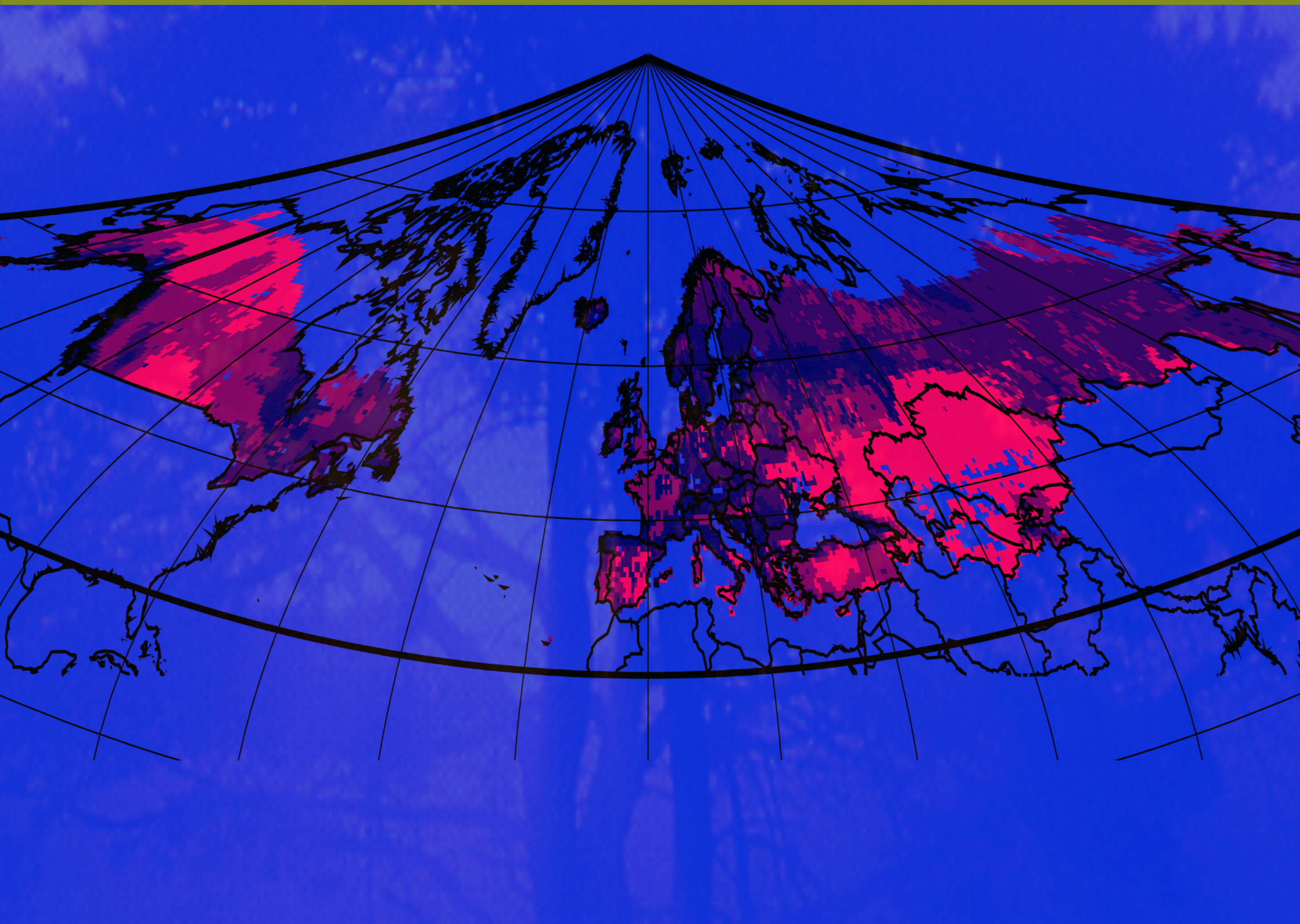
Modelling and Mapping *the Impacts of Atmospheric Deposition of Nitrogen and Sulphur*

CCE Status Report 2015



Modelling and Mapping the Impacts of Atmospheric Deposition of Nitrogen and Sulphur

CCE Status Report 2015

Colophon

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The Coordination Centre for Effects (CCE: www.wge-cce.org), located at RIVM, is the Programme Centre of the International Cooperative Programme on Modelling and Mapping (ICP M&M: www.icpmapping.org) of Critical Loads and Levels and Air Pollution Effects, Risks and Trends under the Working Group on Effects (WGE) of the Convention on Long-range Transboundary Air Pollution (LRTAP Convention: www.unece.org/env/lrtap/welcome.html).

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Publiekssamenvatting

De effecten van atmosferische depositie van stikstof- en zwavelverbindingen gemodelleerd en in kaart gebracht

Als stikstof vanuit de lucht op de bodem terechtkomt, werkt dat als een voedingsstof. Door te veel stikstof kunnen bepaalde plantensoorten verdwijnen of juist gaan overheersen. In internationale politieke gremia is daarom de vraag gesteld bij welke hoeveelheden stikstof (stikstofoxides en ammoniak) in de lucht natuurgebieden intact blijven. Het internationale Coordination Centre for Effects (CCE) helpt deze vraag te beantwoorden door een Europese database te beheren en te analyseren waarin de limieten ('kritische belastingsgrenzen') per type natuurgebied staan weergegeven. Landen uit het CCE-netwerk leveren hiervoor informatie.

Er zijn meerdere methoden om de kritische belastingsgrenzen te bepalen: op basis van de stikstofconcentratie in het bodemvocht (in de bodemlaag waar de wortels zitten) en op basis van de direct waargenomen effecten van stikstofdepositie op de natuur. Een aanvulling hierop is de relatief nieuwe methode die is gebaseerd op het gemodelleerde verlies aan biodiversiteit. Hierbij wordt een relatie gelegd tussen de planten die een bepaald soort vegetatie typeren en de omstandigheden in de bodem waaronder deze planten optimaal gedijen.

Dit jaar is voor het eerst aan de landen data gevraagd over belastingsgrenzen die zijn gebaseerd op het verlies van biodiversiteit. Duitsland en in beperkte mate het Verenigd Koningrijk hebben hieraan een bijdrage geleverd. Vijf andere landen hebben aangegeven in een volgende ronde deze methode ook te gaan passen.

Het CCE informeert beleidsmakers over de effecten van luchtverontreiniging op verschillende ecosystemen, wat de gevolgen daarvan zijn en wat het rendement van maatregelen is. De concentratie stikstof neemt al jaren af, maar is nog steeds hoog. Dit is ook als fundamenteel onderzoeksthema ingebracht in het 7th Framework-project ECLAIRE ('Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems') van de EU.

Kernwoorden: Biodiversiteit, CCE, ecosysteem effecten, luchtverontreiniging, kritische depositie waarde

Summary

Modelling and Mapping the Impacts of Atmospheric Deposition of Nitrogen and Sulphur

This report consists of three parts. The two chapters in Part 1 contain contributions to the update of the European critical loads database in 2015 based on the Call for Data issued in 2014 and the data submissions by 13 Parties to the LRTAP Convention.

In Chapter 1, the changes are described in comparison with the previous version of the critical loads database (2012), while the exceedances for the year 2010 are addressed using both the previous and current version of the critical loads databases for acidification and eutrophication. The exceedance by total nitrogen deposition of critical loads from the database of 2015 is higher than the same from the 2012 database. Overall, the European ecosystem area at risk of excessive nitrogen deposition is 61 %, compared with 55 % for the 2012 database.

Chapter 2 gives a detailed analysis of the results of the 2014/15 Call for Data, leading to the update of the critical loads database, with a focus on comparing the national submissions with the European 'background database'. This is relevant because this background dataset is used for countries that did not submit national data. The critical loads for nitrogen in the background database are generally lower than country submissions. Preliminary results of the regional application of biodiversity-based critical loads are discussed as well. Finally, the critical load for eutrophication (CLeutN) is introduced and compared with the empirical (CLempN) and the modelled critical load for nitrogen (CLnutN). The most striking changes since the 2011/2012 submissions can be noted with respect to the critical loads for acidification in Germany, the coastal regions of France and in Switzerland.

Part 2 consists of Chapters 3 and 4, which address progress made with the modelling and assessment of critical loads for biodiversity. In Chapter 3 an updated version of the PROPS model (described in Chapter 4) is used, in conjunction with the simple mass balance model, to compute the biodiversity response to nitrogen and sulphur deposition in a number of habitats on a regional scale. This response is quantified by the habitat suitability index (HSI), an indicator agreed upon by the Task Force on Modelling & Mapping in 2014 to facilitate transboundary comparisons of critical loads for biodiversity. Furthermore, methods to derive critical loads of nitrogen and sulphur from HSI calculations are described. European data and maps of biodiversity critical loads are presented and discussed. In particular, they are compared with the 'classical' acidity critical loads of N and S (see Part 1). Finally, open issues are listed that need to be resolved before biodiversity critical loads can be used in integrated assessment.

Chapter 4 describes the PROPS model used to compute the occurrence probabilities of about 4,000 European plant species as a function of pH, N and climate parameters. The underlying data (relevés) and the statistical methods used to derive the model parameters are discussed.

Furthermore, the BioScore European habitat map and database are introduced. These are used to assign plant species to habitat-EUNIS class combinations over Europe and thus enable the computation of the HSI and biodiversity critical loads for a European background database. Finally, in Part 3 the National Focal Centre reports are reproduced, describing the methods and data used for their submission to the 2014/15 Call for Data.

Keywords: Air pollution, biodiversity, CCE, critical load, eutrophication, ecosystem effects

Contents

Part 1 Progress CCE

1 Assessments using the 2015 critical loads database

Jean-Paul Hettelingh, Maximilian Posch, Jaap Slootweg

1.1 Introduction

In this chapter, the new European critical loads database is described. The Working Group on Effects (WGE) of the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) decided at its 32nd session that it is fit for use by the Task Force on Integrated Assessment Modelling (TFIAM) and is included in the GAINS system (Amann 2011) for the support of integrated assessments of policy alternatives.

In 2014, a Call for Data was issued at the request of the Working Group on Effects with the aim to:

a. Adapt the critical loads database to the 0.50° × 0.25° and 0.1° × 0.1° longitude-latitude grids used by EMEP to ensure compatibility of the European critical loads database with these new EMEP grid resolutions;
b. Offer the possibility to the National Focal Centres (NFCs) to update their national critical loads data on acidity and eutrophication;
c. Apply novel approaches to calculate nitrogen and sulphur critical load functions, taking into account their impact on biodiversity. For this, the National Focal Centres are encouraged to use the 'Habitat Suitability Index' (HS-index) agreed at the Modelling and Mapping Task Force meeting.

Technical information regarding critical loads in general can be found in De Vries et al. (2015), while the contribution by NFCs to and the results of this Call for Data are described in Chapter 2 and Part 3 of this report. A more detailed description of the modelling of and assessments with biodiversity critical loads can be found in Chapters 3 and 4.

The European critical loads database 2015 consists of critical loads of acidity (CLaci), critical loads of nutrient nitrogen and, not yet available in the 2012 European critical loads database (Slootweg et al. 2012), the critical load for eutrophication (CLeutN; see Chapter 2). In contrast with definitions used in the past, whereby the term 'critical load for eutrophication' was used interchangeably with 'critical loads of nutrient nitrogen', CLeutN is defined as either the empirical (CLempN) or modelled (CLnutN) critical load of eutrophying N or – if a site has assigned both values – the minimum of the two. This means that the default of the European critical loads database of 2015 used for policy support is now using CLeutN instead of CLnutN (as in past versions of the database)[1]. However, CLnutN and CLempN remain available for specific exercises.

[1] For applications of the 2015 critical loads database in integrated assessment, Austria and Germany stipulated side-constraints to the assignment of critical loads to their ecosystems. This results in a European critical loads database for use in the GAINS model, whereby CLempN is not used for German ecosystems nor for Austrian forests.

In this chapter, the distribution of CLeutN over Europe is compared with that of CLnutN in the critical loads databases of 2015 and 2012, respectively. Secondly, the exceedances of the acidity critical loads are compared using the sulphur and nitrogen depositions for the year 2010 for both critical loads databases.

1.2 The European critical loads database 2015

The European critical loads database 2015 consists of data submitted by 13 Parties to the Convention (see Chapter 2). Critical loads for the other Parties to the Convention have been obtained from the European Background Database (see Chapter 2 for references) used under the LRTAP Convention, which is maintained and held at the Coordination Centre for Effects (CCE). The number of critical loads records ('sites') and the area for which critical loads have been computed are listed by country in Table 1.1.

Table 1.1 shows that acidity critical loads for Europe have been computed for an ecosystem area of about 3.4 million km^2, whereas for nutrient N critical loads the ecosystem area varies between ~2.8 and ~3.1 million km^2 (Europe is about 10 million km^2).

Table 1.1 The number of ecosystem records ('sites') and the ecosystem area for acidity (CLaci), nutrient nitrogen (CLnutN), empirical nitrogen (CLempN) and eutrophication (CLeutN) critical loads.

Party	CLaci		CLnutN		CLempN		CLeutN	
	# recs	1,000 km^2	# recs	1,000 km^2	# recs	1,000 km^2	# recs	1,000 km^2
Albania	6	18	6	18	4	13	6	18
Austria*	16	39	16	39	25	50	27	51
Belarus	17	65	17	65	16	62	17	65
Belgium*	26	5	28	6	0	0	28	6
Bosnia & H.	13	34	13	34	12	31	13	34
Bulgaria	45	51	45	51	38	46	45	51
Croatia	14	34	14	34	11	29	14	34
Cyprus	1	2	1	2	0	1	1	2
Czech Rep.*	1	7	1	7	1	7	1	7
Denmark	6	5	6	5	5	4	6	5
Estonia	21	27	21	27	17	24	21	27
Finland*	†145	236	†145	236	31	41	31	41
France*	22	180	22	180	21	177	22	180
Germany*	554	105	554	105	377	73	554	105
Greece	64	67	64	67	30	32	64	67
Hungary	25	27	25	27	21	24	25	27
Ireland	23	56	23	56	20	54	23	56
Italy*	32	101	32	106	79	97	32	106
Kosovo	1	4	1	4	1	4	1	4
Latvia	32	38	32	38	27	34	32	38
Lithuania	20	21	20	21	18	20	20	21
Luxembourg	1	1	1	1	1	1	1	1
Macedonia FYR	6	15	6	15	5	13	6	15
Malta	0	0	0	0	0	0	0	0
Moldova	1	4	1	4	1	4	1	4
Montenegro	3	8	3	8	3	8	3	8
Netherlands*	60	4	73	5	†7	15	73	5
Norway*	14	320	†77	206	80	304	80	304
Poland*	141	74	141	74	141	74	141	74
Portugal	31	37	31	37	22	27	31	37
Romania	60	105	60	105	57	102	60	105
Russia	156	820	156	820	156	820	156	820
Serbia	10	31	10	31	10	30	10	31
Slovakia	22	24	22	24	22	24	22	24
Slovenia	9	14	9	14	8	13	9	14
Spain	180	235	180	235	104	137	180	235
Sweden*	16	395	†185	300	9	217	9	217
Switzerland*	11	10	11	10	19	15	29	24
Ukraine	26	95	26	95	26	95	26	95
United Kingd.*	365	77	113	16	268	57	381	73
EU	1,932	1,968	1,863	1,816	1,361	1,379	1,852	1,608
non-EU	264	1,424	327	1,310	332	1,398	348	1,422
Europe	2,196	3,392	2,190	3,126	1,693	2,778	2,200	3,030

*National data submitted by NFC; †No NFC data; European Background Database used

1.2.1 *Critical loads for acidification*

Comparison of the distribution of CLaci between the critical loads databases of 2012 and 2015 by displaying the 5th, 50th (median) and 95th percentile indicates an increase of the European area with low critical loads (Figure 1.1). The geographical pattern of 5th percentile critical loads, the value of which protects 95% of ecosystems in a grid cell, indicates ranges that are lower in the updated 2015 database than in the 2012 database.

For example, this is the case in western and southern France and in the border areas of the Netherlands, with more critical loads below 100 eq $ha^{-1}a^{-1}$ (red shadings) occurring in 2015. In Germany, CLaci ranges exceeding 1,000 eq $ha^{-1}a^{-1}$ (blue) and between 400-700 eq $ha^{-1}a^{-1}$ (dark green) in 2012 shift to lower ranges in 2015 (light green and yellow).In south-western Sweden, larger areas with 5th percentile critical loads of acidity lower than 100 eq $ha^{-1}a^{-1}$ occur in the 2015 critical loads database. The occurrence of relatively lower critical loads can also be seen in Sweden, where the median critical loads (protecting 50% of the ecosystems against acidification) in 2015 include areas with CLaci in the range of 100-200 eq $ha^{-1}a^{-1}$.

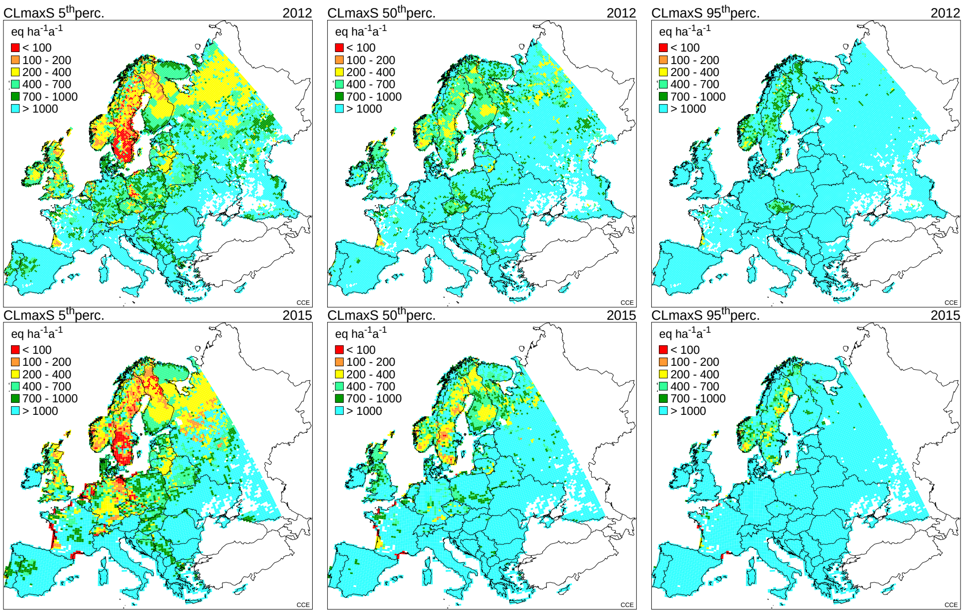

Figure 1.1 The 5th percentile (left), 50th percentile (centre) and 95th percentile (right) critical load of acidity of the European critical loads database in 2012 (top) and 2015 (bottom).

1.2.2 *Critical loads for eutrophication*

In Figure 1.2, the European CLnutN critical loads database of 2012 is compared with the CLeutN critical loads database of 2015. It illustrates that CLeutN in 2015 tends to be equal or lower than CLnutN in 2012, with a noticeable decrease in 2015 compared with 2012 of the 5th, 50th and 95th percentile critical loads in, for example, Ireland, northern Germany and western Austria. The extent to which this affects the exceedances in Europe is explored in the next section. A more detailed investigation into the update can be found in Chapter 2.

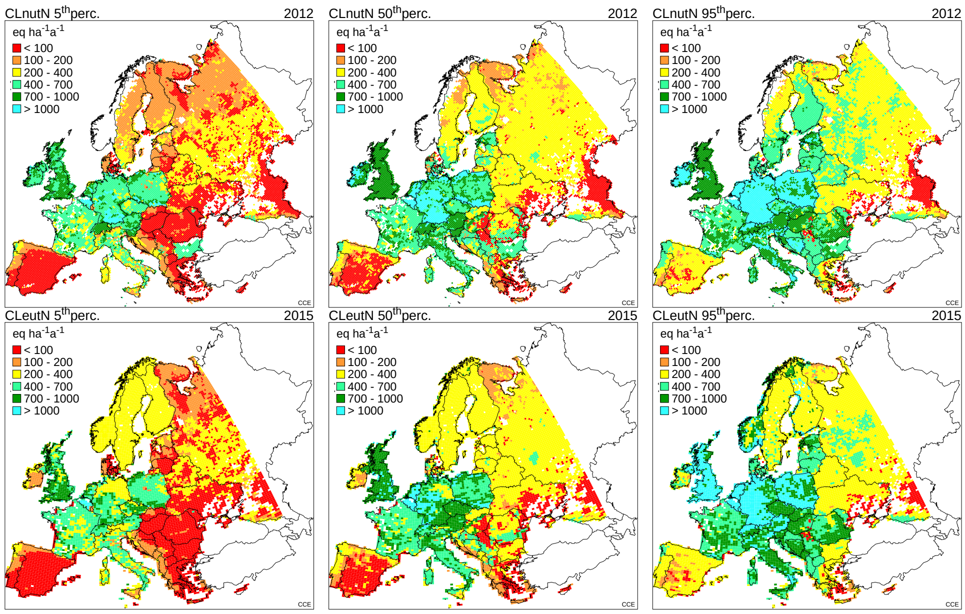

Figure 1.2 The 5th percentile (left), 50th percentile (centre) and 95th percentile (right) critical load of nutrient nitrogen (CLnutN) of the European critical loads database in 2012 (top) and the critical load for eutrophication (CLeutN) in 2015 (bottom).

1.3 Exceedances of European critical loads

1.3.1 Computing exceedance

All exceedances shown in this chapter are average accumulated exceedance (AAE; Posch et al. 2015). In a grid cell (or any region), the AAE is obtained by (i) computing the exceedance of the critical load by the deposition for every site, and (ii) taking the area-weighted average over all ecosystems in that grid cell (or region).

In this chapter, we report exceedances using modelled deposition for the year 2010. These depositions were computed from country emissions, developed for the Thematic Strategy on Air Pollution (TSAP; e.g. Amann et al 2014) and the source-receptor matrices, prepared by EMEP (www.emep.int) and used in the GAINS model (Amann et al. 2011).

1.3.2 Exceedance of critical loads of acidification

In Figure 1.3, exceedances of CLaci computed with the European databases of 2012 and 2015 are compared using depositions for the year 2010.

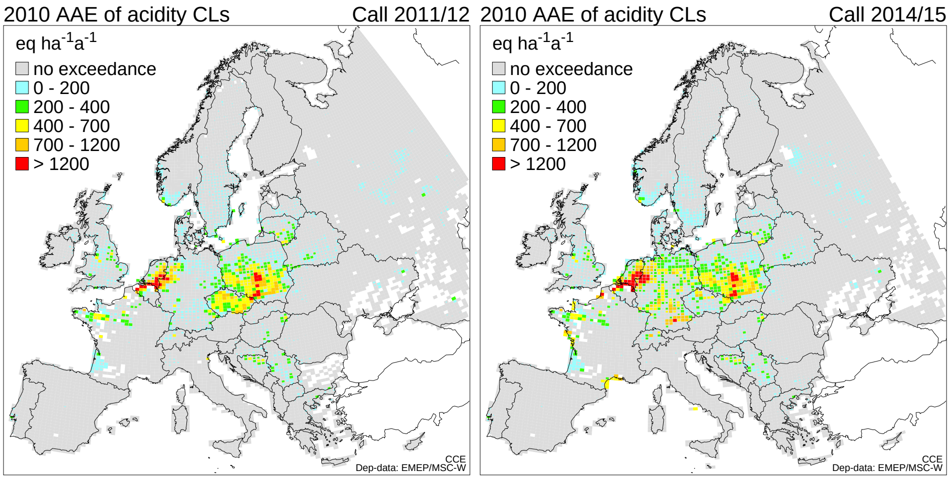

Figure 1.3 The average accumulated exceedance (AAE) computed with 2010 nitrogen and sulphur deposition using the critical loads databases of acidity CLs of 2012 (left) and 2015 (right).

Exceedances for acidification above 400 eq $ha^{-1}a^{-1}$ occur particularly in Germany and Poland for the critical loads database of 2015 (Figure 1.3, right). Indeed, 38% (Table 1.2) of the German ecosystem area is computed to be at risk of acidification in 2010, compared with about 22% when the 2012 CL database and 2010 EMEP depositions are used (Table 1.3). In the Czech Republic, exceedances of the 2015 critical loads for acidity cover 33% of the ecosystem area (Table 1.2), considerably lower than the 78% of area exceeded when using the 2012 CL database (Table 1.3). Overall in Europe, the area at risk of acidification is 7% (8% in the EU) using the 2015 critical loads database (Table 1.2). This implies that a larger area is at risk of acidification than that computed with the 2012 critical loads database, in which it was computed to be 5% (8% in the EU) (Table 1.3).

1.3.3 *Exceedance of critical loads of eutrophication*

The AAE of nutrient nitrogen is computed from the total deposition of nitrogen in 2010, using CLnutN and CLeutN from the European critical loads databases of 2012 and 2015, respectively (Figure 1.4).

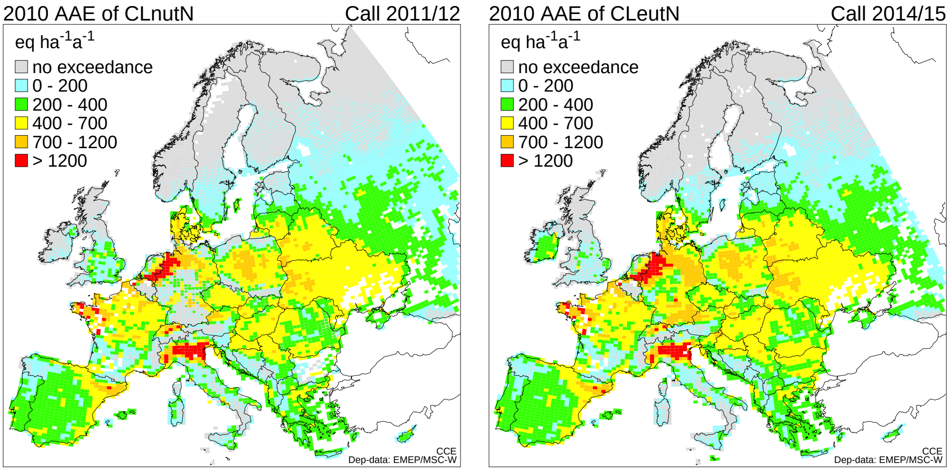

Figure 1.4 The average accumulated exceedance (AAE) by total nitrogen deposition in 2010 of CLnutN and CLeutN using the European critical loads databases of 2012 (left) and 2015 (right), respectively.

As can be expected from the changes in the magnitude of critical loads (Figure 1.2), the exceedance caused by total nitrogen deposition of critical loads from the database of 2015 is higher than it is when using the 2012 database in several countries. For example, in Ireland, Germany and the Czech Republic, the area at risk of nitrogen deposition exceeding CLeutN is 86%, 96% and 100% (see Table 1.2) of the national ecosystem area, respectively, compared with 16%, 56% and 92%, respectively, when CLnutN from the 2012 critical loads database is used (see Table 1.3).

Overall, using the 2015 critical loads database, the European ecosystem area at risk of excessive nitrogen deposition over CLeutN is 62% (75% in the EU) and 58% (EU: 65%) if CLnutN is used (Table 1.2). In comparison, when using the 2012 critical loads databases for CLnutN, it turns out that 55% (EU: 63%) of the area is at risk (Table 1.3).

*Table 1.2 Using the **2015** Critical load database: Ecosystem area at risk (%), i.e. area where the acidity (CLaci), nutrient nitrogen (CLnutN), empirical (CLempN) and eutrophication (CLeutN) critical loads have a positive average accumulated exceedance (AAE in eq ha^{-1} a^{-1}).*

Parties	**exceedance of CLaci**		**exceedance of CLnutN**		**exceedance of CLempN**		**exceedance of CLeutN**	
	%	*AAE*	*%*	*AAE*	*%*	*AAE*	*%*	*AAE*
Albania	0	0	89	247	20	22	89	247
Austria	0	0	65	221	74	233	76[a]	250
Belarus	14	35	100	509	93	378	100	524
Belgium	3	7	1	2	86	204	3	7
Bosnia & Herz.	13	75	72	205	31	55	76	208
Bulgaria	1	5	100	348	39	54	100	349
Croatia	4	18	89	364	53	131	91	383
Cyprus	0	0	100	277	10	8	100	277
Czech Rep.	33	141	42	100	100	468	100	473
Denmark	22	32	100	551	71	298	100	576
Estonia	0	0	47	45	55	62	80	92
Finland	0	0	10	5	7	3	7	3
France	10	26	83	329	46	105	85	337
Germany	38	261	67	483	98	498	96[a]	628
Greece	2	6	98	283	21	29	98	284
Hungary	7	34	95	453	69	193	100	504
Ireland	0	0	84	230	14	14	86	232
Italy	0	0	63	260	64	339	63	260
Kosovo	10	27	78	182	13	17	78	182
Latvia	9	12	90	186	37	67	96	214
Lithuania	32	146	99	434	77	287	100	466
Luxembourg	13	73	100	708	68	449	100	764
Macedonia FYR	6	10	87	235	12	11	87	236
Malta	0	0	97	364	100	259	100	378
Moldova	0	0	100	361	47	78	100	378
Montenegro	0	0	64	98	41	48	71	106
Netherlands	84	1,277	89	900	56	439	89	900
Norway	8	10	4	3	5	6	5	6
Poland	49	277	77	391	87	323	89	427
Portugal	1	2	100	250	17	36	100	253
Romania	1	3	94	333	39	58	98	341
Russia	2	2	47	73	14	23	47	73
Serbia	24	95	91	377	36	71	92	379
Slovakia	6	24	94	402	67	141	99	424
Slovenia	0	0	89	364	86	346	100	497
Spain	0	0	97	319	34	91	98	322
Sweden	9	11	24	28	20	37	20	37
Switzerland	16	93	73	466	48	197	58	304
Ukraine	2	4	100	540	73	193	100	540
United Kingdom	8	19	42	87	8	12	15	28
EU	8	36	65	231	44	140	75[a]	283
non EU	5	10	50	136	21	50	47	130
Europe	**7**	**25**	**58**	**191**	**33**	**95**	**62[a]**	**211**

[a] 65% (Austria) and 67% (Germany) when side-constraints apply (see para. 1.1) with respect to the use of their critical loads data in integrated assessments, resulting in a European area at risk of 61% (73% in the EU).

*Table 1.3 Using the **2012** Critical loads database**:** Ecosystem area at risk (%), i.e. where the acidity (CLaci), nutrient nitrogen (CLnutN) and empirical (CLempN) critical loads have a positive average accumulated exceedance (AAE in eq ha^{-1} a^{-1}).*

Parties*	**exceedance of CLaci**		**exceedance of CLnutN**		**exceedance of CLempN**	
	%	*AAE*	*%*	*AAE*	*%*	*AAE*
Albania	0	0	89	243	21	23
Austria	0	0	69	234	62	191
Belarus	14	36	100	498	94	379
Belgium	7	19	1	3	48	105
Bosnia & Herz.	13	75	70	179	31	56
Bulgaria	0	0	63	128	21	22
Croatia	4	19	88	348	53	132
Cyprus	0	0	100	266	3	2
Czech Republic	78	387	92	416	74	402
Denmark	22	33	100	538	74	307
Estonia	0	0	35	33	57	64
Finland	0	0	9	5	6	3
France	8	15	83	324	45	103
Germany	22	66	56	358	99	653
Greece	2	6	98	286	22	30
Hungary	8	37	96	471	70	200
Ireland	0	0	16	22	1	0
Italy	0	0	59	232	64	344
Latvia	11	15	92	185	38	69
Lithuania	34	166	99	419	78	296
Luxembourg	13	80	100	694	69	456
Macedonia FYR	6	8	85	225	12	12
Moldova	0	0	100	358	46	74
Netherlands	74	958	89	877	89	898
Norway	6	7	3	2	5	6
Poland	49	282	75	376	69	257
Portugal	1	2	100	245	17	36
Romania	1	3	95	330	41	62
Russia	1	1	48	70	12	16
Serbia&Montenegro	18	62	80	280	34	60
Slovakia	6	23	95	398	67	142
Slovenia	0	1	71	124	36	54
Spain	0	0	97	312	35	91
Sweden	10	12	31	45	19	24
Switzerland	9	33	74	507	42	165
Ukraine	2	4	100	530	72	190
United Kingdom	8	18	42	87	6	9
EU	8	31	63	220	41	136
non EU	3	6	48	110	17	37
Europe	**5**	**18**	**55**	**163**	**28**	**80**

*For Europe, the AAE and % area at risk of critical load exceedance is not affected by changed country borders of some Parties. Therefore, European totals in Table 1.3 can be compared to Table 1.2.

1.4 Concluding remarks

In this chapter, a summary is provided of the European critical loads database compiled by the Coordination Centre for Effects in 2015 in collaboration with National Focal Centres under the International Cooperative Programme on Modelling and Mapping and adopted by the Working Group on Effects and EMEP at their 1st joint session (Geneva, 14-18 September 2015).

This European Critical Loads Database 2015 is the basis for the substitution of the database from 2012 (Slootweg et al. 2012) currently implemented in the GAINS model for applications to support European air pollution abatement policies (see, e.g. Hettelingh et al. 2015) in the Task Force on Integrated Assessment Modelling under the LRTAP Convention and under the European Commission.

A comparison of the 2015 and the 2012 critical loads databases indicates that updates have been submitted by National Focal Centres that include increased sensitivity for acidification in areas located in France, Germany, the Netherlands and Sweden. Overall in Europe, the area at risk of acidification is 7% of the ecosystem area (8% in the EU) using the 2015 critical loads database. This implies that a larger area is at risk of acidification than that computed using the 2012 database (i.e. 5% of the ecosystem area in Europe and 8% in the EU).

Overall, using the 2015 critical loads database, the European ecosystem area at risk of eutrophication due to excessive nitrogen deposition over CLeutN is 62% of the ecosystem area (75% in the EU), while CLnutN exceedance occurs in 58% (EU: 65%) of this area. In comparison, 55% (EU: 63%) of the area is at risk when using the 2012 critical loads databases for CLnutN.

The area at risk of CLeutN exceedance changes somewhat when Austrian and German side-constraints to the use of critical loads in scenario analysis by the GAINS model are included. In that case, the European ecosystem area at risk becomes 61% of the total.

In conclusion, the use of the 2015 critical loads database to assess the area at risk of acidifying and eutrophying emissions in 2010 leads to an increase of the European ecosystem area at risk of approximately 2 and 6 percentage points, respectively, in comparison with the use of the 2012 database. This places greater urgency on the identification and analysis of those (protected) ecosystems to which obligations for protection already apply (e.g. Natura 2000 areas in the EU).

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2 Summary of National Data

Jaap Slootweg, Maximilian Posch, Jean-Paul Hettelingh

2.1 Introduction

At its 33rd session (Geneva, 17-19 September 2014), the Working Group on Effects "…requested the CCE to organize the new call for data …" (paragraph 44; ECE/EB.AIR/WG.1/2014/2) with the following aims:

- Ensure the compatibility of the European critical loads database with the revised EMEP grid, in which critical loads exceedances were computed using the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) Model.
- To allow countries to update their critical loads, possibly including a biodiversity indicator.
- To test applications of the Habitat Suitability Index on a regional scale.

This chapter describes the national data from the country submissions to this Call for Data. In Appendix A, a reprint of the instructions to the countries, with all technical details, can be found. Part of the submission is National Focal Centre (NFC) documentation to justify the data used in support of (European) air pollution abatement policies. These NFC reports can be found in Part III of this report. More information about the methods applied can be found in the Mapping Manual (ICP M&M 2015).

2.2 Overview of the responses of NFCs

The call enabled NFCs to submit critical loads of acidity, biodiversity, nutrient nitrogen and empirical critical loads. The call for data was sent to 30 Parties (i.e. member countries) to the LRTAP Convention, 13 of which responded with a submission (Table 2.1).

Table 2.1 List of countries and submitted number of ecosystems with critical loads for (nutrient and empirical), for acidification, as well as for biodiversity.

Country	Nutrient	Empirical	Acidity	Biodiv.
Austria (AT)	15,971	24,895	15,644	
Belgium (BE)*	27,814	136	25,542	
Czech Republic (CZ)**	1,201	1,201	1,201	
Finland (FI)		31245		
France (FR)	22029	21469	22,029	
Germany (DE)	553,980	377,162	553,980	55,3980
Italy (IT)	31,965		32,445	
Netherlands (NL)	72,553		63,409	
Norway (NO)		79,596	13,987	
Poland (PL)	224,358	224,358	222,900	
Sweden (SE)		16,537	16,346	
Switzerland (CH)	10,632	18,514	10,732	
United Kingdom (GB)	113,155	268,061	365,334	40
Total (13 countries)	1,073,658	1,063,174	1,343,549	554,020

*The Belgian submission covers only ecosystems in Wallonia.

**The Czech Republic has not provided documentation for their submission.

The European database of critical loads that can be used for integrated assessment consists of the data of these national submissions (without the biodiversity CLs) completed by the 'European background database' (EU-DB; see Section 2.9 and Posch and Reinds 2005; Reinds 2007; Slootweg et al. 2011). A complete list of ecosystems in the critical loads database is given in Annex 2.1 of this chapter.

The ecosystem types for which critical loads were submitted are classified by the EUNIS classification system (see http://eunis.eea.eu.int). Figure 2.1 shows the coverage of submitted ecosystems as a percentage of the country area for acidification (A), biodiversity (B), empirical (E) and modelled nutrient (N) nitrogen. For example, Austria (AT) has submitted data on empirical critical loads covering 59% of the country (47% forests, 8% grasslands, 4% scrubs and some other ecosystems, which are too few to see on the graph).

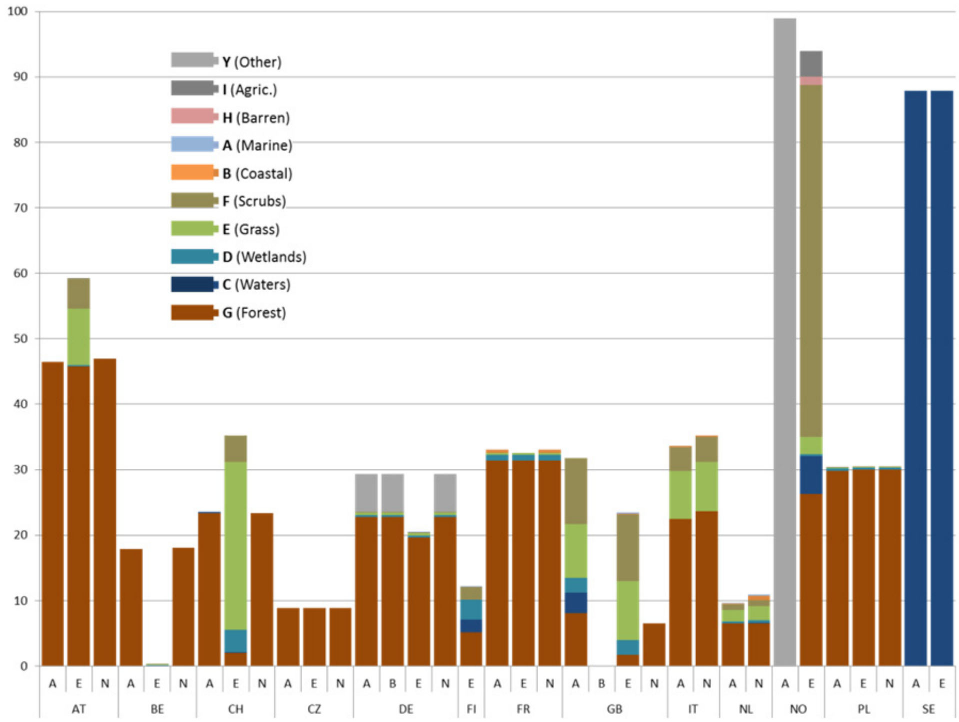

Figure 2.1 Coverage of submitted ecosystems (by EUNIS class) as a percentage of the country area related to critical loads for acidification (A), biodiversity (B) and empirical (E), as well as modelled critical loads of nutrient (N) nitrogen.

2.3 The revised EMEP grid

The European critical loads database is now on a 0.10° × 0.05° longitude-latitude grid. It is therefore compatible with both the 0.50° × 0.25° and 0.1° × 0.1° longitude-latitude grids used by EMEP to report depositions.

Screen or printer resolution is generally not sufficient to distinguish the 0.10° × 0.05° grids. The percentile maps in this chapter are shown on the 0.50° × 0.25° grid, calculated from the 0.10° × 0.05° longitude-latitude grid (merging up to 25 grid cells before calculating the percentile). The grids in the map are approximately 28 km × 28 km and comparable with the 25 km × 25 km grid maps from the 2012 database.

2.4 Critical loads of nutrient nitrogen

The right-hand map in Figure 2.2 shows the 5th percentile critical loads of modelled nutrient nitrogen (CLnutN). Plotted in the map is the 5th percentile (area weighted) of all critical load values in the 0.50° × 0.25° longitude-latitude grid cells. The European dataset has been completed by the background database for the countries that did not submit critical loads of nutrient nitrogen. For comparison, the corresponding map from the 2012 CL database is plotted at left in Figure 2.2 (on the 25 km × 25 km EMEP grid). The 2012 map is taken from the 2012 CCE Status Report (Posch et al. 2012).

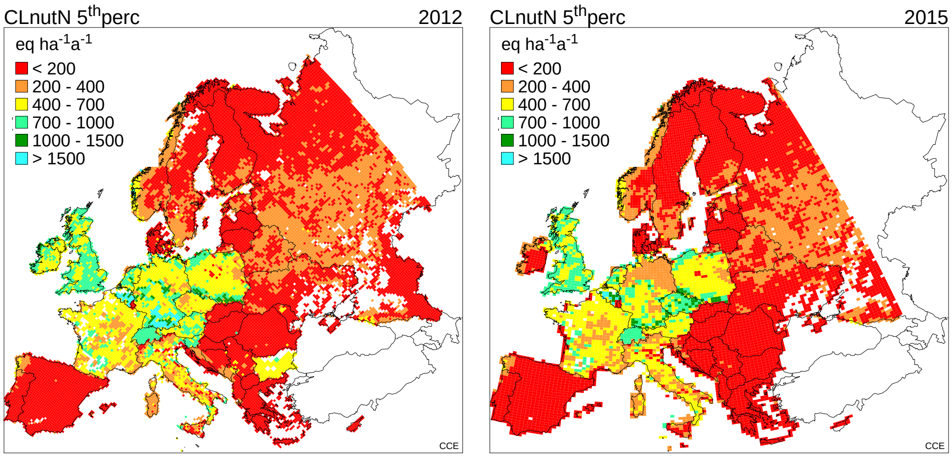

Figure 2.2 5th percentile critical loads of nutrient nitrogen (right) compared with the 2012 submission (left).

There is a clear difference between the critical loads of countries that submitted data in 2012, but not in 2015. This is due to the fact that the European background database is used for these countries in 2015. Critical loads in the European background database are generally lower than national submissions (see Section 2.10). This leads to lower critical loads in 2015 than in 2012, as can be seen in Bulgaria, Ireland and Slovenia, countries that submitted data in 2012, but not in 2015.

Changes in countries that made submissions in both years are noticeable, for example, in the northern part of Germany and the coastal regions of France.

2.5 Empirical critical loads of nitrogen

Similar to the previous section, the maps of empirical critical loads of nitrogen of the latest two database versions (2015 and 2012) are placed side by side in Figure 2.3.

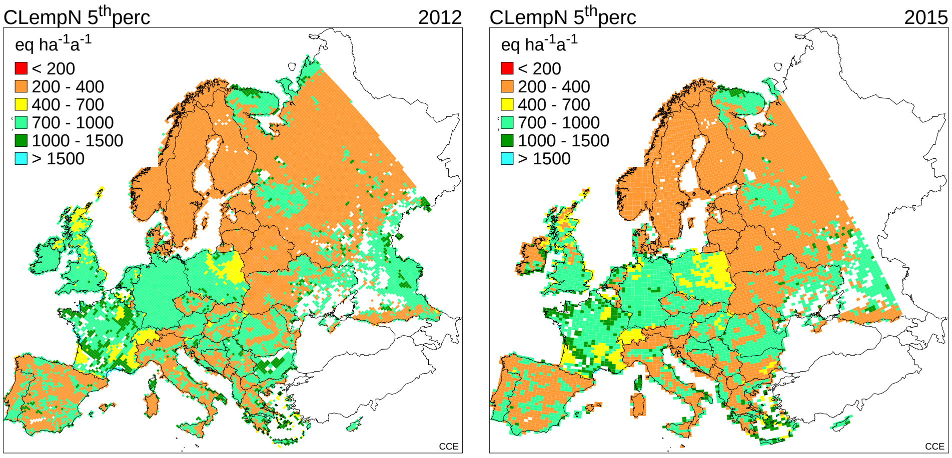

Figure 2.3 5th percentile Empirical critical loads of nitrogen in 2012 (left) compared with the 2015 submission (right).

Eleven countries submitted data for empirical critical loads (Table 2.1). The changes are less prominent than they are for nutrient nitrogen. Slight changes can be seen in the United Kingdom, especially in northern England and Scotland. Three countries, for which the background database is now used (Ireland, Slovenia and the Netherlands), have lower critical loads than they did in their 2012 submission. The reason why the critical loads differ can be twofold. Countries can apply local knowledge about the ecosystems and the ecosystem types selected can differ. Empirical values are assigned to ecosystem types, classified according to the EUNIS classification system (Bobbink and Hettelingh 2011). It therefore becomes very relevant what types of ecosystems need protection according to the submitting country. For the background database there is no selection. All ecosystems present in the harmonized land-use map (Cinderby et al. 2007) for which the empirical range is known are considered. Figure 2.4 shows the EU-DB empirical CLs in comparison with the countries that submitted empirical critical loads.

The cumulative distribution functions (cdf, see textbox) show more extreme values and more specific ecosystem types for the submitted data than the background database, such as the aquatic ecosystems (EUNIS class C) in Austria, Finland, Norway and Sweden.

A cumulative distribution function (cdf), as used in this chapter, shows the sorted variable values on the x-axis and the relative area of all ecosystems with a lower or equal value on the y-axis. In this chapter, the area (on the y-axes) are normalized to 1 for each of the EUNIS classes (A-I, X, Y) separately. For more (mathematical) explanations, see the Mapping Manual (ICP M&M 2015).

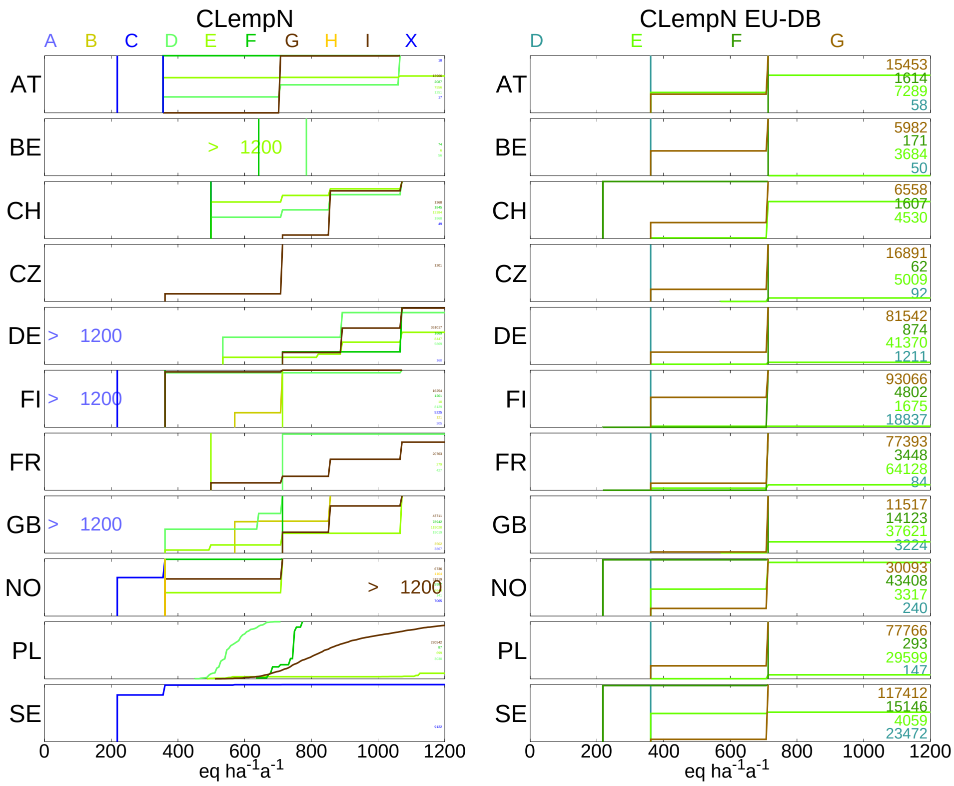

Figure 2.4 Submitted empirical critical loads for nitrogen by country and ecosystem type (left) and empirical critical loads from the background database for the same countries (right).

2.6 Critical loads for eutrophication

In this paragraph, we introduce critical loads for eutrophication. Empirical critical loads for nitrogen have been part of the calls for data since 2005 and are used instead of modelled critical loads for nutrient nitrogen by some NFCs. In the CCE workshop (20-23 April 2015, Zagreb), it was proposed that both methods be accepted as equally suitable for integrated assessment and that the combined dataset be made available for this. This leads to the introduction of the critical load for eutrophication (*CLeutN):*

The **critical load for eutrophication** of an ecosystem is either the empirical or the modelled critical load of nitrogen. For an ecosystem for which both critical loads are derived, the lower value is taken. Empirical critical load ranges are expert judgements and relate directly to observed effects of N depositions and N addition experiments. Modelled critical loads of nutrient nitrogen are based on the soil chemistry and a chemical criterion, which relates to harmful effects. These different approaches give different, but comparable results. This is demonstrated in Figure 2.5. On the left are the cdfs of the empirical (blue) and modelled critical loads (red) and in black is the resulting critical load for eutrophication; on the right, the cdfs of the critical loads for eutrophication for the different ecosystem types are shown for NFCs that submitted data.
Note that many NFCs submit empirical and modelled critical loads for other ecosystem types. The cdfs of CLeutN can be viewed in conjunction

with Figure 2.4 (left) to see the contributions of the empirical critical loads.

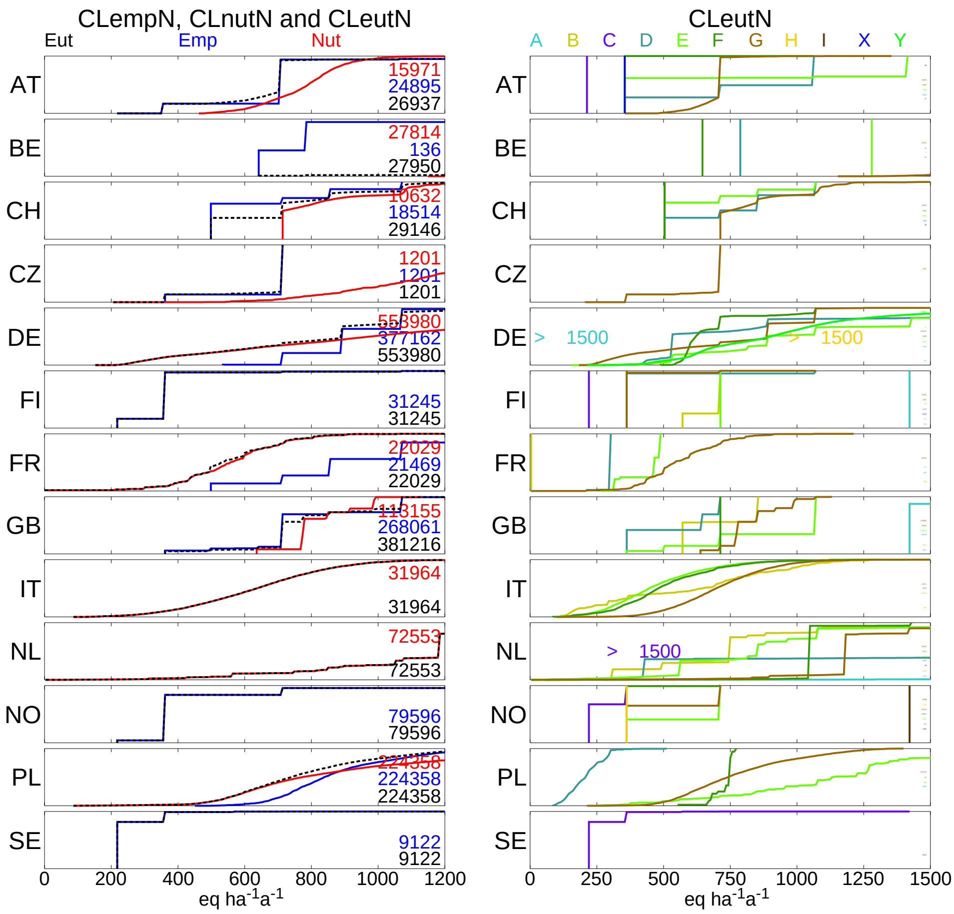

Figure 2.5 The cdfs of empirical and modelled critical loads of nitrogen, together with the critical loads for eutrophication (CLeutN; left) and cdfs of CLeutN, split by ecosystem type (right).

Figure 2.6 demonstrates the differences between empirical and modelled critical loads in another way; the plot on the left shows the difference between empirical and modelled critical loads for nitrogen for ecosystems with both critical loads submitted. The plot on the right shows the distributions of both methods for all the ecosystem types combined, together with the minimum of the two methods, CLeutN. Some distributions show a bias towards more sensitivity of one method over the other, but overall, the distributions are in the same range. For instance, for Germany, the difference between empirical and modelled critical loads for nitrogen for individual ecosystems are quite high (left side of Figure 2.6), but the distributions are comparable (right side of Figure 2.6).

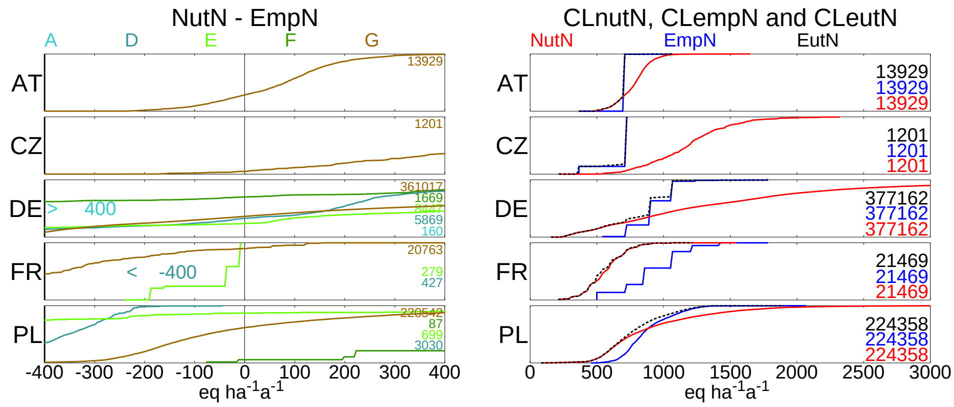

Figure 2.6 Difference between modelled and empirical critical loads of nitrogen for ecosystems for which <u>both</u> were submitted (left) and the distributions of CLnutN, CLempN and CLeutN for the same selection of ecosystems (right).

2.7 Critical loads for biodiversity

Vegetation modelling can be used to establish limits of chemical variables (e.g. a minimum pH and/or maximum N concentration) at which typical/desired/key plant species for a habitat/ecosystem can thrive/survive. Values for N and S deposition combinations, i.e. critical loads, can then be derived with soil-chemical models (e.g. SMB) and associated data. The Habitat Suitability (HS) index is used as metric to measure the extent to which a habitat can support its typical plant species. It is defined as the arithmetic mean of the normalized probabilities of occurrence of the species of interest.

The aim of the 2014/15 Call for Data was to test the applications of the HS index on a regional scale. However, that this part of the call for data was not intended to lead to critical loads for biodiversity fit for use in integrated assessment modelling and policy support. At this stage, the ICP M&M aims at scientifically sound developments and testing of new approaches to use biodiversity as an endpoint for critical loads.

Only two countries, Germany and the United Kingdom, submitted biodiversity critical loads. Austria, France, Switzerland, the Czech Republic and the Netherlands have indicated that they are working on it, but were not yet able to submit results. Figure 2.7 shows the maximum critical load of nitrogen (CLNmax) and maximum critical load of sulphur (CLSmax) for the submissions from the United Kingdom and Germany (see Posch et al., 2014, for the definition of the biodiversity CL function).

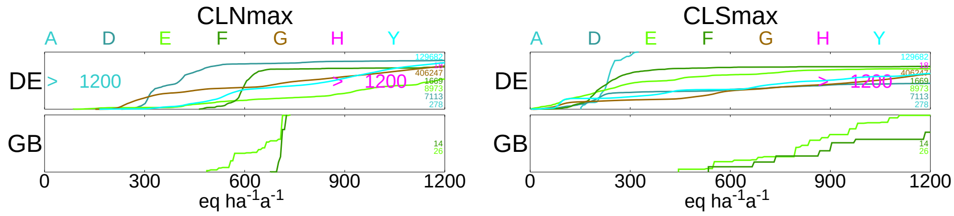

Figure 2.7 Biodiversity-based critical loads: maximum critical load of nitrogen (CLNmax) and maximum critical load of sulphur (CLSmax).

Germany submitted data for many ecosystems with values in the same range as for empirical and modelled nutrient nitrogen. The United Kingdom submitted data for a limited number of ecosystems with values within their empirical critical load range. This makes sense, because their critical limit relates to empirical critical loads.

The critical value of the HS index from Germany, the United Kingdom and (at the right) from the European background database are plotted in Figure 2.8. Germany applies the BERN model to calculate the 'possibility' (fuzzy set theory), which differs from a probability. Their criterion for the HS index is 1. For the European background database, the probability of occurrence, as applied for the index calculation, is based on presence/absence data under comparable abiotic circumstances anywhere in Europe, leading to much lower indices. The British derived a 'prevalence' from presence/absents data and set a threshold based on empirical critical loads. Currently, the HS index is derived from different modelling concepts on how to quantify plant species occurrence and its inter-comparability needs further investigation. For more details on the methods applied, see Chapter 3 and the respective national reports.

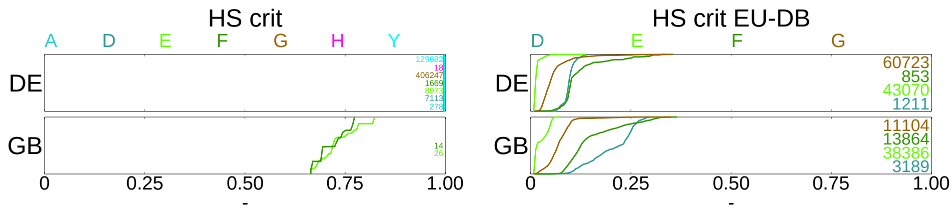

Figure 2.8 Critical value of the Habitat Suitability Index for Germany and the United Kingdom (left), and from the European background database (right).

2.8 Critical loads of acidity

The right map in Figure 2.9 shows the 5th percentile of the maximum critical loads of sulphur on a resolution of 0.50° × 0.25°. The left map shows the same information for the 2012 critical load database. Significant updates were made by Germany and Switzerland. Germany has updated data for precipitation surplus, deposition of base cations and uptake. Switzerland has set the critical limit for Bc:Al to 7. 'Values in the range of 5-10 would be more appropriate to protect forests from acidification, considering the observed storm-induced damage' (see their national report).

Also the Netherlands submitted some changes towards more sensitive values, as did France for their coastal regions. The Czech Republic, Sweden and Norway have made minor updates. The acidity CLs from the

EU-DB for countries that did not submit data in 2015, but did so in 2012, are similar to those NFC submissions, except for Ireland. The Irish 2012 submission shows much more sensitive ecosystems than the 2015 EU-DB dataset (see Posch et al. 2012). The background database has been slightly updated; for instance, the northern European part of Russia, which was and is background data, shows both lower and higher critical load values compared with 2012.

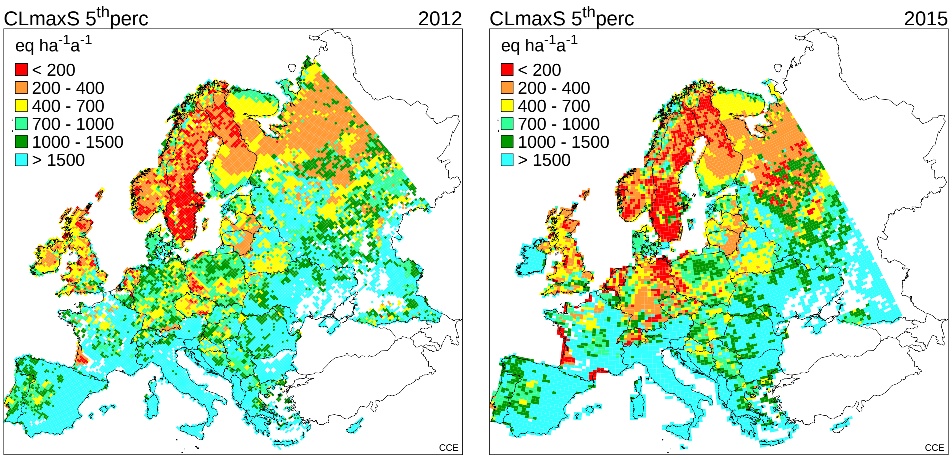

Figure 2.9 Map of the 5th percentile of the maximum critical load of sulphur (right) compared with the 2012-submission (left).

2.9 The European Background Database

The European Background Database (EU-DB) for critical loads (and related information) is maintained at the CCE and used to provide CLs for countries that do not submit national information in response to an official Call for Data. The EU-DB currently used is essentially the same as in 2012 (see, e.g. Posch and Reinds 2005; Reinds 2007; Slootweg et al. 2011). In addition to now distinguishing *all* individual successor states of former Yugoslavia, there has been an update of the forest growth data in countries of the former Soviet Union, leading to (minor) changes in the net uptake of N and base cations in those countries and thus in the critical loads of acidity and nutrient nitrogen.

In 2015, critical loads for biodiversity have been added to the EU-DB. They have been derived with the PROPS vegetation model and the HS-index (see Posch et al. 2014) using the Bioscore Habitat Suitability Map (see Chapter 4). Details are given in Chapter 3 of this report, in which maps of the EU-DB biodiversity critical loads are shown, as well as comparisons with the other CLs. Note that the biodiversity CLs are not yet used for integrated assessment under the LRTAP Convention, but they have been analysed and discussed within the EU 7th Framework project ECLAIRE ('Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems').

2.10 Variables for modelling nutrient nitrogen

The critical loads for nutrient nitrogen that countries submitted are higher than the values from the European background database. These values vary between receptors and depend on input variables such as net nitrogen uptake (Figure 2.10), acceptable nitrogen concentration in the soil solution (Figure 2.11) and net nitrogen immobilization (Figure 2.12).

The nitrogen uptake in the SMB calculations excludes the nitrogen that is cycling within the ecosystem, for instance the part that is in leaves, which are shed during the autumn and are (eventually) taken up again by the vegetation. Figure 2.10 shows the net nitrogen uptake submitted by the countries (left) and, for the same regions, the background database (right).

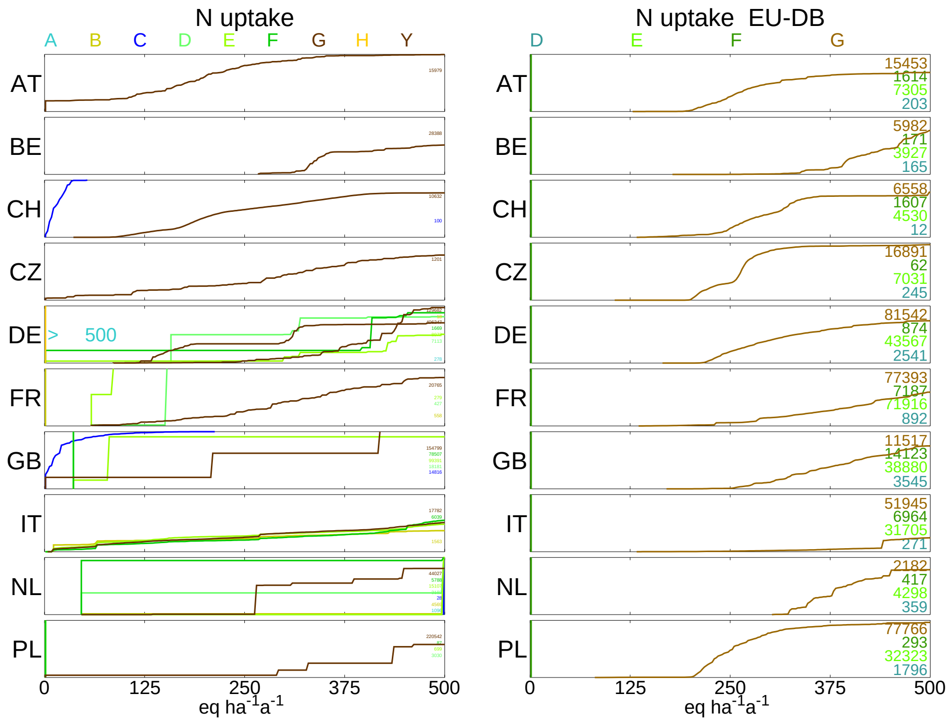

Figure 2.10 Net nitrogen uptake submitted by the countries (left) and, for the same countries, from the European background database (right).

In the EU-DB, only the harvesting of wood causes a net uptake (due to harvesting), other ecosystem types have no net uptake. Some of the ecosystems in the country submissions have nitrogen uptake of hundreds of equivalents. Notice a small fraction in some countries (e.g. Austria) where there is no harvesting of forests (G) and therefore no uptake.

The limit for nitrogen concentration in the soil moisture leaving the root zone (the leaching flux in SMB) relates to the effect from which the ecosystem needs to be protected (see Mapping Manual). The EU-DB applies 0.2 and 0.4 gN m^{-3} for coniferous and deciduous forest, respectively, and 0.3 gN m^{-3} for mixed forests, applying the precautionary principal by using the lowest values suggested in the Manual. Figure 2.11 indicates these values with vertical black lines. The distributions of the limits of the countries are higher in most of the area and vary widely. The limit in Belgium is higher than 100 gN m^{-3} for all of their ecosystems.

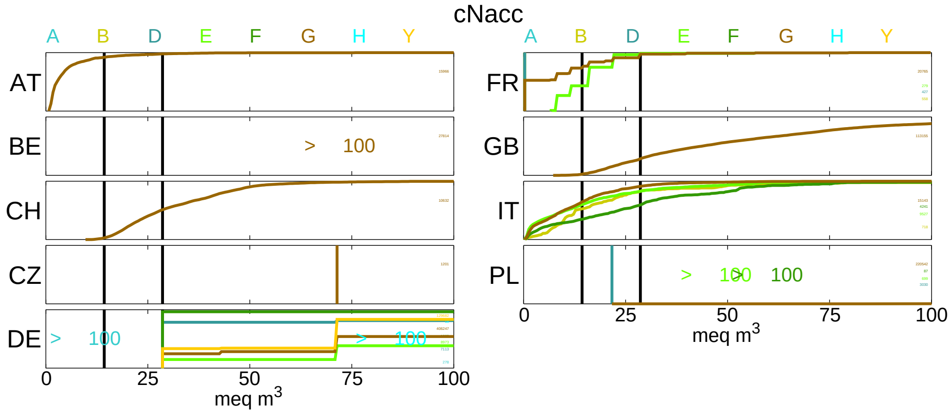

Figure 2.11 Acceptable nitrogen concentration in the leaching flux. The vertical black lines are at 0.2 and 0.4 gN m^{-3} used in the EU-DB.

The long-term net immobilization is 0.5 kgN ha^{-1} yr^{-1}, according to the Manual. Only French coastal ecosystems (B) and some freshwater bodies in Switzerland (CH) and the United Kingdom (GB) are below this limit.

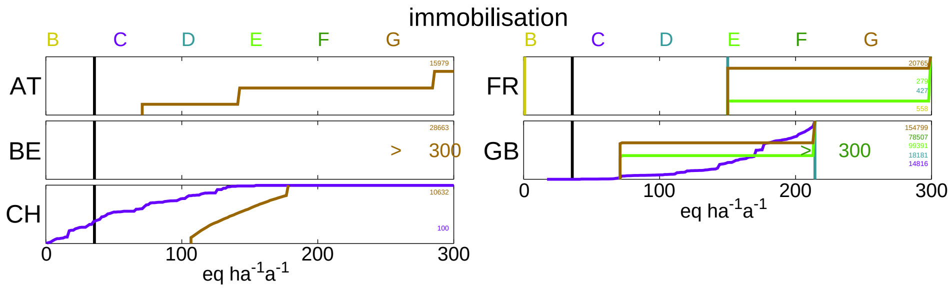

Figure 2.12 Net immobilization of nitrogen as submitted by the countries with an additional black vertical line at 0.5 kgN ha^{-1} yr^{-1}

The European background database contains data and information regarding these variables following the recommendations documented in the Mapping Manual. Because most countries assume uptake from ecosystems other than forests, and apply higher limiting concentration and/or assume higher net immobilization, the resulting critical loads have significantly higher values.

2.11 Other variables

Other variables don't differ much between the country data and the European background database. Figure 2.13, for instance, shows minor differences in the distribution of clay content, which can also be explained by the accuracy of the data; e.g. only distinct classes are used in the EU-DB. In the German submission, the clay content varies clearly between the ecosystem types; wetlands (D) contain very little clay.

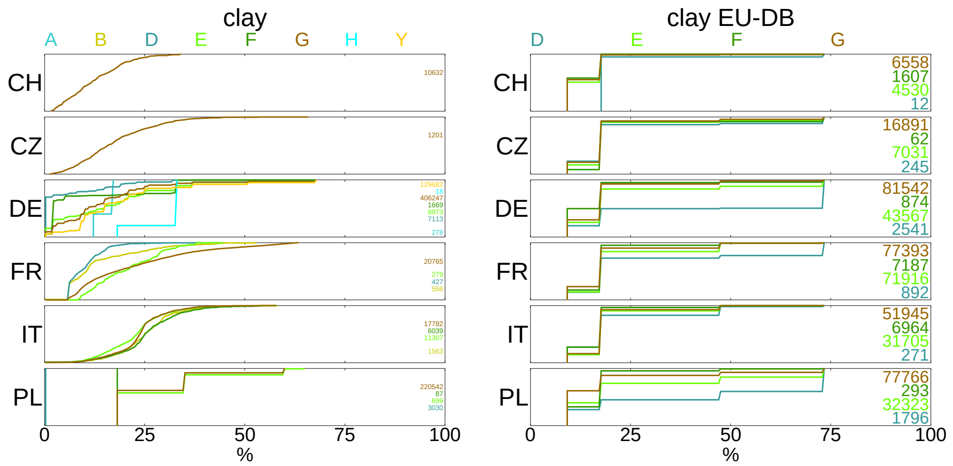

Figure 2.13 Clay content (%) as submitted by countries compared with the European background database.

The slope and aspect (measured counter-clockwise starting from North) of a site influences the local vegetation. Figure 2.14 shows that most ecosystems have a slope lower than 20%, but Switzerland (CH) in particular chooses many ecosystems with steeper slopes as receptor. The right side of the figure shows that Italian south slopes are more frequently of interest as receptor than northern slopes.

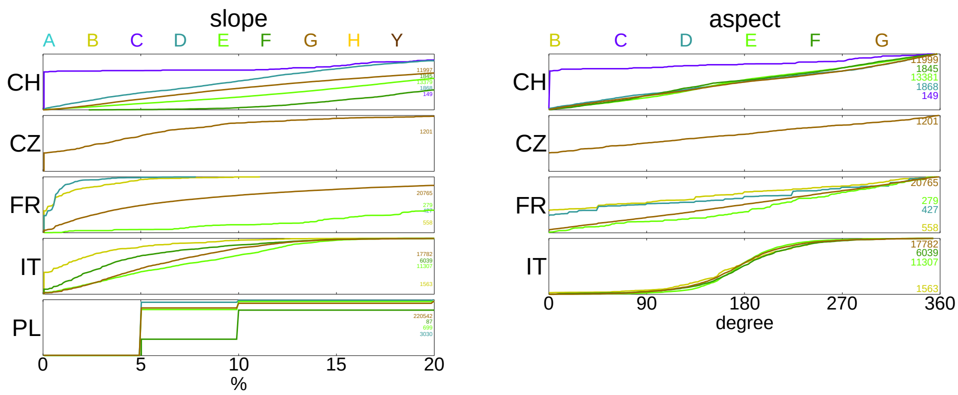

Figure 2.14 Slope and aspect of submitted receptors.

Notice that not all countries that submitted critical loads also submitted the additional variables, as asked for in the call.

2.12 Conclusions

Thirteen (13) countries responded to the 2014/15 Call for Data. The most striking changes since the 2011/2012 submissions are found in Germany and the coastal regions of France and in Switzerland for the critical loads for acidification. The novel but tentative, regional application of critical loads for biodiversity led to results submitted by Germany and the United Kingdom. Although the Task Force on Modelling & Mapping recommended the common habitat suitability index as a measure for comparison, the way it was used by the three teams (DE, GB and EU-DB) still requires further harmonization.
Empirical critical loads are well-established because effects of their exceedances have been associated with documented findings in the field

(Bobbink and Hettelingh 2011). Therefore, the **critical load for eutrophication** (CLeutN) is introduced as the minimum of empirical and modelled critical loads of nitrogen. The values for these two loads differ, but their distributions are comparable where both have been submitted; however, the modelled critical loads of nitrogen in the European background database are significantly lower than national submissions. The combined dataset of critical loads of acidification and eutrophication has been approved by the convention for use in integrated assessment (see Chapter 1).

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Annex 2A. Ecosystem areas in the European Critical Loads Database

Table 2A.1 Ecosystem areas in the European database from (N)ational data and the (E)uropean background database.

Country	**Acidification**		**Biodiversity**		**Empirical**		**Modelled Nut**	
EUNIS	*N*	*E*	*N*	*E*	*N*	*E*	*N*	*E*
AL								
G		6,910		6,545		6,910		6,910
D		15						15
E		6,876		6,664		4,837		6,876
F		4,373		976		978		4,373
AT								
G	38,957			1,412	38,380		39,332	
C					0			
D				22	135			
E				9,778	7,278			
F				2,314	3,823			
X					1			
BA								
G		21,005		17,402		21,005		21,005
D		40						40
E		9,978		6,611		7,825		9,978
F		2,930		1,725		1,725		2,930
BE								
G	5,447			3,061			5,530	
D				45	58			
E				5,658	6			
F				122	53			
BG								
G		35,702		29,992		35,702		35,702
D		94		13		13		94
E		13,494		13,247		8,366		13,494
F		1,763		1,748		1,748		1,763
BY								
G		58,272		16,389		58,272		58,272
D		2,726						2,726
E		3,665		3,665		3,665		3,665
F		90		84		90		90
CH								
G	9,648			9089	849		9,648	
C	86				42			
D					1,382			
E				3,026	10,594			
F				1	1,666			
CY								
G		654				654		654
E		408				86		408
F		575						575
CZ								
G	6,973			18,700	6,973		6,973	
D				32				

Country	**Acidification**		**Biodiversity**		**Empirical**		**Modelled Nut**	
EUNIS	*N*	*E*	*N*	*E*	*N*	*E*	*N*	*E*
E				7,256				
F				25				
DE								
G	81,200		81,200		70,253		81,200	
D	1,108		1,108		967		1,108	
E	1,410		1,410		1,338		1,410	
F	292		292		292		292	
A	34		34		21		34	
H	3		3				3	
Y	20,758		20,758				20,758	
DK								
G		2,651		1,735		2,651		2,651
D		352		176		176		352
E		1,183		1,157		722		1,183
F		398		381		398		398
EE								
G		19,755		3,897		19,755		19,755
D		1,214		772		772		1,214
E		6,356		6,339		2,993		6,356
F		96		93		96		96
ES								
G		85,817		32,753		85,749		85,817
D		552		5		5		552
E		93,117		91,718		43,532		93,117
F		55,645		8,164		8,207		55,645
FI								
G		170,283		78,190	17,340			
C					6,645			
D		18,851		18,747	10,160			
E		37,378		37,342	0			
F		9,352		9,229	6,859			
B					11			
A					125			
FR								
G	170,470			119,249	170,467		170,470	
D	5,123			41	5,123		5,123	
E	1,568			157,924	1,568		1,568	
F				2,710				
B	2695						2,695	
GB								
G	19,701				4,093		15,790	
C	7,661							
D	5,391				5,513			
E	20,002		7		21,891			
F	24,663		6		24,780			
B					323			
A					421			
GR								
G		21,972		18,227		21,972		21,972
D		204						204

Country	**Acidification**		**Biodiversity**		**Empirical**		**Modelled Nut**	
EUNIS	*N*	*E*	*N*	*E*	*N*	*E*	*N*	*E*
E		27,320		25,886		10,308		27,320
F		18765		5		5		18765
HR								
G		19,083		16,572		19,083		19,083
D		142						142
E		13,274		12,267		8,977		13,274
F		1,997		883		890		1,997
HU								
G		15,955		14,772		15,955		15,955
D		816		121		121		816
E		9,892		9,287		8,349		9,892
F		16		16		16		16
IE								
G		2,233		2,078		2,233		2,233
D		10,879		10,712		10,756		10,879
E		42,776		42,689		40,028		42,776
F		535		527		535		535
IS								
D		8,471				8,471		8,471
E		11				11		11
F		53,595				53,595		53,595
IT								
G	67,793			27,074			71,320	
E	22,178			26,570			22,585	
F	10,899			4,549			11,632	
B	527						410	
LT								
G		15,539		3,242		15,539		15,539
D		418		319		319		418
E		5,169		5,169		3,699		5,169
F		23		19		23		23
LU								
G		813		315		813		813
E		408		408		389		408
LV								
G		23,925		12,836		23,925		23,925
D		1,253		1,118		1,118		1,253
E		13,292		13,290		9,050		13,292
MD								
G		1,751		1,059		1,751		1,751
E		1,800		1,800		1,799		1,800
F		38		37		38		38
MK								
G		7,614		7,199		7,614		7,614
D		11						11
E		5,551		4,942		3,439		5,551
F		1,946		1,751		1,751		1,946
NL								
G	2,691			1,696			2,725	

Country	Acidification		Biodiversity		Empirical		Modelled Nut	
EUNIS	*N*	*E*	*N*	*E*	*N*	*E*	*N*	*E*
C							2	
D	132			71			191	
E	746			11,844			906	
F	343			313			358	
B	44						285	
A	7						69	
NO								
G				58,234	84,972			
C					18,965			
D				601	691			
E				6,323	8,817			
F				2,242	173,956			
H					3,946			
I					12,682			
Y	320,450							
PL								
G	93,309			60,568	93,793		93,793	
D	1,036			48	1,036		1,036	
E	330			24,731	330		330	
F	34			156	34		34	
PT								
G		20,479		16,205		20,479		20,479
D		9						9
E		12,400		12,357		4,123		12,400
F		4,122		2,332		2,334		4,122
RO								
G		69,733		59,975		69,733		69,733
D		10		9		9		10
E		31,576		25,677		29,063		31,576
F		3,325		3,325		3,325		3,325
RU								
G		660,688		341,147		660,688		660,688
D		4		4		4		4
E		140,342		136,722		140,332		140,342
F		19,388		19,167		19388		19,388
SE								
G				213,802				
C	395,226				395,226			
D				20,941				
E				31,390				
F				386				
SI								
G		10,618		6,364		10,618		10,618
D		23						23
E		2,562		1,835		2,310		2,562
F		304		303		303		304
SK								
G		19,099		15,902		19,099		19,099
D		37		3		3		37
E		3,660		1,135		3,454		3,660

Country	**Acidification**		**Biodiversity**		**Empirical**		**Modelled Nut**	
EUNIS	*N*	*E*	*N*	*E*	*N*	*E*	*N*	*E*
F		1,207		1,207		1,207		1,207
UA								
G		72,692		29,803		72,692		72,692
D		56						56
E		21,107		19,608		21,104		21,107
F		993		955		993		993

Part 2 Progress in Biodiversity Modelling

3 Critical Loads for Plant Species Diversity

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3.1 Introduction

In response to the request in 2007 of the Executive Body of the LRTAP Convention to the Working Group on Effects 'to consider further quantification of policy-relevant effect indicators such as biodiversity change, and to link them to integrated modelling work' (ECE/EB.AIR/91, paragraph 31), the members of the Task Force on Modelling and Mapping and the CCE embarked on developing methodologies and databases for deriving critical loads (CLs) and other indicators for plant species diversity. A vegetation model (the PROPS model) was developed at Alterra and used by the CCE to derive CLs for plant diversity on a European scale. This has been documented in Chapters 3 and 4 of the 2014 CCE Status Report (Slootweg et al. 2014). In this chapter, we present an update of the PROPS model and the CL derivation, which has been used to derive biodiversity CLs for Europe in response to the 2014/15 Call for Data (see also Chapters 2 and 4).

3.2 The PROPS model

The PROPS model has been described in the 2014 Status Report (Reinds et al. 2014; Posch et al. 2014) and updated in Chapter 4. Its basic principles and the functional shape of the probability functions are the same, but the number of explaining abiotic variables has changed from 4 to 5, with NO_3 concentration in soil replaced by soil C:N ratio and N deposition, representing a slow and fast changing N variable, respectively. The probability *p* of occurrence of a plant species is modelled as:

(3-1) $$p = \frac{1}{1+\exp(-z)}$$

where *z* is a quadratic polynomial:

(3-2) $$z = a_0 + \sum_{i=1}^{n} a_i \cdot x_i + \sum_{i=1}^{n}\sum_{j=1}^{n} a_{i,j} \cdot x_i \cdot x_j$$

with $a_{i,j} = a_{j,i}$ for all *i* and *j*. In the updated PROPS model (see Chapter 4), the number of (normalized/log-transformed) variables x_i is n = 5: pH, soil C:N, N deposition, annual average precipitation and temperature. The 21 coefficients for as many plant species possible are derived from relevés with both biotic and abiotic observations (see also Chapter 4 in this Report).

In Figure 3.1, isolines of occurrence probabilities for a single plant species (*Calluna vulgaris*) are shown in the pH-N_{dep} plane and the pH-C:N plane, respectively, keeping the other 3 parameters at constant values. The species shown has a clear preference for habitats with low pH, low N deposition and high C:N ratios. The white (vertical and horizontal) lines indicate the range of the respective variable for which the function (eq.3.1) is valid, i.e. for which there are observations; for values outside the range the value at the boundary is taken. This avoids an overestimation or underestimation of the probabilities for input values for which no observations were available.

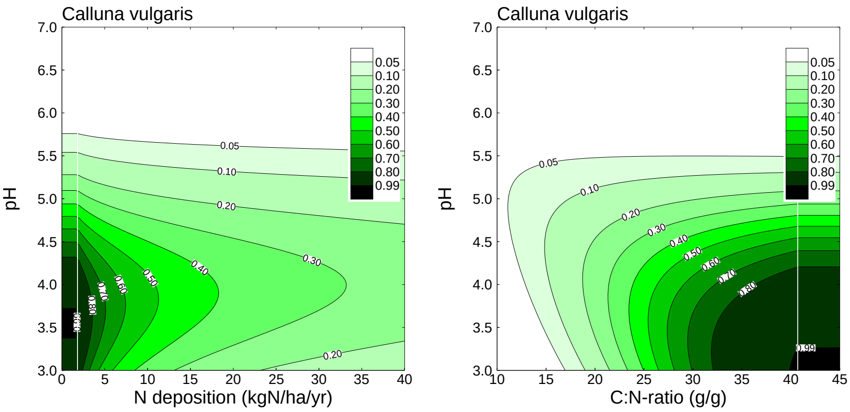


Figure 3.1 Isolines of normalized occurrence probabilities computed with the PROPS model as a function of the pH and N concentration (left) and pH and C:N (right) for Calluna vulgaris (P=700 mm/yr, T=7°C; left: C:N=22 g g^{-1}, right: N_{dep}=10 kgN $ha^{-1}yr^{-1}$).

3.3 The Habitat Suitability Index

At the 2014 CCE Workshop and ICP Modelling & Mapping Task Force meeting (Rome, 7–10 April), it was agreed that the Habitat Suitability index (HS index or HSI) should be used for the comparison of model results on a European scale. The HS index is defined as the arithmetic mean of the 'normalized' probabilities of occurrence of the species of interest. In mathematical form, this reads as:

$$(3.3) \quad HS = \frac{1}{K}\sum_{k=1}^{K}\frac{p_k}{p_{k,max}}$$

where K is the number of species, p_k the occurrence probability of species k, and $p_{k,max}$ the maximum occurrence probability of that species. For the sake of convenience, the HS index can be normalized to 1, i.e. divided by its maximum value. The species entering into the equation should be 'typical', 'characteristic' or 'desired' species for the respective habitat, the choice of species being the responsibility of each country.

For a given vegetation unit/habitat/ecosystem, the normalized probabilities of all typical/desired species are computed and the HS index is determined according to eq. 3.3. In Figure 3.2 (left), the isolines of the HS index in the N_{dep}–pH plane are shown for the habitat H6520 ("Mountain hay meadows"), consisting of 13 characteristic species for which data are available in the PROPS model. The figure shows that the maximum HS index would be achieved for a soil solution pH of around 6.4 and an N deposition of about 14 kgN $ha^{-1}yr^{-1}$.

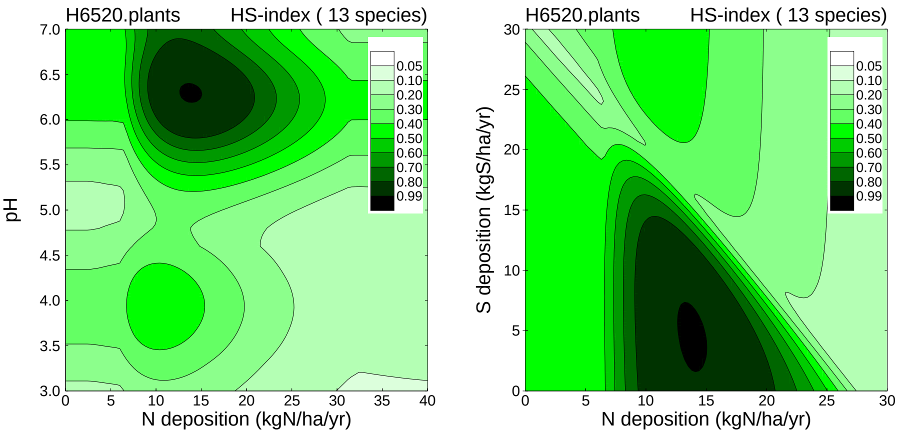


Figure 3.2 Isolines of the normalized Habitat Suitability Index for 'Mountain hay meadows' as a function of N deposition and pH (left) (PROPS model; C:N=22 g g^{-1}, T=7°C, P=700 mm yr^{-1}) and as a function of N and S deposition using the SMB model (with 'average' parameters) to link pH and S_{dep} (right).

To be used in emission reduction assessments, the PROPS variables have to be converted into N and S depositions. Nitrogen deposition is already a PROPS variable and the relation to S deposition can be conveniently derived with, e.g., the SMB model. The leaching flux of N, N_{le}, is derived as:

(3.4) $$N_{le} = (1 - f_{de})(N_{dep} - N_i - N_u)$$

where N_i and N_u are the long-term average immobilization and net uptake (removal) of N, and f_{de} the is denitrification fraction (see ICP M&M 2014). The S deposition, S_{dep}, is obtained by using $[H^+]$ (from *pH*) to compute the ANC leaching, ANC_{le}, and from the charge balance (fluxes in moles of charge):

(3.5) $$S_{dep} = BC_{le} - Cl_{le} - ANC_{le} - N_{le}$$

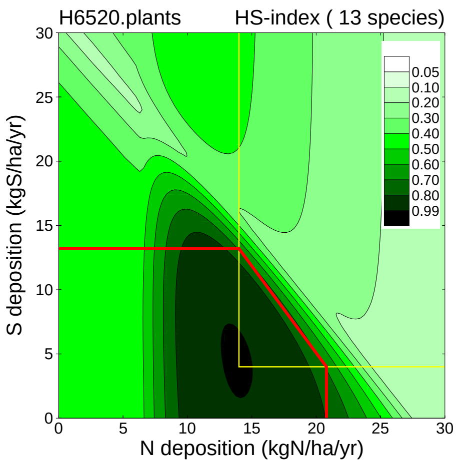
where BC_{le} and Cl_{le} are the leaching fluxes of base cations and chloride, respectively.

In Figure 3.2 (right), isolines of the HS index for 'Mountain hay meadows' (characterized by 13 typical plant species) are displayed as a function of N and S deposition, computed with the SMB model from the data displayed in Figure 3.2 (left) using 'average' site parameters (N_i+N_u=0). It shows that, in this case, the HS index is maximal for an S deposition of about 5 kgS $ha^{-1}yr^{-1}$ (and N_{dep}≈14 kgN ha^{-1} yr^{-1}).

3.4 Deriving critical loads

A possible derivation of critical loads of N and S (actually, a critical load function) from the HSI data is illustrated in Figure 3.3 (left): starting from the optimal point (N_{opt}, S_{opt}), i.e. the value of N and S deposition for which the HSI of a habitat is maximal, one proceeds (for constant

S_{dep}=S_{opt}, horizontal yellow line in Figure 3.3) along increasing N deposition until a pre-defined fraction of the optimal HSI is reached (here: 80% of the optimum). In the same fashion, a point along the line N_{dep}=N_{opt} (vertical yellow line) is reached for 80% of the HSI. These points define a trapezoidal function in the N_{dep}-S_{dep} plane (shown in red), called the N-S critical load function (CLF) of biodiversity. It is defined by 2 points (4 values; see right panel of Figure 3.3): (CLN_{min},CLS_{max}) and (CLN_{max},CLS_{min}). Note that, as a consequence of the way we constructed the CLF, we have CLN_{min}=N_{opt} and CLS_{min}=S_{opt}). Note the similarity of CLF with the critical load function of acidity that has been used for the support of European air pollution abatement policies (Posch et al. 2015).



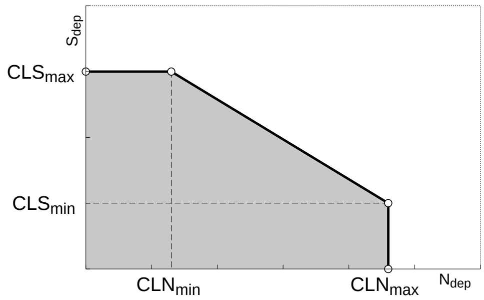


Figure 3.3 Left: A nitrogen-sulphur critical load function (N-S CLF) derived from the chosen HS index limit value (80% of maximum) (red line).
Right: The N-S critical load function defined by two points (four values): (CLN_{min},CLS_{max}) and (CLN_{max},CLS_{min}).

In Chapter 3 of the 2014 CCE Status Report (Posch et al. 2014), we outlined a method for deriving the CLF from isolines of the HSI. In fact, this might be a superior method to the one outlined above, but it requires routines for computing isolines and, moreover, in large-scale applications (see below), it is quite CPU-time-consuming. Therefore, the above method was used to compute CLFs on a European scale for the Call 2014/15.

3.5 The European Background Database for biodiversity critical loads

The above methodology has been used to derive critical loads for biodiversity on a European scale. The BioScore maps showing the probability of habitats over Europe (see Chapter 4 for details) were overlaid with the data layers of the European Background Database, providing for every 'site' the most probable habitat and the abiotic variables (soil characteristics, climate) needed for the SMB model. The PROPS model was applied to the characteristic plant species for every habitat at a given location, the HSI computed for 50×50 N_{dep}-S_{dep} combinations and the critical load function determined as described above. This results in approximately 1.3 million records ('sites' with a CLF) covering about 2.4 million km^2.

In Figure 3.4, the 5th and 50th percentiles (median) in the 0.50°×0.25° grid cells covering Europe of the biodiversity CLN_{max} are displayed (see Figure 3.3 for the definition of the CL quantities). In large parts of Europe (with the exception of northern Europe), the 5th percentile lies in the range of 400–1,000 eq ha^{-1}yr^{-1}, whereas the median values exceed 1,500 eq ha^{-1}yr^{-1} in most of Europe. Figure 3.5 shows the same percentile maps for the biodiversity CLS_{max} sulphur critical load. While the 5th percentile shows rather low values in large parts of Europe (except in the Nordic countries), the median values are mostly above 1,500 eq ha^{-1}yr^{-1}.

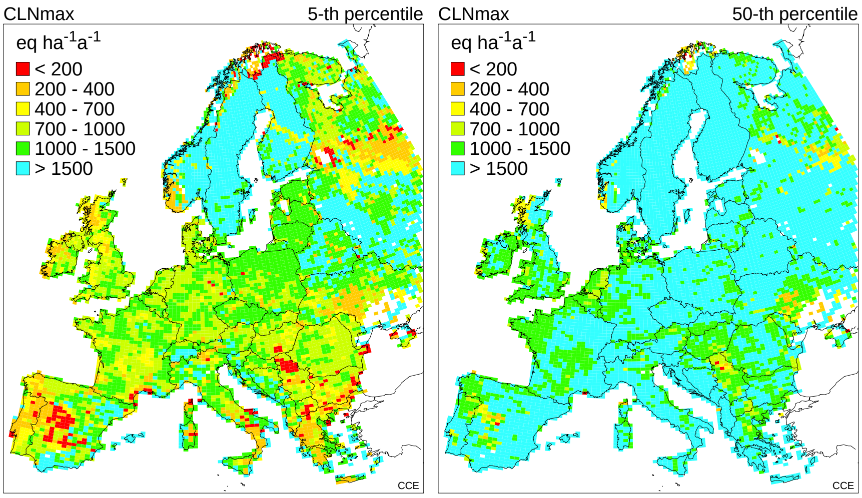


Figure 3.4 Left: 5th percentile (left) and 50th percentile (median; right) of the biodiversity CLN_{max} on the 0.50°×0.25° grid over Europe (see Fig. 3.3).

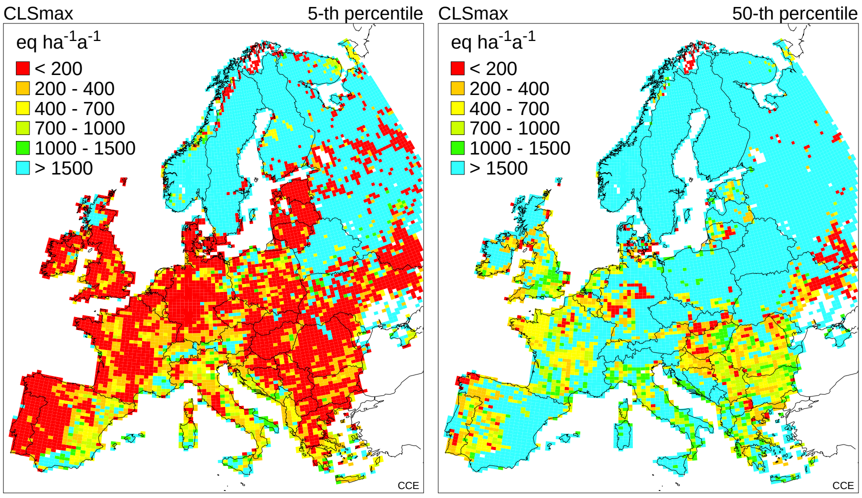


Figure 3.5 Left: 5th percentile (left) and 50th percentile (median; right) of the biodiversity CLS_{max} on the 0.50°×0.25° grid over Europe (see Fig. 3.3).

The work on critical loads for biodiversity is by no means finalized. For example, the following points need some more consideration:

1. Which maximum probability in the computation of the HSI (eq.3.3) should be used: The overall (5-dimensional) maximum probability that for a given site certainly includes some values that are never achievable for, say, the climatic parameters? Or a

more 'realistic' maximum probability, keeping the climate variables at the site quantities, for instance? Maybe taking into account future climate change?

2. Obviously, the limit value for deriving the CL function (here: 80% of the maximum) is arbitrary – How can this be made more 'objective'? Also, the way the CLF is derived (see Figure 3.3) might be improved. Maybe using isolines (as suggested earlier) would be more suitable, albeit technically more demanding. And, given the highly variable shapes of the HSI-isolines, maybe the CLF should be allowed a more general shape?
3. Integrated assessment modellers expressed concern when communicating the meaning of the HSI and the CLs derived from it. They expressed a preference for a quantity related to 'species number' as a function of deposition.

All these issues are currently being investigated and therefore the biodiversity critical loads cannot yet be used for integrated assessment under the LRTAP Convention.

3.6 Exceedances of the critical loads of biodiversity

The European database of critical loads for biodiversity was also used to analyse their exceedances for scenarios developed for the European Seventh Framework Programme "Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems" (ECLAIRE).

From the nine deposition scenarios that have been developed under ECLAIRE, four are used here: two consisting of measures to abate air pollution (by 2010 and 2050, respectively), one reflecting de-carbonization of the economy (counter-acting climate change) and the fourth including 'all' those control measures: "Current Legislation in 2010" (CLE2010), "Current Legislation in 2050 (CLE2050), "Decarbonization in 2050" (DECARB), and Maximum Control Effort in 2050" (MCE2050). The depositions of total N and S for the two 'extreme' scenarios, CLE-2010 (the 'present' situation) and MCE-2050 (the most stringent abatement by 2050), are shown in Figure 3.6. The maps show that (i) as early as 2010, N is the dominant pollutant in most of Europe, (ii) both pollutants will be considerably reduced by 2050 under MCE, and (iii) N will remain a substantial problem in 2050.

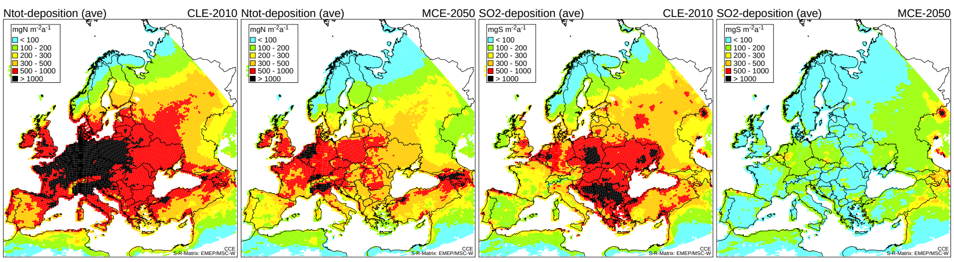


Figure 3.6 Deposition of grid-average total nitrogen (mgN $m^{-2}yr^{-1}$; left 2 maps) and sulphur (mgS $m^{-2}yr^{-1}$; right 2 maps) under Current Legislation (CLE) in 2010 and Maximum Control Effort (MCE) in 2050.

The depositions described above are used to compute the exceedance (expressed as average accumulated exceedance, AAE (see e.g. Posch et al. 2015), of the critical loads for biodiversity. Exceedances above 200 eq $ha^{-1}yr^{-1}$ cover considerable parts of Europe under CLE-2010 and peaks (400–1,200 eq $ha^{-1}yr^{-1}$) persist for ecosystems in the Netherlands, western parts of Germany, Belgium and northern Italy, both under the DECARB-2050 and MCE-2050 scenarios (Figure 3.7).

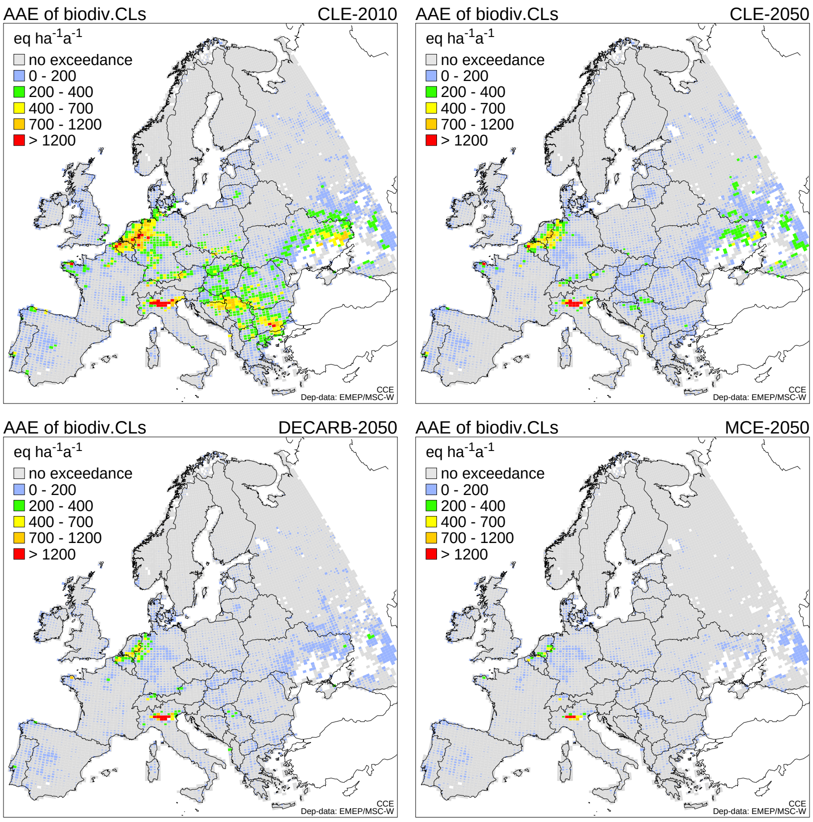


Figure 3.7 The exceedance (AAE) of critical loads for biodiversity under ECLAIRE scenarios CLE-2010 (top left), CLE-2050 (top right), DECARB-2050 (bottom left) and MCE-2050 (bottom right). Note: The size of the grid shading reflects the ecosystem area exceeded.

3.7 Robustness analysis of exceedances of ECLAIRE scenarios for N-S critical loads

Both the critical loads for biodiversity (CLbio) and the acidity critical loads (CLaci) depend on N and S depositions, i.e. they are characterized by a critical load function. It is therefore of interest to compare these two N-S CLs, and respectively their exceedances. While the exceedances of CLbio for four ECLAIRE scenarios are shown in Figure 3.7, the corresponding exceedances of CLaci, taken from the European background database for all countries (see Chapter 2), are shown in Figure 3.8. And in Table 3.1, the exceeded areas under the four scenarios are listed for the sake of comparison.

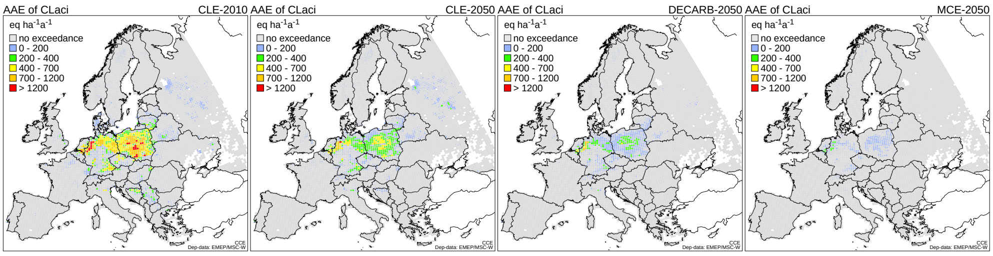


Figure 3.8 Exceedances (AAE) of critical loads of acidification (from the EU-DB) under ECLAIRE scenarios CLE-2010 (left), CLE-2050 (centre left), DECARB-2050 (centre right) and MCE-2050 (right). Note: The size of the grid shading reflects the ecosystem area exceeded.

Table 3.1 European ecosystem area (in %) where CLbio and CLaci (both from the EU-DB) are exceeded under ECLAIRE scenarios CLE-2010, CLE2050, DECARB-2050 and MCE-2050

Exceedance of:	**CLE** *2010*	**CLE** *2050*	**DECARB** *2050*	**MCE** *2050*
CLbio	18	12	8	4
CLaci	8	6	3	1

The robustness of exceedances (Hettelingh et al. 2015) was derived by analogy to the way in which uncertainties are addressed in the Fourth Assessment Report of the IPCC, as described in IPCC (2005). According to this logic, the robustness of an assessment that ecosystems are at risk can range on a scale from ‘exceptionally unlikely’ to ‘virtually certain’. In this chapter, the robustness analysis of ecosystem impacts of the four ECLAIRE scenarios is based on the analysis of the location and magnitudes of exceedances for the CLaci and CLbio critical load functions.

The method is based on the analysis of the location, coverage and the magnitudes of exceedances of CLaci and CLbio, following the principle of ensemble assessment (Hettelingh et al. 2015), whereby an exceedance is more likely when it occurs using different methods (here: two types of critical load functions). Here, the robustness analysis focuses on the question of whether the combination of sulphur and nitrogen deposition causes scenario-specific exceedances to point in the same direction. The consideration of different endpoints (soil chemistry for CLaci and plant species diversity for CLbio) leads to two sets of critical load functions. This leaves an ecosystem area with the following possibilities of being at risk of atmospheric deposition of sulphur and nitrogen:

- None of the critical loads are exceeded (i.e. exceedance is ‘unlikely’).
- Exactly one of the critical load functions is exceeded (i.e. exceedance is ‘as likely as not’).
- Both critical load functions are exceeded:
  a. with a likelihood in the interval (0, 0.33] ; i.e. the ecosystem is ‘likely’ to be at risk;
  b. with a likelihood in the interval (0.33, 0.67]; i.e. the ecosystem is ‘very likely’ to be at risk;
  c. with a likelihood > 0.67; the ecosystem is ‘virtually certain’ to be at risk.

- The likelihood of an exceedance in a grid cell is defined as the square root of the product of the percentages of exceeded ecosystem areas (i.e. their geometric mean) with respect to CLaci and CLbio (Hettelingh et al. 2015).

Following this approach, it can be concluded that the likelihood of exceedances under CLE-2010 varies between 'as likely as not' (green shading) in many parts of Europe and 'virtually certain' (red shading) in broad areas in Central-Western Europe, in particular (Figure 3.9, left). Note that the 2010 exceedance in Central Europe was driven mostly by the exceedance of CLaci (see Figure 3.8, left), especially in Poland, while high exceedances of CLaci *and* CLbio concentrated at the Dutch-German border made the likelihood of exceedances 'virtually certain'.

Under the ECLAIRE scenario MCE-2050, a different picture emerges (Figure 3.9, right), as exceedances are 'as likely as not' (green) in most European countries, with 'likely', 'very likely' and 'virtually certain' (yellow to red) exceedances scattered over Poland, northern and southern Germany, Switzerland, and on the Dutch-German border area, in particular. From the comparison between the AAE of CLaci under MCE-2050 (Figure 3.8, right) and the AAE of CLbio (Figure 3.7, bottom right), it is noted that the robustness of the estimated European area at risk seems driven mostly by CLbio exceedances.

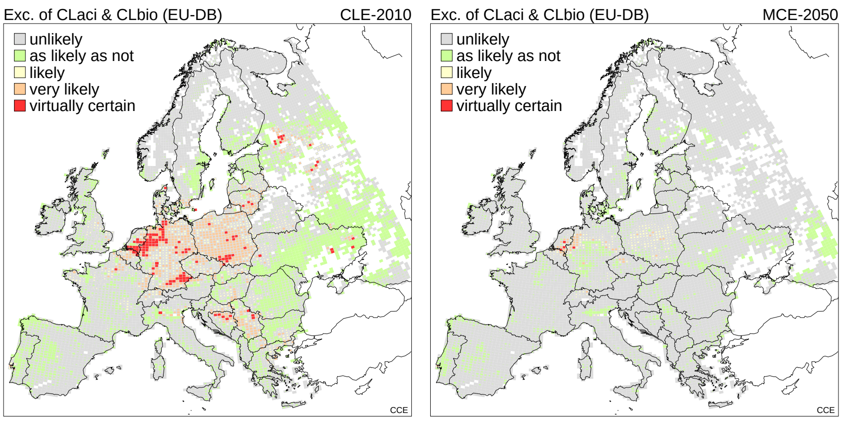


Figure 3.9 The likelihood of a positive exceedance (AAE) under CLE-2010 (left) and MCE-2050 (right), i.e. that the respective grid cell contains at least one ecosystem of which CLaci and/or CLbio is exceeded. Note: The size of the grid shading reflects the ecosystem area exceeded.

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4 Probability of Plant Species (PROPS) model: Latest Developments[2]

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The PROPS model was developed to enable assessments of changes in plant species diversity caused by changes in air pollution and climate, and to compute critical loads of sulphur and nitrogen for biodiversity. This model computes the probabilities of the occurrence of ground vegetation species as a function of abiotic conditions, such as pH and C:N ratio in the soil and climatic variables such as temperature and precipitation. The model also computes an aggregate indicator for species occurrence in a habitat, i.e. the Habitat Suitability Index (HS index, see Chapter 3) for use in the computation of the critical load for biodiversity (see Chapter 3). When linked to a model (e.g. VSD+) that simulates abiotic conditions such as pH and C:N ratio as a function of the deposition of acidifying and eutrophying air pollutants (sulphur (S) and nitrogen (N) compounds), the PROPS model aims to be useful for the support of policies in the field of air pollution abatement and nature protection.

In this chapter, we first summarize the basics of the PROPS modelling followed by an overview of the latest developments with emphasis on a further analysis of PROPS results, species selection for habitats and European mapping of the HS index.

4.1 The PROPS model

The PROPS model estimates the occurrence probability of plant species as a function of soil chemistry and climate, using a logistic regression technique (Ter Braak and Looman 1986). In this technique, the occurrence probability of a plant is estimated based on presence-absence data (Figure 4.1; every dot with y-value equal to 1 indicates that the plant species is present (for parameter x), when the species is not present, the value is 0).

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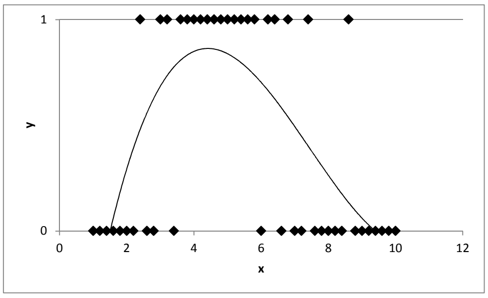

Figure 4.1 Example of occurrences of plant species against an abiotic parameter x. When the value is 1, the species occurs and for value 0, it doesn't (from Reinds et al. 2012).

If *p* is the probability of a species to occur, then the odds that a species does occur is *p*/(1–*p*). The logit of *p*, defined as the log of the odds, varies between –∞ and ∞, and is approximated (fitted) by a quadratic polynomial:

(4.1) $$z = \mathrm{logit}(p) = \log\frac{p}{1-p} = a_0 + \sum_{i=1}^{n} a_i \cdot x_i + \sum_{i=1}^{n}\sum_{j=1}^{n} a_{i,j} \cdot x_i \cdot x_j$$

with $a_{i,j} = a_{j,i}$ for all i and j, and n is the number of abiotic variables. The (normalized) variables x_i include precipitation (P), temperature (T), N related variables (e.g. soil N concentration [N], carbon to nitrogen ratio in the soil (C:N ratio), N deposition (Ndep)), all log-transformed, and soil pH. The explanatory variables need to be normalized:

(4.2) $$x_{norm} = \frac{x - x_{mean}}{x_{std}}$$

where *x* is the log-transformed value of the explanatory variable, x_{mean} is the average value and x_{std} the standard deviation of the explanatory variable from the database that is used to fit the model. From eq.4.1 the probability *p* is then obtained as:

(4.3) $$p = \frac{1}{1 + \exp(-z)}$$

Using the probabilities p_k of species *k* (*k*=1,…,*K*), the HS index is defined as:

(4.4) $$HS = \frac{1}{K}\sum_{k=1}^{K}\frac{p_k}{p_{k,\max}}$$

where $p_{k,max}$ is the maximum occurrence probability of that species.

4.2 Recent developments of the PROPS model

Over the past year, work on the PROPS model consisted of:

- Introducing a new set of explanatory variables.
- Further analyses of model results.

4.2.1 New set of explanatory variables:

In PROPS, effects of nitrogen (N) on species occurrence probability were modelled using N concentration and C:N ratio as explanatory variables. The C:N ratio represents the long-term effect of N enrichment on species, whereas N concentration is a proxy for short-term influences. The number of N concentration measurements in the dataset (held at Alterra) is, however, very limited: for only 2,330 of the 12,300 plots are measurements available. Furthermore, these measurements are confined to Western Europe (mostly the Netherlands, Ireland and the UK) where N deposition and N concentrations are relatively high. Consequently, the species responses tended to be biased towards frequent occurrences at high N concentrations. The latest PROPS model, therefore, now uses N deposition instead of N concentration to incorporate the short-term N effect on species occurrence probability. Values for N deposition used to fit the response curves were derived from a European data set by Schöpp et al. (2003), using vegetation-specific (forest/short vegetation) values. An average value for total (NO_x+NH_3) N deposition was computed over the period ranging from two years before to two years after the vegetation at the plot was recorded.

4.2.2 Further analysis of the model results:

To assess the impact of N and S deposition on habitat types with PROPS, the overall response of the species in the habitat to abiotic variables was needed. This was achieved by constructing e.g. isolines of equal probability occurrence of this set of species as a function of pH and N deposition at given C:N and climate. Reviewing these responses, it turned out that, for a number of habitats, the response of occurrence probabilities (expressed by the HS index, see eq.4.4) to pH and N deposition can show multiple optima, one at very high and one at very low pH when we fix the C:N ratio at a low value and temperature (T) and precipitation (P) at average values (Figure 4.2, left). This pattern changes when we substitute a high value for the (fixed) C:N ratio: the pronounced optima at low pH is (almost) absent (Figure 4.2, right).

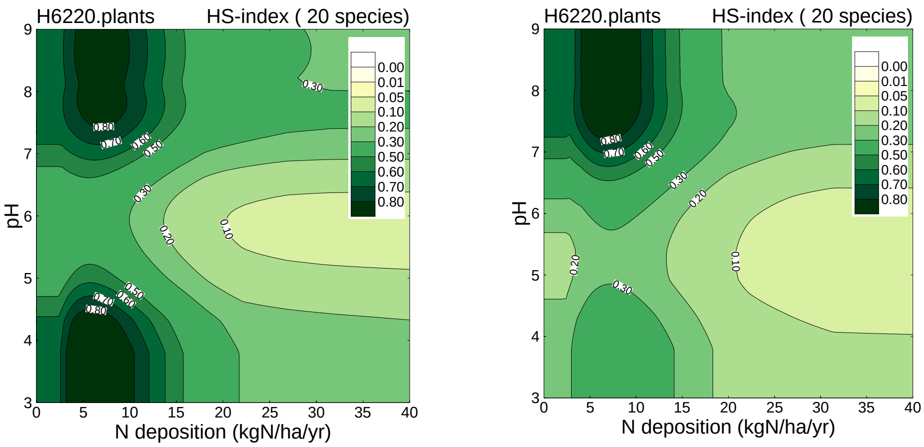

Figure 4.2 Isolines of HSI for Habitat 6220 ("Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea") using PROPS with a C:N ratio of 12 (left) and with a C:N ratio of 25 (right); temperature and precipitation set to average values for this habitat in Europe (12.5°C and 614 mm yr^{-1}).

The occurrence of two optima can be explained by looking at the response of individual species. If we fix C:N at a low value (e.g. 12 which is at the lower limit of observed C:N ratios in European forest soils (Vanmechelen et al. 1997)), Ndep, T and P at average values for this species and then compute occurrence probability in response to pH, the highest computed probability occurs at the lower limit of the pH range (pH~3, Figure 4.3 left), although the observations indicate the highest occurrence probability at approximately pH 6.5-7.0 and low occurrence at low pH. At higher C:N values, the curve shows a less pronounced effect of pH on probability, although it still increases with decreasing pH. If, however, we apply the same equation to each site in the database, but now use the C:N ratio, N deposition, temperature and precipitation *at the site* instead of a fixed value for all sites, then the function describes the observed probability very well (Figure 4.3, right).

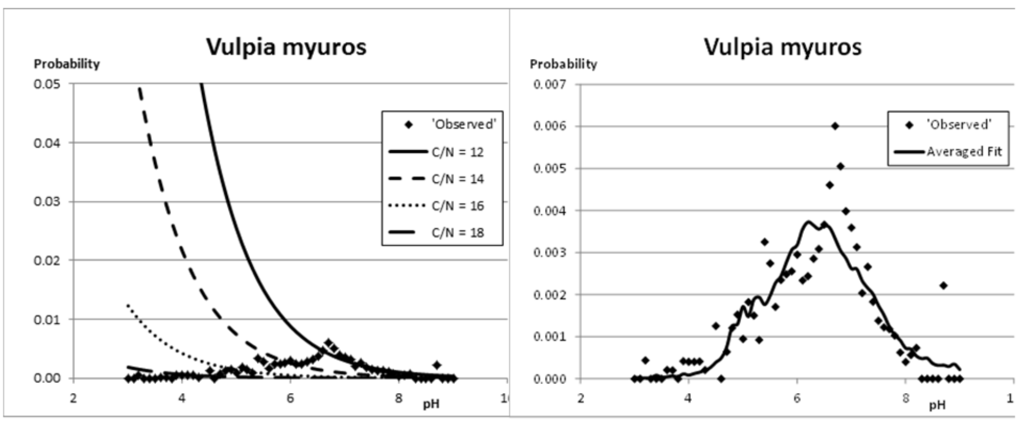

Figure 4.3 Observed and fitted response of occurrence probability as a function of pH for Vulpia Myuros using PROPS with a fixed low C:N ratio (left) and with plot-specific values for C:N, Ndep, T and P (right).

Obviously, pH and C:N ratio are correlated in the database and combining a low pH with a low C:N ratio in equation 4.1 leads to unrealistic results. Principally, the model should not be applied using combinations of inputs outside the domain of the data the model is based on. Figure 4.4 shows the percentage of plots for the ten possible combinations of all abiotic variables obtained from the database (about 400,000 relevés), showing that pH and C:N ratio are correlated and combinations of low pH (<4) and low C:N (< 15) are virtually absent (Figure 4.3, top left). Low C:N ratios do occur at varying ranges of Ndep, precipitation and temperature (Figure 4.3, second row); high C:N ratios (> 35) are sparse in the database and most observations are between C:N 10 and 20 with associated Ndep values between 10 and 20 $kg.ha^{-1}.yr^{-1}$. Furthermore, as expected, high Ndep values are mainly associated with temperature values typical for Western Europe (between 7 and 12 °C; Figure 4.3, third row).

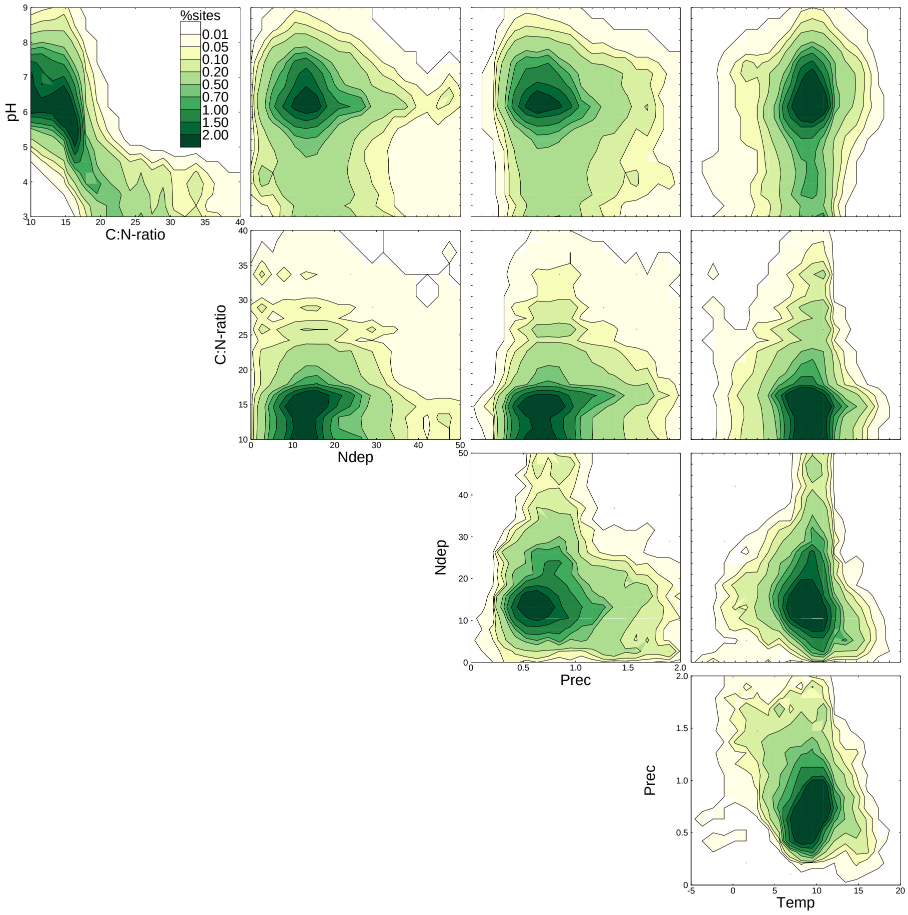

Figure 4.4 Percentage of plots in the PROPS database for combinations of abiotic variables (20×20 bins).

Further work is required to adapt the PROPS model in such a way that its application is confined to those areas in the 5-dimensional space (pH, N deposition, C:N ratio, T, P) such that the model is actually based on available data. Further refinement of the fitting procedure may also improve the PROPS model, as would measurements, especially from southern Europe.

4.3 Species selection and habitat mapping

For both local and regional applications in EUROPE, the PROPS model is applied to describe the impact of phenomena such as air pollution and climate change on combinations of soil type and EUNIS class or habitat type. Soil type is used to parameterize the soil model (e.g. VSD+), providing the abiotic parameters, whereas EUNIS class or habitat type can be used to define a set of relevant ground vegetation species. In previous years, this species selection was based on the Map of the Natural Vegetation of Europe (EVM; Bohn et al. 2000/2003), which provides lists of species that are characteristic for each mapping unit. Linking these mapping units to EUNIS classes provides the desired species list. This map, however, provides the *potential* natural vegetation, which means that it does not map the *actual* natural vegetation; semi-natural grasslands, for example, that are currently found in lowland areas in Europe are not (well-) represented on the map. For site applications, this does not pose a problem as one can manually select a mapping unit suited to a site. For regional applications, however, the map is less suitable: when overlaying the EUNIS map and the EVM map, there is very little correspondence between the mapping units in those countries in which the current vegetation deviates from the potential natural vegetation, which makes it difficult to assign region-specific species to EUNIS classes in a reliable manner.

This can be remedied by using results of the BioScore project (Van Hinsberg et al. 2014) in which sets of typical species are defined for 40 ANNEX1 habitat types covering most geo-biographical regions. Species selection in the Bioscore project was based on the Interpretation Manual of European Union habitats (EC 2013) and various literature sources. BioScore also provides detailed gridded maps with predicted habitat suitability across Europe, based on the relationship between habitat suitability and climate, soil type, land use and external drivers such as agricultural intensity and forest management type. For each grid cell, Figure 4.5 shows the habitat selected from the 40 available habitats that has the highest predicted habitat suitability.

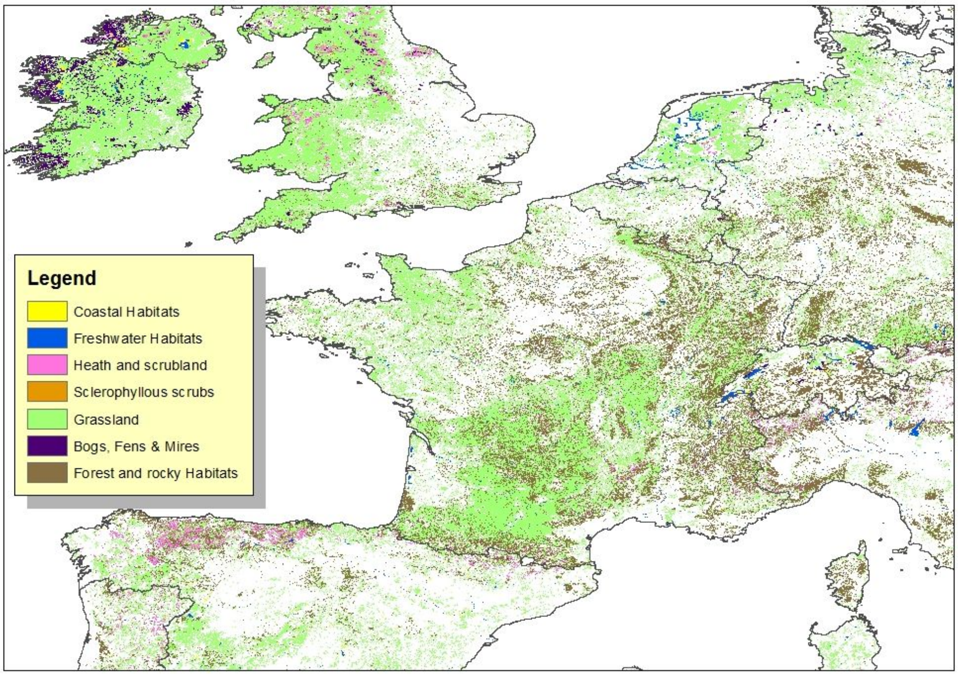

Figure 4.5 Habitat per grid cell with the highest predicted habitat suitability.

For regional assessments with PROPS, this detailed grid was combined with the EUNIS map for Europe to arrive at combinations of EUNIS and ANNEX 1 habitats. A translation table of habitat types to EUNIS classes was constructed based on expert judgement to enable the assignment of species lists to EUNIS classes. The set of combinations was then cleaned to eliminate implausible combinations of EUNIS and ANNEX 1 habitats caused by map inaccuracies. In the European mapping, all relevant habitat types were assigned to the EUNIS class based on the map overlay and the list of plausible combinations of habitat type and EUNIS class, using the computed habitat suitability as a weighing factor in the subsequent calculations.

4.4 Conclusion and further work

Based on a large data set with observed plant species occurrences and abiotic variables, PROPS response curves have been derived for about 4,000 European plant species. PROPS response curves for C:N were compared with results from Finland (Heikkinen and Mäkipää 2010) for about 40 species, showing similar responses. By combining responses for species that are typical for a habitat, biodiversity metrics such as the HS Index can be derived. Using a soil model (e.g. VSD+) results that describe the effects of reduced S and N deposition on soil chemistry and scenarios for climate change, PROPS can be applied to show the time-development in the HS index. Alternatively, PROPS results combined for a habitat can be used to obtain critical loads for S and N (see Chapter 3). Additional work would be needed to refine PROPS, mainly related to constraining the model application to combinations of abiotic variables present in its underlying database and including more data from countries in Southern Europe. Further developments should also include further investigations into how critical loads can best be derived from the combined PROPS curves and alternative metrics for biodiversity, such as the (relative) number of plants likely to occur under given abiotic conditions.

The PROPS model enables the incorporation of biodiversity issues into the effects-based support of European emission reduction policies. By its focus on habitats in areas of special protection (e.g. Natura 2000 areas), the PROPS model is well-suited for the support of European nature policies.

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Part 3 NFC Reports

This part brings together the reports by the National Focal Centres (NFCs) documenting their country’s submission of data and assessments in response to the CCE’s Call for Data, issued in 2014 (see also Appendix A).
In an annex to the Swedish national report, a report on a comparison of critical load methods for freshwaters in Norway and Sweden, produced together with the Norwegian NFC, is re-printed.
The Czech Republic did not substantiate their submission with a national report.
The reports have not been thoroughly edited, only minor corrections and harmonisations have been carried out. However, the responsibility for the substance of the National Reports remains with the National Focal Centres and *not* with the National Institute for Public Health and the Environment.

Austria

National Focal Centre	Collaborating Institutions
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Introduction

In response to the 2014/15 call for data the NFC for Austria provides updated critical load data on acidity (Simple Mass Balance, SMB) and eutrophication (SMB and empirical critical loads) in the new grid resolution of 0.10° Lon × 0.05° Lat. Protection status is reported in addition to the EUNIS habitat type. Biodiversity critical loads are not reported, but will be available in the year 2016 because they are in the focus of an ongoing Austrian research project (CCN-Adapt).

Method

For acidity and nutrient N critical loads the SMB was applied with the methods described in ICP Modelling & Mapping Manual (2014). Only forested areas were taken into account.

Nutrient critical loads were derived from climatic, and soil maps, Austrian-wide forest yield data in a resolution of 0.5°Lon x 0.25°Lat, and a detailed habitat map (Umweltbundesamt, 2015). Following the Swiss method (Posch et al., 2003), we used an altitude-dependent acceptable (critical) N leaching (Nle(acc)). Nle(acc) is linearly decreasing between 500 m with 4 kg N/ha/yr (= 285 eq/ha/yr) and 2000 m with 2 kg N/ha/yr (= 143 eq/ha/yr).

In order to calculate acidity critical loads we additionally used forest point data with soil profile information (FBVA, 1992). We calculated lgKAlox based on the percent organic matter (OM) of the soils:

$$lgKAlox = 9.8602 - 1.6755 * \log(OM) \qquad \text{for } 1.25 < OM < 100;$$

$$lgKAlox = 9.7 \qquad \text{for } OM <= 1.25$$

Base cation weathering was derived following De Vries et al. (1993) by using soil texture, parent material, and annual mean temperature. Base cation uptake was taken from the forest yield data.

The empirical critical loads were derived with the habitat map together with minimum critical load values given in Bobbink & Hettelingh (2011).

Minor adjustments of empirical values were done for some habitat types based on expert knowledge (Table AT.1).

Table AT.1 Empirical Critical Loads (kg N ha-1yr-1) for N effects in ecosystems. Minimum and maximum values as given in Bobbink & Hettelingh (2011), and adjustments

EUNIS	Name (EUNIS, own)	Min Cl_{emp}	Max Cl_{emp}	AT Cl_{emp}
C1.4	Permanent dystrophic lakes, ponds and pools	3	10	3
D1	Raised and blanket bogs	5	10	5
D1.1	Raised bogs	5	10	5
D1.2	Blanket bogs	5	10	5
D2	Valley mires, poor fens and transition mires	10	15	10
D2.3	Transition mires and quaking bogs	10	15	10
D3	Aapa, palsa and polygon mires	5	10	5
D4	Base-rich fens and calcareous spring mires	15	30	15
D4.1	Rich fens, including eutrophic tall-herb fens and calcareous flushes and soaks	15	30	15
D4.2	Basic mountain flushes and streamsides, with a rich arctic-montane flora	15	25	15
D4.22	Alpine riverine [Carex maritima] ([Carex incurva]) swards	15	25	15
	X04: Raised bog complexes			5
E1	Dry grasslands	15	25	15
E1.1	Inland sand and rock with open vegetation	15	25	15
E1.12	Euro-Siberian pioneer calcareous sand swards	15	25	15
E1.2	Perennial calcareous grassland and basic steppes	15	25	15
E1.22	Arid subcontinental steppic grassland ([Festucion valesiacae])	15	25	15
E1.23	Meso-xerophile subcontinental meadow-steppes ([Cirsio-Brachypodion])	15	25	15
E1.24	Central alpine arid grassland ([Stipo-Poion])	15	25	15
E1.26	Sub-Atlantic semi-dry calcareous grassland	15	25	15
E1.27	Sub-Atlantic very dry calcareous grassland	15	25	15
E1.29	[Festuca pallens] grassland	15	25	15
E1.2B	Serpentine steppes	15	25	15
E1.2C	Pannonic loess steppic grassland	15	25	15
E1.7	Closed non-Mediterranean dry acid and neutral grassland	10	15	10
E1.76	Dry sub-continental acid steppic grasslands	10	15	10
E1.831	Iberian montane [Nardus stricta] swards	15	25	15

E1.9	Open non-Mediterranean dry acid and neutral grassland, including inland dune grassland	10	15	15
E1.99	Pannonic inland dunes	10	15	10
E1.D	Unmanaged xeric grassland	10	15	10
	E1_calc: calcareous dry grassland	15	25	15
	E1_acid: acid and neutral dry grassland	10	15	10
E2.2	Low and medium altitude hay meadows	20	30	20
E2.3	Mountain hay meadows	10	20	20
E3.5	Moist or wet oligotrophic grassland	15	25	15
E4	Alpine and subalpine grasslands	5	10	5
E4.3	Acid alpine and subalpine grassland	5	10	5
F2	Arctic, alpine and subalpine scrub	5	15	5
F2.2	Evergreen alpine and subalpine heath and scrub	5	15	5
F2.222	Pyrenean rusty alpenrose heaths	5	15	5
F2.3	Subalpine deciduous scrub	5	15	15
F2.4	Conifer scrub close to the tree limit	5	15	5
F4.2	Dry heaths	10	20	10
G1	Broadleaved deciduous woodland	10	20	10
G1.6	[Fagus] woodland	10	20	10
G1.7	Thermophilous deciduous woodland	10	20	10
G1.8	Acidophilous [Quercus]-dominated woodland	10	15	10
G1.A	Meso- and eutrophic [Quercus], [Carpinus], [Fraxinus], [Acer], [Tilia], [Ulmus] and related woodland	15	20	15
G3	Coniferous woodland	5	15	10
G3.1	[Abies] and [Picea] woodland	10	15	10
G3.2	Alpine [Larix] - [Pinus cembra] woodland	5	15	10
G3.3	[Pinus uncinata] woodland	5	15	10
G3.4	[Pinus sylvestris] woodland south of the taiga	5	15	5
G3.5	[Pinus nigra] woodland	15	15	15
G3.D	Boreal bog conifer woodland			5
G3.E	Nemoral bog conifer woodland			5
G4	Mixed deciduous and coniferous woodland	10	20	10

Data sources
Habitat map: we used a new habitat map with a resolution of 100 x 100 m of entire Austria including EUNIS Level 2 and Level 3 habitats (Umweltbundesamt, 2015).
Loss of N and base cations via forest harvest: Forest yield data was derived for a 0.5° Lon x 0.25° Lat grid from the Austrian forest inventory. The data covers the period between 1981 and 2009. For each cell, biomass removal of conifers and deciduous trees was calculated, while element contents given in Jakobsen et al. (2003) were used to derive N and base cation loss per unit area. Additional to biomass removal, the per-cell area percentage of forests out of use, and thus without element loss, was calculated. The respective share of forest areas were distributed evenly among all forest types.

Base cation deposition: Deposition of base cations was taken from Van Loon et al. (2005) for the year 2000 and was provided by the Coordination Centre of Effects. The original 50 x 50 km resolution was statistically downscaled to the 0.10° Lon × 0.05° Lat grid with mapped precipitation data.
Forest soil plots: We used 496 soil profiles from the forest soil status inventory (FBVA 1992) as additional input for the SMB to calculate acidity critical loads. The soil data was collected between the years 1987–1991 in a rectangular 8.7 km grid over entire Austria.
Reported data sets
Critical loads of acidity (CLacid): CLmaxS, CLminN and CLmaxN as computed with the SMB model. Only forest sites with an area >0.01 km² are included
Critical loads of nutrient nitrogen (CLnut): also here the SMB was applied. Only forest sites with an area >0.01 km² are included
Empirical critical loads (CLemp): based on a habitat map and empirical values given in Bobbink & Hettelingh (2011). Only forest sites with an area >0.01 km² are included.
Instruction for the use of Critical Loads
In broader applications of the N critical loads by the CCE the following procedure should be applied. Since for the same 'ecord' different critical load methods were applied, a decision has to be made as to which to use. For Austria only for forests different methods have been applied. Therefore, for all but forests empirical critical loads for eutrophication effects (CLemp) should be used. For forests, mass balance critical loads (CLnut) should be used because the detail in EUNIS forest types was too coarse to differentiate sufficiently.

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Belgium (Wallonia)

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Regional Data Produced
Critical loads data have been produced for forests (coniferous, deciduous, mixed forests) and natural vegetation in Wallonia.

Mapping procedure Wallonia
From Walloon Land Cover Map, 27,344 forest ecosystems areas (>1 ha) were extracted and overlaid with thematic maps in order to calculate critical loads parameters. From Corine Land Cover 2005, four natural ecosystem types (representing 136 ecosystems area) were extracted and assigned to a theoretical value according to ecosystem type. Next, critical loads maps were overlaid with new EMEP grid (0.50° x 0.25°) in order to load CCE database as requested.

Calculation methods & results Wallonia

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):

$$CL_{max}(S) = BC_{we} + BC_{dep} - BC_u - ANC_{le(crit)}$$
$$CL_{max}(N) = N_i + N_u + CL_{max}(S)$$
$$CL_{nut}(N) = N_i + N_u + N_{le} + N_{de}$$
$$ANC_{le(crit)} = -Q_{le}\ ([Al^{3+}] + [H^+] - [RCOO^-])$$

Where:

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

$[H^+]$ = concentration of $[H^+]$ at critical pH (Table BE.2).

$[RCOO^-]$= 0.044 molc/molC x $DOC_{measured}$ (Table BE.2)

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.1) 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.1 and BE.2 summarise the values given to some of the parameters.

Table BE.1 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.2. Constants used in critical load calculations in Wallonia

Parameter	Value
N_i	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
$N_{le\ (acc)}$	2.5 mg N L^{-1} for coniferous forest
	3,5 mg N L^{-1} for deciduous forest
	3 mg N L^{-1} for mixed forest
N_{de}	Fraction of (N_{dep} – Ni – Nu)

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

In *Wallonia*, the **base cation weathering rates** (BC_{we}) were estimated for 10 different representative soil types (table BE-3) 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 BC_{we}, a acid input was fixed at 900 $eqH^{+}\ ha^{-1}\ yr^{-1}$ in order to keep a long term balance of base content in soils.

$$\mathbf{N_{le} = Q_{le} \ \ cN_{(acc)}}$$

The **flux of drainage water leaching(Q_{le})** from the soil layer (entire rooting depth) was estimated from EPICgrid model (Faculté Universitaire des Sciences Agronomiques de Gembloux). The results of the EPICgrid model are illustrated in Fig. BE.1.

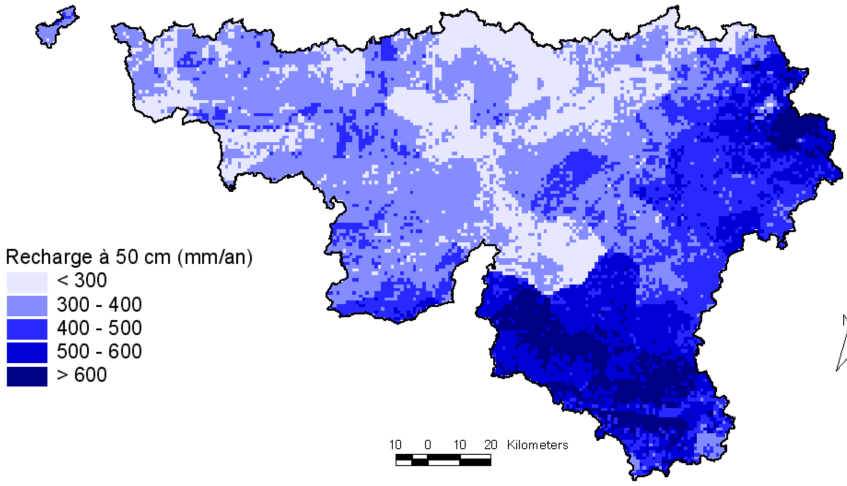

Figure BE.1 Flux of drainage at 50 cm depth in Wallonia for the 2001-2005 period.

The **critical (acceptable) N concentration, $cN_{(acc)}$,** comes from the CCE/Alterra Report (De Vries et al. 2007):

Coniferous forest	2.5-4 mgN L^{-1}
Deciduous forest	3.5-6.5 mgN L^{-1}

The minimum recommended values are applied for the calculation of CLnutN (Table BE.2).

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; Frédéric André et al., 2010; Frédéric André, Quentin Ponette, 2003).

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 cation deposition

In Wallonia, actual throughfall data collected in 8 sites, between 1997 and 2002, were used to estimate BC_{dep} 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 BC_{dep} data of the 8 sites was extrapolated to all Walloon ecosystems depending on the location and the tree species.

Results
In Wallonia, the highest CL values were found for 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.

Natural vegetations
For Walloon ecosystems, considering the lack of accurate input data, we use critical values established in Flanders with SMB method (Meykens & Vereecken, MIRA/2001/04). The critical loads for N and S deposition to natural vegetations are reported in Table BE.3.

Table BE.3 Critical loads for natural vegetations in Wallonia

Ecosystem type	EUNIS code	CLmax N	CLmax S	CL nut
Natural grassland	E1	4572	1893	1286
Moors and heathland	F4.2	2185	1645	643
Inland marshes	D5	2339	1655	786
Peat bogs-Fens	D2	2339	1655	786

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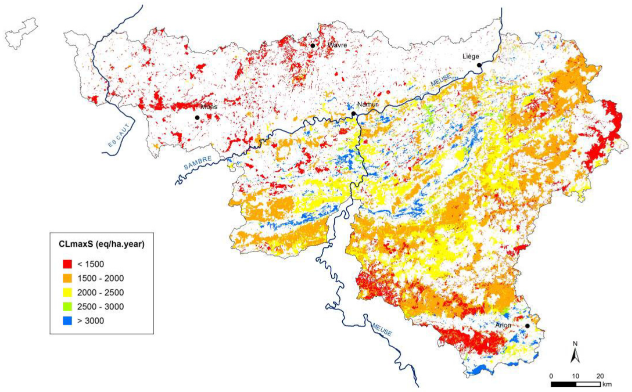

Figure BE.2 Maximum critical loads of sulphur for forests, CLmax(S)

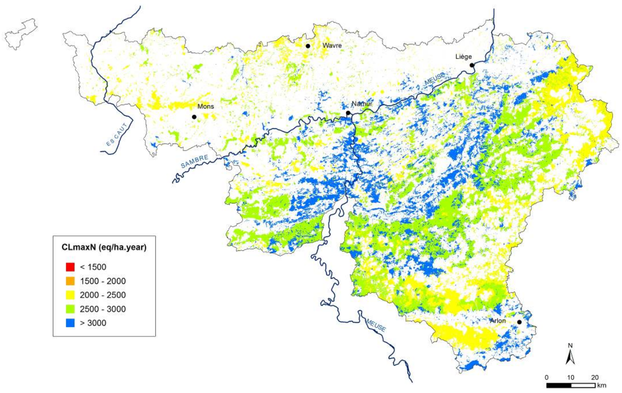

Figure BE.3 Maximum critical loads of nitrogen for forests, CLmax(N)

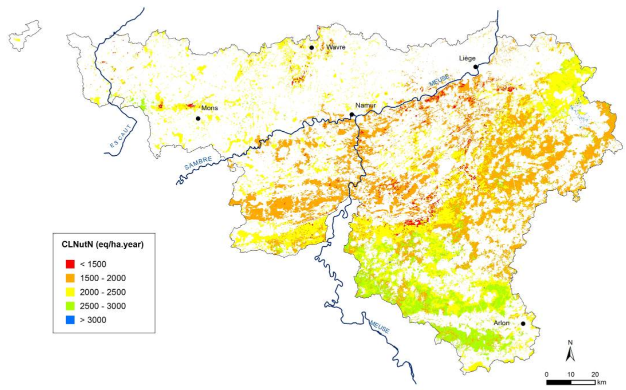

Figure BE.4 Critical loads of nutrient nitrogen for forests, CLNut(N)

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Empirical critical loads of nutrient nitrogen for Finnish Natura 2000 sites

Empirical critical loads of nutrient nitrogen ($CL_{emp}N$) were first assigned for Finnish Natura 2000 sites in response to the CCE call for data 2010–2011 (Holmberg et al. 2011). In response to the CCE call for data 2014–2015, the empirical critical loads of nitrogen were updated using new information on land cover (Härmä et al. 2015). The $CL_{emp}N$ values were assigned for 25 habitat types within the Finnish Natura 2000 sites (Airaksinen and Karttunen 2001, Natura 2000, Metsähallitus 2012). The Natura 2000 GIS data set of the Finnish Environment Institute (Natura 2000 GIS) was used, in accordance with the reporting for the Habitats and Birds Directives (EC 2015). A distinction was made between sites protected within the Birds Directive (SPA), the Habitats Directive (SCI) or by both directives simultaneously (SPA and SCI). Landcover information for Finnish Natura 2000 sites was obtained from the 25 m Corine 2012 database (Härmä et al. 2015). Only area features of the Natura 2000-areas were included, not linear or point features. The landcover classes of the Corine 2012 database were interpreted to EUNIS habitats using expert judgment, in combination with indicative cross-references (Moss and Davies 2002). To distinguish between different mire habitats the mire database of Metsähallitus (Parks and Wildlife Finland) was used.

The land cover information was combined with a 0.10º × 0.05º longitude–latitude grid, in the WGS84 coordinate system. In this grid, there are 25,460 grid cells covering Finnish territory. Within each grid cell, the area for each protection category (SPA, SCI, SPA and SCI) was summed separately for each EUNIS habitat type. Areas smaller than 1 ha were not included. The resulting number of records is 31,245, covering a total area of 41,141 km^2. The total areas of each protection category in each EUNIS habitat are given in Table FI.1. The values of empirical critical loads of nutrient nitrogen were based on the recommendations by the 2010 meeting in Nordwijkerhout (Bobbink et al. 2011, UNECE 2010). The lower values of the suggested ranges were used to reflect the sensitivity of northern boreal ecosystems.

Table FI.1 Empirical CL N values used for Finnish Natura 2000 sites and total area per protection type.

Eunis code		CLNemp (kg ha^{-1} yr^{-1})	Natura sites (km^2)	SPA (km^2)	SCI (km^2)	SCI/SPA (km^2)
A2	Littoral sediments	20	125	12	6.3	107
B1	Coastal dune and sand habitats	8	1.3	0	0.4	1.0
B1.3	Shifting coastal dunes	10	1.3	0	0.6	0.7
B1.4	Coastal stable dune grassland	8	1.6	0	0.7	0.9
B1.5	Coastal dune heaths	10	1.0	0	0.7	0.4
B1.7	Coastal dune woods	10	5.7	0	2.7	2.9
B1.8	Moist and wet dune slacks	10	0.6	0	0.03	0.6
C1	Surface standing waters	3	1 508	24	865	619
C1.1	Permanent oligotrophic lakes	3	3 546	10	2 375	1 161
C1.3	Permanent euthrophic lakes	3	29	13	5.5	11
C1.4	Permanent dystrophic lakes	3	1 562	98	1 209	255
D1	Raised and blanket bogs	5	1 729	19	575	1 134
D1.1	Raised bogs	5	1 077	0.5	548	529
D3.1	Palsa mires	5	376	0	105	271
D3.2	Aapa mires	5	6 519	11	1 954	4 554
D4.1	Rich fens	15	460	0.5	110	350
E2.2	Low and medium altitude hay meadows	10	0.2	0	0.1	0.1
E2.3	Mountain hay meadows	10	0.1	0	0.1	0.01
F2	Arctic, alpine and subalpine scrub habitats	5	6 859	0.1	1 930	4 929
G1	Broadleaved deciduous woodland	10	542	3.4	146	393
G1.9	Non-riverine woodland with *Betula*	5	3 900	0	1 533	2 367
G1.A	Meso- and eutrophic *Quercus* woodland	15	0.6	0.02	0.3	0.3
G3	Coniferous woodland	5	10 952	26	5 453	5 473
G4.1	Mixed swamp woodland	5	145	2	72	71
G4.2	Mixed taiga woodland with *Betula*	5	1 800	11	540	1 249
Total area			41 141	231	17 431	23 479

Exceedance of empirical critical loads of nutrient nitrogen for Finnish Natura 2000 sites

Exceedances were calculated as the positive differences between the N deposition and the $CL_{emp}N$ values. For N deposition, the sum of oxidized and reduced N deposition was used. The deposition was provided by the CCE in the 0.5° longitude by 0.25° latitude grid, calculated by the EMEP model version rv4.3beta and the scenarios according to the revised

Gothenburg Protocol (Simpson et al. 2012). In calculating exceedances for the habitats in EUNIS classes A, B and C, the grid average deposition was used, while the deposition to semi-natural vegetation was used for habitats in EUNIS classes D, E and F, and the deposition to forest was used for habitats in EUNIS classes G.
The critical loads were exceeded for the aquatic habitats (C1), raised bogs (D1.1) and aapa mires (D3.2), coniferous (G3) and mixed woodland (G4) (Table FI.2, Figure FI.1). No exceedances were projected for the other habitats. The exceedances are largest for the year 2005, and decrease considerably for the year 2020.

Table FI.2 Natura 2000 sites, area for which empirical CL N values are exceeded in 2005, 2010 and 2020.

EUNIS code	EUNIS description	Area in Natura sites (km^2)	Area exceeded 2005 (km^2)	Area exceeded 2010 (km^2)	Area exceeded 2020 (km^2)
C1	Surface standing waters	1 508	455	296	62
C1.1	Permanent oligotrophic lakes	3 546	1695	1195	178
C1.3	Permanent euthrophic lakes	29	21	19	19
C1.4	Permanent dystrophic lakes	1 562	980	471	62
D1	Raised and blanket bogs	1 729	3.3	0.6	
D1.1	Raised bogs	1 077	34	6.1	
D3.2	Aapa mires	6 519	1.1	0.4	
G3	Coniferous woodland	10 952	570	413	65
G4.1	Mixed swamp woodland	145	15	8.6	2.5
G4.2	Mixed taiga woodland with *Betula*	1 800	101	78	13
Total area		41 141	3876	2489	401

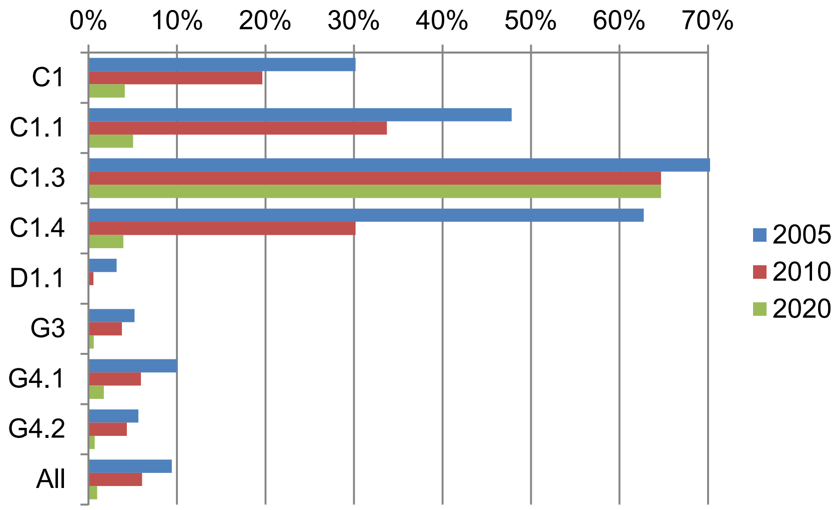

Figure FI.1 Percentage of area in each EUNIS class for which empirical CL N are exceeded.

Summary

Empirical critical loads of nutrient nitrogen, $CL_{emp}N$, were assigned for an area covering about 41,000 km^2 representing 25 habitat types of Finnish Natura sites. While the $CL_{emp}N$ values were exceeded in almost 10% of the total area, or about 4,000 km^2, with the 2005 deposition, the 2020 deposition exceeds the $CL_{emp}N$ values in only about 400 km^2, or less than 1% of the area of the Finnish Natura sites. In relation to their total area, the lake habitats are proportionally more affected by $CL_{emp}N$ exceedances than other habitats. This is because the lakes were assigned the lowest $CL_{emp}N$ values (3 kg ha^{-1} yr^{-1}).

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Introduction

The 2015 Call for Data aimed to compute updated maps of critical loads with a high spatial resolution, standardized for all countries that participate to this Call. For that purpose, new grids were constructed to get a 0.10° x 0.05° regular fishnet for Europe. Within this updating process, new biodiversity critical load maps are supposed to be computed in order to take into account the impact of global changes on plant biodiversity.
To reach these objectives, some important steps have to be completed: i) first, to adapt and update the input data and results to the new grid, in order to propose new CL maps compliant with CCE instructions; ii) to collect and compute data of plant biodiversity evolution on long-term periods, to calculate biodiversity critical loads as well as a Habitat Suitability Index; iii) to progress in modelling the influence of nitrogen deposition and climate change on biodiversity by using combined geochemical-ecological models that consider various N atmospheric deposition scenarios. The achievement of these steps implies the completion of the model calibration for both biogeochemical and ecological modules; iv) finally, to collect data relative to ecosystems protection status (French and European directives). Information concerning protection status would let to improve maps of biodiversity critical loads, and also to highlight a potential protection status effect – through the simulations – on forest ecosystems, in a context of global change.
To reach these objectives, we used input data from three very well documented forest sites belonging to the French ICP Forest network (RENECOFOR, National network of forest health survey from the National Forest Office), which is part of the European network for forest health survey since 1992 (ONF, 2015).

Update of current critical loads maps

Higher resolution grid

Until the last Call for Data, the EMEP 50 km x 50 km grid was used to map steady-state mass balance empirical critical loads and critical loads of sulfur and nutrient nitrogen. The nitrogen deposition data used as inputs to model and map empirical, sulfur and nutrient nitrogen critical loads and evaluate their exceedances were from the EMEP model. So, in order to ensure the compatibility with the new grid, the French critical load database was adapted by computing and using the new 0.10° x 0.05° grid. Resolution differences between old and new EMEP grids are presented in Figure FR.1.

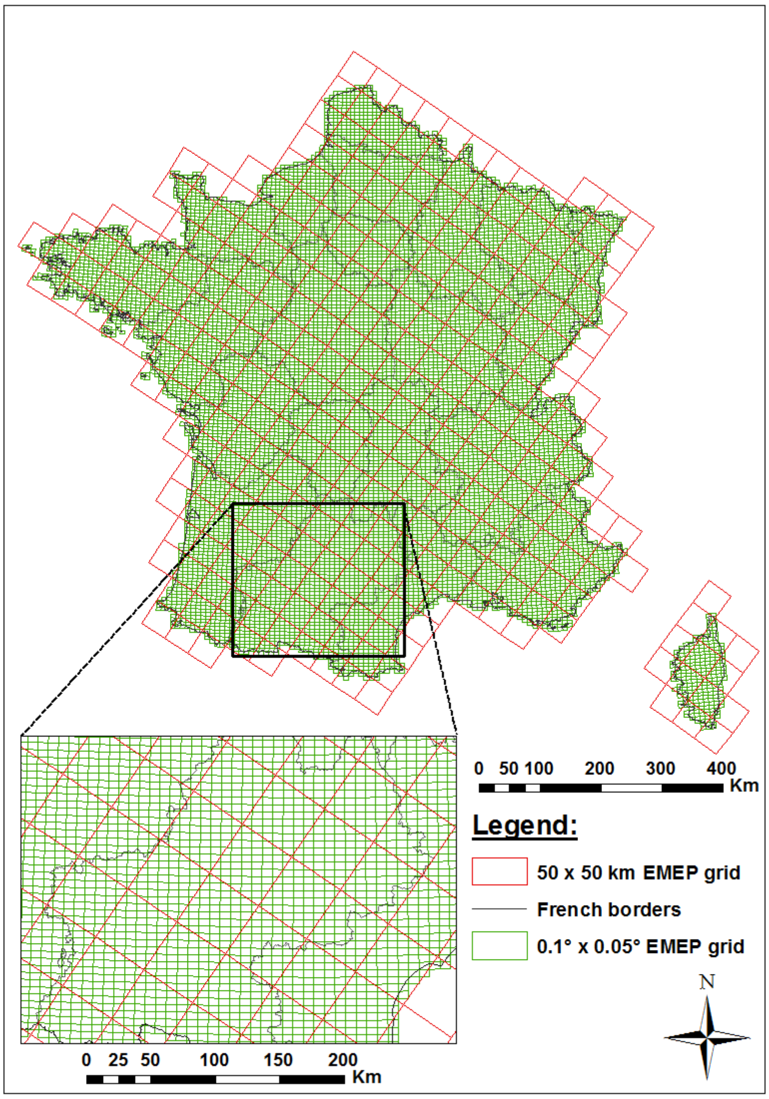

Figure FR.1 European EMEP grids at two different resolutions, adjusted on France boarders. The new grid appears in green and the old one in red.

New resolution CL maps

No new deposition data were added to the database since the last Call for Data for which steady-state mass balance for Sulfur (CLS) and nutrient Nitrogen (CLnutN) and empirical critical loads of nutrient nitrogen (CLempN) were modelled and mapped relatively to the old EMEP grid (50km x 50km) (Party et al., 2001; Probst et al., 2008). New data were collected from the French National Center for Meteorological Research and the ICP-Forest network national database (Brêthes et Ulrich 1997; Archaux et al. 2009), respectively. They concern meteorological data and sites characteristics such as soil texture, percolation rates, on the extended period running up today. However, they need to be carefully checked up or adapted before using in the critical load models. As a consequence, no significant modifications in critical loads values were expected for now when computing maps using the new resolution grid.

The new French CL maps were computed and spatialized using the new resolution grid through the projection of historical experienced data.

These new resolution maps are presented in Figure FR.2, together with the location of the 102 studied ICP Forest sites, which provide the main environmental input data for critical load calculations for forest ecosystems.

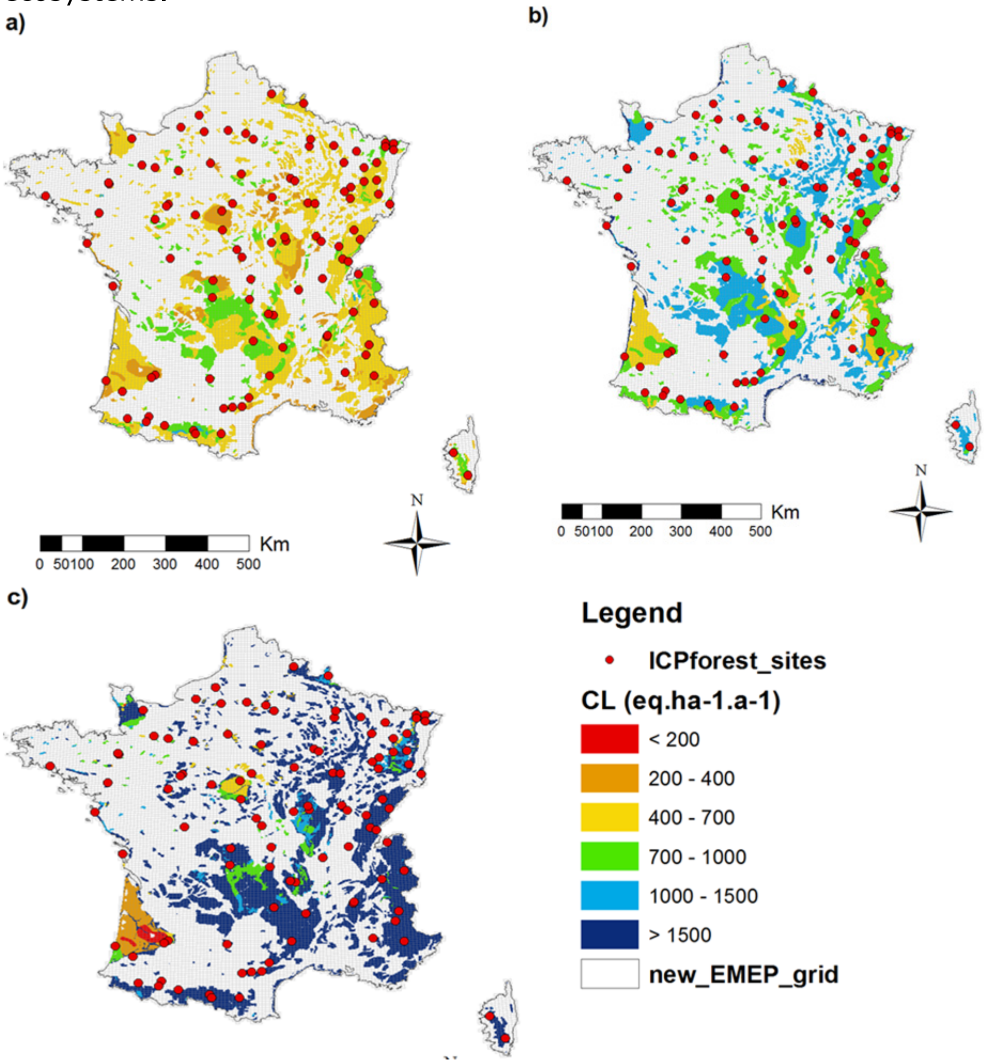

Figure FR.2 Critical Loads maps for France: a) CL of nutrient nitrogen CLnutN; b) CL of sulfur CLS; c) empirical CL of nutrient nitrogen CLempN. Only forested areas are considered.

Critical loads for nutrient nitrogen map indicates that the most sensible ecosystems are localized in the Mediterranean arc, in the Landes region (SW), in the eastern part of the Paris basin and in the very northern part of the Massif Central. On the contrary, less sensible ecosystems are located in mountainous areas (Northern Alps, Pyrénées and in the central part of the Massif Central) (Figure FR.2a).
Empirical critical loads of nutrient nitrogen for French ecosystems were determined on the basis of the method described in chapter V of the Mapping Manual (Posch et al., 2004). Computation details were described in the CCE Status Report 2008 (Probst et al., 2008). Like SMB Critical Loads for nutrient Nitrogen, the most sensitive areas of empirical Critical Loads 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.2b). Less sensitive ecosystems, like for SMB Critical Loads of Nutrient Nitrogen, are localized in some parts of the mountainous regions, such as the Pyrénées, the northern part of Alps, the south of the Vosges region, and the central part of the Massif Central.
Compared with the SMB critical loads of nutrient nitrogen, most of the ecosystems shifted to a higher critical load class with the empirical method (+1 class for 47% of the ecosystems, +2 classes for 35% and +3 classes for 3%). Consequently, the sensitivity of the ecosystems to nutrient nitrogen is supposed to be lower when derived from the empirical method.
According to the Sulphur Critical Loads map, most of the French ecosystems are not very sensitive to sulphur acidification. However, some very sensitive areas do exist, and are, for a very large majority, localized in the Landes (SW) and in the Vosges mountains (North of Alsace region) and, to a lesser extent, in Sologne (Southern Paris Basin) and the Picardie regions (Northern part of France) (Figure FR.2c). As the french ICP-forest sites are well spread across the territory and cover a wide range of Critical Loads classes, forest management can thus be adapted widely in the future in relation to ecosystem sensitivity.
Indeed the use of this new resolution grid allows mapping critical loads at a very local scale more accurately. With such accuracy, it is possible to predict ecosystems sensibility taking into account stand and site characteristics. Regarding to the considered 2 ha ICP-forest sites (Camaret et al., 2004), this resolution will be more efficient to spatialize Critical Loads in relation to field observations.

Modelling the impact of nitrogen deposition and climate change on biodiversity

Calibrating the ecological part of the model

For the last call for data, the calibration of the biogeochemical ForSAFE model was achieved for three well-known sites of the French ICP-forest network. This calibration was done considering the model simulations and measured data for various parameters, and more particularly for pH, stem biomass and ions concentration in the soil solution (Gaudio et al. 2015). The impact of climate change and atmospheric Nitrogen deposition scenarios on soil solution composition could thus be simulated on a long time scale (over one century). With the objective to calculate a Habitat Suitability Index for the concerned ecosystems, the ecological model *Veg* used in combination to *ForSAFE* to simulate plants diversity evolution, needs to be calibrated as well (Sverdrup et al. 2007; Belyazid et al. 2011; Probst et al., 2015). The main objective of this current work is to parameterize at least five environmental factors for each of the 476 representative plants species of selected French ecosystems. In order to complete this calibration, we followed two different methods.
The first method consists in using verified data from the literature, ICP-forest field measurement campaigns, and expert judgement. These data concern plants response to important environmental parameters such as pH, soil water content, temperature, light and nitrogen concentration. The main principle of this method is to parameterize for a given plant species considered, its response to each of these environmental parameters under French climatic conditions. In the following step, we adjusted the parameterisation to calibrate the model after each run, using field measurements over a period of 20 years. Plants response

was analysed by comparing simulated and measured cover percentages. An example of this calibration is presented for an oak dominated site localized in the central part of France (Figure FR.3).

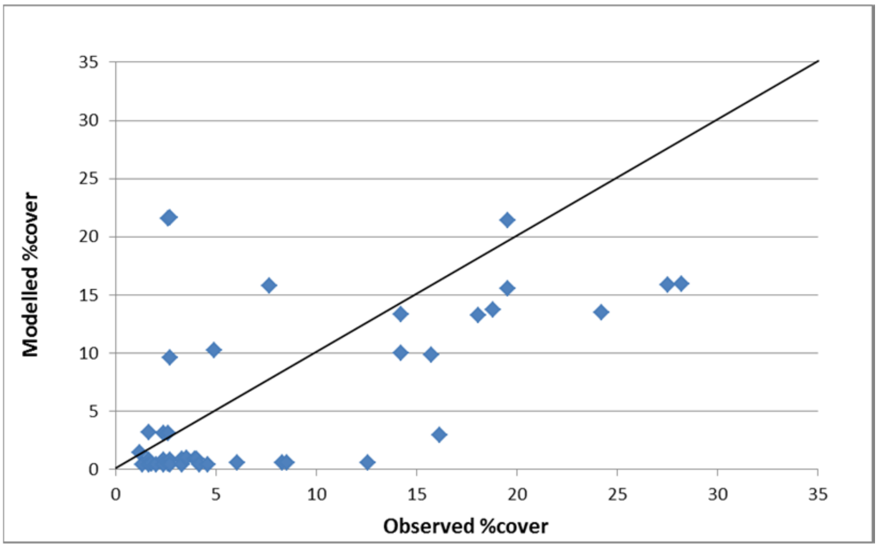

Figure FR.3 Comparison between simulated and measured plants cover response to five environmental factors on sessile oak dominated site

The Czekanowski similarity index was 64%, which indicated a rather good similarity between observed and simulated data. However, differences still exist and concern specific species. The calibration work will thus continue in order to target species that are not well parameterized yet, and to identify specific environmental factors we have to focus.

The second method consists in calibrating the vegetation model Veg using the ecological database EcoPlant developed in France (Gégout, 2001). EcoPlant is a floristic database containing thousands of phytoecological measures sampled in more than 14,000 French forested sites (Figure FR.4). For each sample, the exhaustive list of plants species observed on the site, and the measurements of ecological and environmental factors among which pH, C/N, mean annual temperature, soil water content and stand structure, are compiled (Coudun et al.,2005; Gegout et al., 2005).

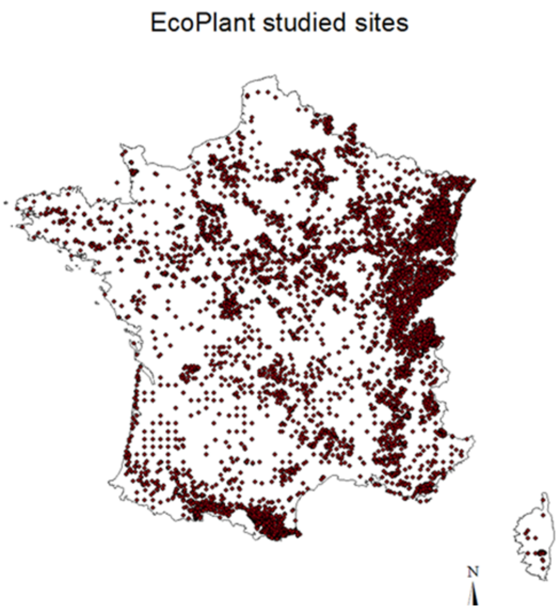

Figure FR.4 Localisation of the 14,000 forest ecosystems sampled into the EcoPlant database

By developing this method, the objective is to calibrate the plants response using measured and experienced data, for the five environmental factors described before. To reach this objective, linear regression models were developed (Coudun et al., 2006) to parameterize the plants response from one to five ecological factors, which have a high influence on the presence probability (amplitude and optimum) for each species. An example of Sessile oak response to the mean annual air temperature for the 1961–1990 period is presented in Figure FR.5.

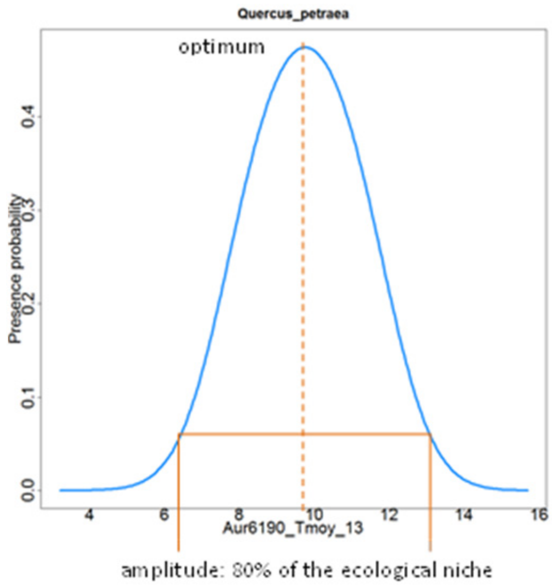

Figure FR.5: Sessile oak presence probability according to the mean annual air temperature (°C)

The Veg model will then use these amplitude and optimum to predict the probability of presence of species relatively to each forest site environmental characteristics computed by the geochemical model ForSAFE.

Example of simulations outputs

These two calibration methods will allow predicting accurately the evolution of plants biodiversity on each considered forest sites. Since climate change and atmospheric deposition scenarios (MFR: Maximum Feasible Reduction and CLE: Current LEgislation as examples) are included, the impacts of global changes on plant species composition can be predicted on a long time scale. As a result, it will be possible to use plants cover estimation to compute a Habitat Suitability Index, and then to map biodiversity critical loads with these output simulation results. This work will be achieved in 2016 (Rizzetto, PhD in progress). Some preliminary simulations were performed without these new calibrations.

Figure FR.6 presents the changes in cover percentages of four vegetation groups in a Spruce dominated site (a) and in a Sessile oak dominated site (b), under two atmospheric nitrogen deposition scenarios (CLE and MFR), from nowadays to 2100 . Details of these changes at the species scale for the Spruce (c) and the Oak (d) dominated sites are also mentioned.

The first results show that on a long time scale, atmospheric nitrogen deposition has an impact on plant species cover, and therefore on site biodiversity, but it depends on the scenario considered. Depending on the characteristics of species that composed each group, the cover percentage varies differently with the two deposition scenarios. For example, the mosses group appears to be more stimulated by the MFR scenario, which is supposed to be the least nitrogen impacting one. Indeed, this trend is driven by the response of *Polytrichum formosum* Hedw., which is the dominant species into the mosses group. Similar observations can be made for the other groups: their global trend is controlled by one or two dominant species response to nitrogen deposition. Consequently, plant specific responses to nitrogen deposition depend on species autecological characteristics.

These results have to be confirmed but the results observed on two well-known forest sites reveal that soil biogeochemistry and plants respond to the scenarios by following the same trends (Figure FR.6).

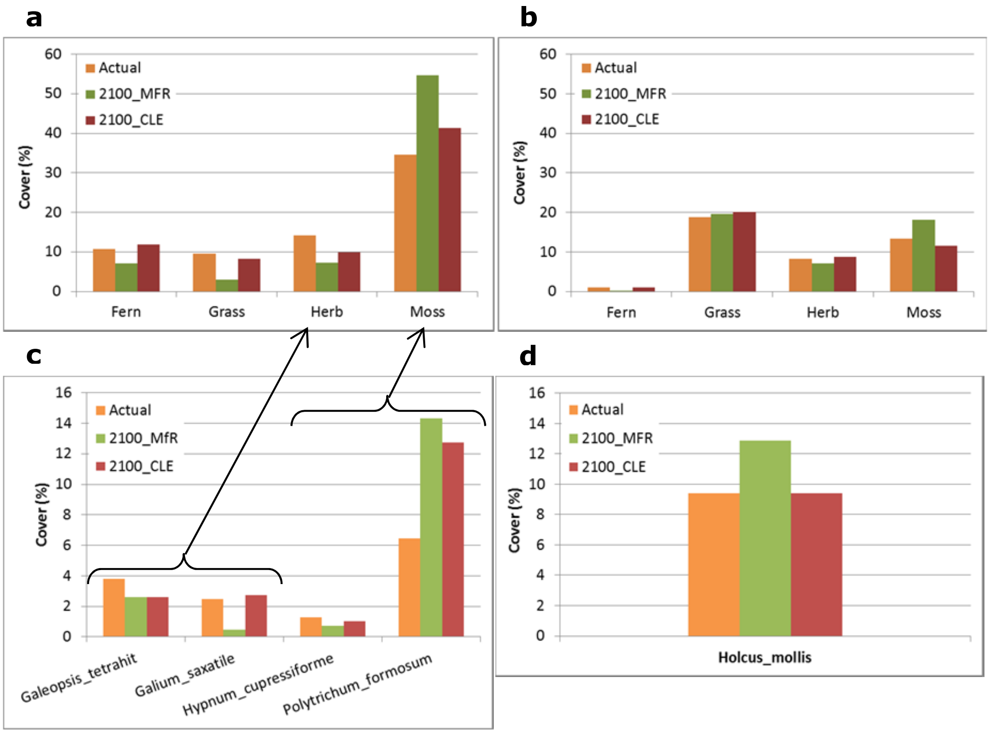

Figure FR.6 Changes on the cover percentage of vegetation group and species observed from nowadays to 2100 under two atmospheric nitrogen deposition scenarios (MFR and CLE) in a Norway spruce dominated site (a & c) and in a sessile oak dominated site (b & d).

The influence of nitrogen deposition on plants cover has also to be evaluated in association with climate change. Figure FR.7 presents the cover evolution of two understorey species (*Hedera helix* L. *and Holcus mollis* L.) in an oak-dominated site, under the influence of two nitrogen atmospheric deposition scenarios (MFR: Maximum Feasible Reduction and CLE: Current LEgislation), without (a) and with (b) climate change.

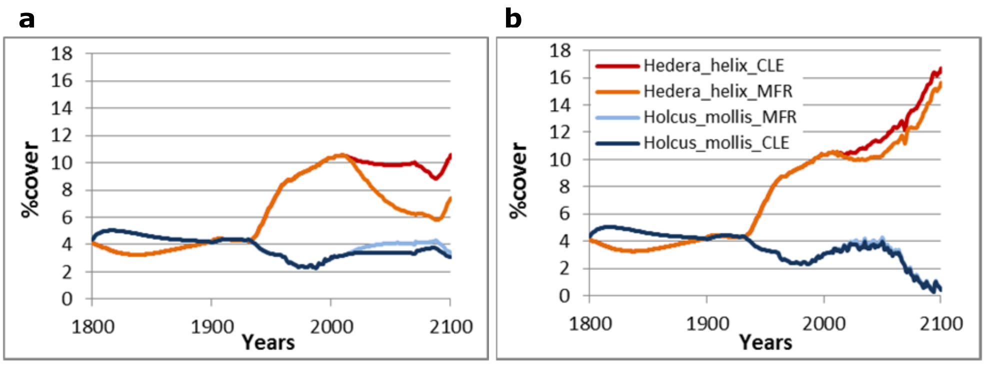

Figure FR.7 Evolution of the cover percentage of two understorey species on a sessile oak dominated site, under two different N atmospheric deposition scenarios (CLE and MFR): a) with no climate change scenario; b) with the A2 - High growth climate change scenario.

Results show that species evolution mainly depends on their affinity to temperature and nitrogen. In a context of both climate and N deposition changes, for these species, climate change has a predominant effect on species cover than N deposition. Indeed, differences observed on species cover between the two nitrogen deposition scenarios (ex. *Hedera helix* which is a nitrophilous species is favoured by CLE with no climate change) becomes reduced under the influence of the A2 climate change scenario.

Nature protection programs

Another important objective of this Call for Data was to give informations relative to ecosystems status of protection. The aim is to know exactly which type of protection applies to French forest ecosystems. This kind of data allows understanding the biodiversity evolution trends according to the protection status. We used data compiled by the INPN (National Inventory of the Natural Patrimony) to propose maps of protection programs applied in France, and more specifically on the 102 studied sites from the French ICP-forest Network. The data we were able to collect, concern European and also national protection programs.

The Natura 2000 European network aims to ensure the long-term survival of species and habitats of special concern, with high stakes of conservation in Europe. It consists of a set of natural sites that shelter rare and fragile species of flora and fauna. The structure of this network includes Special Protection Area (SPAs) for the conservation of wild birds species (7.9×10^6 ha), and Special Area of Conservation (SACs) for the conservation of habitat types and plants and animal species (7.5×10^6 ha) (INPN website, 2015). Distribution of SPAs and SACs in France are presented in the following maps (Figure FR.8 (a) and FR.8 (b) respectively).

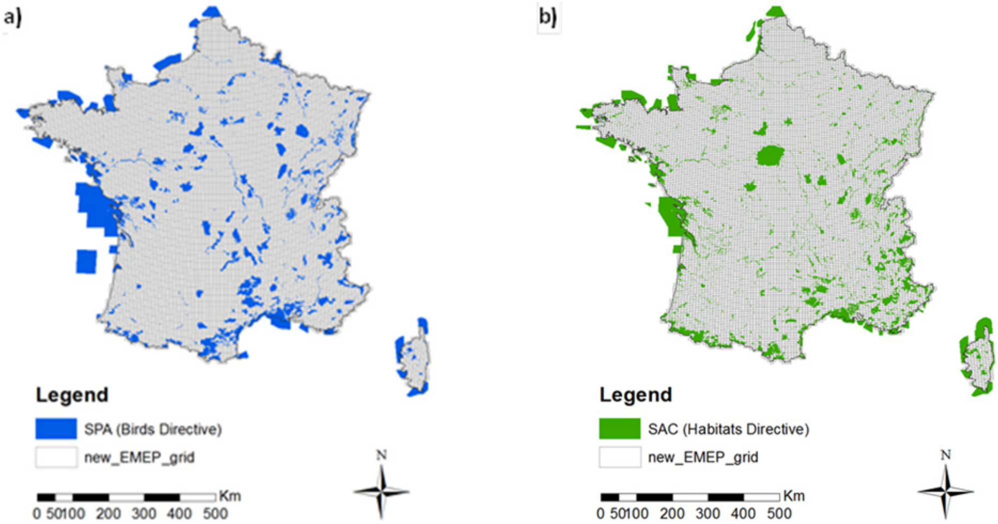

Figure FR.8 (a) National repartition of Natura 2000 SPAs. (b) National repartition of Natura 2000 SACs. Both maps show this repartition relatively to the new 0.1 x 0.05 ° EMEP grid.

Both Natura 2000 protection statuses are applied on terrestrial and marine ecosystems. The comparison of these two maps indicates that Special Protection Areas are larger than Special Areas of Conservation. Actually, SPA average area is around 20,000 ha, whereas SAC is 5,500 ha (both marine and terrestrial ecosystems are considered). But, SAC are four times more numerous than SPA. These differences are linked to the nature of the target group of species to be protected. On the one

hand, Birds Directive brings protection to areas known to be useful for birds species, i.e. nesting zones, or stop-over zones during migration. Large areas are needed to ensure this role. On the other hand, Habitat Directive targets remarkable habitats because of the species they shelter. Depending on the species protected, for SACs the areas vary from 10^{-2} to 4.5×10^{5} ha in France, where the smallest area corresponds to a building where a colony of bats is hosted (INPN website, 2015). Because of the diversity of plants and animals species to protect, SACs are very fragmented, very specific and defined at a very local scale. For these reasons, SPAs are defined at regional scale where necessary large area sites are identified, whereas SACs are smaller zones locally established that target specific species to protect.
Considering terrestrial habitats in metropolitan France, 4.4×10^{6} ha are classified as SPA, and 4.7×10^{6} ha as SAC. In other words, 12.6 % of terrestrial ecosystems and 18.9 % of forest ecosystems from metropolitan France are under Natura 2000 protection status. Among the 102 studied sites from the ICP-forest network, there are 19 Special Protection Areas, 20 Special Areas of Conservation. Nine sites are both SPA and SAC (Figure FR.10).
In addition to these status defined according to European programmes for environment protection, some national directives are also applied on French forest ecosystems. One of the main French protection programs is the inventory of Natural Areas of Ecological Fauna and Flora Interest (ZNIEFF), which purpose is to identify and describe areas with strong biological capabilities and a good state of conservation. Two types compose this inventory: ZNIEFF type I concerns areas of great biological or ecological interest; ZNIEFF type II denotes large, rich and slightly modified landscapes, providing significant biological potential (INPN website, 2015). The distribution of ZNIEFF protection areas is presented in Figure FR.9.

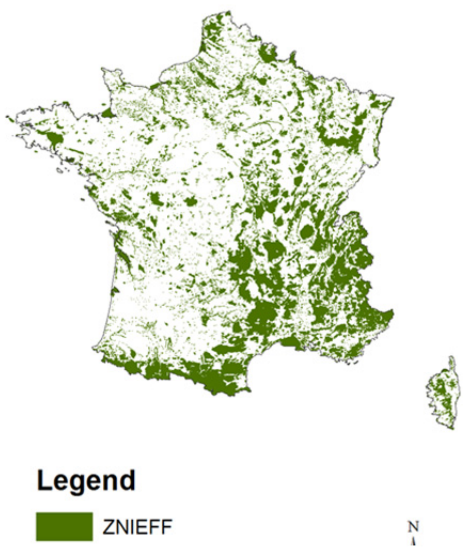

Figure FR.9 Distribution of ZNIEFF protection areas of metropolitan France

ZNIEFF protection measures are applied on more than 16 million hectares in metropolitan France, which represents 47 % of forest ecosystems (IFN website, 2015). The average ZNIEFF area is 1100 ha, that is much lower than the average surface of SAC and SPA for terrestrial ecosystems. So, the national ZNIEFF program let to apply

protection measures on local ecosystems to protect specific species, and thus represents a good complement of European programs.
Figure FR.10 is a map combining the 102 ICP-forest sites with Natura 2000 and ZNIEFF protected areas.

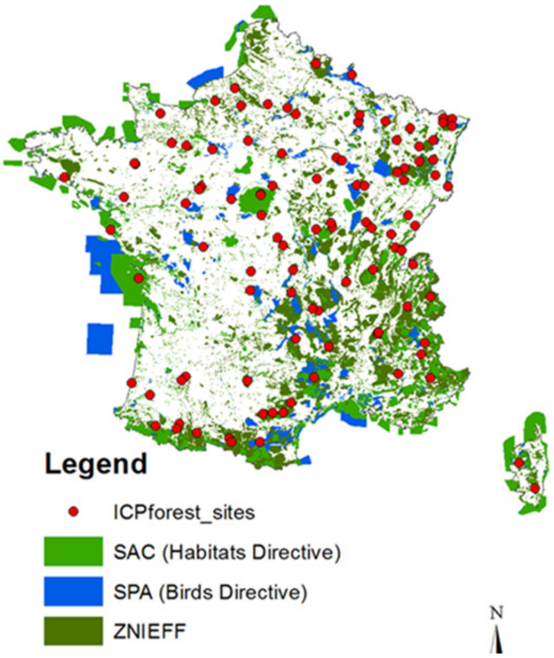

Figure FR.10 Distribution of N2K and ZNIEFF protection areas in relation with the French ICP-forest sites

Among the 102 considered ICP-forest sites, 30 sites are concerned by one or two of the SPA and SAC classifications, and 78 sites are under ZNIEFF protection.
All these protection programs target specific species, i.e. the Special Protection Areas of the Natura 2000 network for the conservation of wild birds species. It is also true for the SACs where protection measures are motivated by one remarkable or rare species present on the site.
However, by taking measures to protect one specific species in a defined area, other species living into the same ecosystem should benefit from this protection through an "umbrella effect" from the protected species. An *umbrella species* can be defined as a species whose conservation confers protection to a large number of naturally co-occurring species (Roberge et al., 2004). Considering this concept of "umbrella species", it is obvious that all these ecosystems protection status are important, especially when different complementary programs are applied such as ZNIEFF and Natura2000.
For our purpose of evaluating critical loads of nitrogen in a context of global change, it is important to consider protected and non-protected areas to run the dynamic biogechemical-ecological model in order to predict biodiversity evolution. By considering the protection status of each site, the impact of anthropogenic activities and forest management should be underlined in addition to changes due to atmospheric global changes. The model simulations should let us know how ecosystems will change at a long time scale, depending on their protection status. Within the next few months, this type of information will be considered while analysing of the simulation results on the different studied sites.

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The response of the German NFC to the Call for Data (CCE 2014) focuses on new-developed critical load based on biodiversity. Despite this, the "classical" critical loads, protecting ecosystems against acidification and/or eutrophication, were submitted as well. The dataset was completed by the empirical critical load values for EUNIS classes relevant for Germany (see Table DE.1). The German dataset consists of 540,019 records representing 29.7 percent of the territory. Unlike the former data submissions, now the critical loads were computed based on polygons instead of the former 1 km² grid structure.

Table DE.1 EUNIS classification for selected receptors of the critical load computation in Germany

EUNIS Code	Proportion of the receptor area [%]	Proportion of German territory [%]	EUNIS Code	Proportion of the receptor area [%]	Proportion of German territory [%]
A.x	0.03	0.01	G1.8	0.16	0.05
D.x	1.06	0.31	G1.A	0.99	0.29
E.x	1.34	0.39	G3.1	28.51	8.36
F.x	0.28	0.08	G3.4	13.06	3.83
G1.2	1.63	0.48	G3.E	0.90	0.26
G1.4	0.29	0.08	G4.3	10.24	3.00
G1.5	0.76	0.22	G4.7	0.18	0.05
G1.6	20.77	6.09	G4.F	19.80	6.20

Mass balance based critical load of sulphur and nitrogen
Critical loads are calculated following the methods described in the Mapping Manual (ICP Modelling & Mapping 2015). New data of long-term annual means of precipitation surplus (1980–2010) were available (BGR 2014a) and with a new land use dependent soil map (BGR 2014b) more detailed information on soil dependent input parameter could be derived. The former 72 soil units now are sophisticated into 674 combination types of soil form, landuse form and climate zone. For each of these combination types a typical soil profile is attached. A lot of soil chemically and physically data are attributed to the horizons, such as

field capacity, row density, pH-class, CEC, class for organic matter and others. Furthermore, new deposition estimates for base cations and chloride were available (Pineti 2015). A sensitivity study for the influence of the changed parameters was conducted (see below).

Criticals load of acidity, $CL_{max}S$ and $CL_{max}N$

The calculation of critical load of acidifying sulphur for forest soils and other (semi-)natural vegetation was conducted according to the simple mass balance equation V.22 of the Mapping Manual. For base cation and chloride deposition the 3-year means (2009–2011) were included in order to smooth large variations of this parameter due to meteorological influences (PINETI 2015). The critical load calculation for each polygon of the dataset was done by using 3 different chemical criteria: the critical aluminium concentration (equation V.29), the critical base cation to aluminium ratio (equation V.31) and the critical pH-value (equation V.35). The minimum value determines the $CL_{max}S$ for the specific ecosystem.
The critical load for acidifying nitrogen, $CL_{max}N$, was computed with equation V.26 of the Manual.

Empirical and mass balance critical loads of nutrient nitrogen, $CL_{emp}N$ and $CL_{nut}N$

The mass balance based calculation of the critical load of nutrient nitrogen is described in detail in the Mapping Manual (equation V.5). Different criteria and, consequently, different protection targets were used for acceptable N concentrations in soil solution for the critical load computation. Following the Manual (Chapter V.3.1.2 and Table V.5) the limit can be set between 0.2 mg N per litre (vegetation change from lichens to cranberry) and 6.5 mg N per litre (upper range for deciduous forest). Specific values for acceptable N concentrations $[N]_{crit}$ were derived on the base of these ranges due to computed specific critical load for NATURA 2000 habitat types in Germany BMVBS 2013. For 1990 various habitat types specific $[N]_{crit(plant)}$ are published (ARGE Stickstoff BW 2014).

Table DE.2 Matrix of applied acceptable N concentrations in soil solution (see also Mapping Manual Table V.5)

Sensitive species of the vegetation type	N_{crit}[mg N/l]
Lichens	0.3
Cranberry	0.5
Blueberry	1.0
Trees with risk on fine root biomass or sensitivity to frost and fungal diseases	3.0
Less sensitively coniferous trees	4.0
Less sensitively deciduous trees	5.0
Rich fens and bogs	2.0
Flood swards	5.0
Grass lands	3.0
Heath lands	4.0
Herbs	5.0

In addition to the calculation of critical loads with the steady-state mass balance approach, empirical critical loads of nitrogen ($CL_{emp}N$) following the updated and reviewed values from the expert workshop in

Noordwijkerhout 2010 (Bobbink & Hettelingh 2011) were broadly assigned to national maps. The difference between these approaches is fundamental and ranges from different levels of uncertainty to protection aims which are not congruent. Therefore, the $CL_{nut}N$ and the $CL_{emp}N$ for Germany should not be mixed or combined in order to derive another critical load dataset. Only the German $CL_{nut}N$ dataset shall be used in integrated assessment modelling and not the minimum value of both, as discussed as the ICP M&M meeting in Zagreb.
Critical loads to protect biodiversity

Description of the model approach
The model BERN (**B**ioindication for **E**cosystem **R**egeneration towards **N**atural conditions) was designed to integrate ecological cause-effect relationships into environmental assessment studies including the derivation of critical load (Schlutow et al. 2015).
Natural plant communities that were observed on reference sites in a reference year, e.g. before major air pollution impact, can be defined as reference communities. They represent the current solution of long-term interaction between their species to each other (competition, coexistence, cooperation) and to the environment. In order to model reactions of plant communities to changes in the environment, the reference realized niches of plant species (currently 1970) and of plant communities (692 communities) with their fuzzy (blurred) thresholds of the suitable site parameters are derived from the BERN database including more than 45,000 relevés at more than 7,600 locations in Europe. It is assumed that these combinations of site parameters represent a dynamic nutrient balance. The plant communities are therefore classified as reference site types.
The BERN model derives the niches of those plant species, which mainly constitute the community, i.e. the constant plant species, which are by definition, the characteristic species and all attendant species that can be found with a similar abundance in more than 70 percent of all vegetation relevés representing the plant community at the same ranges of the site parameters. The assemblage of constant plant species of a community does not vary significantly within a climatic region or at a short time scale, if the site state parameters do not vary significantly in space or time.
The possibility for a plant community should be defined in a way that it reaches the highest values at the point where most constant species have their maximum values too.

The following site parameters are used in the BERN database to characterize reference site types (in the shape of trapezoidal functions):

- Soil water content at field capacity [m^3 m^{-3}];
- Base saturation [%];
- pH value (in H2O);
- C/N ratio [g/g];
- Climatic water balance [mm per vegetation period]: precipitation minus potential evaporation;
- De Martonne-Index of continentality [precipitation in vegetation period per mean temperature in vegetation period + 10];
- Length of vegetation period [d a^{-1}]: number of days of the year with an average daily temperature above 10°C;

- Available energy from solar radiation during the vegetation period [kWh m^{-2} a^{-1}]: depends on latitude, slope, aspect, cloudiness, and the shading caused by overlapping vegetation layers and their coverage in the plant communities;
- Temperature [°C]: The trapezoid function was defined by the following indicators: minimum (frost hardiness), minimum and maximum of optimum (beginning and ending of photosynthesis) and maximum (heath hardiness).

Input parameters from the BERN model for biodiversity critical loads

The parameters in the BERN database for which critical thresholds for the preservation of plant communities can be estimated are similar to the parameters used in the "Simple Mass Balance" (SMB) method for critical load computations, e.g. C/N ratio, base saturation, pH value. A reasonable threshold value is the degree of possibility at the intersection point of the optimum plateau border line with the site gradient for nutrient imbalance with decreasing C/N-ratio and decreasing base saturation caused by eutrophication and acidification (see Figure DE.1). Complying with these values, the natural reference plant community just can exist at the maximum possibility of its occurrence (100 percent). We define the values as critical limits.

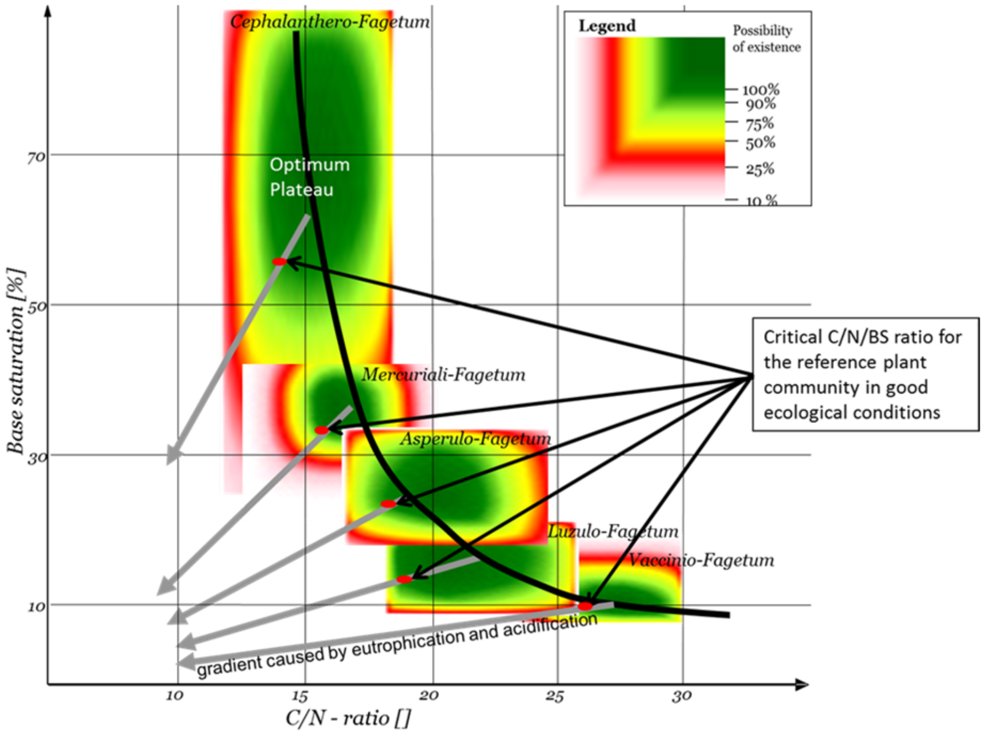

Figure DE.1 *(1) The red-to-green fields show the distribution of the possibility function of all beech communities in the planar-subatlantic region with a plane relief, groundwater distance >2 m; (2) the black line shows an obviously regular arrangement of the natural plant communities, which demarcates an indirect proportional correlation between the base saturation and C/N-ratio at the optima of possibility ranges; (3) the grey arrows indicate the trend of nutrient imbalance after acidification and eutrophication; (4) the red points define the critical limits of the communities.*

Biodiversity critical loads for acidification
The substitution of the critical limits found in the "classical" critical load calculation with threshold determined by plant communities allows the application of the SMB approach as described in the mapping manual. For the threshold of acid deposition (CLS_{max}) the critical base saturation ($BS_{crit(biodiv)}$) e.g. could be used in equation V.38. In addition the critical acid neutralization capacity ($ANC_{le(crit)}$) was computed using the empirical GAPON exchange coefficients (deVries and Posch, 2003) as well as the relation H^+/Al^{3+} (Table V.9 of the Manual).

Biodiversity critical load for eutrophication
Biodiversity related critical load of nitrogen (CLN_{max}) are based on the fact that the C/N ratio is a rather solid parameter which changes with nitrogen deposition continuously and reflects the site conditions very well. The critical C/N ratio needs a transformation to a critical nitrogen concentration $[N]_{crit(biodiv)}$ in order to fit into the simple mass balance equations according to the manual (eq. V.6). The following approach is proposed.

$[N]_{crit(biodiv)} = \frac{N_{\min(crit)}}{\theta \cdot z}$ with $N_{\min(crit)} = N_{t(crit)} - N_u - N_{de} - N_{org}$

with

$[N]_{crit(biodiv)}$ =	critical nitrogen concentration in soil water of the rooting zone as long-term annual mean [kg N m^{-3}]
$N_{min(crit)}$ =	critical amount of mineral nitrogen as long-term annual mean [kg N m^{-2}]
θ =	average content of water in the rooting zone [m^3 m^{-3}]
z =	depth of the rooting zone [m] (as minimum of the potential depth determined by the rooting potential of the soil and the potential rooting depth of the dominant plant species of the occurring plant community)
$N_{t(crit)}$ =	critical amount of total nitrogen in soil and soil water as long-term annual mean [kg N m^{-2}]
N_{org} =	amount of organic nitrogen as long-term annual mean [kg N m^{-2}]
N_u =	annual nitrogen uptake of biomass as long-term annual mean [kg N m^{-2}]
N_{de} =	annual nitrogen loss by denitrification as long-term annual mean [kg N m^{-2}]

$N_{t(crit)} = \frac{C_{org}}{C/N_{crit(biodiv)}}$ with $C_{org} = \frac{OM \cdot \rho \cdot z}{f_{C/OM}}$

with:

C_{org} =	amount of organically fixed carbon as long-term annual mean [kg C m^{-2}]
OM =	share of organic matter in the soil [%]
$f_{c/OM}$ =	transformation factor ($f_{c/OM} \approx 1{,}72$ for mineral soils and $f_{c/OM} \approx 2$ for peats and humus soil layers)
ρ =	dry bulk density of the soil [g cm^{-3} = 1000 kg m^{-3}]

$N_{org} = N_t \cdot (1 - f_{min})$

with:

f_{min} = factor (0-1) describing the share of N_{min} to N_t (linked to the clay content in the soil)

The data for θ, z, OM, ρ and clay content was derived by the horizon specific data of reference soil types in Germany. The f_{min} was derived by the clay content, but is an indicator for soil moisture and pH in soil water as well. This landuse specific database is provided by the BGR (2014b).
The plant communities described in the BERN database were linked to their typical reference soil profiles and the deduced data.
Regularly a plant community can be typical for various reference soil types leading to different $[N]_{crit(biodiv)}$ for the same community; therefore the values for the $[N]_{crit(biodiv)}$ needed aggregation to one value. The 50th percentile (median) was chosen as threshold representing a rather conservative approach since the maximum values still contain vital plant communities. The choice for median was made in order to reduce data uncertainties which might lead to unrealistic results.
The results for natural and semi-natural plant communities range between 0.07 mg l^{-1}(5th percentile) and 4.7 mg l^{-1} (95th percentile) with a median of 1.2 mg l^{-1}.

Results

The regional distribution of resulting critical load to protect biodiversity is shown for sulphur, CLS_{max} in Figure DE.2 and nitrogen, CLN_{max} in Figure DE.3 and the results for the "classical" critical load is shown in Figure DE.4 and Figure DE.5.

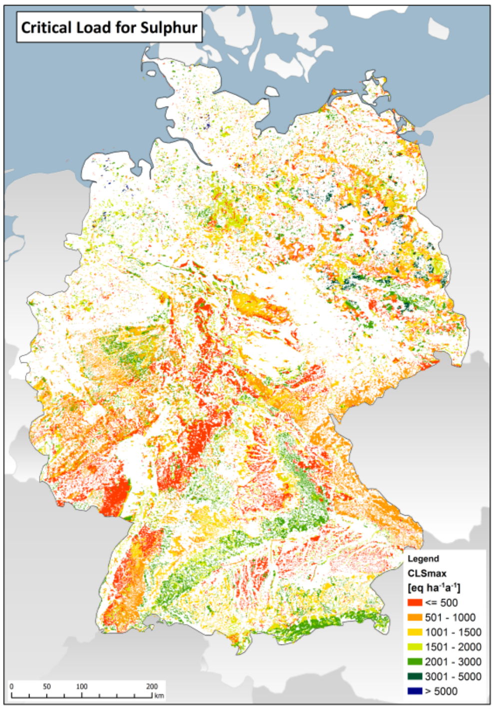

Figure DE.2 CLS_{max}

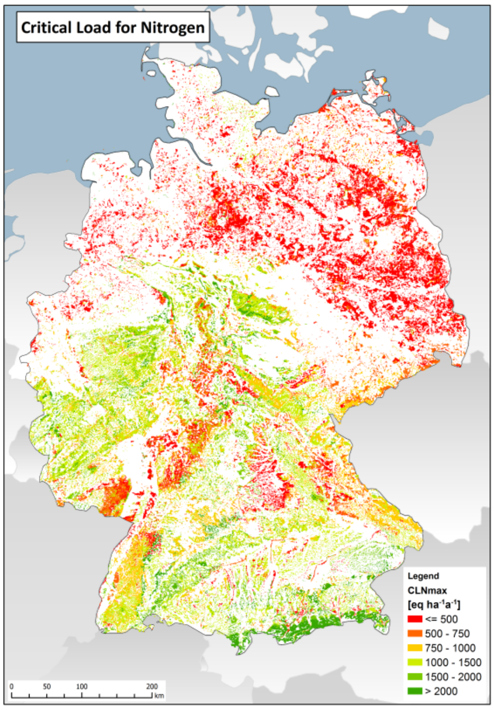

Figure DE.3 CLN_{max}

In comparison with the "classical" critical load computed with critical limits according to Table DE.1 the application of new critical limits to protect biodiversity derived from the BERN database result in a higher sensitivity of acid and nitrogen deposition. Ecosystems with high risk for acidification (CLS_{max} below 500 eq $ha^{-1}a^{-1}$) were identified for about 25 percent of receptor area instead of 17 percent without biodiversity limits. And more than 30 percent of the ecosystems showed biodiversity critical load for nitrogen deposition below 500 eq N $ha^{-1}a^{-1}$ (see Table DE.2). In addition Figure DE.6 shows the overall distribution of the resulting datasets and underpins the trends described above.

Table DE.3 Results for different critical load approaches (share of the receptor area in [%])

Range [eq ha^{-1} a^{-1}]	CLmaxS (1)	CLSmax (2)	CLmaxN (1)	CLnutN (1)	CLNmax (2)	CLempN (3)
< 500	17.56	25.21	0.06	21.53	30.84	0.00
500 - 1000	11.49	24.94	15.04	29.57	27.39	63.39
1000 - 1500	18.92	21.06	11.04	19.31	27.18	36.13
1500 - 2000	18.17	11.74	9.98	12.55	7.96	0.48
2000 - 3000	27.18	15.39	63.88	17.04	6.64	0.00
3000 - 5000	4.81	1.52				
> 5000	1.87	0.13				

(1) "Classical" critical load applying the SMB method as described in Chapter V.3 of the Mapping Manual (data submitted with the CFD 2015)
(2) Critical load of biodiversity resulting from the BERN model (data submitted with the CFD 2015)
(3) Empirical critical load according to Bobbink & Hettelingh 2011

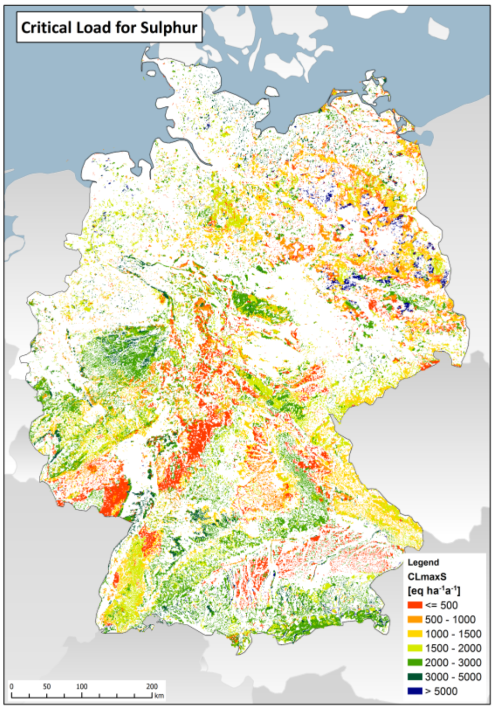

Figure DE.4 CLmaxS

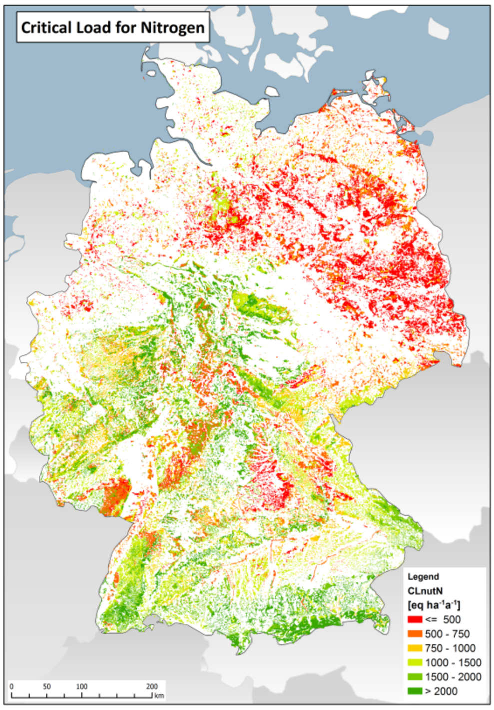

Figure DE.5 CLnutN

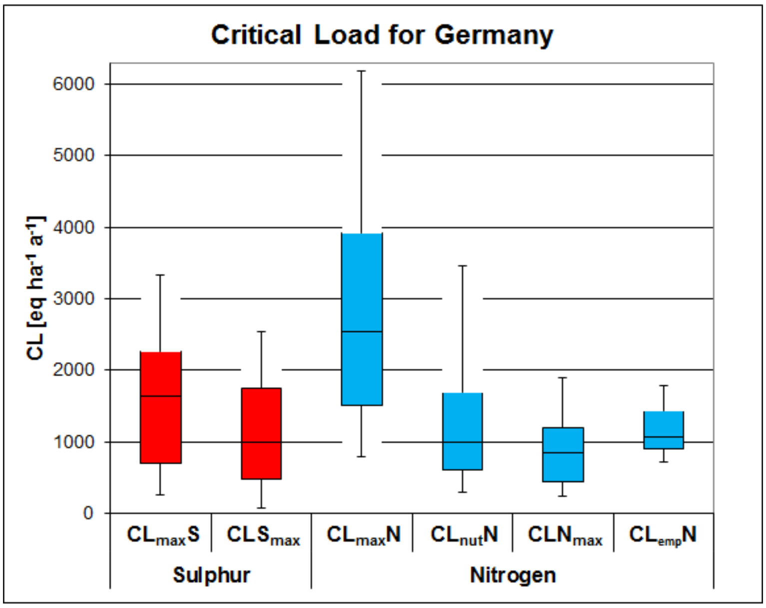

Figure DE.6 The distribution of the submitted critical load datasets

Critical load sensitivity
Updates for several input parameter are available and were included in the latest critical load computation. In order to estimate the impact of the changed input parameters a sensitivity analysis was carried out. The results of a reference run (calculation with the current method and most recent input data) were compared with results for simulation runs (calculation with the current method but changed input parameters). The changed considered parameters are:

- precipitation surplus, PS [mm]
- deposition of base cations, BCdep [eq ha^{-1} a^{-1}]
- uptake of base cations and nitrogen, Bcu [eq ha^{-1} a^{-1}], Nu [eq ha^{-1} a^{-1}].

For the selection of the subset for the sensitivity analyses the GIS data sets of the most recent input data of PS and BCdep were intersected with the old input data for critical load submission (see CCE Status Report 2012 p.81 ff.). The changes of PS and BCdep from one dataset to another are not evenly distributed and contain spatial patterns. Therefore cumulative distribution functions (CDFs) of the changes were calculated. Such CDFs allow the identification of the real-value of the parameter at different levels of probability (percentiles). Values at the 5th and the 95th percentile and equal to the median were selected for the sensitivity analysis. The absolute parameter values at the percentiles thresholds (5^{th}, 50^{th} and 95^{th}) for PS and BCdep were chosen as the constant absolute change of the input parameters for the simulation. This approach implies a constant absolute change of the input parameters (for the different percentiles) but realistic variations in the reference input data. The change of the uptake input data doesn't show spatial patterns and was therefore excluded from this identification process.
The method and input data of the reference run is equal to the data of the recent call for data submission. Four critical load calculations were performed in addition to the reference run. Three runs with single

variating input parameter (PS, BCdep and Bcu/Nu) and one run with variation of all parameters.
Firstly, the absolute change of the input parameter was compared with the absolute change of the critical load (CLnutN and CLmaxS). In order to identify the direction of the dependency and to qualify the sensitivity of critical load based on changes in the input data. Therefore, the slope of the linear regression was chosen as indicator for the strength and direction of the impact of the parameter. Secondly, the resulting critical load in the reference and the simulation run was compared directly. This analysis provided the standard deviation of the absolute (SD abs. in [eq ha^{1} a^{-1}]) and relative (SD rel. in [%]) critical load change as indicator for the spread around the mean critical load change. A further parameter out of this direct comparison is the coefficient of determination (R^2) of the regression.

Table DE.4 Influence of precipitation surplus (PS), deposition of base cations (BCdep) and uptake of nitrogen/base cations (Nu/Bcu) on the change of calculated critical loads. Table shows calculated results for the slope of the linear regression between absolute change of input parameter and critical load (Slope), the coefficient of determination (R^2) and the standard deviation (absolute/relative) of the change of critical load results (SD (abs./rel.)) for the critical load of eutrophication (CLnutN) and acidification (CLmaxS).

	CLnutN				CLmaxS			
	Slope	R^2	SD abs. [eq ha^{-1} a^{-1}]	SD rel. [%]	Slope	R^2	SD abs. [eq ha^{-1} a^{-1}]	SD rel. [%]
PS	2.5	0.98	266	8.9	0.5	1.00	64	6.0
BCdep	0.0	1.00	0	0.0	1.5	0.97	234	75.3
Nu/Bcu	1.0	0.99	138	12.0	-1.2	0.91	374	49.6
All		0.97	305	13.5		0.90	403	253.1

The results in Table DE.4 are based on the analysis of all five runs and give indication about impact strength and direction of the different input parameters (single and as combination) as well as the relative and absolute impact on the critical load.
The PS is positively correlated with the CLnutN and CLmaxS, which was expected because the amount of leaching water determinates the amount of accepted nitrogen leaching (Nle(acc.)) and the ANCle. The data also indicates, that the effect on CLnutN is stronger (see slope) and is more scattered (see SD (abs./rel.) and R^2) than the effect on CLmaxS. The reason for these differences is caused by differences in the equations for the Nle and ANCle within the calculation of the CLnutN and the CLmaxS, respectively.
The BCdep has of course no effect on the CLnutN and a strong positive effect on CLmaxS which is easy to comprehend. The more base cations are available the more acid neutralization potential has an ecosystem. The changes in the BCdep have rather small impact in absolute numbers (see SD abs.) but quite high impact on the relative change of the critical load (see SD rel.). This might be an indication for higher sensitivity to changes of BCdep on sites with rather low critical load for acidification.

The effect of a changing site growth potential (Nu and Bcu) is bidirectional. While the effect on CLnutN is positive, the effect on CLmaxS is the opposite and seems to be a bit stronger. Again this was anticipated since higher uptake of nitrogen means higher site potential of nitrogen fixing. On the other hand higher uptake of base cations means less site potential of acid neutralization.
Comparing the strength of the different parameters it seems that the PS has the highest impact on the CLnutN (Slope = 2.5). On the other hand the PS has the lowest impact on the CLmaxS, while the deposition of base cations (BCdep) has the highest influence (Slope = 1.5).
Applying changes on all selected input parameters show the highest scatter (SD and R^2). Especially the relative impact on the CLmaxS (see SD rel.) is remarkable and gives an indication for increased sensitivity to changes of the input parameters on sites with low critical load. An overall trend for the recent critical load dataset (e.g. generally higher/lower than the previous one) was not detected. The trend varies from region to region since not only average numbers of the input parameters changed but also the spatial pattern within Germany.

Conclusions
The critical load approach offers a number of tools to parameterize biodiversity targets. Obviously, the determination of the protection aim is the most crucial part. This report proposes a method combining ecological niches of 226 German plant communities with specific limits of soil properties (C/N ratio, base saturation) to ensure high vitality and sustainability of these site specific reference species compositions. These specific limits are used to calculate critical load for biodiversity, which are generally more sensitive. Uncertainties of this approach lay in (a) the generalized approach of attribution of soil parameters to vegetation data within the BERN database. Several relevés are combined only with verbal descriptions of the site factors, therefore values for soil chemical and climate parameters were assigned from similar sites. (b) Secondly uncertainties lay in the generalized approach of attribution of vegetation communities to land cover and soil maps. However the approach can be seen as a first step to map broad-scale biodiversity critical loads. If it is a valuable approach for integration into integrated assessment modelling, has to be proven yet.

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Introduction
The *updating* and renewing of the critical load database by considering the need to adapt it to the change of the EMEP grid from 5x5km cells into the new 0.1° x 0.05° longititude-latitude grid have been the work of this year.
All parameters requested were estimated in the new sampling size by selecting them from the raster layers by using the new grid and average values for each of the EUNIS categories.

Data sources
Table IT.1 show detailed information about data included in the re-analysis, for all parameters included in the critical loads estimation.

Table IT.1 Parameters for the Critical Loads, the calculation, their sources and algorithms

Variable	Explanation	Units	Source	Algorithm
SiteID	Identifier of the site (see ecords Table)			
CLminN	Minimum critical load of nitrogen	eq/ha/a		CLmin(N) = Ni + Nu
CLmaxS	Maximum critical load of sulphur	eq/ha/a		CLmax(S)=BCdep-Cldep+BCw-BCu-Alkle
CLmaxN	Maximum critical load of nitrogen	eq/ha/a		CLmax(N)=CLmin(N)+CLmax(S)/(1-f_{de})
ClnutN	Critical load of nutrient nitrogen	eq/ha/a		CLnut(N)=Ni+Nu+Nde+Nfire+Neros+ Nvol+Nle-Nfix
cNacc	Acceptable (critical) N concentration for CLnutN calculation	meq/m^3		Nle(acc) = Q×[N]acc

Variable	Explanation	Units	Source	Algorithm
thick	Thickness (root zone!) of the soil	meq/m^3	ESDB	
bulkdens	Average bulk density of the soil	g/cm^3	ESDB	
nANCcrit	The quantity-ANCle (crit)	eq/ha/a		
Cadep	Total deposition of calcium	eq/ha/a		
Mgdep	Total deposition of magnesium	eq/ha/a		
Kdep	Total deposition of potassium	eq/ha/a		
Nadep	Total deposition of sodium	eq/ha/a		
Cldep	Total deposition of chloride	eq/ha/a		
Cawe	Weathering of calcium	eq/ha/a		Naw×2.3
Mgwe	Weathering of magnesium	eq/ha/a		Naw×0.9
Kwe	Weathering of potassium	eq/ha/a		Naw×0.6
Nawe	Weathering of sodium	eq/ha/a		BCw×0.3
Caupt	Net growth uptake of calcium	eq/ha/a		Bcu %
Mgupt	Net growth uptake of magnesium	eq/ha/a		
Kupt	Net growth uptake of potassium	eq/ha/a		
Qle	Amount of water percolating through the roof zone	mm/a		
lgKAlox	Equilibrium constant for the Al-H relationship (log10) (var. formaly known as Kgibb			
expAl	Exponent for the Al-H relationship (=3 for gibbsite equilibrium)			
cOrgacids	Total concentration of organic acids (m*DOC)	eq/m^3		
Nimacc	Acceptable nitrogen immobilised in the soil (eq/ha/a			
Nupt	Net growth uptake of nitrogen	eq/ha/a		
fde	Denitrification fraction (0≤fde<1)(-)			
Nde	Amount of nitrogen denitrified	eq/ha/a		
Slope		°	GIS	

Variable	Explanation	Units	Source	Algorithm
Aspect	Angle between North and the perpendicular line of slope (degrees up to 360°, measuring clockwise) (°)		GIS	
Altitude	Above sea level	M		
Prec	Precipitation	mm/a		
TempC	Temperature	T°		
Theta	Water/moisture content	m^3/m^3	ESDB	
Corg	Organic carbon content (%)		ESDB	
sand	% sand in soil		ESDB	
clay	% clay in soil		ESDB	
bsat	Base saturation (-)		ESDB	
Cpool	Amount of carbon in topsoil	g/m^2	ESDB	
CNrat	C/N ratio in topsoil	g/g	ESDB	

Many local to regional studies have shown that chronic N deposition leads to a shift in the plant species composition of the forest floor and eventually to diversity loss. Actually, biodiversity indices are to be estimated for some Italian test-sites in order to verify a direct or an indirect relationship between nitrogen critical loads and plant diversity at the herbaceous level. Preliminary results highlighted very different correlations among different biodiversity indices and N critical loads and absence of relationships in some test-sites in central Italy.

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Introduction

Nitrogen deposition in the Netherlands, which often exceeds the critical loads, is a large threat to protected habitats and species. Environmental policies follow a two way approach to reduce problems due to critical load exceedances. On the one hand there is the international policy to reduce emissions at the international level as stated in the LRTAP Convention. On the other hand there is a Dutch Programmatic Approach to Nitrogen (PAN; Ministerie van Economische Zaken & Ministerie van Infrastructuur en Milieu, 2015) to reduce emissions on national, sub-regional and local levels, give or reject permits for plans and projects which may influence emissions, and to take restoration measures in sensitive Natura 2000 areas. Both policies use critical loads for nitrogen which are based on calculations with SMART2 combined with empirical critical loads (Van Dobben et al, 2006; Van Dobben et al., 2014).
In response to the 2014-2015 Call for Data, the Dutch NFC delivered an updated map of critical loads for use under the LRTAP convention, calculated by VSD$^+$, on a grid of 250 x 250 m^2. Input parameters depend on soil type (soil physical parameters, organic matter, CEC, base saturation and weathering) and vegetation type (nutrient uptake, litter production). Precipitation, upward seepage and base cation deposition were location specific. The calculated critical loads were compared with the critical loads used in the Dutch PAN.
Besides, the Dutch NFC has tested a method to calculate critical loads for dry heath with critical limits derived with the PROPS model, based on the Habitat Suitability Index (HSI) described in the call-for-data.

Critical load map NL

The updated map of critical loads for the Netherlands is calculated with VSD$^+$, instead of previously used SMART2 model. The critical conditions are based on protection of plant associations of nature target types against eutrophication and too much acidification. Nitrogen availability and pH were used to describe these critical conditions (Van Hinsberg and Kros, 2001). The calculated critical load is the maximum N deposition where the most strict (binding) critical condition is met (either maximum N availability or minimum pH).

Model input
Litter fall and N seepage are considered to be the most important input parameters which influence the critical load in situations where N availability is the binding condition. N in seepage water is taken from a map (Pastoors, 1993; Bolsius et al., 1994). Litter fall is calculated with SUMO (Wamelink et al., 2009) and used for calibration. Compared to default input, extra litter fall in forests is assumed due to litterfall from ground vegetation and litter fall in selected (managed) grasslands is reduced because the default values only applies to wet, productive, grasslands and not to e.g. dune grasslands and other poor systems.

In cases where pH is the binding condition, critical loads are more affected by input and output fluxes. Most important input fluxes are deposition, weathering, upward seepage and mineralisation. The main output fluxes are downward seepage and net uptake. A yearly water balance is needed to calculate the seepage fluxes. Deposition (of base cations and Cl) and rainfall are regionally variable and described in maps. Other parameters are considered to vary per soil type (CEC, base saturation, exchange constants, weathering rates), vegetation type (mineralisation rates, nutrient contents) or a combination of both (transpiration).Compared to the former critical load map delivered to the CCE, weathering rates have been improved for löss. Upward seepage was assigned per nature target type.

Results
A database with critical loads on 250 m x 250 m gridcells was delivered to the CCE. In Table NL.1, a comparison is made between the average critical load for nitrogen in the CCE database, the empirical ranges and the critical loads used in the Dutch policy (PAN). The critical loads in the database correspond quite well with the PAN critical loads for the given the habitat types and they are all within the empirical ranges. For some habitat types however (data not shown), the results are less consistent with the PAN, which indicates the limitations of the current database for local analyses. The reasons for the difference between the critical loads in the CCE database and the PAN critical loads lies in the calculation method itself, model input and critical conditions. Whereas the critical loads of the CCE database are all calculated with VSD+ alone, the PAN critical loads are a combination of empirical critical loads and calculations. Moreover, the calculated critical loads for PAN were computed with an optimization routine in SMART2 to find the N deposition where the maximum N availability was not exceeded and the pH was not too low, whereas the calculation with VSD+ is really a steady state calculation from condition to critical load. So the calculation method itself is different which causes different results. In addition, the model input for the CCE calculations is partly considered local variable (rainfall and base cation deposition), whereas the PAN critical loads were calculated for average conditions. Additional effort is needed to further tune model input and both databases. It is also important to harmonize the maps of plant associations, i.e. the maps of habitat types protected in the PAN and the maps of nature targets used for calculation of the CCE database.

Table NL.1 Critical loads in database CCE compared with empirical ranges and critical loads used in Dutch policy (PAN)

Type	Habitat type	Critical load (kg N ha^{-1} yr^{-1})		
		Database CCE	Empirical range	PAN
Bogs	H7110	6	5-10	7
Dune grasslands	H2130	9	8-15	10
Dry heathland	H4030	15	10-20	15
Salt marches	H1330	29	20-30	22
Dry nutrient-poor forest	habitat for protected animals	16	10-20	15

Modelling CLs using HSI
The protected habitat types in the Habitat Directive are in the Netherlands formally described in terms of abiotic ranges, lists of typical species and lists of desired plant associations. The abiotic ranges for habitat types are however only broadly defined, with terms like 'nutrient poor' and 'medium acid'. For calculations of critical loads a stricter and clearer definition is needed. By combining PROPS with lists of typical species and lists of species belonging to desired plant associations, such stricter and quantitative conditions can be defined.

Method
For the habitat type dry heath (H4030) we have calculated critical loads with critical conditions derived from the Habitat Suitability Index (HSI), calculated with PROPS. Critical loads were computed for three different selections of species: one with all wanted species in H4030 (all typical species and species belonging to plant associations with good quality), one with the typical species listed in the habitat description, including mosses and lichens, and one with typical species according to Schaminée et al. (2011).
All sets of species show optima at low NO_3 contents and at a pH roughly between 3.5 and 5.5 (Figure NL.1). The species set of Schaminée has the highest probability (isoline with highest value of 0.5 instead of 0.3 for the other two sets). Based on these figures, several combinations of critical pH and NO_3 contents on the isoline HSI=0.3 have been selected to calculate critical loads. To calculate critical loads, NO_3 content (mg kg^{-1}) had to be converted to N-concentration (eq m^{-3}) using:
N-conc = $NO_3 * \rho / \theta / 62$
Where *N-conc* is the critical N-concentration in soil solution (eq m^{-3}), *NO3* is NO_3-content (mg kg^{-1}) read from the isolines graph, ρ is bulk density, θ is soil moisture content. Bulk density and soil moisture content were set at resp. 1.416 g cm^{-3} and 0.148), being default values for this soil type. With this conversion an uncertainty is introduced, because the critical limit is linearly correlated with bulk density and soil moisture content, and both these soil parameters are somewhat uncertain.

Results
Figure NL.1 shows the critical loads for nitrogen for the selected combinations of NO_3 and pH on the isoline of HSI=0.3. The critical loads,

i.e. the N deposition where the most strict (binding) critical condition is met (either NO_3 content or pH), are depicted in the various boxes in this figure. Different species selections result in different patterns of isolines and thus different critical limits.
For the selected combinations of critical pH and NO_3, the critical loads vary between 6 and 9 kg N ha^{-1}, except for one point on the isoline with a pH of 5.6: such a high pH can only be reached with very low depositions. The various calculated critical loads with this method at HSI=0.3 are close to the lowest value of the empirical range but lower than the critical load in PAN and the average critical load in the database (see Table NL.1). The question is, however, which list of species should be considered and which HSI should be used for the calculation of critical loads. Mosses and lichens should, may be, be excluded or treated separately, since they are affected by pH and NO_3-contents from shallower soil depths than the soil depth where plant roots grow. Also a HSI of 0.3 might be too low to fully obtain optimal conditions for *all* desired species.

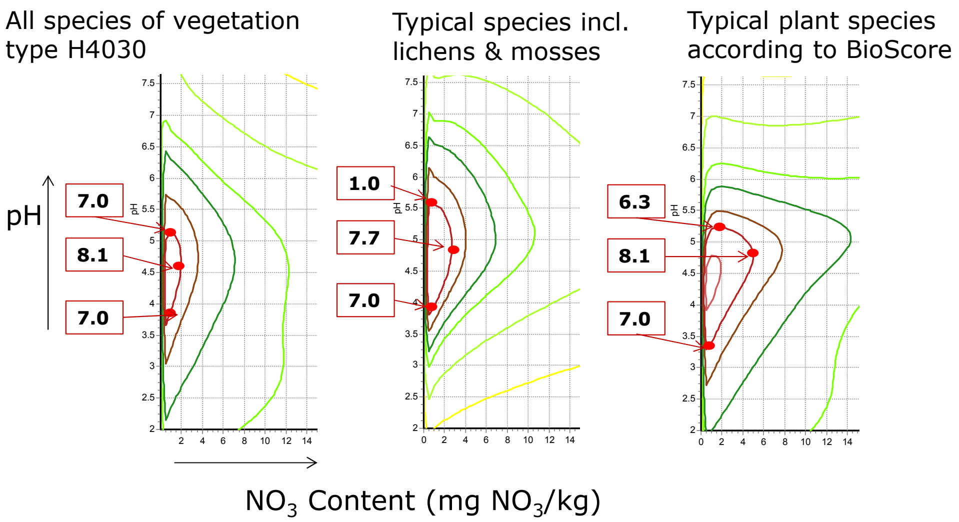

Figure NL.1 HSI isolines for three selections of species for vegetation type H4030 with calculated critical loads (kg N ha-1 yr-1) belonging to the dots on the isoline HSI = 0.3.

Conclusions

- A more detailed CL-map for EU-wide scenario analyses is now available for the Netherlands. However, this current map is not appropriate to draw site specific conclusions, since the critical loads do not always correspond with local site conditions.
- More work is needed to improve the correspondence of the CL map with local information. The improvement will be focussed on consistency of input maps (habitat maps) and calibration of system in- and outputs like litter fall and uptake.
- Since the current critical limits for habitat types are defined broad and not quantitative, the HSI bases modelling might be a fruitful way to go. It is recommended to define a common list of species per habitat type to make the results between different areas or countries comparable. An automatized procedure is needed to apply this technique on a larger scale.

- For the investigated habitat type, the HSI based critical loads were close to the empirical range. Uncertainties in HSI based method are caused by the conversion from NO_3 content to critical N concentration, the critical value for the HSI used, and the selection of species.

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Methods and data

Norway has updated the critical loads to fit with the new 0.10°×0.05° longitude-latitude grid, according to the Call for Data 2014/15. Minor modifications have also been made to the calculation method for critical loads of acidity for surface waters. Norway has not developed biodiversity critical loads, and no changes have been done to the dynamic modelling. In connection with the Call, Sweden and Norway have compared the calculation methods for critical loads for surface waters (see separate report under the national report from Sweden).

Critical loads for surface waters

The database for critical loads for surface waters is based on a 0.25°×0.125° longitude-latitude grid (Henriksen 1998). The chemistry of surface water within a grid cell was set by comparing available water chemistry data for lakes and rivers within each grid cell. The water chemistry data were primarily results from the national lake survey conducted in 1986 (Lien et al. 1987). The chemistry of the lake that was judged to be the most typical was chosen to represent the grid cell. If there were wide variations within a grid cell, the most sensitive area was selected, if it amounted to more than 25% of the grid cell area. Sensitivity was evaluated on the basis of water chemistry, topography and bedrock geology. Geology was determined from the geological map of Norway (1:1 million) prepared by the Norwegian Geological Survey (NGU). The critical loads of the original grid were assigned to the new 0.10°×0.05° longitude-latitude grid without further data collection. The mid-point critical loads values of the new grid cells were used as critical load for the entire grid cell. When the mid-point was at the border between two original grid cells or at the corner of four original grids cells, the average critical load of the original grid cells in question was used.

The methodology for Norway was described by Henriksen (1998) and the application later updated in Larssen et al. (2005; 2008). A variable ANClimit as described by Henriksen and Posch (2001) is used, but adjusted for the strong acid anion contribution from organic acids after Lydersen et al. (2004). [BC]0* was originally calculated by the F-factor approach, using the sine function of Brakke et al. (1990), but in recent applications [BC]0* has instead been estimated from MAGIC model (Cosby et al. 1985; Cosby et al. 2001) runs used for calculating target loads (Larssen et al. 2005). Here MAGIC was applied to 131 lakes in Southern Norway, of which 83 lakes were acidified (ANC < the variable ANClimit). A linear regression of MAGIC modelled [BC]0* ([BC]1860*) vs [BC]1986* for these 83 lakes is used to estimate [BC]0* for each grid cell. For the current call, a minor error in the regression was corrected, and potassium was included in BC, which has not traditionally been done.

Nitrogen removal in harvested biomass was estimated by Frogner et al. (1994) and mapped for the entire Norway according to forest cover and productivity. Nitrogen immobilisation was kept constant at 0.5 kg N a^{-1} (CLRTAP 2004). The de-nitrification factor (fde) was kept constant at 0.1 and the fraction of peat in the catchments ignored in the national scale applications. Mass transfer coefficients were kept constant at 5 m a^{-1} and 0.5 m a^{-1} for N and S, respectively and chosen as the mid-value of the ranges proposed by Dillon and Molot (1990) and Baker and Brezonik (1988), respectively. Mean annual runoff data were taken from runoff maps prepared by the Norwegian Water Resources and Energy Directorate (NVE). The lake to catchment area was set constant to 5%.

Dynamic modelling of surface water acidification

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, sulphate 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). 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 in the Norwegian critical loads database (2304 cells) using a matching routine called "MAGIC library" (IVL 2015) (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. Input data and data sources are described in the CCE Status Report 2008 (Hettelingh et al. 2008).

Empirical critical loads for nitrogen
The vegetation map of Norway was updated with the new empirical critical loads from the "Workshop on the review and revision of empirical critical loads and dose-response relationships" (Bobbink and Hettelingh 2011) in 2011 (see CCE Status Report 2011 (Posch et al. 2011)). The empirical critical loads map was overlaid with the new 0.10°×0.05° longitude-latitude grid. In 2011 the mid-point values of the grid cells were used as empirical critical load values for the cells. For the 2014/15 call, empirical critical loads are reported for each ecosystem type within the grid cells (ecords).

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Introduction

In response to the CCE "call for data 2014-15", the Polish NFC submitted an updated critical loads of CLmax(S), CLmin(N), CLmax(N), CLnut(N) and CLemp(N) as well as input parameters for their calculation, in following tables:

- Table 1 "ecords"
- Table 2 "CLacid"
- Table 3 "CLnut"
- Table 4 "CLemp"
- Table 6 "SiteInfo"

Critical loads were calculated for six terrestrial habitats types, identified according to the EUNIS classification as: (G1) broad-leaved forests, (G3) coniferous forests and (G4) mixed forests, (D) mire, bog and fen habitats, (E) grasslands and (F) heathland, scrub and tundra habitats. The Table 5 "CLbdiv" was not calculated and submitted.

New EMEP grid for CLs calculations in Poland

The new grid for CLs calculations for Polish ecosystems was prepared according to new CCE grid, based on 0.1° x 0.05° longitude latitude EMEP grid. The final spatial resolution for Polish ecosystems was set on 0.02° x 0.01° what gave grid dimensions from (long x lat): 1.113 km x

1.285 km in Northern Poland to 1.113 km x 1.626 km in Southern Poland.
The CLC2006/EUNIS-SEI ecosystems database was used for establish ecosystems spatial range, type and area within the calculation grids. Final database covered 95191.870 km^2 of ecosystems area, with one or more habitats in each grid cell and contains 224358 records (for EcoArea larger than 1 ha).

Table PL.1 Ecosystem database for Poland

EUNIS code	EUNIS habitat name	Area [km^2]
D	Mire, bog and fen	1 035.326
E	Grasslands and tall forbs	329.738
F	Heathland scrub and tundra	33.775
G1	Broad-leaved forests	14 316.940
G3	Coniferous forests	56 693.539
G4	Mixed forests	22 782.552
Total		95 191.870

Critical Loads of Acidity
Critical loads of acidity calculations were based on the SMB model as it was described in CLRTAP Manual [CLRTAP 2004].
The properties and spatial distribution of soils was obtained from European Soils Database (ESDB), with some supplementary data taken from the ICP Forest II-level monitoring system. Precipitation and temperature data was derived from New et al. 2002.
The base cation depositions were obtained from national monitoring stations (5 year averages) and spatially distributed. Chemical criterion used for CL of acidy calculations was: molar [Bc]:[Al].
Above procedure was previously used for CLmax(S) calculations in Poland [Pecka et al. 2013].
Average values of calculated CLmax(S) by EUNIS ecosystems are shown in Table PL.2. Spatial distribution of CLmax(S) is presented in Figure PL.1.

Critical Loads of Eutrophication
Critical loads of eutrophication calculations were based on the SMB model as it was described in CLRTAP Manual [CLRTAP 2004].
Nitrogen uptake was obtained from State Forest Inventory as forest biomass (stems and branches) removed from forest ecosystems.
The acceptable nitrogen leeching (Nacc) was calculated with data establish in Sweden and the Netherlands (Table 5.7 from CLRTAP Manual, as updated in 2007). For the lower threshold value of the growing season, Nacc empirically determined in Scandinavia were used while for the upper threshold Nacc reported for the Netherlands were taken. The values of Nacc between the both threshold values of growing season were calculated for considered ecosystems using simple linear functions. The growing season length for Poland was calculating according to the Huculak and Makowiec (1977) method and basing on the detailed meteorological data for the period of 6 years.

Above procedure was previously used for CLnut(N) calculations in Poland [Pecka et al. 2013].
CLnut(N) values should be used as eutrophication risk indicators for simulations in GAINS model.
Average values of calculated CLnut(N) by EUNIS ecosystems are shown in Table PL.2. Spatial distribution of CLnut(S) is presented in Figure PL.2.

Critical Loads of Nitrogen
For calculations of Empirical Critical Loads of Nitrogen information provided in the "Review and revision Empirical Critical Loads and dose-response relationship" [Bobbink et al. 2011] were used.
The lower and upper limits of CLemp(N) for each EUNIS classes were calculated with modifying factors – precipitation, temperature and base cation availability. Modifying factors for each grid were obtained from the cumulative distribution functions (CDFs) calculated for each EUNIS class for ecosystems in Poland, based on Polish CL input parameters database. Above procedure was previously used for CLemp(N) calculations in Poland [Pecka et al. 2011].
Average values of calculated CLemp(N) by EUNIS ecosystems are shown in Table PL.2. Spatial distribution of CLemp(S) is presented in Figure PL.3.

Table PL.2 Average values of calculated critical loads for terrestrial ecosystems in Poland

EUNIS code	CLmax(S)	CLnut(N)	CLemp(N)
D	2203.6	192.8	567.4
E	2250.3	1289.0	1771.4
F	2430.0	3271.5	728.0
G1	1034.0	1270.0	1129.8
G3	1323.6	774.8	780.4
G4	1213.0	1071.2	980.0

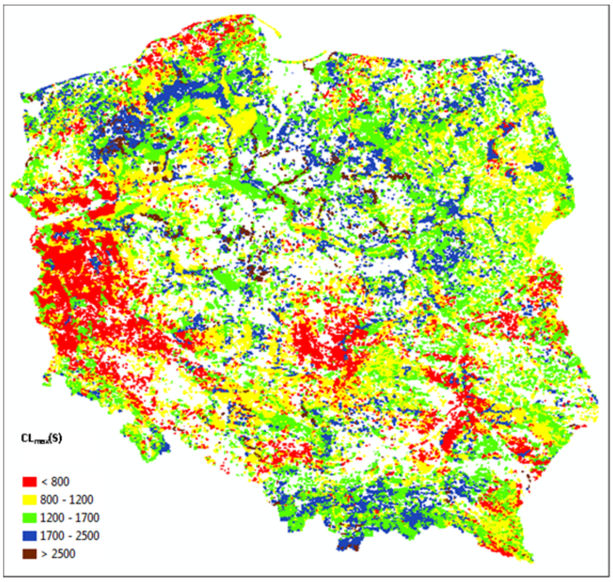

Figure PL.1 Spatial distribution of CLmax(S) values for terrestrial ecosystems in Poland.

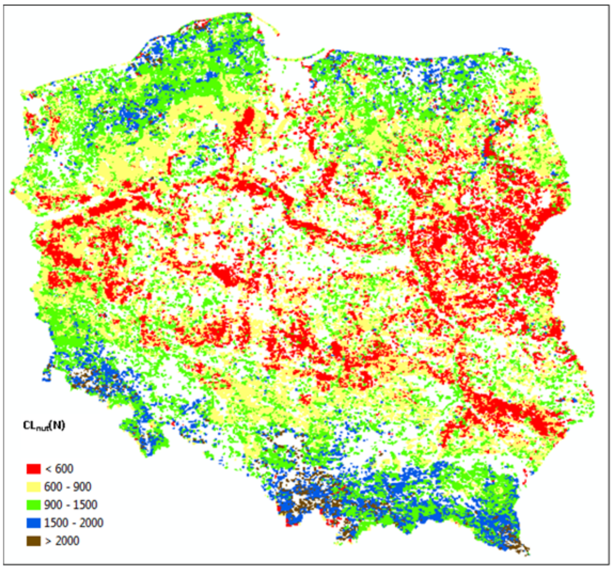

Figure PL-2 Spatial distribution of CLnut(N) values for terrestrial ecosystems in Poland.

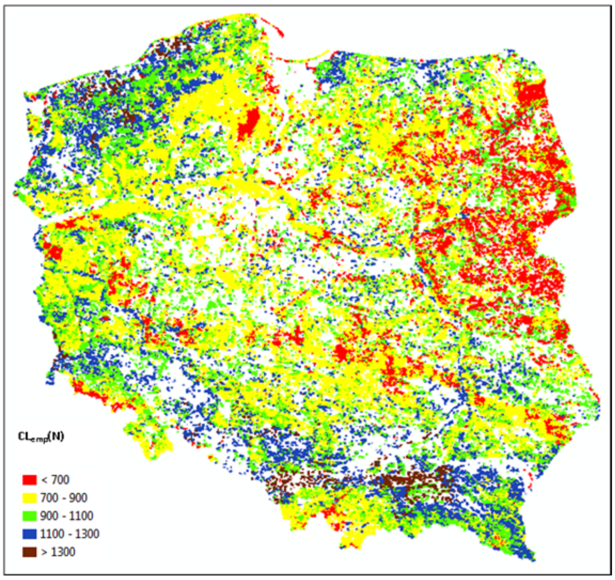

Figure PL.3 Spatial distribution of CLemp(N) values for terrestrial ecosystems in Poland.

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Summary

A Call for Data 2014-2015 on critical loads and biodiversity indicators was adopted by the Working Group on Effects at its 33rd session in Geneva in September 2014 and later issued by the Coordination Centre for Effects under ICP Modelling and Mapping with a delivery deadline of March 2015.

The aims of the Call for Data were: (1) to adapt the critical load database to the new longitude-latitude grid to ensure compatibility with EMEP depositions; (2) a possibility for the NFCs to update their national critical load data on acidity and eutrophication; (3) to apply novel approaches to calculate nitrogen and sulphur critical load functions taking into account their impact on biodiversity.

The Swedish NFC response answers to points 1 and 2 above. Our response consists of a re-gridding of the previously (2014) reported critical loads for acidity and re-gridding empirical critical loads established in 2014 by Swedish habitat experts at 3798 Swedish Natura 2000 sites. For acidity the calculations are based on lakes and apply for both lakes and their catchments, in the same way as in data submissions in 2012 and 2014. A database with the results of the new calculations is submitted simultaneously.

After the data submission in 2012, the Swedish and Norwegian NFCs realized a shift in exceedance of critical loads for acidity running along the Swedish-Norwegian border (Posch et al., 2012). The Swedish ecosystems appeared generally more sensitive than Norwegian ecosystems in the same geographical region. The two NFCs teamed up to compare the methodologies applied in respective country to explain the difference. Three key differences in the critical loads calculations were identified. The Swedish calculations considered more intense forestry practices than the Norwegian, making the Swedish forest ecosystems more sensitive. This has been re-evaluated already in the 2014 data submission and both 2014 and 2015 data submissions take

into account future forest harvesting on a lower level than the 2012 submission. The second main difference lays in the choice of target lake water alkalinity expressed as ANClimit. In Norway ANClimit is set to protect fish population while in Sweden it is set to protect also littoral invertebrates. Both the Swedish and Norwegian ANC criteria relates target ANC to estimated pre-historical ANC, however the Norwegian target ANC is never set higher than 50µeq/l while the Swedish target ANC could be much higher than that for lakes with historically high ANC. The third difference is in the application of the precautionary principle with regards to future nitrate leaching where Norway considers smaller long term immobilisation of nitrogen than Sweden. Several aspects of the different approaches are discussed in the joint Swedish-Norwegian report (see the annex to this national report).

Introduction
In Sweden the impact of air pollution on ecosystems is of major concern, both with respect to acidification and eutrophication of soils and waters. In response to the Call for data Swedish NFC re-gridded critical loads for acidity on lakes and empirical critical loads at Natura 2000 areas the $0.10^0 x 0.05^0$ degrees longitude and latitude grid. The submitted critical loads reflect our view on acceptable level of air pollution which – if not exceeded – provides sufficient level of protection of Swedish ecosystems from harmful effects of acidification and eutrophication due to N deposition. Due to that and due to limited availability of resources, the response does not also answer the part of the Call concerned with establishing critical loads based on biodiversity change.

Critical loads for acidity
In 2014 Sweden revised the calculations of critical loads for acidity in surface waters (Slootweg et al., 2014). In the current submission the same calculations are re-submitted in the new geographical grid. Relative to the 2014 submission, ecosystem area of each grid was re-assessed. Hereby submitted ecosystem area (ECOarea) is reduced by excluding of nine largest Swedish lakes (same as in 2014) along with densely populated areas and agricultural land. Thus the area assessed for critical loads of acidification (395 226 km^2) is ca 88% of the total area of Sweden (449 964 km^2).
Critical loads for acidity are based on calculations at 5084 lakes as described in CCE Status Report 2014 (Slootweg et al., 2014). For the grid cells with no assessed lakes in it we have used inverse distance weighting interpolation (IDW). IDW determines cell values using a linearly weighted combination of a set of sample points. The weight is a function of inverse distance. This method assumes that the variable being mapped decreases in influence with distance from its sampled location. Between 3 and 10 lakes within 30 km radius were considered for interpolation for each grid. For the grid cells with assessed lakes in it we have used the critical loads at these lakes. Geographical distribution of the areas most sensitive to acidification follow the same pattern as observed in previous CL submissions.
Re-gridding and slight adjustment of the ECOarea (see above) did not have a major impact on exceedance of critical loads. Preliminary calculation showed exceedance at about 8% of the considered ECOarea

compared to 9% based on 2014 critical loads submission (Slootweg et al., 2014).

Empirical critical loads
Empirical critical loads established at 82 habitats represented in 3798 Natura 2000 areas covering 58 688 km2 (Figure SE.1) were re-gridded to the new coordinates without any other adjustments compared to the 2014 submission (Slootweg et al., 2014). Remaining 273 Natura 2000 areas of the total 4071 were not relevant in this context (caves, large lakes, marine ecosystems etc.).

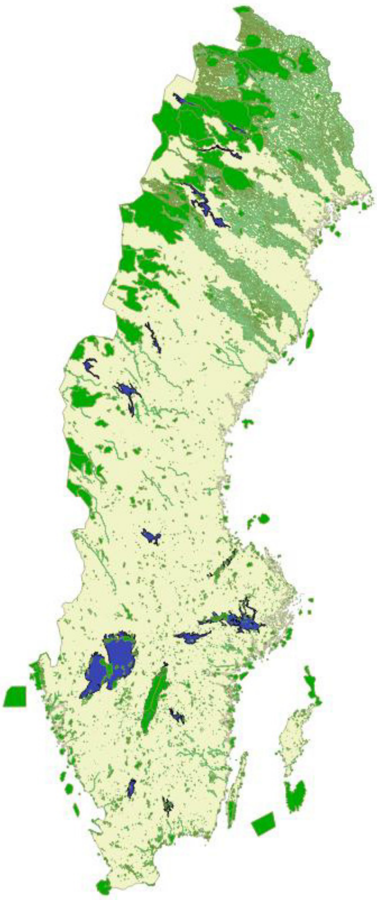

Figure SE.1. Map of Sweden showing geographical location of Natura 2000 areas in green.

Comparison of critical load methods for freshwaters in Norway and Sweden[3]

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Background
From the critical load maps for Europe produced by the Coordination Centre for Effects (CCE) in the 2012 Status Report (Posch et al., 2012), it was apparent that there was a systematic difference between Sweden and Norway. The critical loads of acidity for surface waters in Sweden were lower than those for Norway, and the border between the two countries showed clearly on the European map (Figure C.1).

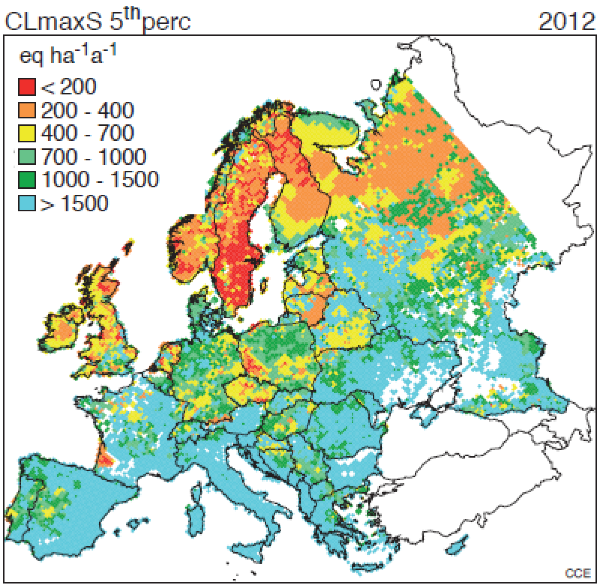

Figure C.1. Map of critical load of acidity (5th percentile of CLmaxS) for in Europe. Note the step change at the border between Norway and Sweden. Units here are eq/ha/yr. 100 eq/ha/yr = 10 meq/m^2/yr. Source: CCE Status Report 2012 (Posch et al., 2012).

The difference in critical loads was most likely due to the different methods used in Sweden vs Norway, as there is no inherent reason why a critical load should be different simply by crossing the border. If anything, critical loads would be expected to be lower in Norway, where

[3] This part of the Swedish national report has been produced as a joint effort between the Swedish and Norwegian National Focal Centres for ICP Modelling and Mapping.

the surface waters are generally more sensitive to acidification due to thinner, less well buffered soils and lower weathering rates. The study reported here is thus an examination of the two methods, their differences and the policy implications for critical loads in Sweden and Norway.

Both sulphur (S) and nitrogen (N) contribute to acidification of surface waters. The countries submit to the CCE three values that enter into the critical load calculations – CLmaxS (the maximum amount of S that can be tolerated given no N deposition), CLminN (sum of N sinks such as immobilisation, uptake and sedimentation), and CLmaxN (the maximum amount of N that can be tolerated given no S deposition). In Sweden and Norway S deposition plays a much more important role for acidification than N. Even though leaching of N to surface waters is currently minor (with a correspondingly small contribution to acidification), the critical load considers the risk of future leaching and therefore the deposition of N is for critical loads calculations as important as deposition of S. For illustration this study focussed on the two different methods for calculating CLmaxS in Norway and Sweden. The work was conducted jointly by the Swedish focal centre at IVL, the Norwegian focal centre at NIVA and the Swedish University of Agricultural Sciences (SLU) with support from the Swedish National Environmental Protection Agency and the Norwegian Environment Agency.

Methods to calculate critical loads

The Swedish method

The critical loads were calculated using the first-order acidity balance (FAB) model (Henriksen and Posch 2001) with the following modifications:

The chemical threshold, ANClimit, was calculated individually for each lake to a value corresponding to a decrease in pH of 0.4 units from reference conditions (i.e. the year 1860) calculated by the dynamic acidification model MAGIC (Moldan et al., 2013). This criterion is derived from empirical data for sensitive fish populations and littoral invertebrates (Fölster et al. 2007). A delta pH > 0.4 is considered “unacceptable biological damage” and is used for classification of ecological status in Sweden (Naturvårdsverket 2007).

The reference leaching of base cation concentration (BC^*_0) used in the FAB-model was the calculated BC* concentration (* = sea-salt corrected) in year 2100 simulated by MAGIC given the emission scenario “current legislation” (CLE) as provided by CCE. The F-factor was not used to estimate the weathering rate. The year 2100 was used instead of 1860 for steady state since modelling indicated that it will not be possible to reach the BC* concentration of 1860 even with a total reduction of acidifying deposition.

Instead of assuming a fixed N immobilisation by the soil ecosystem, calculations of N immobilisation were based on Gundersen et al (1998). Excess N deposition was calculated as deposition minus forest N uptake. Nitrogen immobilisation was set to 100% for excess deposition up to 2 kg N/ha, 50% for the fraction between 2 and 10 kg N/ha and 0 % for the excess deposition above 10 kg N/ha. In addition to this, leaching of organic nitrogen calculated from the lake concentration of total organic nitrogen (TON) was regarded as non-acidifying.

The critical loads were calculated for 5084 lakes within the national lake survey program (Fölster et al. 2014). For each lake, data for $BC*_0$ (2100) and delta ANC were obtained from the most similar lake with a MAGIC simulation (Moldan et al. 2013) using the tool "MAGIClibrary" (www.ivl.se/magicbibliotek). ANC1860 was calculated as ANCt + dANCMAGIClibrary and pH1860 was calculated from ANC1860 by using the model of Hruska et al. (2003) for organic acids and assuming that total organic carbon (TOC) has been constant over time. Finally the ANClimit was calculated from pH1860 – 0.4 according to the criterion for acidification.

The Norwegian method

The methodology for Norway was described by Henriksen (1998) and the application later updated in Larssen et al (2005; 2008a). The FAB model was applied, to a large extent in line with Henriksen and Posch (2001). The main deviation was in the calculation of the pre-industrial base cation concentration ($BC*_0$), which was not calculated using the F-factor, but by a regression of present against pre-industrial BC* derived from the MAGIC model (Cosby et al. 1985; 2001). As base cation concentration, the sum of calcium, magnesium and sodium concentration was used.

A variable ANClimit was used, where ANClimit was calculated as a function of $BC*_0$, with higher ANClimit for lakes with higher $BC*_0$. The ANClimit varies between 0 and 50 μeq/l, where the lower boundary is conceptual (it is assumed that at a critical load of zero, the ANClimit is zero), and the upper boundary is based on potential damage to fish populations (no damage to fish populations at ANC above ca 50 μeq/l (Lien et al., 1996)). The variable ANClimit was also adjusted for organic acids (see below).

Nitrogen removal in harvested biomass was estimated by Frogner et al. (1994) and mapped for the entire Norway according to forest cover and productivity. Nitrogen immobilisation was kept constant at 0.5 kg N yr^{-1} (CLRTAP, 2004). The denitrification factor (fde) was kept constant at 0.1 and the fraction of peat in the catchments ignored in the national scale applications. Mass transfer coefficients in the lakes were kept constant at 5 m yr^{-1} and 0.5 m yr^{-1} for N and S, respectively and chosen as the mid-value of the ranges proposed by Dillon and Molot (1990) and Baker and Brezonik (1988), respectively.

Modifications to the calculation methods

To make outcomes from Swedish and Norwegian methods comparable, both methods were modified relative to the methods actually used when officially reporting critical loads. Adjustments were as follows:

For the official reporting of critical loads Norway applies an organic acid adjustment to the variable ANClimit (Hindar and Larssen, 2005). The critical loads thus calculated are directly comparable to other critical loads, but the organic acid adjusted ANClimit cannot be directly compared to other ANClimits, as it is not actually an ANC value. Hence, to be able to compare also the individual elements of the critical loads calculation, the organic acid adjustment was not applied.

In the actual Norwegian critical loads calculation, the regression equation used to estimate $BC*_0$ is based on MAGIC output from 1986 and 1860 for 83 lakes for which MAGIC has been calibrated (Larssen et al., 2005). This regression equation is then applied to all the grid

squares of Norway (with water chemistry data from 1986, but no MAGIC calibration). In the present exercise the regression was based on MAGIC output for years 1800 and 1860 and measured data from 1995 and 2007 for Norwegian and Swedish lakes, respectively (see below). The regression was thus based on data from all the lakes for which the BC^*_0 was estimated. This was done because the regression equation from the 83 lakes is only valid for Norway and only when "present" BC data are from 1986, which was not the year used in this exercise.
For the official reporting Sweden uses modelled base cation concentration in year 2100 (BC^*_0 (2100)) instead of historical modelled value (BC^*_0 (1860)). For this comparison we used BC^*_0 (1860) to make it more comparable to BC^*_0 (1800) used in Norway.

Comparing the methods
Each method was run on each of two sets of lakes data, one from Sweden and one from Norway (Figure C.2). The resulting calculated CLmaxS values were compared to identify differences due to differences in the lake datasets and differences due to the critical load method used. In addition, individual factors used in the calculations were compared to evaluate the causes of the differences.
Two key factors which are treated differently in the two countries and which would potentially cause systematic differences in calculated critical loads are:
Choice of biological damage criterion (delta pH in Sweden, threshold ANClimit in Norway)
The procedure for estimating BC^*_0 (regression based on MAGIC results in Norway, using MAGIC results directly in Sweden; exclusion of potassium in Norway, not in Sweden)

Data
The Norwegian lake data were taken from the 1995 regional lake survey of 1500 lakes (Skjelkvåle et al., 1996). The 989 lakes used for the comparison were among the 1007 statistically selected lakes in the survey, and were the ones for which MAGIC was calibrated in 2008 (Larssen et al, 2008b) (the remaining 18 lakes could not be calibrated with MAGIC). While year 1860 MAGIC outputs were used to calculate BC^*_0 and ANC_0 for the Swedish lake data, year 1800 outputs were used for the Norwegian data. In Sweden, 1860 is used as the reference condition year when it is assumed that there were minor anthropogenic impacts on surface waters. The implications of using year 1800 in Norway and 1860 in Sweden are marginal, as anthropogenic deposition was at that time low. For illustration the hindcast ANC for the Norwegian lakes changed by only 5 µeq/l from 1800-1860 on average.
The data for the 3239 Swedish lakes comes from three datasets (any duplicates removed) consisting of "trend" lakes, "synoptic" lakes, and "liming reference" lakes. The trend lakes have been sampled four times annually since the mid-1980s; the liming reference lakes were sampled in 2007-2008; the synoptic lakes were sampled in nationwide surveys conducted in 1995, 2000 and 2005 and since 2007 one sixth of these lakes have been sampled each year. Results of MAGIC modelling on these lakes are described by Moldan et al. (2013).

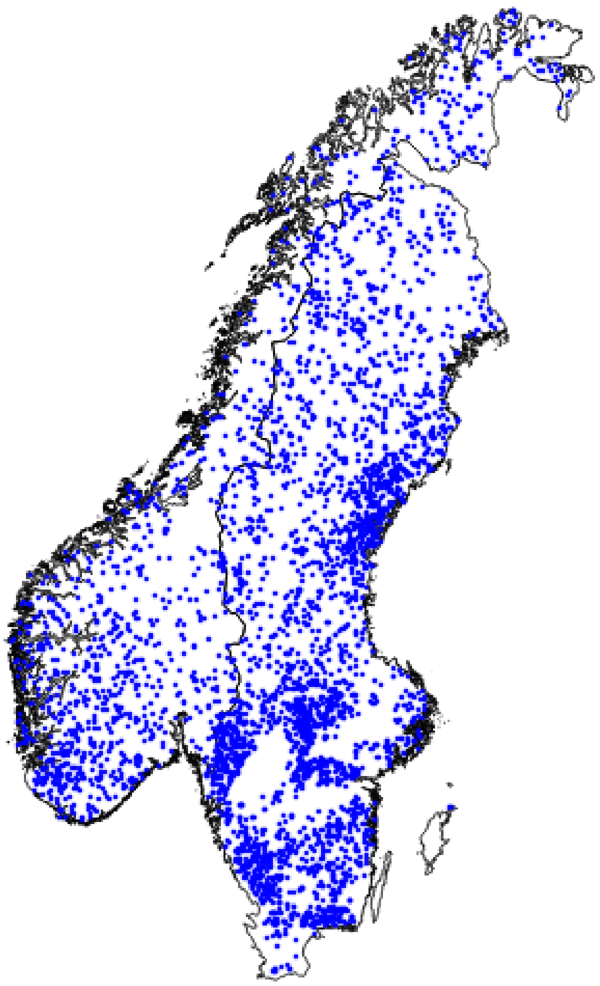

Figure C.2. Map of Norway and Sweden showing location of lakes.

The critical loads used by the CCE to construct the European map (Figure C.1) were submitted in 2012, and since then Sweden has revised the manner in which forestry has been treated in simulations of future conditions in the MAGIC calculations. In the more recent critical loads submission (in 2014, Slootweg et al., 2014) less intense future forestry was assumed. Consequently, the calculated Swedish critical loads increased (and critical loads exceedance decreased). This is due to lower demand for base cations by the forests in the future. Lower BC uptake leaves more of the soil buffering capacity in the soils available to counteract acidifying deposition. European critical loads exceedance maps have not yet been constructed using the 2014 Swedish submission, and it is most probable that the difference between Sweden and Norway would become less acute relative to that based on the Swedish 2012 submission.

Results

On average the Swedish method gives lower CLmaxS for lakes in both Norway and Sweden (Table C.1) relative to the Norwegian method applied to both datasets. On average the Norwegian method gives higher CLmaxS by 13 meq/m^2/yr for Norwegian lakes and 58 meq/m^2/yr for Swedish lakes. For individual lakes, the Swedish method gives lower CLmaxS in most cases (88% of the lakes) (Figure C.3). Above CLmaxS 600 meq/m^2/yr (both methods) the Swedish method always gives lower CLmaxS, and the discrepancy increases with increasing CLmaxS.

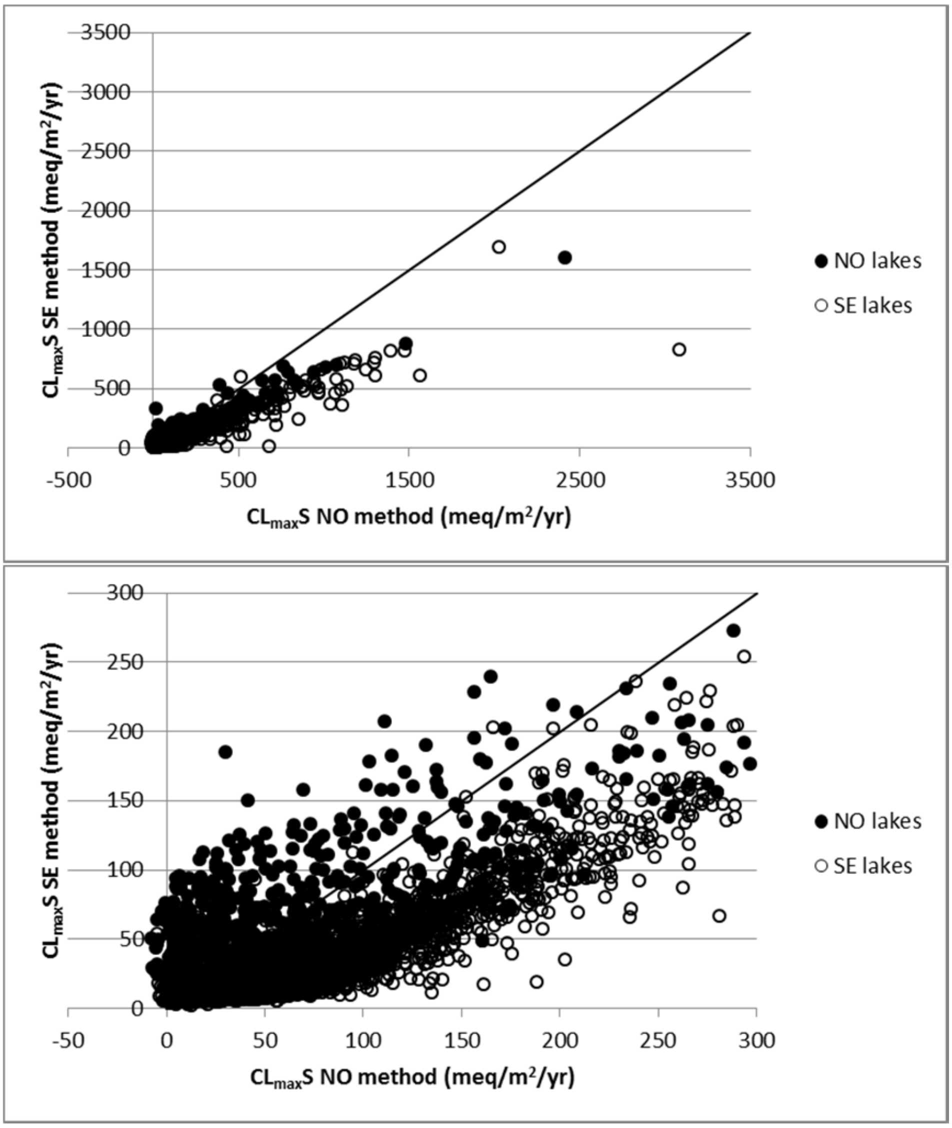

Figure C.3. CLmaxS calculated using the Swedish method vs CLmaxS calculated using the Norwegian method. Upper panel - all the lakes; lower panel – lakes with CL below 300 meq/m2/yr. 1:1 line shown.

Table C.1. Mean values of CLmaxS and two key parameters calculated by the Swedish and Norwegian methods for lakes in Norway and Sweden.

	Unit	**Norwegian method**		**Swedish method**	
		989 NO lakes	3239 SE lakes	989 NO lakes	3239 SE lakes
BC^*_0	μeq/l	125	289	138	304
ANClimit	μeq/l	16	19	59	161
CLmaxS	$meq/m^2/yr$	87	108	74	50

Most (78%) of the lakes for which the Norwegian method gives the lowest CLmaxS are Norwegian. This mainly happens in the lower CLmaxS range (Figure C.3), indicative of higher sensitivity to acid deposition. Norwegian lakes are on average more sensitive, due to poorer and thinner soils in the catchments and lower weathering rates, which gives lower buffering capacity. This is manifest by lower BC^*_0 for Norwegian lakes compared to Swedish lakes, independent of method (Table C.1). However, this inherent difference between Norwegian and Swedish lakes cannot explain the difference in CLmaxS calculated by the two countries (Figure C.1). Figure C.4 shows that when CLmaxS is calculated for all lakes by the same method, there is no marked country border; lakes in close proximity, but at each side of the national border have similar CLmaxS. The maps also confirm the generally lower CLmaxS calculated by the Swedish method. The main exceptions are lakes in the southwestern part of Norway, for which the Norwegian method gives lower CLmaxS. Lakes in this region have particularly low BC^*_0, so this corresponds well with the impression from Figure C.3 that lower CLmaxS with the Norwegian method occurs mainly for lakes that are more sensitive to acidification. The higher CLmaxS for these lakes calculated by the Swedish method may also explain why the average CLmaxS is higher for Norwegian than Swedish lakes with the Swedish method (Table C.1) despite the Norwegian lakes being generally more sensitive.

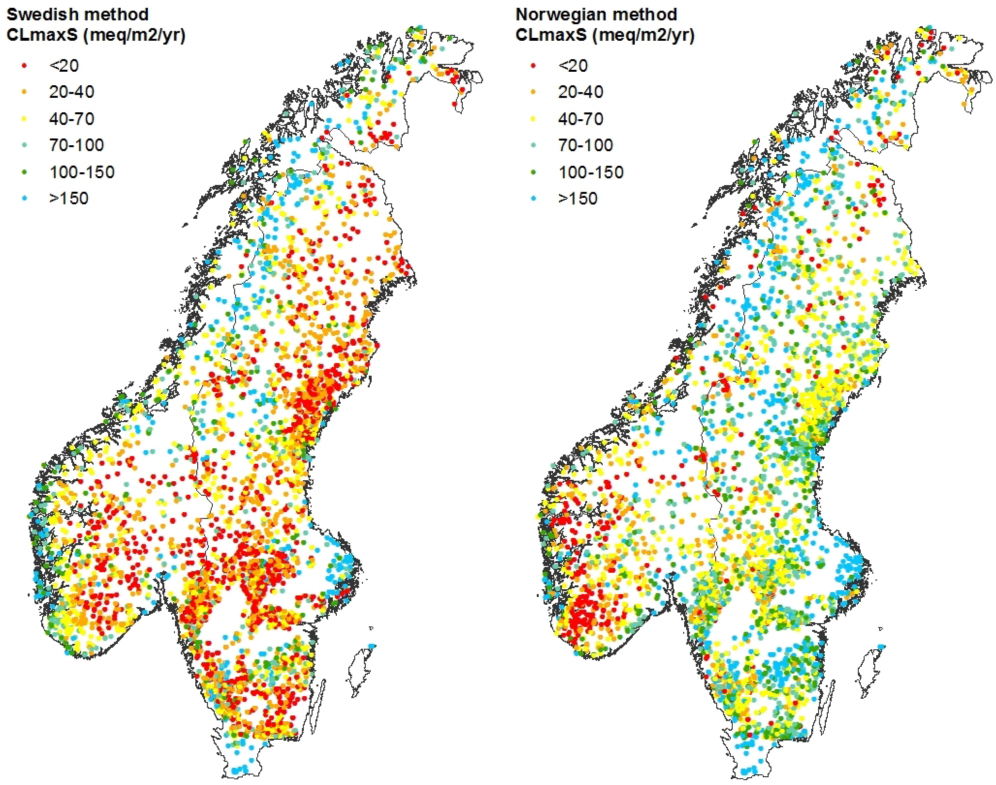

Figure C.4. Map of Norway and Sweden showing the calculated CLmaxS for the different lakes, when applying the Swedish (left) or the Norwegian (right) method.

The differences between the two methods in treatment of more vs less sensitive lakes become clearer if the lakes are stratified according to their historical ANC (ANC_0) (Figure C.5). ANC_0, like BC^*_0, is an indicator of the buffering capacity, but is estimated in the same way in both countries (by MAGIC). Only lakes with CLmaxS <100 meq/m^2/yr are included, as lakes with higher CLmaxS are not likely to have critical load exceeded, and are thus less relevant. Figure C.5 shows that for lakes with ANC_0<50 µeq/l the Norwegian method gives lower CLmaxS, whereas for higher ANC_0 lake classes the Norwegian method gives higher CLmaxS as compared with the Swedish method. Most of the lakes in the two lower categories are Norwegian, while most of the lakes in the three upper ANC_0 categories are Swedish (Table C.2).

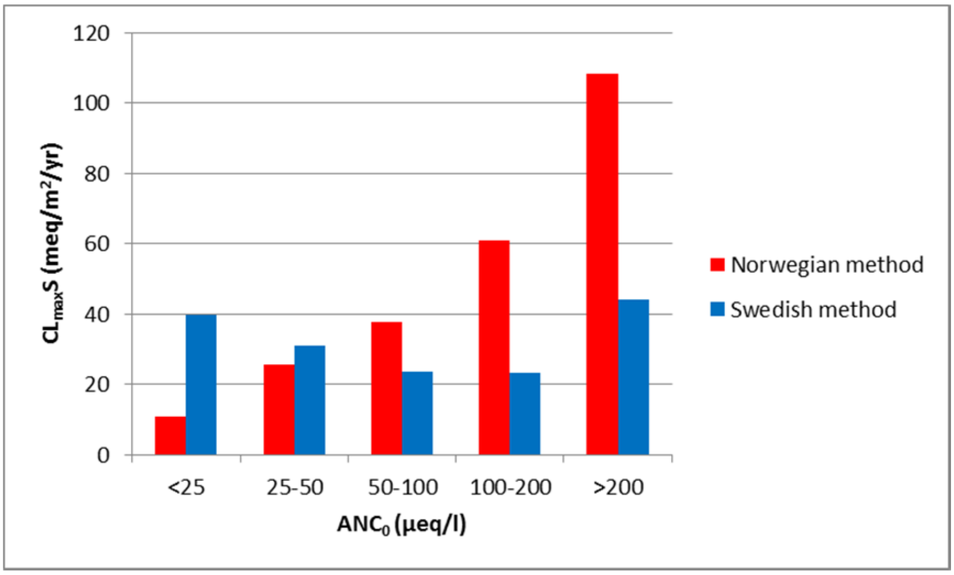

Figure C.5. Average CLmaxS for groups of lakes in different ANC_0 categories, using the Norwegian method (red bars) and the Swedish method (blue bars). Only lakes with CLmaxS Swedish method <100 meq/m^2/yr are included (n=3704).

Table C.2. Number of Swedish and Norwegian lakes in the ANC_0 categories applied in Figures C.5, C.6, C.7 and C.9.

	ANC_0 (µeq/l)				
	<25	25-50	50-100	100-200	>200
Lakes in Sweden	18	76	333	1306	1184
Lakes in Norway	228	176	190	126	67
Total	246	252	523	1432	1251

Differences in CLmaxS calculated by the two methods can be due to differences in calculations of both ANClimit and BC^*_0. The observed discrepancy in BC^*_0 between the methods is generally so small that it has very little effect on the average CLmaxS (Table C.1). However, for the low ANC lakes, even small differences in BC^*_0 can be important. For the <25 µeq/l ANC_0 category, the much higher CLmaxS calculated by the Swedish method can partly be explained by higher BC^*_0. This is to some extent also true for the 25-50 µeq/l ANC_0 category. In the low ANC_0 range the relative difference in BC^*_0 between the methods is larger (Figure C.6), and the BC^*_0 is also relatively more important in the CLmaxS calculation because the ANClimit is small. The difference in BC^*_0 is mainly due to the exclusion of potassium in the Norwegian method. If potassium is included in the Norwegian method the estimated BC^*_0 with the Norwegian (regression) approach gives nearly identical results to the Swedish method (MAGIC) on average. Also for the three upper ANC_0 categories the difference is negligible when adding potassium. However, for the two lower categories, in particular for the <25 µeq/l ANC_0 category, the relative difference between the methods, although far smaller than when potassium is excluded in the Norwegian method, is sufficiently big to have an effect. Hence, the regression approach in itself does have some impact for these lakes.

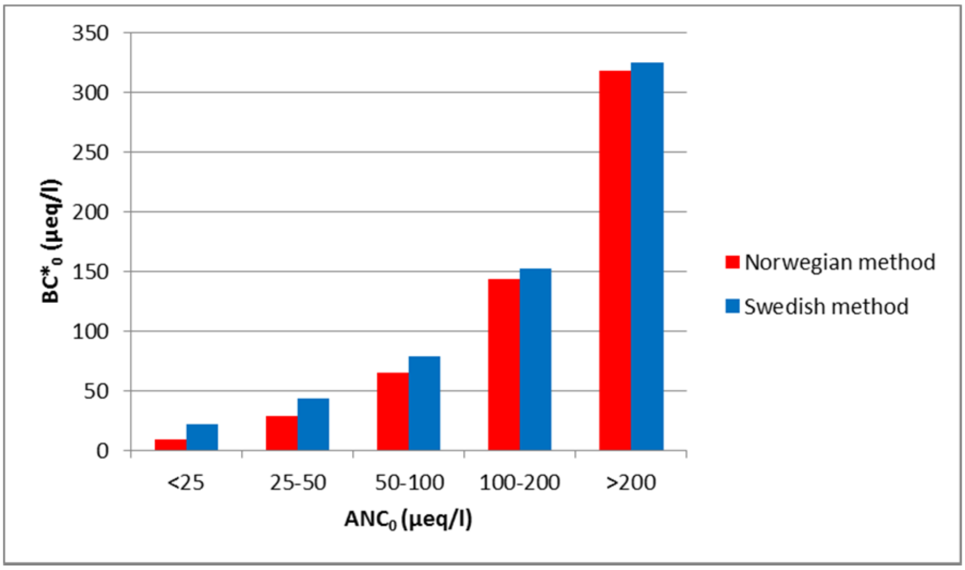

Figure C.6. Average BC^*_0 for groups of lakes in different ANC_0 categories, using the Norwegian method (red bars) and the Swedish method (blue bars). Only lakes with CLmaxS Swedish method <100 meq/m2/yr are included (n=3704).

The difference in ANClimit between the two methods is large (Table C.1). Stratifying by ANC_0 categories (Figure C.7) shows that apart from the lowest ANC_0 category, the average ANClimit is higher with the Swedish method, and the difference increases with increasing ANC_0. This explains that the Swedish method gives lower CLmaxS for the three upper ANC_0 categories, and that the discrepancy between the methods increases with increasing ANC_0 (and with increasing CLmaxS, cf. Figure C.3).

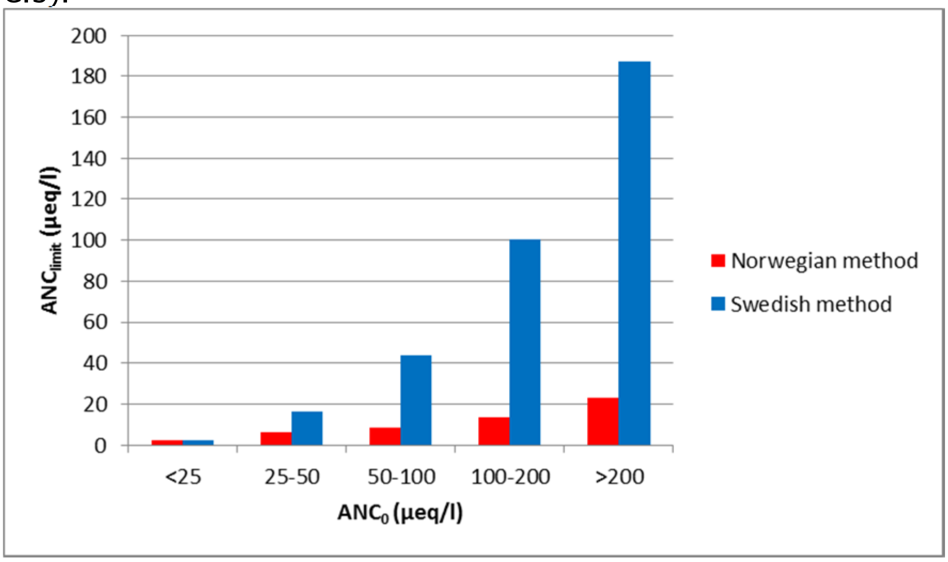

Figure C.7. Average ANClimit for groups of lakes in different ANC_0 categories, using the Norwegian method (red bars) and the Swedish method (blue bars). Only lakes with CLmaxS Swedish method <100 $meq/m^2/yr$ are included (n=3704).

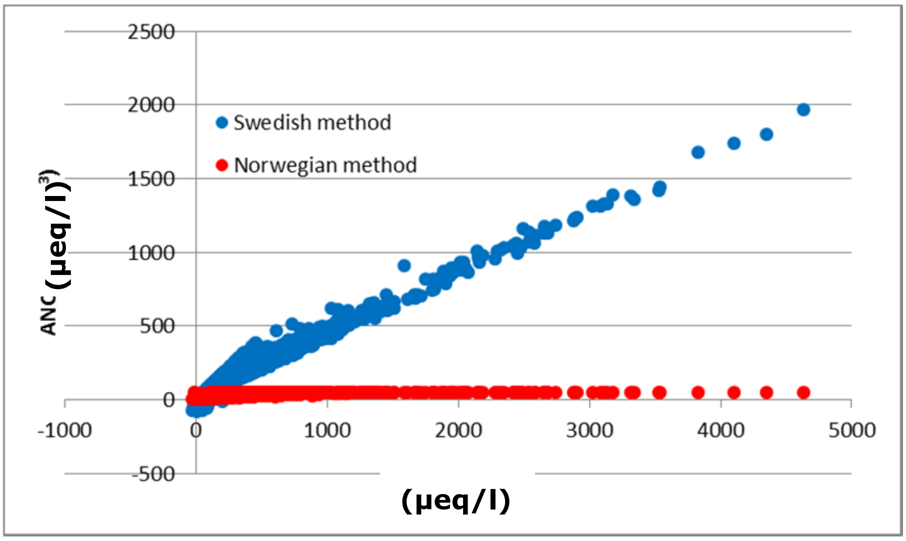

Figure C.8. ANClimit calculated by the Swedish method (blue) and Norwegian method (red) vs. ANC_0 (year 1800 NO/1860 SE).

One difference between the methods is that the Norwegian method has a fixed ANClimit with a range of 0-50 µeq/l, while there are no such boundaries with the Swedish method. This explains why the average Swedish ANClimit is in many cases much higher than the Norwegian, especially for higher ANC_0 lakes (Figure C.8). It also explains why the Swedish method can result in negative ANClimit at lakes with very low ANC_0 (Figure C.8). However, the upper threshold of 50 µeq/l with the Norwegian method does not explain the large discrepancy between the methods seen in Figure C.7, as all the lakes included here have ANClimit below 50 µeq/l with the Norwegian method anyway, i.e. without using a 50 µeq/l cut-off. Moreover, removing the upper threshold in the calculation only gives minor changes to the average values given in Table C.1 (ANClimit 20 and 23 µeq/l and CLmaxS 82 and 106 meq/m^2/yr for Norwegian and Swedish lakes, respectively). Figure C.9 shows that also within the 0-50 µeq/l ANClimit range, with some exceptions the Swedish method generally gives higher ANClimit. Most of the (few) cases where the ANClimit is higher with the Norwegian method are found at low ANClimit.

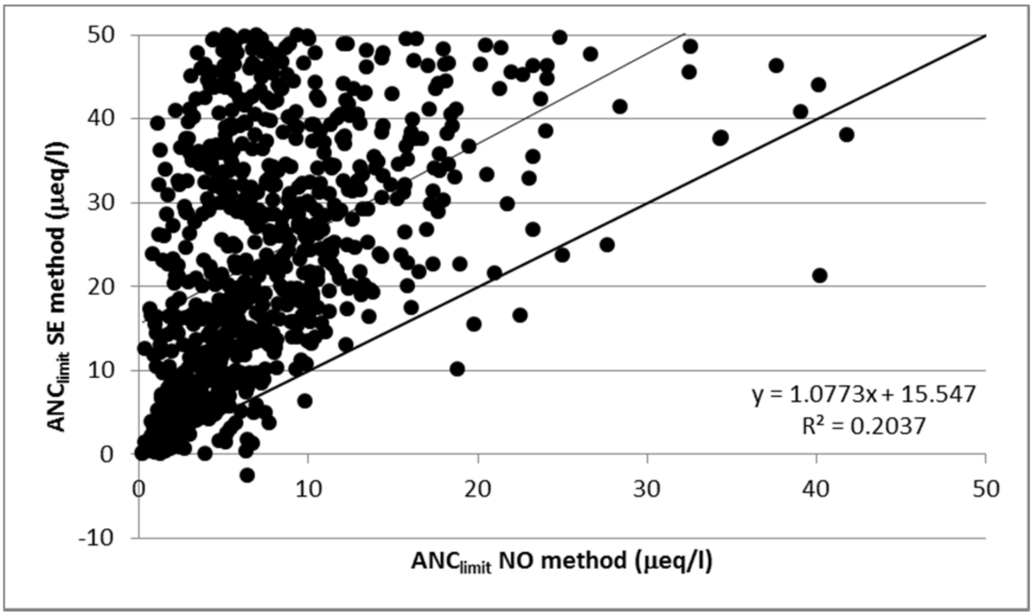

Figure C.9. ANClimit calculated using the Swedish method vs ANClmit calculated using the Norwegian method. Only lakes where ANClimit is in the range 0-50 µeq/l with both methods are included (n=769). 1:1 line and regression line also shown.

Remarks on the methodological differences between the comparison exercise and the actual critical loads calculation

The CLmaxS calculated with the Norwegian method were generally only slightly lower on average when using the organic acid adjustment (10 meq/m^2/yr including all lakes). Grouping the lakes in the same way as in Figure C.5 (Figure C.10), shows that CLmaxS calculated with the Norwegian method is closer to the Swedish method when using organic acid adjustment (except in the 25-50 category). However, for all categories except the 50-100 µeq/l category, the difference between the Norwegian method with organic acid adjustment and the Swedish method is still larger than the difference between the two Norwegian methods. This means that the general impression from the comparison exercise remains the same.

It should be noted that negative CLmaxSoaa was observed for 122 Swedish lakes (all part of the data set shown in Figure C.10; for Norwegian lakes CLmaxSoaa was only negative for 20 lakes with negative BC^*_0). These lakes had on average higher TOC and far lower BC^*_0 than the overall average for the Swedish lakes. The results indicate that the organic acid adjustment developed on empirical relationships for Norwegian lakes is not always applicable for Swedish lakes, probably because many of the Swedish lakes are outside the range of TOC and BC^*_0 experienced in Norway. The organic acid adjustment concept is applicable in principle but would need to be adapted based on empirical relationships observed in Swedish lakes to reflect generally higher both TOC and BC^*_0 levels observed in Sweden.

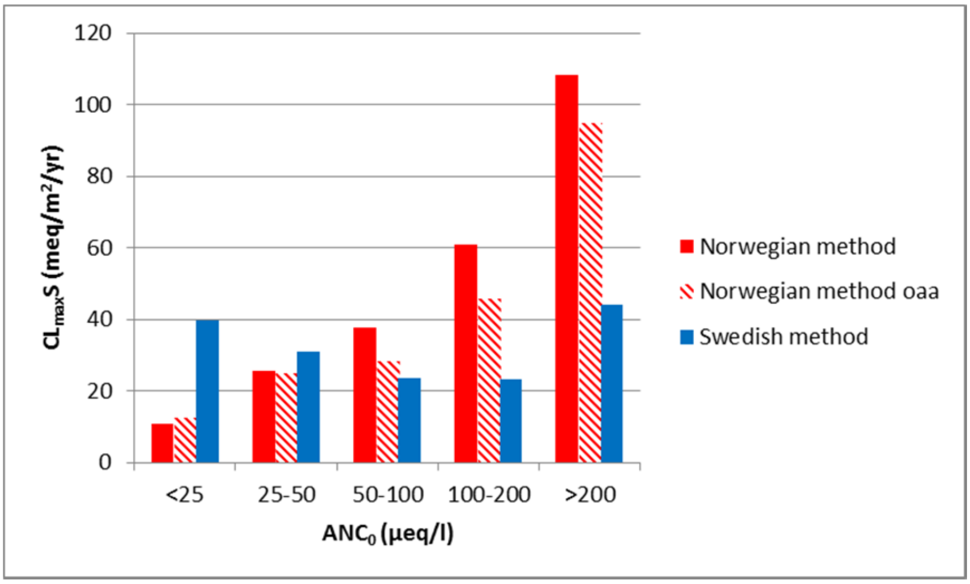

Figure C.10. Average CLmaxS for groups of lakes in different ANC_0 categories, using the Norwegian method without (red filled bars) and with (red diagonal striped bars) organic acid adjustment and the Swedish method (blue bars). Only lakes with CLmaxS Swedish method <100 meq/m^2/yr are included (n=3704).

To test the effect of the choice of regression equation, BC^*_0 was estimated for the 989 Norwegian lakes using the MAGIC output from 1986 as "present" BC* and either the regression equation for the 83 lakes (the actual Norwegian procedure) or a regression equation based on MAGIC output from years 1986 and 1800 for the 989 lakes. There was good correlation, but with a consistent bias, with the BC^*_0 estimated from the 83 lakes regression being about 17 meq/m^3 higher on average. This gives a CLmaxS 17 meq/m^2/yr higher on average, i.e. the difference between the Norwegian and Swedish methods would have been slightly larger if the actual Norwegian method was used.
It is not clear, however, if the BC^*_0 calculation based on 989 lakes is more accurate than that based on 83 lakes. Conceptually, the regression approach is less accurate than applying the MAGIC output directly. However, the MAGIC calibration of the 83 lakes is probably better than that of the 989 lakes, as it is based on time series, not just a single data point. The number of lakes is low, but all the lakes are acidified, so thus relevant. Moreover, using the 83 lakes regression produced no negative BC^*_0, while the MAGIC modelled BC^*_0 for the 989 lakes was negative at four lakes, indicating a negative bias. As a test, the MAGIC modelled BC^*_0 for the 989 lakes was applied in calculating CLmaxS for the Norwegian critical load grid cells (after matching the 989 lakes with the grid cells using the MAGIC library routine) instead of using the regression approach. This gave 34 out of 2304 grid cells with negative CLmaxSoaa (after setting negative BC^*_0 values to zero), again indicating that this MAGIC calibration underestimates BC^*_0. Hence, until an improved MAGIC calibration (preferably based on new data from the 989 lakes) is in place, the current regression approach is considered to be the best option.
In the Swedish official submission of critical loads calculations, base cations concentrations in year 2100 were used as a BC^*_0 (BC^*_0 (2100)). For comparability to the Norwegian method in this exercise this has

been changed and BC^*_0 (1860) was used. This adjustment has only a very minor impact on critical loads calculations since by the year 2100 BC* concentrations in the lake water are in general approaching the levels close to modelled state in 1860 (Figure C.11).

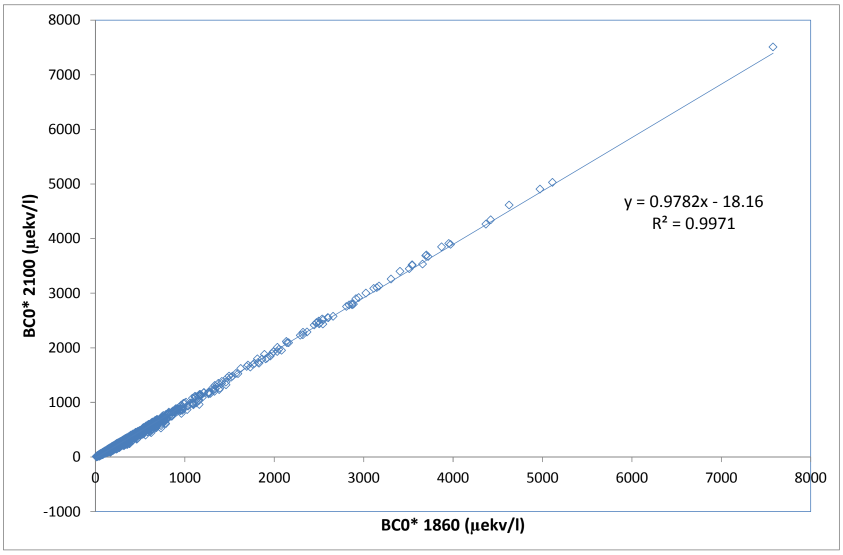

Figure C.11. Comparison of BC^*_0 (1860) and BC^*_0 (2100) for Swedish lakes. Relative differences are greatest at lower concentrations.

Discussion

The step change in CLmaxS at the border between Norway and Sweden on the CCE maps of CLmaxS in Europe is by and large due to differences in calculation methods applied and not due to differences in lake properties. The major difference between the two methods is the specification of the ANClimit. With both the Norwegian and Swedish method the ANClimit increases with increasing original historical pH (or ANC). In the Swedish case the damage criterion is delta pH=0.4 below historical lake pH. It is a continuous variable that assumes that unacceptable biological damage occurs over the entire pH (and ANC) range in lakes. By the Swedish criterion it is equally unacceptable to acidify a lake from pH 7.2 to 6.8 as it is to acidify a lake from pH 5.5 to 5.1. In the Norwegian case the ANClimit is defined not as a change from historical conditions but as a linear function of historical base cation concentration. Moreover, the ANClimit has an upper threshold of 50 µeq/l, above which no biological damage is expected regardless of historical ANC, and a lower ANC limit of 0 µeq/l, below which damage is always assumed to occur. The Swedish method can result in high ANClimit at well buffered lakes and the differences in ANClimit and in CLmaxS calculated by the two methods at these lakes could be substantial.

The relationship between pH and ANC is not linear (Figure C.12). pH is a logarithmic variable, whereas ANC is a linear variable. At a given ANC the pH in lakes is largely determined by the combined effects of pCO2 and the inorganic carbon equilibria, the dissociation of organic acids, and the dissolution and speciation of aluminium. At low pH (below about 4.5) dissociation of Al buffers changes in pH, while at high pH (pH above 6.5) bicarbonate buffers changes in pH. Organic acids buffer across the entire pH range. Buffering is lowest at intermediate pH (4.5-6.5) – in this

range relatively small changes in ANC are needed to give rise to a change in pH of 0.4 units (Figure C.12).
In summary, both methods give relatively similar ClmaxS in the ANC_0 range of 25 – 100 µeq/l, while at high ANC and at low ANC, the ClmaxS calculated by the two methods will be more different. At high ANC, the Swedish method will tend to tolerate less S deposition because of the high ANClimit and the opposite is true for the least-well buffered lakes with lowest historical ANC, where the Norwegian method will tolerate less S (Figure C.5).

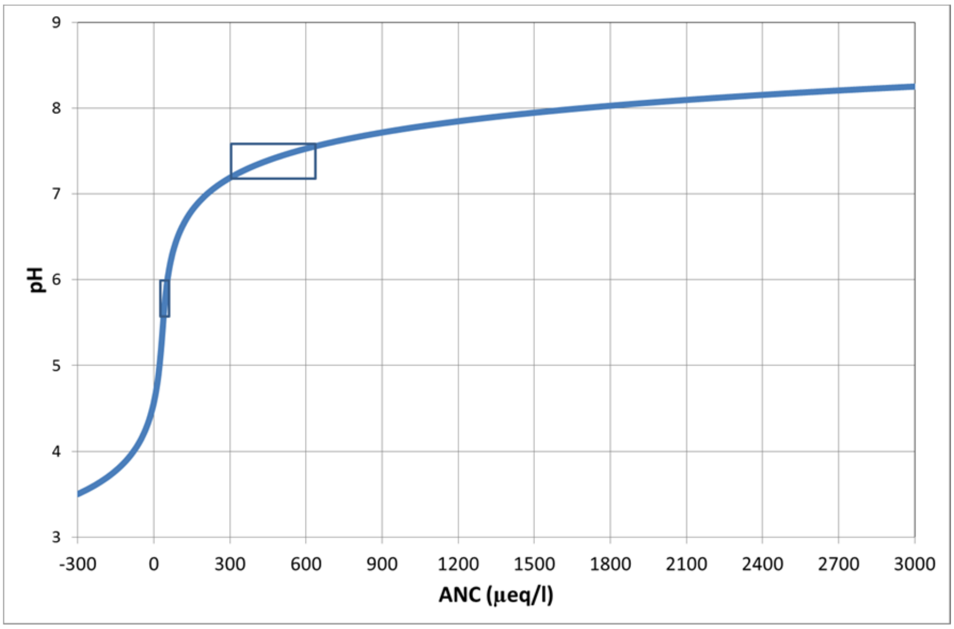

Figure C.12. The relationship between ANC and pH for a lake with a TOC of 5 mgC/l, and a pCO2 of 4x atmospheric pressure. The change in ANC required to produce a 0.4 unit change in pH (boxes) is very small in the pH range 5-7 as compared to the pH range above 7.

The delta pH criterion used in Sweden addresses several organism groups. Littoral invertebrates are considered as well as fish species. The ANClimit thresholds used in Norway is based principally on fish species, mainly brown trout (Lien et al., 1996). Other fish species such as roach may have damage at ANC levels up to ca 70 µeq/l (Lien et al., 1996). Moreover, other organism groups, which may or may not be less tolerant, are not considered in the Norwegian method. The decision as to which organisms need to be protected is political rather than scientific. Further exploration of biological and chemical data could reveal whether the upper threshold of 50 µeq/l in the Norwegian method is too low, and give a better basis for the political decisions. However, as has been shown, the upper threshold does not affect lakes in the lower CLmaxS range, i.e. even removing it completely is not likely to have major effects on exceedances.
The Norwegian approach assumes that once the ANC is above the ANClimit, further improvements in ANC will not affect the fish populations and is therefore not required. In the Swedish approach the goal is to obtain pH levels within 0.4 units of the original reference condition for the lake. This conceptual difference is the major

explanation of the discrepancy between the methods. It is thus not a question of which method is better, as the methods have different objectives with respect to the criterion for ecological protection. The choice of method is again more political than scientific.

Recommendations
Based on this comparison, the differences in methods applied in Norway and in Sweden when calculating critical loads for acidity for lakes are to a large extent due to decisions on how to relate desired water quality to pre-industrial status, and partly also which organisms to protect. The report illustrates the consequences of decisions taken on both sides of the Swedish – Norwegian border. Both methods serve the purpose and differences found are understandable. Both countries might in the future give consideration to findings in this report and re-open discussions about the choice of criteria for acidification assessment on lakes and rivers. Such discussion would have consequences for both critical loads and for national acidification assessments, not the least in connection with Water Framework Directive and assessment of Good Ecological Status. Some minor adjustments to critical loads methodology however could be considered even within the current concept:

For Sweden:

- in cases where historical ANC_0 is very low, restrictions (e.g. lowest ANClimit set to 0 µeq/l) could be considered to avoid setting negative ANClimit: new organic acidity coefficients that give more credible ANC values at high TOC and low pH could also be used
- use of BC^*_0 (1860) as BC^*_0 instead of currently used BC^*_0 (2100) makes very little practical difference and would be more in line with the common methodology described in the ICP M&M Mapping Manual

For Norway:

- Potassium should be included in the base cation calculations. This has recently been done for the 2014-15 Call for data. (see the Norwegian national report in this volume)
- The ANClimit upper boundary may be revisited in light of the EU "no net loss of biodiversity" target: Does the 50 µeq/l limit protect all organisms in a satisfactory way? Does it keep the biodiversity intact?

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Overview of Critical Load Data

This document gives a summary of data sources and methods used to calculate Swiss critical loads, and highlights changes since the previous data submission (Achermann et al. 2011). As in 2011, 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). For the CCE data call 2014/15 all methods and data were updated, except the critical loads for alpine lakes did not change:

The SMB method for calculating critical loads of nutrient nitrogen (CLnutN) was applied on 10,632 forest sites. 10,331 of these sites originate from the National Forest Inventory (NFI 1990/92), which is based on a 1x1 km^2 grid. They are complemented by 301 sites with soil profiles (which are partly identical with the NFI-sites).

The empirical method for mapping critical loads of nutrient nitrogen (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^2 grid combining several input maps of nature conservation areas and vegetation types. The total sensitive area amounts to 14,532 km^2.

A variant of the SMB was used for assessing critical loads of acidity on 301 forest sites, where full soil profiles were available. Net-uptake fluxes were modelled with the model MakeDep.

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).

With regard to the use of the "Habitat Suitability Index" progress was made in gathering input data but no results could be submitted so far.

The Swiss critical loads database is constructed on the base of sampling points and modelling sites in such a way that ecosystem areas are consistent with the new EMEP longitude-latitude grids (0.50° x 0.25° or 0.1° x 0.1°). Figure CH.1 gives an overview of the ecosystems and methods used for mapping.

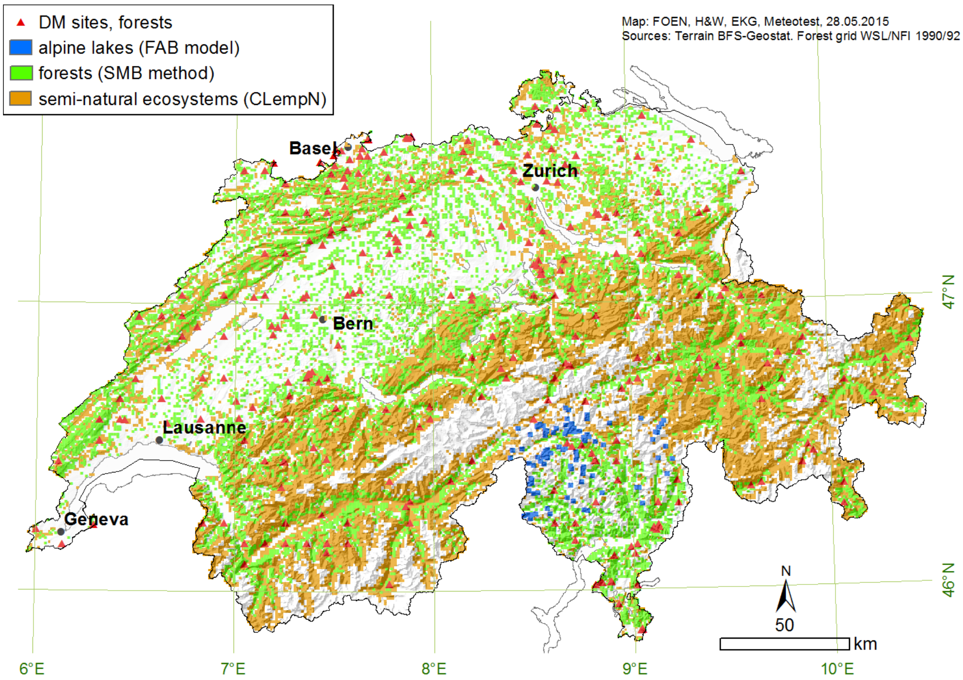

Figure CH.1 Overview of ecosystems: forest monitoring sites used for dynamic modelling (DM sites), alpine lakes, forest sites from the NFI and semi-natural ecosystems from various data sources (Hegg et al. 2003; national inventories of raised bogs, fens and dry grassland (TWW), biodiversity monitoring network (BDM)).

Some essential results of the update are shown in Figure CH.2 as cumulative frequency distributions: CLnutN for forests (SMB method), CLnutN for (semi-)natural ecosystems (empirical method) as well as the maximum critical load of sulphur (CLmaxS) for forests (MakeDep/SMB models) and Alpine lakes (FAB model).

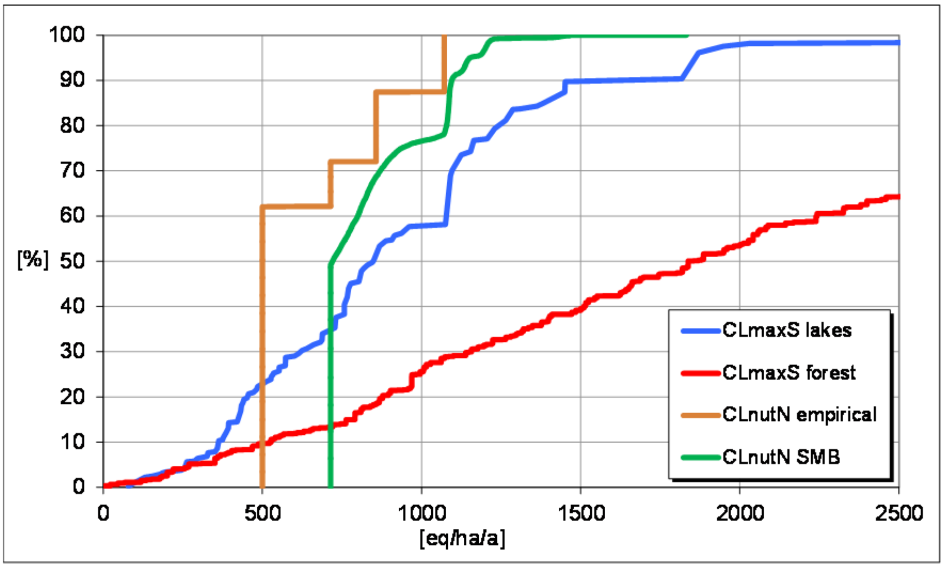

Figure CH.2 Cumulative frequency distributions of CLnutN (SMB and empirical method) and CLmaxS (forests and alpine lakes).

Critical loads of nutrient nitrogen (SMB method)

Procedure

In a first step, CLnutN was calculated by the SMB method for 301 forest sites used in dynamic modelling and for 10,331 sites of the National Forest Inventory (NFI). Table CH.1 gives a summary of the input parameter values. Thereby, only NFI-sites with a defined mixing ratio of deciduous and coniferous trees are included (NFI 1990/92). This corresponds approximately to the managed forest area as brush forests and inaccessible forests are excluded.

In a second step, 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 used for forests). This means, all values of CLnutN below 714 eq ha^{-1} a^{-1} were set to 714. This is 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} yr^{-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
Nle (acc)	4 kg N ha^{-1} yr^{-1} at 500 m, 2 kg N ha^{-1} yr^{-1} at 2000 m altitude, linear interpolation in-between	Acceptable N leaching. Leaching mainly occurs by management (after cutting), which is more intense at lower altitudes.
Ni	1.5 kg N ha^{-1} yr^{-1} at 500 m, 2.5 kg N ha-1 yr^{-1} at 1500 m altitude, linear interpolation in-between	N immobilization in the soil. At low temperature (correlated with high altitude) the decomposition of organic matter slows down and therefore the accumulation rates of N are naturally higher.
Nu	0.5 – 14.7 kg N ha^{-1} yr^{-1}	N uptake calculated on the basis of long-term harvesting rates.
fde	0.2 – 0.7 depending on the wetness of the soil	Denitrification fraction. For NFI-sites, information on wetness originates from soil map 1:200’000. For DM-sites it is a classification according to the depth of the saturated horizon.

Acceptable nitrogen leaching

Instead of using precipitation surplus (Q) and acceptable N concentrations in soil water ([N]acc) as proposed in the mapping manual, Nle(acc) was calculated as a function of altitude (see Table CH.1). The rationale for this procedure was presented in a former CCE Status Report (Achermann et al. 2007). The proposed values for [N]acc were tested with the Swiss dataset. Some of the proposed values led to implausible high N leaching and CLnutN, mainly in high precipitation areas, which was judged to be unacceptable with respect to the risk of acidification and concomitant nutrient (base cation) losses.

Therefore it was decided to continue using the acceptable N leaching rates (Nle(acc)), which were used already in former data submissions. They are basically drawn from the 1996 version of the Mapping Manual

(UBA, 1996). They reflect an average long-term N leaching rate which is caused by management, mainly after cutting or other disturbances. Forest management is generally more intense at lower altitude than at high altitude (see also Section Nitrogen Uptake).
The submitted values of acceptable N concentration were calculated as: [N]acc = Nle(acc) / Q.

Nitrogen immobilization
At high altitudes, the decomposition of organic matter slows down due to lower temperatures and therefore the accumulation rates of N in the soil are naturally higher. The values shown in Table CH.1 are somewhat higher than the proposal in the Mapping Manual. This means that a 'conservative' calculation of CLnutN is made.

Net growth uptake of nitrogen
For the DM-sites, net-uptake fluxes were modelled with MakeDep (Alveteg et al., 2002) using biomass data from the 3rd National Forest Inventory (http://lfi.ch, WSL, 2013), tree genera-specific logistic growth curves, site productivity index, nutrient contents in the various compartments of the tree, and average annual harvesting rates stratified according to the five NFI-regions (Table CH.2).
The uptake for the other forest sites was derived from the DM-sites by a linear regression with altitude (z) within each region (Table CH.2).

Table CH.2 Net nitrogen uptake (Nu) in the five NFI-regions (kg N ha^{-1} a^{-1}).

Region	**Average**	**Function of altitude z (m a.s.l.)**
1. Jura	5.3	6.99 - 0.00300 z
2. Central Plateau	8.5	--
3. Pre-Alps	4.3	7.60 - 0.00322 z
4. Alps	2.9	3.58 - 0.00064 z
5. Southern Alps	1.6	2.29 - 0.00056 z
Average CH	4.4	--

Denitrification fraction
For calculating CLnutN, fde was determined according to wetness class information from the digital soil map BEK (SFSO, 2000) as shown in Table CH.3. On the DM-sites, information from the soil profiles was used to determine the depth of the water saturated horizon.

Table CH.3 Values of fde selected for the BEK classes of soil wetness.

Wetness class BEK	Description	Depth of saturated horizon	fde
0	Unknown	--	0.2
1	No groundwater	--	0.2
2	Moist	below 90 cm, but capillary rise	0.3
3	Slightly wet	60-90 cm	0.4
4	Wet	30-60 cm	0.6
5	Very wet (not occurring on the digital map)	<30 cm	0.7

Empirical critical loads of nutrient nitrogen
The application of the empirical method is based on vegetation data compiled from various sources and aggregated to a 1x1 km^2 raster (see Figure CH.1). Overall, 44 sensitive vegetation types were identified and included in the critical load data set:

- 1 type of raised bog; source Federal Inventory of Raised and Transitional Bogs of National Importance (EDI 1991), see Table CH.4;
- types of fens; source Federal Inventory of Fenlands of National Importance (WSL 1993), see Table CH.4;
- 21 types with various vegetation worthy of protection (Hegg et al. 1993) including rare and species-rich forest types, grasslands and alpine heaths, see Table CH.4;
- 1 type of mountain hay meadow in montane to sub-alpine altitudinal zones with more than 35 species $(10\ m2)^{-1}$ (Roth et al. 2013), source Biodiversity Monitoring (BDM, http://www.biodiversitymonitoring.ch/en/data/indicators/z/z9.html), see Table CH.4.
- 18 types of dry grassland; source National Inventory of Dry Grasslands of National Importance (TWW, FOEN 2007); see Table CH.5.

The values for the empirical critical loads for nitrogen (CLempN) have been based on the outcome of the Workshop in Noordwijkerhout (Bobbink and Hettelingh 2011). In addition, the relative sensitivity of the ecosystems was reassessed by Burnand (2011).

On the basis of recent results from the assessment of relationships between nitrogen deposition and species diversity in mountain hay meadows (EUNIS class E2.3) and (sub-)alpine scrub habitats (EUNIS class F2.2) in Switzerland it was concluded that the empirical critical loads for nitrogen proposed for these habitats at the workshop in Noordwijkerhout (Bobbink and Hettelingh 2011) should be set at lower values (Roth et al 2013, Achermann et al 2014). For mountain hay meadows a range for CLempN of 10-15 kg N $ha^{-1}yr^{-1}$ (instead of 10-20 kg N $ha^{-1}yr^{-1}$) and for (sub)alpine scrub habitats a range of 5-10 kg N $ha^{-1}yr^{-1}$ (instead of 5-15 kg N $ha^{-1}yr^{-1}$) is used now.
The critical loads database was adapted accordingly and complemented with new sites of the BDM. Furthermore, EUNIS codes and empirical critical loads were specified for some grassland ecosystem types.
The TWW data set complements well the grassland types mapped by Hegg et al. (1993). It contains 18 vegetation groups, which partially also occur in the inventory of Hegg et al. The two inventories are used here in a complementary way, because they answer different purposes: the atlas of Hegg et al. gives an overview of the occurrence of selected vegetation types, while TWW focuses on the precise description of objects with national importance.
If more than one sensitive ecosystem type occurs within a 1x1 km^2 grid-cell the lowest value of CLempN was selected for this cell.

Table CH.4 The empirical method: selected ecosystems, critical load values applied in Switzerland (kg N ha^{-1} a^{-1})

Ecosystem type	CLN range	Relevant vegetation types in Switzerland	CLempN	EUNIS code
Coniferous forests	5-15	Molinio-Pinetum (Pfeifengras-Föhrenwald)	12	G3.44
		Ononido-Pinion (Hauhechel-Föhrenwald)	12	G3.43
		Cytiso-Pinion (Geissklee-Föhrenwald)	12	G3.4
		Calluno-Pinetum (Heidekraut-Föhrenwald)	10	G3.3
		Erico-Pinion mugi (Ca) (Erika-Bergföhrenwald auf Kalk)	12	G3.44
		Erico-Pinion sylvestris (Erika-Föhrenwald)	12	G3.44
Deciduous forests	10-20	Quercion robori-petraeae	15	G1.7
		(Traubeneichenwald)	15	G1.71
		Quercion pubescentis (Flaumeichenwald)	15	G1.73
		Fraxino orno-Ostryon (Mannaeschen-Hopfenbuchwald)		
Arctic and (sub)- alpine scrub habitats	5-10	Juniperion nanae	7	F2.23
		(Zwergwacholderheiden)	7	F2.21
		Loiseleurio-Vaccinion (Alpenazaleenheiden)		
Sub-atlantic semi-dry calcareous grassland	15-25	Mesobromion (erecti) (Trespen-Halbtrockenrasen)	15	E1.26
Molinia caerulea meadows	15-25	Molinion (caeruleae) (Pfeifengrasrieder)	15	E3.51
Mountain hay meadows	10-15	Grassland types 4.5.1-4.5.4 (Delarze et al. 2008)	12	E2.3
(sub)-alpine grassland	5-10	Chrysopogonetum grylli (Goldbart-Halbtrockenrasen)	10	E4.3
		Seslerio-Bromion (Koelerio-Seslerion) (Blaugras-Trespen-	10	E4.4
		Halbtrockenrasen)	10	E4.4
		Stipo-Poion molinerii (Engadiner Steppenrasen), sub-alpine	7	E4.42
		Elynion (Nacktriedrasen), alpine		
Shallow soft-water bodies	3-10	Littorellion (Strandling-Gesellschaften)	7	C1.1
Poor fens	10-15	Scheuchzerietalia (Scheuchzergras)	10	D2.21
		Caricion fuscae (Braunseggenried)	12	D2.2
Rich fens	15-30	Caricion davallianae (Davallsseggenried)	15	D4.1
Raised bogs	5-10	Sphagnion fusci (Hochmoor)	7	D1.1

Table CH.5 Empirical critical loads for nitrogen assigned to 18 types of dry grasslands (TWW) of the national inventory of dry grasslands (FOEN 2007), in kg N ha^{-1} a^{-1}. Some types are also included in the dataset by Hegg et al. (2003), see remarks.

TWW-code		Vegetation type	EUNIS	Remarks	CLempN
1	CA	Caricion austro-alpinae	E4.4	(sub-)alpine grassland	8
2	CB	Cirsio-Brachypodion	E1.23	similar to TWW 18, also used as hay meadow	12
3	FP	Festucion paniculatae	E4.3	similar to TWW 13; also mapped by Hegg et al.	7
4	LL	(low diversity, low altitude)	E2.2	contains different types, promising diversity when mown, therefore lower range chosen	15
5	AI	Agropyrion intermedii	E1.2	transitional type	15
6	SP	Stipo-Poion	E1.24	pastures/fallows in large inner-alpine valleys; CLempN based on national expert-judgment (Hegg et al. 1993)	10
7	MBSP	Mesobromion / Stipo-Poion	E1.26	similar to TWW 18, 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)	12
9	MBXB	Mesobromion / Xerobromion	E1.26	similar to TWW 18	12
10	LH	(low diversity, high altitude)	E2.3	contains different types of dry grassland at high altitude	12
11	CF	Caricion ferrugineae	E4.41	(sub-)alpine grassland; also mapped by Hegg et al.	7
12	AE	Arrhenatherion elatioris	E2.2	often used as meadows, lower range chosen as it occurs at all altitude levels	12
13	FV	Festucion variae	E4.3	(sub-)alpine grassland, middle of the range chosen	7
14	SV	Seslerion variae	E4.43	alpine grassland, middle of the range chosen; also mapped by Hegg et al.	7
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	15
18	MB	Mesobromion	E1.26	genuine semi-dry grassland	12

Critical loads of acidity for forests

Critical loads of acidity were assessed by means of a variant of the Simple Mass Balance (SMB) model also considering the extensions listed in the Mapping Manual (Chapter 5.3, UNECE, 2004). To allow weathering rates to be consistently calculated for conditions at critical load, the Sverdrup-Warfvinge Weathering (SWW) algorithm (i.a. Sverdrup & Warfvinge, 1995) was linked to the SMB (version March 23, 2013, M. Posch, CCE, pers. comm.).

Critical chemical limits

On the basis of results from the long-term monitoring of forest sites (inter-cantonal long-term forest monitoring network, including i.a. soil profile analysis, soil solution analysis, forest condition assessment, ground vegetation relevés) and on the basis of published results on relationships between base saturation and storm-induced forest damages as well as fine root conditions (Braun et al. 2003, Braun et al. 2005) we came to the conclusion that a critical limit value of the Bc/Al ratio of 1 allows for too much acidification and weakening of forests stands in Switzerland. Taking the Bc/Al ratios resulting from soil solution monitoring and considering its relation to base saturation (Braun 2013) we concluded that a critical limit value for Bc/Al of 5-10 would be more appropriate to protect forests from acidification since it would not allow, like for Bc/Al=1, a development of base saturation towards values substantially below 20%. Thus, our revised critical loads of acidity for forests are based on calculations with a critical limit value for Bc/Al ratio of 7.

Input

Due to the extension of the SMB with the SWW algorithm, the list of needed input parameters got slightly larger than in earlier assessments (see Table CH.6). Compared to the submission in 2011, an additional 51 sites (current total 311) were considered in the modelling and a series of basic data was brought up-to-date in recent years entailing changes in the model input.

Climate input was drawn from revised site-specific monthly climate data (Remund et al., 2014) for a past 1961-1990 and future 2045-2074 period adopting an IPCC A1B scenario. For critical loads calculations the data were annualized for each of the 30 years period (i.e. input is 30 years annual average).

Wet and dry deposition rates for base cations (Bc), Na and Cl were interpolated by spatial regression on the basis of monitoring results from the Long-term Forest Ecosystem Research Programme of WSL (http://www.wsl.ch/info/organisation/fpo/lwf/index_EN). They represent an average of the period 2006-2009 (Rihm et al. 2013). Deposition of base cations is input to MakeDep, which was used to simulate forest growth and management and resulting nutrient cycle. Annual harvest and corresponding nutrient contents were taken from an up-to-date MakeDep run. Net uptake of base cations and nitrogen was calculated as the sum of tree compartment mass removed from the plot (harvest) times the average nutrient contents of the compartments. Since critical loads are being used to set future emission/deposition targets and to remain consistent with the climate input, it was decided to use average annual deposition and nutrient flux output from MakeDep for the period 2045-2074.

In the course of integrating the 51 new sites into the database and in conjunction with the implementation of the weathering calculation routine, soil input required by the extended SMB was completely revised. For the current submission we considered (cp. Phelan et al., 2014)

- a modification of the assessment of the major rooting zone, which defines the single soil compartment required by the SMB,
- a modification of the weatherable surface area estimation,
- a modification of the area weighting of the mineralogy,
- the introduction of a stoichiometry correction for base cation depleted clay minerals,
- a harmonisation of the assessment of long-term average soil moisture content and porosity, which determine water saturation and thereby wetted mineral surface.
- Finally, instead of averaging the layered soil input within the rooting zone, transfer functions used to get from soil raw data to the requested soil input were now applied to averaged raw data.

Table CH.6 List of input parameters required to run the SWW/SMB.

Key word	Unit	Comment
SiteInfo	-	string with info on the site (max.128 chars)
useSWW	-	flag; 0=weathering rates given; 1=steady-state weathering rates computed with SWW
AciCrit	-	Criterion for acidity CLs; 1=Al:Bc (mol mol^{-1}); 2=[Al] (molc m^{-3}); 3=bsat (fraction); 4=pH (mol L^{-1}); 5=[ANC] (molc m^{-3})
Vacicrit	-	Critical value for criterion 'AciCrit'; units as given under 'AciCrit'
NutCrit	-	Criterion for CLnutN; 1=[N]acc (mgN L^{-1}); 2=Nle,acc (molc m^{-2} a^{-1})
Vnutcrit		Critical/acceptable value for criterion 'NutCrit'; units as given under 'NutCrit'
thick	M	thickness of the soil compartment
porosity	m^3 m^{-3}	porosity of the soil
Theta	m^3 m^{-3}	volumetric water content of the soil
lgKAlox	(mol $L^{-1})^{-2}$	log10 of equilibrium constant in [Al] = KAlox*$[H]^3$
lgKAlBC	-	log10 of Gapon selectivity constant for Al-Bc exchange
lgKHBC	-	log10 of Gapon selectivity constant for H-Bc exchange
pCO2fac	-	CO_2 pressure in soil solution as multiple of pCO_2(atm) in air
cRCOO	mol m^{-3}	total concentration of organic acids (m*DOC); (0=no organic acids simulated)
TempC	°C	soil temperature
percol	m a^{-1}	percolation (precipitation surplus) (m/a)
f_de	-	denitrification fraction (0<=f_de<=1)
Nim_acc	molc m^{-2} a^{-1}	'constant' (acceptable, minimum) N immobilized
Ca_dep	molc m^{-2} a^{-1}	deposition of Ca
Mg_dep	molc m^{-2} a^{-1}	deposition of Mg
K_dep	molc m^{-2} a^{-1}	deposition of K
Na_dep	molc m^{-2} a^{-1}	deposition of Na
Cl_dep	molc m^{-2} a^{-1}	deposition of Cl
Ca_upt	molc m^{-2} a^{-1}	net uptake of Ca
Mg_upt	molc m^{-2} a^{-1}	net uptake of Mg
K_upt	molc m^{-2} a^{-1}	net uptake of K

Key word	Unit	Comment
N_gupt	molc m^{-2} a^{-1}	net uptake of N
Ca_we	molc m^{-2} a^{-1}	weathering rate for Ca
Mg_we	molc m^{-2} a^{-1}	weathering rate for Mg
K_we	molc m^{-2} a^{-1}	weathering rate for K
Na_we	molc m^{-2} a^{-1}	weathering rate for Na
surface	m^2 m^{-3}	soil particle surface area
MinDat	-	Path to PROFILE-style 'mineraldata' file {mineraldata}
M_groups	-	number of mineral groups used (first M_groups of those in MinDat)
M_fracts	m^2 m^{-2}	surface area fractions of minerals in M_groups

Determining the ecosystem area

Critical loads of acidity were successfully calculated for 301 DM-sites. These are not regularly distributed within the country. The NFI-sites (National Forest Inventory), however, are a systematic sample, each representing a forest area of 1 km^2. Therefore, the area of forest represented by one DM-site was determined by those NFI-sites situated within the respective Thiessen-polygon constructed for the DM-sites, and all acidity parameters were copied from a DM-site to the affiliated NFI-sites. In consequence, EcoArea was set to 1.0 km^2 for all resulting sites with critical loads for acidity.

However, if a NFI-site was situated on a 1x1 km grid cell containing also a site with empirical critical loads, EcoArea was set to 0.8 km^2 for the NFI-site and to 0.2 km^2 for the empirical site. Thus, double area counts were excluded.

Critical loads of acidity for alpine lakes

Critical loads of acidity for alpine lakes were left unchanged. They 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.

Habitat Suitability Index

Progress was made in preparing the required vegetation data for the well-monitored forest sites. Well-monitored sites were selected with the purpose to be able to compare modelling results with field observations. There are now two sets of vegetation data, one showing the current site-specific vegetation composition according to recent relevés and the other highlighting the vegetation composition reflecting the natural “undisturbed” situation for the respective habitat type according to expert judgement. The plant species were parameterized according to the ecological indicator values given in the Swiss Flora Indicativa (Landolt et al 2010). A customised Veg database will be established on the basis of this parameterization. Dynamic modelling with VSD-Veg and VSD+-Veg, respectively, and calculation of biodiversity critical loads with SMB-Veg is planned.

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Introduction

In response to the "CCE Call for Data 2014-15" the UK NFC has:

- carried out minor updates to the UK critical load database;
- applied the MADOC-MultiMove model chain to calculate critical loads based on a habitat quality metric for 40 sites. Details are provided below.

Updates to UK critical load database

The UK critical loads data for terrestrial habitats are mapped nationally on a 1x1km grid of the Ordnance Survey British National Grid. For the data submission these data are referenced by the longitude-latitude for the centre point of each 1x1km grid square. The critical loads data for the 1752 freshwater catchments have been sub-divided to the same grid resolution for consistency with the terrestrial data and to ensure future compatibility with the new EMEP grid resolutions.

In previous years the NFC has submitted empirical nutrient nitrogen critical loads for the designated features of Natura 2000 sites, i.e. Special Areas of Conservation (SACs) and Specially Protected Areas (SPAs) (Hall et al., 2011). However, as these sites can overlap with the UK broad habitat critical loads data they could not be used by the CCE due to double counting of habitat areas in assessments. To overcome this, the nutrient nitrogen critical loads for broad habitats, SACs and SPAs have been integrated into a single database, without duplicating the areas. This has been achieved by:

- Identifying the designated features that are the same EUNIS class as the UK broad habitats.
- Identifying the 1x1km squares that contain individual UK broad habitats and all or part of any SAC and/or SPA.
- Assigning the appropriate nutrient nitrogen critical load for each relevant EUNIS class to each 1x1km square, using the lowest value if there are differences between the values for the broad habitat, the SAC and/or SPA.
- Assuming that the habitat area for the designated feature habitat within the 1x1km square is the same as the area that has been

mapped for that broad habitat. This is necessary as spatial data on the location and areas of designated feature habitats within sites is not available.
- Setting the "protection" score for the 1x1km squares according to the codes provided by the CCE (1: SPA, 2: SAC, 3: SPA and SAC, -1: protection status unknown).

The critical load values applied to feature habitats of UK SACs and SPAs are values (within the published ranges) agreed nationally for use in air pollution impact assessments. For some habitats these values will be the same as the "UK mapping values" applied to broad habitats and based on UK evidence; where no UK evidence exists, the values may be based on expert opinion or set to the minimum of the published range.
It should be noted that the resulting database tables do not include: (a) designated feature habitats that are not mapped nationally; (b) areas of SACs/SPAs that fall outside of the broad habitat areas mapped nationally. In total 13.3% of the UK 1x1km critical load records submitted for nutrient nitrogen represent the designated feature habitats of SACs and/or SPAs (Table GB.1).

Table GB.1: The percentage of UK 1x1 km broad habitat grid squares that contain designated feature habitats of SACs and/or SPAs.

EUNIS class	% of 1x1 km squares in the following categories:				
	Broad habitat only	Broad habitat + SPA	Broad habitat + SAC	Broad habitat + SAC + SPA	Broad habitat + any site combination
A2.5	32.9	17.0	12.3	37.7	67.1
B1.4	60.5	11.4	14.2	14.0	39.5
D1	70.1	2.8	9.1	18.0	29.9
E1.7	94.0	0.5	5.5	0.0	6.0
E3.52	96.6	1.3	2.1	0.0	3.4
E4.2	64.4	2.3	19.5	13.8	35.6
E1.26	94.6	0.0	5.4	0.0	5.4
F4.11	79.3	4.7	6.8	9.2	20.7
F4.2	84.7	1.8	9.7	3.9	15.3
G4	97.1	1.1	1.5	0.3	2.9
G1.6	92.2	0.0	7.8	0.0	7.8
G1.8	88.1	0.0	11.9	0.0	11.9
G3.4	71.4	0.0	28.6	0.0	28.6
All the above	86.7	2.2	6.5	4.6	13.3

The UK database includes acidity and nutrient nitrogen critical loads for a number of different woodland categories (EUNIS classes G1, G1.6, G1.8, G3, G3.4, G4). The methods used to derive the critical loads for these (and all other UK habitats) are described in detail in Hall et al. (2015). The UK NFC has received acidity and nutrient nitrogen critical loads data for 167 UK forest plots from ICP Forests, however these have not been incorporated into the UK database since they currently lack additional data and information to enable a full comparison to be made between the UK methods and results and those used by ICP Forests.

Biodiversity-based critical loads

The methods and results applied to calculate biodiversity-based critical loads are summarised here. A more complete description of the study can be found in Rowe et al. (2015).

Introduction

Air pollution by sulphur (S) and nitrogen (N) causes soil acidification, and nitrogen has additional effects on ecosystems through mechanisms such as eutrophication and formation of ground-level ozone. Substantial reductions in S pollution since the 1980s have led to a widespread recovery from acidification (Emmett et al., 2010) except on some weakly-buffered soils (Evans et al., 2012). Nitrogen pollution has also decreased, but by a smaller proportion. The current approach to assessing effects of N pollution is based on its contribution to acidification, using a comparatively simple mass-balance approach; and on its eutrophying and other effects, which are summarised using the "empirical critical load" approach. Empirical critical loads for N have been established by assessing evidence from experiments and some survey studies (Bobbink and Hettelingh, 2011). However, experimental studies may not capture the medium-term and long-term effects of N, since the effects of N deposition can be persistent and cumulative, and at many sites changes induced by N are likely to have already occurred when the experiment started. Also this approach does not adequately represent the combined effects of N and S pollution. For these reasons, the CCE has encouraged the development of dynamic modelling approaches that capture the combined effects of air pollution on biodiversity (e.g. Hettelingh et al., 2008). Progress was initially slow due to lack of consensus on how the outputs from such models (e.g. changes in habitat-suitability for each of a large set of plant and lichen species) should be interpreted in terms of policy targets such as "no net loss of biodiversity". However, work funded by Defra under the AQ0828 and AQ0832 projects (Rowe et al., 2014a; Rowe et al., 2014b) has defined an index of Habitat Quality (HQI) for use in this context, i.e. mean habitat-suitability for positive indicator-species. Here we describe the application of this index to the dynamic modelling of N and S impacts.

The third aim of the CCE Call for Data 2014-15 was to "Apply novel approaches to calculate nitrogen and sulphur critical load functions taking into account their impact on biodiversity. For this, National Focal Centres are encouraged to use the 'Habitat Suitability Index' (HS - index) agreed at the M&M Task Force meeting". This aim was met by applying the habitat quality metric (HQI) developed in the AQ0828 and AQ0832 projects. The MADOC-MultiMOVE model (Butler, 2010; de Vries et al., 2010; Rowe et al., 2014c) was used to determine combinations of N and S likely to cause habitat quality to decline below a threshold, i.e. biodiversity-based critical load functions. This report outlines the approach taken and illustrates this approach for a set of example sites.

Methods

The basis of the study is the capacity to predict changes in habitat suitability for species under different pollutant deposition scenarios, which has been developed by linking dynamic models of biogeochemical change with regression models of habitat-suitability for individual species. The biogeochemistry model used in the current study was

MADOC (Rowe et al., 2014c), essentially a combination of the Very Simple Dynamic (VSD) acid-base chemistry model (Posch and Reinds, 2009) with a simple model of carbon (C) dynamics (Tipping et al., 2012). It is analogous to the VSD+ model (Bonten et al., 2010) which is being developed using a different model of C dynamics to extend VSD, but in the UK model more emphasis has been placed on processes that are important in upland systems and more C-rich soils, such as the production of dissolved organic C. The MADOC model responds to several environmental drivers such as the deposition loads of N and S, and was used to predict changes in soil pH, soil total C/N ratio, and the annual flux of available N from deposition and release from soil organic matter.

The habitat-suitability model used in the current study was MultiMOVE (Butler, 2010). This predicts the suitability of a site for each of around 1300 plant and lichen species, depending on the current environmental conditions. These conditions are expressed using four indicators that are based on trait-means for the species present (mean "Ellenberg R" for alkalinity; mean "Ellenberg N" for eutrophication; mean "Ellenberg F" for wetness; mean "Grime Height" for vegetation height) and three climate-based indicators (minimum January and maximum July temperature, and annual precipitation). The habitat-suitability values predicted by MultiMOVE were rescaled by prevalence in the training dataset, using the method of Real et al. (2006). Values rescaled in this way are comparable among species and can be used to reconstruct a plausible set of plant species for a given site (Rowe et al., 2014a).

Habitat-suitability for a large set of species could be analysed and interpreted in many different ways. The AQ0828 and AQ0832 studies established that the most suitable indicator of overall habitat quality that can be calculated from these outputs is the mean habitat suitability for positive indicator-species. This conclusion was reached following a detailed consultation with habitat specialists of the Statutory Nature Conservation Bodies (Rowe et al., 2014b). In the current study, species-level model outputs were summarised using this Habitat Quality Index (HQI). To calculate N and S critical load functions using such an index requires definition of a threshold value below which the site should be considered to be in damaged or unfavourable condition. To establish this threshold, the value of HQI was calculated under a scenario where N deposition was set to the empirical N critical load (CLempN), using the 'mapping value' for CLempN as determined for each site by the UK National Focal Centre, and no anthropogenic sulphur deposition. The CLempN was originally set, on the basis of evidence and/or expert judgement, at a level intended to avoid damage in the near- and long-term. By running the model chain forward at the critical load for an extended period, the resulting value of HQI can be assumed to correspond to a threshold or critical value. The model chain was run forward to 2100 as recommended by the CCE. This date is a compromise between capturing the effects of N persisting over many decades (although with diminishing impacts) and the increasing uncertainty associated with predicting effects in future centuries.

Using the threshold established in this way, a more complete picture of N and S effects on ecosystems can be obtained, by running the model chain at different rates of N and S deposition to determine which combinations cause HQI to decline below the threshold. The combinations that give HQI = HQIcrit were assumed to correspond to the 'biodiversity-based' Critical Load function which was the goal of the exercise. Such a CL function is illustrated for a hypothetical site in Figure GB.1.

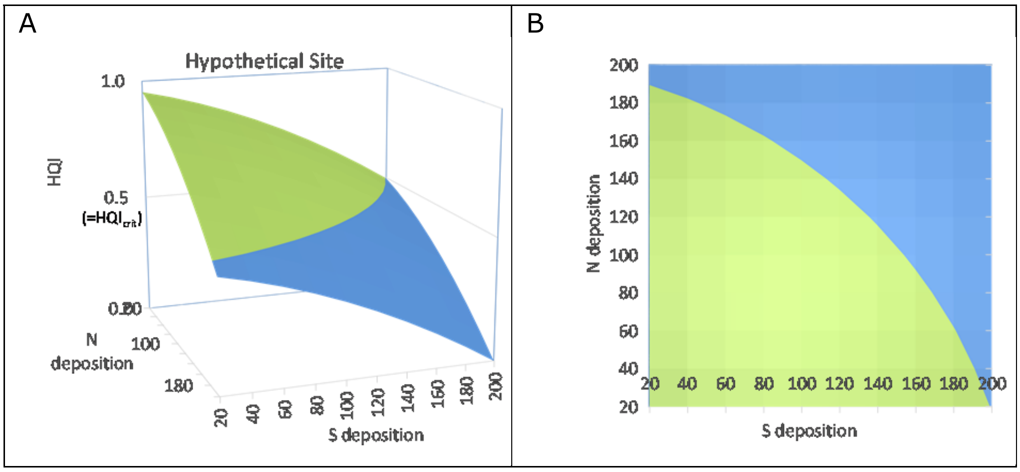

Figure GB.1 Hypothetical response, illustrated in A) three and B) two dimensions, of a habitat quality index (HQI; vertical axis in graph A) to variation in S and N deposition. The light green area represents combinations which maintain HQI above a threshold value, HQIcrit, assumed here to be 0.5. The contour where HQI = HQIcrit corresponds to a 'biodiversity-based' critical load function.

Responses to the Call for Data were requested in the form of two points on the plot, defined by values on each of the S and N deposition axes: CLNmin, CLSmax, CLNmax and CLSmin, as illustrated in the Call for Data instructions (see Appendix A). Clearly such a simple function can only be an approximation of a curvilinear function.
Example sites (Figure GB.2) were chosen from the database of Special Areas for Conservation (SACs) maintained by the NFC. Example SACs with either E1.7 'Closed non-Mediterranean dry acid and neutral grassland' or F4.11 'Northern wet heaths' were selected at random from the database. The MADOC model was set up using deposition sequences for S and N provided by EMEP, and values collated by the UK NFC for climate and soil parameters. The model was calibrated to match present-day values of two key observations, soil pH and soil total C/N ratio, by adjusting parameters whose true value is unknown. The target values for pH and C/N used in the current study were mean values for the broad habitat corresponding to the EUNIS class for the site, as observed in Countryside Survey 2007 (Emmett et al., 2010). Soil pH was matched by adjusting calcium weathering rate or the density of exchangeable protons on dissolved organic carbon. Soil total C/N ratio was matched by adjusting the rate of N fixation during the pre-industrial period. The calibrated model was then run again with N and S deposition set, for the period 1980-2100, to CLempN. The simulated environmental conditions in 2100 were used to calculate habitat-suitability for positive indicator-species, and thence HQI. The HQI under this Critical Load scenario was assumed to correspond to a threshold level for the site, HQIcrit. The model chain was then re-run, to find combinations of N and S deposition below which this HQIcrit value was exceeded.

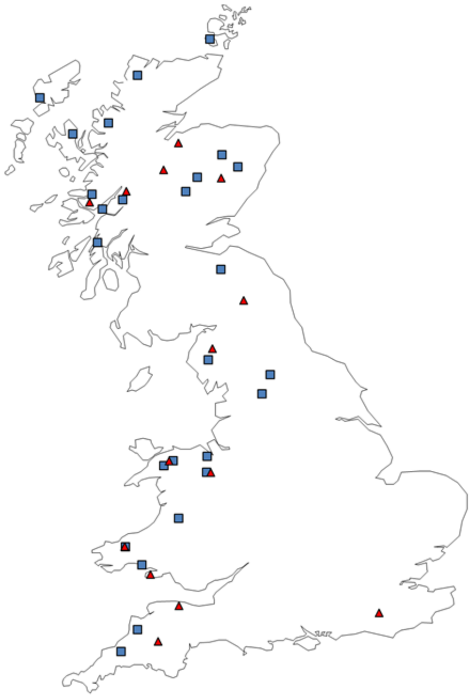

Figure GB.2. Locations of Special Areas for Conservation for which biodiversity-based Critical Load functions were submitted in response to the CCE Call for Data. Blue squares = E1.7 Dry acid grassland; red triangles = F4.11 Wet heath.

The biogeochemical conditions predicted by MADOC for 2100 under three Critical Load scenarios were then used to estimate positions on each of the gradients that define habitat-suitability for species in the MultiMOVE model. These gradients are mean values for floristic traits – for wetness (EW), alkalinity (ER), fertility (EN) and vegetation height (GH). Together with climate variables (maximum July temperature, minimum January temperature and total annual precipitation), these trait-means define the environmental conditions at a site. Biogeochemical conditions were related to trait-means using relationships established from empirical data (Table GB.2).

Table GB.2. Conversion equations used to estimate floristic trait-means (used to predict habitat-suitability for species) from biogeochemical conditions. EW = mean Ellenberg 'moisture' score for species present; ER = mean Ellenberg 'alkalinity' score for present species; EN = mean Ellenberg 'fertility' score for present species; GH = mean Grime 'height' score for present species; MC = soil moisture content, g water 100 g^{-1} fresh soil; pH = soil pH; Nav = available N, g N m^{-2} yr^{-1}; CN = CN ratio, g C g^{-1} N; H = canopy height, cm; Cplant = total plant biomass C. Mean GH was weighted by observed cover or occurrence frequency; other trait-means were not weighted.

Value to be estimated	Calculated as	Source
EW	$\dfrac{ln\left(\frac{\mathrm{MC}}{100-\mathrm{MC}}\right)+3.27}{0.55}$	Smart et al. (2010)
ER	$\dfrac{\mathrm{pH}-2.5}{0.61}$	Smart et al. (2004)
EN	$0.318\,log_{10}\,\mathrm{N_{av}}+1.689+\dfrac{284}{\mathrm{CN}}$	Rowe et al. (2011)
G_H	$\max(1, 1.17\times ln\,\mathrm{H}-1.22)$	Rowe et al. (2011)
H	$\left(\dfrac{\mathrm{C_{plant}}}{14.21\times 3}\right)^{1/0.814}$	derived from Parton (1978) and Yu et al. (2010)

Whether changes in vegetation height should be included is debatable. If management intensity increases to compensate for extra herbage production, the vegetation height may not change. However, faster closure of gaps is probably a key driver of species loss as systems become more productive, since the diversity of strategies for colonising new gaps is an important factor in maintaining overall plant diversity. This argues for the inclusion of an effect on ground-level light availability of extra biomass production, even if the vegetation height changes little, and this approach was taken in the current study, with simulated changes in height taken into account in the species modelling.

The MultiMOVE model was used to determine the suitability of the site for positive indicator-species for the habitat, under the conditions projected to occur in 2100 under the CL scenario. The model predicts habitat-suitability using several statistical modelling techniques in an ensemble approach (Butler, 2010), and for the current study the model-average habitat-suitability was used. Raw suitabilities predicted by the model were standardised for prevalence in the training dataset using the method of Real et al. (2006). Habitat suitability was estimated for all species that were: a) positive indicator-species for the habitat (see below); and b) present in the surrounding 10x10 km square. The lists of indicator species used to calculate HQI in the original AQ0832 study (Rowe et al., 2014b) were derived from common standards monitoring (CSM) guidance documents (e.g. JNCC, 2006). Judgements were made as to which species to include or exclude as positive indicators. Since that study, lists of positive indicator-species have been made available as a result of a combined effort by the Joint Nature Conservation Committee (JNCC) and the Botanical Society of the British Isles (BSBI) (Kevin Walker, pers. com.) and were used in the current study. However, neither the CSM guidance nor the more recent effort lists species for EUNIS habitat classes, which need to be used for the CCE data submission. To obtain suitable lists for these classes we used

correspondence tables developed under the JNCC AND-UP project (Jones et al., in prep). At a given site, particular positive indicator-species might not be present due to unsuitable climate rather than because of the effects of pollution. To avoid underestimating the overall habitat-suitability for positive indicator-species, species that had never been recorded from a particular grid-square were excluded when calculating the mean habitat-suitability. The records used for this filtering were obtained from the Botanical Society of the British Isles, the British Lichen Society and the British Bryological Society.
Following calibration of MADOC to match the C/N ratio and soil pH values obtained from the NFC database for the soil and vegetation type, this model was run forward to 2100 with deposition set to each of the CL combinations of N and S. The resultant abiotic conditions were used to predict the value of HQI under each of these CL combinations, and the mean value was used as HQIcrit for the site. Ideally, the new CL function would be established by determining the exact combinations of N and S deposition that result in HQI = HQIcrit. Routines to do this could be developed, but would require calibration of the whole MADOC-MultiMOVE chain, which would currently be too time-consuming. Instead, the model chain was run using 10 x 10 combinations of N and S deposition, evenly covering ranges from 20% to 200% of CLempN and CLmaxS, respectively. This allowed the response surface to be plotted, and a contour-fitting routine was applied to interpolate the new CL function. This function was simplified into the two-node form required for responding to the CCE Call for Data, by positioning these nodes so that differences from the interpolated function were minimised within these deposition ranges.

Results

The dynamic effects of different air pollution scenarios extended over the 21st century are illustrated below for the Snowdon acid grassland site. The time course of N and S deposition is shown for i) current legislated emissions, and for three Critical Load combinations: ii) S deposition at CLmaxS; iii) S deposition at CLmaxS together with N deposition at CLminN; and iv) the empirical N critical load, CLempN (Figure GB.3). At this site CLempN is greater than the current legislated emissions, so the CLempN scenario causes relative increases in C/N (due to stimulated production of plant litter with a high C/N), N availability and vegetation height (Figure GB.4). Sulphur pollution was reduced in all the CL scenarios, so these showed increases in pH, although N leaching in the CLempN scenario caused pH to decrease in the longer term.

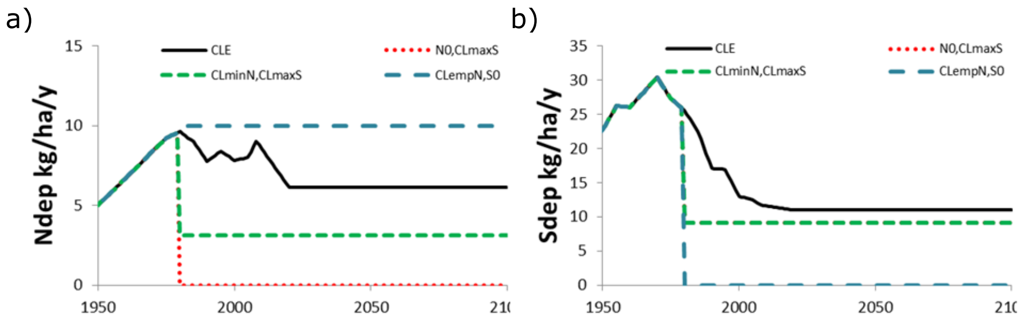

Figure GB.3. Deposition rates of a) nitrogen and b) sulphur at Snowdon (a Welsh acid grassland site) under four scenarios: i) deposition predicted with current legislated emissions under the Gothenberg protocol; ii) N = 0, S = CLmaxS; iii) N = CLminN, S = CLmaxS; and iv) N = CLempN, S = 0.

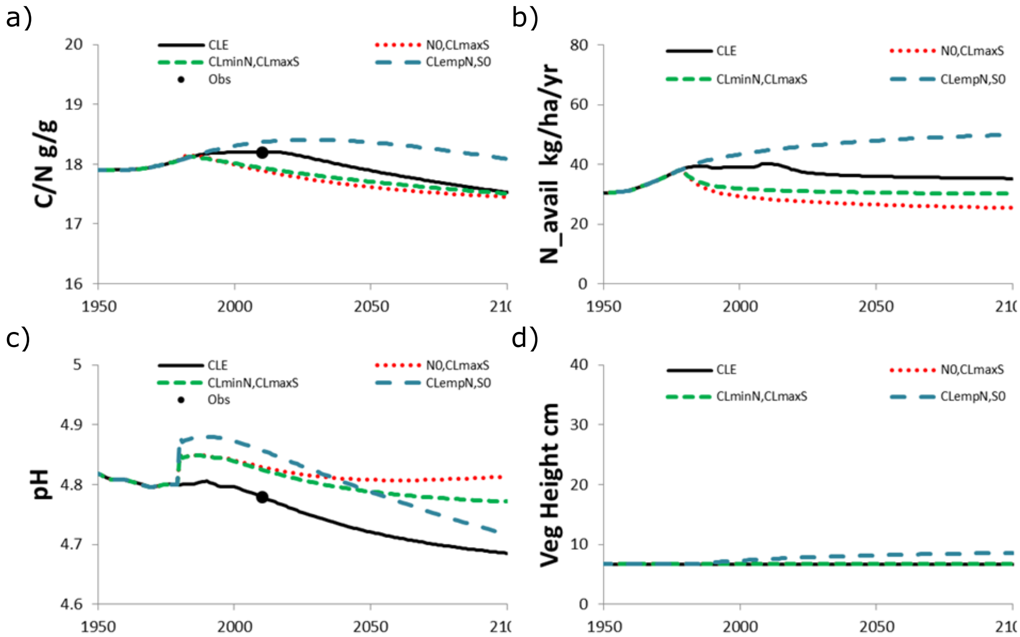

Figure GB.4. Simulated responses at Snowdon (a Welsh acid grassland site) of a) soil C/N, g g^{-1}, b) available N, kg N ha^{-1} yr^{-1}, c) soil pH and d) vegetation height, cm, to four N and S deposition scenarios: i) deposition predicted with current legislated emissions under the Gothenberg protocol; ii) N = 0, S = CLmaxS; iii) N = CLminN, S = CLmaxS; and iv) N = CLempN, S = 0.

The sensitivity of the MADOC-MultiMOVE model chain was explored by varying N deposition over the range of 20-200 % of CLempN and S deposition over the range of 20-200 % of CLmaxS (Figure GB.5). Increases in both N and S caused pH to decline. Soil C/N ratio and plant-available N both increased with greater rates of N deposition but were not affected significantly by S deposition.

a) b) c)

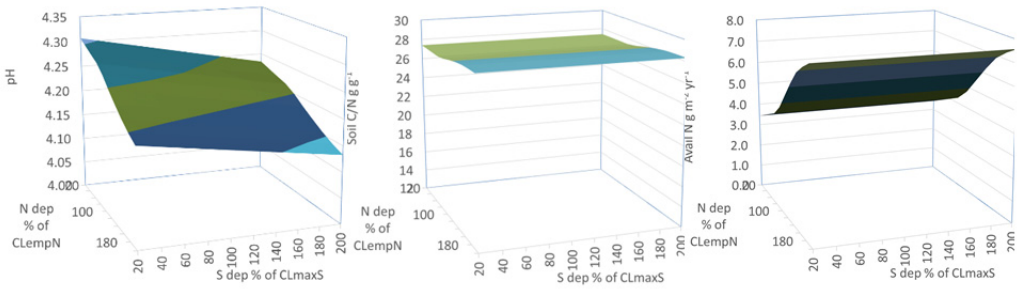

Figure GB.5. Simulated sensitivity of biogeochemical properties: a) pH; b) C/N ratio, g C g^{-1} N; c) plant-available N, g N m^{-2} yr^{-1}, to variation in nitrogen and sulphur deposition at the Whim Moss blanket bog site.

The habitat-suitability for individual species is calculated on the basis of floristic trait-means, the values of which are inferred from biogeochemical properties (see Table GB.2). The sensitivity of the three trait-means that are most responsive to N and S deposition was assessed over ranges of 20-200 % of CLempN and 20-200 % of CLmaxS (Figure GB.6). The response of the alkalinity trait to N and S was similar to the pH response. Trait-means representing fertility and vegetation height both increased with more N deposition but were hardly affected by S deposition.

a) b) c)

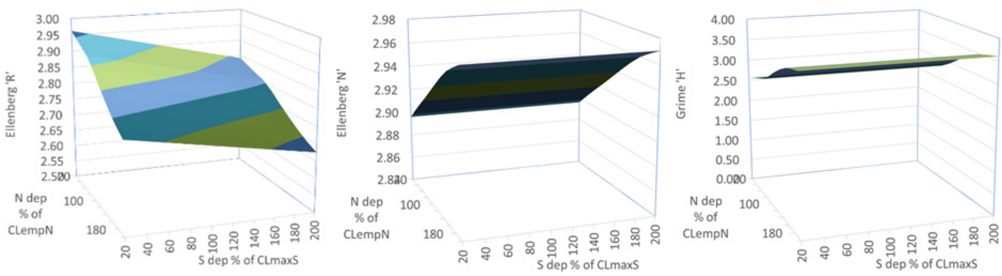

Figure GB.6. Simulated sensitivity of mean values for floristic traits: a) Ellenberg R i.e. alkalinity; b) Ellenberg N i.e. fertility; c) Grime H i.e. height, to variation in nitrogen and sulphur deposition at the Whim Moss blanket bog site.

The trait-mean values calculated above were used to explore the sensitivity of individual species to variation in N and S pollution. Three of the positive indicator-species for blanket bog were selected to illustrate different types of response (Figure GB.7). Habitat-suitability for all three species declined with more N deposition, steeply in the case of Drosera rotundifolia. This species was relatively insensitive to S deposition. The other two species illustrated show contrasting responses to increased S deposition, which made the site more suitable for Vaccinium myrtillus but less suitable for Trichophorum cespitosum.

a) b) c)

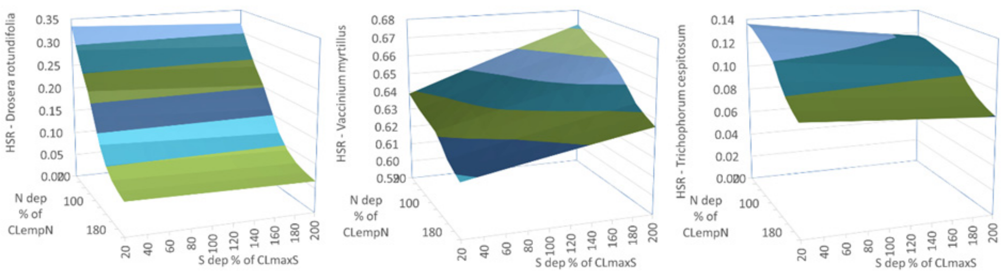

Figure GB.7. Simulated sensitivity of habitat suitability (rescaled by prevalence) for selected positive indicator-species: a) Round-leaved sundew; b) Bilberry; c) Deergrass, to variation in nitrogen and sulphur deposition at the Whim Moss blanket bog site.

The overall response of the habitat was summarised using the HQI metric, i.e. mean habitat suitability (rescaled by prevalence) for all locally-occurring positive-indicator species. The sensitivity of HQI to variation in N and S deposition was assessed over ranges of 20-200 % of CLempN and 20-200 % of CLmaxS for the Whim Moss blanket bog site (Figure GB.8). Although other positive indicator-species were included, the response is similar to the surface that would be obtained by averaging the responses for the three species illustrated in Figure GB.7. Clearly positive and negative responses to S deposition (i.e. principally to acidification) cancelled out, and there was no overall response of HQI to variation in S deposition within this range. By contrast there was a strong overall response of HQI to N deposition (i.e. principally to eutrophication and increased vegetation height), with clear decline at greater N deposition rates. At the Glensaugh wet heath site, HQI declined with both N and S deposition.

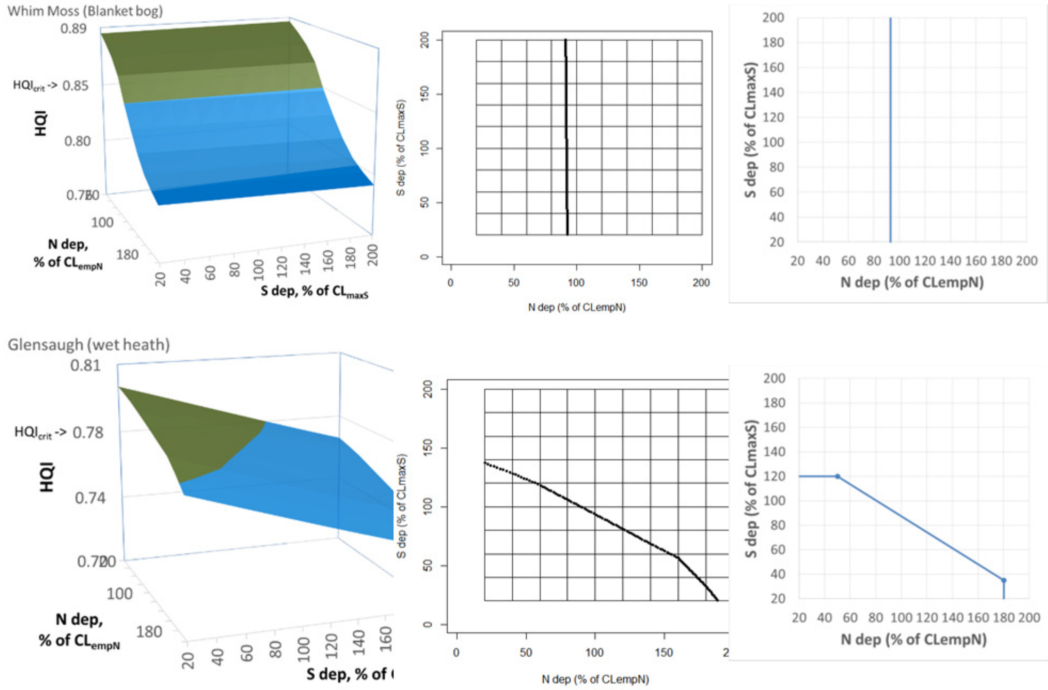

Figure GB.8. Simulated sensitivity of an overall habitat quality index HQI, the mean habitat-suitability (rescaled by prevalence) for locally-occurring positive indicator-species, to variation in nitrogen and sulphur deposition at the Whim Moss (blanket bog) and the Glensaugh (wet heath) sites.

A threshold value for the habitat quality metric, HQIcrit, was determined by calculating the HQI value in 2100 under a scenario with N deposition set to the empirical critical load. Combinations of N and S deposition that result in HQI values below this threshold were assumed to be in exceedance of the biodiversity-based critical load. The biodiversity-based CL function was derived as the line of combinations of N and S deposition that gave an HQI value of exactly HQIcrit. This function is illustrated for in the left-hand and middle columns of plots in Figure GB.8. An approximation of each function, required for the CCE Call for Data response and made by fitting two points on the N x S plane to minimise differences from the exact function, is shown in the right-hand column of plots in Figure GB.8.

The responses of HQI to N and S pollution at the wet heath site illustrated were broadly as expected, in that HQI values declined with both N and S deposition, and it was possible to make an approximate function of the form required for the Call for Data response. At the blanket bog site, HQI declined with N pollution, but changes in S pollution had little effect on HQI. This is presumably due to the combination of two effects. Firstly, the soil pH at the site was calibrated to a typical value for UK blanket bog, 4.51 (Emmett et al., 2010). This is quite acid, and because pH is measured on a negative logarithmic scale, further decreases in pH require substantial additions of acid anions. Secondly, many of the species that are positive condition indicators for blanket bog are typical of acid environments. Although habitat-suitability for such species is expected to decline at very low pH values, these low values were not represented in the MultiMOVE training dataset so the niche models do not show a decline at low pH. It is probably true that naturally-acid habitats are not extremely susceptible to acid pollution, but the model chain may underrepresent the effects of large S loads that reduce pH to unnatural levels.

Following an initial analysis of example sites, it was decided to prepare the revised Call for Data response for only two habitats, E1.7 'Dry acid grassland' and F4.11 'Northern wet heath'. This decision was made partly due to time constraints – biodiversity-based CL functions could be developed for other habitats but would require more exploratory work. The E1.7 and F4.11 habitats are those for which there is currently most confidence in the simulated responses of HQI and in the derived CL functions. Critical loads functions were derived and submitted in the Call for Data response for 26 E1.7 Dry acid grassland sites and 14 F4.11 Wet heath sites. A selection of representative examples is shown in Figure GB.9. Of the wet heathland sites, ten had responses similar to that in Figure GB.9a (i.e. when N deposition is 20% of CLempN, the CLbdiv function was exceeded with S deposition of less than 200% of CLmaxS) and four had responses similar to that in Figure GB.9b (i.e. when N deposition is 20% of CLempN, CLbdiv was exceeded only with S deposition > 200% of CLmaxS). Of the dry acid grassland sites, 18 had responses similar to that in Figure GB.9c, four similar to Figure GB.9d, and four similar to Figure GB.9e (i.e. very sensitive to S pollution, such that CLbdiv was exceeded with only 20% of CLmaxS).
The CLbdiv functions for all sites were approximated using two points on the N x S plane (see Figure GB.8) and these points were submitted on 18th May 2015 as part of the UK response to the Call for Data 2014-15. The submitted data are reproduced in Table GB.3.

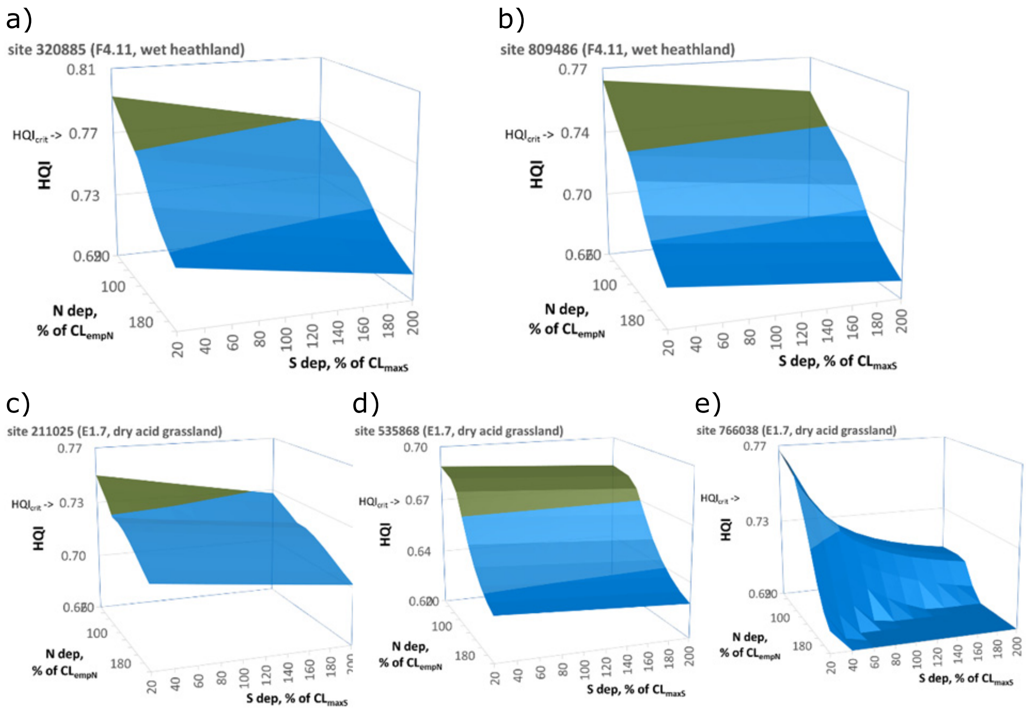

Figure GB.9. Examples of biodiversity-based Critical Load functions, defined as the line where a habitat quality index reaches a critical value and shown as the boundary between green and blue areas in the above plots, for: a) & b) a wet heath site; c), d) and e) a dry acid grassland site. See text for discussion of response types.

Table GB.3. Biodiversity-based Critical Load functions and critical values for the habitat quality metric submitted to the CCE in response to the Call for Data 2014-15. SiteID = UK National Focal Centre code for the 1 x 1 km gridcell and EUNIS habitat; CLNmin, CLSmax = coordinates of first point defining the CLbdiv function; CLNmax, CLSmin = coordinates of second point defining the CLbdiv function; HScrit = critical value for the habitat quality metric, referred to as HQIcrit in the text.

E1.7 Dry acid grassland sites					
SiteID	**CLNmin**	**CLSmax**	**CLNmax**	**CLSmin**	**HScrit**
211025006	178.5	682.5	536	137	0.732
248732006	357	441	643	10.5	0.710
272407006	107.1	798	678	4.2	0.758
299095006	7.14	682	536	4.8	0.754
310949006	7.14	528	571	16	0.775
332841006	7.14	1080	714	144	0.683
345461006	107.1	874	714	19	0.696
356609006	107.1	840	714	20	0.719
371994006	178.5	800	657	140	0.705
374674006	7.14	704	500	110	0.770
380313006	7.14	815	700	16.3	0.777
390087006	357	1326	607	780	0.734
425781006	7.14	612	486	1.7	0.729
453939006	142.8	900	714	180	0.680
549823006	7.14	986	714	16.8	0.787

E1.7 Dry acid grassland sites

SiteID	CLNmin	CLSmax	CLNmax	CLSmin	HScrit
565303006	7.14	1012	714	138	0.705
585592006	214.2	900	657	205	0.725
651322006	7.14	1107	714	492	0.721
656178006	7.14	1056	735	6.4	0.782
661066006	7.14	798	557	5.7	0.823
668121006	335.58	962	664	156	0.666
717085006	7.14	552	571	4.6	0.727
747117006	7.14	684	521	4.5	0.753
766038006	7.14	810	714	40.5	0.768
834632006	285.6	1008	657	252	0.752
857711006	249.9	668	621	267	0.784

F4.11 Wet heath sites

SiteID	CLNmin	CLSmax	CLNmax	CLSmin	HScrit
320885009	7.14	873	707	3.8	0.768
348866009	7.14	533	714	4.1	0.746
358040009	7.14	672	714	8.2	0.729
371918009	7.14	738	728	4.7	0.697
383071009	7.14	1188	714	4.4	0.668
486869009	7.14	907	700	5.6	0.773
537929009	7.14	979	714	5.1	0.662
656873009	7.14	1209	714	7.7	0.669
668827009	7.14	2327	700	16.5	0.747
747116009	7.14	1346	707	4.5	0.703
776549009	7.14	3255	678	15.5	0.756
809486009	7.14	1519	714	3.1	0.736
817443009	7.14	1829	714	3.1	0.753
847259009	7.14	1403	728	8. 4	0.696

Discussion

The MADOC-MultiMOVE model was successfully applied to the task of deriving simple functions that describe combinations of N and S deposition above which the habitat is likely to be damaged. Inevitably the results are a simplification, in that environmental conditions and pollution history at a site are imperfectly known, species occurrence is affected not only by habitat-suitability but by dispersal and extinction processes, and interpretation of species change in terms of conservation targets is inevitably somewhat subjective. Nevertheless, the approach reproduces to a large extent the expected effects of N and S pollution, with changes to individual species dependent on their sensitivity to acidification, eutrophication and/or shading, and, generally, declines in overall habitat quality with greater rates of pollution.

Nitrogen pollution consistently caused declines in habitat quality. Sulphur pollution often had a relatively weak effect, causing little decline in simulated HQI even at rates of 200 % or more of CLmaxS at some sites. This may be because the habitats studied (dry acid grassland and wet heath) are relatively insensitive to acidification – even though soil pH in these habitats can be pushed to low levels by acid pollution, their positive indicator-species are not greatly affected by low pH. However, an alternative explanation is that the models do not capture the negative impacts of very low pH, in particular when very acid sites have not been included in the datasets used to derive species niche models. More exploration of individual species' responses would help in assessing which of these explanations is more correct. However, the negative effects of N via eutrophication and shading seem to be well-captured by the model chain.

The study demonstrated that a model chain that predicts changes in habitat-suitability for individual species can be used to assess the likelihood of biodiversity loss under different pollution scenarios. The model was applied using data held by the UK NFC, showing that predictions can be obtained for any UK 1 km grid square that has been mapped as containing an acid-sensitive or N-sensitive habitat. The biodiversity-based Critical Load functions derived in the study are plausible, showing strong effects of N pollution on habitat quality, and effects of S pollution that depend on the site and habitat's sensitivity to acidification. These effects were not inevitable, but rather emerged from the evidence provided by the responses of individual species. Uncertainties remain with many aspects of the model chain, but considerable progress has been made with applying MADOC-MultiMOVE, and with summarising outputs into forms that can be used in policy analysis and development.

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Appendix A Call for Data 2014/15: Instructions

This appendix is a reprint of the instructions for the 2014-2015 Call for Data

Introduction
At its 33rd session (Geneva, 17-19 September 2014) the Working Group on Effects "…requested the CCE to organise the new call for data and report its results to the thirty-first meeting of the ICP Modelling and Mapping Task Force to be held in Zagreb (Croatia) in 2015 and to the Working Group at its thirty-fourth session" (para. 45; ECE/EB.AIR/WG.1/2014/2 in press)

The document in front of you contains the instructions on how to reply to this Call for Data 2014/15. The call aligns the critical load database to the new longitude-latitude deposition grid of EMEP and comprises critical loads determined by Simple Mass Balance (acidification and eutrophication), empirical critical loads, and critical loads for protecting plant species diversity.

Aims of the Call for Data
The aims of the Call for Data are:
To adapt the critical load database to the 0.50° × 0.25° and 0.1° × 0.1° longitude-latitude grids, used by EMEP, to ensure compatibility of the European critical loads database with these new EMEP grid resolutions;

To offer the possibility to NFCs to update their national critical load data on acidity and eutrophication;

Apply novel approaches to calculate nitrogen and sulphur critical load functions taking into account their impact on biodiversity. For this, National Focal Centres are encouraged to use the 'Habitat Suitability Index' (HS-index) agreed at the M&M Task Force meeting.

Deadline, documentation and other general information
Deadline for submissions is 23 March 2015.
Please email your submission to jaap.slootweg@rivm.nl. The data can be attached to the email, but large data files can also be uploaded; in this case contact Jaap Slootweg for instructions.

All information is also available on our website www.wge-cce.org under News. It is suggested to look there occasionally for updates.
To facilitate the integration into the European database at the CCE, you should use the Access Database template developed by the CCE. This template is described in Section 7.

Note: At the 33rd session of the Working Group on Effects it was "… noted that national data under the ICP Forests on critical loads and underlying variables would be provided to National Focal Centres of the ICP Modelling and Mapping in the coming weeks enabling them to complete the national critical loads data for inclusion in the European critical loads database" (para. 33e; ECE/EB.AIR/WG.1/2014/2 in press).

Therefore, Mapping and Modelling NFCs that receive critical load and underlying data from National Focal Points of the ICP-Forest are encouraged to include this data in their submission to the CCE, provided that the ICP-F data is complete and consistent with the submission instructions from the CCE. For this, NFCs are requested to add the attribute "measured" that the CCE introduced since 2008 to facilitate the incorporation of external data and indication of its origin. For reasons of quality control and assurance the CCE must stress, however, that only the NFCs of the ICP M&M have the responsibility/authority to submit national critical load data to the CCE for incorporation in the European critical loads database.

As with earlier calls for data, NFCs are requested to provide documentation of their submission that will be reported by the CCE in the 2015 reporting. This documentation is of importance for the justification of the use of your data in support of (European) air pollution abatement policies. Your documentation should focus on the data sources and methods applied in your country. You can consider the methods described in the Mapping Manual (www.icpmapping.org) as thoroughly documented; there is no need to repeat them; please enter as reference for the Mapping Manual 'ICP M&M, [yr]. Mapping Manual, www.icpmapping.org, accessed [day mon yr]', like in this document.

The CCE reporting requirements are best served by sending a Word document with a plain single-column layout. Please avoid complicated formatting of your text, tables and figures: E.g., no special fonts; also, figure captions should be plain text and not part of the figure! Please use the dot as a decimal separator and the comma as thousands separator.

The RIVM publication department has formulated stricter demands for figures to improve the electronic accessibility of your contribution by visually impaired readers: Please apply sufficient contrasting colours in your figures. In addition to a figure caption, please add an additional description of the figure especially for the blind; this text should have no references to colours, but a plain description of what is in the picture, e.g. "A map showing the heathlands in the centre of the Netherlands." This text should be entered as 'Alt Text', which can be found (in Word) by right-clicking the mouse on the figure, selecting 'Format…' and select the Alt-text tab when formatting a picture. You also may submit a separate document containing these extended captions.
The final layout will be done under CCE supervision by the RIVM publication department.

Types of Critical Loads and how to submit them
In the history of critical loads, the ICP M&M developed several methods of determining the maximum load for sulphur and/or nitrogen that would not harm an ecosystem. The 'classical' critical loads protected ecosystems against acidification or eutrophication. The limits to the maxima of the critical loads were (most often) concentrations in the soil solution involving the elements H, Al and the base cations for acidification. A simple mass balance (SMB) allows then to calculate the maximum of S and N depositions. Since 2005 we include empirical critical loads in the databases. In recent years, we aimed at developing

critical loads based on biodiversity. We now distinguish four types of critical loads (variable names are also used in the Tables):

Critical loads of acidity (CLacid): The maxima for S and N (See Figure 5.1 in the Mapping Manual (ICP M&M) are given by CLmaxS, CLminN and CLmaxN and generally computed by the SMB model.

Critical loads of nutrient nitrogen (CLnut): For eutrophication the nitrogen deposition is limited by a maximum concentration of N in the soil solution; also here an SMB is applicable. This deposition is denoted as CLnutN.

Empirical critical loads (CLemp): Based on observed effects in ecosystems at different nitrogen depositions (mostly by addition experiments) a maximum nitrogen deposition was agreed upon by experts for more than 40 ecosystem types. These depositions are called empirical critical loads, CLempN (Bobbink and Hettelingh, 2011).

Biodiversity critical loads (CLbdiv): Vegetation modelling can be used to establish limits of chemical variables (e.g., a minimum pH and maximum N concentration) at which typical/desired/key plant species for a habitat/ecosystem can thrive/survive. Values for N and S deposition combinations, i.e. critical loads, can then be derived with soil-chemical models (e.g. SMB) and associated data. These biodiversity N and S critical loads are named (in analogy to acidification) CLNmin, CLSmax, and CLNmax, CLSmin (see Figure 1).

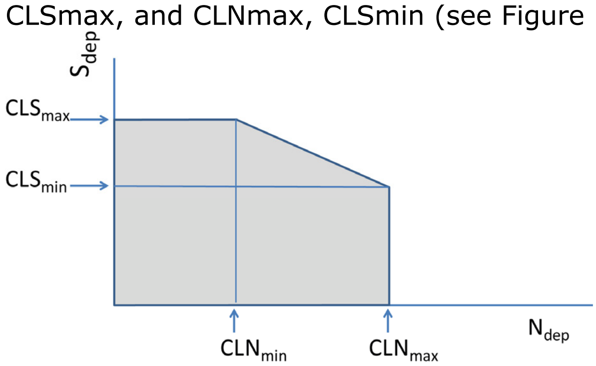

Figure 1. Critical load function for plant diversity, characterised by the two points (CLNmin,CLSmax) and (CLNmax,CLSmin).

Integrating critical loads into a single nitrogen-sulphur critical load function

The four types of critical loads distinguished in the previous paragraph are defined by 9 variables in total (see Figure 2). In a strict interpretation of the definition of a CL of N and S, the final critical load would be the minimum of the individual CLs. Such a final N-S critical load function is illustrated in Figure 2: the darkest area represents combinations of Ndep and Sdep for which none of the critical loads is exceeded. NFCs should not perform this minimization (unless for national purposes). This minimization might be carried out by the CCE when investigating the exceedances of the different CLs.

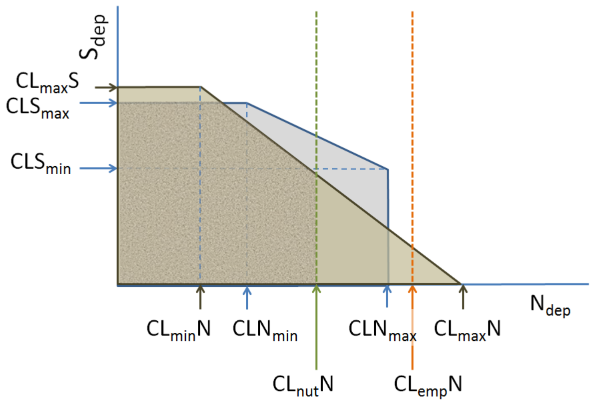

Figure 2. The 9 critical load quantities asked in this Call (see Tables below): CLnutN, CLempN, the 3 quantities defining the acidity CL function (CLmaxS, CLminN, CLmaxN) and the 4 quantities defining the new biodiversity CL function (CLNmin, CLSmax, CLNmax and CLSmin). The darkest area shows the 'minimal critical load function', i.e. all combinations of Ndep and Sdep, for which none of the CLs is exceeded.

The grid system

An ecord is the part of an ecosystem that lies entirely in a single 0.10°×0.05° Longitude-Latitude grid cell. A grid cell is referred to by its lower-left (south-west) grid coordinates in decimal degrees. You will need to overlay the new grid with your maps containing the data to determine the locations (and potentially divisions) of your critical loads. Most countries apply ArcGIS for the spatial operations. In the Annex 1 to this document[4] you can find detailed instructions for generating the grid, including useful attributes. Following this procedure for your country will help integrating the data into the CCE data base.

Access Database template

The Tables in the database have different purposes and are listed below.
ecords – General site data like coordinates.
CLacid, CLnut, CLemp, CLbdiv – Critical loads, one table for each type, with its related limits.
SiteInfo – General background data for the site.

Table 1. Attributes 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)
EcoArea	Area of the ecosystem within the grid cell (km2)	3)

[4] Not printed in this report

Variable	Explanation	Note
Method	The sum of: 1 – Classical SMB for acidification 2 – Classical SMB for nutrient nitrogen 4 – Empirical critical load 8 – Critical loads derived from plant species diversity protection	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) 4: SPA or SAC (1 or 2) [don't know which one(s)] 9: A national nature protection program applies (but not 1 to 4!) -1: protection status unknown	
EUNIScode	EUNIS code, max. 6 characters	5)

Notes on Table 1 (see last column):
Use integer values only (4-bytes)!
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. 48.533 for 48º31', etc.);
Please remove spurious records with an ecosystem area smaller than 1 ha, unless it has relevance other than for exceedance calculations (e.g. a Natura 2000 site). Furthermore, make sure that the ecosystem area does not exceed the size of the land area of your country in the respective grid cell;
Consider all methods and add the number. E.g., if you applied SMB for acidification and empirical critical load (but neither estimated a CLnutN nor a CLbdiv) this should be 5;
You can find information on EUNIS at http://eunis.eea.eu.int/

Table 2. Attributes of the database-table 'CLacid'.

Variable	**Explanation**
SiteID	Identifier of the site (see ecords Table)
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})
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'

Table 3. Attributes of the database-table 'CLnut'.

Variable	**Explanation**
SiteID	Identifier of the site (see ecords Table)
CLnutN	Critical load of nutrient nitrogen (eq ha^{-1} a^{-1})
cNacc	Acceptable (critical) N concentration for CLnutN calculation (meq m^{-3})

Table 4. Attributes of the database-table 'CLemp'.

Variable	**Explanation**
SiteID	Identifier of the site (see ecords Table)
CLempN	Empirical critical load of nitrogen (eq ha^{-1} a^{-1})

Table 5. Attributes of the database-table 'CLbdiv'.

Variable	**Explanation**
SiteID	Identifier of the site (see ecords Table)
CLNmin	Minimum critical load of nitrogen (eq ha^{-1} a^{-1})
CLSmax	Maximum critical load of sulphur (eq ha^{-1} a^{-1})
CLNmax	Maximum critical load of nitrogen (eq ha^{-1} a^{-1})
CLSmin	Minimum critical load of sulphur (eq ha^{-1} a^{-1})
HScrit	Critical value of the Habitat Suitability Index

Table 6. Attributes of the database-table 'SiteInfo'.

Variable	**Explanation**
SiteID	Identifier of the site (see ecords Table)
nANCcrit	The quantity –ANCle(crit) (eq ha^{-1} a^{-1})
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})
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 (log10) (The variable formerly known as Kgibb)
expAl	Exponent for the Al-H relationship (=3 for gibbsite equilibrium)
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 ($0 \leq fde < 1$) (-)
Nde	Amount of nitrogen denitrified (eq ha^{-1} a^{-1})
Slope	(°)
Aspect	Angle between North and the perpendicular line of the slope (degrees up to 360°, measuring clockwise) (°)
Prec	Precipitation (mm a^{-1})
TempC	Temperature (°C)
Theta	Water/moisture content [m^3 m^{-3}]
Altitude	Above sea level (m)
Corg	Organic carbon content (%)
Sand	% sand in the soil
Clay	% clay in the soil
Bsat	Base saturation (-)
Cpool	Amount of carbon in the topsoil (g m^{-2})

Variable	Explanation
CNrat	C/N ratio in the topsoil (g g^{-1})
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 programme. (if more than one of the listed possibilities applies, add the numbers!)

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