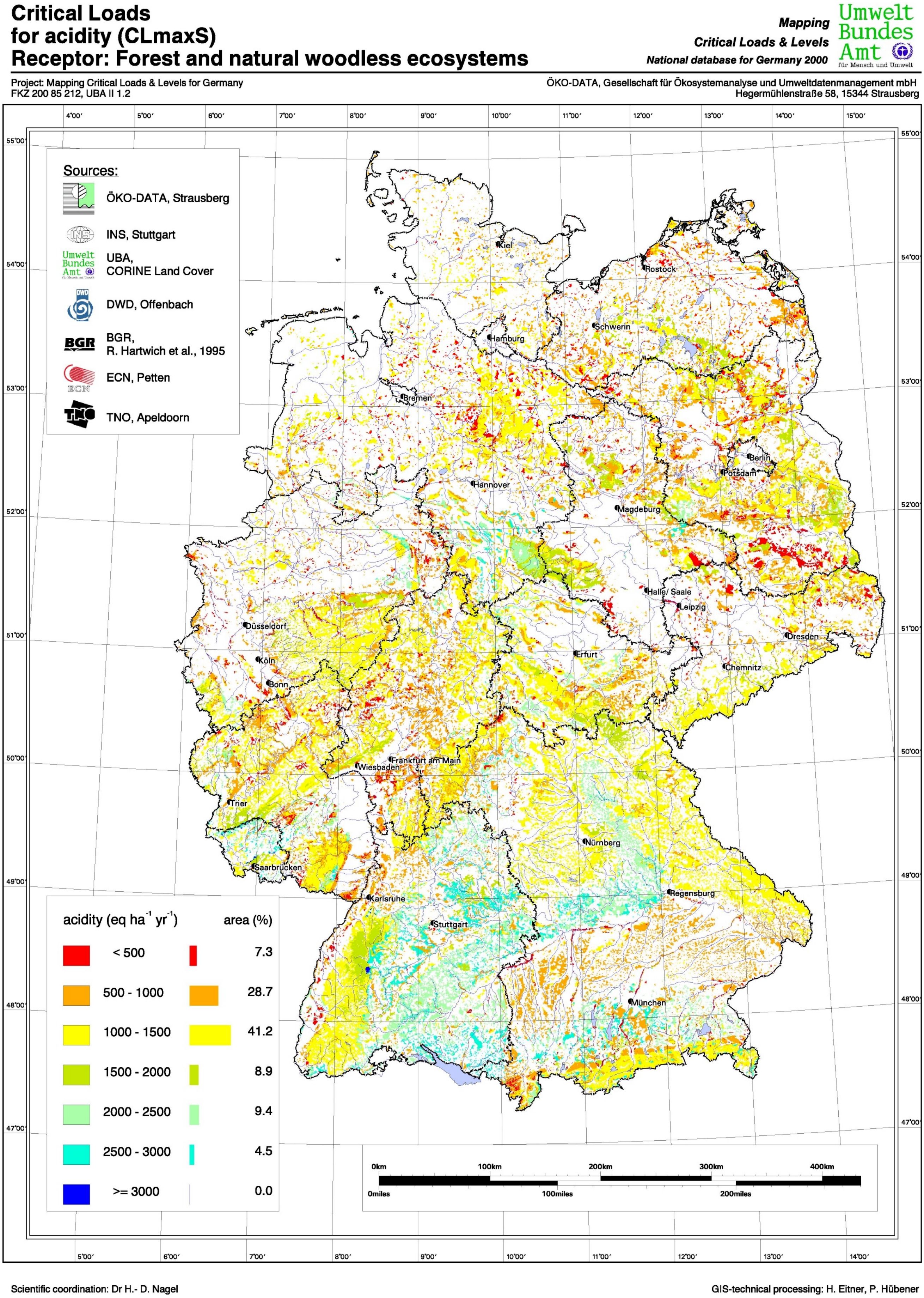

Figure DE-1. Critical loads of acidity (in eq ha^{-1} a^{-1}).

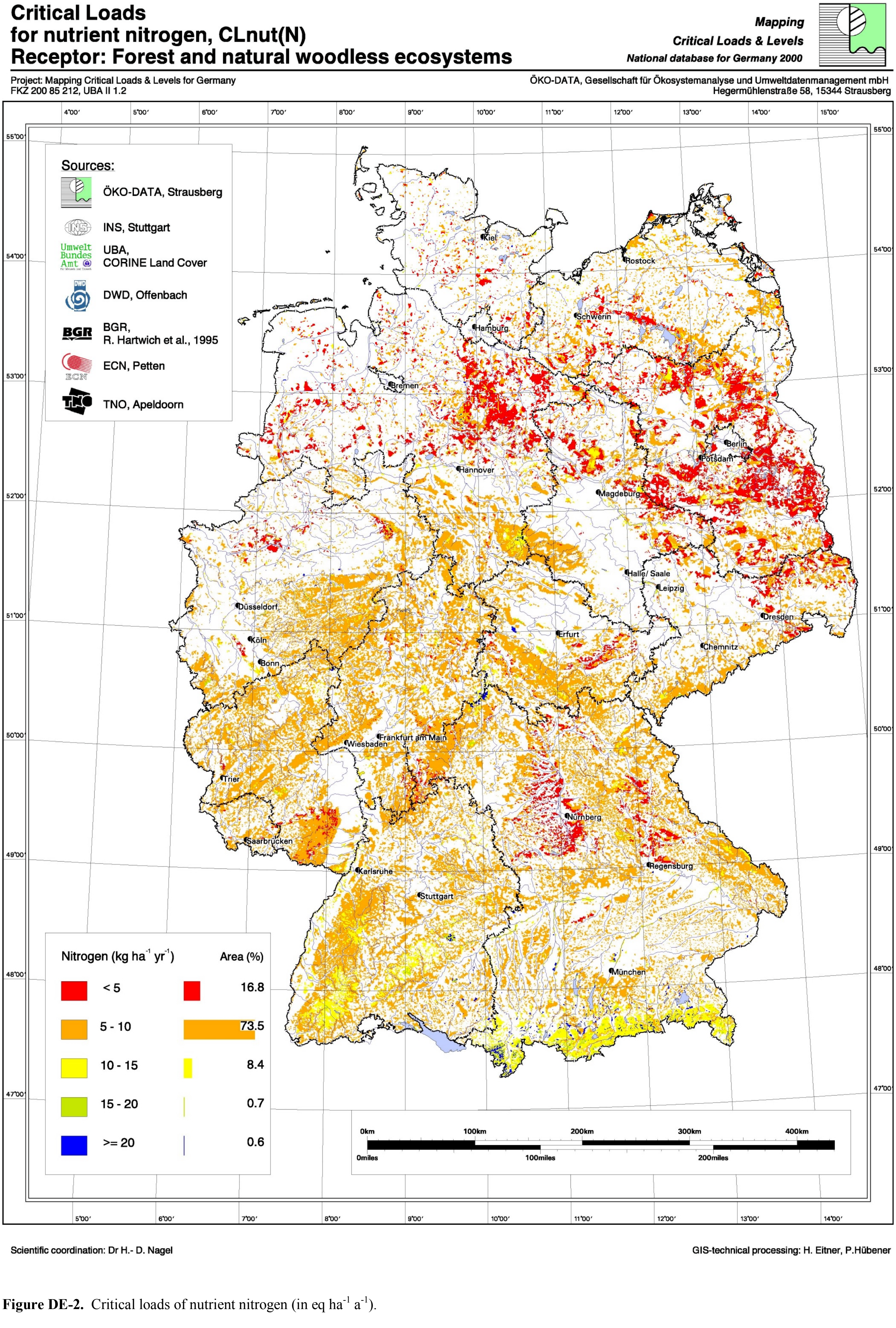

Figure DE-2. Critical loads of nutrient nitrogen (in eq ha^{-1} a^{-1}).

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Calculation methods

The Hungarian National Focal Centre provided critical load data to the CCE for two forest ecosystem types. Critical loads of acidity for forests ecosystems were calculated using dynamic modelling and SMB approach, implemented in the SAFE/MAKEDEP models. Critical ANC leaching was determined at the critical Al:Bc molar ratio. Critical load functions were calculated according to the Mapping Manual (UBA 1996).

Dynamic modelling exercise

Simulations were performed for deciduous and coniferous forest ecosystems. Input data required for modelling on nutrient content of trees were estimated at 14 forest catchments, while biomass growth parameters from the Michealis-Menten model were uniform throughout the country. Mineralisation rates were set to 20% in all ecosystem types, while a 15% litter mineralisation rate was used in coniferous and 90% in deciduous forests (Table HU-1). Ages of forests ranged from 31 to 96 years; catchments were established at unmanaged sites (see Table HU-2). Wet and throughfall deposition data were measured directly at each site. The soil profile was divided into five layers in the SAFE model. For the soil variables CEC, base saturation, bulk density, and soil texture were used as measured parameter, while CO_2 pressure and DOC were according to literature values. Base saturation varied between 64–100% on the forest catchments. Weathering rates appropriate to Hungarian soil types were used according to Sverdrup (1990) and Holmqvist (2001). $CL_{nut}(N)$, $CL_{max}(S)$, $CL_{max}(N)$ and $CL_{min}(N)$ have been calculated according to the Mapping Manual (UBA 1996).

Data sources

The analyses were conducted on the data gathered by the Hungarian Institute for Forestry. Mineralogy was converted from the Map of Clay Mineral Associations in Hungarian Soils (Stefanovits and Dombovarine 1986).

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Table HU-1. Nitrogen and base cation content in tree compartments.

		Contents (g $100g^{-1}$ dry weight)											
		Canopy				Branch				Stem			
Site	Species	N	K	Ca	Mg	N	K	Ca	Mg	N	K	Ca	Mg
1	*Fagus silvatica*	2.71	0.73	1.12	0.15	0.66	0.11	0.98	0.04	0.25	0.08	0.65	0.02
2	*Picea abies*	1.37	0.62	1.31	0.08	0.54	0.10	0.53	0.02	0.08	0.04	0.35	0.01
3	*Quercus petraea*	2.81	0.84	1.05	0.17	0.49	0.19	1.20	0.03	0.19	0.14	0.79	0.01
4	*Pinus silvestris*	2.08	0.47	0.84	0.17	0.34	0.07	0.66	0.02	0.05	0.05	0.44	0.01
5	*Pinus silvestris*	1.82	0.42	1.03	0.17	0.41	0.05	0.49	0.02	0.06	0.04	0.32	0.01
6	*Pinus nigra*	1.46	0.55	0.42	0.18	0.39	0.05	0.47	0.02	0.30	0.02	0.31	0.01
7	*Quercus robur*	3.34	0.82	1.00	0.28	0.81	0.23	1.74	0.08	0.31	0.17	1.15	0.04
8	*Quercus robur*	3.19	0.92	1.22	0.28	0.76	0.22	1.32	0.08	0.29	0.17	0.87	0.04
9	*Fagus silvatica*	2.51	0.78	1.47	0.22	0.45	0.14	0.95	0.04	0.17	0.11	0.63	0.02
10	*Quercus petraea*	2.86	0.80	0.75	0.22	0.65	0.17	0.93	0.06	0.25	0.13	0.61	0.03
11	*Pinus silvestris*	1.60	0.41	0.59	0.10	0.45	0.07	0.63	0.03	0.07	0.05	0.42	0.02
12	*Quercus petraea*	2.38	0.83	0.86	0.21	0.51	0.16	1.01	0.04	0.19	0.12	0.67	0.02
13	*Fagus silvatica*	2.58	0.72	1.12	0.22	0.48	0.15	1.25	0.05	0.18	0.11	0.83	0.02

Table HU-2. Forest catchments corresponding to ecosystem types.

Longitude (UTM, m)	Latitude (UTM, m)	Altitude (m)	Ecosystem type	Age (yrs)
475331	195731	560	Dec	89
475338	195705	560	Con	35
475146	195801	660	Dec	63
473327	192326	240	Con	35
465801	193312	120	Con	30
465746	193252	120	Con	62
472029	210542	90	Dec	72
472024	210546	90	Dec	67
473920	162903	460	Dec	96
474007	163011	470	Dec	96
464924	162417	260	Con	48
464920	162415	260	Dec	72
463600	164747	240	Dec	69
465757	193318	120	Dec	31

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Calculation methods

The maximum critical loads of sulphur, minimum critical loads of nitrogen, maximum critical loads of acidifying nitrogen and critical loads of nutrient nitrogen have been estimated for four ecosystem types: coniferous forests (2449 km^2), deciduous forests (1805 km^2), natural grasslands (2050 km^2) and heathlands (2631 km^2). The calculation methods are in accordance with the Mapping Manual (UBA 1996):

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

where

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

The weathering rate of base cations (BC_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 BC_w has been produced (Aherne and Farrell 2000a,b). The critical leaching of acid neutralising capacity (ANC_{crit}) is calculated as described by Hettelingh et al. (1991), p. 35. 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 (Aherne et al. 2001, Aherne and Farrell 2002). The H^+ critical limit of pH = 4.2 is based on work by Ulrich (1987).

$$CL_{min}(N) = N_u + N_i + N_{de}$$
$$CL_{max}(N) = CL_{min}(N) + CL_{max}(S)$$

The empirical approach (UBA 1996) has been used to calculate critical loads of nutrient nitrogen for deciduous forests (1070 eq ha^{-1} a^{-1}), natural grasslands (1070 eq ha^{-1} a^{-1}), and heathlands (1070 eq ha^{-1} a^{-1}). For coniferous forest ecosystems the critical loads of nutrient nitrogen was estimated as the minimum of the mass balance and empirical approach, where the mass balance was estimated as:

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

The empirical value for coniferous forests is set at 1070 eq ha^{-1} a^{-1}.

A detailed description of the data and methods is given in Table IE-1. For further discussion on the data sources and methods see Aherne and Farrell (2000a,b, 2002) and Aherne et al. (2001).

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).

Altitude: Digital Elevation Model (DEM) derived from the twenty-five 1:126,720 Ordnance Survey maps of Ireland (contour intervals of 100 feet).

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

Precipitation surplus: Precipitation minus evapotranspiration and surface runoff. Evapotranspiration is estimated from interpolation (kriging) of long-term mean annual evapotranspiration volume, 1951–1980 (14 sites, see Aherne and Curtis 2003). Surface runoff is inferred from soil permeability classes derived from the general soil map of Ireland.

Table IE-1. Irish critical load database: calculation methods and data sources.

Parameter (units)	EC[1]	Value[2] (min–max)	Calculation method	Data source
$CL_{max}(S)$ ($eq\ ha^{-1}\ a^{-1}$)	C	180–5657	$= BC_w + ANC_{crit} + BC_{dep} - Bc_u$	Method source: UBA (1996).
	D	314–6186		
	G	292–5882		
	H	341–5964		
$CL_{min}(N)$ ($eq\ ha^{-1}\ a^{-1}$)	C	524–739	$= N_u + N_i + N_{de}$	Method source: UBA (1996).
	–	235–500		
$CL_{max}(N)$ ($eq\ ha^{-1}\ a^{-1}$)	C	751–6210	$= CL_{min}(N) + CL_{max}(S)$	Method source: UBA (1996).
	D	814–6520		
	G	792–6223		
	H	840–6392		
$CL_{nut}(N)$ ($eq\ ha^{-1}\ a^{-1}$)	C	759–954	Minimum of empirical (1070) and mass balance ($=N_u + N_i + N_{de} + N_{le}$) approaches.	Method source: UBA (1996).
	–	1070	Empirical approach.	
BC_{dep} ($eq\ ha^{-1}\ a^{-1}$)	C	102–728	Interpolation of mean annual bulk precipitation concentrations for ~20 sites (1985–1994); filter factor of 2.0 for forests and 1.5 for heathlands.	Aherne and Farrell (2000b).
	D	102–652		
	G	54–392		
	H	87–590		
Bc_u ($eq\ ha^{-1}\ a^{-1}$)	C	179–390	Minimum of available base cations ($=Bc_w + Bc_{dep} - Bc_{le}$) and estimated uptake (= yield class × wood density × stem concentration).	COFORD (1996). Emmett and Reynolds (1996).
	–	45	Removal via occasional grazing.	
BC_w ($eq\ ha^{-1}\ a^{-1}$)	–	100–4000	Skokloster classification: the mid-value of each of the five classes is used to define soil weathering, except for the final (non-sensitive) class, which is set at 4000.	Nilsson and Grennfelt (1988). Hornung et al. (1995a). Aherne and Farrell (2000a).
Q ($mm\ a^{-1}$)	–	110–2278	Precipitation minus evapotranspiration and surface runoff.	Met Éireann.
K_{gibb} ($m^6\ eq^{-2}$)	–	9.5–300	Organic soils: 9.5 Peaty podzol and gley soils: 100 Remaining soils: 300	Gardiner and Radford (1980). UBA (1996).
$ANC_{le(crit)}$ ($eq\ ha^{-1}\ a^{-1}$)	C	155–2762	$= Q\cdot([Al]_{crit} + [H]_{crit})$ A pH of 4.2 was selected for $[H]_{crit}$ and $[Al]_{crit}$ estimated via the gibbsite relationship.	UBA (1996). Ulrich (1987).
	D	152–3081		
	G	183–3154		
	H	188–3154		
N_i ($eq\ ha^{-1}\ a^{-1}$)	–	143–214	Organic and podzolic soils:143 Remaining soils: 214	Gardiner and Radford (1980). Hornung et al. (1995). Downing et al. (1993).
N_u ($eq\ ha^{-1}\ a^{-1}$)	C	142–310	See Bc_u comments.	COFORD (1996). Emmett and Reynolds (1996).
	–	71	Removal via occasional grazing.	
N_{de} ($eq\ ha^{-1}\ a^{-1}$)	–	71–214	Organic and gley soils: 214 Remaining soils: 71	Gardiner and Radford (1980) Hornung et al. (1995)
$N_{le(acc)}$ ($eq\ ha^{-1}\ a^{-1}$)	–	214	All ecosystems: 214	Hornung et al. (1995)

[1] Ecosystem codes (EC): coniferous (C), deciduous (D), natural grasslands (G) and heathlands (H). Note: '–' indicates all ecosystems or remaining ecosystems in parameter category.
[2] A range is not shown when minimum and maximum values are equal.

Table IE-1 (continued). Irish critical load database: calculation methods and data sources.

Parameter (units)	EC[1]	Value[2] (min–max)	Calculation method	Data source
Thick (m)	–	0.05–1.22	Derived from the modal profile of the principal soil for each of the forty-four soil associations on the general soil map of Ireland. Bulk density and volumetric water content were estimated using transfer functions.	General soil map of Ireland and soil survey bulletin (Gardiner and Radford 1980).
ρ (g cm^{-3})	–	0.06–1.52		
θ (m^3 m^{-3})	–	0.10–0.91		
CEC (meq kg^{-1})	–	62–888		
EBC (%)	–	21–98		
yearEBC	–	1980		
Cpool (g m^{-2})	–			
CNrat0	–			
clay (%)	–	8–60		
sand (%)	–	3–38		
Corg (%)	–	1–24		
pH	–	3.4–7.8		
Prec (mm a^{-1})	–	726–3517	Interpolation of long-term mean annual values (1951–1980).	Fitzgerald (1984).
Temp (°C)	–	8.7–10.5		Keane (1984).
Alt (m)	–	15–1041	Digitised from published maps.	Ordnance Survey of Ireland.

Temperature: Interpolation (kriging) of long-term mean annual temperature for approximately 60 sites for the period 1951–1980 (Keane 1984).

***Base cation deposition**:* Interpolation (kriging) of mean 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 2.0 for forests and 1.5 for heathlands (Aherne and Farrell 2000b).

Nutrient uptake: It is assumed that all coniferous trees are Sitka spruce; yield class is the average for Sitka spruce in Ireland (COFORD 1996) and stem concentrations are for Sitka spruce in Wales (Emmett and Reynolds 1996).

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Calculation methods

The methods for calculating critical loads of acidity and of nutrient nitrogen have been previously described in the NFC report of Italy (in Posch et al. 1999) except for the critical load of acidity, $CL(A)$, is now calculated using the Steady-state Mass Balance methodology.

Table IT-1 provides an overview of the data methods and sources used to calculate critical loads. Table IT-2 provides details on the runoff values used for various ecosystem types.

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Table IT-1. Data methods and sources.

Critical load parameter (units)	Ecosystem type[1]	Data sources/Methods used	Justification
$CL(A)$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= BC_w + ANC_{le(crit)}$	Mapping Manual (UBA 1996).
$CL_{max}(S)$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= CL(A) + BC_{dep} - BC_u$	Mapping Manual.
$CL_{min}(N)$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= N_u + N_{le(crit)}$	Mapping Manual.
$CL_{max}(N)$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= CL_{min}(N) + CL_{max}(S)$	Mapping Manual.
$CL_{nut}(N)$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= N_i + N_u + N_{le} / (1 - f_{de})$	Mapping Manual.
Bc_u (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	From Italian experts.	Based on volume increment basic wood density concentration in wood (Bonanni et al. 2001).
BC_{dep} (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= \begin{cases} 2 \cdot BC_{wet} & \text{where } BC_{wet} < 250 \\ 250 + BC_{wet} & \text{where } BC_{wet} \geq 250 \end{cases}$	Downing et al. 1993, Annex II.
BC_w (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	Based on soil type. For soils with very high BC_w, the default mean value 8896 was used.	Mapping Manual.
$ANC_{le(crit)}$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= \left(1.5 \cdot \frac{Bc_{le}}{K_{gibb}}\right)^{1/3} \cdot Q^{2/3} + 1.5 \cdot Bc_{le}$ with $Bc_{le} = 0.8BC_w + BC_{dep} - Bc_u - 0.015Q$	Mapping Manual.
N_u (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	From Italian experts.	Based on volume increment basic wood density concentration in wood (Bonanni et al. 2001).
$N_{i(crit)}$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	$= N_i + N_{fire} + N_{vol} - N_{fix}$ where: N_i = N immobilised in soil organic matter; N_{fire} = N losses in smoke; N_{vol} = N losses via NH_3 volatilisation; N_{fix} = N fixed by biological fixation	Mapping Manual.
$N_{le(acc)}$ (eq ha^{-1} a^{-1})	mf, ag, td, bf, t, tc	Based on runoff values as listed in Table IT-2.	Bonanni et al. 2001.
f_{de}	mf, ag, td, bf, t, tc	Based on soil type and soil texture.	Mapping Manual.
Q (mm a^{-1})	mf, ag, td, bf, t, tc	$= P - E - R$, where: P = precipitation; E = evapotranspiration; R= surface runoff	Bonanni et al. 2001.
K_{gibb} (m^6 eq^{-2})	mf, ag, td, bf, t, tc	Mean default value for soils with poor organic matter.	Mapping Manual.

1. Ecosystem Codes: mf = Mediterranean forest; ag = acid grassland; td = temperate deciduous; bf = boreal forest; t = tundra; tc = temperate coniferous

Table IT-2. Ranges of runoff and $N_{le(acc)}$ values (eq N ha^{-1} a^{-1}).

	Ecosystem Type					
Q **(runoff) (mm a^{-1})**	**Tundra**	**Boreal forest**	**Temperate coniferous forest**	**Temperate deciduous forest**	**Mediterranean forest**	**Acid Grassland**
0 – 300	7	143	71	71	36	71
300 – 600	54	179	125	125	45	107
600– 900	107	214	179	179	–	143
900 – 1100	–	250	–	–	–	179
1100 – 1300	–	286	–	–	–	–

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Calculation of critical loads

In evaluating the Dutch acid rain abatement strategies in 2002, critical loads were computed for:

1. Ground water quality, protecting 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).
5. Updated values for precipitation over the period 1970–2000 instead of 1950–1980, bulk density, soil moisture content and organic matter content.

Due to these changes in input data, the maps that were submitted to CCE in 2001 and reported in the CCE Status Report 2001, deviate slightly from the newly submitted critical loads. Although new values for the (initial) pH have also been submitted, for the recalculated critical loads (SMB) the 2001 values for the pH were used.

Data derivation for the dynamic soil models VSD and SMART

Limitations of the approach:
Calculations with the dynamic models VSD and SMART allow comparisons with critical load calculations by the SMB model including critical loads related to: (i) forests (soils) against nutrient imbalance due to elevated foliar N content (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) and (ii) groundwater quality, protecting against contamination by nitrate (critical N load) and Al (critical acid load). We do not consider a detailed N cycle in the VSD model and therefore, we cannot calculate critical loads related to plant species composition in terrestrial ecosystems against eutrophication (critical N load) and acidification (critical acid load). This is even more so for the plant species composition in fens against eutrophication (critical N load). Thus the VSD results are not related to the critical loads in the Netherlands that are considered most important.

Spatial resolution and vegetation types and soil types distinguished:
As with the SMB model, both VSD and SMART perform calculations at a 250×250m^2 grid scale. Specification of the vegetation-soil combination in each 250×250m^2 grid was 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 sixteen major soil types were distinguished. For vegetation types, we distinguished three groups of tree species (deciduous forests, pine forests and spruce forests), grassland and heathland. Table NL-1 describes the ecosystems for which data were derived.

Table NL-1. Vegetation types for which calculations can be carried out

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*.

Soil types were differentiated into sixteen major groups, including two non-calcareous sandy soils and one calcareous sandy soil, three loess soils, four non-calcareous clay soils, a calcareous clay soil and five peat soils (Van der Salm 1999). All these soil types were further sub-divided 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 VSD and SMART. More information on the soil types distinguished is given in Table NL-2.

Derivation of data

For both critical load calculations and dynamic modelling, 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.

Base cation deposition: Bulk deposition data for base cations for a 1km by 1km 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 sea-salt 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 sea-water, namely 1.165 for Cl and 0.116 for SO_4. Both Cl and sea-salt SO_4 were subtracted from the total base cation deposition values to derive seasalt-corrected base cation inputs.

Weathering of base cations: Base cation weathering rates for non-calcareous sandy soils were taken from De Vries (1994), who derived weathering rates on the basis of one-year batch experiments that were scaled to field observations. Weathering rates for calcareous soils were derived from De Vries et al. (1994a). For the distinguished loess, clay, and peat subsoil types, weathering rates 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: Uptake rates of nitrogen and base cations were calculated based on the concept of nutrient-limited uptake, which is defined as that uptake that can be balanced by a long-term supply of base cations. This amount, referred to as the critical base cation uptake, 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 eq ha^{-1} a^{-1} for Ca and Mg and 0 eq ha^{-1} a^{-1} for K. From the critical base cation uptake, the corresponding critical N uptake is 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 long-term 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), with 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 eq ha^{-1} a^{-1}. These values correspond well with a range of between 2 and 5 kg ha^{-1} a^{-1} mentioned in the Mapping Manual (UBA 1996).

Nitrogen immobilisation: The long-term 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 ha^{-1}) in this layer is calculated by multiplying the thickness of the soil layer (0.3 m), with 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

Table NL-2. Soil groups.

Major soil group	Dutch code	Major FAO type Name	Code	Examples of major soil types included (by Dutch soil code)
Sandy soils:				
–Poor sandy soils	SP	Cambic Podzols	PZb-4ab	zV,zWp,iV,iWp,Y,Hn,Hd,pZn,tZd,cZd,Zn,Zd,Zb,MZz, FG,G,ABH,AD,AS,AZ,AAK,AVk
–Rich sandy soils	SR	Umbric Gleysols	GLu-4a	uV,Wp,uWp,iWz,uWz,zWz,EZg,bEZ,zEZ,cY,cHn,cHd ,pZg,BZn,BZd,MZk,ABz,AFz,AK,AR,AQ, AM +SP23
–Calcareous sandy soils	SC	Calcaric Arenosols	ARc-4b	EZ,pZg,Zn,Zd,Zb,Sn -A
Loess soils:				
–Sandy loam	L1	Unbric Gleysols	GLu-5a	Ln5,Ld5,Lh5,pLn5,Ldd5,Ldh5,Ln5,Lnd5
–Silty loam	L1B	Haplic Luvisols	LVh-5a	BLd5,BLn5,BLh5,EL5
–Clayey loam	L2	Haplic Luvisols	LVh-5a	BLd6,BLb6,Ld6,Ldh6,BLh6,BLn6,Lh6,Ldd6
Clay soils:				
–Sandy clay	C1	Eutric Fluvisols	Fle-7a	pMv51,pMo50,pRn59,KRn1,KRn2,KRd1,KRd7,BKd2 5,BKd26
–Light clay	C2	Mollic Fluvisols	Flm-7a	pRv81,pRn86,KRn8,pRn89,AEk9
–Heavy clay	C3	Mollic Fluvisols	Flm-8a	pMv81,pMo80, AEm5,AEm8,'AWo'
–Very heavy clay	C4	Eutric Fluvisols	FLe-9a	KK,KM,KS,ALu, Rv01C,Rn44C,Rn14C
–Calcareous clay	CC	Calcaric Fluvisols	FLc-7a	MOo,MOb,ROb,AO,AWg, pMn,Mv,Rv,Mo,Ro,Mn,MOo,MOb,ROo,ROb,Rn,Rd,A Em,AEp -A
Peat soils:				
–Peat	P1	Terric Histosols	HSs-1a	aVc,aVz,aVp,Vz,Vp
–Light clayey peat	P2	Terric Histosols	HSs-1a	hVz,kWp,Vc,Vb,Vd,Vk,Vr
–Clayey peat	P3A	Terric Histosols	HSs-7a	hVc,hVk,hEV,hVd,hVr,kVd,kVr
–Peaty clay	P3B	Terric Histosols	HSs-8a	kWz,pVz
–Peaty heavy clay	P4	Terric Histosols	HSs-9a	pVs,kVz,kVb,kVs,kVc,kVk,pVk

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 eq ha^{-1} a^{-1}. These values correspond well with the of 2 and 5 kg ha^{-1} a^{-1} mentioned in the Mapping Manual (UBA 1996).

Denitrification: Denitrification fractions were derived for each soil type based on data 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).

Runoff: Runoff was calculated as the difference between precipitation and evapotranspiration. Precipitation estimates have been derived from an overlay with 30-year average (1970–2000) results of 280 weather stations in the Netherlands. Interception fractions, relating interception to precipitation, have been derived from literature data 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).

Al release constants: Al release is not described in VSD and SMART (and also not in SMB) by using the gibbsite equilibrium approach but by using a general formula relating Al to protons as described in the mapping manual (Posch et al. 2003):

$$[Al] = KAl_{ox} \cdot [H]^{\alpha} \qquad (1)$$

where α is a soil type-dependent exponent. For α=3 this is the familiar gibbsite equilibrium.

For both the SMB and VSD application an empirical relation between Al and H concentrations was constructed using 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 have been used (Leeters et al. 1994, Klap et al. 1999). For these sites Al^{3+} activities were calculated from the total concentration of Al and dissolved organic carbon (DOC) using the speciation software 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 of KAl_{ox} and α is given in Table NL-3.

Table NL-3. Overview of the values of KAl_{ox} and α used to calculate critical acidity leaching.

Soil type	Vegetation type	Soil depth (cm)	Log KAl_{ox} [log $(mol^{-2}\ l^2)$]	α (–)
Sand	Forest	60–80	5.20	2.51
Sand	Grass/Heath	30	2.14	1.88
Clay	Forest	100	7.88	2.65
Clay	Grass/Heath	30	7.88	2.65
Loess	Forest	100	4.55	2.17
Loess	Grass/Heath	30	3.29	1.90
Peat	Forest	50	-1.06	1.31
Peat	Grass/Heath	30	-1.06	1.31

Derivation of additional soil data needed for dynamic modelling

Data sets that were used to derive the various soil data include:

- 12 forest stands on non-calcareous sandy soils sampled in 1992: the humus layer and the depths of 0–10, 10–30, 30–60 cm and 60–100cm (De Vries et al. 1994b).
- 48 stands on non-calcareous sandy soils in the Dutch dune area sampled in 1991: the depths of 0–10, 10–30, 30–60 and 60–100cm (De Vries 1993, De Vries unpublished data).
- 150 forest stands on non-calcareous sandy soils sampled in 1990: the depths of 0–30 cm (De Vries and Leeters 2001).
- 200 forest stands on non-calcareous sandy soils sampled in 1995: the humus layer and the depths of 0–10 cm (Leeters and De Vries 2001).
- 100 forest stands sampled between 1992 and 1993 in approximately 40 loess soils, 30 clay soils and 30 peat soils: the depths of 0–10, 10–30, 30–60 and 60–100 cm (Klap et al. 1999).
- 50 forest stands in the "Drentse Aa" area, sampled in 1994: 44 sandy soils, 4 clay soils and 15 peat soils; the mineral topsoil with depths varying from 0–10 cm and 0–30 cm (Klap et al. 1997).

More information about the derivation is given below.

Depth of the rooting zone: The depth of the rooting zone varies by soil and vegetation type as given in Table NL-3. These values were based on expert judgement by foresters.

Soil bulk density: According to the Dynamic Modelling Manual (Posch et al. 2003), values for the soil bulk density of sandy soils and clay soils ($C_{org} \leq 15\%$) were estimated from the measured organic carbon content and clay content in the various data sets, according to (Hoekstra and Poelman 1982):

$$\rho = \frac{1}{a_0 + a_1 \cdot C_{org} + a_2 \cdot clay} \qquad (2)$$

where ρ is the bulk density (g cm^{-3}) and C_{org} is the organic carbon content (%). Values used for a_0 and a_1 in sandy soils range between 0.601 and 0.646 and between 0.025 and 0.03 respectively, depending upon groundwater level. The value of a_2 in sandy soils is set to 0. Values used for a_0, a_1 and a_2 in clay soils range between 0.572 and 0.618, between 0.0053 and 0.023 and between 0.00067 and 0.0039, respectively, depending upon soil horizon. The assignment of results to the various soil types was done by an averaging procedure.

For loess, bulk densities were assigned per horizon according to A, E, AC, B, C horizon as 1420, 1428, 1486, 1542 and 1553 kg m^{-3} based on measurements in those horizons. For peat soils ($C_{org} > 15\%$), the bulk density was estimated according to (Van Wallenburg 1988):

$$\rho = 0.826 - 0.337 \cdot \log(2 \cdot C_{org}) \qquad (3)$$

Again, the assignment of results to the various soil types was done by an averaging procedure for peat soils.

Volumetric water content at field capacity: The volumetric soil moisture content (m^3 moisture m^{-3} soil) of sandy soils, loess soils and peat soils was based on measured soil moisture contents on a dry weight basis (m^3 moisture kg^{-1} soil) multiplied by the estimated bulk density of the soil (kg m^{-3}) with the following maxima: 35% for sandy soils, 45% for loess soils and 90% for peat soils. For clay soils, no data were available. According to the modelling manual (Posch et al. 2003), an approximation of the annual average soil moisture content was made for those soils as a function of the clay content according to:

$$\theta = 0.04 + 0.077 \cdot clay \qquad (4)$$

Equation 4 holds for clay contents up to 30%. Above 30% a constant value of 0.27 was assumed.

Cation exchange capacity and base saturation: The CEC value was measured at the actual (unbuffered) pH in the above mentioned soil data sets. Especially acid soils (non-calcareous sandy soils, most loess and peat soils) this implies that the cation exchange constants are only applicable in the limited pH range of the soils considered (mainly between pH 3 and 5). As a first estimate the CEC at pH 6.5 was made by calculating the CEC as a function of the clay and organic carbon content, accounting for the impact of pH according to (Helling et al. 1964):

$$CEC = (0.44 \cdot pH + 3.0) \cdot clay + (5.1 \cdot pH - 5.9) \cdot C_{org} \quad (5)$$

We updated the CEC calculation at pH 6.5 according to:

$$CEC_{updated(pH\,6.5)} = CEC_{calculated(pH\,6.5)} \cdot \frac{CEC_{measured}}{CEC_{calculated(measuredpH)}}$$

The base saturation was calculated as the ratio of the amount of bases divided by the CEC at pH 6.5 as derived above.

***Carbon pool and C:N ratio in the topsoil**:* The C:N ratio in the topsoil were related to the topsoil and based on measured values the in the above mentioned soil data sets. The C pool was calculated for the same depth by multiplying the measured C content with the estimated bulk density and a soil depth of 20 cm. As with other soil data, the assignment of results to the distinguished soil types was done by an averaging procedure.

General information

Soil type: See Table NL-2.

Soil characteristics: This refers to the contents of organic matter (carbon), clay and sand and the pH. The organic carbon content, clay content and pH were based on measurements in the above-mentioned data sets and averaged to the soil types distinguished. Sand contents were not available

Climatic characteristics: Both precipitation and temperature data were derived from 280 weather stations in the Netherlands, using interpolation techniques to obtain values for each grid.

Exchange constants are not yet included in this data report. We do take the constants from the same dataset described in this document. More information on the derivation of those constants and the results is given in De Vries and Posch (2003).

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National maps produced

Critical load maps for surface waters, forest soils and nutrient nitrogen to vegetation have been produced. There are changes in the data for surface water and for soil since the 2001 CCE Status Report (Posch et al. 2001). Data are calculated on the following grid size: each 1° longitude by 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 2304 grids has been calculated.

Surface waters

The SSWC method was used to calculate the critical loads of acidity using variable ANC_{limit} (Henriksen and Posch 2001). $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ were computed according to the FAB model (Henriksen and Posch 2001).

The contribution to sulphur deposition from atmospheric, non-anthropogenic, non-marine sources has been updated according to EMEP background deposition estimates. Calculation of background sulphate is done according to the equation (Henriksen and Posch 2001):

$$[SO_4]_0^* = a + b\,[BC]_t^*$$

After the update a = 3, while b still equals 0.17.

Data from national regional lake surveys and monitoring programmes were used. No changes were made to the input data since the 2001 CCE Status Report (Posch et al. 2001).

The 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.

All grids have critical loads for surface waters. They have been assigned EUNIS Level 1 classification C.

Forest soils

Critical loads for forest soils were calculated using the SMB model. There have been considerable updates in the forest critical loads data since the 2001 CCE Status Report (Posch et al. 2001).

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 was developed as described above. Vegetation uptake data was estimated based on forest monitoring data (Norwegian Institute for Forest Research). The procedure for lumping soil data and principles of the calculations have been reported previously (Frogner et al. 1993, 1994).

Weathering rates were taken from calculations with the dynamic acidification model MAGIC (Cosby et al. 1985), calibrated to observed soil chemistry and surface water chemistry. The criterion used was Ca:Al molar ratio of 1.0 in the upper 60 cm of the soil. The MAGIC model has also been used previously in calculating critical loads for forest soils in Norway (Frogner et al. 1994). There are several updates in the model calibration for this call. The updates are related to updates in several assumptions, including treatment of organic acids and background sulphate in the modelling approach.

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. All grids with forest have been assigned EUNIS Level 1 classification G.

Nutrient nitrogen - vegetation

The critical loads have been estimated from empirically derived relationships between N deposition and vegetation type (Esser and Tomter 1996). There have been no updates in the natural vegetation critical loads data since the 2001 CCE Status Report (Posch et al. 2001). The following vegetation types and critical load values have been used:

Ombrotrophic bog:	5 kg N ha^{-1} a^{-1}
Coniferous forest:	7 kg N ha^{-1} a^{-1}
Deciduous forest:	10 kg N ha^{-1} a^{-1}
Calluna heath:	15 kg N ha^{-1} a^{-1}
Others	20 kg N ha^{-1} a^{-1}

Critical loads for natural vegetation are reported for 1610 of the total 2304 grids. They have not been classified according to EUNIS and were given the code Y in the response to the 2002 call for data.

Assignment of areas to 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 adds up to 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

The receptor types considered are broadleaved deciduous woodland and coniferous woodland. Maximum critical loads of sulphur, minimum critical loads of nitrogen, maximum critical loads of nitrogen and critical loads of nutrient nitrogen have been calculated by the Simple Mass Balance method outlined in the Mapping Manual (UBA 1996), and partly by the PROFILE model.

The primary revision of the critical loads calculation approach reported in 2001 for both woodland ecosystem types is the substitution of EMEP base cation deposition data with data derived from the national precipitation chemistry monitoring network. This monitoring network covering all of Poland has been operated by the Institute of Meteorology and Water Management, Wroclaw Branch since 1996. The initial results from the wet deposition measurements were published in 1999, and the most recent reported data are for 2001. The total deposition values of base cations were estimated from the recently reported wet deposition data multiplied by dry deposition factors derived from throughfall data provided by the integrated monitoring surveys.

Comments

As a result of the above-mentioned data modifications, calculations of maximum critical loads of sulphur and nitrogen have changed noticeably compared to maps presented in the 2001 Status Report. Although maps of minimum critical loads of nitrogen and critical loads of nutrient nitrogen have not changed, they are also presented here to provide a complete overview of the results (see Figures PL-1 to PL-4).

The Polish NFC is intensely working on implementing dynamic modelling under Polish conditions. As part of this task, the existing Polish critical load database has been extended with additional input data required by dynamic models. Successful test runs have been performed with the simple VSD model and the complex SAFE model. A number of useful methodological suggestions and practical hints resulted from these tests, particularly in terms of acquiring the necessary input data. The main conclusion from this preliminary dynamic modelling exercise was that the availability of input data from the Polish integrated monitoring network is satisfactory to perform full-scale dynamic modelling.

Calculations of critical loads of heavy metals are in the initial stages, and the first maps are expected by the end of 2003.

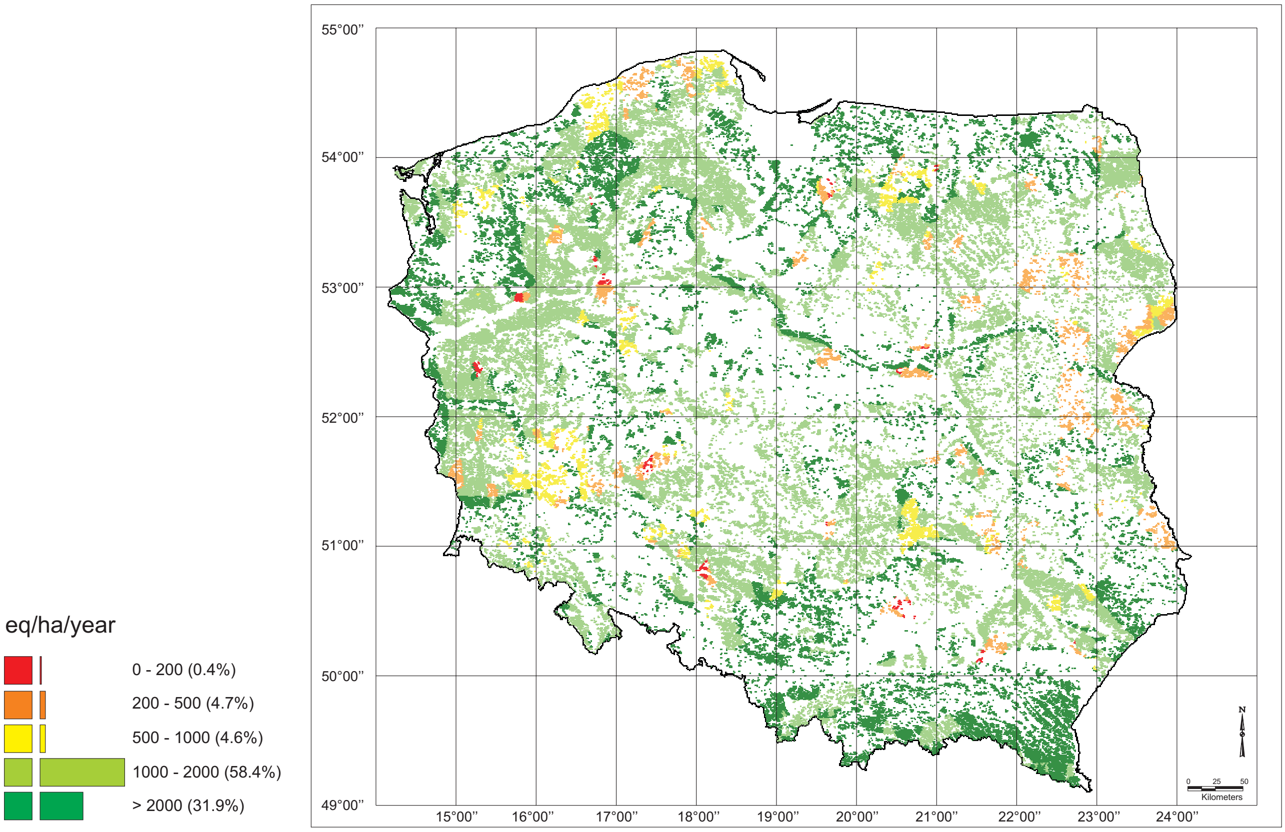

Figure PL-1. Maximum critical loads of sulphur.

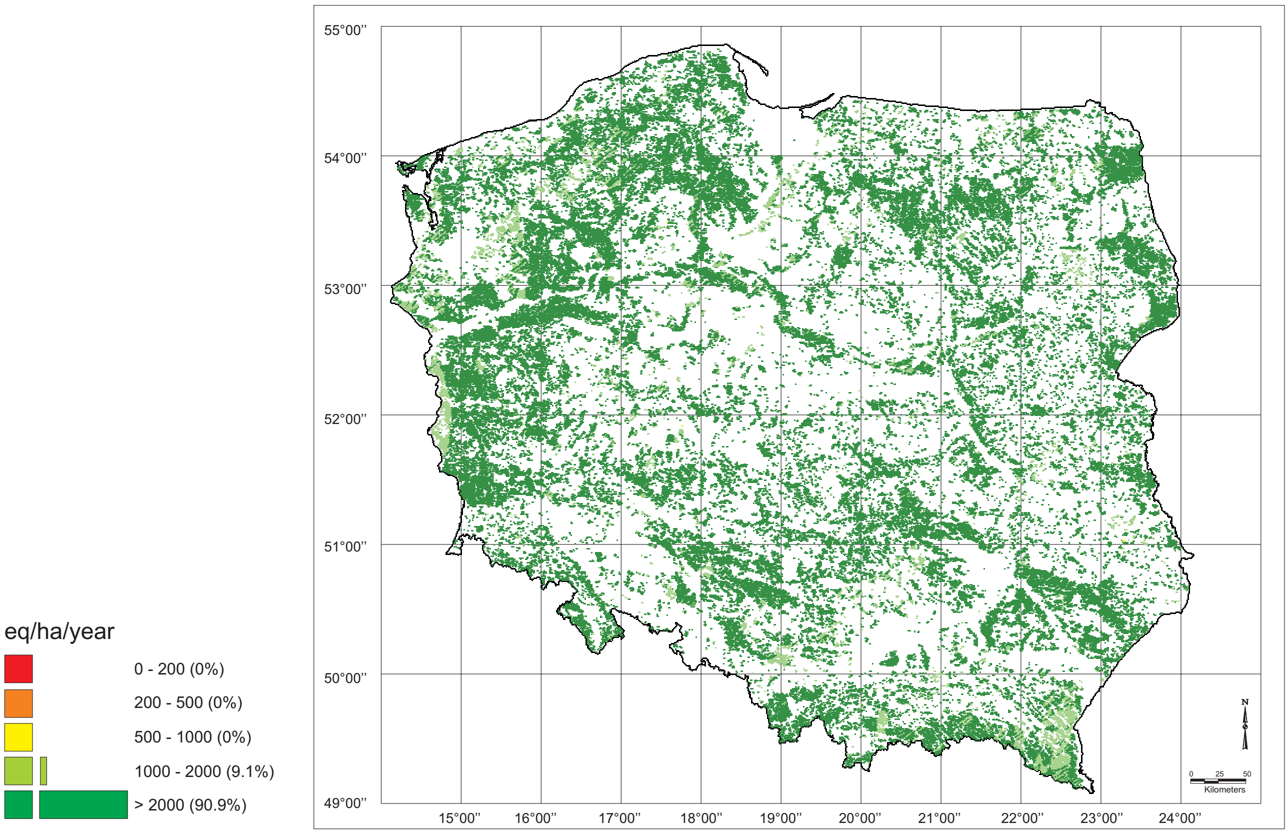

Figure PL-2. Maximum critical loads of nitrogen.

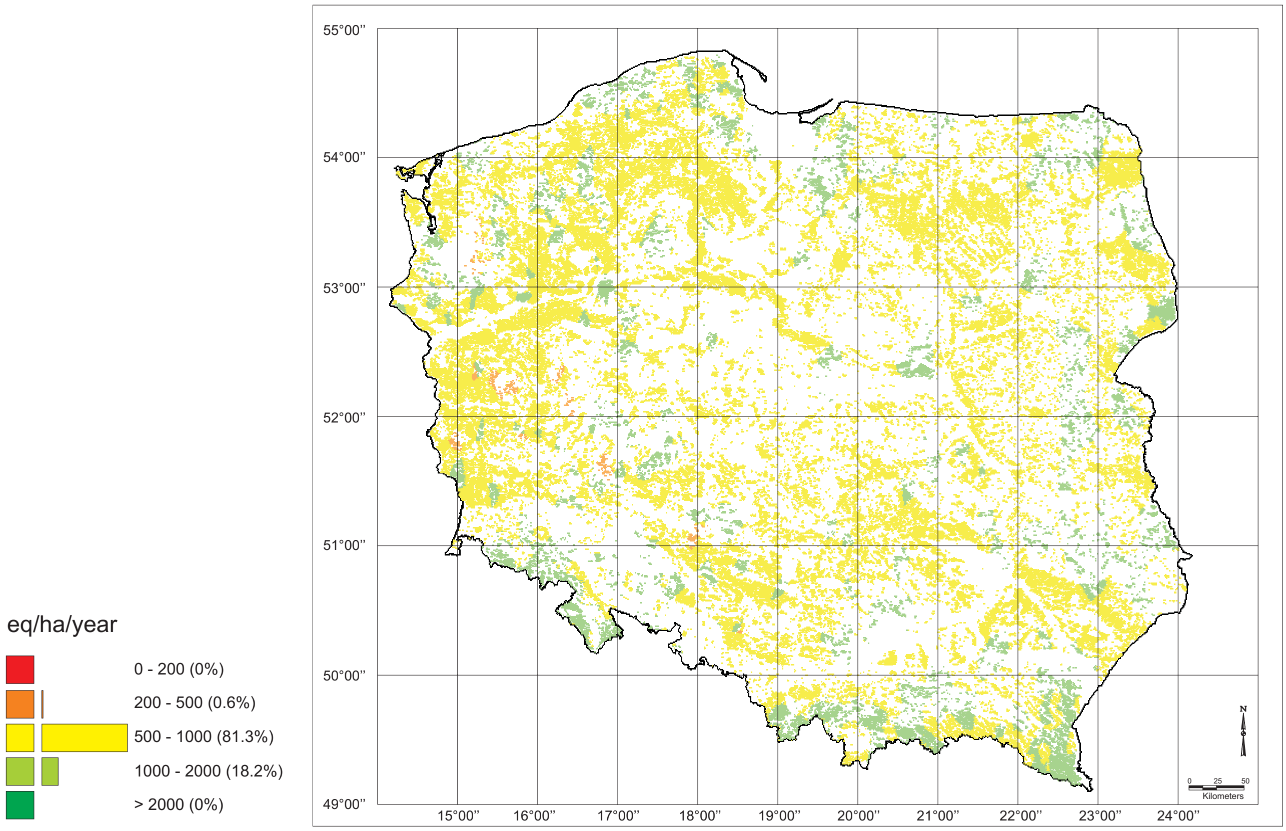

Figure PL-3. Minimum critical loads of nitrogen.

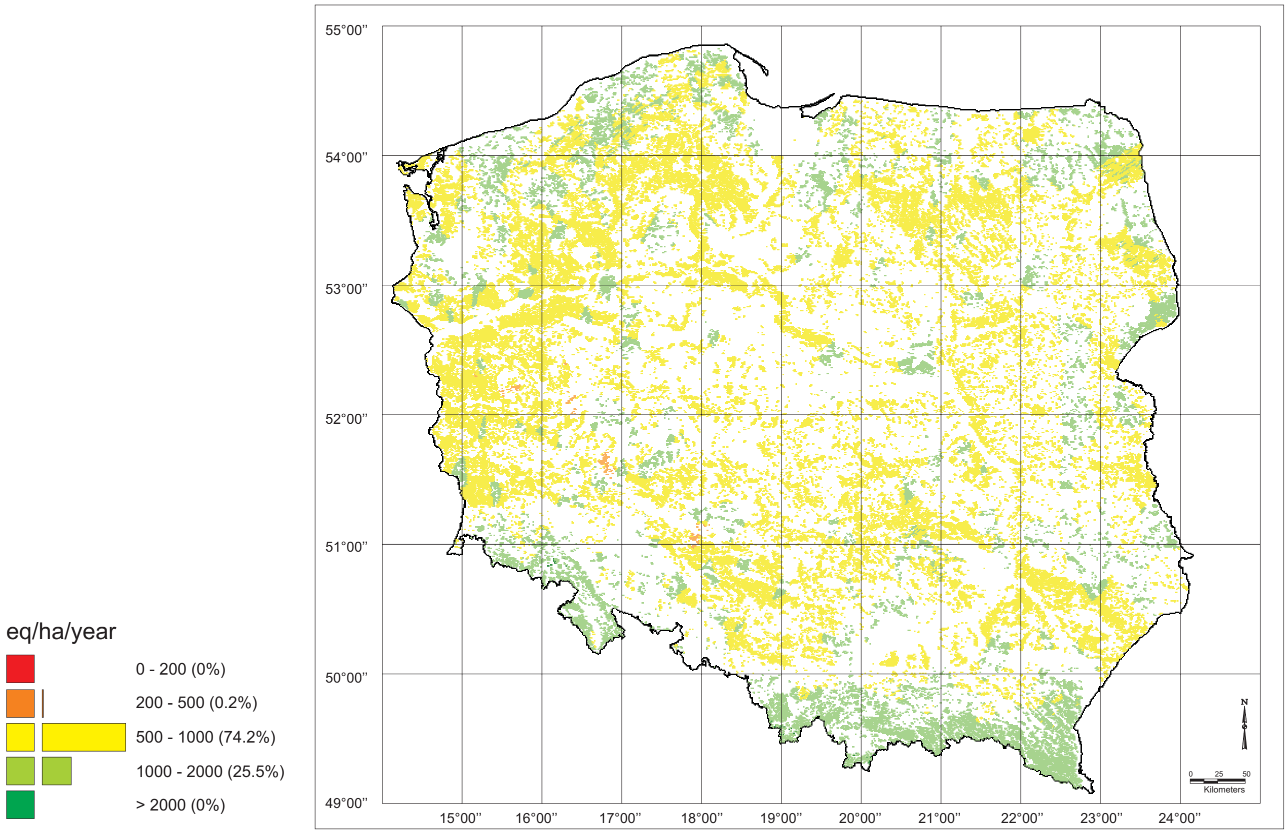

Figure PL-4. Critical loads of nutrient nitrogen.

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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 coordinate 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

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

Dynamic modelling of soil response to atmospheric deposition: The first stage of dynamic modelling activities has been started in Slovakia, following the methods outlined in the Dynamic Modelling Manual (Posch et al. 2003). We used the VSD model as the simplest extension of the SMB model for critical loads.

Data sources

Critical loads of acidity: Based on new field data on nitrogen, base cation, and heavy metals concentration in the wood and bark of forest trees, values for nitrogen and base cation uptake have been updated in the Slovak database of critical loads for forest soils.

Input data for dynamic modelling: The first stage of dynamic modelling began with forest monitoring plots. There are 111 forest monitoring plots in Slovakia (Figure SK-1) that represent the variability of site conditions and tree species composition of Slovak forests.

Many parameters were measured or estimated for these plots, including all parameters required to apply the steady-state mass balance approach to calculate critical loads. Additional parameters needed for the VSD model application were derived as follows:

Soil input data:

- Measured (from 1998 soil survey's forest soil monitoring database): CEC, E_{BC}, C, N (data for the 10–20 cm soil layer as a "medium" layer of the soil compartment used for calculation).
- Calculated using pedotransfer function: soil bulk density, soil moisture content.
- Derived from data in VSD model help files according to soil texture/soil type: exchange constants, Al exponent.
- Estimated (combination of measured data at some of plots and estimated data at some of plots): clay content.

In some cases (CEC, base saturation), the input data are not fully comparable due to some differences in methodology ($BaCl_2$ extract instead of ammonium acetate extract).

Deposition data:

- Wet deposition data were derived from the element concentrations and precipitation totals (Slovak EMEP stations).
- Total deposition were calculated from wet deposition data and deposition enrichment factors depending on altitude and tree species.
- Deposition scenarios were defined as a portion of current deposition values as follows: Sulphur: year 0 = 10%, year 200 = 30%, year 300 (current) = 100%, year 500 = 20%.

Nitrogen: year 0 = 10%, year 200 = 20%, year 300 (current) = 100%, year 500 = 20%.

For spatial modelling and interpretation of the dynamic modelling results, we also prepared the input data in the grid resolution 250×250m^2 with the same structure as for SMB method (S, N, heavy metals). Spatial distribution of the input parameters was derived from the supporting GIS layers (soil types, DTM, soil textures) and point-measured data by using GIS tools (interpolation methods).

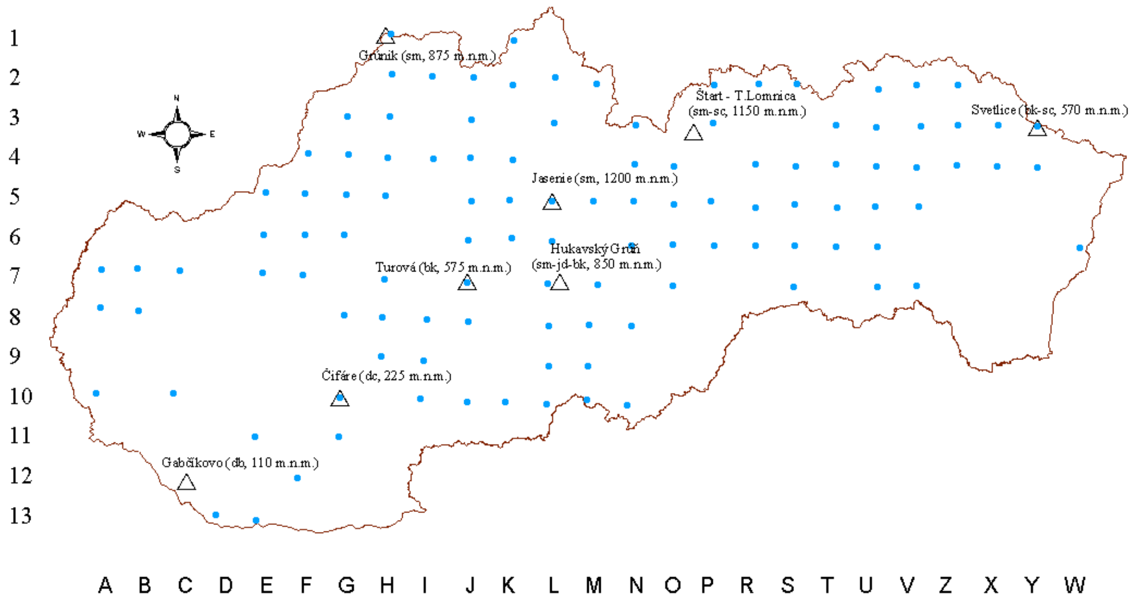

Figure SK-1. Spatial distribution of forest monitoring plots (level I and level II) in Slovakia.

Results

Preliminary results for dynamic modelling:
Calculations were carried out for all 111 forest monitoring plots. Detailed analyses have been conducted for 5 plots (see Table SK-1). Table SK-1 presents the soil pH values calculated from the VSD model for the present time, along with measured pH soil data (10–20 cm). The largest differences were obtained for plot A7 (an Arenosol soil type), and the best results were for plot S5 (a Rendzic Leptosol). Examples of calculations with the VSD model are presented for two plots with different soil conditions (Figs SK-2, SK-3).

Conclusions

The first preliminary results from VSD model calculations showed that this model could be a useful tool for future improvements in critical load calculations. In the near future the following activities should be carried out in Slovakia:

- verification and detailed analysis of input data for VSD model.
- sensitivity analysis of input parameters within the different site conditions.
- improvements of deposition scenarios.

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UBA (1996) Manual on methodologies and criteria for mapping critical levels/loads and geographical areas where they are exceeded. UNECE Convention on Long-range Transboundary Air Pollution. Federal Environmental Agency (Umweltbundesamt) Texte 71/96, Berlin.

Table SK-1 Preliminary results from VSD model calculations for 5 forest monitoring plots in Slovakia.

Plot	Parent material	Soil type	pH measured (depth 10–20 cm)	pH VSD model
A7	Eolic sands	Arenosol	4.90	4.40
J7	Andesitic pyroclastics	Eutric Cambisol	5.16	5.20
L2	Flysch/sandstone	Dystric Cambisol	4.02	4.40
S5	Limestone	Rendzic Leptosol	7.37	7.38
Y3	Flysch	Dystric Cambisol	4.88	4.70

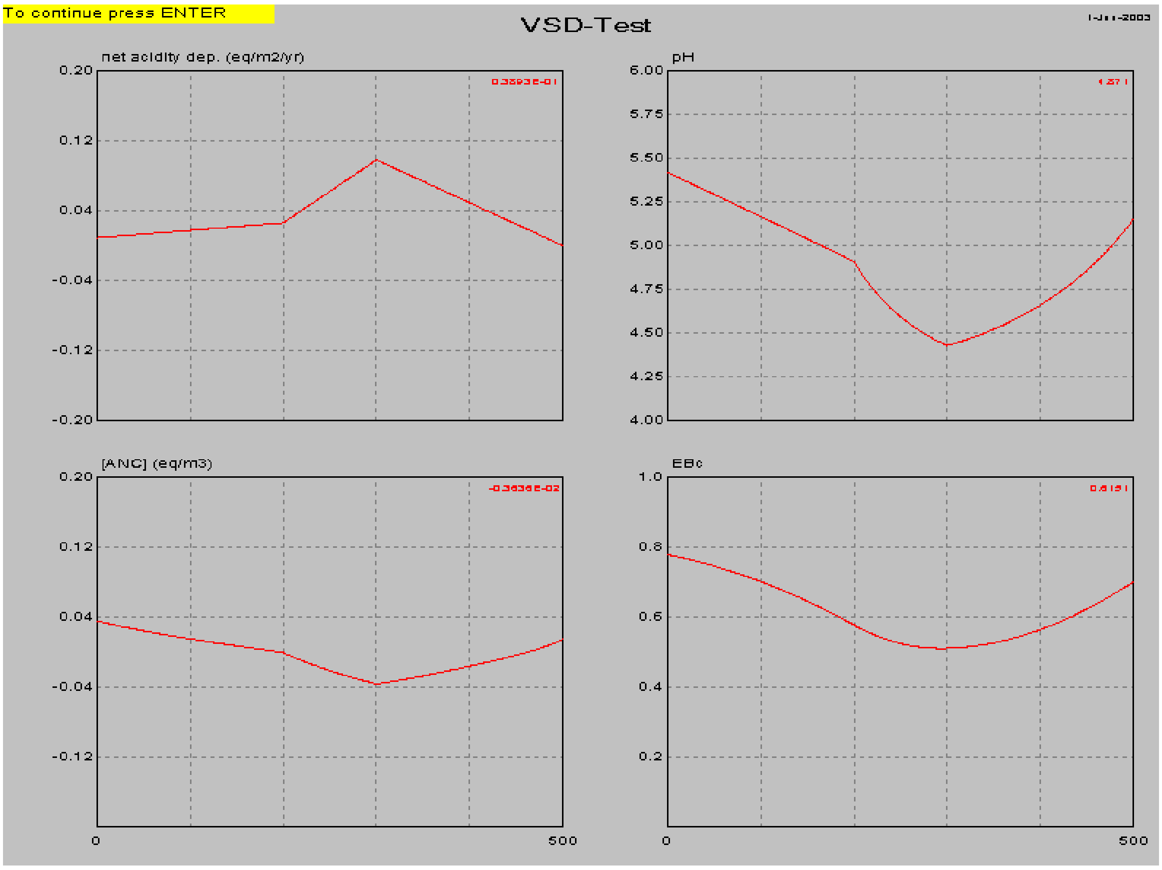

Figure SK-2. Selected calculated values from VSD model for plot A7 (Arenosol).

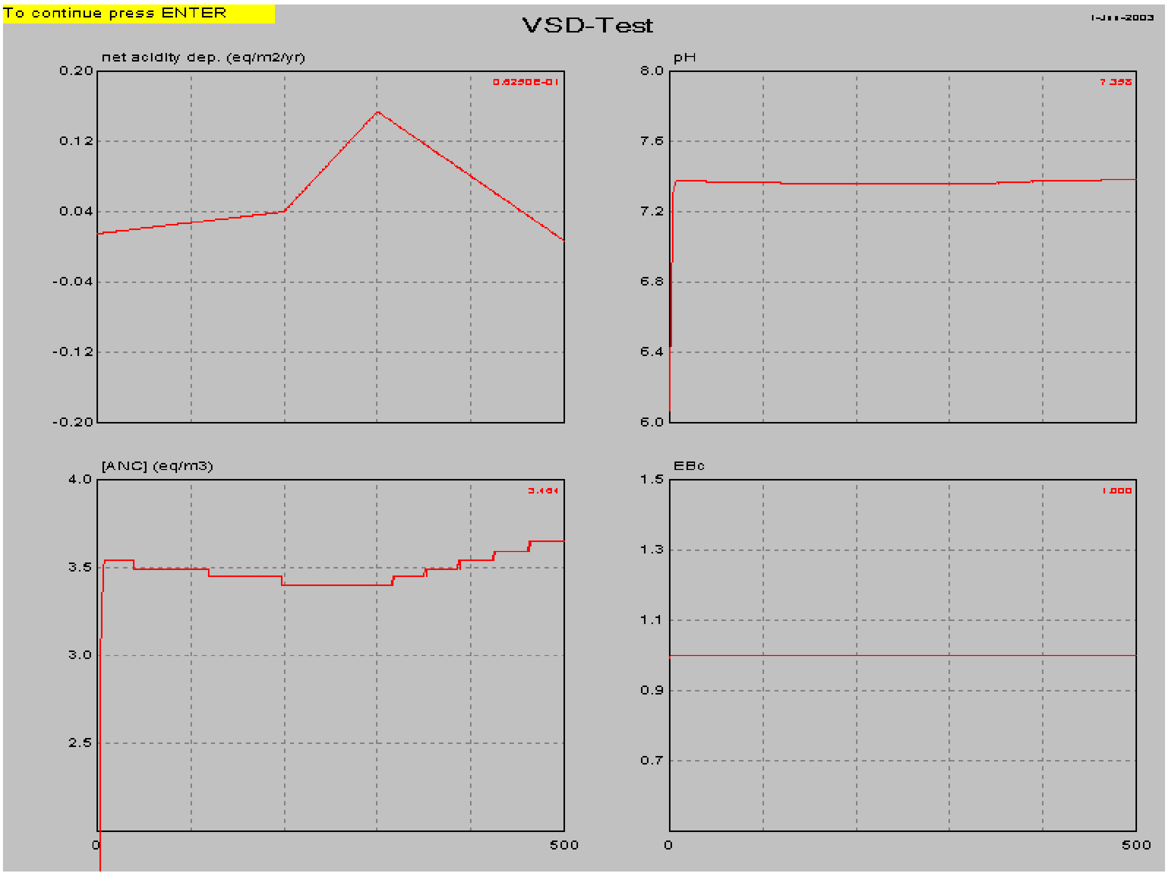

Figure SK-3. Selected calculated values from VSD model for plot S5 (Rendzic Leptosol).

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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 prior critical loads database has been converted to the latest format 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 1997.

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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 critical load of acidity, $CL(A)$, was calculated as:

$$CL(A) = BC_w - ANC_{le(crit)}$$

where BC_w is the weathering rate and $ANC_{le(crit)}$ is the critical leaching of ANC.

The soil profile of each site 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. A molar ratio of 1.0 was used for all forest ecosystems. Thereafter the critical load functions ($CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$) were calculated according to the Mapping Manual (UBA 1996). The denitrification rate was, by contrast with the Manual, included in $CL_{max}(N)$ instead of $CL_{min}(N)$ as there were for constant nitrogen 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:

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

where:
N_u = long-term net uptake by the forest
N_{de} = denitrification
N_i = N immobilisation
$N_{le(acc)}$ = acceptable N leaching

The calculation of $CL_{nut}(N)$ was made in parallel and integrated with the calculation of critical loads of acidity using the PROFILE model. 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 (Warfvinge et al. 1992). A criterion, expressed as the minimum quotient between concentrations of base cations (calcium, magnesium and potassium) and nitrogen in the trees, was introduced to avoid long-term nutrient imbalances in forest trees.

N immobilisation was determined as:

$$N_i = k_i \cdot [N] \cdot f(pH) \cdot z(T)$$

where $[N]$ is the total concentration of N in the soil solution and the functions $f(pH)$ and $z(T)$ represent the influence of pH and temperature on the immobilisation rate.

The calculation model was calibrated using empirical data on immobilisation for two extreme situations in Sweden; one is the northern part of Sweden with an estimated long-term mean immobilisation of 0.5 kg N ha^{-1} a^{-1}, a soil pH of 5.5 and a deposition of 2 kg N ha^{-1} a^{-1}. The other extreme represents southernmost Sweden with an immobilisation of 15 kg N ha^{-1} a^{-1} at pH 4.2 and a deposition of 20 kg N ha^{-1} a^{-1} (Ineson at al. 1996, Nilsson et al. 1998).

Denitrification was calculated using the Sverdrup-Ineson equation as given in the Mapping Manual (UBA 1996). Acceptable N leaching was calculated in a separate procedure, a critical concentration of 0.3 mg N l^{-1} in leaching water was multiplied with the runoff at each site. A concentration of 0.3 mg N l^{-1} represents unpolluted conditions (upper limit of the lowest concentration class) according to the environmental quality criteria for surface waters in Sweden (Swedish Environmental Protection Agency 1999).

Surface water ecosystems:
Critical loads of sulphur and acidifying nitrogen were calculated using the first-order acidity balance (FAB) model as described in Henriksen and Posch (2001). The chemical threshold, ANC_{limit}, was set to 20 μeq l^{-1} in cases where $[BC^*_0] > 25$ μeq l^{-1}. In other cases, ANC_{limit} was set to $0.75[BC^*_0]$ to allow for naturally low ANC concentrations. N immobilisation was set to a maximum of 2 kg N ha^{-1} a^{-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). In contrast with the literature and the Mapping Manual, net uptake of base cations was taken into account when calculating $CL_{max}(S)$.

Mapping

The area assigned to each lake and/or forest site 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. To account for differences in number of sampled lakes in relation to the total number of lakes within a region, the number of sampled lakes of a certain lake size class, and county, was weighted to the total number of lakes of the same class (Wilander et al. 1998). For instance, if two lakes were sampled in one of Sweden's 22 counties, out of 20 lakes of the same size class, the sampled lakes were given weights equal to 10. Thus the assumption is that the sampled lakes represent the properties of all lakes within a county. For forest sites, weights based on the Swedish Forest Inventory were used. To account for cell areas not at risk from acid deposition, 10% of each cell area was subtracted when calculating the cell ecosystem area.

Data sources

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 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. Deposition data from 1997 was used in the calculations.

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 tree species composition. All input data were derived according to Warfvinge and Sverdrup (1995).

Surface water ecosystems:
Water chemistry data were taken from the 1995 Swedish Lake Survey (Wilander et al. 1998).

In total, 2983 lakes were included in the calculation, consisting of 2068 unlimed lakes and an additional 915 lakes which were corrected for liming by assuming a constant Ca:Mg ratio for nearby lakes and assuming that the Mg concentration was not affected by liming. A long-term average (1961–90) of runoff data from the Swedish Meteorological Institute (SMHI) was used. Land use data and the long-term average of nutrient uptake were derived from the Swedish Forest Inventory 1983–92.

Deviations from the Manual:
All variables were derived according to the Mapping Manual unless otherwise stated in the text.

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Status of Swiss critical loads data

In response to the CCE call for data in November 2002, two new data sets for critical loads were provided:

- data for forests (critical loads of acidity and nutrient nitrogen, and input parameters dynamic modelling).
- data for natural and semi-natural ecosystems (CL for nutrient nitrogen, empirical method only).

These files replace previous data provided in February 2001. No data are presently available for alpine lakes.

The main changes since February 2001 are:

- DBF instead of ASCII files were submitted.
- New file format – the names of the 36 fields correspond to the call for data.
- New data for dynamic modelling has been supplied for 642 ecosystems.
- Information on ecosystem types is now provided as EUNIS codes.
- The critical loads values ($CL_{max}(S)$, $CL_{nut}(N)$, …) have *not* changed since 2001.

The data set for forests

The data set contains 691 records related to the sampling points on the 4x4 km^2 sub-grid of the National Forest Inventory (NFI). Therefore, each point represents 16 km^2, resulting in a total area of forest of 11,056 km^2.

All records have values for acidity critical loads, ($CL_{max}(S)$ and $CL_{max}(N)$). ANC_{le} and BC_w are the results of the regional PROFILE model application conducted by Kurz (SAEFL 1998). The relatively high values (>6000 eq ha^{-1} a^{-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_{le} and BC_w (>5000 eq ha^{-1} a^{-1}) are very rare.

$CL_{nut}(N)$ is calculated with f_{de} according to the Mapping Manual. There are 42 records with no values ("–1") for $CL_{nut}(N)$. These are NFI sites that are supposed to be unmanaged (inaccessible sites, bush forest) and therefore, N_u and BC_u are zero. $CL_{nut}(N)$ for natural unmanaged forests are included in the data set.

BC_{dep}, runoff, N_i, N_u, BC_u and f_{de} are described in FOEFL (1994). Values for these parameters have not changed since March 2001. The logarithm of K_{gibb} has been set to a value of 8 for all records. This is an appropriate value for SMB or VSD applications;, but different values have been used to calculate ANC_{le} for each of the four soil layers modelled in PROFILE: 6.5, 7.5, 8.5 and 9.2 (SAEFL 1998).

$N_{le(acc)}$ is chosen as follows: 4 kg N ha^{-1} a^{-1} in the lowlands (500 m a.s.l) with a linear decrease to 2 kg N at 2000 m a.s.l.

The ecosystem types are coded following the EUNIS method:
G3 = coniferous woodland (>95% coniferous trees)
G1 = broadleaved deciduous woodland (< 5% coniferous trees)
G4 = mixed woodland (5–95% coniferous trees).

This is not completely in accordance with the definition in EUNIS, where the limit for G3 is set at >75% coniferous crown cover, but more precise information is not available.

The VSD parameters *thick*, *ρ, θ*, CEC and E_{BC} come from the database acquired for the critical load calculations with the PROFILE model (SAEFL, 1998). The standardised 4-layer input has been recalculated to the one-layer input according to Kurz and Posch (2002, Chap. 4). The parameters C_{pool}, CNrat0, clay and sand are not available (values = –1).

Soil type information comes from the Swiss soil map 1:500,000 (Swisstopo 1984) which was converted to the FAO classification (www.fao.org/ag/AGL/agll/key2soil.stm) as described in FOEFL (1994, Chap. 4.4). Fourteen different soil types were identified: Bd, Be, Bg, Bh, E, G, H, I, J, Lo, O, P, Rc and Rd.

C_{org} and pH are data from the NFI. They are both related to the topsoils (0–20cm) and pH is pH_{CaCl2}. *Prec*, *Temp* and *Alt* are also data from the PROFILE application. For dynamic modelling in Switzerland the parameter P_{CO2} was set to a value of 15 times atmospheric pressure.

Data set for natural and semi-natural ecosystems

The data sources and procedures for implementing the empirical method are described in FOEFL (1996). The file contains 11,482 records, which represent 1 km^2 each, and is a compilation of various vector and raster data sets (FOEFL 1996). Spatial overlays with the NFI sites were eliminated. Thus, there is no double-counting of ecosystem areas. Ecosystem types are coded using the EUNIS classification (mrw.wallonie.be/dgrne/sibw/EUNIS/home.html). Table CH-1 lists the EUNIS codes with their ecosystem area in Switzerland.

Further work

New critical load data for alpine lake catchments are expected to be available by the end of 2003. In addition, it is planned to establish a new database for forests by March 2004. This database will be based on 250 full soil profiles and will replace the present data, which are based mainly on topsoil samples.

Table CH-1. Area and EUNIS codes of ecosystems included in the empirical method for mapping $CL_{nut}(N)$.

Area (km^2)	EUNIS code
38	C1.1
568	D1.1
310	D2.2
153	D2.21
317	D4.1
718	E1.2
301	E1.24
845	E1.26
275	E3.51
6	E4.3
899	E4.41
801	E4.42
3932	E4.43
1016	F2.21
496	F2.23
27	G1.7
195	G1.71
16	G1.73
38	G3.3
13	G3.4
50	G3.43
468	G3.44
11,482 km^2 total	

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Introduction

This update to the critical loads of acidity and nutrient nitrogen for sensitive UK habitats has been made in light of new research findings and revisions to (i) the habitats mapped, (ii) the underlying data, and (iii) the methods used for calculating critical loads. This report provides an overview of the revisions made. A detailed report (Hall et al. 2003) and maps are available on the UK NFC web site (critloads.ceh.ac.uk).

Mapping sensitive habitats

In earlier data submissions UK critical loads were mapped for six general ecosystem types: acid grassland, calcareous grassland, heathland, coniferous woodland, deciduous woodland and freshwaters. The distributions of these ecosystems were defined from the CEH Land Cover Map 1990 (Fuller et al. 1994) and additional data sets (e.g. species distribution data). The CEH Land Cover Map 2000 (Fuller et al. 2002a,b) has been used in this update, together with species distribution data and other data sets (e.g. soils, altitude), to map critical loads for the terrestrial UK Biodiversity Action Plan (BAP) Broad Habitats, where both appropriate and feasible. In this way critical loads and exceedances can be considered in relation to habitats of high conservation value in the UK.

Critical loads are now mapped for: acid grassland, calcareous grassland, dwarf shrub heath, bog, coniferous woodland, broadleaved and mixed woodland, montane and coastal dune habitats. It should be noted that the areas of the habitats now mapped differ from those previously submitted (Hall et al. 2003).

For freshwaters, critical loads are calculated using water chemistry samples taken from lakes and streams in acid-sensitive areas of the UK. From the original data set of samples for over 1500 sites, rigorous screening of the data in this update has produced a set of 1163 lakes and streams for which critical loads have been calculated.

In order to harmonise the naming and classification of habitats across Europe, habitat codes from the EUNIS habitat classification scheme (Davies and Moss 1999, 2002) have been assigned to each of the habitat types for which critical loads are mapped. Table GB-1 shows the relationship between the UK BAP Broad Habitats and the EUNIS habitat classes. This shows that more than one EUNIS class may be mapped for a single UK BAP Broad Habitat. In addition, we have created two EUNIS classes:

Table GB-1. Relationships between UK BAP Broad Habitats and EUNIS habitat classes.

UK BAP Broad Habitat	EUNIS class (with additional specification in brackets)	Mapped for acidity	Mapped for nutrient nitrogen
1. Broadleaved, mixed and yew woodland	G1 Broadleaved woodland (managed (productive) broadleaved woodland)	yes	yes
	G1-LA Broadleaved woodland (effects on epiphytic lichens only)	no	yes
	G1&G3 Broadleaved and coniferous woodland (unmanaged ancient & semi-natural woodland)	yes	yes
	G1&G3-GF Broadleaved and coniferous woodland (unmanaged ancient & semi-natural woodland) (effects on ground flora only)	no	yes
2. Coniferous woodland	G3 Coniferous woodland (managed (productive) coniferous woodland)	yes	yes
7. Calcareous grassland	E1.26 Sub-Atlantic semi-dry calcareous grassland	yes	yes
8. Acid grassland	E1.7 Non-Mediterranean dry acid and neutral closed grassland	yes	yes
	E3.5 Moist or wet oligotrophic grassland	yes	yes
10. Dwarf shrub heath	F4.11 Northern wet heaths (Calluna-dominated upland & Erica-dominated lowland wet heaths)	yes	yes
	F4.2 Dry heaths	yes	yes
12. Bogs*	D1 Raised and blanket bogs	yes	yes
13. Standing open water & canals	C1 Surface standing waters	yes	no
14. Rivers and streams	C2 Surface running waters	yes	no
15. Montane*	E4.2 Moss and lichen dominated mountain summits (Racomitrium heath)	yes	yes
19. Supralittoral sediment*	B1.3 Shifting coastal dunes & B1.4 Coastal stable dune grassland (not mapped separately)	no	yes

* Habitats not previously mapped.

NB. It should be noted that the UK BAP Broad Habitats and the EUNIS classes are not entirely interchangeable; some broad habitats may include a wider habitat than defined by the EUNIS classes and conversely some EUNIS classes may include habitats not wholly in the broad habitat.

G1-LA for the effects of nitrogen on epiphytic lichens in broadleaved (Atlantic oak) woods, and G1 and G3-GF for the effects of nitrogen on woodland ground flora. The methods used to map each Broad Habitat and each EUNIS class are given in Hall et al. (2003).

Revisions made to critical loads data

The changes made to the data sets underlying the critical load calculations are listed below:

National soils data: The UK empirical critical loads of acidity for soils are based on weathering rate and mineralogy of the dominant soil type in each 1km grid square (Hornung et al. 1995a). Revisions have been made to the national soil databases for England, Wales and Scotland, leading to changes in some of the percentage areas of the different soil types in each 1km grid square. As a consequence, the dominant soil type upon which the acidity critical loads are based, has changed in some squares leading to changes in the acidity critical loads map. This map is used to assign acidity critical loads to non-woodland terrestrial habitats in the UK. These changes also have implications for other data sets that are based on soil information and used in the calculation of critical loads, namely:

- Base cation and calcium weathering rates used in the Simple Mass Balance (SMB) equation for calculating acidity critical loads for woodland habitats.
- Classification of mineral, organic and peat soils, required for applying the SMB and calculating acidity critical loads for peat soils.
- Soil nitrogen immobilisation and denitrification values.

National deposition data: The SMB calculations of acidity critical loads require total (i.e. marine plus non-marine) calcium deposition and the calculations of $CL_{max}(S)$ require non-marine base cation and chloride deposition. These data sets have been updated to the measured values for 2000.

Individual input parameters: A number of specific input values (e.g. N and base cation uptake, acceptable nitrate leaching) have been updated based on the availability of new or additional data. The new values are given in Table GB-2, which also includes the input parameters assigned to the new habitats, not previously mapped in the UK, as well as the justification for all values and methods used.

Table GB-2. Summary of UK critical load values and justification for their use.

Critical loads parameter (units)	EUNIS code	Min. value	Max. value	Data sources/ Methods used	Justification	Uncertainty
$CL_{max}(S)$ (eq ha^{-1} a^{-1})	G1	1	12,070	$= CL(A) + (BC^*_{dep} - Cl^*_{dep}) - BC_u$	Mapping Manual (UBA 1996).	Not calculated.
	G3	1	11,575			
	G1&G3	248	12,906			
	F4.11	130	4450			
	F4.2	150	4570			
	E1.26	3,818	4348			
	E1.7	140	4470			
	E3.5	130	4470			
	E4.2	170	4420	$= L_{crit} / (1 - \rho_S)$		
	D1	130	4400			
$CL_{min}(N)$ (eq ha^{-1} a^{-1})	G1	562	920	$= N_u + N_i + N_{de}$	Mapping Manual.	Not calculated.
	G3	352	710			
	G1&G3	142	500			
	F4.11	499	857			
	F4.2	1249	1607			
	E1.26	856	1214			
	E1.7	223	581			
	E3.5	223	581			
	E4.2	178	536			
	D1	178	536			
	C1	64	583	$= f N_u + (1 - r)(N_i + N_{de})$		
	C2	146	565			
$CL_{max}(N)$ (eq ha^{-1} a^{-1})	G1	563	12,632	$= CL_{max}(S) + CL_{min}(N)$	Mapping Manual.	Not calculated.
	G3	428	12,019			
	G1&G3	533	13,048			
	F4.11	639	5155			
	F4.2	1419	5819			
	E1.26	4674	5290			
	E1.7	393	4766			
	E3.5	363	4872			
	E4.2	348	4541			
	D1	318	4827			
	C1	271	75,681			
	C2	383	38,385			
	D	647	11,751			
	W	143	201,500			
$CL_{nut}(N)$ (eq ha^{-1} a^{-1})	G1	776	1134	N mass balance $= N_u + N_i + N_{de} + N_{le(acc)}$	Mapping Manual (UBA 1996).	Not calculated.
	G1-LA	714	714	Empirical (epiphytic lichens in Atlantic oak woods): 10 kg N ha^{-1} a^{-1}.	Achermann & Bobbink (2003), Hall et al. (2003).	Uncertainty range: 0–25 kg N ha^{-1} a^{-1}.
	G3	638	996	N mass balance $= N_u + N_i + N_{de} + N_{le(acc)}$.	Mapping Manual (UBA 1996).	Not calculated.
	G1 & G3-GF	857	857	Empirical (woodland ground flora): 12 kg N ha^{-1} a^{-1}.	Achermann & Bobbink (2003), Hall et al. (2003).	Uncertainty range: 5–20 kg N ha^{-1} a^{-1}.
	F4.11	1071	1071	Empirical (wet heaths): 15 kg N ha^{-1} a^{-1}.		Uncertainty range: 0–30 kg N ha^{-1} a^{-1}.
	F4.2	857	857	Empirical (dry heaths): 12 kg N ha^{-1} a^{-1}.		Uncertainty range: 10–20 kg N ha^{-1} a^{-1}.
	E1.26	1429	1429	Empirical (calcareous grass) 20 kg N ha^{-1} a^{-1}.		Uncertainty range: 15–25 kg N ha^{-1} a^{-1}.
	E1.7	1071	1071	Empirical (dry acid grassland) 15 kg N ha^{-1} a^{-1}.		Uncertainty range: 5–25 kg N ha^{-1} a^{-1}.
	E3.5	1071	1071	Empirical (wet acid grassland) 15 kg N ha^{-1} a^{-1}.		Uncertainty range: 5–25 kg N ha^{-1} a^{-1}.

Table GB-2 (continued). Summary of UK critical load values and justification for their use.

Critical loads parameter (units)	EUNIS code	Min. value	Max. value	Data sources/ Methods used	Justification	Uncertainty
$CL_{nut}(N)$ (eq ha^{-1} a^{-1})	E4.2	500	500	Empirical (mountain summits) 7 kg N ha^{-1} a^{-1}.	Achermann & Bobbink (2003), Hall et al. (2003).	Uncertainty range: 0–15 kg N ha^{-1} a^{-1}
	D1	714	714	Empirical (bogs) 10 kg N ha^{-1} a^{-1}.		Uncertainty range: 5–12 kg N ha^{-1} a^{-1}.
	C1 C2	NA NA	NA NA	$CL_{nut}(N)$ not assigned to freshwaters sampled in UK.	Freshwaters sampled tend to be P-limited, not N-limited.	NA
	B1.3/ B1.4	1071	1071	Empirical (coastal dunes) 15 kg N ha^{-1} a^{-1}.	Achermann & Bobbink (2003), Hall et al. (2003)	Uncertainty range: 5–25 kg N ha^{-1} a^{-1}.
BC^*_{dep} – Cl^*_{dep} (eq ha^{-1} a^{-1})	G1 G3 G1&G3	40 50 50	770 650 770	Updated from measured mean data for 1986–1991 to mean data for 2000 for woodland habitats.	Mapping Manual (UBA 1996), Hall et al. (2003).	Not calculated for UK data. Draaijers et al. (1996) quote the following uncertainty estimates for European base cation deposition data: random errors: 50–70%; systematic errors: 40–55%.
	F4.11 F4.2 E1.26 E1.7 E3.5 E4.2 D1	30 30 40 30 30 70 30	500 570 570 500 500 490 500	Updated from measured mean data for 1986–1991 to mean data for 2000 for low-growing vegetation.		
	C1 C2	– –	– –	Not used in FAB.		NA
Bc_u (eq ha^{-1} a^{-1})	G1	315	410	New values for managed broadleaved woodland. Minimum value for Ca-poor soils and maximum value for Ca-rich soils. (Previous values: 400 and 850 eq ha^{-1} a^{-1})	Values from Forest Research, based on site-specific measurements from 10 ICP Forests Intensive Forest Health monitoring sites (Level II) in the UK. (Hall et al. 2003). Values used in $CL_{max}(S)$ only, estimates of calcium uptake used in SMB for mineral soils.	CV ± 14%
	G3	270	270	New values for managed coniferous woodland. (Previous value 250 eq ha^{-1} a^{-1}).		CV ± 23%
	G1&G3	0	0	Unmanaged woodland, uptake set to zero assuming no harvesting.		NA
	F4.11 F4.2	0 0	0 0	Set to zero; uptake negligible.	Rawes & Heal (1978), Reynolds et al. (1987).	NA
	E1.26	222	222	Includes removal by sheep grazing.	Published data.	Not calculated.
	E1.7 E3.5 E4.2 D1	0 0 0 0	0 0 0 0	Set to zero; uptake negligible.	Rawes & Heal (1978), Reynolds et al. (1987).	
	C1 C2	– –	– –	Not used in FAB.		NA

Table GB-2 (continued). Summary of UK critical load values and justification for their use.

Critical loads parameter (units)	**EUNIS code**	**Min. value**	**Max. value**	**Data sources/ Methods used**	**Justification**	**Uncertainty**
ANC_w (eq ha^{-1} a^{-1})	G1 G3 G1&G3		4000 4000 4000	Based on mid-range empirical acidity critical loads for soils. ANC_w set to zero for peat soils. Note: calcium weathering only used in calculation of $ANC_{le(crit)}$.	Methods agreed by UK experts. Hall et al. (1998, 2001a, 2003).	Uncertainties (mid-range values): 100 eq ha^{-1} a^{-1}=±100% 350 eq ha^{-1} a^{-1}=± 43% 750 eq ha^{-1} a^{-1} =± 33% 1500 eq ha^{-1} a^{-1}=±33% 4000 eq ha^{-1} a^{-1}=±50%
	F4.11 F4.2 E1.26 E1.7 E3.5 E4.2 D1	– – – – – – –	– – – – – – –	SMB not used – only applied to woodland habitats in UK. Empirical critical loads of acidity for soils applied to non-woodland terrestrial habitats, therefore ANC_w not assigned.	Hornung et al. (1995a), Hall et al. (1998, 2001a, 2003).	NA
	C1 C2	– –	– –	Not used in FAB.		NA
$ANC_{le(crit)}$ (eq ha^{-1} a^{-1})	G1 G3 G1&G3	0 0 148	7710 7579 8136	Calculated via SMB equation: mineral soils: critical molar ratio Ca:Al = 1 in soil solution; organic soils: critical pH 4.0 in soil solution; peat soils: critical soil solution pH 4.4 (Al conc. set to zero for peat soils).	Mapping Manual (UBA 1996), Hall et al. (1998, 2001a, 2001b, 2001c, 2003).	Not calculated.
	F4.11 F4.2 E1.26 E1.7 E3.5 E4.2 D1	– – – – – – –	– – – – – – –	SMB not used for non-woodland terrestrial habitats in UK; $ANC_{le(crit)}$ not calculated.	Hornung et al. (1995a), Hall et al. (1998, 2001a, 2003).	NA
	C1 C2	– –	– –	For freshwaters ANC_{crit} is set = 0 μeq l^{-1}	Value selected for 50% probability of damage to brown trout populations.	NA
N_u (eq ha^{-1} a^{-1})	G1	420	420	New value (equivalent to 5.88 kg N ha^{-1} a^{-1}) for managed broadleaved woodland. (Previous value 7 kg N ha^{-1} a^{-1}).	Values from Forest Research, based on site-specific measurements from ten ICP Forests Intensive Forest Health monitoring sites (Level II) in the UK (Hall et al. 2003).	CV ± 7%
	G3	210	210	New value (equivalent to 2.94 kg N ha^{-1} a^{-1}) for managed coniferous woodland. (Previous value 7 kg N ha^{-1} a^{-1}).		CV ± 27%
	G1&G3	0	0	Unmanaged woodland, uptake set to zero assuming no harvesting.		NA
	F4.11 F4.2	36 36	36 36	New value, equivalent to 0.5 kg N ha^{-1} a^{-1}. (Previous value 4 kg N ha^{-1} a^{-1}).	Perkins (1978), Rawes and Heal (1978), Reynolds et al. (1987), Batey (1982), Gordon et al. (2001).	Uncertainty range: 0.05–4.0 kg N ha^{-1} a^{-1}

Table GB-2 (continued). Summary of UK critical load values and justification for their use.

Critical loads parameter (units)	EUNIS code	Min. value	Max. value	Data sources/ Methods used	Justification	Uncertainty
N_u (eq ha^{-1} a^{-1})	E1.26	714	714	Equivalent to 10 kg N ha^{-1} a^{-1}.		Not calculated.
	E1.7 E3.5	81 81	81 81	New value, equivalent to 1.14 kg N ha^{-1} a^{-1} (Previous value 1 kg N ha^{-1} a^{-1}).	Frissel (1978).	CV ±44%
	E4.2 D1	36 36	36 36	Equivalent to 0.5 kg N ha^{-1} a^{-1}.	UK experts agreed to apply the same value as used for F4.11 & F4.2.	Uncertainty range: 0.05–4.0 kg N ha^{-1} a^{-1}
	C1 C2	0 0	212 195	= f N_u based on values for G1 & G3 and percentage forest in catchments.		Not calculated.
N_i (eq ha^{-1} a^{-1})	G1 G3 G1&G3	71 71 71	214 214 214	N_i values assigned according to soil type.	Mapping Manual (UBA 1996), Hornung et al. (1995b), Curtis (2002).	Uncertainty range: 1–2 kg N ha^{-1} a^{-1} for more mineral soils. Uncertainty range: 2–3 kg N ha^{-1} a^{-1} for more organic soils.
	F4.11	392	535	= N_i + N_{fire}, (N_{fire} = 4.5 kg N ha^{-1} a^{-1}).	Inclusion of N_{fire} UBA (1996). N_{fire} values: Chapman (1967), Allen (1964).	N_{fire} uncertainty range: 2.3–6.4 kg N ha^{-1} a^{-1}
	F4.2	1142	1285	= N_i + N_{fire} (N_{fire} = 15 kg N ha^{-1} a^{-1}).		N_{fire} uncertainty range: 9.1–26 kg N ha^{-1} a^{-1}
	E1.26 E1.7 E3.5 E4.2 D1	71 71 71 71 71	214 214 214 214 214	N_i values assigned according to soil type.	Mapping Manual (UBA 1996), Hornung et al. (1995), Curtis (2002).	Uncertainty range: 1–2 kg N ha^{-1} a^{-1} for more mineral soils; 2–3 kg N ha^{-1} a^{-1} for more organic soils.
	C1 C2	11 71	214 214	N_i values catchment weighted by soil type.		Not calculated
$N_{le(acc)}$ (eq ha^{-1} a^{-1})	G1	214	214	New value, equivalent to 3 kg N ha^{-1} a^{-1} (Previous value 6 kg N ha^{-1} a^{-1}).	Emmett (submitted).	Uncertainty range: 1–3 kg N ha^{-1} a^{-1}.
	G3	286	286	New value, equivalent to 4 kg N ha^{-1} a^{-1} (Previous value 6 kg N ha^{-1} a^{-1}).	Emmett et al. (1993), Emmett & Reynolds (1996).	Uncertainty range: 1–5 kg N ha^{-1} a^{-1}.
	G1&G3 F4.11 F4.2 E1.26 E1.7 E3.5 E4.2 D1	– – – – – – – –	– – – – – – – –	Empirical nutrient nitrogen critical loads used, therefore $N_{le(acc)}$ not assigned.		NA
	C1 C2	– –	– –	Not used in FAB.		NA
N_{de} (eq ha^{-1} yr^{-1})	G1 G3	71 71	286 286	N_{de} values assigned according to soil type.	Mapping Manual (UBA, 1996), Hornung *et al.* (1995), Curtis (2002).	Uncertainty range: 0–1 kg N ha^{-1} yr^{-1} for aerated soils; 2–3 kg N ha^{-1} yr^{-1} for sites with waterlogged soils and low deposition; 4–5 kg N ha^{-1} yr^{-1} for sites with waterlogged soils and high deposition.
	G1&G3 F4.11 F4.2 E1.26 E1.7 E3.5 E4.2 D1	71 71 71 71 71 71 71 71	286 286 286 286 286 286 286 286	N_{de} values assigned according to soil type. Only used in $CL_{min}(N)$ as empirical nutrient nitrogen critical loads applied.		

Table GB-2 (continued). Summary of UK critical load values and justification for their use.

Critical loads parameter (units)	EUNIS code	Min. value	Max. value	Data sources/ Methods used	Justification	Uncertainty
N_{de} (eq ha^{-1} yr^{-1})	C1	11	285	Uses catchment weighted N_{de} values (based on soil type) instead of f_{de}		Not calculated.
	C2	71	286			
Precipitation surplus Q (mm)	G1	57	3130	1km runoff data based on 30-year (1941–1970) mean rainfall data.	Used in SMB equation for acidity critical loads for wood-land habitats.	Not calculated on data used. Arnell et al. (1990) quote CV ± 23% for UK catchment runoff.
	G3	100	3393			
	G1&G3	83	3631			
	F4.11	–	–	SMB not used, therefore Q not assigned.		NA
	F4.2	–	–			
	E1.26	–	–			
	E1.7	–	–			
	E3.5	–	–			
	E4.2	–	–			
	D1	–	–			
	C1	101	3364	1km catchment-weighted runoff based on mean rain-fall data for 1941–1970 for GB and 1961–1990 for NI.	Used in FAB.	Not calculated.
	C2	212	2877			
K_{gibb} (m^6 eq^{-2})	G1	9.5	950	Minimum value applied to organic soils and maximum value applied to mineral soils.	Mapping Manual (UBA 1996), Hall et al. (2001a, 2003).	Not calculated. Suutari et al. (2001) quote uncertainty ± 20%.
	G3	9.5	950			
	G1&G3	9.5	950			
	F4.11	–	–	SMB not used, therefore K_{gibb} not assigned.		NA
	F4.2	–	–			
	E1.26	–	–			
	E1.7	–	–			
	E3.5	–	–			
	E4.2	–	–			
	D1	–	–			
	C1	–	–	Not used in FAB.		NA
	C2	–	–			

Changes in critical loads calculation methods

The following changes have been made to the methods for calculating critical loads in the UK:

Acidity critical loads for peat soils: Changes in the number and distribution of peat-dominated 1km squares have resulted from the revision of the GB soil databases, including revisions to those soils classified as peat soils in Scotland. The method used to calculate acidity critical loads for peat soils has been reviewed and a new method adopted. The updated method sets the critical load to the amount of acid deposition that would give rise to an effective rain pH of 4.4. This pH reflects the buffering effects of organic acids upon peat drainage water pH (refer to Hall et al. 2003 for further details). The critical loads are thus calculated as:

$$CL(A) = Q \cdot [H]_{crit}$$

where:
Q = runoff, m
$[H]_{crit}$ = critical H concentration equivalent to pH 4.4

The runoff data used (mean values for 1941–1970) are the same as those used in the SMB and FAB models.

Acidity critical loads for freshwaters: Whilst no changes have been made to the methods used, several of the parameters used to define nitrogen processes have been updated. The critical chemical criterion also remains unchanged (i.e. ANC_{crit} = 0 μeq l^{-1}). However, questions remain over the most appropriate value of ANC_{crit} for UK surface waters because of their great variety in terms of water chemistry and catchment hydrology. UK experts acknowledge there is a growing body of evidence to suggest that the current value does not provide adequate protection for freshwater biota, and further research is being conducted. The changes in mean values of $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$ for all habitats are given in Table GB-3.

Table GB-3. Summary of changes in the mean values of $CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$.

Critical load	Broad habitat[1] (EUNIS class)	Previous (Feb 2001) mean value (eq ha^{-1} a^{-1})	Updated (Feb 2003) mean value (eq ha^{-1} a^{-1})	Difference between previous and updated means[2]
$CL_{max}(S)$	Acid grassland (E1.7 & E3.5)	1046	824	Decrease 21.2%
	Calcareous grassland (E1.26)	2768	3920	Increase 29.4%
	Dwarf shrub heath (F4.11 & F4.2)	843	843	No change
	Coniferous woodland (managed) (G3)	1828	1965	Increase 7.0%
	Broadleaved woodland (managed) (G1)	2101	2660	Increase 21.0%
	Unmanaged woodland (G1&G3)	Not mapped	3243	
	Bogs (D1)	Not mapped	901	
	Montane (E4.2)	Not mapped	557	
	Standing open waters, rivers & streams (C1 & C2)	5019	3636	Decrease 27.6%
$CL_{min}(N)$	Acid grassland (E1.7 & E3.5)	351	367	Increase 4.4%
	Calcareous grassland (E1.26)	214	889	Decrease 26.8%
	Dwarf shrub heath (F4.11 & F4.2)	580	851	Increase 31.8%
	Coniferous woodland (managed) (G3)	782	478	Decrease 38.9%
	Broadleaved woodland (managed) (G1)	747	663	Decrease 11.2%
	Unmanaged woodland (G1&G3)	Not mapped	245	
	Bogs (D1)	Not mapped	343	
	Montane (E4.2)	Not mapped	318	
	Standing open waters, rivers & streams (C1 & C2)	288	307	Increase 6.2%
$CL_{max}(N)$	Acid grassland (E1.7 & E3.5)	1397	1192	Decrease 14.7%
	Calcareous grassland (E1.26)	3687	4809	Increase 23.3%
	Dwarf shrub heath (F4.11 & F4.2)	1424	1695	Increase 16.0%
	Coniferous woodland (managed) (G3)	2611	2443	Decrease 6.4%
	Broadleaved woodland (managed) (G1)	2599	3323	Increase 21.8%
	Unmanaged woodland (G1&G3)	Not mapped	3488	
	Bogs (D1)	Not mapped	1244	
	Montane (E4.2)	Not mapped	874	
	Standing open waters, rivers and streams (C1 & C2)	8031	5308	Decrease 33.9%

[1]The "broadleaved, mixed and yew woodland" broad habitat is separated into "broadleaved woodland (managed)" and "unmanaged (ancient and semi-natural) coniferous and broadleaved woodland" abbreviated to "Unmanaged woodland" above; the latter includes Atlantic oak woods and unmanaged coniferous woodland.
[2]An increase or decrease in the mean critical load values does not necessarily mean that all values for that habitat have increased or decreased, some may have increased in value and others decreased in value.

Empirical critical loads of nutrient nitrogen: The empirical critical loads of nutrient nitrogen used for UK mapping have been revised in light of the conclusions of the Bern workshop held in November 2002 (Achermann and Bobbink 2003, Hall et al. 2003). The critical loads for managed woodlands are calculated using the nitrogen mass balance equation, i.e.:

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

The changes made to the nutrient nitrogen critical load values assigned in the UK are summarised in Table GB-4.

Uncertainties in the calculation of critical loads

A preliminary analysis of the uncertainties in some of the input parameters required for calculating critical loads is presented below. Where values have been taken from default ranges given in the literature, these ranges have been used to calculate the percentage uncertainty around the value used. Where input parameters are based on experimental data, these have been analysed to give a coefficient of variation (CV). In a few cases uncertainty ranges have been taken directly from the literature or expert judgement has been used.

Table GB-4. Summary of changes in nutrient nitrogen critical loads applied in the UK.

UPDATE (Feb 2003)			PREVIOUS (Feb 2001)		
Broad Habitat[1]	**EUNIS classes**	**$CL_{nut}(N)$ values (kg N ha^{-1} a^{-1})**	**Ecosystem**	**$CL_{nut}(N)$ categories**	**$CL_{nut}(N)$ values (kg N ha^{-1} a^{-1})**
Acid grassland	Dry acid and neutral closed grassland (E1.7) Moist or wet oligotrophic grassland (E3.5)	15 15	Acid grassland[3]	Neutral-acid species rich grassland Montane sub-alpine grassland Peat (bog)	25 12.5 10
Calcareous grassland	Semi-dry calcareous grassland (E1.26)	20	Calcareous grassland	Calcareous species-rich grassland	50
Dwarf shrub heath	Northern wet heaths (F4.11) Dry heaths (F4.2)	15 12	Heathland[4]	Lowland wet & dry heaths Species-rich heath/acid grassland Upland Calluna moor Arctic & alpine heath Peat (bog)	17 17 15 10 10
Bogs	Raised and blanket bogs (D1)	10		Mapped as part of acid grassland and heathland as above	
Coniferous woodland (managed)[2]	Coniferous woodland (G3)	8.9 – 13.9 mean 10.7	Coniferous woodland	Minimum of empirical value (13 kg N) or N mass balance (higher values)	13
Broadleaved woodland (managed)[2]	Broadleaved woodland (G1)	10.9 – 15.9 mean 12.3	Deciduous woodland	Minimum of empirical value (17 kg N) or N mass balance	15 – 17 mean 16.1
Unmanaged woodland	Broadleaved woodland (G1&G3-GF) (effects on ground flora)	12	Not mapped		
Broadleaved woodland (Atlantic oak woods)	Broadleaved woodland (G1-LA) (effects on epiphytic lichens)	10	Not mapped		
Montane	Moss & lichen dominated summits (E4.2)	7	Not mapped	Not specifically mapped - areas included in acid grassland & heathland	
Supralittoral sediment	Shifting coastal dunes (B1.3) Stable dune grassland (B1.4)	15 15	Not mapped		

[1]The “broadleaved, mixed and yew woodland” broad habitat is separated into “broadleaved woodland (managed)”, “broadleaved woodland (Atlantic oak woods)” and “unmanaged (ancient & semi-natural) coniferous and broadleaved woodland” (excluding Atlantic oak woods) abbreviated to “Unmanaged woodland” above; the latter includes unmanaged coniferous woodland.

[2] Nitrogen mass balance used (i.e. $N_u + N_i + N_{de} + N_{le(acc)}$).

[3] Mean value on acid grassland map = 22.7 kg N ha^{-1} a^{-1}.

[4] Mean value on heathland map = 14.9 kg N ha^{-1} a^{-1}.

This is preliminary work, reported here for information only. The uncertainty estimates do not form part of the official call for data and are not intended for use in any policy or emission scenario negotiations.

The following paragraphs describe the methods used and the uncertainty values are summarised in Table GB-2.

Base cation deposition: At the present time estimates of uncertainties in the UK base cation deposition data are not available. Draaijers et al. (1996) have used error propagation to estimate the random and systematic errors in total (wet and dry) deposition for an average 10×20km^2 grid cell in Europe. Their worst case uncertainty ranges for random and systematic errors are quoted in Table GB-2.

Base cation weathering: The UK uses the mid-range empirical soil acidity critical loads values to define base cation weathering (Hall et al. 2003). These values are consistent with work on soil weathering rates by Langan et al. (1995). Therefore the ranges provide a good estimate of uncertainty. The median uncertainty is ±50%, which corresponds with the expected uncertainty using the "Skokloster" method of assigning empirical critical loads given in Sverdrup et al. (1990).

Base cation, calcium and nitrogen uptake in woodland habitats: These uptake values are based on measurements made at the ten UNECE (ICP Forests) intensive forest health monitoring sites (Level II) in the UK. Uncertainties in the values used are determined by calculating the CVs for the three broadleaf and seven conifer plots, and these then represent the uncertainty estimates for the respective woodland types. For unmanaged woodlands, uptake terms are set to zero on the assumption that no harvesting takes place.

Base cation uptake in non-woodland habitats: Base cation uptake is assumed to be negligible for all non-woodland habitats mapped, except calcareous grassland (see Table GB-2). Uncertainty in the latter cannot currently be quantified.

Nitrogen uptake in non-woodland habitats: For acid grassland, the uncertainty is expressed as a CV, based on data for six sites from Frissel (1978). For the dwarf shrub heath, bog and montane habitats the uptake values were based on Perkins (1978), Rawes and Heal (1978) and Reynolds et al. (1987) and the uncertainty expressed as the total range of the published values.

Nitrogen losses through fire: Values for nitrogen losses through fire could only be quantified and applied to the dwarf shrub heath habitat. Uncertainty ranges for this term were calculated by assuming burn frequencies of between 7 and 20 years as suggested by Allen (1964) for blanket peats in the Pennines.

Nitrogen immobilisation: The nitrogen immobilisation values are based on the dominant soil type in each 1km grid square (Hall et al. 1998). Uncertainty ranges have been defined from default ranges published Sverdrup et al. 1990 (page 55).

Denitrification: Denitrification values are also based on the dominant soil type in 1km grid square (Hall et al. 1998). Uncertainty ranges have been defined from values published in Appendix 1 of Grennfelt and Thörnelöf (1992).

Acceptable nitrogen leaching: The nitrogen leaching value applied to coniferous woodland is at the upper end of the range given by Hornung et al. (1995b). For broadleaved woodland a range of 1–3 kg N ha^{-1} a^{-1} is suggested by Emmett (pers. comm. 2002) with a recommendation to use the top of this range.

Precipitation surplus (runoff): Although uncertainty has not been estimated on the actual data set used, Arnell et al. (1990) have calculated a median coefficient of variation for annual runoff of 23% for UK catchments.

Gibbsite equilibrium constant: The uncertainty in this term has not been determined by the UK NFC. Instead, the uncertainty range of ±20% (Suutari et al. 2001) is assumed to apply to the UK.

Empirical nutrient nitrogen critical loads: These critical loads (Achermann and Bobbink 2003) are expressed as a range, indicating the variation in sensitivity within an ecosystem, for example, because of differences in nutrient status, management etc. The uncertainty in the critical load range is expressed qualitatively, by assessing the critical load as being "reliable ##", "quite reliable #" and "expert judgement (#)". Experts in the UK believe an estimate of uncertainty needs to combine both aspects. Hence, uncertainty has been estimated using triangular functions, with the selected UK mapping value as the maxima of the distribution, and the ends of the ranges representing the tails of the distributions. Additionally, the critical load ranges have been extended for each reliability category to incorporate an element of uncertainty, as follows:

##	range as published
#	±5 kg N ha^{-1} a^{-1} beyond the range
(#)	±10 kg N ha^{-1} a^{-1} beyond the range

The only exception to this rule was the critical load for bogs (EUNIS class D1), where the UK is using the upper limit of the range as its mapping value (i.e. 10 kg N ha^{-1} a^{-1}); to deal with this the maximum of the range was increased to 12 kg N ha^{-1} a^{-1} to provide a reasonable estimate of uncertainty.

Further work on uncertainties in the calculation of critical loads and their exceedances in the UK is ongoing.

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Appendix A. The polar stereographic projection (EMEP grid)

To make critical loads useful for the work under the LRTAP Convention, one has to be able to compare them to deposition estimates. Deposition of sulphur and nitrogen compounds have earlier been reported by EMEP on a 150×150 km^2 grid covering (most of) Europe, but in recent years depositions have also become available on a 50×50 km^2 grid. Both are special cases of the so-called polar stereographic projection, which is described in the following.

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 in the CCE Status Report 2001, p. 182.) Consequently, the coordinates x and y are obtained from the geographical longitude λ and latitude ϕ (in radians) by the following equations:

$$x = x_p + M \tan\left(\frac{\pi}{4} - \frac{\phi}{2}\right)\sin(\lambda - \lambda_0) \tag{A.1}$$

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 centre 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 ($i\pm\frac{1}{2}$, $j\pm\frac{1}{2}$).

The 150×150 km2 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 50×50 km2 grid (EMEP50 grid):

The eulerian dispersion model of EMEP/MSC-W produces concentration and deposition fields on a 50×50 km^2 grid with the parameters (see also www.emep.int):

$$d = 50\text{km}, \quad (x_p, y_p) = (8{,}110), \quad \phi_0 = \frac{\pi}{3} = 60°\text{N}, \quad \lambda_0 = -32°\,(\text{i.e. } 32°\text{W}) \tag{A.8}$$

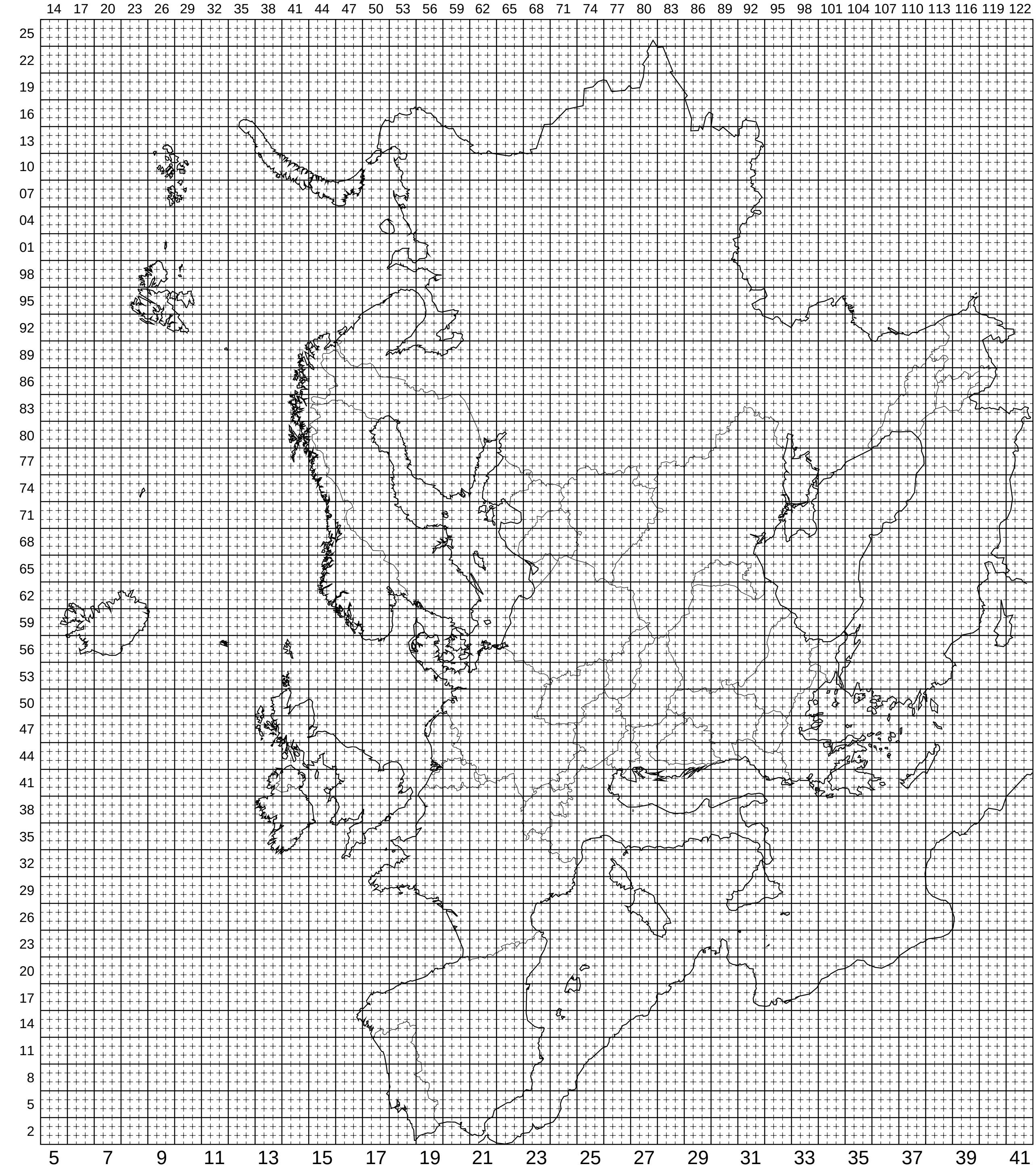
yielding M=237.7314...

An EMEP150 grid cell (i,j) contains 3×3=9 EMEP50 grid cells (m,n) with all combinations of the indices $m=3i-2$, $3i-1$, $3i$ and $n=3j-2$, $3j-1$, $3j$. The part of the two EMEP grid systems covering Europe is shown in Figure A-1.

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 subroutine  llemep
```

EMEP150 coordinates are obtained by calling the above subroutine with *par*(1)=150, *par*(2)=3 and *par*(3)=37; and the EMEP50 coordinates are obtained with *par*(1)=50, *par*(2)=8 and *par*(3)=110.



F
igure A-1. The EMEP150 grid (solid lines) and EMEP50 grid (dashed lines). The numbers at the bottom and right are EMEP150 grid indices; those at the top and left are EMEP50 grid indices.

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 == 0. .and. ey == 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 subroutine  emepll
```

The area of an EMEP grid cell:

As mentioned above, the stereographic projection does not preserve areas, e.g. a 50×50 km^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 CCE Status Report 1997:

$$I(u,v) = \iint \frac{2dudv}{(1+u^2+v^2)^2} = \frac{v}{\sqrt{1+v^2}} \arctan \frac{u}{\sqrt{1+v^2}} + \frac{u}{\sqrt{1+u^2}} \arctan \frac{v}{\sqrt{1+u^2}} \tag{A.10}$$

These two equations allow the calculation of the area of the EMEP grid cell (i,j) by setting (x_1,y_1)=(i–½, j–½) and (x_2, y_2)=(i+½, j+½).

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 function  aremep
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 function femep
```

Reference:

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.

Appendix B. Correcting for sea salts

Since acidity critical loads are generally compared with *anthropogenic* S (and N), i.e. the deposition due to sea spray is not included, the base cation and chloride deposition in the charge balance – from which critical loads are derived with the SMB model – have to be corrected for sea salt contributions as well. The aim of this Appendix is to provide the historical background and original sources of data for the composition of sea salts as well as a general formula to carry out a sea-salt correction.

The constancy of ratios between the salts in ocean water unaffected by land drainage was firmly established by Dittmar (1884). Dittmar's results were so consistent that later investigations introduced only minor changes, mostly with respect to more accurate atomic weights(!). Here we report the values given in the classic textbook by Sverdrup et al. (1946), which are in turn based on data by Lyman and Fleming (1940). Table B-1 lists the amounts of the six major ions in seawater, their atomic weights and the calculated equivalents (see Eq. C.1 in Appendix C).

Table B-1. Major ions in the seawater and their abundance.

Ion	Amount in seawater[a)] ($g\ kg^{-1}$)	Molecular weight of ion[b)] ($mol\ g^{-1}$)	Equivalents in seawater ($eq\ kg^{-1}$)
Ca^{2+}	0.4001	40.078	0.01997
Mg^{2+}	1.2720	24.305	0.10467
K^{+}	0.3800	39.098	0.00972
Na^{+}	10.5561	22.990	0.45916
Cl^{-}	18.9799	35.453	0.53545
SO_4^{2-}	2.6486	96.064	0.05514

[a)]Sverdrup et al. (1946; p. 173); [b)]Weast et al. (1989)

The equivalent sum of base cations does not exactly match that of chloride and sulphate, since other ions such as Br, F, Sr, boric acid and bicarbonate, which occur in traces in seawater, are not included here.

Depositions of base cations, sulphur and chloride (given in equivalents) are corrected by assuming that either all sodium or all chloride is derived from sea salts, using the formula

$$X^*_{dep} = X_{dep} - r_{XY} \cdot Y_{dep} \tag{B.1}$$

where X=Ca, Mg, K, Na, Cl or SO_4, Y=Na or Cl, r_{XY} is the ratio of ions X to Y in seawater and the star denotes the sea-salt corrected deposition. Ratios r_{XY} can by computed from the last column of Table B-1 and are shown in Table B-2 with 3-decimal accuracy.

Table B-2. Ion ratios r_{XY}=[X]/[Y] (in $eq\ eq^{-1}$) in seawater (computed from Table B-1).

	X					
Y	**Ca**	**Mg**	**K**	**Na**	**Cl**	**SO_4**
Na	0.043	0.228	0.021	1	1.166	0.120
Cl	0.037	0.195	0.018	0.858	1	0.103

Note that for arbitrary ions X, Y and Z the relationships $r_{YX} = 1/r_{XY}$ and $r_{XY} \cdot r_{YZ} = r_{XZ}$ hold. If Na (Cl) is chosen to correct for sea salts, $Na^*_{dep} = 0$ ($Cl^*_{dep} = 0$).

References:

Dittmar W (1884) Report on researches into the composition of ocean water, collected by H.M.S. *Challenger*, during the years 1873–1876. *Phys. Chem.* 1: 1-251.

Lyman J, Fleming RH (1940) Composition of sea water. *Journal of Marine Research* 3: 134-146.

Sverdrup HU, Johnson MW, Fleming RH (1946) *The Oceans – Their Physics Chemistry and General Biology.* Prentice-Hall, New York, 1087 pp.

Weast RC, Lide DR, Astle MJ, Beyer WH (eds) (1989) *CRC Handbook of Chemistry and Physics (70th edition).* CRC, Boca Raton, USA.

Appendix C. Unit conversions

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:

$$1\,\mathrm{g\,X} = \frac{1}{M}\,\mathrm{mol\,X} = \frac{z}{M}\,\mathrm{eq\,X} \tag{C.1}$$

Obviously, moles and equivalents are the same for z=1. For depositions one has:

Table C-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 C-2. Conversion factors for **nitrogen** deposition (g stands for grams of N; M=14, z=1). For conversion multiply by the factors given in the table.

From: To:	mg/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

$$1\,\mathrm{ppm} = 1000\,\mathrm{ppb} = \frac{M}{V_0}\,\mu\mathrm{g/m}^3 \tag{C.2}$$

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. C.2 by:

$$V_1 = V_0\,\frac{T_1}{T_0}\cdot\frac{p_0}{p_1} \tag{C.3}$$

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 C-3. Conversion factors for concentrations of common pollutants at two different temperatures (1 ppm=1000 ppb).

		From ppb to μg/m^3, multiply by:		From μg/m^3 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 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:

$$[\mathrm{A}^{m\pm}]^x = K[\mathrm{B}^{n\pm}]^y \tag{C.4}$$

where the square brackets [...] denote concentrations in mol/L (where L stands for liter), implying for the equilibrium constant K the units (mol/L)$^{x\text{-}y}$. If the concentrations are to be expressed in eq/V, where V is an arbitrary volume unit with 1L=10^{c}V, then the equilibrium constant in the new units is given by

$$K' = K \cdot 10^{c(y-x)} \frac{m^x}{n^y}\ (\mathrm{eq/V})^{x-y} \tag{C.5}$$

Note: To convert to mol/V, set m=n=1 in the above equation; and to covert 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 1: The gibbsite equilibrium is given by $[\mathrm{Al}^{3+}]$=$K[\mathrm{H}^+]^3$, 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)$^{-2}$.

The above reasoning can also be used for converting exchange constants. For example, the Gapon equation for Al-Bc exchange can be written as

$$\frac{E_{Al}}{E_{Bc}}[Bc^{2+}]^{1/2} = k_{AlBc}[Al^{3+}]^{1/3} \tag{C.6}$$

and since exchangeable fractions are dimensionless, Eq. C.5 can be used.

Example 2: Let $\log_{10}k_{AlBc}$=–2; then k_{AlBc}=10^{-2}=0.01(mol/L)$^{1/6}$ (x=1/2, y=1/3). Since m=2 and n=3, one gets when converting to eq/m^3, i.e. c=–3, $k_{AlBc}=10^{-2}\cdot10^{-3\cdot(1/3-1/2)}\cdot(2^{1/2}/3^{1/3})$ = 0.03100806 (eq/m^3)$^{1/6}$. And for k_{HBc} the multiplier to obtain (eq/m^3)$^{-1/2}$ is 0.002$^{1/2}$=0.0447213.