CCE Status Report 2008

Critical Load, Dynamic Modelling and Impact Assessment in Europe

Netherlands Environmental Assessment Agency

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Cover design: Jaap Wolters

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wge Working Group on Effects of the Convention on Long-range Transboundary Air Pollution

Convention on Long-range Transboundary Air Pollution

ICP M&M Coordination Centre for Effects

Netherlands Environmental Assessment Agency

Life

LRTAP Long-range Transboundary Air Pollution

Critical Load, Dynamic Modelling and Impact Assessment in Europe
CCE Status Report 2008

Report 500090003

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ISBN: 978-90-6960-211-0

This research has been performed by order and for the account of the Directorate for Climate Change and Industry of the Dutch Ministry of Housing, Spatial Planning and the Environment within the framework of PBL project M/500090, 'Coordination Centre for Effects (CCE)'; for the account of the European Commission LIFE III Programme within the framework of PBL project E/555065 'European Consortium for Modelling Air Pollution and Climate Strategies (EC4MACS)' and for the account of (the Working Group on Effects within) the trust fund for the partial funding of effect-oriented activities under the Convention on Long-range Transboundary Air Pollution.

A comprehensive reference to the report reads as 'Hettelingh JP, Posch M, Slootweg J (eds.) (2008) Critical load, dynamic modelling and impact assessment in Europe: CCE Status Report 2008, Coordination Centre for Effects, Netherlands Environmental Assessment Agency, www.pbl.nl/cce (www.mnp.nl/cce until 2009).

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Acknowledgements

The methods and results presented in this report are the products of a collaboration within the Effects Programme of the UNECE Convention on Long-range Transboundary Air Pollution, involving many institutions and individuals throughout Europe. National Focal Centres, whose reports regarding modelling and mapping activities appear in Part II, are acknowledged for their contribution.

In addition, the Coordination Centre for Effects (CCE) acknowledges:
- The Directorate for Climate Change and Industry of the Dutch Ministry of Housing, Spatial Planning and the Environment, and Mr J. Sliggers in particular, for their continued support,
- The Working Group on Effects, Task Force of the International Co-operative Programme on the Modelling and Mapping of Critical Levels and Loads and Air Pollution Effects, Risks and Trends, in particular, for their collaboration and assistance,
- The EMEP Meteorological Synthesizing Centres East and West and the EMEP Centre for Integrated Assessment at the International Institute for Applied Systems Analysis, for their collaboration in the field of atmospheric dispersion and integrated assessment modelling,
- The UNECE secretariat of the Convention on Long-range Transboundary Air Pollution, for its valuable support, including the preparation of official documentations,
- The European Commission's LIFE III Programme, for co-funding the participation of the CCE in the European Consortium for Modelling Air pollution and Climate Strategies (EC4MACS),
- The Nature and Biodiversity Unit of the European Commission's Environment Directorate-General, for supporting the availability of data on Natura 2000 areas.
- Karin van Doremalen, for managing the institutional routing and printing, Annemieke Righart, for editing the language in chapter 4, 5 and 7, and Jaap Wolters, Valentijn van Hees and Filip de Blois of the RIVM graphics department for taking care of the lay-out of this report.

Summary

Critical load, dynamic modelling and impact assessment in Europe: CCE Status Report 2008

This report describes the 2008 European database on spatially-explicit critical loads and dynamic modelling data (2008 CL database). It analyses the underlying fundamentals of the 2008 CL database, and provides examples of its use in the assessing of the magnitude and location of the risk of current and future impacts of nitrogen and sulphur on ecosystems in Europe, including the Natura 2000 areas. The report emphasises the risk of impacts caused by the deposition of oxidised and reduced nitrogen.

This 2008 CL database is important because it is designed to support the revision – expected to commence in the coming year – of the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone, under the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) of the United Nations Economic Commission for Europe. The information is also available to the Economic Commission, in support of its Thematic Strategy on Air Pollution and to the European Environment Agency for the update of its core set of indicators.

In the autumn of 2007, the Coordination Centre for Effects (CCE) was requested by the Working Group on Effects under the LRTAP-Convention, to revise the existing (2006) European database on critical loads and dynamic modelling parameters. This 2006 CL database was used for the review of the Gothenburg protocol and the National Emission Ceiling Directive of the European Commission. Twenty parties to the Convention – nineteen from Europe plus Canada – responded to the CCE call for data, all of which submitted spatially-specific modelled critical loads for acidification, 19 of which provided data on critical loads for eutrophication, 13 on empirical critical loads and 12 on dynamic modelling.

Spatially-specific critical loads for the remaining parties in the EMEP domain were calculated and compiled from the CCE background database, using most recent information on critical load input variables and land-cover data.

The 2008 CL database covers a broader area of sensitive ecosystems within Europe than the 2006 CL database, mainly because the CCE background database now also includes critical loads for semi-natural vegetation.

Emission data, compiled by the EMEP Centre for Integrated Assessment Modelling (CIAM), and information on deposition from EMEP Meteorological Synthesizing Centre West, were used by the CCE to identify areas at risk of eutrophication and acidification.

The 2008 CL database was used to re-calculate the size of the natural areas within Europe which were at risk of eutrophication in 2000. This has shown that these areas were about 3% larger – covering 49% of nature – than was previously calculated with data from the 2006 CL database. For the EU25 countries, in the same year, this area was even 12% larger, covering about 77% of the ecosystems.

The natural areas at risk of acidification do not vary dramatically between both CL databases. In 2000, the areas at risk of acidification within Europe and the EU25 countries, cover about 11% and 18%, respectively, based on the 2008 CL database. A complete description of this analysis is included in Chapter 1, while a detailed review of the data in the 2008 CL database can be found in Chapter 2.

Bearing in mind that, for eutrophication, the exceedances of critical loads are not likely to disappear under current legislation, this report elaborates the results of using dynamic models to improve the understanding of the consequences of long-term exceedance. Chapter 2 summarises data and results of the dynamic modelling part of the 2008 CL database, including data obtained from the CCE background database for other countries. Chapter 3 provides an illustrative use of dynamic modelling in the context of integrated assessment modelling, to support European air pollution abatement agreements.

In addition to these practical results, which were obtained through the application of the European critical loads database, this report also describes the progress made in the research to help quantify the effects of air pollution on the change in biodiversity, on a European scale. These preliminary results are largely based on assessments of recorded impacts around Europe and tentative dose–response relationships which are subject to exceedances of empirical critical loads (Chapter 4). Preliminary findings on the relationship between critical load exceedance and species loss in the Netherlands are described in Chapter 5.

Chapter 6 describes the possible use of the dose–response relationships from Chapter 4 in the context of integrated assessments and scenario analyses, in support of European air pollution abatement agreements.

The increasing need under the LRTAP Convention for collaboration with Eastern European Caucasian and Central Asian (EECCA) parties to the Convention, is reflected also in this CCE report, in which data were compiled on critical loads of heavy metals, in collaboration with the Wageningen University, and on the exceedances of critical loads of heavy metals, in collaboration with EMEP's Meteorological Synthesizing Centre-East (Chapter 7).

Part II of this report includes the results of the 2007/2008 collaboration with National Focal Centres.

Finally, Appendix A summarises the derivation of European N and S deposition histories and scenarios that are used for this report, Appendix B consists of the 'Instruction to the Call for Data' which was provided to National Focal Centres in support of their data submission, whereas Appendix C provides examples of the relationship between nitrogen deposition and ecosystem services that affect human well being.

Key words: Acidification, air pollution effects, biodiversity, critical loads, dose–response relationships, dynamic modelling, eutrophication, exceedance, LRTAP Convention.

Rapport in het kort

Kritische drempel-, dynamische modellering- en impactanalyse in Europa: CCE Status rapport 2008.

Dit rapport behandelt de 2008 Europese database van ruimtelijk bepaalde kritische drempels en dynamische modelleringgegevens (2008-CL database). Het beschrijft de achtergrond van de 2008-CL database en het gebruik ervan voor de analyse van de huidige en toekomstige omvang en locatie van effectrisico's van luchtverontreiniging op Europese ecosystemen, inclusief Natura 2000 gebieden.

De 2008 CL-database is belangrijk omdat het is ontwikkeld ter ondersteuning van de revisie – die komend jaar zal beginnen – van het Gotenburg protocol voor de reductie van verzuring, vermesting en troposferisch ozon onder de Conventie voor Grootschalige Grens Overschrijdende Lucht Verontreiniging (LRTAP Conventie) van de Verenigde Naties Economische Commissie van Europa. De met de database af te leiden informatie over de gevolgen van luchtverontreiniging wordt ook gebruikt door de Economische Commissie bij de ondersteuning van zijn Thematische Strategie Luchtverontreiniging en door het Europese Milieu Agentschap voor de 'core set of indicators'

In de herfst van 2007 verzochten relevante werkgroepen onder de LRTAP Conventie aan het Coordination Centre for Effects (CCE) om de bestaande (2006) Europese CL-database te reviseren. De 2006 CL-database was gebruikt voor de review van het Gotenburg protocol (CLRTAP-Conventie) en van de Nationale Emissie Plafond onder de Europese Commissie. Twintig partijen onder de Conventie – negentien Europese landen en Canada – beantwoordden de door het CCE methodologisch onderbouwde call voor gegevens.

Twintig landen leverden berekende kritische drempels voor verzuring, waarvan negentien landen ook berekende kritische drempels voor vermesting verschaften. Twaalf landen leverden gegevens over dynamische modellering en dertien landen verschafte empirische critical loads. Alle gegevens zijn voorzien van ruimtelijk relevante informatie op Europese schaal.

Voor de overige landen in het EMEP-domein (geografisch gebied onder de LRTAP Conventie waarbinnen de dispersie van luchtverontreiniging wordt berekend en ruimtelijk bepaald) werd de gereviseerde CCE-achtergrond database gebruikt, inclusief recente input data voor berekeningsmodel van kritische waarden en landbedekking.

De 2008 CL-database beslaat een grotere oppervlakte van gevoelige ecosystemen dan de 2006 CL-database. Een belangrijke reden is dat de 2008 CL-database voor veel landen nu ook kritische drempels voor semi-natuurlijke vegetatie bevat.

De overschrijding van critical loads door atmosferische depositie zijn door het CCE berekend en ruimtelijk weergegeven. Daartoe is gebruik gemaakt van emissie- en dispersie data van EMEP centra op het gebied van respectievelijk geïntegreerde modellering (Centre for Integrated Assessment Modelling, CIAM te IIASA, Oostenrijk) en dispersiemodellering (Meteorologische Synthesizing Centre West, MSCW te Noorwegen).

Resultaten geven aan dat het risico voor vermesting dat is berekend met de 2008 CL-database hoger is in vergelijking tot de 2006 CL-database. De Europese oppervlakten van de natuurlijke gebieden met vermestingrisico in 2000 lag ca. 3% hoger terwijl in de EU25 het percentage ca. 12% hoger ligt. Voor verzuring zijn de verschillen tussen het gebruik van de 2006 en 2008 CL-database niet opvallend.

De overschrijdingen van kritische drempels voor vermesting zullen naar waarschijnlijkheid lange tijd aanhouden. Daarom is het belangrijk te kunnen analyseren hoe veranderingen in de hoogte en ruimtelijke locatie van overschrijdingen doorwerken in de tijd op eindpunten die relevant zijn vanuit het oogpunt van de bodemchemie en de biodiversiteit. Daartoe zij dynamische en comparatief statische methoden ontwikkeld waarvan resultaten en illustratieve exercities in het rapport zijn opgenomen.

Onder de LRTAP Conventie is er toenemende aandacht voor ratificatie en implementatie processen van diverse protocols, waaronder het zware metalen protocol door Eastern European Caucasian and Central European (EECCA) landen. In samenwerking met de Universiteit Wageningen and het EMEP Meteorologische Synthesizing Centre East (Moskou) worden eerste resultaten gepresenteerd van zware metalen critical loads en overschrijdingen in natuurlijke gebieden in EECCA landen.

In Deel II van dit rapport zijn rapportages opgenomen van de landen ter onderbouwing van hun bijdragen aan de 2008 CL-database.

Tenslotte, beschrijft Appendix A de historische ontwikkeling en scenarios van Europese zwavel en stikstofdepositie die voor dit rapport zijn gebruikt, verschaft Appendix B inzicht in de aanbevelingen van het CCE om bijdragen van National Focal Centres aan de Europese databank van critical loads te helpen harmoniseren, en worden in Appendix C voorbeelden gegeven van het verband tussen stikstof depositie en ecosysteem diensten die het menselijk welzijn beïnvloeden.

Trefwoorden: Biodiversiteit, dosis-response relaties, dynamische modellering, kritische drempels, LRTAP Conventie, effecten van lucht verontreiniging, overschrijding, vermesting, verzuring

PART I
Status of Methods and Databases of European Critical Loads

1 Status of the Critical Loads Database and Impact Assessment

Jean-Paul Hettelingh, Maximilian Posch, Jaap Slootweg

1.1 Introduction

This chapter summarizes the 2008 European database on spatially specific critical loads and dynamic modelling data (*2008 CL-database*), including critical loads for ecosystems in Canada. The 2008 CL-database is important because it is designed to support the revision – expected to commence in 2009 – of the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (Gothenburg Protocol) under the Convention on Long-range Transboundary Air Pollution (LRTAP Convention). Therefore, this chapter includes information on European areas at risk using the 2008 CL-database and recent information on deposition. Also included are comparisons to the 2006 CL-database (UNECE, 2006), which was used to support the review of the 1999 Gothenburg protocol and of the National Emission Ceiling Directive of the European Commission.

1.2 Background of the 2008 CL-database

The Working Group on Effects, at its twenty-sixth session, approved the proposal of the ICP Modelling and Mapping to request its Coordination Centre for Effects (CCE) to make a call for data on empirical and computed critical loads for nitrogen (N) and dynamic modelling parameters, in preparation for use in a possible revision of the Gothenburg Protocol. The results of this call for data are presented here in accordance with item 3.7 of the 2008 work plan for the implementation of the Convention (UNECE, 2007a) adopted by the Executive Body at its twenty-fifth session.

The CCE issued a call for critical loads data on 11 November 2007, after an early notification to the national focal centres (NFCs) of ICP Modelling and Mapping in June 2007. The deadline for data submission was set to 10 March 2008. The call took into account lessons learned from the call for voluntary data issued in 2006, which was designed to allow NFCs to test new scientific and technical knowledge and reported in Slootweg et al. (2007) and in UNECE (2007b).

In support of the call, the CCE had:

a. Finalized a harmonized land cover database in collaboration with the Stockholm Environment Institute. It covered the domain of the Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants (EMEP);
b. Produced an updated version of the Very Simple Dynamic (VSD) model;
c. Established deposition trends in Europe for use in dynamic modelling by NFCs. The periods included were 1880–2010 using historic emissions and as of 2020 using two national emission scenarios (see Appendix A): ‘Current Legislation’ (CLE) and ‘Maximum Feasible Reductions’ (MFR). The scenarios were prepared in collaboration with the Centre for Integrated Assessment Modelling (CIAM) in November 2007. These scenarios could also be used by CCE to interpolate other emission scenarios, e.g. regarding dynamic modelling of aspirational targets that might be formulated by the Working Group on Strategies and Review and the Task Force on Integrated Assessment Modelling;

d. Finalized comprehensive software, interactive database management queries and instructions to assist NFCs in their response to the call for data.

NFCs were requested to participate in the application of:
a. Broad range of critical limits in simple mass balance calculations to address biodiversity, as proposed in De Vries et al. (2007);
b. Empirical critical loads for all ecosystems for which NFCs provided computed critical loads, including European Union's (EU) Natura 2000 ecosystems. The ecosystems were classified following the European Nature Information System (EUNIS, http://eunis.eea.europa.eu);
c. Dynamic modelling of acidification and eutrophication.

1.3 Response to the call for data

The response by NFCs to the 2007/08 call for data is presented in Table 1-1.

Twenty parties to the Convention – nineteen from Europe (fourteen from the EU27) and Canada, responded to the call for data, all of which submitted spatially explicit modelled critical loads for acidification, while 19 countries provided data on critical loads for eutrophication, 13 on empirical critical loads and 12 on dynamic modelling. Romania submitted data for the first time.

Table 1-1 Data submissions from countries (denoted with 'X') as a response to the 2007/2008 call for data.

COUNTRY	Critical loads			Dynamic modelling
	Acidity	Nutrient nitrogen (empirical)	Nutrient nitrogen (modelled)	
Austria (AT)	X	X	X	X
Belarus[1] (BY)	X		X	
Belgium (BE)	X		X	X
Bulgaria (BG)	X	X	X	
Canada (CA)	X			
Finland (FI)	X	X	X	
France (FR)	X	X	X	X
Germany (DE)	X	X	X	X
Ireland (IE)	X	X	X	X
Italy (IT)	X		X	
Netherlands (NL)	X	X	X	X
Norway (NO)	X	X	X	X
Poland (PL)	X	X	X	X
Romania (RO)	X		X	
Russia (RU)	X		X	
Slovenia (SI)	X	X	X	X
Sweden (SE)	X	X	X	X
Switzerland (CH)	X	X	X	X
Ukraine[2] (UA)	X		X	
United Kingdom (GB)	X	X	X	X
Total	**20**	**13**	**19**	**12**

[1] Preliminary data requiring further clarification.
[2] Data for the Crimea only; data requiring further clarification.

Many Parties submitted reports to substantiate their results (see Part II). Spatially explicit critical loads for the remaining Parties in the EMEP domain were computed and compiled from the CCE-background database, using most recent information on critical load input variables and land cover data. This included, for example, the computation of temperature-dependent nitrogen immobilization over Europe (see Chapter 2).

The updated European critical load maps and data statistics were presented at the eighteenth CCE workshop, held on 21–23 April 2008 in Berne, and the twenty-fourth meeting of the Task Force of ICP Modelling and Mapping, held on 24–25 April 2008 in Berne. In comparison to the European critical load database of 2006, which was used for the review of the Gothenburg Protocol, more and improved information became available to support effects-oriented integrated assessments under the Task Force on Integrated Assessment Modelling and the Working Group on Strategies and Review. This included information on critical loads for Natura 2000 areas and, for the first time in the CCE background database, also on semi-natural vegetation.

However, the 2008 CL-database not only consists of computed critical loads. The United Kingdom informed the CCE to include all ecosystems for which empirical critical loads should be included in the 2008 European critical loads database, which would be used by the Task Force on Integrated Assessment Modelling. For France the empirical critical loads for ecosystems with EUNIS class B1.4 have also been included for use by the Task Force. This is in line with the historic critical load database of both countries. Norway indicated that for Norwegian critical loads for nitrogen, the European background database of computed critical loads should be used.

1.4 Critical load maps

Figure 1-1 shows critical loads for eutrophication and acidification that protect 95% of forests, semi-natural vegetation or surface waters in Europe (see Chapter 2). Areas most sensitive to eutrophication are in the north, east and south of Europe. Regarding acidification, most sensitive areas are in northern Europe where 95% of the natural areas are protected provided that acid deposition is lower than 200 eq $ha^{-1}yr^{-1}$. Figure 1-2 shows critical loads for acidification in Canada. Most sensitive areas are in the Coastal Mountains in the west, in Saskatchewan in the centre and in the northern part of Ontario and southern part of Quebec.

Mapping empirical critical load submissions and the European background map of empirical critical loads for other countries yields Figure 1-3. Compared to map of modelled critical loads of nutrient nitrogen (left map in Figure 1-1) it can be seen from Figure 1-3 that empirical critical loads are higher. Both maps show that most sensitive ecosystems are in northern Europe.

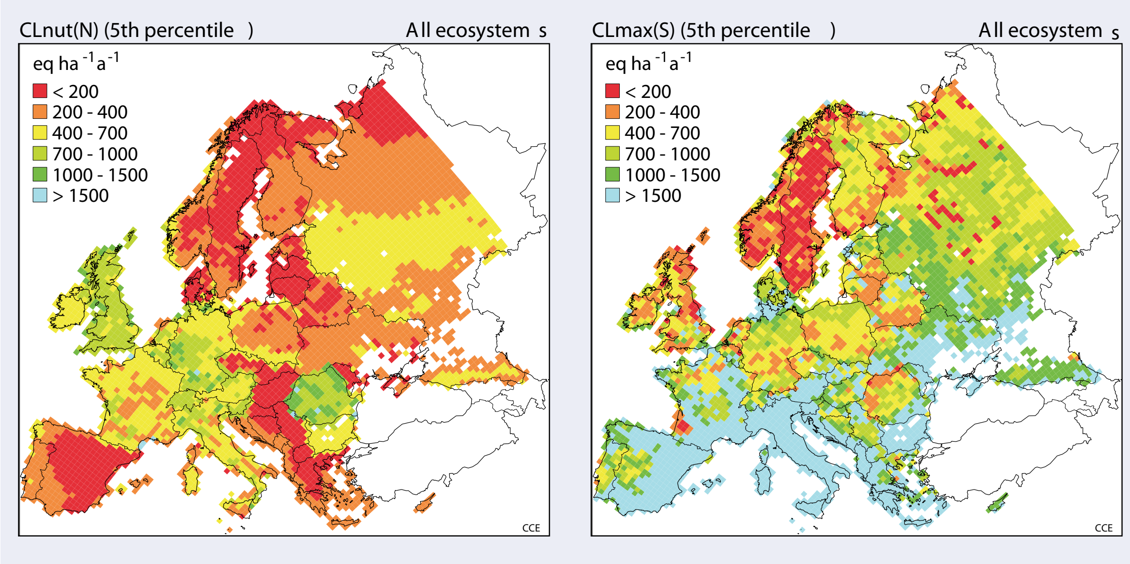

Figure 1-1 European maps of critical loads for eutrophication (left) and acidification (right) which protect 95% of natural areas in 50x50 km² EMEP grid. Red shaded areas illustrate grid cells where deposition needs to be lower than 200 eq $ha^{-1}a^{-1}$ to reach this protection target.

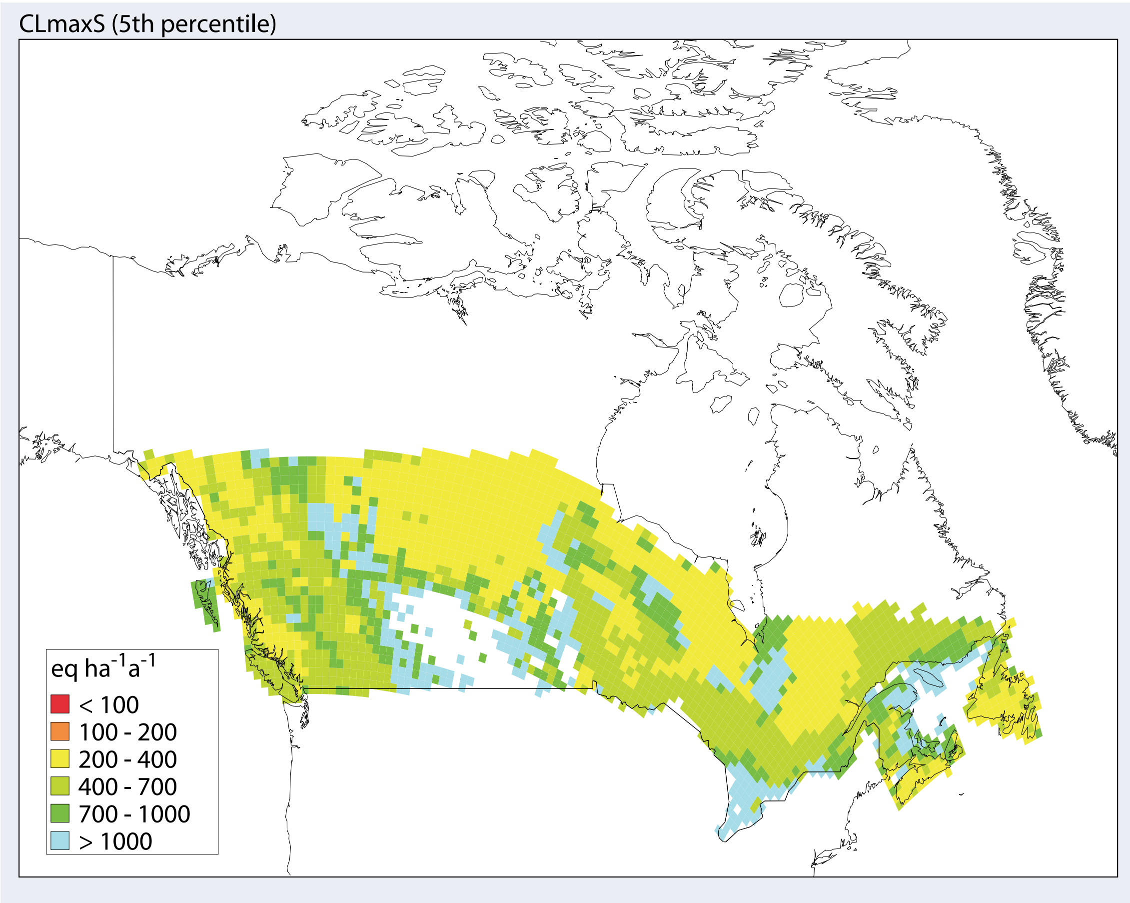

Figure 1-2 Critical loads of acidity in (the southern parts of) Canada illustrating the protection of 95% of the soils in the mapped grid cells.

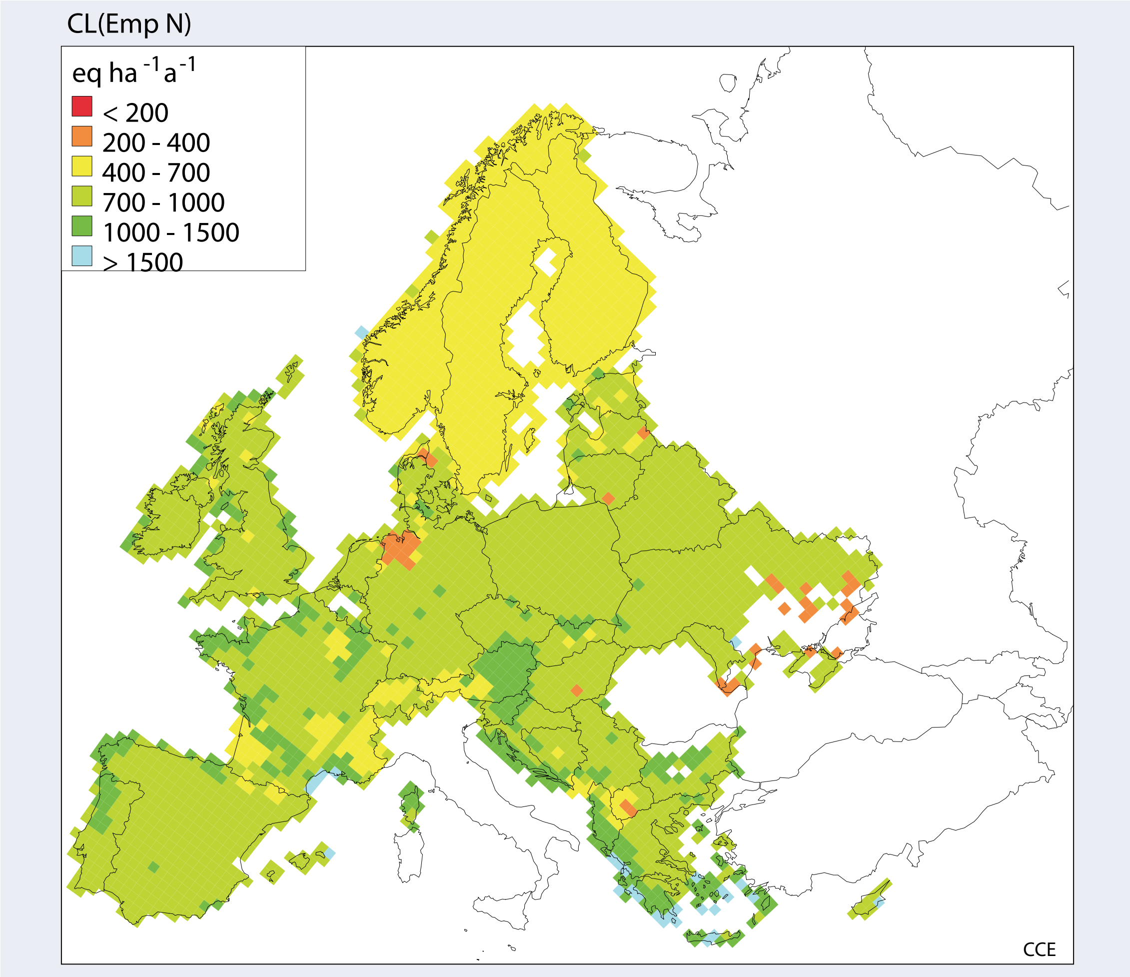

Figure 1-3 Empirical critical loads illustrating the protection of 95% of the ecosystems. Most sensitive regions are in the orange and yellow shaded areas.

1.5 Critical load exceedances

An overview of ecosystem areas at risk of acidification and of excessive nutrient N deposition in countries within the domain of EMEP is given in Table 1-2. Results were computed using the 2008 CL-database. Depositions were made available by CIAM in autumn 2007. They were based on two emission scenarios: Current Legislation (CLE) in 2010 and 2020 and Maximum Feasible Reduction (MFR) in 2020.

Table 1-2 shows that the computed European area at risk of acidification decreases from 11% in 2000 to 6% and 1% in 2020 for CLE and MFR, respectively. Even for MFR, 60% of the ecosystem area in the Netherlands was computed to be at risk of acidification in 2020. For all other Parties the ecosystem areas at risk of acidification were well below 10 %. In the EU27 the areas at risk of acidification under CLE in 2000, 2010 and 2020 cover 19%, 11% and 9% respectively. The application of MFR reduces the area at risk in the EU27 cover only 1% of its ecosystems.

From Table 1-2 it can be seen that the computed European area at risk of eutrophication decreases from 49 % in 2000 to 47 % and 17 % in 2020 for CLE and MFR, respectively. In the EU27 the areas at risk of eutrophication under CLE in 2000, 2010 and 2020 cover 74 %, 69% and 67% respectively. The application of MFR implies an area at risk in the EU27 that covers 28% of its ecosystems.

Table 1-2. Percentage of natural ecosystem area at risk of acidification (left) and of eutrophication for Parties to the Convention within EMEP modelling domain in 2000 and for two emission scenarios: current legislation (CLE) in 2010 and 2020, maximum feasible reductions (MFR) in 2020

	Acidification					Eutrophication				
Country	Area (km^2)	2000 (% at risk)	CLE 2010 (% at risk)	CLE 2020 (% at risk)	MFR 2020 (% at risk)	Area (km^2)	2000 (% at risk)	CLE 2010 (% at risk)	CLE 2020 (% at risk)	MFR 2020 (% at risk)
AL	16,954	0	0	0	0	16,954	100	99	99	43
AT	35,746	2	1	0	0	40,255	100	94	78	5
BA	31,892	17	15	10	0	31,892	89	81	77	40
BE	6,250	29	21	19	4	6,250	100	99	94	37
BG	48,330	0	0	0	0	48,330	94	91	80	18
BY	64,023	18	17	16	0	64,023	99	99	99	56
CH	9,805	9	5	3	1	9,625	99	96	91	21
CY	2,461	0	0	0	0	2,461	68	68	68	17
CZ	27,626	28	22	20	5	27,626	100	100	100	99
DE	102,891	58	32	24	5	102,891	84	67	58	36
DK	3,584	50	42	37	2	3,584	100	100	100	99
EE	24,728	0	0	0	0	24,728	67	57	47	5
ES	187,115	3	0	0	0	187,115	95	93	90	48
FI	273,634	3	2	2	0	240,403	47	41	36	2
FR	177,359	12	8	6	1	180,099	98	95	91	41
GB	81,815	39	19	15	7	92,244	26	19	17	9
GR	53,671	3	1	1	0	53,671	98	97	97	60
HR	31,698	5	3	3	0	31,698	100	100	99	81
HU	20,805	26	8	7	0	20,805	100	100	100	56
IE	8,935	23	8	6	2	2,449	88	81	77	73
IT	124,788	0	0	0	0	124,788	69	61	55	14
LT	19,018	34	32	32	4	19,018	100	100	100	92
LU	1,015	15	13	13	0	1,015	100	100	99	98
LV	35,823	20	14	12	0	35,823	99	99	96	44
MD	3,483	1	0	0	0	3,483	100	100	100	72
MK	13,945	12	1	0	0	13,945	100	100	100	53
NL	6,968	76	71	71	60	4,447	94	88	88	76
NO	179,158	16	11	10	3	137,701	22	14	11	0
PL	90,330	77	61	50	3	90,330	100	100	99	68
PT	31,121	8	3	3	0	31,121	97	83	69	6
RO	97,964	46	22	12	0	97,964	19	20	15	0
RU	1,821,560	1	1	2	0	1,821,560	21	24	28	2
SE	443,660	17	10	9	2	150,865	56	47	43	13
SI	10,996	7	0	0	0	10,996	98	92	82	0
SK	20,532	18	9	8	0	20,532	100	100	100	83
UA	72,200	5	3	4	0	72,200	100	100	100	92
YU	41,108	18	9	3	0	41,108	97	95	92	34
EU27	1,937,164	19	11	9	2	1,619,811	74	69	64	28
EMEP domain	4,222,991	11	7	6	1	3,864,000	49	48	47	17

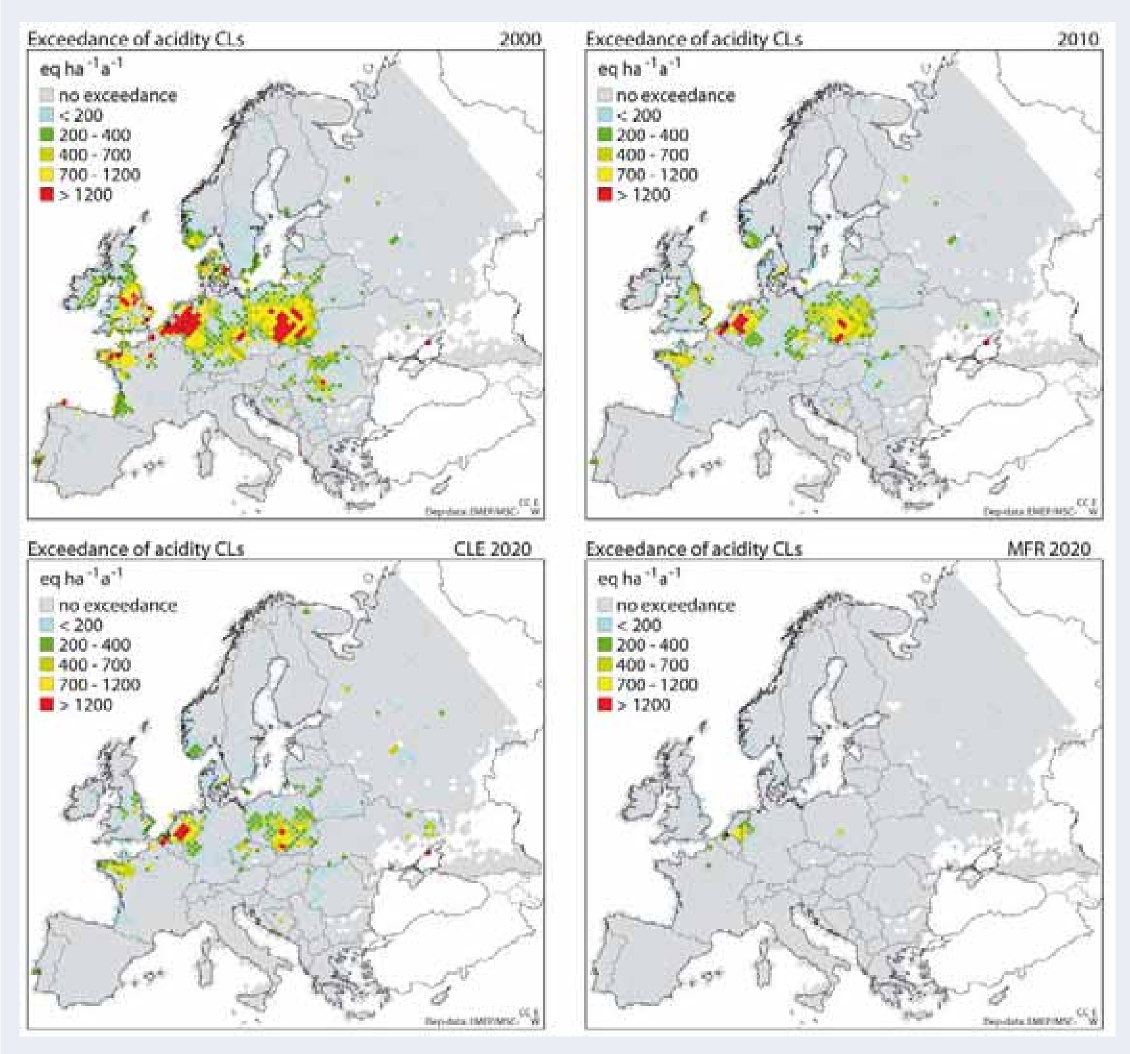

Figure 1-4 Exceedance of critical loads for acidification by depositions in 2000 (top left), 2010 (top right), and 2020 (bottom left) under Current Legislation to reduce national emissions, and in 2020 (bottom right) under Maximum Feasible Reductions.

The distribution over Europe of the locations of the areas at risk of acidification and eutrophication is shown in Figures 1-4 and 1-5 respectively. Exceedances are expressed as average accumulated exceedances (AAE) per grid cell. From Figure 1-4 it is seen that areas with exceedances higher than 1200 eq $ha^{-1}a^{-1}$ (red shaded) are mostly in Belgium, Germany, the Netherlands and Poland in 2000. Under CLE in 2020 the size of the exceeded area is considerably reduced. In 2020 under MFR it is obvious that many areas in Europe are no longer at risk of acidification, and that the highest peaks in the interval between 700 and 1200 eq $ha^{-1}a^{-1}$ are for ecosystems in the Netherlands.

From Figure 1-5 it is clear that rather large areas with the highest exceedances of critical loads of nutrient N (red shaded) in 2000 are in the West of Europe, following the coastal regions from north-western France to Denmark, the south-eastern part of the United Kingdom, while in southern Europe high exceedances are found in northern Italy. The reduction of the area with exceedances above 1200 eq $ha^{-1}a^{-1}$ in 2010 hardly changes under CLE in 2020, while exceedances in this highest range do not occur following the application of MFR (see lower right map in Figure 1-5). However, in the latter case still broad area in Europe remain at risk of eutrophication with exceedances that range from 200 to 1200 eq $ha^{-1}a^{-1}$.

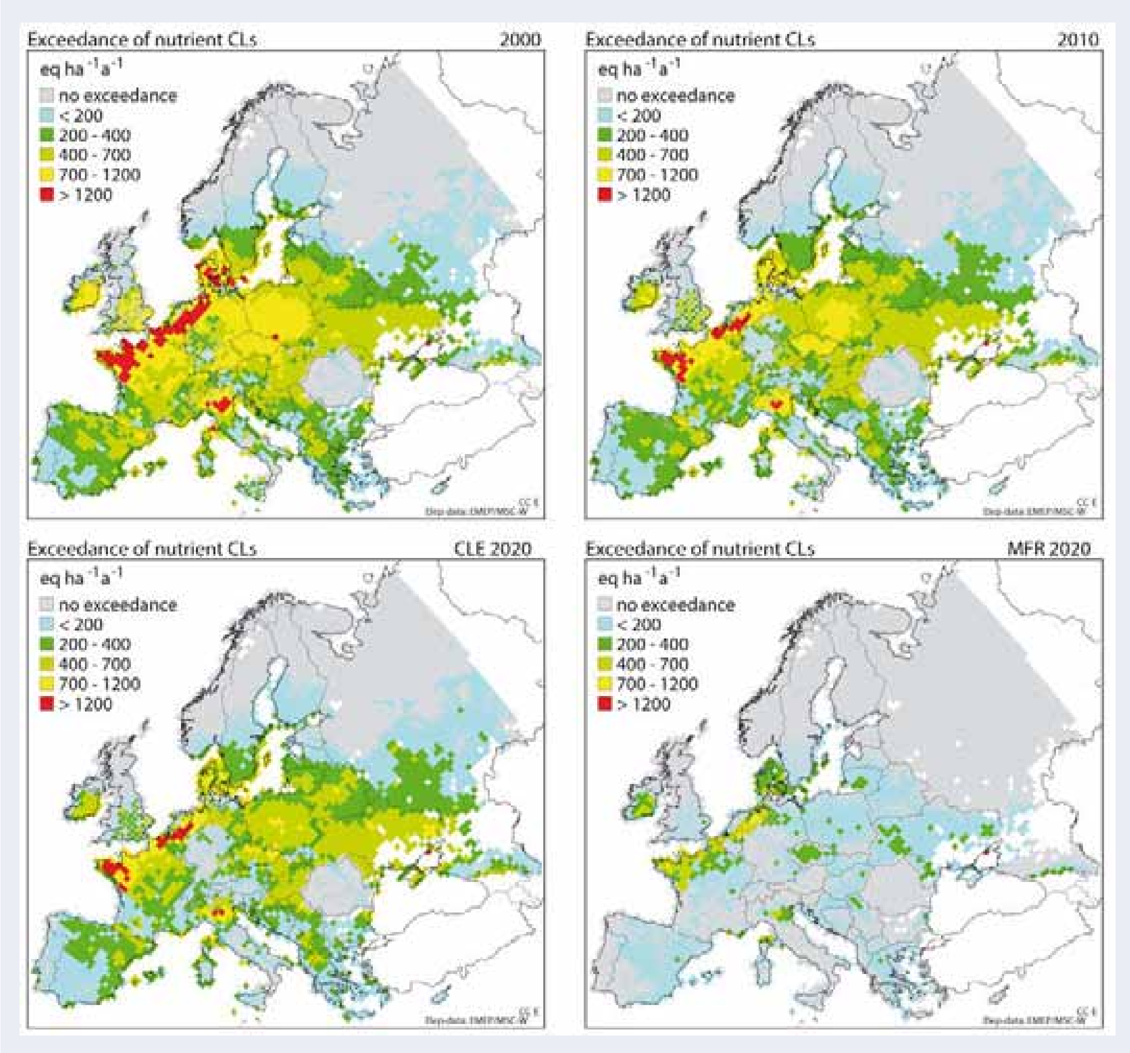

Figure 1-5 Exceedance of critical loads for eutrophication by depositions in 2000 (top left) and in 2010 (top right), 2020 (bottom left) under Current Legislation to national emissions, and in 2020 (bottom right) under Maximum Feasible Reductions.

1.6 The 2008 CL-database compared with the 2006 CL-database

The comparison between the 2008 CL-database and the 2006 CL-database is important because the latter database has been used for the review of European air pollution policy agreements. The 2008 CL-database is now available for the revision of the Gothenburg protocol under the LRTAP Convention and for the possible revision of the Thematic Strategy on Air Pollution of the European Commission. The distribution of critical loads for acidification turns out not to have significantly changed. However, the distribution of critical loads for eutrophication has been affected. An important reason (see chapter 2 for a more complete overview) for this change in the 2008 CL-database in comparison to 2006 CL-database is the inclusion of semi-natural vegetation in many more countries, i.e. the CCE background database. The inclusion of semi-natural vegetation as an ecosystem class leads to the occurrence of more areas with relatively low critical loads for eutrophication. This is illustrated in Figure 1-6 showing the cumulative distribution of critical loads for all ecosystems in 2006 CL-database (grey), for forests in the 2008 CL-database (green) and for all ecosystems (including semi-natural vegetation) in the 2008 CL-database (red).

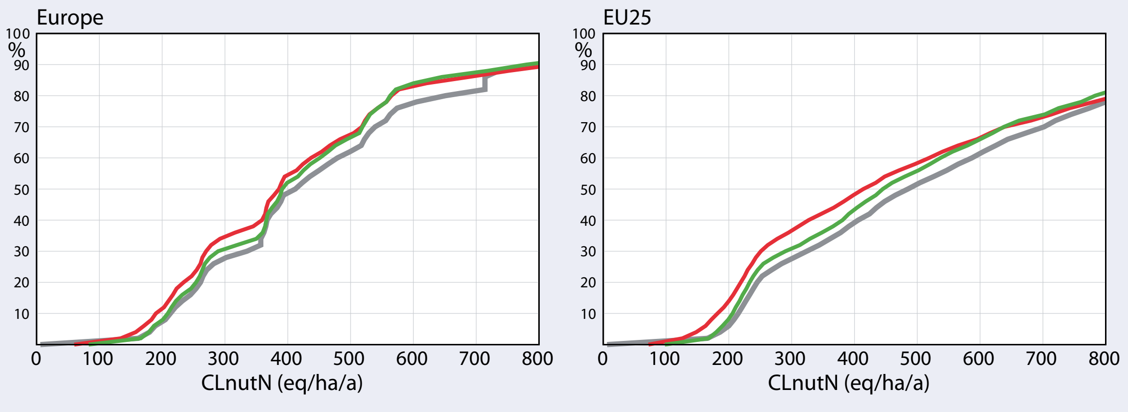

Figure 1-6 Cumulative distribution of critical loads for eutrophication for all ecosystems in the 2006 CL-database (grey), for forests in the 2008 CL-database (green) and for all ecosystems in the 2008 CL-database (red) in Europe (left) and the European Union of 25 member States (the EU27 did not exist in 2006). The CCE background database in 2008 CL-database includes semi natural vegetation.

Figure 1-6 shows for example that the areas with critical loads below 200 eq $ha^{-1}a^{-1}$ cover less than 10% in Europe and the EU25. The critical loads for forests in that range in 2008 CL-database covers about the same percentage. This is to be expected since the 2006 CL-database contains information mostly on forest ecosystems, especially in the part of the 2006 CL-database that has its data from the CCE background database. However, after the inclusion of semi-natural vegetation the areas with critical loads below 200 eq $ha^{-1}a^{-1}$ cover now more than 10% of the ecosystems in Europe and more than 15% in the EU25.

The difference between 2006 CL-database and 2008 CL-database is also reflected in the exceedance, although the evolution since 2006 of computed deposition data also plays a role. The European area at risk of acidification computed with the 2006 CL-database is similar to the results using the 2008 data. For EU25 the area at risk using the 2006 CL-database is slightly lower.

The European area at risk of eutrophication computed with the 2006 CL-database is about 3% lower in 2000, 2010 and 2020 (CLE and MFR). For the EU25, the area at risk using the 2008 CL-database is significantly higher. Using the 2006 CL-database the areas at risk in the EU25 have been reported (WGE, 2006) to be 65%, 60%, 56% and 25% in 2000, 2010, 2020 (CLE) and 2020 (MFR) respectively. The area at risk computed in 2006 may increase when more recent 2007 EMEP depositions are used (about 3% additional area at risk in 2000). With the 2008 CL-database the percentages of the areas at risk become 77%, 71%, 67% and 31%, respectively.

Taking these differences into account it is recommended to use the 2008 CL-database in integrated assessment, if only because it covers more ecosystem classes that are relevant from the point of view of nature protection.

1.7 Dynamic modelling results

Most parties submitted results on dynamic modelling using the VSD model for terrestrial ecosystems. The Netherlands included results of the application of a dynamic vegetation model on terrestrial ecosystems. Dynamic vegetation models are also tested in Austria, Germany, Sweden, Switzerland and the United Kingdom, but results have not yet been included in the data submission. Norway, Sweden and the United Kingdom also used a dynamic model on aquatic ecosystems.

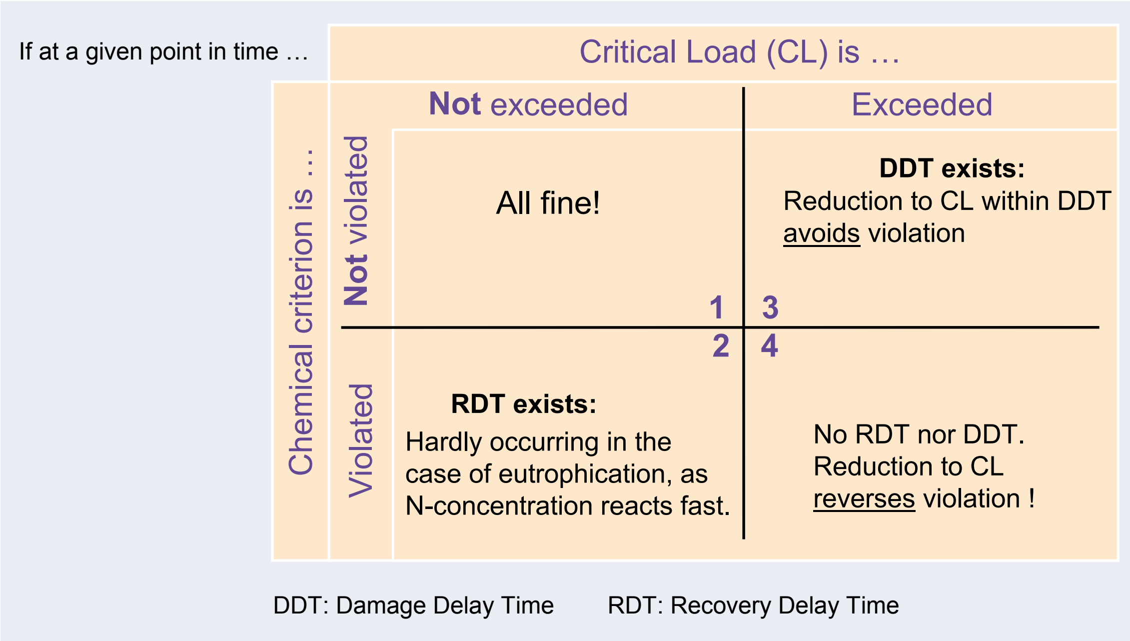

Figure 1-7 Four combinations of critical load (non-)exceedance and criterion (non-)violation.

Dynamic modelling data submitted by these countries, and data from the background database, allow assessments of the occurrence of (non-)exceedance and (non-)violation of the critical limits. Four cases are distinguished in Figure 1-7 by combining (non-) exceedance of critical loads with (non-)violation of critical limits.

Case 1 applies to deposition that does not exceed a critical load while the critical limit is not violated. In case 2 there is no exceedance, however the critical limit is violated. In case 3 the critical load is exceeded but the critical limit not violated, while finally case 4 applies to the situation where both the critical load is exceeded and critical limit is violated. Case 4 includes ecosystems that are subject to immediate risk of damage by N deposition. Case 1 implies full protection. Cases 2 and 3 included ecosystems for which recovery delay times (RDT) and damage delay times (DDT) can be identified, respectively.

Investigation of combinations of (non-)exceedance of critical loads of nutrient N with (non-) violation of a critical limit value of 0.3 mg N L^{-1} was conducted using the CCE background database. This critical limit is associated with vegetation changes, but also with nutrient imbalances in deciduous forests (de Vries et al., 2007).

Results indicated that for CLE in 2010 and 2050 about 56% and 55% of European ecosystems were in case 4 (critical load exceeded and critical limit violated). Case 1 (critical load not exceeded and critical limit not violated) applied to 35% in 2010 and increased to 37% of the ecosystems in 2050. The latter percentage was close to being attained already in 2020, confirming the fast response of N concentration to changes in N deposition.

For MFR scenario most of the ecosystems moved from case 4 to case 1 than for CLE. Case 1 (critical load not exceeded and critical limit not violated) applied to about 65%, while case 4 (critical load exceeded and critical limit violated) covered 29% of the ecosystems in 2050. Again, first time attainment of these area percentages occurred soon after 2020.

The computations for the violation of the critical limit (cases 2 and 4) confirmed the early attainment before 2030 of the area that would be safe for MFR in 2050. For example, violation was computed by the French NFC for an area of 96% in 2010, which diminished to 91% for CLE and 57% for MFR in 2020 and to 90% for CLE and 41% for MFR in 2030. For the German NFC these percentages were 54% in 2010, 43% (CLE) and 26% (for MFR) in 2020 and 43% (CLE) and 23% (MFR) in 2050. Hardly any additional area becomes protected after 2030.

This type of analysis could be performed for any reasonable deposition scenario which the Task Force on Integrated Assessment Modelling might wish to explore for the revision of the Gothenburg Protocol. Chapter 3 gives an illustrative example of the manner in which dynamic modelling can be used to provide more knowledge on area at risk of acidification and eutrophication in the context of integrated assessment.

1.8 Ecosystems and human well-being

Assessments of the impacts on human health caused by (ultra-)fine particles or ozone are well established as part of integrated assessment in support of European air pollution policies. Effect-based policies also include the assessment of impacts on ecosystems, but another weight is assigned to this part of impact assessment. A clear distinction is made between human health effects and ecosystem effects in the minds of many end-users of integrated assessment results. As a matter of fact, there tends to be an economic focus which is easier to address by human health effects than by environmental effects, for which a market price is more difficult to agree upon. Important elements to approximate the latter include relative "benefits" between emission reduction scenarios of not exceeding critical loads, of identifying recovery (through dynamic modeling) or of becoming better versed in the identification of biological endpoints (see Figure 1-9). While progress is made with cost-effectiveness analysis based on ecosystem impacts, the public health *edge* – so to say – of costs and benefits is still under development. However, another reason for the backseat position of assessments of ecosystem effects is perhaps also the lack of knowledge about – for example – the relationships between biological diversity and human well-being, including human health.

The WHO prepared a report as a contribution to the Millenium Ecosystem Assessment to address the issue of ecosystems and human well-being (Corvalan et al., 2005). The report starts by stating that "ecosystems are essential to human well-being and especially to human health - defined by the World Heltah Organization as a state of complete physical, mental and social well being" (Corvalan et al. 2005, pp. 12). To underpin this statement a logic is used whereby ecosystem services, i.e. provisioning services, regulating services, supporting services and cultural services, is related to well-being as shown in Figure 1-8.

Figure 1-8 illustrates how benefits obtained from ecosystems include food, natural fibres, clean water, regulation of some pests and diseases, medicinal substances and protection from natural hazards. It is clear, that more dose-drivers and effect-receptors are involved, than those related to the context of this section, i.e. air pollution in general and nitrogen in particular. Examples of the role of nitrogen in provisioning, regulating and cultural services is provided in Appendix C.

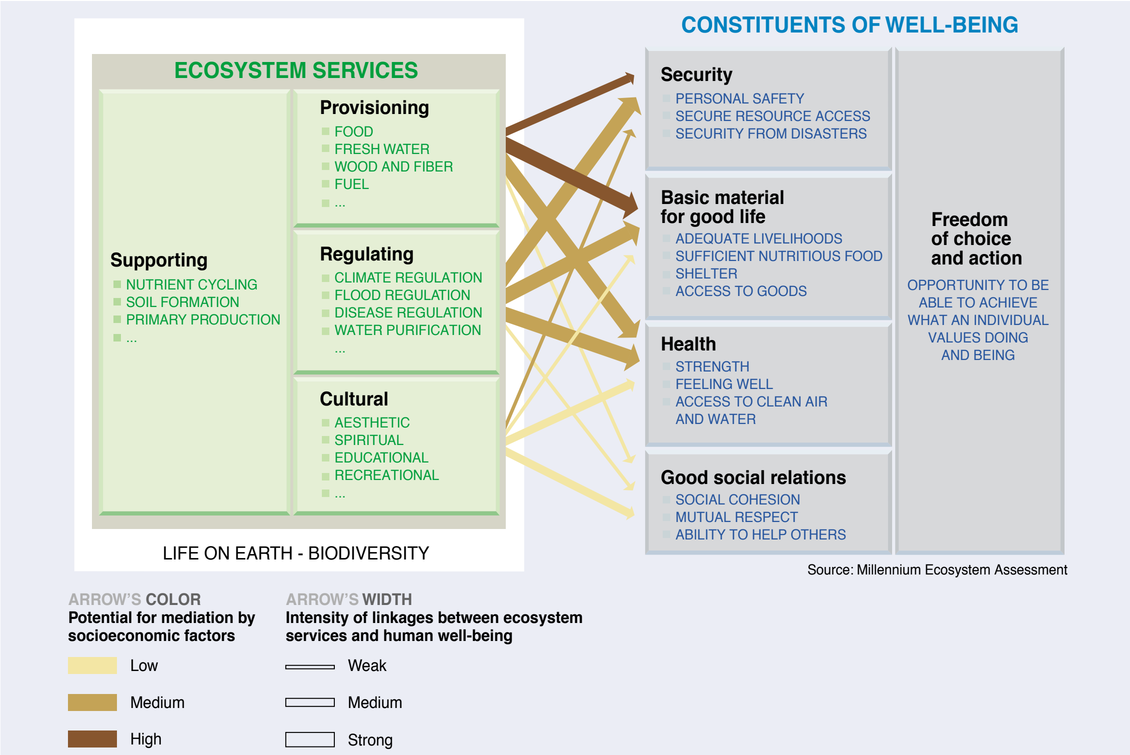

Figure 1-8 Linkages between commonly-encountered categories of ecosystem services and components of human well-being including the intensity of linkages (arrow's width) and potential for mediation of the linkage (arrow's colour). (Source: Millennium Ecosystem Assessment report by Corvalan et al. 2005, pp. 15). See Appendix 3 for examples of the role of nitrogen on ecosystem services.

These examples of the relationships between the exceedance of nitrogen critical loads and the adverse effects of ecosystem services are part of many other causal links between environmental change and human health. The influence of nitrogen would be more comprehensive in the context of the Millenium Ecosystem Assessment if we would manage to better address the effect of nitrogen deposition on biodiversity indicators rather than its effect on the change of exceedances. A first identification and application of useful indicators is described in chapters 4 and 5, while a tentative regional application is described in chapter 6.

1.9 Proposed methodology for the Protocol revision work

The following proposes a methodology for collaboration with the EMEP Centre on Integrated Assessment Modelling (CIAM) to support the work on the revision of the Gothenburg Protocol. As of 2008, CCE is able to deliver the following knowledge to CIAM, the Task Force on Integrated Assessment Modelling and the Working Group on Strategies and Review. It consists of three elements:

a. Modelled critical loads, exceedances and information on delay times of damage and recovery using dynamic modelling for any given emission scenario;
b. Empirical critical loads, exceedances and preliminary information on changes to species richness as a first indicator of biological impacts to vegetation for any given emission scenario. The quantification of an indicator which is relevant to biodiversity was requested by the Executive Body of the LRTAP Convention in its 25th session (EB, 2008)
c. Improved robustness of the identification of areas at risk, using the Ensemble Assessment of Impacts approach (see CCE Progress Report 2007).

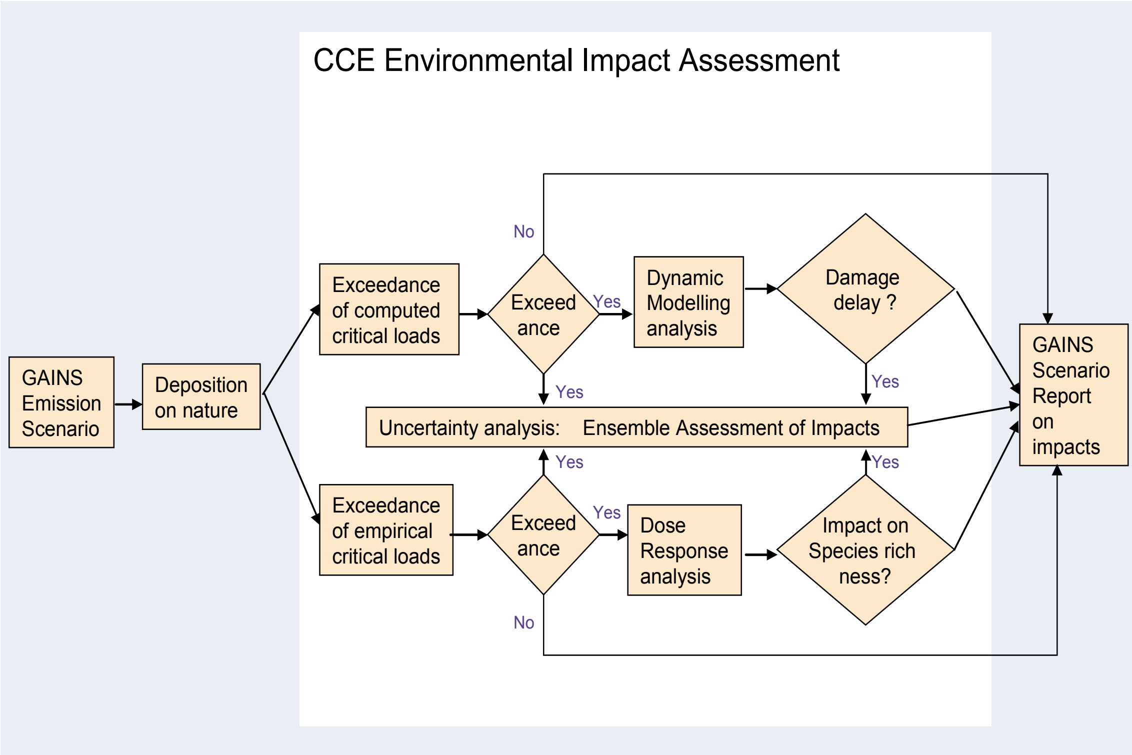

Figure 1-9 A simplified flowchart of the framework for the assessment of impacts of excess nitrogen deposition in the context of integrated assessment modelling, e.g in the GAINS model (source: EC4MACS, 2008).

The Task Force of ICP Modelling and Mapping acknowledged that CIAM and CCE needed to collaborate closely for the use of effects-oriented information to support the revision of the Gothenburg Protocol and further work in support of the thematic strategy on air pollution of the European Commission. Two groups of methods can be distinguished for use under integrated assessment, i.,e. “modelled critical loads and dynamic modelling” and “empirical critical loads and dose-response relationships”. The CCE has combined the two groups of methods in a framework of effect indicator applications for integrated assessment which is shown in Figure 1-9.

Figure 1-9 shows that environmental impacts of excess nitrogen deposition can be analyzed in two ways. The first (upper pathway) is to compute exceedances of computed critical loads and perform dynamic modelling on delay times as appropriate (see chapter 3). For reasons of simplification Figure 1-9 illustrates the case of analyzing Damage Delay Times only. The lower pathway illustrates the manner in which exceedances of empirical critical loads can be used in conjunction with the analysis of impacts to species richness (see chapter 6). Empirical critical loads for many EUNIS classes have been reported in Achermann and Bobbink (2003), while regionalization to Europe is conducted by the CCE on the basis of the European land cover map.

Critical load exceedances can be used by CIAM directly for target setting and optimization with GAINS, as appropriate. The regional use of dynamic models and dose response curves can be done ex-post by the CCE with the outcomes of GAINS scenarios. Finally, CIAM and CCE can join their results in reports on integrated assessment of emission abatement alternatives.

1.10 Other Activities

The collaboration between the CCE and EMEP is currently focussing on the computation of exceedances on 25×25 km^2 rather than on 50×50 km^2 EMEP grid cells. This work extends the EMEP domain to include some countries of the EECCA. A first example of this work is published in the 2008 Status report of EMEP/MSC-W.

The CCE established first maps of critical loads of acidity, nutrient N, cadmium, lead and mercury for EECCA countries in collaboration with Wageningen University. Exceedance maps of critical loads of heavy metals in EECCA countries were produced in collaboration with the EMEP MSC-E (see Chapter 7).

Effects-oriented modelling and mapping activities are increasingly addressing ecosystems on a hemispheric scale, as is illustrated on the front cover of this report.

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2 Summary of National Data

Jaap Slootweg, Maximilian Posch, Jean-Paul Hettelingh

2.1 Introduction

The Working Group on Effects, at its 26th session (Geneva, 29-31 August 2007), approved the proposal made at the 23rd Task Force meeting of the ICP-M&M (Sofia, 26-27 April 2007) to issue a call for data on empirical critical loads, critical loads for acidification and eutrophication, and for data on dynamic modelling (EB.AIR/WG.1/2007/2 para. 12j). This data is assembled into a new European database which will be submitted to the to Task Force on Integrated Assessment Modelling (TFIAM) with the intention to use it in the revision of the Gothenburg Protocol, and for possible use in support of policy processes under the European Commission.

The results have been presented and discussed at the CCE workshop held back-to-back with the Task Force M&M (Bern, 21-25 April 2008). Some parties have updated their data shortly after these meetings, and these updates are included in the results reported here.

The call for data was issued for critical loads, including empirical critical loads, as well as input variables, values of chemical variables from dynamic model runs in historic and in future years for deposition scenarios that were provided, and a document describing the sources and methods used to produce the data. It has been made clear that *none* of 2007 (or earlier) submitted data will be used. To obtain full coverage of the Pan-European (EMEP-) domain the (updated) European background database is used for the countries that did not react to this year's call. A full list of all variables and a complete description of the call can be found in the *Instructions for Submitting Critical Loads of N and S and Dynamic Modelling Data*, reprinted in Appendix B of this report.

In this chapter you will find maps and statistics of critical loads, an analysis of the most important variables leading to the critical loads, on the exceedances of the critical loads and a description of dynamic modelling results.

2.2 Status of national data in relation to the European background database

Eighteen parties submitted data that will be used in future assessments. Most of the submissions contained both modelled and empirical critical load data and the majority applied dynamic modelling (see Table 1-1). Forest ecosystems are considered by all parties, but other receptors (ecosystem types) considered vary considerably. Most cross-country comparisons in this chapter differentiate between ecosystem types, allowing more detailed comparisons. The deposition on forests is relative higher than on other vegetation and on areas with no vegetation, which justifies the focus on forests. Table 2.1 shows the relevant ecosystem types according to the EUNIS classification, with a short description and the deposition class used to calculate exceedances.

Table 2-1. Ecosystem types used for critical loads (EUNIS codes, up to level 2) with a brief description and their associated deposition class.

EUNIS classes	Description	Deposition
G, G1, G2, G3	Forests	Forests
C3	Littoral zones	Vegetation
D, D1, D2, D4, D5, D6	Mire, bog and fen habitats	
E, E1, E2, E3, E4	Grassland and tall forb habitats	
F, F1, F2, F3, F4, F5, F7, F9	Heathland, scrub and tundra habitats	
I1	Agriculture	
Y	Unknown	
A, A2, A4	Marine habitats	Other (Average)
B1, B3	Coastal habitats	
H4, H5	Inland unvegetated or sparsely vegetated habitats	
C1,C2	Inland surface water habitats	

A full overview of the number of records, the areas of the submissions by country and by EUNIS classes for eutriphication (modelled and empirical), acidification and dynamic modelling is given in Table 2-2. Dynamic modelling is more frequently done for ecosystems for which acidification is also considered. For many ecosystems empirical critical loads are available, but (still) relatively little emphasis is given to nitrogen dynamic modelling, maybe due to the lack of widely-accepted simple models of nitrogen dynamics.

For countries that did not submit data, the CCE applies the European background database (Reinds, 2007). This database contains all critical loads and dynamic modelling results for the EUNIS classes D, E, F and G, and is in use for Albania, Bosnia and Herzegovina, Belarus, Cyprus, the Czech Republic, Denmark, Estonia, Spain, Greece, Croatia, Hungary, Lithuania, Luxembourg, Latvia, Moldova, Macedonia, Portugal, Slovakia, Ukraine and Serbia and Montenegro.

Table 2-2 Number of ecosystems and area of the country submissions for modelled nutrient nitrogen, empirical, acidification critical loads and dynamic modelling.

		Modelled Nutr. N		Empirical N		Acidification		Dynamic Modelling	
		#records	Area (km²)	#records	Area (km²)	#records	Area (km²)	#records	Area (km²)
AT	D			2720	339				
	E			2570	8297				
	G	18314	40255	7108	40308	496	35745	496	35745
BE	G	3094	6250			3094	6250	1725	4977
BG	A			7	237				
	B			7	207				
	C			46	1553				
	D			19	183				
	E			42	335				
	F			25	110				
	G	83	48330	83	48330	83	48330		
CA	C					492	6728		
	G					138415	1648716		
CH	C			49	42	100	180		
	D			2057	1513				
	E			12889	10243				
	F			1816	1645				
	G	10608	9625	1607	1015	10608	9625	260	260
	Y			41	31				
DE	A	21	21			21	21		
	B	65	65	133	133	65	65		

		Modelled Nutr. N		Empirical N		Acidification		Dynamic Modelling	
		#records	Area (km²)	#records	Area (km²)	#records	Area (km²)	#records	Area (km²)
	C	36	36	33	33	36	36		
	D	1177	1177	675	675	1177	1177		
	E	1493	1493	1168	1168	1493	1493		
	F	300	300	3	3	300	300		
	G	99799	99799	99903	99903	99799	99799	97729	97729
FI	C					1450	33231		
	D			3129	26029				
	F			234	9062				
	G	3083	240403	11104	252928	3083	240403		
FR	B			156	2741				
	D	67	5123	67	5123	67	5123	67	5123
	E	81	1580	81	1580	81	1580	81	1580
	G	3839	170655	3837	170620	3839	170655	3839	170655
GB	A			44	7246				
	B			10421	4068				
	C			64	1269	1751	15024	310	1153
	D			19342	8946	18682	5455	17041	5197
	E			113642	20970	93977	16957	66568	13682
	F			79237	28670	78550	24669	67323	22789
	G	113169	15793	38786	5282	150208	19748	85502	12323
IE	E			6895	2050	6895	2050	6895	2050
	F			6847	2631	6847	2631	6847	2631
	G	9195	2449	17242	4254	17241	4254	17241	4254
IT	A	1	35			1	35		
	B	15	371			15	371		
	C	3	60			3	60		
	E	180	22751			180	22751		
	F	204	12664			204	12664		
	G	697	88907			697	88907		
NL	A	1096	69	456	29	976	61	1096	69
	B	4385	274	4385	274	3467	217	4385	274
	C	417	5						
	D	3182	199	3182	199	2908	182	3182	199
	E	15107	944	15107	944	9489	593	15107	944
	F	5788	362	5788	362	5551	347	5788	362
	G	44027	2752	43942	2746	91525	5720	87979	5499
NO	C			273	19045	2304	322152	2304	322152
	D	112	628	12	694				
	E	1618	6763	288	9508				
	F	16803	163312	367	175378				
	G	14882	76904	474	85933				
	H			77	3947				
	I			126	12865				
PL	D	3956	2114	3319	1779	3956	2114	3956	2114
	E	1145	577	730	386	1145	577	1145	577
	F	128	78	9	5	128	78	128	78
	G	161295	87561	128426	70139	161295	87561	161295	87561
RO	G	97964	97964			97964	97964		
RU	G	31043	1821560			31043	1821560		
SE	C					1974	430536	1974	430536
	D			512	44021				
	G	16120	221837	786	298718	16120	221837	16120	221837
SI	F	256	164	256	164	256	164	256	164
	G	12436	10832	12435	10832	12436	10832	12436	10832
Total		**697284**	**3263041**	**665079**	**1507740**	**1082487**	**5527528**	**689075**	**1463346**

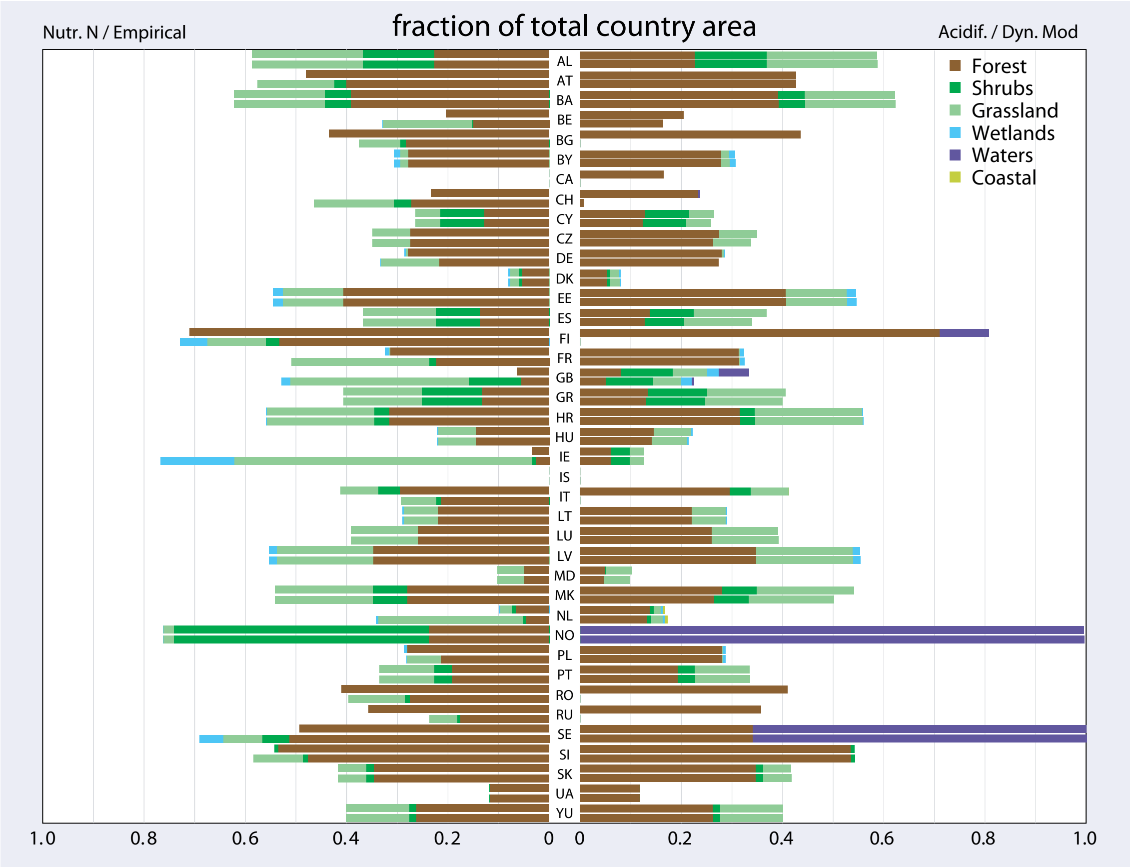

Figure 2-1 National distributions of ecosystem types as % of the total country area for the whole of the country.

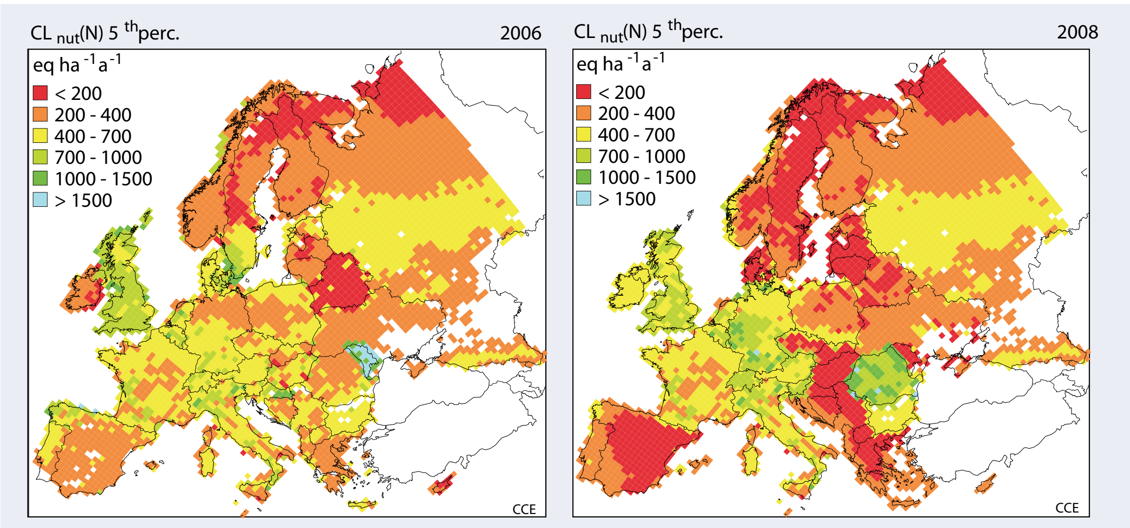

Figure 2-2 Critical loads of nutrient nitrogen for the 2006 data (left) and the data for this year's call (right).

The ecosystem areas of all countries are plotted as fraction of the respective total country areas in Figure 2-1. The bars are stacked for the EUNIS level-1 classes and show on the left-hand side the fractions for nutrient N and empirical critical loads (respectively top and bottom bar for each country). On the right-hand side the top bar of each country shows the fractions for ecosystem for which critical loads for acidity are present, whereas the second bar indicates the fractions for dynamic modelling. Some countries consider forest areas also as catchments of rivers, which potentially leads to a coverage of over 100%. In these cases the ecosystem areas of all ecosystems in that country are scaled back to sum up to the total country area. Forests are (traditionally) well represented in most countries, but especially for empirical loads other ecosystem types are considered.

2.3 Critical loads

Critical loads have changed since the submission of 2006, especially for *CLnutN*. These changes have several reasons: (a) the European background database is for the first time used for all countries that have not responded to the last call, i.e. the Belarus, Croatia, Czech Republic, Denmark, Estonia, Latvia, Lithuania, Moldova, Slovakia, Spain and the Ukraine; (b) in the background database now also wetlands, grasslands and scrubs (EUNIS classes D, E and F) are included as receptors, and critical loads for N for these ecosystems are low compared to forest; (c) Norway has requested the CCE to use the background database for nutrient N; (d) Ireland and Germany updated their critical loads; (e) Romania and Slovenia have officially submitted data for the first time. Due to all these reasons the maps of *CLnutN* from 2006 (left map in Figure 2-2) is quite different from the one for 2008 (right map in Figure 2-2).

The modelled critical loads of nutrient N are nearly everywhere in Europe lower than the empirical critical loads of N, shown in Figure 2-3 for the four most commonly considered ecosystem types. This figure shows that wetlands and scrubs contain more sensitive ecosystems (in the range of 200 to 400 eq ha^{-1} a^{-1}) than forests and grasslands.

Like the critical loads for nutrient N, also the maximum critical loads of sulpher (*CLmaxS*) have changed since 2006. The reasons are generally the same, but the changes are less remarkable for most countries. Still there are clear differences in sensitive areas for Belarus, Romania, Poland, Germany, Austria and Switzerland.

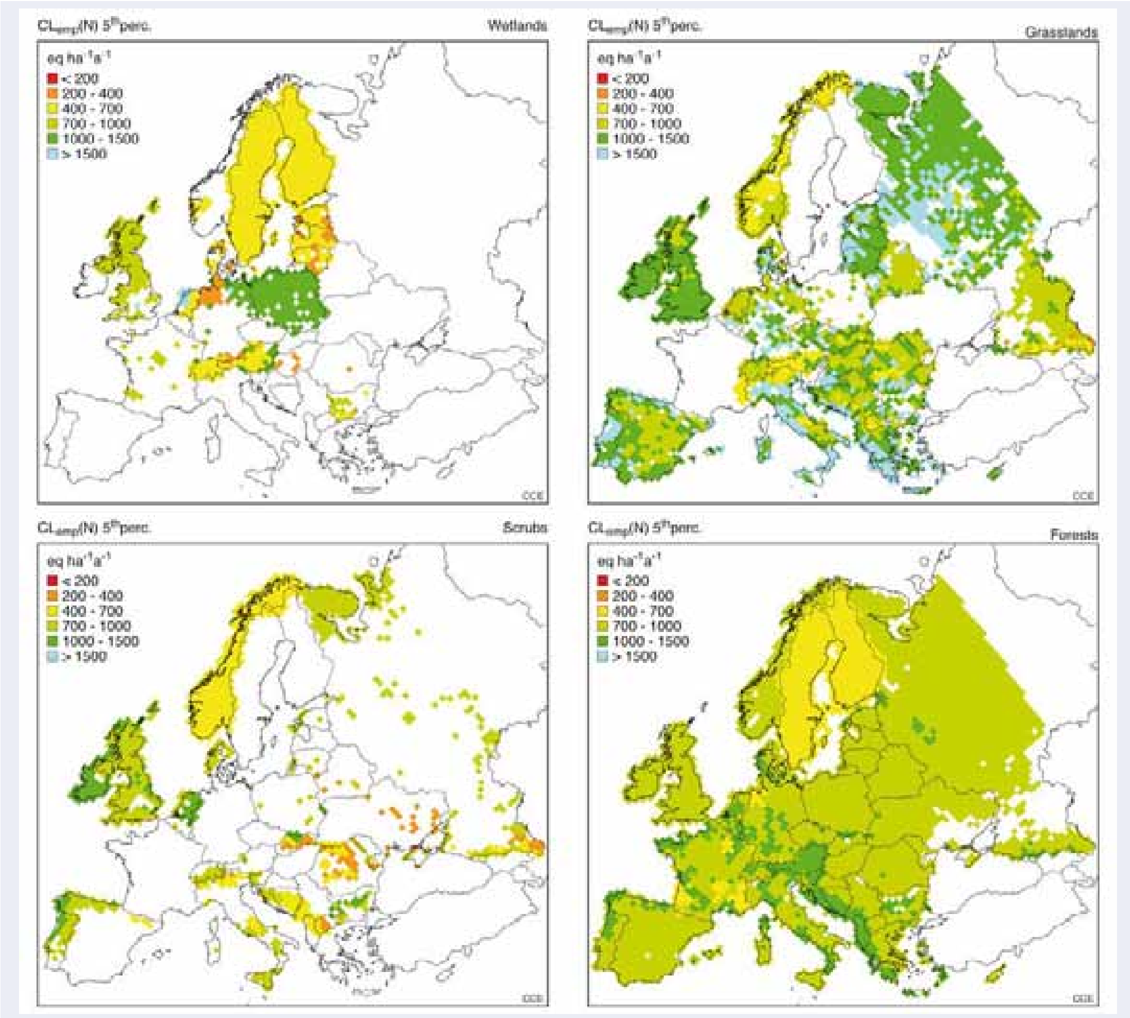

Figure 2-3 Empirical critical loads of nitrogen for the ecosystem types Wetlands (Top-left), Grasslands (Top-right), Scrubs (Bottom-left) and Forests (Bottom-right).

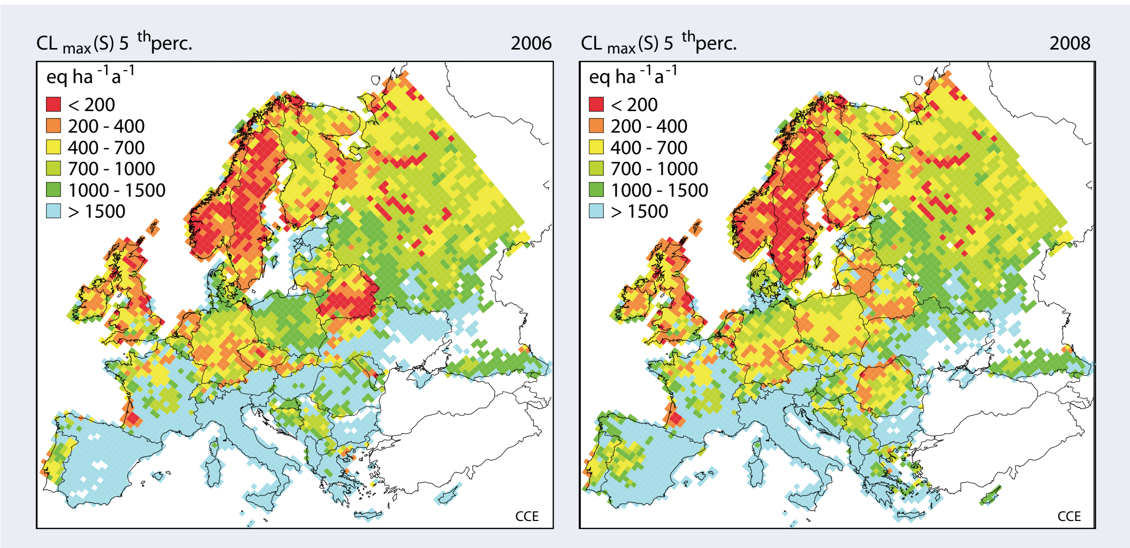

Figure 2-4 Maximum critical loads of sulphur (CLmaxS) for the 2006 data (left) compared the data for this year's call (right).

2.4 Exceedance of critical loads

Maps of exceedances of the updated critical loads are shown in Figures 1-4 and 1-5. Figure 2-5 displays the exceedances of *CLnutN* as cumulative distribution functions, separately for the EUNIS level-1 classes, and Figure 2.6 the exceedances of *CLempN*. The graphs show many exceedances of *CLnutN* and *CLempN* for ecosystems other than forests. Note that many ecosystems are exceeded by more then 1000 eq ha^{-1}a^{-1}. The fraction of ecosystems with this high exceedance can be seen from the intersection of the cdf at the right side of each graph.

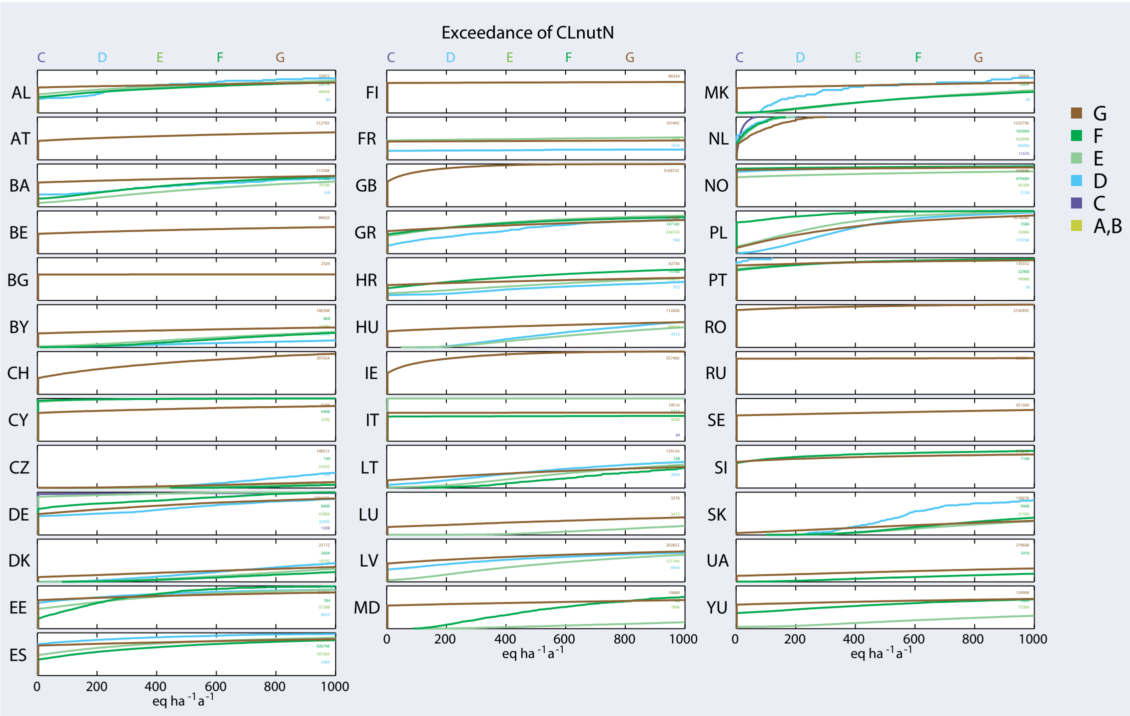

Figure 2-5 Distributions of exceedances of *CLnutN* for all countries and nearly all ecosystem types at EUNIS level 1.

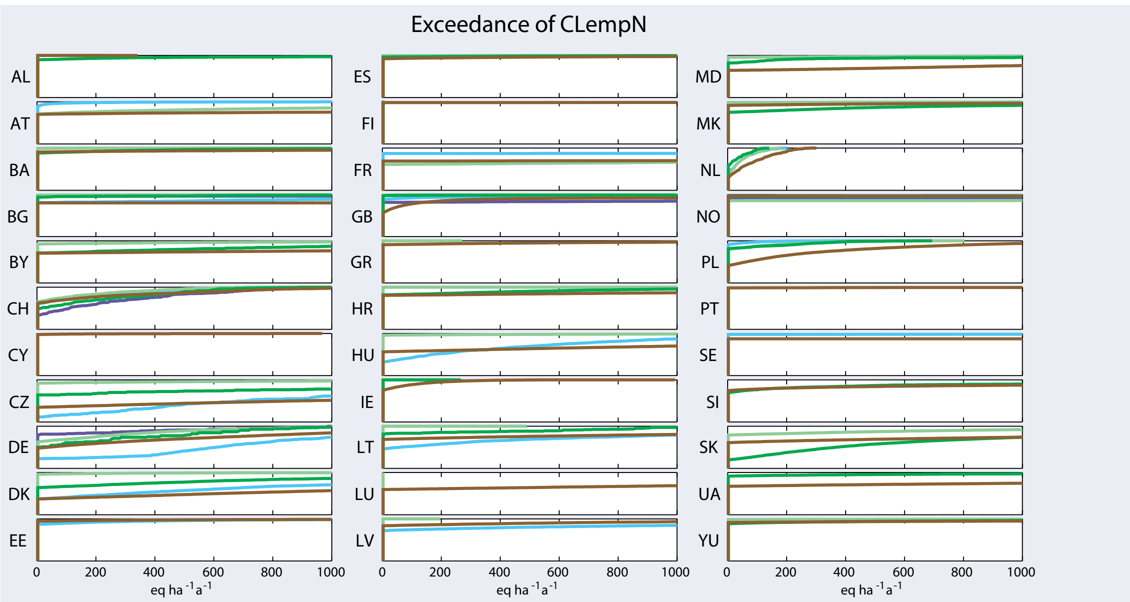

Figure 2-6 Distributions of exceedances of *CLempN* for all countries and nearly all ecosystem types at EUNIS level 1.

2.5 Dynamic modelling results

Twelve NFCs carried out dynamic modelling and provided modelling output for seven chemical variables in selected years in the past, 1980(10)2010, and for 14 deposition scenarios in the years 2020(10)2050 and 2100 (see Appendix B). For terrestrial ecosystems most countries used the VSD model (Posch and Reinds 2008), but also the SAFE model (Warfvinge et al. 1993) has been used (Sweden, Switzerland); for surface waters the MAGIC model (Cosby et al. 2001) has been used throughout. For the rest of the European countries the VSD model was used on the European Background Database (EU-DB; see Posch and Reinds 2005, Reinds et al. 2008 and elsewhere in this Chapter). The 12 NFCs produced dynamic modelling output for almost 700,000 sites (see Table 2-2), and from the EU-DB close to 400,000 sites are used for the other European countries.

For acidification (of soils) the molar Al:Bc ratio is a widely used criterion, and in Figure 2-7 the temporal development of the median and 5th and 95th percentile of this (derived) variable ('percentile traces') are shown for every country in Europe for the CLE and MFR scenarios. As can be seen that in all countries the Al:Bc ratio declines, below Al:Bc = 1 in most cases. Only in the Netherlands more than half of the ecosystems retain Al:Bc>1 even under the MFR scenario.

While acidification has declined substantially over the last decade, the interest in nitrogen with respect to its role in eutrophication and biodiversity has risen in prominence. Thus, in Figure 2-8 the percentile traces for the total N concentration in soil/lake water are displayed. For Belgium and the Netherlands even the medians are high. Despite the constant future deposition (after 2020), [N] is (very) slowly increasing in several countries owing to the slow filling-up of N pools and consequent increased leaching.

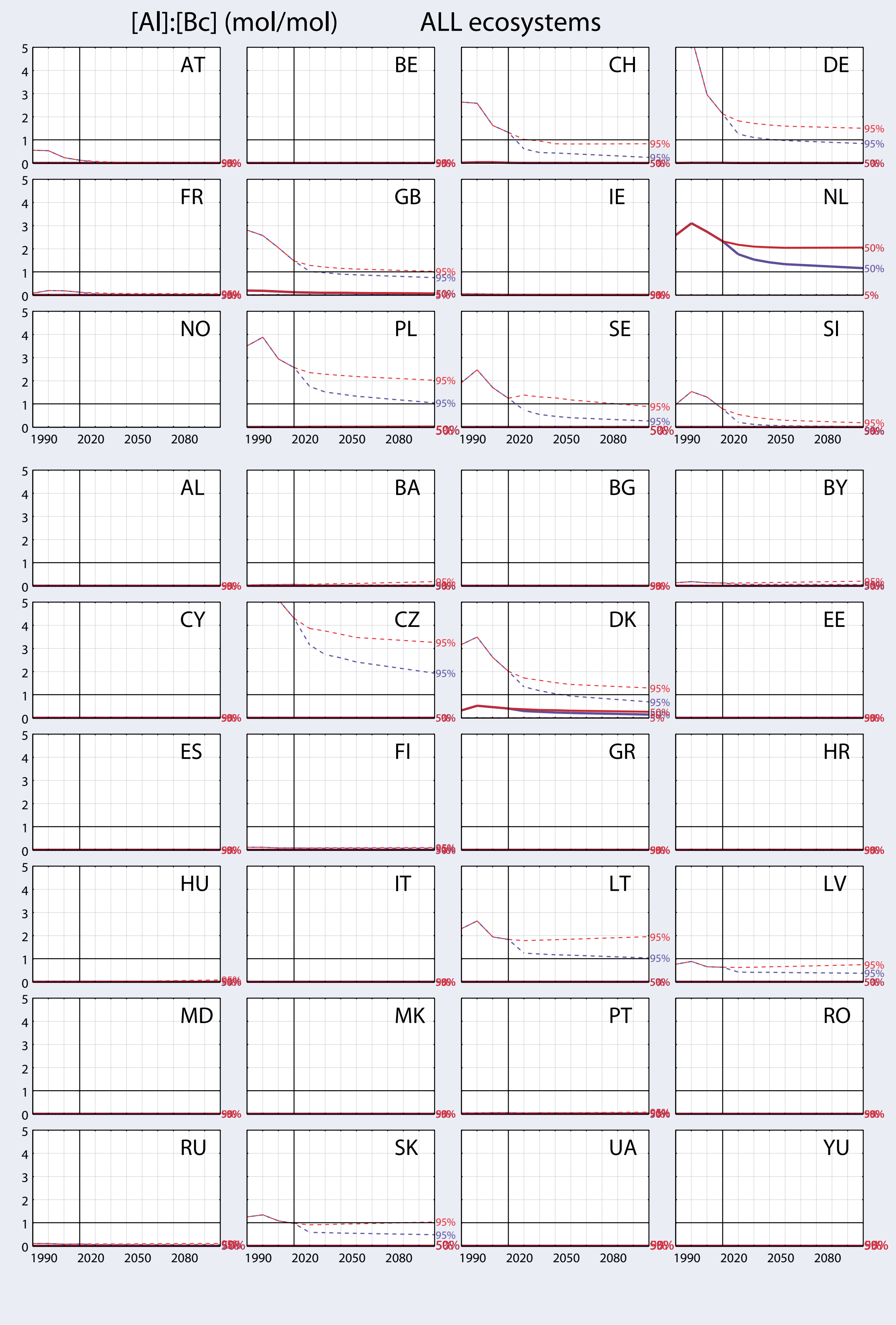

Figure 2-7 Temporal development of the median (solid line) and 5th and 95th percentile (dashed lines) of the molar Al:Bc ratio in European countries for two scenarios: CLE (red) and MFR (blue). The top 3 rows are the results from NFC simulations, whereas the rest are VSD simulations carried out on the European background database.

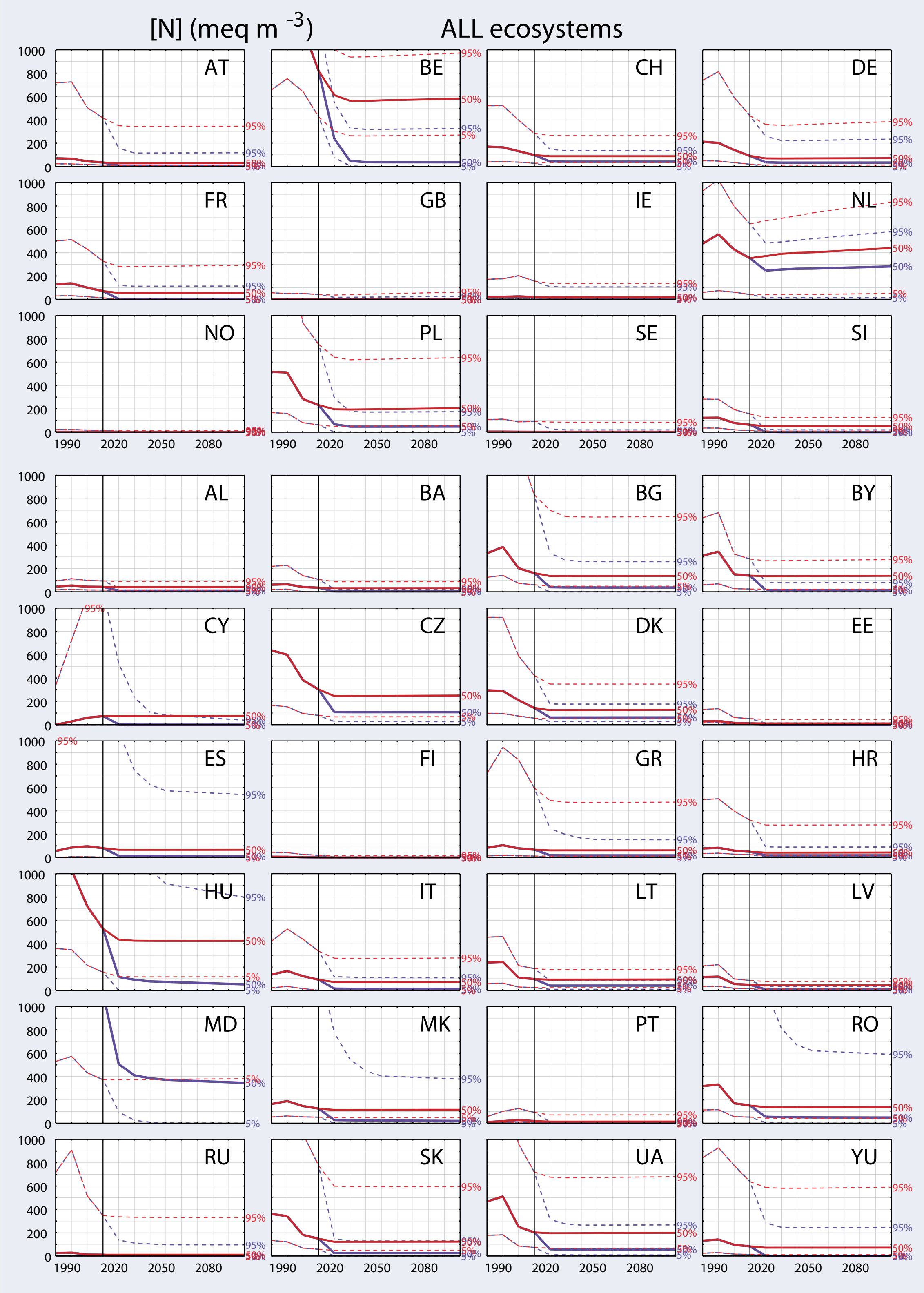

Figure 2-8 Temporal development of the median (solid line) and 5th and 95th percentile (dashed lines) of the total N concentration in European countries for two scenarios: CLE (red) and MFR (blue). The top 3 rows are the results from NFC simulations, whereas the rest are VSD simulations carried out on the European background database.

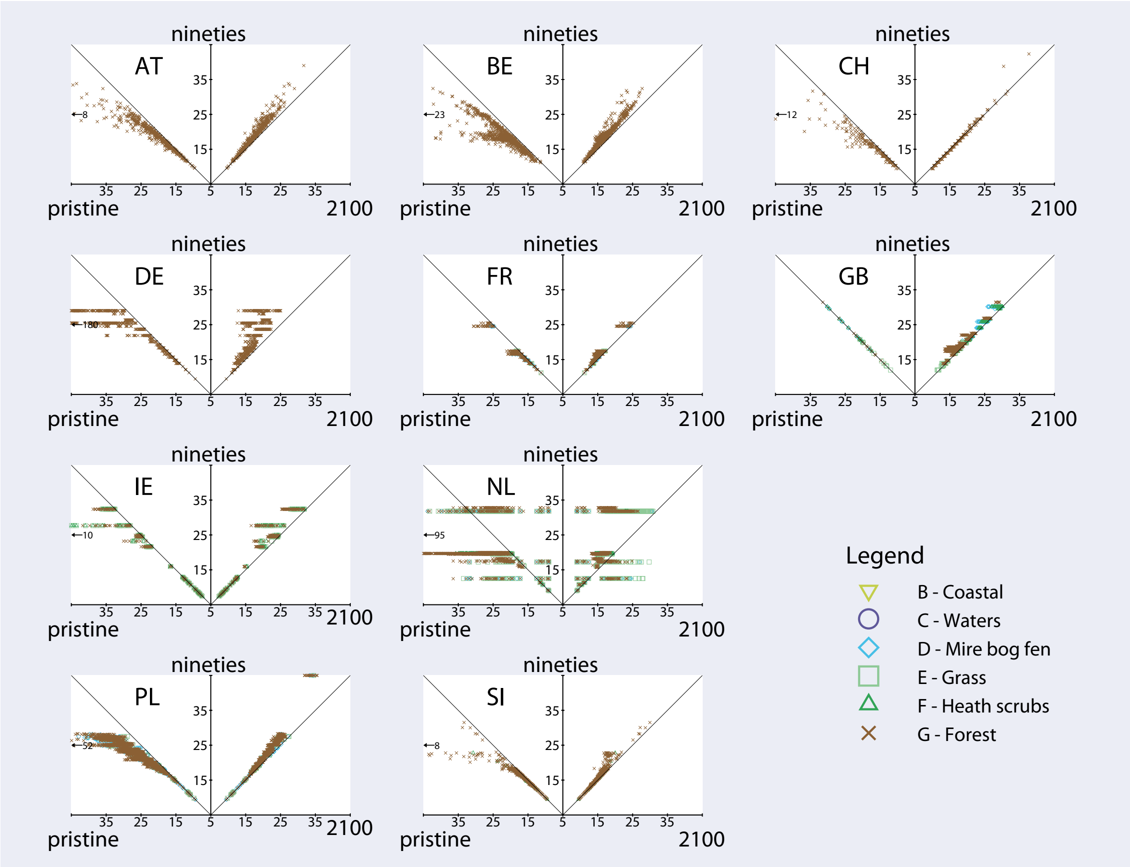

Figure 2-9 Correlation of C:N ratio between pristine and nineties (left-hand side of each plot) and between nineties and 2100 (CLE scenario; right-hand side).

While the nitrogen concentrations in general react relatively fast to changes in N input, the C:N ratio in the soil changes very slowly. To see significant changes time-scopes of a century or more are needed. This data call included the C:N ratio in 'pristine' conditions, i.e. at the start of the simulations. This quantity is usually derived by calibrating from a measured or estimated C:N ratio at present and a deposition history. 'Present' here means a recent survey or estimate. Figure 2-9 shows the correlation between C:N ratios under pristine conditions and the present ratio on the left-hand side of each plot, and between the present and the ratio in 2100 under the CLE scenario on the right-hand side for countries that submitted all ratios. Except for the Netherlands, the C:N ratio does not go up anywhere. The now included C:N ratio (CN_{seq} see Appendix B) in the VSD model (Evans et al. 2006) has apparently not been utilised by the submitting countries. Also interesting are the decreases in C:N ratio in the future, but lack of change in the past in the United Kingdom.

The concentration of nitrogen in soil solution is related to ecosystem effects via the critical (acceptable) N concentration, $[N]_{acc}$. This concentration depends on the ecosystem type (forest, grassland, etc.) and on the protection target (growth, vegetation change, etc.). It is therefore impossible to derive from Figure 2-8 the percent of ecosystem for which the criterion is (not) violated (unless there is a single $[N]_{acc}$ for the whole country). Comparing each ecosystem to its individual $[N]_{acc}$, however, yields these percentages, and they are displayed in Figure 2-10. It shows that the ecosystem area for which the criterion is violated is and remains high in most countries and that even the MFR scenario brings a substantial reduction in the violated area for about half the countries.

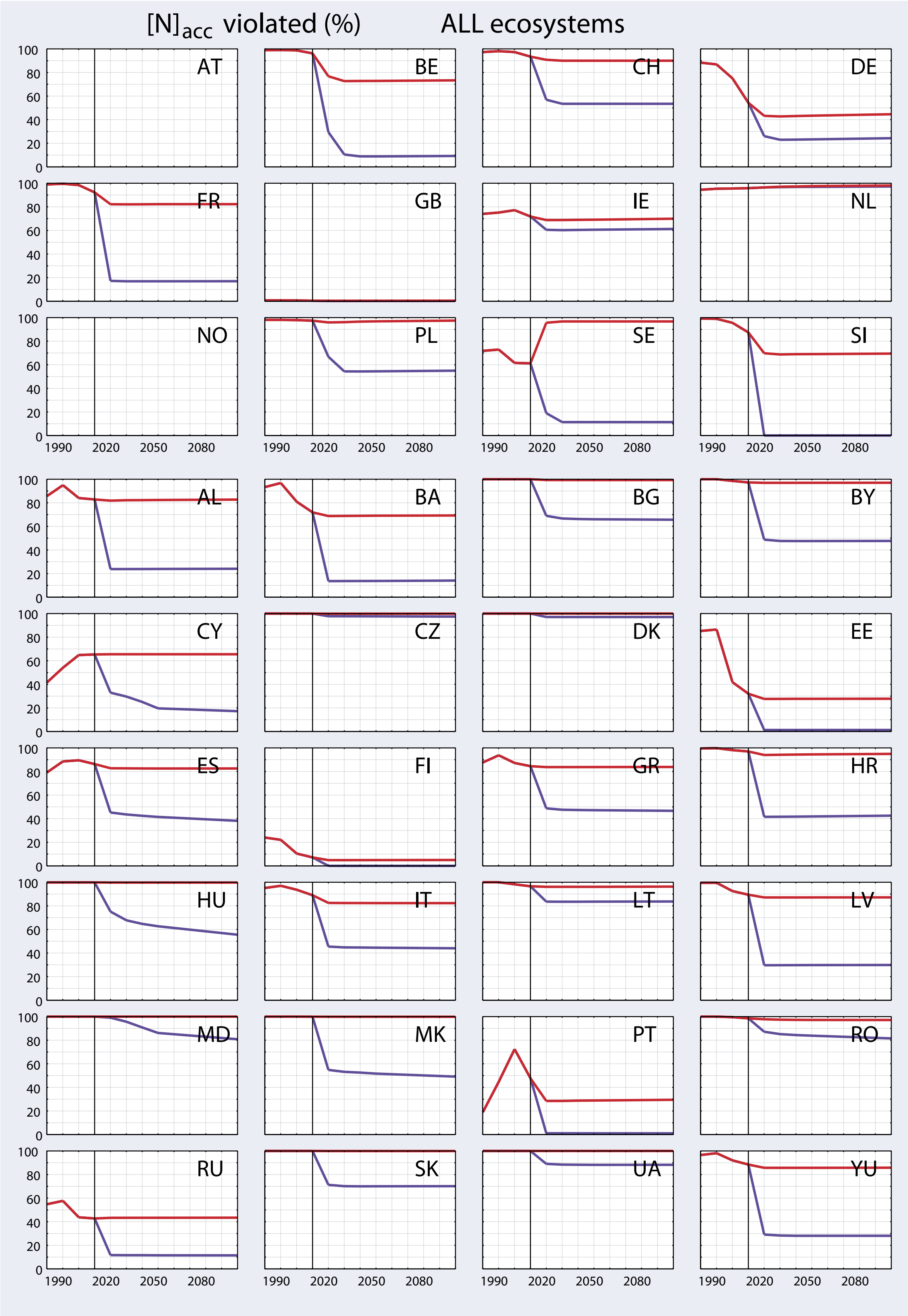

Figure 2-10 Temporal development of share (in %) of ecosystems violating the [N] criterion, i.e. with solution concentrations exceeding $[N]_{acc}$, for the CLE (red) and MFR (blue) scenario. The top 3 rows are the results from NFC simulations, whereas the rest are from VSD simulations on EU-DB.

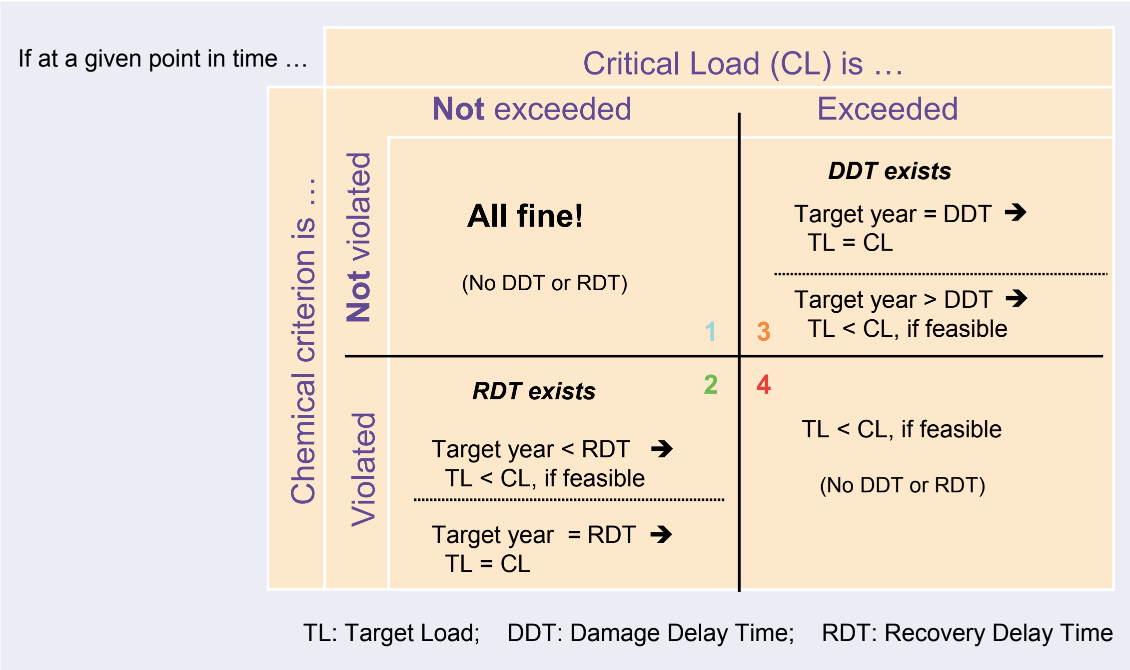

Figure 2-11 Categories 1–4 for which damage or recovery delay can occur for all combinations of critical load (non-) exceedance and criterion (non-)violation. The conditions for the existence of a target load are also shown (modified from Posch et al. 2005).

At any given point in time there are four possible states ('category') for an ecosystem with respect to critical load (non-)exceedance and criterion (non-)violation, and they are illustrated in Figure 2-11. Depending on the category, a damage delay or recovery times exist or a target load (<CL) can be computed. In the following we investigate these categories for eutrophication, i.e. using $[N]_{acc}$ as a critical limit. Table 2-3 shows the percent ecosystem area in each of the four categories under the CLE and MFR scenario in 2030 and 2100 for the whole of Europe.

As shown in Posch et al. (2007), for eutrophication either category '1' (blue) or category '4' (red) dominate (at least for the VSD model) since N concentrations rather quickly equilibrate with N inputs, thus making the other categories less likely. For the individual countries the same information is summarised in Figure 2-12. Note that an ecosystem is only included in these graphs if, in addition to the simulated N concentration, also the critical load *CLnutN* is available.

Table 2-3 Percent of European ecosystem area for the CLE and MFR scenarios in 2030 and 2100 in the four categories defined in Figure 2-N with respect to $[N]_{acc}$ and CLnutN.

Category	CLE scenario		MFR scenario	
(see Fig. 2-N)	2030	2100	2030	2100
1	30.8	31.7	41.1	69.0
2	2.6	2.5	0.6	1.2
3	7.7	7.6	8.3	5.8
4	58.9	58.2	50.0	24.1

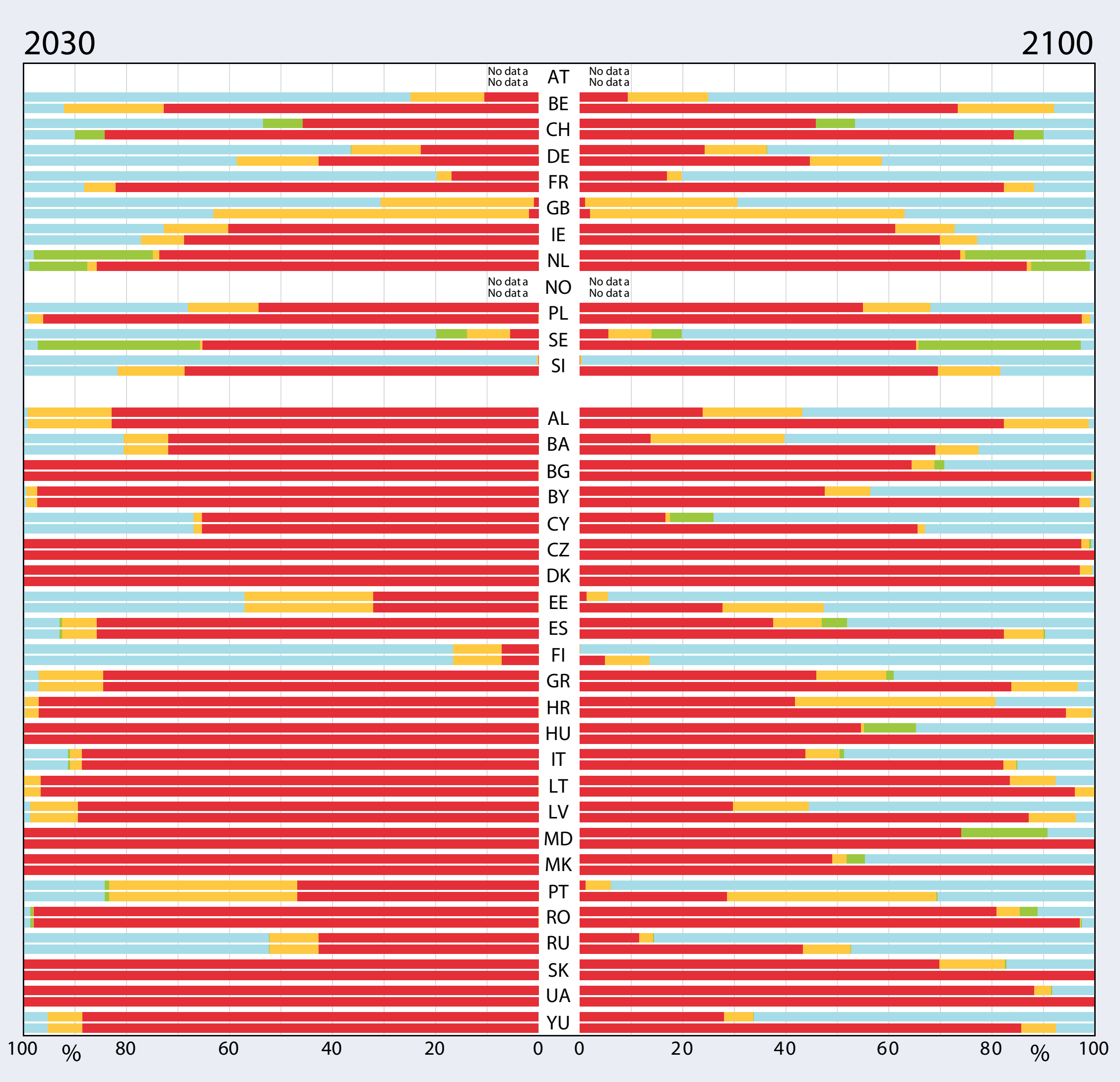

Figure 2-12 Percent of ecosystem area in the four categories defined in Figure 2-11 with respect to $[N]_{acc}$ and CLnutN for European countries for the MFR (upper bar) and CLE (lower bar) scenarios in 2030 (left) and 2100 (right). For the 12 countries at the top (separated by a blank line from the rest) NFC data are used.

2.6 Likelihood of exceedance using ensemble assessment

In Hettelingh et al. (2007) an approach to assess the likelihood of exceedances has been introduced, based on the methodology introduced by the IPCC (2005). This approach combines the areas exceeded regarding *CLnutN* and *CLempN* for each EMEP grid. Applying this approach to the recent CL data yields the maps of Figure 2-13. The maps show the likelihoods of exceedances of critical loads of nitrogen for (clockwise, starting with the upper left map) 1990, 2010 with current legislation (CLE), 2020 CLE, and 2020 with a scenario implementing a maximum of feasible reductions (MFR). Obviously exceedances become less likely between 1990 and 2010, and with MFR. With CLE exceedances are less likely in 2020 compared to 2010 for some regions, for instance central Europe, but increase in parts of Eastern Europe.

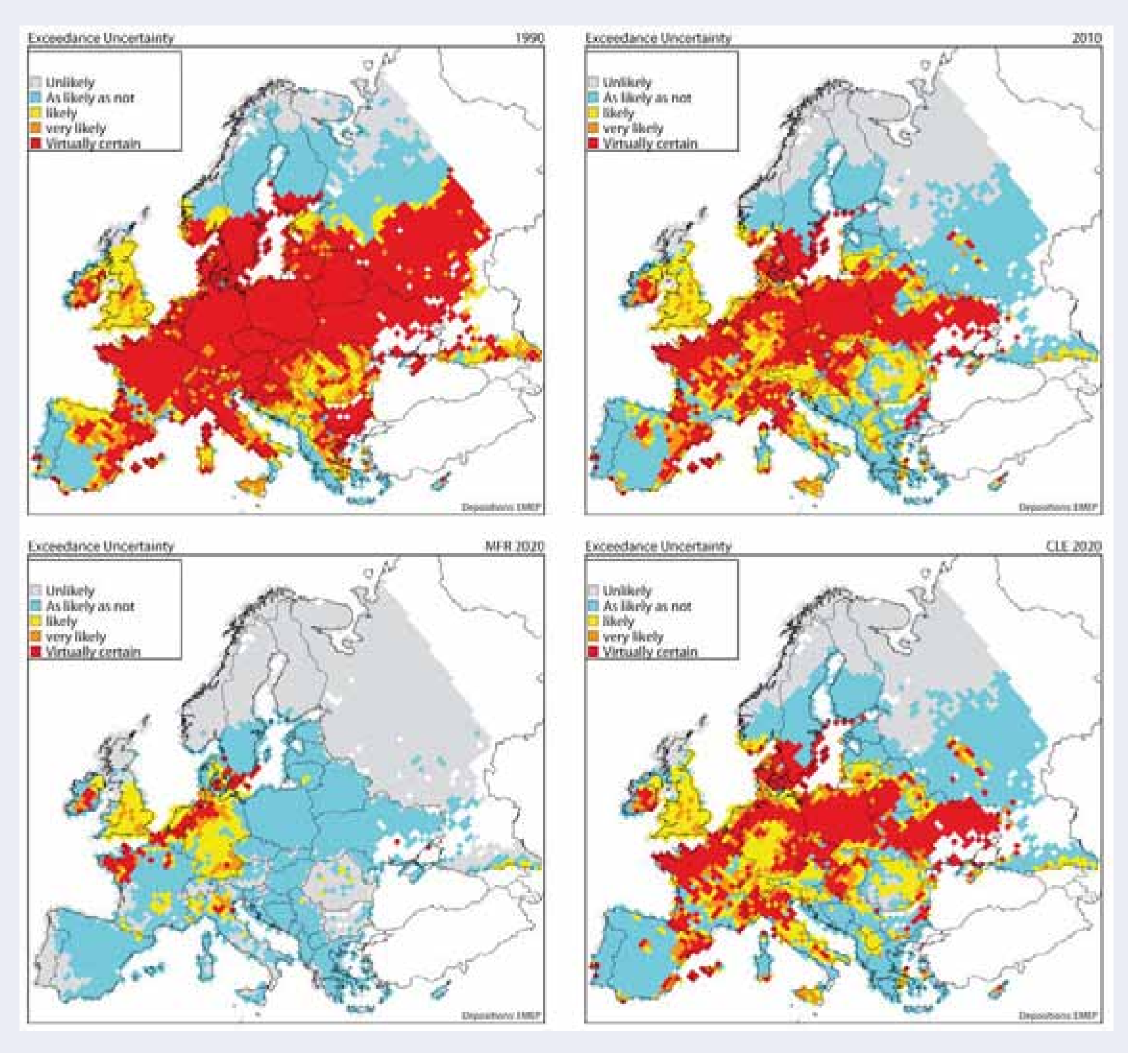

Figure 2-13 Likelihoods of exceedances of critical loads of nitrogen for (clockwise, starting with the upper left map) 1990, 2010 with current legislation (CLE), 2020 CLE, and 2020 with a scenario implementing maximum feasible reductions (MFR).

2.7 Updates of the European Background Database

The latest version of the European Background Database (EU-DB) maintained at the CCE is, *inter alia*, based on the LRTAP land cover map described in Cinderby et al. (2007) and has been comprehensively described in Reinds et al. (2008). This database is used to compute critical loads and carry out dynamic modelling (with the VSD model) for countries which did not deliver national data in response to the call for data. Since last year a few changes have been implemented in EU-DB, which are described in the sequel.

In applications of EU-DB up to now only forest ecosystems (EUNIS class G; Davies et al. 2004) had been considered. As of 2008 also semi-natural vegetation (EUNIS classes D, E and F) is included in EU-DB. Also for these categories critical loads are calculated with the SMB model as described in the Mapping Manual (UBA 2004), the only difference being that net plant uptake (= harvesting) of base cations and nitrogen is assumed zero. Considering only receptors larger than 1 km^2, EU-DB now contains about 700,000 records (= receptors). The criterion used in the calculation of critical loads for acidity is Al/Bc = 1 mol/mol for all receptors. For critical loads of nutrient N the critical (acceptable) N concentration used varies between 0.2 and 0.3 gN m^{-3}, thus staying on the precautionary side.

To date, for critical load calculations with the SMB model, the constant long-term net immobilisation N_i has been set to 1 kgN ha^{-1}a^{-1} for the whole of Europe, in accordance with the Mapping Manual (see p.V-11). This number is an 'extrapolation' of estimates of N accumulation in Swedish soils since the last glaciation in the range 0.2–0.5 kgN ha^{-1}a^{-1} (Rosen et al. 1992). This has been changed to a temperature-dependent formulation:

$$N_i(T) = N_i(T_0) \cdot \exp\left[A\left(\frac{1}{T_0} - \frac{1}{T}\right)\right] \qquad (2\text{-}1)$$

where T is the annual mean (soil) temperature in Kelvin, T_0 = 275 K (about the average Swedish annual mean temperature), and $N_i(T_0)$ = 0.35 kgN/ha/a, the mid-point of the observed Swedish values. Setting A = 12000 K^{-1}, yields N_i = 1 kgN ha^{-1}a^{-1} for 9°C and 3 kgN ha^{-1}a^{-1} for 16°C, thus still underestimating the choices of N_i by many NFCs.

Figure 2-14 shows a graphical representation of eq.2-1, and in Figure 2-15 the 5th and 95th percentile of N_i in every EMEP-50 grid over Europe is shown as computed from eq.2-1 and the climate information in EU-DB. In-grid variation comes from the high-resolution temperature data in EU-DB, which are described in New et al. (2000), and which is most pronounced in mountainous areas.

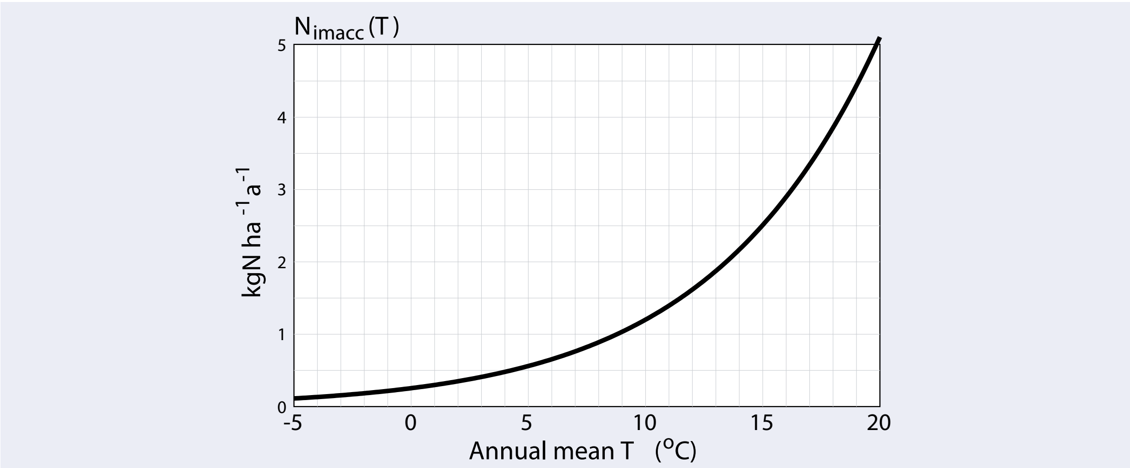

Figure 2-14 Temperature-dependence of the long-term net immobilisation used for critical load calculations in EU-DB.

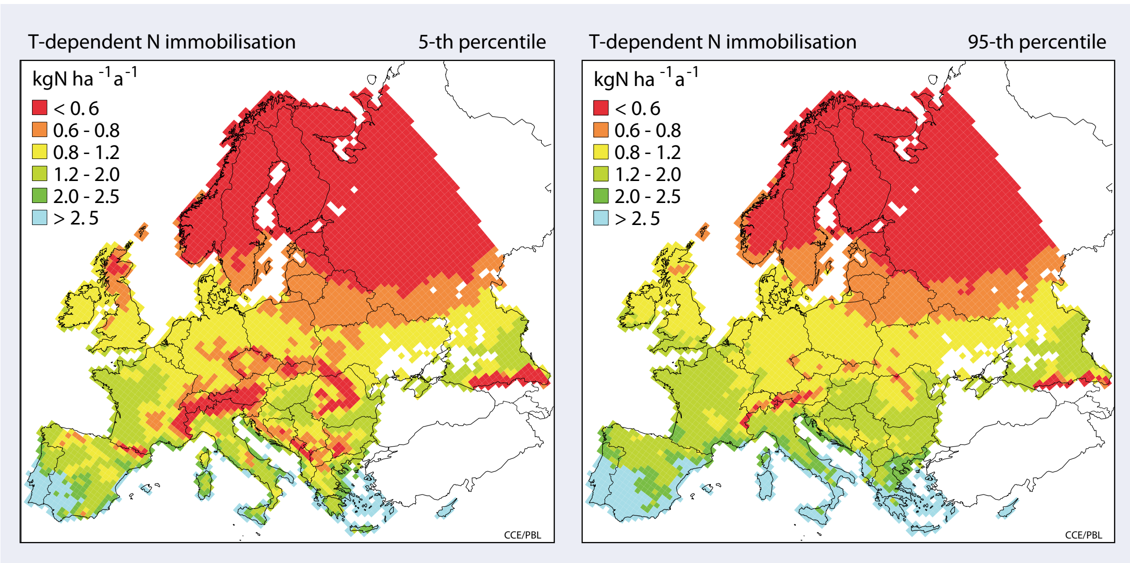

Figure 2-15 Spatial distribution (5th and 95th percentile) of the long-term net immobilisation in EU-DB.

It is fair to say that the term N_i in the SMB model is one of the least known and most difficult to determine, especially since the enhanced N input to ecosystems over the last decades has led to increased *immobilisation* of N, but opinions differ whether this is sustainable or not in the long run.

In the LRTAP land cover map (Cinderby et al. 2007) a EUNIS ecosystem code has been assigned to every land cover class (De Bakker et al. 2007), and therefore it is possible to assign empirical critical loads of N to them, since they are also classified according to EUNIS (Achermann and Bobbink 2003 and Chapter 5.2 of the Mapping Manual). However, in all cases empirical critical loads are given as an interval, thus necessitating a criterion to arrive at a unique critical load for every receptor. The Mapping Manual provides some qualitative guidance on selecting a critical load value from the given intervals (see Table 5.2 in UBA 2004). Two such criteria are soil wetness (lower CL for drier sites) and base cation availability (lower CL for lower availability). This has been made quantitative by using the precipitation surplus (Q_{le}) as indicator for soil wetness and the sum of Bc (= Ca+Mg+K) deposition and Bc weathering as indicator of Bc availability (Bc_{av}). The joint distributions for each of the for EUNIS classes D–G of these two quantities, derived from the data of every receptor in EU-DB, are used to obtain a unique modifying factor f_{mod} ($0 \le f_{mod} \le 1$) for any site; and the empirical critical load is then computed as:

(2-2) $$CL_{emp}(N) = CL_{lo} + f_{mod} \cdot (CL_{up} - CL_{lo})$$

where CL_{lo} and CL_{up} are the lower and upper limit of the empirical N critical load given in Table 5.1 in the Mapping Manual (UBA 2004).

Figure 2-16 shows isolines of the modifying factors for the EUNIS classes D–G derived and used in the European background database.

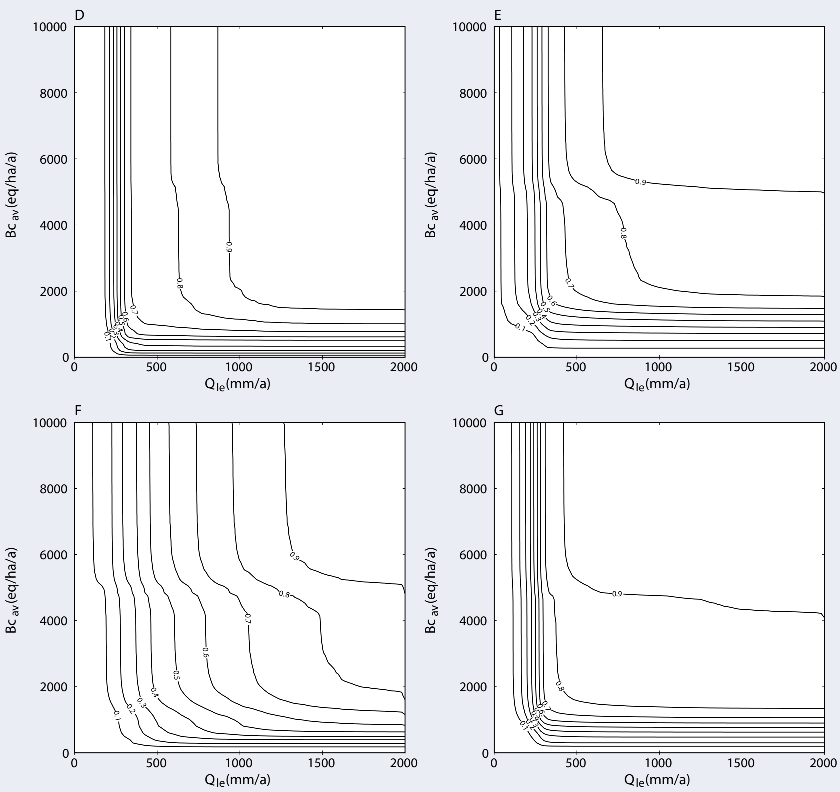

Figure 2-16 Isolines of modifying factors as function of leaching (Q_{le}, in mm/a) and base cation availability (BC_{av}, in eq/ha/a) for the EUNIS classes D (top left) through G (bottom right).

It has to be noted that the derivation of these modifying factors depends on the interpretation of 'low' and 'high' for the modifying quantities: If one takes the data for a single country only, the joint distributions will look different, and thus the modifying factors as well. Unfortunately, the Mapping Manual does not provide guidance on how to interpret 'low' or 'high' soil wetness, Bc availability etc. (i.e. low compared to what?); and this requires further discussions.

2.8 Input variables for critical loads and dynamic modelling

In the mass balance for nitrogen (Mapping Manual, eq.5.3) the four 'sinks' considered are denitrification, net uptake, net immobilisation and leaching. These quantities (at critical load) are given in Figures 2-17 and 2-18. Each country-plot has four quadrants showing the correlation diagram between (starting for the first quadrant, in clockwise order) denitrification and uptake, uptake and leaching, leaching and immobilisation, and finally immobilisation and denitrification. The colours of the dots (each representing an ecosystem) are given by the exceedance of nutrient nitrogen in 2010 (see Figure 1-5). For Norway the values of the background database are shown, because these are used to calculate *CLnutN*.

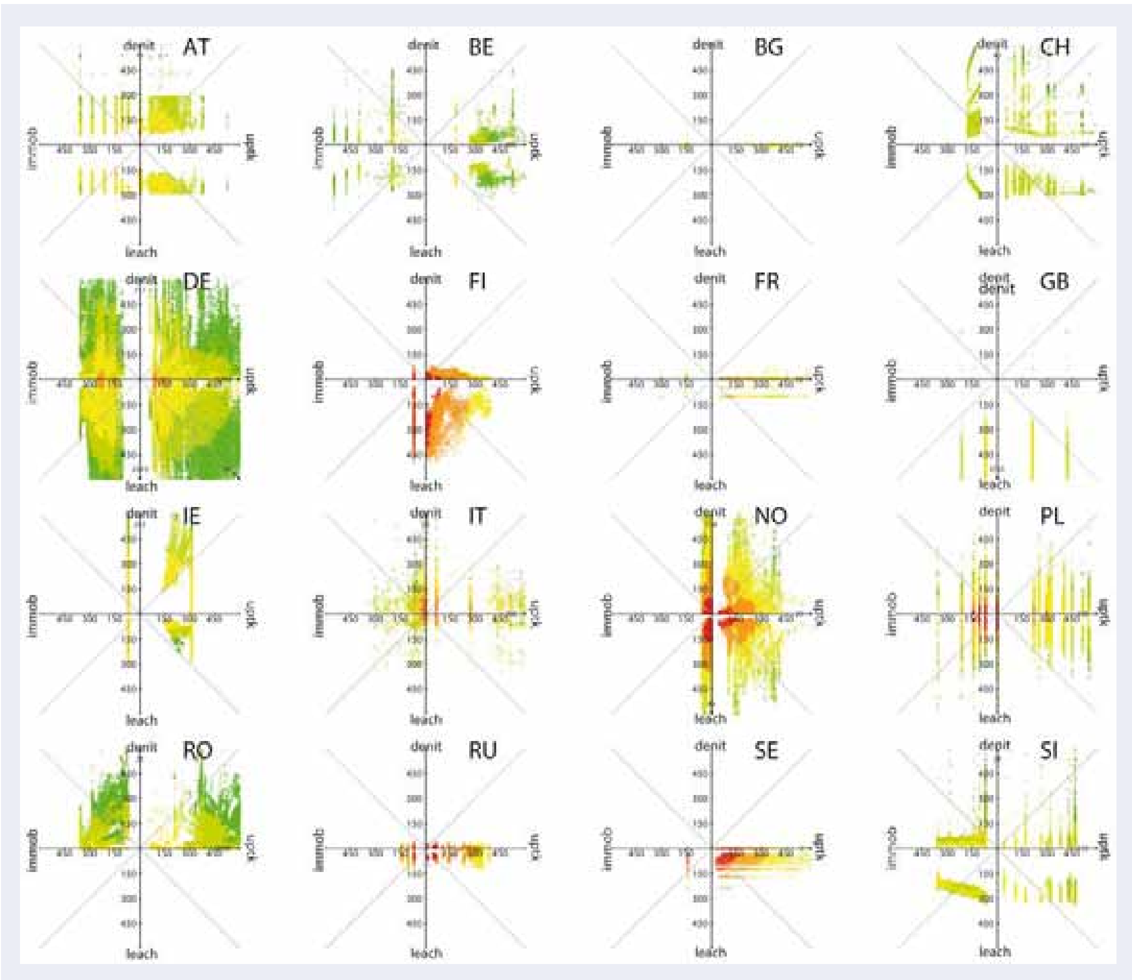

Figure 2-17 Correlations between the sinks of nitrogen: denitrification, uptake, leaching and immobilisation for the data submitted by NFCs. The colours of the dots reflect the exceedance of CLnutN in 2010.

As can be expected the most exceeded areas (in red) are close to the origin of the plots, where the sinks have low values. It is not the primary intention of this graph to correlate the quantities shown, but it does give an impression of the distributions. This impression can be deceiving, especially if many ecosystems have the same value(s) and obscure each other, like uptake in the background database which is zero for all but forests. In this respect one should realise that in each quadrant the same number of ecosystems is plotted. The fact that each quantity is plotted twice, on each side of its axes, can help estimate the characteristics of the ecosystems. Dutch data is missing for all sinks but leaching, and is therefore not plotted at all.

Generally uptake has the largest values, but also leaching (for areas with high precipitation) and denitrification can be significant. There is a considerable difference between the background database and the submitted data, also for countries in the same region of Europe, with higher values in the submitted data.

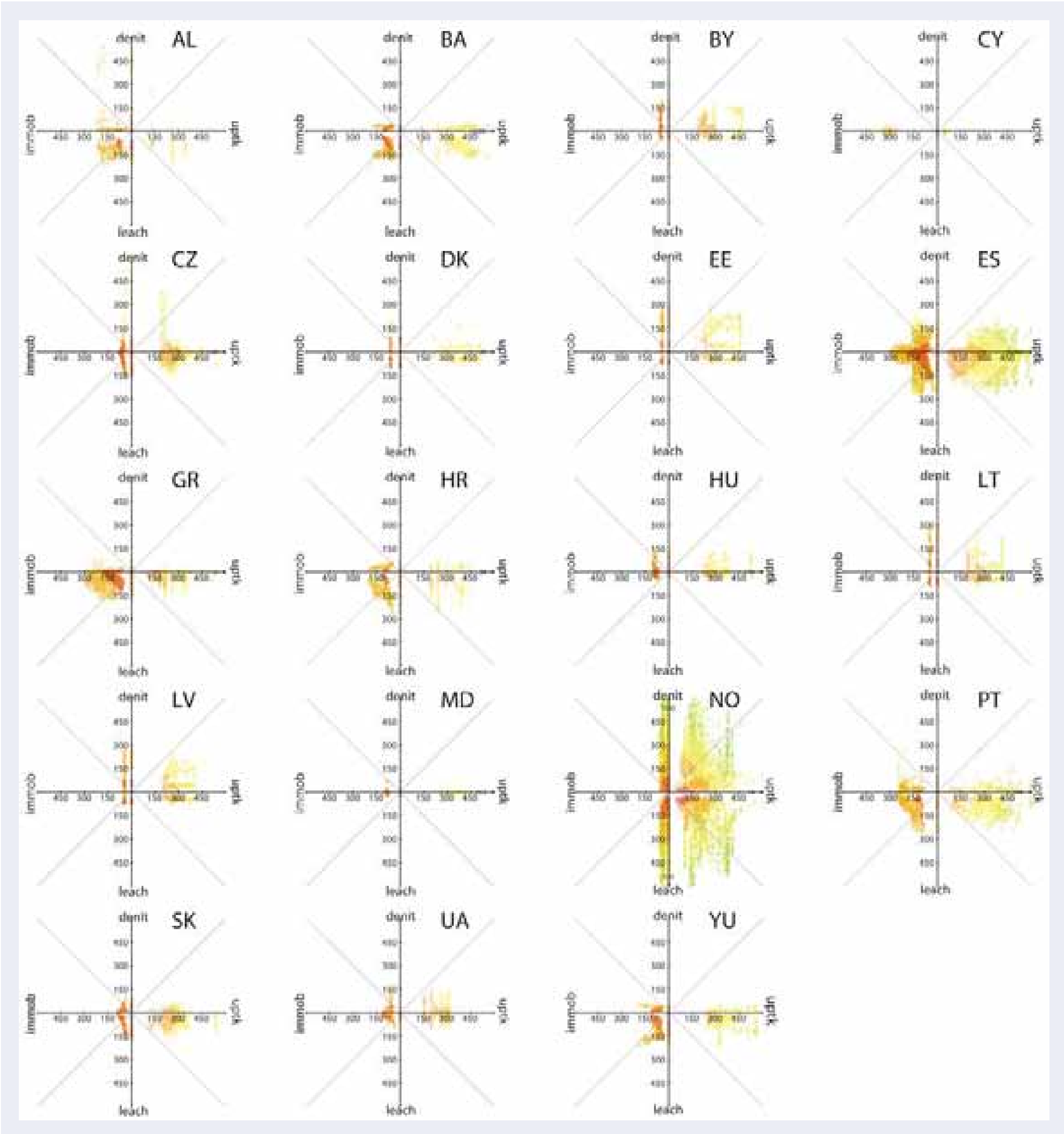

Figure 2-18 Correlations between the sinks of nitrogen: denitrification, uptake, leaching and immobilisation for countries for which the EU-DB data are used. The colours of the dots reflect the exceedance of CLnutN in 2010.

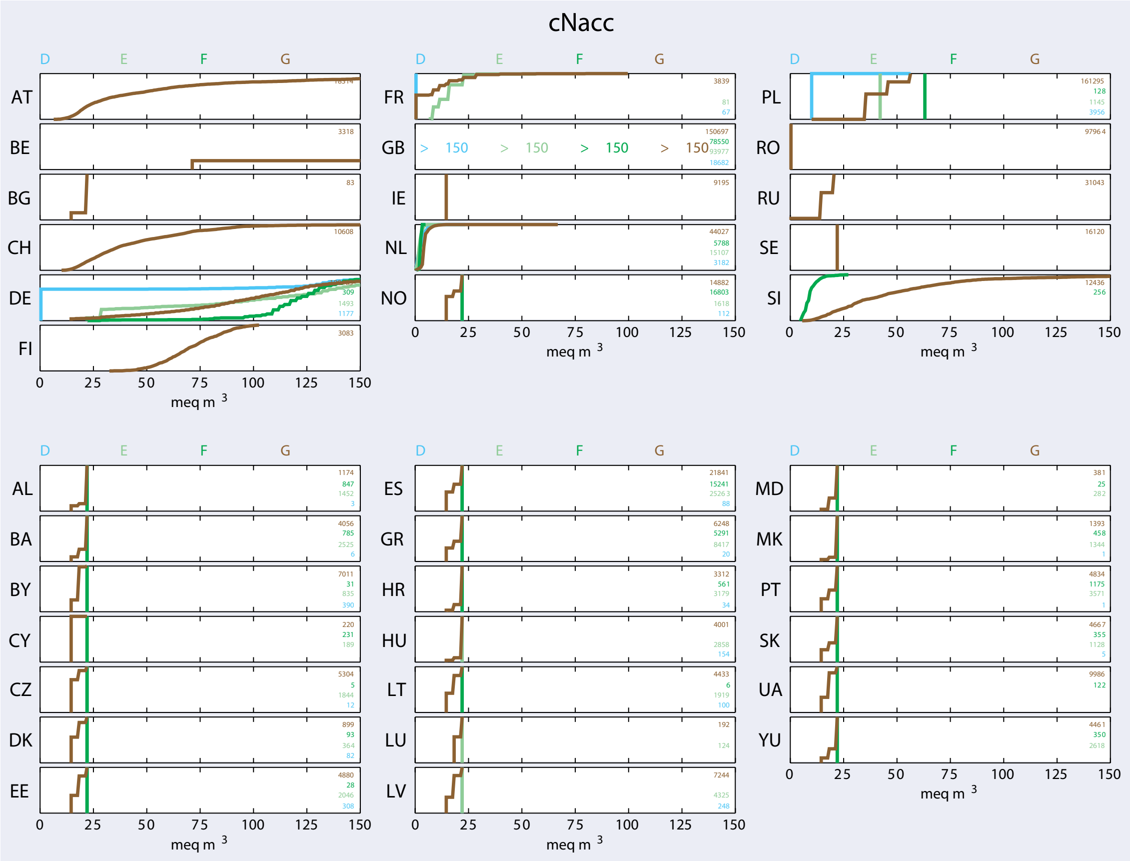

Figure 2-19 Distribution of $[N]_{acc}$ of submitted data (top) and the European background database (lower graphs).

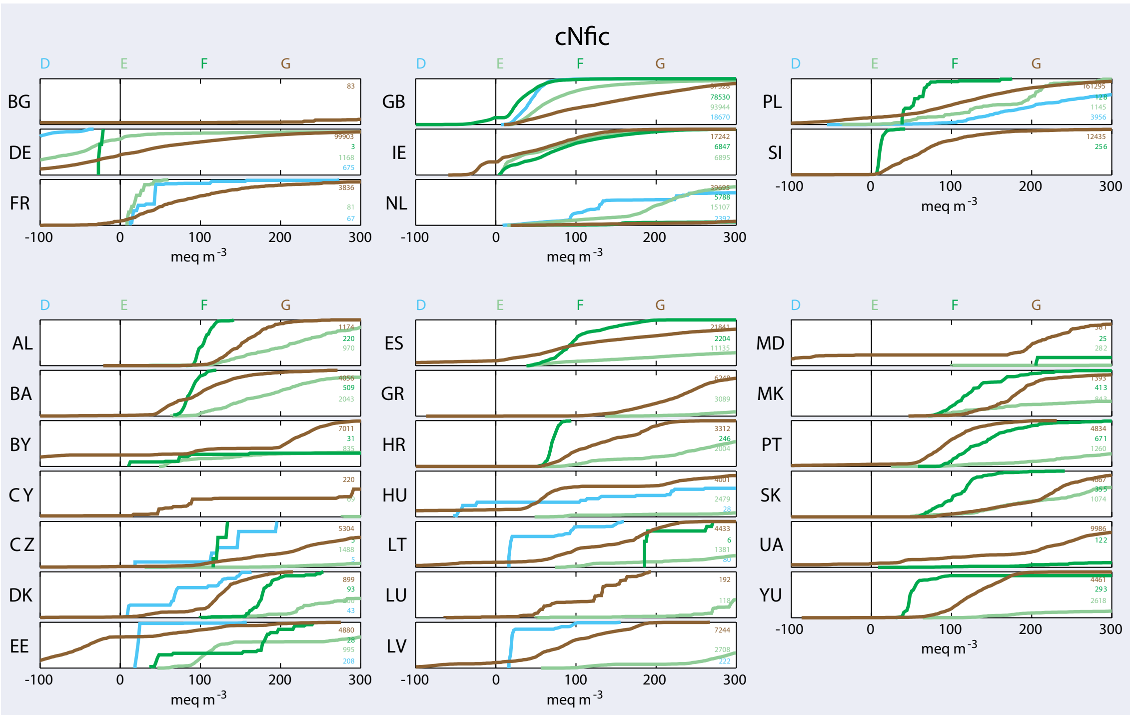

Figure 2-20 Fictive nitrogen concentrations, $[N]_{fic}$, based on empirical critical loads and SMB input parameters for national submissions (top) and EU-DB entries (lower graphs).

The leaching of nitrogen depends on the amount of percolating water and the concentration which is considered to be the limit for eutrophication effects. This limit, $[N]_{acc}$, is perceived differently by the submissions, as can be seen in Figure 2-19. It is a revealing exercise to derive the N concentration corresponding to the empirical critical load of N using the uptake, denitrification, immobilisation and the flow of percolating water from the *CLnutN* calculations:

$$[N]_{fic} = \left((1-f_{de})\cdot CL_{emp}(N) - N_i - N_u\right)/Q \qquad (2.3)$$

This 'fictional' concentration is plotted in Figure 2-20 for countries for which all the necessary data were available. Results range from negative values (total of sinks of nitrogen supersede the empirical critical load) to values larger than 300 meq m^{-3}.

2.9 Conclusions

Submissions by National Focal Centres have been incorporated in the European critical load database of 2008 (2008 CL-database). This database is a significant update of the last official critical load database of 2006 (2006 CL-database).

This chapter includes an extensive analysis of the differences between the 2008 CL-database and 2006 CL-database. Reasons for these differences include first time submissions, the use of an updated European background database for countries that did not submit data and varying interpretations of the information provided in the Mapping Manual regarding e.g. immobilisation, denitrification and acceptable nitrogen concentrations. Further work is needed to strengthen consensus on these parameters.

Despite these differences, exceedances of critical loads of nitrogen still remain virtually certain all over Europe under the current legislation scenario. Exceedances persist in central Europe, even under the maximum feasible reduction scenario.

The concentration of nitrogen in groundwater responds quickly to changing depositions, but equilibrium can take ages to be reached, as shown by national applications of dynamic models.

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3 Illustrative Dynamic Modelling Applications for Use in Support of European Air Pollution Abatement Policies

Maximilian Posch, Jean-Paul Hettelingh, Jaap Slootweg, Rob Maas

3.1 Introduction

In this chapter we want to show with illustrative examples the use of dynamic models. The purpose is two-fold: To illustrate what can (or cannot) be done with dynamic models and to show how dynamic model results can be used in support of European air pollution abatement policies. Although our statements will in general be true for any dynamic model, we use VSD model (Posch and Reinds 2008) simulations on the European background database (see Chapter 2 and Posch and Reinds 2005) throughout this chapter to illustrate our points. We also draw the attention to the chapter on "critical loads and dynamic modelling of nitrogen" in the 2007 CCE Progress Report (Posch et al. 2007).

3.2 Approaching steady state

Critical loads are (by definition) steady-state quantities, i.e. they do not change over time, and in this way they are an idealization of the true world, which is subject to permanent change, not only with respect to deposition, but also in other driving forces such as climate. Neglecting the future change of others drivers in this chapter, also deposition scenarios are in general only given for a few decades into the future – in fact only until 2020 in the case of scenarios prepared for assessments under the LRTAP Convention. Therefore, to simulate the long-term response of an ecosystem (soil or surface water) the simplifying assumption is made that depositions do not change after 2020 (for a given scenario). The constancy of any scenario deposition after a certain year (called the 'implementation year') is also the basis of many concepts developed in the context of dynamic modelling, such as target load and (damage and recovery) delay times (see Posch et al. 2003). Therefore it seems legitimate to ask how fast different output variables of dynamic models approach steady state, since this determines, e.g., how fast a chemical criterion used in critical load calculations is attained and thus helps to judge how 'effective' critical loads are in avoiding 'harmful effects'.

Different variables will react with different speed to a constant input. Solution concentration will react much faster than soil pools; but the interaction between pools and solution will also influence the long-term values of solution variables. E.g., in the case of sulphate, which does not interact with any pool in the VSD model, the solution concentration attains the steady-state value commensurate with the S deposition input within a few years or at most decades. Base cation or nitrogen concentrations, on the other hand, are influenced by the slow changes of the exchangeable base cation pool or the C and N pools in the soil, and thus for the long-term development of soils the solid phase pools are most important. Therefore we will investigate in this chapter the long-term temporal development of base saturation (characterising the base cation pool) and the C/N ratio (characterising the N pool).

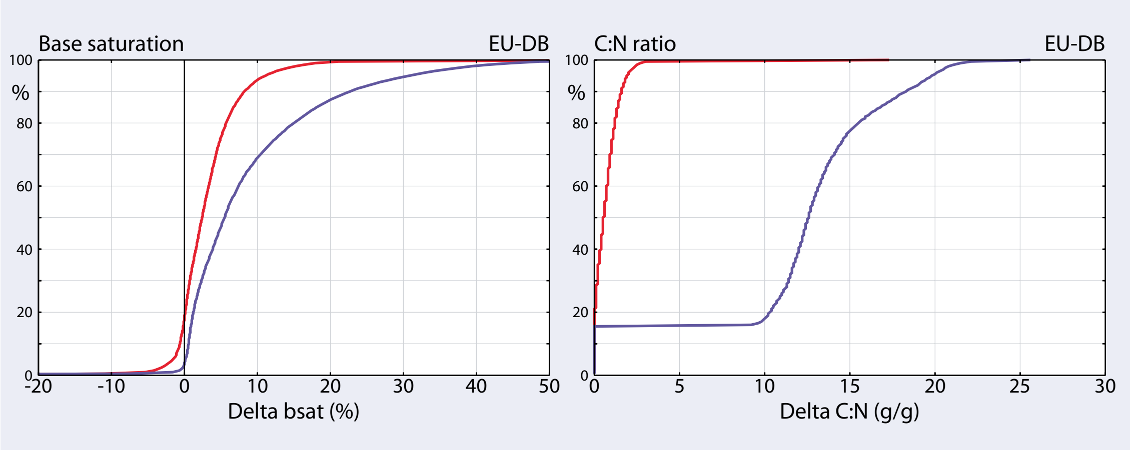

Figure 3-1. Left: CDFs of the difference between base saturation in 2100 and 2500 (red line) and 2500 and steady state (blue) under the CLE scenario deposition (since 2020) for non-calcareous soils in Europe (EU-DB). Right: same for the C/N ratio.

We ran the VSD model until the year 2500 for about 570,000 non-calcareous soil sites in Europe using the CLE deposition scenario and compare base saturation and C/N ratio in 2100 with the values in 2500 and with the steady-state values of these quantities. This is done by displaying CDF of the differences in base saturation and C/N ratios (2100 minus 2500 and 2500 minus steady-state; Figure 3-1). Steady-state base saturation and C/N ratios are computed with a special module of the VSD model which computes the steady-state values of all variables for any given constant input, thus not relying on having reached steady state by simulating 'a couple of hundred years' into the future. As can be seen, even in 2500, i.e. after 480 years of constant N and S deposition, a large number of sites has not reached steady state; in fact, for many sites there is more change after 2500 than there is between 2100 and 2500 (especially for the C/N ratio)! Although the differences between steady state in values in 2500 might be negligible for many sites, this shows nevertheless that one has to be cautious to speak of having reached steady state after a few decades or centuries of simulations.

Since the attainment of a steady state can take arbitrary long for a number of sites, we use in the following the values for the year 2100 as 'long-term values' in the comparison of different scenarios, keeping in mind that this might be far from steady state for some sites.

3.3 Long-term development of soil variables under different scenarios

Scenario analyses, i.e. the answer to 'what-if' questions related to effects of changes in exceedances of critical loads, is especially important when emission reductions are not sufficient to achieve depositions that are equal to or below critical loads. Emission reduction scenarios that are prepared for integrated assessment lead to different regional patterns of exceedances in European ecosystems, depending on the amount and location of abated emissions. However, we know from Figure 1-7 that the occurrence of an exceedance is not sufficient to answer the 'what-if' question, as non-exceedance does not mean that an ecosystem has recovered (i.e. the chemical criterion remains violated).

The use of dynamic models extends the assessment of the regional pattern of exceedances *now* to also include the regional pattern of critical limit (non-)violation in the future, i.e. 2100 in this chapter. Therefore, emission reduction scenarios come with specific regional patterns of (non-)exceedances as well as of (non-)violations of critical limits. For every emission reduction scenario the regional distribution of simulated chemical values – including (non-) violations of chemical criteria – changes over time. Therefore, in the context of integrated assessment the performance of an emission reduction scenario can also be assessed by comparing the distribution of critical limits between scenarios for years of interest.

The analysis of the long-term development of soil variables in this chapter focuses on the comparison – between emission scenarios – of the cumulative distribution of critical limit values. For acidification we focus on base saturation (B_{sat}), while for eutrophication we address the carbon to nitrogen (C/N) ratio. As examples for 'safe' critical limit values we use the range of B_{sat} between 5% and 20% (see, e.g., Augustin 2005, Černý and Pačes 1995, p.85; see also Braun et al. (2003) for the importance of base saturation). Dise et al. (1998) found a significant relationship between nitrogen input and the C/N ratio. Different datasets support a threshold of C/N ratio 25, below which nitrate leaching seems to increase (Gundersen et al. 1998, MacDonald et al. 2002).

Six examples are chosen to illustrate the analysis of long-term effects of acidification and eutrophication in integrated assessment using the European background database.

1. What is the historic distribution of B_{sat} and C/N in 1900 and in 1980?

To get a point of reference for assessing future distributions of B_{sat} and C/N we look at their distributions in the historic years 1900 (close to the start of the simulations) and 1980 (around the maximum of deposition in most European countries). The deposition history is based on Schöpp et al. (2003; see Appendix A for details). The cumulative distribution (cdf) of B_{sat} in 1980 (black curve in left panel of Figure 3-2) is lower than that for 1900 (grey) This is to be expected because soils have acidified between 1900 and 1980, especially because emissions dramatically increased over that period. The two cdfs show the overall shift in base saturation, but do not say how dramatic (or not) the changes are at individual plots. This can be seen on the right panel of Figure 3-2. It also shows that a few (fringe) sites even increase in base saturation between 1900 and 1980. Figure 3-2 shows that 10% of the area has a B_{sat} lower than 20% in 1900 (e.g. 90% are 'safe') compared to about 15% (85% 'safe') in 1980. The difference between 1900 and 1980 in the effect of eutrophication also shows the effect of increased nitrogen stress until 1980. More than 50 % of the area has a C/N ratio higher than 25 in 1900. The area with an acceptable C/N ratio decreases to less than 50% in 1980 (see Figure 3-3).

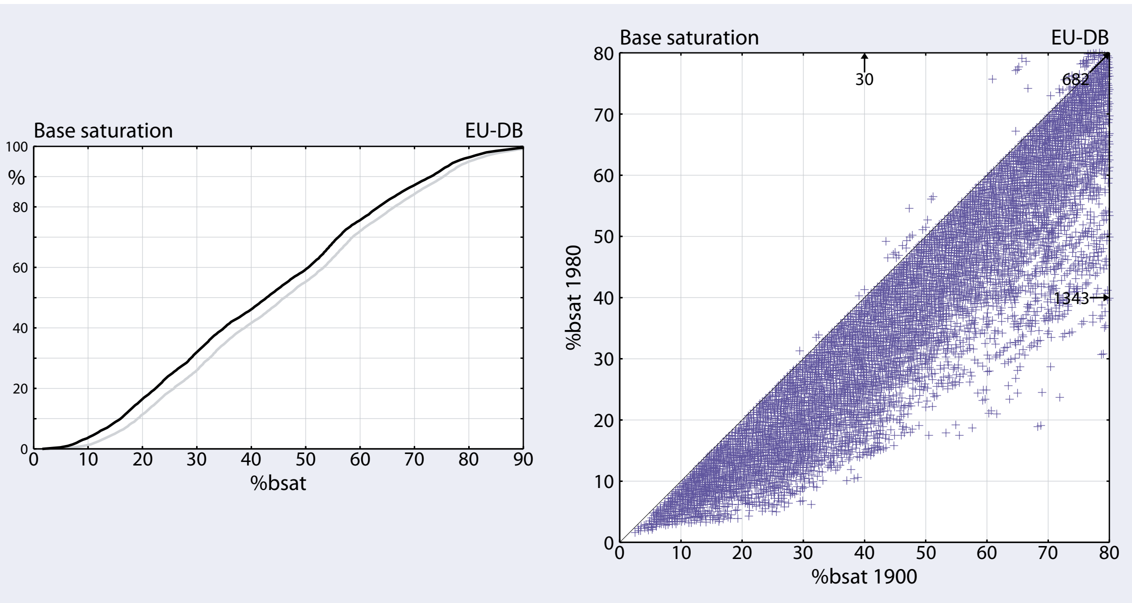

Figure 3-2 Left: CDFs of base saturation (in %) of non-calcareous soils in Europe in 1900 (grey), 1980 (black). Right: Scatter-plot of the same data.

2. What is the distribution of B_{sat} and C/N in 2100 based on 1900 depositions?

Here we explore the values of the critical limits in 2100, assuming a scenario (called 'C00') where the 1900 emissions, and therefore depositions, are kept constant thereafter, to explore the impact of a long-term low deposition. From Figure 3-3 we see that this scenario results in a distribution (yellow line) over Europe of B_{sat} and C/N which lies between the 1900 (grey) and 1980 (black) distributions, rather close to the 1980 distribution. This means that even 1900 depositions cause further acidification in the long run (2100). The C/N ratio in 2100 is hardly distinguishable from the 1980 distribution. One is tempted to conclude that 1900 emissions cause a similar risk of eutrophication in 2100 as emissions from 1880-1980 to the eutrophication in 1980. However, one should remember that a cumulative distribution looses information about the location of the sites; the distribution may be similar, but the site location of eutrophication values may have changed over Europe.

3. What is the distribution of B_{sat} and C/N in 2100 based on 1980 depositions?

This case addresses the acidification and eutrophication effects of 1980 emissions which are kept constant thereafter until 2100 (before 1980 historic depositions are used). This high-emission scenario leads to a distribution of B_{sat} and of C/N (red lines), which are the lowest of all scenarios. 35% of the ecosystems have a B_{sat} lower than 20%. The C/N ratio in 2100 is computed to be 'safe' for about 40% of the ecosystems only (i.e. about 60% has a C/N lower than 25).

4. What is the distribution of B_{sat} and C/N in 2100 based on the Current Legislation scenario?

This scenario is familiar in the context of integrated assessment modelling to support the development of the National Emission Ceilings Directive and the review/revision of the Gothenburg protocol. Current legislation is assumed to be implemented in 2020 and kept constant thereafter. Figure 3-3 (blue line) shows a considerably higher (more to right) distributions of B_{sat} and C/N compared to the previous case of keeping 1980 depositions constant (red line). Now only 20% of the area has a B_{sat} lower than 20%, while less than 60 has a C/N lower than 25.

5. What is the distribution of B_{sat} and C/N in 2100 based on the Maximum Feasible Reduction scenario?

This scenario is also well known in integrated assessment modelling under the LRTAP Convention. It assumes that emissions are abated with the best available techniques in 2020 and we assume that it is kept constant thereafter. The result (Figure 3-3) shows that the distribution of B_{sat} (green line) in 2100 is about the same or lower than the distribution in 1980 caused by historic emissions (black line). It suggests that the application of MFR can almost halt ('on average') further acidification. Of course the MFR scenario yields a distribution of B_{sat} in 2100 that is higher than resulting from C80 and CLE. The distribution of C/N in 2100 under MFR is worse than historic values and the application of the C00 scenario, but better than the result of C80 and CLE.

6. What is the difference between Damage Delay Times (DDT)?

This case compares – between scenarios – the time between the first occurrence of critical load exceedance and the first violation of the chemical criterion (base saturation or C/N limit). In Figure 2-11 this means moving from Case 3 to Case 4. Compared to the earlier analysis no target year is set, because the switch of a site from case 3 to case 4 can take place at any year in the simulation period (up to 2500 in this example). Therefore, for each scenario a cumulative distribution of site specific DDTs can be compiled. Figure 3-4 shows the cumulative distribution of the *differences* of the DDT between the CLE and C80 scenario to reach a B_{sat} of 5%, 10% and 20% coming from a higher (safer) value. The three distributions look surprisingly similar. It can be seen that 20% of the area consists of sites where the time to deteriorate to a B_{sat} of 20, 10 and 5% due to C80 scenario is shorter by about 110, 130 and 150 years, respectively, than under the CLE.

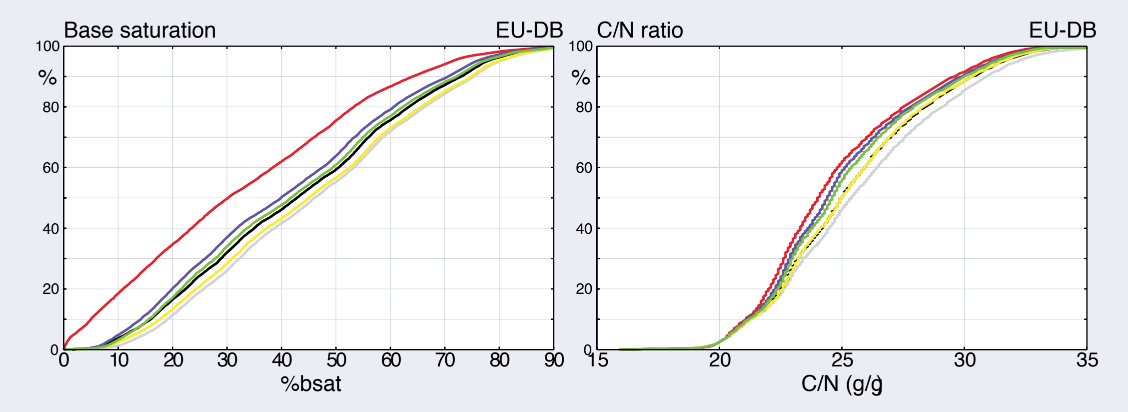

Figure 3-3 Left: CDFs of base saturation (in %) of non-calcareous soils in Europe in 2100 for MFR (green), CLE (blue), for constant 1900 deposition (C00, yellow) and constant 1980 deposition (C80, red). Right: same for C/N ratio (in g/g). Also shown are the 1900 (grey) and 1980 (black) distributions from Figure 3-2.

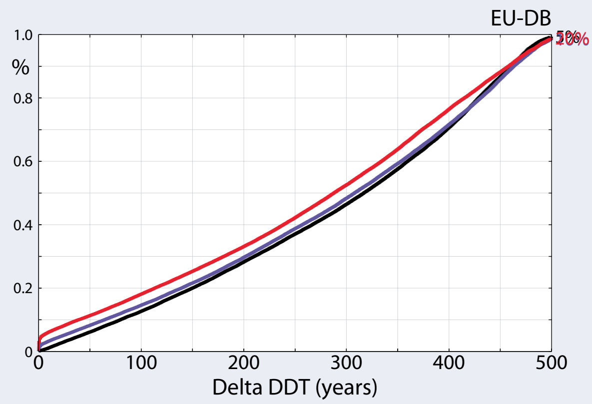

Figure 3-4 CDFs of differences in DDT (CLE minus C80 scenario) to reach a base saturation (from above) of 5% (black), 10% (blue) or 20% (red).

The use of dynamic modelling in the context of integrated assessment models can, apart from addressing certain time periods as seen above, also 'zoom in' from a European to a country perspective. As an example we carry out the above analysis for the Netherlands. Figure 3-5 shows results for Dutch ecosystems in the past (1900 and 1980) and future effects of the same scenarios (C00, C80 , CLE and MFR).

Comparison of Figure 3-5 with Figure 3-3 shows that the Dutch cumulative distributions of B_{sat} are significantly lower than the European distribution, ranging from 0 to 40% even for about 99% of the sites under C00. The distribution of C/N has a range of which the minimum is comparable (about 20) and the maximum about 27 in the Netherlands compared to about 35 in Europe. The case of C80 yields a distribution in 2100 of B_{sat} and of C/N of which more than 95% of the sites are 'not safe' i.e. lower than about 10% and 25 g/g, respectively. The case of MFR yields a cumulative distribution of B_{sat} and C/N in the Netherlands of which about 50% and more than 95%, respectively, are 'not safe'. It is challenging to see that the C/N ratio would require depositions to be at 1900 levels (C00) to protect about 40% of the sites with a C/N ratio greater than 25 g/g. Similarly, the C00 scenario results in 60% of the sites with a B_{sat} of more than 10%.

As an example for a country, the analysis for the Netherlands of the differences in DDT shows that under CLE it takes 400 years more than under C80 for 20% of the ecosystems to reach a B_{sat} of 5% (Figure 3-6). In an additional 200 years under CLE all ecosystems in the Netherlands would sink to this level that, as mentioned above, is not considered a 'safe' level.

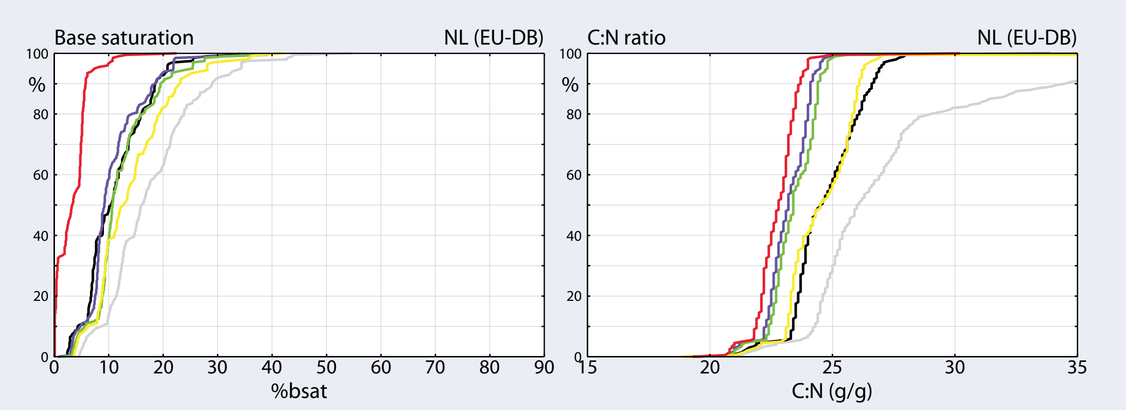

Figure 3-5 Left: CDFs of base saturation (in %) of non-calcareous soils in the Netherlands in 1900 (grey), 1980 (black) and in 2100 for MFR (green), CLE (blue), for constant 1900 deposition (C00, yellow) and constant 1980 deposition (C80, red). Right: same for C/N ratio (in g/g). Note the much lower overall base saturation compared to the European ones (compare Fig. 3-3).

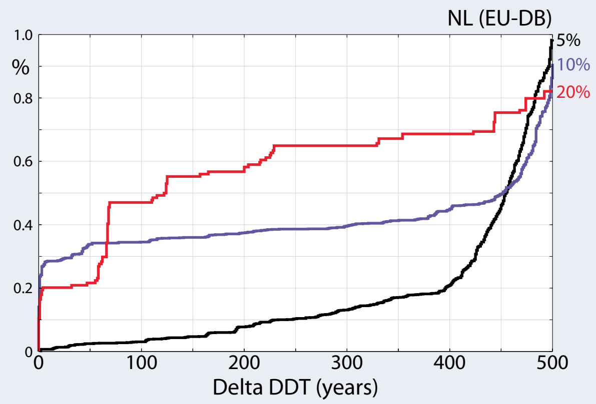

Figure 3-6. CDFs of differences in DDT (CLE minus C80 scenario) for Dutch sites to reach a base saturation (from above) of 5% (black), 10% (blue) or 20% (red) (compare Fig.3-4).

Exploration of the differences in DDT to reach a B_{sat} of 10% shows that 35% of the ecosystems take 50 years more under CLE than under C80. Subsequently, it takes 450 years more under CLE for 50% of the ecosystems with B_{sat} that diminishes to 10%. With a difference of 500 years 80% of the ecosystems have a B_{sat} that diminishes to 10%.

Exploring the downfall of B_{sat} to the 'safe' level of 20% we see a jump from 20% of the ecosystems to about 48% under CLE that occurs about 60 to 70 years later under CLE than under C80. Note that no meaningful spatial observations can be made with respect to the crossing of the cumulative distributions.

Compared to C80, we have gained, broadly spoken, about 300 years delay for the B_{sat} to crash to any of the 3 critical limit values in 50% of the ecosystem area in Europe. Specifically for the Netherlands we gain about 120 years for 50% of the ecosystems-B_{sat} to sink to 20% and 450 years of the median of the ecosystems to have B_{sat} decrease to 10% or 5%.

Moving from alternative 'histories' (after all, 1980 depositions have not and will not occur) to a comparison of possible future scenarios, Figure 3-7 shows the difference in DDT to reach a base saturation of 20% from above with the MFR and the CLE scenarios. It shows that the implementation of MFR instead of CLE would result in a delay by about 200 years 'on average' (actually: median) of reaching the critical value of 20% base saturation.

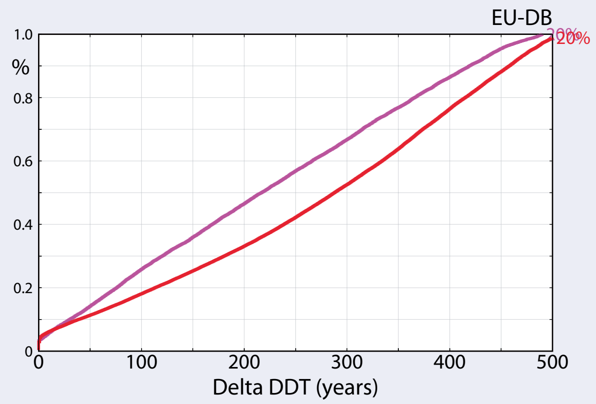

Figure 3-7. CDFs of differences in DDT to reach a base saturation of 20% (from above) for the MFR minus CLE scenarios (pink) compared with the CLE minus C80 scenario (red, see Figure 3-4).

3.4 Interpolating for arbitrary scenarios

The examples discussed above are made for a limited set of pre-specified scenarios, and the VSD model is run for all of them. In practice, however, it is too tedious and time-consuming to run the VSD model (or any other dynamic model) for the about 670,000 sites for every new scenario, especially in an exploratory fashion. Furthermore (and more importantly) it is impossible for the CCE to run the country-specific model(s) for sites for which dynamic modelling has been carried out by NFCs. Thus NFCs were asked in the call for data to provide dynamic modelling output for seven chemical variables (relevant for their connection to impacts) for 14 N and S deposition scenarios arranged as shown in Figure 3-8 (see also Appendix B).

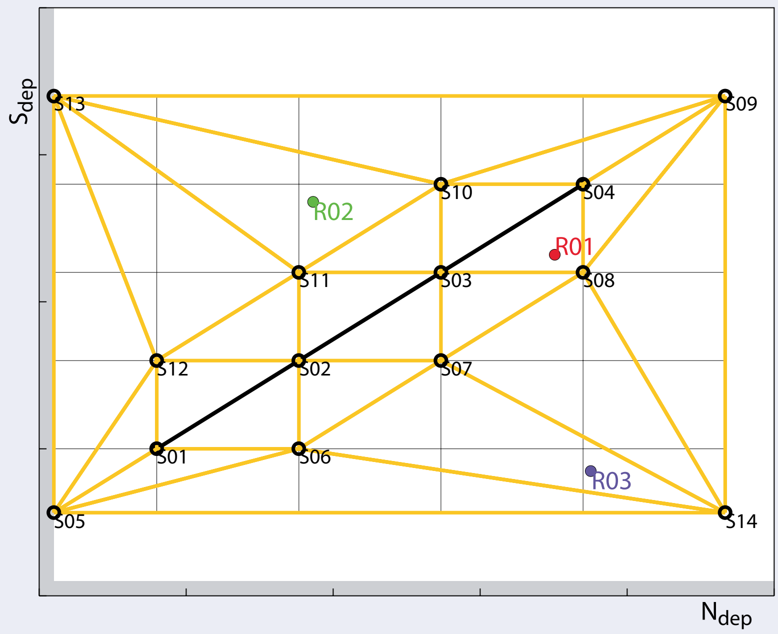

Figure 3-8. N and S deposition scenarios (S01-S14) for which dynamic model output has been provided by NFCs (and EU-DB). S01 is the deposition values of the MFR and S04 those of the CLE scenario in a given grid cell. The yellow tri-/rect-angles show the mesh for interpolation, and scenarios R01-R03 have been run by VSD for EU-DB to test interpolation procedures. The magnitude of this configuration is different for every EMEP grid cell.

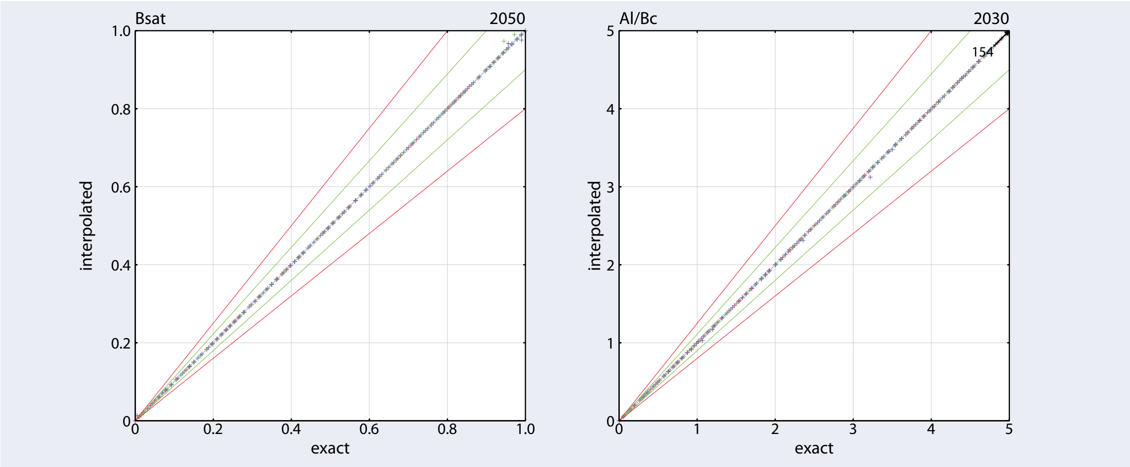

Figure 3-9. Exact versus interpolated values for about 670,000 sites in EU-DB for the scenarios R01 (red), R02 (green) and R03 (blue), as defined in Figure 3-8. The green and red lines define the 10 and 20% deviation from the 1:1 line.

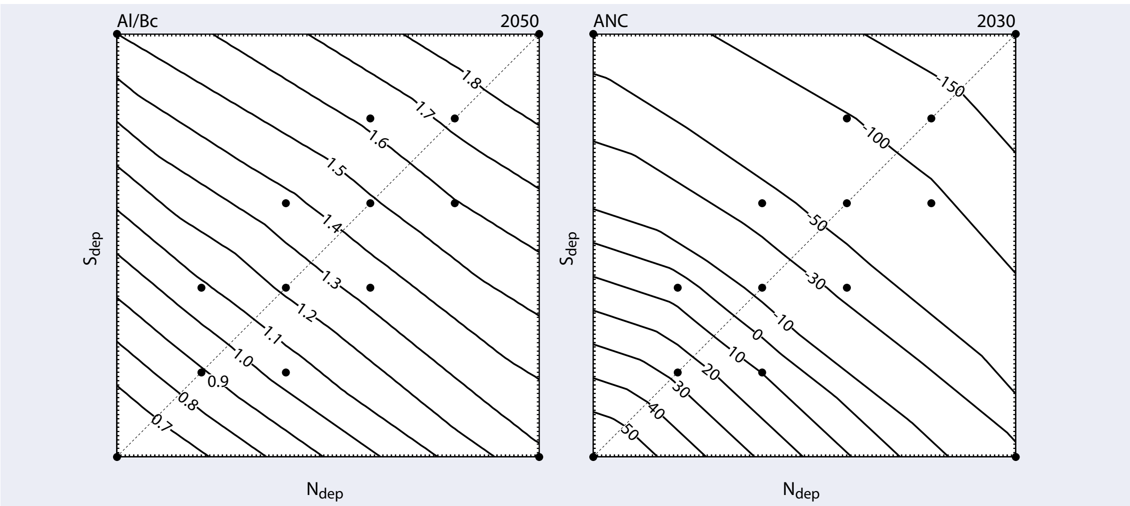

Figure 3-10. Isolines of Al/Bc (mol mol⁻¹) in 2050 and ANC (meq m⁻³) in 2030 at two European sites interpolated from the values given at the 14 scenario points (shown as black dots, see also Figure 3-8).

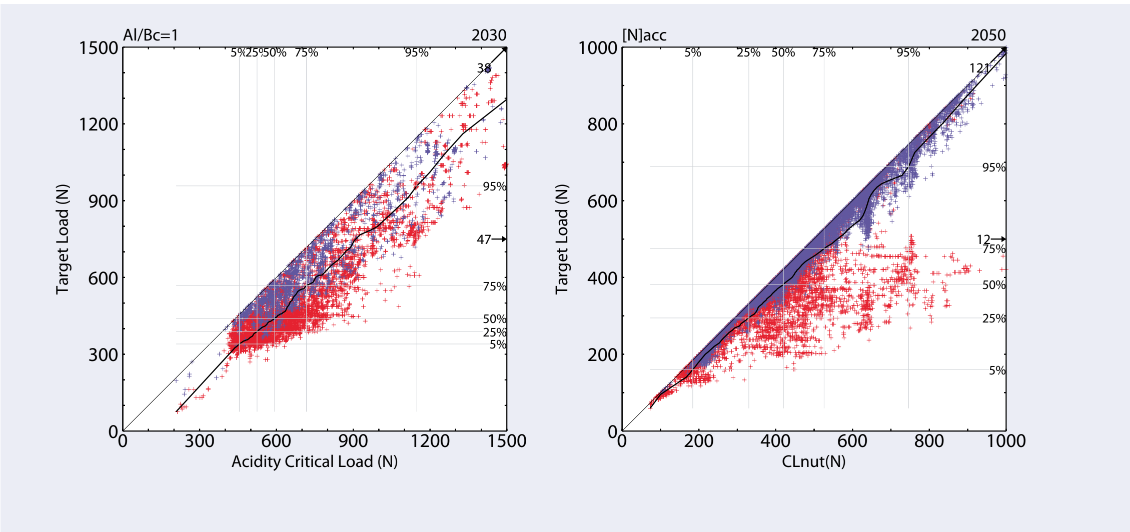

Figure 3-11. N critical loads vs. target loads for acidity in 2030 (left; Al/Bc = 1) and nutrient N in 2050 (right) using EU-DB. Only target loads smaller than critical loads are shown; red dots indicate target loads smaller than the MFR N deposition. For acidity the TLs are computed along the MFR-CLE axis.

The data provided for the 14 scenarios shown in Figure 3-8 can be used to obtain the chemical values for any scenario by interpolation as long as the N and S deposition does not lie outside the domain, i.e. as long as depositions are not too far beyond CLE or MFR. To test the interpolation scheme we used the exact results obtained by VSD for three 'random' scenarios (R01-R03 in Figure 3-8) and compared them to values obtained by interpolation from values at the adjacent points of the 14 'standard' scenarios (Figure 3-9). As can be seen, for base saturation in 2050 and Al/Bc (a derived quantity) in 2030 interpolation results are excellent, almost all points lying on the 1:1 line; and this gives confidence that also other interpolations give reliable results.

3.5 Target loads

The results from the interpolation tests discussed in the previous section encourage the use of interpolation for any given N and S deposition scenario, In Figure 3-10 isolines derived from a field of 100×100 interpolated values for Al/Bc and ANC at two different sites in Europe are shown. They turn out to be fairly smooth, although the geometry of the exact values (at the black dots) can be occasionally discerned. It should be noted that the windows shown in Figure 3-10 are determined by the minimum and maximum deposition at every site.

Selecting the isoline (Figure 3-10) with the critical limit of the chemical value used to calculate the critical load at a site gives the pairs of N and S deposition for which this value is obtained for the given year – in other words, the *target load function* (TLF) for the given target year. Thus, the interpolation procedure outlined above allows the (approximate) determination of any target load for all sites in Europe. As examples, the target load of acidifying N in 2030 (using Al/Bc = 1 as criterion) and nutrient N in 2050 (using the site-specific critical N concentration) have been derived (Figure 3-11). Note, that in case of acidification the N-deposition is computed and shown at which the target load function intersects the line connecting MFR with CLE (see Figure 3-8). In line with the definition of a target load (see Posch et al. 2003, UBA 2004) they are only calculated if they are lower than the critical load (= target load for target year 'infinity') of the site.

To aid interpretation, a 'percentile grid' – i.e. selected percentiles values (5%, 25%, median, 75% and 95%) as thin dashed lines for both variables – is shown together with the 'percentile-percentile curve' connecting all percentiles of the respective distributions of the two variables. This so-called 'P-P-plot' reveals both the correlation structure of the two variables and allows an (approximate) determination of any percentile.

By definition, target loads are smaller (or equal to) critical loads; and in Figure 3-11 those target loads which are smaller than the deposition of the MFR scenario are indicated in red. The percentile-percentile grid in that Figure, e.g., shows that target loads are concentrated in the range 400–600 eq $ha^{-1}a^{-1}$.

The spatial distribution of the target loads (their 5th percentile in every grid cell) shown in Figure 3-11 can be seen in the maps of Figure 3-12 (for acidifying N) and Figure 3-13 (for nutrient N).

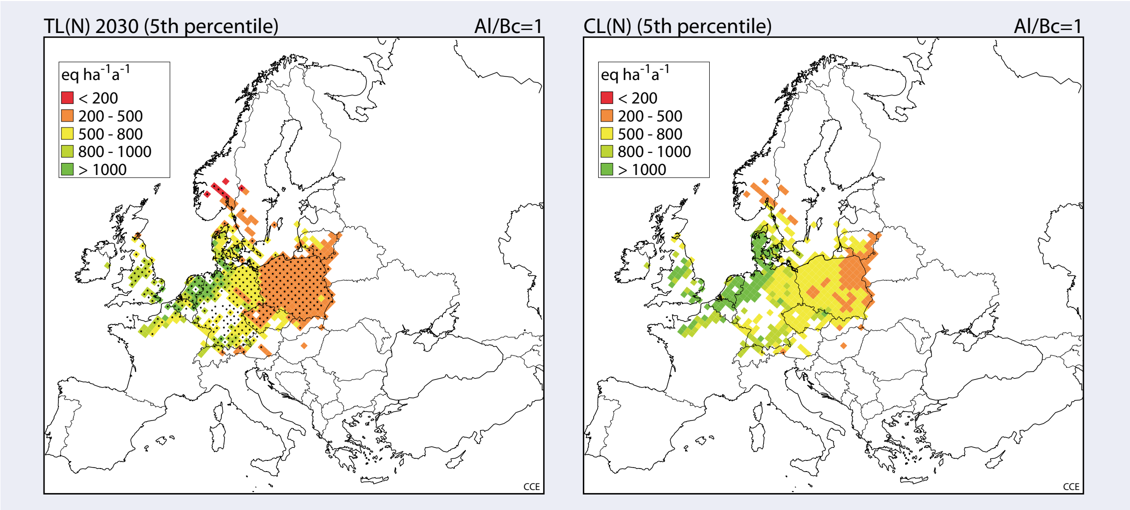

Figure 3-12. Spatial distribution of 2030 N-acidity target loads (left) and critical loads (right) from the left panel of Figure 3-11 (5th percentile). A black dot in a grid cell indicates that there are (also) ecosystems with (technically) infeasible target loads.

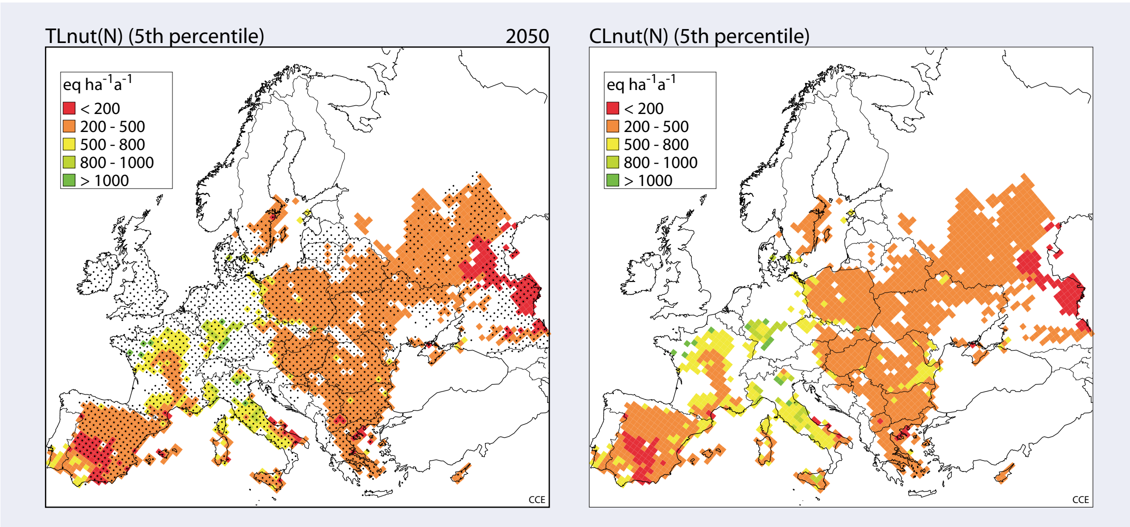

Figure 3-13. Spatial distribution of 2050 nutrient N target loads (left) and critical loads (right) from the right panel of Figure 3-11 (5th percentiles). A black dot in a grid cell indicates that there are (also) ecosystems with (technically) infeasible target loads.

A target load for a given target year need not exist. It can happen that even the reduction to zero deposition is not enough to attain the desired chemical status by the target year. In this case we say the target load is infeasible; and in Figures 3-12 and 3-13 an EMEP grid cell is marked with a black dot, if there exist sites with infeasible target loads. This shows that, e.g., in central-western Europe there are large areas where it is impossible to each non-violation of the criterion by 2050 even with the most radical emission reductions.

Many more analyses can be carried out with the wealth of information available from dynamic models. One of the main tasks remaining is to implement the various options for the dynamic modelling data submitted by NFCs to be able to carry out ex-post analyses in the context of integrated assessment (see Chapter 1).

3.6 Conclusions and recommendations

While in theory the precautionary principle is meaningful when emissions are abated such that exceedances vanish, the reality is that emission abatement policy scenarios do not attain this aim simultaneously for all ecosystems in Europe.

The use of dynamic modelling in integrated assessment can help to address questions such as "what is the long-term effect of a change – between emission reduction scenarios - of exceedances ?" The answer to this kind of questions is addressed in this chapter in an illustrative manner by inspecting the temporal development of abiotic variables in the soils. For the comparison of acidification effects between scenarios scenario's we looked for example at base saturation and for eutrophication we inspected the C/N ratio.

We used the VSD model on the CCE background database for the illustrative comparison of the effects of the Current Legislation (CLE) with the Maximum Feasible Reduction (MFR) scenario. Under MFR the area with a "safe" base saturation (we assume between about 5-20%) or a "safe" C/N ratio (about 25) increases by about 2-5% compared to CLE on a European scale. Implementing MFR would, on average, result in a delay by about 200 years to reach the critical value of 20%. This kind of analysis can also be zoomed in to ecosystems within a country.

Emission reduction goals are often the starting point in integrated assessment for the design of a scenario. Thus, computed depositions follow from national emissions. The other way around, i.e. compute required emissions from deposition goals, has also been applied in the so-called "optimization mode" of models such as RAINS. Dynamic Modelling can help in making the choice of deposition goals by addressing the question "What is the required deposition to reach recovery in a future target year". We successfully tested a method to derive these target loads for arbitrary scenarios. Illustrations of target load maps to reach acidification and eutrophication recovery goals in 2030 and 2050 respectively are included in this chapter. However, VSD computations also show that in many areas in Europe even a complete reduction of N and S depositions will not lead to recovery in 2050.

Further work is needed to address other dynamic modelling indicators including biological endpoints for use in integrated assessment.

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4 The Derivation of Dose-response Relationships between N Load, N Exceedance and Plant Species Richness for EUNIS Habitat Classes

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4.1 Introduction

The emissions of ammonia (NH_3) and nitrogen oxide (NO_x) strongly increased during the second half of the 20th century. Ammonia is volatilized from intensive agricultural systems, such as dairy farming and intensive animal husbandry, whereas nitrogen oxides originate mainly from the burning of fossil fuel by traffic and industry. Because of short- and long-range transport of these nitrogenous compounds, atmospheric nitrogen (N) deposition has clearly increased in many natural and semi-natural ecosystems across the world. Areas with high atmospheric nitrogen deposition, nowadays, are central and western Europe, eastern USA and, since the 1990s, East Asia (e.g. Galloway & Cowling 2002).

The availability of nutrients is one of the most important abiotic factors which determine plant species composition within ecosystems. Nitrogen is the limiting nutrient for plant growth in many natural and semi-natural ecosystems, especially in oligotrophic and mesotrophic habitats. Most of the plant species living under such nutrient-poor conditions have adapted and can only survive or compete successfully on soils with low nitrogen availability (e.g. Tamm 1991; Aerts & Chapin 2000). The series of events which occur when N input increases in an area with originally low background deposition rates, are highly complex. Many ecological processes interact and operate at different temporal and spatial scales. As a consequence, high variations in sensitivity to atmospheric nitrogen deposition have been observed between the different natural and semi-natural ecosystems. Despite this diversity, the most obvious effects of increased N deposition are significant changes in nitrocycling, vegetation composition and biodiversity (see Bobbink *et al.* 1998 and Bobbink et al. 2002 for more details)

Within the UN/ECE Convention on Long-range Transboundary Air Pollution, empirical procedures have been developed to set critical loads for atmospheric N deposition. Based on observed changes in the structure and function of ecosystems, as reported in a range of publications, empirical N critical loads were established for European natural and semi-natural ecosystems in 1992, 1996 and – extensively updated – in 2002 (Achermann & Bobbink 2003). It is clear that future exceedance of the set empirical N critical loads should be prevented, to protect the structure and function of natural and semi-natural ecosystems. At present, the causal relationship between the level of such an exceedance and biodiversity is hardly quantified.

The aim of this study is to reveal the empirical relationship between the amount of N load and the exceedance of the N critical loads in European ecosystems (EUNIS classes), and its effect on the loss of plant species. Because data are mainly available for plant life only, this chapter is restricted to changes in *plant species richness* (number) with respect to N deposition.

4.2 Methods

The basis for this study was the extended database of publications, prepared for the empirical N critical load studies in 1992, 1996 and 2003. International bibliographic systems (BIOSIS; SCI-Web of Science) were used to obtain the most recent studies that were not covered before (November 2002 – August 2008). Peer-reviewed publications, book chapters, nationally published papers and “grey” reports of institutes (if available) were incorporated. Studies providing insights into ecosystem reactions to increases in nitrogen loads, have been conducted for a variety of reasons. This has resulted in many different experimental designs. In this study, the outcomes of *field addition experiments* were only used if they were in accordance with the following criteria:

- independent, sole nitrogen treatments;
- no applications of additional nutrients, such as P, K or lime;
- below 150 kg N ha^{-1} a^{-1} experimental addition;
- experimental period longer than 2 years (thus incorporating at least 3 growing seasons);
- located in Europe (west of 400 longitude);
- containing data on plant species richness (number of plant species per plot)

Number of plant species per control plot or N-treated plot were either directly obtained from the papers, or calculated from presented full species tables or extrapolated from given figures. Per used study, data were averaged per N treatment per investigated site (varying between 1 – n per publication) to avoid pseudo-replication and overrepresentation. The used experimental N loads (in kg N ha^{-1} a^{-1}) were also noted. For each field addition experiment, the background deposition at the location and date of the experiment was derived from historical deposition trends, described in Appendix A. A map of the locations of the field addition experiments, including their deposition ranges, has been plotted in Figure 4-1.

To prevent the effects of differential plots sizes, used in the studies, of a specific EUNIS habitat type upon species numbers and to make the outcome independent of plot size, the species richness ratio (per N treatment) has been calculated, thus:

Species richness ratio: S_N / S_C
with: S_N = species number in N-treated situation
S_C = species number in control situation

If this ratio is 1, the number of species in the N-added situation equals that of the control situation; above 1 the number of species after N addition are higher than in the control situation. If the value of this ratio is below 1, the species number after N enrichment are lower than in the control situation. This species richness ratio is relatively simple to interpret: a ratio of 0.75 indicates 25 % reduction in species number after N treatment, compared with the control situation.

The available data on the species richness ratio were ordered per level 1 and level 2 EUNIS European classes (http://eunis.eea.europa.eu/habitats) and analysed with 1x1 regression methods (including line fit procedures) against the total N load (experimental N addition plus background deposition). Furthermore, their relationship with N exceedances, that is, the total N load minus the minimum (Exc-min) or maximum value (Exc-max) of the empirical N critical load for the specific habitat class (Achermann & Bobbink 2003), was calculated.

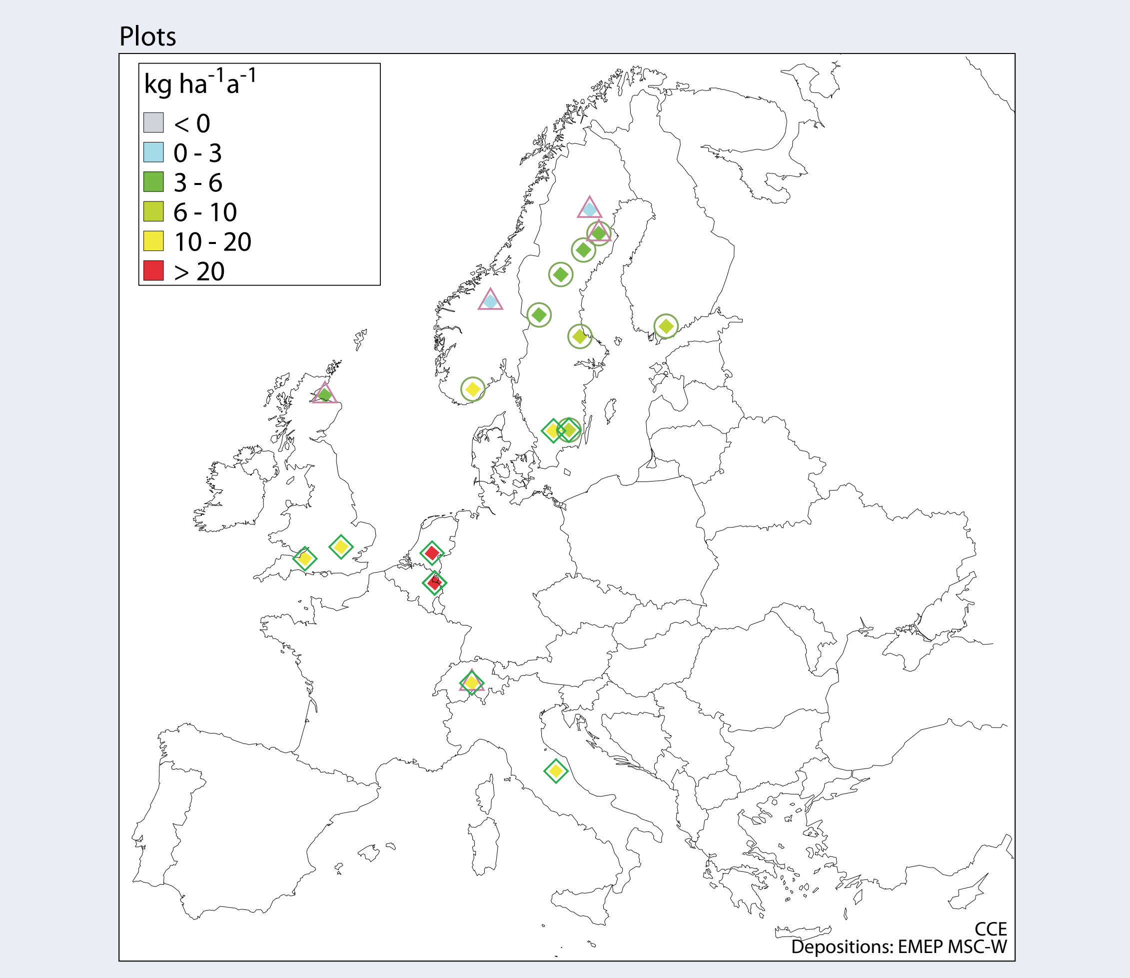

Figure 4-1 Deposition ranges in EMEP grid cells in which the plots are located. The dark green circles indicate the forests, green diamond grasslands and red triangles the arctic-alpine scrubs. An EMEP grid cell can contain more than one plot.

The changes in species composition of the understorey vegetation in boreal forests could be analysed in detail, because of the availability of a full digital data set of long-term N fertilisation experiments in Swedish forests (kindly provided by Dr Han W. van Dobben, Alterra. Wageningen, The Netherlands). Besides plot species richness, the similarity index of Sorensen could be calculated with MVSP 3.0, based on presence and absence of species in the experimental plots in comparison with the respective control situation (7 locations across boreal Sweden, n = 12). This index quantifies the (dis)similarity between two vegetation samples in a simple scale from 0 to 1, with 1 meaning full similarity (all species in common), and 0 meaning no similarity (no species in common). This approach was used because it was impossible to classify all the observed plant species as characteristic or non-characteristic species for the studied habitat.

4.3 Results

The outcome of the nutrient-addition studies are presented per level 1 and 2 EUNIS habitat class. Unfortunately, the number of data per EUNIS habitat class was only sufficient for the following three habitat classes: Grasslands and tall forbs habitats (E), Arctic, alpine and subalpine scrub habitats (F2) and Coniferous boreal woodland (G3.A-C). For all other classes, the data set was much too restricted to perform a regression analysis (below n = 5 per habitat class).

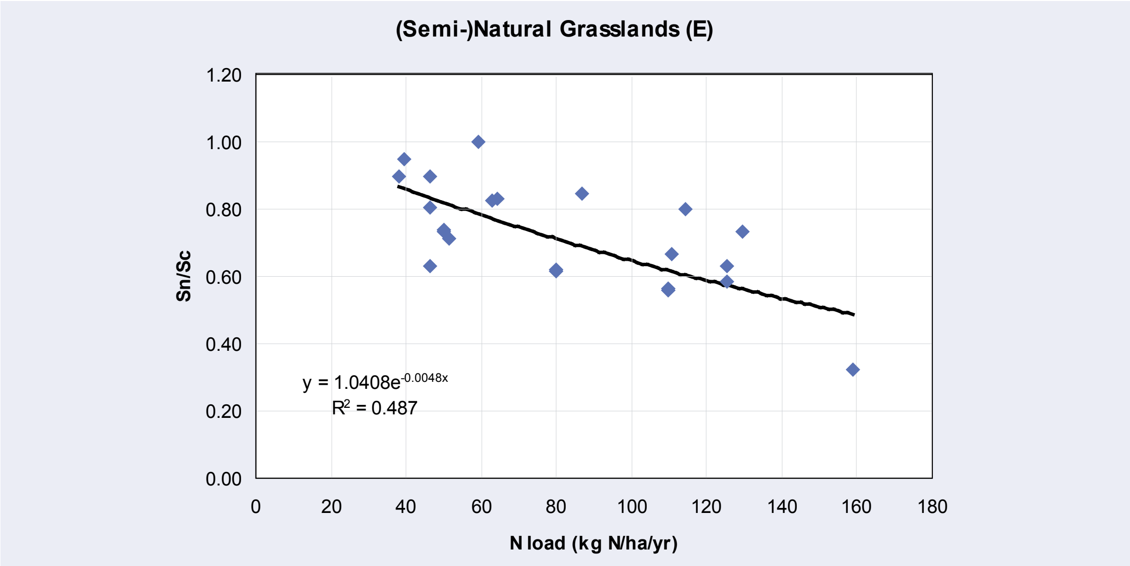

Figure 4-2 Relation between the species richness ratio (S_N/S_C) and total N load (N addition plus background deposition) in grassland habitats (6 countries; n=22; p<0.001).

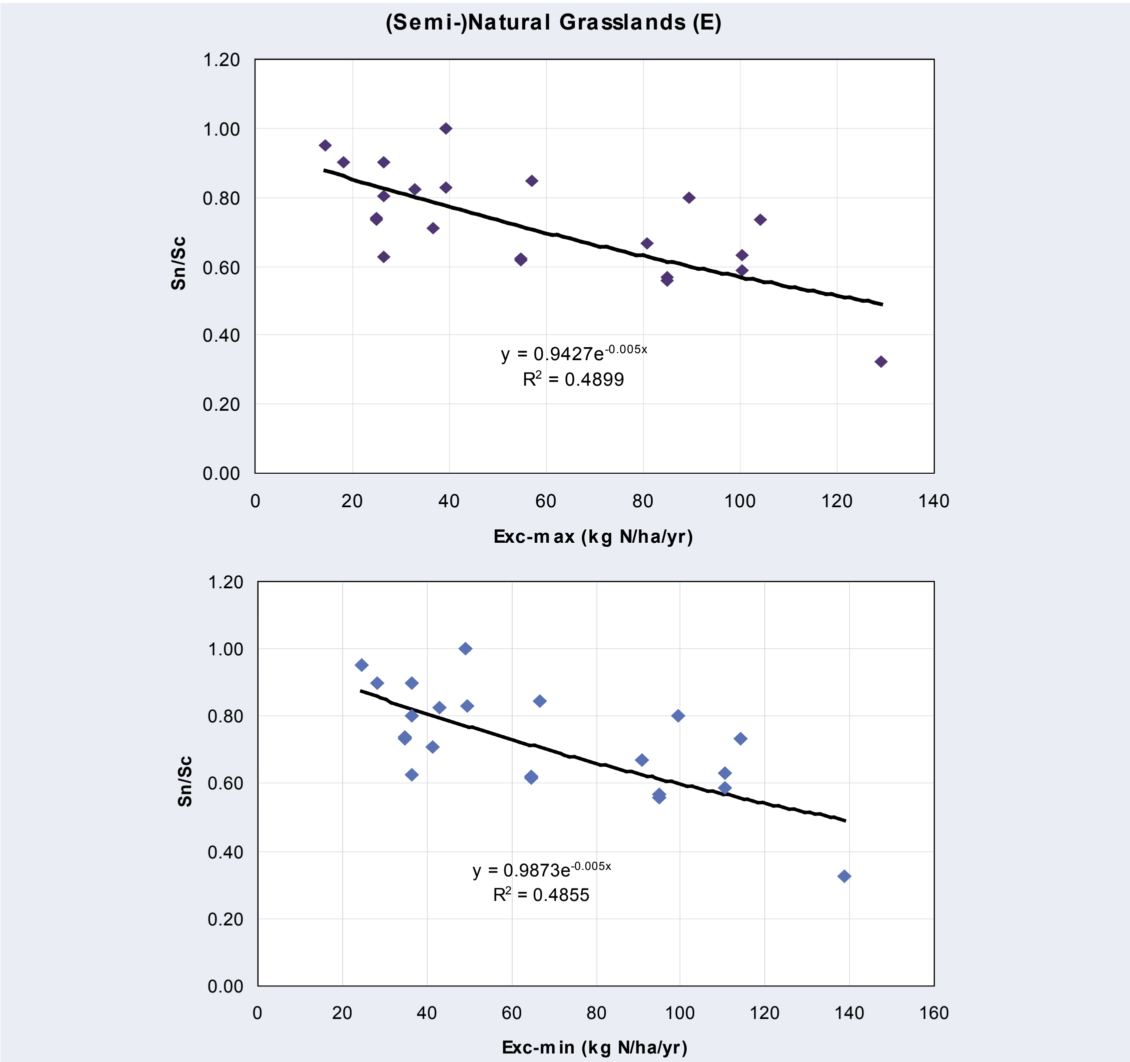

Figure 4-3 Relationship between the species richness ratio (SN/SC) and N exceedance (total load addition minus minimum (top) or maximum (bottom) values of critical load) in grassland habitats (6 countries; n=22; p<0.001).

Grasslands and tall forbs habitats (E)

The effects of nutrients on (semi-)natural species-rich grasslands have been studied in several European countries. The literature study on the effects of sole N treatments on plant species richness revealed the largest regression data set of this chapter (n = 22). Results included studies under both dry and wet conditions, or with very different soil pH (acid – calcareous) (E1, E2, E3 & E4) across Europe (6 countries) (Lüdi 1959; Bobbink 1991; Tallowin et al. 1994; Berlin 1998; Jacquemin et al. 2003; Crawley et al. 2005; Bonanomi et al. 2006; Beltman et al. 2007). A significant negative relationship (negative exponential fit) between the species richness ratio and the total N load was found for these grasslands (Figure 4-2). In addition, similar negative relationships were obtained for the N exceedances, both based upon minimum and maximum values of the empirical N critical load (Figure 4-3).

Arctic, alpine and subalpine habitats (F2)

In general, plant productivity of arctic, alpine and subalpine (scrub) ecosystems is highly influenced by N, within the specific climatic constraints. Because of the remoteness of most of these ecosystems and, thus, with (still) low atmospheric N inputs, possible impacts of N additions on the vegetation has only recently received attention. Fortunately, results from some relatively long-term studies on this habitat class (F2; n = 11, 4 countries) could be used to obtain a first impression of the relation between the level of N exceedance and the species richness ratio in arctic-alpine vegetation (Lüdi 1959; Gordon et al. 2000, Nilsson et al. 2002; Fremstad et al. 2005; Britton and Fisher 2007). These results showed a significant negative linear relationship between the species richness ratio and total N load ($p < 0.05$) and between the species richness ration and N exceedances ($p < 0.05$) (Figure 4-4). For example, these figures suggest a 20% reduction in plant species richness at an exceedance maximum of 25 kg N $ha^{-1}a^{-1}$. However, the data set is rather restricted, at this moment, and should be extended in the near future to obtain a more reliable regression.

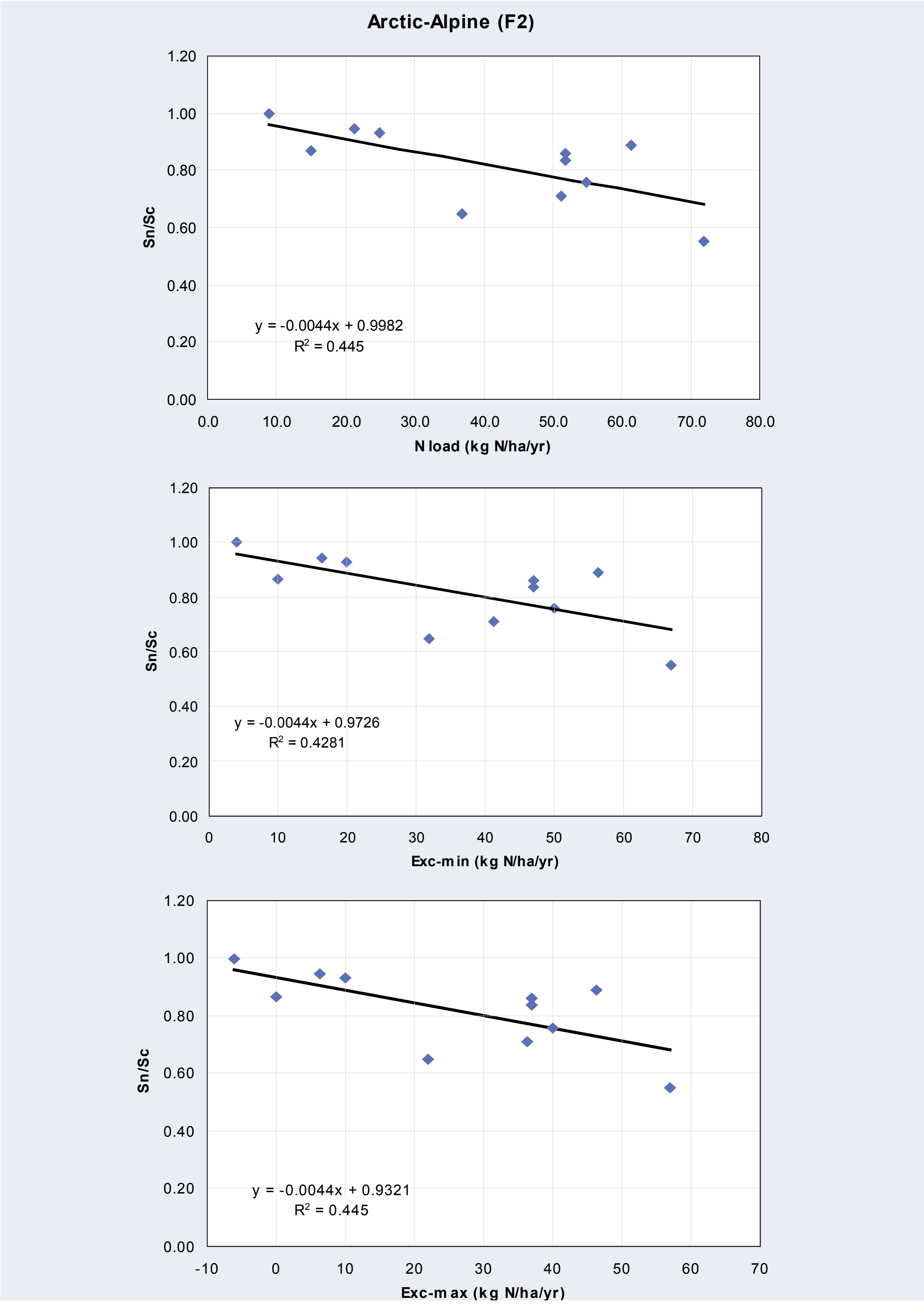

Figure 4-4 Relationship between the species richness ratio (SN/SC) and total N load with N exceedance (total load addition minus minimum or maximum values of critical load) in arctic and (sub)alpine scrub habitats (4 countries; n=11; all lines p<0.05).

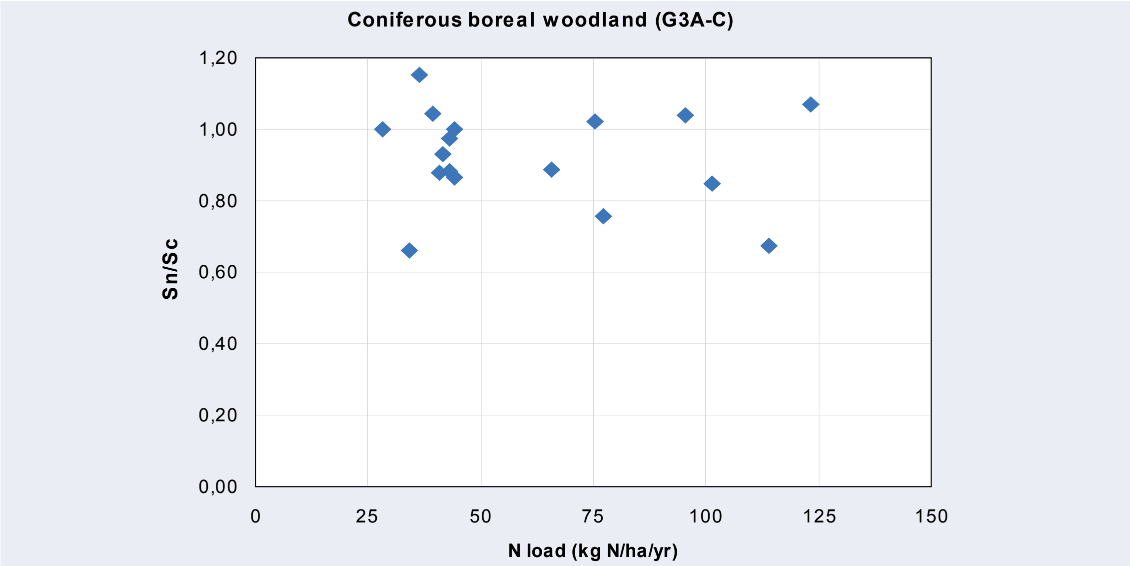

Figure 4-5 Relationship between the species richness ratio (SN/SC) and N exceedance (N addition minus mean critical load) in boreal coniferous woodlands (3 countries; n= 18; not significant).

Boreal coniferous woodlands (G3 A-C)

Boreal coniferous woodlands make up the largest forest zone of European vegetation. Numerous studies have been performed with respect to the influence of acidic deposition. In the last decades, the number of studies on the impacts of N deposition increased but, mostly, the consequences for tree growth and soil have been determined. In the last 2 decades, the impacts on the understorey vegetation received more attention, but the number of published studies which provides the number of species per plot or full vegetation are, however, scarce and fully restricted to the three Scandinavian countries (Van Dobben 1993; Kellner 1993; Mäkipää 1998; Van Dobben et al. 1999; Shrindo & Okland 2002). Because of the possibility of analysing the original data set by Van Dobben (n = 12), the number of data points became rather large (n = 18). The results for this EUNIS habitat type clearly showed that the species richness ratio was not influenced, at all, by the total N load exceedance (Figure 4-5), although many studies reported a sharp decline in lichen diversity, a change in bryophyte composition and lower abundance of typical dwarf shrubs. These results suggest that the decline in typical species is counterbalanced by increases in invasive or non-characteristic species from more nutrient-rich habitats.

The data, previously shown, demonstrates that plant species richness as an indicator is too restricted for showing the N effects on understorey vegetation in these boreal woodlands. However, the detailed comparison of the species composition of the understorey in Swedish boreal woodlands (7 locations, n = 12) showed that the similarity of the N treated vegetation significantly decreased (negative exponential curve) with the total load of N exceedance (Figure 4-5).

In addition, a clearly negative relationship has been found for both N exceedances (Fig. 4-7). These results indicate that ca 30 % of the species have changed at an exceedance of ca 30 kg N ha^{-1} a^{-1}, whereas at an exceedance of above 80 kg N ha^{-1} a^{-1}, 45 to 50 % of the species composition is different. Since the species richness itself did not change (Figure 4-5), this means that the original species were replaced by others!

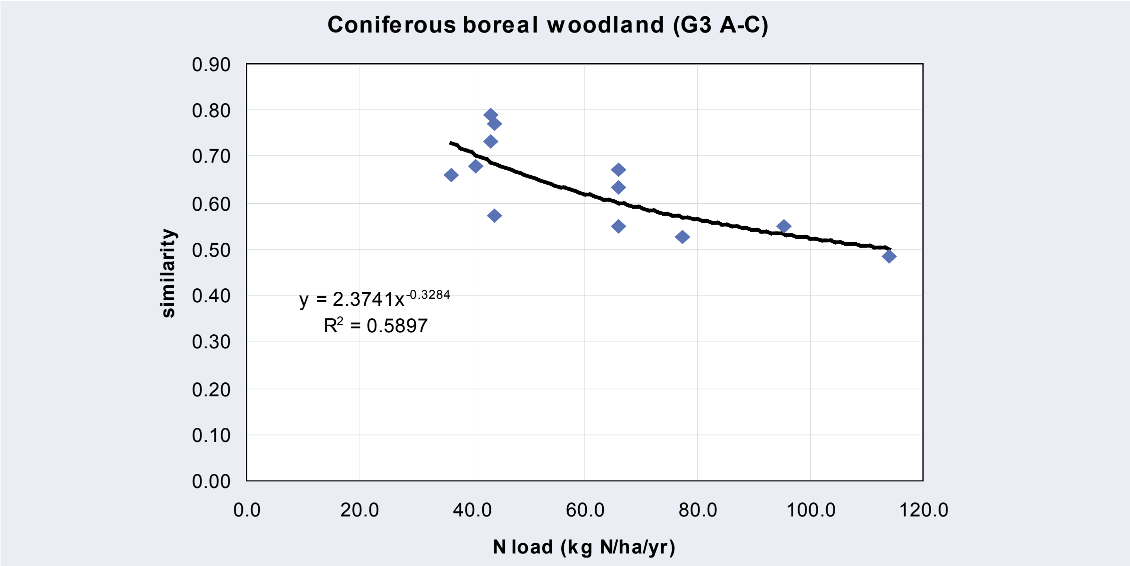

Figure 4-6 Sorensen's similarity index of N treated understorey vegetation, compared to the control vegetation in Swedish boreal forests (7 locations; (p <0.01; n=12) against the total N load.

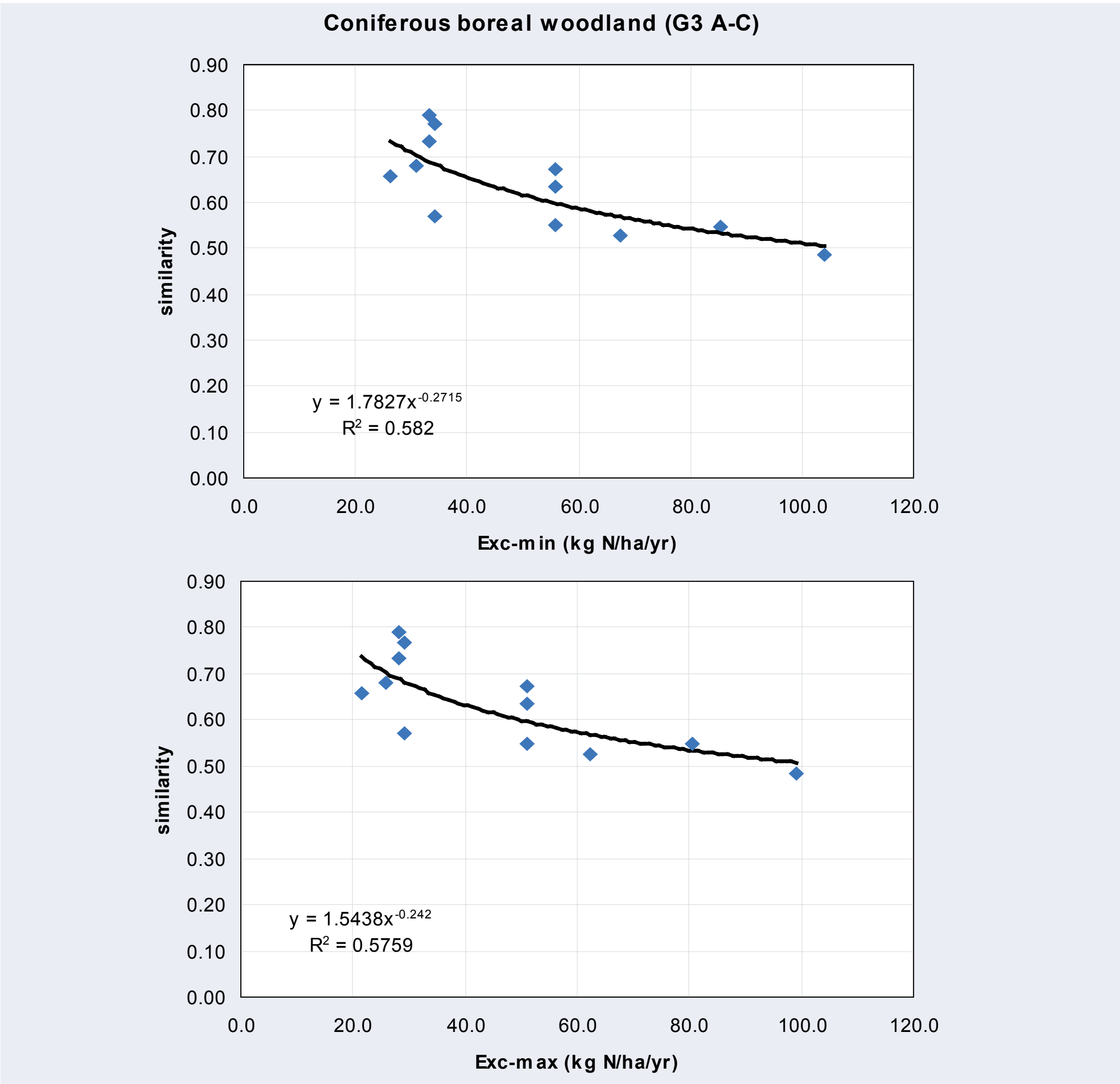

Figure 4-7 Sorensen's similarity index of N treated understorey vegetation, compared to the control vegetation in Swedish boreal forests (7 locations; (both p <0.01; n=12) against the N exceedance (N addition minus minimum (top) or maximum value of the critical load).

4.4 Conclusions

Within the UN/ECE Convention on Long-range Transboundary Air Pollution, empirical procedures were developed to set critical loads for atmospheric N deposition. Based on observed changes in the structure and function of ecosystems, empirical N critical loads have been established for European natural and semi-natural ecosystems (e.g. Achermann & Bobbink 2003). It seems clear that the exceedance of the set empirical N critical loads should be prevented to protect the structure and function of natural and semi-natural ecosystems. At present, the causal relationship between the level of exceedance of these N critical loads and the biodiversity in these systems, is hardly quantified. The central aim of this inventory study was to reveal this relationship for major EUNIS habitat classes. As expected, this study had to be restricted to plant species richness, because experimentally quantified data for fauna is extremely rare.

The effects of experimentally-enhanced plant species richness were expressed as the species richness ratio, that is, the ratio between the number of plants in the N treated situation and in the control situation (Sn/Sc). Any significant, negative relationship between the exceedance of the empirical N loads and the species richness ratio, was found only for grassland habitats (E) and arctic/alpine scrubland habitats (F2). Remarkably, the quantified relationships between N exceedances and species richness across a range of grasslands in Europe, strongly reflected the observations in temperate grasslands of the USA (e.g Haddad et al. 2000). As in grassland habitats, the number of studies in boreal coniferous woodlands (G 3 A-C) was large enough to indicate a quantitative relationship between the species richness ratio and the exceedance. The species richness ratio was clearly not related to increasing N, in this group of habitats. However, as a result of the availability of a complete species data set, it became obvious – after comparing the similarity in species composition of the N treated situation to the control vegetation – that a significant negative relationship could be demonstrated. This clearly suggests that typical “nutrient-poor” species may be replaced by “invasive or N-loving” species, without changing the overall species richness. It is suggested that the number of characteristic or nutrient-poor plant species is a more sensitive indicator of plant diversity in a habitat class, with respect to environmental N stresses. However, it was not possible to analyse most publications in this way; this clearly would need the availability and/or digitalisation of the original data sets. Due to this fact, it is rather likely that the slope of the present curves would be steeper if data on these aspects of species diversity were available. Furthermore, it became obvious that for most EUNIS habitat classes hardly any or no data were available. Finally, for some (relatively) well-studied systems (temperate heathlands and boglands) the effects on species richness were hardly published, and could even be of restricted use, because of the species-poor characteristics of these habitats.

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5 Relation between Critical Load Exceedance and Loss of Protected Species

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5.1 Introduction

Nitrogen is an important nutrient for life on earth. At the same time, excess of nitrogen is thought to be one of the major threats to global biodiversity. Although nitrogen is the element that is most abundantly present in the world's atmosphere, only very few organisms – such as nitrogen-fixing bacteria – can utilise the gaseous nitrogen, directly. Most plants can absorb nitrogen only in the form of nitrites (NO_2^-), nitrates (NO_3^-), or ammonium (NH_4^+) iones from a soil solution. Animals obtain their nitrogen through amino acids, produced by plants. Biologically available nitrogen is often the growth-limiting nutrient in temperate terrestrial ecosystems (Vitousek & Howarth, 1991; Bobbink et al., 2008). Experiments have shown that nitrogen addition leads to enhanced plant growth and increased vegetation productivity in many ecosystems, thereby changing the original vegetation structure and plant species composition. Most of the plant species from oligotrophic and mesotrophic habitats are adapted to nutrient-poor conditions and are outcompeted by high and dense vegetation growth when the availability of nitrogen increases (e.g. Tamm, 1991; Bobbink et al., 1998; Aerts & Chapin, 2000). Evidence suggests that increasing nitrogen availability often causes an overall decline in plant species diversity (Tilman, 1987; Bobbink et al., 1998) even at long-term low nitrogen input (Clark & Tilman, 2008). However, at low nitrogen input, an increase in plant species diversity has (also) been observed due to an increase in the number of nitrophilic species (Emmett, 2007). In boreal forests, species' numbers seem unaffected by nitrogen enrichment, despite a drastic change in their composition. Here, also, an increase in the number of common nitrophilic species compensates the decline in a number of other species (Bobbink, 2004). Changes in vegetation structure and composition might also affect other taxonomic groups. A decrease in the density of food plants will have affect on herbivore density. Changes might also be more indirect. Butterfly larvae, which feed on green plants, for example, depend on a warm microclimate for their development. Such microclimates are present within short vegetations (Stoutjesdijk & Barkman, 1992). An increase in vegetation height, caused by high nitrogen deposition, may provide a cooler microclimate, which negatively affects larval development (WallisdeVries & Van Swaay, 2006).

Under natural conditions, biologically available nitrogen enters terrestrial ecosystems through nitrogen fixing bacteria, lightning, the weathering of inorganic soils and the decomposition of organic matter. Fossil fuel combustion and modern agriculture have both increased atmospheric nitrogen deposition on natural areas from 50-200 $mol.ha^{-1}.yr^{-1}$, to 500 $mol.ha^{-1}.yr^{-1}$ in central and eastern USA, to 1200 $mol.ha^{-1}.yr^{-1}$ in central Europe, and up to 3000 $mol.ha^{-1}.yr^{-1}$ in parts of the Netherlands (e.g. Galloway et al., 2004, 2008). Atmospheric deposition has not only become an important source of biologically available nitrogen within natural areas, but is sometimes even the dominant source (Galloway et al. 2008). On a global scale, current nitrogen emission scenarios project an increase in deposition rates for most regions (Dentener et al. 2006), which is causing concern about significant impacts on global biodiversity (Vitousek et al. 1997, Sala et al. 2000, Phoenix et al. 2006). Under the Convention on Biological Diversity, the increase in nitrogen deposition is considered a major threat to biodiversity (UNEP, 2005). Information on exceedance of critical load depositions on natural ecosystems is used for defining emission reduction targets in international pollution abatement policy (e.g. EC, 2001). These critical loads, being 'quantitative estimates of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson and Grennfelt, 1988), are based on experiments of nitrogen addition and observational research along geographic nitrogen deposition gradients (Bobbink et al., 2008) or ecosystem modelling (De Vries et al., 2008). In Europe, the critical load exceedance by current deposition levels is now being used as an indicator for monitoring the threats to biodiversity (EEA, 2007). However, the indirect link to biodiversity loss itself is seen as a drawback of this indicator (EEA, 2007).

To identify the dose-effect functions, which could be used to relate critical load exceedances to biodiversity loss, three major methods can be applied. First, it is possible to derive dose-effect functions from nitrogen addition experiments, as has been shown in Chapter 4 and by Bobbink et al., 2008. Bobbink found a clear negative-log relationship between the exceedance of empirical critical loads and the plant species richness in a set of experiments of nitrogen addition across various ecosystems. Another way of deriving relationships between critical load exceedance and species occurrence is by generalising the output of the various dynamic ecosystem models, as described in De Vries et al., 2008. In this chapter, we describe a third possible way of using regression models based on field observations of species distributions and detailed maps of nitrogen deposition. The nitrogen deposition in the Netherlands is relatively high, which provides a good opportunity for finding effects of deposition on species occurence if present (i.e. 70% of the natural area experiences critical load exceedance). The variation in deposition level, which is low in the northern and western part and high in the south-eastern part of the country, further increases the chance of finding a relation between critical load exceedance and number of species. Based on field data, we show that in several habitats a high critical load exceedance corresponds with a situation where fewer protected and/or threatened species of plants, butterflies and birds are present. We argue that this method, in combination with information from experiments and dynamic ecosystem models, yields functions which might help to translate a pressure-based indicator, such as critical load exceedance, into more biodiversity-relevant indicator.

5.2 Materials and methods

Data on critical load exceedance

Maps of atmospheric nitrogen deposition in the Netherlands were calculated by using the OPS model (Operational Priority Substances), a Lagrangian atmospheric dispersion model (Van Jaarsveld, 1995). These calculations were based on the national emissions database from 2002, which includes estimates of the ammonia emission from agricultural sources, on a 250 × 250 m scale. Depositions from other sources were calculated at a courser resolution. The total amount of nitrogen deposition was calculated as being the total of ammonia and nitrogen oxide deposition, from both national and international sources.

The deposition map was combined with a habitat map showing the locations of dry and wet heathlands, bogs, dry forests on poor sandy soils, forests on rich sandy soils and nutrient-poor wet grasslands. These habitat types were selected because of their high sensitivity to nitrogen (indicated by their relatively low critical load values) and their importance to nature conservation. All of the selected habitat types are (partly) protected under the EU Habitats Directive. Next, critical load exceedances were calculated as being the difference between deposition and critical load (Table 5.1).

Table 5.1 Description of the variation in critical load exceedances (mol.ha^{-1}.yr^{-1}) in grid cells with forests on poor sandy soils, forests on rich sandy soils, dry and wet heathlands and nutrient-poor wet grasslands.

	Median critical load exceedance	5 percentile	95% percentile
Forests on poor sandy soils	1680	730	2600
Forests on rich sandy soils	1300	520	1940
Dry heathlands	1030	720	1730
Wet heathlands	430	210	1550
Bogs	1330	1150	2200
Nutrient-poor wet grasslands	730	0 (no exceedance)	1730

Critical loads were set to 1300 for dry forests on poor sandy soils, 1400 for forests on rich sandy soils, 1100 for dry heathlands, 1400 for wet heathlands and 400 for bogs. For oligotrophic grasslands, critical loads were set depending on soil type. Critical load values were based on the European critical load ranges, published by Bobbink et al. (2002), and the Dutch modelled critical loads (Van Dobben et al., 2006). Negative values (i.e. locations with no exceedances) were set to zero.

Species distribution

For each of the selected habitat, the protected and/or threatened plant, butterfly and bird species were identified (Bal et al., 2002). Information on the distribution of these species was derived from the nation-wide survey data of the Dutch Butterfly Conservation, SOVON and FLORON, for butterflies, breeding birds and vascular plants, respectively. The Dutch Butterfly Conservation provided us with maps on a 250 x 250 m scale, of the total number of protected and/or threatened butterfly species for each of the selected habitat types. SOVON provided us with similar information on bird species on a 1000 x 1000 m scale. The number of species within each grid cell was assigned only to those locations were the specific habitats were present, according to the habitat map. Information on plant species occurrence was provided by FLORON, on a 1000 x 1000 m scale. It was assumed that the total number of plant species of a specific habitat type within each 1000 x 1000 m grid cell, varied between underlying 250 x 250 m grid cells according to the suitability of the soil and the hydrology for that habitat type. Plants found within a 1000 x 1000 m grid cell were assigned to those underlying 250 x 250 m grid cells that had the most suitable vegetation structure, soil type and groundwater table. The suitability of soil and hydrology was based on Runhaar (1999). The Dutch Butterfly Conservation had used the same information on suitability, to derive the distribution maps of butterfly species.

Statistical methods

From the data set of critical load exceedances and species distribution, we selected those 250 x 250 m grid cells in which the area of the specific habitat was larger than 75%. From these grid cells we selected only the cells in which the presence of that particular habitat could be confirmed by a second habitat map. The focus on narrow defined habitat types improved the possibility of accurately linking habitat maps with both species distribution maps and critical loads. By focusing on habitats with similar soil types and hydrological conditions, we also diminished the chance of finding effects of deposition purely because of differences in deposition levels between eco-regions. For plants, the analysed region of the higher sandy soils is identified as a separate eco-region (Van der Meijden, 1990). Although, there are five separate flora districts characterised within this eco-region, Van der Meijden (1990) states that the plant species composition within these districts are quite similar, especially with respect to the oligotrophic vegetations, such as the forests on poor sandy soils, heathlands, bogs and sandy grasslands. For bird and butterfly species, eco-regional differences in species composition of the selected habitat types are thought to be of minor importance (SOVON, 2002; Bos et al. 2006).

After selection, the data within each 1000 x 1000 m grid cell was averaged. This was done to counteract possible pseudo replication, introduced by the downscaling of the distribution maps. The analysed data set consisted of 1036 data points (1000 x 1000 m grid cells) for dry forests on poor sandy soils, 412 for forests on rich sandy soils, 274 for dry heathlands, 143 for wet heathlands, 51 for bogs, and 295 data points for nutrient-poor wet grasslands.

The effect of critical load exceedance on the number of observed protected species was analysed by using logistic regression, with the maximum number of species as binomial denominator. Where necessary, overdispersion was accounted for by inflating the binomial variance by an unknown factor (McCullagh and Nelder, 1989). Generalised additive models gave results that were similar to simple linear logistic regression and, therefore, only the latter is reported here. The fitted model is Logit(f) = α + β·Critical Load exceedance, with f being the fraction of species present. The dimension of critical load exceedance is $kmol.ha^{-1}.yr^{-1}$. To calculate the fraction of species present given a particular critical load exceedance, the model can be formulated as:

$$f = \frac{1}{1+e^{\left(-\alpha-\beta \bullet exceedance\right)}}$$

5.3 Results

The results from the logistic regression analyses are presented in Table 5.2. In nearly two thirds of the combinations of examined habitats and taxonomic groups, a significant negative effect of critical load exceedance was found. This indicates that high exceedance levels correspond with low numbers of species. Negative effetcs were found in all examined habitats and for all groups of species. Negative effects were also found when overall species numbers were analysed. In both wet heathlands and bogs, significant negative effects were only found for birds. In these habitat types, no significant relationships were found for vascular plants and butterflies, despite the large critical load exceedances.

Table 5.3 shows that the predicted loss in the number of species at exceedance levels of 500 $mol.ha^{-1}.yr^{-1}$, ranges between 5% and 35%, depending on which habitat type and species group is looked at. At an exceedance of 1000 $mol.ha^{-1}.yr^{-1}$ the number of species were reduced to levels of between 10% and 58%, compared to predicted values at critical load. At 1500 $mol.ha^{-1}.yr^{-1}$ this increases to between 14% and 73%. At current deposition levels, 23% of the Dutch area of selected habitats experience exceedances between 500 and 1000 $mol.ha^{-1}.yr^{-1}$. In 25% of the area the exceedance is 1000 - 1500 $mol.ha^{-1}.yr^{-1}$.

5.4 Conclusion and discussion

Effects of deposition

In all examined habitats we were able to show significant negative effects of critical load exceedance on the occurrence of threatened/protected species. This conclusion is in line with the results from nitrogen addition experiments (Bobbink et al., 2008) and dynamic ecosystem modelling (Van Dobben et al., 2006), which both identify the examined habitats as sensitive to nitrogen deposition. Experiments and modelling, however, have focused on the effects on plants and vegetations (Achermann & Bobbink, 2003; Bobbink et al., 2008; De Vries et al., 2008). Our results indicate that nitrogen deposition might have effects on a much broader range of species, including fauna species, such as butterflies and birds.

Table 5.2 Parameters from simple linear logistic regression of protected/threatened butterfly, breeding bird and vascular plant species against the critical load exceedance. Parameters denoting significant negative effects are presented in bold.

	α			β		
	α	s.e.	p	β	s.e.	P
Forests on poor sandy soils (n = 1036)						
Butterflies	-2.505	0.135	0.000	-0.027	0.085	0.751
Breeding birds	-0.204	0.051	0.000	**-0.311**	0.033	0.000
Plants	-2.671	0.144	0.000	**-0.388**	0.097	0.000
Total	-0.971	0.041	0.000	**-0.257**	0.027	0.000
Forests on rich sandy soils (n = 412)						
Butterflies	-3.098	0.223	0.000	0.186	0.148	0.208
Breeding birds	-0.179	0.082	0.030	**-0.189**	0.057	0.001
Plants	-1.376	0.242	0.000	**-0.662**	0.184	0.000
Total	-0.970	0.074	0.000	**-0.215**	0.052	0.000
Dry heathland (n = 274)						
Butterflies	-1.832	0.207	0.000	**-0.389**	0.173	0.024
Breeding birds	-0.358	0.101	0.000	**-0.168**	0.082	0.040
Plants	-0.994	0.154	0.000	**-0.411**	0.128	0.001
Total	-0.959	0.074	0.000	**-0.279**	0.061	0.000
Wet heathland (n = 143)						
Butterflies	-2.063	0.173	0.000	0.060	0.194	0.757
Breeding birds	-0.152	0.086	0.078	**-0.428**	0.102	0.000
Plants	-1.617	0.168	0.000	0.116	0.187	0.533
Total	-1.209	0.086	0.000	-0.073	0.099	0.460
Bogs (n = 51)						
Butterflies	-3.535	0.752	0.000	0.717	0.442	0.104
Breeding birds	1.180	0.338	0.000	**-0.729**	0.210	0.001
Plants	-1.760	0.664	0.008	-0.029	0.412	0.943
Total	-0.898	0.292	0.002	-0.222	0.183	0.225
Nutrient-poor wet grasslands (n = 295)						
Butterflies	-2.882	0.422	0.000	**-0.910**	0.467	0.050
Breeding birds	0.029	0.080	0.721	**-0.310**	0.082	0.000
Plants	-2.064	0.141	0.000	0.008	0.130	0.950
Total	**-1.217**	**0.076**	**0.000**	**-0.272**	**0.076**	**0.000**

Table 5.3: Percentage of species loss at 500, 1000 and 1500 mol.ha^{-1}.yr^{-1} critical load exceedance relative to a situation with no exceedance (i.e. 0 mol.ha^{-1}.yr^{-1}), using the estimated logistic regression models. ns = not significant.

	Forests on poor sandy soils n= 1036	Forests on rich sandy soils n= 412	Dry heathland n = 274	Wet heathland n = 143	Bogs n = 51	Nutrient-poor wet grasslands n = 295
Butterflies	Ns	ns	-16/-29/-41	Ns	Ns	-35/-58/-73
Birds	-8/-17/-25	-5/-10/-15	-5/-10/-14	-11/-22/-33	-9/-20/-32	-8/-15/-23
Plants	-17/-30/-42	-24/-43/-58	-14/-27/-38	ns	Ns	ns
Total	**-9/-18/-25**	**-8/-15/-22**	**-10/-19/-27**	**ns**	**Ns**	**-10/-19/-28**

Quantified data regarding effects on fauna species are quite rare in scientific literature, although it is widely recognised that changes in vegetation structure and/or plant species composition, theoretically, might effect fauna by influencing the possibilities for finding food and/or shelter. The fact that the high natural variation between sites, caused by factors such as weather, local soil conditions, management(history) and habitat size, did not overrule the chance of finding significant negative effects, further strengthens the idea that nitrogen deposition is an important threat to biodiversity. Figure 5.1 shows the calculated species loss, based on the regression functions, which can indeed be quit substantial. The focus on protected and/or threatened species may have helped to find such large negative effects. However, experimentally derived dose-effect functions, based on the total number of plant species – and not only on the threatened and/or protected species (see Bobbink et al., 2008) – yield similar reduction percentages. At exceedance levels of 500 - 1000 mol.ha^{-1}.yr^{-1} plant species richness was reduced by 10% to 20% (Bobbink et al., 2008). At higher exceedance levels, the functions derived by Bobbink et al (2008) yield somewhat smaller reduction percentages, than those found by us, but larger reduction percentages also have been published (Stevens et al., 2004). In a transect of 68 acid grasslands across Great Britain, Stevens et al (2004) show that, on average, one plant species was lost per 4 m^2 quadrant for every 2.5 kg N.ha^{-1}.yr^{-1} (~175 mol N. ha^{-1}.yr^{-1}). At exceedance levels of 500 - 1000 mol.ha^{-1}.yr^{-1} this would mean that plant species richness will be reduced by 44 - 51%, applying a critical load of 1100 mol.ha^{-1}.yr^{-1}

Contrary to the results from Bobbink et al. (2008), we did not find any effects on plants within moist habitats. Wassen et al (2005) argued that enhanced phosphorus concentrations are more likely to be the cause of plant species loss than nitrogen enrichment in wet and moist habitats (grasslands), by showing that, within a gradient of declining levels of atmospheric nitrogen deposition, many more endangered plant species persisted under phosphorus-limited conditions than under those with limited nitrogen. We, however, did find effects on butterflies and birds in wet habitats (bogs, wet heathlands and nutrient-poor wet grasslands), showing the sensitivity of these habitats, but we did not find significant effects on plants. The absence of such significant effects on plants in these habitats, is most likely caused by our experimental design. The lack of effects on bog plants (and butterflies) might be due to the relative high exceedances at most sites and to the lack of sites with small exceedances (Table 5.1). This limited variation in exceedance might have decreased the statistical power to find any significant effects. Another explanation might be that analyses on a scale of 1000 x 1000 m might be too coarse for describing the variation in plant occurrences in wet habitats. Effects on birds and butterflies might be easier to detect on this scale, because of a better correspondence with the scale of territory sizes and/or home range of these species.

Applicability of the regression functions

The calculated regression functions, might help to relate the pressure based biodiversity indicator (i.e. critical load exceedance), to more generally understandable indicators, such as the loss of threatened and/or protected plants, birds and butterflies. This might help to improve the current policy indicators used in the convention on the protection of biodiversity (UNEP, 2005; EEA, 2007) and in conventions on air pollution (e.g. EC, 2001). However, the regression functions are derived by correlative research and thus do not describe a cause-effect relationship between deposition and species number. Although evidence from dynamic modelling and nitrogen-addition experiments supports our findings, the regression functions must be used with caution. It remains questionable whether a reduction of nitrogen automatically leads to an increase in protected/threatened species. Up until now, the few studies on the dynamics of plant diversity recovery after reductions of nitrogen, show divergent results. Recovery will often take time and is dependent on many factors, such as nature management and proximity of populations. Studies on dynamic ecosystems might help in testing the applicability of our regression functions as dose-effect relations. Moreover, dose-effect functions might differ, regionally, and applicability of our functions need to be tested, by using data from other countries.

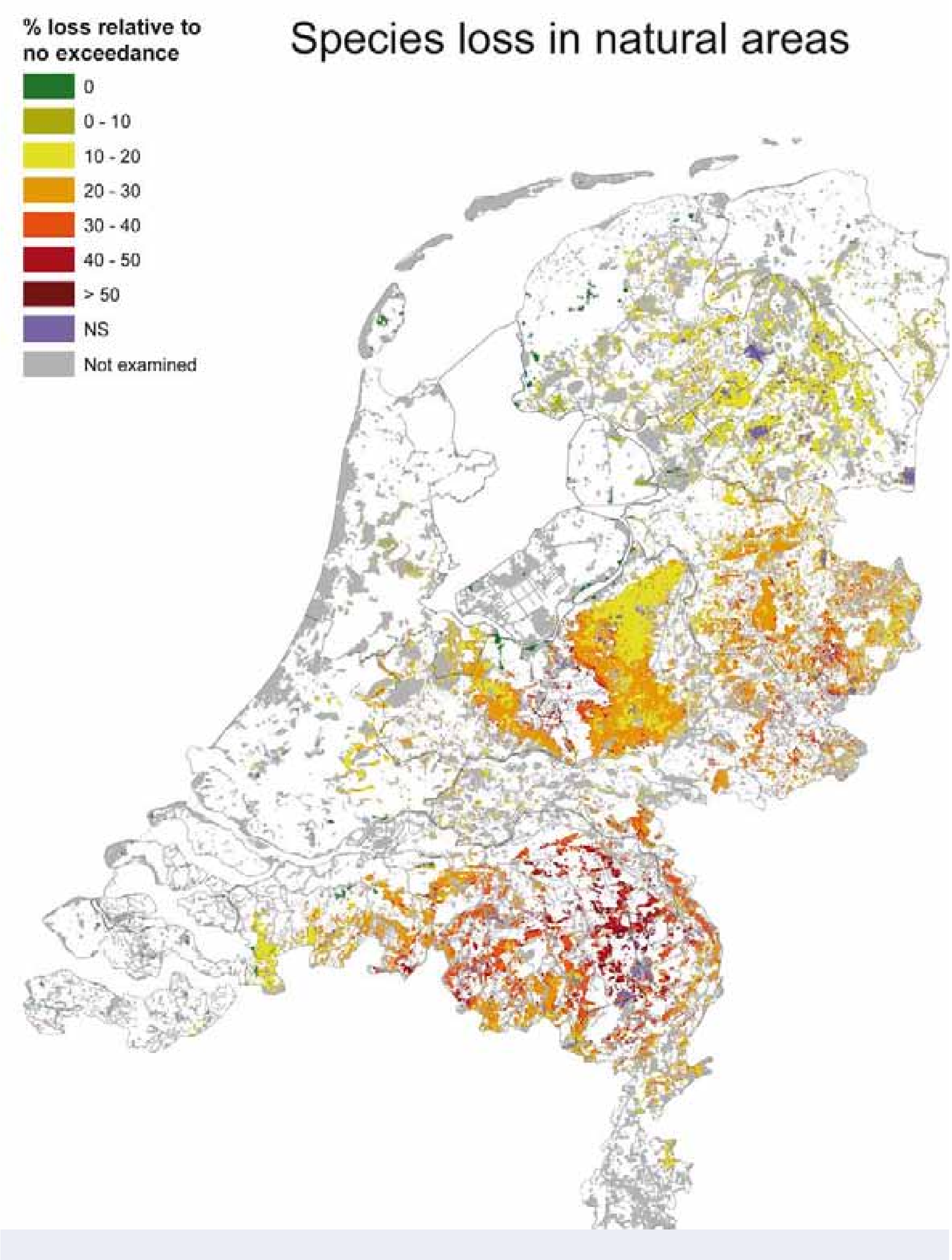

Figure 5.1. Percentages of species loss, relative to a situation without critical load exceedance for nitrogen, as calculated from the derived regression functions and the critical load exceedances in examined habitats within the Netherlands.

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6 Tentative Dose-response Function Applications for Integrated Assessment

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6.1 Introduction

Integrated assessment of air pollution abatement policies includes an effect-based component whereby exceedances of critical loads of nutrient nitrogen are compared between scenarios. In this context, a scenario may be preferred to another when it yields lower exceedances (e.g. in a EUNIS class) or a smaller European ecosystem area at risk. Thus, exceedance targets have been used together with economic and technical constraints to support the development of emission abatement policies. However, policy analysts have recently invited the effect-based community to consider further quantification of policy relevant effect indicators such as biodiversity changes, and link them to the integrated modelling work (UNECE 2008). Biodiversity protection is also a priority issue of the environmental policy of the European Union. This includes the establishment of the Natura 2000 network by 2010, but also – as stated by DG Environment of the European Commission – by better integrating biodiversity considerations into other but environmental policy areas (Murphy 2008). The EEA recently published results of the first phase of the Streamlining European 2010 Biodiversity Indicators (SEBI 2010) project on the development of indicators to monitor progress towards, and help achieve the European target to halt the loss of biodiversity by 2010 (EEA 2007).

European empirical critical loads have been developed mainly based on long-term field N-addition experiments, from which ranges of critical loads for many EUNIS classes were derived from the lowest nitrogen addition for which biological effects occurred (Achermann and Bobbink 2003). Therefore, the inclusion of empirical critical loads could move us forward in addressing biodiversity in integrated assessment. However, the meaning of 'biodiversity change' is not easily caught in a single indicator. The definition of biological diversity is provided by the Convention on Biodiversity (CBD 1992) as the variability among living organisms from all sources including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems. Other definitions include a distinction between (from Dougherty 2000/01):

- genetic diversity: the variety of different versions of the same gene within a species;
- species diversity: the variety of species within an area, which includes *species richness* (number of species) and *species abundance* (number of individuals per species);
- ecosystem diversity: the variety of communities of interacting organisms and their environments;
- cultural diversity: the variation in behaviour between groups of the same species.

As stated by the Ecological Society of America, "out of convenience or necessity, biodiversity is usually quantified in terms of numbers of species, and this perspective has greatly influenced conservation goals. It is important to remember, however, that benefits of that biodiversity supplies to humanity are delivered through populations of species residing in living communities within specific physical settings – in other words, through complex ecological systems (Daily and Ehrlich, 1995)" (ESA 1997, p.5).

A recent publication of the International Union for the Conservation of Nature (IUCN) on the Red List of threatened species (Vié et al. 2008) indicates that about 25% of mammal species is threatened with extinction. This and other publications that address threats to biological diversity confirm the importance of biodiversity indicators in environmental policy. Of course, there are more culprits than nitrogen, and the contribution of nitrogen affects many links in the chain of causes to effects. Evidence shows that also low rates of nitrogen contribute to the loss of species (Clark and Tilman 2008, Emmett 2007, Nordin et al. 2005). It remains challenging to identify appropriate indicators that are applicable on a European scale under the effect-based policy development of the LRTAP Convention. However, needs must – also to follow up on the request by the Executive Body – useful and usable biodiversity related indicators have to be developed. Therefore, the contents of this chapter should be viewed as a first attempt to introduce a biodiversity indicator that can be applied on a regional scale to compare impacts between scenarios in integrated assessment.

Functional relationships have been derived (see chapter 4) between (a) the nitrogen load and indicators for loss of biodiversity, and (b) the exceedance of empirical critical loads of nitrogen and the loss of biodiversity. This chapter describes an illustrative regionalized application of these dose-response (D-R) functions to produce European maps of scenario dependent biodiversity loss. This approach could be used to evaluate N emission reduction scenarios by the Task Force on Integrated Assessment Modelling (TFIAM) under the LRTAP Convention, but also to support scenario development by the European Consortium for Air Pollution and Climate Strategies (EC4MACS) under the EU LIFE programme.

6.2 An illustrative regional application of dose-response functions from chapter 4

Ranges of empirical critical loads have been established for various ecosystem types over recent years (Achermann and Bobbink 2003). These ranges have been adapted for EUNIS classes, which are also used in the harmonized land cover database (De Bakker et al. 2007). Dose-response functions have not been derived for all of these ecosystem classes. Species richness has been chosen as response variable for grasslands and scrubs, while the similarity index has been used for forests. Table 6-1 lists the ecosystem types that have been distinguished, including their range of empirical critical loads and the dose-response functions applied in this chapter.

Table 6-1 Ecosystems for which loss of biodiversity has been estimated with their respective range of empirical critical loads and applied dose-response functions (DR(x); in kgN/ha)

EUNIS Code	CLemp(N)		D-R (x)		
	min	max	x = N-load	x = Exc_{min}	x = Exc_{max}
E1.2	15	25	$1.0408e^{-0.0048x}$	$0.9873e^{-0.005x}$	$0.9427e^{-0.005x}$
E1.7 or E1.9	10	20			
E2 without E2.3	20	30			
E2.3	10	20			
E3	10	25			
E4	5	15			
F1	5	10	$-0.0044x + 0.9982$	$-0.0044x + 0.9726$	$-0.0044x + 0.9321$
F2	5	15			
F4	10	20			
G	10	20	$2.3741x^{-0.3284}$	$1.7827x^{-0.2715}$	$1.5438x^{-0.242}$

Table 6-1 shows that we tentatively assumed that the D-R function derived for boreal coniferous woodlands can be applied to all woodlands (EUNIS class G). The D-R function for arctic alpine and sub-alpine scrub is tentatively assumed to be applicable to all heathland, scrub and tundra (EUNIS class F), referred to as 'scrubs' in the remainder of this chapter. The results of the regional application of the D-R functions subject to N-deposition or exceedances are mapped as a EUNIS class specific protection of species richness within an EMEP grid cell, expressed as percentage (100% is full protection). In case of different critical loads within a single grid cell, the median protection percentage was taken. The depositions are obtained from the EMEP model under the Current Legislation (CLE) and Maximum Feasible Reduction (MFR) scenarios (see Appendix A). A response per EUNIS class in each EMEP grid cell is obtained by subjecting the D-R functions described in chapter 4 to the CLE or MFR deposition. The focus is on the application of the function that describes the relationship between total nitrogen deposition (addition + background deposition) and the change of species richness, DR(N_{dep}). This function is *not* related to empirical critical loads established, but obviously embeds a threshold below which effects are not likely. In addition, also the functions describing the relationship between effect and the dose expressed as the exceedance of the minimum, DR(Exc_{min}), and maximum, DR(Exc_{max}), empirical critical load respectively are computed.

6.3 Results

Exceedances of the minimum empirical critical loads are illustrated in Figure 6-1 for 1990 (top), under CLE in 2020 (CLE2020), under MFR (MFR2020), and – as an example of low historic deposition – in 1900 (bottom). Figure 6-1 shows the exceedances in grasslands, scrubs and forests (from left to right). Both the magnitude (red is highest) and geographical distribution of exceedances decreases, particularly in forest areas, from 1990 to 2020 under MFR. This is also true for the effect on species diversity of grass and scrubs.

Figure 6-2 shows percentage (area weighed median) of species abundance in 1990 in forests (top) and species richness in scrubs and grasslands (bottom) due to nitrogen deposition (left), the exceedance of the minimum empirical critical load (centre) and the exceedance of the maximum empirical critical load (right). The red shading indicates areas where the estimated biodiversity indicator percentages are lower than 80%, while green shadings indicate areas where this percentage is between 95 and 100%. We can see that for 1990 depositions DR(N_{dep}) indicates an effect for grasslands between 95 and 100% in many countries, while DR(Exc_{min}) indicates no effect in the same region. This would indicate that the minimum empirical critical load is not sufficiently low. This could have many reasons, including the inappropriateness of extrapolating local dose response function over Europe. However, there may be a need to review the influence of background deposition in the establishment of empirical critical loads.

Finally, Figure 6-3 is the species richness (grass lands and scrubs) and species abundance (forests) using N deposition (DR(N_{dep}). It illustrates that these percentages in forests (right) under MFR2020 are computed to be lower than 80%, in comparison to non-affected forests especially in Belgium, the Netherlands. However, most losses can be seen for scrubs where many areas show a species diversity of between 90 and 95% between 1990 and MFR2020. The highest losses in biodiversity occur in forests, possibly caused by the higher deposition on forest compared to other ecosystem types. It should be noted, that the patterns of sensitive areas in the exceedance maps (Figure 6-1) are broadly similar to the patterns of biodiversity losses (Figure 6-3).

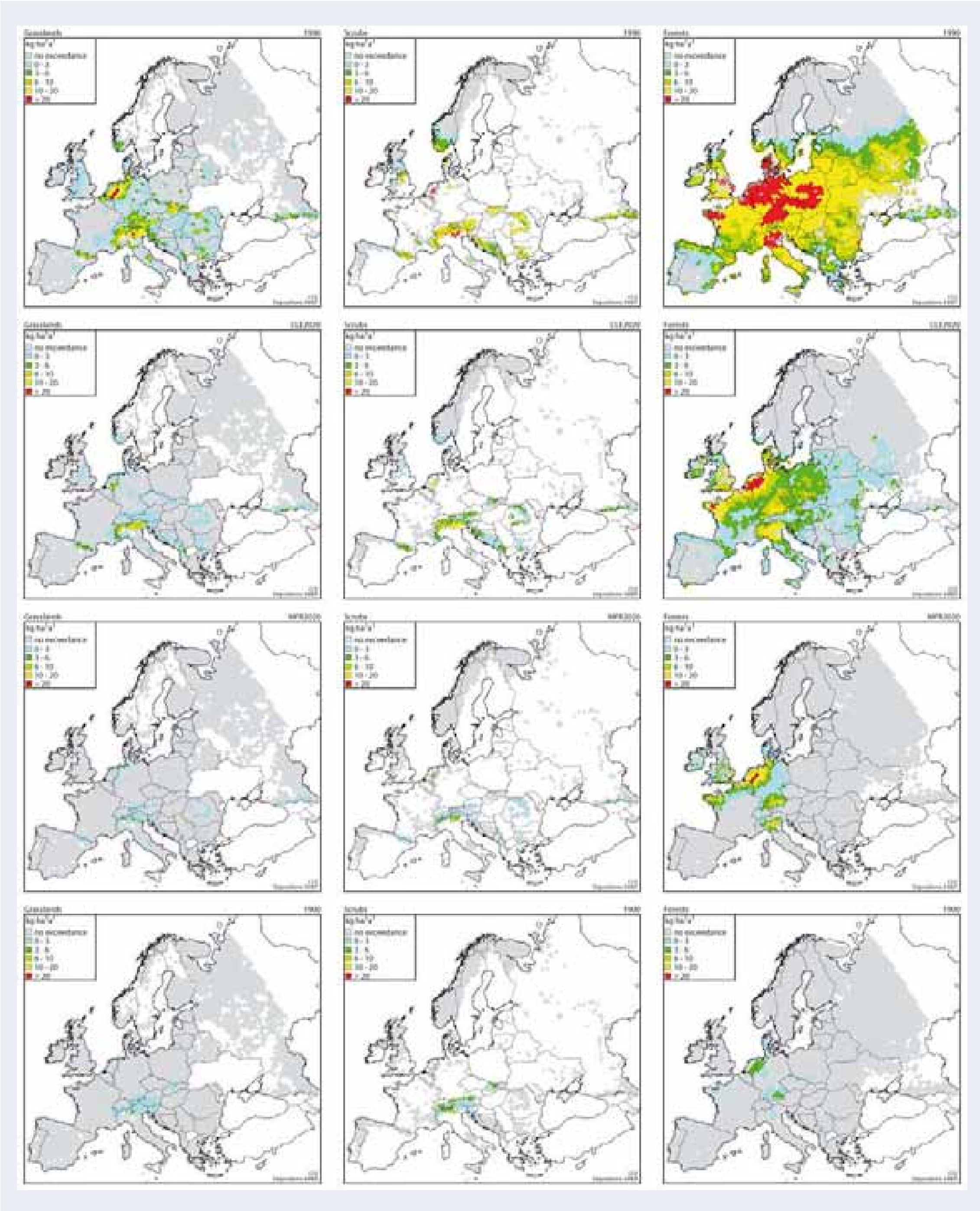

Figure 6-1. Exceedances of the minimum empirical critical loads for grasslands (left), scrubs (centre) and forests (right) in 1990 (top), in 2020 under CLE (CLE2020), in 2020 under MFR (MFR2020), and in 1900 (bottom).

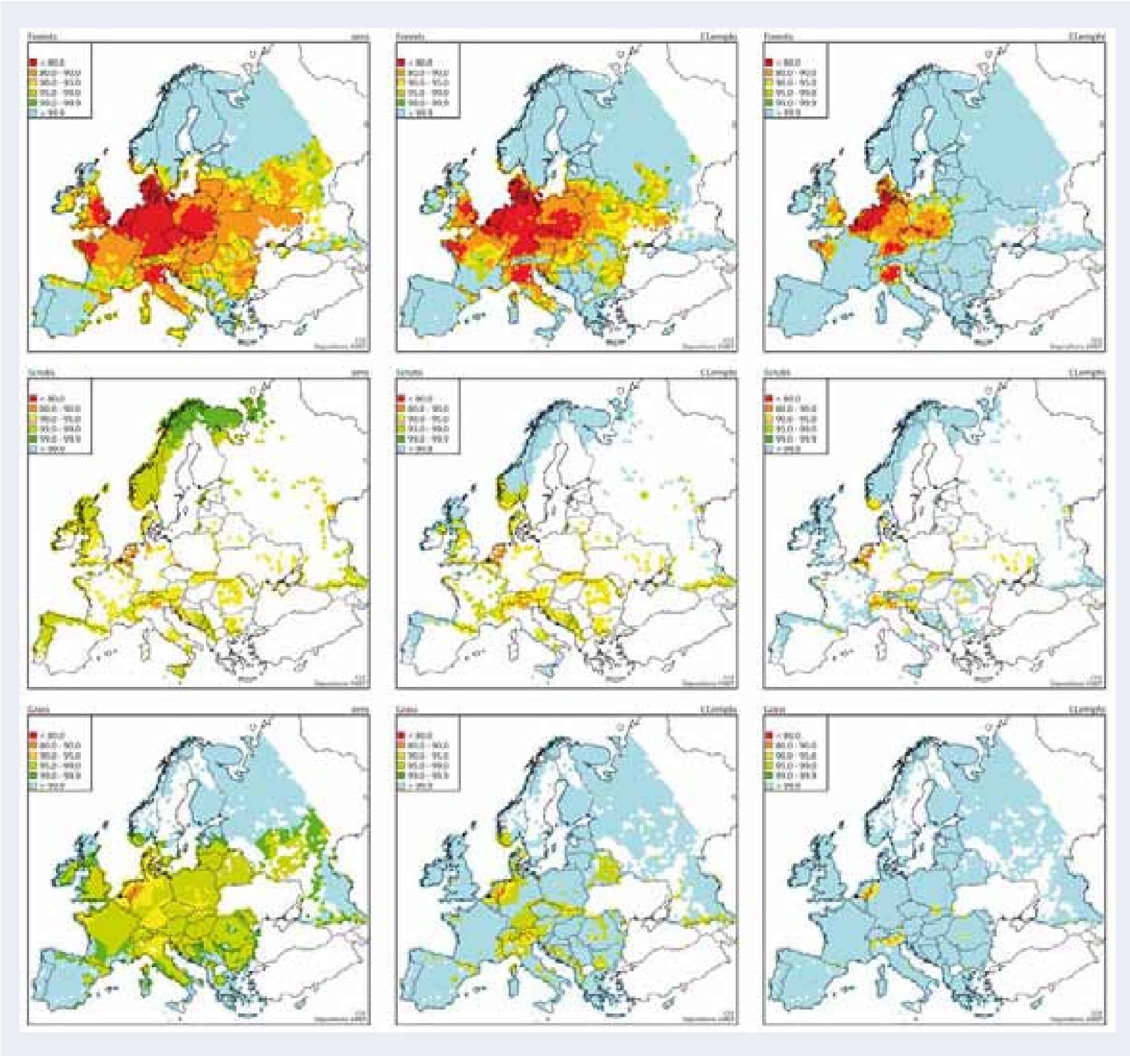

Figure 6-2. Percentage of species abundance in 1990 in forests (top) and species richness in scrubs (centre) and grasslands (bottom) due to N deposition (left), the exceedance of the minimum empirical critical load (centre) and the exceedance of the maximum empirical critical load (right).

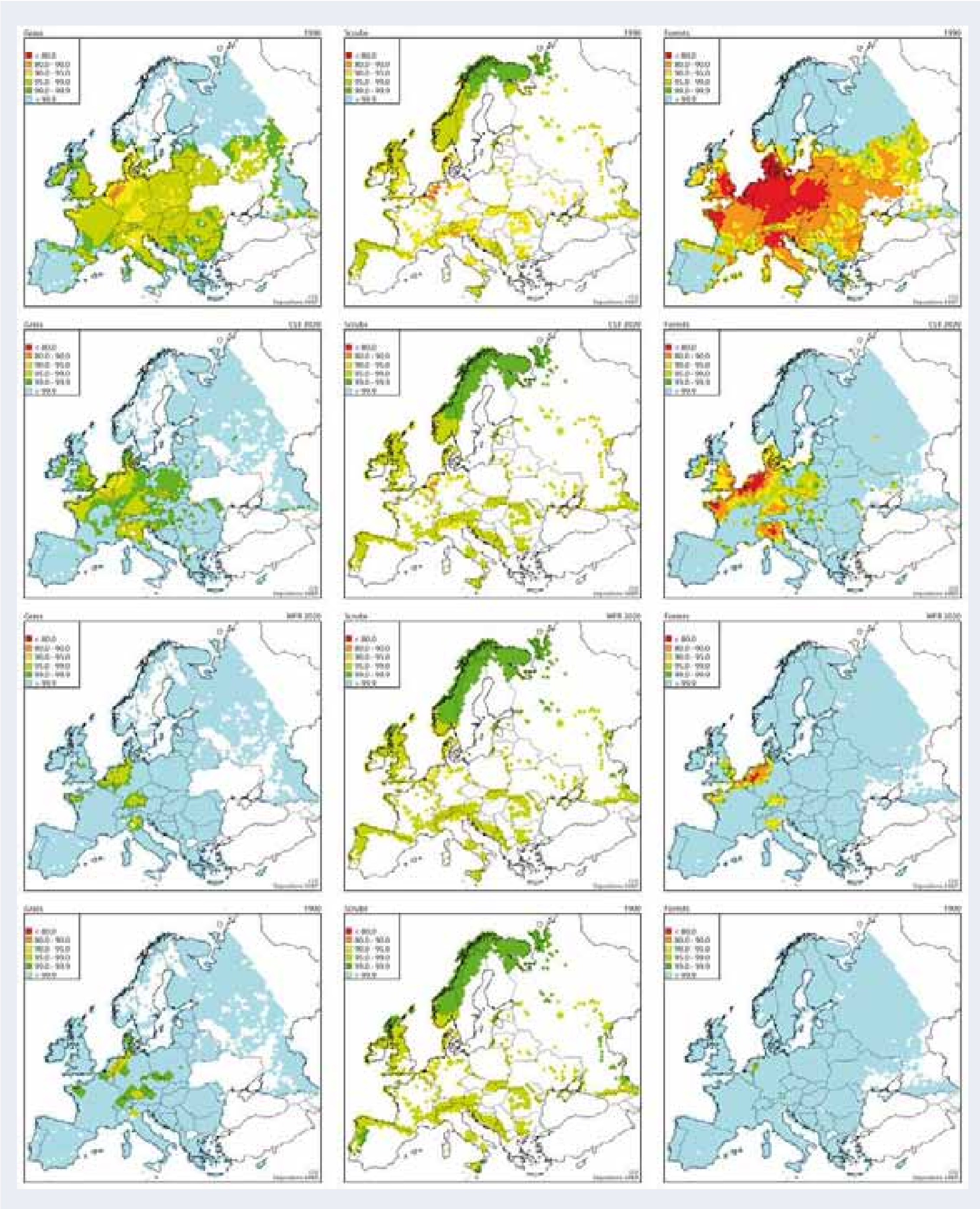

Figure 6-3. Percentage of species richness in grass lands (left), scrubs (middle) and species abundance in forests (right) in 1990 (top), under Current Legislation in 2020 (CLE2020), Maximum Feasible Reductions in 2020 (MFR2020) and in 1900 (bottom).

6.4 Conclusions and recommendations

This chapter describes results of the tentative allocation of effects of N deposition on species diversity (grass lands and scrubs) and species abundance (forests). For this the EUNIS classification has been used, which has been mapped over Europe using the harmonised land cover map under the LRTAP Convention. Dose-response functions were derived from documented nitrogen addition experiments including ambient deposition at the experimental plots during the N-addition time period (see chapter 4). These dose-response functions were then applied to the relevant EUNIS classes and subjected to EMEP computed deposition in 1990, CLE2020, MFR2020 and 1900. While results show a decrease in the biodiversity indicator, caution is in order, not the least because no distinction is made between nitrophobous and nitrophilous species and related changes due to, e.g., invasive species. However, the exercise described in this chapter can be useful in a relative context, i.e. for the assessment of the performance of one scenario in comparison to another for use in integrated assessment of European emission abatement alternatives.

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7 Critical and Present Loads of Cadmium, Lead and Mercury and their Exceedances, for Europe and Northern Asia

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7.1 Introduction

Over the last decade, a number of studies have been carried out to assess critical loads of heavy metals, in particular for cadmium (Cd), lead (Pb) and mercury (Hg). In the 2001 CCE Status Report, maps of critical loads of Cd and Pb for European forest soils were presented (Reinds et al. 2001) based on methods which were later laid down, in a somewhat adapted form, in the Mapping Manual (UBA 2004). In 2004 the Working Group on Effects of the LRTAP Convention requested the CCE to issue a call for data on critical loads for Cd, Pb and Hg. Results from the 16 countries that responded to the call, as well as critical-load data obtained from the European Background database for heavy metals, were published in Slootweg et al. (2005). Exceedance maps were produced by comparing the critical loads with modelled heavy metal deposition (Ilyin and Dutchak 2005). An effect-based assessment of both priority (Cd, Pb, Hg) and other heavy metal emission-reduction scenarios for Europe, was prepared as a contribution to the review of the 1998 heavy-metals protocol (Hettelingh et al., 2006). Recently, the Background database of the CCE was extended to the east and now also covers the whole of Russia and the EECCA countries (Reinds 2007). Since the domain of the deposition modelling also has been extended to the EECCA countries and to most of Russia, a new assessment was made of critical loads and critical-load exceedances for Cd, Pb and Hg. The following sections describe the methods and data used for calculating critical loads, the modelling of heavy metal deposition and show critical loads and exceedances.

7.2 Critical Loads

Methods

Critical loads for cadmium (Cd), lead (Pb) and mercury (Hg) for forests were calcutaled by using the methods described in the Mapping Manual (UBA 2004). The critical load is defined as:

$$CL(M) = M_u + M_{le(crit)} \tag{7-1}$$

where:

$CL(M)$ = critical load of heavy metal M (g ha^{-1} a^{-1})

M_u = removal of heavy metals by biomass harvesting or net uptake in forest ecosystems from the mineral topsoil (g ha^{-1} a^{-1})

$M_{le(crit)}$ = critical leaching of heavy metals from the mineral topsoil (g ha^{-1} a^{-1})

Thus, the critical load consists of metal uptake and a critical metal leaching determined by a critical metal concentration, related to effects on the ecosystem.

Metal uptake was calculated from data on forest growth (obtained from existing databases; see Reinds et al. 2008) multiplied by metal contents in stem wood, obtained from the Mapping Manual. Contents of Cd and Pb in all trees were taken as average for the ranges listed in the manual's Table 5.19: 0.3 mg kg^{-1} for Cd and 5.0 mg kg^{-1} for Pb.

Critical leaching rates were calculated as being the water flux leaving the root zone multiplied by a critical metal concentration. Water fluxes were calculated with a regional hydrological model using detailed meteorological data; details can be found in Reinds et al. (2008). The critical concentration for Cd and Hg was based on eco-toxicological effects on soil organisms, and calculated as being a function of soil pH and the concentration of dissolved organic carbon (DOC). For Hg, the critical concentration was derived from a critical Hg content in soil organic matter and a fractionation ratio that relates the concentration of Hg in soil solution to the concentration in solids. Another critical limit for Hg, the critical concentration in precipitation, is not addressed in the context of this chapter. Further details on the methodology for the derivation of critical loads of heavy metals can be found in the Mapping Manual (UBA 2004).

7.3 Data

The most recent version of the European background database was used for calculating the critical loads. This database is described in Reinds et al. (2008). Critical loads were calculated for forests only. When variables were not found in existing databases, the recommendations and transfer functions in the Mapping Manual (UBA 2004) were followed, as closely as possible.
A map with computational units was created by overlaying maps on soil, vegetation and forest growth regions. Overlaying these maps and merging polygons with common soil, vegetation and regional characteristics within grid cells of 10×10 km^2 resulted in about 3.8 million computational units. In the model runs, we only used computational units which were larger than 1 km^2, thus, reducing their number to 1.3 million, together still covering 96% of the total area.

Organic carbon content and pH were derived for each soil type from a pan-European database on forest soils (Vanmechelen et al. 1997). Organic matter content was assumed to be twice the organic content.

The dissolved organic carbon (DOC) concentration was calculated from a linear regression between DOC, pH and soil texture, using data from intensively monitored forest sites in Europe.

7.4 Results

For each of the EMEP 50×50 km grid cells, the frequency distribution of critical loads was calculated. From this distribution, we used the 5^{th} percentile critical load to represent the grid cell, to protect 95% of forested areas. The critical loads for Cd and Pb show strong regional patterns (Figure 7-1). The lowest critical loads are found in areas with low metal uptake and leaching: arid regions in southern Europe and in the New Independent States, with high pH values in the soils. The highest critical loads are found in areas with large amounts of excess precipitation, such as in north-western Europe and in areas with acidic soils where the critical concentration is relatively high, such as in parts of Germany, Poland and Russia.

Critical loads of Hg are high in areas with a large amounts of excess precipitation, for example, along the west coast of Europe (Figure 7-2). Critical concentrations of Hg decrease with decreasing DOC concentration in the soils, so that low critical loads are also found in areas with poor sandy soils, which are poor in organic matter. In areas with organic rich soils, critical loads are high, for example, in parts of Finland and the United Kingdom.

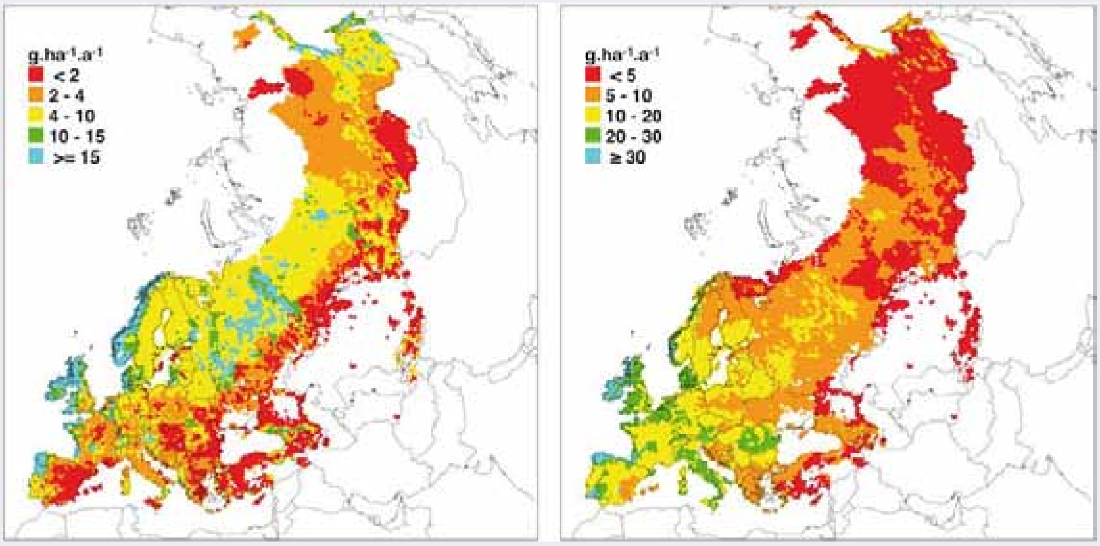

Figure 7-1 5th percentile critical loads of Cd and Pb for forests.

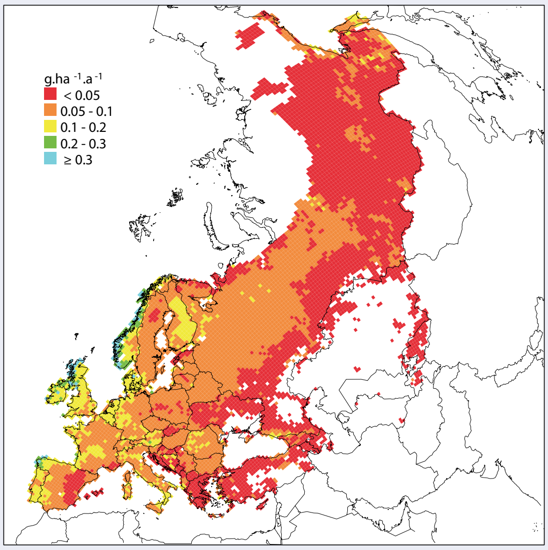

Figure 7-2 5th percentile critical load of Hg for forests.

7.5 Modelling of lead, cadmium and mercury depositions within the extended EMEP domain

For the evaluation of critical-load exceedances of heavy metals, information on atmospheric depositions is needed. This section summarises the main results from the modelling of atmospheric depositions of lead, cadmium and mercury, relating to 2006. The section also includes concise information on input data involved in modelling and the verification of calculated results by comparing them with the measurement data.

A brief description of the model and input data

According to the Action Plan for the EECCA countries, approved by the Executive Body of the Convention (ECE.EB.AIR/91), the EMEP domain was extended eastward, in 2008, to also cover the territories of the Central Asian countries (Kazakhstan, Kyrgyzstan, Tajikistan, Uzbekistan and Turkmenistan). An evaluation of ecosystem-dependent depositions over the extended EMEP domain was performed for 2006, by using the European-scale atmospheric transport model MSCE-HM. This model is actively used for operational calculations of heavy metal transboundary pollution within the EMEP region (ECE/EB.AIR/GE.1/2006/4). The spatial resolution of the model is 50×50 km². The main information output of the model includes fields of concentrations and depositions, ecosystem-dependent depositions, long-term trends of pollution levels and source–receptor relationships. A detailed description of the model is available in Travnikov and Ilyin (2005).

The input data for the model include meteorological fields, emissions of heavy metals and distribution of land-cover types. Meteorological data represents six-hour, three-dimensional fields of meteorological parameters (wind components, air temperature, precipitation etc), processed by the MM5 meteorological driver. The meteorological input data used for calculating pollution levels for 2006, are based on ECMWF analyses.

The emission data applied for calculating lead, cadmium and mercury levels, were based on official information, officially reported to the UNECE Secretariat by EMEP Parties, and supplemented with unofficial emission data. For European countries, in particular, the unofficial data on emissions of lead, cadmium and mercury, for 2006, were obtained by interpolation of emissions for 2000 and emission projections for 2010, available from the TNO emission data set (Denier van der Gon et al. 2005). For Central Asian countries, emissions were estimated on the basis of expert emission estimates, such as (Denier van der Gon et al. 2005, Pacyna et al. 1995 http://www.ortech.ca/cgeic/index.html, Pacyna et al. 2006 http://amap.no/Resources/HgEmissions/). Moreover, the emission of heavy metals to the atmosphere from natural emissions and wind re-suspension is considered. A description of the procedure that was applied for the emission estimation, is presented in the EMEP status report (Ilyin et al. 2008).

Land-cover data are required for the evaluation of dry deposition velocities and assessment of ecosystem-specific depositions. Currently, a preliminary land-cover dataset which was developed – within the Working Group of Effects (WGE) – for the Convention, is used in the model. This data set was applied to the area within the 'old' EMEP domain (135x111 grid cells). Over the extended part of the EMEP region, land-cover data were utilised, as prepared by the Geological Survey of the United States (USGS) (Guo and Chen 1994 http://edcsns17.cr.usgs.gov/glcc/) and described in (Ilyin et al. 2008, Gusev et al. 2007).

Evaluation of modelling results

Modelling results were evaluated by comparing them with observation data (concentrations in air, in precipitation and in wet deposition fluxes) collected from the EMEP monitoring network. Most of the measurement stations are located in the northern, western and central parts of Europe, while in Central Asia information on background monitoring is not available, yet. A summary of the verification of modelling results is described in this section, whereas the details are available in Ilyin et al. (2008).

The concentrations of lead in precipitation and wet deposition fluxes are underestimated by the model, by about 30% (Figure 7-3). However, the model performance at individual stations varies considerably. The concentrations and wet deposition fluxes for stations in Belgium, Germany, the United Kingdom, Poland, Sweden and the Netherlands were reproduced reasonably well. At these stations, the modelled and measured wet depositions and concentrations in precipitation typically correspond within around 40%. The model managed to reproduce spatial variability of the observed concentrations and fluxes: the correlation coefficients between measured and calculated parameters are about 0.7.

Annual mean concentrations of cadmium in precipitation and wet depositions fluxes were underestimated by the model, on average by a factor of two (Figure 7-4). Nevertheless, spatial variability was reproduced satisfactorily: the correlation coefficient between modelled and measured concentrations in precipitation is around 0.8 and for wet deposition fluxes about 0.7. The model performance for cadmium concentrations in precipitation varies between stations. The measured concentrations at German, Dutch and UK stations were simulated relatively well by the model: the correspondence between measured and modelled concentrations for precipitation is mostly kept within around 30%, and for fluxes within around 40%. Some underestimation of the concentrations in precipitation and fluxes is noted for Swedish, Belgium and Lithuanian stations. A higher underestimation is noted for Latvian, Polish and Finnish stations.

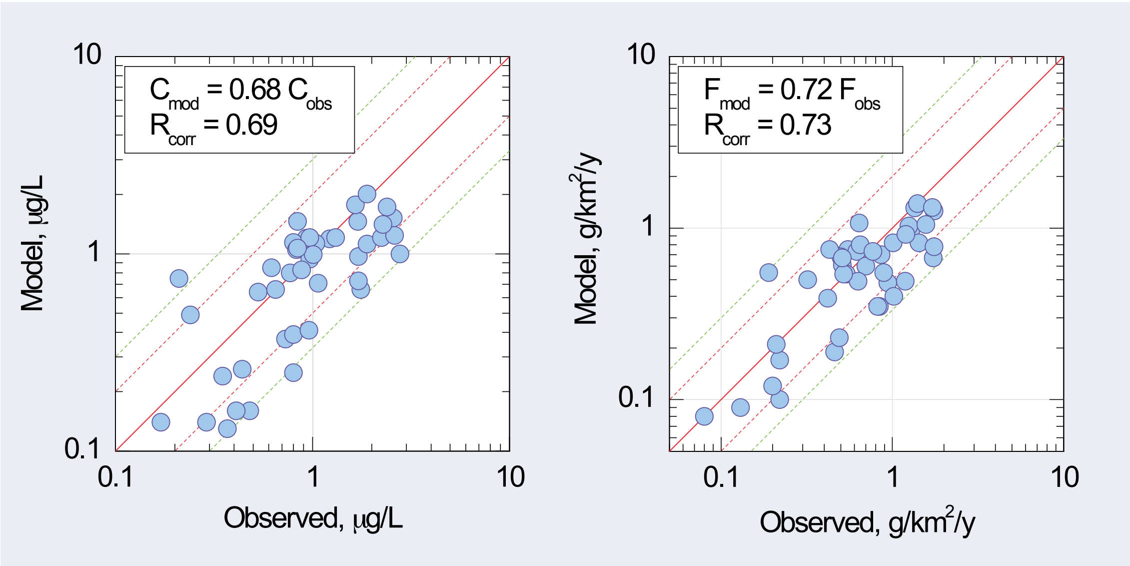

Figure 7-3 Modelled and measured annual means of concentrations (µg L⁻¹) in precipitation (left) and wet deposition fluxes (kg km⁻²a⁻¹) of lead, in 2006. The red solid line delineates the 1:1 ratio, the red and green dashed lines show the deviation by a factor of two and three, respectively.

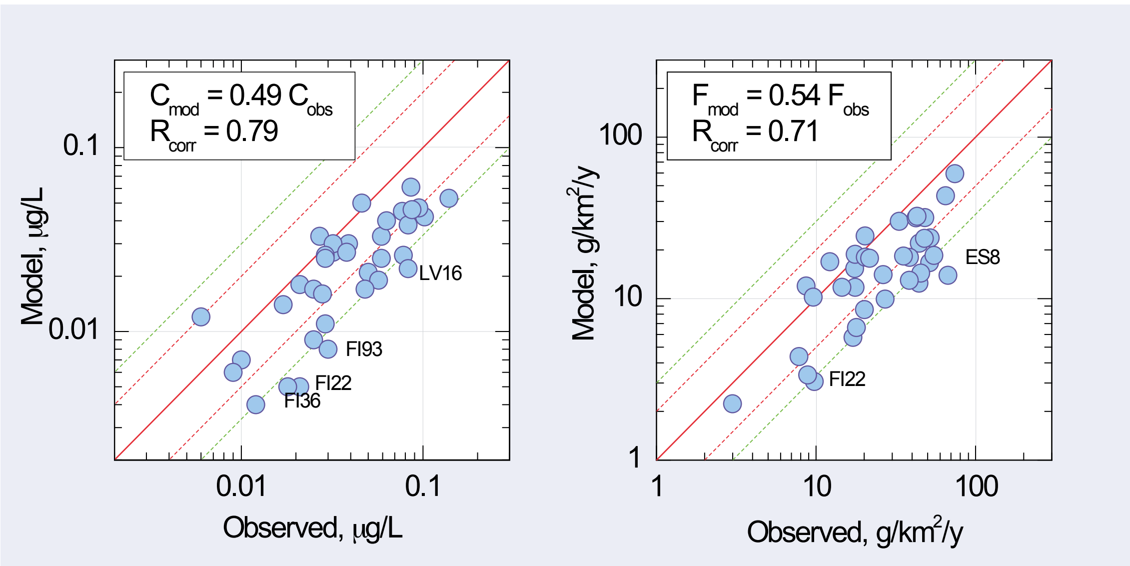

Figure 7-4 Modelled and measured annual means of concentrations (µg L⁻¹) in precipitation (left) and wet deposition fluxes (g km⁻²a⁻¹) of cadmium, in 2006. The red solid line delineates the 1:1 ratio, the red and green dashed lines show the deviation by a factor of two and three, respectively.

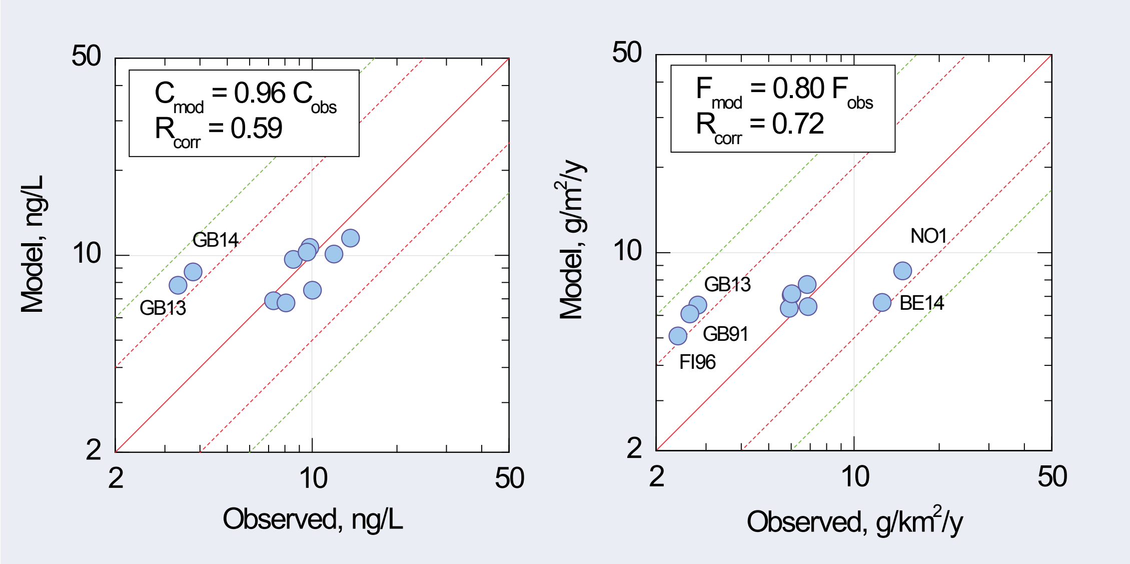

Figure 7-5 Modelled and measured means of concentrations (ng/L) of mercury in precipitation (left) and wet deposition fluxes (g km^{-2}a^{-1}). The red solid line delineates the 1:1 ratio, the red and green dashed lines show the deviation by a factor of two and three, respectively.

The correspondence between annual mean calculated and measured concentrations of mercury in precipitation lies within around 25%, and the correlation coefficient between modelled and measured values is 0.6 (Figure 7-5). The exceptions are two UK stations: GB13 and GB91, where modelled concentrations exceed the observed ones by a factor of 2. Annual wet deposition fluxes of mercury were on average underestimated by the model by about 20% (Figure 7-5b). At some stations, the correspondence is higher between the concentrations in precipitation than those in wet deposition fluxes, because of differences in precipitation amounts measured at stations and processed by MM5. The model represented the spatial variability in the observed wet deposition fluxes, relatively well: the correlation coefficient is about 0.7.

Computed deposition and concentration fields

The annual deposition fluxes in heavy metals were calculated for all land-cover types. The examples of spatial distributions of lead and cadmium depositions are demonstrated for coniferous forests and croplands. The deposition fluxes in lead to coniferous forests within the European and Central Asian countries range from 2 g ha^{-1}a^{-1} in the northern part of the extended EMEP domain to 40 g ha^{-1}a^{-1} in the southern part (Figure 7-6a). The highest deposition fluxes (higher than 40 g ha^{-1}a^{-1}) are associated with regions with significant emissions (Benelux region, Poland and the eastern part of the Ukraine). The depositions to ecosystems with low vegetation, such as croplands, are lower than to those with coniferous forests. The depositions to ecosystems with low vegetation typically vary from 2 to 15 g ha^{-1}a^{-1}, and their spatial pattern is similar to that of the forests (Figure 7-6b).

The cadmium depositions to coniferous forests in Europe and Central Asia range from 0.1g ha^{-1}a^{-1} in the northern part of the EMEP domain to 0.8 g ha^{-1}a^{-1} in the central and southern part (Figure 7-7a). In some regions, where anthropogenic emissions are large, such as in Poland, the Balkan countries, the Benelux region, the eastern Ukraine and the south-western part of Russia, the deposition fluxes exceed 1.5 g ha^{-1}a^{-1}.

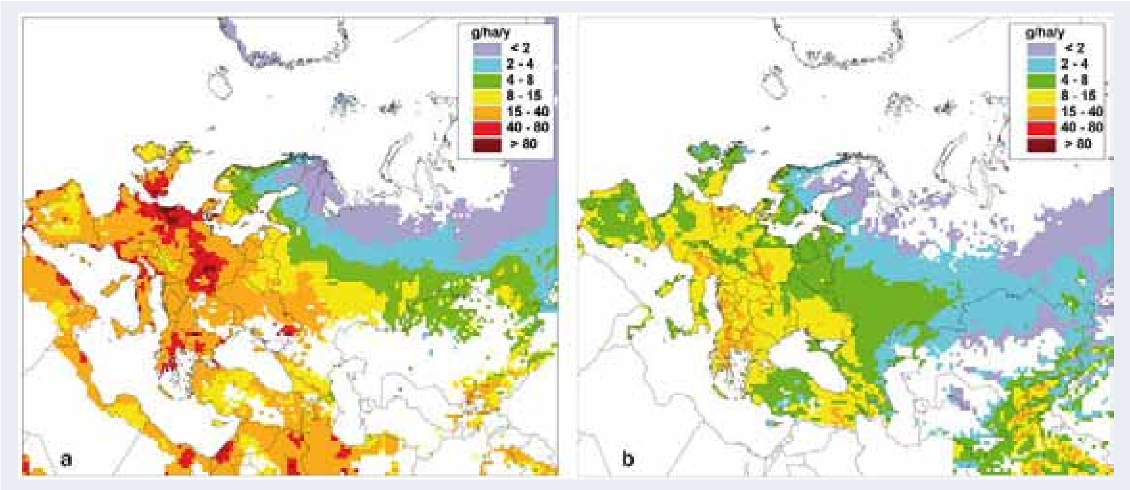

Figure 7-6 Total annual depositions of lead to coniferous forests (a) and to croplands (b), in 2006.

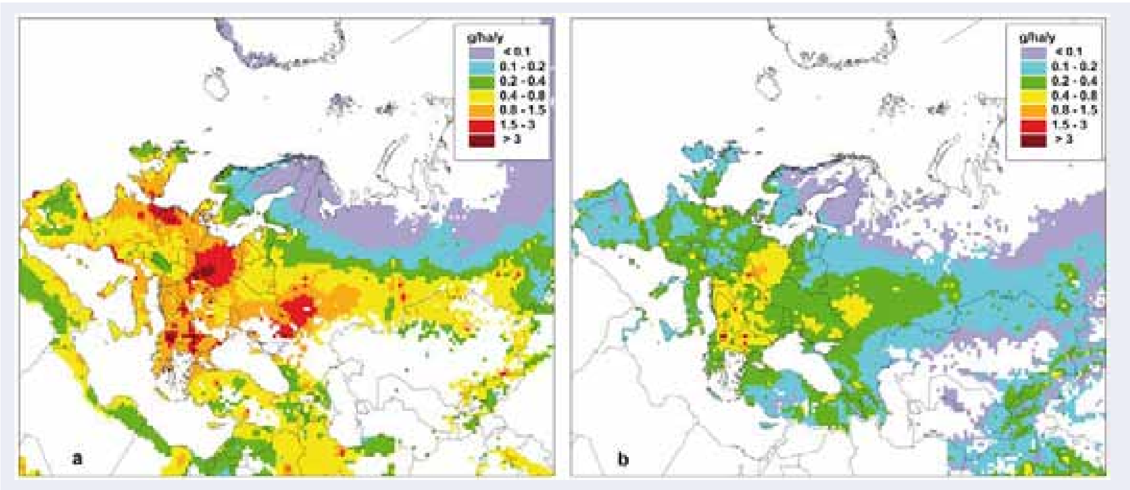

Figure 7-7 Total annual depositions of cadmium to coniferous forests (a) and to croplands (b), in 2006.

Similar to that of lead, the deposition of cadmium to croplands is smaller than to coniferous forests. It typically ranges from 0.1 to 0.4 g $ha^{-1}a^{-1}$. In regions with high emissions, the fluxes exceed 1.5 g $ha^{-1}a^{-1}$ (Figure 7-7b). The depositions to croplands within the Central Asian region are relatively low, compared to those within Europe, although the emission fluxes are of similar order (Ilyin et al. 2008). These low depositions are explained by typically low amounts of precipitation in the Central Asian region.

The mercury depositions to coniferous forests vary from 0.1 to 0.3 g $ha^{-1}a^{-1}$, and for croplands they vary from 0.1 to 0.2 g $ha^{-1}a^{-1}$, within most areas of Europe and Central Asia (Figure 7-8). Similar to that of lead and cadmium, elevated depositions of mercury are noted for regions with significant emission sources. In the Central Asian region, relatively high depositions are computed for the northern part of Kazakhstan, due to the estimated large emission fluxes within this area. Besides, relatively high deposition fluxes are indicated along the border between Kazakhstan and Kyrgyzstan. They are explained by elevated emissions, as well as by relatively large amounts of precipitation originating from windward mountain slopes.

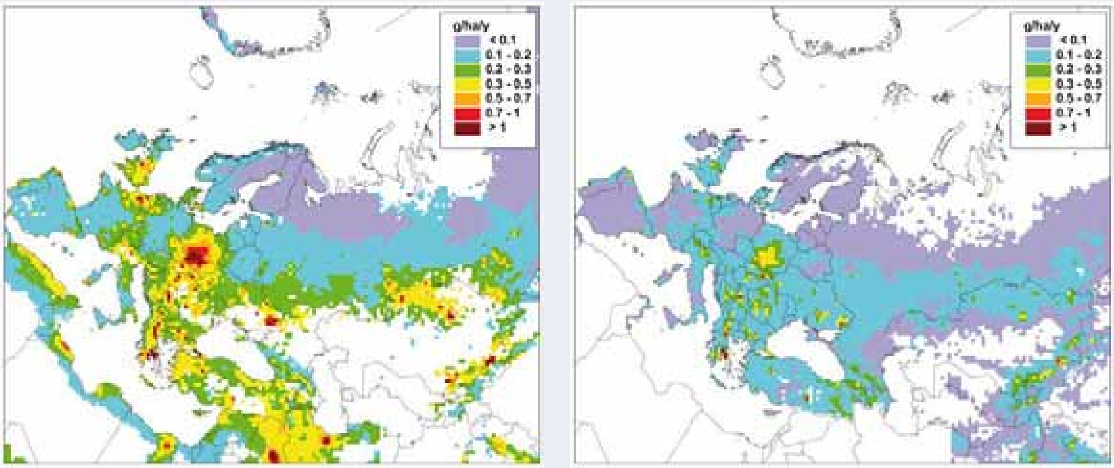

Figure 7-8 Total annual depositions of mercury to coniferous forests (a) and to croplands (b), in 2006.

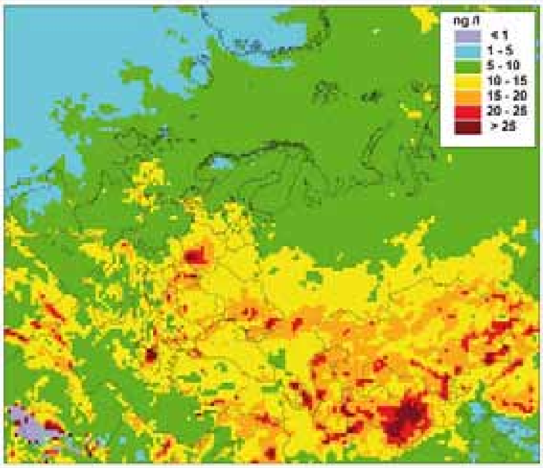

Figure 7-9 Annual mean mercury concentrations in precipitation, in 2006.

If critical loads of Hg would also include the critical concentration in precipitation (not in this chapter), then calculations of exceedances would also require information on concentrations in precipitation. The spatial distribution of annual mean mercury concentrations in precipitation is shown in Figure 7-9. It is assumed that the amounts of precipitation and, hence, the concentrations in precipitation, do not depend on type of land cover. Within most of Europe, the concentrations in precipitation range from 5 to 15 ng L^{-1}. The relatively high concentrations observed for Poland, Hungary and the eastern part of the Ukraine, are explained by considerable emission sources affecting these countries. In the Central Asian region, the concentrations of mercury in precipitation are relatively high, compared with European levels. There are two main reasons for this; the estimated emission values for the Central Asian countries are comparable with those of Europe, and low amounts of precipitation imply higher concentrations of mercury.

Pollution levels, simulated by the MSCE-HM model, are more reliable for the European region than for Central Asia. Firstly, the availability of emission data for the Central Asian region is insufficient. Secondly, actual measured data from this region are not available, which makes verification of the modelling results complicated, compared to those from Europe. Therefore, the calculated depositions to the Central Asian region should be considered to be preliminary, from the viewpoint of the evaluation of critical-load exceedances.

7.6 Exceedances

For each of the metals, the Average Accumulated Exceedance (AAE) was calculated. The AAE is the area-weighted average exceedance over all ecosystems in an EMEP grid cell. Exceedances of critical loads for cadmium are limited to areas with high depositions and low critical loads, such as parts of central Europe and the Ruhr Area in Germany (Figure 7-10). Critical loads for lead are exceeded within large parts of central and south-eastern Europe, where depositions are high. The exceedances along the European Mediterranean coast are also noticeable, caused by relatively high lead depositions.

For Hg, critical load exceedances occur within almost all of Europe and northern Asia (Figure 7-11). Since the regional variability in critical loads for Hg is limited, the magnitude of the average accumulated exceedances for Hg strongly follows the deposition patterns with the highest exceedances within central and southern Europe, southern Russia and the New Independent States.

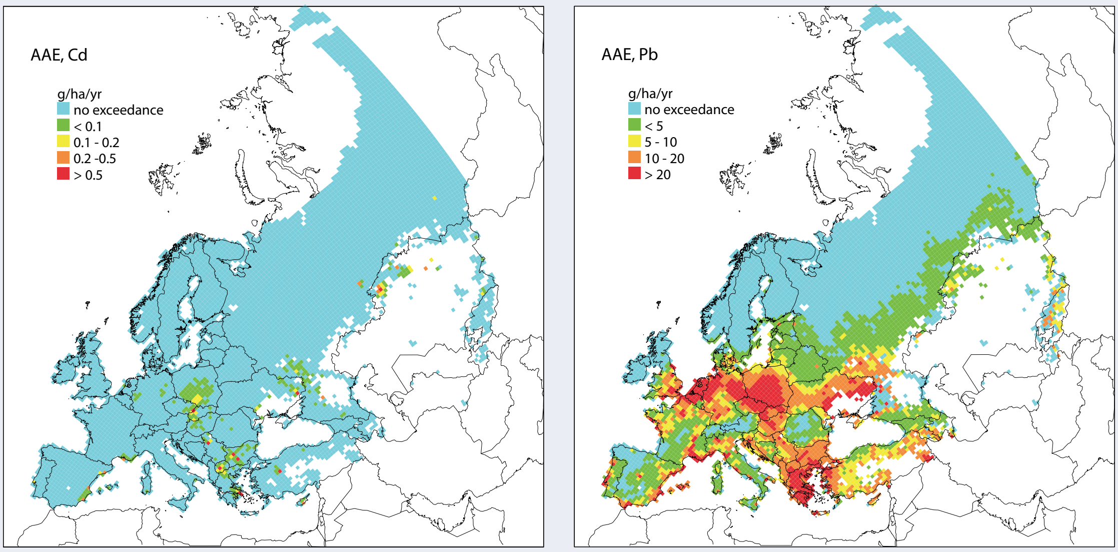

Figure 7-10 Average accumulated exceedances for Cd and Pb.

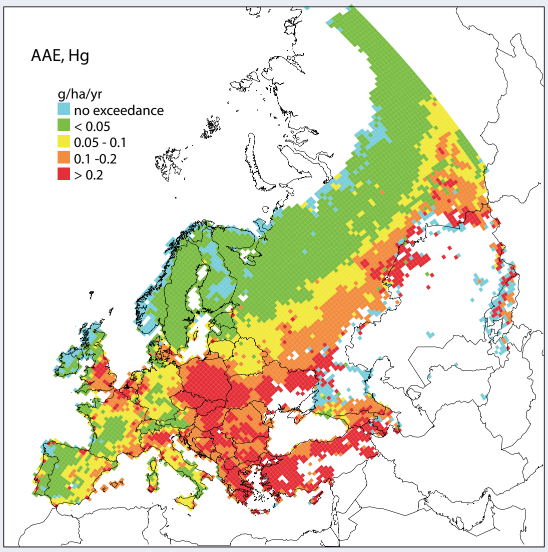

Figure 7-11 Average accumulated exceedance for Hg.

7.7 Conclusions

The new CCE background database formed a proper basis for calculating critical loads of Cd, Pb and Hg. Ranges and patterns of Critical loads in this study correspond with the heavy metal critical-load maps for Europe, provided by Slootweg et al. (2005). The uncertainties in the calculated critical loads are substantial, because critical concentrations are calculated as a function of pH, and DOC concentrations are defined as a function of soil type, but both pH and DOC are known to vary greatly, within soil types as a function of, for instance, climate and parent material.

The testing of the modelled deposition by comparing it to the measurements, shows that the correspondence between the model and measurements is generally in the range of 25 to 40%, although at some measurement stations the deviation is larger. Exceedance maps show that critical loads of Cd are exceeded in very limited areas. Critical loads for lead are exceeded in large parts of western, central and southern Europe, as well as in southern Russia. The area with critical load exceedances for Hg strongly resembles that for Pb; large areas within Europe and Russia and the New Independent States show exceedances. Exceedances for Cd are similar to earlier critical load exceedances, in 2000, calculated with the European background database (Slootweg et al., 2005). However, in this study, exceedances for Pb are somewhat higher than those reported in Slootweg (2005), due to updates and improvements in the background database and in deposition estimates.

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PART II
National Focal Centre Reports

This part consists of the reports on national data on critical loads and dynamic modelling calculations submitted to the Coordination Centre for Effects by the National Focal Centres (NFCs) following the CCE call for data of November 2007. The reports have been edited for format and clarity, but have not been reviewed.

Russia indicated the validity of their latest national report which can be found in the CCE Progress Report 2007.

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Status

In response to the call for data of November 2007 a new dataset of critical loads and dynamic modelling is provided. Three different approaches for the calculation of critical loads are applied. *Critical Loads of acidity* (CLmaxN&S) and dynamic modelling output are calculated using the VSD model and soil data from 496 soil monitoring sites. The calculation of *Critical Loads of nutrient nitrogen* (CLnutN) is also done using the mass balance approach, but is based on the Corine Landcover 2000 dataset and other maps instead of soil monitoring sites. This is possible because of the reduced data requirements of the CLnutN calculation and it allows the production of CLnutN maps. At least, the *Empirical Critical Loads* dataset (CLempN) is also based on the Corine Landcover 2000 dataset.

Every record of the *inputs/CLdata-* and the *EmpNload*-tables has a unique link to the *ecords*-table with information describing the location (one-to-one relation). Due to restrictions of the data structure, overlapping areas of the three approaches do not point to a common location record, so summing up ecosystem areas is meaningful only within one of the three approaches.

Critical loads of acidity

Data Sources

Soils: Soil information is based on the Austrian Forest Soil Inventory from the Austrian Federal Office and Research Centre for Forests (Forstliche Bundesversuchsanstalt 1992). About 500 sample plots were investigated in an 8.7 x 8.7 km grid between 1987 and 1990. Most of the soil input parameters to calculate critical loads and target loads were taken from this dataset. The data are part of the Soil Information System BORIS, maintained at the Federal Environment Agency.

Nutrient uptake: Information on biomass uptake is derived from data of the Austrian Forest Inventory, sampled by the Austrian Federal Office and Research Centre for Forests - BFW (Schieler et al. 2001). Mean harvesting rates for the years from 1986 to 1996 were aggregated on EMEP grid cell basis. Grid cells with too few sample points were combined with neighbouring cells. Base cation and nitrogen contents were taken from Jacobsen et al. 2002. No nutrient uptake takes place at unmanaged protection forests.

Ecosystem: Four forest ecosystem types have been investigated according to EUNIS classification: G1 (Fagus sylvatica, Quercus robur), G3 (Picea abies, Pinus sylvestris, Larix decidua), G4 and G3.1B, which is used to indicate unmanaged protection forests. The ecosystem area was identified by dividing the known ecosystem area per grid cell (from forest inventory) by the number of soil inventory points located in this ecosystem type.

Depositions: New sulphur and nitrogen deposition time series provided by the CCE 2008 ('Review of the 1999 Gothenburg Protocol', Executive Body for the Convention [2007], ECE/EB.AIR/WG.5/2007/7); Base cation depositions: van Loon et al. 2005

Calculation Method

The calculations and assumptions are generally in accordance with the Mapping Manual (ICP M&M (2004) and the CCE Status Reports. A detailed description of the parameters and the data and methods used for their derivation is given in table AT-1.

The Access version of VSD was used for critical loads calculation and dynamic modelling. For the cation exchange the Gapon model was used, the exchange constants were calibrated. Theta was set to be 0.3, CNmin and CNmax were set to be 10 resp. 40. Oliver constants for the organic acid dissociation model were set to be 4.5, 0, 0.

Base cations were lumped together in the Ca column for weathering and uptake. Due to the lack of spatial distributed information on organic acids, default values for all records were used.

Calcareous soils occur at 30% of the sample points representing about 40% of the ecosystem area.

Table AT-1. Data description, methods and sources for the CL of acidity calculation

Variable	Explanation and Unit	Description
CLmaxS	Maximum critical load of sulphur (eq ha^{-1} a^{-1})	calculated by VSD
CLminN	Minimum critical load of nitrogen (eq ha^{-1} a^{-1})	calculated by VSD
CLmaxN	Maximum critical load of nitrogen (eq ha^{-1} a^{-1})	calculated by VSD
nANCcrit	The quantity –ANCle(crit) (eq ha^{-1} a^{-1})	calculated by VSD
crittype	Chemical criterion used	used: molar Al/Bc (1)
critvalue	Critical value for the chemical criterion	used: 1
thick	Thickness of the soil (m)	mostly 0.5 m, sometimes less, depending on soil inventory data
bulkdens	Average bulk density of the soil (g cm^{-3})	Mapping Manual 6.4.1.3 eq. 6.27
Cadep	Total deposition of calcium (eq ha^{-1} a^{-1})	total depositions for forest ecosystems (van Loon et al. 2005)
Mgdep	Total deposition of magnesium (eq ha^{-1} a^{-1})	total depositions for forest ecosystems (van Loon et al. 2005)
Kdep	Total deposition of potassium (eq ha^{-1} a^{-1})	total depositions for forest ecosystems (van Loon et al. 2005)
Nadep	Total deposition of sodium (eq ha^{-1} a^{-1})	total depositions for forest ecosystems (van Loon et al. 2005)
Cldep	Total deposition of chloride (eq ha^{-1} a^{-1})	Nadep * 1.166 (Nadep from van Loon et al. 2005)
Bcwe	Weathering of base cations (eq ha^{-1} a^{-1})	Mapping Manual 5.3.2.3, eq. 5.39; Table 5-14 (WRc = 20 for calcareous soils; factor 0.8 for Na reduction)
Bcupt	Net growth uptake of base cations (eq ha^{-1} a^{-1})	[average yearly yield rate * base cation content], data from Austrian forest inventory, base cation contents from Jacobsen et al. 2002 (no uptake from unmanaged protection forests)
Qle	Amount of water percolating through the root zone (mm a^{-1})	Hydrological Atlas of Austria-v.2
lgKAlox	Equilibrium constant for the Al-H relationship (log10)	[9.8602 - 1.6755 * log(OM) for 1.25 < OM < 100; 9.7 for OM < 1.25]; SAEFL 2005 (OM = Organic Matter [%])
expAl	Exponent for the Al-H relationship	used: 3 (gibbsite equilibrium)
pCO2fac	Partial CO_2-pressure in soil solution as multiple of the atmospheric CO_2 pressure (-)	[log10pco2 = -2.38 + 0.031 * Temp (°C)]; atmospheric CO2 pressure = 0.00037 atm; equation recommended by CCE
cOrgacids	Total concentration of organic acids (m*DOC) (eq m^{-3})	used: 0,01 (recommended by Max Posch)
Nimacc	Acceptable amount of nitrogen immobilised in the soil (eq ha^{-1} a^{-1})	decreasing from 5 kg N in the highlands (< 5° C mean Temp) to 1 kg N in the lowlands (> 8° C mean Temp); see German NFC Report in Posch et al. 2001, p.142, Table DE-7
Nupt	Net growth uptake of nitrogen (eq ha^{-1} a^{-1})	[average yearly yield rate * N content], data from Austrian forest inventory, N contents from Jacobsen et al. 2002
fde	Denitrification fraction (0≤fde<1) (–)	from 0.1 (dry) to 0.7 (wet) according to soil moisture class; information from soil inventory
CEC	Cation exchange capacity (meq kg^{-1})	information from soil inventory; calibrated to pH 6.5 (Mapping Manual 6.4.1.3 eq. 6.29)
bsat	Base saturation (–)	information from soil inventory
yearbsat	Year in which the base saturation was determined	year of soil inventory (1987-1990)
lgKAlBc	Exchange constant for Al vs. Bc (log10)	calibrated by VSD; initial value 0
lgKHBc	Exchange constant for H vs. Bc (log10)	calibrated by VSD; initial value 3
Cpool	Initial amount of carbon in the topsoil (g m^{-2})	[thick * bulkdens * Corg(%) * 10 000]; for mineral topsoil (0-10 cm) + organic layer; information from soil inventory
CNrat	C/N ratio in the topsoil	Cpool / Npool
yearCN	Year in which the CNratio and Cpool were determined	year of soil inventory (1987-1990)
Measured	On-site measurements included?	all sites: ICP-Forests (1)
EUNIScode	EUNIScode of ecosystem	information from soil inventory: G1, G3, G4, G3.1B (unmanaged protection forests)
Protection	Type of nature protection (SAC, SPA...)	status unknown at all sites (-1)
EcoArea	Area of the ecosystem within the EMEP grid cell (km^2)	calculated from Austrian forest inventory data (virtual area)

Critical loads of nutrient nitrogen

Data Sources and Calculation Method

The calculation of CLnutN is based on about 18 000 forest patches of the Austrian Corine Landcover dataset. Generally data sources and calculation method are comparable to the *CL of acidity method*, although some changes were necessary due to the different spatial approach and the data availability.

Denitrification: The denitrification fraction is based on the soil type units of the soil map 1:1 000 000 of the Hydrological Atlas of Austria as no better spatial distributed information on soil moisture in forests is available. The assignment of fde-values to soil types is based on an analysis of soil moisture classes within soil types of the Austrian forest soil inventory dataset.

Table AT-2. Assignment of fde-values to soil type units

Soil type unit	fde
Rendzina, Lithosol, orthic Luvisol	0.3
Chernosem, Cambisol, gleyic Luvisol, Regosol, Podzol, Solonetz	0.4
Fluvisol, Planosol	0.5
Histosol	0.7

Leaching: As the acceptable leaching does not depend on the precipitation surplus and the critical nitrogen concentration but on the altitude (see Swiss NFC Report in Posch et al. 2001), the cNacc is back-calculated from the acceptable leaching and Q leading to very high (at low Q values) and very low (at high Q values) acceptable nitrogen concentrations.

Ecosystem: EUNIS type G3.1B, which is used to indicate unmanaged protection forests, cannot be identified within the Corine Landcover dataset. On the other hand, the area per EMEP grid cell is known from the forest inventory. As protection forests are mostly located at higher altitudes, G3-patches (coniferous forests) were successively added to the G3.1 area, moving downwards from the highest parts of the EMEP grid cell , until the area of protection forest reported in the Austrian Forest Inventory was reached.

A description of the parameters and the data and methods used for their derivation is given in table AT-3.

Table AT-3. Data description, methods and sources for the CLnutN calculation

Variable	Explanation and Unit	Description
CLnutN	Critical load of nutrient nitrogen (eq ha^{-1} a^{-1})	Mapping Manual 5.3.1.1, eq. 5.5
cNacc	Acceptable (critical) N concentration (meq m^{-3})	back-calculated from Nleacc and Qle
Nleacc	Acceptable nitrogen leaching (eq ha^{-1} a^{-1})	decreasing from 4 kg N in the lowlands (500 m a.s.l.) to 2 kg N at 2000 m a.s.l. (see Swiss NFC Report in Posch et al. 2001)
Qle	Amount of water percolating through the root zone (mm a^{-1})	Hydrological Atlas of Austria-v.2
Nimacc	Acceptable amount of nitrogen immobilised in the soil (eq ha^{-1} a^{-1})	decreasing from 5 kg N in the highlands (< 5° C mean Temp) to 1 kg N in the lowlands (> 8° C mean Temp); see German NFC Report in Posch et al. 2001, p.142, Table DE-7
Nupt	Net growth uptake of nitrogen (eq ha^{-1} a^{-1})	[average yearly yield rate * N content], data from Austrian forest inventory, N contents from Jacobsen et al. 2002
fde	Denitrification fraction (0≤fde<1) (-)	from 0.1 (dry) to 0.7 (wet) according to the soil type of the soil map 1:1 Mio. of the Hydrological Atlas of Austria-V.2
Measured	On-site measurements included?	all sites: no measurements (0)
EUNIS code	EUNIS code of ecosystem	Corine Landcover 2000; G1, G3, G4, G3.1B (unmanaged protection forests - information from Austrian forest inventory)
Protection	Type of nature protection (SAC, SPA, ...)	status unknown at all sites (-1)
EcoArea	Area of the ecosystem within the EMEP grid cell (km2)	Corine Landcover 2000 patch size

Empirical Critical Loads

Data Sources and Calculation Method

The Austrian Corine Landcover 2000 dataset is the main data source for this study. Additionally, the Austrian mire conservation database is used to update the small-scale CLC2000 data with mire, bog and fen habitats.

EUNIS-codes are applied and CLempN values are assigned to the habitats according to the recommendations made in the Mapping Manual. The mean value of the recommended range is used as CL (table AT-4), no further adaptation to abiotic factors according to Table 5-2 of the Mapping Manual is done due to the restricted data availability and the poor knowledge of the quantitative influence of these factors.

Table AT-4. Ecosystem, Corine2000 code, EUNIS code, recommended CL range and applied CLemp(N) value

Ecosystem	CLC2000	EUNIS	CLNrange	CLemp(N)
Raised and blanket bogs	a)	D1	5-10	7,5
Oligotrophic fens	a)	D2.1	10-15	12,5
Mesotrophic fens	a)	D2.2	15-20	17,5
Eutrophic fens	a)	D4.1	15-25	20
Mountain hay meadows	321	E2.3	10-20	15
Moss and lichen dominated mountain summits	333	E4.2	5-10	7,5
Broadleaved deciduous woodland	311	G1	10-20	15
Coniferous woodland	312, 322	G3	10-20	15
Mixed deciduous and coniferous woodland	313, 324	G4	10-20	15

a) Ecosystem information from Austrian mire conservation database

Table AT-5. Data description, methods and sources for the CLempN calculation

Variable	Explanation and Unit	Description
CLempN	Empirical critical load of nitrogen (eq ha-1 a-1)	values used: see table AT-4
EUNIS code	EUNIS code of ecosystem	Corine Landcover 2000; see table AT-4
Protection	Type of nature protection (SAC, SPA, ...)	status unknown at all sites (-1)
EcoArea	Area of the ecosystem within the EMEP grid cell (km2)	Corine Landcover 2000 patch size

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Critical load and dynamic modelling data sources and methods

Flanders

Critical loads of acidity and nutrient nitrogen for ecosystems in Flanders were calculated using the Simple Mass Balance (SMB) model as described in the Mapping Manual (UBA 2004). Compared to the results presented in the CCE Status Report 2003 (Posch et al. 2003), critical loads for forests were updated and the first dynamic modelling results were produced using the VSD model (Staelens et al. 2006). For grassland and heather (Meykens et al. 2001), no updates were made compared to Posch et al. (2003) and hence, data sources and methods for these ecosystems are not repeated here.

Critical loads for forests

Critical loads of acidity and of nutrient nitrogen were calculated for 1438 forest locations in Flanders. Table BE-1 summarizes the methods and data sources used. The critical loads of acidity were not corrected for the seasalt derived sulphur deposition.

Table BE-1. Crititical loads and dynamic modelling data sources and methods for forests in Flanders

Parameter	Term	Unit	Description
Critical loads of acidity	$CL_{max}(S)$	eq ha^{-1} a^{-1}	UBA (2004) Eq. 5.22
	$CL_{min}(N)$	eq ha^{-1} a^{-1}	UBA (2004) Eq. 5.25
	$CL_{max}(N)$	eq ha^{-1} a^{-1}	UBA (2004) Eq. 5.26
Critical load of nutrient N	$CL_{nut}(N)$	eq ha^{-1} a^{-1}	UBA (2004) Eq. 5.5
Critical leaching of acid neutralising capacity	$ANC_{le,crit}$	eq ha^{-1} a^{-1}	Critical molar ratio of Al:Bc = 1
Acceptable N leaching	$N_{le,acc}$	eq ha^{-1} a^{-1}	100 eq ha^{-1} a^{-1}
Thickness of the root zone	z	m	0.5 m
Average bulk density of the soil	bulkdens	g cm^{-1}	Data of 83 Level I and II forest plots
Bc, Na^+ and Cl^- deposition	Bc_{dep}, Na_{dep}, Cl_{dep}	eq ha^{-1} a^{-1}	Based on throughfall data of five Level II forest plots (2000-2004) corrected for canopy exchange
Weathering of base cations	Bc_w	eq ha^{-1} a^{-1}	Soil type - texture approximation
Net growth uptake of Bc and N	Bc_u, N_u	eq ha^{-1} a^{-1}	Tree species specific growth and nutrient content data from Belgian and Dutch literature
Precipitation surplus	Q_{le}	mm a^{-1}	Based on interpolated rainfall data and species specific interception and evapotranspiration
Exponent for Al-H relationship	expAl	-	3 (i.e. assumed gibbsite equilibrium)
Equilibrium constant for the Al-H relationship (log_{10})	$log(K_{Alox})$	$(eq\ m^{-3})^{1-expAl}$	8.5 for mineral soils 7.5 for organic soils
Partial CO_2 pressure in soil solution	pCO_2	-	15 times the atmospheric CO_2 pressure
Total concentration of organic acids	m·DOC	eq m^{-3}	0.09 (based on a mean measured [DOC] of 1.35 mol C m^{-3} and *m* value in de Vries et al. (2001))
Dissociation constant organic acids	K_1	mol L^{-1}	Oliver model, UBA (2004) Eq. 5.47
Acceptable N immobilisation in soil	$N_{i,acc}$	eq ha^{-1} a^{-1}	Method of Van Hinsberg and de Vries (2003)
Denitrification fraction	f_{de}	-	Based on soil type and drainage status
Cation exchange capacity (at pH 6.5)	CEC	meq kg^{-1}	Data of Level I and II forest plots, standardized at pH 6.5 by UBA (2004) Eq. 6.28 and 6.29
Base saturation (at pH 6.5)	Bsat	-	Exchangeable base cation data of Level I and II forest plots (as a fraction of the CEC at pH 6.5)
Exchange constant for Al vs. Bc (log_{10})	$lg(k_{AlBc})$	$(mol\ L^{-1})^{1/6}$	Calibrated by the VSD model
Exchange constant for Al vs. H (log_{10})	$lg(k_{HlBc})$	$(mol\ L^{-1})^{-1/2}$	Calibrated by the VSD model
Actual amount of C in top soil (0-20 cm)	Cpool	g m^{-2}	Data of Level I and II forest plots
C/N ratio in topsoil (0-20 cm)	CNrat	-	Data of Level I and II forest plots
Nitrogen and S deposition	N_{dep}, S_{dep}	eq ha^{-1} a^{-1}	Time series from Schöpp et al. (2003) for the EMEP 50 x 50 km grid as provided by the CCE
Cation exchange model	-	-	Gapon exchange model
Maximum and mininum C/N ratio in topsoil	CN_{max} CN_{min}	-	40 12
Soil moisture content	theta	m^3 m^{-3}	0.12 (VSD is unsensitive to this parameter)

The following changes were made compared to the former critical load calculations:
Including the dissocation of organic acids and bicarbonate leaching in the SMB model.
Mineral weathering estimated based on both the clay and sand soil content.
Including atmospheric deposition of Na^+ and Cl^-.
Updated (lower) atmospheric deposition of K^+, Ca^{2+} and Mg^{2+} based on recent throughfall measurements (2000-2004) that were corrected for canopy exchange.
Precipitation surplus calculated based on data from more weather stations and on actual evapotranspiration values per tree species.
Long-term acceptable nitrogen immobilisation calculated by accepting a change of 0.2% of N in the organic matter pool in the upper soil layer (0-20 cm) during a period of 100 years (cf. Van Hinsberg and de Vries 2003). When no data on carbon soil pool were available, the median value of a representative subset of forest soils was used.

Dynamic modelling for forests

Dynamic modelling has been executed for 83 non-calcareous Level I and II forest plots in forests using the Microsoft Office Access version of the Very Simple Dynamic (VSD) model. According to the critical load calculations, these 83 plots can be considered as a representative subset of the larger critical load database (n = 1438) for Flanders. The plots were chosen because of the availibility of recent soil data needed for dynamic modelling (CEC, base saturation, C pool and C/N ratio).

Table BE-1 describes the main data sources used in this first attempt to the dynamic modelling of forest soil acidification and recovery for the Flemish region. Soil solution concentrations measured in five Level II plots were used to determine an empirical relationship between aluminum- and proton concentrations. However, when the derived parameters *expAl* and pK_{Alox} were used to calculate target loads, the criterion Al:Bc = 1 could never be reached before 2100, and therefore a gibbsite equilibrium was assumed. Critical loads of acidity were much less sensitive to the assumed relationship between *pAl* and *pH* than target loads of acidity.

Calibration of initial C:N ratio and exchange constants are based on the EMEP nitrogen and sulphur deposition time series provided by the CCE, as recommended, even though the EMEP sulphur depositions were on average 40% higher during the period 1990-2004 than a regional emission-based deposition model (OPS). For reasons of consistency, the same - relatively low - base cation depositions as for the critical load calculations were used for dynamic modelling throughout the entire simulation period (1880-2100). However, the past higher sulphur depositions onto forest vegetation likely were accompanied by higher calcium depositions than at present. Consequently, more reliable results might be accompliced if EMEP deposition time series would also be available for base cations, and when a time dependent base cation deposition could be taken into account in the Access version of the VSD model.

Wallonia

Maps have been produced for coniferous, deciduous and mixed forests.

Mapping procedure Wallonia

Digitized maps with a total of 1900 ecosystems were overlaid by a 5 x 5 km² grid to produce the resulting maps. In Wallonia, the critical value given for a grid cell represents the average of the critical values weighted by their respective ecosystem area (coniferous, deciduous or mixed forests).

Calculation methods & results Wallonia

A. Forest Soils

Calculation methods

Critical loads for forest soils were calculated according to the method as described in UBA (1996) and Manual for Dynamic Modelling of Soil Response to Atmospheric Deposition (2003):

$CLmax(S) = BCwe + BCdep – BCu – ANCle(crit)$

$CLmax(N) = Ni + Nu + CLmax(S)$

$CLnut(N) = Ni + Nu + Nle + Nde$

$ANCle(crit) = -Qle ([Al^{3+}] + [H^+] - [RCOO^-])$

Where :

$[Al^{3+}] = 0.2\ eq/m^3$

$[H^+]$ = concentration of $[H^+]$ at the pH critique (table BE-4).

$[RCOO^-]$= 0.044 molc/molC x DOCmeasured (table BE-4)

The equilibrium $K = [Al^{3+}]/[H^+]3$ criterion: The Al^{3+} concentration was estimated by 1) experimental speciation of soil solutions to measure rapidly reacting aluminium, Alqr (Clarke et al.,1992); 2) calculation of Al^{3+} concentration from Alqr using the SPECIES speciation software. The K values established for 10 representative Walloon forest soils (table BE-2) were more relevant than the gibbsite equilibrium constant recommended in the manual (UBA, 1996). The difference between the estimated Al^{3+} concentrations and concentration that causes damage to root system (0.2 eq Al^{3+}/m^3 ; de Vries et al., 1994) gives the remaining capacity of the soil to neutralise the acidity.

The Tables BE-2 and BE-3 summarise the values given to some of the parameters.

Table BE-2 : Aluminium equilibrium and weathering rates calculated for Walloon soils.

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

(1) deciduous or (2) coniferous forest

Table BE-3. Constants used in critical loads calculations in Wallonia

Parameter	Value
Ni	5.6 kg N ha^{-1} yr^{-1} coniferous forest
	7.7 kg N ha^{-1} yr^{-1} deciduous forest
	6.65 kg N ha^{-1} yr^{-1} mixed forest
Nle (acc)	4 mg N L^{-1} for coniferous forest
	6.5 mg N L^{-1} for deciduous forest
	5.25 mg N L^{-1} for mixed forest
Nde	Fraction of (Ndep – Ni – Nu)

Soils : In Wallonia, 47 soil types were distinguished according to the soil associations map of the Walloon territory, established by Maréchal and Tavernier (1970). Each ecosystem is characterised by a soil type and a forest type.

Weathering rate : In Wallonia, the base cation weathering rates (BCwe) were estimated for 10 different representative soil types (table BE-2) through leaching experiments. Increasing inputs of acid were added to soil columns and the cumulated outputs of lixiviated base cations (Ca, Mg, K, Na) were measured. Polynomial functions were used to describe the input-output relationship. To estimate BCwe, an acid input was fixed at 900 eqH^{+} ha^{-1} yr^{-1} in order to keep a long term balance of base content in soils.

Nle = Qle cN(acc)

The flux of drainage water leaching, Qle , from the soil layer (entire rooting depth) was estimated from lysimetric measurement on 10 different representative soil types (table BE-4) (Catholic University of Louvain, 2005).

Table BE-4 : Flux of drainage water through entire root layer Qle, concentration of organic acids ($RCOO^{-}$) and pH critique in Walloon soils.

Sites	Soil types	RCOO-	pH crit	Qle (m yr^{-1})
		(eq/m^3)		at 0,5m
Bande (1-2)	Podzol	0.103	3.95	0,138
Chimay (1)	Cambisol	0.038	4.10	0,046
Eupen (1)	Cambisol	0.105	4.36	0,045
Eupen (2)	Cambisol	0.094	3.70	0,045
Hotton (1)	Cambisol	0.031	4.38	0,108
Louvain-la-Neuve (1)	Luvisol	0.099	4.17	0,039
Meix-dvt-Virton (1)	Cambisol	0.037	4.35	0,049
Ruette (1)	Cambisol	0.007	4.47	0,045
Transinne (1)	Cambisol	0.078	4.41	0,053
Willerzie (2)	Cambisol	0.038	4.37	0,044

(1) deciduous or (2) coniferous forest

Precipitation surplus : The actual methodology can not be compared with the previous methodology because the definition of the precipitation surplus is modified. In the previous methodology the surplus was defined as the total amount of water leaving the root zone (total run off). In the present methodology the precipitation surplus doesn't take into account of the horizontal flux but considers only the amount of water percolating through the root zone (mm a^{-1}). In forest growing on abrupt locations, a non negligible fraction of the precipitation runs off on the top soil.

Net growth uptake of Base cations and nitrogen : In Wallonia, the net nutrient uptake (equal to the removal in harvested biomass) was calculated using the average growth rates measured in 25 Walloon ecological territories and the chemical composition of coniferous and deciduous trees. The chemical composition of the trees (Picea abies, fagus sylvatica, Quercus robus, Carpinus betulus) appears to be linked to the soil type (acidic or calcareous) (Duvigneaud et al., 1969; Bosman et al., 2001; Unité des Eaux et Forêts, may 2001).

The net growth uptake of nitrogen ranges between 266 and 822 eq ha^{-1} yr^{-1}, while base cations uptake values vary between 545 and 1224 eq ha^{-1} yr^{-1} depending on trees species and location in Belgium.

Base cations deposition: In Wallonia, actual throughfall data collected in 8 sites, between 1997 and 2002, were used to estimate BCdep parameters. The marine contribution to Ca^{2+}, Mg^{2+} and K^{+} depositions was estimated using sodium deposition according to the method described in UBA (1996). The BCdep data of the 8 sites was extrapolated to all Walloon ecosystems as a function of the location and the tree species.

Results

In Wallonia, The highest CL values were found in calcareous soils under deciduous or coniferous forests. The measured release rate of base cations from soil weathering processes is high in these areas, and thus provides a high long-term buffering capacity against soil acidification.
More sensitive forest ecosystems are met on sandy-loamy or loamy gravelly soils. The lowest CLnut values were found in Ardennes. In this zone, Picea abies L.Karts. frequently show magnesium deficiency symptoms, which have been exacerbated by atmospheric pollution (Weissen et al, 1990).

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Modelled critical loads

Data sources

This report presents recent results of the team-work of the Bulgarian experts of Executive Environmental Agency and the Bulgarian scientific team as parts of the ICP Modelling and Mapping on the dynamic assessment of exceedances of critical loads for acidifying pollutants in Europe.

The report is entirely based on the “Calculation and mapping of critical loads for sulphur, nitrogen, acidity and heavy metal for forest ecosystems in Bulgaria”, Report 998/22.12.2006, Ministry of Environment, Sofia, 96 pp. included in the list of references.

Current critical loads data for acidification and eutrophication are described as well justifying methods and data applied.

Critical loads of acidifying sulphur and nitrogen are calculated for main forest tree species using the Steady State Mass Balance method in accordance with the latest recommendations provided in the last version of the Mapping Manual (UBA, 2004).

The database involve maximum critical loads of sulphur (*Manual, equation* 5.22), maximum critical loads of nitrogen (*Manual, equation* 5.26), minimum critical loads of nitrogen (*Manual, equation* 5.25), nutrient nitrogen (*Manual, equation* 5.5) and all related data.

Critical loads have been calculated for all major tree species using soil data base of the content of the organic mater (%), the clay content for the fraction 0,01 mm in the soil (%), soil bulk density, cation exchange capacity CEC, Base saturation, C / N ratio and the pH of the soil. in grid cells of 16 km / 16 km (Ignatova et al., 2001).

Runoff of water under root zone has been measured in grid cells of 10 / 10 km for the entire country (Kehayov, 1986).

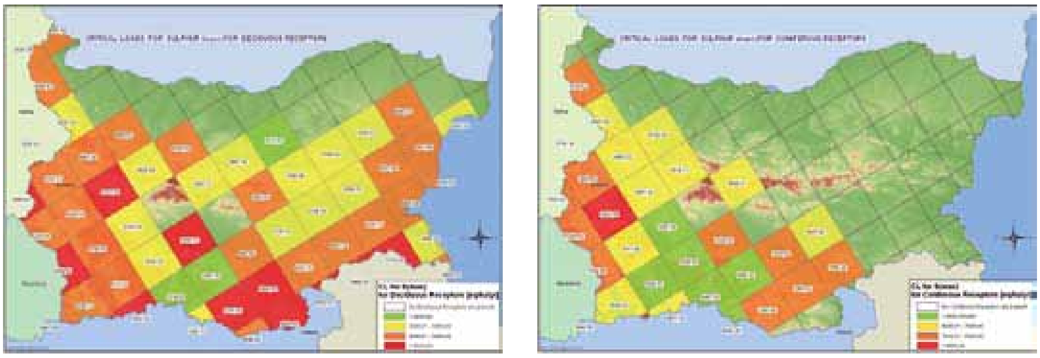

Figure BG-1. CL max S for broadleaved (left) and coniferous (right) forests in Bulgaria.

A network of pened collectors for atmospheric deposition by precipitation have been used for base cations, sulphur and nitrogen depositions.

Nitrogen and base cations net uptake rates are obtained by multiplying the element contents of the stems (Na, Ca, K, Mg and N) with annual harvesting rates (Ignatova et al., 1997). Data on biomass removal for forests have been derived from the State Forests 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 all., 2001)

In the absence of more specific data on the production of basic cations through mineral weathering for most of 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 contents of the Bulgarian forest soils (UBA, 1996).

Chemical criterion used is a molar ratio [Al] / [Bc] = 1 (Manual, equation 5.31). Identifiers of the site for critical loads calculation of acidifying nitrogen and sulphur, and the integers in the submission of the empirical critical loads of nitrogen are not identical because of different number of sites under consideration in two submissions but they correlate each to other by the EMEP grid cells indices and geographical coordinates.

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 km² grid cell when the forest is a mixture of both tree types, in accordance with the area fractions of the tree species.

National maps

Soil type information - soil map of Bulgaria (M 1 :1 000 000), FAO classification;
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

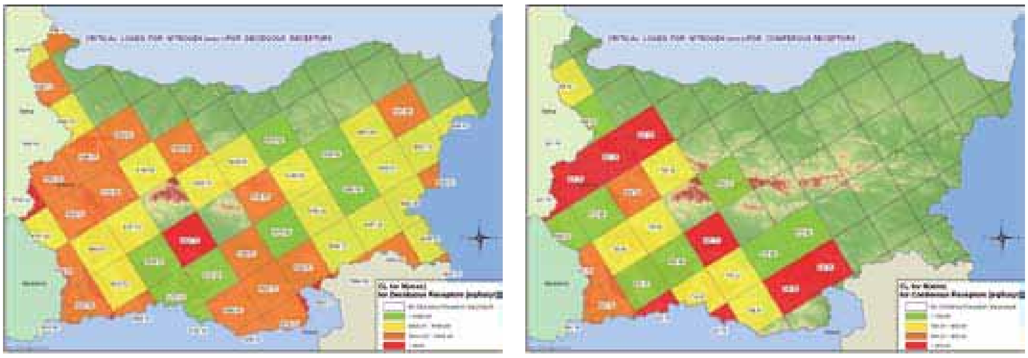

Figure BG-2. CL maxN for broadleaved (left) and coniferous (right) forests in Bulgaria.

Calculated values for CLmaxS vary between 4489 eq ha^{-1} a^{-1} and 8052 eq ha^{-1} a^{-1} for coniferous, and between 2774 eq ha^{-1} a^{-1} and 5687 eq ha^{-1} a^{-1} for broadleaved forests (Figure BG-1). CLmaxN (5411 eq ha^{-1} a^{-1} and 8625 eq ha^{-1} a^{-1} for coniferous, and between 3275 eq ha^{-1} a^{-1} and 6243 eq ha^{-1} a^{-1} for broad-leaved forests) are similar but a little higher than CLmaxS. (Figure BG-2). On the contrary , critical load values for nutrient nitrogen are lower and ranged between 581 and 941 eq ha^{-1} a^{-1} for conifer-ous, and between 398 and 776 eq ha^{-1} a^{-1} for deciduous forests. The lowest critical loads are calcu-lated for CLminN (between 573 and 926 eq ha^{-1} a^{-1} for coniferous, and between 394 and 768 eq ha^{-1} a^{-1} for deciduos forests).

In general, all calculated critical loads values for all over the country are higher for coniferous forests than for brood leaved ones, due to the lower mean values of critical loads parameters used for the computing (base cations weathering, deposition and uptake).

For the minimum critical loads of nitrogen as well as the critical loads of nutrient nitrogen the vari-ability of computed individual data is much smaller, which reflects on the average values (789 eq ha a for coniferous ecosystems for minimum critical loads of nitrogen with 797 eq ha and for broad-leaved ones, and 801 eq ha a for coniferous for nutrient nitrogen against 544 eq ha a for broad leaved forests) (Table BG-1).

Table BG-1. Average, maximum and minimum values of critical loads of sulphur, nitrogen ass alkalinity for broad-leaved and coniferous forests in Bulgaria (in eq ha^{-1} a^{-1}).

	Coniferous			Broadleaved		
	Min	Max	Average	Min	Max	Average
CLmaxS	4489	8052	6644	2774	5687	4394
CLminN	573	926	789	394	768	534
CLmaxN	5411	8625	7433	3778	6243	4928
CLnutN	581	941	797	398	776	544
nANCcrit	2712	4851	4006	1700	3455	2653

Critical loads of maximum nitrogen and sulphur for 2007 - Bulgaria (Figure BG-3).

Data in the current report is based on the proposal of the working group in 2007 (France and England) for the use of permanently open collectors for collecting dry and wet depositions during the whole year, which approaches the real deposition on a unit of ground surface in the forest ecosystems.

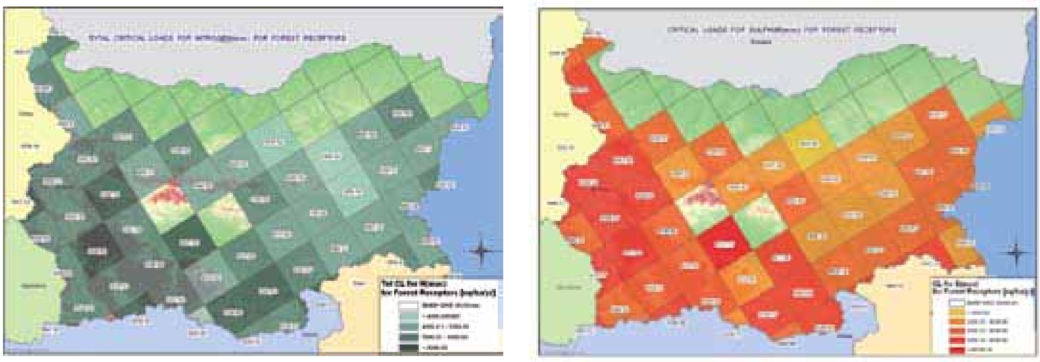

Figure BG-3. CL maxN (left) and CL maxS (right) for forest receptors in Bulgaria – 2007 year.

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COORDINATION CENTRE OF EFFECTS 2007/2008 DATA CALL

Critical Loads of N and S and Dynamic Modelling Data

This file contains brief information regarding the second Canadian NFC submission of critical loads data to the CCE in response to the 2007/2008 data call. The submission provides terrestrial (forest) critical loads data for approximately 138,415 records covering all provinces in Canada (below 60°N) and dynamic modelling data for 496 lakes in eastern Canada. The ecosystems represent a total area of 1,655,444.25 km2 (approximately 17 % of the total area of Canada). NOTE: The lake data have been previously described in the 2005 CCE status report and 2007 CCE progress report.

Data have been provided within a Microsoft Access database (template supplied by the CCE) containing six tables: ecords, CLdata, inputs, EmpNload, scenvars and h20inputs.

1. ecords TABLE

SiteID: Forest ecosystems are numbered from 1 to 138,415 and lakes lake ecosystems from 200,001 to 200,500.

Longitude, Latitude, EMEP50-i and EMEP50-j: no explanation needed!

Ecosystem area, protection and EUNIScode: The data submission includes forest ecosystem (EUNIS code G) and surface waters (EUNIS code C). National and provincial park areas were overlaid to generate protection class 9.

2. CLdata TABLE

Critical loads data have only been supplied for CLmaxS (estimated following the NEG-ECP protocol, 2001). Where data are available nANCcrit was estimated as CLmaxS minus base cation weathering and base cation deposition.

3. inputs TABLE

NOTE: Total soil weathering is specified entirely as calcium weathering, harvesting removals have been ignored and nitrogen processes were assumed to be negligible. As such, the following variables have been set to zero: Mgwe, Kwe, Nawe, Caupt, Mgupt, Kupt, Nimacc, Nupt, Nde and cOrganics.

No data have been submitted for cNacc, CEC, bsat, yearbsat, Cpool, CNrat and yearCN.

crittype and critvalue: set to 7 and 10 respectively following the NEG-ECP protocol.

thick and bulkdens: CLmaxS was estimated using a maximum depth for forest soils of 0.5 m. Bulk density were estimated from the Soil Landscape of Canada v2.1 map (URL: sis2.agr.gc.ca/cansis/nsdb/slc/v2.1) deposition (Ca, Mg, K, Na and Cl): Average total wet and dry deposition for the period 1994 to 1998 supplied by Environment Canada at a grid resolution of 35 km by 35 km.

Qle: estimated as precipitation minus modelled evapotranspiration (original data supplied by M. Posch, CCE)

lgKAlox and expAl: set to 9 and 3 respectively following the NEG-ECP protocol.

4. EmpNload TABLE

No data have been submitted.

Note: The Canadian NFC hope to submit data under this category in the future.

5. scenvars TABLE

No data have been submitted.

Note: The Canadian NFC hope to submit data under this category in the future.

6. h2oinputs TABLE

Data for 496 surface waters. See response to the 2006/2007 data call for further details.

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Critical loads of acidity and nutrient nitrogen

Introduction

In response to the call for data issued in November 2007, Finland submitted mass balance critical loads of acidification and nutrient nitrogen and empirical critical loads of nutrient nitrogen. The methods and data for mass balance critical loads of acidification and nutrient nitrogen have remained the same since 2001, while empirical critical loads of nitrogen are now submitted for the first time. Also information on Natura 2000 protection areas is reported with this submission.

Methods and data sources

Critical loads of acidification

Critical loads of acidification and nutrient nitrogen are calculated in Finland using mass balance methods (Johansson 1999; Johansson ym. 2001, Holmberg and Forsius 2004, UBA 2004). The mass balance method accounts for soil mineral weathering and the uptake of base cations by vegetation as well as leaching of sulphur, nitrogen and base cations. For lake acidification, the protection criterion is related to the acid neutralizing capacity of leaching soil water ([ANC]*limit* of 20 μeq L^{-1}). For forest soils, the protection criterion is related to the concentrations of aluminium and base cations in soil solution (molar Al:BC ratio of 1.0). Critical loads of acidification are determined for lakes and forest soils (spruce-, pine- and deciduous forests).

Critical loads of mass balance nutrient nitrogen

Nutrient nitrogen critical loads are calculated only for forest soils. The mass balance model accounts for nitrogen uptake by vegetation, immobilisation, denitrification and leaching of nitrogen. The protection criterion for forest soil nutrient nitrogen is related to the concentration of nitrogen in soil solution (0.3 mg N L^{-1}). Elevated concentrations of nitrogen in forest soil solution is considered to alter the nutrient balance of vegetation and thereby influence vegetation health and species distribution. Mass balance critical loads of nutrient nitrogen are 2 – 5 kg N $ha^{-1}yr^{-1}$ for coniferous and 2 – 7 N $ha^{-1}yr^{-1}$ for deciduous forest soils.

Land cover classes

The harmonized land cover map for the bodies under the LRTAP convention (Cinderby et al. 2007) represents 96 % of FI territory in 20 classes of land use. For the submission of empirical critical loads of nitrogen the land cover information was aggregated to approximately 10x10km^2 resolution.

Critical loads of mass balance acidification and nutrient nitrogen for Finland have been reported for soils in three types of forest ecosystems (spruce, pine, deciduous). A satellite image based land use data set has been used to estimate total forest area in each EMEP50 grid cell. Data from the seventh national forest inventory, collected by the Finnish Forest Research Institute 1977-1984 have been used to assign total forest area (240 400 km^2) to spruce, pine and deciduous tree species (Johansson 1999, Johansson et al 2001). The total forested area in the Finnish massbalance CL submissions, 240 400 km^2, is close to the sum of the EUNIS classes G1, G3, G4 (deciduous, coniferous and mixed woodland 253 500 km^2). In the Finnish massbalance CL submissions, the total area of spruce and pine forests amounts to 215 000 km^2, compared to 129 600 km^2 in class G3 in the harmonized land cover map. The area of deciduous woodland is 25 500 km^2 in the Finnish massbalance CL database while class G1 covers only 8 700 km^2 in the land cover map. The land use classes D1 (raised and blanket bogs) and F2 (artic, alpine and subalpine scrub) together cover 35 000 km^2 in Finland.

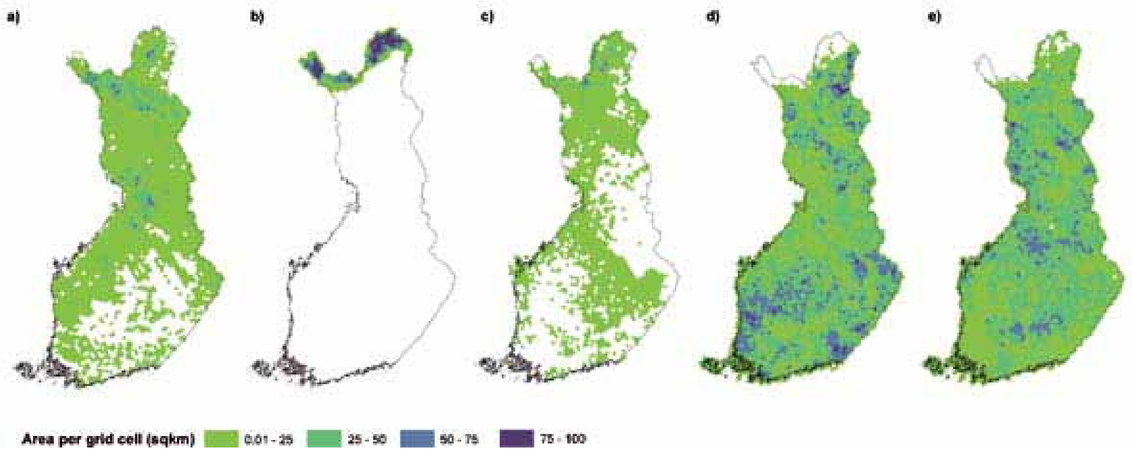

Figure FI-1. Distribution of land use classes in 10x10km^2resolution. a) D1 Raised and blanket bogs b) F2 Arctic, alpine and subalpine scrub; c) G1 Broadleaved deciduous woodland; d) G3 Coniferous woodland; e) G4 Mixed deciduous and coniferous woodland.

Natura 2000 areas

The areas protected under the Natura 2000 programme were determined at different spatial resolutions. The results were submitted at approximately 10x10 km^2 resolution. For comparison the proportion (%) of protected grid cells for two alternative resolutions 1 x 1 km² and 50 x 50 km² were determined for the land cover class G4 mixed deciduous and coniferous woodland (Table FI 1). Although some information on small protection areas is lost with the 10 km grid cell length, this resolution was considered to be sufficiently detailed for the purpose of the critical loads data base.

Table FI 1. Proportion (%) of protected grid cells for land cover class G4 mixed deciduous and coniferous woodland at varying spatial resolutions.

Natura 2000 protection programme	1 km	10 km	50 km
No Natura 2000 protection	22 %	56 %	91 %
SPA (Special Protection Area, Birds Directive applies)	0.07%	0.04%	0.00%
SAC (Special Area of Conservation, Habitats Directive applies)	1.62%	1.05%	0.00%
SPA *AND* SAC	77 %	43 %	9 %

Empirical critical loads of nutrient nitrogen

Values of critical loads of nitrogen were compiled by Achermann and Bobbink (2003) from the Bern workshop November 2002 on empirical studies on the response of terrestrial ecosystems to nitrogen deposition. Values suggested by the workshop to the new harmonized land cover map can be assigned using the interpretation by de Bakker et al. (2007). The most sensitive land use classes in Finland are raised and blanket bogs and artic, alpine and subalpine scrub and grasslands and the Bern workshop suggested values for empirical CLN for these land use classes 5-10 or 15 kg N $ha^{-1}yr^{-1}$. The suggested empirical CLN for coniferous, and mixed deciduous and coniferous woodland are 10-20 kg N $ha^{-1}yr^{-1}$ (de Bakker et al 2007). With the mass balance method, the corresponding values for Finnish forested ecosystems are 2-7 kg N $ha^{-1}yr^{-1}$.

In Finland, five land cover classes have been assigned values of empirical critical loads of nitrogen (Table 1). We set empirical critical loads of nitrogen to the value 8 kg N ha^{-1} a^{-1} to all land cover classes based on the recommendations of the Workshop on effects of low-level nitrogen deposition (UN ECE 2007) and in consideration of Finnish and Swedish investigations on vegetation effects of nitrogen.

In Finland, empirical studies on the effect of additions of low levels of nitrogen have been conducted mainly in combination with analysis of acidification (Shevtsova and Neuvonen 1997). In general, there was an increase in the amount of grasses and a decrease in the amount of lichens in response to the addition of nitrogen. There was also an impact on the berries, especially *Vaccinium vitis idae* carried fewer berries. Changes in soil microbiota were also recorded (Pennanen et al. 1998). Other empirical studies on nitrogen addition have primarily been concerned with fertilization and consequently involved high levels of nitrogen addition. It is clear, however, that different plant species react differently to nitrogen additions. For example, three common forest bryophyte species showed differences in the growth responses, when exposed to nitrogen in a chamber experiment (Salemaa et al. 2008). Even low levels of nitrogen addition may alter the species composition and fungi and herbivores may respond.

Swedish investigations have shown that low levels of nitrogen deposition (<10 kg) may cause changes in species distribution. Nitrogen application resulted in increased damage to *Vaccinium myrtillus* by natural enemies (Nordin et al. 1998). It is also likely that the response depends on the form of nitrogen. The grass *Deschampsia flexuosa* increased in response to nitrogen additions and especially if the nitrogen was added in the form of nitrogen oxide (Nordin ym. 2006).

Table FI 2. Overview of values of empirical critical load of nitrogen CLNemp used for land cover classes in Finland, minimum and maximum values (Achermann and Bobbink 2003) and area of each land use class of LRTAP harmonized Land Use Map (Cinderby et al. 2007).

EUNIS class	Area (km2)	CLN_{emp} kgN/ha^{-1}/yr^{-1}	FI Mapping Value kgN/ha^{-1}/yr^{-1}
D1 Raised and blanket bogs	26 000	5-10	8
F2 Arctic, alpine and subalpine scrub	9 000	5-15	8
G1 Broadleaved deciduous woodland	8 700	10-20	8
G3 Coniferous woodland	129 600	10-20	8
G4 Mixed deciduous and coniferous woodland	115 300	10-20	8

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Modelled critical loads and dynamic modelling data

The objectives of this call for data were to submit updated critical loads and to provide time series of modelled chemical variables for different deposition scenarios, i.e. dynamic modelling results. In 2005, the French National Focal Centre (NFC) provided updated critical load values for nitrogen (acid and nutrient) and sulphur as well as dynamic modelling results (Probst et al., 2005). In 2007, the French NFC: (1) tested the updated critical concentrations for the calculation of critical loads of nutrient nitrogen proposed by the Coordination Centre for Effects (CCE) and, (2) sent data for dynamic modelling (Probst and Leguédois, 2007). In comparison with 2005, the only major change was the removal of costal ecosystems (EUNIS code B1.4) from the dynamic modelling database as, for those ecosystems, the empirical critical loads were more consistent with observations (Probst et al., 2005).

For the 2008 call, the main differences concerned the structure of the database used for submission and the application of more conservative rules for the calibration of the dynamic model. To convert the 2007 data into the 2008 database structure we used the append queries provided by the CCE with VSD-Access version.

With the new calibration rules we had to change the values of the weathering rates (*Cawe*, *Mgwe*, *Kwe* and *Nawe*) and the base saturation (*bsat*) for calcareous soils in order to enable the dynamic model to calibrate. For these highly weatherable soils (weathering rates for base cations > 2,500 eq. $ha^{-1}.a^{-1}$), we had to increase the base saturation values to 0.9 and 0.99 for sites with weathering rates at 2,500 and 7,500 eq.$ha^{-1}.a^{-1}$ respectively. These adjustments were not sufficient for all the calcareous sites, so we also decreased the weathering rate down to 2,500 eq.$ha^{-1}.a^{-1}$. As we had no detailed information about the new calibration rules it was difficult to follow strict guideline for these number changes.

The 2008 results for dynamic modelling were significantly different from the 2007 results (Wilcoxon rank test performed on all the modelled variables *cAl*, *cBc*, *pH*, *ANC*, *bsat*, *CNrat* and *cN*, n = 244,290 and $p < 3$ %). Because of the way VSD-Access is built, it was not possible to assess the specific effect, on one side, of the value changes for base saturation and weathering rates, and, on the other side, of the modification of the deposition scenarios. However these changes shouldn't affect the target loads results as the concerned soils have a high buffering capacity.

Calculation method

The data were computed following the method used in 2005 by the French NFC (Probst et al., 2005) which is in accordance with the Mapping Manual (UBA, 2004). For steady state critical loads, the Simple Mass Balance (SMB) model (Sverdrup and de Vries, 1994) was applied on the soil top-layer (0–20 cm). VSD (Posch et al., 2003) was used for dynamic modelling. For soils with high buffering capacity, the results obtained with VSD show significant differences with more complex models (Probst et al., 2003; Probst et al., 2005). However VSD allows better consistency for impact assessment at large scale like across the European territory (Probst et al., 2005).

Data sources

Table FR-1: Sources for the parameters used in the critical load and dynamic modelling.

Variable	Explanation	Unit	Data sources or settings
crittype, critvalue	Chemical criterion used for critical loads for acidity and corresponding critical value	-	See Table FR-3.
CNacc	Aceptable (critical) N concentration	$meq.m^{-3}$	Derived from the acceptable nitrogen leaching (0 for plain deciduous forest; 50 for plain coniferous forest; 100 for mountain forest ecosystems — Party and Thomas, 2000) and the amount of water percolating through the root zone.
Cadep, Mgdep, Kdep, Nadep	Total deposition of base cations	$eq.ha^{-1}.a^{-1}$	RENECOFOR network measurements (ICP forest level I) extrapolated at the national scale (Ulrich et al., 1998; Croisé et al., 2002)
Cawe, Mgwe, Kwe, Nawe	Weathering rate of base cations	$eq.ha^{-1}.a^{-1}$	PROFILE simulations (Party, 1999)
Caupt, Mgupt, Kupt, Naupt	Net growth uptake of base cations	$eq.ha^{-1}.a^{-1}$	Calculated from base cation concentrations in vegetation (Party, 1999) and net uptake of biomass by harvesting (national survey, see IFN, 2002)
Nupt	Net growth uptake of nitrogen	$eq.ha^{-1}.a^{-1}$	Calculated from nitrogen concentration in vegetation (Party and Thomas, 2000) and net uptake of biomass by harvesting (national survey, IFN, 2002)
fde	Denitrification fraction	$eq.ha^{-1}.a^{-1}$	Adapted from the Mapping Manual data (UBA, 2004) for French soil conditions (see Table FR-2)
All soil parameters			From RENECOFOR network (ICP forest level I) data (Brêthes *et al.*, 1997) and CCE network data (Badeau and Peiffer, 2001). See Table FR-4.

Table FR-2: Setting of the values for the denitrification fraction (adapted from UBA, 2004).

Soil type	f_{de}
Non hydromorphic soil	0.05 to 0.2
Hydromorphic silt or sandy soil	0.3
Hydromorphic clay	0.4
Peat soil and marshes	0.5

Table FR-3: Values for the chemical criterion and the critical limit used in the calculation of the critical loads for acidity.

Soil and bedrock type	Chemical criterion	*crittype*	*critvalue*
Soft calcareous sediments	Molar [Al]/[BC]	1	1,2
Hard calcareous sediments	Molar [Al]/[BC]	1	1,2
Soft acid sediments			
Sands	pH	4	4,6
Sandy silex formations	pH	4	4,6
Others	Molar [Al]/[BC]	1	1,2
Hard acid sediments			
Schists	pH	4	4,6
Sandstones	pH	4	4,6
Others	Molar [Al]/[BC]	1	1,2
Metamorphic rocks			
Acid granite	pH	4	4,6
Others	Molar [Al]/[BC]	1	1,2
Volcanic rocks	Molar [Al]/[BC]	1	1,2

Table FR-4: Value of the soil parameters (from Brêthes et al., 1997).

Variable	Explanation	Units	Min	Max	Median
bulkdens	Bulk density	$g.cm^{-3}$	0.732	1.400	0.915
cOrgacids	Total concentration of organic acids	$eq.m^{-3}$	0	0.02436	3.48×10^{-5}
CEC	Cation exchange capacity	$meq.kg^{-1}$	15	380	140
bsat	Base saturation	-	0.12	0.99	0.90
Cpool	Amount of carbon in the topsoil	$g.m^{-2}$	3920	13 800	9 878
CNrat	C/N ratio on the topsoil	$g.g^{-1}$	12	28	15

The total concentration of organic acids in soil solution is calculated from DOC (Dissolved Organic Carbon) which is estimated from pH and clay content in soil layer. Due to the lack of data on partial CO_2-pressure in soil solution (*pCO2fac*), only one value (5 at) was considered for the topsoil.

Results

Critical loads of nutrient nitrogen

The most sensitive areas for nitrogen eutrophication are located in Sologne (Centre part of France) and the Landes (SW), the northern part of Massif Central and the eastern Mediterranean area (Figure FR-1).

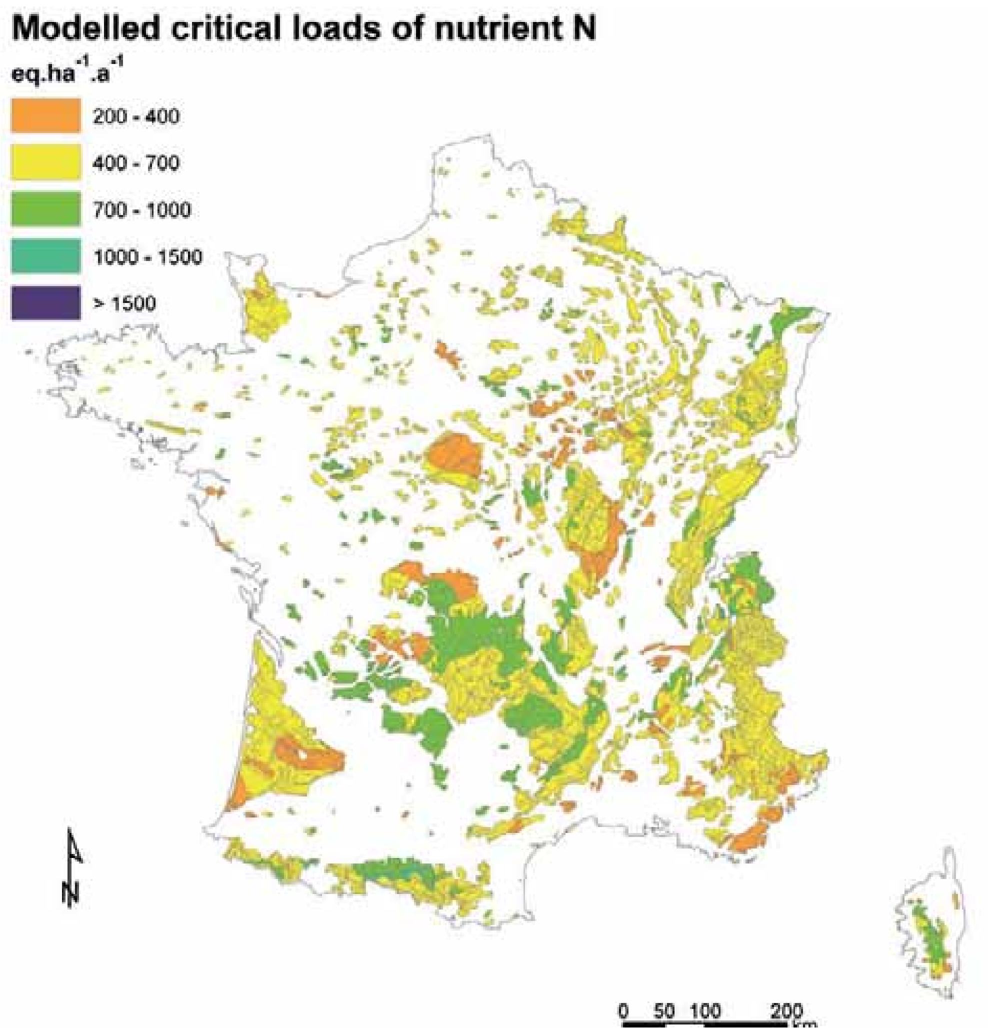

Figure FR-1: Map of modelled critical loads of nutrient nitrogen.

The critical loads of nutrient nitrogen show a global sensitivity of the French ecosystems. The lowest critical load values (< 400 eq.ha^{-1}.a^{-1} or < 5.6 kgN.ha^{-1}.a^{-1}) represent 22,185 km^2 (12.5 % of the studied area). Critical load values < 700 eq.ha^{-1}.a^{-1} (< 9.8 kgN.ha^{-1}.a^{-1}) represent 140,375 km^2 (79.2% of the studied area, i.e. forests and natural grasslands).

Critical loads of sulphur

Critical loads of sulphur were updated in 2005 (Figure FR-2) with a new computation of cations throughfall deposition, taking into account the cycle between biomass uptake and redeposition. Globally, the French ecosystems are not very sensitive to acidification by sulphur. The most sensitive areas are the Landes (SW), Sologne, some parts of the Massif Central as well as the Vosges mountains.

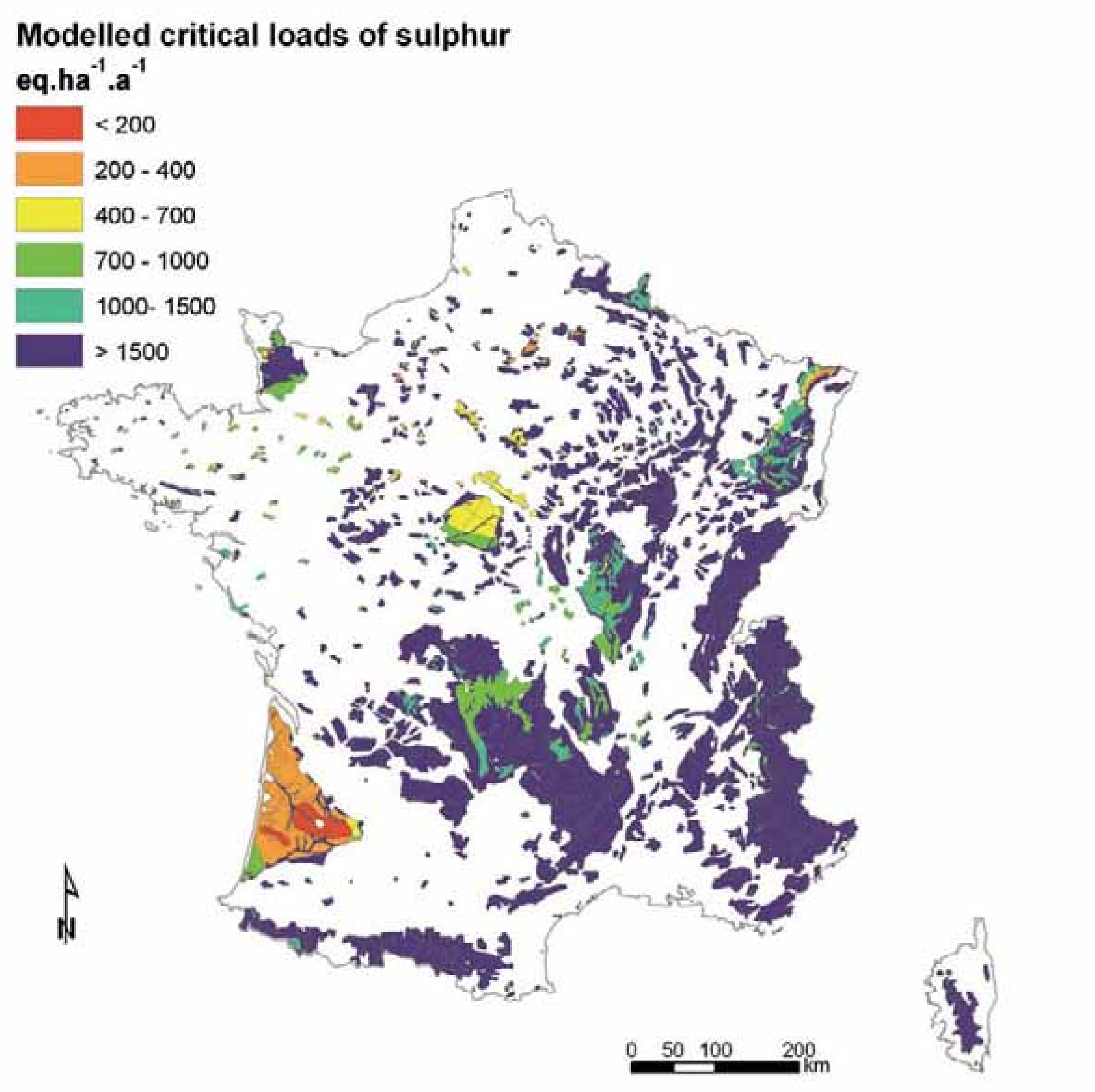

Figure FR-2: Map of the modelled critical loads of sulphur

Empirical critical loads of nutrient nitrogen

Method

The determination of empirical critical loads of nitrogen for French ecosystems was based on the method described in chapter 5.2 of the Mapping Manual (UBA, 2004). The values given in table 5.1 of the Mapping Manual (UBA, 2004) were adapted to the French terrestrial ecosystems (Party et al., 2001). The adaptation was based on: (1) the information available on the potential vegetation and the land use for each ecosystem and, (2) the adaptation rules given in table 5.2 of UBA (2004) using temperature, frost period and base cation availability estimated by expert judgement. The subsequent empirical critical loads are given in Table FR-5.

Table FR-5: Empirical critical loads, in eq.ha-1.a-1, derived for the French ecosystems (adapted from Party et al., 2001).

Potential vegetation	**Land use**			
	Coastal dune	Grassland	Upland meadow	Forest
Coastal dunes and heathlands	1786			
Swamps, bogs and wet heathlands	1786	714		714
Quercus robur dominated woodlands		1214		714
Quercus-Carpinus or *Ulmus* woodlands with *Quercus petraea*	1214	1214	500	857
Quercus petraea and *Q. pubescens* woodlands	1429	1429		1214
Quercus petraea, *Q. robur* or *pubescens* and *Q. pyrenaica* woodlands		1214		Per. + Bord.: 1214 SW + Nantes: 714
Mixed *Fagus-Quercus* and *Fagus* woodlands	1214	1214	500	1071
Quercus pubescens woodlands		K: 1789 A: 714		Corsica: 1071 Out Cors.: 1429
Quercus ilex woodlands	K: 1789 A: 714	K: 1789 A: 714		Corsica: 1071 Out Cors.: 1429
Quercus suber woodlands		714		Corsica: 1071 Out Cors.: 1429
Pinus halepensis and *P. nigra laricio corsicana* Mediterranean woodlands		857		1071
Pinus pinaster woodlands		714		500
Abies and mixed *Abies-Fagus* woodlands		714	714	857
Picea woodlands		714		714
Pinus sylvestris woodlands		714		500
Pinus uncinata and *P. cembra* woodlands		500		500
Larix woodlands		714		714
Alpine and subalpine grasslands			500	

K: calcareous ecosystem; A: acidic ecosystem;Out Cors.: outside Corsica; Per.+Bord.: Perigord and Bordeaux regions;SW+Nantes: South-West and Nantes regions.

Data Sources

The French ecosystem classification and map was updated in 2003 for calculation and mapping of the critical loads of acidity and nutrient nitrogen (Probst et al., 2003 ; Moncoulon et al., 2004). The map of potential vegetation was synthesised for the French territory by Party (1999) from various vegetation maps (Dupias and Rey, 1985; Houzard, 1986; Ozenda and Lucas, 1987). Land use was derived from the map of forested and grassland areas in de Monza (1989) as well as the Digital Elevation Model GTOPO30 (USGS, 1996).

Results

The most sensitive areas to nitrogen deposition are located in the Landes (SW), the eastern part of the Paris basin, the eastern part of the Massif Central as well as in the Alps (Figure FR-3). Empirical critical loads of nitrogen are higher than critical loads for nutrient nitrogen determined with the Steady State Mass Balance (SMB) model (see Figure FR-1 as well as Party and Thomas, 2000). Consequently, the sensitivity of the ecosystems is lower when derived from the empirical method. Comparatively to the SMB model, most of the ecosystems shifted to a higher critical load class with the empirical method (+ 1 class for 49 % of the ecosystems and + 2 classes for 35 % of the ecosystems).

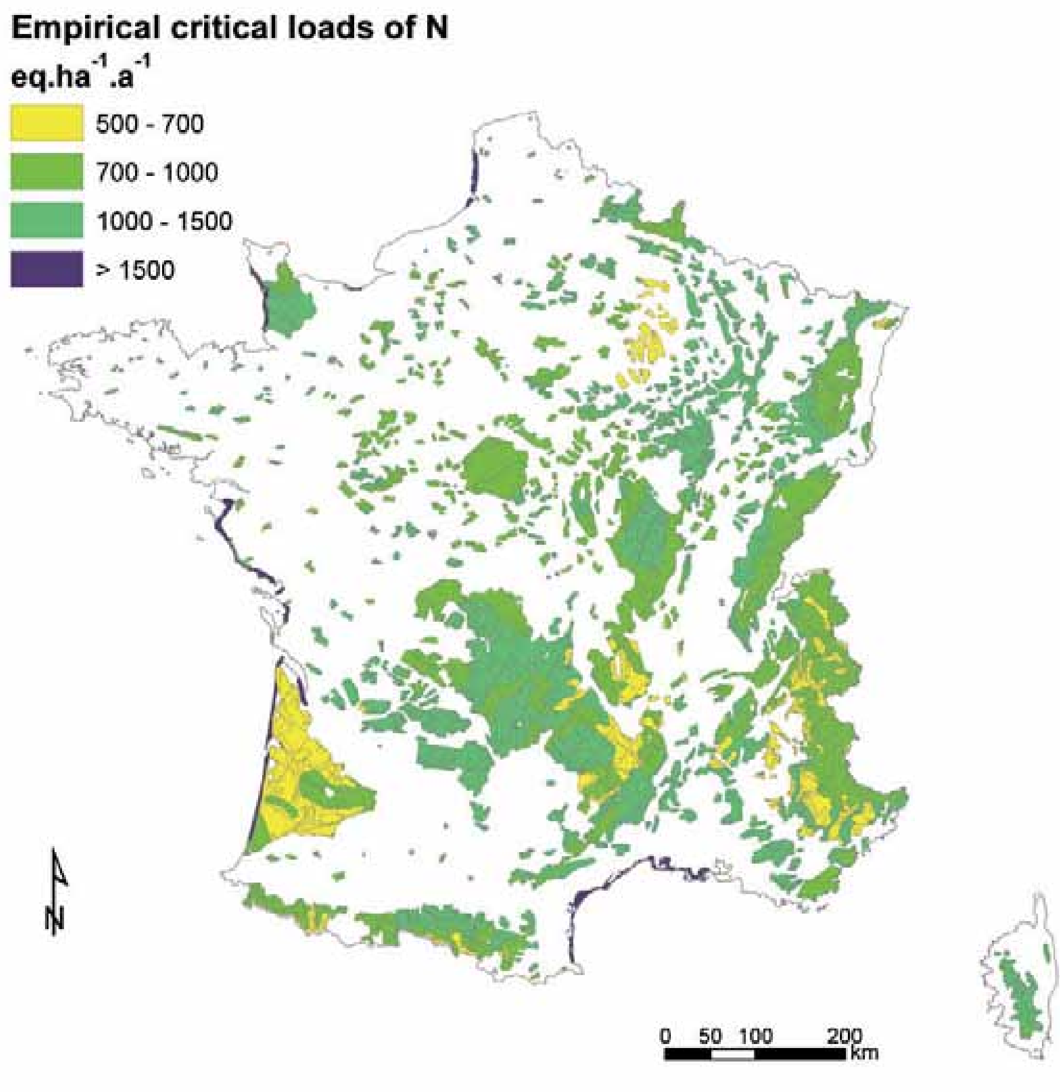

Figure FR-3: Map of the empirical critical loads of nutrient nitrogen.

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Data sources

Critical Loads of Nitrogen and Sulphur, and Dynamic Modelling Data

The German NFC provides an update of the national critical load data of sulphur and nitrogen, and results of the dynamic model application (VSD model). Critical loads are calculated completely using the VSD model based on methods described in the Mapping Manual (UBA 2004) and following the instructions of the CCE for data submission (CCE 2007). The German critical load database consists of 97,729 records.

In comparison with previews data submissions (2005, 2007) some changes can be observed concerning the critical loads of sulphur (Fig. DE-1) and the critical loads of nutrient nitrogen (Fig. DE-2).

Compared to the 2007 data critical loads of sulphur, $CL_{max}(S)$, have a wider range of values and show less overall ecosystem sensitivity. While identical input parameters were used the outcome of the application of the VSD model in 2008 show critical loads below 1 keq ha^{-1} a^{-1} for about 30 % of ecosystem receptors, whereas the figures of the 2007 calculation showed critical loads below 1 keq ha^{-1} a^{-1} for about 60 % of ecosystem receptors. The regional distribution of critical loads of sulphur is shown in Figure DE-3.

Only very small changes between 2007 and 2008 data can be observed as critical loads of nutrient nitrogen, $CL_{nut}(N)$, are concerned. Between 2005 and 2007 significantly greater changes, due to applying the suggested update of critical concentrations in soil solution (CCE 2007a), have occurred. Therefore a national approach was derived using the vegetation period for assignment of different concentration values in Northern and Western Europe (CCE 2007b). A box plot of submitted data is given in Figure DE-2, showing the empirical critical load values, $CL_{emp}(N)$, the VSD computed critical load data, $CL_{nut}(N)$, as calculated in 2005 using the original critical N concentrations given by the Mapping Manual of 2004, the suggested (national modified) update of the 2007 call for data ($CL_{nut}(N)$ 2007) and the results of $CL_{nut}(N)$ 2008. The regional distribution of critical loads of nutrient nitrogen is shown in Figure DE-4.

The dynamic model VSD was successful implemented. Results for the given scenarios "Current Legislation" (CLE) and "Maximum Feasible Reduction" (MFR) are shown in Figure DE-5 and Figure DE-6. As one of the most sensitive indicators the pH value was selected and the distribution trend over time is presented in the box plots.

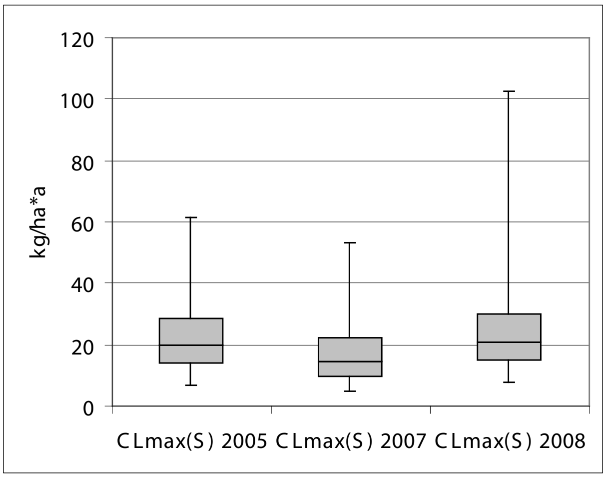

Figure DE-1: Comparison of submitted national data sets for critical loads of sulphur.

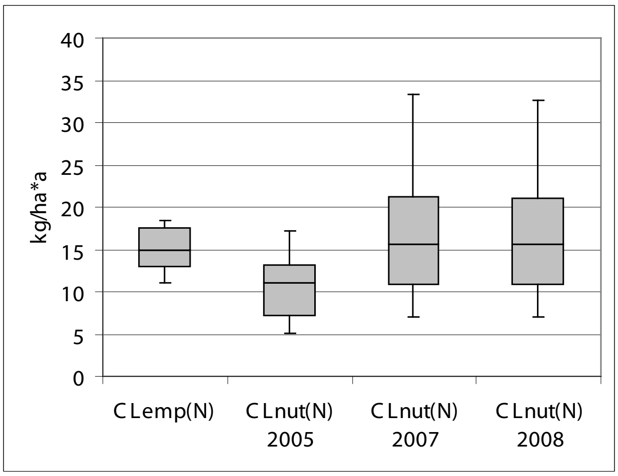

Figure DE-2: Comparison of submitted national data sets for critical loads of nutrient nitrogen.

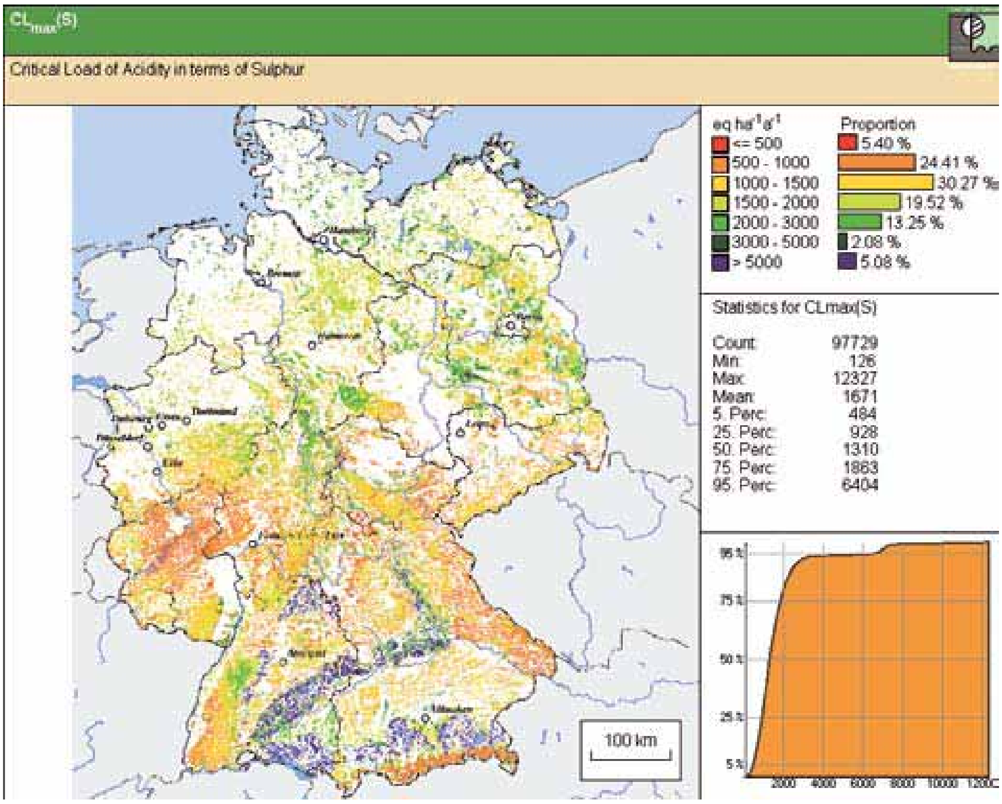

Figure DE-3: Critical loads of sulphur, $CL_{max}(S)$.

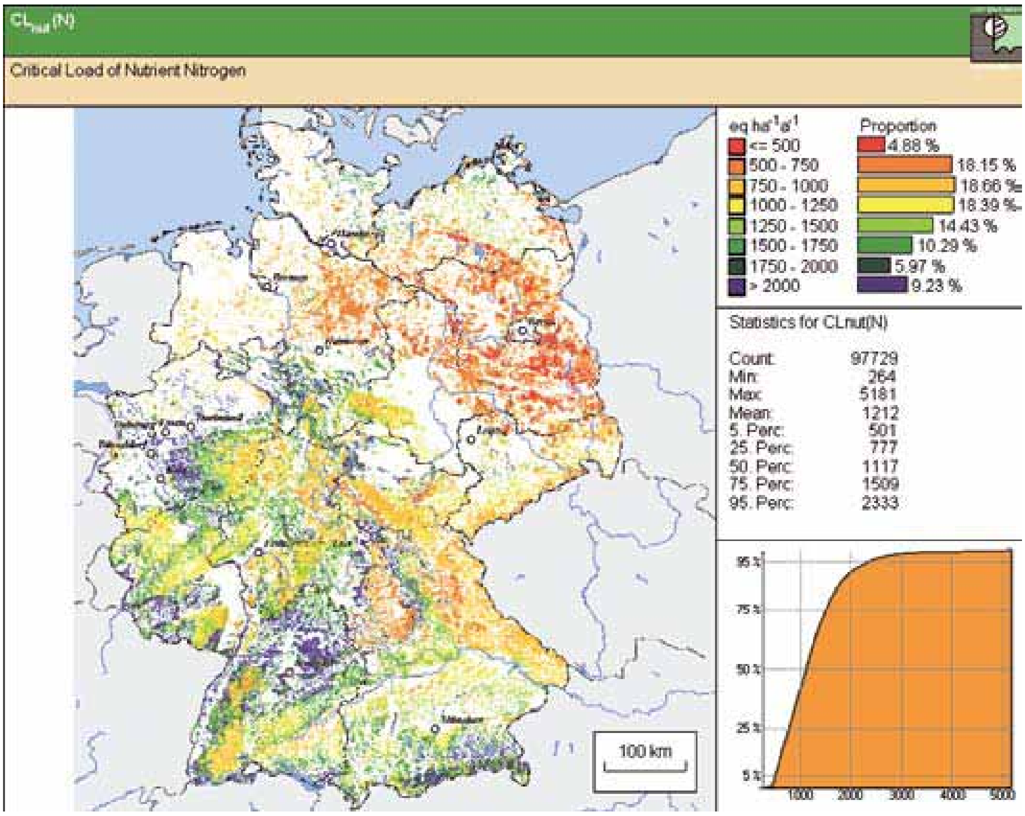

Figure DE-4: Critical loads of nutrient nitrogen, $CL_{nut}(N)$.

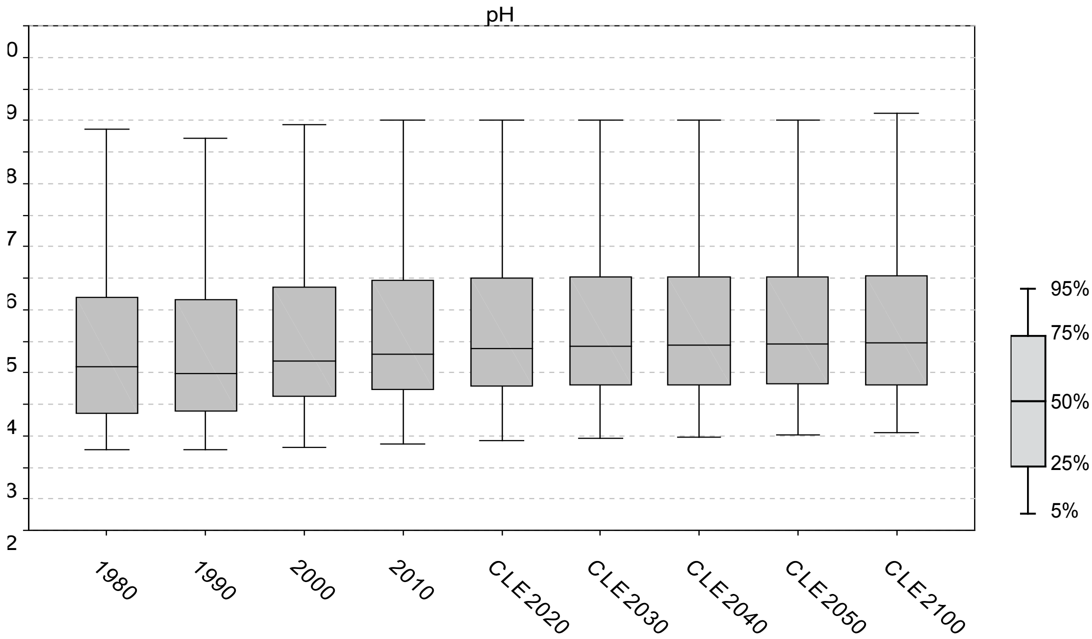

Figure DE-5: Trend of the distribution of pH values in Germany following the "current legislation" deposition scenario.

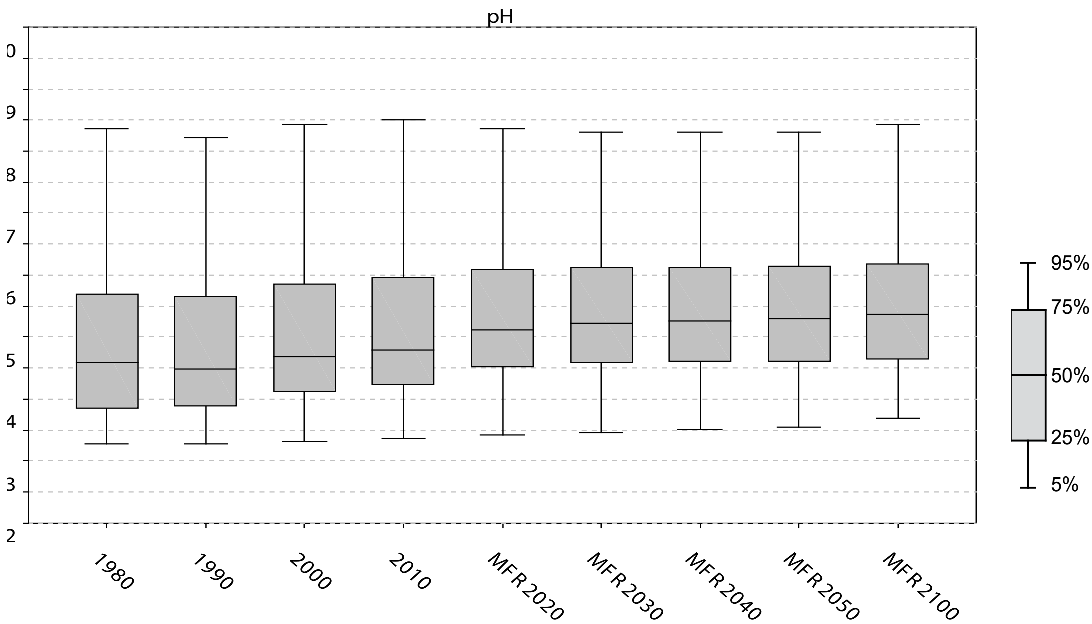

Figure DE-6: Trend of the distribution of pH values in Germany following the "maximal feasible reduction" deposition scenario.

Empirical Critical Loads of Nitrogen

In addition to the calculation of critical loads with the VSD model, empirical critical loads of nitrogen were calculated for the complete national dataset, consisting of 102,560 records, in accordance to the methods described in the Chapter 5.2 of the Mapping Manual (UBA 2004) and following the recommendations of the workshop "Empirical Critical Loads for Nitrogen" (Achermann and Bobbink 2003). A regional distribution of this dataset is shown in Figure DE-7.

Critical load ranges given in Table 5.1 of the Mapping Manual were specified by applying the BERN model (Schlutow and Kraft 2006). A typical plant community with a unique empirical critical load value could be defined for each EUNIS code (see CCE 2007c). Empirical critical loads of nitrogen for terrestrial ecosystems in Germany range between 5 and 38 kg N ha^{-1} a^{-1} with a mean of 15 kg N ha^{-1} a^{-1}.

As additional information the protection status of all grid cells with empirical critical loads of nitrogen was checked. The European Habitats Directive (FFH) applies at nearly 28 percent (28,806) of mapped grids, 10,532 of them are also Special Protection Areas (SPA) for which the Birds Directive applies. About 5 percent of the grid cells are SPA areas only.

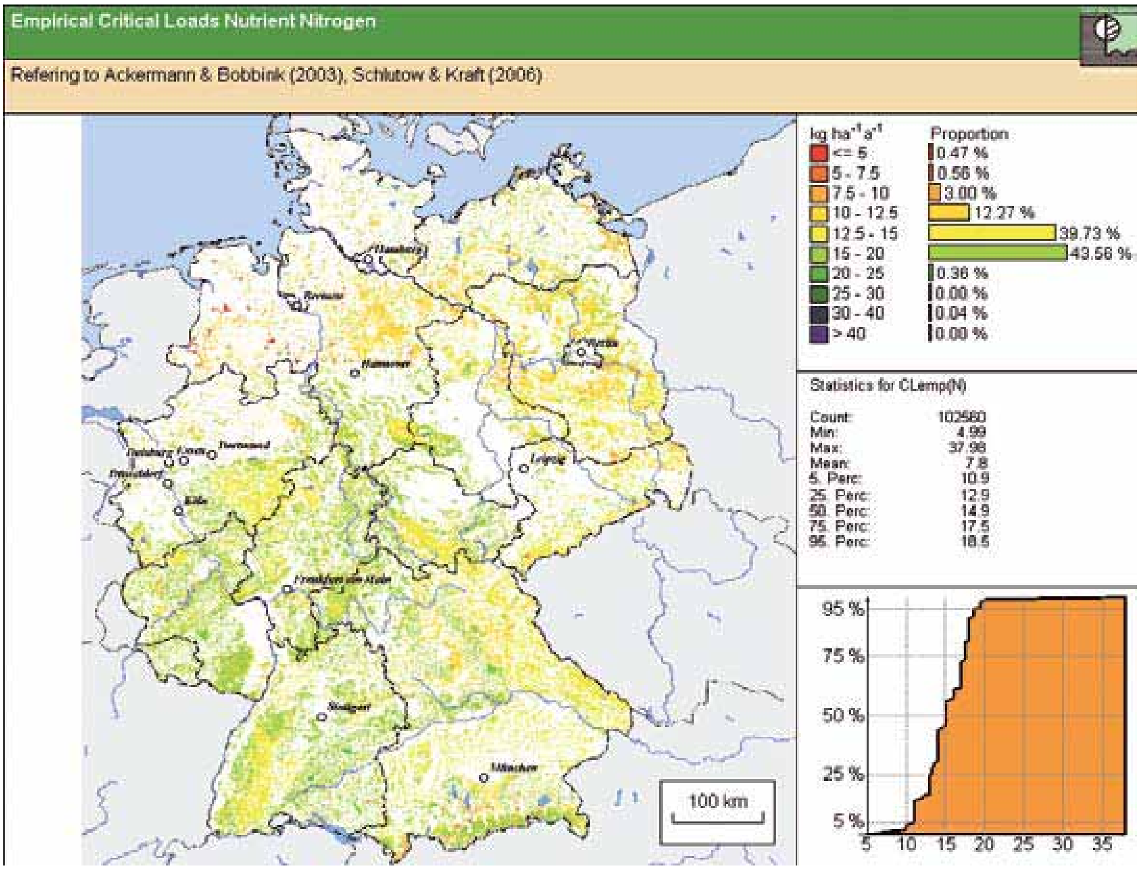

Figure DE-7: Empirical critical loads of nitrogen, CL_{emp}(N).

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CCE (2007a) Progress Report 2007 Coordination Centre for Effects, Appendix B, Table 4, p. 196 Bilthoven, Netherlands, Report No. 500090001/2007, ISBN No. 978-90-6960-175-5

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COORDINATION CENTRE OF EFFECTS 2007/2008 DATA CALL

Critical Loads of N and S and Dynamic Modelling Data

This file contains information regarding the Irish critical loads data submitted to the CCE 2008. The submission includes updated critical loads data in response to an error in the previous (March 2005) data submission and NEW dynamic modelling data. In 2005, the mass balance approach for nutrient nitrogen was applied to all ecosystems in error; it should have only been applied to coniferous ecosystems, and an empirical critical load set for the remaining eco–systems. This submission includes the correct CLnutN data and other updates as required (NOTE: the CL data is identical to that submitted under 2007 voluntary data call). Dynamic model simulations under historic and multiple future scenarios have been carried out using the VSD model. All data have been submitted using the Microsoft Access template provided by the CCE. The following five tables have been submitted: ecords, CLdata, inputs, EmpNload and scenvars.

Critical load mapping is carried out on a 1 by 1 km grid, therefore all data records refer to 1 km2 grids. The database contains one table: critical loads and dynamic model input data. A description of the submitted data follows below.

ECORDS TABLE (30984 records)

1. Site Id: Unique ID used across all tables (simple umeric sequence)

2-5. Longitude, Latitude, EMEP50-i and EMEP50-j: no explanation needed!

6. Ecosystem area

Four ecosystems were selected for mapping, coniferous forests, deciduous forests, natural grasslands, and moors and heathlands. The CORINE land cover map for Ireland was used to define the distribution of the mapped terrestrial ecosystems. A database is held which gives the percentage of each ecosystem in every 1 km2 (range 0.01-100%). The selected ecosystems were derived from the following CORINE classes:
Coniferous - 3.1.2 Coniferous forest; [9195 records], EUNIS code: G3
Deciduous - 3.1.1 Broad-leaved forest, EUNIS code: G1
3.1.3 Mixed forest, EUNIS code: G4
3.2.4 Transitional woodland scrub; [8047 records], EUNIS code: G5
Grasslands - 3.2.1 Natural grasslands; [6895 records], EUNIS code: E1/E3
Heathland - 3.2.2 Moors and heathlands; [6848 records], EUNIS code: F4
Total number of records is 30984 representing an area of approximately 8936 km².

7. Protection

NOTE: this has been set to zero BUT i is anticipated that this will be updated to reflect the true protection status.

8. EUNIScode

EUNIS G3: CORINE 3.1.2 Coniferous forest
EUNIS G1: CORINE 3.1.1 Broad-leaved forest
EUNIS G4: CORINE 3.1.3 Mixed forest
EUNIS G5: CORINE 3.2.4 Transitional woodland scrub
EUNIS E1/E3: CORINE 3.2.1 Natural grasslands
EUNIS F4: CORINE 3.2.2 Moors and heathlands

This classification was simplified to four codes representing coniferous (G3), mixed deciduous (G4), grassland (E3) and heathland (F4) for the current submission

CLDATA TABLE (30984 records)

2-5. Critical loads

Maximum critical load of sulphur, CLmax(S)
Minimum critical load of nitrogen, CLmin(N)
Maximum critical load of acidifying nitrogen, CLmax(N)
Critical load of nutrient nitrogen, CLnut(N)

Critical loads were estimated using the VSD-MDB tool (2005), which includes the current critical load methodology as present in the mapping manual (see www.icpmapping.org). This ensured consistency between critical and target loads. Critical loads for sulphur have not change since the 2003 submission; however, the critical loads for nitrogen have changed in line with recommendations in the mapping manual and update methodologies. A mass balance approach for nutrient nitrogen has been applied to coniferous forests only [EUNIS code: G3], all other ecosystem records have NULL values as empirical critical loads have been set [see EmpNload database].

6. Critical leaching of alkalinity, nANCcrit

This was calculated as described in CCE STATUS 1, July 1991, page 35. ALle(crit)=Q.[Al]crit and Hle(crit)=Q.[H]crit, where Q is precipitation surplus (m3/ha/yr) and [H]crit and [Al]crit are critical hydrogen and aluminium concentration (molc/m3). A pH of 4.2, was selected as the H+ concentration limit and used to estimate the Al critical limit via the gibbsite relationship. The H+ critical limit of pH = 4.2 was selected based on work by Ulrich (1987) which states that at pH lower than 4.2 AL3+ is in solution.

7-9. lgKAlBc, lgKAlBc and CNrat0

All enteries were estimated using the VSD model.

INPUTS TABLE (30984 records)

2. Acceptable nitrogen concentration, cNacc

The acceptable nitrogen concentration was set to 0.0143 eq/m3 or 14.3 meq/m3 for coniferous forests. All other ecosystem records have NULL values as empirical critical loads have been set [see EmpNload database].

3-4. Chemical criterion (crittype) and critical value (critvalue)

Soil solution pH was selected as the chemical criterion, with a critical value of 4.2 for all ecosystems (see 11 above).

5. Soil thickness, thick

All soil variables were derived from the modal profile of the principal soil for each of the soil associations on the general soil map of Ireland (44 associations). Soil thickness was estimated as the sum of O/A/E/B horizons for each model profile.

6. bulk density, bulkdens

Average bulk density was estimated from percent organic carbon in each horizon and weighted by horizon for each profile.

7-10. Base cation deposition, Cadep, Mgdep, Kdep, Nadep

Base cation deposition was calculated from interpolated (kriging) point source bulk concentration measurements (approx. 20 points) and interpolated (kriging) rainfall volumes (approx. 600 points). A filter factor of 2 was used to scale from bulk deposition to total deposition to forest and 1.5 for total deposition to heathlands. NOTE only anthropogenic deposition used. All deposition was assigned to Cadep which is the component of base cation deposition (magnesium and sodium are predominantly of marine origin)

11. Total chloride deposition, Cldep

This was set to zero as only non-marine deposition was submitted (it is assumed that all chloride is marine origin).

12-15. Weathering of base cations, Cawe, Mgwe, Kwe, Nawe

Base cation weathering was calculated using the Skokloster classification ranges for mineral soils, the mid-value of each of the five classes was used to define soil weathering, except for the final (non-sensitive) class which was set at 4000 eq/ha/yr. Organic soils were allocated to the lowest Skokloster class. A default of 20% of the total weathering was allocated to sodium weathering (Nawe) and the remainder was assigned to Cawe.

16-18. Base cation uptake, Caupt, Mgupt, Kupt

All base cation uptake (Bc) was assigned to Caupt. Uptake for coniferous forests was estimated from the following:

min(BCavail, Bcu)

where BCavail is the available base cation flux estimate according to:
BCavail = (BCw + BCdep - BCleaching), BCleaching was set equal to 0.02 eq/m3.

BCu was calculated using a yield class of 16 m3/ha/yr, a wood density of 390 kg/m3 and stem concentrations of ca=0.056, mg=0.021 and k=0.0665 % [390 eq/ha/yr]. It was assumed that all coniferous trees were sitka spruce, the yield class is the average yield class for sitka spruce for Ireland (COFORD, 1996) and the stem concentrations are for sitka spruce in Wales (Emmett and Reynolds, 1996).

For deciduous forests, natural grasslands and moors and healthlands an uptake of 45 eq/ha/yr was selected to account for uptake by grazing.

19. Runoff, Qle

Runoff or precipitation surplus was estimated as the difference between rainfall and evapotranspiration plus surface runoff. Rainfall and evapotranspiration are GIS layers interpolated from long-term measurements. Surface runoff was estimated from soil type.

20-21. Equilibrium constant for Al-H relationship (lgKAlox) and exponent (expAl)

Both parameters are based on the ‘classic’ gibbsite (Kgibb) equilibrium relationship, as such the exponent was set to 3. The spatial distribution of Kgibb was defined by reclassifying the general soil map of Ireland into three classes: Kgibb 9.5 m6/eq2 for organic soils; Kgibb 100 m6/eq2 for peaty podzols and peaty gleys; and Kgibb 300 m6/eq2 for the remaining soils. These represent log10 values of 6.5, 7.523 and 8, respectively.

22. Partial CO2 pressure, pCO2fac

For consistency with previous critical load methodologies pCO2fac was set to a null value of 1, which essentially ‘switches off’ the process in the VSD model. This is a simplification; however, in soils with pH lower than 5.0 (the soils of interest in relation to critical/target loads) the process is negligible. As such, this exclusion has very minor implications for critical/target load estimates.

23. Total concentration of organic acids, cOrgacids

Organic acids have previously not been included in Irish critical load calculations and due to limited data were excluded from the current submission. Organic acids were set to a low value, which essentially switches off the process in the VSD model.

24. Nitrogen immobilisation, Nimm

Following recommendations in the mapping manual (www.icpmapping.org) this was set to a default value of 71 eq/ha/yr for all ecosystems.

25. Nitrogen uptake, Nupt

For coniferous ecosystems the same method as above (26-28) was used (actually the BC uptake was used to scale the N uptake where Nu was estimated using a yield class of 16 m3/ha/yr, a wood density of 390 kg/m3 and stem concentrations of N=0.05 % [310 eq/ha/yr]).

For deciduous forests, natural grasslands and moors and healthlands an N uptake of 71 eq/ha/yr was selected to account for uptake by grazing.

26. Denitrification fraction, Fde

Following the method of Reinds et al. (2002) Fde was assigned to each soil type on the general soil map of Ireland based on drainage class.

Drainage (Fde): excessive (0), good (0.1), moderate (0.2), imperfect (0.4), poor (0.7) and very poor (0.8).

27. Nitrogen denitrified, Nde

Null. Not required as Fde was submitted.

28-30. Cation exchange capacity (CEC), base saturation (Bsat) and year for base saturation (yearbsat)

These variables were derived from the modal profile of the principal soil for each of the soil associations on the general soil map of Ireland (44 associations). For each soil association, the variables (per horizon) were weighted by depth and bulk density to produce one average per profile. The handbook that accompanies the general soil map of Ireland (and describes the physico-chemical properties of the principal soil types) was published in 1980. This was taken to be the default year for all base saturation measurements.

31-33.Carbon pool (Cpool) and C/N ratio (CNrat) in the topsoil and year for C/N ratio (yearCN)

These variables were derived from the modal profile of the principal soil for each of the soil associations on the general soil map of Ireland (44 associations). For each soil association, the variables (per horizon) were weighted by depth and bulk density to produce one average per O and/or A horizons only. The measurement was set to match the base saturation measurement year.

34. Measured

Status unknown.

EMPNLOAD TABLE (30984 records)

2. Empirical critical load of nitrogen

The critical load for nutrient nitrogen (CLnutN) was set to 857 eq/ha/yr for coniferous and deciduous forest ecosystems (EUNIS code G3, G1, G4 and G5) and 1071 eq/ha/yr for grasslands, and moors and heathlands (EUNIS code E1, E3 and F4). These values represent the mid-points of the 'recommended' empirical critical load ranges (Achermann and Bobbink, 2003), protecting against changed species composition for forest ecosystems (10-15 kg N/ha/yr) and increased N mineralisation, nitrification and leaching for grassland ecosystems (10-2 kg N/ha/yr).

SCENVARS TABLE (30984 records)

All parameters and scenarios were simulated using the VSD-MDB tool.

References

Achermann, B. and Bobbink, R., 2003. Empirical Critical Loads for Nitrogen Environmental, Documentation No. 164, SAEFL, Berne, Switzerland, 327 pp.

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Introduction

The critical loads database were processed according to the SMB methodology, suggested by CCE and described in the Mapping Manual 2004 (CCE, 2004). The most relevant differences from the suggested methodology are mentioned in the following explanation

Receptors: the classification of land use proposed by CORINE Land Cover 2000 has been applied to the mapped receptors. They were defined geometrically by CORINE database, while vegetation characteristics were assessed by an intersection with the Map of the real vegetation provided by the Ministry for the Environment. The resulted data were reclassified into the EUNIS Habitat system (15 habitats of first level and 26 habitats of second level classification), as shown below.

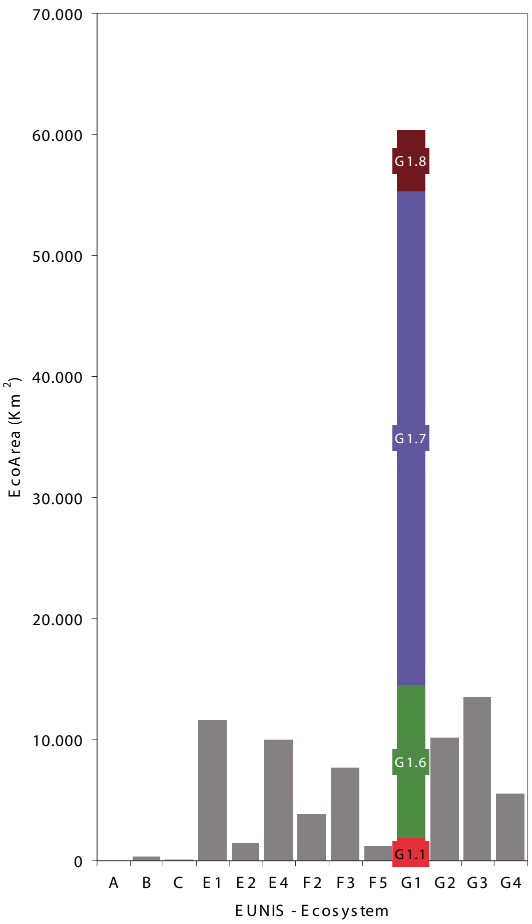
Meteorology: the information regarding temperature and precipitation (yearly mean of about one decade of measurements) were updated by data provided by Climate Research Unit of the University of East Anglia.

Soil parameters: all the information regarding soil parameters were derived from the european database, EU Soils (European Soil Portal).

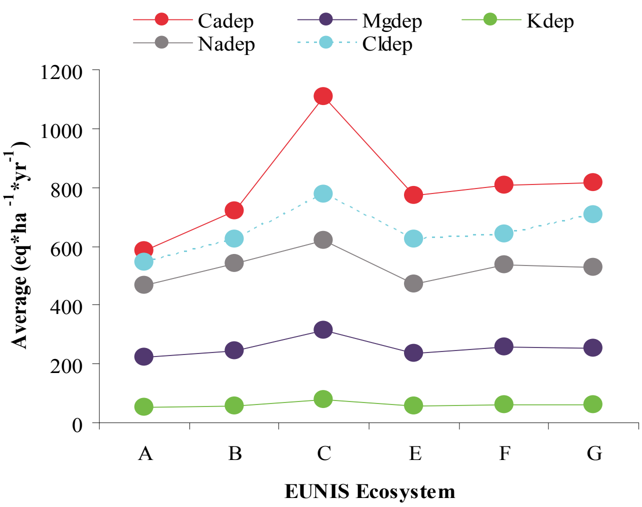
All maps have been processed by Geographic Information System (GIS) software.

Data sources

EUNIS Ecosystems distribution



Level 1	2	3	Habitat
A	A4	A4.5	Shallow sublittoral sediments dominated by angiosperms
B	B1	B1.4	Coastal stable dune grassland
	B3	B3.3	Rock cliffs, ledges and shores, with halophytic angiosperms
C	C1	C1.2	Permanent mesotrophic lakes, ponds and pools
	C3	C3.2	Water-fringing reedbeds and tall helophytes other than canes
E	E2	E2.3	Mountain hay meadows
	E1	E1.2	Perennial calcareous grassland and basic steppes
		E1.3	Mediterranean xeric grassland
		E1.5	Mediterraneo-montane grassland
		E1.8	Mediterranean dry acid and neutral closed grassland
	E4	E4.3	Acid alpine and subalpine grassland
		E4.4	Calciphilous alpine and subalpine grassland
F	F2	F2.3	Subalpine and oroboreal bush communities
	F3	F3.1	Temperate thickets and scrub
		F3.2	Mediterraneo-montane broadleaved deciduous thickets
	F5	F5.2	Maquis
	F7	F7.4	Hedgehog-heaths
G	G1	G1.1	Riparian [Salix], [Alnus] and [Betula] woodland
		G1.5	Broadleaved swamp woodland on acid peat
		G1.6	[Fagus] woodland
		G1.7	Thermophilous deciduous woodland
		G1.8	Acidophilous [Quercus]-dominated woodland
	G2	G2.1	Mediterranean evergreen [Quercus] woodland
	G3	G3.1	[Abies] and [Picea] woodland
		G3.2	Alpine [Larix] - [Pinus cembra] woodland
		G3.4	[Pinus sylvestris] woodland south of the taiga
		G3.5	[Pinus nigra] woodland
		G3.7	Lowland to montane mediterranean [Pinus] woodland (excluding [Pinus nigra])
	G4	G4.6	Mixed [Abies] - [Picea] - [Fagus] woodland



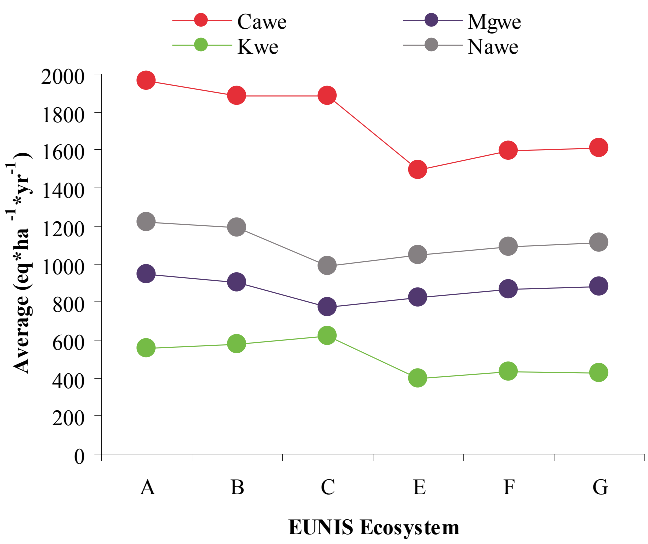
Average deposition Ca, Mg, K, Na and Cl

Base cation deposition (Ca, Mg, K, Na) has been corrected for marine contribution and for canopy interception. The latter according to

$$BC_{dep} = \begin{cases} 2 * BC_{wet} & , \text{if } BC_{wet} < 250 \\ 250 + BC_{wet} & , \text{if } BC_{wet} \geq 250 \end{cases}$$

Data source:
Italian Network for the assessment of atmospheric deposition and ENEL monitoring stations (1987-1999)

Map of mean yearly precipitation in Italy 1955-2000



Mapping Manual:

Soil type – texture approximation (eq. 5.39, modified)

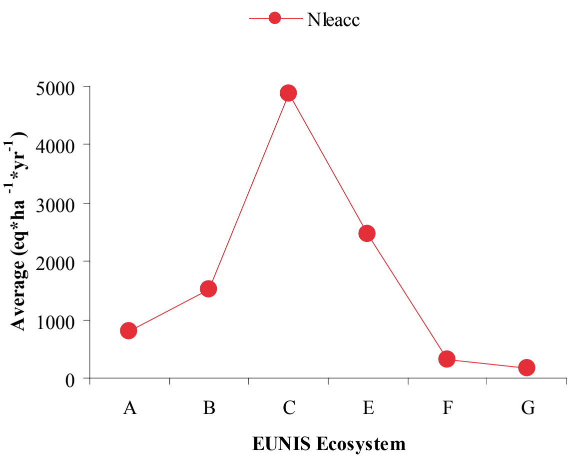
Data source:
Soil Map of the European Communities (ECE, 1985)

Map of mean yearly temperature in Italy 1955-2000

Manual:
De Vries et al., 1993

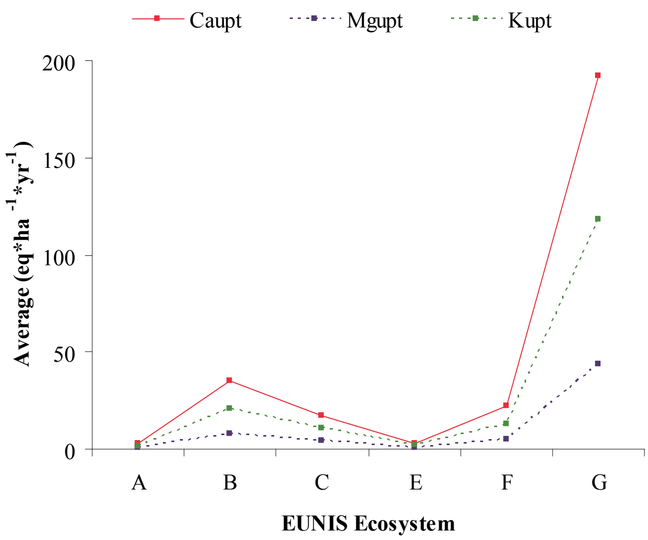
Hettelingh J. P. and de Vries W., 1990

Average weathering Ca, Mg, K and Na



Method
Based on Runoff and Nacc for different habitat by Mapping Manual 2004

Average Nleacc



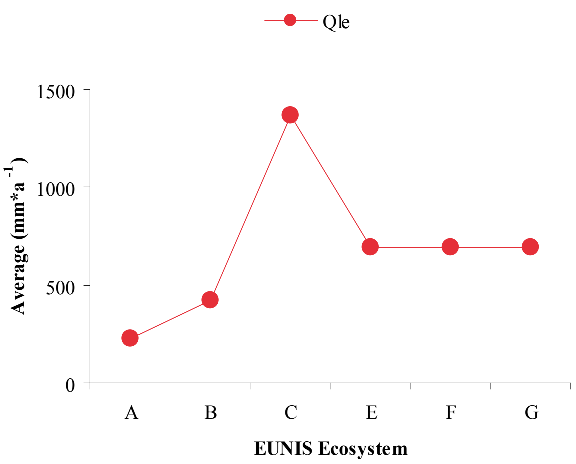
Average uptake Ca, Mg and K

Data source

National Forest Inventory

Method

Mapping Manual, based on average annual stem growth x wood density x element concentration in wood. (Bonanni P. et al. 2001)



Average Qle

$$Q_{le,zb} = P_m - 0.8 * P_m^{-2} + (e^{(0.063*Tm)} * E_{m,pot})^{-2})^{-1/2}$$

with

Qle,zb = mean yearly water flux (in m yr-1)

Pm = mean yearly precipitation (in m yr-1)

Tm = mean yearly temperature (in °C)

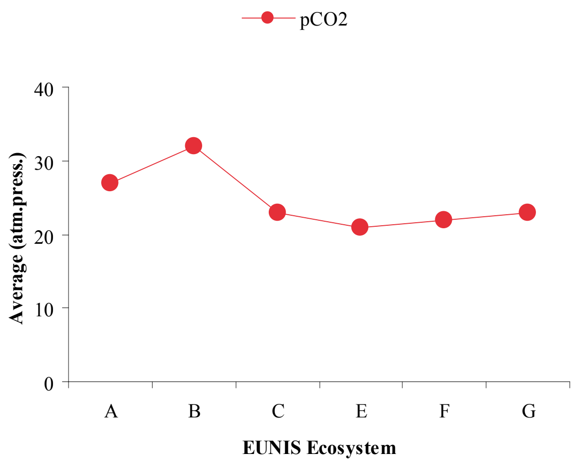
Em_{pot} = mean yearly potential evapotraspiration (in m yr^{-1})

Data source:

Map of mean yearly temperature in Italy 1955-2000

Map of mean yearly precipitation in Italy 1955-2000

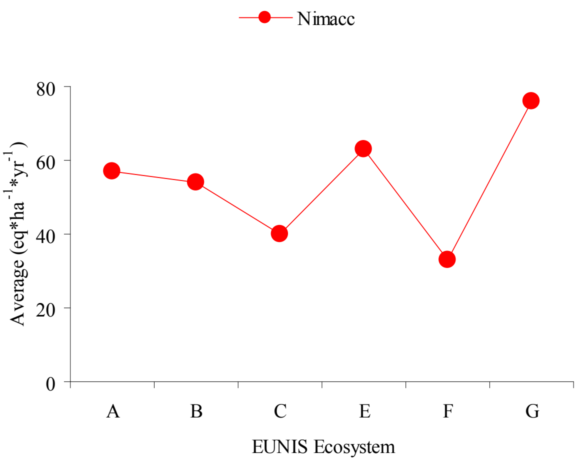
Mapping Manual 2004



Method:
Mapping Manual 2004 eq 5.44
FOEFL, 1994

Data source:
Map of mean yearly temperature in Italy 1955-2000
$Log10P_{co2} = -2.38 + 0.031 \cdot T$

Average pCO_2

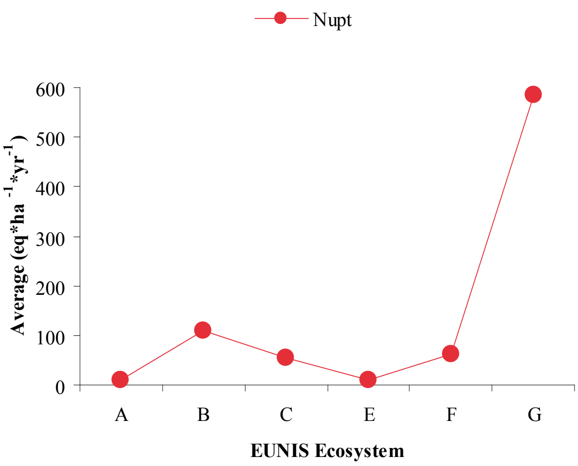


Nimacc=Ni+Nfire+Nvol-Nfix

with:
Ni= immobilised N in soil organic matter
Nfire = N losses in fire
Nvol = N losses via NH3 volatilisation
Nfix = N fixed by biological fixation

Data source:
Forest fires statistics (ISTAT 1978 – 2000)
Manual UBA , 1996

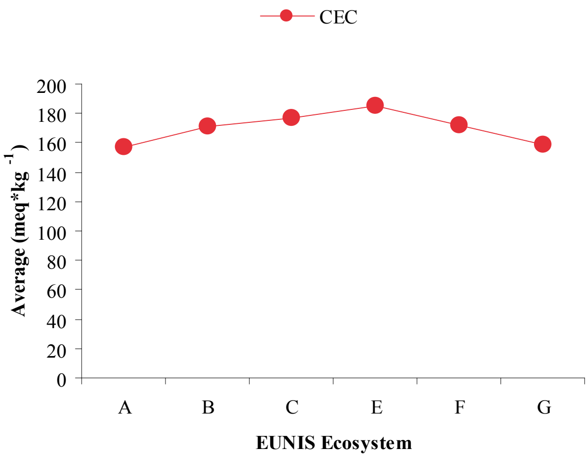
Average Nimacc



Data source:
National Forest Inventory

Mapping Manual:
Based on average annual stem growth x wood density x element concentration in wood. (Bonanni P. et al. 2001)

Average Nupt

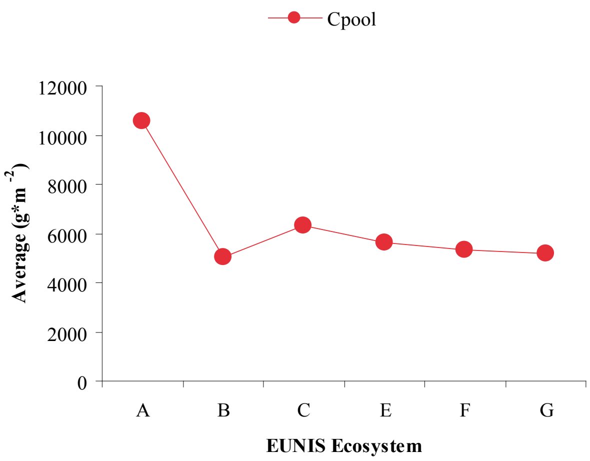


CEC= (0.44* pH+3.0) clay +
(5.1 pH −5.9) Corg

(Eq. 5.2 M&M Manual)

Data source:
European Soil Database EUSOILS
Manual for Dynamic Modelling of Soil Response to Atmospheric Deposition

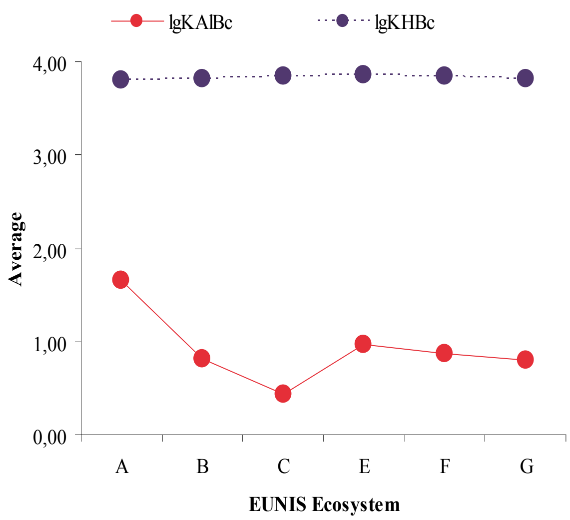
Average CEC



Cpool = 10^6*rho* soil thickness * C_{org}/100

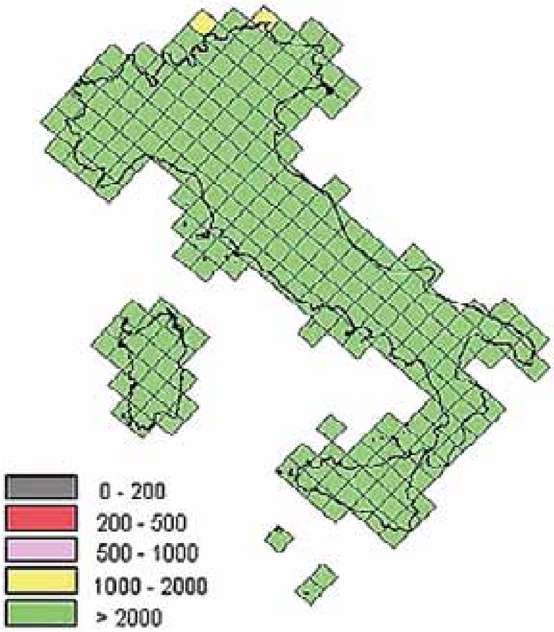
Data source:
European Soil Database EUSOILS
Manual for Dynamic Modelling of Soil Response to Atmospheric Deposition

Average Cpool

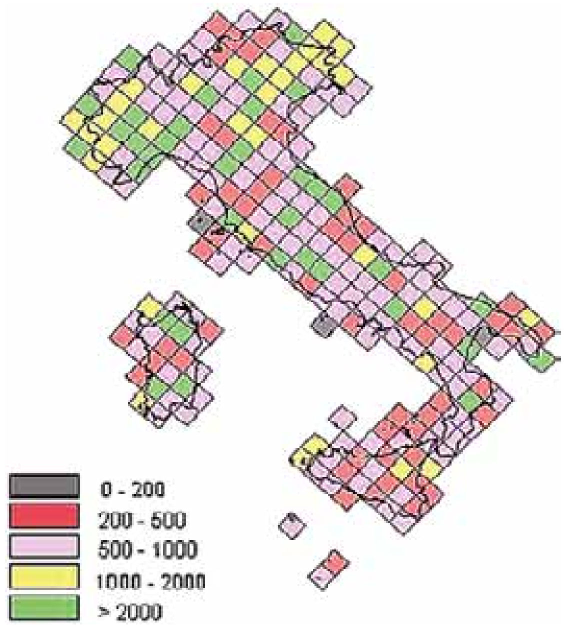


Method:
Average value weighted by ecosystem area
Manual for Dynamic Modelling of Soil Response to Atmospheric Deposition

Average lgKAlBc and lgKHBc



Critical Load of Acidity (eq*ha^{-1}*yr^{-1})



Critical Load of Nutrient Nitrogen (eq*ha^{-1}*yr^{-1})

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National Critical Load Maps

The Dutch data set on critical loads of acidity and nutrient nitrogen contains critical loads for protection of:
Forests (soils) against root damage due to elevated Al/Bc ratios and soil quality by requiring no depletion of the soils' Aluminium pool.
Plant species composition in terrestrial ecosystems (both forests and other semi-natural vegetations) against eutrophication and acidification.
Plant species composition in small heathland lakes against eutrophication.

The methods to calculate these critical loads are described in a report the evaluation of the Dutch acid rain abatement strategies (Albers et al., 2001) and in various CCE reports since 2001. Critical acid loads for the protection of forest soils were calculated with SMB. Critical loads for the protection of heathland lakes were calculated with the dynamic model, AquAcid (Albers et al., 2001). The critical loads for the protection of terrestrial vegetations were calculated with a steady-state version of SMART2-MOVE/NTM (De Vries et al., 2007).

In 2008, both the critical loads for plant species composition in terrestrial ecosystems and the critical loads for forest soils were updated by using new H-Al equilibrium constants (see below). Critical loads for small heathland lakes remained unchanged. In response to the call for data, also empirical critical loads were mapped and dynamic scenario analyses were carried out with VSD and the dynamic version of SMART2-MOVE/NTM..

Used data

Critical load and dynamic simulations were carried out for a number of different ecosystems (Table NL-1). VSD/SMB was applied for forest soils only, whereas SMART2 was used to compute critical loads and analyze scenarios for various semi-natural vegetations including forest understorey vegetations. Calculations were carried out for individual 250×250 m grid cells. The models were parameterized for different combinations of soil and vegetations types. For each individual 250x250 m grid cell the present combination of vegetation type and soil type was specified, by using an overlay of the 1:50,000 soil map and vegetation map. Soil types were differentiated into 16 major groups, including two non-calcareous sandy soils and one calcareous sandy soil, three loess soils, four non-calcareous clay soils, one calcareous clay soil and five peat soils (Van der Salm, 1999). All these soil types were further sub- divided into five hydrological classes, depending on the seasonal fluctuations of the groundwater table. With respect to vegetation types, we distinguished three groups of forests (deciduous forests, pine forests and spruce forests), and grassland and heath land (Van Hinsberg and Kros, 2001).Table NL-2 provides an overview of the various input parameters used in the different models. Parameterization of processes included in both SMB and SMART2 was kept as uniform as possible. Detailed information on model parameters can be found in the status report of 2003 (Van Hinsberg and de Vries, 2003), while an overview of the SMART2 model, along with its parameterization, is provided in Kros (1998).

Updated critical loads

The H-Al equilibrium constants in VSD and SMART2 were calibrated by comparing modelled pH change in the period 1950 -1990 with observed changes (Kros and Mol Dijkstra, 2001). This was done by selecting those constants and associated exponents from the range of values given in Dutch literature (Kros et al., 1995; Van der Salm and De Vries, 2001), which provide a close fit between modelled and measured pH values and the changes therein between 1950 and 1990.

Empirical critical loads

Ranges of empirical critical loads (Achermann and Bobbink, 2003) were assigned to the different nature targets types. Critical loads computed with SMART2-MOVE/NTM or AquAcid were used to assign one value to each target type, when calculated critical loads were inside the range of empirical critical loads. When calculated critical loads were outside of the empirical ranges, the nearest limit was used to set the empirical critical load. No critical loads were set for those nature target types for which no empirical ranges are available (i.e. fluvial, riparian or swamp woodlands and reedlands).

Dynamic modelling

Based on the software and data provided by CCE, 15 scenarios of combined N and S deposition were derived for the Netherlands. For forests, the VSD model was applied to evaluate these scenarios, for plant species composition the dynamic version of SMART2-MOVE/NTM was used.

Both models were calibrated on spatial patterns of base saturation, Cpools and C/N ratio's; for each grid cell the model was calibrated such that is reproduces the measurements in the year of observation. For the calibration, an improved map of 'observed' base saturation was constructed as described in the CCE Progress Report of 2007 (Slootweg et al., 2007).

Table NL-1 Area of receptors

Eunis codes	Short description	Area (km²)
A2	Coastal low-mid salt marshes	68
B1	Coastal habitats (dunes)	272
C3	Inland surface water bodies	2
D1	Raised and blanket bogs (in combination with F4.11: wet heaths)	76
D2+D4	Valley mires, poor fens, transition mires or base rich fens	121
E1	Dry grasslands	390
E2	Mesic grasslands	306
E3	Seasonally wet and wet grasslands	241
F4	Temperate shrub heathland	359
G1+G3	Woodlands	2729

Table NL-2 Inputs for critical loads and dynamic modelling

Variable	SMB	SMART2 (Steady State)
CNACC	Not used	Derived from N availability
CRITTYPE	Combined Al/Bc – no Al depletion	Combined pH-N availability
CRITVALUE	Forest type dependent	Nature target dependent
THICK	0.4 m	0.4 m
BULKDENS	Soil group dependent	Soil group dependent
BCDEP	Interpolated for 1×1 km cells	Interpolated for 1×1 km cells
BCWE	Soil group dependent	Soil group dependent
BCUPT	Function of growth and nutrient contents (for combinations of forest/soil types)	Modeled from growth and BC availability
QLE	Derived from a hydrological model	Derived from a hydrological model
LGKALOX	Soil group dependent	Soil group dependent
EXPAL	Soil group dependent	Soil group dependent
LHKHBC	Calibrated using observed base saturation	Calibrated using observed base saturation
LGKALBC	Calibrated using observed base saturation	Calibrated using observed base saturation
PCO2FAC	Separate values for sand, clay and peat soils	Separate values for sand, clay and peat soils
NIMACC	Soil group dependent	Soil group dependent
NUPT	Function of growth and nutrient contents (for combinations of forest/soil types)	Modeled from growth, N demand and N availability
FDE	Soil group dependent	Soil group dependent
CEC	Soil group dependent	Soil group dependent
BSAT	Spatial patterns based on regression with environmental factors	Spatial patterns based on regression with environmental factors
YEARBSAT	1995	1995
CPOOL	Soil group dependent	Soil group dependent
CNRAT0	Soil group dependent	Soil group dependent
YEARCN	1995	1995

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Dynamic modelling

Modelling of aquatic ecosystems (lakes) have been carried out for the entire country using the MAGIC model (Cosby et al., 1985; Cosby et al., 2001). The model was calibrated to observational data from 990 of the 1007 statistically selected lakes in the 1995 National lake survey (Skjelkvåle et al., 1996). (17 lakes of the total 1007 lakes in the survey were disregarded due to very high phosphorus concentrations (and ANC) from local pollution, extremely high sea salt concentrations or inconsistencies in the catchment characteristics data available.) The model was calibrated to observed water chemistry for each of the lakes and to soil base saturation from nearest available (or most relevant) sample. In the automatic calibration routine of MAGIC the following switches were set: BC optimizer (weathering calibration): on, SO4 adsorption optimizer: off, soil pH optimizer: on, N dynamics optimizer: off (this means that nitrogen uptake in the catchment was assumed proportional (with a constant proportion) to the input at all times).

Var	Unit	Min	Max	Assumptions, data sources and justifications
EcoArea	km²	2.7	204.6	We consider 100% of the land area to contain watersheds for lakes and rivers. The reported values are based on grid cells (see text) reported for each grid cell. This has not been split on different EMEP grid cells in cases were our small grid cells are split between two EMEP grid cells.
CLmaxS	eq ha^{-1} a^{-1}	56.7	20404	Calculated with FAB model (according to Mapping Manual, except BC^*_0 taken from MAGIC calibrations (1860))
CLminN	eq ha^{-1} a^{-1}	32.0	504	
CLmaxN	eq ha^{-1} a^{-1}	122	24873	
CLnutN	mgN m^{-2} a^{-1}	500	2000	Empirical values taken as minimum of range suggested in mapping manual
crittype		5	5	ANC is used as criterion for all lakes
critvalue	µeq L^{-1}	1.3	50	Variable ANC_{limit}
SoilYear		1985	2000	
ExCa	%	3.0	69.4	Lake catchment split into 4 categories: i) Forest area, taken from nearest relevant soil sampling locations (National Forest Inventory) for the percent forest in the lake catchment. ii) Peat area, taken from Langtjern soil pits no. 2 and 3 (1991 and 2000 average). iii) Non-forested upland, assigned county-wise, according to available data. iv) Open water, including lake itself. See details in text below.
ExMg	%	0.7	26.8	
ExNa	%	0.3	13.7	
ExK	%	0.7	11.0	
thick	m	0.11	1.3	
BulkDens	g cm^{-3}	0.19	1.18	
CEC	meq kg^{-1}	8.4	533	
Porosity	%	50	50	Assumption. Constant value used for all sites.
UptCa	meq m^{-2} a^{-1}	0.00	37.3	Based on National Forest Inventory. Same as in critical loads database: value for the 12x12 km² grid cell in which the lake was located.
UptMg	meq m^{-2} a^{-1}	0.00	7.0	
UptK	meq m^{-2} a^{-1}	0.00	8.1	
UptNa	meq m^{-2} a^{-1}	0.00	0.00	
UptSO4	meq m^{-2} a^{-1}	0.00	0.00	
HlfSat	µeq L^{-1}	0.00	0.00	Assumption. Constant value used for all sites.
Emx	meq kg^{-1}	0.00	0.00	Assumption. Constant value used for all sites.
Nitrif	%	100	100	Assumption based on the fact that ammonium concentrations are very low.
Denitrf	%	0.00	0.00	
DepYear		1995	1995	
Cldep	eq ha^{-1} a^{-1}	17.5	11502	Deposition flux of chloride, sat equal to catchment output flux
Cadep	eq ha^{-1} a^{-1}	0.7	432	Calculated from [Cl^-] using standard sea salt ratios and assuming no non-sea salt deposition.
Mgdep	eq ha^{-1} a^{-1}	3.4	2208	
Nadep	eq ha^{-1} a^{-1}	15	8257	
Kdep	eq ha^{-1} a^{-1}	0.3	216	
NH4dep	eq ha^{-1} a^{-1}	4.8	530	Calculated from observed ratios in deposition to SO_4. NO_3 deposition was increased to make net flux always negative or 0 (1 case of the 60). SO4 deposition was calculated from runoff flux and background deposition from CCE scenarios adjusted for SO_4 weathering in cases were excess SO_4 output flux were >100 meq/m²/yr (56 of the 990 lakes).
NO3dep	eq ha^{-1} a^{-1}	9.1	897	
LakeYear		1995	1995	1995 National lake survey.
Calake	µmol L^{-1}	2.0	1801	
Mglake	µmol L^{-1}	0.7	821	
Nalake	µmol L^{-1}	3.5	905	
Klake	µmol L^{-1}	0.5	81.8	
NH4lake	µmol L^{-1}	0.0	11.6	
SO4lake	µmol L^{-1}	4.2	583	
Cllake	µmol L^{-1}	5.6	832	
NO3lake	µmol L^{-1}	0.07	97.1	
DOC	µmol L^{-1}	8.3	1208	
RelArea	%	0.24	78.1	Data for each catchment
RelForArea	%	0.0	95.7	Data for each catchment
RetTime	yr	0.5	20	Assumption. 3 classes, by lake size.
Qs	m	0.16	5.2	Runoff taken from digital 30-year normal (1961-1990) runoff database.

Var	Unit	Min	Max	Assumptions, data sources and justifications
expAllake		3.00	3.00	Assumption. Constant value used for all sites.
pCO2	%	0.06	0.06	Assumption. Constant value used for all sites.
Cased	m a^{-1}	0.00	0.00	Assumption. Constant value used for all sites.
Mgsed	m a^{-1}	0.00	0.00	
Nased	m a^{-1}	0.00	0.00	
Ksed	m a^{-1}	0.00	0.00	
SO4sed	m a^{-1}	0.00	0.00	
Clsed	m a^{-1}	0.00	0.00	
NH4sed	m a^{-1}	0.00	0.00	
NO3sed	m a^{-1}	0.00	0.00	
UptNH4lake	%	0.00	0.00	Assumption. Constant value used for all sites.
UptNO3lake	%	0.00	0.00	Assumption. Constant value used for all sites.

Atmospheric deposition history was provided by CCE for EMEP grid cells and a sequence for each grid cell assigned to the lakes with each cell. After calibration, all 14 scenarios were run for all 990 lakes. In order to get a reasonable coverage within each EMEP grid cell, the calibrated lakes were then used to assign scenarios to all grid cells (1/4*1/8 degree) in the Norwegian critical loads database (2304 cells) using a matching routine called "MAGIC library" (IVL, 2007) (see also country report for Sweden). The 2304 grid cells were matched to the 990 lakes to which the model was calibrated according to a Eucledian distance routine based on water chemistry and location. Each of the 2304 grid cells was thus assigned a MAGIC modelled lake. The Norwegian grid cells were only matched to Norwegian lakes (unlike in the 2007 submission, where they might also have been matched to Swedish lakes. Input data ranges and data sources are described in table NO-1.

Ranges of model inputs and parameters (for the 990 calibrated lakes) and comments on their sources and justifications are listed in table NO-1.

Soil data calculations and assignments

Forest soil data available from the national forest inventory on a grid cell basis. Of the 990 lakes, 342 were located in a grid cell were forest data were available (and hence forest present). Additional 221 lakes had forest in their catchments; for these lakes soils data were taken from nearest neighbor with data. 427 of the lakes had no forest in the catchments.

Mountain/upland/heath soils: Assigned from data available from different research projects.

Data sources used:

- Vest-Adger and Rogaland: About 30 sites in heathlands sampled in 1998 by NIJOS using the same protocol as for the national surveys of forest soils. Grid based site selection. In addition one site (Stavsvatn) in Telemark. Reference: (Wright et al., 1999)
- Yndesdal, Hordaland/Sogn og fjordane. Catchment weighted average of data from about 8 soil samples collected in 2003. Reference: (Bjerknes et al., 2004)
- Dalelva, Finnmark. Soil data from national monitoring sampled in 1990 with original data in the SFT 1991 annual report. Aggregated data used here are from the MAGIC calibration to Dalelva. Reference: (Wright and Traaen, 1992)
- Storgama, Vikedal, Gaular, Naustal, Sogndal, Risdalsheia. Soil data from national monitoring sampled in 1980s and with original data reported in SFTs annual reports. RAIN data from NIVA, in ARRR. Aggregated data used here are from the MAGIC calibrations. Reference: (Wright et al., 1990)

- Vosso, Hordaland. Catchment weighted averages from about 6 soil samples collected in 2000. Reference: (Kroglund et al., 2002)
- Hardangervidda. (Prikkaureprosjekt). Averages from 6 soil samples collected in 2000. Reference: (Fjellheim et al., 2002)
- Rondane. Data from Dahl 1982 (Dahl, 1982), averaged for MAGIC calibration. Reference: (Skjelkvåle et al., 1997)
- Assignment of data for the heathland/mountain area was done based on county (fylke) in which each lake was located.
- Finnmark: Dalelva
- Hordaland, Rogaland, Agder: “nearest neighbor”
- Østfold, Hedemark: Rondane
- Telemark, Vestfold, Akerhus: “nearest neighbor”.
- Buskerud: Hardangervidda for northern sites, Stavsvatn for southern sites (nearest neighbor)
- Oppland: Rondane
- Møre og Romsdal: Naustdal
- Trøndelag, Nordland: Daleva

Peat soil data were taken as the Langtjern average peat soil for all locations. A depth of 50 cm was used. The three types of soil for each site were area-weighted for each parameter.

Critical loads for surface waters

Calculations were carried out with the FAB model in accordance with the Mapping Manual.

Empirical critical loads for nutrient nitrogen

The empirical critical loads for nutrient nitrogen were updated using the harmonised land use map by SEI provided by the CCE and the lower limits of the critical load values given in the Mapping Manual (UBA, 2004).

Map code category	EUNIS code	Critical limit (mg m^{-2} yr^{-1})
0	0	0
301	C1	500
302	C2	500
401	D1	500
501	E1	1000
502	E2	1000
503	E3	1000
504	E4	500
601	F1	500
602	F2	500
603	F3	500
604	F4	1000
701	G1	1000
703	G3	1000
704	G4	1000
804	H4	500
805	H5	500
901	I1	2000
1000	J	

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Modelled critical loads and dynamic data

Introduction

In response to the current CCE call for data the Polish NFC is submitting critical loads calculations results and dynamic modelling outputs for six distinct terrestrial habitats (Table PL-1). The spatial resolution applied is determined by 1 km² grid squares which contains 1 ha or more of the habitat.

Table PL-1: Ecosystems subject to critical load calculations and dynamic modelling

Ecosystem	EUNIS code	Area km²	No of grid cells	Percentage of receptor area
Broad-leaved forest	G1	16056	30153	17.8
Coniferous forest	G3	48398	88151	53.6
Mixed forest	G4	23107	42992	25.6
Natural grasslands	E	577	1145	0.6
Moors and heath land	F	78	128	0.1
Mire, bog and fen habitats	D	2114	3956	2.3
	Total	**90330**	**166524**	**100.0**

Table PL-2:Data description, methods and sources

Critical loads parameter	Method or value used	Data source
$CL_{max}(S)$ [eq $ha^{-1}a^{-1}$]		Calculated by VSD
$CL_{min}(N)$ [eq $ha^{-1}a^{-1}$]		Calculated by VSD
$CL_{max}(N)$ [eq $ha^{-1}a^{-1}$]		Calculated by VSD
$CL_{nut}(N)$ [[eq $ha^{-1}a^{-1}$]		Calculated by VSD
$nANC_{crit}$ [eq $ha^{-1}a^{-1}$]		Calculated by VSD
lgKAlBc		Calculated by VSD
lgKHBc		Calculated by VSD
CNrat0		Calculated by VSD
cNacc [meq m^{-3}]		Revised values from Mapping manual Table 5.7
crittype	2	
critvalue	[Al]=0.2 [eq m-3]	
thick	0.5 [m]	
bulkdens [g cm^{-3}]		II-level Forest Monitoring System operated by the Forest Research Institute (Wawrzoniak et al., 2005)
BC_{dep} [eq $ha^{-1}a^{-1}$]	The bulk deposition values of base cations were estimated from the reported wet deposition data multiplied by dry deposition factors derived from throughfall data provided by the integrated monitoring surveys	Data for 2006 from monitoring network operated by the Institute of Meteorology and Water Management – Wroclaw Branch under the authority of Main Inspectorate of Environment Protection
BCu [eq ha-1a-1]	New data on stem and branches harvesting for the period 1985-1998 from the Central Statistical Office – Section Forestry database were used.	The mean element contents for stems and branches were taken from the Mapping Manual (UBA, 2004).
BC_w [eq $ha^{-1}a^{-1}$]	Soil type – texture approximation according to Mapping Manual (UBA, 2004).	Soil types identified from the Soil Geographical Database of Europe
Q_{le} [mm/a]	Actual evapotranspiration was calculated by the method of Federer (1982) based on the Monteith (1964) equation. Interception was computed according to (deVries at al., 2003)	Historical time series of meteorological data acquired from Climate Change Unit of the University of East Anglia (New,et al., 2002)
lgKAlox [m^6/eq^2]	300 [m^6/eq^2] for mineral soils	Mapping Manual (UBA, 2004).
expAl	3	Mapping Manual (UBA, 2004).
pCO_2fac	15	Mapping Manual (UBA, 2004).
cOrgacids [eq m^{-3}]	Based on DOC values estimated form soil properties and vegetation types	II-level Forest Monitoring System operated by the Forest Research Institute (Wawrzoniak et al., 2005)
Nimacc [eq $ha^{-1}a^{-1}$]	A temperature dependent long-term immobilization factor was applied, ranging from 71 to 356 [eq ha-1a-1]	CCE Status Report'2001 (Posch et al., 2001)
Nu [eq $ha^{-1}a^{-1}$]	New data on stem and branches harvesting for the period 1985-1998 from the Central Statistical Office – Section Forestry database were used.	The mean element contents for stems and branches from the Mapping Manual (UBA, 2004).
fde	Depending on soil clay content, values from 0.1 to 0.8 were applied	CCE Status Report'2001(Posch et al., 2001)
Nde [eq $ha^{-1}a^{-1}$]	Ignored	
CEC [meq kg^{-1}]		II-level Forest Monitoring System operated by the Forest Research Institute (Wawrzoniak et al., 2005)
bsat		II-level Forest Monitoring System operated by the Forest Research Institute (Wawrzoniak et al., 2005)
yearbsat	2004	
Cpool [g m^{-2}]		
CNrat		II-level Forest Monitoring System operated by the Forest Research Institute (Wawrzoniak et al., 2005)
yearCN	2004	

Methods and data sources

Critical loads were calculated using the VSD model and dynamic modelling was performed by use of the VSD/ACCESS model version supplied by the CCE. In general the input parameters were estimated in accordance with the Mapping Manual procedures. A detailed description of the input data and calculation methods used is given in Table PL-2.

The main source of soil data was the II-level Forest Monitoring System operated by the Forest Research Institute within the National Monitoring of Environment funded by the Chief Inspectorate of Environment Protection. Data from 148 forest monitoring sites were regionalized to fit to a grid system determined by a 1 km^2 size grid cell.

Critical load maps

The resulting critical load maps for $CL_{max}S$ and $CL_{nut}N$ are shown in Figures PL-1.

Figure PL-1. Critical loads.

Institute of Environmental Protection
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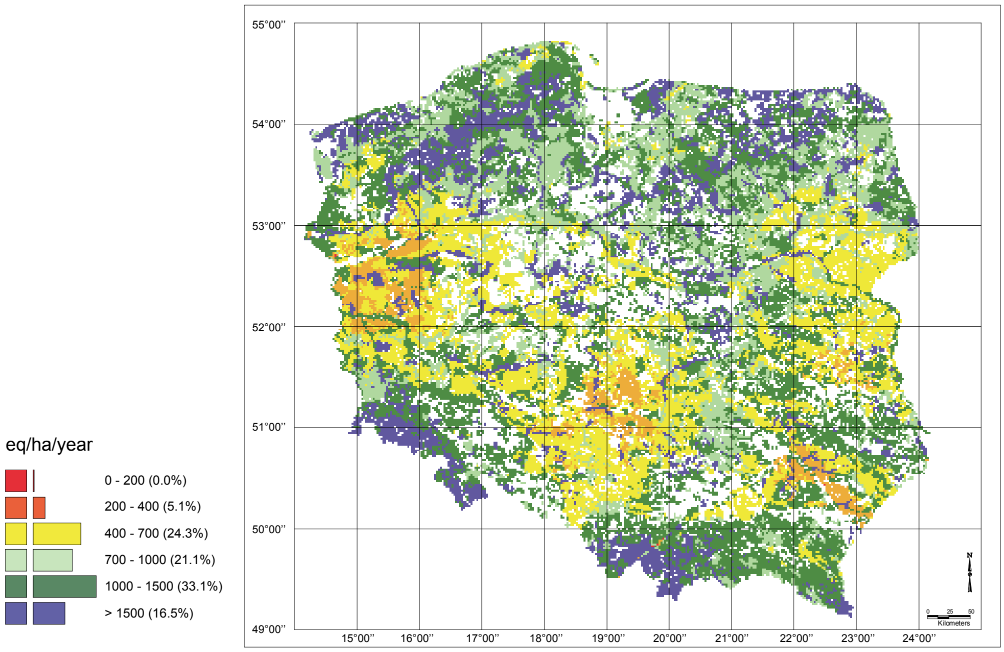
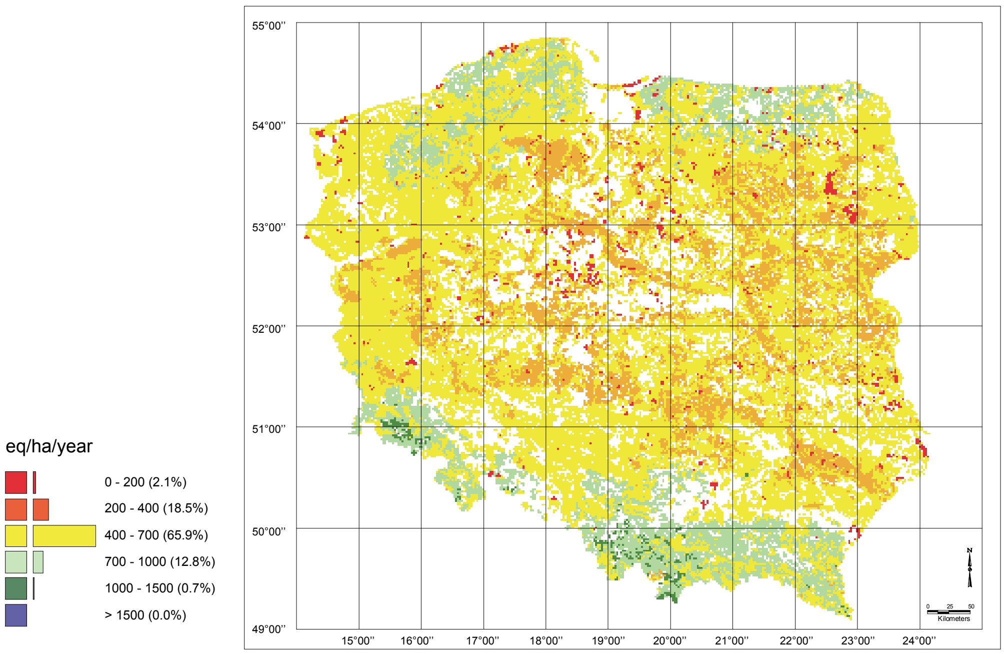
Grunwaldzka Str. 7b/2, 41-106 Siemianowice Sl., Poland

Maximum critical loads of sulphur 2008



Critical loads of nutrient nitrogen 2008

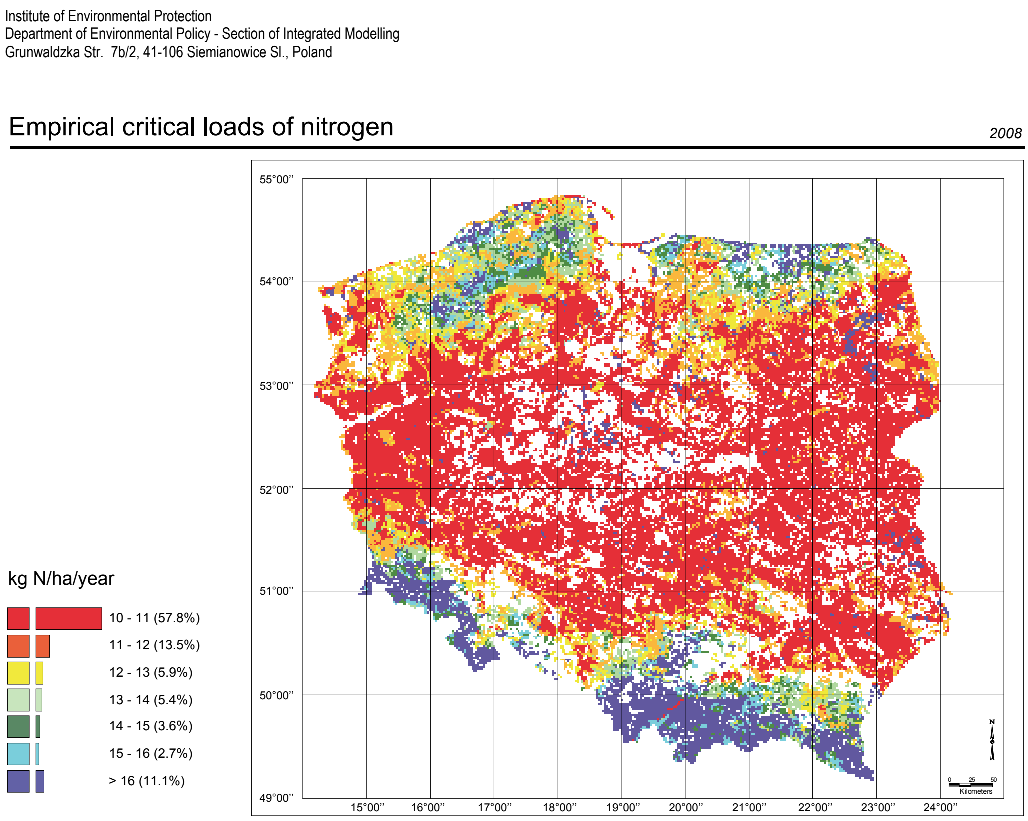


Empirical critical loads of nutrient nitrogen

Empirical critical loads of nutrient nitrogen were slightly modified against those submitted in 2007 following the recently updated content of Table 5.1. of the Mapping Manual and with regard to the adaptation rules summarized in Table 5.2. of this manual.

Figure PL-2 presents the spatial distribution of empirical critical loads values of nutrient nitrogen.

Figure PL-2. Critical loads.



Data measured

Atmospheric depositions of Ca, Mg, K, Na and Cl are all measured by the Institute of Meteorology and Water Management the Wroclaw Branch within the State Monitoring of Environment.

Soil bulk density, CEC, base saturation and C:N ratio are measured by the Forest Research Institute within the II-level Forest Monitoring Net functioning under the ICP Forests.

Acknowledgements

The Polish Ministry of Environment and the National Fund for Environment Protection and Water Management provided formal foundation and financial support to this study what is gratefully acknowledged. The help of the CCE staff in deriving some of input data and the implementation of the VSD model is particularly appreciated.

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Bucharest

Data sources

Calculation methods for critical loads of acidity and nutrient nitrogen

The first national critical load dataset of Romania was computed using the steady-state mass balance approach. About 42 % of the area of Romania is covered by forests for which critical loads of acidity and nutrient nitrogen are calculated in accordance to the methods described in the Mapping Manual (UBA 2004, updated version 2007). The critical load database of Romania consists of 97 964 records, a detailed description of the data and the methods for derivation is given in Table RO-1.

Critical loads of acidity, $CL_{max}(S)$:

The highest critical loads of acidity [$CL_{max}(S)$] with values up to 10 keq ha^{-1} a^{-1} are observed in the riverside tree galleries along Dunarea, Siret, Mures and their tributaries. High weathering rates of alluvial wet clay soils in addition to low sensitivity of the tree species of grey oak-ash-poplar forests result in very high $CL_{max}(S)$. In the Romanian Land situated north of Danube and south of the Southern Carpathians and Dobrogea the less sensitive mixed oak forests on soils (rendzina, chromic luvisols) with high weathering rates of base cations and relatively high deposition rates of base cations coming with the south wind result in high critical loads of acidity with values from 3 to 10 keq ha^{-1} a^{-1}. Medium high critical loads (about 1.5 – 3 keq ha^{-1} a^{-1}) are located in the lower (colline) mountains of Carpatii Meridionali (Southern Carpathians) and in Transylvania region. Here sensitive dystric cambisol soils combined with less sensitive beech and mixed beech forests cause a medium critical load. The higher mountains of Carpatii Meridionali, Carpatii Orientali (Eastern Carpathians) and in Transylvania region with dystric cambisol soils are combined with more sensitive spruce forests (critical loads from 0,5 to 1,5 keq ha^{-1} a^{-1}). The lowest critical loads (<0,5 keq ha^{-1} a^{-1}) have to be allocated to the lowlands of Transylvania and Moldova regions because of relatively low deposition rates of base cations behind the mountain barrier against the south wind. The regional distribution of critical loads of acidity is shown in Figure RO-1.

Critical loads of nutrient nitrogen, $CL_{nut}(N)$:

The highest critical loads of nutrient nitrogen (>25 kg N ha^{-1} a^{-1}) can be observed on rich loess soils with very high acceptable leaching rates caused by high precipitation at the lower west and south of Carpatii Meridionali.

Similar to the critical loads of acidity the lower (colline) mountains of Carpatii Meridionali and in the Transylvania region also have high critical loads of nutrient nitrogen (about 20-25 kg N ha^{-1} a^{-1}). A high uptake by harvesting of the oak and beech trees on rich stagno-luvisols is accompanied by a relatively high denitrification rate. Medium high critical loads (15- 20 kg ha^{-1} a^{-1}) are located in Moldova region and in the lowlands between the Carpathian Mountains and Transylvania. Eutric regosols and gleyic luvisols could cause a medium growth rate, high denitrification and a medium leaching rate. The lowest critical loads values (7,5 - 15 kg ha^{-1} a^{-1}) are observed in the mountains of Carpatii Meridionali, Carpatii Orientali and Transylvania region with poor dystric cambisol soils and low uptakes, but high immobilisation rates caused by low temperature in the high mountains. The regional distribution of critical loads of nutrient nitrogen is shown in Figure RO-2.

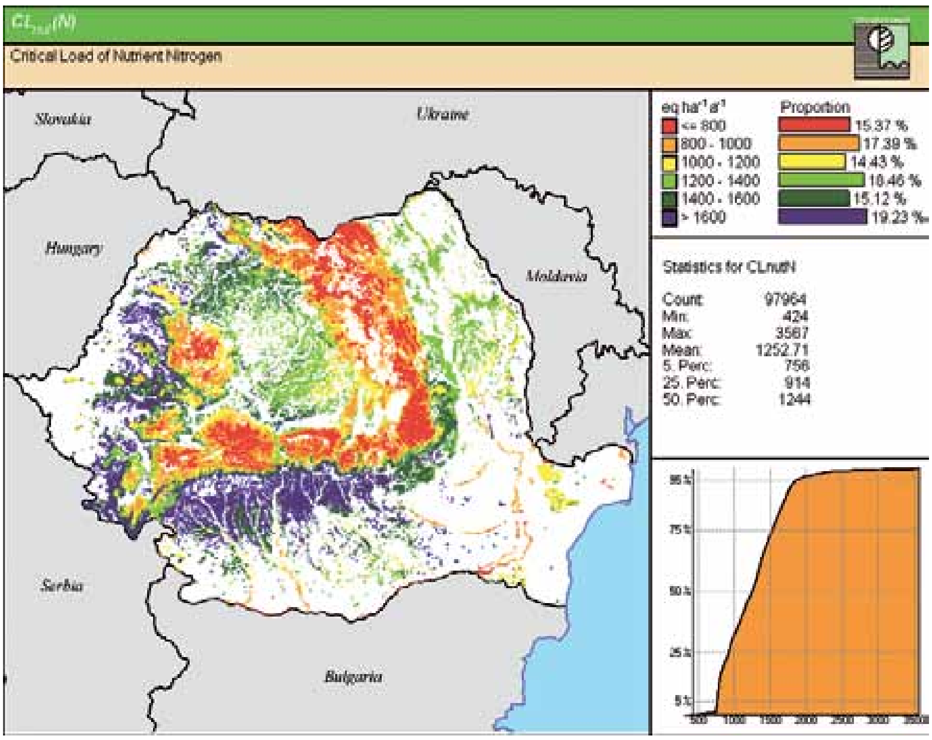

Figure RO-1. Critical loads of acidity, $CL_{max}(S)$ in Romania, receptor forest ecosystems.

Table RO-1: National critical load database and calculation methods / approaches

Parameter	Term	Unit	Description
Critical load of acidity	$CL_{max}S$	eq ha^{-1} a^{-1}	Manual, equation 5.22
	$CL_{min}N$	eq ha^{-1} a^{-1}	Manual, equation 5.25
	$CL_{max}N$	eq ha^{-1} a^{-1}	Manual, equation 5.26
Critical load of nutrient nitrogen	$CL_{nut}N$	eq ha^{-1} a^{-1}	Manual, equation 5.5
Acid neutralisation capacity leaching	$nANC_{(crit)}$	eq ha^{-1} a^{-1}	Manual; the minimum value of the following approaches using different chemical criteria was taken for the calculation (see crittype in the call for data): 1 [Al]:[Bc] – equation 5.31 2 [Al] – Derived from $Al_{le(crit)}$ in equation 5.32-5.34 by Al_{le}/Q_{le} 4 pH – equation 5.35 6 [Bc]:[H] – equation 5.36
Acceptable nitrogen leaching	$N_{le(acc)}$	eq ha^{-1} a^{-1}	Manual, equation 5.6; see Table 6 of the CCE instructions for $[N]_{crit}$ values
Thickness of soil layer	thick	m	Actually rooted zone
Bulk density of the soil	bulkdens	g cm^{-3}	Romanian general soil map (FAO classes), ICPA – Romanian Institute for Soil science
BC deposition	Ca_{dep}	eq ha^{-1} a^{-1}	EMEP Deposition Data
Weathering of base cations	Ca_{we}	eq ha^{-1} a^{-1}	Manual, equation 5.39, Manual, table 5.12-5.14
Uptake of base cations by vegetation	Ca_{upt}	eq ha^{-1} a^{-1}	Manual, equation 5.8 (without branches) Manual, table 5.8 for element contents, Jacobson et al 2002
Amount of water percolating through the root zone	Qle	mm a^{-1}	Hydrological Data from the National Meteorological Administration (INMH)
Nitrogen immobilisation	N_{imm}	eq ha^{-1} a^{-1}	Vegetation period dependent, coniferous forest 2-5 kg ha^{-1} a^{-1}, all other vegetation types 1-4 kg ha^{-1} a^{-1}
Nitrogen uptake by vegetation	N_{upt}	eq ha^{-1} a^{-1}	Manual, equation 5.8 (without branch) Manual, table 5.8 for element contents, Jacobson et al 2002
Denitrification factor	f_{de}	-	Depending on pore volume for pF>4.2, influence of (ground) water on the horizons and nutrient availability according to Manual Table 5.9
Exchange constant for Al vs. Bc	lgKAlBc		Gapon, based on Manual, Table 6.4
Exchange constant for Al vs. H	lgKAlH		Gapon, based on Manual, Table 6.4
EUNIS code	EUNIScode		Forest Inventory Data from Forest Research and Management Institute (ICAS) and Schlutow (2004)

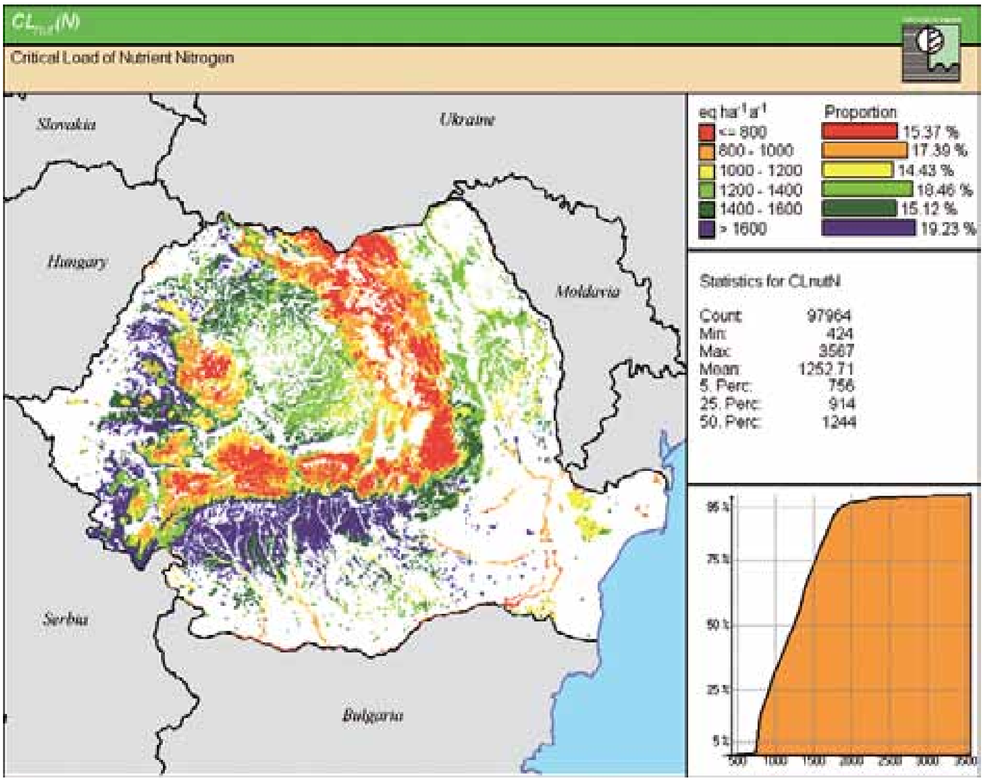

Figure RO-2: Critical loads of nutrient nitrogen, $CL_{nut}(N)$ in Romania, receptor forest ecosystems.

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C. Iacoban -Forest Research and Management Institute of Romania – ICAS Câmpulung Moldovenesc: Deposition data set for the Level II ICP Forests plots established in Romania (period 2002 – 2005);

National Meteorological Administration, Bucharest, Romania - Department of Climatology. Precipitation map (period 1961-1990)

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Acknowledgements

Many thanks to people from CCE: Jean-Paul Hettelingh, Max Posch and Jaap Slootweg for all their support, valuable comments and technical help and to the project "Developing expertise, data and models for establishing a Slovenian NFC for the support of European effect-based air pollution abatement policies", case number: 901 7.06.1 068, funded by the Dutch Ministry of Housing, Spatial Planning and the Environment and the Slovenian Ministry of Environment and Spatial Planning.

Table SI-1: EUNIS forest habitat types identified in Slovenia which all receptors were ascribed to. Additionally, some environmental characteristics (ordinal scale: 1 small value, 5 high value) of EUNIS categories are shown.

EUNIS code	Habitat type description	Temperature class	Soil moisture class	Base cation availability class	N and P limitation	Management intensity
F2.42	Outer Alpine *Pinus mugo* scrub	1	3	3	3	1
F2.47	Pelago dinaride *Pinus mugo* scrub	1	3	3	3	1
G1.1112	Eastern European poplar-willow forests	3	4	2	1	1
G1.1211	Alpine grey alder galleries	3	4	3	3	3
G1.2111	Sedge ash-alder woods	3	5	3	3	3
G1.6334	Southeastern Alpine bittercress beech forests	2	3	3	3	3
G1.6351	Sub-Pannonic beech forests	4	3	2	2	4
G1.676	Pre-Alpine hop-hornbeam beech forests	4	2	3	4	3
G1.6C1	Illyrian woodrush-beech forests	3	3	2	2	4
G1.6C21	Illyrian collinar neutrophile beech forests	3	3	5	4	5
G1.6C22	Illyrian montane fir-beech forests	3	3	5	4	5
G1.6C223	Illyrian high montane fir-beech forests	2	3	2	2	4
G1.6C31	Illyrian coastal beech forests	4	2	4	4	3
G1.6C4	Illyrian subalpine beech forests	1	3	3	3	1
G1.7431	Illyrian hop-hornbeam mixed oak woods	4	1	3	3	2
G1.7432	Illyrian black pea sessile oak woods	3	3	4	4	3
G1.7C14	Illyrian hop-hornbeam woods	4	1	3	3	2
G1.A1A1	Illyrian sessile oak-hornbeam forests	3	4	4	4	4
G1.A1A2	Illyrian pedunculate oak-hornbeam forests	4	5	3	3	5
G1.A463	Illyrian ravine forests	3	4	3	3	2
G3.11221	Illyrian neutrophile spruce fir forests	2	4	3	3	5
G3.124	Dinaric calcareous block fir forests	2	3	3	3	2
G3.1322	Illyrian acidophile fir forests	3	3	2	2	4
G3.135	*Bazzania* fir forests	2	4	2	2	4
G3.1B21	*Adenostyles glabra* subalpine spruce forests	2	3	3	3	2
G3.1C2	Calciphile montane inner Alpine spruce forests	2	3	2	4	2
G3.1F3	Peri-Alpine bazzania spruce forests	2	4	1	3	4
G3.1F42	Illyrio-Alpine montane beech spruce forests	3	3	2	3	3
G3.1F51	Illyro-Dinaric cold station spruce forests	1	3	3	3	2
G3.425	Eastern Alpine acidophilous Scots pine woods	3	3	2	2	3
G3.441	Alpine spring heath Scots pine forests	3	2	3	3	2
G3.4C52	Dinaric dolomite Scots pine forests	4	2	3	3	2
G3.5215	Illyrian sub-Mediterranean *Pinus nigra* forests	5	1	3	3	2
G3.E	Nemoral bog conifer woodland	1	5	1	5	1

Introduction

The current report on critical loads and dynamic modelling of air pollutants under the LRTAP convention is the first attempt of Slovenian project team (SFI & BF) in cooperation with CCE (J. Slootweg, M. Posch) to evaluate forest ecosystems of Slovenia for their susceptibility to atmospheric depositions of nitrogen and sulfur, to derive maps of possible exceedances and to compute the target loads for different deposition scenarios. This early attempt inevitably has some flaws due to lack of high precision data (particularly soil data) which makes the final results less accurate. Some refinement in weathering rates, uptake quantities, and especially in depositions of base cations and pollutants is necessary in the future.

Modelling was done exclusively on forest ecosystems of Slovenia. Due to high percentage of forests in Slovenia the area being included in the modelling is high and reaches 54.2% of the national territory. The % of forest area is smaller (cca. 6%) due to older version of forest community map. We are planning to include the semi-natural grasslands in the modeling process in the future due to their high ecological value.

Receptor definition

Receptor is the fundamental unit which all the calculations and estimations regarding critical loads are made on. It is a patch of forest with its distinct plant community composition and soil and other environmental properties. Receptor responds presumably homogeneously to environmental factors e.g. air pollutants. Spatial data of Slovenian forest inventory was used to define receptors. Forest patches (plus *Pinus mugo* alpine shrublands) were classified into 34 EUNIS habitat categories (figure SI-1, table SI-1). Overlaying forest community map with the EMEP 50×50 km grid and discarding polygons smaller than 1 ha resulted in altogether 12691 receptors.

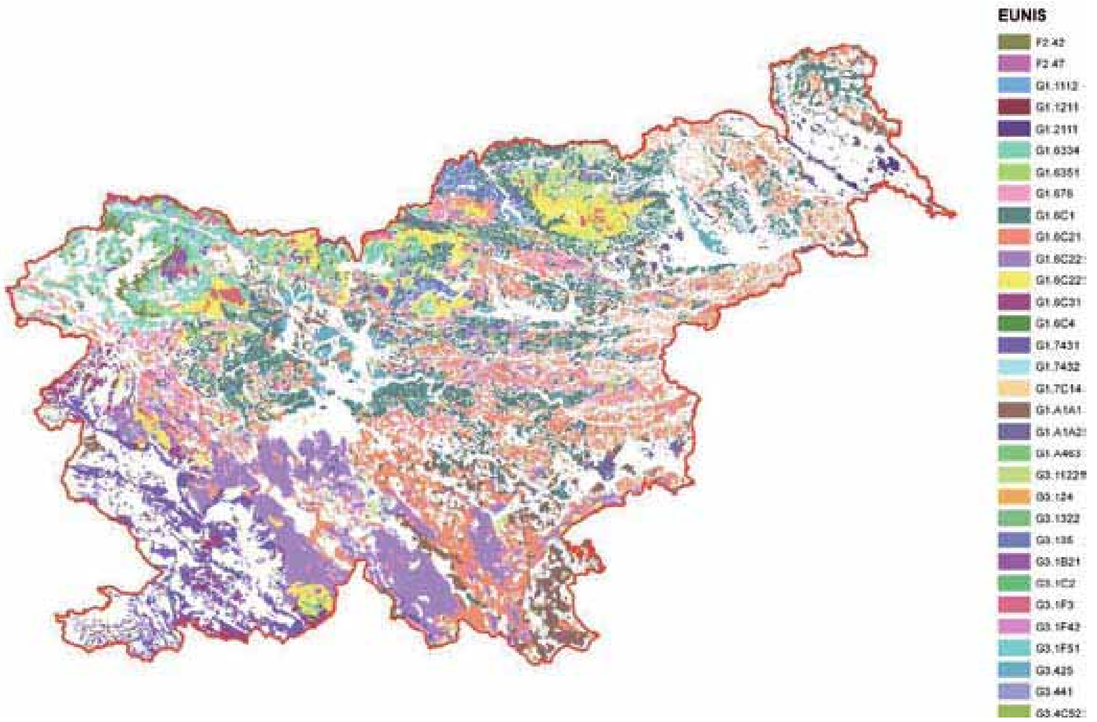

Figure SI-1: EUNIS forest habitat types and their distribution in Slovenia.

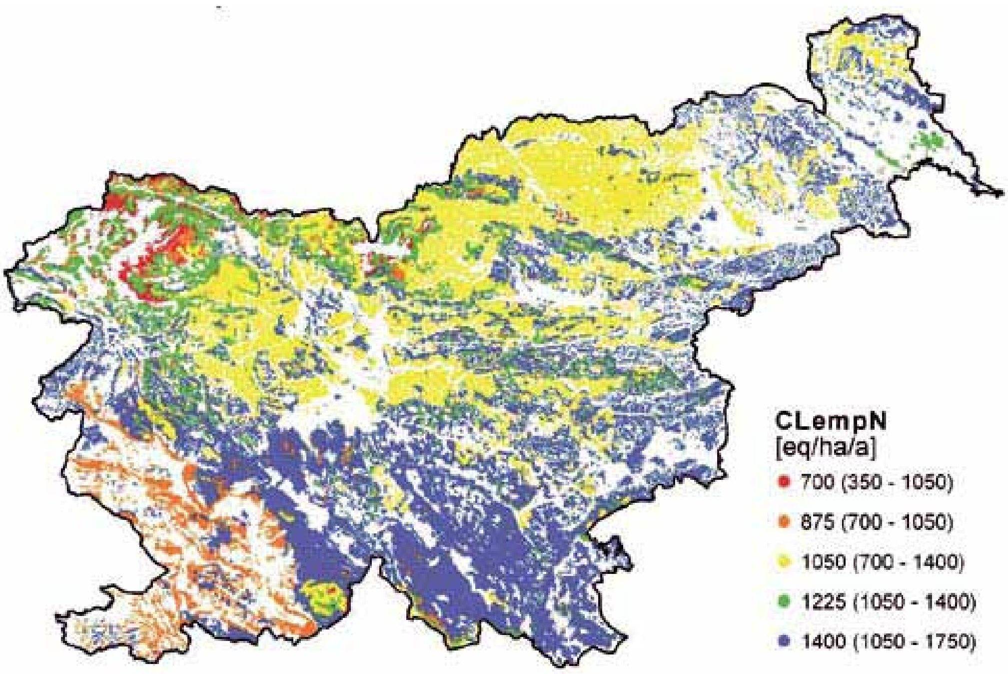

Figure SI-2: Empirical critical loads of nitrogen ($CL_{emp}(N)$) for forest ecosystems of Slovenia.

Empirical critical loads of nitrogen

In contrast to modelling activities which are based on objective, mathematical equations empirical critical loads comprise higher degree of subjectivity when deriving the values for individual (habitat) types. The mayor advantage of empirical loads is that their derivation is connected to the more relevant biological effects of air pollutants on the contrary to the modelling approach where the chemical effects of pollution are still on the forefront (despite latest improvements towards biological response models, e.g. CALLUNA, ERICA (Posch et al. 2003). Empirical critical loads are still relevant and can also be used to compare empirical values with the ones produced by the modelling approach.

For EUNIS forest habitat types found in Slovenia the empirical critical loads for nutrient nitrogen were acquired from Bobbink et al. (2002) and the Mapping Manual (2004). The results are in the table SI-2. The values of empirical nitrogen critical loads span from 5 kg/ha/a for some low tree and shrub communities from the alpine region and bog woodlands to 25 kg/ha/a for some types of beech and oak forests with the deeper soil profile. The majority of forest types are in range of maximum 10-20 kg/ha/a of N deposition.

Modelled critical loads and dynamic modelling

Input data for critical loads and dynamic modelling

Critical load calculation and dynamic modelling of air pollutants effects demand various information on climate, soil factors, base cation and air pollutant deposition, weathering rates, and nutrient uptake by plants. The majority of data need spatial denotation.

Since forest soils are the primary target of modeling critical loads (and not the forest biotic community as it should be), each receptor needs to be precisely characterized regarding its soil characteristics. Soil databases were the primary source of information.

Table SI-2: Empirical critical loads of nutrient nitrogen for each EUNIS forest habitat type found in Slovenia.

EUNIS code	Habitat type description	Critical load eq N/ha/a	Critical load kg N/ha/a
F2.42	Outer Alpine *Pinus mugo* scrub	350 - 1000	5 - 15
F2.47	Pelago dinaride *Pinus mugo* scrub	350 - 1000	5 - 15
G1.1112	Eastern European poplar-willow forests	700 - 1000	10 - 15
G1.1211	Alpine grey alder galleries	700 - 1000	10 - 15
G1.2111	Sedge ash-alder woods	1000 - 1400	15 - 20
G1.6334	Southeastern Alpine bittercress beech forests	1000 - 1400	15 - 20
G1.6351	Sub-Pannonic beech forests	700 - 1400	10 - 20
G1.676	Pre-Alpine hop-hornbeam beech forests	1000 - 1400	15 - 20
G1.6C1	Illyrian woodrush-beech forests	700 - 1500	10 - 20
G1.6C21	Illyrian collinar neutrophile beech forests	1000 - 1800	15 - 25
G1.6C22	Illyrian montane fir-beech forests	1000 - 1800	15 - 25
G1.6C223	Illyrian high montane fir-beech forests	1000 - 1400	15 - 20
G1.6C31	Illyrian coastal beech forests	1000 - 1800	15 - 25
G1.6C4	Illyrian subalpine beech forests	700 - 1000	10 - 15
G1.7431	Illyrian hop-hornbeam mixed oak woods	700 - 1000	10 - 15
G1.7432	Illyrian black pea sessile oak woods	1000 - 1400	15 - 20
G1.7C14	Illyrian hop-hornbeam woods	700 - 1000	10 - 15
G1.A1A1	Illyrian sessile oak-hornbeam forests	1000 - 1800	15 - 25
G1.A1A2	Illyrian pedunculate oak-hornbeam forests	1000 - 1800	15 - 25
G1.A463	Illyrian ravine forests	1000 - 1400	15 - 20
G3.11221	Illyrian neutrophile spruce fir forests	700 - 1400	10 - 20
G3.124	Dinaric calcareous block fir forests	700 - 1000	10 - 15
G3.1322	Illyrian acidophile fir forests	700 - 1400	10 - 20
G3.135	*Bazzania* fir forests	700 - 1400	10 - 20
G3.1B21	*Adenostyles glabra* subalpine spruce forests	700 - 1000	10 - 15
G3.1C2	Calciphile montane inner Alpine spruce forests	350 - 1000	5 - 15
G3.1F3	Peri-Alpine bazzania spruce forests	700 - 1400	10 - 20
G3.1F42	Illyrio-Alpine montane beech spruce forests	700 - 1000	10 - 15
G3.1F51	Illyro-Dinaric cold station spruce forests	1000 - 1400	15 - 20
G3.425	Eastern Alpine acidophilous Scots pine woods	700 - 1400	10 - 20
G3.441	Alpine spring heath Scots pine forests	700 – 1000	10 - 15
G3.4C52	Dinaric dolomite Scots pine forests	700 – 1000	10 - 15
G3.5215	Illyrian sub-Mediterranean *Pinus nigra* forests	700 - 1000	10 - 15
G3.E	Nemoral bog conifer woodland	350 - 1000	5 - 15

Since the existent databases in Slovenia were not established to be used for such detailed analyses it was not possible to use original, raw data. A total of 886 soil profiles were averaged to the level of 26 main soil types, present in forest ecosystems of Slovenia. These soil types were attributed to adequate EUNIS classes. Soil data included information on soil depth, bulk density, cation exchange capacity, base saturation, C:N ratio of the topsoil, soil carbon pool, soil texture, soil N and P availability (ordinal scale) and drainage class.

Additional data included:
mean annual temperature map of Slovenia 1×1 km raster grid
mean annual precipitation map of Slovenia 1×1 km raster grid
geological map of Slovenia 1:100.000
digital terrain model of Slovenia 12,5×12,5 m raster grid
data on average annual forest growth and harvesting rates for all forest types (data provided by Slovenia Forest Service, Forest inventory)

vegetation data of forest types (EUNIS classes), with the emphasis on tree species composition of forest communities 1:100.000
base cation deposition data was acquired from the modeling work of Van Loon *et al*. (2005). The values for each EMEP 50×50 km grid cell were provided by the CCE
N and S deposition time series were provided by the CCE and were included in the 2007 call for data database.

Critical loads calculation

The steady state simple mass balance model (SMB) was used to calculate critical loads of acidifying S and N compounds and eutrophying N compounds. Three critical loads connected with soil acidification were computed: maximum critical load of sulfur ($CL_{max}(S)$), maximum critical load of nitrogen ($CL_{max}(N)$) and minimum critical load of nitrogen ($CL_{min}(N)$). For individual receptor the latter three CL's define the critical load function, which delimitates "safe" deposition from the deposition which leads to the exceedances of a certain chemical criterion (e.g. Al/BC ratio).

SMB model regarding soil acidification is based on the following equations (Mapping Manual, 2004):
$CL_{min}(N) = N_u + N_i$
$CL_{max}(N) = CL_{min}(N) + CL_{max}(S)$
$CL_{max}(S) = CL(A) + BC_{dep} - Bc_u$
$CL(A) = BC_w - ANC_{le,\,crit}$

where:
N_u : net growth uptake of nitrogen (eq/ha/a)
N_i : nitrogen immobilization into the soil (eq/ha/a)
BC_{dep}: deposition of base cations (eq/ha/a)
Bc_u: uptake of base cations (except from Na+) by vegetation (eq/ha/a)
BC_w: base cations weathering rate (eq/ha/a)
$ANC_{le,\,crit}$ - defined according to one of the chemical criteria, e.g. Al/BC ratio =1, which yields:
$ANC_{le,\,crit} = 1.5 \times Bc_{le} / (Bc/Al)_{crit}$

where: $Bc_{le} = Bc_{dep} - Bc_w - Bc_u$
Regarding eutrophication effect of nitrogen deposits the following equations are needed:
$CL_{nut}(N) = N_u + N_i + N_{le(acc)} / (1 - f_{de})$
$N_{le(acc)} = Q \times [N]_{acc}$

where:
$N_{le(acc)}$ = leaching of nitrogen at critical load (eq/ha/a)
$[N]_{acc}$ = concentration of nitrogen in the soil solution at critical load (eq/m3)

Dynamic modelling

For dynamic modelling Very Simple Dynamic model (VSD) (Posch et al. 2005) was used. VSD is the simplest extension of the SMB model with the time-dependent mass balances, cation exchange and C:N ratio dependent N immobilization as a group of parameters which make the model dynamic. The purpose of dynamic modelling approach is to define target loads of air pollutants and to determine the response of an ecosystem or more precisely soil solution to the variation in atmospheric deposition of (acidifying) compounds. The Access version of the VSD model was used. The model was calibrated on spatial data of base saturation, soil carbon pools and C:N ratio. Temporal calibration was performed by setting the appropriate year of soil data measurements (set to 1995). The model was run in Gapon "mode" (i.e. Gapon echange equations were used). Parameters needed to run the model and their methods of derivation/acquisition are described in table SI-3.

Table SI-3: Parameters needed to run the VSD model, their description and methods and sources of their calculation / acquisition.

PARAMETER	Description	Data source / computation method
EmpSiteID	ID of the site	
Lon	Geographic longitude	
Lat	Geographic latitude	
I50	EMEP50 horizontal coordinate	
J50	EMEP50 vertical coordinate	
EcoArea	Receptor area	Based on forest vegetation maps of Slovenia
CLmaxS	Max critical load of sulfur	Computed by VSD
CLminN	Max critical load of nitrogen	Computed by VSD
CLmaxN	Min critical load of nitrogen	Computed by VSD
CLnutN	Critical load of nutrient nitrogen	Computed by VSD
nANCcrit	Critical leaching of base cations	Computed by VSD
cNacc	Acceptable concentration of nitrogen in the soil solution	Computed from the max acceptable quantity of nitrogen leaching from the soil profile (Nleacc in eq/ha/a) and Qle (see below), for Nleacc
Crittype	Chemical criterion	Used : molar Al/Bc ratio
Critvalue	Chemical criterion value	Used : Al/Bc = 1
Thick	Soil thickness (m)	Based on the soil databases of Slovenia, averaged for the soil type
Bulkdens	Soil bulk density (g/cm3)	Based on the soil databases of Slovenia; computed from soil organic carbon and soil clay content. Mapping manual 6.4.1.3., Eq. 6.27
Cadep	Total deposition of calcium (eq/ha/a)	EMEP deposition database
Mgdep	Total deposition of magnesium (eq/ha/a)	EMEP deposition database
Kdep	Total deposition of potassium (eq/ha/a)	EMEP deposition database
Nadep	Total deposition of sodium (eq/ha/a)	EMEP deposition database
Cldep	Total deposition of chloride (eq/ha/a)	EMEP deposition database
Cawe	Weathering rate of calcium (eq/ha/a)	O,3×BCw, where BCw is the cumulative weathering rate for base cations. For BCw see mapping manual 5.3.2.3, Eq. 5.39 amd Table 5-14 (for calcareous soils WRc=20 was used)
Mgwe	Weathering rate of magnesium (eq/ha/a)	0.3×BCw
Kwe	Weathering rate of potassium (eq/ha/a)	0.2×BCw
Nawe	Weathering rate of sodium (eq/ha/a)	0.2×BCw
Caupt	Net uptake of calcium (eq/ha/a)	Average yearly yield rate × wood calcium content. Only dominant tree species of the forest type was considered. Data on average yearly yield rate are based on Slovenian forest inventory of the last ten years.
Mgupt	Net uptake of magnesium (eq/ha/a)	Average yearly yield rate × wood magnesium content. For details see Caupt.
Kupt	Net uptake of potassium (eq/ha/a)	Average yearly yield rate × wood potassium content. For details see Caupt.
Qle	Amount of water percolating through the root zone (mm/a)	Mapping manual 5.5.2.1.3., Eq. 5.91a and b. Annual mean temperature and annual mean precipitation data based on Meterological database of Slovenia
lgKAlox	Equilibrium constant for the Al-H relationship (log10)	Used: 8 (gibbsite equilibrium)
expel	Exponent for the Al-H relationship (-)	Used: 3 (gibbsite equilibrium)
pCO2fac	Partial of CO2 pressure in soil solution as multiple of the atmospheric CO2 pressure (-)	Ratio between soil CO_2 pressure and atmospheric CO_2 pressure (370 ppm). The former one being estimated on the basis of mean annual (soil) temperature (Mapping Manual Eq. 5.44)
cOrgacids	Total concentration of organic acids (eq/m3)	Used: 0.01
Nimacc	Acceptable amount of nitrogen imobilised in the soil (eq/ha/a)	Based on German NFC (see Posch et al., 2001: 142, table DE-7)
Nupt	Net nitrogen uptake by plants (eq/ha/a)	Average yearly yield rate × wood nitrogen content. Only dominant tree species of the forest type was considered. Data on average yearly yield rate are based on Slovenian forest inventory of the last few years.
Fde	Denitrification fraction (between 0 and 1)	Value set by soil science expert on the basis soil drainage characteristics and clay content , value span from 0.8 for wet, clay rich soil (gleysol) to 0.1 for dry soil (ranker, leptosol)
CEC	Cation exchange capacity (meq/kg)	Based on the soil databases of Slovenia, averaged for the soil type
Bsat	Base saturation (-)	Based on the soil databases of Slovenia, averaged for the soil type
Yearbsat	Year of base saturation determination	Based on the soil databases of Slovenia
lgKAlBc	Exchange constant for Al vs. Bc (log 10)	Parameter calibrated by VSD, starting value 0
lgKHBc	Exchange constant for H vs. Bc (log 10)	Parameter calibrated by VSD, starting value 3
Cpool	Soil carbon content (g/m2)	Computed from soil thickness, soil bulk density and soil carbon content. The latter one was averaged for the whole soil layer; data based on the soil inventories of Slovenia, averaged for the soil type
CNrat	C:N ratio in the soil	Based on the soil databases of Slovenia, averaged for the soil type
yearCN	Year of C:N ratio determination	Based on the soil databases of Slovenia
DMstatus	Static / dynamic model switch	Both modes used in the modeling process
EUNIScode	EUNIS code	EUNIS forest habitat types identified in Slovenia
CC	Country code	SI

Results of the modelling

Critical loads of acidity

Calculated $CL_{max}(S)$ values vary between 450 and 15850 eq/ha/a (7.2 and 253 kg/ha/a). 50% of receptors (quartile range) are in range between 2310 and 7330 eq/ha/a. The upper values are extremely high and some other toxic effects of such high sulfur depositions might appear before soil acidification takes place. Huge differences in variability of $CL_{max}(S)$ among EUNIS habitat types and within these categories are seen. Some of the habitats show very different critical loads, e.g. the receptors of certain beech and oak forests from the submediterranean region have some of the highest and lowest CL's computed. Taking into consideration only the quartile range the following habitat types appeared the most susceptible: Sub-Pannonic beech forests, Illyrian acidophilous fir forests and *Bazzania* fir forests, all of them thriving on acid soils. For more details see table SI-4.

Maximum critical load of acidifying nitrogen - $CL_{max}(N)$ (load permitted at no sulfur deposition) shows very similar spatial pattern as it is the case with $CL_{max}(S)$. This similarity is the consequence of the critical load function which links both acidifying pollutants (the larger critical load of sulfur, the smaller critical load of nitrogen). The values of $CL_{max}(N)$ span from 1060 to 33060 eq/ha/a (or 14.8 to 465 kg/ha/a), quartile range is between 3470 and 9020 eq/ha/a (48.5 and 126 kg/ha/a). For more details see table SI-4. Computed critical loads of nitrogen as an acidifying pollutant far exceed the critical loads of nitrogen as an eutrophicating factor in all the receptors studied ($CL_{min}(N) < CL_{nut}(N) < CL_{max}(N)$). In that respect N_{dep} can only be as high as $CL_{nut}(N)$.

Minimum critical load of nitrogen ($CL_{min}(N)$) is the load of acidifying N where $N_{dep} = N_i + N_u$. Decreasing critical load value below this level would in a longer term lead to impoverishment of the ecosystem and decrease in organic matter production i.e. forest growth. The interval of values of $CL_{min}(N)$ is much narrower comparing to the $CL_{max}(N)$ interval and spans between 180 and 950 eq/ha/a (2.5 and 13.3 kg/ha/a). For more details see table SI-4.

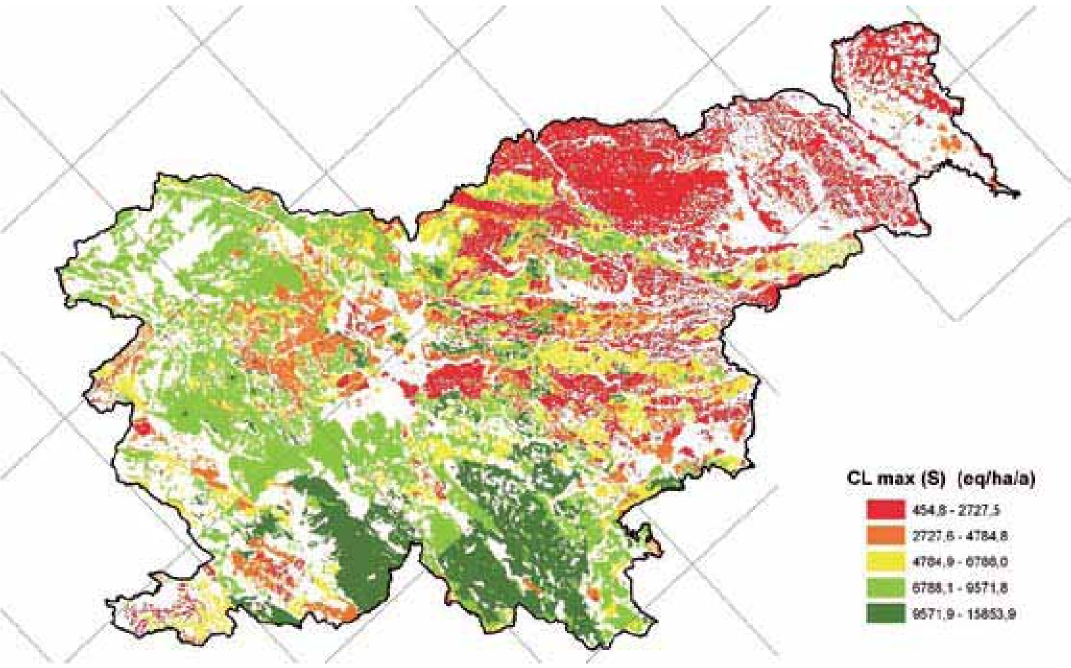

Figure SI-2: Maximum critical loads of sulfur ($CL_{max}(S)$) for forest ecosystems of Slovenia.

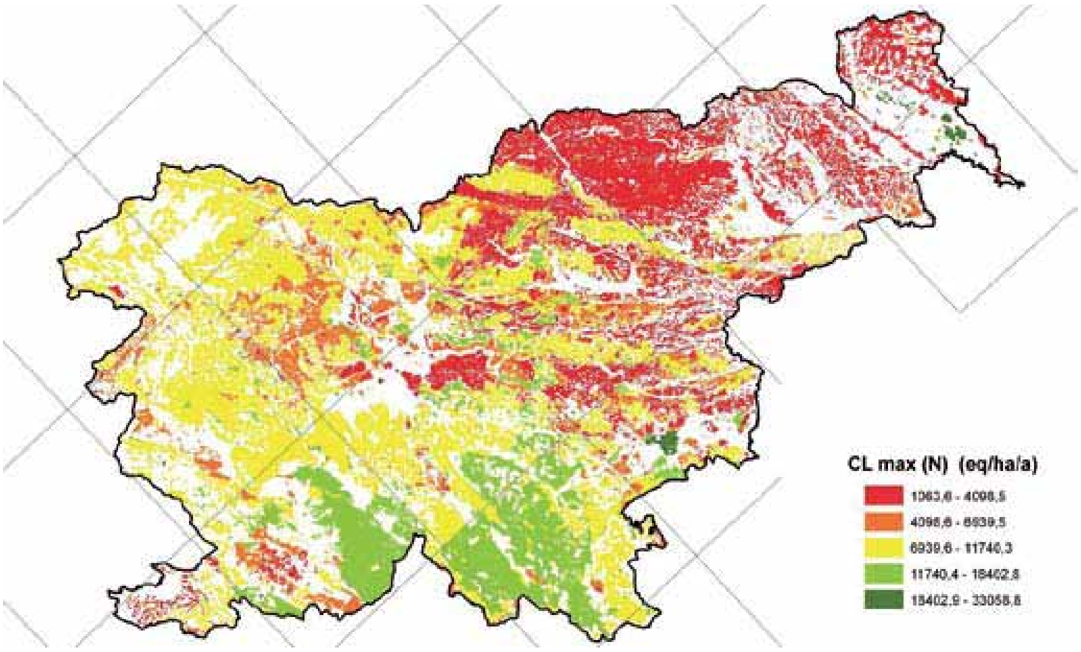

Figure SI-3: Maximum critical loads of nitrogen (CL_{max}(N)) for forest ecosystems of Slovenia.

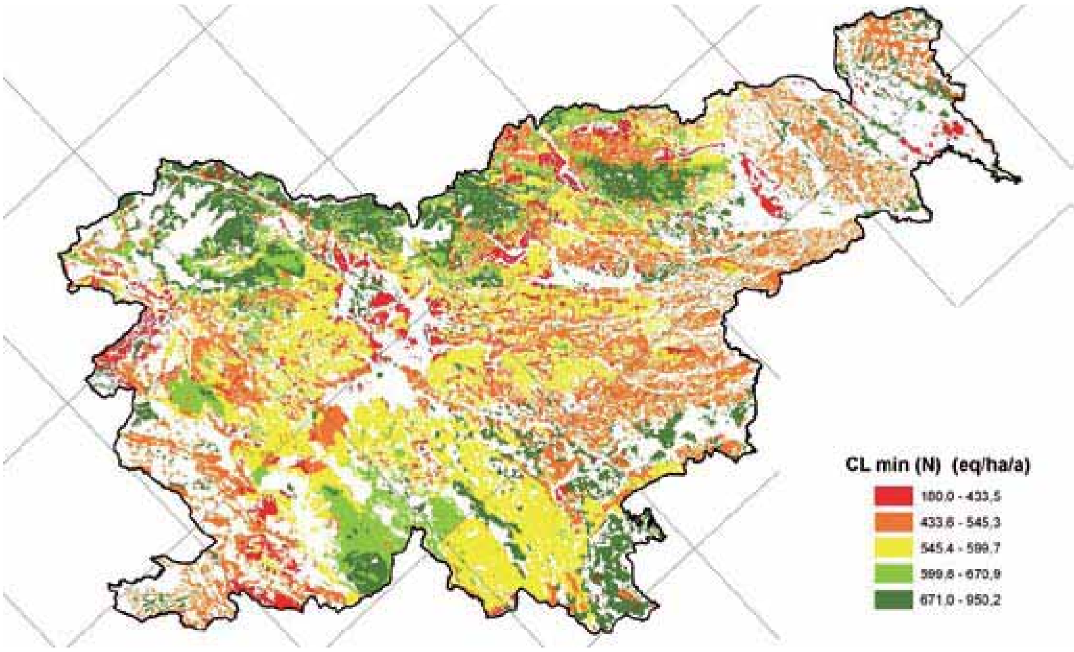

Figure SI-4: Minimum critical loads of nitrogen (CL_{min}(N)) for forest ecosystems of Slovenia.

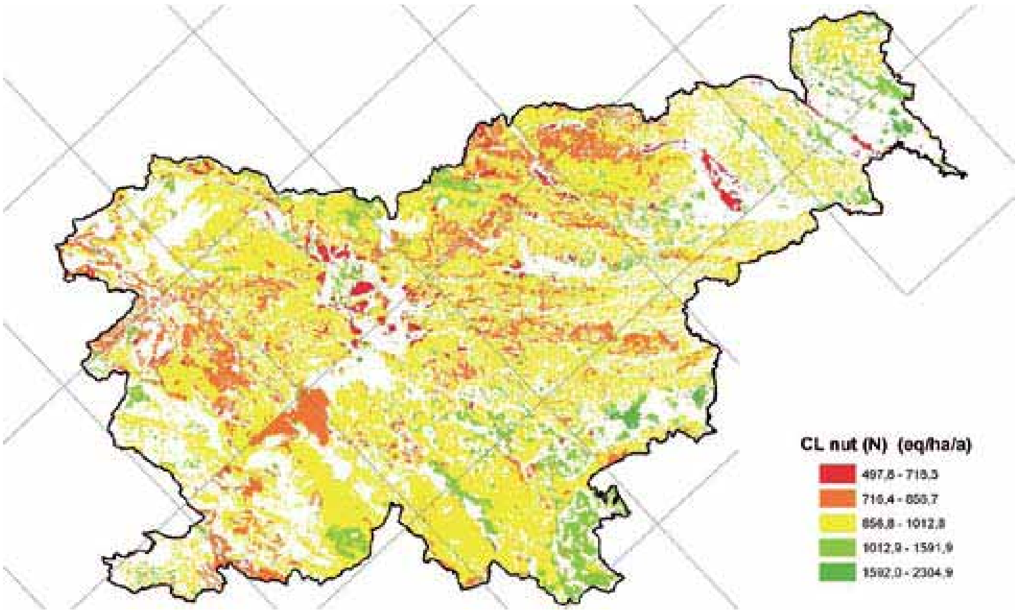

Figure SI-5: Critical loads of nutrient nitrogen ($CL_{nut}(N)$) for forest ecosystems of Slovenia.

Critical loads of nutrient nitrogen

Apart from acidification effects nitrogen compounds also cause eutrophication of (forest) ecosystems. The results of $CL_{nut}(N)$ modelling differ considerably from critical loads of acidifying N. There is no clear west-east gradient in $CL_{nut}(N)$, which shows that quite different soil and environmental parameters influence the critical loads of eutrophication. The values of $CL_{nut}(N)$ vary between 500 and 2300 eq/ha/a (7.0 and 32.3 kg/ha/a) with 50% of receptors in range between 845 and 950 eq/ha/a (11.8 and 13.3 kg/ha/a). The variability of CL's between receptors of particular habitat type is considerably lower than variability of CL's of acidity. The EUNIS habitat types most susceptible to eutrophication are the following: *Pinus mugo* scrubs of the alpine and dinaric region, Dinaric dolomite Scots pine forests, Illyrian sub-Mediterranean *Pinus nigra* forests, Eastern Alpine acidophilous Scots pine woods and Alpine spring heath Scots pine forests. These forest types are located in different regions of Slovenia; their main similarity is the shallowness of the soil profile. The results of CL modelling are well congruent with empirically derived critical loads.

Critical load exceedances

On the basis of deposition data provided within the CCE call for data 2008, the difference between deposition of S, N, and A (acidity) and critical loads can be calculated. The difference is known as the exceedance (in eq/ha/a). Regarding static models of soil chemistry, knowing the deposition exceedances are really the subject that matters and not the CL's themselves. Due to the time-dependence of atmospheric deposition of pollutants, exceedances are theoretically speaking only valid for a given moment in time. Consequently the time, for which the exceedances have been calculated, has to be reported. We calculated the exceedances for the deposition data of year 2000.

The simplest case is to compute the exceedance of critical load of nutrient nitrogen, where $Ex(N_{dep}) = N_{dep} - CL_{nut}(N)$. The result is shown in figure SI-6. The exceedances of year 2000 nitrogen deposition appeared in altogether 266 receptors covering less than 1.5% of the area under modelling or less than 0.7 % of national territory. The exceedance was observed in 26 different EUNIS habitat types but most often in two of them: in Eastern Alpine acidophilous Scots pine woods and in Dinaric dolomite

Scots pine forests. If taking into consideration deposition from the year 2000 the plant communities of Scots pine forests appear to be the most affected from current nitrogen depositions. Lot of these forests can be found in the vicinity of the capital city Ljubljana, where the largest intensity of traffic in the country is observed, but it is also the area of intensive agricultural production.

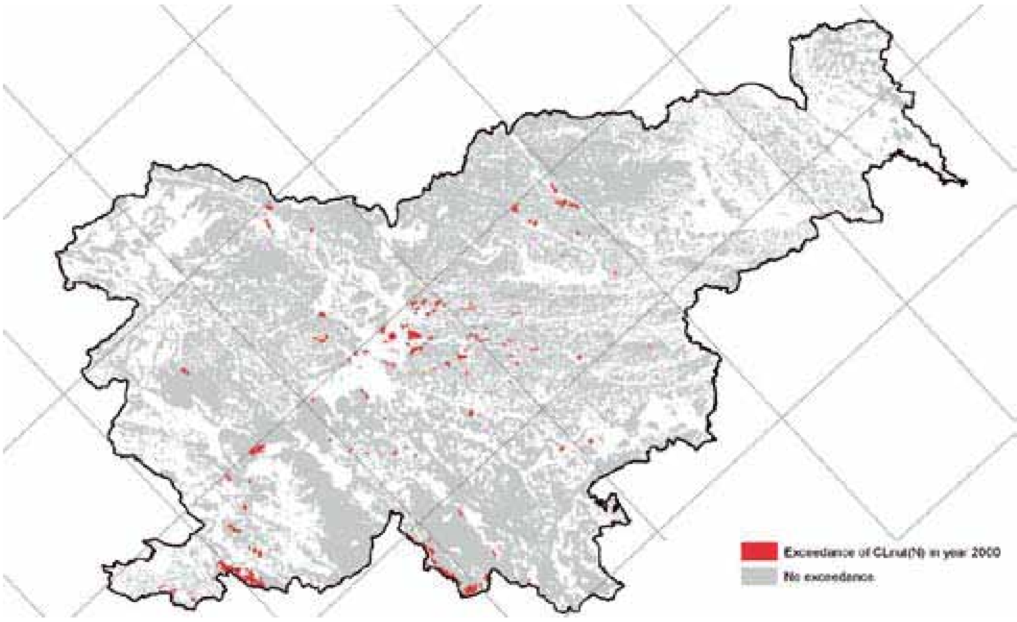

Figure SI-6: Exceedances of critical load of nutrient nitrogen calculated from the year 2000 deposition data.

Since there is no unique critical load of N or S (numerous pairs of combinations of CL's for N and S pollutants lying on the CL function lead to the same critical leaching of acid neutralizing capacity (ANC)) there are also no unique exceedances of acidifying S and N. One of the pollutants needs to be fixed and then so called conditional critical load for another pollutant can be defined. It is also possible to calculate exceedance of critical load of acidity CL(A) or to calculate exceedance functions (exceedance isolines). For this report we calculated three different conditional acidity exceedances: exceedance of S deposition over critical load of sulfur at current (year 2000) N deposition ($CL(S|N_{dep})$, exceedance of N deposition over critical load of nitrogen at current (year 2000) S deposition ($CL(N|S_{dep})$) and exceedance of S deposition over critical load of sulfur at critical load of nutrient nitrogen ($CL(S|CL_{nut}(N))$). The latter has also been termed exceedance of minimum critical load of sulfur ($CL_{min}(S)$) (UBA, 2004). These exceedances show similar spatial pattern which is largely the effect of low resolution of deposition data (EMEP grid cell size). This indicates the urgent need to refine the deposition data in Slovenia with some additional measurements. The majority of exceedances of 2000 deposition in Slovenia appeared in EMEP 78,45 grid cell, where two main Slovenian thermal power stations are located. The western part of the country appears safe as far as soil acidification is concerned, which is supposedly a consequence of predominant calcareous rocks and absence of very large SO2 pollutant.

Forest ecosystems of Slovenia do not appear very sensitive to soil acidification of sulfur and nitrogen atmospheric deposition. In comparison with the northern countries of Europe maximal critical loads of acidity are high and consequently the exceedances are present in only a small fraction of the forest ecosystems. This is largely the effect of predominance of calcareous rocks in Slovenia (44% of the area). Weathering of base cation rich limestone is an efficient neutralizer of soil acidifying compounds. In that respect modelling of critical loads for acidity is not really important for limestone and dolomite rich areas. However the west-east gradient in the susceptibility to acidification is apparent which follows the geological map (western part – eutric, calcareous soils, eastern part – district, silicate soils).

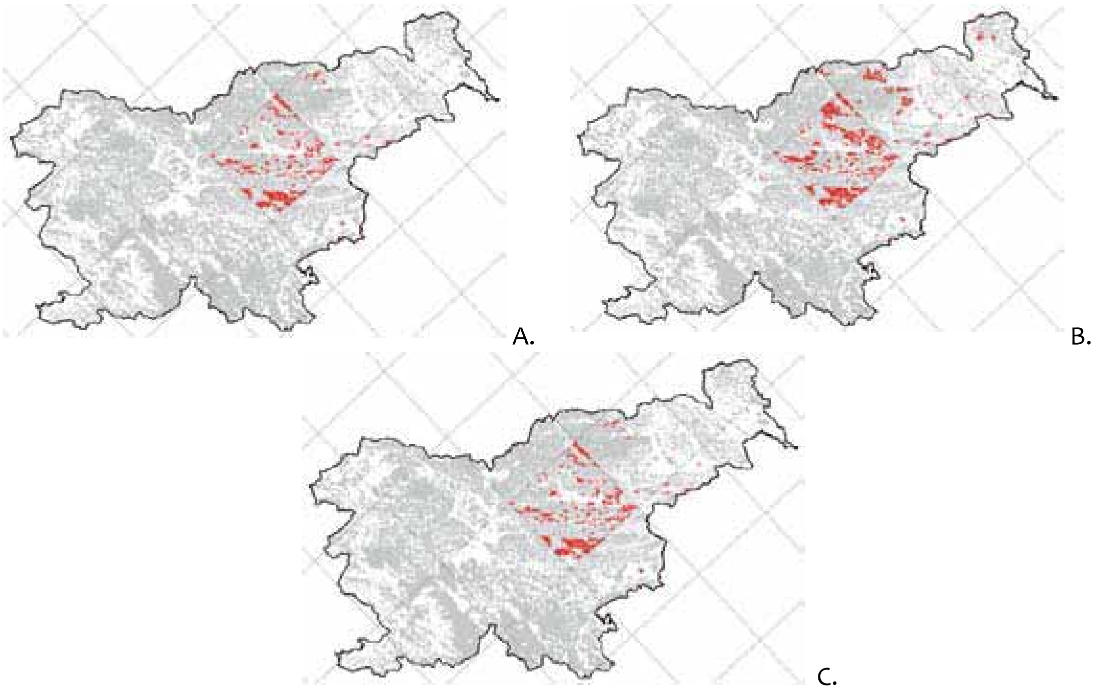

Figure SI-7: Exceedances of conditional critical loads of acidity calculated from the year 2000 deposition data. A.) Exceedance of $CL(S|N_{dep})$; B.) Exceedance of $CL(S|N_{nut})$; C.) Exceedance of $CL(N|S_{dep})$.

Table SI-4: Minimum, maximum, median and 1st and 3rd quartiles of critical loads for individual EUNIS habitat categories.

EUNIS code	F2.42	F2.47	G1.111	G1.121	G1.211	G1.633	G1.635	G1.676	G1.6C1	G1.6C2	G1.6C3	G1.6C4	G1.743	G1.7C1	G1.A1A
Area	16224	225	621	35	5036	57005	130	64709	192825	447987	28664	20	69860	2897	96763
% Area	1.5%	0.0%	0.1%	0.0%	0.5%	5.2%	0.0%	5.9%	17.5%	40.7%	2.6%	0.0%	6.4%	0.3%	8.8%
No. of receptors	250	6	33	1	148	476	1	1111	1757	4620	263	1	733	53	1685
$CL_{max}(S)$ (eq/ha/a)															
Minimum	3289.2	9008.0	728.6	7475.5	1108.4	1938.2	1511.3	1230.2	516.4	611.5	1912.8	8247.3	1407.8	1890.5	454.8
Q25	8092.1	9102.8	2101.9	7475.5	2175.4	6590.7	1511.3	6250.8	1667.8	2348.2	3676.9	8247.3	3512.4	7466.7	1708.9
Median	8666.6	9131.8	4353.3	7475.5	2313.2	7208.2	1511.4	7065.5	2321.5	5933.1	7408.6	8247.3	6709.1	7668.5	2685.3
Q75	8971.4	9145.8	6874.7	7475.5	3969.5	7489.1	1511.4	7699.9	4243.9	7323.6	7875.2	8247.3	7330.5	7804.3	6501.9
Maximum	13751.1	9145.8	7935.3	7475.5	7911.1	11186.9	1511.4	13056.7	12324.5	13449.0	13635.0	8247.3	13201.5	8123.7	15853.9
$CL_{min}(N)$ (eq/ha/a)															
Minimum	189.3	309.4	335.0	550.0	335.0	507.5	543.7	387.8	537.3	496.8	387.8	583.7	535.4	449.0	683.0
Q25	385.5	325.8	335.0	550.0	335.0	655.5	543.7	449.0	539.6	534.8	387.8	583.7	535.4	449.0	701.0
Median	397.5	326.4	335.0	550.0	335.0	722.8	543.7	485.0	546.3	534.8	420.8	583.7	535.4	495.8	701.0
Q75	397.5	397.5	371.0	550.0	335.0	792.2	543.7	521.0	589.9	571.5	459.8	583.7	571.0	522.7	701.0
Maximum	404.8	397.5	478.0	550.0	335.0	793.5	543.7	735.0	825.6	793.5	745.8	583.7	708.2	664.0	950.2
$CL_{max}(N)$ (eq/ha/a)															
Minimum	3686.7	10406.4	1063.6	8856.1	1443.4	2799.7	2238.1	1815.9	1185.3	1155.3	2316.5	9747.3	2133.2	2373.2	1166.7
Q25	9345.6	10458.8	2436.9	8856.1	2747.6	8172.6	2238.1	7492.9	2683.1	3490.5	4984.6	9747.3	4964.0	8748.5	2975.1
Median	10027.1	10478.7	5261.4	8856.1	2905.2	8769.2	2238.1	8545.8	3526.1	7283.8	8788.2	9747.3	8098.1	9042.3	4350.7
Q75	10369.0	10488.5	8486.7	8856.1	6894.5	9073.6	2238.1	9151.7	6046.4	8982.2	9284.3	9747.3	8798.6	9222.0	9670.9
Maximum	17557.6	10539.9	9152.0	8856.1	23868.5	14492.3	2238.1	16805.9	23846.5	30151.5	17518.9	9747.3	28401.4	9640.6	27255.0
$CL_{nut}(N)$ (eq/ha/a)															
Minimum	499.8	537.7	635.2	808.4	637.0	828.5	865.7	705.2	845.5	765.9	661.8	811.5	825.3	751.1	1003.1
Q25	588.9	545.0	650.4	808.4	680.8	939.4	865.7	785.7	908.8	875.8	729.9	811.5	873.6	785.3	1076.9
Median	603.0	552.8	669.1	808.4	687.1	984.2	865.7	804.9	921.1	903.0	747.3	811.5	890.2	796.4	1094.0
Q75	619.0	599.2	687.4	808.4	952.5	1012.9	865.7	827.0	944.9	922.6	776.6	811.5	915.5	812.5	1212.1
Maximum	655.1	621.7	752.6	808.4	1925.9	1078.9	865.7	985.2	2132.4	2122.8	1003.9	811.5	1972.2	942.1	2304.9
nANCcrit (eq/ha/a)															
Minimum	2485.9	5874.5	506.9	4687.8	756.1	1354.8	996.8	855.1	381.6	432.0	1413.0	5496.1	969.9	1509.0	319.6
Q25	5190.2	5910.1	1461.4	4687.8	1313.7	4207.8	996.8	3755.3	1107.3	1497.4	2548.3	5496.1	2390.1	4788.0	1081.6
Median	5570.9	6055.0	2725.9	4687.8	1619.8	4618.9	996.8	4406.0	1485.1	3566.7	4788.9	5496.1	4120.7	4986.7	1695.7
Q75	5864.5	6068.7	4371.3	4687.8	2478.5	4887.9	996.8	4891.1	2730.8	4595.6	5150.4	5496.1	4635.7	5091.2	3924.5
Maximum	8415.4	6068.7	5174.3	4687.8	5148.8	6746.6	996.8	7716.0	7182.1	8369.9	8473.2	5496.1	8000.6	5438.4	9229.4

Dynamic modeling results

Dynamic modelling was carried out on all of the 12692 receptors which were also subjected to calculating critical loads. Target loads were computed except in cases where CL's were not exceeded in 2010 and where chemical criterion was less than specified limit (in our case [Al]:[Bc]=1). Dynamic modelling data were calculated for target years 2020, 2030, 2040, 2050 and 2100; each year for 14 different deposition scenarios, provided by the CCE. The results are summarized in table SI-4.

Table SI-4: Summary of dynamic modelling for forest ecosystems of Slovenia. The number of receptors and the percentage of modelled area where the chemical criterion is not met are shown for each target year. Only the results for moderate, low and high deposition scenarios are shown.

Target year	Moderate deposition scenario		High deposition scenario		Low deposition scenario	
	No. of receptors	% of modelled area	No. of receptors	% of modelled area	No. of receptors	% of modelled area
2020	311	1,61	903	4,70	20	0,03
2030	72	0,20	701	3,36	5	0,01
2040	36	0,09	528	2,79	4	0,01
2050	28	0,07	466	2,35	2	0,00
2100	14	0,03	313	1,14	0	0,00

G1.A46	G3.112	G3.124	G3.132	G3.135	G3.1B2	G3.1C2	G3.1F3	G3.1F4	G3.1F5	G3.425	G3.441	G3.4C5	G3.521	G3.E	EUNIS code
699	42042	2911	927	21802	7426	809	10276	732	1457	21979	2977	2063	462	53	Area
0.1%	3.8%	0.3%	0.1%	2.0%	0.7%	0.1%	0.9%	0.1%	0.1%	2.0%	0.3%	0.2%	0.0%	0.0%	% Area
25	566	49	37	254	64	15	94	26	20	270	50	66	15	2	No. of receptors
															CL_{max} (S) (eq/ha/a)
3159.1	1102.2	7285.4	1589.8	1204.0	3387.7	3439.7	1559.7	3374.9	6541.9	1809.0	7336.6	3428.4	7579.8	3983.4	Minimum
6289.0	2075.0	7598.1	1702.9	2149.0	6385.3	7608.6	2327.9	7120.7	8155.4	3062.5	7492.8	7071.6	8423.5	3983.4	Q25
6581.2	2951.7	7772.0	2223.5	2389.6	7602.5	7928.5	2520.5	7356.7	10512.2	3576.4	8061.7	7870.8	8462.7	4107.7	Median
8281.1	4994.6	8057.4	2389.5	3326.5	7942.6	7958.0	4772.2	7667.1	13752.7	4749.6	8264.7	8867.8	8536.1	4231.9	Q75
13455.6	13059.9	13579.4	2543.2	10794.8	12857.3	8044.5	8138.5	12417.8	13871.2	13413.4	8536.6	13677.3	8753.7	4231.9	Maximum
															CL_{min} (N) (eq/ha/a)
507.5	388.6	388.6	432.5	388.6	534.9	461.0	388.6	495.4	460.6	239.2	252.0	180.0	195.3	397.5	Minimum
507.5	424.6	448.3	531.6	424.6	657.9	674.6	545.2	529.9	583.1	239.2	316.9	216.0	252.0	397.5	Q25
643.0	450.5	465.5	535.2	438.0	674.6	674.6	603.6	552.2	605.6	239.2	323.0	216.0	287.8	397.5	Median
656.9	482.7	531.6	603.6	460.6	674.6	674.6	665.9	674.6	671.9	275.2	413.3	244.1	323.0	397.5	Q75
793.5	674.6	674.6	674.6	674.6	674.6	674.6	674.6	674.6	674.9	389.5	466.0	252.0	330.4	397.5	Maximum
															CL_{max} (N) (eq/ha/a)
4456.4	1649.4	8483.5	2231.9	1813.4	4879.1	4974.3	2426.9	4804.7	8313.0	2354.8	8437.0	4222.6	8709.8	7704.7	Minimum
7752.0	3041.5	8901.3	2425.0	3141.2	8315.5	9071.9	3539.1	8448.1	9638.9	4191.9	8662.2	8076.1	9613.8	7704.7	Q25
8030.1	4115.8	9096.1	3321.4	3410.4	9131.7	9485.9	4035.3	8694.7	13482.1	4725.3	9302.4	9121.2	9704.3	9825.0	Median
9851.1	6707.0	9627.3	3590.5	4572.0	9504.2	9516.9	6774.6	9193.6	17793.1	7241.9	9675.7	11056.6	9740.6	11945.3	Q75
17595.6	30396.5	17577.9	4040.9	13954.2	16733.7	9613.0	26052.1	16017.9	17989.2	33058.8	9880.1	17267.5	10049.3	11945.3	Maximum
															CL_{nut} (N) (eq/ha/a)
838.3	709.4	718.6	738.7	719.8	829.6	761.9	738.5	789.4	742.7	553.9	551.5	503.9	497.8	775.8	Minimum
877.6	766.1	741.5	798.9	775.1	889.9	888.7	841.1	819.9	815.3	612.9	591.1	529.4	532.3	775.8	Q25
914.5	785.9	764.2	835.6	786.1	897.7	898.9	885.7	849.8	870.3	627.7	620.1	542.9	566.4	905.1	Median
958.9	810.1	810.4	872.5	803.3	917.9	904.9	919.2	883.0	916.3	661.6	654.6	557.7	587.6	1034.3	Q75
1050.5	1904.5	928.8	928.2	1019.4	938.1	932.3	1417.8	909.4	946.0	1760.7	717.0	719.4	613.3	1034.3	Maximum
															nANCcrit (eq/ha/a)
2066.6	766.0	4483.3	1131.9	851.6	2450.9	2502.9	1132.9	2314.5	4416.2	1210.2	4547.6	2217.2	4759.5	2895.3	Minimum
3941.4	1385.6	4733.0	1241.9	1437.3	4227.6	4871.9	1626.0	4554.2	5414.8	1956.2	4671.3	4252.9	5452.5	2895.3	Q25
4178.8	1961.7	5005.3	1560.3	1600.0	4863.7	5187.3	1795.1	4623.9	6700.6	2342.5	5071.3	4751.7	5484.4	2944.1	Median
4883.7	3261.0	5221.4	1669.1	2304.9	5186.1	5219.3	3182.5	4926.7	8516.3	3006.3	5233.1	5300.6	5549.9	2992.9	Q75
8377.2	7875.8	8340.3	1809.1	6338.4	7850.6	5305.6	5258.5	7623.9	8628.8	7966.9	5447.3	8062.0	5758.6	2992.9	Maximum

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Introduction

In response to the call for data November 2007, the following datasets have been produced (Table SE-1):
Critical loads of S and N for lakes and soils
Dynamic modelling results for lakes and soils
Empirical critical loads of nitrogen
A document describing the sources and methods used to produce the data (this document).

Table SE-1. Datasets produced for the call.

	Number of sites	Model
Critical loads for lakes	1974	FAB, MAGIC library
Dynamic modelling for lakes	1974	MAGIC, MAGIC library
Critical loads for soils	15920	PROFILE
Dynamic modelling for soils	15920	SAFE
Empirical critical loads of N	1298	-

Data sources and calculation methods

Lakes

Critical loads

The lakes with submitted critical loads are part of a Swedish national synoptic lake survey performed in 2005 including 1974 lakes > 1 ha selected by a stratified random selection. Lake water chemistry was taken from this survey. Limed lakes were corrected by assuming a constant Ca:Mg ratio for nearby lakes and a constant Mg concentration of the liming agent.

For freshwaters the critical loads were calculated using the first-order acidity balance (FAB) model as described in Henriksen et al. (1993), Posch (1995) and Rapp et al. (2002) with some modifications described below. The BC_{le} used in the FAB-model was the calculated BC concentration 2100 according to MAGIC simulations using the CLE scenario. Thus the F-factor for estimating the weathering rate was not used. Nitrogen immobilisation was set to 100% for deposition up to 2 kg N/ha, 50% for the part of the deposition exceeding 2 kg/ha up to 10 kg/ha and 0% for the deposition exceeding 10 kg N/ha. The calculations of nitrogen immobilisation was partly justified by Gundersen et al (1998). In addition to this, leaching of organic nitrogen calculated from the lake concentration of Total Organic Nitrogen, was regarded as immobilised. The chemical threshold, ANC_{limit}, was calculated individually for each lake to a value corresponding to a change in pH of 0.4 units from reference conditions calculated by MAGIC. This threshold is used as a definition of acidification in the Swedish Environmental Quality Criteria and for the fulfilment of Good Ecological Status within the EU Water Frame Directive (Fölster et al, 2007). When MAGIC was not run on the lake itself, the data used in the FAB model was taken from a similar lake within a database of MAGIC simulated lakes by a matching procedure (MAGIC library). Less than 10% of the lakes did not get any match, since no similar lakes were in the library. Those lakes were in most cases well buffered and were unlikely to be acidified even at a very high deposition. CLmaxS, CLmaxS and nANCcrit was then set to high values (10000, 10000 and 5000 eq/ha/yr, respectively) and the critical value of ANC was set to a negative value (-0.2 meq/l) to ensure that the critical load for those lakes were not exceeded in further calculations.

Dynamic modelling

The lakes modelled with MAGIC are part of national lake surveys (in the years 1985, 1990, 1995 and 2000) and long term lake monitoring programmes focussing on acidified lakes. Lake water chemistry was obtained from these sources. Long-term averages (1961-1990) of annual runoff volumes provided by the Swedish Meteorological Institute (SMHI) were used. Land use data were obtained from the Swedish National Land Survey. Soil chemistry and properties for the catchments were taken from the Swedish survey of forest soils and vegetation. Soil bulk densities estimated by Karltun (1995) was used and averaged over the soil profiles. Soil water DOC was assumed to be 8 mg/l for all catchments (based on data from permanent forest monitoring plots in Sweden, ICP Forests, level II). Long-term averages of nutrient uptake were derived from the Swedish Forest Inventory 1983-92 and from the ASTA database. Pre-industrial nutrient uptake was set to 0.5 times present day for lake catchments in southern Sweden and zero for lake catchments in northern Sweden, based on existing information about Swedish forests and forestry from the 1870/80-ies. Present day deposition data was estimated from the MATCH model (Robertson et al. 1999, www.smhi.se) in a 20 x 20 km square grid over Sweden for the years 1997-1998 or 2002-2004. For the lakes, the deposition was scaled to a calibration year (calibration year varied from lake to lake; depending on data availability we used 1985, 1990, 1993, 1995, 1997 or 2000) and adjusted using the observed lake water chemistry to account for the local variation within the 20x20 km squares (Moldan et al. 1997). The total deposition of Cl^-, SO_4^{2-} and base cations was adjusted at each site using lake water chemistry. It was assumed that, as a result of the declining SO_4^{2-} deposition in the years 1985 to 2000, an estimated percentage of the output flux of SO_4^{2-} from the lakes had been desorbed from catchment soils or from the lake sediment. The percentage used was 0-35%, depending on the rate of decline in SO_4^{2-} deposition in the calibration year. The modelled deposition of N species was adjusted to account for variations in dry deposition by assuming that the ratio between the adjusted deposition and the deposition given by the MATCH model was the same for the N species and SO_4^{2-} at each lake. Historical deposition sequences were derived from updated EMEP150 grid specific deposition histories 1880- 2000 over Europe according to Schöpp et al (2003) and were not (yet) updated with the new historic sequences from CCE which were provided for the purpose of the Call for data issued in November 2007. Future CLE and MFR deposition scenarios were those provided by CCE. The 12 intermediate deposition scenarios were interpolated between the CLE and MFR according to the instructions in the call. For practical purposes the future deposition scenarios were grouped together and averaged in 7 geographical regions with similar sequences.

The nitrogen dynamics for lakes and lake catchments was modelled in a simplified way without coupling between N deposition and the long term development of the ability of ecosystems to assimilate nitrogen. It was assumed that the percentage of N deposition leached in runoff will remain constant in the future for all scenarios. This assumption is probably reasonably accurate for the majority of the modelled lakes and their catchments with given N deposition scenarios (no significant increase in N deposition) and for a given time (up to years 2030, 2040, 2050 and with more uncertainty up to 2100). It needs to be pointed out, however, that this is an optimistic view of future N changes where nitrogen saturation does not progress and where future N deposition does not cause increased leaching of NO_3^-. If a less optimistic view of future effects of N deposition is adopted (such as e.g. precautionary principle), it could change the outcome of the dynamic modelling in a major way towards worse surface water quality in terms of higher NO_3^-, higher inorganic aluminium, lower pH and lower ANC.

Climate was assumed not to change over the modelled period. To what extent a changing climate will affect the future surface waters quality in response to the 14 modelled deposition scenarios is beyond the scope of the response to the call. However, it needs to be noted that this is – together with the fate of deposited N - another source of uncertainty in the model predictions because the combined effect of changing air pollution and climate could be significantly different from each of the two major driving factors alone.

The dynamic model runs were performed in two steps – first the 14 scenarios were assessed for 325 lakes in Sweden, using previously existing calibrations with the MAGIC model (Cosby et al., 1985, 2001). Those model results were then assigned to all 1974 lakes with calculated critical loads using the MAGIC library. The MAGIC library is a web based tool (www.IVL.se/magicbibliotek) developed at IVL in 2003 – 2007 to do lake acidification assessment in lakes and running waters. The MAGIC library consists of two main parts: a catalogue of lakes and running waters with existing MAGIC model calibration and a so called matching tool. The matching tool is used for comparing any given lake (*evaluation lake,* in this case one of the 1974 lakes with calculated critical load) described by its key parameters in the matching questionaire with all lakes included in the MAGIC library lake catalogue (*library lakes,* in this case the 325 modelled lakes in Sweden) and selecting which library lakes are most similar to the evaluation lake. If a similar lake is found among the library lakes (a good match) it is assumed that MAGIC calibration and scenario outputs calculated for the library lake are also valid for the evaluation lake. In this way the MAGIC model calculation for the 14 future deposition scenarios modelled at 325 lakes were extrapolated to the larger dataset of 1974 lakes.

For 181 of the (1974) reported lakes the MAGIC library did not contain any lake similar enough to justify the assumption that the two lakes (the evaluation and the library lake) share the same acidification history and future. These 181 lakes either have a high ANC or are polluted by other sources than anthropogenic deposition and therefore the dynamic modelling parameters for these lakes were assumed to be non-acidified and equal to that of the least acidified lake in the MAGIC library.

The total area of Sweden is regarded as ecoarea for lakes, since the lake water quality is a result of processes in the catchment. Only the nine largest, very well buffered lakes are excluded from the total area. The sum of the ecoareas for the 1974 lake stations is equal to the total land area for Sweden with the nine largest lakes subtracted. The ecoareas that each of the sampled 1974 lakes represents, is calculated according to the principles for the stratified random selection of the lake survey (Wilander and Fölster, 2007).

Forest ecosystems

Critical loads

Critical loads of acidity and nutrient N was calculated with the steady state soil chemistry model PROFILE (Sverdrup & Warfvinge, 1993). The chemical criteria molar ratio BC/Al in the soil solution was used, where BC is molar concentration of base cations Ca^{2+}, Mg^{2+} and K^{+} and the Al is a charged weighted sum of the molar concentrations of inorganic Al in the soil solution. The critical limit was set to BC/Al=1, corresponding to a growth reduction of spruce by 20% (Warfvinge and Sverdrup, 1995). The root weighted BC/Al was used, since it is a more relevant measure than previously used BC/Al in the soil layer with the lowest BC/Al. The BC/Al ratio was weighted over the soil horizons according to the root content of each layer. The soil layer specific root content was represented by an estimated fraction of BC uptake from each layer. The root zone was assumed to be 0.5 m, except for a few sites with shallow soils, were it was assumed to be 0.15 m.

The CL for nutrient nitrogen, CLnutN, was based on the critical nitrogen concentration limit of 0.3 mgN/l in the water leaching from the root zone. This concentration represents the upper limit for Class 1 (low concentrations) according to the Swedish environmental quality standards for lakes (www.naturvardsverket.se) and corresponds also to the critical concentration for vegetation changes of sensitive vegetation (CCE Instructions for submitting... 2007, Table 6). Calculations were performed on sites within the National Forest inventory (Hägglund, 1985). Successful calculations were made on 15 920 forest sites.

Deposition data were derived from the MATCH model (Robertson et al. 1999, www.smhi.se), in a 20 x 20 km square grid over Sweden. The average for three years, 2003-2005 was used. Mineralogy was based on soil data from the Swedish Geological Survey (Lax & Selenius, 2005). Forestry data, e.g. uptake of base cations and nitrogen, originates in the National Forest Inventory (Hägglund, 1985). More detailed information about the input data can be found in Akselsson et al. (2004).

The ecoarea for the forest sites were derived from the National Forest Inventory database. The sum of the ecoarea for all the sites corresponds to the total area of forest in Sweden.

Dynamic modelling

The dynamic soil model SAFE (Alveteg et al. 1998) was used to assess the 14 deposition scenarios defined in the Call at 632 forest sites distributed over the whole country. The model was run without considering nitrogen dynamics in soil beyond nitrification, which means that all nitrogen that is not taken up is assumed to nitrify. Denitrification and N immobilisation were not considered. The model was calibrated based on measured soil base saturation. The simulations include the 14 deposition scenarios defined in the call for data, and optionally a simple run with no change in deposition after 2020. The scenarios were simulated from 2010 to 2100. The format for submitting the data to CCE did not allow for separate reporting of static critical loads and dynamic modelling results. To overcome this technical difficulty the results of the 14 deposition scenarios were assigned to the geographically nearest points with calculated critical loads. Thus were the scenario calculations of the 632 sites assigned to the 15 920 sites with calculated critical load.

Empirical critical loads of N

Empirical critical loads of N were defined for two land cover classes in Sweden, forests and mires. The critical loads were set to 8 kg ha^{-1} y^{-1} for both forest and mires, based on the recommendations from the Workshop on effects of low-level nitrogen deposition in Stockholm in March 2007 (United Nations 2007). Ground vegetation changes have been observed at this load in both forests and mires (e.g. Nordin et al., 2005; Gunnarsson et al., 2002). The empirical critical loads of N for forests and mires were presented on an EMEP grid resolution. Each combination of land use class and protection class in a grid cell received a unique ID, with corresponding ecosystem area and land use class.

For the empirical critical load calculation, the sum of the ecoarea of forest and wetlands within each EMEP50 square was calculated, representing different levels of protection. Five protection levels were represented; 0) no protection, 1) Special Protection Area (SPA), Birds Directive (Natura 2000), 2) Special Area of Conservation (SAC), Habitats Directive (Natura 2000), 3) SPA and SAC (also Natura 2000) and 9) national nature protection.

Comments and conclusions

Freshwater ecosystems

Critical loads

For lakes, the median critical load of S deposition is 413 eq/ha/year, the Nmin is 321 eq/ha/year (i.e. the amount of N deposition that is taken up by the ecosystem and does not cause any acidification) and the Nmax is 1322 eq/ha/year (i.e. the maximum amount of N deposition the ecosystem could take without inacceptable acifification, if the S deposition is zero). The differences between lakes are large, with some lakes that cannot recover with any deposition under present day conditions and many lakes that cannot become acidified under present day conditions. The area with exceedance of critical loads was 19% in 2002-2004 (Figure SE-1).

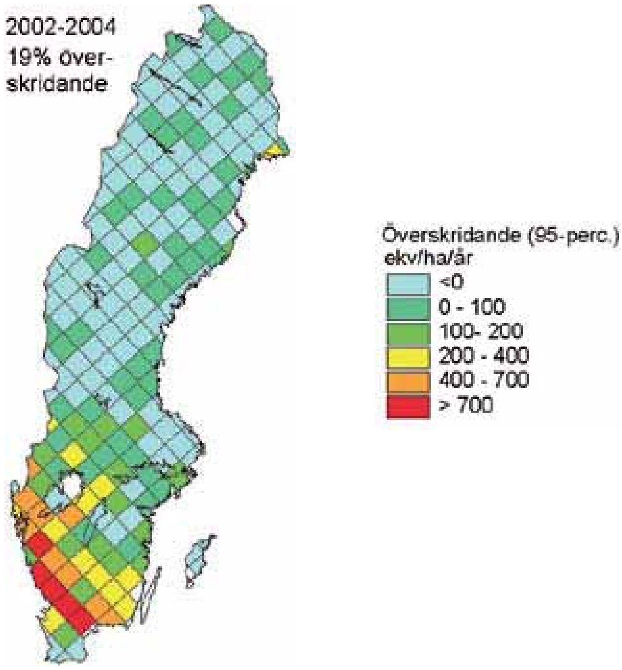

Figure SE-1. Exceedence of CL for lakes in Sweden 2002-2004. For each square the 95 percentile of exceedence is given, i.e. the exceedence of the 5 percent of the area with the highest exceedence.

Dynamic modelling

There is a large variation in ANC among the 1974 evaluated lakes (Table SE-2). In 2010, 2% of the lakes will have a negative ANC, 7% will have an ANC between 0 and 50μeq/l, 47% will have an ANC between 50 and 200μeq/l and 44% above 200μeq/l. There are lakes with low, intermediate and with high ANC in most parts of Sweden, although the eastern parts of Sweden have a more uniformly high ANC. In 2050 according to the CLE scenario, ANC will have increased somewhat in many lakes, but decreased in others. The average difference between 2010 and CLE 2050 is very close to zero. In 2050 according to the MFR scenario almost all lakes will have a small increase in ANC, on average 8 μeq/l. According to the MFR scenario, there are less than 1% lakes with negative ANC in 2050, all in southwestern Sweden (Table SE-2).

The average small differences between years and scenarios are expected since a majority of the Swedish lakes have never been severely acidified. This means that they never have experienced any large decrease in ANC and thus no great ANC increase should be expected in these cases. The lakes with a strong increase in ANC (under the MFR scenario) are mostly found in the southern part of Sweden, where acidification has caused the largest ANC decline. There are also many sensitive lakes all over Sweden where ANC will either remain more or less unchanged or even decline further under the CLE scenario (672 lakes) or even under the MFR scenario (22 lakes). But for many sensitive lakes in high deposition areas, the deposition decrease under the MFR does give a significant and large increase in ANC and pH.

Table SE-2. Percentiles of lakes in different ANC classes in 2010, 2050 under the MFR and 2050 under the CLE scenarios.

ANC (μeq/l)	2010	2050 CLE	2050 MFR
<0	1.9%	0.7%	1.4%
0-50	7.4%	8.4%	7.8%
50-200	47.4%	43.5%	47.3%
>200	43.3%	47.5%	43.5%

Forest ecosystems

Critical loads

The critical load of acidity was exceeded at 12.5% of the Swedish sites (Figure SE-2). Exceedances can be explained by weak mineralogies that lead to low base cation weathering rates, in combination with relatively high acid deposition.

The exceedance of the critical load of nutrient N, calculated by the PROFILE model, followed the N deposition gradient, with the highest exceedance in the southwest (> 6 kg ha^{-1} y^{-1}) and no exceedance in the northernmost parts (Figure SE-3). The exceedance is generally somewhat higher than the exceedance of the empirical critical load of N (Figure SE-5). For Sweden, the critical N load is based on the risk for N leaching to surface waters (Figure SE-3) but is also an important criteria for the protection of the marine environment. The sources for eutrophication of the Baltic Sea are mainly land use and sewage. The N (and P) input to has already changed the ecosystems of the Baltic Sea significantly. Further deterioration of the marine ecosystem is a major concern in Sweden.

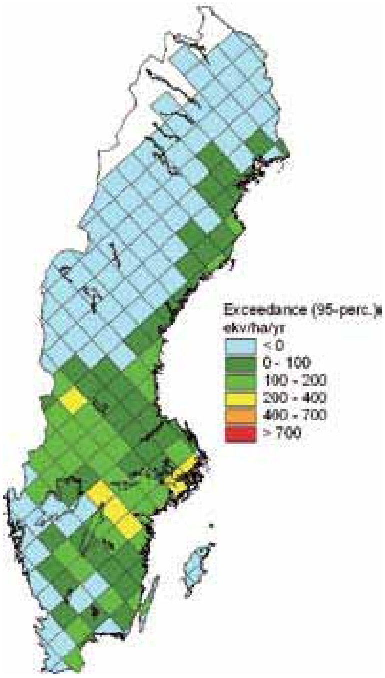

Figure SE-2. Exceedance of critical load of acidity, with the criteria molar BC/Al = 1 (root weighted), on 15920 sites in Sweden. Exceedance calculated for an average deposition in years 2003 - 2005. For each square the 95 percentile of exceedence is given, i.e. the exceedence of the 5 percent of the area with the highest exceedence.

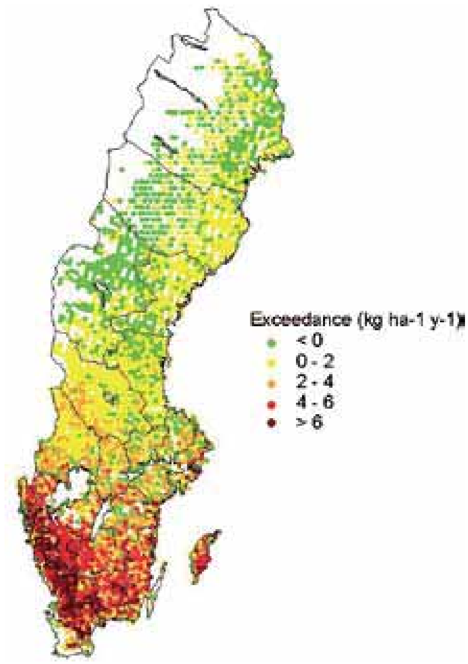

Figure SE-3. Exceedance of critical load of nutrient N calculated with PROFILE with the criteria Critical N concentration in soil solution = 0.3 mg l^{-1} on 15920 sites in Sweden. Exceedance calculated for an average deposition in years 2003 - 2005.

Dynamic modelling

The 14 deposition scenarios give valuable information about the response of the soils to different deposition levels (Figure SE-4).

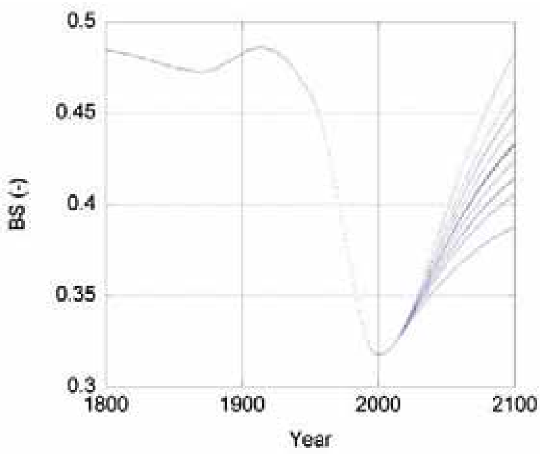

Figure SE-4. An example of a soil BS response to the 14 deposition scenarios.

There is a rather clear difference in development of the BC/Al (root density weighted average) over time under the MFR and CLE emission scenarios. While the number of points with BC/Al <1 decrease from 2010 to 2050 only modestly under the CLE scenario from 27% to 23% of the 15 920 sites (Table SE-3), the MFR would improve the conditions in the soil water dramatically leaving only 4% of the modeled sites with a BC/Al<1.

Table SE-3. Percentage of 15 920 modeled with BC/Al ratios divided into four categories.

BC/Al	2010	2050 CLE	2050 MFR
< 0.1	0.1%	0.1%	0.0%
0.1-1	26.7%	22.5%	3.6%
1-10	53.2%	53.1%	48.5%
>10	20.0%	24.3%	47.9%

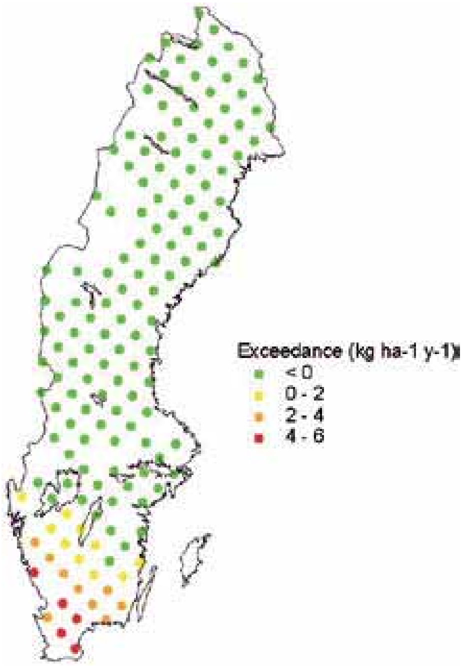

Figure SE-5. Exceedance of empirical critical load of nitrogen.

Empirical critical loads of N

The same value (8 kg ha-1 y-1) for the critical load was used for the two considered types of ecosystems (forests and mires) all over Sweden. This leads to a gradient in exceedance of critical loads which follows the N deposition gradient, with the highest exceedance in the southwest (up to 6 kg ha-1 y-1) and no exceedance in the central and northern part of the country (Figure SE-5).

No long-term experimental data from Swedish forest and mires are available where lower N loads than 8 kg ha/yr have been used, and thus it is unclear (but not unlikely) if ground vegetation changes occur at even lower N loads. Moreover, there are no data that makes it possible to differentiate between different forest types. Forests and mires cover most of the land area with natural and semi-natural vegetation in Sweden. It is desirable to include also mountain areas, and grasslands which are probably sensitive to N loads, but at present we have no data. The empirical critical load for N as a nutrient should be updated when new experimental data becomes available.

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Overview

In the CCE Status Report 2005 (Posch et al. 2005), data sources and methods used to calculate Swiss sulfur and nitrogen critical loads were described in detail. The contribution of the Swiss NFC in 2007 (in Slootweg et al. 2007) mainly focused on items changed since 2005. This document gives again a short summary of the data sources and methods and highlights the changes since the last data submission one year ago.

As in the data submission of 2007, the Swiss data set on critical loads of acidity and nutrient nitrogen is compiled from the output of four modelling and mapping approaches (see Figure CH-1):
The dynamic models SAFE and VSD (very simple dynamic model) were used for assessing acidifying effects of air pollutants on forest soils. The multi-layer model SAFE was calibrated and applied on 260 sites, where full soil profiles were available. For calculating critical loads of acidity and deposition scenarios with VSD, the required flux input-data were calculated by the SAFE model.

The SMB method for calculating critical loads of nutrient nitrogen (*CLnutN*) was applied on 10'608 forest sites. 10'348 of these sites originate from the National Forest Inventory (NFI, see LFI 1990/92), which is based on a 1x1 km grid. They are complemented by the 260 sites with soil profiles (which are partly identical with the NFI-sites).
The empirical method for mapping *CLempN* includes different natural and semi-natural ecosystems, such as raised bogs, fens, species-rich grassland, alpine heaths and poorly managed forest types with rich ground flora. The mapping was done on a 1x1 km grid combining several input maps of nature conservation areas and vegetation types. The total sensitive area amounts to 18'460 km².
Critical loads of acidity were calculated for 100 sensitive alpine lakes in Southern Switzerland applying a generalized version of the FAB model (first order acidity balance).

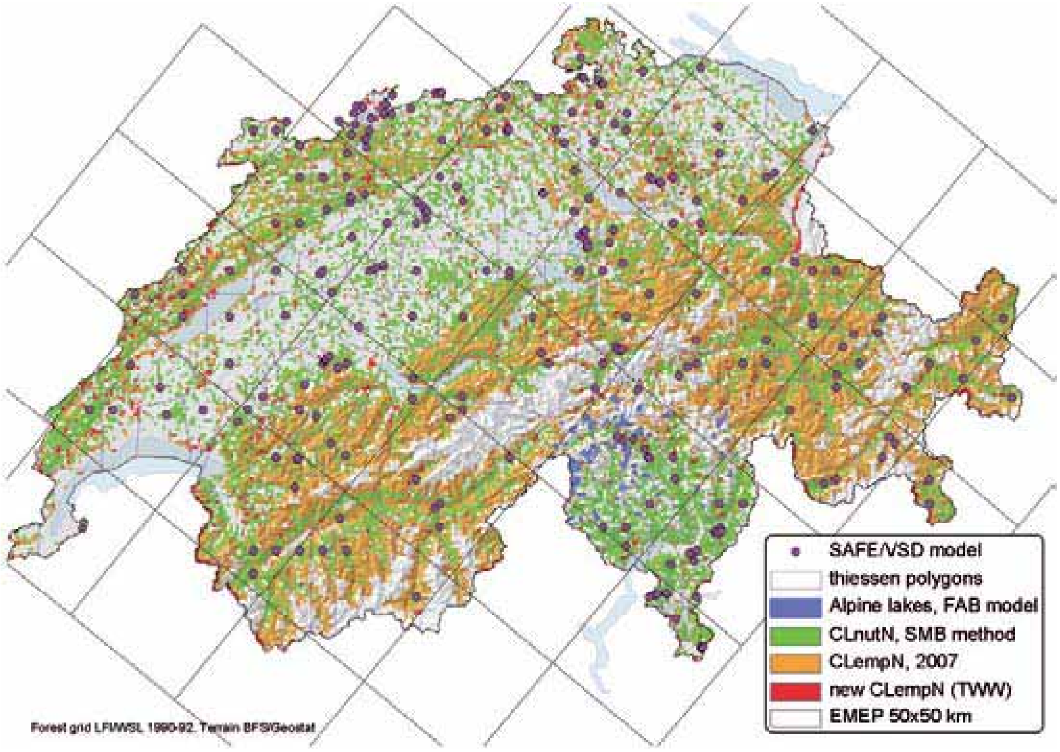

Figure CH-1. Overview of sensitive ecosystems and modelling approaches in Switzerland.

Dynamic modelling and critical loads of acidity for forests

Deposition

Wet and dry deposition rates were modeled or interpolated on the basis of results from various monitoring sites. The deposition of N, S, Bc, Na and Cl was calculated with a generalised combined approach for the reference year 2000. Thimonier et al. (2004) give a description of the methods related to N and S deposition in forests. These site-specific modelled depositions for the reference year were used to scale the deposition trend data supplied by the CCE.

As integral part of the call for data 2007/2008, the CCE provided a major update of sulfate, nitrate and ammonia historic and future deposition trends (1880-2020) for all 50x50km EMEP grids covered by the Swiss territory (M. Posch, pers. commun. Nov 30, 2007). The new deposition data were tested and evaluated (Kurz 2008). Most significant changes in the deposition trends are found with sulfate and ammonia. Peak sulfate deposition in the 1970s and 1980s is now estimated to have been substantially higher in most of the grids. Ammonia depositions are both lower and higher compared to earlier used data. Unlike sulfur and nitrate, also the values at the start of the period covered by data have changed for ammonia. The updated deposition trends for nitrate do not substantially deviate from the trends earlier used.

The new deposition data had to be extended beyond the time interval given because a common simulation period runs from 1500 to 2500. We have assumed for all three compounds deposition trends for years beyond 2020 to be constant until the end of the simulation period. Deposition trends of SO_x and NO_y prior to 1880 were obtained from scaling the Rothamsted sulfur deposition trend (C. Walse, Lund University, pers. comm., 1996) to the given 1880 SO_x and NO_y depositions. Resulting grid average background loads of the early 18th century, 27 to 40 $mol_c\ ha^{-1}\ a^{-1}$ for SO_4 and 7 to 11 $mol_c\ ha^{-1}\ a^{-1}$ for NO_3, were finally extended into the past as far as needed. The 1880 ammonia depositions were extended unchanged to the beginning of the simulation period also due to a lack of reliable alternatives. Finally, values for years between the provided five-year intervals were obtained by linear interpolation.

Time series of deposition of base cations and Cl have not changed, since their derivation is essentially based on national emission inventories (Kurz 2008, SAEFL 1998, BUWAL 1995).

Consequences of the use of the new deposition data were:
Biomass calibration failures due to an inconsistency between nitrogen availability (too low) and nitrogen demand to sustain the wanted growth at two sites. This can be avoided by shifting the afforestation year from 1600 to 1500 and by additionally using the site-specific instead of the regional average biomass at one site (2088).
The need for a recalibration of the sites soil chemistry to reproduce the currently measured base saturations.

Combined application of SAFE and VSD

The modelling of critical loads of acidity and the dynamic model runs of the provided 14 deposition scenarios is based on 260 forest plots (see Figure CH-1) for which the layered soil input is available. The sources of all input data required for the SAFE model runs were listed in the CCE Status Report 2005.

PRESAFE and SAFE are used to simulate input data for the VSD model, especially flux data such as weathering, nutrient uptake and deposition rates. N-processes other than uptake and leaching (immobilisation, denitrification) are yet only considered in VSD. For the current submission, the following modifications were made:
A new version of the SAFE model (V.2008, S. Belyazid, pers. commun. Feb 29, 2008) was used for preparing the input to VSD.
In former submissions, the multi-layer output of SAFE was aggregated to one layer in order to run the single-layer model VSD. Now, already the calculations in PRESAFE/SAFE are made in a one-layer mode.
Fluxes for VSD are drawn from SAFE model runs at critical load deposition (instead of current legislation deposition), using the criterion $Bc/Al^{3+} = 1$.
Due to the results of the data checks performed with the CCE-software (Access data base) input data were improved for a small number of sites, e.g. carbon and nitrogen pools as well as C:N ratios.

Determining the ecosystem area

Critical loads of acidity are calculated for 260 SAFE-sites that are not regularly distributed within the country. The NFI-sites (National Forest Inventory) however are a systematic sample representing a forest area of 1 km², each. Therefore, the area of forest represented by one SAFE-site was determined by those NFI-sites situated within the respective Thiessen-polygon constructed for the SAFE-sites (see Figure CH-1), and all acidity parameters were copied from a SAFE-site to the affiliated NFI-sites. In consequence, *EcoArea* was set to 1.0 km² for all resulting sites with critical loads for acidity.

If a NFI-site is situated on a 1x1 km grid cell containing also a site with empirical critical loads, *EcoArea* is set to 0.8 km² for the NFI-site and to 0.2 km² for the empirical site. Thus, double counts are avoided.

Critical loads of nutrient nitrogen (SMB method)

CLnutN are calculated by the SMB method for the 260 forest sites used in dynamic modelling and for 10'348 sites of the National Forest Inventory (NFI). Thereby, only NFI-sites with a defined mixing ratio of deciduous and coniferous trees are included (LFI 1990/92). This corresponds approximately to the managed forest area; for example brush forests and inaccessible forests are excluded.

The input for calculating the nitrogen process was presented in the CCE Report 2007. Table CH-1 gives an overview of the input parameters used . There was no change since then. However, the lower limit of *CLnutN* calculated by the SMB was set to 10 kg N ha^{-1} a^{-1} (corresponding to the lower limit of *CLempN* for forests). This means, all values of *CLnutN* below 714 eq ha^{-1} a^{-1} were set to 714. This was done with respect to the fact that so far no empirically observed harmful effects in forest ecosystems were published for depositions lower than 10 kg N ha^{-1} a^{-1} and for latitudes and altitudes typical for Switzerland. Therefore, the critical loads calculated with the SMB method were adjusted to empirically confirmed values.

Table CH-1. Range of input parameters used for calculating CLnutN with the SMB method.

Parameter	Values	Comment
fde	0.2 – 0.7 depending on the wetness of the soil	For NFI-sites, information on wetness originates from soil map 1:200'000. For SAFE-sites it is a classification according to the depth of the saturated horizon.
Nle(acc)	4 kg N ha-1 yr-1 at 500m altitude, 2 kg N ha-1 yr-1 at 2000m altitude	linear interpolation in-between. Acceptable leaching mainly occurs by management (after cutting), which is more intense at lower altitudes. Q and [N]acc are not used (for explanations see CCE Progress Report 2007).
Ni	1.5 kg N ha-1 yr-1 at 500m altitude, 2.5 kg N ha-1 yr-1 at 1500m altitude	linear interpolation in-between. At high altitudes the decomposition of organic matter slows down due to lower temperatures and therefore the accumulation rates of N and C are naturally higher.
Nu	0.7 – 7.0 kg N ha-1yr-1	present uptakes calculated on the basis of estimated long-term harvesting rates and average element contents in stems.

Empirical critical loads of nutrient nitrogen

As described in the CCE Progress Report 2007, the application of the empirical method is based on vegetation data compiled from various sources and aggregated to a 1x1 km raster. Until the last submission, 25 sensitive vegetation types were identified and included in the critical load data set (see Figure CH-1, orange colour): 1 type of raised bog (EDI 1991), 3 types of fens (WSL 1993) as well as 21 types with various vegetation worthy of protection (Hegg et al. 1993) including species-rich forest types, grasslands and alpine heaths. If more than one type occurs within a 1x1 km grid-cell the lowest value of *CLempN* was selected for this cell.

For this submission, an additional data source was used: the national inventory of dry grasslands of national importance (TWW, FOEN 2007), which was recently completed. This data set complements well the grassland types mapped by Hegg et al. (1993). Figure CH-1 shows the additional grid cells covered by TWW (red colour). The inventory contains 18 vegetation groups. EUNIS-codes and values for *CLempN* were assigned to 17 vegetation groups (see Table CH-2) according to Achermann and Bobbink (2003). Those vegetation types partially also occur in the inventory of Hegg. The two inventories are used here in a complementary way, because they cover different purposes: the atlas of Hegg gives an overview of the occurrence of selected vegetation types, while TWW focuses on the precise description of objects with national importance.

Table CH-2. Empirical critical loads for nitrogen assigned to 17 types of dry grasslands (TWW) of the national inventory of dry grasslands (FOEN 2007), in kg N ha^{-1} a^{-1}.

TWW-code		Vegetation type	EUNIS	Remarks	CLempN
1	CA	Caricion austro-alpinae	E4.4	alpine grassland	10
2	CB	Cirsio-Brachypodion	E1.23	similar to TWW 18 (E1.26), also used as hay meadow	15
3	FP	Festucion paniculatae	E4.3	similar to TWW 13	12
4	LL	(low diversity, low altitude)	E2.2	contains different types, promising diversity when mown, therefore lower range chosen	20
5	AI	Agropyrion intermedii	--	transitional type, no CLempN defined	--
6	SP	Stipo-Poion	E1.24	pastures/fallows in large inner-alpine valleys; CLempN based on national expert-judgment (Hegg et al. 1993)	12
7	MBSP	Mesobromion / Stipo-Poion	E1.26	similar to TWW 18 (E1.26), pastures	15
8	XB	Xerobromion	E1.27	meadows/pastures/fallows in large inner-alpine valleys; CLempN based on national expert-judgment (Hegg et al. 1993)	15
9	MBXB	Mesobromion / Xerobromion	E1.26	similar to TWW 18	20
10	LH	(low diversity, high altitude)	E2.3	contains different types of dry grassland at high altitude	15
11	CF	Caricion ferrugineae	E4.41	similar to E4.4, alpine grassland	10
12	AE	Arrhenatherion elatioris	E2.2	often used as meadows, lower range chosen as it occurs at all altitude levels	20
13	FV	Festucion variae	E4.3	middle of the range chosen	12
14	SV	Seslerion variae	E4.43	alpine grassland	10
15	NS	Nardion strictae	E1.71	meadows, subalpine	12
16	OR	Origanietalia	E2.3	meadows/fallows	15
17	MBAE	Mesobromion / Arrhenatherion	E1.26	similar to TWW 18, slightly more nutrient-rich than Mesobromion	20
18	MB	Mesobromion	E1.26	genuine semi-dry grassland	20

Alpine lakes

Critical loads of acidity for alpine lakes were calculated with a generalised FAB-model (Posch et al. 2007). The model was run for the catchments of 100 lakes in Southern Switzerland (see Figure CH-1) at altitudes between 1650 and 2700 m (average 2200 m). To a large extent the selected catchments consist of crystalline bedrock and are therefore quite sensitive to acidification. The data are submitted since 2005. In this submission, some minor errors in the data formatting and conversion were solved for *CLmaxN*.

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Introduction

In response to this call for data the UK are re-submitting steady-state critical loads for acidity and nitrogen and empirical critical loads for Special Areas of Conservation (SACs); these data remain largely unchanged from previous data submissions, but are re-submitted using the new table formats required by the CCE. In addition dynamic modelling outputs for the deposition scenarios provided by the CCE are submitted for 310 surface water sites and for acid sensitive regions of nine terrestrial EUNIS habitat classes.

Critical loads for terrestrial broad habitats

The methods and data used to calculate critical loads for broad habitats in the UK are described in detail in Hall et al (2003a,b & 2004a,b) and will not be repeated here. However, for this data submission the following changes or additions have been made:
The values of nitrogen immobilization for the heathland habitats (EUNIS classes F4.11 and F4.2) incorporate losses of N through fire (N_{fire}). Different values of N_{fire} are used for areas of wet and dry

heathland (Hall et al, 2004b). The value for N_{fire} in dry heathland (EUNIS class F4.2) has been revised downwards from 15 kg N ha^{-1} $year^{-1}$ to 10 kg N ha^{-1} $year^{-1}$ on the basis of recent measurements (Power et al, 2004; Terry et al, 2004). Therefore the values submitted for Nimacc, CLminN and CLmaxN for this EUNIS class have been updated.
The values of Ca, Mg, K, Na and Cl deposition submitted are non-marine (ie, seasalt corrected); in the previous submission "total" (marine plus non-marine) values had been submitted although non-marine data are used in the critical load calculations, with the exception of the acidity critical loads for woodland habitats where "total" Ca deposition is used in deriving $ANC_{le(crit)}$ (Hall et al, 2004a).
The values of K uptake for managed woodlands on organo-mineral and peat soils have been updated to reflect the application of phosphate and potassium fertilizers (primarily rock phosphate and muriate of potash) and their contribution to the base cation budget (Table UK-1). Whilst included in the methodology used to calculate critical loads previously the values had been omitted from the data submitted.

Table UK-1. The contribution of phosphate and potassium fertilizers to the base cation budget for managed woodlands.

Woodland type	Soil type	Addition of rock phosphate (eq ha^{-1} $year^{-1}$)
Managed broadleaf (G1)	Organo-mineral	80
Managed conifer (G3)	Organo-mineral	177
Managed conifer (G3)	Peat	417

Critical loads are required to protect these managed habitats and to protect the land under managed conifer forest for future non-forest use and possible reversion to semi-natural land uses.

The EUNIS categories assigned to the woodland broad habitats have been revised with G1 and G3 still being used for the managed broadleaved woodland and managed coniferous woodland respectively; and all the unmanaged woodland being assigned to G4 (mixed woodland). For G1 and G3 both acidity and nutrient nitrogen critical loads are based on a simple mass balance approach. For G4 the simple mass balance is used for acidity, but the empirical approach is used for nutrient nitrogen with different values set for (i) the effects of nitrogen on the ground flora of unmanaged ancient and semi-natural woodland (broadleaf and conifer), and (ii) the effects of nitrogen on epiphytic lichens in Atlantic oak woodlands. The methodology and values remain unchanged from previous submissions.

A "protection" code of 9 (ie, "a national nature protection program applies") is given to all the habitat data submitted as these "Broad Habitats" are of conservation value in the UK.

Critical loads of acidity for freshwaters

The previous submission included acidity critical loads for 1717 freshwaters (EUNIS classes C1 and C2) calculated using the FAB model. For this submission new data for 51 sites representing acid sensitive streams and ponds in the North York Moors area of the UK were added to the existing freshwater data set. Following screening 35 new sites were added to the data set for FAB modelling and critical loads have been submitted for these, bringing the total for this habitat type to 1752 sites. These sites are also assigned a "protection" code of 9 as they are also treated here as a broad habitat.

Critical loads for Special Areas of Conservation (SACs)

In 2007 site-relevant empirical critical loads of nutrient nitrogen were submitted for the designated features of SACs in the UK. These data have been re-submitted this year to provide the data in the updated table format; the methods and habitats considered are described in Hall et al, 2007.

Dynamic modelling of surface water habitats

The MAGIC model was applied to 310 previously calibrated UK lakes and streams covering the acid sensitive regions of Wales (Snowdonia, Cambrian Mountains), England (Lake District, South Pennines), Scotland (Galloway, Cairngorms) and Northern Ireland (Mourne Mountains). These are the same 310 sites for which dynamic model outputs were submitted in 2007 and the methods and data sources used to calibrate the model are described in Hall et al, 2005. For this submission MAGIC has been run using the latest set of deposition scenarios requested by the CCE.

Dynamic modelling of terrestrial habitats

Steady-state critical loads of acidity are calculated for all 1x1 km squares of the UK that contain at least 1ha of sensitive habitat and for which all the relevant data are available. The additional data required to run the VSD are largely based on a survey by Evans et al (2004) of the acid-sensitive regions of the UK and hence the model has only been applied to 1x1 km habitat squares where the soil base cation weathering is less than 1000 eq ha^{-1} $year^{-1}$ (Table UK-2).

Table UK-2. The number of 1x1 km squares where the base cation weathering rate is < 1000 eq ha^{-1} $year^{-1}$ and soils data are available for dynamic modelling and outputs have been submitted for terrestrial habitats in the UK.

EUNIS class	Habitat	Number of habitat 1x1 km squares modelled	Percentage of total habitat 1x1 km squares
D1	Bog	17041	91%
E1.7 & E3.5	Acid grassland	66568	86%
E4.2	Montane	3018	55%
F4.11 & F4.2	Dwarf shrub heath	67323	86%
G1	Managed broadleaved wood	38401	51%
G3	Managed coniferous wood	26576	71%
G4	Unmanaged wood	20525	55%

Parameters for the VSD have been assigned on the basis of soil type alone, or both habitat and soil type. Soil type was defined (as for critical loads) as the dominant soil association or map unit within each 1x1 km grid square and the soil group (eg, podzol), sub-group (eg, brown podzol), or broad soil class (ie, mineral, organo-mineral, peat) to which the soil type belongs. Where both habitat and soil type were required, the parameters for the unmanaged woodland habitat (that consists of both broadleaved and coniferous woodland) were set to those for broadleaved woodland. The parameters used are summarised in Table UK-3.

Table UK-3. Summary of parameters for the VSD "inputs" table for EUNIS classes D1, E1.26, E1.7, E3.5, E4.2, F4.11, F4.2, G1, G3, G4.

Parameter	EUNIS class	Value used	Notes ([] indicates source listed under References)
SiteID			Unique IDs for habitat and location
cNacc	E1.7, E3.5, E4.2	214	Values used in the VSD but not yet formally adopted in the UK.
	D1, F4.11, F4.2	321	
	G1, G4	357	
	G3	232	
Crittype	D*, E*, F*	ANC & CritpH	VSD run using ANC criteria for all squares[10]
	G*	Ca:Al & CritpH	VSD run using ANC criteria for all squares[10]
Critvalue	D*, E*, F*	ANC = 0	Habitat on mineral or organo-mineral soils[10]
		CritpH = 4.4	Habitat on peat soils[10]
			But VSD run using ANC0 for all squares
	G*	Ca:Al = 1	Trees on mineral or organo-mineral soils[10]
		CritpH = 4.4	Trees on peat soils[10]
			But VSD run using ANC0 for all squares
Thick	All	0.5	Default value
Bulkdens	All		Mean values by soil group (NSRI data)
Cadep	All		CBED non-marine values 1998-2000[20, 21]
Mgdep	All		CBED non-marine values 1998-2000[20,21]
Kdep	All	Set to zero	Data not available
Nadep	All	Set to zero	Data not available
Cldep	All		CBED non-marine values 1998-2000[20, 21]
Cawe	All		Dependent on soil type as fraction of BCwe[8]
Mgwe	All	(BCwe-Cawe)/3	Assumed to be 1/3 of BCwe - Cawe
Kwe	All	(BCwe-Cawe)/3	Assumed to be 1/3 of BCwe - Cawe
Nawe	All	(BCwe-Cawe)/3	Assumed to be 1/3 of BCwe - Cawe
Caupt	E1.26	222	[8]
	D*, F*, E1.7, E3.5, E4.2	Zero	Assumed to be zero[8]
	G1	195 eq ha^{-1} yr^{-1}	for trees on Ca-poor soils[8]
		290 eq ha^{-1} yr^{-1}	for trees on Ca-rich soils[8]
	G3	160 eq ha^{-1} yr^{-1}	[8]
	G4	Zero	Unmanaged woodland, assumes no harvesting[8]
Mgupt	D*, E*, F*	Zero	Assumed to be zero[8]
	G1	35 eq ha^{-1} yr^{-1}	For trees on Ca-poor soils[based on data in 3]
		41 eq ha^{-1} yr^{-1}	For trees on Ca-rich soils[based on data in 3]
	G3	54 eq ha^{-1} yr^{-1}	[based on data in 3]
	G4	zero	Unmanaged woodland, assumes no harvesting[10]
Kupt	D*, E*, F*	Zero	Assumed to be zero[8]
	G1	85 eq ha^{-1} yr^{-1}	For trees on Ca-poor soils[based on data in 3]
		79 eq ha^{-1} yr^{-1}	For trees on Ca-rich soils[based on data in 3]
	G3	56 eq ha^{-1} yr^{-1}	[based on data in 3]
	G4	zero	Unmanaged woodland, assumes no harvesting[10]
Qle	All		1x1 km runoff data based on 30-year (1941-1970) mean rainfall data
lgKAlox	All	8.5	Habitats on mineral soils[10, 23]
		7.6	Habitats on organo-mineral soils[10, 23]
		6.5	Habitats on peat soils[10, 23]
expAl	All	3	Default value
pCO2fac	All	40	Habitats on mineral soils[M.Billett pers.com.]
		100	Habitats on organo-mineral soils[M.Billett pers.com.]
		100	Habitats on peat soils[M.Billett pers.com.]
cOrgacids	All	25 eq m^{-3}	Habitats on mineral soils[C.Evans, unpublished data]
		32 eq m^{-3}	Habitats on organo-mineral soils[C.Evans, unpublished data]
		65 eq m^{-3}	Habitats on peat soils[C.Evans, unpublished data]
Nimacc	All except F*		Assigned by soil type[8, 14]
	F4.11		Assigned by soil type and additionally includes N_{fire} of 4.5 kg N ha^{-1} yr^{-1} [1, 4, 8, 14, 23]
	F4.2		Assigned by soil type and additionally includes N_{fire} of 10 kg N ha^{-1} yr^{-1} [8, 14, 16, 22, 23]
Nupt	D1, E4.2, F4.11, F4.2	36 eq ha^{-1} yr^{-1}	[2, 7, 8, 15, 17, 18]
	E1.26	714 eq ha^{-1} yr^{-1}	[8]
	E1.7, E3.5	81 eq ha^{-1} yr^{-1}	[6, 8]
	G1	420 eq ha^{-1} yr^{-1}	[8, 10]
	G3	Zero	Unmanaged woodland, assumes no harvesting[8,10]
fde	All	Nde * -1	For running VSD
Nde	All		Assigned by soil type[8, 14]
CEC	All		[5]
Bsat	All		[5]
Yearbsat	All	2004	[5]
Cpool	All		Mean values by soil group (NSRI data)
CNrat	All		[5]
yearCN	All	2004	[5]

The VSD was run in Access using the "CalcDM" form with the following settings:
Oliver constants: defaults as supplied by the CCE
Sea-salt correction: set to zero (no sea-salt correction)
Exchange kinetics: Gaines-Thomas
Minimum and Maximum values of CN ratio set according to Rowe et al (2006):

CNrat_min:	7.5 gC/gN for all habitats
CNrat_max:	20.8 gC/gN for E1.7, E3.5, E4.2, G1, G4
	43.6 gC/gN for D1, F4.11, F4.2, G3

It should be noted that as the VSD is not currently set up to run with the criteria of Ca:Al or critical pH, it was run for all habitat squares using the criterion of ANC set to zero. This enabled scenario outputs to be generated for all sensitive habitat squares for which data were available. The table "CLdata" contains the original UK critical loads data and not outputs from the VSD. The "inputs" table contains the criteria types and values used in the UK as defined in Table UK-3 above.

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PART III
Appendices

Appendix A Historical and Future Depositions

While critical loads are ecosystem properties which do not depend on the deposition of N and S, the temporal development of these depositions is needed for dynamic modelling as driving force (input) to simulate the bio-geochemical state of an ecosystem (terrestrial or aquatic) over time. Here we describe the historic and future N and S deposition (scenarios) for the period 1880–2100, as provided for use in dynamic modelling in response to the 2007/08 call for data.

a. For the historic period 1880–1990 N and S deposition data are taken from the compilation described in Schöpp et al. (2003). Annual grid-average depositions are computed with transfer matrices (11-year meteorological average) derived from the (old) lagrangian EMEP model and published historical emissions (sources listed in Schöpp et al. 2003). Since these depositions are given on the EMEP-150 grid, identical depositions are assigned to the 3×3 50×50 grid cells contained in a 150×150 cell.
b. For 1990 and 1995 the EMEP-50 grid-average depositions computed with the (current) eulerian atmospheric transport model by EMEP/MSC-W, which are available on the EMEP website (www.emep.int), have been used (see also Tarrasón et al. 2007).

For the year 1990 the two data sets were 'glued together' in the following way: The 50×50 depositions from EMEP/MSC-W were assumed to be true. A historical deposition for the year t (≤1990) in a given grid cell was multiplied with the following factor f_{corr}:

$$f_{corr} = 1 + \left(\frac{Dep_{EMEP,90}}{Dep_{hist,90}} - 1 \right) \cdot \frac{t - 1880}{1990 - 1880} \tag{A-1}$$

where $Dep_{EMEP,90}$ and $Dep_{hist,90}$ are the eulerian and historical 1990 depositions, resp., for the given grid cell. Multiplication which these factors makes the (modified) historical depositions coincide with the eulerian depositions in the year 1990 and leaves the 1880 depositions unaltered – it is a kind of rotation around 1880 so that 1990 values coincide. Thus, the further back you go the closer the depositions are to Schöpp et al. (2003); and in 1880 they are the original Schöpp-depositions (3×3 identical EMEP-50 grid values in every EMEP-150 grid cell). This modification of the historical depositions is carried out separately for all three pollutants (SO_2, NO_x and NH_3). Note, that this method differs from the widely-used practice of multiplying a historical deposition sequence by a constant factor to match it with observations in a given year, i.e. performing a translation instead of a rotation, with the consequence that early (and thus low) depositions can change by very large relative amounts!

c. For 2000, 2010 and 2020 emissions from IIASA were used and the resulting depositions calculated with the 50×50 transfer-matrices prepared at IIASA using simulation data provided by EMEP/MSC-W. These transfer matrices are available for a 5-year meteorological average (1996–98, 2000 and 2003 meteorology) and separately for grid-average, forest and semi-natural land cover in every grid cell. For 2010 the 'Current Legislation' (CLE) scenario was used and for 2020 both the CLE and the Maximum Feasible Reduction' (MFR) scenario; and the emissions are based on best knowledge of November 2007. To obtain landcover-specific depositions for all years, the ratio between grid-average and forest (resp. semi-natural) deposition in every grid cell for 2000 was assumed to hold for all years and these ratios were used to obtain landcover-specific depositions back to 1880.

The 2020 CLE and MFR depositions onto forests in every land-based EMEP-50 cell are shown in Figure A-1 together with some statistical information. Clearly, NH_3 depositions are the least affected by stronger (MFR) reductions measure and S-depositions are reduced the most in MFR (compared to CLE); in fact, S deposition is in many cases smaller than each of the N depositions.

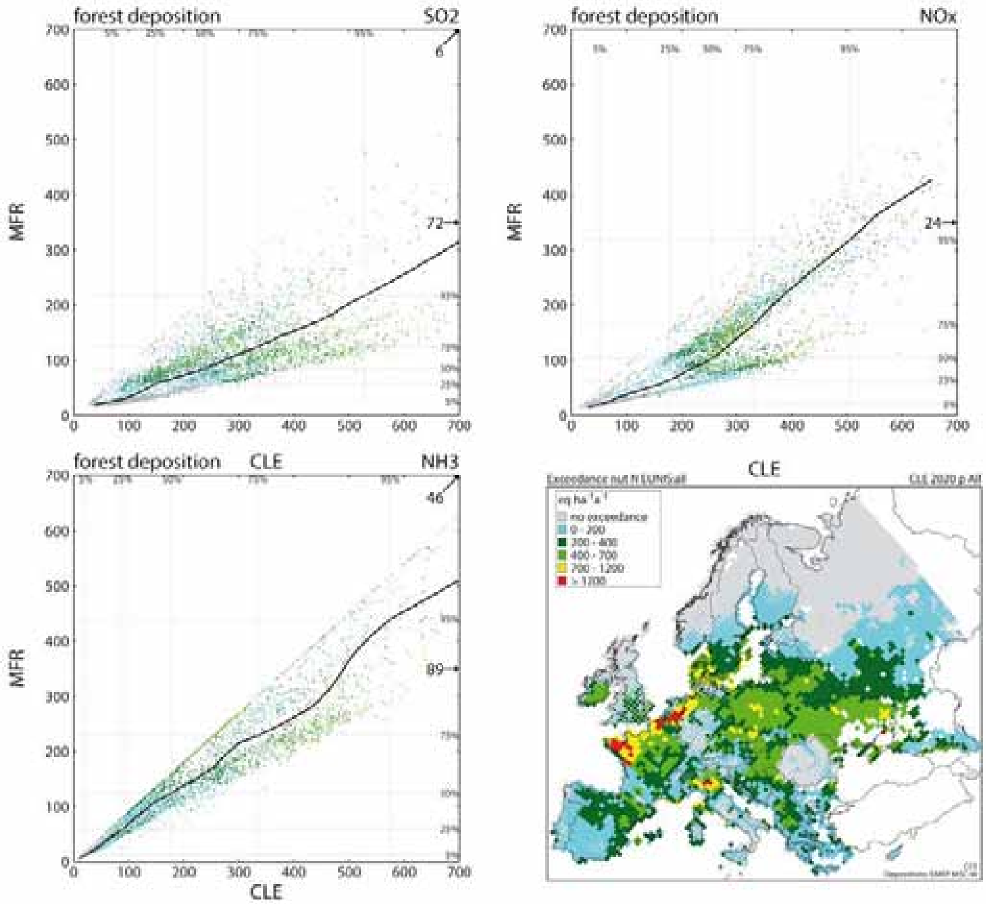

Figure A-1. Correlations between CLE and MFR depositions of SO_2, NO_x and NH_3 onto forests in the land-based EMEP-50 grid cells covering Europe. Data points are coloured according to the exceedance of N critical loads in 2020 (CLE scenario). Also shown are 'percentile grids', i.e. selected percentiles (5 %, median, 95 %) as thin dashed lines and the 'percentile-percentile curve' (solid line) connecting the resp. percentiles of the two variables, from the 1st percentile in the lower left to the 99th in the upper-right corner. These 'P-P-plots' reveal the correlation structure of the variables and allow an (approximate) determination of any percentile.

The two scenarios for 2020 (CLE and MFR) were used to systematically generate 14 scenarios, i.e. pairs of total N and S deposition, in every EMEP-50 grid cell for use in dynamic modelling. Details of the scenario generation are given in the instructions for the 2007/08 call for data, which is reproduced in Appendix B.

Future improvements of the deposition paths presented here c/should include: (a) the use of depositions calculated with the eulerian model (1980 and 1985 would be available, albeit from an older model version); (b) exact model simulations (instead of calculations with transfer matrices) for 2005 and scenario years. The most important improvement, however, would result from simulations with the eulerian model using updated historic emissions, especially for nitrogen species.

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Appendix B Instructions for Submitting Critical Loads of N and S and Dynamic Modelling Data

This Appendix is a reprint of the instructions as it was sent to the National Focal Centres with the call for data
Coordination Centre for Effects (CCE), Bilthoven, November 2007

Introduction

This document contains the instructions for the submission of data to the CCE on empirical and modelled critical loads of nitrogen and sulphur, as well as dynamic modelling data.

Your submission should contain the following key outputs:
1. **Updated critical loads, including empirical critical loads,** as well as input variables to allow consistency checks and inter-country comparisons
2. **Values of chemical variables from dynamic model runs in historic and in future years for deposition scenarios that are provided,** which can also be used (by the CCE) to infer target loads.
3. **A document describing the sources and methods used to produce the data.**

What's new and/or important to know?

- **None of 2007 submitted data will be used.** Please submit your data using the new format even if you haven't updated it. This provides an opportunity for you to check your (existing) data ☺.
- **Deadline** for submissions is **10 March 2008**.
- **The structure of the tables** has been improved to support multiple datasets, please read the section **below** on this topic.
- Indication of a site being an area which is protected by European or national legislation is part of the 'ecords' table, and applies also to modelled critical loads.
- The preferred **file format** is an Access database file (mdb), but Excel files or comma-separated ASCII files are also accepted. The easiest way is to use the Access database made available by the CCE.
- Please email your **submission** to jaap.slootweg@mnp.nl . The data can be attached to the email, but large data files can also be uploaded to ftp://ftp.mnp.rivm.nl/cce/incoming/ using ftp. If you have used ftp, please inform Jaap Slootweg by an email.
- Historic depositions and the **deposition scenarios** for nitrogen and sulphur are from the 'Review of the 1999 Gothenburg Protocol', Executive Body for the Convention (2007), ECE/EB.AIR/WG.5/2007/7, made available from the CCE upon request.
- It is important to use 'null' (i.e. "nothing") to indicate **missing or no value**, and ***not*** (e.g.) '–1' or '–999' or '0'.
- The **software** provided by the CCE has extended possibilities for performing consistency checks on your critical load database.
- All information is also available on our website www.mnp.nl/cce/ under ***News***. It is suggested to look occasionally for updates.
- Updated versions of chapters of the Mapping Manual are available at www.icpmapping.org.

Since August 2007, when a preliminary version of the instructions was distributed, additional highlights are:

- The 'inputs' table now contains an attribute to indicate if the data of the variables for the calculation of critical loads of the site are based on measurements.
- The "pristine" C:N ratio is asked for in the 'CLdata' table.
- Some less frequently used entries have been removed from the 'h2oinputs' table.

Data structure

The easiest way to assemble and submit data is to use the **Access** database, which is available from the CCE website www.mnp.nl/cce/ under *News*. The CCE also prepared a new version of the dynamic modelling software (VSD model) for this call. This software – tailored to every NFC (incl. national depositions) – is available from the CCE upon request (contact jaap.slootweg@mnp.nl). NFCs who wish to make more detailed analyses of individual sites can use the (updated) 'VSDStudio' software, developed in collaboration with Alterra and also available on the CCE website.

The data structure has changed in order to improve consistency over different datasets a NFC may use. Whereas in earlier versions the attributes related to the location of an ecosystem were present in the inputs tables for both modelled and empirical terrestrial data, and also in the inputs table for waters ('h2oinputs'), there now exists a single table with information describing the location. This new table ('ecords') contains the attributes listed in Table 1. A submission for a country may contain multiple datasets, but only a single 'ecords' table.

Table 1. Structure of the database-table 'ecords'.

Variable	Explanation	Note
SiteID	Unique(!) identifier of the site	1)
Lon	Longitude (decimal degrees)	2)
Lat	Latitude (decimal degrees)	2)
I50	EMEP50 horizontal coordinate	3)
J50	EMEP50 vertical coordinate	3)
EcoArea	Area of the ecosystem within the EMEP grid cell (km²)	4)
Protection	0: No specific nature protection applies 1: Special Protection Area (SPA), Birds Directive applies 2: Special Area of Conservation (SAC), Habitats Directive applies 3: SPA *and* SAC (1 *and* 2) 9: a national nature protection program applies (but *not* 1 or 2!) -1: protection status unknown	
EUNIScode	EUNIS code, max. 6 characters	5)

Notes on Table 1 (see last column):

1. Use integer values only (4-bytes)!
2. The geographical coordinates of the site or a reference point of the polygon (sub-grid) of the receptor under consideration (in decimal degrees, i.e. 48.5 for 48°30', etc.)
3. Indices (2-byte integers) of the 50km×50km EMEP-grid cell in which the receptor is located. It is the grid with the North Pole at (8,110); see also chap.8 of the Mapping Manual (www.icpmapping.org).
4. Please remove (spurious) records with an ecosystem area smaller than 1 ha.
5. You can find information on EUNIS (updated 2005) at http://eunis.eea.eu.int/

A second change to the data structure, suggested by some NFCs, is the splitting of input variables from the results of critical load and dynamic models. The '*inputs*' table now only contains the (most important) variables needed for the models, and a new table called '*CLData*' contains the modelled critical loads, and selected other output variables (e.g., obtained by calibration). NFCs who applied the MAGIC model are requested to make the same split between '*CLdata*' and '*h2oinputs*'. Please note the changes in the tables since last year. The table with the output variables for dynamic modelling, '*scenvars*', remains unchanged.

The data structure is summarized in Tables 2 to 5 below. The database you submit should contain at least 2 tables of which one should be '*ecords*'. It may also include tables with the structure of '*EmpNload*', '*CLdata*' in combination with '*inputs*' and/or '*h2oinputs*' and a table '*scenvars*'. Dynamic modelling results can only be used meaningfully if also the corresponding critical loads and the criterion used are known, therefore ecosystems that occur in the '*scenvars*' table should have an entry in either '*inputs*' or '*h2oinputs*'.

Routines in the software provided by the CCE allow you to perform consistency checks on your data. It is strongly recommended to carry out these checks! They (may) generate screen messages, which should be followed up.

1 Data structure of modelled critical loads and input data:

Table 2. Attributes of the table '*CLdata*'.

Variable	Explanation
SiteID	Identifier of the site
CLmaxS	Maximum critical load of sulphur (eq ha^{-1} a^{-1})
CLminN	Minimum critical load of nitrogen (eq ha^{-1} a^{-1})
CLmaxN	Maximum critical load of nitrogen (eq ha^{-1} a^{-1})
CLnutN	Critical load of nutrient nitrogen (eq ha^{-1} a^{-1})
nANCcrit	The quantity $-ANC_{le(crit)}$ (eq ha^{-1} a^{-1})
lgKAlBc	Exchange constant for Al *vs* Bc (log_{10})
lgKHBc	Exchange constant for H *vs* Bc (log_{10})
CNrat0	Pristine CNrat (i.e. at beginning of simulation)

Table 3. Attributes of the table '*inputs*'.

Variable	Explanation
SiteID	Identifier of the site
cNacc	Acceptable (critical) N concentration for CLnutN calculation (meq m^{-3})
crittype	Chemical criterion used for acidity CL calculations: =1: molar [Al]:[Bc]; =2: [Al] (eq m^{-3}); =3: base sat.(-); =4: pH; =5: [ANC] (eq m^{-3}); =6: molar[Bc]:[H]; =7: molar [Bc]:[Al]; = 8 molar [Ca]:[Al]; =11: molar [Al]:[Bc] AND [Al] > 0.1meq/L; = –1: other
critvalue	Critical value for the chemical criterion given in 'crittype'
thick	Thickness (root zone!) of the soil (m)
bulkdens	Average bulk density of the soil (g cm^{-3})
Cadep	Total deposition of calcium (eq ha^{-1} a^{-1})
Mgdep	Total deposition of magnesium (eq ha^{-1} a^{-1})
Kdep	Total deposition of potassium (eq ha^{-1} a^{-1})
Nadep	Total deposition of sodium (eq ha^{-1} a^{-1})
Cldep	Total deposition of chloride (eq ha^{-1} a^{-1})
Cawe	Weathering of calcium (eq ha^{-1} a^{-1})

Variable	Explanation
Mgwe	Weathering of magnesium (eq ha^{-1} a^{-1})
Kwe	Weathering of potassium (eq ha^{-1} a^{-1})
Nawe	Weathering of sodium (eq ha^{-1} a^{-1})
Caupt	Net growth uptake of calcium (eq ha^{-1} a^{-1})
Mgupt	Net growth uptake of magnesium (eq ha^{-1} a^{-1})
Kupt	Net growth uptake of potassium (eq ha^{-1} a^{-1})
Qle	Amount of water percolating through the root zone (mm a^{-1})
lgKAlox	Equilibrium constant for the Al-H relationship (log_{10}) (The variable formerly known as Kgibb)
expAl	Exponent for the Al-H relationship (=3 for gibbsite equilibrium)
pCO2fac	Partial CO_2-pressure in soil solution as multiple of the atmospheric CO_2 pressure (-)
cOrgacids	Total concentration of organic acids (m*DOC) (eq m^{-3})
Nimacc	Acceptable nitrogen immobilised in the soil (eq ha^{-1} a^{-1})
Nupt	Net growth uptake of nitrogen (eq ha^{-1} a^{-1})
fde	Denitrification fraction (0≤fde<1) (-)
Nde	Amount of nitrogen denitrified (eq ha^{-1} a^{-1})
CEC	Cation exchange capacity (meq kg^{-1})
bsat	Base saturation (-)
yearbsat	Year in which the base saturation was determined
Cpool	Amount of carbon in the topsoil (g m^{-2})
CNrat	C/N ratio in the topsoil (g/g)
yearCN	Year in which the CNrat and Cpool were determined
Measured	On-site measurements included in the data for CL calculations: 0 – No measurements, 1 ICP – Forest, 2 – ICP Waters, 4 – ICP Integrated Monitoring, 8 – ICP Vegetation, 16 – Other Measurement Program. (If more than one of the listed possibilities applies, the numbers should be added!)

2 Data structure for empirical critical loads:

Table 4. Attributes of the table '*EmpNload*'.

Variable	Explanation	Note
SiteID	Identifier for the site	1)
CLempN	Empirical critical load of nitrogen (eq ha^{-1} a^{-1})	

3 Data structure for dynamic modelling output:

To be able to analyse changes over time of the chemical status of ecosystems, dynamic modelling has to be carried out. From these model runs the following 7 variables for 4 'historic' years (1980, 1990, 2000, 2010) and 5 future years (2020, 2030, 2040, 2050, 2100) for a number of deposition scenarios (see below) are requested (concentrations are from the soil solution or, when aquatic systems are modelled, in the surface water) (meq m^{-3}):

1. [Al^{3+}] (meq m^{-3} = μeq L^{-1})
2. [Bc]=[Ca+Mg+K] (meq m^{-3})
3. pH
4. ANC concentration (meq m^{-3})
5. Base saturation (fraction)
6. C:N ratio (g g^{-1})
7. Nitrogen (= [NO_3]+[NH_4]) concentration (meq m^{-3})

These variables are independent of any scenario from 1980 until 2010 – and thus calculated only once (and labelled 'His') – while for the years thereafter output is required for every deposition scenario. The scenarios are composed of two base scenarios (CLE and MFR) and a number of deposition paths derived from it; and a description is given below.

Table 5. Attributes of the table '*scenvars*'.

Variable	Explanation
SiteID	Identifier for the site (relate to Table 1)
ScenName	Name of the scenario (3 characters)
Year	Year
depN	total nitrogen deposition in that year (eq ha^{-1} a^{-1})
depS	sulphur deposition (excluding sea-salt fraction) in that year (eq ha^{-1} a^{-1})
cAl	aluminium concentration (meq m^{-3})
cBc	base cation concentration (meq m^{-3})
pH	pH (–)
ANC	ANC concentration (meq m^{-3})
Bsat	base saturation (fraction)
CNrat	C:N ratio (g g^{-1})
cN	nitrogen ($[NO_3]$ + $[NH_4]$) concentration (meq m^{-3})

4 Aquatic ecosystems:

As mentioned above, the results of the critical load calculation is no longer part of the '*h2oinputs*' table, but resides in the 'CLdata' table (see Table 2). Note that some (barely used) variables, e.g. for sedimentation, are no longer in the table!

Table 6. Attributes of the table 'h2oinputs'.

Variable	Explanation
SiteID	Identifier for the site
crittype	Criterion used: =5: [ANC] (eq/m^3); see table 3 for other criteria
critvalue	Value of the criterion used
SoilYear	Year for soil measurements
ExCa	Exchangeable pool of calcium in given year (%)
ExMg	Exchangeable pool of magnesium in given year (%)
ExNa	Exchangeable pool of sodium in given year (%)
ExK	Exchangeable pool of potassium in given year (%)
Thick	Thickness of the soil (m)
Porosity	Soil pore fraction (%)
bulkdens	Bulk density of the soil (g cm^{-3})
Nimacc	Acceptable amount of nitrogen immobilised in the soil (eq ha^{-1} a^{-1})
CEC	Cation exchange capacity (meq kg^{-1})
HlfSat	Half saturation of SO4 ads isotherm (ueq L^{-1})
Emx	Maximum SO4 ads capacity (meq kg^{-1})
Nitrif	Nitrification in the catchment (meq m^{-2} a^{-1})
Denitrf	Denitrification rate in catchment (meq m^{-2} a^{-1})
Cpool	Amount of carbon in the topsoil in the given yearCN(g m^{-2})
Npool	Amount of nitrogen in the topsoil in the given yearCN(g m^{-2})
CNRange	C/N ratio range where N accumulation occurs (mol mol^{-1})

Variable	Explanation
CNUpper	Upper limit of C/N ratio where N accumulation occurs (mol mol^{-1})
CaUpt	Net growth uptake of calcium (meq m^{-2} a^{-1})
MgUpt	Net growth uptake of magnesium (meq m^{-2} a^{-1})
KUpt	Net growth uptake of potassium (meq m^{-2} a^{-1})
NaUpt	Net growth uptake of sodium (meq m^{-2} a^{-1})
SO4Upt	Net growth uptake of sulphate (meq m^{-2} a^{-1})
NH4Upt	Net growth uptake of ammonia (meq m^{-2} a^{-1})
DepYear	Year for deposition measurements
Cadep	Total deposition of calcium (eq ha^{-1} a^{-1})
Mgdep	Total deposition of magnesium (eq ha^{-1} a^{-1})
Kdep	Total deposition of potassium (eq ha^{-1} a^{-1})
Nadep	Total deposition of sodium (eq ha^{-1} a^{-1})
Cldep	Total deposition of chloride (eq ha^{-1} a^{-1})
NH4dep	Total deposition of ammonia (eq ha^{-1} a^{-1})
NO3dep	Total deposition of nitrate (eq ha^{-1} a^{-1})
LakeYear	Year for lake measurements
Calake	Measured concentration of calcium in lake(μmol L^{-1})
Mglake	Measured concentration of magnesium in lake(μmol L^{-1})
Nalake	Measured concentration of sodium in lake(μmol L^{-1})
Klake	Measured concentration of potassium in lake(μmol L^{-1})
NH4lake	Measured concentration of ammonia in lake(μmol L^{-1})
SO4lake	Measured concentration of sulphate in lake(μmol L^{-1})
Cllake	Measured concentration of chloride in lake(μmol L^{-1})
NO3lake	Measured concentration of nitrate in lake(μmol L^{-1})
RelArea	The area of the lake relative to the catchment (%)
RelForArea	The area of the forest relative to the catchment (%)
RetTime	Retention time in the lake (a)
Qs	Annual runoff flux (m a^{-1})
expel	Exponent for the Al-H relationship ()
pCO2	Partial CO2-pressure in the lake in relation to the atmospheric CO2 pressure (%atm)
DOC	DOC concentration in the lake (μmol L^{-1})
Nitriflake	Nitrification in the lake (%)
UptNH4lake	Uptake of ammonia in the lake (in % of measured value)
UptNO3lake	Uptake of Nitrate in the lake (in % of measured value)
Measured	On-site measurements included in the data for CL calculations: 0 – No measurements, 2 – ICP Waters, 4 – ICP Integrated Monitoring, 16 – Other Measurement Program. (If more than one of the listed possibilities applies, the numbers should be added!)

Updated critical concentrations for $CL_{nut}(N)$ calculations

Following the CCE/Alterra Report (De Vries et al., 2007) the critical concentrations have been updated and are now also revised in the Mapping Manual (see www.icpmapping.org). These latest values are also listed in Table 7. NFCs are urged to re-visit their $CL_{nut}(N)$ calculations and update them if appropriate. The values in Table 6 are converted to meq m^{-3} by multiplying them with 1000/14=71.428; and it is these multiplied values which should be entered under 'cNacc' in Table 3.

Table 7. Critical (acceptable) N concentrations in soil solution for calculating CLnutN.

Impact	Critical N concentration (mg N L^{-1})
Vegetation changes (data established in Sweden) [1]	
Lichens to cranberry (lingonberries)	0.2-0.4
Cranberry to blueberry	0.4-0.6
Blueberry to grass	1-2
Grass to herbs	3-5
Vegetation changes (data established in The Netherlands) [1]	
Coniferous forest	2.5-4
Deciduous forest	3.5-6.5
Grass lands	3
Heath lands	3-6
Other impacts on forests	
Nutrient imbalances	(0.2-0.4)
Elevated nitrogen leaching/N saturation	1
Fine root biomass/root length	1-3
Sensitivity to frost and fungal diseases	3-5

[1] Note that these values should be used with caution, e.g. in areas with high precipitation.

See also the report of the workshop on effects of low-level nitrogen deposition (Stockholm) at http://www.unece.org/env/wge/26meeting.htm and the minutes of the Taskforce meeting (http://www.icpmapping.org/pub/workshop/sofia/ICPMM_final_Minutes_23TFM.pdf).

Deposition scenarios

The dynamic model should be run with the historic deposition until 2010 and from 2020 till 2100 with ***14 scenarios***, with a linear transition between 2010 and 2020 and staying constant thereafter. The scenarios are derived from 2 given scenarios: "Current LEgislation" (**CLE**), assuming implementation of current legislation (Gothenburg Protocol, NEC directive, etc.) by 2010 and "Maximum Feasible Reductions" (**MFR**), assuming the implementation of maximum technically feasible reductions. These two scenarios are used to derive the other scenarios: First the distance between MFR (=S01) and CLE (=S04) is divided into 3 intervals; then this interval length is used to go beyond CLE and below MFR. This defines the grid shown in Figure 1, which also shows those points (= deposition pairs) which constitute valid scenarios. Note that the lower- and left-most grid cells can be smaller than the others, depending how close MFR is to the origin (minimum distance of 10 eq/ha/a indicated by the grey stripe).

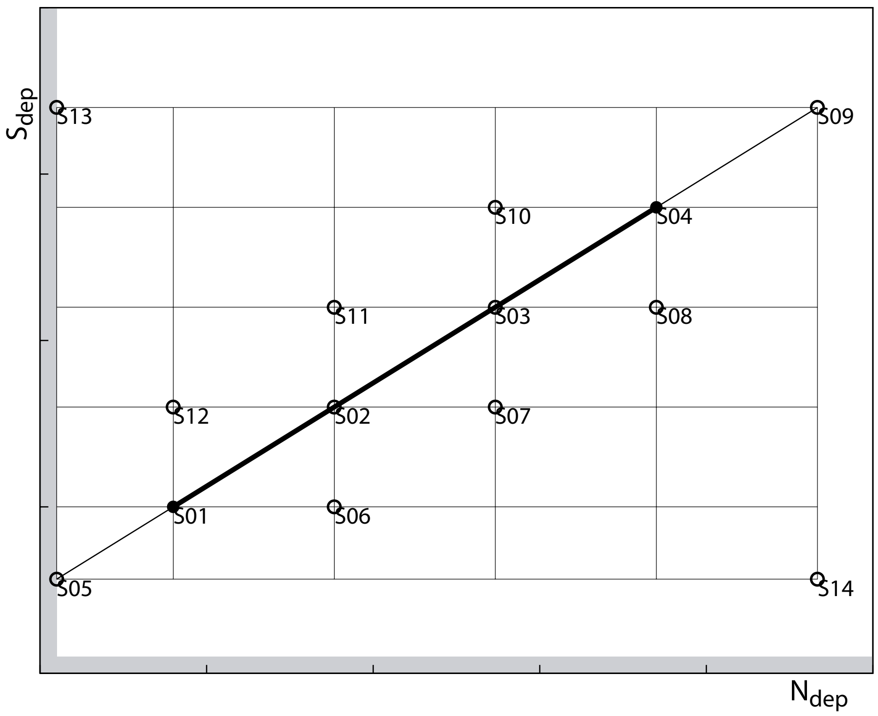

Figure 1: Scenarios S01–S14, i.e. deposition pairs (N_{dep},S_{dep}), derived from given MFR (=S01) and CLE (=S04) scenarios.

The depositions (S, NO_x-N, NH_3-N) for all scenarios and all EMEP50 grid cell covering a country can be provided by the CCE in three files, one per land-cover class (grid-average, forest, semi-natural vegetation), upon request. In the VSD-Access software provided by the CCE this is done 'automatically'. Note that the NFC application of these deposition scenarios will also enable the CCE to compute Target Loads.

Documentation

Please provide the CCE with documentation to substantiate and justify sources and methods applied in response to the call for data. It is strongly recommended to apply agreed methods as described in Chapter 5 of the Mapping Manual (www.icpmapping.org) and only list and describe the deviations from the Manual.

The CCE reporting requirements are currently best served by using the WORD-template provided or with a plain single-column WORD layout. Please avoid complicated formatting of your text and tables and figures: E.g., no special fonts; also, figure captions should be plain text and not part of the figure!

The final uniform layout will be done by the CCE.

Appendix C Examples of Links between (excessive) Nitrogen Deposition and Ecosystem Services affecting Human Well-being[1)]

This Appendix is related to the context of section 8 in Chapter 1. It gives examples of relationships between (excessive) nitrogen deposition and ecosystem services. Tables C-1, C-2 and C-3 provide examples of effects of nitrogen deposition on provisioning, regulating and cultural services, respectively. The first, second and third column of each of the tables specify an ecosystem service, a related nitrogen effect and the processes involved in relating these effects to nitrogen deposition. A more detailed treatment of relationships between nitrogen inputs in general and human health effects can be found in, e.g., Townsend et al. (2003).

Table C-1 distinguishes the production of food, fiber/wood and fresh water as examples of *provisioning services*. With respect to food it is noticed that nitrogen deposition can cause a decline of food quality as well as an increase of food quantity. The processes involved are summarized in column 3, i.e. acidification leading to an excessive content of cadmium in crops and nitrogen deposition serving as fertilizer in N-limited systems.

Table C-2 lists some effects of nitrogen deposition on *regulating services*, such as water quality. A recent article by Rupert (2008) substantiates that long-term exposure to nitrate may adversely affect human health at concentrations that are low, e.g. in comparison to the WHO guideline for drinking water quality (nitrate and nitrite, 50 mg L^{-1} total nitrogen).

The importance of *cultural services* of nature to human well-being is extensively described in the synthesis report of the Millenium Ecosystem Assessment (2005). Table C-3 only gives an example of an effect of nitrogen in terrestrial and aquatic natural systems, i.e. the increased growth of stinging nettle and algal bloom respectively, of which the adverse effects on amenity services (aesthetic enjoyment, recreation, artistic and spiritual fulfilment, intellectual development; see Millenium Ecosystem Assessment, 2005, pp. 46) are clear to everyone. The importance of *cultural services* on cultural identity and social stability is a more generic way of addressing the benefits of ecosystems to human well-being.

Table C-1 Links between the exceedance of nitrogen critical loads and the impairment of *provisioning services*

Ecosystem service: *Provisioning services*	Examples of nitrogen effects	Causal link with nitrogen deposition
Food	Decline in food quality	Nitrogen induced soil acidification increases Cd availability leading to Cd contents in crops exceeding food quality criteria.
	Increase in food production	Nitrogen deposition increases crop growth in N limited systems (low N fertilizer inputs).
	Decrease in food production	Excess atmospheric N oxides. driving high levels of tropospheric ozone may cause crop losses.
Fiber/wood	Increased growth Loss of timber	In N limited systems, nitrogen increases forest growth and wood production. Acidification may increase uprooting of trees by storms (Braun et al. 2003).
Fresh water	(see regulating services)	

1) Wim de Vries (Alterra, Wageningen University and Research Centre (WUR), Wageningen, The Netherlands) is acknowledged for his dedicated involvement in the compilation of the Tables in this Appendix during a CCE brainstorm (Bilthoven, 3 July 2008) on the relationship between nitrogen deposition and human well-being. However, the responsibility for any lack of accuracy or completeness of this Appendix lies with Jean-Paul Hettelingh.

Table C-2 Links between the exceedance of nitrogen critical loads and the impairment of *regulating services*

Ecosystem service: *Regulating services*	Examples of nitrogen effects	Causal link with nitrogen deposition
Air quality regulation	Decline in air quality	Nitrogen deposition is correlated with increased concentrations of nitrogen oxides (NO_x), ozone (O_3) and particulate matter (PM10 and PM2.5), all affecting human health.
Water quality regulation	Decline in ground water and surface water (drinking water) quality	Nitrogen eutrophication and N induced soil acidification increases NO_3, Cd and Al availability, leading to NO_3, Cd and Al concentrations in groundwater and surface water exceeding drinking water quality criteria. There is increasing consensus about links between aluminium in drinking water and dementia. Higher levels of nitrate-N fed to infants may cause the "blue baby syndrome". Long term exposure to nitrate (nitrite plus nitrate as nitrogen) at concentrations as low as 2-4 mg L^{-1} in drinking water has possible links to bladder and ovarian cancer (Weyer et al. 2003 cited in Rupert 2008) and non-Hodgkin's lymphoma (Ward et al. 1996 cited in Rupert 2008). Fish dieback by algal blooms and anoxic zones (eutrophication) and aluminium on fish gills (acidification).
Soil quality regulation	Decrease in acidity buffer	Nitrogen induced soil acidification decreases the exchangeable pool of base cat ions, that may ultimately lead to reduced forest growth (unsustainable use) and affects the aluminium pool reducing soil structure.
Water quantity regulation	Increased flooding	Excess nitrogen may cause a lower Leaf Area Index due to defoliation caused by aluminium toxicity or pests/diseases, leading to an increased precipitation excess.
	Increased drought stress	Excess nitrogen causes an increased need for water by an increased growth and an increased sensitivity for drought stress by an increase in the ratio of above versus below ground biomass.
Climate regulation Green house gas balance	Increased carbon sequestration in forests	In N limited systems, nitrogen increases forest growth and the related carbon sequestration. It also causes a reduced decomposition, leading to soil carbon sequestration.
	Increased carbon sequestration in peat lands	At low N deposition, additional atmospheric nitrogen deposition may stimulate net primary productivity.
	Decreased carbon sequestration in peat lands	At high rates of N deposition, species composition changes lead to loss of peat land forming species and changed microbial activity causing degradation of peat land.s
Pest/disease regulation	Increased human allergic diseases	Increasing N availability can stimulate greater pollen production, causing human allergic responses, such as hay fever, rhinitis and asthma.

Table C-3 Links between the exceedance of nitrogen critical loads and the impairment of *cultural services*

Ecosystem service: *Cultural services*	Examples of nitrogen effects	Causal link with nitrogen deposition
Recreation/aesthetic values	Stinging nettles Algal blooms	Nitrogen induces the increase in nitrophilic species such as stinging nettles and algal blooms reducing recreational and aesthetic values of nature.

Recommendations

Further work is required to quantify relationships between for example the exceedance of critical loads and regulation of air, water and soil quality. This implies the comparison of critical load exceedances to related pollutant concentrations in these three media. For example the exceedance by deposition of NO_x and NH_3 should be compared to the air quality criteria for concentrations of NO_x and NH_3, which are 40 μg m^{-3} and 2 μg m^{-3}, respectively. Water quality is related to concentrations of aluminium, nitrate and to pH (and thereby to concentrations of e.g. cadmium for a given cadmium soil content). Soil quality can be expressed in terms of base cation weathering rates in view of base cation depletion and aluminium weathering rates in view of aluminium depletion.

More in general, questions need to be addressed on whether exceedances of nitrogen critical loads in areas in Europe (a) are the highest risk (in comparison to e.g. pollutant concentrations, see above) in terms of the precautionary principle, (b) are varying under climate change, and how or (c) are having synergetic or antagonistic relationships with other system elements and processes that cause erosion of biodiversity, e.g. in Natura 2000 areas.

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European nature sensitive to the deposition of nitrogen compounds

This report addresses the 2008 European database on spatially-specific critical loads and dynamic modelling parameters (2008 CL database). This report emphasises the risk of impacts caused by the deposition of oxidised and reduced nitrogen and includes a comparison with the former database, which was compiled in 2006 (2006 CL database).

The 2008 CL database is important because it is designed to support the revision – which is expected to commence in the coming year– of the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone, under the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) of the United Nations Economic Commission for Europe. The information on impacts of air pollution is also available to the Economic Commission in support of its Thematic Strategy on Air Pollution and to the European Environment Agency for the update of its core set of indicators.

The 2008 CL database was used to re-calculate the size of the natural areas within Europe which were at risk of eutrophication in 2000. This has shown that these areas, in total, were about 3% larger – covering 49% of nature – than was previously calculated with data from the 2006 CL database. For the EU25 countries, in the same year, this area was even 12% larger, covering about 77% of the ecosystems. This increase can be largely explained by the inclusion of European semi-natural vegetation in the 2008 CL database.

The report also describes the adverse effects of excessive nitrogen deposition that have been tentatively calculated to occur in the future, with respect to geo-chemistry and biodiversity.

The Coordination Centre for Effects (CCE) located at the Netherlands Environmental Assessment Agency (PBL) is responsible for the development of methodologies and databases of the ICP-Modelling and Mapping in support of the effect oriented work under the LRTAP Convention.