

Developments in deriving critical limits and modeling critical loads of nitrogen for terrestrial ecosystems in Europe

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ABSTRACT

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This collaborative report of Alterra and the Coordination Center for Effects (MNP-CCE), in cooperation with various participants of the International Cooperative programme on Modelling and Mapping (ICP-MM) includes:

1. A summarizing overview of adverse nitrogen deposition effects on terrestrial ecosystems in terms of impacts on plant species and faunal biodiversity, forest nutrient status in relation to impacts on soil and solution chemistry and on ground water quality.
2. An overview of integrated dynamic biogeochemical models with plant species diversity models, that allow the assessment of critical loads and target loads of nitrogen in view of plant species diversity impacts.
3. A review of currently used critical limits for N concentrations in soil solution and derivation of new critical limits, based on field (literature) data and integrated soil vegetation models, that can be used in the computation of critical loads by steady state soil models.

This updated knowledge of N effects, critical N load methodologies (integrated models) and critical N limits can contribute to a more appropriate data submission on critical N loads by the National Focal Centres to the CCE, specifically in view of biodiversity impacts, to be used for support of the UNECE and EU air pollution policies.

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Preface

This report is the result of a collaborative project between Alterra and the Coordination Centre for Effects (CCE), including comments and texts from international experts. It presents an overview of:

- Adverse impacts of elevated nitrogen deposition on terrestrial ecosystems and related empirical critical nitrogen loads (Chapter 2).
- Modelling approaches for the derivation of critical nitrogen loads for plant species diversity impacts (Chapter 3).
- Critical nitrogen limits: evaluation of currently used values and possible updated values for use in critical load assessments (Chapter 4).

A first draft of this report was presented during a two-day nitrogen session organized by the CCE at its 15th workshop (Berlin, 25-27 April 2005). The focus of that session was to update knowledge on the effects of nitrogen (Chapter 2) and to review presently used critical limits (Chapter 4). Issues which were of interest to the session included: (i) establish what are harmful effects of nitrogen; present observation trends and (ii) review of currently used critical limits in the computation of critical loads, including proposals for a possible revision using material from experiments and observations and (iii) how new limits can be derived from dynamic models and used in steady- models for the assessment of effects of air pollution, including e.g. changes of biodiversity. Participants to the workshop were then invited to comment on the first draft and/or provide contributions in order to improve the document.

The second draft was presented at the “UN-ECE International Workshop on Nitrogen” in Brighton, UK, November 2005. This version included revisions based on comments made by NFCs (especially from Switzerland) and descriptions of models used in Sweden and Germany, in addition to Dutch modelling approaches (Update of Chapter 3). A third draft version of this report was input to the 16th CCE workshop (Slovenia, 3-5 April 2006), which focused on methods and data regarding nitrogen critical loads and dynamic modelling. Compared to the second draft, it also included a description of a model chain used in the UK. That report thus presented a complete overview of modelling approaches used to predict plant species diversity in the Netherlands, UK, Germany and Sweden. Furthermore, the validation status of these models was described and their potential to calculate critical nitrogen loads (information in various annexes).

This final draft includes mainly an update of: (i) the assessment of nitrogen deposition impacts (chapter 2) and the derivation of critical limits, as presently used in the manual and in the model FOrSAFE-VEG (Chapter 4 and the Annexes 8 and 9). The report includes an overview of new critical limits that can be considered by NFCs for the review and possible revision of national critical loads for eutrophication. Furthermore, a systematic overview is given of the data requirements of the various models.

The report presents the state of current knowledge in the modelling of critical loads and of impacts of changes of exceedances. Its objective is to support CCE calls for data from National Focal Centres of Parties under the Convention on LRTAP and EC Member States in view of a possible revision of the Gothenburg protocol (CLRTAP) and Thematic Strategy for Air Pollution (EC).

The overarching aim of the new work ahead of us is to improve the knowledge of nitrogen impacts on biology-endpoints such as biodiversity. NFCs can benefit from the information in this document for:

- Preliminary applications of a broader range of critical limits in SMB modelling to address biodiversity as proposed in the N-background-document.
- The exploration of the possibility for dynamic modelling applications of changes in exposure to eutrophication applying based on models described in this document.

The responsibility for the various chapters is as follows

- Chapter 1; Introduction: Wim de Vries and Jean Paul Hettelingh
- Chapter 2; Adverse impacts of elevated nitrogen deposition on terrestrial ecosystems: Wim de Vries, Hans Kros, Roland Bobbink, Bridget Emmet and Arjen van Hinsberg.
- Chapter 3; Modelling approaches for the derivation of critical nitrogen loads for plant species diversity impacts; Section 3.1: Wim de Vries
- Section 3.2; The SMART2-SUMO-MOVE/NTM model with Annex 1 and 2: Wiegner Wamelink, Hans Kros, Gert Jan Reinds, Max Posch and Wim de Vries
- Section 3.3; The MAGIC-SUMO-GBMOVE model with Annex 3: Simon Smart, Chris Evans and Wim de Vries
- Section 3.4; The VSD/SMB-BERN model with Annex 4 and 5: Angela Schlutow, Philipp Kraft and Wim de Vries
- Section 3.5; The ForSAFE-VEG model with Annex 6, 7 and 8: Harald Sverdrup, Salim Belyazid and Wim de Vries
- Chapter 4; Critical nitrogen limits with Annex 9: evaluation of currently used values and possible updated values for use in critical load assessments: Wim de Vries, Hans Kros, Gert Jan Reinds and Harald Sverdrup.
- Chapter 5; Discussion, conclusions and recommendations: Wim de Vries, Hans Kros, Arjen van Hinsberg, Simon Smart, Chris Evans

We expect that this report will contribute to new results of the International Cooperative Programme on the Modelling and Mapping of critical levels and Loads and Air Pollution Effects, Risks and Trends and its support of European Air Pollution Abatement Policies in the coming years.

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Summary

Background of the report

Using currently accepted critical loads it turns out that acidification has diminished since 1990, in particular due to the reduction of sulphur emissions. This has led to a decrease of ecosystem areas in pan-Europe where critical acid loads are exceeded from about 48% in 1980 to about 12 % in 2000 using latest deposition patterns available from EMEP, the latest critical load database and IIASA assessments of current legislation in 2006. However, nitrogen emissions continue to be high, specifically in areas with large emissions of ammonia. Ecosystem areas that are at risk of eutrophication according to currently available critical N loads still cover about 45 % of Pan-European ecosystems in 2000 (38% in 1980). Of ecosystem areas in the EU25 the percentages are even 65 % in 2000 compared to about 80% in 1980.

Considering the importance of N deposition, this report presents an overview of:

- Adverse nitrogen deposition effects in terms of impacts on terrestrial ecosystems, with special emphasis on forests and related critical nitrogen loads.
- Integrated dynamic biogeochemical models, that allow the assessment of critical loads and target loads of nitrogen in view of plant species diversity impacts.
- Currently used and updated critical limits for N concentrations in soil solution, that can be used in the computation of critical loads by steady state soil models and for the evaluation of scenarios by dynamic models.

Nitrogen deposition effects and empirical critical nitrogen loads

The report presents an overview of impacts of nitrogen deposition on plant species diversity on forests, grasslands, heath lands, coastal habitats, bog and fen habitats, distinguishing between information included in the overview report of Achermann and Bobbink (2003) and new evidence. Furthermore, an overview is given of impacts on soil quality and forest nutrient status and of other impacts (faunal species diversity and ground water quality). A summary of the various impacts on plant species diversity, soil quality and forest nutrient status shows that the results of Achermann and Bobbink (2003) are still adequate for use in critical load mapping (see Mapping Manual Chapter 5.2: www.icpmapping.org). The empirical critical N loads thus derived are affected by: (i) indicators of change used for impacts on biodiversity or ecosystem function and (ii) the type of studies involved in deriving the critical load. In most cases N manipulation experiments.

Even though it is clear that N deposition does affect faunal species diversity, a relationship between impacts and critical N concentrations in soil solution is not feasible. Regarding the critical N concentrations in ground water in view of human health impacts (ground water quality criteria), the criteria in the manual still hold.

Use of integrated dynamic biogeochemical models and vegetation models to assess critical nitrogen loads

The various modelling approaches described in this report consist of a combination of a biogeochemical model of nitrogen behaviour in the soil, connected with a vegetation model predicting nitrogen impacts on biodiversity. The biogeochemical models discussed are SMART2 (either or not in connection with SUMO), MAGIC, VSD and ForSafe. These models differ with respect to the included processes and management options, that are presented in the main text. These models can be used for the analysis of Damage Delay Time Recovery Delay Time. Dynamic models can also be used to compute critical loads, e.g. by back calculating the nitrogen deposition starting from safe indicator values in the future, or using other approaches described in detail in this report. The vegetation models and succession models predicting nitrogen impacts on biodiversity are MOVE/NTM, BERN, VEG and SUMO, respectively. A summary of the weaknesses and strengths of the various approaches is also given in the main text.

The VSD model has been distributed to all NFCs as an option for the comparative-static analysis of the effect of deposition scenarios in general – and exceedances in particular - on a critical indicator such as nitrogen concentration for a number of target years until 2100.

Present and updated critical nitrogen limits to assess critical nitrogen loads and evaluate scenarios with steady state and dynamic soil models

A comparison of the currently used critical limits for dissolved N concentrations compared to the findings in this study is given in the Table below (Table 24 in the main text). More information on the background of the data is given in the main text. The table does not contain the critical limits for ground water. Note that the EC target for drinking water remained 11.3 mg.l^{-1} and the target value remains 5.6 mg.l^{-1} . The updated critical limits can be used to (see also discussion):

- Calculate critical loads using the steady state SMB models
- Assess the impact of N deposition scenario's using dynamic biogeochemical models such as VSD, SMART, MAGIC and ForSafe.

Discussion and conclusions

Potential use of models: Linked biogeochemistry-biodiversity models for nitrogen have great potential for application in support of European policies to reduce excessive nitrogen inputs in general and European air pollution policies in particular. There are large similarities between the models, but there are also several important differences, including: (i) use of different abiotic variables for N, (ii) use of different variables for acidity, (iii) prediction of individual plant species versus plant communities and (iv) calibration to different (national) soil and vegetation datasets.

In deriving critical loads, the definition of reference conditions and damage thresholds for terrestrial biodiversity represents a major challenge, if linked biogeochemical-biodiversity models are to be used. As a first step, the comparison of the calls for empirical critical loads and critical loads for N and S can provide more information on possible ecosystem specific effects of exceedance.

Acceptable N concentrations in soil solution as used in the mapping manual and derived in this study

Impact	Critical N concentration (mg N.l ⁻¹)	
	UN/ECE (2004)	This study
Vegetation changes in Northern Europe		
- Lichens to cranberry (lingonberries)	0.2-0.4	0.2-0.4
- Cranberry to blueberry	0.4-0.6	0.4-0.6
- Blueberry to grass	1-2	1-2
- Grass to herbs	3-5	3-5
Vegetation changes in Western Europe		
- Coniferous forest		2.5-4
- Deciduous forest	-	3.5-6.5
- Grass lands		3
- Heath lands	-	3-6
Other impacts on forests		
- Nutrient imbalances	0.2-0.4	-
- Elevated nitrogen leaching/N saturation	-	1
- Fine root biomass/root length	-	1-3
- Sensitivity to frost and fungal diseases	-	3-5

Recommendations: Plant species responses to environmental variables form the basis of all developed models and is also widely used to estimate abiotic conditions. Data need to be collected in Europe of both a vegetation description and variables affecting the species diversity, including soil acidity, nutrient status and water availability. Based on this, plant species response per abiotic variable can be estimated, ecologically reviewed and then tested on independent datasets. Apart from further development of integrated biogeochemical-biodiversity impact models, further testing of these models on long-term monitoring, long-term experimental, and large-scale survey data is of crucial importance.

1 Introduction

1.1 Background of the report

In the 1980s the work of Ulrich and co-workers (e.g. Ulrich & Matzner, 1983) drew the attention towards the phenomenon of 'acid rain' and its association with imminent large-scale forest dieback. In response to the concern about the risk for forests and other ecosystems, several methods have been developed to estimate the critical load per ecosystem. According to a widely accepted definition, formulated by Nilsson and Grennfelt (1988), a critical load is 'a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge'. The concept is most commonly used in connection with the deposition of atmospheric pollutants, particularly nitrogen, acidity and heavy metals, and the critical load is the maximum flux (in $\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$, $\text{kmol H}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ or $\text{g HM}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) that an ecosystem is able to sustain. Critical loads have played an important role since 1994 in support of European air pollution policies (Hettelingh et al., 2001; EEA, 2003a).

Using currently accepted critical loads it turns out that acidification has diminished since 1990, in particular due to the reduction of sulphur emissions. This has led to a decrease of ecosystem areas in pan-Europe where critical acid loads are exceeded from about 48% in 1980 to about 12 % in 2000 using latest deposition patterns available from EMEP, the latest critical load database and IIASA assessments of current legislation in 2006 (see Table 1) Recovery from acidification has been recorded in the literature in particular in surface waters. However, nitrogen emissions continue to be high, specifically in areas with a large emission of ammonia, such as the Netherlands and large parts of Denmark, UK and Germany. This leads both to acidification and N-effects such as eutrophication.

The diminished risk of acidification may have contributed to the fact that large-scale forest die-back did not yet occur. This and the persisting high nitrogen pressure on terrestrial and aquatic ecosystems shifted the attention from effects of S-deposition and acidification towards effects of N-deposition and eutrophication. Effects of N deposition and resulting eutrophication appear to be much more widespread than the current risk effects of S deposition and resulting further acidification. Effects of earlier acidification are still widespread. Ecosystem areas that are at risk of eutrophication according to currently available critical N loads still cover about 30% of Pan-European ecosystems in 2000 (38% in 1980). Of ecosystem areas in the EU25 the percentages are even 71% in 2000 compared to about 80% in 1980 (Table 1).

Table 1 Exceedances of the critical load for acidification (left) and for eutrophication (right) as % of the European ecosystem area for which critical loads are available (including the CCE back ground database), using depositions computed with the EMEP-Unified Model from 1980 to 2010 on the basis of the BL_CLE scenario (totals include Andorra, Liechtenstein and San Marino).

Country	Area NOT protected from acidification (% at risk)					Area NOT protected from eutrophication (% at risk)				
	Area (km ²)	CLE-	CLE-	CLE-	MTFR	Area (km ²)	CLE-	CLE-	CLE-	MTFR
		2000	2010	2020	2020		2000	2010	2020	2020
AL	6,334.00	0	0	0	0	6,334.00	100	100	100	16
AT	35,745.50	1	0	0	0	35,745.50	97	88	74	7
BA	10,240.90	53	45	31	0	10,240.90	100	100	100	20
BE	70,52.30	78	40	19	2	7,052.30	95	94	91	39
BG	48,330.47	0	0	0	0	48,330.47	99	99	98	20
BY	107,841.30	64	58	47	7	107,841.30	59	60	57	21
CH	11,776.60	19	13	9	1	22,790.00	82	72	62	4
CY	4,061.75	0	0	0	0	4,061.75	66	66	74	17
CZ	11,178.37	79	47	21	3	11,178.37	100	99	96	43
DE	104,195.00	62	41	29	7	104,195.00	98	97	95	73
DK	3,148.60	32	8	4	0	3,148.60	94	85	82	53
EE	21,450.00	0	0	0	0	22,411.00	45	34	29	1
ES	85,225.00	1	0	0	0	85,225.00	88	82	74	38
FI	266,829.82	2	1	1	0	240,403.40	36	28	22	2
FR	180,101.69	15	8	5	1	180,101.69	98	97	93	41
GB	77,668.75	34	16	12	2	74,204.02	28	24	23	10
GR	9,326.00	11	7	4	0	9,326.00	100	100	100	69
HR	6,931.20	11	1	0	0	7,009.00	52	44	41	9
HU	10,447.69	0	0	0	0	10,447.69	98	87	69	11
IE	8,935.51	25	13	9	2	8,935.51	88	85	81	62
IT	125,838.00	0	0	0	0	125,838.00	71	65	58	14
LT	17,650.70	77	68	55	2	17,650.70	100	100	100	96
LU	820.70	33	22	20	18	820.70	100	100	100	98
LV	27,013.65	7	1	0	0	27,013.65	96	95	95	45
MD	11,985.00	3	3	3	0	11,985.00	0	0	0	0
MK	5,068.20	43	17	4	0	5,068.20	100	100	100	5
NL	9,170.61	81	77	76	58	6,208.83	92	87	85	59
NO	389,160.68	15	11	10	4	318,762.00	3	1	1	0
PL	88,383.00	58	39	12	0	88,383.00	98	97	95	60
PT	21,220.50	11	5	4	0	21,220.50	94	92	87	1
RO	62,807.00	7	6	2	0	62,807.00	99	99	99	81
RU	1,821,560.00	1	1	1	0	1,821,560.00	32	35	36	2
SE	519,343.08	14	8	5	2	225,264.25	18	10	8	4
SI	5,264.30	2	0	0	0	5,264.30	100	100	100	48
SK	19,253.46	24	13	8	0	19,253.46	100	100	89	15
UA	63,600.20	28	23	18	0	63,600.20	100	100	100	100
YU	21,307.10	43	31	13	0	21,307.10	100	100	99	3
EU25	1,659,324.12	18	11	7	2	1,333,353.25	65	60	56	25
Pan-Europe	4,226,442.50	12	8	6	1	3,841,164.50	46	46	44	14

The vast majority of ecosystems, at least those that are at highest risk, are nutrient-limited. And often, the most limiting nutrient is nitrogen (Vitousek & Howarth, 1991). As a consequence, even small additions of nitrogen may lead to a shift in the competitive relationship between species, and thereby to a shift in species composition. A common pattern is that at increasing nitrogen availability, many species adapted to nitrogen deficient circumstances are out competed by one or a

few species that use the available nitrogen more efficiently. Evidence suggests that increasing N availability often causes overall declines in plant species diversity. At present, widespread ecological effects of nitrogen deposition have been described for forests, grasslands (including tundra, montane and Mediterranean grasslands), oligotrophic wetlands (mire, bog and fen), heathland, aquatic habitats and coastal and marine habitats. In Europe, the effect of N deposition is now therefore considered as the most relevant effect of air pollution on floral biodiversity having consequences also for faunal diversity. In addition, N deposition affects groundwater quality and green house gas emissions.

Considering the importance of N deposition, there is an increased attention to review the methods to derive critical loads for N on terrestrial ecosystems in view of their direct and indirect effects on soil /soil solution and the biodiversity of terrestrial ecosystems. The revision of critical loads for N should not only enable an improved assessment of risk (exceedance or non exceedance of critical loads), but also of the time lags by which recovery or damage can occur when exceedances change. Critical loads, only refer to a steady-state situation of the receptor (soil, lake etc.). The critical load concept does not give any indication about the time it takes to reach that steady state, nor does it allow any assessment of the time when an emission reduction measure will lead to chemical conditions not considered harmful. To answer questions concerning time horizons, dynamic models are the most suitable tools, as such models can be used to (i) assess the impact of deposition scenarios on future values of bio-geochemical indicators, and (ii) derive target loads, for nitrogen and acidity. A target load is the deposition (path) which ensures recovery by having the prescribed chemical (or, ideally, biological) criterion be met in a given year and maintained thereafter.

The review and possible revision of critical N loads requires appropriate critical limits of nitrogen in view of their impacts, followed by an appropriate use of these limits in steady state and dynamic models for the assessment of critical loads and exceedances, respectively. At present, the critical limits used in view of terrestrial biodiversity impacts, being values for a critical N concentration in soil solution (UBA, 2004), need to be updated with current knowledge. This includes other critical limit indicators, such as nutrient concentrations in tree leaves and needles, ionic concentrations or concentration ratios in the soil and groundwater quality that have not yet been used in the European modelling and Mapping activities under the LRTAP Convention with the exception of mentioning criteria for ground water quality (UBA, 2004). However, in general, critical limits for biodiversity are the most sensitive one, i.e. yielding the lowest critical load values, thus requiring specific emphasis to improve critical limits for these effects. Furthermore, the indirect effects of nitrogen deposition on green house gas emissions is of importance to gain insight in the side effects of nitrogen abatement strategies on green house gas emissions and related climate change effects.

1.2 Aim of the report

The objective of this report is to present:

- A summarizing overview of adverse nitrogen deposition effects in terms of impacts on plant species and faunal biodiversity of terrestrial ecosystems, forest nutrient status in relation to impacts on soil and solution chemistry and on ground water quality.
- An overview of integrated dynamic biogeochemical models with plant species diversity models, that allow the assessment of critical loads and target loads of nitrogen in view of plant species diversity impacts.
- A review of currently used critical limits for N concentrations in soil solution and derivation of new critical limits, based on field (literature) data and integrated soil vegetation models, that can be used in the computation of critical loads by steady state soil models and for the evaluation of scenarios on soil solution chemistry by dynamic models.

The report aims to contribute to the modelling and mapping of the risk for nitrogen effects in general and of biodiversity in particular. More specifically, it aims to provide tools and limits for improved critical load and target load assessments in European Modelling and Mapping activities, to be used in the support of European air pollution policies. These results enable the Coordination Centre for Effects (CCE) and their collaborating ICP-M&M National Focal Centres (NFCs) to respond to requirements for improved knowledge on multiple-source multiple effect relationships involving nitrogen.

1.3 Contents of the report

In chapter 2 an overview is given of adverse direct impacts of elevated N inputs on soil and soil solution quality, forest nutrition/vitality and species diversity (both of flora and fauna) of terrestrial ecosystems, including a summary of available empirical critical N loads. An overview of integrated soil vegetation models is presented in Chapter 3. This includes a combination of dynamic soil models (SMART2, MAGIC, VSD, SAFE) with plant species diversity models (MOVE/NTM, GBMOVE, BERN, VEG). These integrated soil vegetation models can be used to predict plant species diversity in response to atmospheric deposition and inversely calculate critical loads for N and acidity in view of an acceptable change. The chapter also summarizes the data requirements of each of the models which will probably necessitate an extension of currently available NFC databases in the near future.

In Chapter 4 an evaluation is presented of currently used critical N limits and a methodology for the derivation of new critical N limits using observations and model interpretations. This chapter also includes examples of the use of the new limits in the assessment of: (i) critical N loads with available steady state models (SMB, SMART2-Steady-state) and (ii) target N loads with dynamic models (SMART2). Finally in Chapter 5, an overall discussion of the results is given, including suggestions for further improvements.

2 Adverse impacts of elevated nitrogen deposition on terrestrial ecosystems and empirical critical nitrogen loads: an evaluation

2.1 Introduction

Adverse impacts of elevated nitrogen deposition

Elevated nitrogen use does cause various widespread and well-documented environmental effects. There are various overview papers describing those effects, including acidification and eutrophication of terrestrial and aquatic ecosystems, with related impacts on plant species and faunal species diversity, surface water pollution, including damage to fisheries in coastal ecosystems, groundwater pollution and global warming by N₂O emissions (e.g. Vitousek et al., 1997; Galloway, 1998; Galloway & Cowling, 2002; Matson et al., 2002; Galloway et al., 2003). There are also diverse consequences for human health (Wolfe & Patz, 2002) and on climate, visibility and materials. These undesirable “cascading effects”, as Galloway et al. (2003) called them, of reactive N moving through aquatic and terrestrial ecosystems and the atmosphere do not stop until the reactive is eventually converted back to N₂ through the process of denitrification.

Nitrogen saturation and (forest) ecosystem responses

Aber et al. (1989) launched the theory on ecosystem nitrogen saturation for forests and the different stages that can be identified in view of: (i) impacts on soil chemical processes such as mineralization, immobilization, nitrification, affecting N leaching, (ii) plant nutrition and forest growth and (iii) plant species diversity. A schematic illustration over the progression of a forest ecosystem from N limitation to N saturation to N excess and the potential effects of N deposition is given in Figure 1, being an update of the original figure by Aber et al. (1989), further updated by Gundersen (1991) and recently by Emmett (2007).

Nitrogen saturation can be defined in several ways (Ågren & Bosatta, 1988; Aber et al., 1989; Stoddard, 1994). The most widely used definition (and most relevant for water quality) is the condition where ‘availability of mineral N may exceed the combined nutritional demands of plants and microbes’ (Aber et al., 1989) which then can be determined as elevated nitrate leaching from the rooting zone. The development of N saturation by increased N inputs involves a complex interaction of the processes in the N cycle (Aber et al., 1989; Aber, 1992; Aber et al., 1998).

In the first phase, originally described for forests, primary production increases. Plants and microbes effectively absorb added N and the N content of plants increase. Retranslocation of N from senescent foliage (and roots) may decrease leading to higher N contents in litter materials and thus increased litterfall N flux. The internal cycling of N is accelerated through increases in litter fall N, net mineralization and tree N uptake. As N availability is increased, the composition of the (forest floor) vegetation may gradually change towards more nitrophilic species.

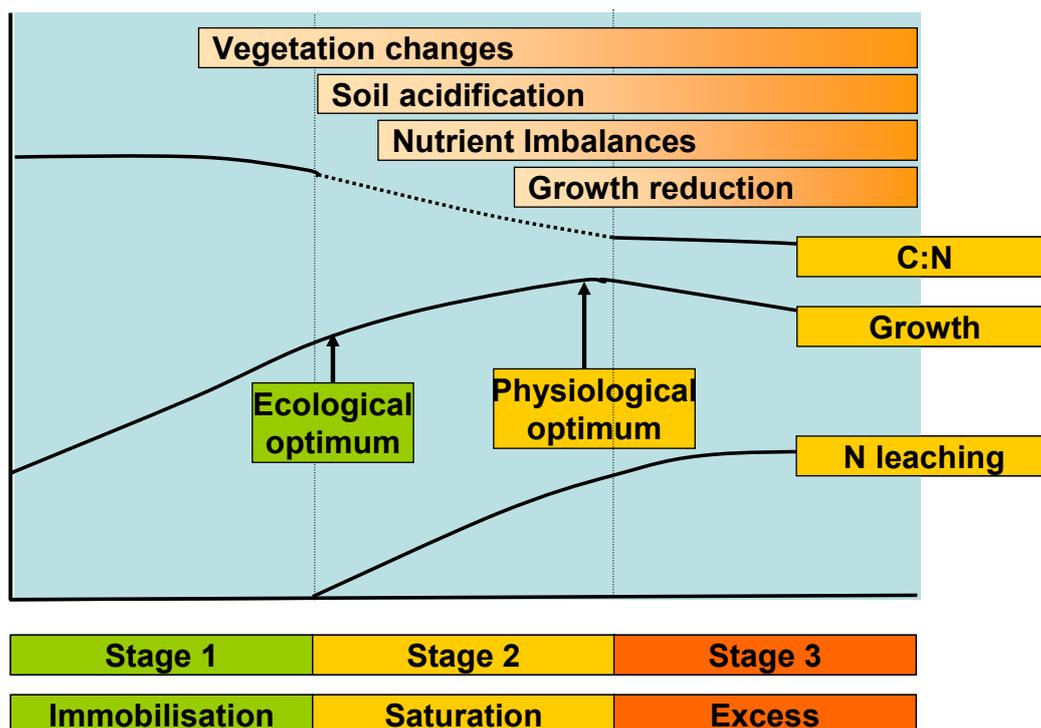


Figure 1 Hypothetical relationship between the stage of N saturation and the effects on forest ecosystems in terms of (relative) changes in soil processes (soil acidification), vegetation changes/ecosystem response, growth, N status (C/N ratio; nutrient imbalances) and input – output relations (N leaching). The time scale for the responses may differ widely between ecosystems and regions. Modified from Gundersen (1991).

In the accelerated N cycle, immobilisation of nitrate by soil microbes declines and also net nitrification (the production of nitrate by soil microbes) becomes important. Both of these processes contribute to the appearance of nitrate in soil water. The ecosystem approaches N saturation. When elevated nitrate leaching becomes a chronic condition, soil acidification from N transformations becomes significant. After reaching N saturated conditions, N leaching will continue to increase with deposition. In an originally N limited, primary production of forests or natural vegetation will increase in response to additional N inputs until a physiological optimum (which is beyond the ecological optimum) is reached. Above that threshold level, production stays constant or even decreases (see Figure 1). In this stage other essential resources than N (P, K, Ca, Mg, or water), indicated by nutrient imbalances may at least periodically limit plant growth.

In extreme cases, destabilisation and possibly forest decline from excess N deposition has been shown in case studies where the nutritional imbalance was important (Roelofs et al., 1985; Schulze, 1989; McNulty et al., 1996). Boxman et al. (1998) even demonstrated a significant growth increase after declining trees were relieved from excess N deposition by an experimental roof under the canopy. Recent synthesis efforts support this general scheme although the understanding of processes and interactions has become more complex and detailed and a need to

expand the framework to identified. For an overview of effects of atmospheric ammonia on terrestrial vegetation, we refer to Krupka (2003).

2.2 Plant species diversity

Nitrogen deposition and plant species diversity loss

In Europe the effect of N deposition is now considered as the most relevant effect of air pollution on plant diversity. During the past two decades the general attention shifted from effects of S deposition and acidification towards effects of N deposition and eutrophication and climate change. Since the recognition of nitrogen deposition as one of the main drivers behind the general loss of biodiversity in Europe, a number of expert workshops have taken place in order to reach agreement among specialist as to the impacts of nitrogen for various ecosystems and related critical loads (Nilsson & Grennfelt, 1988; Bobbink et al., 1992; Hornung et al., 1995; Bobbink et al., 1996; Achermann & Bobbink, 2003).

Evidence suggests that increasing N availability often causes overall declines in plant species diversity (Tilman, 1987; Bobbink et al., 1998). In some cases, especially under very nutrient poor conditions, however, an increase in biodiversity has been observed due to the expansion in the range of more nitrophilic species (Emmett, 2007). In boreal forests, it has been shown that the overall species number was not affected by N enrichment, despite a drastic change in species composition. This was caused by the increase in the number of common N-loving species, that increased parallel with the decline of the number of typical ground flora species (Bobbink, 2004).

In oligotrophic to mesotrophic ecosystems, nitrogen is generally the most important growth-limiting element, and their species are adapted to a nitrogen-deficient environment. If the availability of nitrogen increases, other species that use the available nitrogen more efficiently will out-compete the unproductive species adapted to nitrogen deficiency. This effect may occur at relatively low deposition rates, and is probably most determinative for the lower end of the critical load range (Heij & Erisman, 1997; Stevens et al., 2004). At higher deposition levels, nitrogen saturation will occur when the deposited nitrogen is no longer completely taken up by the vegetation or immobilised in the soil, and leaching may occur (Tamm et al., 1999). Nitrogen saturation is accompanied by increased levels of nitrogen in foliage, which in turn may increase susceptibility of plants to frost or diseases. This effect is well documented for forest trees (e.g. Aronsson, 1980; Balsberg-Pålsson, 1992), but also occurs in natural vegetation as heaths (e.g. Berdowski, 1987).

These impacts of N deposition can be detected at the regional and national scale from field monitoring schemes, with data often indicating a shift towards more N-loving species. As an example, Figure 2 presents results from two long term monitoring programmes in the UK which both suggest that there have been wide-ranging changes in species occurrence in this country during the latter half of 20th century associated with increased N availability (Emmett, 2007). In the first study, use is made of data of the New Plant Atlas of the UK (Preston et al., 2002), to make an analysis of species occurrence records to identify which species are changing their

geographical range. This analysis provides evidence for: (i) a decline in the frequency of occurrence of plant species characteristic of low nutrient availability between 1930-69 and 1987-99 and (ii) an increase in the geographic range of species associated with high nutrient availability (Figure 2a). In the second study, use is made of a stratified sampling of the countryside on a grid which contain permanent vegetation plots to monitor species changes. Reported results from repeated surveys of higher plant species data from permanent quadrats from 1978-1990 and 1990-1998 suggest a shift towards plant species associated with high nutrient availability particularly in low nutrient habitats such heathland and infertile grasslands (Figure 2b). In both studies, the greatest change when analysed for trends was an increase in plant species which favour high N availability and a decline in plant species which favour low N availability in terms of Ellenberg 'N' values. Ellenberg values are essentially a scoring system to indicate the environmental preferences for plants with the Ellenberg 'N' index, often used to indicate species preference for N availability (Ellenberg et al., 1991).

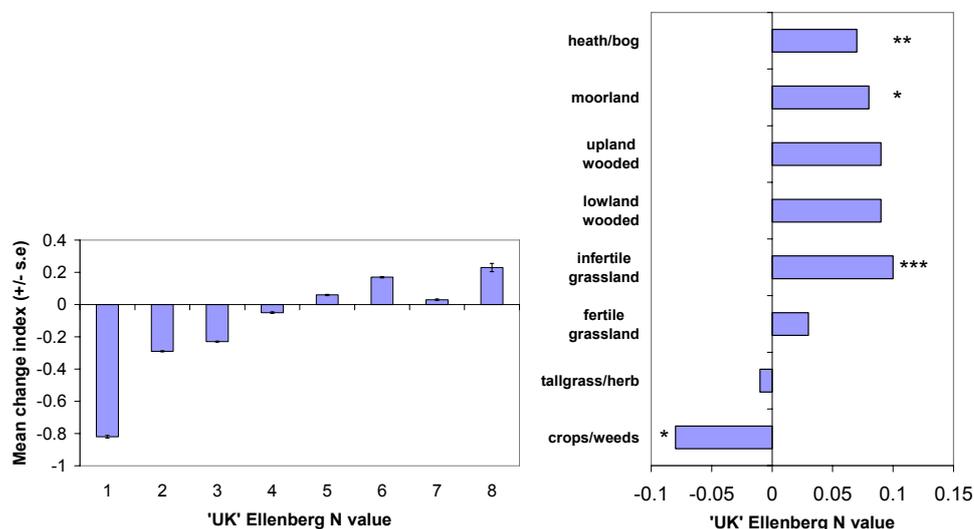


Figure 2 Changes in vegetation composition in the UK according to their UK Ellenberg Nutrient (N) values (Hill et al., 1999). A low value is associated with low N fertility and high value with high N fertility: (a) The mean change index for occurrence of plant species classified according to their UK Ellenberg (N) values between 1930-69 and 1987-99 as reported in the New Atlas of the British and Irish Flora (Drawn from data presented in Preston et al., 2002). Standard errors are indicated. (b) The change in UK Ellenberg N score for different habitats between 1990 and 1998 taken from the most recent UK Countryside Survey (redrawn from Haines-Young et al., 2003). Statistically significant changes are marked as * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$

Effects of nitrogen deposition are now recognised in nearly all oligotrophic and mesotrophic (semi-)natural ecosystems; these include aquatic habitats, forests, grasslands (including tundra and Mediterranean grasslands), oligotrophic wetlands (mire, bog and fen), heathland, and coastal and marine habitats (Achermann & Bobbink, 2003). An overview of major effects on plant species diversity, including impacts on mosses, lichens and mycorrhizae, for major terrestrial ecosystems in Europe (grasslands, heathland, coastal habitats wetlands and forests using the EUNIS classification) is presented in Table 2.

Table 2 Indicators for the effects of elevated N deposition and related empirical critical loads ($\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) for major ecosystem types (according to the EUNIS classification) occurring in Europe (from Achermann & Bobbink, 2003).

Ecosystem type (EUNIS class)	EUNIS- code	Effect indicators	Empirical critical load
<i>Forest habitats (G)</i>			
Mycorrhizae	-	Reduced sporocarp production, reduced belowground species composition	10-20
Ground vegetation	-	Changed species composition, increased nitrophilous species; increased susceptibility to parasites (insects, fungi, virus)	10-15
Lichens and algae	-	Increase of algae; decrease of lichens	10-15
<i>Grasslands and tall forb habitats (E)</i>			
Sub-atlantic semi-dry calcareous grassland	E1.26	Increased mineralization, nitrification and N leaching	15-25
Non-mediterranean dry acid and neutral closed grassland	E1.7	Increase in nitrophilous graminoids, decline of typical species	10-20
Inland dune grasslands	E1.94, E1.95	Decrease in lichens, increase in biomass, increased succession	10-20
Low and medium altitude hay meadows	E2.2	Increased tall grasses, decreased diversity	20-30
Mountain hay meadows	E2.3	Increase in nitrophilous graminoids, changes in diversity	10-20
Moist and wet oligotrophic grasslands	E3.5.1/ E3.5.2	Increase in tall graminoids, decreased diversity, decrease of bryophytes	10-25
Alpine and subalpine grasslands	E4.3 and E4.4	Increase in nitrophilous graminoids, changes in diversity	10-15
Moss and lichen dominated mountain summits	E4.2	Effects on bryophytes and lichens	5-10
<i>Heathland habitats (F)</i>			
Northern wet heaths	F4.11	Decreased heather dominance, transition heather to grass, decline in lichens and mosses	10-20
Dry heaths	F4.2	Transition heather to grass, decline in lichens	10-20
Arctic, alpine and subalpine scrub habitats	F2	Decline in lichens, mosses and evergreen shrubs	5-15
Dry heaths	F4.2	Transition heather to grass, decline in lichens	10-20
<i>Coastal habitat (B)</i>			
Shifting coastal dunes	B1.3	Increased biomass, increased N leaching	10-20
Coastal stable dune grasslands	B1.4	Increase in tall grasses, decreased prostrate plants, increased N leaching	10-20
Coastal dune heaths	B1.5	Increase in plant production, increased N leaching, accelerated succession	10-20
Moist to wet dune slacks	B1.8	Increase in biomass and tall graminoids	10-25
<i>Mire, bog and fen habitats (D)</i>			
Raised and blanket bogs	D1	Changed species composition, N saturation of <i>Spagnum</i>	5-10
Poor fens	D2.2 ^d	Increase sedges and vascular plant, negative effects on peat mosses	10-20
Rich fens	D4.1 ^e	Increase in tall graminoids, decreased diversity, decrease of characteristic mosses	15-35
Mountain rich fens	D4.2	Increase in vascular plants, decrease of bryophytes	15-25

Empirical data are taken from Achermann and Bobbink (2003).

The information is based on a review by Bobbink et al. (2003), included in Achermann and Bobbink (2003). This review also includes information on “Inland surface water habitats” and “Marine habitats” that is not part of this overview. For forests, the impacts on soil processes and trees are not included as this does not refer to impacts on plant species diversity. These impacts are described in the next paragraph. Table 2 also includes information on the related empirical critical N loads, based on the same overview by Bobbink et al. (2003), included in Achermann and Bobbink (2003).

In this approach, the critical load is practically equal to the highest input of nitrogen that does not lead to adverse physiological changes (on the individual level) or loss in biodiversity (on the ecosystem level). In the empirical approach, medium to longer term (i.e., > 1 year) effects of nitrogen addition (< 100 kg N.ha⁻¹.yr⁻¹) to existing vegetation play a central role. Such addition experiments are mostly carried out in the field, or sometimes in the laboratory in so-called ‘mesocosms’ (i.e., pieces of vegetation directly taken from the field). Because of the time and labour-intensive nature of such studies, results are only available for a rather limited group of ecosystems. In some cases, experimental results are supplemented by observational studies e.g. time series under a known increase in deposition.

Below, we present a short summary of the major effects of N deposition and plant species diversity loss in the various ecosystems, focusing on impacts in the Netherlands where significant species diversity changes have occurred in the past due to nitrogen deposition. In most cases, use was made of the overview by Bobbink et al. (2003). The review is updated with more recent information on nitrogen deposition impacts and critical nitrogen loads since that time, specifically in the Nordic countries.

Impacts of nitrogen deposition on forest ecosystems

Available evidence

There is a large number of observations across Central Europe showing an increase in abundance of nitrophilous species in forests, either as time series or within a nitrogen deposition gradient. In parallel, the average N indicator number (after Ellenberg, 1988) has been shown to rise, whereas a lot of endangered species are plants with a low N indicator value (Ellenberg, 1985). Examples of such studies are given below.

Circumstantial evidence is available for large changes in the forest ground vegetation in the Netherlands from 1950 to 1990. A pilot study carried out by De Vries (1982) in an area of pine forests on dry, sandy soil, where vegetation maps from 1957 were available showed a complete change in ground vegetation, from a moss and lichen dominated type to a grass dominated type. Changes in the species composition of the ground vegetation of Dutch pine forests in a more recent period were studied by comparing vegetation descriptions made in 177 permanent plots in 1984 and in 1993 (Van Dobben et al., 1994). This study showed a significant decrease in the cover of heathland species (*Erica tetralix* and *Calluna vulgaris*) and a strong increase of many nitrophilous species.

Species changes in the ground vegetation of forests towards nitrophilic species have also been recorded in other parts of Europe. These changes entail: (i) a decline of terrestrial lichens ('reindeer lichens') and of ectomycorrhiza mushrooms, (ii) an increase of grasses, notably *Deschampsia flexuosa* and (iii) a general increase of mosses and vascular plants that typically occur on N rich soils (Bobbink et al., 1998). In a recent review, Bobbink et al. (2003) concluded that these changes occur at N loads above 10-15 kg N.ha⁻¹.yr⁻¹.

Large scale monitoring data have shown a positive correlation between the abundance of nitrophilous species and the current N deposition in forests in Nordic countries. For example, in Sweden, *Quercus robur* stands in two geographical areas with different nitrogen deposition (6-8 and 12-15 kg N.ha⁻¹.yr⁻¹, respectively) were compared with special emphasis on nitrogen indicator species (Tyler, 1987). The following nitrophilous species were more common in the most polluted site: *Urtica dioica*, *Epilobium angustifolium*, *Rubus idaeus*, *Stellaria media*, *Galium aparine*, *Aegopodium podagraria*, *Sambucus spp.*. Comparable observations were reported by Falkengren-Grerup (Falkengren-Grerup, 1986; Falkengren-Grerup et al., 1998) who examined the changes in soil and vegetation in repeated studies (10-40 years) in deciduous forests in southern Sweden where nitrogen deposition has doubled since 1955 to 15-25 kg N.ha⁻¹.yr⁻¹. A marked increase in frequency was found for almost 15 species, including *Aegopodium podagraria*, *Epilobium angustifolium*, *Rubus idaeus*, *Stellaria nemorum*, *S. holostea*, *Dryopteris filix mas*, *Urtica dioica*, all considered to be nitrophilous species after Ellenberg (1988). (Rosén et al., 1992) found a significant positive correlation between *Deschampsia flexuosa* dominated coniferous forests in the past 20 years in Sweden and the pattern of nitrogen deposition, based upon comparisons of ground vegetation surveys in the Swedish Forest Inventory between 1973/77 and 1983/87. *Deschampsia flexuosa* increased significantly in this period. These changes occurred above a nitrogen deposition of 7-11 kg N.ha⁻¹.yr⁻¹, being in line with the range of 5-10 kg N.ha⁻¹.yr⁻¹ mentioned above based on N addition experiments.

New evidence

Recent N addition experiments, varying between 3 and 50 kg N.ha⁻¹.yr⁻¹, have shown, however, that in boreal forests in Scandinavia critical loads of 10-15 kg N.ha⁻¹.yr⁻¹ are too high (Nordin et al., 2005). In these forests, N additions above approximately 8-12 kg N.ha⁻¹.yr⁻¹ after a 10 year period experiments already caused significant effects. It causes an increase grasses, such as wavy hair-grass (*Deschampsia flexuosa*) induced by an increased light penetration that results form damage to shrubs. Furthermore it causes a decrease in lichens, bilberries (*Vaccinium myrtillus*) and cowberries (*Vaccinium vitis-idaea*). The change in the *Vaccinium* species (bilberries and cowberries) occurs in such a way that it favours the attack by commonly occurring parasitic fungi, thus leading to an increased occurrence of pests. The reduced abundance of bilberries and cowberries in the N addition experiments is in accordance with large scale monitoring data that show a negative correlation between the abundance of these *Vaccinium* species and the current N deposition in coniferous forests in Sweden (Nordin et al., 2005). Based on these results, a critical load of 5-10 kg N.ha⁻¹.yr⁻¹ seem most appropriate for boreal forests.

Impacts of nitrogen deposition on grasslands, heathlands and coastal habitats

Available evidence

In Dutch calcareous grassland: massive expansion of *Brachypodium* and drastic reduction in species diversity has been observed due to nitrogen addition (Bobbink, 1991; Willems et al., 1993). In many Dutch dune grasslands with relatively high nitrogen deposition (about 25 kg N.ha⁻¹.yr⁻¹) many grasses have increased, whereas in coastal areas of Western England with relatively low deposition (about 10 kg N.ha⁻¹.yr⁻¹) dune grasslands are still species-rich (Bobbink et al., 1998). In the Netherlands more than 35% of the heathland has been altered into grassland. Even in southwest Norway, the area with the highest N deposition in Norway, similar changes have been observed (see e.g. Bobbink et al., 1998). The importance of N deposition, especially in the early phase of heathland development is also confirmed by field experiments (Aerts et al., 1990). Specifically in wet heathlands in the Netherlands, which are generally richer in plant species compared to dry heathlands, a drastic change in species composition has been observed. Most wet heathlands in the Netherlands are now mainly dominated by *Molinia* in stead of *Erica* (e.g. Bobbink et al., 1998). Some authors also ascribe the decline of the diversity of grasslands on poor, sandy soil to acidification (De Graaf et al., 1997). However, in many of these cases also other factors besides acidification seem to be responsible for the reported decline. The decline of acidic grassland species is probably caused by a combination of acidification and eutrophication, where toxicity of Al ions and a shift in the NH₄ / NO₃ ratio are the triggers (De Graaf et al., 1998).

Calcareous grasslands are among the most species-rich plant communities in Europe and contain a large number of rare and endangered species. These grasslands decreased strongly in area during the second half of this century (e.g. Wolking & Plank, 1981; Ratcliffe, 1984). Some remnants became nature reserve in several European countries. To maintain the characteristic calcareous vegetation, a specific management is needed in most situations to prevent their natural succession towards woodland (Wells, 1974; Dierschke, 1985). Many semi-natural grassland types occur in the mountain regions across Europe, containing many rare and endangered plant and animal species (e.g. Ellenberg, 1996). The elevated nitrogen deposition in mountainous regions in central Europe (e.g. Hesterberg et al., 1996; Rihm, 1996) has most likely caused effects of eutrophication, but relevant studies are not available. The effects of nitrogen in montane grasslands has thus been identified as a major gap in knowledge (Bobbink et al., 1992; Grennfelt & Thörnelöf, 1992; Hornung et al., 1995).

New evidence

A recent nation wide inventory of 68 acid grasslands across the UK also showed that increased N deposition results in a decrease of floristic diversity, at least in acid grassland communities (Stevens et al., 2004). Based on regression analysis Stevens et al. (2004) found that the floristic diversity declines as a linear function of the rate of inorganic nitrogen deposition, with a reduction of one species per 4 m² for every 2.5 kg N.ha⁻¹.yr⁻¹ of nitrogen deposition (Figure 3). The regression line shown is: Plant species richness = 23.3 - 0.408 (N-dep) ($r^2 = 0.55$, $p < 0.0001$).

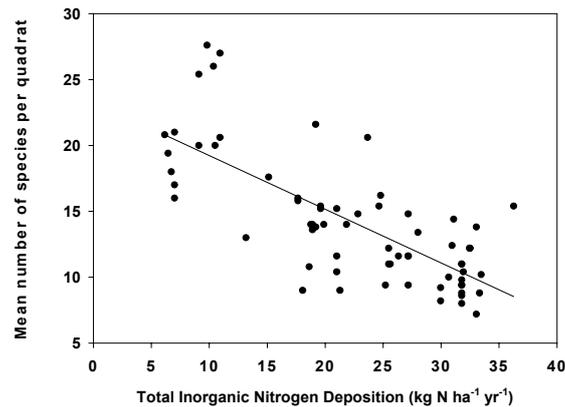


Figure 3 Acid grassland species richness plotted against N deposition for 68 field sites visited in summer 2002 and 2003. Thus for every 2.5 kg N/ha/y of sustained elevated N deposition, a mean of 1 additional species is excluded from a 4-m² quadrat (after (Stevens et al., 2004).

At the mean nitrogen deposition of central Europe (17 kg N.ha⁻¹.yr⁻¹), this implies an average number of species of 16.3 compared to an average number of species of 21.3 on grasslands receiving the lowest levels of nitrogen deposition (5 kg N.ha⁻¹.yr⁻¹). This implies reduction of being 23%, By setting an acceptable average reduction of species one may derive a critical N load from this information.

A summary of N-addition experiments in grasslands in Europe (8 countries) showed already early rapid loss of species diversity as presented in Figure 4 (after Bobbink, 2004). Haddad et al. (2000) observed a similar change in the plant species diversity of abandoned prairie grassland in response to N input. By defining an acceptable change one may again define a critical load from this information.

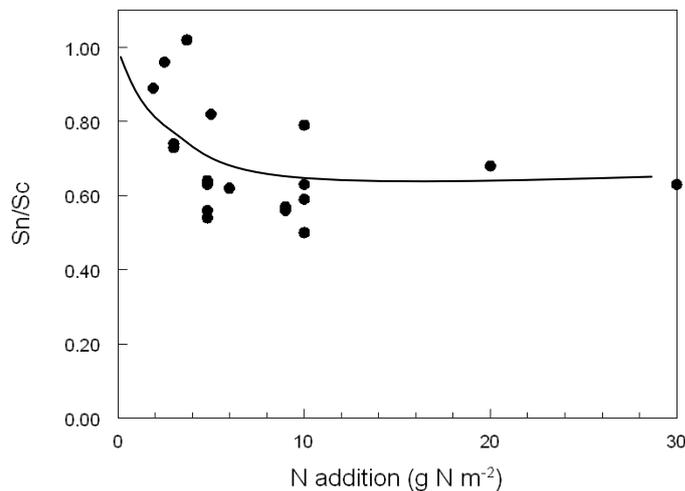


Figure 4 Summary of N-addition experiments expressed as the ratio between species-richness with N-addition (Sn) over species richness of control (Sc), showing early rapid loss of species diversity (after Bobbink, 2004).

A similar analysis of North American fertilisation experiments examined the plant traits which could be used to predict winners and losers in the presence of N fertilisation (Pennings et al., 2005; Suding et al., 2005). N fertilisation was found to increase production and reduce species richness in all experiments. Two hypotheses were tested that:

- change in biodiversity related to increased productivity and chance of loss related to initial abundance. (random loss of rare species)
- change in biodiversity related to shift in traits optimal for resource use (functional trait-base loss)

They concluded, species of low abundance appear to be particularly at risk (Suding et al., 2005).

Finally, N reduction experiments can provide information on changes which have already occurred and may be used to define a critical load limit. One example is a mesocosm experiment where N input were reduced below ambient (20 kg N.ha⁻¹.yr⁻¹) and also increased to 55 kg N.ha⁻¹.yr⁻¹. Results indicate again a strong decline in abundance of an important moss species *Racomitrium lanuginosum* which suggest much of the change in response by this species at least has already occurred (Figure 5).

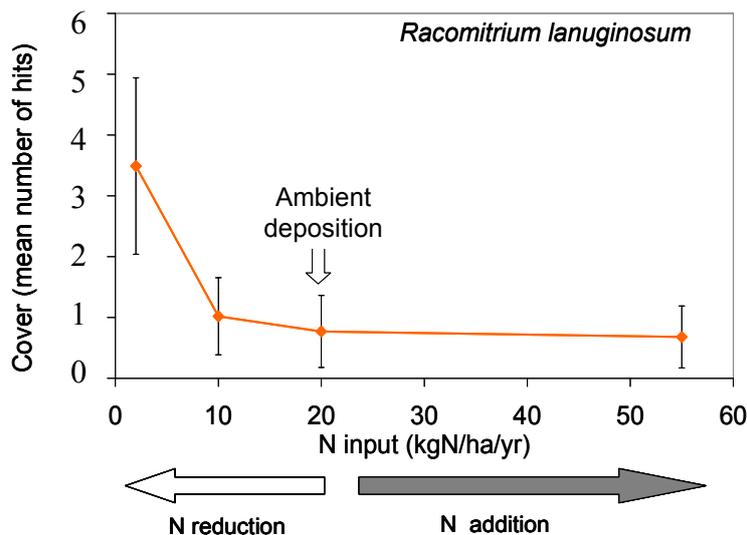


Figure 5 Change in cover as detected by pin pointing (hits) of the moss *Racomitrium lanuginosum* to both N additions (> 20 kgN/ha/yr) and N reductions (< 20 kgN/ha/yr) in acid grassland mesocosms in an experimental misting facility which excluded ambient N deposition (redrawn from Jones, 2005 in Emmett, 2007). Standard errors are shown.

Impacts of nitrogen deposition on bog and fen habitats

Available evidence

Clear effects of nitrogen eutrophication have been observed in Dutch ombrotrophic bogs. The composition of the moss layer in the small remnants of the formerly large bog areas has markedly changed in recent decades as nitrogen loads have increased to 20-40 kg N.ha⁻¹.yr⁻¹, especially as ammonium/ammonia; the most characteristic *Sphagnum* species have been replaced by more nitrophilous moss species (Greven,

1992). The effects of atmospheric nitrogen deposition on ombrotrophic bogs have also been intensively studied in Britain (e.g. Lee & Studholme, 1992). Many characteristic *Sphagnum* species have become largely absent from affected ombrotrophic bog areas in Britain, such as the southern Pennines of England, where atmospheric nitrogen deposition has increased to ca. 30 kg N.ha⁻¹.yr⁻¹.

Although these, and other, studies strongly indicate the detrimental effects of high nitrogen deposition rates on the development of the bog-forming *Sphagnum* species, there is also evidence of growth stimulation in response to small increments in nitrogen deposition. Field experiments by Aerts et al. (1992) at a site with low atmospheric deposition (0.4 kg N.ha⁻¹.yr⁻¹) in northern Sweden showed that *Sphagnum balticum* increased growth four-fold following addition of nitrogen (20 and 40 kg N.ha⁻¹.yr⁻¹), whilst no effect was found on *Sphagnum magellanicum* at sites with higher atmospheric deposition (7-9 kg N.ha⁻¹.yr⁻¹). Because of the increase in the nitrogen deposition during the decades before the experiment, the growth of *Sphagnum* at the site with high atmospheric nitrogen deposition rates had become phosphorus limited (Malmer, 1990).

New evidence

In northwest European raised bogs the increased nutrient deposition has resulted in the invasion of birch (*Betula* spp.) and purple moor grass (*Molinia caerulea*) (Risager, 1998; Limpens et al., 2003a; Tomassen et al., 2003; Tomassen et al., 2004). These changes in the vegetation composition and its structure will have affected the fauna species assemblages, e.g. spiders and beetles living in the originally open bog vegetation and ground breeding birds. The increased nutrient availability has also resulted in an increase of the nutrient content of plant material (Limpens et al., 2003a; Tomassen et al., 2004). Also the growth of algae might be stimulated by an increase of N and P availability (Gulati & DeMott, 1997; Limpens et al., 2003b). These changes have consequences for the herbivorous and detritivorous invertebrates that eat this material. Also carnivorous invertebrate species, like aquatic beetles (Dytiscidae) are affected (Van Duinen et al., 2004).

2.3 Soil quality and forest nutrition

Impacts of nitrogen on forest ecosystems

Until a certain threshold level is reached, forests will react to additional N inputs by an increased biomass production until a physiological optimum (which is beyond the ecological optimum), but above that, production stays constant or even decreases (see Figure 1). Below the threshold level for growth, however, changes in the ecosystem are observed, especially the forest biodiversity may gradually change towards more nitrophilic species (Ellenberg, 1985; Bobbink et al., 1996; Bobbink et al., 1998). In forested plots with a continuous high N input, the ecosystem may approach 'N saturation' (Aber et al., 1989). In this stage, the N leaching will increase above (nearly negligible) background levels, associated with soil acidification in terms of elevated leaching of base cations or aluminium, causing a decrease in acid neutralizing capacity. At the stage of "N saturation" or "N excess", the ecosystem may be destabilised by the interaction of a number of factors. Release of aluminium

by soil acidification and imbalances of ammonium to base cations may cause absolute or relative nutrient deficiencies, which may be aggravated by a loss of mycorrhiza or root damage (Figure 1). Furthermore, strong accumulation of N in foliage (e.g. as amino acids) may affect frost hardiness and the intensity and frequency of insect and pathogenic pests. It may also cause water stress as a result of increased canopy size, increased shoot/root ratio, and loss of mycorrhizal infection. An overview of possible effects on forests as a result of increased atmospheric acid and N deposition and/or exposure to air pollutants is presented by Erisman and De Vries(2000), as summarised in Table 3. Below we illustrate several of those effects, including recent information from an Intensive Forest Monitoring Programme in Europe (see e.g. De Vries et al., 2000; 2001; 2003b; 2003d). Furthermore, in many cases, use was made of the excellent overview made by Bobbink et al. (2003).

Table 3 Possible effects of increased atmospheric N loading and exposure to NO_x and NH₃ on forest ecosystems (after Erisman & de Vries, 2000).

Forest compartment	Effects	
	Chemistry	Ecosystem
Soil (solution)	<ul style="list-style-type: none"> - elevated N concentrations in soil (solution) - elevated ratios of NH₄⁺ and Al³⁺ to base cations 	<ul style="list-style-type: none"> - increase in NO₃ leaching - increase in nitrophilous species/ decrease in biodiversity - root damage and mycorrhiza decline - inhibition of uptake (nutrient imbalances)
Trees (foliage)	<ul style="list-style-type: none"> - elevated arginine concentrations in foliage - elevated N concentrations in foliage 	<ul style="list-style-type: none"> - growth reduction - nutrient deficiency absolute or relative (to N) / discoloration - increased biomass production/ water demand (risk of drought) - increased ratio of foliage to roots (risk of drought and nutrient deficiency) - increased frost sensitivity - increased parasite injury (insects, fungi, virus)

Nitrogen leaching and soil acidification

One of the first indications of adverse impacts of N inputs in forest ecosystems is elevated leaching of N (NO₃) that may cause acidification of ground and surface water (see Figure 15). At more than 100 intensive monitoring plots in Europe, the input of and output of different N compounds (total N, NH₄ and NO₃) has been derived, using methods described in detail in De Vries et al. (2001). Results of the leaching of total N and NO₃ against the total N deposition show that the leaching of N is generally negligible below a total N input of 10 kg.ha⁻¹.yr⁻¹ (Figure 6A). The same is true for NO₃, that dominated the N leaching (Figure 6B). These results are in accordance with those found by e.g. Dise et al. (1998a; 1998b) and Gundersen et al. (1998). At N inputs between 10 and 20 kg.ha⁻¹.yr⁻¹, leaching of N is generally elevated, although lower than the input indicating N retention at the plots. At N inputs above 20 kg.ha⁻¹.yr⁻¹, N leaching is also mostly elevated and in several cases (seven plots), it is near or even above (for two plots) the N deposition (Figure 6A, B).

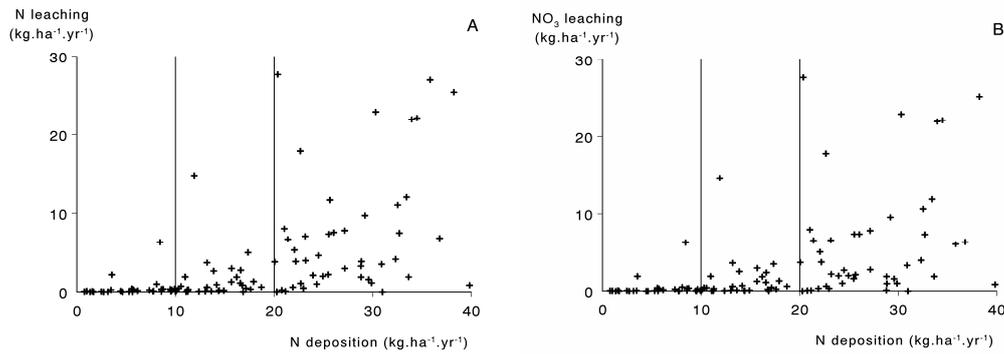


Figure 6 Scatter plots of the leaching of total N (A) and NO₃ (B) against the total N deposition.

The increase in N leaching in response to elevated N loads (above 20 kg.ha⁻¹.yr⁻¹) appeared to be larger for deciduous trees than for conifers. Statistical analyses of input-output relationships for N suggest that apart from the N input, the C:N ratio is also significantly correlated to the N output. Sites broadly break into high and low C:N categories and sequential regressions on leaching data show the ‘best’ (highest r₂, lowest MSE) division at C:N 23. This is shown in Figure 7, using data from both the IFEF and level II sites (Dise et al., 2006). At N-enriched sites (C:N < 23), the relationship between N leaching and N input is relatively strong, and a good estimate of N-out is ca ¾ of (N_{in}).

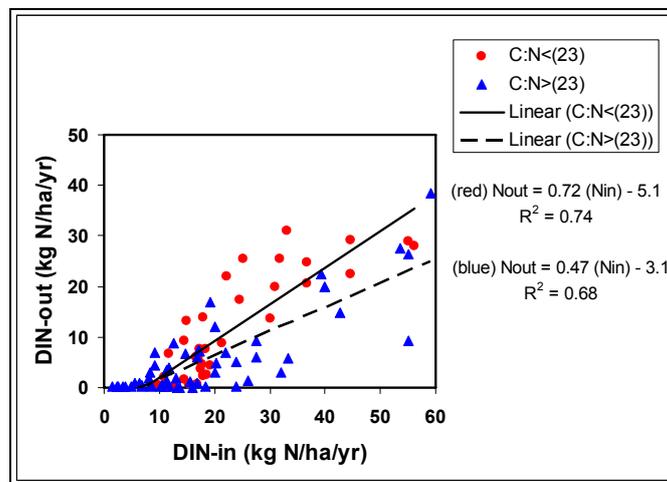


Figure 7 Scatter plots of the leaching of total inorganic N (DIN) against the N input as a function of the C/N ratio in the organic layer (After Dise et al., 2006).

In acidic soils, atmospheric deposition of S and N compounds do lead to elevated Al concentrations, in response to elevated concentrations of sulphate (SO₄) and nitrate (NO₃), and also to accumulation of NH₄ in situations where nitrification is (strongly) inhibited. This may cause nutrient imbalances, since the uptake of base cation nutrients (Ca, Mg, K) is reduced by increased levels of dissolved Al and NH₄ (Boxman et al., 1988). This effect may be aggravated in systems of low N status,

where an elevated input of N will increase forest growth, thus causing an increased demand for base cations. Observations of increased tree growth of European forests (Spiecker et al., 1996) may be the effect of increased N inputs. The effect of increased N inputs in combination with soil acidification has been emphasised in several studies (Schulze et al., 1989; Huttel, 1990; Olsthoorn & Tiktak, 1991). Nitrogen may stimulate tree growth and increase the demand of, for example, Mg, which has to be taken up (i) from a decreasing soil pool, (ii) by a root system which may be damaged by Al toxicity or be less effective due to a decline of mycorrhiza, and (iii) possibly in competition with NH_4 in elevated concentrations (Schulze, 1989). From several of the N saturated sites, deficiencies of elements such as Mg (Kazda, 1990; Probst et al., 1990) and K (Roelofs et al., 1988) are observed. These deficiencies may limit the capacity of the vegetation to retain N inputs hereby causing NO_3 leaching.

The possible impact of acid deposition on Al release and of N deposition on NH_4 accumulation is given in Figure 8. The release of Al in response to elevated SO_4 and NO_3 concentrations in subsoils with a low pH (below 4.5) is shown in Figure 8A. In those soils, more than 80% of the variation in Al concentration could be explained by a variation in SO_4 and NO_3 concentrations, which in turn were strongly related to the deposition of S and N, respectively. Although SO_4 is important in releasing Al, results showed that NO_3 concentrations were mostly higher, reflecting the increasing role of N in soil acidification. The NH_4/K ratio in the mineral topsoil in response to elevated N deposition is shown in Figure 8B.

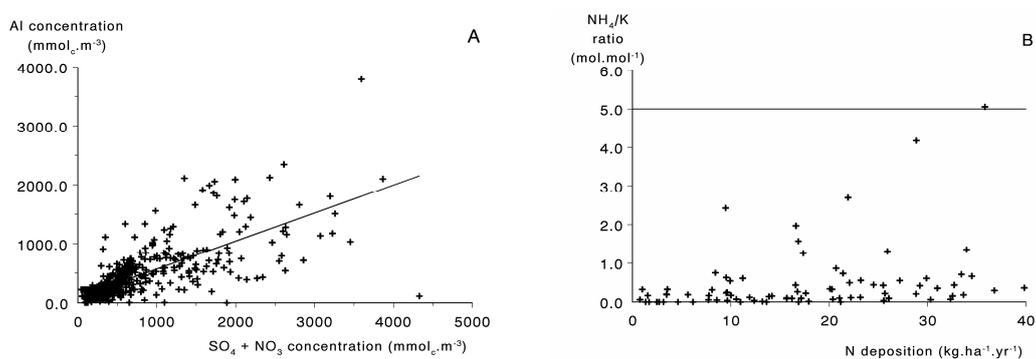


Figure 8 Scatter plots of the concentration of total Al against total $\text{SO}_4 + \text{NO}_3$ in the subsoil of Intensive Monitoring plots with a $\text{pH} < 4.5$ (A) and of the NH_4/Mg ratio in the mineral topsoil against the total N deposition (B). The line in A represents a regression line equal to: $\text{Al} = -95 + 0.74 (\text{SO}_4 + \text{NO}_3)$ ($R^2 = 0.86$).

Results do indicate that below an N deposition of approximately $10 \text{ kg.ha}^{-1}.\text{yr}^{-1}$, the NH_4/K ratios are hardly elevated, whereas they do increase above this value. The critical NH_4/K ratio of 5 is only exceeded once in the topsoil at an N input near $30 \text{ kg.ha}^{-1}.\text{yr}^{-1}$. The results hardly indicate a clear critical load for N in relation to N accumulation, but one could use $25 \text{ kg.ha}^{-1}.\text{yr}^{-1}$ as a reasonable precautionary value (De Vries et al., 2003b).

Impacts on the growth of shoots and roots

There are many publications showing that nitrogen fertilization increased growth or had no effect (e.g. Spiecker, 1991; Nilsson & Wiklund, 1992; Emmett, 1999). Braun et al. (1999) showed a significant correlation between stem increment of beech and modelled nitrogen deposition in Switzerland. It was, however, restricted to plots with sufficient phosphorus supply. When other nutrients are not deficient, increased N concentrations in foliage in response to elevated N inputs are generally associated with an increased growth unless the toxicity range is reached and growth is depressed, sometimes associated with visual symptoms of toxicity (see Figure 1).

Criteria for nutrient concentrations in foliage, including N, are mostly based on the fact that, within a certain range of foliar concentrations, positive relationships exist between the nutrient concentration and the growth and outer appearance of plants (i.e. the occurrence of visual symptoms of deficiency). One approach to classify ranges of foliar mineral nutrient concentrations is: deficiency, low, normal, optimal, high and toxicity range. In practice, however, foliar N concentrations hardly ever occur in the toxicity range. Data by Van den Burg (1988) suggest that concentrations above 20-25 g.kg⁻¹ are high for spruce and pine respectively, whereas concentrations above 30 g.kg⁻¹ are in the toxicity range.

Nellemann and Thomsen (2001), who analysed data from increment cores of >31,000 spruce forest plots in southern Norway for the time period 1954-1996, however, observed increases in growth during 1960-1970, followed by a subsequent decline in the 1980-1990 in plots with modelled wet nitrogen deposition >15 kg N.ha⁻¹.yr⁻¹ (Figure 9).

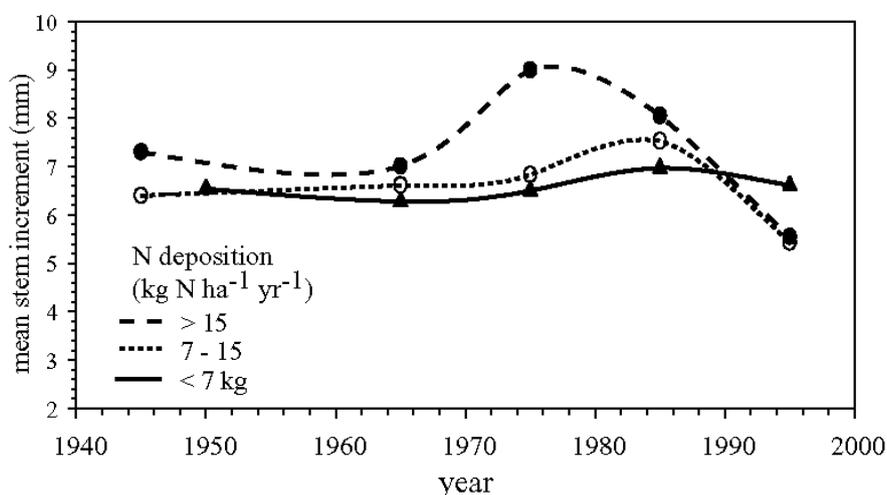


Figure 9 Stem increment of spruce in Norway from 31,606 increment cores grouped according to modelled wet nitrogen deposition. Growth increase in the highest deposition class as well as the decrease in the two highest classes are significant at $p < 0.01$ (After Nellemann & Thomsen, 2001).

This pattern of initial growth stimulation, followed by subsequent growth decline, may be due to the limitation of other nutrients such as phosphorus or base cations as suggested by Flückiger and Braun (1999a).

Unlike shoot growth, there is strong evidence that increased nitrogen deposition reduces fine root biomass and root length. Matzner and Murach (1995) found that total fine root biomass of Norway spruce saplings decreased significantly when the total inorganic N ($\text{NO}_3 + \text{NH}_4$) concentration in soil water was $>2 \text{ mg N l}^{-1}$. Furthermore, increasing root biomass and root vitality was reported when trees were protected from nitrogen deposition in roof experiments for Scots pine (Boxman et al., 1995), Douglas fir (Murach & Parth, 1999) and Norway spruce (Persson & Ahlström, 2002).

Nutritional imbalance due to elevated N concentrations in foliage

Increased N concentrations in foliage are generally associated with an increased growth unless the toxicity range is reached and growth is depressed, sometimes associated with visual symptoms of toxicity. In practice, however, foliar N concentrations hardly ever occur in the toxicity range and they are even seldom in the high range, except for nitrogen. Data by Van den Burg (1988) suggest that concentrations above $20\text{-}25 \text{ g.kg}^{-1}$ are high for spruce and pine respectively, whereas concentrations above 30 g.kg^{-1} are in the toxicity range. Note, however, that these ranges are specifically related to growth.

High foliar N concentrations in response to increased N deposition may however cause a nutritional imbalance, i.e. deficiencies of the macronutrients K, P, Mg and Ca, and possibly of micronutrients, B, Mn and Mo relative to N in needles. Increased growth rate and increased N concentrations in foliage may dilute the pool of other nutrients in absolute and/or relative terms. The relative shortage of P, K, Mg and Ca compared to an increased N content can be aggravated by the acidifying impact of both N and S compounds. As illustrated before, in soils with a low-base saturation (most sandy forest soils in Europe) an elevated input of S and N compounds, will cause the release of toxic Al that may reduce the availability of base cation nutrients (Ca, Mg, K) by affecting both root growth and root uptake (e.g. Sverdrup & Warfvinge, 1993). This effect may be aggravated in systems of low N status, where an increased input of N will increase forest growth, thus causing an increased demand for base cations. Observations of increased tree growth of European forests (Spiecker et al., 1996) may be the effect of increased N inputs. The effect of increased N inputs in combination with soil acidification has been emphasised in several studies (Schulze et al., 1989; Huttli, 1990; Olsthoorn & Tiktak, 1991). Nitrogen may stimulate tree growth and increase the demand of, for example, Mg, which has to be taken up (i) from a decreasing soil pool, (ii) by a root system which may be damaged by Al toxicity or be less effective due to a decline of mycorrhiza, and (iii) possibly in competition with NH_4 in increased concentrations (Schulze, 1989). From several of the N saturated sites, deficiencies of elements such as Mg (Kazda, 1990; Probst et al., 1990) and K (Roelofs et al., 1988) are observed. These deficiencies may limit the capacity of the vegetation to retain N inputs hereby causing NO_3 leaching.

Elevated atmospheric deposition of N and S compounds in the Netherlands during the period 1960-1990 has led to an increase in the N content and a decrease in the P,

K and Ca content in foliage. This can be derived from a study by Van den Burg and Kiewiet (1989), who compared the foliar composition of stands of Scots pine, black pine and Douglas fir in 1956 and 1988 in the 'Peel' area with intensive animal husbandry. Surprisingly, the Mg content did not decrease during that period. However, even in 1956, the Mg content was already low. Furthermore, as with P, K and Ca, the Mg supply relative to N decreased (Table 4).

Table 4 Average N content and ratios between K⁺, Mg²⁺ and N concentrations in half-year-old needles in 1956 and 1988 (Van den Burg & Kiewiet, 1989).

Tree species	N content (% dry weight)		Nutrient ratio x 100 (g g ⁻¹)			
	1956	1988	K/N		Mg/N	
			1956	1988	1956	1988
Scots pine	1.5	2.3	34	27	3.0	2.7
Corsican pine	1.2	1.7	58	35	4.0	3.8
Douglas fir	1.4	2.2	68	24	6.1	5.0

As reported in Bobbink et al (2003), experimental nitrogen addition to saplings of beech and Norway spruce in young stands on both acidic and calcareous soils induced nutrient imbalances and deficiencies, that were significant at added nitrogen loads of ≥ 10 -20 kg N.ha⁻¹.yr⁻¹ after 4-6 years of nitrogen treatment (modelled atmospheric deposition 12-20 kg N.ha⁻¹.yr⁻¹). On acidic soil, nitrogen treatment led to acute Mg deficiency (Figure 10) whereas on calcareous soil K and P became limiting. (Flückiger & Braun, 1999b).

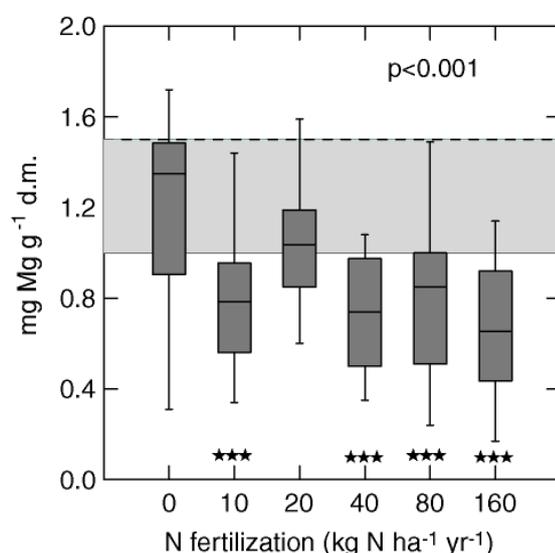


Figure 10 Magnesium concentration in beech leaves in a nitrogen fertilization experiment on acid soil. Significant differences to control are indicated with *** $p < 0.001$, overall linear regression $p < 0.001$ (After Flückiger & Braun, 1999a) Grey field: range for optimum nutrient concentration after Stefan et al. (1997).

Addition of 35 kg N.ha⁻¹.yr⁻¹ to an ambient nitrogen input of 15-20 kg N.ha⁻¹.yr⁻¹ in a spruce forest at Klosterhede led to increased needle nitrogen concentrations and decreased P and Mg concentrations in the foliage during the 3 years of treatment (Gundersen, 1998).

Insight in the possible impact of N deposition on a nutritional imbalance in foliage in Europe has further been derived for more than 100 Intensive Monitoring plots in European Forest with information on both the chemical composition of the foliage and total N deposition (De Vries et al., 2003b). Ranges in N deposition at plots with a balanced and unbalanced ratio of P, K, Ca or Mg to N are given in Table 5. More information on the criteria related to balanced and unbalanced ratios is given in De Vries et al. (2000) and Flückiger and Braun (2003).

The number of plots with a clearly unbalanced nutrition is approximately 50%, with relative P deficiencies being the most important reason for an imbalance, followed by relative Mg deficiencies. When considering all elements P, K, Ca, Mg, there was an unbalanced ratio of one or more of those elements at 57 of the 109 plots.

Table 5 Ranges in N deposition at 109 Intensive Monitoring plots in Europe with a balanced and unbalanced ratio of the base cations K, Ca or Mg to N.

Element	N deposition kg.ha ⁻¹ .yr ⁻¹							
	Unbalanced				Balanced			
	Nr of plots	50%	5%	95%	Nr of plots	50%	5%	95%
P	46	21	6.9	34	63	11	1.5	34
K	15	23	14	37	94	14	1.7	33
Ca	4	28	20	35	105	16	1.9	34
Mg	24	22	11	35	85	13	1.7	33
All	57	21	7.8	34	52	9.6	1.4	32

The results clearly indicate a larger N deposition at the plots with an unbalanced ratio. Considering all base cations, the median N deposition is 10 kg.ha⁻¹.yr⁻¹ (range of 2-32 kg.ha⁻¹.yr⁻¹) at the plots with a balanced nutrition and 21 kg.ha⁻¹.yr⁻¹ (range of 8-34 kg.ha⁻¹.yr⁻¹) at the plots with an unbalanced nutrition. These results do at least suggest that an unbalanced nutrition hardly ever occurs at an N deposition of 10 kg.ha⁻¹.yr⁻¹, thus suggesting a critical load of approximately 10-20 kg.ha⁻¹.yr⁻¹ in view of tree nutrition.

Increased sensitivity to frost, drought and fungal diseases

Reports on the effects of N on frost sensitivity are contradictory, ranging from positive (+, increasing frost hardiness) to negative effects (-, decreasing frost hardiness), with 0 denoting no change: Aronsson (1980) (-) and Jönsson (2000) (-); DeHayes et al. (1989) (+), Klein et al. (1989) (+), L'Hirondelle et al. (1992) (+), and Perkins et al. (2000) (cold hardiness +, winter injury -); Hellergren (1981) (0), Thomas and Ahlers (1999) (0), Jönsson et al. (2004) (0) and Fløistad (2002) (0).

Flückiger and Braun (1999a) found an increased sensitivity to drought. in response to N inputs. In a nitrogen addition experiment (0, 25, 50, 100, 200, 400 kg N.ha⁻¹.yr⁻¹) in pots, the shoot/root ratio of beech was significantly increased from 25 kg N.ha⁻¹.yr⁻¹ onwards and the leaf surface per tree from 50 kg N.ha⁻¹.yr⁻¹ onwards after six year's treatment. This led to more rapid water loss, as shown by a negative correlation between water content of the soil one day after watering and nitrogen fertilization (significant from 100 kg N.ha⁻¹.yr⁻¹ onwards). Thomas et al. (2002) found that planted

beech saplings in a nitrogen fertilization experiment showed significantly increased leaf necroses after short drought periods in the second and third treatment year at more than 20 kg N.ha⁻¹.yr⁻¹ (Figure 11).

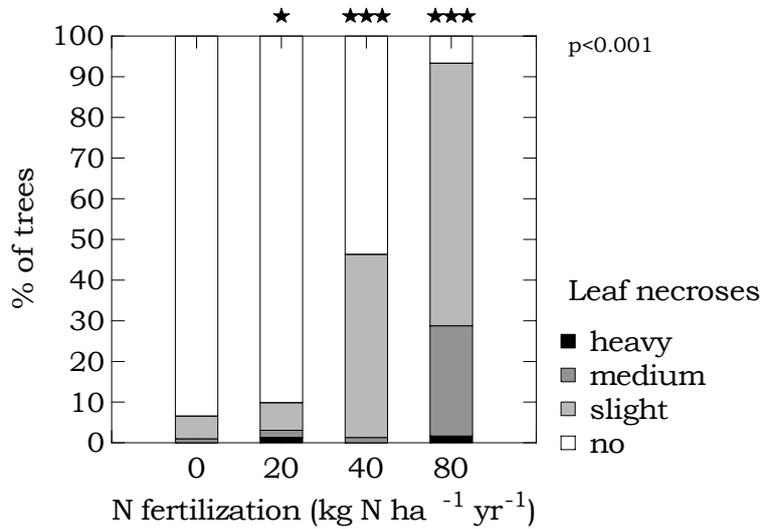


Figure 11 Leaf necroses in beech caused by drought in a nitrogen addition experiment. Differences to control significant at * $p < 0.05$, *** $p < 0.001$, general linear trend $p < 0.001$ (After Thomas et al., 2002).

Increased N concentrations in foliage may cause an increased sensitivity to diseases and plagues, such as attacks of the fungi *Sphaeropsis sapinea* and *Brunchorstia pinea* (Roelofs et al., 1985; Van Dijk et al., 1992; Flückiger & Braun, 1998). These authors reported N concentration values of approximately 18-20 g.kg⁻¹ as critical values. For most coniferous tree species, an N concentration in this range is considered optimal for growth (e.g. McNulty et al., 1991).

Figure 12 shows the relationship between N contents in first year needles of Scots pine and total N deposition at 68 plots in Europe (De Vries et al., 2003b). The empirical relationship between N deposition and foliar N content indicates that at an N load (in throughfall) of approximately 15-25 kg.ha⁻¹.yr⁻¹, an approximate critical N concentration of 1.8% for pine is mostly exceeded (Figure 12). This coincides with the empirical range of 15-20 kg.ha⁻¹.yr⁻¹ reported by Bobbink et al. (2003), which is partly based on the results presented here. These values are exceeded in high N deposition areas located in (parts of) the Netherlands, Belgium, Denmark and Germany.

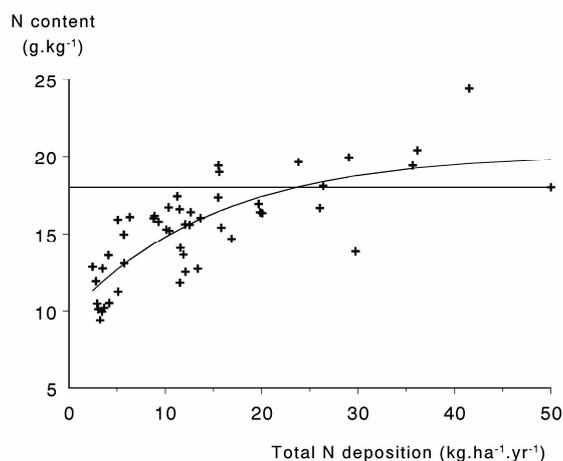


Figure 12 Relationship between N contents in first year needles of Scots pine and total N deposition at 68 plots in Europe.

A summary of the critical N loads related to impacts on trees and soil processes is given in Table 6 with a comparison of the results presented in Bobbink et al. (2003).

Table 6 Summary of the range in critical N loads related to impacts on trees and soil processes based on the results predicted in this report compared to those given in Bobbink et al. (2003).

Compartment	Effect	This study	Bobbink et al. (2003)
Soil processes	Increased mineralization, nitrification and N leaching	10-20	10-15
Trees	Increased susceptibility to frost drought, pathogens and pests	15-25	15-20
	Decreased P, K and Mg to N ratios in foliar tissue	10-20	15-20

2.4 Other impacts

Apart from the impacts on soil chemical processes, plant nutrition (forest growth) and plant species diversity, as summarized in Figure 1 and described above, there are additional impacts on the terrestrial ecosystem. Most relevant are the impacts on faunal species diversity and ground water quality, as summarized below. For these impacts, it is not possible (faunal species diversity) or relevant (ground water quality) to derive a critical nitrogen load

2.4.1 Faunal species diversity

Nitrogen deposition and faunal species diversity loss

Until recently, research on acidification, eutrophication, as well as restoration management has mainly focussed on vegetation and abiotic processes. Research on fauna is complicated, as different species use the landscape at different spatial scales and animal species outnumber plant species by about 25 to 1. Consequently, research on the effects of N deposition on fauna is largely lacking. There is however a clear

impact by affecting food and environmental conditions, including micro-climate, but also heterogeneity in landscape and vegetation structures, needed by animal species to complete their life-cycles.

Elevated N deposition causes changes in nutrient content of plant organic matter and plant species composition and thereby alters the micro-climate (temperature and moisture regimes) experienced by animals. As an example, increased N deposition has consequences for herbivorous animals like caterpillars, as their host plants may decrease or increase in abundance, or because of changes in food quality (Bink, 1992; Soontjens & Bink, 1997; Kerslake et al., 1998). Changes in nutrient content of dead organic matter has also consequences for detritivores, as e.g. Vos et al. (2000; 2002) showed for aquatic invertebrates.

Because of elevated N deposition, landscape heterogeneity is also often declining due to e.g. extensive grass encroachment. The occurrence of animal species is related to landscape heterogeneity by at least three mechanisms. First, species may depend on specific conditions, which are only present in transitions between different biotopes. Second, many animal species require different parts (biotopes) of the landscape for reproduction, resting, foraging, etc. Third, heterogeneity creates the possibility of risk spreading, leading to a higher persistence of species (Verberk et al., 2002; Verberk & Esselink, 2004). Therefore, N deposition affects fauna diversity not only directly (e.g., changes in food quality and micro-climate), but also indirectly through changes in landscape configuration and heterogeneity. Below, two examples are given of the impacts of N deposition on faunal diversity in the Netherlands.

Decline of ground beetles in dry coastal dune grasslands

The ground beetle (Carabidae) assemblages of dry open sandy coastal dune grasslands is characterised by species preferring drought and higher temperatures. N deposition, however, results in grass encroachment. Consequently, the characteristic micro-climate of coastal dune grasslands (very warm during day time, but fairly cold at night and continuously dry) changes to a buffered micro-climate (continuously cool and moist). Comparison of the ground beetle assemblage between 15 coastal dune grasslands on the Waddensea isles Ameland and Terschelling showed that encroachment with the grasses *Calamagrostis epigejos* and, to a lesser extent, *Ammophila arenaria* results in a change in the relative numbers of drought vs. moisture preferring species. Grazing by sheep is one of the nature management measures used to combat grass encroachment. Grazing results in a strong decrease of tall grasses, but the ground beetle assemblage remains dominated by moisture preferring species, instead of the warmth and drought preferring species dominating in more intact dune grasslands (Nijssen et al., 2001, Figure 14).

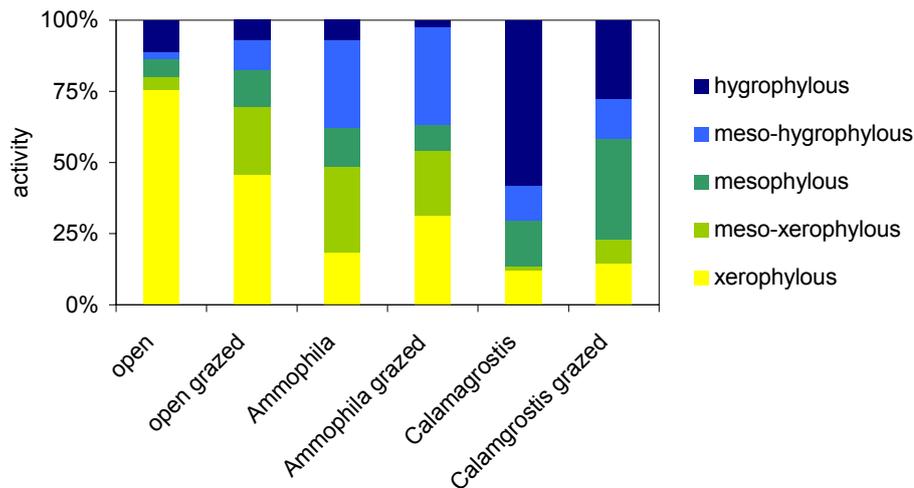


Figure 13 Relative activity of ground beetles classified according to their preference for moist (hygrophilous) to dry (xerophilous) conditions in different coastal dune grasslands on the Dutch Waddensea isles Ameland and Terschelling (From Nijsen et al., 2001).

Decline of red-backed shrike

The decline of the red-backed shrike (*Lanius collurio*) illustrates how the effects of elevated N deposition have repercussions across the entire food web (Beusink et al., 2003). This bird species strongly declined from 1900 onwards throughout Western Europe. It has currently disappeared from the coastal dunes of the Netherlands and it is disappearing from the coastal dunes of northern Germany and southern Denmark. Only in the coastal dunes of northern Denmark the population of red-backed shrikes is still stable (Figure 14).

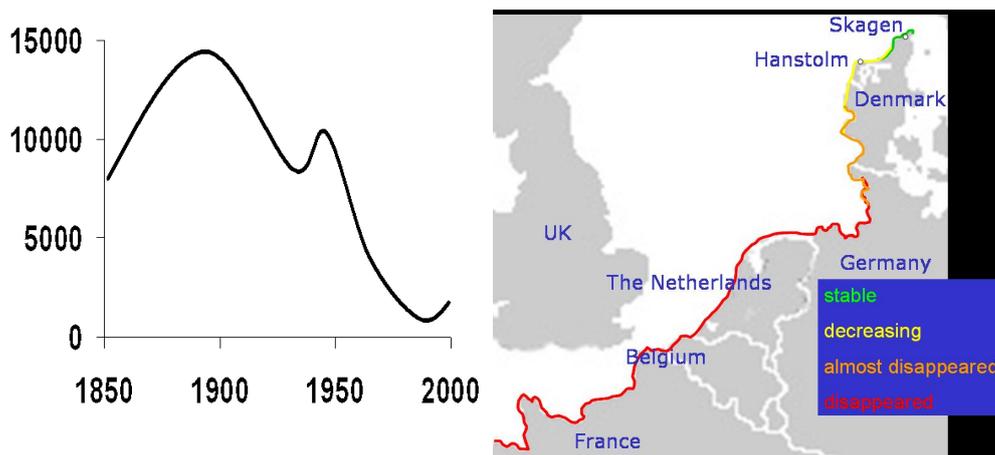


Figure 14 The change in the number of breeding pairs of the red-backed shrike in the Netherlands (left) and population trends in coastal dunes of Western Europe (right).

This pattern in population trends is clearly correlated to atmospheric N deposition levels, although the occurrence of this bird species can of course not directly be

related to higher N availability. Red-backed shrikes feed on large insects and small vertebrates (e.g. lizards) and carry only a single prey item to the nest at a time. Prey demands of the nestlings have to be met during the day under different weather conditions and also during the whole breeding period. To ensure a constant and sufficient energy supply, the red-backed shrikes require a high diversity of large prey species which in turn depends on landscape heterogeneity (Esselink et al., 1994). In Dutch coastal dunes, increased N deposition led to the encroachment by tall grasses and bushes, a decrease of open sandy areas and a loss of succession stages rich in species.

The decline in landscape heterogeneity seriously affected the prey availability for red-backed shrikes. Especially the lack of sufficient large prey species in the Netherlands is considered as an important factor (Beusink et al., 2003).

2.4.2 Ground water quality

Water quality is a major concern throughout Europe. Nitrogen pollution of ground water mitigates its use for drinking water. Nitrate concentrations in drinking water should not exceed 50 mg l⁻¹ (Drinking Water Directive). In several countries (e.g. the Netherlands) the target value is even set at 25 mg l⁻¹. However, exceedances are a common problem across Europe, particularly from shallow wells. It is often a problem in rural water supplies. For example, in Belgium 29% of 5000 wells examined had concentrations in excess of the limit value (OECD, 1997) and in Bulgaria it was estimated that, in the early 1990's, up to 80% of the population was exposed to nitrate concentrations greater than the limit value (OECD, 1995). Exceedances of the limit value were found in about a third of the groundwater bodies for which information is available. In general, there has been no substantial improvement in the nitrate situation in European groundwater and hence nitrate pollution remains a significant problem (EEA, 2003b).

Groundwater nitrate contamination associated with fertiliser use is common in both developed and developing regions (Oenema et al., 1998; Agrawal et al., 1999). Severe instances of groundwater contamination are, however, often associated with intensive livestock production, particularly swine and poultry (Mallin, 2000). The potential health effects of high nitrate levels are diverse, including reproductive problems (Kramer et al., 1996), methemoglobinemia, and cancer. Infants are especially at risk for methemoglobinemia ("blue-baby" syndrome). In this context, groundwater nitrate contamination is a serious problem due to its poor reversibility (Van Lanen & Dijkema, 1999).

While little conclusive evidence exists for this disorder at levels below 10 ppm, higher values found throughout the world can significantly elevate the risk (Gupta et al., 2000). The World Health Organisation thus adopted a 10 mg NO₃-N.l⁻¹ standard for safe drinking water, being close to the EU standard 11.3 mg NO₃-N.l⁻¹ (NO₃ is 50 mg.l⁻¹). Recent evidence suggests that nitrate levels even below the WHO standard of 10 ppm may stimulate formation of N-nitrosoamines (Van Maanen et al., 1996), compounds strongly implicated in cancer risks. In Iowa, rising nitrate levels well

below the WHO standard were associated with an increased risk of bladder and ovarian cancers (Weyer et al., 2001). Long-term consumption of water with nitrate-N concentrations above 6.3 mg NO₃-N.l⁻¹ has been linked to a higher risk for Non-Hodgkin's lymphoma (NHL), a cancer disease that has increased dramatically in the US (Ward et al., 1996).

In the Netherlands, the EU standard 11.3 mg NO₃-N.l⁻¹ is often exceeded in upper groundwater and this is especially the case with the target value of 5.6 mg NO₃-N.l⁻¹ (NO₃ is 25 mg.l⁻¹). High nitrate concentrations in drinking water wells, due to elevated nitrate leaching to ground water, occur specifically in well drained sandy soils below intensive agricultural areas (Fraters et al., 2004, Figure 16A). Apart from agricultural soils, high concentrations also occur below non-agricultural (specifically forest) soils in the Netherlands, because of high N deposition, low denitrification rates and low precipitation surpluses (De Vries et al., 1995, Figure 16B).

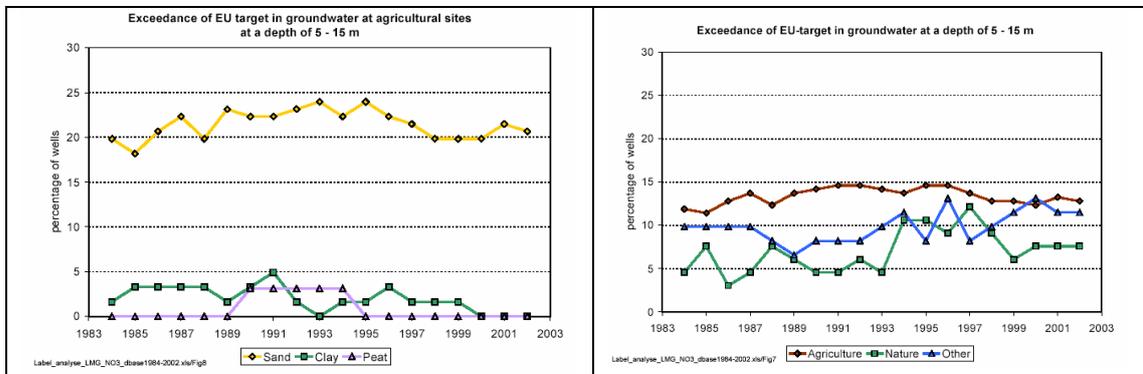


Figure 15 Exceedance of the EU target value of 50 mg.l⁻¹ for nitrate in groundwater within the agricultural areas of the Netherlands (left, A) and for all type of land use (right, B) at a depth of 5-15 m below the surface level for the 1984-2002 period (Source: Fraters et al., 2004).

Even worldwide, the 10 mg NO₃-N.l⁻¹ standard is often exceeded. In the US, where the Safe Drinking Water Act regulates this standard, regional studies suggest that 10–20% of groundwater sources may exceed 10 mg NO₃-N.l⁻¹ (is 10 ppm).

2.5 Evaluation of the presented critical loads

Based on observed changes in the structure and function of ecosystems, reported in a range of publications, empirical critical loads for nitrogen were set for specific receptor groups of natural and semi-natural ecosystems in 1992, 1996 and 2002 (Bobbink et al., 1992; 1996; 2003). The comparability and reliability of these critical loads depends on the (i) comparability in the type of effects which have been studied and (ii) methodology that has been used to derive the critical loads. Both aspects are shortly discussed below, while ending with an overview of the most important gaps in knowledge.

Indicators of change used for biodiversity impacts

In deriving empirical critical loads, being the highest input of nitrogen that does not lead to adverse effects, it is important to define the indicators of change, that are considered relevant for biodiversity impacts. In various definitions, an excess of critical loads has been related to the occurrence of changes in ecosystem structure and function. Changes in ecosystem function refer, for example, to soil processes, such as decomposition and mineralization and related processes, such as leaching and accumulation. Changes in ecosystem structure refer to (i) species characteristics (e.g. physiological or morphological changes in plants), (ii) vegetation structure, such as vegetation height, productivity/biomass and (iii) species composition, such as higher plants, mosses, lichens, mycorrhiza. The studied effects in deriving critical N loads range from (see also Table 2):

- changes in chemical processes (leaching, mineralization, accumulation),
- physiological or morphological changes in plants,
- changes in vegetation structure and/or plant species composition.

The first effect is related to ecosystem function and the latter two effects to ecosystem structure. Changes in species composition are most relevant with respect to the UNEP/CBD and EU goals on protection of biodiversity, since those changes are directly related to the loss of diversity on the species level. Changes in chemical processes and or plant physiological or morphological changes are less relevant in that respect. This is reflected in the indicators that are used to derive critical N loads. The derived critical loads for 27 ecosystems are mainly based on species composition (25), often combined with vegetation structure (20) and to a lesser extent with a combination of impactys on species characteristics (3) or chemical processes such as leaching (5) and mineralization, (2). Changes in species composition is thus the dominant indicator (see also Table 2).

However, diversity in plant species is only one aspect of biodiversity. Other aspects are also relevant, given the set of headline indicators on ‘status and trends of the components of biological diversity’ and ‘sustainable use’ used by the EU and UNEP for reporting on progress towards the 2010 biodiversity target. Furthermore, with respect to ‘ecosystem structure’ it is important to focus on protected species and characteristic species. With respect to ‘ecosystem function’ it is important to focus on ‘sustainable use/management’ and ‘species in functional groups’. There is an emerging agreement, that the effects on ecosystem function should be attributed to the functional traits of species, rather than to species number (see Nordin, 2007 and the references therein). Due to the complications related to classifying species into relevant functional groups, it has sometimes been assumed that plant species richness can serve as a surrogate for functional richness. Although the two types of richness often are correlated, it has been argued, however, that this relationship is not sufficiently universal to justify using species richness as a reliable proxy for functional richness (see Nordin, 2007 and the references therein). So far, not many studies have addressed whether nitrogen induced species loss also have affected ecosystem function (see Nordin, 2007 and the references therein).

Reliability of empirical critical loads in view of its derivation

The available studies not only differ in the type of effects which have been studied, but also in the methodology used to derive critical loads. In this context, a distinction can be made in N manipulation experiments and observational studies. In N manipulation experiments, the level of N input is manipulated, either in the field or in the laboratory in 'mesocosms' (pieces of vegetation directly taken from the field). In observational studies, the critical load is derived by (Sutton et al., 2003):

- Comparing species composition of areas with contrasting N deposition or
- Assessing responses of e.g. species composition in time series under a known increase in atmospheric N deposition.

Empirical critical loads that are based on observational (correlative or retrospective) field studies alone are highly uncertain, because of uncertainty in causal relationships between deposition and effects and since the uncertainty in N deposition assessment (mostly modelled deposition).

From the perspective of reliability, the N manipulation experiments are thus to be preferred, since the uncertainty in the background level of N deposition makes a smaller contribution to the effects level than those by observational studies. The current empirical critical loads have in principle been based on addition field experiments and mesocosms. In addition, the results from correlative or retrospective field studies have been used, but only: (i) as additional evidence to complement the results from experimental nitrogen addition studies (done for critical loads considered as reliable) or (ii) as an indication for expert judgement.

Statistically and biologically significant outcomes of long term field addition experiments and mesocosm studies, with realistic nitrogen loads, are most appropriate to assess reliable empirical nitrogen critical loads. Realistic nitrogen loads and durations are loads below $100 \text{ kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ that have lasted for more than 1 yr. Because of the time and labour-intensive nature of such studies, results are only available for a rather limited group number of broadly defined ecosystems. Even when using experimental studies in pots and mesocosms or field addition experiments, it is important to realize the uncertainties involved including the:

- Uncertainty in background N deposition, which has to be summed with the added N.
- Level of the lowest N addition where effects occur, which can be too high due to the chosen interval in N additions.
- Occurrence of high background N deposition in certain areas. This may lead to high N additions before effects show up, since they have mainly occurred already and it is needed to use N removal experiments.

Gaps in knowledge and needed further research

Although considerably progress has been made in setting critical loads for ecosystem types from 1996 to 2005, still serious gaps in knowledge exist. In Bobbink et al. (2003) ranges of empirical critical loads could be set for 29 different ecosystem types using the EUNIS classification. Additional research/data collection was considered as required to establish a critical load for especially steppe grasslands, Mediterranean vegetation types, wet-swamp forests, different types of mire and fens, several coastal habitats and high altitude systems and this holds until now. Moreover, with the

available information reliable critical load ranges could be set for only 5 different ecosystem types (Bobbink et al., 2003). Critical loads were depicted as reliable, when a number of published papers of various studies show comparable results. For the majority of the ecosystem types the critical loads were classified as ‘quite reliable’ (8x) or as ‘expert judgement’ (16x). Critical loads were depicted as expert judgement, when no or very limited experimental data was available but international experts agreed that critical loads could be set, sometimes based on correlative or retrospective) field studies (Bobbink et al., 2003).

It is still very important to increase the link between exceedance of critical nitrogen loads and effects on biodiversity, such as species richness. This will increase the value of their use in deriving national emission ceilings for NO_x and NH₃ and by the UNEP-Convention on Biological Diversity (CBD) and the European Habitat directive. An example of such a link is given in Figure 3 and 4. Another example for various ecosystems, including several grassland types, wetlands, (sub)arctic and alpine vegetation and temperate forests, is presented in Figure 16 (Bobbink et al., submitted). The figure shows a clear link between exceedance of critical nitrogen loads and the species richness ratio in various European N addition experiments.

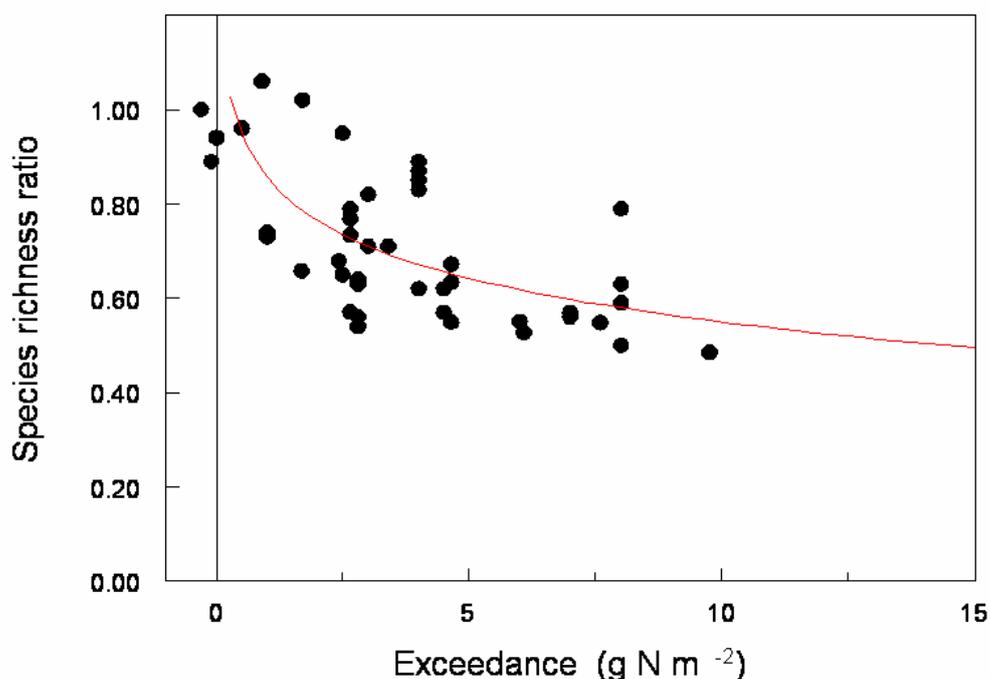


Figure 16 The species richness ratio (i.e. the ratio of the mean number of plant species in the N-treated vegetation and in the controls) and the exceedance of the empirical critical nitrogen loads in European N addition experiments in several grassland types, wetlands, (sub)arctic and alpine vegetation and temperate forests. (n=44; addition for two or more years) (Bobbink et al., submitted).

In this context, research/data collection is required to (see also Bobbink et al., 2003; Dorland & Bobbink, 2006):

- Establish a critical load for steppe grasslands, wet-swamp forests, many mire & fens, several coastal habitats, several high altitude systems and all Mediterranean vegetation types;
- Derive a more reliable critical load for all distinguished EUNIS items with critical loads based on expert judgement or few research;
- Assess impacts of nitrogen enrichment in (sensitive) freshwater and shallow marine ecosystems;
- Allocate observed nitrogen effects to the appropriate EUNIS forest subtypes;

Other gaps in knowledge and possible ways to solve them are (see also Bobbink et al., 2003; Dorland & Bobbink, 2006):

- Clarification/adjustment of the EUNIS classification with respect to some grasslands groups, Nordic bogs and mires and surface water habitats;
- Study of possible differential effects of the deposited nitrogen species (NO_y or NH_x), which are currently insufficiently known to make a differentiation between these nitrogen species for critical load establishment;
- Use of long-term (>3-5 yrs) nitrogen addition experiments with a high resolution of treatments between 5 and 50 $\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ at low background regions or in mesocosms. This would increase the certainty of deriving critical loads when the lowest treatment level considerably exceeds the critical load.

3 Modelling approaches for the derivation of critical nitrogen loads for plant species diversity impacts

3.1 General approach

Here, we describe the possibilities that multi-plant species models in combination with a dynamic soil vegetation model may have for directly calculating critical N loads. In principle, integrated models can be used to predict plant species composition as a function of atmospheric deposition, as illustrated in Figure 17. The principle of the approach is that a dynamic soil model, such as SMART2, VSD or (For)SAFE, predict the changes in soil and soil solution chemistry, whereas a statistical model (NTM/MOVE, BERN) or process based model predict changes in the plant species composition or forest succession (SUMO, VEG).

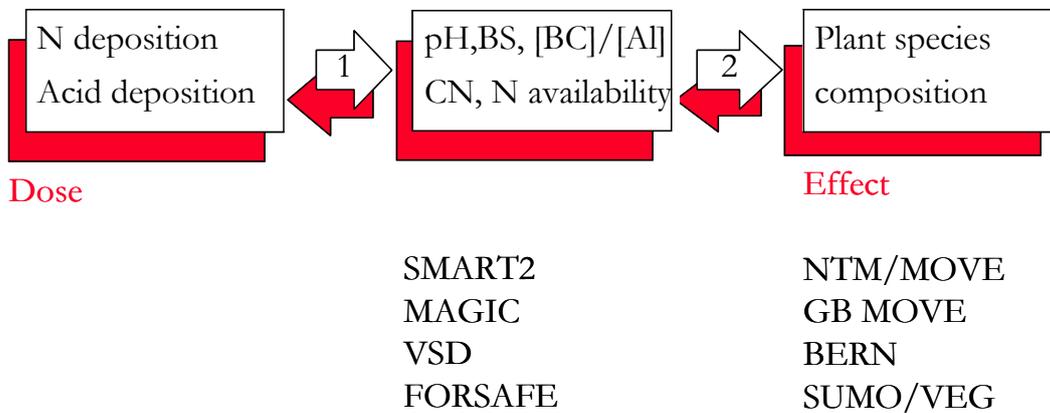


Figure 17 Method to predict plant species composition as a function of atmospheric deposition. In the first step predictions are made related to nitrogen (e.g. nitrogen availability or C/N ratio) and acidity (e.g. soil pH or base saturation) using a dynamic soil model and in the second step the plant species composition is predicted using plant species specific information on habitat preferences.

In principle, these models can also be used in an inverse way as given in Figure 18. Based on a targeted vegetation type, critical values for abiotic factors (e.g. N availability or C/N ratio and pH or base saturation) are derived (step 1) and subsequently used in nutrient cycling models (step 2) to calculate the critical N load.

This approach was first applied in the Netherlands in which the:

- critical pH and N availability for the plant species composition is based on Ellenberg indicator values for N (EN) and acidity (ER), that are used to indicate the preference of each species (MOVE model, Latour & Reiling, 1993) or each vegetation type (NTM model, Schouwenberg et al., 2000).
- dynamic soil model SMART2 (Kros et al., 1995) was used to calculate the critical loads at which the above critical limits were not exceeded. It was assumed that under the critical deposition levels the critical limit for neither nitrogen availability nor soil pH were exceeded.

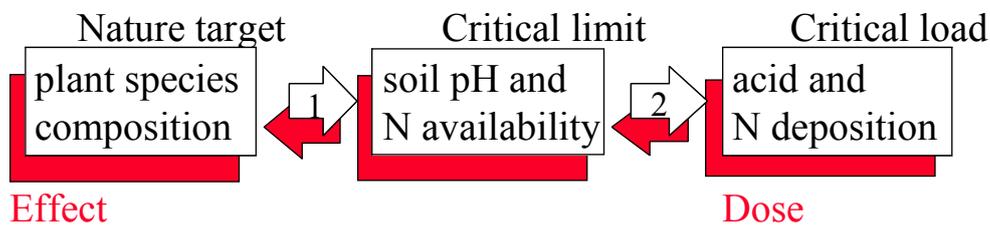


Figure 18 Method for calculating critical loads for nitrogen and acid deposition. In the first step critical limits for different nature conservation targets are derived from plant species specific information on habitat preferences for nitrogen availability and soil pH. In the second step a dynamic soil model is used to back calculate the critical loads.

Normally, the model SMART2 is used to estimate the soil conditions that determine biodiversity (pH and nitrogen availability) at given deposition levels of sulphur and nitrogen compounds (Kros, 2002). For this application SMART2 was ‘inverted’ to SMART2⁻¹, to produce deposition levels that lead to a given pH and N availability. It should be noted however that any other model (e.g. SAFE, Sverdrup et al., 1995) can be used as long as it is able to calculate a critical load. Originally, an iterative procedure was used that searches the nitrogen deposition level that yields the predefined values for soil pH and N availability in successive dynamic SMART2 runs with varying deposition levels. This procedure is also used for the calculation of target loads (Van Hinsberg & Kros, 1999). However, this methods does not yield a formal steady state, which is by definition needed for a critical load. Hence, a steady state version of SMART2 has recently been developed to calculate a critical N load as function of critical pH and N availability, derived from the Ellenberg indicators E_N and E_R .

Alternative approaches include the use of critical limits for other abiotic parameters, such as the C/N ratio (used in the Bern model by Schlutow & Hübener, 2004) or of soil N, P, BC availability, soil moisture, pH, light and grazing pressure (used in the ForSafe-VEG model: (Wallman et al., 2005). Furthermore, other soil acidification models than SMART2 can be used, such as VSD or SAFE used in combination with the BERN model and ForSAFEVEG to derive critical N loads in view of plant species diversity impacts on the basis of critical abiotic impact parameters, such as N availability, C/N ratio and pH. The various approaches that are presently explored are presented below, including an application example illustrating the modelling approach and an evaluation of the weaknesses and strengths.

3.2 The SMART2-SUMO-MOVE/NTM model

General approach

The SMART2-SUMO-MOVE/NTM model is developed and used in the Netherlands and the UK. It consists of: (i) relationships between species diversity and the abiotic factors pH and N availability, derived indirectly via Ellenberg indicator values, as included in the MOVE and NTM models and of (ii) the soil acidification model SMART2. SMART2 can be used both in its original form, allowing the calculation of target loads, and as a steady state version, allowing the calculation of critical loads. In this section we first describe how relationships between abiotic conditions and plant species occurrence are included in the MOVE and NTM models, then the model SMART2 and the adapted versions for the calculation of critical loads (SMART2 steady state) and target loads followed by an application of these models for the Netherlands. In Annex 1, background information is given on the calculation of critical loads and target loads with the model chain SMART2-NTM; a reverse version of the vegetation succession model SUMO is not available yet.

The MOVE/NTM models to assess relationships between abiotic conditions and plant species occurrence

Use of species response curves

Both MOVE and NTM are based on response curves in which the probability of plant species occurrence is determined by, mainly abiotic, site conditions, including soil pH and N availability. The probability of occurrence is visualised as a simplified bell-shaped optimum curve, representing species occurrence along a single environmental gradient (Figure 19).

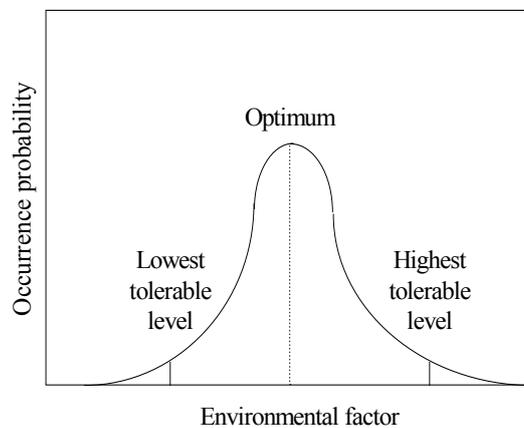


Figure 19 A hypothetical species response function visualised as a bell-shaped optimum curve, representing species occurrence along a single environmental gradient. The range between the lowest and highest tolerable level indicates the range of suitable environmental conditions.

Impacts of nitrogen and (related) acid deposition on plant species diversity are due to changes in pH and N availability (the environmental factor on the x-axis), thus causing changes in the occurrence probability of individual species. Inversely, critical

limits for the nature conservation target types can be calculated from the list of species to be protected within a target type and the ranges of suitable environments for species occurrences. However where MOVE uses response curves for individual plant species NTM uses response curves for vegetation associations).

Since the plant species and vegetation occurrence depends on more than one environmental factor the real curves are in fact multidimensional bell-shaped. The species response curves can be used to determine the range of suitable environments at the species levels. Latour et al. (1994) described the use of the 10 and 90 percentiles as measures for risk assessment, analogous to NOECs (No Observed Effect Concentrations). In this view the 10 and 90 percentiles correspond to sub-optimal environmental conditions with reduced occurrence probability due to “limitation” or “intoxication”, respectively. Between these percentiles the environmental conditions are suitable for plant species occurrence (MOVE). This principle is also applied for vegetation types used in NTM.

The critical pH and N availability for the plant species composition is based on Ellenberg indicator values for N (E_N) and acidity (E_R), that are used to indicate the preference of each species (MOVE model, Latour & Reiling, 1993) or each vegetation type based on the species present in relevés representing the vegetation type (NTM model, Schouwenberg et al., 2000; Wamelink et al., 2003a) for these environmental factors. Ellenberg (1991) built a database of species responses to abiotic circumstances, mostly based on expert judgement. Essentially, the optimal values of a number of abiotic factors have been estimated for a large number of species, and scored in an arbitrary nine-point scale. In order to use these values as criteria for critical load calculations, we hypothesized that (i) species with equal Ellenberg scores have an equal response, and (ii) there is a monotonous relation between the Ellenberg estimates and the ‘real’ measurable values. If those conditions are met, the Ellenberg values can be translated into measurable values if a training set consisting of relevés with known abiotic conditions is available (Ertsen et al., 1998; Wamelink et al., 2002).

Assessment of the relationship between Ellenberg indicator values and the actual pH and N availability requires that these abiotic conditions have been measured for all vegetation relevés considered in the nature targets to be protected. However, direct measurements of abiotic circumstances are lacking for the vast majority of these relevés. Therefore a two-step procedure was used in the Netherlands to derive the critical conditions:

- First, the critical conditions in terms of Ellenberg’s indicator values were derived from the species composition of the relevés (original species in the MOVE model and vegetation types in the NTM model) followed by aggregation into nature target types;
- Second, the indicator values were translated into measurable entities, using a separate training set of relevés whose abiotic conditions had been measured (Ertsen et al., 1998; Wamelink et al., 2002).

Derivation of Ellenberg's indicator values from species composition

Step 1: Both MOVE and NTM are developed to evaluate the effects of a changes in soil pH, N availability and ground water level on species diversity for plant species. Within the context of critical N load calculations we are only interested in soil pH and N availability. Ground water level is not considered as a limiting factor within the context of critical nitrogen load calculations. The pH, however, is included as limiting factor because nitrogen deposition influences the pH. With regression statistics the probability of occurrence of a species can be calculated for each combination of soil factors or for each environmental variable separately resulting in species-response curves. For MOVE, species-response curves of about 900 plant species have been determined for soil moisture, nutrient availability and soil acidity (Wiertz et al., 1992) using Gaussian logistic regression models. Although, it is known that species diversity is affected by several nutrients (cf. Olde Ventering, 2000), so far only N has been considered. Based on the same principle, the response of vegetation associations is estimated in NTM, resulting in vegetation type response curves.

Regression results of both MOVE and NTM were based on an extensive database developed for a revision of the Dutch classification of plant communities (Schaminée et al., 1989). This database consists of more than 100 000 vegetation relevés. The regression functions can be visualized as bell-shaped optimum curves, representing species occurrence along a single abiotic gradient. Since MOVE and NTM focus on more than one abiotic factor, the real curves are, in fact, multi-dimensional and bell-shaped. Where MOVE uses Gaussian response curves, NTM uses the more sophisticated spline technique to estimate responses. Because this analysis use only floristic information to assess the abiotic site factors, any (historical) vegetation relevé can be included in the analysis. Subsequently the ranges for species are used to determine the range of suitable environments for the different nature target types, distinguished in the nature policy in the Netherlands (Bal et al., 2001). The critical limits are calculated as the highest nitrogen availability and lowest soil pH at which 80 per cent of the total number of plant species of a nature target type could be present.

Calibration equations of soil factors to Ellenberg indicators

Step 2: Ellenberg indication values were calibrated with quantitative values for the abiotic soil factors using combined samples of vegetation and environmental variables. This calibration connects SMART2 with Ellenberg based criteria. For this purpose a database has been compiled with combined samples for pH (N = 2759) and N availability (N = 266). For the pH a satisfying relations with Ellenberg values was found by using a nonlinear model (Alkemade et al., 1996; Ertsen et al., 1998; Wamelink et al., 2002):

$$\text{pH}(\text{H}_2\text{O}) = 0.424 + \frac{39.38}{12.901 - E_R} \quad R^2 = 0.54 \quad N = 2759 \quad (1)$$

The relationship between N availability and the Ellenberg indicator for nitrogen, E_N , was derived in an indirect way (Alkemade et al., 1996; Ertsen et al., 1998). Based on direct measurements such as C/N and N concentration in soil solution no

satisfactory relation could be found. However, using data at 256 plots a fairly good relation ($R^2 = 0.54$) was found for standing crop (Y):

$$Y = -7.62 + 3.04 \cdot E_N \quad (2)$$

Where Y is standing crop in $\text{ton} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$. This relation was combined with a relation between standing crop and nitrogen production, which was used as a proxy for N availability (Van Hinsberg & Kros, 1999):

$$Y_N = 0.87 + 11.1 \cdot Y \quad (3)$$

Where Y_N is nitrogen production in $\text{kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$. Combining equation (2) and (3), while assuming that N availability equals the N production Y_N gives:

$$N_{av} = -84.1 + 33.9 \cdot E_N \quad (4)$$

It must be noted that the relation for yield is mainly based on (partly) fertilized grassland data. Most of the data are in the range of E_N between 5 and 7, which corresponds with a N production between 8 and 20 ton standing crop (Ertsen et al., 1998). The relation between yield and nitrogen availability, however, is derived from non fertilized grassland (Van Hinsberg & Kros, 1999). So far no validation of the use of N availability based on nitrogen production has been performed. Especially the validation for other systems than grassland, which were used for the above mentioned relation. Validation of critical limits in terms of nitrogen availability is, however, cumbersome due to the lack of sufficient and adequate field measurements (Ertsen et al., 1998; Wamelink et al., 1998).

A reasonable relationship ($R^2 = 54\%$) between mean ground water level in spring, used in NTM and MOVE, and the Ellenberg indicator for soil moisture, F, was derived by a linear regression of measurements of both parameters at 193 plots (Alkemade et al., 1996; Ertsen et al., 1998):

$$\text{MGW} = 221.2 - 26.3 \cdot F \quad (5)$$

where:

MGW = the mean ground water level in spring in cm minus the surface level.

Improvement of the above mentioned relationships is crucial. In an analysis of error propagation in Dutch soil and vegetation model chains, the residual variation in the calibration equations given above contributed the greatest uncertainty to predictions of change in species composition (Schouwenberg et al., 2000; Van Dobben et al., 2004). Furthermore, small changes in critical soil pH for vegetation types, already can lead to huge differences in critical load values (Wamelink & van Dobben, 2003). Wamelink et al. (2002) showed that the variation explained in calibration equations could be improved dramatically if equations were developed separately for each vegetation type. The reasons why 'global' cross-community calibrations should

perform less well relates to the fact that additional explanatory variables that are correlated with vegetation type are not included in each calibration (Wamelink et al., 2002; Smart & Scott, 2004).

The SUMO model to simulate biomass growth

SUMO (SUccession MOdel) is a process based model that simulates biomass growth under given soil, climate and management conditions (Wamelink et al., 2005). The basis of the model is a linear growth equation consisting of a series of reduction factors that constrains maximum growth. These factors convey the effect of changes in the availability of light, nitrogen, phosphorous and water. Biomass growth is also a function of temperature and management.

The processes that are modelled are competition for light and nutrients by five functional types of plant (climax trees, pioneer trees, shrubs, dwarf shrubs and herbs). The competitive balance between functional types is governed by canopy height and biomass of roots and leaves, which in turn reflect management and initial abiotic conditions. However these conditions change during yearly time steps as a result of the growth and death of functional types or by interventions in the form of changing pollutant deposition, climate or management. Soil dynamics are modelled by SMART2. SUMO2 is an integrated part of SMART2; the models exchange information about N and P, litter and vegetation type on a yearly basis.

SUMO model incorporates the effects of management on vegetation growth and successional stage because impacts such as grazing or mowing remove nutrients and also allow more light to be available for the growth of shorter types of plants such as herbs or dwarf shrubs. SUMO is not formally coupled to plant species niche models, but it can be used in interaction with SMART2 to estimate biomass and hence litterfall, being a crucial input for the latter model. Both models are dynamic because they mimic processes that operate over time such as biomass accumulation, decomposition and N mineralization. Hence, predictions can be made explicitly over a 10, 20 or 100 year interval. More information on SUMO is given in Annex 2. For calculating critical loads, a reverse version of SUMO should be developed, as was done for SMART2.

The SMART2 model and adapted versions for the calculation of critical loads and target loads

SMART2: SMART2 (Kros et al., 1995; Kros, 2002) is a process-oriented soil model used for calculating the effects of deposition of NO_x, NH_y, SO_x and base cations (BC²⁺), as well as the hydrological effects (upward seepage and groundwater level) on soil chemistry. SMART2 is a one-compartment, soil-acidification and nutrient-cycling model that includes the major hydrological and biogeochemical processes in litter and mineral soil. It is a relatively simple model for predicting long-term effects of deposition and hydrology on a regional scale. The model has a high degree of process aggregation to minimize data requirements for application on regional scale. It consists of a set of mass-balance equations, describing the soil input-output relationships, and a set of equations describing the rate-limited and equilibrium soil processes. Apart from pH, the model predicts changes in aluminium, base cation,

and nitrate and sulphate concentrations in the soil solution and solid phase. The soil-solution chemistry in SMART2 depends solely on the net element input from the atmosphere (deposition) and groundwater (seepage). Canopy interactions (foliar uptake, foliar exudation), geochemical interactions in the soil (CO_2 equilibria, weathering of carbonates, silicates and/or Al hydroxides, SO_4^{2-} sorption and cation exchange) and a complete nutrient cycle (litterfall, mineralization, root uptake, immobilization, nitrification and denitrification) for base cations and N are included in the model. Litterfall and uptake by vegetation are provided by SUMO. There is feedback between SMART2 and SUMO; information exchange takes place in each time step (one year). In addition, SMART2 has also an internal simplified growth module which enables the model to calculate nutrient cycling detached from SUMO. This detached version of SMART2 was used for the calculation of critical loads and target loads (see below).

Soil interactions are described either by simple rate-limited (zero-order) reactions (silicate weathering) or by equilibrium reactions (carbonate and Al-hydroxide weathering and cation exchange). Influence of such environmental factors as pH and moisture content is included for mineralization, nitrification and denitrification. Solute transport is described by assuming that the element input is completely mixed within one homogeneous soil compartment of constant density and fixed depth. The time step of the model is one year, so seasonal variations are not considered.

SMART2 steady state for the calculation of critical loads: Until last year, critical loads with SMART2 were derived from regression equations based on dynamic SMART2 runs with constant deposition. Recently, however, a true steady state version of SMART2 was derived, able to compute critical loads. Apart from a critical load function that defines pairs of N and S deposition that lead to the desired pH, it also computes the critical N load that in steady state leads to the desired N-availability. Details are given in Reinds et al. (2005). The steady state solution for N availability is relatively simple and the critical load depends on:

- Litterfall rate
- N content in litterfall
- Ratio of above ground and below ground biomass
- Mineralization constant
- Fraction of roots in the litter layer
- Reallocation of N

The steady state solution for pH is complex and involves numerous cases. For nitrification fractions smaller than one, the critical load function depends e.g. on the ratio between NO_x and NH_3 in the deposition and can be considered a critical load surface in the 3-dimensional (NO_xdep , NH_3dep , Sdep) space. By assuming a constant NO_x to NH_3 ratio in deposition equal to the ratio in the reference year, the critical load function is reduced to the usual 2 dimensional function (see also Annex 1).

SMART2 for the calculation of target loads: For the computation of target loads, a procedure was developed to iteratively run SMART2 until the N and S deposition used, lead to the desired pH and N availability. This procedure strongly resembles

the one used to compute target loads with VSD (Posch & Reinds, 2005). In detail the procedure is as follows. For sites with an exceedance of the critical load, SMART2 is run with this critical load. If the pH and/or N-availability criterium is still violated in the reference year, a target load computation is needed. If so, SMART2 is run with (almost) zero S and N deposition. If both criteria are met in the target year, a valid target load exists, if not, the target cannot be reached (not even with zero deposition). If a valid target load exists, SMART2 is run first with zero S deposition and varying N deposition to establish the N deposition at which both the pH and the N availability criteria are met. Then, the model is run for 8 steps in N deposition which is linearly reduced in each step (to arrive at zero N deposition in the 8th step). For each step, the S deposition is varied until the combination of the (fixed) N and S deposition leads to the desired pH. Since N deposition used in these steps is lower than the N deposition at which N availability is obtained, the N availability criterium is automatically met, and only the pH criteria needs to be explicitly fulfilled in this part of the target load function.

Assessment of critical loads and target loads with the SMART2-MOVE model for the Netherlands in comparison to empirical critical N loads

The calculation of critical loads using the SMART2/MOVE models consists of two steps: (i) deriving critical limits for the different groups of species and describing the species composition of low-impact ecosystems and (ii) using SMART2 steady state, to calculate the critical loads at which the above critical limits were not exceeded. The critical limits are based on plant-species specific information on habitat preferences for nitrogen availability and soil pH, which are available in MOVE (see above). The MOVE regression functions are used to determine the range of suitable environments for the different nature target types, distinguished in the nature policy in the Netherlands (Bal et al., 2001). The critical limits are calculated as the highest nitrogen availability and lowest soil pH at which 80 per cent of the total number of plant species of a nature target type could be present. For the calculation of critical loads from these critical limits, SMART2 steady state was used for the relationship between deposition level and abiotic site conditions. For the derivation of target loads the dynamic version of SMART2 was used.

Compared to the standard parameterization of SMART2 for the Netherlands (Kros, 2002) litterfall fluxes and the N content therein were modified. In the original parameterization, the N input by litterfall does not depend on the nature target, and is e.g. equal for all grasslands. For computing nature target specific critical loads, this is not satisfactory because different types of grassland produce different N litterfall fluxes. For example, the N litterfall flux in a poor dune grassland will be much lower than in a species rich grassland located on moist clayey soil. N litterfall fluxes per nature target were based on expert judgement and literature data in conjunction with simulated N contents and litterfall fluxes from the SUMO model, by running this model for a 100 year time period with a low N deposition.

The empirical critical loads and the calculated critical loads correspond reasonably well. Except for dry and neutral grasslands (E1.7), the modelled critical load is always within the range of the empirical values. For dry grassland a lower value was

calculated. Modelled critical loads often vary within a single EUNIS-ecosystem (Table 7), but the variation was generally less than the variation in the empirical values. Within the grassland ecosystem EUNIS classes no variation was found. The nature target types that were assigned to those classes having the same limits for pH and N availability. In addition all other crucial model parameters such as litterfall rate and mineralization were also equal. Consequently, identical critical loads were calculated per cell. The widest ranges were found for forest (G) and dry heathlands (F4.2). In general it can be stated that the SMART2-MOVE methodology yields plausible results for the Netherlands.

Table 7 Empirical (Bobbink et al., 2003) and average with SMART2 steady state modelled critical N loads and target N loads for 2030 and 2100 (in kg.ha⁻¹.yr⁻¹) for EUNIS classes.

EUNIS Class	Empirical CL	Modelled CL ¹⁾	Modelled target load (2030)	Modelled target load (2100)
Forest (G)	10-20	16.8 (12.9 - 18.2)	8.4 (7.4 - 16.8)	14.0 (13.0 - 16.8)
Raised bogs (D1)	5-10	6.1 (6.1 - 6.1)	4.5 (3.8 - 6.1)	5.7 (5.0 - 6.1)
Salt marsh (A2.64/65) ²⁾	30-40	30.0 (30.0 - 34.1)	33.7 (29.9 - 33.9)	34.1 (34.0 - 34.1)
Dry and neutral grasslands (E1.7) ²⁾	10-20	8.0 (8.0 - 8.0)	1.4 (0.2 - 3.1)	7.9 (4.4 - 10.9)
Semi-dry calcareous grasslands (E1.26) ³⁾	15-25	12.4 (12.4 - 12.4)	-	-
Moist and wet oligotrophic grasslands (E3.5)	10-20	12.6 (12.6 - 12.6)	1.4 (0.5 - 6.7)	1.2 (0.4 - 12.6)
Coastal dune heaths (B1.5) ⁴⁾	10-20	15.5 (14.4 - 15.5)	3.3 (3.1 - 5.0)	12.9 (12.6 - 12.9)
Dry heaths (F4.2)	10-20	11.2 (9.4 - 17.1)	19.8 (17.0 - 21.7)	19.8 (18.5 - 21.7)

¹⁾ Values in bracket refer to the 5 and 95 percentile

²⁾ Consists of a few nature types only with similar requirements regarding N status, leading to very similar values for the various percentiles.

³⁾ Consists of one nature type only, so all critical nutrient N load computations yield equal results

⁴⁾ Consists of a few receptors only, leading to strongly skewed distribution

Figure 20 shows the cumulative frequency distributions of target loads for nitrogen for all Dutch forests, based on SMART2 calculations with the new limits for N availability and pH in view of plant species diversity impacts. Results are presented for the target years 2030, 2050, 2100 and at steady state (critical loads). Similarly, results are given in Table 8 divided in coniferous and deciduous forests. Both Figure 20 and Table 8 show that the target loads are moving into the direction of the critical loads as the target year is increasing.

Table 8 Median values of target critical loads for N preventing vegetation changes for the year 2030, 2050, 2100 and at steady state (critical load).

Forest type	Target N loads (kg N.ha ⁻¹ .yr ⁻¹)			
	2030	2050	2100	Steady state
Coniferous	8.3	10.7	13.9	20
Deciduous	15.6	15.5	15.7	20
All	8.4	10.8	14.0	20

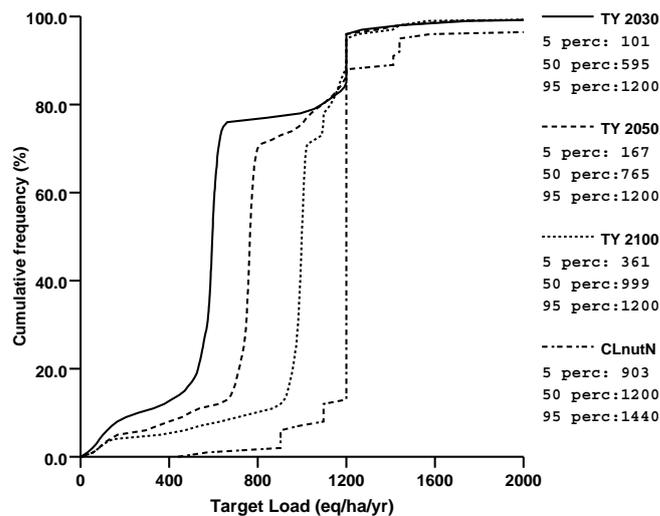


Figure 20 Cumulative frequency distribution of target loads for N preventing vegetation changes for the years 2030, 2050 and 2100 and the critical load

Evaluation of the SMART2-NTM/MOVE approach

The strengths of the SMART2-NTM approach are:

- There is a field basis between species composition of the vegetation and Ellenberg indicators for soil moisture, nutrient availability and soil acidity. The relationships are based on species-response curves of about 900 plant species (Wiertz et al., 1992). Regression was based on an extensive database consisting of 100 000 vegetation relevés (Schaminée et al., 1989). By using this extensive database of vegetation relevés, it was possible to calculate significant regression functions for many different higher plant species and critical limits for a large set of different plant groups (e.g. nature targets).
- In contrast to previously presented methods to derive Ellenberg based critical loads for the Netherlands (Van Hinsberg et al., 2003; Van Dobben et al., 2004) we used a steady state version of the dynamic nutrient cycling model SMART2. This procedure has actually several advantages compared to the previous studies: (i) The method is fully compatible with SMB in that it calculates critical N loads at steady-state. When using the same criteria, SMART2 steady state yields the same critical N load as the SMB, (ii) The newly calculated critical N loads are related to a steady state, whereas the previous ones are biased by the assumed time horizon. Van Hinsberg and Kros (1999) used 10 years whereas Van Dobben et al. (2004) used 30 years, (iii) For almost every ecosystem critical N loads could be calculated whereas the previous studies failed to find critical N loads for 30-50% of systems.

The weaknesses of the SMART2-NTM/MOVE approach are:

- The relationship between Ellenberg indicators for nitrogen availability and acidity and field data for these abiotic variables is weak. Due to the absence of measured abiotic variables in most of the relevés, it was only possible to use indirect estimates of the abiotic conditions in terms of the mean Ellenberg

indicator values of plants within the relevés. An additional data set of vegetation relevés, in which abiotic conditions were also measured, was needed to link the averaged indicator values to abiotic conditions. With the help of this second data set we could significantly correlate the respective estimates for moisture, acidity and nutrient availability with the water level in the spring, and with the soil pH and nitrogen availability (Ertsen et al., 1998). However, for nitrogen availability, the relationship is weak. The relation between F_N and N availability has been made on the basis of productivity. Furthermore, this extra step introduces additional errors in the calculation of critical limits and critical loads (Schouwenberg et al., 2000; Wamelink et al., 2002; Wamelink & van Dobben, 2003). Ideally the use of Ellenberg indicator values should be avoided and response curves should be estimated on field measurements itself, e.g. soil pH (Wamelink et al., 2005).

- The applicability of the model in other countries depends on the availability of critical criteria (Ellenberg values) and the translation of Ellenberg values to physical values. It is not likely that the relations derived for the Netherlands are generally valid for other countries. Characteristic species that do not occur in the Netherlands will be missing and also the relation between site factors and species diversity might be different. In addition other site factors such as C/N ratio might be relevant. Therefore, for use in other countries it necessary to analyse vegetation relevés in order to assign critical site factors to ecosystems.
- Output is the potential vegetation on a site, the observed vegetation will differ due to time lag effects.
- For the second step, the derivation of the corresponding critical load, the model SMART2 may be used. However, this model must be parameterized with country specific data amount which the most critical are litterfall rate, mineralization rate constant and nitrogen content in the foliage. These data may be limited. In this context, a link with the SUMO model is possible (see Annex 2), but this model is presently only parameterized for the Netherlands and the UK.
- For grassland N removal causes problems. This causes an uptake being higher than the and the derivation of a critical N concentration is thus not possible. This effect is mainly caused by the implicitly introduced management, i.e. mowing followed by removal. This resulted in a relatively large uptake and a large turnover of biomass, in turn resulting in a large N availability. For mown grasslands, being common practice in Europe, there is thus no link between dissolved N concentration and species composition at all. The effects of management practices such as mowing can be dealt with in the model SUMO, but in this case a reverse version of SUMO is needed.

3.3 The MAGIC-SUMO-GBMOVE model

General approach

The components of the integrated soil vegetation model used in the UK are: (i) the soil model (MAGIC) that mimics the cycling of nutrients in the soil, (ii) the succession model (SUMO) that takes nutrients (N and P) out of the soil as plants grow but returns nutrients as plants die and (iii) GBMOVE that predicts changes in the favourability of abiotic conditions for individual species by a series of multiple

regression equations that define the realized niche of each plant species. GBMOVE is based on the principles of the Dutch MOVE model.

Changes in species composition in a particular place are modelled by firstly simulating the effect of N and S deposition on the soil by the MAGIC soil model. This model produces estimates of soil pH and C/N ratio for each yearly time step. Soil pH and soil C/N ratio are then translated into mean Ellenberg R and N values respectively using calibration equations (Smart et al., 2003). The mean Ellenberg R and N values are terms in the GBMOVE regression equations. Hence, for given values of soil C/N, soil pH, % soil moisture and cover-weighted canopy height, each equation for each relevant species is solved at each time step resulting in a changing predicted probability of species occurrence as time passes. The initial set of GBMOVE equations were generated using extensive vegetation survey data representing the range of plant communities found in Great Britain. Regression coefficients were derived for a range of environmental gradients, particular parts of which will favour different species.

The Dutch SUMO model is also part of the model chain. However, as with the model chain SMART2-SUMO-NTM/MOVE, SUMO does not have to be coupled to the GBMOVE plant species niche models. However, it can be used in combination with MAGIC to predict canopy height, which is then used in GBMOVE providing values for the cover-weighted mean canopy height term as an indicator for succession and disturbance. A number of modifications to SUMO were implemented and tested to make the model more applicable to British ecosystems (Smart et al., 2005b) but a complete coupling of SUMO with GBMOVE and MAGIC awaits further development and testing. The MAGIC-SUMO-GBMOVE model chain further includes a series of additional empirical models and filters to take account of further influences on vegetation species composition or to increase the relevance of the modelling approach to established conservation policy targets for rare species and indicator variables. These additional model components include methods to: (i) estimate the immigration potential of species known to be present in the local species pool, based on dispersal traits plus national survey data, and (ii) predict changes in probability of occurrence of rare and subordinate species (Smart et al., 2005b). Again, an integrated architecture combining these filters with the model chain is under further development (see Smart et al., 2005b).

The GBMOVE model to assess relationships between abiotic conditions and habitat suitability for plant species

Derivation of Ellenberg's indicator values from species composition

As with MOVE, multiple logistic regression was used to construct empirical equations that predict habitat suitability for higher and lower plants representative of British plant communities, based on their abundance along key environmental gradients as recorded by extensive botanical quadrat data (e.g. Roy et al., 2000). Each equation consists of regression coefficients that apply to either four or seven explanatory variables, depending on whether climate variables (Minimum January temperature, maximum July temperature and precipitation) are included or not (Table 9). Changes in soil pH and C/N ratio are predicted with MAGIC. Canopy

height can be changed arbitrarily using pre-existing knowledge of the pace of succession in a particular location, or on a more process-linked basis, by the SUMO succession model. Climate variables can be changed to mimic expectations under different climate change scenarios. Likewise, soil moisture can also be changed to mimic drainage or drought.

Table 9 Explanatory variables used in multiple logistic regression equations to define each species realised niche.

Drivers of change to which explanatory variables are responsive	Explanatory variable	Linked by calibration equation to measured...
Atmospheric N deposition, NPK originating directly or indirectly from agriculture	Mean unweighted Ellenberg fertility	Soil C/N ratio
SOx deposition, liming	Mean unweighted Ellenberg pH	Soil pH
Drainage, drought, flooding	Mean unweighted Ellenberg wetness	% soil moisture
Succession and disturbance	Cover-weighted mean canopy height	not applicable
Climate change	Minimum January temperature	not applicable
	Maximum July temperature	not applicable
	Precipitation	not applicable

The data used to derive each equation were assembled from a variety of sources so as to maximise the number of plant species covered (Table 10). 1217 quadrats had no grid reference and so were omitted from GBMOVE models that included climate variables.

Table 10 Datasets and sample numbers used to build GBMOVE models for British higher and lower plants.

Datasets	Number of quadrats
Key Habitats 1992	548
Countryside Survey 1998	7221
Broadleaved woods 1971	1648
National Vegetation Classification (various years)	31266

Each logistic regression was then based on presence/absence data for each plant species in each plot paired with values of each of the explanatory variables, calculated during model processing with the target species omitted. Variable selection was carried out by first testing the explanatory power of each variable separately and then entering those that were significant into a stepwise procedure. Both linear, quadratic and two-way interaction terms were tested. The result is an equation that produces a probability of the plant species being present under different sets of conditions specified by the values of the explanatory variables. The resulting GBMOVE models constitute an empirical, statistical description of the realized niche of each species. The final number of higher and lower plants having models is shown below (Table 11).

Table 11 Number of species having GBMOVE regression models. The count is based on models with no climate variables. Figures in brackets indicate the number of species that have models but for which no optima and hence no maximum occurrence probability could be calculated

	Bryophytes	Higher plants	Lichens
Coastal		75 (13)	
Non-coastal	233 (72)	971 (182)	74 (28)

Calibration equations of soil factors to Ellenberg indicators

A critical component in the model chain are the three calibration equations used to convert soil C/N, pH H₂O and % soil moisture into mean unweighted Ellenberg fertility, acidity and wetness values. Calibration equations were constructed to enable soil C/N ratio, soil pH H₂O and % soil moisture to be estimated for quadrats in which no soil measurements are recorded. Equations were constructed that used mean Ellenberg scores to explain soil measurements. For this step, only the quadrats with soil measurements could be used. GBMOVE regression models were constructed using all available quadrat data but with mean unweighted Ellenberg values as explanatory variables plus climate variables and mean cover-weighted canopy height. The calibration equations were then used to translate soil C/N and soil pH H₂O estimates from MAGIC into values of explanatory variables to solve each GBMOVE equation. Calibration equations were all constructed using paired soil measurements and mean Ellenberg values from the Countryside Survey 1998 database (Smart et al., 2003). Results are presented in Equation 6-8 and Figures 21-23.

$$\ln(\text{C/N ratio}) = 3.61 - 0.63 \ln E_N \text{ (mean Ellenberg fertility).} \quad R^2 = 62\% \text{ (6)}$$

$$\text{Soil pH} = 2.5 + 0.61 E_R \text{ (mean Ellenberg acidity)} \quad R^2 = 61\% \text{ (7)}$$

$$\ln(\text{M\%/100-M\%}) = -3.27 + 0.55 E_X \text{ (mean Ellenberg wetness)} \quad R^2 = 72\% \text{ (8)}$$

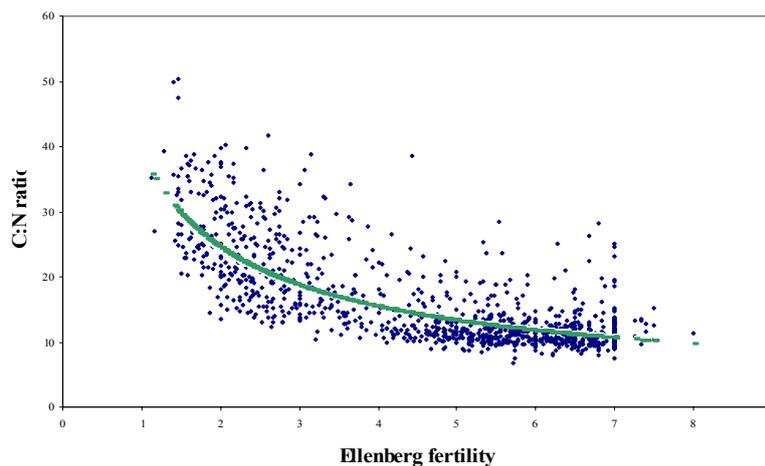


Figure 21 Relationship between soil C/N ratio and mean Ellenberg fertility ($\ln \text{C/N ratio} = 3.61 - 0.63 \ln E_N$; $R^2 = 62\%$).

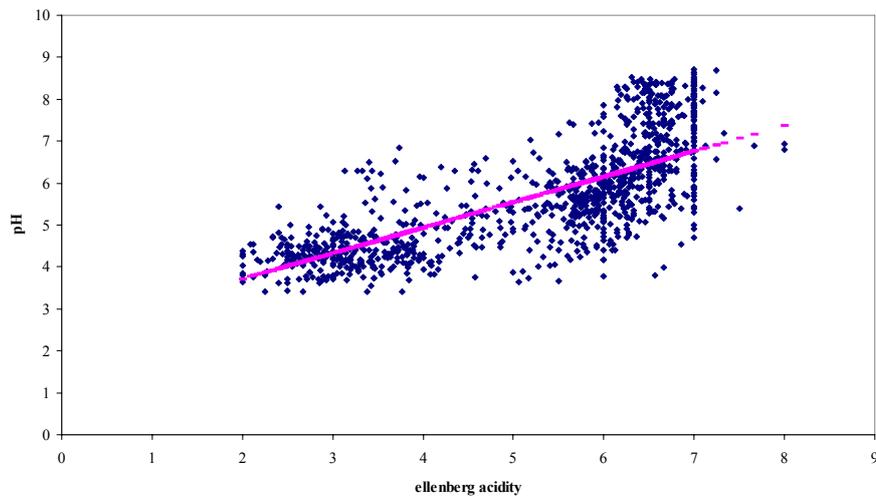


Figure 22 Relationship between soil pH H₂O and mean Ellenberg acidity (soil pH = 2.5 + 0.61. E_R; R²= 61%,)

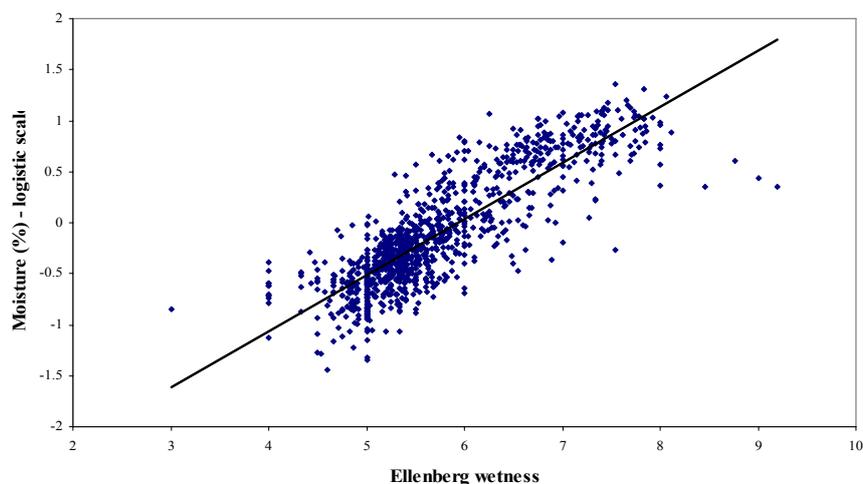


Figure 23 Relationship between moisture content and mean Ellenberg wetness ($\ln(M\%/100-M\%) = -3.27 + 0.55. E_{ss}; R^2 = 72\%$)

The calibration equations developed for British soils and plant communities have higher R² values than their Dutch equivalents. As presented before, Ertsen et al (1998) found an R² of 54% for Ellenberg fertility and standing crop (62% for British calibration between soil C/N and mean Ellenberg fertility), 54% for soil pH (61% for British calibration) and 51% for soil moisture (72% for British calibration). However, explained variation is only moderate and the low predictive power obtained has a critical impact on the accuracy of MAGIC-GBMOVE predictions. In particular, the exponential form of the British calibration between mean Ellenberg N

and soil C/N resulted in very large increases in mean Ellenberg value with only very small reductions in soil C/N. An obvious conclusion is that soil C/N on its own is a poor predictor of mean Ellenberg N and this lack of explanatory power is worst toward the lower end of the soil C/N range, which includes neutral grasslands. Hill & Carey (1997) concluded that Ellenberg N values are better treated as overall indices of fertility rather than explicitly N availability, hence much higher explanatory power was achieved in predicting annual biomass than soil properties.

The predictive power of the GBMOVE chain has been tested against species presence data in an independent sample of 244 monitoring quadrats located in English Environmentally Sensitive Areas (Critchley et al., 2002). Predicted occurrence probabilities were generated for 200 plant species using GBMOVE either with mean Ellenberg values from the sample plots as model inputs or using observed soil data for pH and C/N ratio (Figure 24). The match between predictions and observations was tested by a linear logistic regression of predicted probabilities onto observed presence. Hence, significant positive outcomes indicate a good match. The results (Figure 24) highlight the uncertainty contributed by the conversion of observed soil pH and C/N into mean Ellenberg values. When only mean Ellenberg values are used, which derive from the total species composition of the plot minus the target species, 88% of species had significant positive associations whilst when soil data were used to solve GBMOVE niche models, only 26% of species probabilities were positively related to observed presence (Figure 24). The results validate the GBMOVE niche descriptions but highlight the weakness of the calibrated links between vegetation composition and soil measurements.

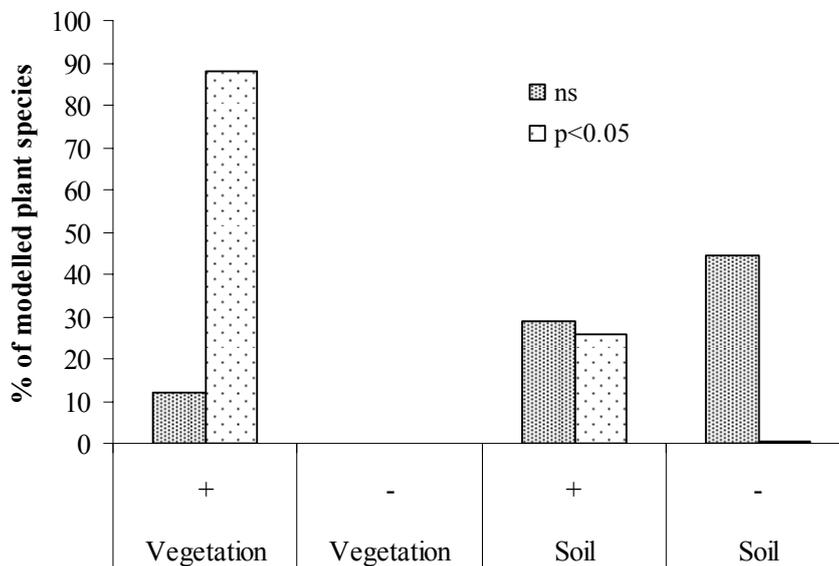


Figure 24 Comparison of predicted species' occurrence probabilities using GBMOVE with observed presences in 244 nature reserve monitoring plots. Predictions were generated using either observed mean Ellenberg values (vegetation) or soil C/N and pH (soil) as model input. High percentages of positive (+) relationships indicate overall good matches between predictions and observations.

The MAGIC model for the calculation of soil changes and critical loads

The MAGIC model (Model of Acidification of Groundwater In Catchments) was first developed during the 1980s (Cosby et al., 1985), and has been revised and updated on several occasions since, most recently to include a more detailed representation of N dynamics (Cosby et al., 2001). MAGIC simulates changes in soil, soil solution and groundwater chemistry resulting from acid and N deposition and land use. The model simulates transfers between several different soil, wetland and stream compartments. In connection with GBMOVE, MAGIC has so far been used in its simplest form, with one soil and one soil solution compartment. Soil properties are averaged over the soil column, and so the model can be applied to a soil, catchment or region with a small set of input data.

MAGIC consists of a set of equations describing equilibrium soil processes, a set of mass balance equations describing input-output relationships for base cations and strong acid anions in precipitation and stream water, and a set of definitions relating the variables in the equilibrium equations to the variables in the mass-balance equations. Key parameters include the input and output fluxes of base cations and strong acid anions, the soil cation exchange capacity, and the fraction of this capacity that is occupied by Ca, Mg, Na and K ions. Nitrogen dynamics are based on empirical relationships between net N retention and the current C/N ratio in the soil. Plant uptake, and other sinks and sources, can be included where necessary.

The model is normally calibrated to present-day measurements of soil, surface water and/or soil water chemistry, and the accuracy of the model simulation can be tested against long-term monitoring records, where available. Calibration of the model to a site involves fitting unknown terms, such as soil cation weathering rates and base cation selectivity coefficients, so that they are consistent with the measured soil and soil solution chemistry. The stream concentrations of SO₄ and Cl ions are calibrated first, normally (for UK soils) assuming that transport through the soil is conservative and so output fluxes are equal to input fluxes. The relationship between soil C/N and net retention is calibrated to match observed soil N and soil solution NO₃ and NH₄ concentrations. Finally, the base cation concentrations are calibrated using an optimisation procedure. The calibrations are performed on simulations run from the pre-industrial period, based on historical deposition sequences.

Assessment of critical loads with the MAGIC-GBMOVE model for the UK in comparison to empirical critical N loads

To date, the MAGIC-GBMOVE model chain has not been applied in 'inverse mode' to estimate critical loads based on biodiversity targets. However the MAGIC model has now been adapted to incorporate a target loads facility, whereby the model can be used to determine the combinations of S and N deposition required to meet specified chemical targets by a given date (or, in principle, at steady state, i.e. critical loads). This facility has been used to define target loads for acidity, but the recent inclusion of the key GBMOVE soil abiotic variables (pH and C/N) provide the capability to define critical loads or target loads on the basis of selected biodiversity criteria. Future work will provide an assessment of model-based critical loads for

biodiversity with existing empirical N critical loads, comparable to the study undertaken in the Netherlands.

Evaluation of the modelling approach MAGIC-GBMOVE

A description of the validation status of the MAGIC-GBMOVE model chain is given in Annex 3. On the basis of these results and more fundamental issues regarding the modelling approach, the strengths and weaknesses of the MAGIC-GBMOVE approach can be summarized as presented below.

The strengths of the MAGIC-GBMOVE approach are:

- As with MOVE and NTM, there is a field basis between species composition of the vegetation and Ellenberg indicators for soil moisture, nutrient availability and soil acidity. The relationships are based on large quadrat datasets covering the majority of plant community types and hence, abiotic and climatic gradients in GB. Thus the strength of the resulting empirical niche models is that the weight of data reduces noise relative to species-environment relationships.
- A large number of higher and lower plant species are covered by GBMOVE. This means that outputs can be related to species action plans while individualistic models allow for flexible prediction of new species assemblages given novel configurations of environmental conditions.
- The approach is compatible with existing critical loads calculated using the SMB, provided the same input data are applied.
- GBMOVE incorporates climate and management-related variables, as well as N and acidity variables, based on the same survey datasets. It therefore has the capacity to consider the integrated biodiversity impacts of multiple drivers, and the impact that climate or management change might have in modifying critical loads.
- MAGIC is a flexible, validated and widely used biogeochemical model. The calibration routine ensures that current conditions are correctly simulated, increasing the probability of predicting the correct current species assemblage.
- MAGIC is a dynamic, process-based model and in combination with the empirical response functions in GBMOVE, employs the best possible combination of both approaches.

The weaknesses of the MAGIC-GBMOVE approach are:

- As with MOVE and NTM, the relationship between Ellenberg indicators for nitrogen availability and acidity and field data for these abiotic variables is rather weak. The relationship between the abiotic measurement of nitrogen enrichment (soil C/N) and Ellenberg N is poorest in high-fertility ecosystems.
- While calibration equations solve an important problem, they contribute uncertainty related to the fact that soil pH, soil C/N and soil moisture do not explain total variation in mean Ellenberg scores. The greater the scatter about each regression line the more likely it is that predictions of mean Ellenberg values from soil measurements will vary from actual observations. Moreover, as stated for the SMART2-MOVE approach, the uncertainty in using Ellenberg indicator values can be large and may influence the end result tremendously (Schouwenberg et al., 2000; Wamelink et al., 2002).

- The GBMOVE models are based on empirical observations recorded at different times in the past 70 or so years across British ecosystems. The resulting regression models assume equilibrium between species and environment. Although there are no obvious solutions to this problem, the niche of each species is static. Hence, the values of each explanatory variable can change but the values of the regression coefficients cannot change.

3.4 The BERN model

General approach

The BERN model developed and used in Germany consists of: (i) direct relationships between species diversity and the abiotic factors base saturation, C/N ratio, soil moisture and climatic parameters and of (ii) an open database interface for linking BERN with geochemical models. The inherent problem of using the indicator values of Ellenberg, requiring that these values are scaled to measured abiotic values, is avoided in the BERN model by using the needed abiotic indicators, lying within the range that compares to the common (overlapping) range of the fundamental niches of all species occurring at this site.

The BERN model can be used as an “Add-On” for dynamic models like VSD and SAFE (tested) or SMART2 and P-NetDNDC (not tested yet) for the integrated assessment of impacts on biodiversity caused by acidification and eutrophication. Furthermore, it can be used to calculate critical loads for eutrophying N and acidity, using an adapted version of the SMB model, in which N immobilisation, critical Bc/Al-ratio and critical pH respectively is based on BERN model results (see also Annex 4).

The BERN model to assess relationships between species diversity and abiotic factors

Background of the model

The Bern model is based on a direct relationship between plant species composition and base saturation, C/N ratio and other relevant site factors. The C/N ratio in the top soil of forests and pasture sites serves as a parameter to indicate changes of the accumulated nitrogen content in the humus over a long time. If nitrogen deposition lies constantly below the critical load (i. e. the deposition an ecosystem can tolerate in the long run), the C/N ratio only changes slowly within a soil dependent typical range. The outer thresholds of this range are explained under two “Points of no return” (C/Nmax, C/Nmin). “No return in this context means that at exceedance of this point results in not reversible changes of the ecosystem. The variation within a soil type depends on climate conditions and on influences of vegetation. It should be regarded as acceptable if the plant species composition equals a type of “natural” vegetation.

In the BERN model the suggestion of Kopp (2003) has been adapted when preparing the forest site mapping in Germany: The starting point of investigations of the anthropogenic change of the current state should be the natural steady state balance between site and vegetation as a starting point. The state variables which are

anthropogenically difficult to influence (“basic state variables with slow dynamics”) have been assigned to the “basic state site type” classes. The vegetation which reflects the basic state variables with slow dynamics is defined as the “basic natural plant community” which is constituted by its typical assemblage of “consistently plant species”. Succession phases and gradual variations are excluded in the BERN model. Only communities in their long term stable species composition, in general not or rarely extensive used are considered, because the species balance due to competition is staying homeostatic over a long time. In these cases the indicator value of changes is mostly driven by pollutions. These basic plant communities are discrete, repeatable vegetation units (Clements, 1916) and have been described and classified comprehensively in the literature about the flora of Europe before 1960 as natural plant communities for woodland, fen and swamp, or as semi-natural communities for extensive dry grassland and heath (in the following summarized to the term “basic natural plant communities”).

The variation of the natural C/N-ratio results in a variation of the potential N-immobilization rate, and this vegetation dependent immobilization rate should be considered as an input variable for calculating critical loads too. To illustrate this statement an example can be provided as follows: At a suboceanic montane site type with eutric cambisol, shady hillside, good base supply (BS around 60 %) the natural woodland community Hordelymo-Fagetum develops. The beech leaves and broad leaves of herbs are quickly decomposable, leading to a steady-state C/N-ratio of around 18 in a mull-moder humus form. In case of a moderate base supply (BS ~ 40) (with similar climate conditions) a Luzulo-Abieto-Fagetum develops. Its fir-needles mixed with beech leaves are slower decomposable. The resulting steady-state C/N-ratio would be about 25 in a typical moder humus. However, if the cambisol has only a base saturation of about 35 %, the best adapted plant community is the Vaccinio-Abietum with predominating fire and dwarf-shrubs. The litter of this vegetation type is very slowly decomposable, therefore the steady-state C/N-ratio is about 28 in a mor humus. A development of a C/N-ratio to beyond the soil-typical range should not be accepted. A sufficient content of base cations (for the nutrition and reproduction of the decomposing soil organisms) provided, reaching the minimum (C/N_{min}), all the available organic matter would be mineralized quickly. In this case the whole net N-immobilization rate equals the temperature dependent N-immobilization rate (Posch & de Vries, 1999). First signs of eutrophication combined with changed conditions for competition for the species in a natural plant community begin to occur. Acidification predominated by sulphur inputs hampers the activity of destruents, thus leading to a widening of the C/N-ratio. The decreasing base saturation and simultaneous sinking pH-value in the topsoil results in a change of the decomposing soil organism composition. (e.g. if the pH-value falls below 4.5 the earthworms or Lumbricidae die). These are, however, mostly responsible for mixing the mineralized nutrients from humus to mineral top soil layer (=“bioturbation”). Only bristle worms (Enchytraeidae) survive in the humus layer. They can not exist in mineral soil layers and therefore mixing does not take place anymore. If a maximum value (C/N_{max}) has been exceeded in a non-hydromorphic terrestrial soil the cycling of nutrients between the humus layer and the mineral top soil layer is interrupted. Then base cations from litterfall do not arrive the mineral

topsoil (Schachtschabel et al., 1998). Only base cations from the weathering of parent material are available in the deeper soil layer rooted by plants. Particularly for some plant species, which are adapted to a good base supply, the possibility of occurrence decreases. In the worst case this means the extinction of the natural basic plant community (Konopatzky & Kirschner, 1997).

The lowest acceptable C/N-ratio and the lowest acceptable BS are determined, by definition, for each natural plant community at the point which exhibits the most remote disharmonic relationship of base saturation to C/N-ratio on the critical limit function $f(\text{BS};\text{C}/\text{N})=0.5$. A reasonable threshold value is the suitability degree of 0.5. At this value of the site variable (Critical Limit), the natural basic plant community has only a 50% of the possibility of its occurrence. At this point the option is provided either to reproduce the steady-state structure of the natural basic plant community or to change furthermore the basic structure up to the extinction of the natural basic community. The polygon that arises by drawing a line along these 0.5 points is the Critical Limit Function of the BERN model. The Critical Limit point is determined by drawing a straight line from the point of optimum of the primary-natural community to the origin of the coordinate system that presents the 0,0 point from base saturation and C/N-ratio (Figure 25). The intersection of this straight line with the Critical limit function is, hence, the extreme disharmonious condition, in which the basic natural plant community is just able to exist, meaning it exists with 50 % possibility. In the following, this point is called (CNBS(crit), BSCN(crit)) respectively.

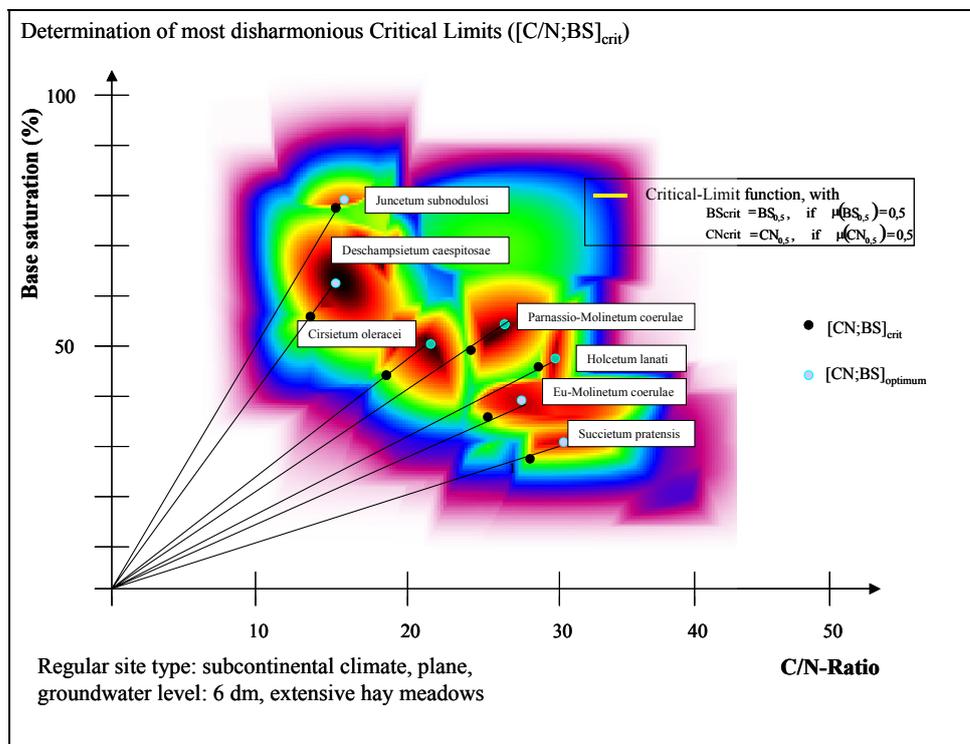


Figure 25 Determination of most disharmonious Critical Limits ($[\text{C}/\text{N};\text{BS}]_{\text{crit}}$) for natural plant communities

The BERN model does not conduct dynamic calculations of competitive relationships between plant species. Its basic approach is to include stable competitive equilibrium states found in nature as the final solution of the natural competition situation. Stable equilibria between populations are classified in plant sociology as plant communities. The ecological niche of a plant community is hereby defined by the combination of the fundamental niches of the individual species which constitute the plant community. In consequence, for the definition of the ecological niche of a community only the constant constituent species of the regarded community are included, i.e. species which stick to certain site factors.

In the result BERN considers changes of plant species composition in terms of shifts from one plant community to another. Which plant community is expected to develop depends on the trends of site variables with fast dynamics. It is not possible to definitely predict the occurrence of plant species on the basis of site variables due to not predictable competition influences (Wamelink & van Dobben, 2003). Theoretically, in order to model the realistic possibility of the occurrence of a species, the points of dynamic competition equivalents between all species at a site had to be considered using the Lotka-Volterra-equation (Shugart, 1984). There is, however, not enough knowledge about these equivalent points of the species among each other (Ellenberg, 1996). But it is possible to predict the potential possibility for a natural plant community to occur (Lortie et al., 2004) because it represents the present final solution of long term competition balance between the species (Callaway, 1995). More information on the background of the Bern model is given in Schlutow & Hübener (2004).

Natural or semi-natural plant communities have been established by adaptation to the site specific balanced steady-state, including harmonious nutrient supply (C:N:Ca+Mg+K) and other variables like water, light and temperature. They reveal stable competitive relationships. Comprehensive knowledge is available about the qualitative relation between site properties and the indicated plant communities. These native plant communities are detailed documented in the literature in Germany (Klapp, 1965; Passarge & Hofmann, 1968; Oberdorfer, 1979; Pott, 1994; Ellenberg, 1996). In order to transform the qualitative but not exact expert knowledge into exact mathematical formulas the BERN model uses the approach of fuzzy relation after Zadeh (1978) of site types to plant species based on empirical knowledge about plant physiology and plant competition. The degree of the fuzzy relation is determined by a distribution function of possibilities of plant occurrence in dependence on one or more site variables with the range between 0 and 1.

This definition in accordance to the definition of the ecological niche after Hutchinson (in: Shugart, 1984 : 185) describes the ecological niche as a n-dimensionally hyper cloud in the functional space of all site factors (Burrows, 1990; Dierschke, 1994; Begon et al., 1998; Martin, 2002). Thus the descriptions of the structure of the natural plant communities (= abundance of the constituent constant species) together with knowledge on site characteristics of the basic site type, where they occurred, can be used for bio indication of site characteristics under conditions scarcely influenced by man. The border of the ecological niche of a plant species is

described as a fuzzy constraint of the environment-plant relation. This constraint is set up for each of the considered dynamic site parameters, like base saturation, C/N, climatic indicators etc. and combined, using a von Liebig-Approach with the Minimum-operator as a fuzzy AND operator. These multidimensional fuzzy constraints or possibility distribution functions (Zadeh, 1978) describe the fundamental niche of a species. As stated before, a basic natural plant community is defined by the simultaneous occurrence of its consistent species. The possibility distribution functions of these species are combined using an AND-like operator (for a detailed description see Schlutow & Hübener, 2004). This kind of combination is shown in Figure 26 below.

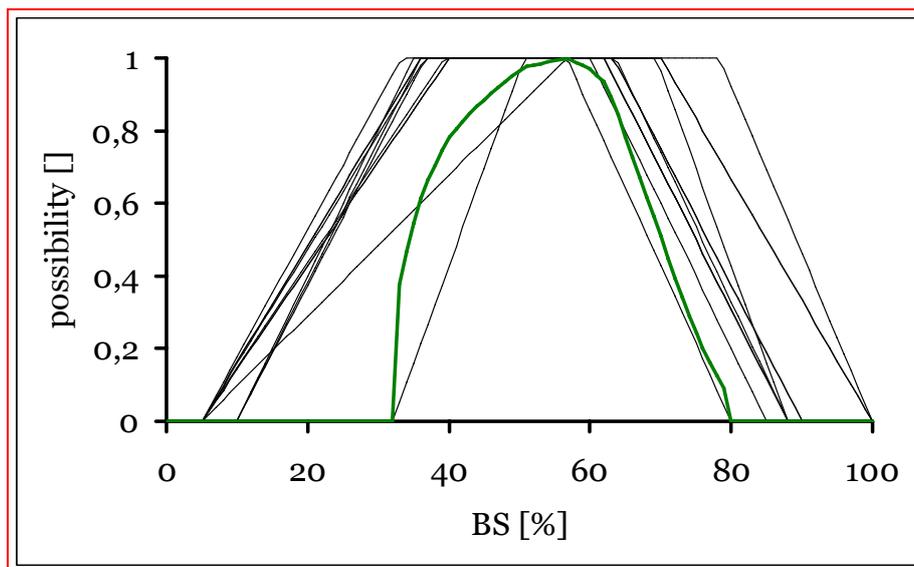


Figure 26 Possibility distribution functions of species (black lines) defining the possibility distribution function of a community (green line).

Data base used for the BERN-model

The assessment of Critical Limits of plant communities by the BERN-model is based on a data set including more or less all basic site types of Germany and their typical basic plant communities in the harmonic ecosystematic steady-state. The area under investigation includes the habitat types of the coastal, submediterranean, planar, colline, montane and alpine regions with oceanic, sub oceanic and sub continental climate conditions.

In the BERN database the natural basic plant communities are assigned to basic site types under consideration of the basic state variables with slow dynamics in geological time-frames:

- type of climatic region,
- relief type,
- exposition type,
- soil type/parent material group.

These plant communities are characterized by lists of the consistently occurring plant species. In the BERN database the consistent plant species of all regarded natural plant communities are described by their fuzzy thresholds for the basic state variables with fast dynamics in geological time-frames:

- degree of moisture,
- humus form,
- land use type,
- C/N-ratio,
- base saturation (or pH respectively)

as well as variables with fast dynamic observed in the last time due to climate change processes (instead of climate region type):

- duration of the vegetation period and
- continentality index (climatic dryness index after deMartonne).

Forests, extensively used grassland, pastures and heaths including bogs and wet heaths were examined. For the area of Germany 28 907 relevés with additional verbal information about site state variables were evaluated. All these relevés have been published before the strong industrialization period in the 1960th (Passarge, 1964; Passarge & Hofmann, 1968; Oberdorfer, 1992; Schmidt et al., 2000; Succow & Joosten, 2001; Walentowski et al., 2004).

The BERN model was validated on results from measurements and observations at 71 German Level-II-plots of the forest monitoring program. The comparison of measured and modelled data of C/N-ratio on 71 Level-II-plots of Germany shows that 100% of the predicted ranges modelled with BERN include the measured data (Figure 27). Therefore the BERN-database of the possibility ranges for soil-C/N of the species occurring in Germany could be characterised as verified.

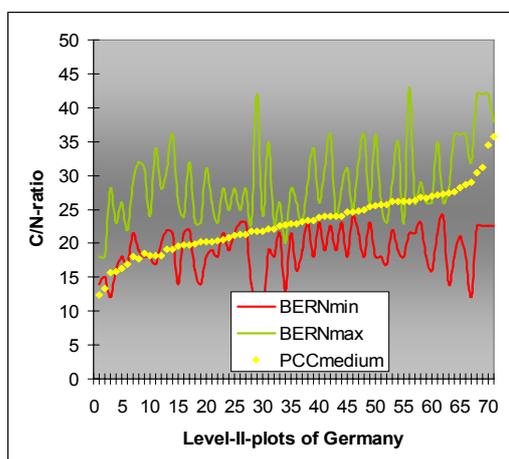


Figure 27 Comparison of measured C/N-ratio at Level-II-Plots (yellow dots) with BERN-modelled ranges of C/N-ratio in Ob/Ab-horizons

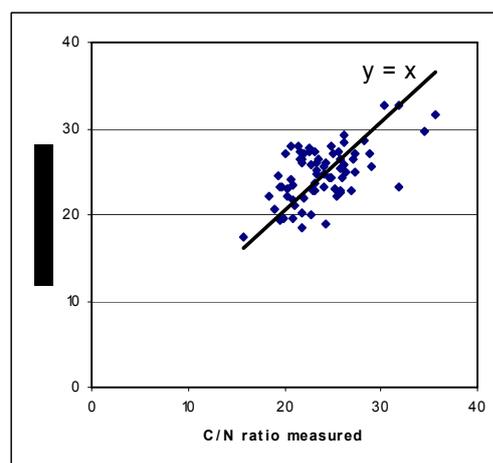


Figure 27b Measured C/N-ratio at Level-II-plots of Germany against BERN-modelled C/N-ratio (medium of ranges)

On the other hand the comparison shows a problem of validation. The measurements took place in the year 1995. According to the definition of natural plant communities and their constant species composition in the BERN model the possibility ranges of the species were obtained from records before 1960. Though a number of measured C/N lies at the lower end of the predicted ranges of the BERN modelled C/N, caused by high N-deposition in the last decades. Otherwise another number of measured C/N lies at the upper end of the BERN modelled ranges, due to high deposition of sulphur caused the decreasing pH-value in the humus layer followed by the deprivation of the destruent activity and then follows an increasing C/N-ratio.

The comparison of pH-values measured and modelled has to be done under consideration of the rooting depth of the various vegetation layers in the soil horizons. While the measurements were taken specifically in each horizon the modelled pH ranges are valid for the whole actually rooted depth. The BERN-database includes the average root length of the plant species. The main rooting zone at the Level-II-plots in German forests including ground vegetation goes to 80 cm. Figure 28 shows that nearly all maxima of measured pH-value of all rooted horizons at a Level-II-plot are lying within the predicted ranges. Only 4 (from 71) sites which were limed in the last few years before the measurements were made show higher pH-values outside the modelled ranges.

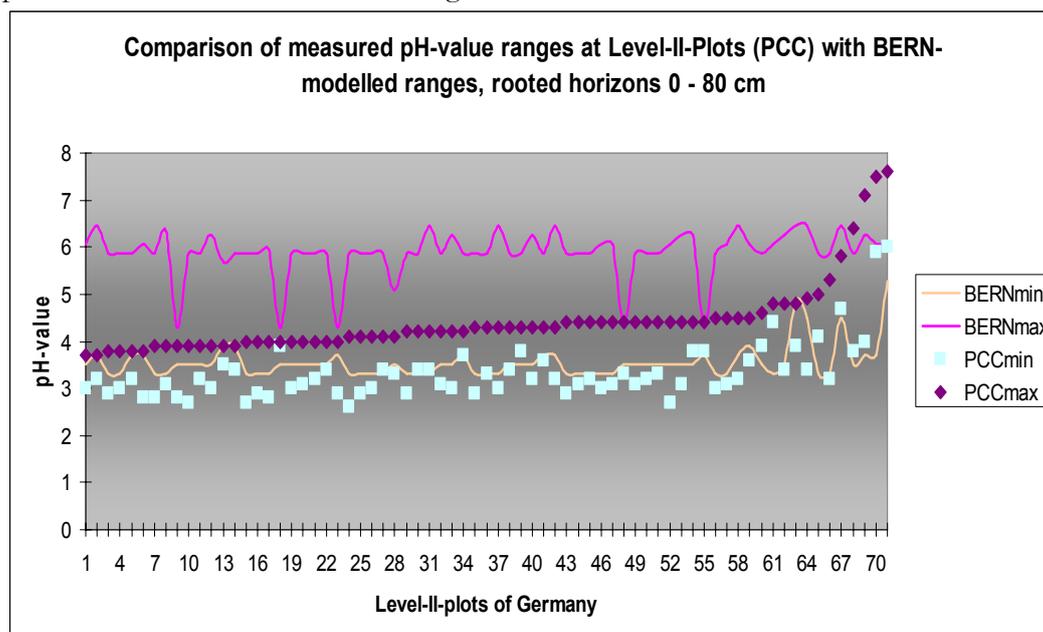


Figure 28 Comparison of measured pH-value at Level-II-Plots (blue dots= minimum and lila dots=maximum of all horizons) with BERN-modelled ranges over all actually rooted horizons (in average 80 cm depth)

The 1998 actually obtained pH minima of all rooted horizons at each Level-II-plot are mostly below the modelled ranges. The reason could be that the plants displace their main root mass into a soil horizon which has already a sufficient supply of base cations after strong acidifying deposition. But this site specific effect the BERN model does not reflect up to now.

Use of the BERN model for the calculation of critical loads with a critical limit function $f(CN,BS)_{crit}$

With the Bern model, the Critical N Load can be calculated from the site specific function $(C/N,BS)=f(N_{dep}, S_{dep})$ at steady state conditions under consideration of:

- Nitrogen availability (N deposition plus N mineralization)
- Gross N mineralization/immobilisation
- Biomass N increment.

The Critical Load is met when a long term balance between incoming deposition, immobilisation, mineralization, uptake, denitrification and leaching leads to a steady state C/N ratio in the soil that equals the critical C/N of the protected plant community. Since mineralization and immobilisation processes work in opposite direction to the other, and both are depending on the same environmental conditions in direct and indirect proportions respectively, the calculation of the long term Critical Load has to be an iterative approximation (Figure 29). This function can be derived with long simulation runs (>100 years), either with the Very Simple Dynamic model or in a better way of medium complex biogeochemical models like SMART2, ForSAFE or any suitable PnET-Family member under constant boundary conditions.

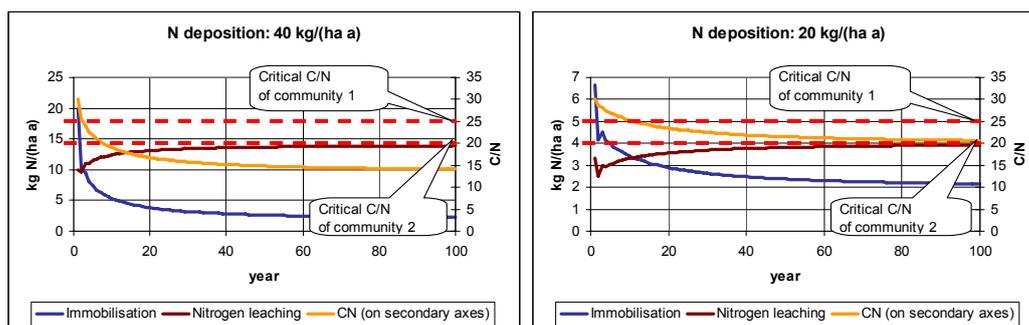


Figure 29 Conceptual N-dynamic in the long run at constant external conditions. With 40 kg N input both communities 1 and 2 get extinct; at 20 kg the steady state C/N-ratio equals the critical C/N of community 2. This means the critical load for community 1 is still exceeded and the critical load for community 2 is exactly met.

In principle the approach of the VSD model (Posch & Reinds, 2005) is particularly suitable for adding the BERN model. It has an open interface and the code is published. It can be used for regions, not only for well researched sites. The needed input data mostly are available. This acidification model has already a linkage to C/N ratio. However, the mathematical concept of this model does not include the mineralization term. For this reason the model results for C/N time series after running the VSD routines seem to be not realistic. Therefore an extension has been developed as described in Annex 4. The Bern model also calculates critical loads for acidity based on a simple mass balance equation for critical loads of acidifying depositions. In this case plant community specific threshold values for base saturation as derived by the BERN model are used. More information on the approach is given in Annex 4.

Apart from quantifying the critical limits and critical loads of N for natural and semi-natural plant communities, the database of the BERN-model can also be used to: (i) assess the current regeneration ability, (ii) determine the dynamic change of vegetation structure in the past and future depending on history and future scenarios of using, depositions and climate change based on results of ForSAFE and/or VSD and (iii) determine the future options of regeneration targets. Examples of such applications are given in Annex 5.

Evaluation of the BERN approach

The strengths of the BERN approach are:

- There is a direct link between species composition of the vegetation and nutrient availability and soil acidity. The database of BERN includes all basic site types which represent country-wide the German coast, low, hilly, montane and alpine regions. It includes results for the allocated basic natural plant communities (145 woodland communities) and semi-natural communities (140 fens, heath or natural grassland communities) with their ecological optima and possibility ranges regarding base saturation (or pH), C/N-ratio, soil moisture, continentality index and length of vegetation duration (approximated by base saturation).
- The constant plant species of these plant communities (1040 plant species are in the data base up to now) with the optima and niche widths for the preferred soil moisture, base saturation (or pH), C/N-ratio ranges, ranges for preferred climate conditions (continentality index and length of vegetation duration). Treating the niches per environmental factor as (fuzzy) constraints, Bern models the stress situation of plants, instead of combining probabilities of related factors
- The method is fully compatible with SMB in that it calculates critical N loads with this model, while adapting the N immobilization and mineralization term. It thus allows the calculation of critical loads for S+N-acidification and N-eutrophication for each (semi)natural plant community.
- The BERN model has an open interface to results of any geochemical model
- The competition between plant species is solved by known constellations in known plant communities

The weaknesses of the present BERN approach are:

- The values for C/N and BS of the historical vegetation relevés are not measured (no field basis) but estimated based on verbal characterization of soil and humus properties. The reliability of the used relations between soil map units in order to derive the C/N and BS of vegetation relevée is however validated but further validation is needed.
- In practise, soil C/N ratios show only a very poor relation with N input (see e.g. De Vries et al., 2003c).
- Output is the potential vegetation on a site, the observed vegetation will differ due to time lag effects.
- Until now, it is not possible to calculate target load functions for the critical $C/N_{BS(crit)}$ limit.
- Base saturation is not the direct environmental factor for intoxication of ground vegetation by acidification, in the future the Bc/Al ratio in the soil solution will be used.

Note also that systems with low C/N ratios can not be recovered when coupling the BERN model with VSD. By a more tight integration with a geochemical model like SMART2 or any PnET/geochemical model combinations like ForSAFE or PnET/BGC, this can be solved.

3.5 The ForSAFE-VEG model

General approach

The ForSAFE-VEG model (Figure 30) developed and used in Sweden consists of: (i) the ForSAFE model, aimed at the dynamic simulation of changes in soil chemistry, soil organic matter, hydrology and tree biomass growth in relation to changes in environmental factors (Wallman et al., 2005); and (ii) the VEG submodel, which simulates changes in the composition of the ground vegetation in response to changes in biotic and abiotic factors such as light intensity at the forest floor, temperature, grazing pressure, soil moisture, soil pH and alkalinity in addition to competition between species based on height and root depth (Sverdrup et al., 2006).

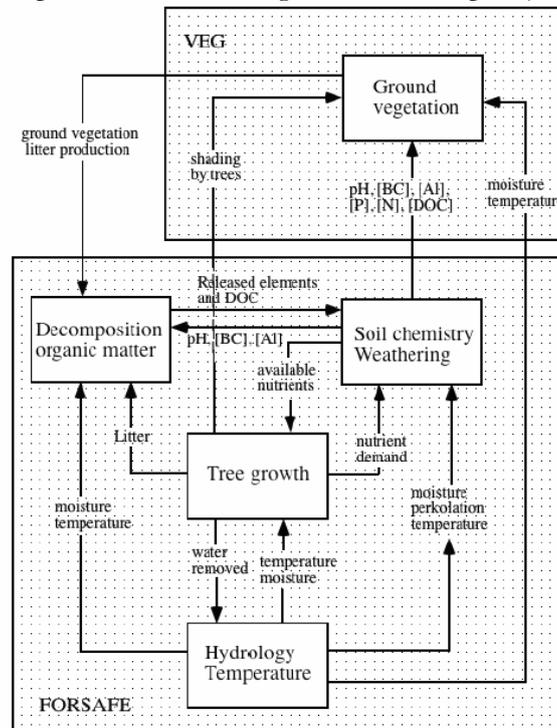


Figure 30 A flowchart of the ForSAFE-VEG model showing the integration of the different modules. The ground vegetation composition module is driven by factors simulated in the ForSAFE model.

Figure 31 illustrates the workflow of calculations adopted in ForSAFE-VEG. For each time step, defined by the resolution of the input data, ForSAFE simulates the changes in state variables in response to environmental changes (temperature and precipitation, atmospheric deposition, forest management). These state variables are read by the VEG module, where the occupancy strength is calculated for each plant

group. The plant groups are defined by the user. The single occupancy strengths are then used to calculate the relative occupancy of each plant group.

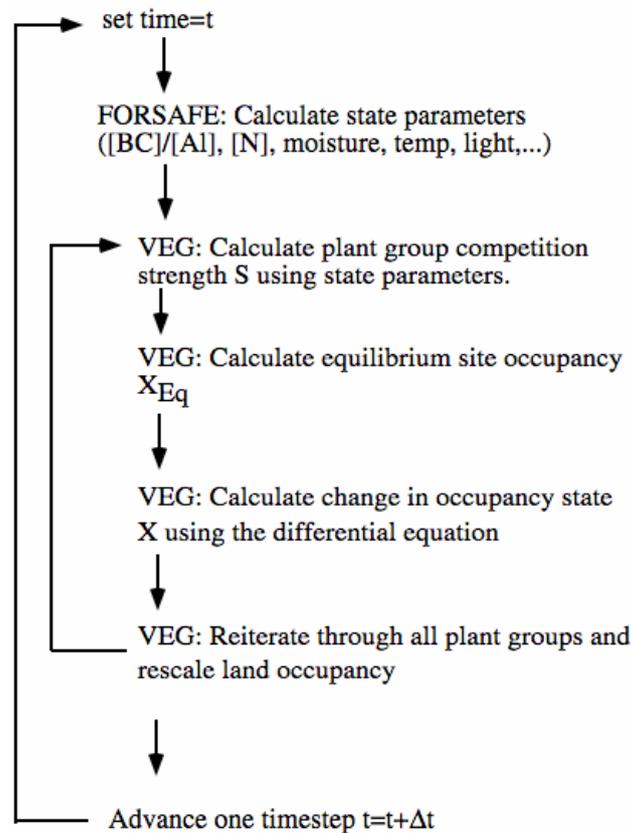


Figure 31 Calculation flow when using ForSAFE-VEG for critical loads calculations of nitrogen

Relationship between species diversity and abiotic factors in FORSAFE

General approach

ForSAFE-VEG simulates the ground vegetation occupancy based on the individual response of plant species to the following drivers:

- Soil solution nitrogen content (kmol.m^{-3})
- Soil acidity ($[\text{H}^+]$, $[\text{BC}^{2+}]$, $[\text{AL}^{3+}]$ (eq.l^{-1})
- Soil water content ($\text{m}^3 \text{ water.m}^{-3} \text{ soil}$)
- Soil temperature ($^{\circ}\text{C}$)
- Light reaching the ground ($\mu\text{mol photon.m}^{-2}.\text{s}^{-1}$)
- Grazing (moose units km^{-2})
- Plant competition based on:
 - Above-ground competition for light (plant height (m/m))
 - Below-ground competition for water and nutrients (root depth (m/m))

Other drivers, being currently developed, are the effects of soil solution phosphorus, the impact of wind chill and wind tatter and air CO_2 concentration. The latter are more detrimental in open fields where the ground vegetation community is exposed to wind.

The change in occupancy of a specific species ($\frac{dX}{dt}$) depends on the actual occupancy of the species (X), the target occupancy (referred to as equilibrium occupancy X_{Eq}), and the specific regeneration time of the species (τ) according to the following equation:

$$\frac{dX}{dt} = \frac{1}{\tau} \cdot (X_{Eq} - X) \quad (9)$$

The regeneration time τ is related to the lifespan of a specific species. If a stress factor would eliminate a certain species, the disappearance of this species will not be instantaneous, but will happen with a delay which depends on the lifespan of the species. The life span in turn is not constant, but depends on the site factors. Drought for example, would shorten the lifespan considerably faster than would a shortage in soil solution nitrogen.

The equilibrium occupancy of a species j , X_{Eqj} , is the ratio between the strength of the species under the specific environmental conditions and the sum of the strengths of all present species according to the following equation:

$$X_{Eqj} = \frac{S_j}{\sum_{j=1}^{j=\text{species}} S_j} \quad (10)$$

Where X_{Eqj} is the fraction of occupied territory, or occupancy of the species j at equilibrium and S_j the individual strength of the species j . It is necessary to note that a species actually represents a plant group, and not a single species. Each species name represent one plant group into which less than 10 up to several hundred individual plant species are assembled. The sum of species strengths is also used as an indicator of the density of the ground cover, referred to as the mass index (MI) and calculated as:

$$MI = \sum_{j=1}^{j=\text{species}} S_j \quad (11)$$

The strength of each species is the product of all the controlling factors:

$$S_i = f_1(N) \cdot f_2(P) \cdot f_3(\text{acid}) \cdot f_4(H_2O) \cdot f_5(G) \cdot f_6(T) \cdot f_7(V) \cdot f_8(I) \cdot f_9(C) \cdot f_{10}(CO_2) \quad (12)$$

Where $f_1(N)$ and $f_2(P)$ are the nitrogen and phosphorus response function respectively, $f_3(\text{acid})$ is the response to soil acidity function, and is also affected by the retardation at high pH and a calcifugity effect for certain plants, $f_4(H_2O)$ is the water response function, $f_5(T)$ is the air temperature response, including wind chill, $f_6(I)$ is the light intensity response function, which reflects the response of the

species to the light reaching the forest floor, $f_7(G)$ is the grazing effect function, $f_8(V)$ is the wind tatter damage and wind chill effect, $f_9(C)$ is the competition function representing above ground competition for light depending on plant height, and belowground competition for water and nutrients and $f_{10}(CO_2)$ is the ambient air CO_2 response function, which is currently being implemented. These effects are multiplicative in synthesis, and have the same weights in affecting the plant strength. When a plant species has roots in more than one soil layer, the belowground response functions (N, water, P, acidity) are weighed according to the plant's root distribution.

Individual response functions

For each plant group indicator that has been selected, response functions were parameterised from published laboratory and field data, by approximations from empirical data, or by scaling the response with respect to other plant groups for which the response is known. However, the basic shape of each response function does not vary between the plant groups. For example, all plant groups will respond positively to an increase in water availability in the soil up until a certain level where anaerobic conditions in the saturating soil may hinder the plant's growth. The distinction between the plant groups is the minimal water content required for survival, optimal water content for growth, and the point at which water becomes damaging. The individual responses functions are described in detail in Annex 6. The ForSAFE-VEG is specifically suited to predict vegetation changes in response to deposition scenario's. Examples of such applications are given in Annex 7. More information on the derivation of limiting N concentration values for individual plant species in the ForSAFE-VEG model is presented in Annex 8.

Assessment of critical N loads with the ForsSafe Veg model for Sweden.

The critical load is determined to be passed at the time we can observe significant shifts in vegetation composition, abundance or the entry/departure of plant groups. This time is used for estimating the deposition of nitrogen at the point in time of significant unwanted vegetation change. Criteria that could be used are e.g.: (i) a permanent shift in composition by more than 5% for each plant group, (ii) a change in ground vegetation index by more than 15% or (iii) the loss of two or more plant groups. To estimate the critical loads of N, a preliminary definition was adopted by which a 95% of the natural ground vegetation composition is preserved. This definition excludes the effect of other factors than N on the ground vegetation composition.

Critical loads estimates for 16 forested sites in Sweden thus derived are given in Table 12. The table presents the year when the just unaccepted change in ground vegetation composition occurred, and the value of the deposition at that year. A reduction from today's deposition values can then be deduced to lower the deposition to the historic value that preceded the undesired change in the ground vegetation composition (Table 12). The estimates set the critical load as the deposition at the time the change occurs, probably leading to a slight overestimate of the critical load.

Table 12 Preliminary critical loads for N based on preservation of the ground vegetation biodiversity according to the set conditions for non-effect.

Site	Time of vegetation response	Critical load deposition kg.ha ⁻¹ .yr ⁻¹	Present deposition kg.ha ⁻¹ .yr ⁻¹	Excess deposition kg.ha ⁻¹ .yr ⁻¹	Required deposition reduction %
Högbränna	1910	1.1	1.5	0.4	27
Brattfors	1890	0.9	2.0	1.1	55
Storulvsjön	1925	2.0	3.5	1.5	43
Högskogen	1928	4.8	7.9	3.2	40
Örlingen	1910	3.6	8.5	3.9	52
Edeby	1918	3.9	7.8	3.9	50
Blåbärskullen	1880	1.6	8.5	6.9	81
Höka	1920	4.0	8.9	4.9	55
Hensbacka	1922	7.4	18.0	10.6	59
Söstared	1868	2.1	20.0	17.9	89
Gyng	1870	2.8	8.3	5.5	66
Fagerhult	1915	3.7	7.5	3.8	51
Bullsäng	1870	2.1	15.0	12.9	86
Timrilt	1889	3.6	23.0	19.4	84
Vång	1910	7.8	17.0	9.2	54
Västra Torup	1866	2.4	27.0	24.6	91

Figure 32 shows a map of all the sites used (left), the time of vegetation change driven by N (middle) and the estimated critical load for nitrogen (right). Below this limit, the biodiversity remains unaffected by N pollution. It can be seen that all sites have significant exceedance, and in order to protect 95% of the area, a 90% reduction of present deposition is required, implying an average atmospheric deposition in southern Sweden of 1.1 kg N.ha⁻¹.yr⁻¹. If we relax the protective level to 50%, still a 55% in present deposition will be required, implying an average deposition in southern Sweden of 2.8 kg N.ha⁻¹.yr⁻¹.

The older approach for Sweden was based on a simple mass balance for nitrogen sources and sinks. Immobilization was ad hoc set at a low value, the connection to ecological effect was very weak or maybe almost non-existent. Uptake was set at the sustainable limit according to mass balance, but no feedbacks from acidity on base saturation dynamics were considered.

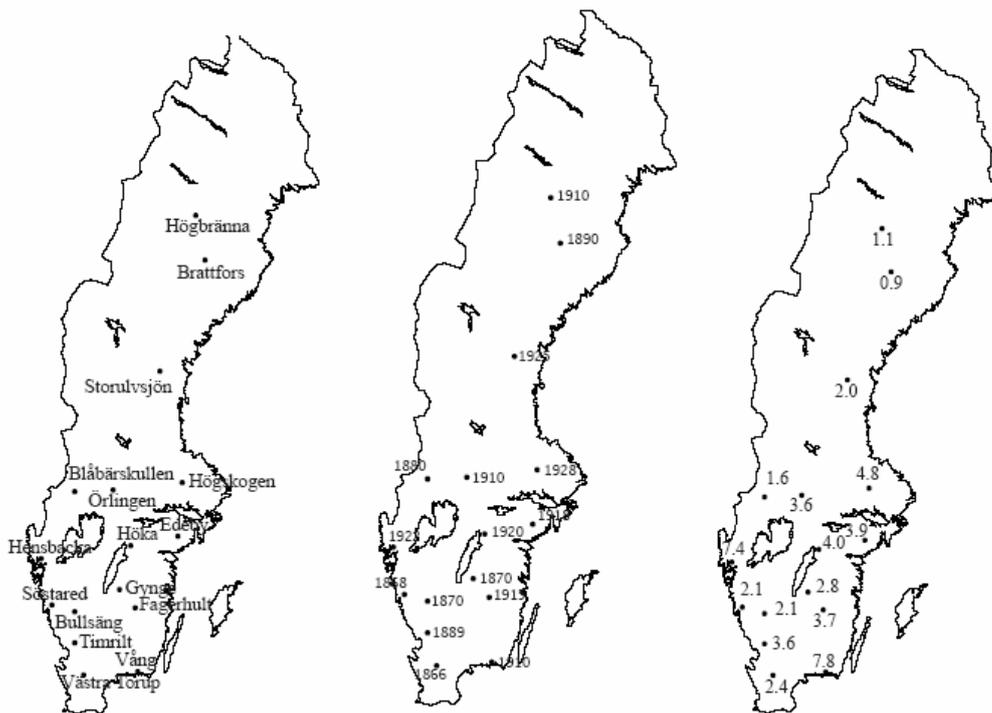


Figure 32 For each of the sites on the map to the left, the year for which a change in the ground vegetation composition has occurred due to N deposition is recorded (middle). Accordingly, the N deposition corresponding to the years in the middle map are used to indicate the critical loads for N on a ground vegetation composition basis.

The impact of using other criteria is presented in the nomogram is shown in Figure 33. The lines represents total nitrogen deposition at different sites in Sweden as a function of time. To summarize, these criteria were used in our example:

- A shift in composition by more than 5% of area cover for each plant group
- A change in ground vegetation mass index by more than 15%
- The loss of two or more plant groups
- The entry of two or more new plant groups

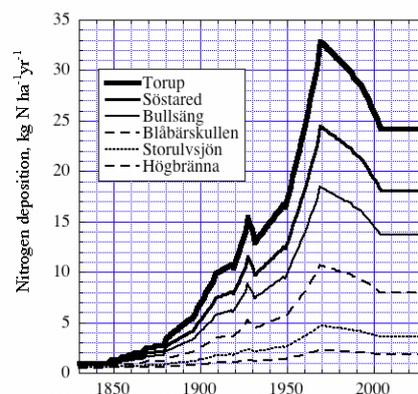


Figure 33 Nomogram used to convert time to a corresponding N-deposition during critical load estimation

At present we have the ability to consider the following ecosystem indicator organisms and functions:

1. Tree species (vitality, population survival/regeneration and nutrient supply aspects) Norway spruce, Scots pine, Birch, Beech, Oak, Maple
2. Ground vegetation species (abundance, survival/regeneration and composition aspects)
 - Individual species comprising 42 groups, including lichens, heather and lings, grass, brackens and ferns, herbs and flowers and bushy plants
 - Ground vegetation composition, mass index and biodiversity index
3. Soil micro fauna: 5 species of colembola (abundance, survival/regeneration and composition aspects)
4. Soil fauna: Earthworms (abundance and survival/regeneration aspects)

The present approach has a new focus on:

1. No long term excess leaching of base cations that will deplete fertility and growth potential for productive forest
2. Critical load based on biodiversity effects driven by nitrogen
 - For ground vegetation
 - For some aspects of soil fauna
 - For some aspects of soil functions

In the new approach the effect based criteria have become accurate and well defined, in our example, with very strict limits for N-induced change in biodiversity. The new and older values are not really comparable, as the old mass balance approached partly lacked ecological robustness and clear environmental protection objectives. In general, the new method tends to give significantly lower values than either the mass balance approach, and significantly lower values than the empirical values. The mass balance approach gives values in the range from 3-12 kg N.ha⁻¹.yr⁻¹, the empirical approach works in intervals with 5 kg (5, 10, 15, 20, ... kg N.ha⁻¹.yr⁻¹), whereas the present approach gives a diversified range from 0.3 to 8 kg N.ha⁻¹.yr⁻¹. Not to much should be made of these differences, because the ecological effects link is quite different in focus and quality between them.

However, the problems with the empirical values are substantial, and the rigor in their derivation still lacks internal consistency. In terms of methodology, the epidemiological approach used is at best very tentative and still has significant flaws. For the Nordic countries it is highly questionable if any of the empirical values derived for Europe are valid, and it is often unclear what they really try to protect.

Evaluation of the model approach

The strengths of the ForSafe-Veg approach are:

- Mechanistic approach relating many abiotic parameters to species diversity
- Inclusion of ground vegetation community competition
- Inclusion of feedbacks from weather and wind
- Inclusion of grazing and feedbacks from large animals
- Full inclusion of forest management

- Full mechanistic integration of the nitrogen cycle with process kinetics and full feedbacks to the chemistry, organic matter decomposition and growth cycles
- No forcing of the model with system outputs like C/N-ratios or forest growth.
- Field tested in Sweden, Norway, Great Britain and Iceland
- The ForSAFE approach is applicable to all sites which have earlier been adapted to the SAFE model

The weaknesses of the ForSafe-Veg approach are:

- High demands on data and quality.
- Critical N deposition is influenced by N deposition history.
- The large complexity of the model makes interpretation of the results quite difficult, as well as the communication of how the different factors like acidity, nitrogen, management and climate change are all linked and very difficult to uniquely separate in quantitative terms.
- For each new ecological distinct ecozone, the ground vegetation parameter file must be parameterized for all plant groups not previously available on existing lists. This requires access to time with very competent plant ecologists.
- No horizontal feedbacks in the landscape
- No handling of slope dynamics in forest stands
- Only ground vegetation aspect of biodiversity considered so far.

3.6 Comparison of modelling approaches and data requirements

Comparison of modelling approaches

The various modelling approaches described in this report consist of a combination of a biogeochemical model of nitrogen behaviour in the soil, connected with a vegetation model predicting nitrogen impacts on biodiversity. The biogeochemical models discussed are SMART2 (either or not in connection with SUMO), MAGIC, VSD and ForSafe. These models differ with respect to the included processes and management options (Table 13).

Models of vegetation succession are included in FORSAFE, and in the model chain SMART2-SUMO-MOVE/NTM, with SUMO being a specific model for vegetation succession. Vegetation succession models are intermediates between biogeochemical models and species composition models since they simulate changes in element budgets and vegetation types. Both SUMO and FORSAFE thus simulate the development of vegetation biomass and stocks of nutrient elements in relation to events such as fire, grazing, mowing or turf stripping. For example, grazing increases light availability and thus favours the growth of short-growing plants. A comparison of the characteristics of the vegetation models and succession models predicting nitrogen impacts on biodiversity (MOVE/NTM, BERN, VEG and SUMO) is given in Table 14. A summary of the weaknesses and strengths of the various approaches is given before.

Table 13 Key processes represented in biogeochemical and vegetation models used in model chains for assessing impacts of nitrogen on biodiversity. ● = modelled dynamically; ○ = modelled indirectly or in a simplified way; k = included as constant or fitted term; - = not modelled.

Process	SMART2	SMART2/SUMO	MAGIC	VSD	ForSAFE-VEG
Photosynthesis / tree growth	k	●	-	-	●
Competition / succession	-	●	-	-	●
Plant N uptake	●	●	○	-	●
Symbiotic nitrogen fixation	k	●	-	-	k
Litterfall	●	●	○	-	●
Decomposition	●	●	○	-	●
N mineralization	●	●	○	○	●
Nitrification	●	●	○	k	●
Denitrification	●	●	○	○	○
Inorganic N leaching	●	●	●	●	●
Organic N leaching	-	-	○	-	○
N immobilization	●	●	●	●	●
Soil carbon dynamics	●	●	○	-	●
SOM pools with different reactivity	●	●	-	-	●
Major ion chemistry/acidity	●	●	●	●	●
Base cation weathering	○	●	k	k	○
Grazing	-	●	○	-	●
Fire	-	●	○	-	●
Sod cutting	-	●	○	-	-
Tree felling	-	●	○	-	●

Data requirements of the various models

To explore the possibility for dynamic modelling applications on eutrophication, applying one or more of the models described in this document, it is needed to have insight in needed input data and the currently available input data. An overview of the data needs of the different models is given in Table 15. The driving variables (Table 15) consist mainly of descriptions of events, in particular the timing and intensity of grazing and other management events. Models of vegetation succession are included in FORSAFE, and in the model chain

Table 14 Comparison of the characteristics of MOVE/NTM, BERN, VEG and SUMO.

Characteristic	MOVE/NTM	BERN	VEG	SUMO
Methodology				
Relation between abiotic conditions and species diversity	Statistical (Logistic & Splines)	Statistical (Fuzzy)	Mechanistic competition model (growth functions)	Mechanistic competition model (growth functions)
Abiotic conditions as single stressors, combined single stressors or multistressors	Multistress (water content, pH, N-availability)	Multistress (water content, BC, C/N, region, Temperature)	Combined single stressors (water content, pH, N, P, light, temperature, grazing)	Combined single stressors (water content, pH, N, P, light, grazing, management)
Crucial factor for critical load calculations	pH & N-availability	Combinations of BS & C/N)	N-concentration in soil solution & pH & B/S & Al-concentration & plant competition (light & nutrients)	N-availability & pH & plant competition (light & nutrients)
Link between environment and biodiversity	(In)direct (correlations between mean Ellenberg-indicator values of plant releveés and abiotic measurements)	(In)direct (direct relations between species occurrences and abiotic measurements & indirect for relations between species occurrences and average conditions per soil type)	Indirect: model-outcome as a result of differences in species-specific growth functions	Indirect: model-outcome as a result of differences in plant type-specific growth functions
Applicability				
Link with biodiversity targets	Direct (via protected species or protected habitat types of EU-habitat directive)	Direct (via habitat types of EU-habitat directive)	Indirect by calculating a relevant indicator.	Indirect: Only possible after link with a species model
Link with SMB for CL-calculations	Possible via acceptable N-leaching	Operational via acceptable N-immobilisation	-	-
Link with dynamic soil models for critical load and target-load calculations	Operational (SMART2)	Link with dynamic models is operational (SAFE, VSD) but can't yet be used for target-load calculations	Link with dynamic models is operational (SAFE) and used for target-load calculations	Link with dynamic models is operational (SMART2) but can't be used for target-load calculations
Countries for which the method is developed and tested/used	Methodology developed for the Netherlands, tested in UK, Denmark	Methodology developed for Germany, tested in Switzerland	Methodology developed and tested in Nordic countries	Methodology developed for Netherlands, tested in UK.

Table 15 Data requirements for biogeochemical and vegetation models used in model chains for assessing impacts of nitrogen on biodiversity. ● = time series or seasonal variation data used; k = constant, average or initial value used; * = data not required, e.g. because modelled dynamically; - = not used.

Group	Variable ¹	SMART2	SUMO	MAGIC	VSD	ForSAFE-VEG
N and C	NO _x and NH _y deposition fluxes	●	●	●	●	●
	Soil or litter total N (or C / N)	●	●	k	k	k
	Soil N in different organic pools	●	●	-	-	*
	C / N leaching thresholds	-	-	k	k	-
Acidity	Cation and anion deposition fluxes	●	●	●	●	●
	Base saturation	●	●	k	k	k
	pH	●	●	k	*	k
Soil water	Soil mineralogy	k	k	-	-	k
	Drainage flux	k	k	k	●	-
Climate	Water holding limits	-	-	-	k	k
	Temperature	●	●	k	-	●
	Precipitation	●	●	●	●	●
	Light flux	-	●	-	-	●
	Wind velocity	-	-	-	-	●
Plant growth	Atmospheric CO ₂ concentration	-	-	-	-	●
	Tree growth parameters	k	●	-	-	k
Management	Plant N uptake	k	*	●	●	*
	N offtake in harvests	k	●	●	●	●
	Grazing intensity	-	●	-	-	●
	Type of grazer	-	●	-	-	k
	Fire events	-	●	●	-	●
	Sod cutting events	-	●	●	-	-
	Tree felling events	-	●	●	-	●

¹) The combination of the vegetation model MOVE with either SMART2 or MAGIC and the combination of the vegetation model Bern with SMART2, MAGIC, VSD or ForSafe does not require any additional data compared to the use of the individual models

4 Critical nitrogen limits: evaluation of currently used values and possible updated values for use in critical load assessments

As stated before, a critical nitrogen load is the maximum flux (in $\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) that an ecosystem is able to sustain. Since critical loads, refer only to a steady-state situation of the receptor, the critical N load is calculated as the sum of N fluxes in a steady state situation according to (SMB model):

$$\text{CL}(\text{N}) = \text{N}_{\text{up}} + \text{N}_{\text{im}} + \text{N}_{\text{de}} + \text{N}_{\text{le}(\text{crit})} \quad (13)$$

where:

- $\text{CL}(\text{N})$ = Critical N load ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)
- N_{up} = Average net yearly uptake during a forest rotation, being equal to the amount removed during harvesting divided by the rotation period ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)
- N_{im} = Long-term average net yearly immobilization ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)
- N_{de} = Denitrification flux at critical N load ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)
- $\text{N}_{\text{le}(\text{crit})}$ = Critical yearly N leaching rate ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)

This approach is based on the idea that when the N leaching rate exceeds a critical value, adverse impacts do occur in terms of impacts on e.g. plant species diversity. Other possible impacts are those on faunal species diversity, forest nutrient status and ground water quality (see chapter 2). The critical N leaching rate is determined by the water flux leaving the ecosystem (precipitation excess) and a critical N concentration in soil water. Below we discuss the currently used critical N concentrations in soil water when deriving critical N loads and possible alternatives based on updated information on nitrogen impacts.

4.1 Evaluation of currently used critical nitrogen limits

Currently used critical nitrogen limits are given in Table 16, based on the mapping manual 2004 (UBA, 2004). Regarding the substantiation of these numbers, the mapping manual already indicates that this is hard in the case of vegetation changes since: (i) no direct relationship between N leaching and vegetation changes can be substantiated, (ii) the critical loads based on these critical N concentrations lead to critical loads that are lower than empirical data on vegetation changes and (iii) it is the increase in N availability through enhanced N cycling that triggers vegetation changes. A further evaluation of these numbers, focusing on the aspects mentioned above is given below.

Critical N concentrations in soil solution in view of vegetation changes

The origin for the limits related to vegetation changes is based on Warfvinge et al. (1992). Until now, the basis for these limits is a statement in Warfvinge et al. (1992)

Table 16 Acceptable N concentrations in soil solution to avoid nutrient imbalances or vegetation changes (UBA, 2004)

Impact	Critical N concentration (mg N.l ⁻¹)	
	UN/ECE (2004)	Warfvinge et al. (1992)
Vegetation changes		
- lichens to cranberry (lingonberries)	0.2-0.4	< 0.2
- cranberry to blueberry	0.4-0.6	< 0.4
- blueberry to grass	1-2	< 1.0
- grass to herbs	3-5	< 2.0
Nutrient imbalances		
- conifers	0.2	-
- deciduous trees	0.2-0.4	-
Ground water quality		
- EC target drinking water	11.3	-
- Target value	5.6	-

that “limiting N soil solution concentrations have been suggested based on preliminary experiences from the Swedish Forest Survey program”. There is thus no real substantiation from the literature for these limits. In Annex 9, however, an overview is given of the derivation of these limits. In principle, the values are based on an inverse use of the SMB model (Eq. 13), by using empirically derived critical N loads and deriving the critical N leaching rate by subtracting derived values for the related N uptake, N immobilization and denitrification and dividing this flux by the water flux, according to:

$$[N]_{(crit)} = (CL(N) - N_{up} + N_{im} + N_{de}) / Q \quad (14)$$

where:

$$[N]_{(crit)} = \text{Critical N concentration (kg.m}^{-3}\text{)}$$

$$Q = \text{water flux (m}^3\text{.ha}^{-1}\text{.yr}^{-1}\text{)}$$

The values for CLN are based on a visual interpretation of maps of vegetation changes in Sweden in the period 1977-1987 (see Annex 9). The derivation of the related N fluxes is also described in detail in this annex. Actually, the limiting or critical N concentrations in soil solution presented in the Mapping Manual (see Table 16), which in turn are quoted from Posch et al. (1993), slightly deviate from those given by Warfvinge et al. (1992), but the basis for this difference is unclear.

It has to be stressed that the critical N concentrations thus derived are limited to the Nordic countries and based on a highly aggregated approach in deriving empirical critical N loads. Furthermore, there are many simplifying assumptions in deriving the critical N concentrations. In general, however, the values from Table 16 will lead to critical loads that are applicable in the Nordic countries (at least comparable to derived empirical critical N loads).

Use of such critical N concentrations in other countries may, however, lead to critical N loads that are lower than empirical values. As an example, Table 17 presents average critical N loads calculated for the Netherlands using the SMB model

(Equation (13)) in which $N_{le(acc)}$ was estimated on the basis of a critical N concentration of $0.05 \text{ mol}_c \cdot \text{m}^{-3}$ ($\approx 0.7 \text{ mg N} \cdot \text{l}^{-1}$) at the bottom of the root zone, being in the centre of the suggested values in Table 14. The critical loads were estimated for all unfertilised $1 \times 1 \text{ km}^2$ grid cells in the Netherlands, using the dominant soil and vegetation types per square as described in Van Dobben et al. (2004). N_{gu} was estimated on the basis of the soil type, the vegetation type and the groundwater level; denitrification was estimated on the basis of the soil type and the groundwater level; N_{im} was set to a constant value of $2.8 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ (near the upper value of $3 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ mentioned in the manual). The denitrification values were derived as a fraction of the N leaching rate and varied from approximately $0.2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ for sandy soils to $4 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ for clay soils. The overall average critical N load varies from approximately 5 (heathland) to 11 (deciduous forests) $\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ (see Table 17), while empirical critical N loads for these ecosystems in the Netherlands vary between 10-25 $\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ (see also Table 2 in Section 2.1). This implies that use of the critical N concentrations in the manual (UBA, 2004) for the non-Nordic countries may lead to an underestimate of the critical N loads.

Table 17 Average and standard deviation for critical N loads per combination of soil type and vegetation type in the Netherlands (excluding peat soils) derived by the SMB method; A = area with this soil and vegetation as the dominant type.

Vegetation type	Soil type	Critical loads ($\text{kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)		A (km^2)
		Mean	Se	
grass land	Sand Poor	6.2	1.8	(3343)
	Sand Rich	6.4	0.8	(2094)
	Sand Calcareous	4.7	3.4	(836)
	Clay	9.9	3.5	(1127)
	Clay calcareous	8.1	4	(1136)
	Loess	7.2	1.4	(112)
Heathland	Sand Poor	4.4	0.9	(1557)
	Sand Rich	5.4	0.7	(179)
Pine forests	Sand Poor	7.2	0.4	(5457)
Deciduous forests	Sand Poor	9.9	0.7	(3980)
	Sand Rich	11.5	0.9	(2398)
	Sand Calcareous	9.7	1.5	(339)
	Clay	11.7	2.4	(1121)
	Clay calcareous	10.5	1.3	(561)
All vegetation types	Sand Poor	7.3	1.8	(15581)
	Sand Rich	8.8	2.5	(5893)
	Sand Calcareous	5.8	3.6	(1271)
	Clay	10.6	3.2	(2367)
	Clay calcareous	8.9	3.6	(1708)
	Loess	10.4	2.3	(389)

Critical N concentrations in soil solution in view of nutrient imbalances

The origin for the limits related to nutrient imbalances is based on Posch et al. (1993) by assuming that “at steady state with a balanced nutrient supply, the nitrogen

leaching should amount to the natural leaching from nitrogen limited stands". For natural N concentrations, reference has been made to NO_3 concentrations of approximately $0.2 \text{ mg NO}_3\text{-N.l}^{-1}$ in stream water of nearly unpolluted forested sites in Sweden (Rosén, 1990). The basis for the natural N concentration below deciduous forests, being twice as high, remains unclear.

More important than the actual value, however, the assumption that nutrient imbalance occurs as soon as the N leaching increases above natural N leaching rates is not substantiated by the literature nor by the concepts of plant physiology. When the N input exceeds the N uptake and natural N leaching of a nitrogen limited stand, it may in first instance lead to a higher N uptake, due to higher biomass production and higher N contents, and thereafter to a state where N is not a limiting nutrient any more. Unbalanced nutrition, in terms of an unbalanced ratio of P, K, Ca or Mg to N, likely occurs at higher N leaching levels than just above the natural level of N limited systems. The described impacts of N deposition on forest nutrition in Section 2.3 confirm this. The occurrence of relative P or Mg deficiencies, which are the most common imbalances, occurs generally at N loads above $10\text{-}20 \text{ kg.ha}^{-1}\text{.yr}^{-1}$ (see Section 2.3; Table 4).

Use of a natural N concentration of 0.2 mg.l^{-1} (most often used in critical load calculations) leads to a critical N leaching rate, $N_{\text{le(crit)}}$, of $0.2\text{-}1.0 \text{ kg.ha}^{-1}\text{.yr}^{-1}$, considering a precipitation excess of $100\text{-}500 \text{ mm.yr}^{-1}$. Using values presented in the manual (UBA, 2004) of $1 \text{ kg.ha}^{-1}\text{.yr}^{-1}$ for long term N immobilization and $0.5 \text{ kg.ha}^{-1}\text{.yr}^{-1}$ for denitrification (typical for well drained forest soils) leads to critical loads that are nearly always below being $10 \text{ kg.ha}^{-1}\text{.yr}^{-1}$ since N uptake is mostly below $7.5 \text{ kg.ha}^{-1}\text{.yr}^{-1}$. As with vegetation changes, this implies that the critical N concentration for N imbalances is most likely (far) too low. A more substantiated approach would be to skip the link with nutrition and include elevated N leaching as a topic in its own and then use the most up to date information on the differentiation between N concentrations in natural conditions and in "leaky sites" (see Section 4.2).

Critical N concentrations in soil solution in view of ground water quality

According to the EC Drinking Water Directive, nitrate concentrations in drinking water should not exceed 50 mg.l^{-1} , being equal to $11.3 \text{ mg.l}^{-1} \text{ NO}_3\text{-N}$. In most countries, this value is set as a maximum for upper ground water being a stringent target since substantial denitrification can still occur between upper ground water and the depth of ground water at which drinking water is extracted. In several countries (e.g. the Netherlands) the target value for upper ground water is even set at 25 mg.l^{-1} , being equal to $5.6 \text{ mg.l}^{-1} \text{ NO}_3\text{-N}$. An even more stringent target would be to require that the dissolved N concentration in soil solution at the bottom of the root zone does not exceed those values. The values of 25 and $50 \text{ mg NO}_3 \text{ l}^{-1}$ has also been mentioned in the manual (UBA, 2004). Despite the stringent approach, even the more stringent target value of $5.6 \text{ mg.l}^{-1} \text{ NO}_3\text{-N}$ is generally higher than the values derived in the following section for critical N concentrations in view of plant species diversity (ranging from 3 to 6 mg.l^{-1}) and forest nutrient status (3 mg.l^{-1} for coniferous forest).

4.2 Assessment of updated critical nitrogen limits in soil solution

In the manual for mapping critical loads, critical N concentrations in soil solution are limited to those related to (i) vegetation changes and (ii) nutrient imbalances, whereas reference is made to (iii) ground water quality criteria. In this section, we give an overview of possible updated values for critical N concentrations based on:

- Relationships between N availability and soil pH and plant species diversity impacts (vegetation changes; see Section 2.2).
- A differentiation between undisturbed forest sites and “leaky” sites regarding nitrogen leaching (see Section 2.4).
- Relationships between dissolved N concentrations and impacts on (i) root growth and (ii) critical plant N contents in leaves in view of nutrient imbalances and elevated natural stresses (drought, frost, pests/diseases; see Section 2.4).

Even though it is clear that N deposition does affect faunal species diversity (See Section 2.3), a relationship between impacts and critical N concentrations in soil solution is not feasible. Regarding the critical N concentrations in ground water in view of human health impacts (ground water quality criteria) further update is not considered necessary (see Section 2.5 and 4.1).

4.2.1 Critical N concentrations in view of plant species diversity impacts

General approach

As mentioned before (Chapter 2), out of all the targets, biodiversity is generally the most sensitive one, thus requiring specific emphasis to improve critical limits for these effects. It has been recognized that the severity of the impacts of atmospheric nitrogen deposition is depending on a number of factors (semi-)natural ecosystem and cannot be addressed by using a generic critical N leaching and N concentration alone. Most important factors are (cf. Bobbink et al., 2003): (1) the duration and total amount of increased nitrogen deposition, (2) the chemical and physical form of the airborne nitrogen input, (3) the intrinsic sensitivity of the plant and animal species present, (4) the abiotic conditions in the ecosystem and (5) past and present land use or management. Long-term multi-factorial experiments can be used to describe the influences of these factors on the sensitivity of ecosystems for nitrogen deposition. However, such experiments are scarce, hampering the calculation of critical loads.

In chapter 3, we describe the additional possibilities that a multi-plant species model in combination with a dynamic soil vegetation model have for directly calculating critical N loads. Here, options for updating critical N concentrations in soil solution based on results of those models are described. Critical load assessments with simple and complex models and the relationship between them is schematically shown in Figure 34.

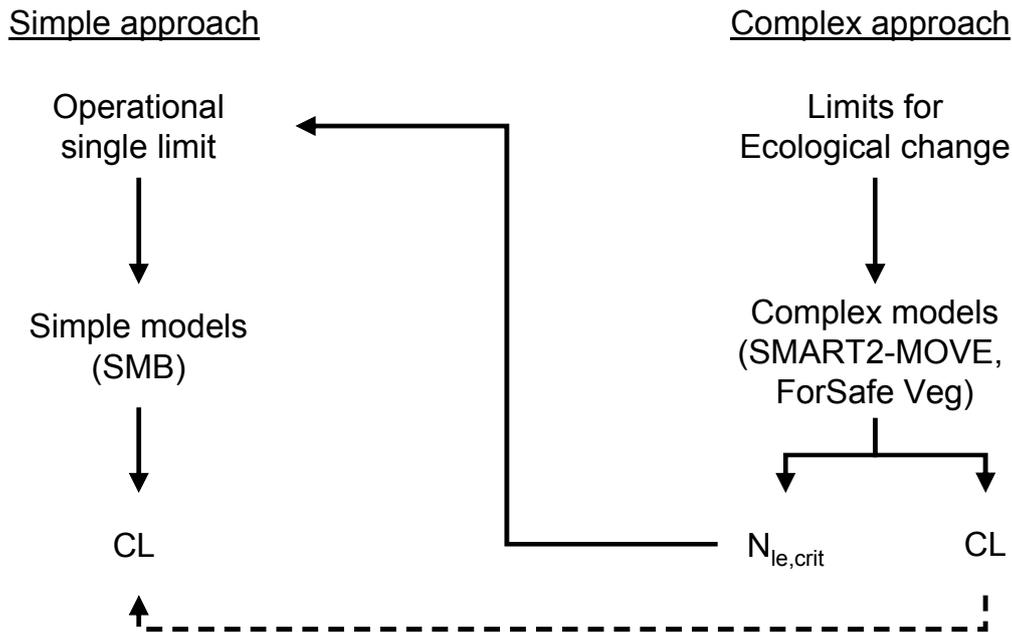


Figure 34 Critical load assessments with simple and complex models and the relationship between them

In principle, a critical N concentration, $[N]_{(crit)}$, can be derived from a calculated critical load with a detailed model, using Eq. (14). This approach has been applied for the Netherlands, using the SMART-MOVE model as illustrated below.

In this context, it is important to realise that the N concentration in soil solution is a result of various processes, including N uptake and N transformations (mineralization/immobilisation etc). In an N limited system, first the production will increase, followed by N immobilisation, before leaching effects occur (Figure 1). The derived critical N concentration can thus vary between near 0 (impacts already take place before leaching occurs) to a certain upper value. Information on the median N critical N concentration in soil solution (and the range) should thus be seen as a surrogate, derived from more adequate N indicators such as N availability, to be used in the SMB model concept.

Model application for the Netherlands

Assessment of critical N concentrations with SMART2-NTM: Van Dobben et al. (2004) used the SMART2 model to calculate the critical N load in an inverse way based on the required N availability for various relevant plant associations in the Netherlands. Simulations were carried out during a 50 year period. For forests, the simulation started with trees at an age of 50 years in case of conifers and 80 years in case of deciduous forests. Critical load values were calculated for terrestrial plant communities in the Netherlands including the N fluxes that make up the critical N load. From information on uptake, immobilization and denitrification and the waterflux, the critical N concentration was derived according to Equation 14.

Denitrification and leaching fluxes are directly calculated in SMART2, but the net uptake and immobilization fluxes were derived from nutrient cycling fluxes calculated in SMART2, according to:

$$N_{up} = N_{ru} + N_{fu} - N_{lf} \quad (15)$$

$$N_{im} = N_{lf} - N_{mi} + N_{imm} \quad (16)$$

where:

- N_{ru} = Average root uptake during the 50 year simulation period ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
- N_{fu} = Average foliar uptake during the 50 year simulation period ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
- N_{lf} = Average litter fall during the 50 year simulation period ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
- N_{mi} = Average mineralization during the 50 year simulation period ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
- N_{imm} = Average immobilization during the 50 year simulation period ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)

The respective N fluxes that make up the critical load according to Eq. (13) for plant communities on sites with no seepage (76 plots) are presented in Table 18. The net uptake during the 50 year simulation period was comparable to values used in SMB, although the net uptake in that period is likely to be a bit lower than the average uptake during the rotation period. In sites with no seepage, denitrification is also comparable with SMB values, specifically in the non-calcareous sandy soils. The values for N immobilization are slightly higher, but the largest difference with the SMB values is the critical N leaching. In sites with no upward seepage, the average value is $11 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$, whereas the values chosen in the SMB model are close to $1\text{-}2 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$. The related average critical dissolved N concentrations vary from 3.0 to $6.4 \text{ mg N} \cdot \text{l}^{-1}$, being close to the upper range of the currently used critical limits (see Table 16)

Table 18 Average N fluxes and critical N loads ($\text{kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$) and critical dissolve N concentrations for plant communities in grasslands, heathlands, deciduous forest and coniferous forests on sand, clay and peat soils on sites with no seepage (76).

Soil type/ Vegetation type	N flux ($\text{kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)					N conc. ($\text{mg} \cdot \text{l}^{-1}$)
	CLN	N_{up}	N_{im}	N_{de}	$N_{le} \text{ (crit)}$	
Grassland	19.0	2.5	2.1	5.8	8.7	3.0
Heathland	25.9	0.9	3.4	1.8	19.7	5.8
Deciduous	29.0	4.7	7.7	5.4	11.1	4.2
Conifers	24.5	1.8	5.6	0.7	16.4	6.4
Average	22.8	2.9	3.9	5.0	11.0	3.8

Results show a large variation in critical N loads for clay and peat soils (mostly from $10\text{-}40 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$, which is mainly due to a large variation in critical N leaching (in peat from approximately $5\text{-}30 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$), as illustrated in Figure 35, and to a lesser extent in denitrification. The variation in the sum of N uptake and N immobilization is relatively small. The variation in critical N leaching implies also a large variation in related critical N concentration as illustrated in Figure 36.

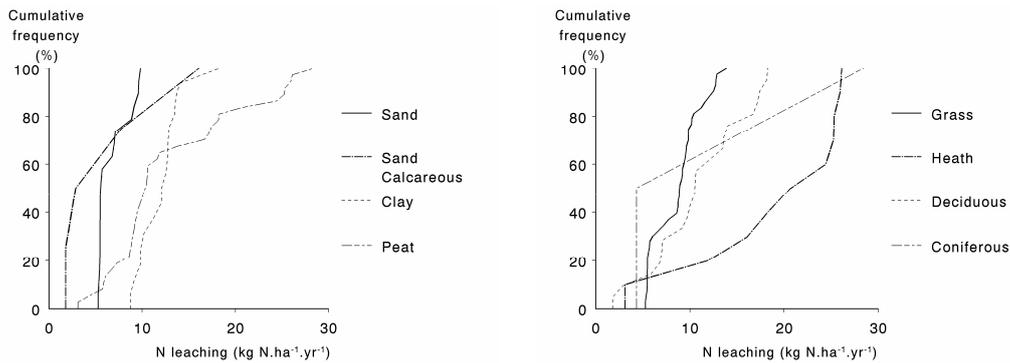


Figure 35 Frequency distribution of the critical N leaching for plots with no seepage.

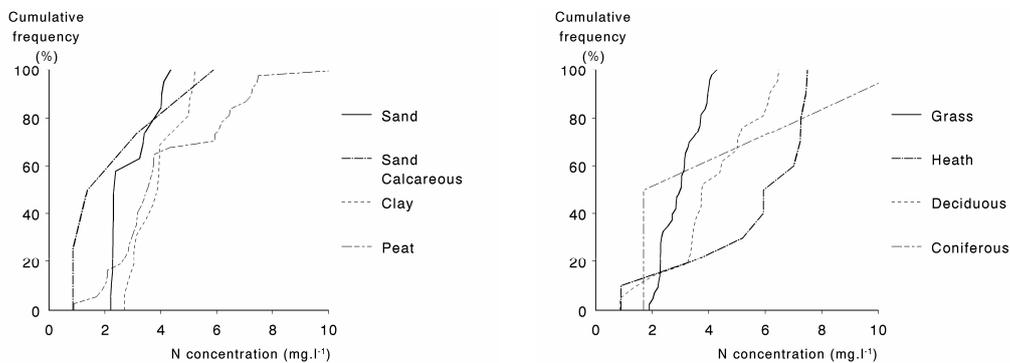


Figure 36 Frequency distribution of the critical N concentration for plots with no seepage.

The plausibility of the derived critical N concentrations for Dutch terrestrial ecosystems can be derived from a comparison of empirical critical loads (Achermann & Bobbink, 2003) and the above mentioned simulated critical loads (Van Dobben et al., 2004) as presented in Table 19. It shows that there is a fair agreement between the empirical and the modelling approach. In general, the empirical critical loads tend to be somewhat lower and to have narrower ranges than the simulated ones. In interpreting the differences it should be noted that the empirical ranges are the result of an interpretation of a large number of studies, and that this interpretation is usually based on a precautionary principle, i.e. it tends to search the lower end of all reported no-effect levels. As a result, the empirical critical load ranges usually pertain to the more sensitive forms of a given ecosystem. On the other hand, the simulated critical loads are determined as an average over all vegetation structures belonging to a given ecosystem, under average environmental conditions for that ecosystem.

In general, the reasonable overlap of the modelled and simulated values, compared to the large difference in those values when using the SMB model with the critical N concentrations in the manual (see Section 4.1) implies that the order of magnitude of the critical N concentrations in Table 18 is better than those present in the manual (UBA, 2004), when applying the model for the Netherlands.

Table 19 Comparison of simulated and empirical critical loads. Unit: kg N.ha⁻¹.y⁻¹. Empirical data are taken from Achermann and Bobbink (2003); calculated values are from Van Dobben et al. (2004). ### = reliable, # = quite reliable, (#) is expert judgement. * = EUNIS class does not occur in the Netherlands. Correspondence between simulated and empirical critical loads is given in the last column.

Ecosystem type (EUNIS class)	Empirical critical load	Reliability	Simulated critical load	Simulated compared to empirical
<i>Grasslands and tall forb habitats (E)</i>				
- Sub-atlantic semi-dry calcareous grassland	15-25	###	15-31	=
- Non-mediterranean dry acid and neutral closed grassland	10-20	#	10-31	=
- Inland dune grasslands	10-20	(#)	10-21	=
- Low and medium altitude hay meadows	20-30	(#)	10-31	=
<i>Heathland habitats (F)</i>				
- Dry heaths	10-20	###	4-31	=
<i>Coastal habitat (B)</i>				
- Shifting coastal dunes	10-20	(#)	15-24	=
- Coastal stable dune grasslands	10-20	#	15-24	=
- Coastal dune heaths	10-20	(#)	33-34	>
- Moist to wet dune slacks	10-25	(#)	10-24	=
<i>Forest habitats (G)</i>				
- Ground vegetation (Temperate and boreal forests)	10-15	#	8-41	=
- Lichens and algae (Temperate and boreal forests)	10-15	(#)	8-9	<
<i>Mire, bog and fen habitats (D)</i>				
- Raised and blanket bogs	5-10	###	26-33	>
- Poor fens	10-20	#	5-30	=
- Rich fens	15-35	(#)	5-30	=
<i>Marine habitats (A)</i>				
- Pioneer and low-mid salt marshes	30-40	(#)	21-24	<

The whole procedure described here should be repeated for vegetation relevés that are important in other parts of Europe while deriving relationships between Ellenberg indicator values and values for N availability and pH. Until then, it seems however much better to use empirical critical N loads than to use an SMB model with the present low N concentrations in solution.

Assessment of critical N concentrations with steady-state SMART2-NTM: The critical loads derived with the SMART2 NTM approach as described in Section 3.2 deviate slightly from those previously calculated by van Dobben et al. (2004) using the dynamic SMART2 approach. This is illustrated in Table 20 in which the results of the calculated critical loads with the related growth uptake of nitrogen, the long-term immobilisation and denitrification and, the critical NO₃ leaching flux is presented. For grassland we implicitly take management into account, resulting in relatively high uptake (see Table 20). This means that all N will be removed either by grazing or mowing. As a result no N is left for leaching. Consequently, for grassland systems this yields a critical N concentration of zero. This makes it difficult to work with critical N concentrations for managed grassland. Comparison with Table 16 shows that the resulting critical loads with SMART2 steady-state are lower. The calculated

median N concentrations are thus also slightly lower than those presented before, as derived by the dynamic SMART2 version. The most important reasons for this are:

- the uptake fluxes for grassland and forest of SMART2 steady-state are higher. These values were adjusted based on SUMO simulations.
- SMART2 steady-state used (slightly) different criteria on abiotic site conditions. The dynamic SMART2 version used criteria assigned to vegetation types based on the NTM model, whereas SMART2 steady-state used nature target type related criteria based on MOVE.
- SMART2 steady-state is focussing at a steady-state, whereas the dynamic SMART2 used a time horizon of 50 year. This period is not long enough to reach steady-state in all systems.

Table 20 Average N fluxes (in SMB terms) and critical N loads ($\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) and critical dissolved N concentrations for plant communities in grasslands, heathlands, deciduous forest and coniferous forests related to vegetation changes.

Soil type/ Vegetation type	N flux ($\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)					N conc. ($\text{mg}\cdot\text{l}^{-1}$)
	CLN	N_{up}	N_{im}	N_{de}	$N_{\text{le (crit)}}$	
Grassland	13.8	10.9	3.5	0	0	0.0
Heathland	15.5	0.9	3.4	0.9	10.3	2.9
Deciduous	20.2	7.1	3.5	1.4	7.2	2.6
Conifers	24.5	3.8	3.5	0.5	7.9	3.6

Figure 37 shows cumulative frequency distribution of critical N loads for forest based on SMB calculations with *old limits* for N concentrations in soil solution (use of $0.2 \text{ mg}\cdot\text{l}^{-1}$ for preventing a change from lichens to cranberries and $1 \text{ mg}\cdot\text{l}^{-1}$ for prevention of a change from blueberries to grass) and SMART2-steady state calculations with *new limits* for N availability and pH in view of plant species diversity impacts. As with forest nutrient impacts, the newly derived critical nitrogen load for vegetation changes are substantial higher than critical loads related to the old values. (Figure 37 and Table 21). The related calculated critical N concentrations are given in Table 22.

Table 21 Ranges in critical loads for N preventing vegetation changes based on current and updated critical N limits

Critical N concentration for vegetation changes ($\text{mg}\cdot\text{l}^{-1}$)	Critical N loads ($\text{kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)					
	Mean	Min	5%	Median	95%	Max
Old						
$0.2 \text{ mg}\cdot\text{l}^{-1}$	9.1	3.3	7.4	8.4	13.0	21.5
$1 \text{ mg}\cdot\text{l}^{-1}$	12.5	5.4	9.6	10.7	18.2	43.7
New						
Ellenberg	17.6	7.5	12.8	16.8	24.6	97.9

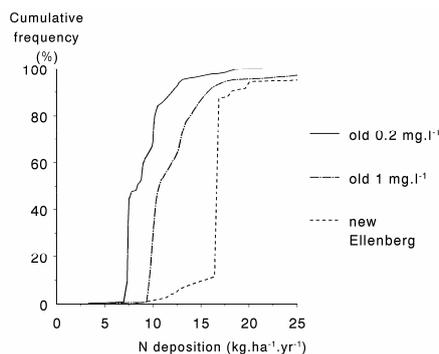


Figure 37 Cumulative frequency distribution of critical loads for N preventing vegetation changes based on current (old) and updated (new Ellenberg) critical N limits.

Note the critical N concentration varies between near 0 (impacts already take place before leaching occurs) to a certain upper value. For heathlands (and most likely unmown grasslands, this upper value is near 5 mg.l⁻¹, being the maximum value given in the manual, whereas the value is near 10 mg.l⁻¹ for forests (Table 22). The median value is near 3 mg.l⁻¹, being a surrogate value that could on average be used in the SMB model concept for Dutch ecosystems. In general, there is an inverse correlation between critical N concentration (critical N leaching) and the other N sinks (uptake, immobilisation and denitrification), implying that use of an average or median value is reasonable when using average values for the other N terms in the SMB model.

Table 22 Calculated median and ranges (50-95%) for critical N leaching rates and critical N concentrations preventing vegetation changes based on updated critical N limits for N availability and pH

Forest type	Critical N leaching (kg N.ha ⁻¹ .yr ⁻¹)		Critical N concentrations (mg N.l ⁻¹)	
	Median	5%-95%	Median	5%-95%
Grassland	0	0 - 0	0	0-0
Heathland	10.3	0 - 22.1	2.9	0-4.9
Deciduous	7.2	0 - 24.9	2.6	0-11.5
Coniferous	7.2	0 - 26.5	3.6	0-8.4

4.2.2 Critical N concentrations in view of a differentiation between undisturbed and “leaky” forest sites

Since 1990, much more information has become available on limits of N concentrations related to pristine sites and sites with clearly elevated N leaching. A nice summary has recently been made by Gundersen et al. (2006). First of all, the mobility of N in soils largely depends on the form of dissolved N (NH₄, NO₃ or dissolved organic nitrogen, DON). Ammonium is absorbed on the soil cation exchange complex and is thus quite immobile in the soil profile. Consequently, ammonium usually contributes less than 5% to the total dissolved N concentration in soil water and NH₄⁺ concentrations are generally below 10 µg NH₄-N.l⁻¹ (Gundersen et al., 2006), except for extremely NH₄ loaded soils (Dise et al., 1998a; 1998b). Concentrations of DON are below 0.6 mg N.l⁻¹ and often even below 0.1 mg N.l⁻¹ in seepage water from well-aerated soils (Andersen & Gundersen, 2000; Michalzik et al.,

2001). Nitrate is thus the constituent in seepage and stream water that responds mainly to flux changes as increased forest N-input. Since nitrate is highly mobile in soils, production in excess of plant and microbial uptake requirements will be transported through the soil profile. Initially, N leaching from a catchment will occur in the dormant season when N uptake and immobilisation is small and limited by temperature. Gradually, this biological control on N leaching may be abolished.

Stoddard (1994) characterised four progressive stages of N saturation based on changes in seasonality and levels of nitrate leaching in streams:

- Stage 0: Strong seasonal pattern in nitrate concentrations. Very low or immeasurable nitrate concentrations at base-flow throughout the growing season. Winter-spring nitrate concentrations at 0.2-0.4 mg N.l⁻¹.
- Stage 1: Still strong seasonality in nitrate concentrations, but some nitrate appears in the growing season. Winter-spring nitrate concentrations peak at up to 1 mg N.l⁻¹.
- Stage 2: Elevated nitrate at base-flow throughout the growing season (0.5-1 mg N.l⁻¹). Seasonal pattern still visible.
- Stage 3: Constant high nitrate concentration (>1 mg N.l⁻¹) with no seasonal pattern.

Gundersen et al. (2006) gave an overview of current water quality in forests by compiling a list of studies of nitrate concentration in seepage water from temperate forests from regional, national and international surveys from the 1990s, including (>500 sites of seepage water from Europe. From the survey data it is difficult to conclude exactly at which level a forest ecosystem can be considered 'leaky' with 'elevated' nitrate leaching but they suggest a level of 1 mg N.l⁻¹ for seepage water (annual mean concentration) as tentative limit values above which nitrate leaching is considered as elevated (and 0.5 mg N.l⁻¹ for streams/catchments). This coincides with the limit in stage 3 given by Stoddard (1994).

4.2.3 Critical N concentrations in view of forest root growth and impacts on foliar N contents

As presented in Section 2.3, there is strong evidence that increased nitrogen deposition reduces fine root biomass and root length. Matzner and Murach (1995) found that total fine root biomass of Norway spruce saplings decreased significantly when the total inorganic N (NO₃ + NH₄) concentration in soil water was >2 mg N.l⁻¹ (Figure 38).

The critical dissolved N concentration in view of adverse impacts on forests can be derived from a critical N concentration in the needles of 18 g.kg⁻¹, above which the sensitivity to frost and fungal diseases increases (Section 2.3). De Vries et al. derived such a relationship on the basis of the results for 120 Intensive Monitoring plots in Europe, as shown in Figure 39 while using the most recent data for both coniferous and deciduous forest. From this graph for coniferous forest a critical limit in the soil solution was estimated as 3 mg.l⁻¹ being the value below which the N contents in foliage are also below 18 g.kg⁻¹. Above 5 mg.l⁻¹ N contents in foliage exceed 18 g.kg⁻¹.

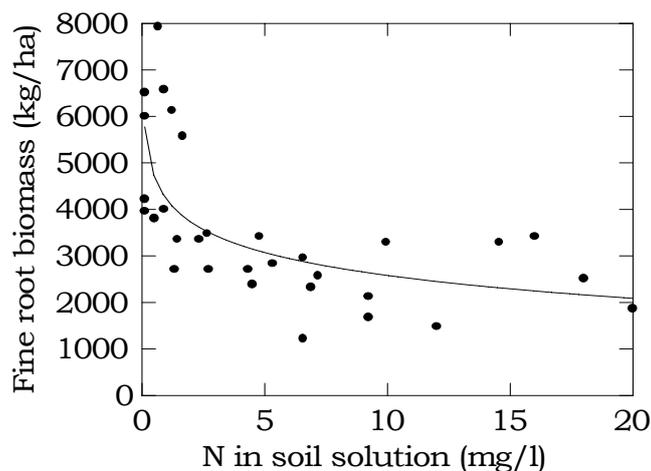


Figure 38 Fine root biomass of spruce in relation to nitrogen in soil solution (After Matzner & Murach, 1995).

This value is quite comparable to the critical median N concentration in soil solution derived from the SMART2-MOVE model in view of biodiversity impacts (This value is quite comparable to the critical median N concentration in soil solution derived from the SMART2-MOVE model in view of biodiversity impacts (Table 22.). For deciduous forest, however, no relation between soil solution N and N concentration in foliage was found (Figure 39).

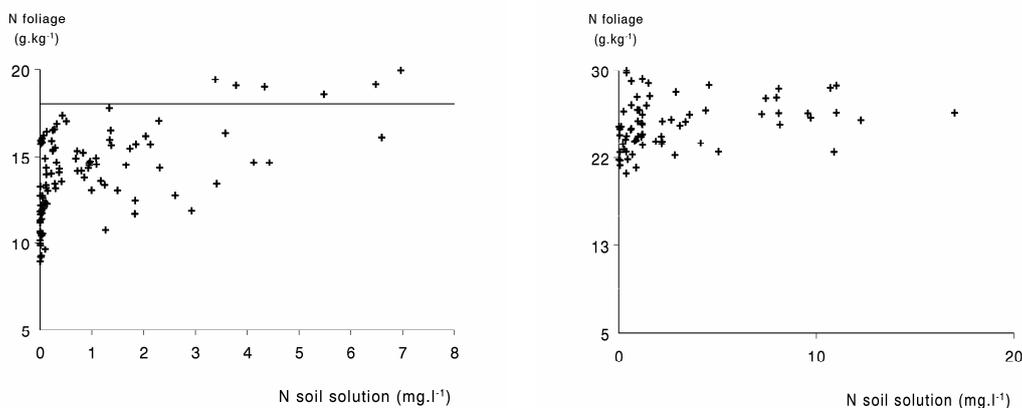


Figure 39 Relationship between measured N content in the foliage of coniferous trees (left) and deciduous trees (right) and the average NO₃ concentration in soil solution for 120 Intensive monitoring plots.

Figure 40 shows cumulative frequency distribution of critical N loads for coniferous forests based on SMB calculations with current critical N concentrations of 0.2 mg.l⁻¹ (coniferous forests) and an updated value of 3 mg.N.l⁻¹ in soil solution in view of nutrient impacts. It also includes the cumulative frequency distribution of critical N loads for deciduous forests based on SMB calculations with current critical N concentrations of 0.4 mg.l⁻¹ for which no update does exist.

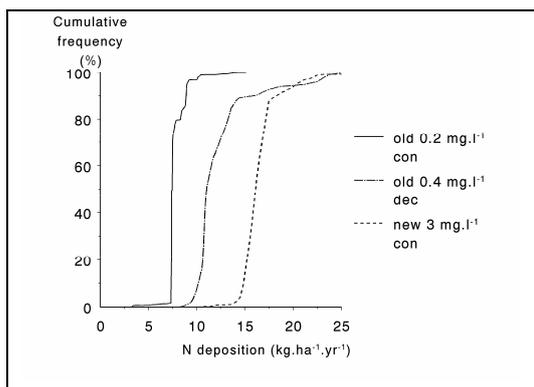


Figure 40 Cumulative frequency distribution of critical loads for N preventing forest nutrient imbalances for coniferous forests (con) and deciduous forest (dec) (right) based on current (old) and updated (new) critical N limits

The SMB-calculations are based on the growth uptake of nitrogen, the long-term immobilisation and denitrification and, the critical NO_3 leaching flux. N uptake is calculated from the product of an average N yield¹ over the rotation period and a minimum N content. The critical nitrogen leaching flux is calculated as the product of the precipitation excess and a critical N concentration. The newly derived critical nitrogen loads for forest nutrient impacts on coniferous forest are substantial higher than critical loads derived by the current values (Figure 40 and Table 23).

Table 23 Critical loads for N forest nutrient imbalances based on current and updated critical N limits

Critical N concentration for forest nutrient impacts (mg.l^{-1})	Critical N loads ($\text{kg N.ha}^{-1}.\text{yr}^{-1}$)					
	Mean	Min	P05	Median	P95	Max
<i>Old</i>						
0.2 mg l^{-1} (coniferous)	7.8	3.3	7.4	7.5	9.0	15.2
0.4 mg l^{-1} (deciduous)	12.3	8.1	9.8	11.1	21.4	27.0
<i>New</i>						
3 mg l^{-1} (coniferous)	16.7	10.1	14.7	16.3	20.5	75.4

4.3 Evaluation of present and updated critical nitrogen limits

A comparison of the currently used critical limits for dissolved N concentrations compared to the findings in this study is given in Table 24. Major differences are:

- The critical limits in view of vegetation changes in the mapping manual vary mostly from 0.2- 2 mg.l^{-1} with an upper value of 5 mg.l^{-1} . These values refer, however, to forest in the Nordic countries and can not simply be used for other countries. Results for the median critical N concentration in soil solution, based on application of the biodiversity impact model NTM in the Netherlands, indicates a range of 0.1- 5 mg.l^{-1} for grass lands and heath lands and of 0.1- 10 mg.l^{-1} for forests, with a median value near 3 mg.l^{-1} when applying a steady state model (see Table 20, Table 22).

¹ Previous calculation for forest nutrient impacts were performed while using optimal growth in stead of average growth.

Table 24 Acceptable N concentrations in soil solution as used in the mapping manual and derived in this study

Impact	Critical N concentration (mg N.l ⁻¹)	
	UN/ECE (2004)	This study
Vegetation changes in Northern Europe ¹		
- Lichens to cranberry (lingonberries)	0.2-0.4	0.2-0.4
- Cranberry to blueberry	0.4-0.6	0.4-0.6
- Blueberry to grass	1-2	1-2
- Grass to herbs	3-5	3-5
Vegetation changes in Western Europe		
- Coniferous forest		2.5-4 ²
- Deciduous forest	-	3.5-6.5 ²
- Grass lands		3 ²
- Heath lands	-	3-6 ²
Other impacts on forests		
- Nutrient imbalances	0.2-0.4 ³	-
- Elevated nitrogen leaching/N saturation	-	1
- Fine root biomass/root length	-	1-3
- Sensitivity to frost and fungal diseases	-	3-5

¹ In the mapping manual, the use of critical N concentrations is not limited to Northern Europe. This study, however, shows that its derivation is limited to forest in Scandinavia (see Annex 9), where empirical critical N loads are generally lower than in the rest of Europe.

² This refers to the range in median values derived with SMART-NTM using a dynamic and a steady state approach. The actual range is much larger and varies often as much as the median value (variation coefficient of 100%). For grassland, the value only refers to the study using a dynamic modelling approach (see text for reasons). The results officially apply only to the Netherlands

³ Actually, the range of 0.2-0.4 mg.l⁻¹ is given for deciduous trees, whereas the value of 0.2 is given for conifers. It is, however, likely that this range also applies for conifers and therefore, both forest types are lumped in this table.

- The critical limits in view of nutrient imbalances in the mapping manual vary from 0.2- 0.4 mg.l⁻¹, based on the assumption that an imbalance occurs as soon as the N leaching increases above natural N leaching rates. There is no clear substantiation for this assumption, neither from the literature nor by the concepts of plant physiology. Nutrient imbalance has been assumed to occur when the availability of base cations (Ca, Mg and K) instead of nitrogen becomes limiting for growth, but this is not clearly related to a critical N concentration in soil solution.
- The manual does not contain critical N concentrations in view of a differentiation between undisturbed and “leaky” N saturated forest sites. An upper limit in this context is 1 mg.l⁻¹.
- Empirical data suggest that critical dissolved N concentration in view of adverse impacts on fine root biomass/root length and an increased sensitivity to frost and fungal diseases varies from 1- 5 mg.l⁻¹, lower values being equal to 1 and 3 mg.l⁻¹. These values are comparable to the median values derived by SMART-MOVE for vegetation changes and to the values in the manual related to the vegetation changes from blueberry to grass and from grass to herbs in the Nordic countries.

The table does not contain the critical limits for ground water. When countries want to use such limits, the EC target for drinking water remained 11.3 mg.l⁻¹ and the target value remains 5.6 mg.l⁻¹.

5 Discussion, conclusions and recommendations

5.1 Discussion

The various aspects that will be discussed in this section are related to: (i) the N variables that are used and can be used to assess impacts on plant species diversity, (ii) the major differences in the various modelling approaches and (iii) the model results, including an intercomparison of the models and a comparison of model results with data (validation). The first two aspects of this discussion are largely based on a background report for a nitrogen workshop in Brighton 2005 by Rowe et al. (2005). For more details on various aspects, the reader is referred to that report.

Relevant nitrogen variables in view of impacts on plant species occurrence

To connect theory on nitrogen dynamics in soil with models of plant species occurrence, a measure of nitrogen exposure, i.e. of plant-available N, is required. There are different measures to integrate N exposure into a single indicator. Some of those indicators give direct information on an N flux to the ecosystem, whereas other indicators only give indirect information based on correlations with fluxes. The discussion presented below is mainly based on (Rowe et al., 2005).

Major direct indicators of N availability are:

- Gross mineralization and nitrification rates: Measurement of gross rates of both mineralization and nitrification reflect the internal cycling of nitrogen and thus potentially the maximum inorganic-N pool available to the plants in competition with microbial uptake. A large ratio of gross nitrification / gross immobilisation has been suggested as indicating ecosystem nitrogen saturation, i.e. excess N over what can be retained by soil and vegetation (Goulding et al., 1998). Measurements of gross fluxes require isotope dilution studies and so may not be practical for wide application (Rowe et al., 2005).
- Measures of N deposition: The N deposition flux does accurately reflect the exposure of species with limited root systems, particularly bryophytes and lichens. In other systems, however, the transformation of nitrogen by soil microorganisms modifies plant exposure and N deposition flux is thus not a complete measure of exposure for plants rooting in soil.

A best indicator might be the sum of N deposition and N mineralization, as used in the SMART2-SUMO-MOVE approach, although indirectly in relationships with a critical Ellenberg N indicator.

The indirect indicators, which are correlated with N availability, can be grouped as follows (Morecroft et al., 2005; Rowe et al., 2005):

- Measures of soil solution N: The soluble nitrogen pool is in principle immediately available to plants. However soil solution only reflects the nitrogen in excess of uptake demands and leaching losses and thus may underestimate total N availability to plants. Concentrations are very dynamic both spatially and

temporally and single measurements of soil solution N concentrations are thus of limited use. Trends, and measures integrated over time such as mean or maximum annual or growing season concentration, are more reliable indicators of N status. Furthermore, species differ in their ability to use different forms of soluble N, i.e. NO_3 , NH_4 and DON and the ratio of ammonium to nitrate in solution may provide information relevant to species occurrence and also the potential for microbial uptake of nitrate. Apart from various other factors, use of a soil solution nitrogen concentration forms the basis of the ForSAFE-VEG model approach

- Measures of soil mineralisable N: Mineralisable N provides information on the net production of inorganic-N. Methods developed to determine the amount of easily mineralisable N are based on measuring soluble N concentrations after incubation under standard conditions. N mineralization is highly variable in space and time, and measurements need to be standardised for soil horizon and season.
- Indicators of soil N reactivity: Many attempts have been to relate N mineralization rate to soil organic matter properties. Large amounts of phenolic compounds, such as lignin, or tannins, inhibit N mineralization. The rate of decomposition of litter, in particular, has been predicted using measures such as the lignin/N ratio or the (lignin + soluble polyphenol)/N ratio.
- Measures of plant chemistry: Increased plant N uptake is likely to lead to increased tissue N concentrations. Tissue concentrations vary considerably among species, with plant part and tissue age/phenological stage, seasonally, interannually, and with nutrient supply, grazing, or other management (e.g. Emmett et al., 2004). Nevertheless, if these factors can be controlled for (e.g. by sampling a standard part, from a single species or group, at a standard time of year) tissue concentrations of N and amino acids may be good indicators of N exposure and in principle could be outputs from biogeochemical models.
- The soil C/N ratio: The organic soil C/N ratio is not a direct control on plant response but represents a readily measurable proxy for important processes (e.g. nitrification/mineralization). In general, the relationship is weak, since total soil N is largely inactive, and is not a good indicator of N availability (Tamm, 1991). It is therefore still useful to measure and to incorporate in models. It forms the basis of the Bern model approach.
- Indicators based on the plant species assemblage: environmental indicators have been defined for European vascular plants and bryophytes (Ellenberg et al., 1992). Mean Ellenberg fertility scores have been shown to be reasonable indicators of soil N availability (Van Dobben, 1993), although the relationship usually shows large variation (Wamelink et al., 2002) and appears to correlate best with annual above-ground biomass production rather than soil nutrient status. Mean scores for these indicators can, however, be used to describe a plant assemblage and for prediction by vegetation type models, as is the case in (GB)MOVE and NTM.

In summary, plants do not respond to a single measurable abiotic variable, and there are some problems with all variables that could potentially be used as input to the vegetation models. Those considered most useful are (see also the conclusions of the

Nitrogen Workshop in Brighton, November 2005): direct measurements of N fluxes, such as N availability (N deposition plus N mineralization), gross N mineralization/immobilisation and N (NH_y and NO_x) deposition, followed by indirect measures such as soil solution N concentrations in the rooting zone, foliar N concentration and soil C/N ratio.

Intercomparison of model results and model validation

Aspects discussed in this section include:

- A comparison of empirical vegetation models based on large-scale vegetation surveys using the Ellenberg approach (MOVE, NTM, GBMOVE) versus a direct relationship (BERN).
- A comparison of the above mentioned empirically oriented approaches versus a process oriented deterministic approach (VEG).
- Need of model testing and application beyond the geographical region for which model dose-response relationships have been parameterised.

Comparison of the NTM/MOVE and Bern model approach

We have made a first comparison of the relations between vegetation types and the model variables that were used as driver as used in NTM/MOVE and the Bern model. The similarity in the approaches used in both models for the relation between vegetation types and a biotic site factors such as acidity and nutrient status opens the way to compare critical limits and find out what causes differences and correspondences in calculated CL.

In order to perform a comparison it was necessary to link the used Dutch nature-target from NTM/MOVE with vegetation types used in the Bern model. This could be done for several types. In order to check the link between both types the optimal F mentioned in the BERN-database was compared with the critical Ellenberg-F from NTM/MOVE (F_zahl stands for feucht or moisture). Both parameters are closely correlated (see Figure 41), $R^2 \sim 0.9$. From this it is concluded that the linkage between the German and Dutch types was satisfying. Given this result the other more relevant parameters (acidity and nutrient status) could be checked.

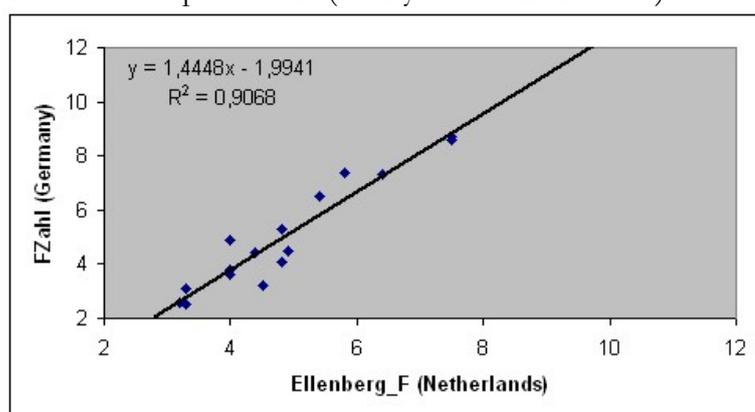


Figure 41 Comparison of the German F_zahl with the Dutch critical Ellenberg_F

The BERN database has information on both the critical pH and BS. In the following graph the critical pH values as used in both methods were compared (see Figure 42). These critical limits are closely correlated ($R^2 > 0.8$).

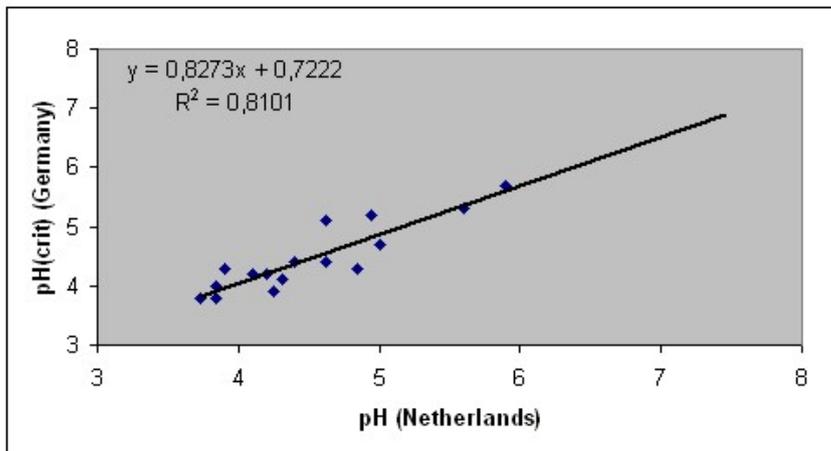


Figure 42 Comparison of the German $pH(crit)$ with the Dutch critical pH

5.2 Conclusions

The potential of linked biogeochemistry-biodiversity modelling approaches

The following conclusions can be drawn regarding the modelling approaches presented in this report:

- Vegetation models, based on either large-scale vegetation surveys (MOVE/GBMOVE, NTM, BERN) or experimental data (VEG), have been developed and integrated with biogeochemical models, such as SMART2 (either or not in connection with SUMO), MAGIC, VSD and ForSafe. Linked biogeochemistry-biodiversity models for nitrogen have great potential for application under the Convention.
- There are large similarities between the models, particularly those based on survey data, but there are also several important differences, including: (i) use of different abiotic variables for N, such as N availability and Ellenberg N indicators in SMART2-MOVE/NTM, soil C/N ratio in Bern and soil solution N in ForSafe, (ii) use of different variables for acidity, such as pH in SMART2-MOVE/NTM and base saturation in Bern, (iii) prediction of individual plant species (e.g. MOVE/GBMOVE) versus plant communities (NTM) and (iv) calibration to different (national) soil and vegetation datasets
- At their current level of development, most integrated models focus primarily on predicting the biodiversity impacts of different emission scenarios. The models, SMART2-MOVE and in potential Bern and FOrSafe-VEG, however, predict biodiversity-based critical loads or target loads and the objective of ongoing research in the future should focus on this application.
- In deriving critical loads or target loads, the definition of reference conditions and damage thresholds for terrestrial biodiversity represents a major challenge, if linked biogeochemical-biodiversity models are to be used for target-setting.

Although the definition of biodiversity targets is an issue for policy-makers, dynamic models can provide valuable information on realistic reference conditions and achievable recovery targets.

- The reliance on Ellenberg Indicator values as a proxy for abiotic conditions in survey-based models such as SMART2-MOVE and MAGIC-GBMOVE adds an additional uncertainty to model predictions. However, Ellenberg values are likely to remain necessary in many areas due to the insufficient coverage of combined vegetation and soil survey data.
- While the models based on survey data have largest potential for country wide mapping of critical loads, the more mechanistic, linked biogeochemical-vegetation models (e.g. FORSAFE-VEG and SMART2-SUMO) can provide more accurate predictions of vegetation change in some ecosystems. Testing and adaptation for other countries/ecosystems are required for larger-scale application.

Model validation status and its comparison with empirical critical loads

The validation status of the various models differs, specifically with respect to the comparisons that have been carried out between measured and changes in plant species composition. In general, the biogeochemical models used (SMART2, MAGIC and ForSafe-veg) have a large validation status. For example, SMART2 has been validated on the Solling plot and hundreds of Intensively monitored forest plots (De Vries et al., 2003a). In this report, we show that MAGIC is able to predict changes in observed N leaching and soil C/N in plot-scale N manipulation studies under different addition levels. We also show that the ForSafe model shows a good correlation simulated and measured values on tree biomass, pools of soil organic C and N, soil pH and BC/Al ratios at 16 Swedish forest sites.

The validation status of the vegetation models is however, much less advanced. The Bern model is not really tested on measured changes in species composition. Biomass growth of SUMO has been validated in a limited way, using data collected at two grassland sites, a heathland site and a forest site in the Netherlands and UK.. regarding (GB)MOVE, a preliminary test was made how successfully GBMOVE in combination with MAGIC could reproduce the observed species composition in test plots, including an unimproved neutral grassland and a blanket bog. Finally, the Veg model in combination with ForSafe has been validated at 16 Swedish forest sites by comparing simulated and measured values on the ground occupancy of the 42 plant groups. Considering the sometimes poor correlations, there is still a need for substantial model validation and model improvement.

To date, the MAGIC-GBMOVE model has not been applied in ‘inverse mode’ to estimate critical loads based on biodiversity targets. The BERN model has also not been applied in ‘inverse mode’ but the critical N load can be calculated at steady state conditions by an iterative approximation. This function can be derived with long simulation runs (>100 years), with available biogeochemical models like VSD, SMART2 and ForSAFE. Similarly, the ForSafe-veg does not run in an inverse mode to derive critical loads. Instead the “critical load” is determined to be passed at the time one can observe unwanted significant shifts in vegetation composition,

abundance or the entry/departure of plant groups. This time is used for estimating the deposition of nitrogen at the point in time of significant unwanted vegetation change. Actually, this value is dependent on the site history and can thus better be called a target load

Unlike the previous models, the SMART2-MOVE model can be used in an inverse mode to derive critical N loads. Both models have thus been applied and results were compared to empirical critical N loads. Results of a SMART2-MOVE application for the Netherlands leads to average critical N load for forests and short vegetation (grassland and heathland) that ranges from approximately 14-25 kg N.ha⁻¹.yr⁻¹. This is comparable with empirical CLs (10 - 25 kg N.ha⁻¹.yr⁻¹; Bobbink et al., (2003).

Critical limits for dissolved nitrogen concentrations

A review of currently used critical limits for dissolved N concentrations shows that:

- The critical limits in view of vegetation changes in the mapping manual refer to forest in the Nordic countries and may not be applicable for other countries. Results for the median critical N concentration in soil solution based on application of the biodiversity impact model NTM in the Netherlands indicates higher levels. The concentrations are comparable to the values in the manual related to the vegetation changes from grass to herbs in the Nordic countries.
- The very low critical limits in view of nutrient imbalances in the mapping manual are based on the assumption that an imbalance occurs as soon as the N leaching increases above natural N leaching rates. There is no clear substantiation for this assumption, neither from the literature nor by the concepts of plant physiology.
- The manual does not contain critical N concentrations in view of a differentiation between undisturbed and “leaky” N saturated forest sites. An upper limit in this context is 1 mg.l⁻¹.
- Empirical data suggest that critical dissolved N concentration in view of adverse impacts on fine root biomass/root length and an increased sensitivity to frost and fungal diseases varies from 1- 5 mg.l⁻¹.

5.3 Recommendations

Measurements of plant species response to environmental variables

Plant species responses to environmental variables form the basis of all developed models and is also widely used to estimate abiotic conditions. The latter gives the opportunity to even estimate abiotic conditions in the past for sites where only the vegetation composition is known. One of the most frequent used systems for this is the Ellenberg indicator system (Ellenberg et al., 1992) or deviations from it (Diekmann, 2003). However the last years it has become clear that there are some major disadvantages using this system; the uncertainty is quite large, it is sometimes unclear what the indicator values represent and they seemed to be biased (Ertsen et al., 1998; Schaffers & Sykora, 2000; Wamelink et al., 2002; Wamelink et al., 2003b; Witte & von Asmuth, 2003; Smart & Scott, 2004; Wamelink et al., 2004). Furthermore, it requires a translation from indicator values into physical units.

The most logical solution to avoid the highly uncertain translation is to replace the expert system by a system based on physical measurements (Wamelink et al., 2002). On a small scale this was carried out for France (Gégout et al., 2003) and the Netherlands (Wamelink et al., 2005). From the results from Wamelink et al. (In prep.) it becomes clear that this yields promising results, but that for a wider application European wide data are needed, instead of only regional data. Data need to be collected in Europe and stored in a database. The data should consist of a vegetation description (containing the species in a certain area; relevés) and at least one of the following measured variables: soil acidity, nutrient status, water availability, or climatical conditions. Based on this, plant species response per abiotic variable can be estimated, ecologically reviewed and then tested on independent datasets. The hypothesis that plant species do have different responses in different regions can then be investigated as well as the hypothesis that species may have different responses in different vegetation types.

In general, the development and testing of both biogeochemical and biodiversity impacts models is critically dependent on long-term monitoring, long-term experimental, and large-scale survey data. The continuation of existing programmes, where possible with improved integration of biotic and abiotic measurements, is essential to the future development of this work.

Priorities for further development and testing/validation of models

The recommendations on the priorities for further development and testing/validation of models are mainly based on the conclusions of the Nitrogen Workshop in Brighton, November 2005. Priorities for future work include:

- Consideration of the relative risk of nitrate leaching under ammonium and nitrate dominated deposition.
- Improved simulation of the links between carbon and nitrogen cycles.
- Incorporation of the effects of climate drivers within the models.
- Continued testing of all models, and model inter-comparison studies

Priorities for future work on modelling nitrogen impacts on biodiversity include:

- The collection of new data to identify and verify the most suitable abiotic N variables for predicting plant response
- Prediction of rare species
- Representation of lag times (e.g. due to species persistence, dispersal)
- Incorporation of feedbacks with biogeochemical models (e.g. changes in litter quality due to species change)
- Consideration of the differential effects of oxidised and reduced nitrogen.

Priorities for testing and validation of the models

- Testing and comparison of different models at the same sites, particularly against long-term datasets.
- Adaptation, testing and upscaling of impacts models beyond the biogeographical regions for which model dose-response relationships have been parameterised particularly areas not included in current model coverage, such as Mediterranean and Alpine regions, and Eastern Europe)

Literature

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Annex 1 Calculation of critical loads for nitrogen with the SMART2 steady state model

The availability of nitrogen (N) for plant growth is a key parameter, e.g., in the MOVE and NTM models (see Chapter 4). Starting from a simple definition of N availability, we derive general expressions for a critical load of N, using a critical N availability as criterion. The SMART2 model is used to illustrate the general concept.

General concept:

We *define* nitrogen availability, N_{av} , as the sum of N supplied by the deposition of N, $N_{dep} = NO_{x,dep} + NH_{3,dep}$, and the N mineralised in the soil, N_{mi} (all in $eq.m^{-2}.yr^{-1}$):

$$N_{av} = N_{dep} + N_{mi} \quad (A1.1)$$

If the N mineralization flux is known, and a critical N availability, $N_{av,crit}$, is specified, a critical load of N, $CL_{av}(N)$, is immediately obtained from Eq. A1.1:

$$CL_{av}(N) = N_{av,crit} - N_{mi} \quad (A1.2)$$

Eq. A1.2 assumes that mineralization is a constant, i.e. does not depend on the deposition of N. This, however, is a quite crude assumption, since one could expect that high N deposition causes an increased mineralization of N. In the simplest case, N mineralization is a linear function of deposition, i.e.

$$N_{mi} = c_{mi} \cdot N_{dep} + N_{mi,0} \quad (A1.3)$$

where $N_{mi,0}$ is the mineralization flux at zero deposition (“minimal nutrient cycle”) and c_{mi} is a dimensionless constant. Inserting Eq. A1.3 into Eq. A1.1 yields then for the critical load:

$$CL_{av}(N) = \frac{N_{av,crit} - N_{mi,0}}{1 + c_{mi}} \quad (A1.4)$$

In a further refinement one could assume that mineralization has also an upper limit. One way to model this is by a piece-wise linear function: Below (above) a minimum (maximum) deposition ($N_{dep,mn}$ and $N_{dep,mx}$, resp.) N mineralization is constant ($N_{mi,mn}$ and $N_{mi,mx}$, resp.), and between these two values it depends linearly on N_{dep} :

$$N_{mi} = \begin{cases} N_{mi,mn} & \text{for } N_{dep} \leq N_{dep,mn} \\ a \cdot (N_{dep} - N_{dep,mn}) + N_{mi,mn} & \text{for } N_{dep,mn} < N_{dep} < N_{dep,mx} \\ N_{mi,mx} & \text{for } N_{dep} \geq N_{dep,mx} \end{cases} \quad (A1.5)$$

with

$$a = \frac{N_{mi,mx} - N_{mi,mn}}{N_{dep,mx} - N_{dep,mn}} \quad (A1.6)$$

Inserting Eq. A5 into Eq. A1 and specifying $N_{av,crit}$ the critical load can be computed: For $N_{av,crit} < N_{mi,mn}$ no positive solution exists (negative CL!); otherwise one obtains (see Appendix A):

$$CL_{av}(N) = \begin{cases} N_{av,crit} - N_{mi,mn} & \text{for } N_{mi,mn} \leq N_{av,crit} \leq N_{mn} \\ b \cdot (N_{av,crit} - N_{mn}) + N_{dep,mn} & \text{for } N_{mn} < N_{av,crit} < N_{mx} \\ N_{av,crit} - N_{mi,mx} & \text{for } N_{av,crit} \geq N_{mx} \end{cases} \quad (A1.7)$$

where we have introduced the following abbreviations:

$$N_{mn} = N_{mi,mn} + N_{dep,mn} \quad \text{and} \quad N_{mx} = N_{mi,mx} + N_{dep,mx} \quad (A1.8)$$

and

$$b = \frac{1}{1+a} = \frac{N_{dep,mx} - N_{dep,mn}}{N_{mx} - N_{mn}} \quad (A1.9)$$

In addition to specifying $N_{av,crit}$ four parameters are needed to calculate $CL_{av}(N)$, i.e. the four parameters $N_{mi,mn}$, $N_{mi,mx}$, $N_{dep,mn}$, $N_{dep,mx}$ defining the deposition-dependence of N mineralization.

Application to the SMART2 model:

N mineralization is not a variable that is easily determined or measured. Thus modelling is used to derive that variable from more basic and easily measurable quantities. Here we present how this is done in the SMART2 model (Kros, 2002). Starting point is the mass balance for the amount of N in the soil, AmN (in eq.m⁻²):

$$\frac{d}{dt} AmN = N_{lf} + N_{rd} - N_{mi} \quad (A1.10)$$

i.e. the change in the amount of N in the soil is due to the input of N through litter fall, N_{lf} and the decay of roots, N_{rd} minus the loss of N due to mineralization. At steady state this amount does not change over time, and thus we obtain for N_{mi} :

$$N_{mi} = N_{lf} + N_{rd} \quad (A1.11)$$

Obviously, the N from litter fall and root decay is obtained by multiplying the respective organic matter fluxes OM_{lf} and OM_{rd} (in kg.m⁻².yr⁻¹) with the corresponding N contents ctN_{lf} and ctN_{rd} (in eq.kg⁻¹):

$$N_{lf} = OM_{lf} \cdot ctN_{lf} \quad \text{and} \quad N_{rd} = OM_{rd} \cdot ctN_{rd} \quad (\text{A1.12})$$

In SMART2, the second equation is not used, but the decay (turnover) of N in the roots is coupled to the N in litter fall by:

$$N_{rd} = ncf \cdot N_{lf} \quad (\text{A1.13})$$

where ncf is the so-called nutrient cycling factor, a constant defined as the ratio of the root turnover (related to N) and the litter fall. Furthermore, the content of N in litter is proportional to the content of N in leaves prior to litter fall, ctN_{lv} , i.e.:

$$ctN_{lf} = (1 - f_{re}) \cdot ctN_{lv} \quad (\text{A1.14})$$

where f_{re} is the re-allocation fraction of N in leaves prior to litter fall. A further assumption is that not all N from root decay is directly available, but only the N from decaying roots in the litter layer, $N_{rd,lt}$:

$$N_{rd,lt} = f_{rt,lt} \cdot N_{rd} \quad (\text{A1.15})$$

where $f_{rt,lt}$ is the fraction of roots in the litter layer. Using this $N_{rd,lt}$ instead of N_{rd} , collecting all terms and inserting them into Eq. A1.11 yields for the N mineralization flux at steady state according to the SMART2 model:

$$N_{mi} = A \cdot ctN_{lv} \quad \text{with} \quad A = (1 - f_{re}) \cdot (1 + f_{rt,lt} \cdot ncf) \cdot OM_{lf} \quad (\text{A1.16})$$

In SMART2 the contents of N in leaves depends on the deposition of N, and this is modelled as a piece-wise linear function. Below (above) a minimum (maximum) deposition ($N_{dep,mn}$ and $N_{dep,mx}$, resp.) the N content is constant ($ctN_{lv,mn}$ and $ctN_{lv,mx}$, resp.), and between these two values it depends linearly on N_{dep} :

$$ctN_{lv} = \begin{cases} ctN_{lv,mn} & \text{for } N_{dep} \leq N_{dep,mn} \\ \alpha \cdot (N_{dep} - N_{dep,mn}) + ctN_{lv,mn} & \text{for } N_{dep,mn} < N_{dep} < N_{dep,mx} \\ ctN_{lv,mx} & \text{for } N_{dep} \geq N_{dep,mx} \end{cases} \quad (\text{A1.17})$$

with

$$\alpha = \frac{ctN_{lv,mx} - ctN_{lv,mn}}{N_{dep,mx} - N_{dep,mn}} \quad (\text{A1.18})$$

Multiplying this with A (defined in Eq. A1.16) gives the deposition-dependence of N_{mi} with the following parameter values:

$$N_{mi,mn} = A \cdot ctN_{lv,mn}, \quad N_{mi,mx} = A \cdot ctN_{lv,mx} \quad \text{and} \quad a = A \cdot \alpha \quad (\text{A1.19})$$

Since this yields exactly the functional form of Eqs. A1.5-6, the critical load with respect to N availability, $CL_{av}(N)$, is given by Eqs. A1.7-9.

In addition to specifying $N_{av,crit}$, eight parameters are needed to calculate $CL_{av}(N)$ according to the SMART2 model: (i) 4 parameters $ctN_{lv,mm}$, $ctN_{lv,mx}$, $N_{dep,mm}$, $N_{dep,mx}$ defining the deposition-dependence of the N content of leaves, (ii) the litter fall flux at steady state, OM_{lp} (iii) the fraction of N re-allocated in leaves prior to litter fall, f_{re} , and (iv) 2 parameters ncf , $fr_{n,lb}$ defining the root decay in the litter layer.

Appendix A to Annex 1:

Here we show how to obtain positive solutions x as a function of the positive parameter d of the equation:

$$d = kx + plf(x) \quad (AA1)$$

where $k > 0$ is a constant and plf is a piece-wise linear function defined as:

$$plf(x) = \begin{cases} y_1 & \text{for } x \leq x_1 \\ a(x - x_1) + y_1 & \text{for } x_1 < x < x_2 \\ y_2 & \text{for } x \geq x_2 \end{cases} \quad \text{with } a = \frac{y_2 - y_1}{x_2 - x_1} \quad (AA2)$$

where the constants fulfil the conditions $0 \leq x_1 < x_2$ and $0 \leq y_1 \leq y_2$.

Geometrically speaking, a solution of Eq. AA1 is the intersection of the straight line $y = -kx + d$ with the function $y = plf(x)$. This is illustrated in Figure AA1. Since k is positive, the straight line is always tilted to the left. Varying d – the value of the straight line at the intersection with the vertical axis – shifts the straight line vertically up and down, and depending on its value, three cases can be distinguished (see Figure AA1).

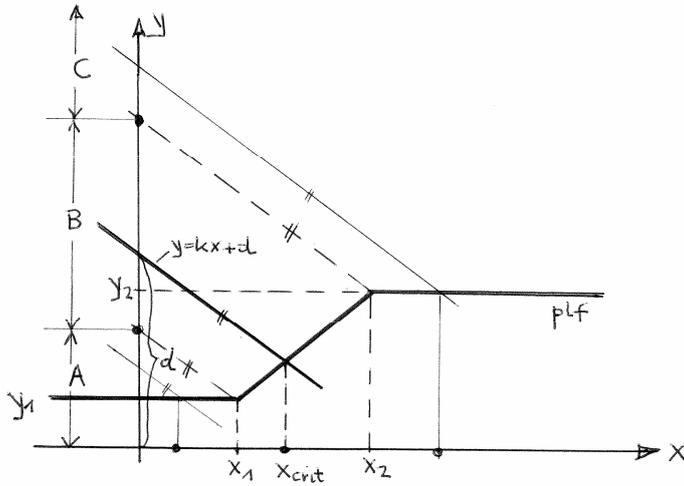


Figure AA1: Schematic diagram illustrating the cases for solving Eq. AA1 for varying d .

Case A: In this case the straight line intersects with the lower constant plateau of plf . A simple geometric consideration shows that this is the case for $d \leq y_1 + kx_1$; and the x -value of the intersection, call it x_{crit} , is obtained as

$$x_{crit} = (d - y_1) / k \quad (AA3)$$

Case B: In this case the straight line intersects with the linear part of plf . Same geometric argument as above shows that this is the case for $y_1 + kx_1 < d < y_2 + kx_2$. And the solution is obtained as the x -value of intersection of the two straight lines, $y = -kx + d$ and $y = a(x - x_1) + y_1$, as:

$$x_{crit} = \frac{d + ax_1 - y_1}{a + k} = b(d - y_1 - kx_1) + x_1 \quad \text{with} \quad b = \frac{x_2 - x_1}{y_2 + kx_2 - y_1 - kx_1} \quad (AA4)$$

Case C: The straight line intersects with the upper constant plateau of plf . This is the case for $d \geq y_2 + kx_2$ and the solution is given by replacing y_1 with y_2 in Eq. AA3.

Summarising the cases, the solution of Eq. AA1 as function of d is given as:

$$x_{crit} = \begin{cases} (d - y_1) / k & \text{for } d \leq y_1 + kx_1 \\ b(d - y_1 - kx_1) + x_1 & \text{for } y_1 + kx_1 < d < y_2 + kx_2 \\ (d - y_2) / k & \text{for } d \geq y_2 + kx_2 \end{cases} \quad (AA5)$$

Note that for $d < y_1$ the solution x_{crit} is negative.

Setting $k = 1$ and identifying d with $N_{av,crit}$, x_1/x_2 with $N_{dep,mn}/N_{dep,mx}$, and y_1/y_2 with $N_{mi,mn}/N_{mi,mx}$, the expression for the critical load $CL_w(N)$ in Annex 1 is obtained from Eq. AA5

Annex 2 Description and validation of the SUMO model predicting vegetation succession

SUMO

The model SUMO (Succesion MOdel) is a process model that was developed to simulate vegetation responses in a changing environment. In combination with SMART2 this model is able to evaluate the effects of management and nitrogen deposition on between plant species competition and on the interaction between soil and vegetation. The simulation of vegetation succession on a regional scale brings limitations. The model has to be able to simulate many cells for periods of ca 25 years within a reasonable amount of time. This limits the number and the detail of the processes in the model. The availability of data for the input is also limited; therefore simulations on a regional scale are done for cells of 250 by 250m with a time step of one year. SUMO was built as an extension to the already existing model SMART2 (Kros, 2002). SUMO is based on the same principles as the much more elaborate and plant species specific model NUCOM (Van Oene et al., 1999a; Van Oene et al., 1999b). First a short general description of SUMO is given and its links to SMART2, followed by a detailed description of each process, including its parameterisation. The description is based on Wamelink et al. (In prep.).

Model description

SUMO simulates the biomass and nitrogen dynamics for five functional types; grasses/herbs, dwarf shrubs, shrubs, pioneer trees, and climax trees. Each functional type is assumed to consist of three organs: root, stem, and leaf. The time step of the model is one year. In each time step the biomass of the five functional types is computed, based on the biomass in the previous time step, biomass growth and death in the present time step, and removal of biomass by management (Equation A2.1).

$$B_{i,j,t} = B_{i,j,t-1} + (A_{act,i,j,t} - D_{i,j,t} - M_t + B_{s,i,j,t}) \quad (A2.1)$$

$B_{i,j,t}$	= biomass (kg.ha ⁻¹)
$A_{act,s,i,j,t}$	= actual growth (kg.ha ⁻¹ .yr ⁻¹)
$D_{i,j,t}$	= biomass mortality (kg.ha ⁻¹ .yr ⁻¹)
M_t	= biomass removal by management (kg.ha ⁻¹ .yr ⁻¹)
$B_{s,i,j,t}$	= input seed biomass (kg.ha ⁻¹ .yr ⁻¹)

For all Equations: t: time step, j: vegetation type, i: functional type, o: organ, g: tree species

Overall description of biomass

The growth is in turn calculated on the basis of an assumed maximum growth, which is reduced by nitrogen availability (provided by SMART2) and water availability (provided by WATBAL) and light interception (Equation A2.2). The dead biomass

(amount of litter fall and corresponding nitrogen content) is returned to the relevant pools in SMART2.

Overall description of growth

$$Aact_{i,j,t} = Amax_{i,j} \cdot RI_{i,j,t} \cdot RNav_{i,j,t} \cdot RWav_{i,j,t} \quad (A2.2)$$

$Aact_{i,j,t}$ = actual growth ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Amax_{i,j}$ = maximum growth ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $RI_{i,j,t}$ = Reduction factor for light availability
 $RNav_{i,j,t}$ = Reduction factor for nitrogen availability
 $RWav_{i,j,t}$ = Reduction factor for water availability

SUMO distinguishes six vegetation types (grassland, heathland, reedland, shrub vegetation, salt marsh and forest). The growth functions are parameterised for each combination of functional type and vegetation type and may therefore differ per combination. In this way differences in species composition per vegetation type can be accounted for. Much attention is given to the simulation of competition between the functional types. The competition for nitrogen and light is assumed to be the driving force for succession. The initial vegetation type has to be derived from prior knowledge. Apart from biomass growth, SUMO also simulates height growth.

For the functional types herbs/grasses, dwarf shrubs, and shrubs, SUMO simulates the total biomass of all species. For the functional types pioneer tree and climax tree the biomass for a specific tree species is simulated, and a different parameterisation is used for each species. The pool of tree species consists of Scots pine (*Pinus sylvestris*), larch (*Larix decidua*), Douglas fir (*Pseudotsuga menziesii*), Norway spruce (*Picea abies*), birch (*Betula pendula* and *Betula pubescens*), ash (*Fraxinus excelsior*), alder (*Alnus glutinosa*), willow (*Salix alba* and *Salix cinerea*), poplar (*Populus spec.*), oak (*Quercus robur* and *Quercus petraea*), northern red oak (*Quercus rubra*) and beech (*Fagus sylvatica*). The two major tree species that are simulated for a site are either given as input or selected from the pool when for instance succession to forest is simulated. In that case the two tree species are selected based on the simulated soil variables.

Only carbon and nitrogen fluxes are simulated by SUMO. Nutrient uptake by the vegetation and litterfall (including the amount of dead roots and dead wood) are provided by SUMO2 to SMART2, whereas SMART2 delivers the nitrogen availability to SUMO2 as the sum of external N input and mineralization. The nitrogen that becomes available through mineralization (simulated by SMART2) and atmospheric deposition is partitioned over the functional types and within each functional type over its organs, using fixed percentage distributions per functional type / vegetation type combination. Nitrogen reallocation before litterfall is also simulated.

The vegetation type can change during a model run. When the management (mowing) of grassland is stopped, succession to heathland or forest may occur. The vegetation type is determined on the basis of the biomass present in the five

functional types. In grasslands the vegetation type changes from grassland into forest, if the stem biomass of the functional types shrubs, pioneer trees or climax trees exceeds a threshold value (Table A2.1). The pioneer tree, the climax tree and the forest type are selected on the basis of the soil type and the groundwater level present at the specific grid (Table A2.1). Tree species specific parameters are then used instead of the common tree parameters. Table A2.1 gives a scheme of all possible succession pathways and the conditions for succession.

Table A2.1 Succession scheme for vegetation types with conditions under which succession takes place (msl = mean spring groundwater level in m below surface, SP = sand poor, SR = sand rich, SC = sand calcareous, CN = clay non-calcareous, CC = clay calcareous, PN = peat non calcareous and LN = loess non-calcareous), and tree species in the new vegetation. Succession will take place when the total stem biomass of the shrubs and trees is above 150 kg.ha⁻¹.

original type	new type	soil type	msl (-m)	pioneer tree	climax tree
grassland	heathland				
grassland	natural forest	SP, SR, CN, CC, LN	< 0.3	alder	Ash
	pine forest	SP	≥ 0.3	birch	Pine
	natural forest	SR, LN	≥ 0.3	oak	Beech
	natural forest	SC		oak	Beech
	natural forest	CN, CC	≥ 0.3	alder	Poplar
	natural forest	PN	< 0.3	alder	Birch
	natural forest	PN	≥ 0.3	alder	Ash
heathland	natural forest	SP, CN, CC, LN	< 0.3	alder	Ash
	pine forest	SP, SR	≥ 0.3	birch	Pine
	natural forest	SR	< 0.3	birch	Oak
	natural forest	SC		oak	Beech
	natural forest	PN	< 0.3	alder	Birch
	natural forest	PN	≥ 0.3	alder	Ash
	natural forest	LN	≥ 0.3	oak	Beech
reedland	natural forest			alder	Ash
shrubland	natural forest			birch	Oak

The processes modelled in SUMO are based on the descriptions made by Berendse (1994a; 1994b), and are extensively described below.

Competition for light

The interception of light is simulated assuming an exponential decrease of light intensity with decreasing height within the canopy, using the Lambert-Beer equation (Equation A2.3, cf. Huisman & Weissing, 1994).

$$RI_{i,j,t} = \sum_{c=1,5} \left[fl_{i,t,c-1} \cdot \left(1 - e^{(-k_{i,j} \cdot fBl_{i,t,c})} \right) \right] \quad (A2.3)$$

- RI_{i,j,t} = reduction factor for light interception (-)
- f_{i,t,c-1} = fraction light remaining after interception in block b-1 and entering canopy layer c (-)
- k_{i,j} = interception factor (-)
- fBl_{i,t,c} = fraction leaf biomass of functional type i in canopy layer c (kg.ha⁻¹.yr⁻¹)
- n = number of functional types present in the canopy layer (-)

The light interception per functional type is based on the biomass and position of the leaves. For each functional type the leaves are considered to be equally distributed over the height of the functional type, except for trees higher than 7 m, where the canopy does not start at the ground level but at a height of 1.5 m. In total five canopy layers are distinguished in which at least one and at most five functional types are present. The highest layer starts at the top of the highest functional type and ends at the height of the second highest functional type. It contains a fraction of the leaf biomass of the highest functional type proportional to the thickness of the layer relative to the height of the functional type. The light that is not intercepted is passed to the next layer. The second layer consists of two functional types, the highest and the second highest, and this layer ends at the height of the third highest functional type. The leaf biomass of the two functional types in this layer is again computed, and added to give the total leaf biomass for the layer. The light interception computed on the basis of this total biomass is subsequently divided over the two functional types according to their proportion in the total leaf biomass in that layer (Equation A2.3). The light interception of the functional types in the other three layers is calculated in a similar way. Per functional type the interception of the layers is summed to give the total light interception. The maximum growth of the functional type is reduced by light interception according to Equation A2.2.

Competition for nitrogen

In SUMO, nitrogen comes from three sources: (1) root uptake from the soil by root uptake, (2) foliar uptake from the atmosphere by uptake and (3) internal reallocation from one organ to another. The nitrogen availability in the soil compartment (including the organic layer) is simulated by SMART2. The root uptake of soil nitrogen is calculated by SUMO, while taking into account the total foliar uptake. Nitrogen that is taken up by the roots of each functional type is assumed to be equal to the proportion of its root biomass in the total root biomass. Foliar uptake of atmospheric nitrogen for the whole canopy is calculated by SMART2, whereas the amount of nitrogen taken up by the canopy is distributed over the functional types by SUMO. For the sake of simplicity the deposition is distributed over the functional types proportional to their light interception (Equation A2.4).

$$N_{int_{i,j,t}} = \left(\frac{RI_{int_{i,j,t}}}{\sum_{i=1,5} RI_{int_{i,j,t}}} \right) \cdot N_{dep_t} \quad (A2.4)$$

$N_{int_{i,j,t}}$ = intercepted nitrogen deposition ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
 $RI_{int_{i,j,t}}$ = light interception (-)
 N_{dep_t} = nitrogen deposition ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)

Data on atmospheric deposition comes from external sources (i.e. deposition models or maps). The actual available atmospheric nitrogen for the vegetation is calculated by SMART2 and depends on the structure of the vegetation (forests catch more deposition than grasslands).

Before litterfall part of the nitrogen in the litter is reallocated, stored and used for growth in the next year (see below). This reallocated nitrogen remains in the functional type and is therefore not available to the other functional types. The total nitrogen availability for each functional type is calculated by summing Eq. (A2.1), (A2.2) and (A2.3), according to:

$$Nav_{i,j,t} = Nint_{i,j,t} + \sum_{o=1,3} (Nrea_{o,i,j,t-1}) + \left\{ \left(\frac{Br_{i,j,t}}{\sum_{i=1,5} Br_{i,j,t}} \right) \cdot Ns_t \right\} \quad (A2.5)$$

$Nav_{i,j,t}$ = nitrogen availability ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Nint_{i,j,t}$ = intercepted nitrogen deposition ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Nrea_{o,i,j,t-1}$ = nitrogen reallocation ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Br_{i,j,t}$ = root biomass ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 Ns_t = nitrogen release from the soil ($kg \cdot ha^{-1} \cdot yr^{-1}$)

The influence of the nitrogen availability on the growth of each functional type is described by a saturation equation based on potential growth, total nitrogen availability, and the minimum nitrogen content per functional type (Equation A2.6). In principle, all available nitrogen is taken up, but the nitrogen uptake of each functional type is limited by its maximum growth and maximum nitrogen content. The nitrogen that is not taken up by the roots remains in the soil.

$$RNav_{i,j,t} = \frac{Nav_{i,j,t}}{Nav_{i,j,t} + (Apot_{i,j,t} \cdot Nmin_{i,j})} \quad (A2.6)$$

$RNav_{i,j,t}$ = biomass growth ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Nav_{i,j,t}$ = nitrogen availability ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Apot_{i,j,t}$ = potential growth; $Apot_{i,j,t} = Amax_{i,j} \cdot RIav_{i,j,t} \cdot RWav_{i,j,t}$ (Equation A2.2), $kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Nmin_{i,j}$ = minimum nitrogen content (-)

The newly taken up nitrogen is divided over the organs (Equation A2.7). The nitrogen content in the organs is calculated after adding the new biomass and new nitrogen, and subtracting litterfall and the nitrogen left in it (Equation A2.8 and A2.9). Part of the nitrogen of the dead biomass is reallocated and is added to the available nitrogen in the next year. The amount of reallocated nitrogen depends on the organ and the nitrogen content of the organ. When the nitrogen content is lower than a threshold value no nitrogen is reallocated (Equation A2.10).

New nitrogen mass functional type

$$Nnew_{o,i,j,t} = Nav_{i,j,t} \cdot fN_{o,i,j} \quad (A2.7)$$

$N_{new_{o,ij,t}}$ = annual nitrogen mass change ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $Nav_{i,j,t}$ = nitrogen availability; $Nav_{i,j,t} = RNav_{i,j,t} \cdot Ared_{i,j,t}$ (Equation A2.12, $kg \cdot ha^{-1} \cdot yr^{-1}$)
 $fN_{o,ij}$ = fraction for the partitioning of nitrogen (-)

Nitrogen mass organs

$$N_{o,ij,t} = N_{o,ij,t-1} - (N_{o,ij,t-1} \cdot fd_{o,ij}) + N_{new_{o,ij,t}} \quad (A2.8)$$

$N_{o,ij,t}$ = nitrogen mass ($kg \cdot ha^{-1}$)
 $fd_{o,ij}$ = biomass mortality fraction (-)
 $N_{new_{o,ij,t}}$ = new nitrogen mass ($kg \cdot ha^{-1} \cdot yr^{-1}$)

Nitrogen content organs

$$NC_{o,ij,t} = \frac{N_{o,ij,t}}{B_{o,ij,t}} \quad (A2.9)$$

$NC_{o,ij,t}$ = nitrogen content (-)

Nitrogen content in litter fall, dead wood and dead roots (i.e. input for SMART2)

$$NCd_t = \frac{\sum_{i=1,5} \left\langle \sum_{o=1,3} \{N_{o,ij,t} - (frea_o \cdot N_{o,ij,t})\} \right\rangle}{\sum_{i=1,5} \left\langle \sum_{o=1,3} (B_{o,ij,t} \cdot fd_{o,ij}) \right\rangle} \quad (A2.10)$$

NCd_t = nitrogen content in litter fall (-)
 $frea_o$ = nitrogen reallocation factor (-)
 $N_{o,ij,t}$ = nitrogen mass ($kg \cdot ha^{-1}$)
 $B_{o,ij,t}$ = biomass ($kg \cdot ha^{-1} \cdot yr^{-1}$)
 $fd_{o,ij}$ = biomass mortality fraction (-)

In SUMO the effect of moisture availability on the vegetation is neglected. It is assumed that no water shortage occurs that limits the growth and that the growth is not reduced due to high groundwater levels.

Biomass

The biomass of each functional type is computed as the result of the biomass in the previous year, the newly grown biomass, the production of dead biomass and the amount of biomass removed by management (Equation A2.1). The newly grown biomass is the result of the reduction of the maximum growth of the functional type by the reduction factors for light interception (Equation A2.3), nitrogen availability (Equation A2.6) and moisture (Equation A2.11- A2.13).

Biomass partitioning over the organs

$$B_{o,i,j,t} = B_{i,j,t} \cdot fB_{o,i,j} \quad (\text{A2.11})$$

$fB_{o,i,j}$ = fraction for the distribution of biomass over the three organs (-)

Dead biomass

$$D_{i,j,t} = \sum_{o=1,3} (B_{o,i,j,t-1} \cdot fd_{o,i,j}) \quad (\text{A2.12})$$

$D_{i,j,t}$ = dead biomass ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)

Height of grasses and herbs

$$H_{i=(gh,d),t} = k \cdot B_{i=(gh,d),t} \quad (\text{A2.13})$$

$H_{i,t}$ = height (m)
 gh = grasses/herbs
 d = dwarf shrubs
 k = regression coefficient; fixed at 0.1 (-)
 $B_{i,t}$ = biomass ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)

The total biomass can also be reduced by the effects of management (see below). Each year, a small amount of biomass is added to each organ of each functional type to simulate seed input ($1 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$). For several processes in SUMO the amount of biomass per organ is required. To this end the newly formed biomass is divided over the organs according Equation A2.11, where the division over the tree organs differs per functional type. As for the total biomass the biomass per organ is corrected for death and management effects.

Litter

Each year part of the biomass dies. The fraction that dies depends on the organ and the functional type, and varies from 1.0 for leaves of herbs, shrubs and deciduous trees to 0.01 for stems of climax trees (Equation A2.12). The nitrogen content of the dead organs is calculated with Equation A2.10. The reallocation of nitrogen gives lower nitrogen content in litter and dead roots, except when the nitrogen content drops below a given threshold value. The biomass of dead roots and leaves is transferred to the litter pool and nitrogen release from the dead plant parts is calculated by SMART2. SMART2 assumes that dead stems do not release nitrogen.

Height

The height of the five functional types is calculated each year. As height growth is assumed to decline with age, SUMO also keeps track of the 'age' (i.e., the number of years since colonisation or plantation occurred) per functional type. For the

functional types herbs/grasses and dwarf shrubs the height is based on the biomass present in the functional type (Equation A2.13).

The height of shrubs is calculated with Equation A2.14. It depends on the age of the stand. Over the years the height growth is decreasing until the maximum height has been reached. The computation of the height of the trees is more complicated. It is based on the height in the previous year, the biomass growth in the current year, a minimum and maximum height growth, and the ‘age’ of the functional type (Equations A2.15 and A2.16).

Height of shrubs

$$H_{i=s,t} = H_{\max} + (-k_{s1} \cdot k_{s2}^{\text{age}_{s,t}}) \quad (\text{A2.14})$$

$H_{i,t}$	= height (m)
H_{\max}	= maximum height; fixed at 3.0 (m)
s	= shrubs
k_{s1}	= regression coefficient 1 (-)
k_{s2}	= regression coefficient 2 (-)
$\text{age}_{s,t}$	= age shrubs (yr)

Minimum height trees

$$H_{\min_{t,g}} = k_{1,g} + (-k_{2,g} \cdot k_{3,g}^{\text{age}_{g,t}}) \quad (\text{A2.15})$$

$H_{\min_{t,g}}$	= minimum height (m)
$k_{1,g}$	= regression coefficient 1 (-)
$k_{2,g}$	= regression coefficient 2 (-)
$k_{3,g}$	= regression coefficient 3 (-)
$\text{age}_{g,t}$	= age trees (yr)

Maximum height trees

$$H_{\max_{t,g}} = k_{4,g} + (-k_{5,g} \cdot k_{6,g}^{\text{age}_{g,t}}) \quad (\text{A2.16})$$

$H_{\max_{t,g}}$	= maximum height (m)
$k_{4,g}$	= regression coefficient 1 (-)
$k_{5,g}$	= regression coefficient 2 (-)
$k_{6,g}$	= regression coefficient 3 (-)
$\text{age}_{g,t}$	= age trees (yr)

The maximum and minimum height growth of the trees is based on growth curves for the Netherlands on rich and poor soil, respectively (Jansen et al., 1996). The realised height growth per tree species per year lies between values determined by the growth curves at the tree’s ‘age’ and the biomass increment in that year, according to Equation A2.17.

$$\text{Hact}_{g,t} = \text{H}_{g,t-1} + \text{Hgrmin}_{g,t} + \left\{ (\text{Hgrmax}_{g,t} - \text{Hgrmin}_{g,t}) \cdot \left(\frac{\text{B}_{g,t} - \text{Bmin}}{\text{Bmax} - \text{Bmin}} \right) \right\} \quad (\text{A2.17})$$

$\text{Hact}_{g,t}$ = Actual height (m)
 $\text{Hgrmin}_{g,t}$ = minimum height growth; difference between $\text{Hmin}_{t,g}$ and $\text{Hmin}_{t-1,g}$ (m)
 $\text{Hgrmax}_{g,t}$ = maximum height growth; difference between $\text{Hmax}_{t,g}$ and $\text{Hmax}_{t-1,g}$ (m)
 $\text{B}_{g,t}$ = biomass growth ($\text{kg}\cdot\text{ha}^{-1}$)
 Bmin = minimum biomass growth ($\text{kg}\cdot\text{ha}^{-1}$)
 Bmax = maximum biomass growth ($\text{kg}\cdot\text{ha}^{-1}$)

The height growth of shrubs and trees is assumed to start at a given threshold biomass ($150 \text{ kg}\cdot\text{ha}^{-1}$ for the sum of stem biomass of shrubs and trees; the threshold values are equal to the threshold values for succession in Table A2.1. If the biomass is below this threshold the height is set to the minimum height given above, and the age is reset to 1 year. A maximum height is explicitly imposed only for dwarf shrubs (1 m). Mark, however, that for shrubs and trees the maximum height is determined by parameters k_1 and k_4 in Equations A2.15 and A2.16. The height and the height growth of the functional types are also influenced by management (see below).

Management

Management implies the removal of biomass and therefore nitrogen from the system. In some cases management also influences the height of the functional types. SUMO can simulate three types of management: mowing (grassland), turf stripping (heathland) and cutting (forests).

- Mowing. In mown grassland the leaf biomass of the functional type grasses/herbs is reduced to $1000 \text{ kg}\cdot\text{ha}^{-1}$. The aboveground biomass of the other functional types is reduced to $2 \text{ kg}\cdot\text{ha}^{-1}$. The biomass of the roots is not reduced. The height of all functional types except for herbs/grasses is reduced to the height of their seedlings. For herbs/grasses the height is corresponding to the remaining biomass (Equation A2.13). The age of the shrubs and trees is set back to 1 year.
- Turf stripping. After turf stripping the total biomass of the functional types is minimised to $2 \text{ kg}\cdot\text{ha}^{-1}$ for dwarf shrubs and to $0.2 \text{ kg}\cdot\text{ha}^{-1}$ for the other functional types. Since after turf stripping the humus layer is also removed, SUMO signals SMART2 to remove the humus layer from the system.
- Cutting. Cutting can influence all functional types. Three different types of forest management are modelled in SUMO. The first is extensive management. In this type of management, every 10 years 10% of the trees are harvested; this is implemented by reducing the biomass of all organs by 10%. The dead roots are transported to the dead roots-pool in SMART2 and mineralised. The rest of the cut biomass is removed from the system. The pool of reallocatable nitrogen is also reduced by 10%. The height of the trees is not influenced by cutting. The second management form is traditional (intensive) forest management with thinning every 5 years and clear-cut at the end of the management cycle (Equation A2.18).

If $rt = tt$

$$B_{i=g,t} = B_{i=g,t} \cdot (1 - kT_{g,t}) \quad (\text{A2.18})$$

rt = runtime (yr)
 tt = thinning time (yr)
 $B_{i=g,t}$ = biomass ($\text{kg}\cdot\text{ha}^{-1}$)
 $kT_{g,t}$ = thinning factor (-)

The amount of thinning depends on the tree species and the age of the stand (after Jansen et al., 1996). The cut biomass and a part of the reallocatable nitrogen are removed from the system, in the case of a clear-cut this is the total aboveground biomass and all the reallocated nitrogen. The biomass of the non-tree functional types is assumed to be destroyed for 90% and added to the litter. The forest is replanted after clear-cut; the height and age are set back to the height of planted young trees. The third management type is coppicing, which is simulated for a few tree species only: willow, ash, alder and oak. For willow, ash and alder the shoots are harvested every seven years, for oak every 30 years. For all tree species 80% of the above ground biomass is harvested. A large part of the functional type herbs is assumed to be destroyed (90%) and is added to the dead roots and litter pools, shrubs are assumed to be actively removed from the system. The height of the remaining stub after management is set to 0.3 m.

Parameterisation

SUMO uses over 1000 different parameters (including the different parameters per organ for five functional types and for ten vegetation types). Almost all parameters are based on extensive literature research (references can be found in Wamelink et al., 2000a; Wamelink et al., 2000b). The data from literature were stored in a database. The parameters were estimated from the database and used for test runs of SUMO using a test-set of vegetation types. Fine-tuning of the parameters took place until the model produced an acceptable outcome. Fine-tuning is necessary because there is often much variation in the data. It was always done within the range of the data found in literature, but most of the parameters were left unchanged. Parameters that are fine-tuned are the maximum growth rate (A_{\max} in Equation A2.2), light interception coefficient (k in Equation A2.3), minimum (N_{\min} in Equation A2.6) and maximum nitrogen content (N_{\max}). Maximum growth rate and light interception were also chosen because for these parameters data are scarce or absent. Parameters involved in the height of the functional type, the influence of moisture on the growth and management were never fine-tuned.

Validation of SUMO in the Netherlands and UK

Biomass growth was validated using data collected at two grassland sites, a heathland site and a forest site in the Netherlands and UK. The nitrogen content of the leaves was validated on a set of forest stands.

The first grassland site is situated near Wageningen (51°58' N, 5°39' E) and is part of a long-term field experiment started in 1958 on former agricultural land (Elberse et al., 1983). The site is mown once a year and not fertilised. Every year the mown biomass was harvested, dried and weighed. The changes in aboveground biomass were simulated using site specific historical deposition data. Due to yearly differences in i.e. rainfall and temperature the measured biomass varies greatly between years, while the simulated biomass does not vary much among the years (Figure A2.1). But the simulated biomass values vary within the range of the measured biomasses. The large difference for the first year is probably caused by the former agricultural use of the land, which has led to a relatively high measured biomass. The effects of former agricultural use of grassland or vegetation in general can not be simulated by SUMO. Both the measured and the simulated biomass show a decrease over the years, due to the yearly biomass removal without manuring.

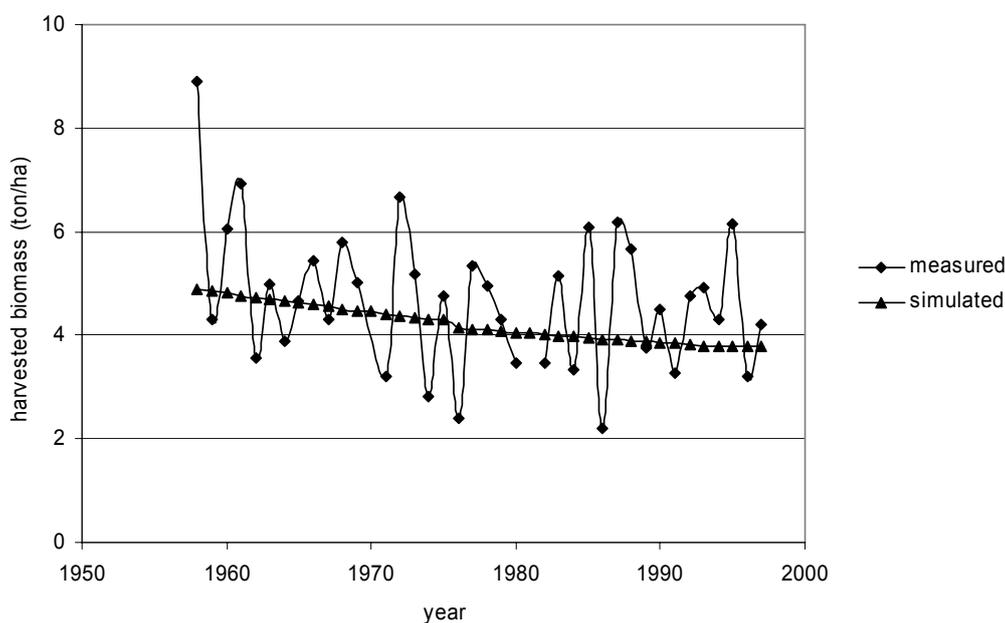


Figure A2.1 Measured and simulated aboveground biomass for a mown grassland site near Wageningen.

The second grassland site is the Parkgrass experimental site at Rothamstead in the UK. The site was mown twice a year and the harvested biomass was weighted and averaged over ten year periods. The experiment started around 1850 and continued until today. The site is extensively described by i.e. Lawes & Gilbert (1880), Cashen (1947), Thurston et al. (1976), Jenkinson et al. (1994). The site was probably extensively manured just after the start of the experiment, but this treatment stopped after a few years (Jenkinson et al., 1994). The model was run for this site with an expert-based estimation of the manuring during the first years, initialised with an estimated amount of biomass and using site specific data for sulphur and nitrogen deposition. Both increased steadily from the start of the experiment, but decreased in recent years; sulphur after approximately 1980, nitrogen after approximately 1990. The nitrogen deposition was more or less stable between 1900 and 1940, which is

accounted for in the model run. The field data were retrieved from Dodd et al. (1994). The results show that the harvested biomass is fairly well simulated by SUMO (Figure A2.2). Only the effect of the nitrogen deposition since approximately 1960 is slightly underestimated. But the reduction in biomass harvest in the early years due to exhaustion of the soil, the stabilisation of the harvest when the effect of nitrogen deposition compensates for the exhaustion, and the increase of the harvest later on due to the further raise of the deposition is simulated quite well.

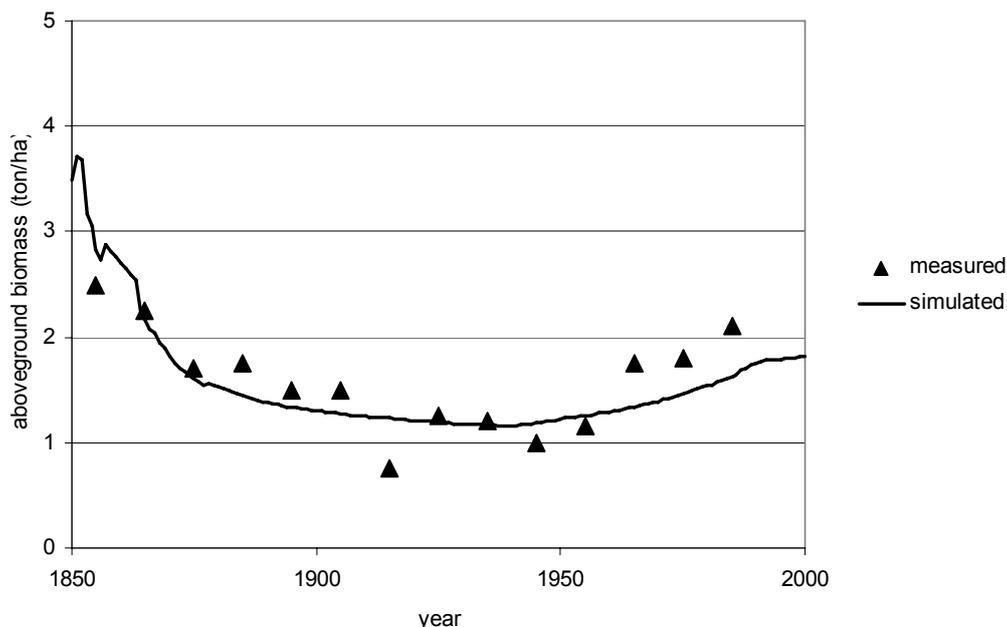


Figure A2.2 Measured and simulated biomass harvest for Rothamstead experimental grassland site in the UK.

SUMO was validated for heathland on a site in the south of The Netherlands near Strabrecht (51°23' N, 5°37' E), which was studied by Berendse (1990). At this site eight plots were used of which the last time of turf stripping, i.e. the removal of all biomass and the litter layer down to the mineral soil layer was known. Above and belowground biomass was harvested in 1984. SUMO was run for each site separately. The runs were started at the moment of turf stripping, which differs for each site, and run up to 1984, using site specific information on soil type, deposition etc. Aboveground biomass simulated by SUMO was compared with the measured aboveground biomass. In general the aboveground biomass is simulated well, although it is overestimated for the plots where the turf was stripped 16 and 18 years before the field data collection (Figure A2.3). For the oldest plot the majority of the simulated biomass is in grasses and herbs, although dwarfshrubs are still present. In the field dwarfshrubs are only present as dead biomass in the litter layer in this plot. In all other plots almost all measured and simulated biomass is present in the functional type dwarfshrubs.

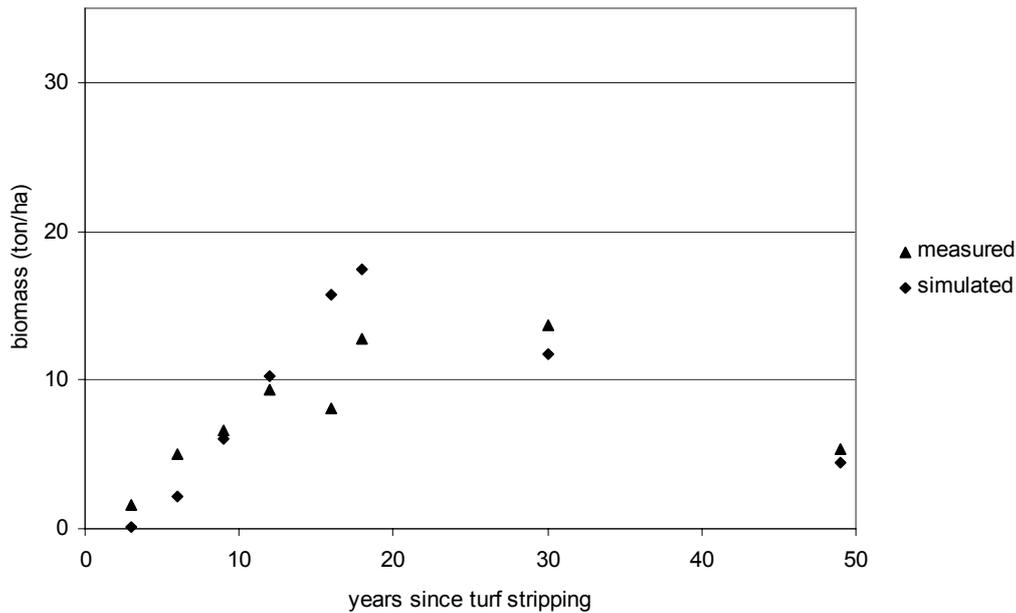


Figure A2.3 Simulated and measured aboveground biomass for a beathland near Strabrecht in The Netherlands. The simulated biomass is the result of separate independent simulations, started at the moment of turf stripping, till the field survey took place.

The forest site that we used for validation is located in the north of The Netherlands near Selligen (52°57' N, 7°03' E). Here small forest plots are situated on former agricultural land. This chronosequence consists of ten plots of 0, 14, 25, 32 and 89 years old in 1999. Of each age stage two independent plots are present. At the beginning of each stage trees were planted. One of the 89-years old plots was cut before 1999 and therefore omitted. Only one site was used for the validation. The remaining stages all have oak as the dominant tree species. The undergrowth is almost absent in some stages, other stages are dominated by grasses. The 0-year stage was used as input for SMART2-SUMO. This 0-year stage is still in agricultural use. The actual biomass of the trees was estimated on the basis of the diameter at breast height and the height. The above ground biomass of the herbs was harvested and weighed (Van Oene et al., 1999a; Wamelink et al., 2001). Historic deposition data were used for the simulation. The simulated values were compared with the measured biomass of the sites (Table A2.2 and Figure A2.4). Some of the successional stages are quite well simulated by SUMO; other successional stages are less well simulated. The simulated biomass in older forest is too low, but the biomass of other functional types is simulated quite well. An exception is the biomass of the pioneer trees in the 89-year-old stage and the biomass of the climax trees in the 25 and 32 year old stage. This difference is probably caused by the thinning regime in the past. Site specific data are not present for the thinning regime.

Table A2.2 Validation results for the chronosequence Selligen. Figures are biomass in $\text{kg}\cdot\text{ha}^{-1}$ ($\cdot 10^3$) dry matter per functional type, measured in the field with (standard error) and simulated by SUMO.

age		herbs (s.e.)	dwarf shrubs (s.e.)	Shrubs (s.e.)	pioneer tree (s.e.)	climax tree (s.e.)
14	measured	0.81 (0.11)	0	0	0.02 (0.03)	28.13 (2.35)
	SUMO	0.79	0.11	0.64	0.64	26.50
25	measured	0.37 (0.16)	0	0	0.09 (0.09)	39.03 (0.72)
	SUMO	0.01	0.04	0.59	0.70	47.23
32	measured	0.14 (0.08)	0	0	0	66.96 (0.61)
	SUMO	0.01	0.02	0.58	0.73	48.95
89	measured	0.77	0	0	4.69	111.36
	SUMO	0.01	0.01	0.44	0.29	98.33

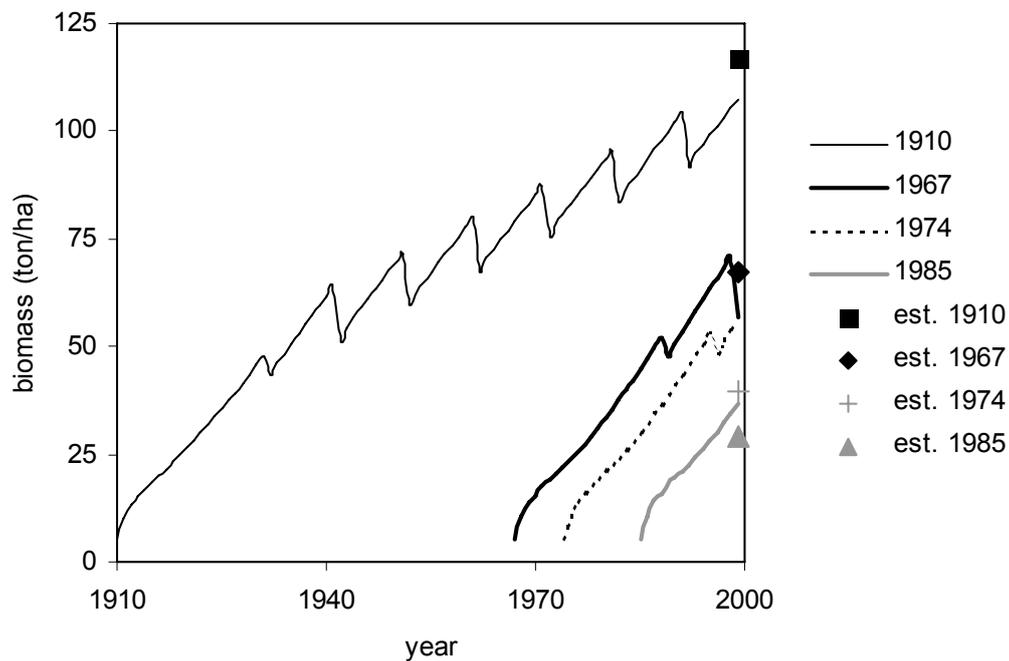


Figure A2.4 Simulated total biomass increment for four different successional stages at the Selligen site. The forests were planted in 1910, 1967, 1974 and 1985 on former agricultural land. The biomass of the sites was estimated in 1999 (est. 1910, est. 1967, est. 1974 and est. 1985 giving the estimations for the respective plant years).

SUMO was also validated for the nitrogen content in the leaves of planted trees on 169 forest sites on all major soil types situated all over The Netherlands. The simulated N contents were regressed on the measured N contents ($R^2 = 39\%$, intercept = 2.1 ± 1.9 ($p = 0.27$), regression coefficient = 0.92 ± 0.09 ($p < 0.001$). When there is no difference between simulated and measured N content one would expect a regression coefficient of 1.0 and an intercept of 0.0. For both the values do not differ significantly. The percentage explained variance indicates that the N contents of the leaves are simulated fairly well.

Annex 3 Prediction and validation of soil and vegetation changes with the dynamic MAGIC-GBMOVE model

Validation of MAGIC simulations on C/N ratios

There are few UK sites at which C/N has been repeatedly sampled over time, and therefore testing model predictions of C/N is problematic. However, data from plot-scale N manipulation studies have been used to test the ability of MAGIC to predict changes in observed N leaching and soil C/N under different addition levels (Evans et al., Environmental Pollution in press). For two sites with high quality soil C and N data (Figure A3.1), the model successfully reproduced observed decreases in C/N under three treatment levels. It should be noted that these simulations incorporated an (observed) increase in C storage as a consequence of N deposition, which slowed down the rate of C/N change. This was hypothesised to reflect additional litter incorporation. The greatest uncertainty in the prediction of future C/N change in small-scale studies is considered to be the magnitude of current and future N deposition (Smart et al., 2005a).

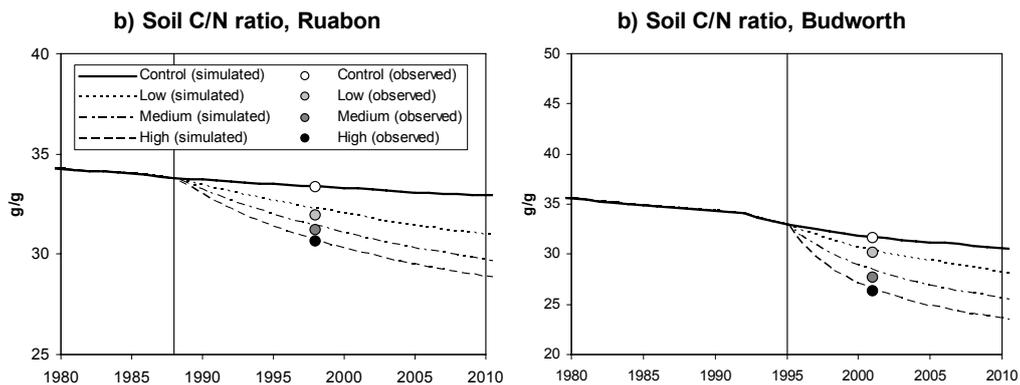


Figure A3.1. Simulated and observed organic soil C/N ratio under ambient N deposition and three levels of long-term NH_4NO_3 addition at two heathland experimental sites. Vertical line indicates start of experiment.

Rothamsted is an example where data on C/N ratios are available for a 100 year period. Use of standard N uptake estimates lead to a mismatch between MAGIC simulations of past C/N ratio on the Rothamsted Park Grass control plots and available measurements. N offtake was calculated by multiplying hay offtake, for which accurate measurements based on decadal averages are available (Dodd et al., 1994), by the proportion of N in hay biomass. For N proportion, a weighted average from three measurement periods (1920-23, 1940-43 and 1956-59) was used (Warren & Johnston, 1964). This calculation method is subject to uncertainty, since N availability during these periods may have differed from that in the earlier and later years of the experiment, leading to differences in N concentration in hay. This uncertainty has a large effect on the net addition (deposition minus offtake) and thus on the accurate simulation of the historic C/N trajectory (Figure A3.2). Note that

raising N deposition would have the same effect as reducing N offtake on the simulation.

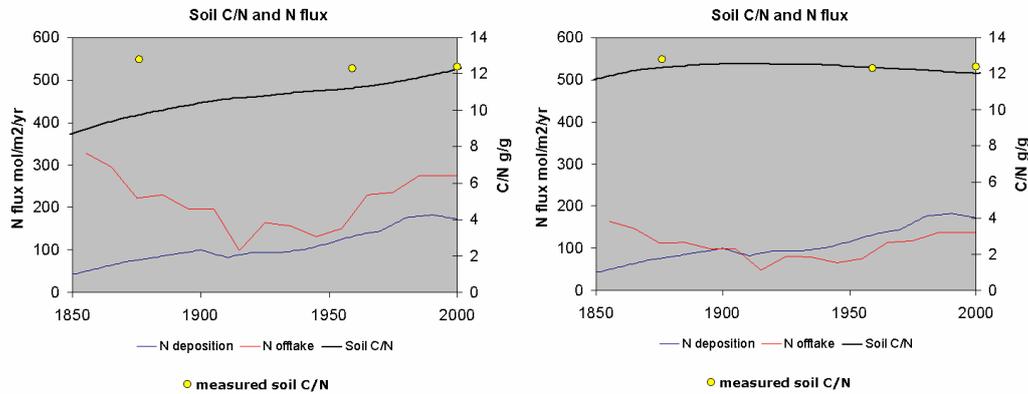


Figure A3.2. a) measured N offtake and deposition history, and comparison of MAGIC-simulated and measured C/N; b) as a) but with N offtake halved.

Validation of MAGIC-GBMOVE on observed species composition and species richness

We determined how successfully MAGIC+GBMOVE could reproduce the observed species composition in test plots. The goal must be near complete prediction of all species present, arbitrarily over 90%, if the models are to be used to reliably predict community assembly. Observations were compared with predictions generated initially by populating a simulated set of quadrats with species conditioned on species composition predicted by GBMOVE and a Poisson distribution of mean species richness with proportional variance predicted by a separate General Linear Mixed Model using the same explanatory variables as GBMOVE. This statistical model is fully described in Smart et al. (2005b).

In order to assess the influence of uncertainty in the calibration equations relating soil properties to mean Ellenberg scores, predictions of species composition based on soil C/N and pH generated by MAGIC were compared with predictions based on observed mean Ellenberg scores. These comparisons were carried out for control plots at two test sites described in Smart et al. (2005b). They comprise the long-term continuous Park Grass hay experiment at Rothamsted (unimproved neutral grassland) and the Hard Hills grazing and burning experiment at Moorhouse National Nature Reserve (blanket bog).

Rothamsted Park Grass

The results for Rothamsted indicated that even when mean Ellenberg scores were used as input to GBMOVE based on observed species composition, substantially less than 90% of species present were predicted to be present. On average only 67% of all species observed were actually predicted when observed mean Ellenberg scores were used to predict species composition. However when predictions were based on MAGIC estimates of soil C/N and pH as input to GBMOVE, the percentage correctly predicted still decreased substantially (Figure A3.3).

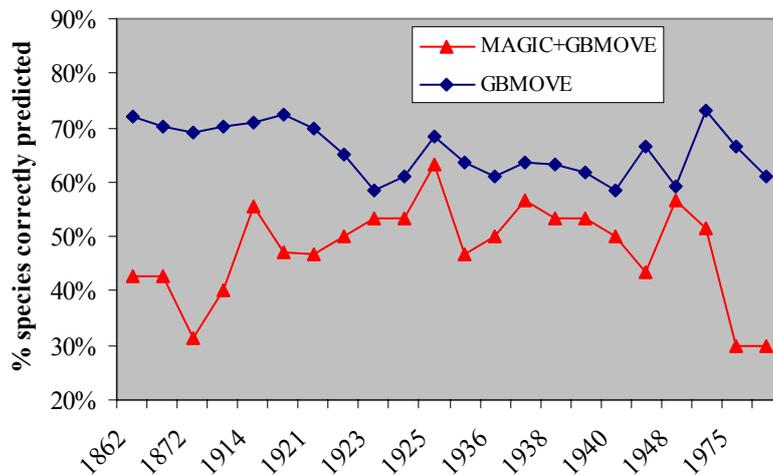


Figure A3.3 Percentage of species correctly predicted in the three Park Grass control plots based on pseudo-quadrats whose richness and composition were predicted by MAGIC+GBMOVE versus predictions based on observed mean Ellenberg scores only as input to GBMOVE.

Although the phyto-sociological community type was correctly predicted by MAGIC+GBMOVE at the start, the observed and modelled decline in soil C/N resulted in a shift to a more fertile community type, which is the reverse of the extensifying trend observed in the control plots (Williams, 1978; Dodd et al., 1995). The main reason for poor performance of MAGIC+GBMOVE appears to be that observed soil changes were actually inconsistent with observed vegetation changes. Since MAGIC calibrates to contemporary soil measurements, it is possible that sampling practice could be responsible for biasing the model simulation. Measured C/N ratios in later years in experimental addition plots carefully avoided a thin mat of persistent litter that had developed in the O horizon over the course of the experiment (Paul Poulton pers.comm. and Warren & Johnston, 1964). This would result in C/N measurements indicating a higher fertility rooting zone than that encountered by at least some of the more shallow rooting species present.

Moorhouse

The same testing strategy was adopted as for Rothamsted. A comparison was made between the proportion of species present in each year that were correctly predicted using observed mean Ellenberg values and observed species richness, versus predictions from MAGIC linked to GBMOVE and a statistical model of above-ground species richness (Figure A3.4).

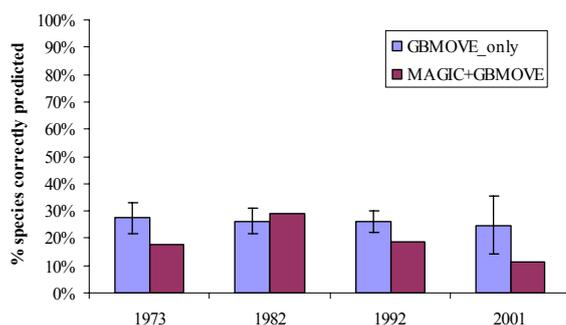


Figure A3.4 Percent of species in control plots that were correctly predicted to be present based on observed Ellenberg values (+/-sdev) i.e. GBMOVE only, versus the percentage correctly predicted in pseudo-quadrats generated by MAGIC+GBMOVE.

When predicted species lists for both GBMOVE and MAGIC+GBMOVE were examined, key absences included a range of bryophytes. However, the key dominants in the vegetation were predicted to be present in both model runs. These included *Calluna vulgaris*, *Eriophorum vaginatum*, *E.angustifolium* the liverwort *Cephalozia connivens* and *Sphagnum capillifolium*. Both predictions lacked the important diagnostic species *Rubus chamaemorus* but this was because this species has no GBMOVE model. As seen at Rothamsted, the linked models did not perform particularly well in predicting species actually observed in control plots. Yet neither did predictions based solely on observed mean Ellenberg values.

From this comparison it can be concluded that model performance varies greatly between sites. While some sites had relatively high success rates (an additional three experimental sites with quadrat data for a single year had an average of 80% of species correctly predicted – (Smart et al., 2005b)), it is unlikely that both generally applicable yet highly accurate models can be developed, because of the dependence of current species composition on site-specific aspects of patch and wider landscape history. Because of this, we no longer attempt to apply predicted probabilities of occurrence from GBMOVE as expectations of species presence. Rather they are interpreted as indices of habitat suitability on the basis that target species ought to be able to persist and increase in population size in the absence of constraints to dispersal and establishment. The focus of further validation work therefore centred on comparing predicted trends through time with observed changes rather than attempting to build complete species assemblages in a specific site.

Validation of MAGIC-GBMOVE on temporal changes among plant species

Changes in species abundance over time comprise cyclic, random and directional components. The expectation is that model predictions based on N and S deposition history driving MAGIC and then GBMOVE ought to explain a significant fraction of directional change, assuming a pollutant deposition signal is not eclipsed by other effects such as succession, sampling error and the weather. A test of observed versus expected change was only carried out for Moorhouse only because of known problems with biased soil sampling. Observed versus predicted species changes over

time were summarised as slope coefficients for each species when a linear regression line was fitted to observed abundance across each year and predicted habitat suitability from MAGIC and GBMOVE across the same time period.

Moorhouse

Observed test data from the Hard Hills control plots coincided with a period during which MAGIC predicted an increase in soil pH and a steady decline in soil C/N ratio (Figure A3.5). While this trend is partly consistent with recovery from previous S deposition and with the eutrophying effect of N, it is also thought to be consistent with a marked trend toward warmer winter and early Spring temperatures (Holden & Adamson, 2002) as well as the deposition of sea salt.

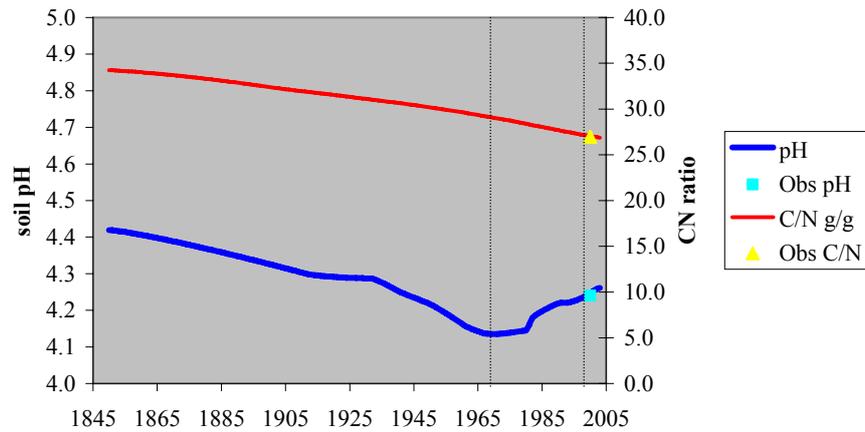


Figure A3.5 MAGIC prediction of change at Moorhouse in response to modelled deposition history. The yellow and blue triangles are observed soil measurements to which MAGIC calibrates. Vertical lines indicate the interval covered by test data from the control plots of the Hard Hills experiment.

Despite considerable scatter there was a positive correlation between observed change in species frequency and predicted change in habitat suitability at Moorhouse (Figure A3.6). The reasons for the residual variation are illustrated by some example plots for individual species in Figure A3.7. A chi-square test of observed versus predicted directions of change was significant ($p=0.016$). While the correlation between observed and predicted slopes was also significant, predicted rates of change covered a narrower range than observed species changes.

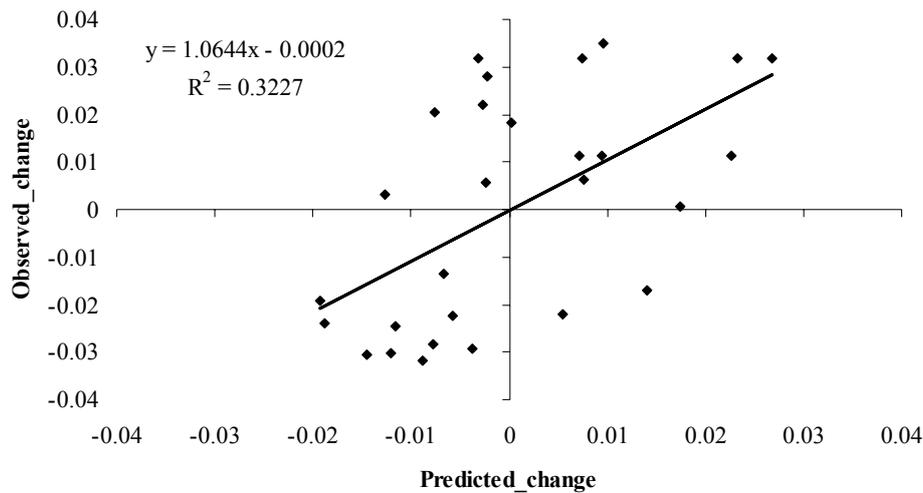


Figure A3.6. Predicted versus observed change in individual species in the Moorhouse Hard Hills control plots. Predicted change is the slope coefficient of a linear regression on occurrence probabilities predicted by MAGIC+GBMOVE for each year between 1973 and 2001. Observed change is the slope coefficient of a linear regression on % frequency in sample plots in each survey year. Pearson correlation coefficient = 0.568, $p=0.002$.

Good model testing data is scarce yet sorely needed to provide further support for the MAGIC-GBMOVE model chain. The tests of temporal trend at Moorhouse provide a degree of support but comparisons of predicted species occurrence using measured soil C/N and pH versus mean Ellenberg scores indicate that the greatest uncertainty and hence, inaccuracy in model predictions, is attributable to the weak calibration relationships, especially that between soil C/N and mean Ellenberg N at high fertility. Current model development is therefore urgently focussed on better prediction of mean Ellenberg N. This is likely to come from reliable modelling of annual biomass and highlights the need to integrate a succession model like SUMO into the model chain.

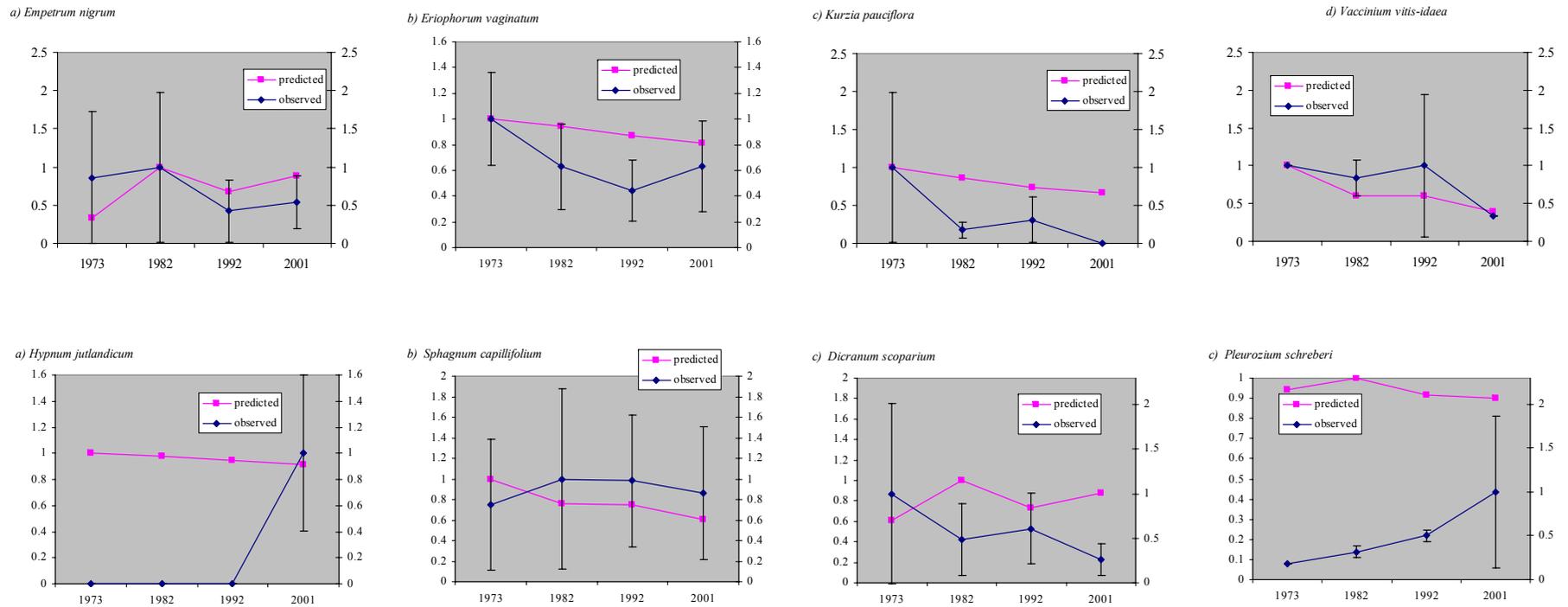


Figure A3.7 Examples of individual species changes at Moorhouse. Frequency and predicted probabilities were standardised to range between zero and 1 across the time series to enable comparability of the direction of change. a-d are good model fits, e-h poor model fits. The standard deviation of observed counts are shown.

Annex 4 Calculation of critical loads for nitrogen and acidity with the SMB –BERN steady state model

The SMB model for the calculation of critical loads.

The aim is to establish the connection between on one hand plant community specific threshold values for the C/N ratio, $CN_{BS(crit)}$, and plant community specific threshold values for the Bc/Al-ratio and on the other hand a limit value for the input of nitrogen and sulphur compounds. Both plant community specific threshold values are derived here by use of the BERN model and the latter is based on a threshold value for base saturation, $BS_{CN(crit)}$. These threshold values have been included in the simple mass-balance equation (Nagel & Gregor, 1999) as described below

Simple mass balance equation for critical loads of nutrient nitrogen

With the Bern model, the Critical N Load is calculated from the site specific function $(C/N, BS) = f(N_{dep}, S_{dep})$ at steady state conditions under consideration of:

- Nitrogen availability (N deposition plus N mineralization)
- Gross N mineralization/immobilisation
- Biomass N increment.

The Critical Load is met when a long term balance between incoming deposition, immobilisation, mineralization, uptake, denitrification and leaching leads to a steady state C/N ratio in the soil that equals the critical C/N of the protected plant community. In principle the VSD model but this model does not include a mineralization term. Therefore an extension has been developed as described below.

In order to derive the yearly changes of the N-content in the soil organic matter (SOM) it needs the calculation after the following equations:

$$\frac{d[N]_{SOM}}{dt} = N_{litterfall} - r_{min}[N]_{SOM} + N_{imm} \quad (A4.1)$$

with:

r_{min}	= mineralization rate (yr^{-1})
$[N]_{SOM}$	= total N content in soil organic Matter ($g.m^{-2}$)
N_{imm}	= immobilisation rate ($g.m^{-2}.yr^{-1}$)
N_{dep}	= N-deposition rate ($g.m^{-2}.yr^{-1}$)
$N_{litterfall}$	= total N-amount in the falling litter ($g.m^{-2}.yr^{-1}$)
$N_{upt,tot}$	= total uptake rate into biomass ($g.m^{-2}.yr^{-1}$)

The potential immobilisation rate of Nitrogen is assumed to be piecewise linear to the actual C/N-ratio:

$$N_{imm} = N_{av} \max \left(\min \left(\frac{CN_{SOM} - CN_{min}}{CN_{max} - CN_{min}}, 1 \right), 0 \right) \quad (A4.2)$$

with:

- N_{av} = Available Nitrogen (see Text below) ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)
- CN_{min} = Lowest steady state (soil type specific) C/N-ratio (constant) in the humus rich top soil layer
- CN_{max} = Highest steady state (soil type specific) C/N-ratio (constant) in the humus rich top soil layer
- CN_{SOM} = actual value of the C/N-ratio in the soil organic matter depending on current vegetation type and deposition history.

Organic soil C/N ratio is not considered to be a direct control on plant response but represents a readily measurable proxy for important processes (e.g. nitrification/mineralization). It can therefore still be useful to measure and to incorporate in models.

A similar equation, sometimes constructed with the inverse N/C, can be found in most dynamic models, the main differences for the system dynamic can be found in different definition of the available Nitrogen. VSD, among others, are using the difference between the deposited N and the N- (Net-) Uptake. In ForSAFE, where the N/C-value is used, and the linear function between CNmin and CNmax becomes a hyperbolic function, only the mineralised N is immobilised. In this approach, we are using the all available Nitrogen from all sources, as available N

$$N_{av} = N_{dep} + r_{min} \cdot [N] \cdot SOM \quad (A4.3)$$

But the microbes are, in the case of N limitation, in a concurrence situation with the higher plants. For each forest ecosystem type, we estimated a potential N uptake of the forest trees for the production of stemwood and litter. For simplicity reasons, a linear concurrence is assumed:

$$N_{imm,pot} = N_{av} \max \left(0, \min \left(1, \frac{CN_{SOM} - CN_{min}}{CN_{max} - CN_{min}} \right) \right)$$

$$N_{imm} = \min \left(N_{av} \frac{N_{imm,pot}}{N_{imm,pot} + N_{upt,pot}}, N_{imm,pot} \right) \quad (A4.4)$$

$$N_{upt} = \min \left(N_{av} \frac{N_{upt,pot}}{N_{imm,pot} + N_{upt,pot}}, N_{upt,pot} \right)$$

where:

- N_{upt} = N-uptake rate (annualised) by trees for production of litter and stemwood ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$)

$N_{\text{upt,pot}}$ = maximum potential N-uptake rate in the case that N-supply is sufficient enough (derived from yield tables)

If N is not limiting the uptake by higher plants and the immobilisation of N by microbes, the N surplus is either denitrified, and leaves the soil to the atmosphere or is nitrified and leaves the soil as nitrate to the ground water. Until the development of new methods to calculate the denitrification, the CLnut(N)-method (see UN/ECE 2004, Chap. 3) is used:

$$\begin{aligned} N_{\text{de}} &= f_{\text{de}} (N_{\text{dep}} + r_{\text{min}}[N]_{\text{SOM}} - N_{\text{upt}} - N_{\text{imm}}) \\ N_{\text{le}} &= (1 - f_{\text{de}}) (N_{\text{dep}} + r_{\text{min}}[N]_{\text{SOM}} - N_{\text{upt}} - N_{\text{imm}}) \end{aligned} \quad (\text{A4.5})$$

where:

N_{de} = denitrification rate ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)
 N_{le} = leaching rate ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)
 f_{de} = Empirical denitrification factor, see Manual on Modelling and Mapping (UN/ECE 2004, Chap. 3)

The dynamic of the above described system depends heavily on the amount of mineralised nitrogen. Since nitrogen is mineralised with the decomposition of organic matter. The rate r_{min} of the decomposition process is the inverse mean time of the decomposition of organic matter in the soil. To calculate the mineralization rate the DECOMP model, first published by Walse et al. (1997) and now integrated in the ForSAFE model (Belyazid, 2006), was used with slight adaptations.

Following the DECOMP model the soil organic matter is differentiated into four independent pools with different decomposition characteristics: (1) easily decomposable materials like proteins and proteids (“EDC”) (2) cellulose, hemicellulose and pectin (“CELL”), (3) lignin and lipid (“LIGN”), (4) resistant matter like tanine, wax etc. (“RES”). Each pool has a different decomposition rate k , which is a function of the decomposition rate under optimal condition and rate limiting factors low soil temperature, dryness and acidity. The potential rate (k_{pot}) ranges from $1/(400 \text{ a})$ to $240/\text{a}$. The limiting function are parameterized for the DECOMP model by Walse et al. (1997). The results were used without further fitting.

Since not all of the decomposed organic matter is removed from the solid phase, a part is transformed in other, less decomposable compounds of the organic matter. Data for these assumed fractions are obtained from iterative curve fitting from field data and published in Berg et al. (2003), as they are used in the ForSAFE model. In future work these values have to be fitted at central European plots also. The carbon balance of the organic matter pools is given therefore by:

$$\frac{dC_K}{dt} = C_{\text{in},K} - k_{\text{act},K} C_K + \sum_{i=\text{EDC}}^{\text{RES}} k_{\text{act},i} t_{K,i} C_i \quad (\text{A4.6})$$

where:

- $C_{in,K}$ = Input of carbon by litterfall for the organic compound K (g/(m².a))
- C_i = Size of carbon pool j (g.m⁻²)
- $k_{act,i}$ = decomposition rate ($k=f(T,\theta,pH)$) (1/a)
- $t_{a,b}$ = transformation matrix, indicates the transformation of one C-pool to another
- K = one of the four litter pools (EDC, CELL, LIGN, RES)

The relative net rate of organic matter leaving the solid phase is therefore given by:

$$r_{min} = \sum \left(\frac{\frac{dC_K}{dt} - C_{in,K}}{C_K} \right) \quad (A4.7)$$

For simplification the model is not simulating the N-content of each pool, but has a single N-pool for all C pools. The mineralization rate of Nitrogen is thus given by:

$$N_{min} = r_{min} \cdot [N] \cdot SOM \quad (A4.8)$$

The possible results from a run of the N-adapted DECOMP/VSD-combination are demonstrated for the example of a typical, spruce site in southern Germany in the following figures (Figure A4.1 to A4.3). Input data are the IIASA-time series for N-deposition history. For the future a N-reduction down to 50% of the Gothenburg protocol is assumed, a spruce vegetation community, the estimated C- and N-content in the tree compartments and their parts of EDC, CELL, LIGN and RES (from literature sources), the soil specific CNmin, CNmax, pH-value, fde, water content at field capacity, annual mean temperature, potential N uptake based on yield tables and the N content in the tree compartments from literature sources.

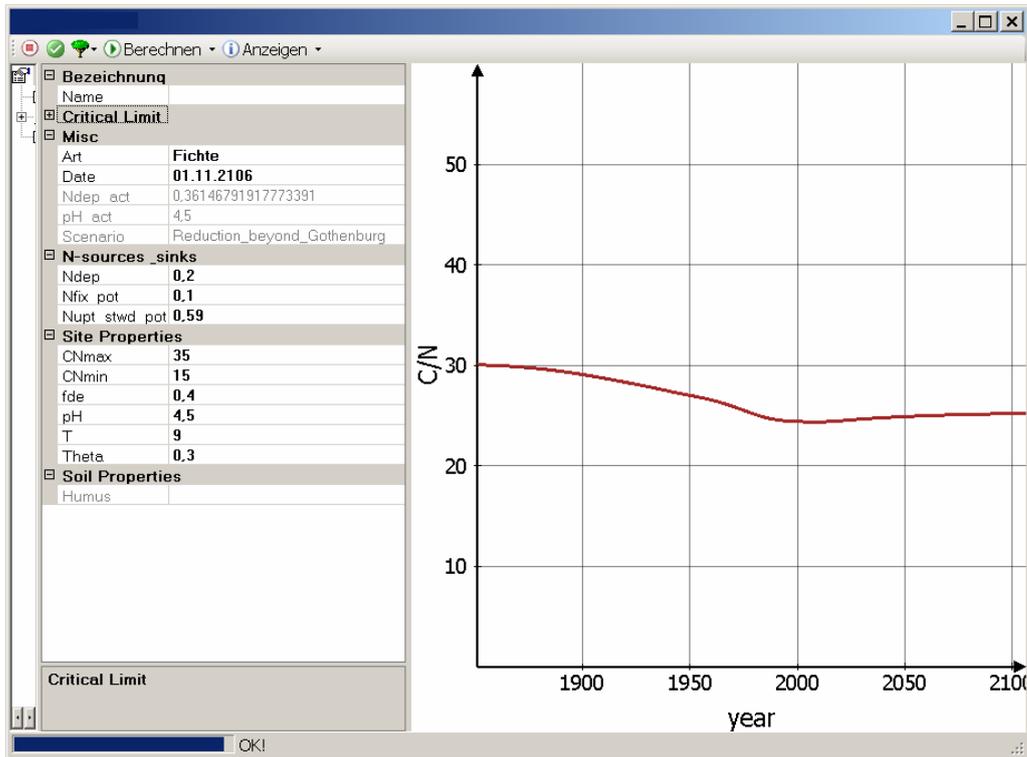


Figure A4.1 Modelled dynamic C/N-ratio at a typical spruce site in southern Germany.

After increasing of incoming N-deposition the mineralization rate, the immobilisation rate, the uptake into the litter and into the stemwood increase (Figure A4.2). During this time the C-pool is increasing, particular the resistant undecomposable C-pool (Figure A4.3). If the maximum uptake rate determined by the genetic potential is reached, the leaching and denitrification rates increase rapidly. The C/N-ratio falls (Figure A4.2). When the deposition begins to decrease the immobilisation rate sinks, and the mineralization rate persists on a high level as well as the uptake rates. The C/N-ratio persists on a low level. After several decades (recovery delay time) the leaching rate begins to decrease and the C/N-ratio adjusts at a constant value. The whole system gets near to steady state conditions with specific pool sizes. With a model, which has clear stable attractors, the C/N-ratio at steady state conditions can be calculated, in dependence of fixed site factors and variable but constant depositions in the long run. The next run shows a dose response function, for the C/N-ratio at steady state conditions in dependence of deposition. The deposition, where a critical C/N ratio, e.g. a BERN derived critical limit, is met, is a Critical Load for biodiversity according to the definition of Critical Loads. Since the natural basic plant community is the *Vaccinio myrtylli-Piceetum*, with a Critical Limit according to the BERN-model of $CN_{crit}=24.5$ this results in a Critical Load of $4,86 \text{ kgN}/(\text{ha a})$, at the same site as above.

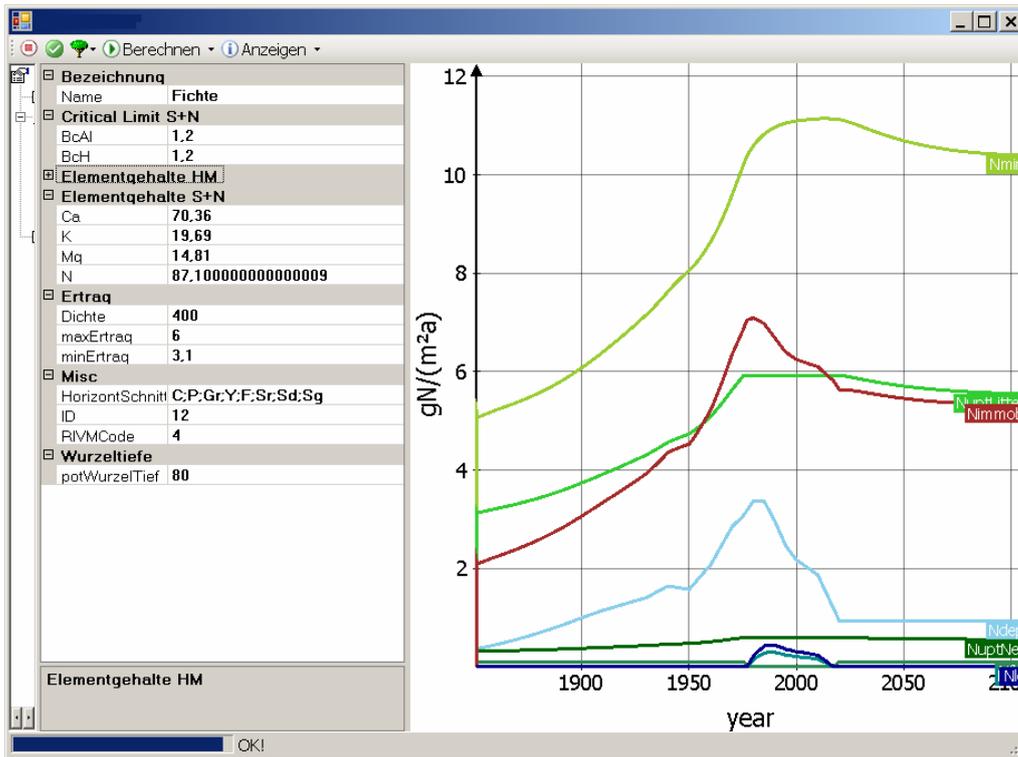


Figure A4.2 Modelled N-fluxes at a typical spruce site in southern Germany

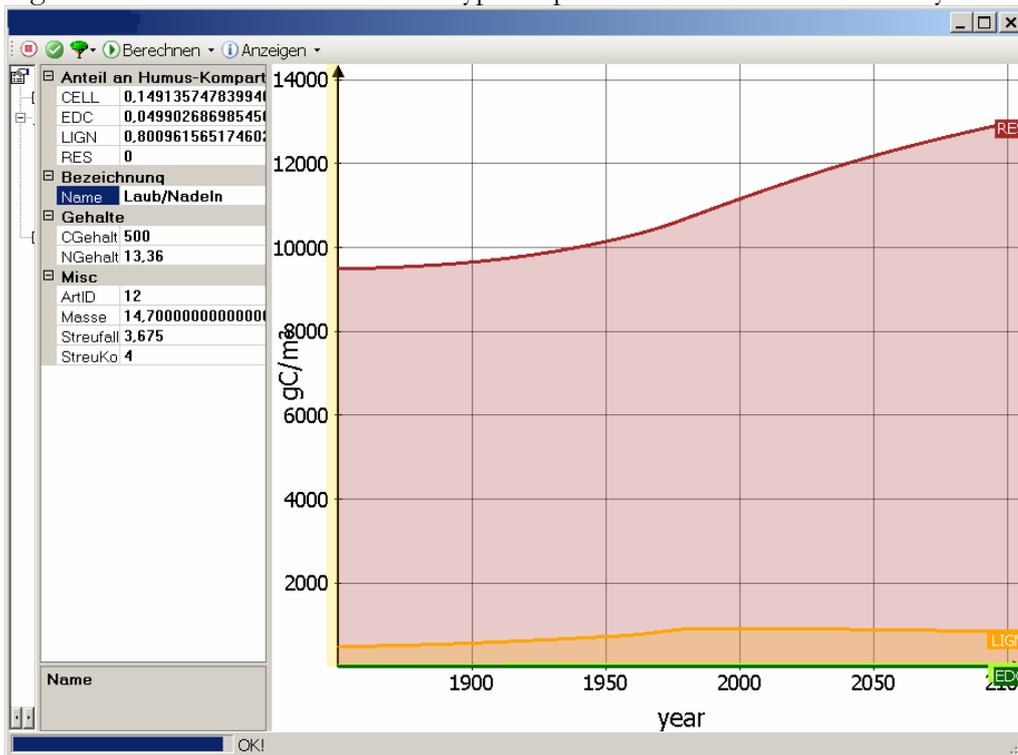


Figure A4.3 Modelled dynamic C-pools in the soil at a typical spruce site in southern Germany

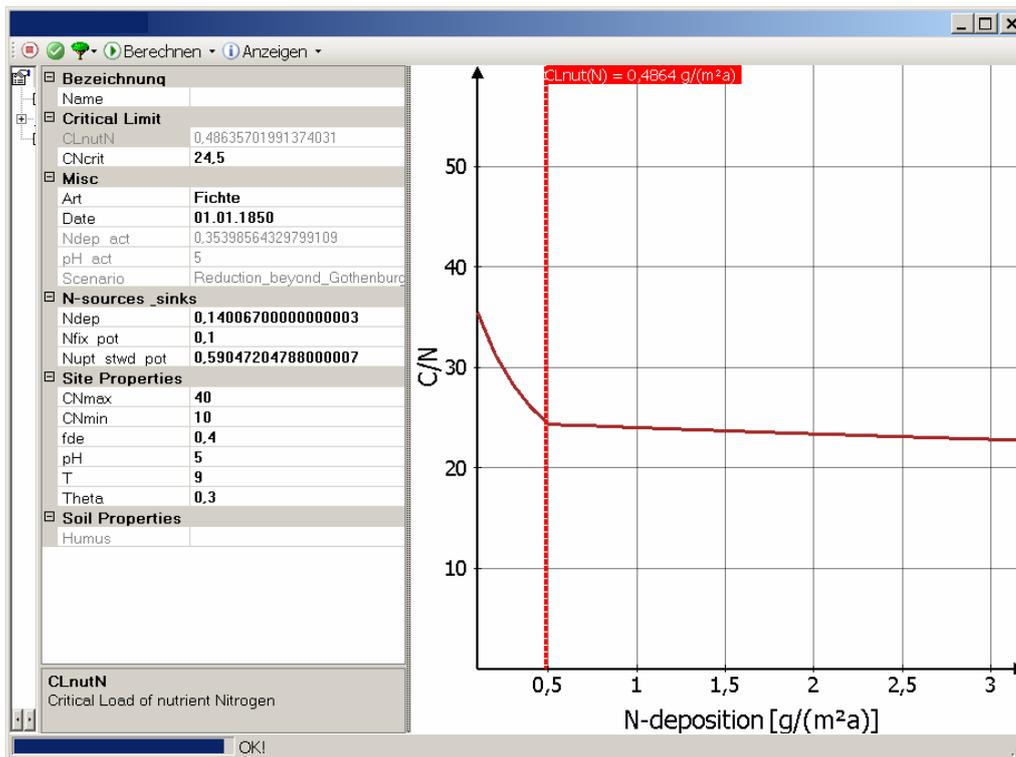


Figure A4.4 Assessment of Critical Load $CL_{nut}(N)$

If we assume another nature target than the basic natural community, with a lower critical C/N ratio, extreme high N-deposition would be tolerable, if the critical C/N ratio is the only chosen critical limit. Since the high concentration of Nitrogen in the groundwater, other ecosystems in the neighbourhood, especially aquatic ecosystems, are at risk.

Future work: The model for the N-dynamic is at an early stage and until now, more a tool to visualize process dynamics, than for predication of real ecosystem states. All parameters used in the model, especially the not reviewed DECOMP parameters have to be fitted at real central European sites. Very sensible parameters, but little investigated, are CNmin/CNmax (aka NimmobA, NimmobB in PnET-CN-like immobilisation models) values. They should be exchanged with measurable values, if a compatible mathematical formulation of the immobilisation process is public available. The main task to be done is the extending of this model for other main nutrients, like Ca, Mg and K and the coupling with a simple geochemical model.

Simple mass balance equation for critical loads of acidifying depositions

Normally in the Simple mass balance the critical Bc/Al-ratio for the main tree species is used. But the BERN Model provides the critical Bc/Al-ratios also for natural plant communities. This value can be derived in the following way directly from the critical limit of base saturation $BS_{CN(crit)}$ (see above):

$$BS_{CN(crit)} = \frac{Bc + Na^+}{Al^{3+}_{crit} + H^+_{crit} + Bc + Na^+} \cdot 100 \quad (\text{in } \%) \quad (A4.9)$$

The portion of Na+ (fNa) within the total quantity of base cations (BC) amounts to 16% (fNa = 0.16) within sand-poor soils. In sand-rich soils, this quantity can reach around 24% (fNa = 0.24). According to Nagel et al. (2001), the Na⁺ proportion can be determined by the soil-type reference values of Central European soils. Thus, Na+ can be computed and eliminated in order to get the critical saturation rate of essential base cations.

$$\frac{[Bc]}{[Al]_{crit}} = \frac{1}{K_{gibb} \cdot K_{Gap}^3 \cdot \left(\frac{1}{BS_{CN(crit)}} - 1 \right)^3 \cdot \sqrt{[Bc]}} \quad (A4.10)$$

Implying the Critical Limits $CN_{BS(crit)}$ and $BS_{CN(crit)}$ in the equation for critical loads of acidifying depositions the following formula results:

$$CL(S + N) = BC^*_{dep} - Cl^*_{dep} + BC_w - Bc_u + N_u + N_{i(T)} + \frac{N_{de}}{1 - (CN_{BS(crit)} - CN_{min}) / (CN_{max} - CN_{min})} - ANC_{le(crit)} \quad (A4.11)$$

The calculation of $[ANC]_{crit}$ by a critical base saturation is given in ICP Modelling and Mapping (2004) (The calculation of $[ANC]_{crit}$ is alterable concerning carbonate and organic acids according to ICP Modelling and Mapping 2004):

$$ANC_{le(crit)} = -Q_{le} ([H]_{crit} + [Al]_{crit}) = -Q_{le} ([H]_{crit} + K_{Gibb} [H]_{crit}^3) \quad (A4.12)$$

$$[H]_{crit} = K_{Gap} \cdot \sqrt{[Bc]} \cdot \left(\frac{1}{BS_{CN(crit)}} - 1 \right) \quad (A4.13)$$

$$K_{Gap} = \frac{1}{k_{HBc} + k_{AlBc} \cdot K_{gibb}^{1/3}}$$

Annex 5 Prediction of vegetation changes with the dynamic BERN model

Implementation of the BERN model into dynamic models for soil changes

The development of vegetation in the past can be reconstructed with the help of the BERN-model. The typical natural plant community for the quite non changeable (fixed) site parameters like soil and (for the present) climate as well as the easily changeable (dynamic) parameters in a harmonious equilibrium of the regular site like soil moisture degree, base saturation and C/N ratio will be calculated from the model for the year 1880.

Based on the known plant history (in forests: composition of species, time of clearcut, afforestation, in grassland: type of management and intensity) it is possible to calculate changes of dynamic parameters triggered by landuse changes until 1960. The BERN-model derives the primary natural plant community.

The dynamic future development of the soil parameter C/N ratio, base saturation, temperature and precipitation determine the dynamic future development of the suitability of the site for different plant communities which could exist on that site because of the fixed site parameters of the soil. The development of such parameters in the future must base on assumptions (scenarios of deposition development, climate change scenarios and changes of ecosystem management). Realistic scenarios are the deposition scenarios in line with the commitments of the Gothenburg Protocol from 1999 (UN/ECE/CCE, 1999) to minimize emissions, the “middle climate scenario” of the International Panel of Climate Change (IPCC, 2001), the “ecological forest conversion”- scenario of the forest departments as well as their liming scenarios.

With the BERN-model it is possible to determine the potential natural plant community under the predicted changes of the nutrient balance and climate. The awaited natural plant community according to the prognosticated site parameters is determined by the model outputting the nearest optimum of a possibility distribution function of one of the natural communities from the same regular type of site. Those possibility spaces which are located closer to the predicted BS/CN dupel and continentality index / duration of vegetation period are determined and shown in Figure A5.1 for a site example. These examples of using BERN in the geochemical modelling were done with the model SAFE and the immobilization algorithm given in VSD / expert judgements. In the future time series of BS/C/N-pairs should be calculated with more sophisticated nitrogen process approaches like ForSAFE or SMART2.

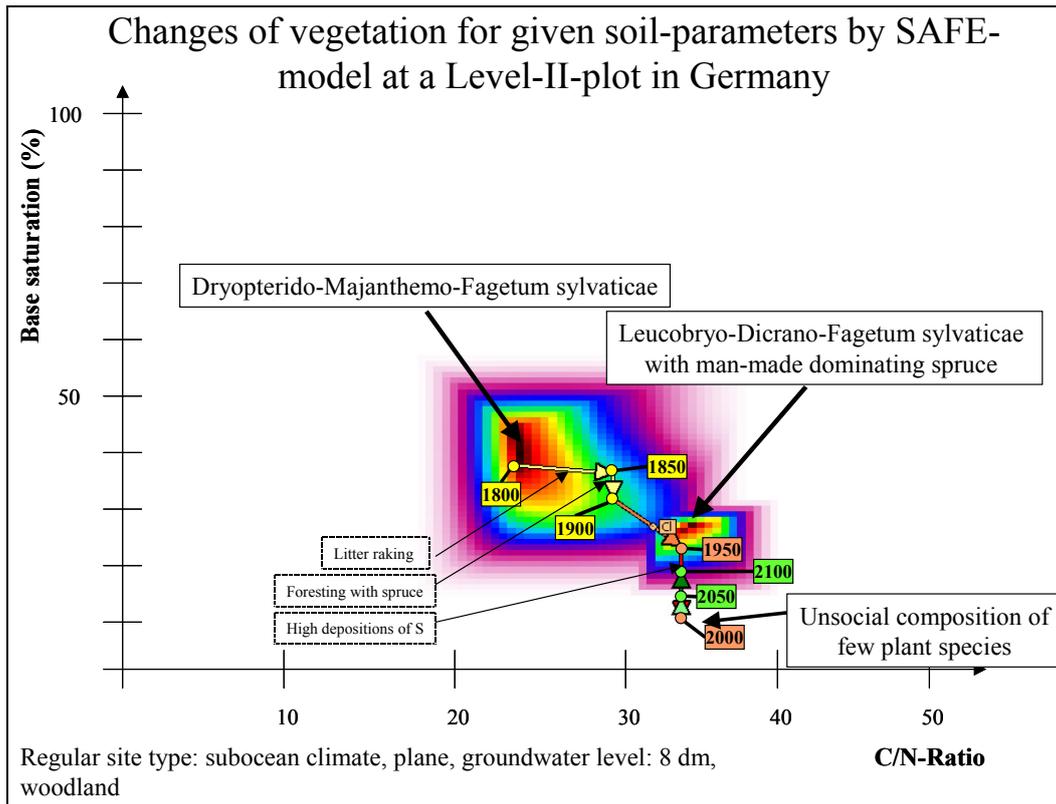


Figure A5.1 Dynamic modelling of vegetation changes using the SAFE-model for the changes of soil chemistry and coupling the BERN-model for an example of Level-II-plot in Germany

Combining effects of climate change and acidic deposition to biodiversity

Figure A5.2a shows the development of the soil chemical status calculated with a SAFE/VSD combination at a Level II plot in Saxonia, depending on the dynamic development of C/N ratio and BS based on the Gothenburg deposition scenario (UN/ECE/CCE, 1999). Figure A5.2b shows the changes of climatic parameters for this site according to (IPCC, 2001). The soil moisture status is assumed as constant. Figure A5.2c shows the changes in possibility measure of existence of certain plant communities. As one can see, no natural plant communities of middle Europe will have good conditions according to this abiotic scenario. In Figure A5.2d gradients of vitality of possibly forest tree species, which could exist, are given for this site.

The Level-II plot shows a reduced base saturation down to 6% in 1985. Simultaneously the Bc/Al-ratio reached 0.23 in the rooted zone down to 90 cm. Thus the beech is predicted not more able to exist here, while the pine and oak can root in deeper soil layers, where the Bc/Al-ratio is still kept at 1.2. The regarded Level-II plot is located in the suboceanic planar climate zone. The predicted development of precipitation and temperature leads to climate conditions in 2050 which can be found now in mediterranean subcontinental climate regions, but not yet in Germany. For this reason it is predicted that the native beech and beech-communities will not recover. Only pioneer species like Scottish pine and English oak with very large ecological niche widths for climate variables and with deep roots

have sufficient possibility for recovering. The influence of C/N-changes is not significant in the shown example.

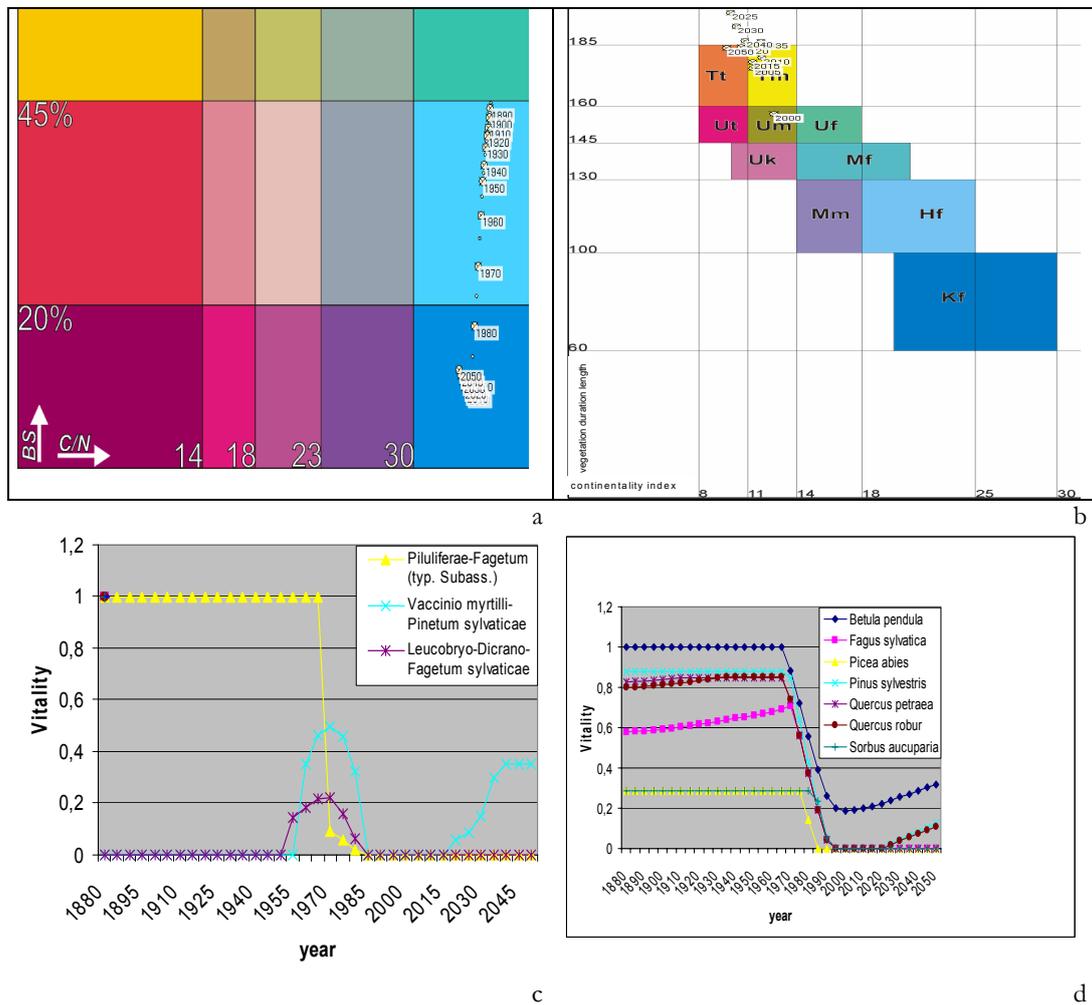


Figure A5.2 a: Site properties depending on base saturation [%] and C/N ratio 1880- 2050; b: Site properties depending on vegetation period [days] and continentality index 1880-2050; c: Vitality of forest plant communities depending on base saturation [%], C/N ratio, duration of vegetation period and continentality index 1880-2050; d: Vitality of tree species depending on base saturation [%], C/N ratio, duration of vegetation period and continentality index 1880-2050

With decreasing site suitability the degree of vitality depending on competition capacity, resistibility against environmental influences like windfall, breakage caused by snow, frost, destructive insects and pests decreases. The amount of trees within the stock increases which doesn't reach their optimal age for economic utilization. If one calculates the inverse value of that vitality factor, one will get an index for latent mortality. Therefore with decreasing site suitability the stability of the expected amount of harvestable wood declines as well. Due to poor knowledge in ecosystem recovery after abating the deposition of acidifying and eutrophying components, the time lag of vegetation recovery is not known.

Determination of the recovery target and the current regeneration potential

If the actual condition lies not far from the primary natural condition (Figure A5.3), then, as a rule, populations of one or several constant dominant species of the primary natural plant community are present to a smaller or larger extent. With decreasing impairment inputs, these species could regain their full vitality, which would initialize the return of the primary natural plant community. This self-regenerating process initiated by decreasing loads will lead to a return to the primary natural conditions within a short period of time.

If the current condition, however, lies far off from the primary natural condition (Figure A5.3), then, as a rule, every dominant constant species and other constant species of the primary natural plant community are extinct.

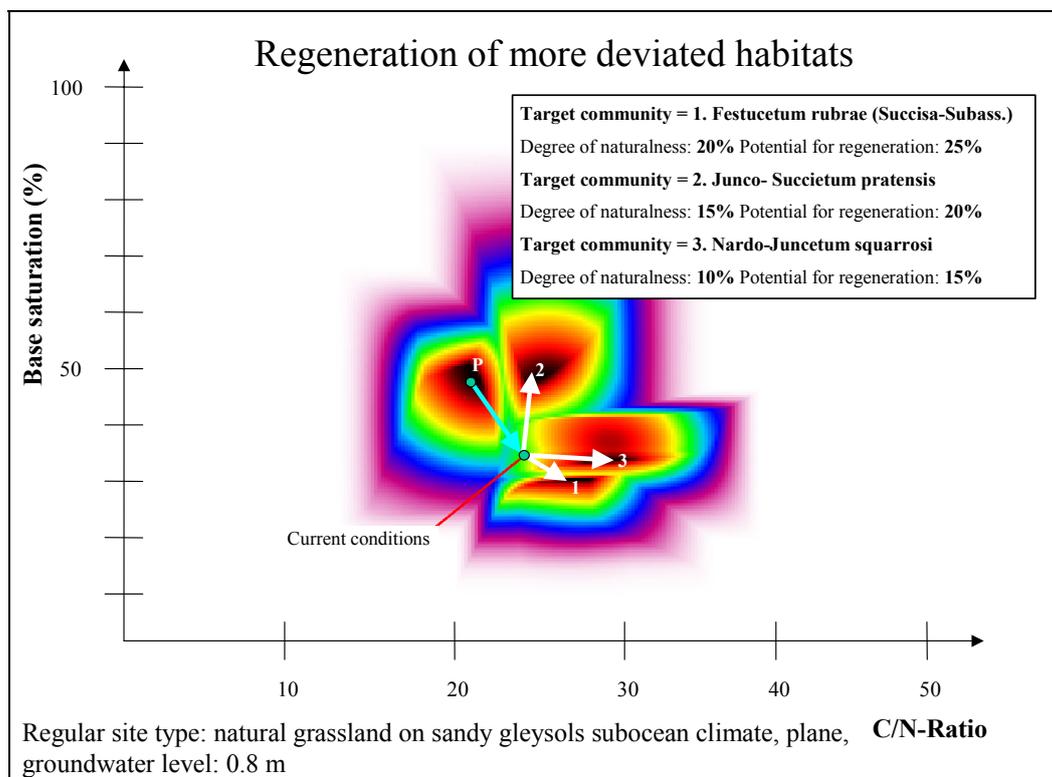


Figure A5.3 Possible regeneration targets for a current habitat, which has greatly deviated from its primary natural condition.

Instead, new species (that are better adapted to the changed soil properties) have immigrated. One or several of the immigrant species, however, could be dominant species of a potentially natural plant community within a series of natural communities on the harmonious C/N- BS vector within a habitat regular type. When this is the case and the loads are decreasing towards a new level of harmonious nature-identical equilibrium, a new natural plant community can be developed, which, like the primary natural community, can exhibit high species diversity and high ecological functionality. Which one, if either, the natural plant community or the semi-natural plant community becomes the “target” of ecosystem management is (in the end) a political decision and depends on the preferred development targets. For

example, in a protected area, fertilization would be completely excluded as a policy option whereas calcareous fertilization could be a meaningful measure in a forest where economical utilization takes place.

For the definition of the recovery target, one condition is the determination of the actually existing degree of remaining “regeneration potential” of the current species composition at the habitat in question. The following definitions apply:

Regeneration potential R_{pot} = proportion of immigrating species to expect, which may arrive during the estimated recovery duration, compared with the total species list of the potential natural plant community (constant species) at the recovery target A_{pot} . It is a function of the soil-chemical regenerative power and the propagation behaviour of the potential dominant and character species.

Possible targets of recovery =

1. Target: (a slightly changed ecosystem): the primary natural habitat condition
2. Target: (a highly changed ecosystem): the re-establishment of the nearest balanced nature-identical equilibrium of N, C and Bc, which is appropriate for the current (disharmonic) condition - the most quickly reached by self-regeneration processes
3. Target: (an irreversible changed ecosystem): re-establishment of nearest balanced nature-identical equilibrium of N, C and Bc, which is appropriate for the current (disharmonic) condition – with the help of ecosystem management

A development target for a highly changed ecosystem - a target that would be reasonable under many criteria - should be, therefore, the re-establishment of a balanced nature-identical equilibrium in the nutrient, water, and energy balance. This can be the nearest equilibrium of N, C and Bc, which is appropriate for the current (disharmonic) condition (Figure A5.3). This condition of equilibrium would be the one reached the most quickly through self-regeneration processes, which would show a high species diversity with a high ecological functionality. A complete, independent return to the primary natural condition becomes, as a rule, a very long in coming proposition, or is completely impossible because of irreversible changes to the soil.

In particular, where the concentration of base cations in the soil solution (Figure A5.4) are strongly diluted and the dilution has already reached into deeper soil layers, one may no longer subsequently assume that sufficient base cations can be delivered from the weathering of parent material into the root-zone. On the one hand, the deep-rooted plant species which could carry the base cations to the surface (e.g., the trees in the forest or the grasses of the meadows and pastures), have experienced such a strong depression in their growth that this performance can hardly be completed. This growth depression is caused when a critical relationship of Bc/Al in the soil solution is reached. The plants, which themselves normally carry out the largest part of the recovery of a harmonious nutrient household in the soil, already are no longer productive or are already dead. On the other hand, soil-chemical

processes have led to a destabilization of the soil content in the area of change- e.g., from aluminium to the iron buffer - which in extreme cases is no longer reversible.

In extreme cases, the nutrient household of some ecological systems is so strongly disharmoniously changed (usually through long and very strong acidification with simultaneous eutrophication) that the potential for self-regeneration to a nature-identical equilibrium has completely disappeared (Figure A5.4). The border of regenerative power is exceeded irreversibly. These ecosystems are characterized by the absence of species, which could function in any potential natural plant community as dominant constant species (degree of “Naturalness” = 0). In such sites, only species with very broad ecological niches occur. Such species can occur very irregularly in many communities of the regular site type. However, these can never arise to a dominant species of a potential natural plant community because they would not be competitive enough in the presence of many other species.

A characteristic of irreversibly changed habitats (with an extremely disharmonious nutrient household) is, in forests, the presence of the humus type “raw humus” with simultaneously high contents of N in the humus layer and a BS in the upper mineral soil layer of less than 10 percent.

It is thus necessary to assess both 1) the maximum stress threshold (= Critical Limit function) in the sense defined so far (UBA, 1996), and 2) the limit of regeneration ability, at which after it is exceeded, no more self-recovery takes place (“line of no return”).

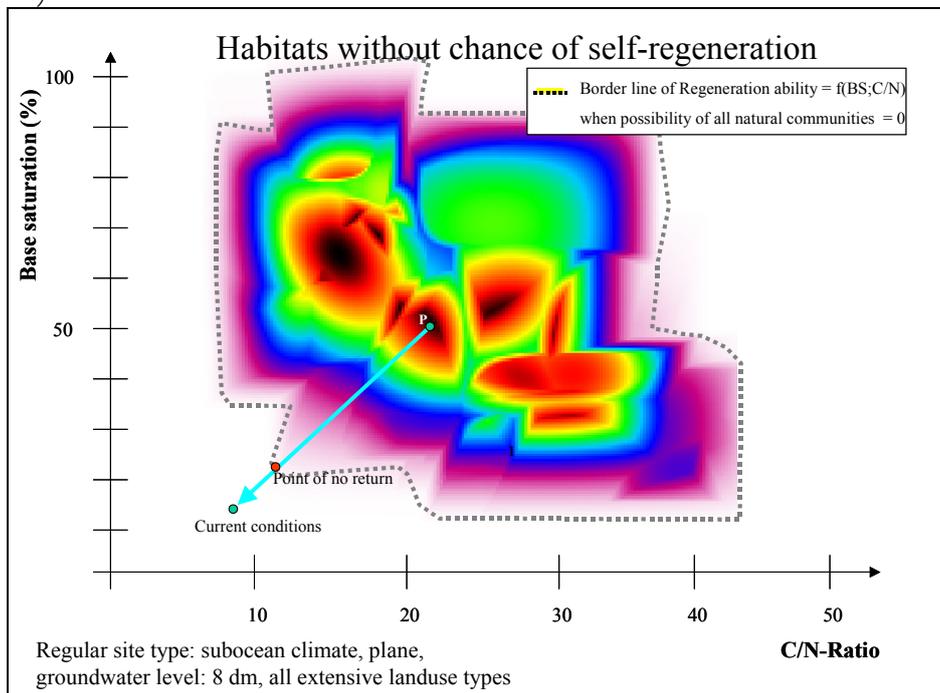


Figure A5.4 Function of regeneration ability. The indicated and a current highly changed habitat has no chance for self regeneration

Annex 6 Assessment of individual responses functions in the ForSAFE-VEG model

In the ForSAFE-VEG model, the strength of each species is the product of 10 controlling factors, according to:

$$S_i = f_1(N) \cdot f_2(P) \cdot f_3(\text{acid}) \cdot f_4(\text{H}_2\text{O}) \cdot f_5(G) \cdot f_6(T) \cdot f_7(V) \cdot f_8(I) \cdot f_9(C) \cdot f_{10}(\text{CO}_2)$$

Where $f_1(N)$ and $f_2(P)$ are the nitrogen and phosphorus response function respectively, $f_3(\text{acid})$ is the soil acidity response function, $f_4(\text{H}_2\text{O})$ is the water response function, $f_5(T)$ is the air temperature response function, $f_6(I)$ is the light intensity response function, $f_7(G)$ is the grazing effect function, $f_8(V)$ is the wind tatter damage effect, $f_9(C)$ is the competition function representing above ground competition for light depending on plant height, and belowground competition for water and nutrients and $f_{10}(\text{CO}_2)$ is the ambient air CO_2 response function. A description of the various functions is given below.

Nitrogen

Nitrogen has two effects on plant establishment and growth at a site. At low nitrogen soil contents, N addition promotes growth, and so has a positive effect. On the other hand, excess nitrogen may cause retardation in growth or even plant damage. The nitrogen response function is thus the product of the promoting and the limiting functions:

$$f(N) = f(N)_+ \cdot f(N)_- \tag{A6.1}$$

Where $f(N)_+$ is the promoting function and is given by:

$$f(N)_+ = a_0 \cdot \frac{[N]^{w_+}}{k_+ + [N]^{w_+}} \tag{A6.2}$$

And $f(N)_-$ is the inhibiting function and is given by:

$$f(N)_- = \frac{k_-}{k_- + [N]^{w_-}} \tag{A6.3}$$

For plants with a fast positive response to nitrogen, $w_+ = 3$, while it is equal to 2 and 1 for normal and slow reacting plants. w_- is set to 3 for fast inhibited plants at N exceedance, and 1 for slow responding plants. Figure A6.1 shows both promotion and retardation curves in response to nitrogen availability for different plant groups. Each group has a specific k value which determines the starting point of the response to N availability, either positively or negatively.

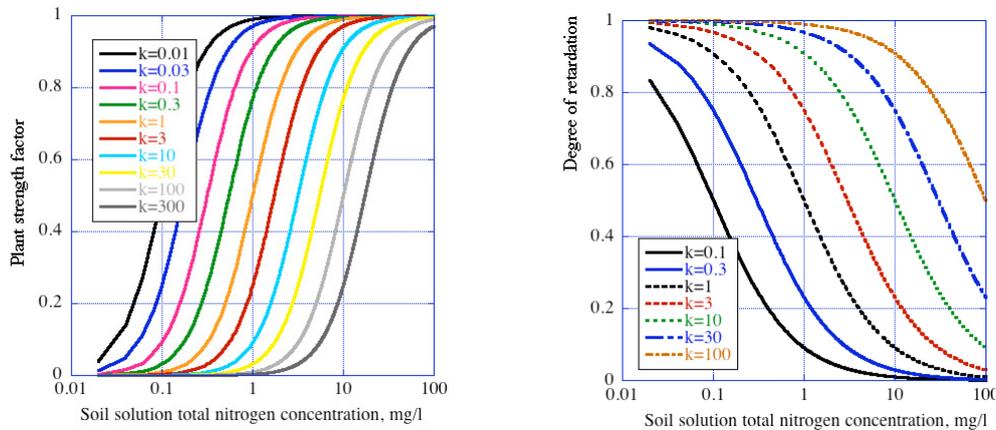


Figure A6.1 The promoting and retarding effects of nitrogen for different plant classes.

Phosphorus

A phosphorus response function has been developed, and phosphorus dynamics are now being incorporated into the model. Phosphorus has a promoting effect on all plant groups represented in the model. In natural habitats, high enough P concentrations that could inhibit growth do not normally happen. For this reason, and considering the early stage of incorporating P in the model at this time, only a promotion effect by P will be considered (Figure A6.2).

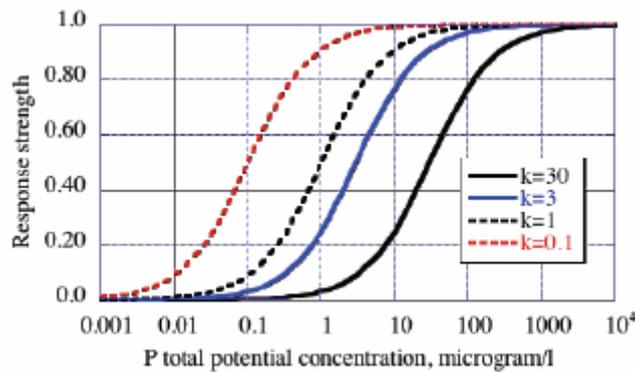


Figure A6.2 The promoting effect of phosphorus, the factor k is presented in Table A7.1 as kP .

Acidity

Acidity affects plant growth negatively both through low pH and high concentrations of Al^{3+} associated with soil acidity. The response function to acidity is thus assumed to be the minimum between the response to low pH and a high Al^{3+} concentration. The negative effect of aluminium is counteracted by the presence of base cations, and the base cations to aluminium ratio (BC/Al) is used to indicate the effect of acidity through high Al^{3+} concentrations.

$$f(\text{acid}) = \min[f(\text{BC/Al}), f(\text{pH})] \quad (\text{A6.4})$$

The BC/Al ratio acts positively on the relative plant strength according to the following equation:

$$f(\text{BC/Al}) = \frac{[\text{BC}]^m}{[\text{BC}]^m + k_{\text{acid}} \cdot (m \cdot [\text{H}^+] + [\text{Al}^{3+}])^n} \quad (\text{A6.5})$$

Where

$$[\text{BC}] = [\text{BC}^{2+}] = [\text{Ca}^{2+}] + [\text{Mg}^{2+}] + 0.5 \cdot [\text{K}^+]$$

The ensuing formula has the formulation of a Michaelis-Menten formula. The coefficient k_{acid} can be found tabulated for approximately 150 different trees, bushes, ground plants and crop plants in Sverdrup and Warfvinge (1993), based on a synthesis of nearly 300 experiments on plants all over the world. The response function for the BC/Al ratio was simplified by setting $m=n=1$, while some plants actually have $m=3$, $n=2$ and salix and coffee have $m=1/3$ and $n=1/2$. $[\text{H}^+]$ and $[\text{Al}^{3+}]$ were related by the following equation:

$$[\text{Al}^{3+}] = k_{G1} \cdot [\text{H}^+]^3 + k_{G2} \quad (\text{A6.6})$$

The constants k_{G1} and k_{G2} change through the soil horizon (Sverdrup et al., 2006). The response function of the BC/Al ratio can then be simplified to the following equation:

$$f(\text{BC/Al}) = \frac{\frac{[\text{BC}]}{[\text{Al}]}}{k_{\text{BC/Al}} + \frac{[\text{BC}]}{[\text{Al}]}} \quad (\text{A6.7})$$

Where $k_{\text{BC/Al}}$ is a plant group specific parameter given in Table A7.1. For plants with no roots, the response functions takes into consideration the effect of H^+ protons instead of Al^{3+} , and the equation becomes:

$$f(\text{BC/Al}) = \frac{\frac{[\text{BC}]}{[\text{H}^+]}}{3 \cdot k_{\text{BC/Al}} + \frac{[\text{BC}]}{[\text{H}^+]}} \quad (\text{A6.8})$$

The factor 3 accounts for the H^+ taking up as much adsorption space as one Al^{3+} . Figure A6.3 below shows the response curves to the BC/Al ratio for some plant groups.

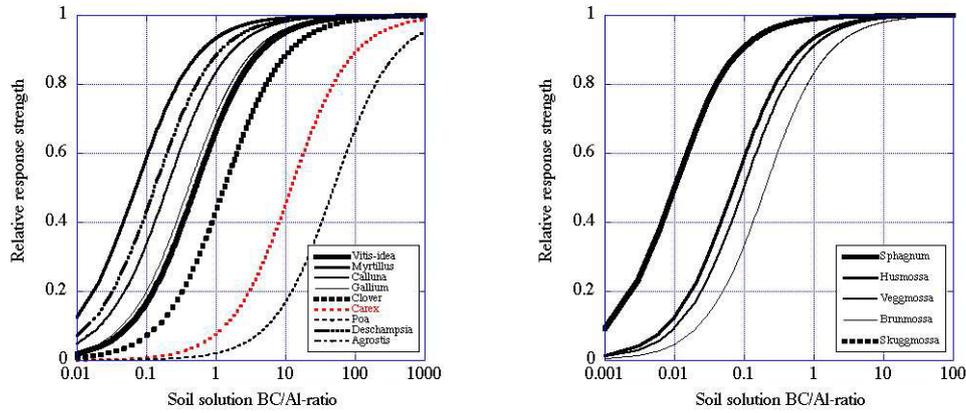


Figure A6.3 The response curves to the soil solution BC/Al-ratio for selected plants.

The effect of acidity can also be accounted for by using pH or $[H^+]$ only. In this case, the response function to acidity becomes:

$$f(\text{pH}) = \frac{1}{1 + k_{\text{pH}} \cdot [H^+]} \quad (\text{A6.9})$$

Where k_{pH} is a plant specific parameter that can be derived from $k_{\text{BC/Al}}$ using the following equation:

$$k_{\text{pH}} = \frac{3 \cdot k_{\text{BC/Al}}}{[BC^{2+}]} \quad (\text{A6.10})$$

Many plants appear to have an upper limit of pH at which growth is impeded, possibly by blocking the uptake of nutrients taken up as anions. An upper limit of pH value (pH_{high}) is defined for each plant where $[OH^-]$ is high enough to reduce NPP by half or more. A retardation pH function is then defined as follows:

$$f(OH) = \frac{[H^+]}{[H^+] + 0.33 \cdot 10^{-\text{pH}_{\text{high}}}} \quad (\text{A6.11})$$

The promoting and limiting effects of pH on plant growth are illustrated in Figure A6.4.

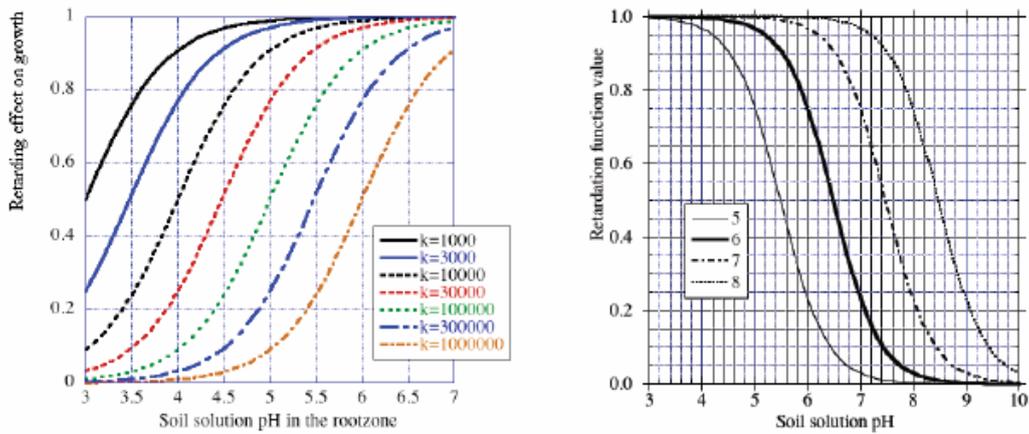


Figure A6.4 pH has both a promoting effect at low values and a retarding effect at high values. The diagram to the right shows response curves for plants with a limiting pH value at 5, 6, 7 and 8.

Calcifugicity

High Ca concentrations in the soil may have a negative effect on the uptake of Mg and P of some plant groups, referred to as calcifuges (Larcher, 1975). To account for the negative impact of high Ca concentrations in the soil solution, the following retardation equation is used:

$$f(BC) = \frac{1}{1 + k_{Ca} \cdot [Ca^{2+}]^2} \quad (A6.12)$$

Where the coefficient $k_{Ca} = 10^{-6}$. the equation does not apply to all plant groups, but only to calcifuge plants such as Hylocomium, Sphagnum, Holcus mollis, and Rhododendron. The retardation response function is shown in Figure A6.5.

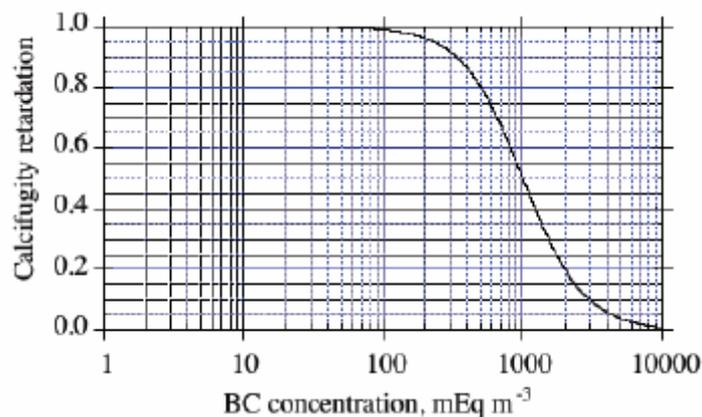


Figure A6.5 The response function of the calcifugicity retardation effect.

Soil moisture, temperature and light intensity

The three abiotic factors of soil moisture, temperature, and light intensity affect plant growth in the same basic manner according to the response function shown in Figure A6.6. For each of these factors, growth is initialised at a threshold value, after which growth responds positively to an increase in the availability of the factor until a saturation point after which no growth enhancement appears as a result of the factors increase. Beyond the saturation point lies a decline threshold after which an increase in the factor causes a decline in plant growth. This decline ultimately reaches an extreme point at which the plant does not survive. While the behaviour of the three factors, moisture, temperature, and light is similar, the threshold points as well as the non-survival points are different between the factors and between the plant groups. Light intensity is assumed to have neither decline threshold nor a non-survival point.

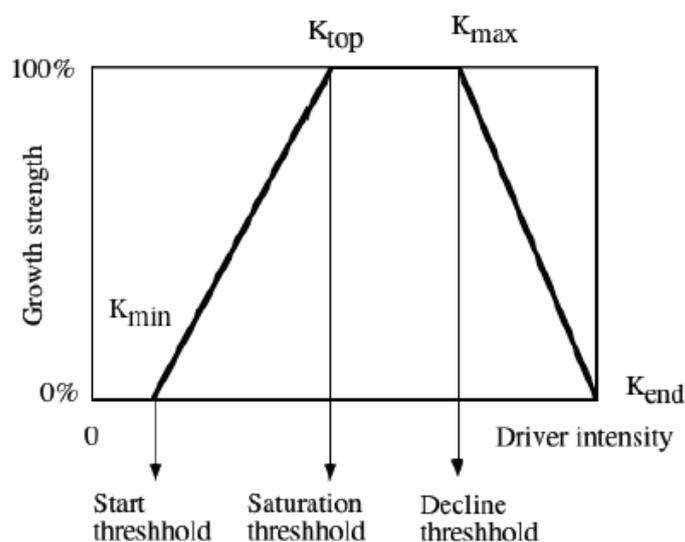


Figure A6.6 The basic shape of the plant response to moisture, temperature and light intensity is characterised by threshold points which delimit the positive, neutral, and negative effects of the drivers.

Browsing and grazing

The grazing pressure was given as input to ForSAFE-VEG, and was separately simulated for Sweden using the HÄLGE model (Sverdrup et al., 2006). HÄLGE was developed to estimate the regional grazing pressure for three regions in Sweden (Figure A6.7). The output from the grazing model was given as input to ForSAFE-VEG. Within the VEG module, a selectivity parameter (kG in Table A7.1) is set on each plant group to account for the grazers' preference for each specific plant. Palatable plants have high selectivity parameters as the animals prefer them, and would thus suffer most from the grazing.

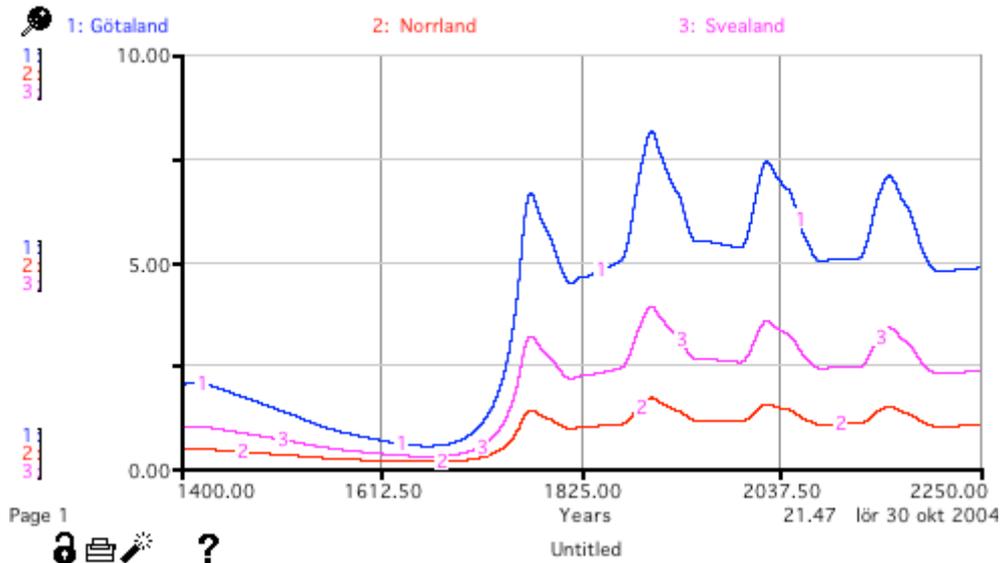


Figure A6.7 Grazing pressure (moos units km^2) for three regions in Sweden: Götaland (south), Norrland (north) and Svealand (centre). The grazing pressure time series were given as input to ForSAFE-VEG.

Windtatter and wind chill

The wind effect function contains two components, first for the lowering the ambient temperature (chill effect), and the second for causing physical damage on the plants (wind tatter). The chill effect reflects the effective lowering of air temperature with increased wind speed. The effective temperature (T_{eff} °C) experienced by plants subjected to wind chill is given by:

$$T_{\text{eff}} = T - \Delta T \quad (\text{A6.13})$$

Where T is the ambient air temperature unaffected by wind and ΔT (°C) is the decrease in temperature caused by the wind. ΔT is a function of wind speed v (m s^{-1}):

$$\Delta T = 1.14 \cdot v + 0.0042 \cdot v^2 \quad (\text{A6.14})$$

The actual wind speed is reduced closer to the ground surface due to friction. To account for this effect, the effective wind speed felt by a plant of height h (m) is given by:

$$v = v_{\text{max}} \cdot \left(1 + 0.5 \cdot \log \frac{h}{\delta}\right) \quad (\text{A6.15})$$

Where v_{max} (m s^{-1}) is the maximum wind temperature unhampered by landscape roughness, and δ (m) is the layer boundary thickness which can change between 1 and 6 m depending on the roughness and topography of the landscape.

Besides the wind chill effect, wind can cause physical damage to plants in a process referred to as windtatter. As with the wind chill, windtatter is a function of the wind velocity, and the mechanical abrasion function is given by:

$$f(V) = \frac{1}{1 + k_v \cdot v^2} \quad (\text{A6.16})$$

Where k_v is a plant group specific windtatter coefficient and v is the effective wind velocity as given above.

Aboveground and belowground competition

Plant height and root depth is used to estimate aboveground and belowground competition respectively. Aboveground, the plants compete for light and shade with each other. This effect is a function of the plant height and is given by:

$$f(C) = 0.0273 \cdot e^{\beta \cdot h} \quad (\text{A6.17})$$

Where h is the height of the plants and β is the exponent in the shape factor and has been set to 3 for all plant groups used in the test simulations.

Belowground, plants compete with their roots for access to water and nutrients. The belowground competition manifests through the root distribution and is imbedded in the responses to soil parameters, such as acidity, N and water availability. Plants with deeper root systems will have an advantage in that they have access to deeper resources. Four groups of root types have been assumed (Table A6.1)

Table A6.1 Root distribution groups as used in ForSAFE-VEG for belowground competition

Root type	% root mass in 0-0.1m	% root mass in 0.1-0.2m	% root mass in 0.2-0.4m	% root mass in 0.4-1.0m
0	100	0	0	0
1	70	20	10	0
2	20	60	20	0
3	20	30	30	20

Ambient air CO₂

Historic contents of atmospheric CO₂ were around the value of 270ppm. According to the IPCC this level increased steadily since the 1800s and is today at 350ppm (year 2000). The predictions by the IPCC expect an increase in atmospheric CO₂ content between 500 and 1200ppm. To account for the possible effect of changes in atmospheric CO₂ over the ground vegetation composition, two different functions are used to account for enhancement in growth of C3 and C4 plants separately. For C4 plants, the growth enhancement is given by:

$$f(\text{CO}_2) = 1.33 \cdot \frac{P_{\text{CO}_2} - 70}{P_{\text{CO}_2} + 30} \quad (\text{A6.18})$$

And for C3 plants, the equation is similar in form but parameterised differently as:

$$f(\text{CO}_2) = 3.33 \cdot \frac{P_{\text{CO}_2} - 70}{P_{\text{CO}_2} + 580} \quad (\text{A6.19})$$

At elevated CO_2 concentrations in the air, C3 plants are favoured.

Annex 7 Prediction and validation of vegetation changes with the dynamic ForSAFE-VEG model

Study sites and model parameterization.

The ForSAFE-VEG model was used to study changes in the soil chemistry, land cover, and changes in the ground vegetation cover at 16 Swedish forest sites (Figure A7.1). The sites are part of the ICP level II monitoring network (International Co-operative Program on Assessment and Monitoring of Air Pollution Effects on Forests, <http://www.icp-forests.org>). Forty two plant groups and nine tree seedling types have been identified to represent the ground vegetation in Sweden. These plant groups are assumed to be potentially present throughout Sweden, but are only expected to manifest where environmental conditions are favourable. For each of the identified plant groups, parameters were defined for all the specific response functions described above. Table A7.1 summarises the parameterisation for Sweden.

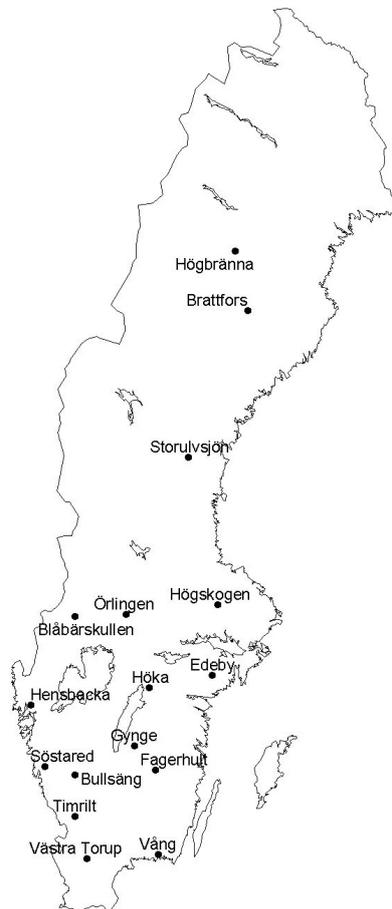


Figure A7.1 16 Swedish sites used to test the ForSAFE-VEG model.

Table A7.1 42 plant groups and 9 seedlings were parameterised for Sweden in order to model the changes due to the different response functions described above.

Latin name	τ years	[N]					[H ⁺]			W			T			I			h (m)	root class	kP	kG
		a0	k+	w+	k-	w-	Kbc/al	kbc	kph	min	top	max	min	top	max	min	max	max				
Cladonia_lichen	20	1	0.01	1	0.003	3	0.07	0	1050	-0.2	0.05	0.25	-2.5	5.5	13.5	500	2500	0.05	0	0.1	0.7	
Hylocomium_mosses	20	1	0.03	1	-	-	0.07	150000	1050	0.05	0.15	0.35	-1	7	15	100	2500	0.02	0	3	0	
Mnium_mosses	20	1	0.3	2	-	-	0.4	0	6000	0.15	0.25	0.6	0	8	16	50	2500	0.02	0	3	0	
Sphagnum_moss	20	1	0.03	1	0.1	3	0.01	150000	150	0.4	0.6	1	-1	7	15	100	2500	0.02	0	1	0	
Calluna_vulgaris	30	1.4	0.2	1	3	3	0.2	0	3000	-0.25	0.15	0.4	-1	7	15	500	5000	0.25	2	1	0.7	
Empetrum_nigrum	15	1.6	0.03	1	0.003	3	0.2	150000	3000	-0.2	0.1	0.4	-1.5	6.5	14	500	5000	0.1	1	1	0	
Erica_tetralix	15	1.6	0.3	1	0.03	3	0.4	0	6000	0.2	0.35	0.6	0	8	16	1000	5000	0.15	1	1	0	
Vaccinium_myrtillus	10	1.6	0.1	1	0.1	3	0.1	0	1500	-0.1	0.15	0.5	-1	5	11	100	2000	0.3	1	1	2.3	
Vaccinium_vitis-idea	15	1.6	0.03	1	0.003	3	0.35	0	5250	-0.2	0.1	0.45	-1.5	4.5	10.5	500	4000	0.15	1	1	0.7	
Agrostis_capillaries	10	1	0.5	2	-	-	0.2	0	3000	0.05	0.15	0.5	3	11	19	750	4000	0.25	2	3	2.3	
Brachiopodium_pennatum	5	1	20	2	-	-	6	0	3500	0.1	0.2	0.35	3	11	19	1000	3500	0.5	1	3	9	
Bromus_benekenii	5	1	20	2	-	-	12	0	180000	0.1	0.2	0.4	5	13	21	250	3000	0.6	2	30	9	
Calamagrostis_arundinasius	5	1	0.5	2	-	-	1.8	0	20800	0.1	0.2	0.4	2	10	18	750	3500	0.5	2	3	0.67	
Deschampsia_cespitosa	5	1	0.5	2	-	-	0.2	0	3000	0.15	0.35	0.6	3	11	19	1000	5000	0.35	2	3	0	
Deschampsia_flexuosa	5	1	0.05	2	-	-	0.13	0	1950	0.05	0.15	0.3	-1	7	15	250	3000	0.2	2	3	2.3	
Festuca_ovina	10	1.4	0.02	2	10	1	0.1	0	1500	-0.25	0.05	0.25	3	11	19	1500	5000	0.1	1	30	0.67	
Milium_effusum	5	1	20	2	-	-	8	0	150000	0.15	0.45	0.6	5	15	20	250	3000	0.5	2	3	9	
Molinia_caerulea	5	1	1	2	-	-	0.2	0	3000	0.2	0.3	0.45	5	13	21	1000	5500	0.4	2	30	2.3	
Nardus_stricta	10	1.2	0.05	2	10	1	0.2	150000	3000	0.15	0.25	0.4	0	8	16	1500	5000	0.15	2	1	0	
Poa_nemoralis	5	1	5	2	-	-	8	0	120000	0.05	0.1	0.2	2	10	20	1250	5000	0.4	2	3	9	
Dryopteris_dilata_coll	20	1	0.5	2	-	-	2	0	30000	0.1	0.3	0.5	3	11	19	150	2500	0.4	2	1	2.3	
Pteridium	20	1	0.5	2	-	-	12	0	180000	0.05	0.2	0.3	2	8	18	750	3250	0.5	2	1	0	
Aconitum_lycoctonum	20	1	5	2	-	-	10	0	150000	0.25	0.55	0.9	2	6	10	1000	5000	1	2	1	0	
Allium_ursinum	2	1	20	2	-	-	40	0	600000	0.25	0.2	0.6	4	12	20	250	5000	0.25	2	30	0	
Anemone_nemorosa	10	1	0.5	2	-	-	0.8	0	12000	0.2	0.3	0.4	2	10	18	250	3500	0.15	1	3	2.3	
Antennaria_diocia	5	1	0.01	2	-	-	0.1	0	1500	0.05	0.1	0.2	0	6	12	2000	5500	0.01	1	1	0	
Arnica_montana	5	1	0.01	2	-	-	0.6	0	9000	0.05	0.1	0.2	7	15	20	2000	5500	0.01	1	1	0	
Epilobium_augustifolium	5	1	1	2	-	-	2	0	30000	0.15	0.2	0.3	0	8	20	1750	5500	0.8	2	3	32	
Galium_odorata	3	1	5	2	-	-	1.2	0	18000	0.15	0.25	0.4	3	11	19	250	3000	0.15	1	1	0.67	
Geranium_sylvestrum	3	1	1	2	-	-	1.8	0	27000	0.15	0.25	0.4	2	10	14	500	3000	0.5	2	3	9	

Latin name	τ years	[N]					[H ⁺]			W				T			I		h (m)	root class	kP	kG
		a0	k+	w+	k-	w-	Kbc/al	kbc	kph	min	top	max	min	top	max	min	max					
Hepatica_nobilis	20	1	1	2	-	-	8		0	120000	0.15	0.25	0.4	2	10	18	375	3000	0.5	1	3	0
Mercurialis_perennis	5	1	5	2	-	-	2		0	30000	0.1	0.25	0.4	5	15	20	1000	5000	0.5	1	1	0
Origanum_vulgare	20	1	0.5	2	30	1	10		0	150000	0.05	0.15	0.25	4	12	20	1500	6000	0.04	2	3	0.67
Oxalis_acetocella	2	1	0.5	2	-	-	0.2		0	3000	0.1	0.2	0.4	0	8	18	250	3000	0.05	1	1	0
Trientalis	2	1	0.5	2	10	1	0.2		0	3000	0.1	0.2	0.4	2	10	18	250	3000	0.15	1	1	0.67
Trifolium_repens	5	1	1	0	-	-	1.3		0	19500	0.2	0.35	0.4	5	15	25	1250	5500	0.3	2	1	32
Urtica_dioica	5	1	5	2	-	-	10		0	150000	0.15	0.25	0.45	2	10	20	500	5000	0.8	1	3	0
Norway spruce	100	1	0.3	2	30	1	0.33		0	5000	0.1	0.4	0.9	5	15	20	400	700	0.25	1	3	0.7
Sitka spruce	110	1	0.1	2	3	1	0.07		0	1050	0.15	0.45	0.9	2	12	17	600	700	0.25	1	1	0.7
Scots pine	150	1	1	2	100	1	0.28		0	4730	0.05	0.3	0.8	3	13	18	1200	2296	0.2	2	1	0.7
Larch	70	1	1	2	30	1	0.5		0	7500	0.05	0.25	0.8	6	16	25	400	700	0.2	2	3	0.7
Birch	60	1	1	2	100	1	0.25		0	4000	0.15	0.45	0.9	2	12	17	800	1600	0.2	2	1	9
Beech	120	1	3	2	300	1	0.22		0	3500	0.15	0.45	0.7	6	16	30	320	600	0.25	3	3	2.3
Oak	160	1	3	2	300	1	0.2		0	3000	0.05	0.3	0.7	6.5	16.5	35	600	800	0.2	3	3	9
Ash	80	1	1	2	-	-	0.25		0	4000	0.15	0.45	0.7	7	17	35	600	1600	0.25	2	3	9
Norway maple	80	1	3	2	-	-	0.25		0	4000	0.05	0.3	0.7	5.5	15.5	25	160	280	0.2	3	3	9
Myrica_gale	10	1	1	2	-	-	0.8		0	12000	0.25	0.35	0.6	3	7	18	1500	4000	0.6	2	1	0.67
Rhododendron_toment	10	1	0.03	2	-	-	0.2	150000	0	3000	0.25	0.35	0.5	-1	5	9	1000	3500	0.5	2	1	0
Rubus_idaeus	5	1	1	2	-	-	1		0	15000	0.15	0.25	0.4	2	10	18	1500	5000	0.8	2	3	9
Salix_lanata	30	1	0.5	1	0.1	3	1		0	9000	0.15	0.35	0.6	-2	2	6	1000	4000	1.2	3	1	2.3
Salix_myrsinifolia	30	1	0.5	2	-	-	0.5		0	9000	0.15	0.35	0.6	-1	5	11	1000	4000	1.2	3	1	9

The parameterization partly rest upon a variety of literature sources (Larcher, 1975; Fridriksson & Sigurdsson, 1983; Bergthorsson, 1985; Hanks & Ritchie, 1991; Kimmins, 1991; Ellenberg et al., 1992; Falkengren-Grerup, 1992; Sverdrup & Warfvinge, 1993; Latour et al., 1994; Tilman, 1994; Hansson, 1995; Roberts & Gilliam, 1995; Lambers et al., 1998; Ten Brink et al., 2000) as well as a Delphi process involving the authors and Prof. Lars Erikson of Umeå University, Sweden. For these single estimates of the critical load of acidity, nitrogen, climate change, or with more detail, critical load response surfaces can be estimated using presently available models and datasets. The regional distribution is 660 sites at present, but with prospects of expansion to 1600 in dynamic mode. In static calculation mode (as for PROFILE), 26,000 sites are available for adaptation in Sweden. In the present study, only ground vegetation plant criteria were used.

The sites cover a wide range of climatic conditions, soils, fire regimes, atmospheric deposition gradients and management histories. ForSAFE-VEG was used to simulate the changes in soil chemistry, hydrology and tree biomass according to these conditions, and the composition of the ground vegetation was subsequently derived. Atmospheric deposition data for $\text{NO}_3^- + \text{NH}_4^+$ and SO_4^{2-} (Figure A7.2), among other elements, were given in input according to the 1999 UN ECE LRTAP Gothenburg protocol (Schöpp et al., 2003).

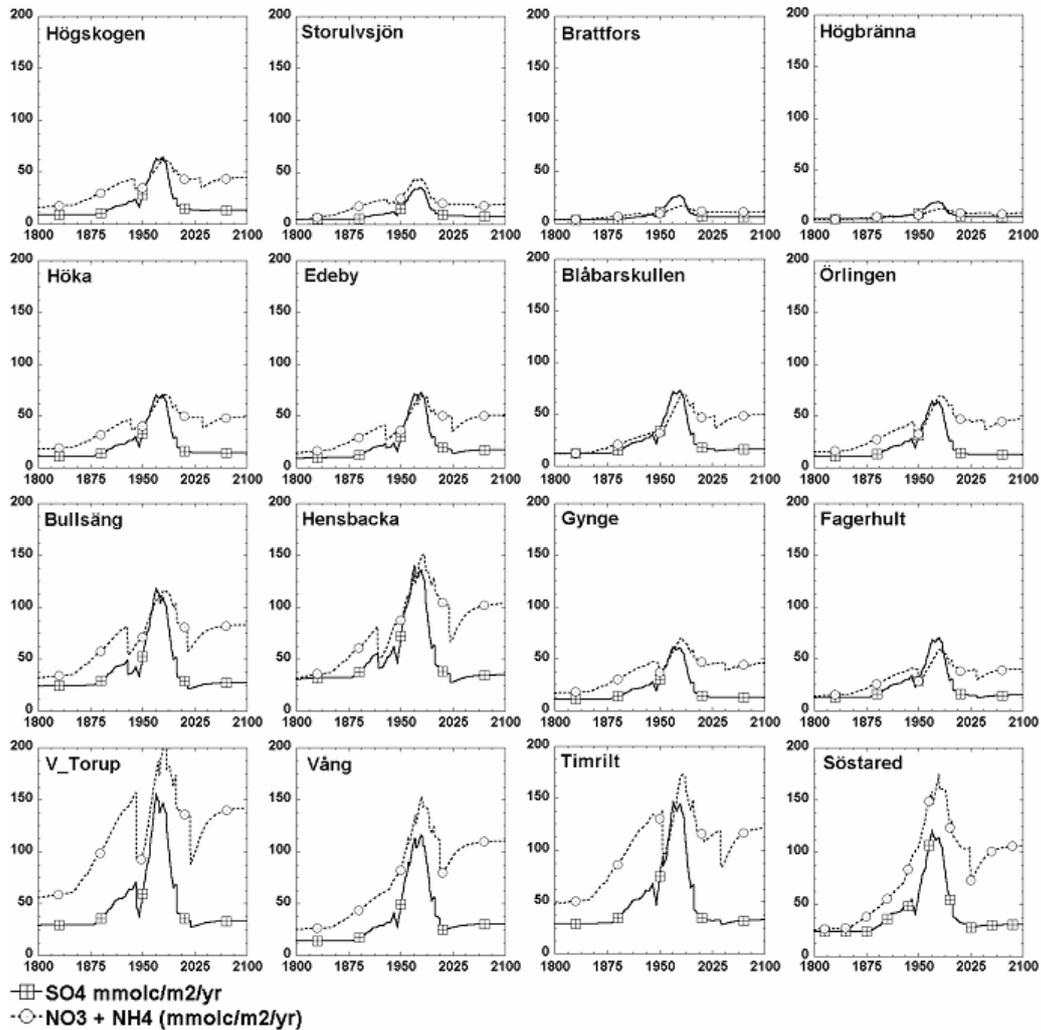


Figure A7.2 The atmospheric deposition of N and S show a clear increase from north to south.

The sites were subject to different histories involving fire regimes, alterations between open fields and forest cover as well as different harvesting regimes depending on the location of each site. The specific site history is summarised in Figure A7.3 below. As will be shown in the results of the study, the land use histories have a great effect on the composition of the ground vegetation.

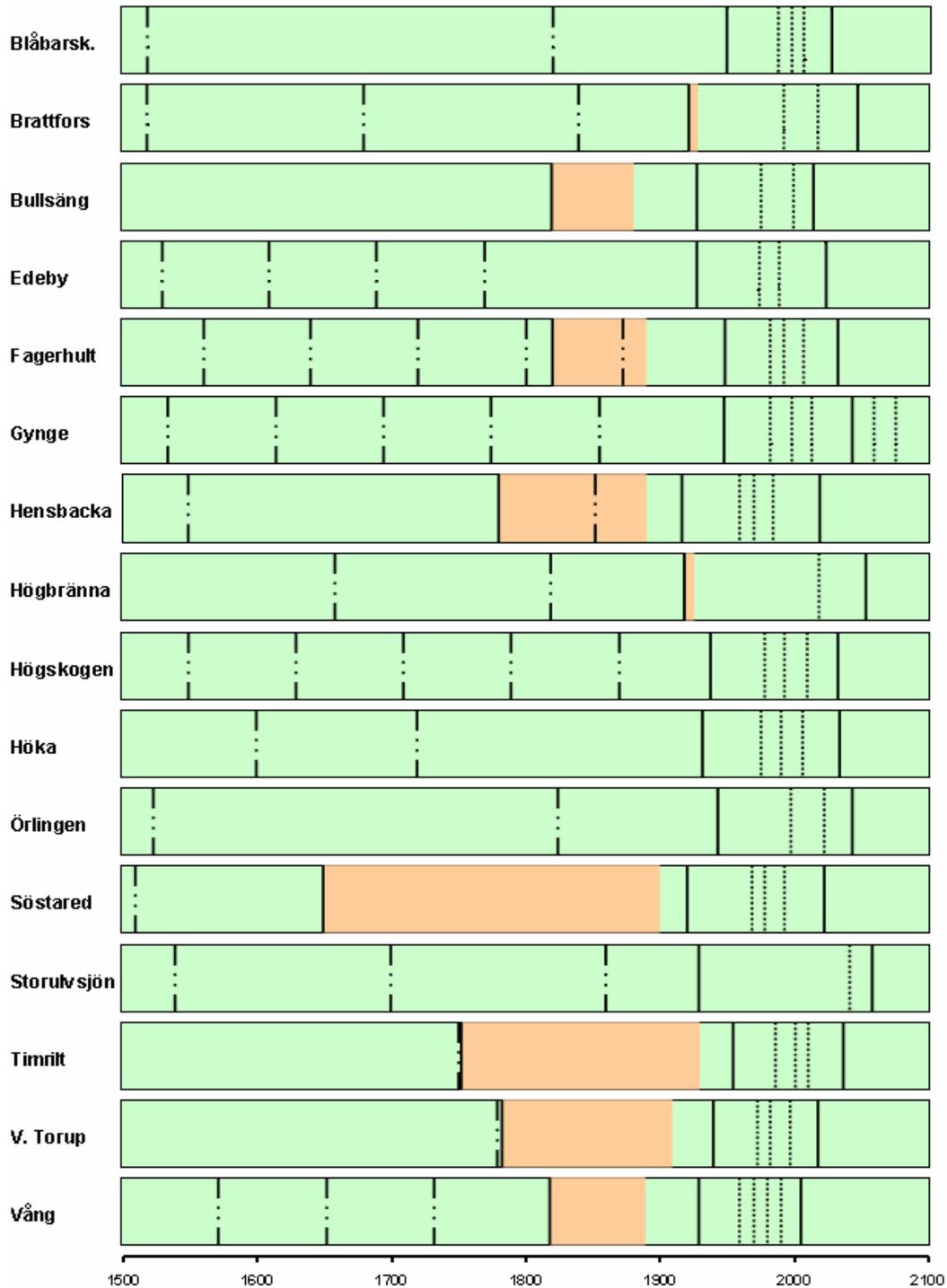


Figure A7.3 The land use histories of the studied sites. Green areas correspond to the presence of forest cover and the brown shade indicates open lands. The solid vertical lines show clear-cuts, the dotted lines show thinnings and the discontinuous lines indicate fire events.

The simulation spanned between the year 1500 and 2100. In the case of lack of data, the assumption was made that the forest tree composition does not change, i.e. if a

forest is made of 100% spruce today, it would have been made of 100% spruce anytime there would have been forest cover at the site. Of the outputs of the simulation, only parameters with a direct influence on the composition of the ground vegetation will be presented below.

Results and validation

Time series for standing tree biomass were obtained, and compared to reported values from the years 1995 and 1996 (Figure A7.4). The relevance of the tree biomass growth in affecting the composition of the ground vegetation lies primarily in the interception of light and the alteration of the soil hydrology and chemistry through uptake and litterfall.

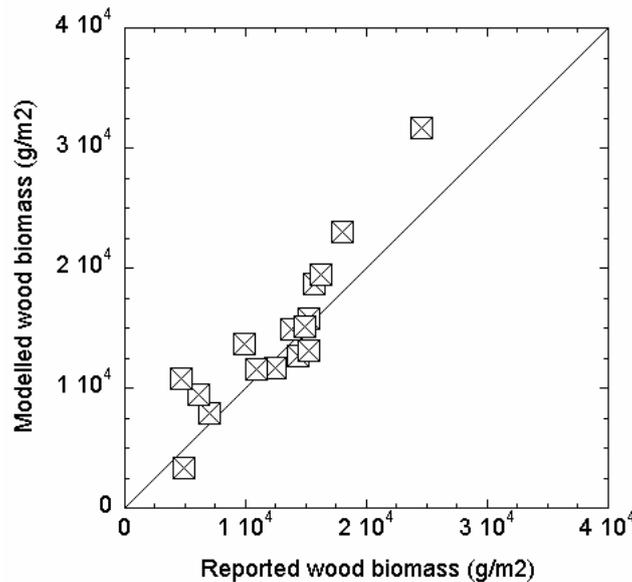


Figure A7.4 Modelled and reported standing wood biomass.

Looking at the simulation of soil organic matter, Figure A7.5. shows a good correlation between the measured and modelled values of soil organic carbon (C) and N. Soil organic matter being the nutrients pathway from the vegetation to the soil, it is vital in estimating the changes in the soil chemistry. Soil solution pH is used as an indicator of the validity of the model output (Figure A7.6). The model reconstructs the pH well, but underestimates the acidity at the deeper soil layers. This inconsistency is probably due to the fact that the model considers only a limited amount of roots at the deep layers, thus underestimating uptake and the presence of organic matter and its decomposition.

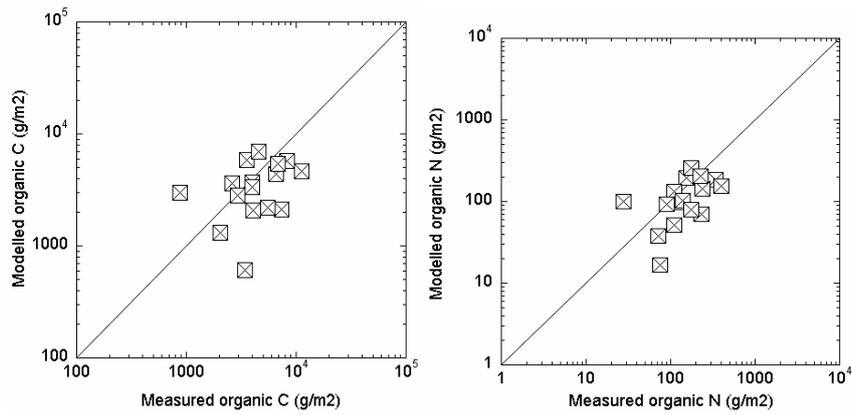


Figure A7.5 Modelled versus measured soil organic C and N at the 16 sites

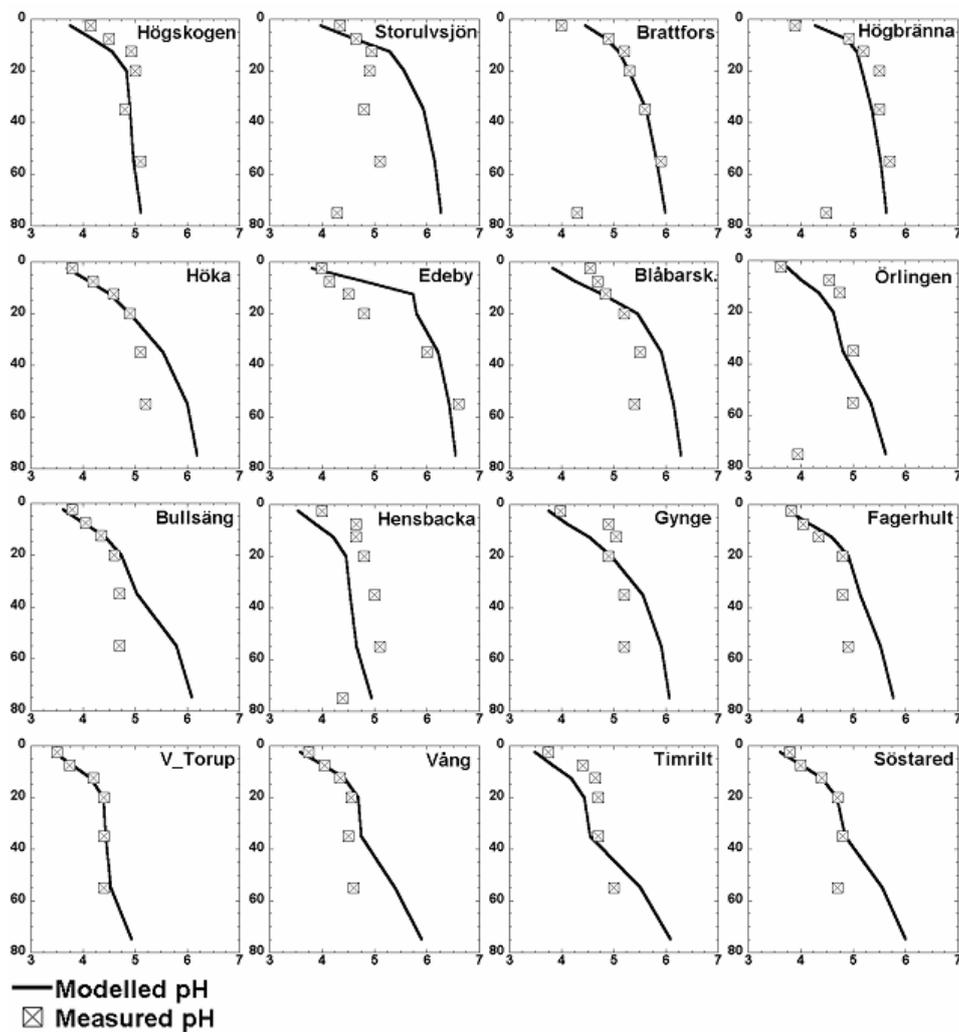


Figure A7.6 Modelled and measured pH values through the soil profile at the 16 study sites show a good correlation for the shallow layers. pH is overestimated by the model in the deep soil.

Also important for the ground vegetation community is the soil base cation to aluminium ratio (BC/Al ratio). Figure A7.7 shows the modelled BC/Al ratios for each of the 16 sites plotted against the measured values. The variation in both the measured and modelled BC/Al ratios is large for most of the sites, but the correlation between the model and the measurements is reasonably good.

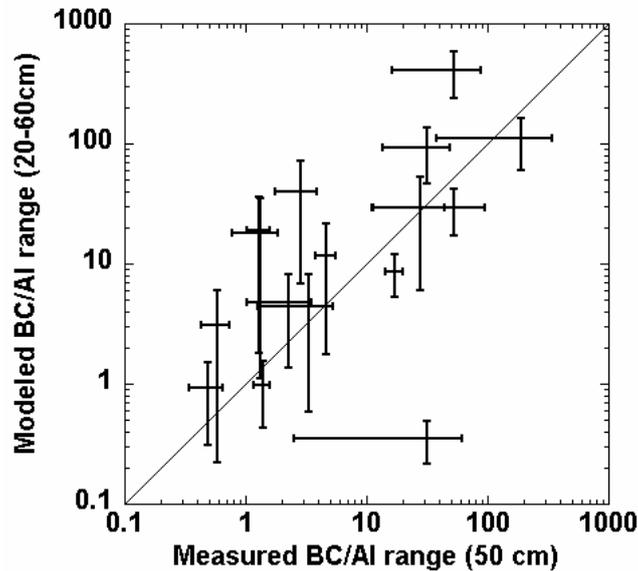


Figure A7.7 The measured and modelled values of BC/Al ratio at a depth of 50 cm. the modelled BC/Al ratios show the values for the soil layer defined between 20 cm and 40 cm and the 50 cm depth lies within this layer.

Reconstructing the ground vegetation composition

The ground occupancy of the 42 plant groups defined in Table A7.1 was calculated in ForSAFE-VEG. The modelled occupancy values for the year 1995 were plotted against measurements from the same year to establish the validity of the model outputs (Figure A7.8). The model predicts fairly well the occupancy of the present vegetation groups. However, the model predicts the presence of some groups where measurements report that these groups do not exist. The reason behind this might be the assumption in the model that all the plant groups are potentially establishable in all the studied ecosystems, and would manifest if the conditions are ripe.

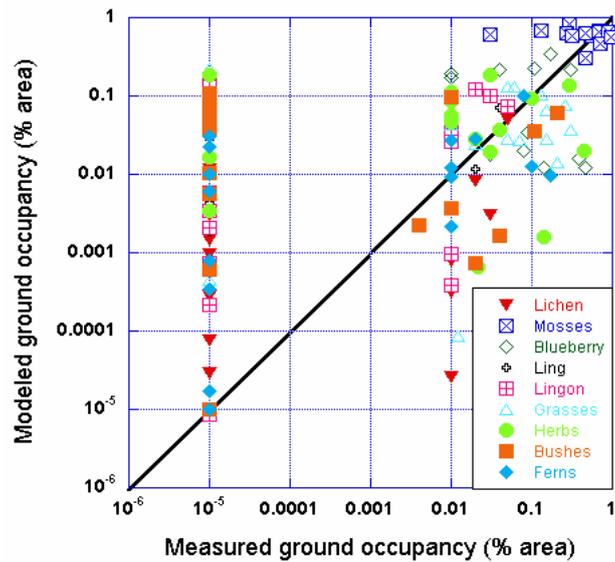


Figure A7.8 Modelled and measured ground vegetation occupancy of different plant groups. The vertical grouping of some groups at a measured occurrence value of 1/10000 represents the absence of these groups in the measurements.

Time dependant changes in the ground vegetation composition following changes in the environmental conditions and land use are shown in Figure A7.9. Clear differences appear between the sites and within each site as the conditions change. The southern sites have a wider diversity of species, while the northern sites are dominated by lichens, mosses, and berries. Open fields are rather quickly colonised by grasses, which can be mainly noticed in the southern sites. The effect of the forest fire cycles enhance the berries as they allow for more light to reach the forest floor as well as freeing more nutrients. On top of these changes, a trend can be observed by which mosses become more dominant primarily as a result of the increased N deposition. This chronic effect is not likely to be reversed unless the deposition of N is reduced. Based on this phenomenon, it is possible to calculate critical loads of N which allow for a minimal disturbance in the historic composition of the ground vegetation.

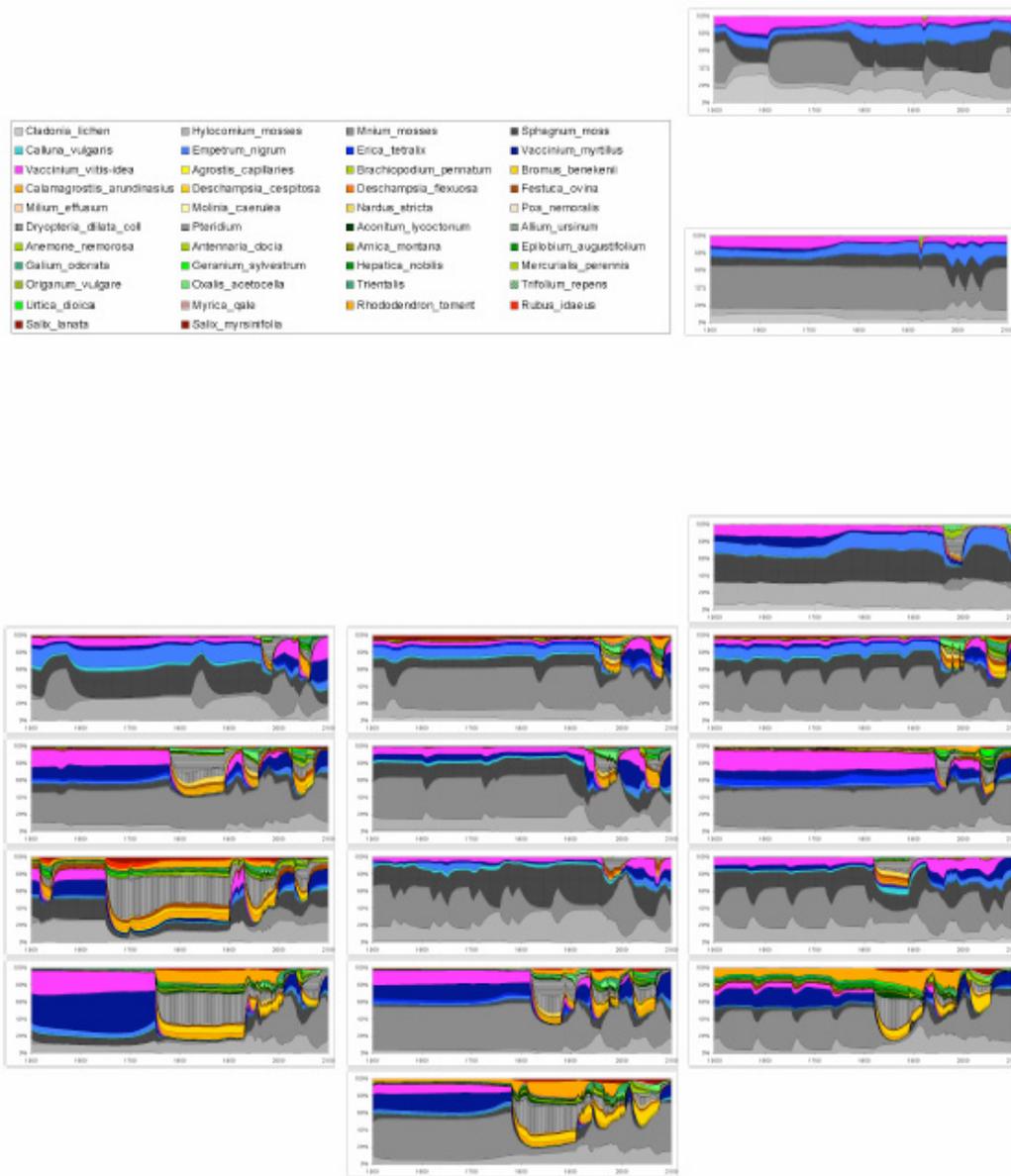


Figure A7.9 The time diagrams of the ground vegetation composition at the 16 Swedish sites. The sites are distributed geographically, with the northern most site in the top of the figure, and the southern most site in the bottom. A key of the plant groups is presented in the figure.

Annex 8 Parameterization of response functions for plant groups in the ForSAFE-VEG model

This Annex provides an overview of the derivation of individual response functions in the ForSafe-Veg model. The principles of ForSafe-Veg and the use of functions and limits in deriving critical loads with the ForSafe-Veg model is described in Section 3.5. Possibilities for deriving operational critical levels for use in SMB on the basis of such critical loads is described in section 4.2.

In ForSAFE, nitrogen is one of several factors that determine the competition success of ground vegetation. Emphasis here is on N, but it should be kept in mind that N does not act alone, and the separation of the causes in the end output from the ecosystem, is not a trivial task. Therefore a very short outline of the parameterization for a few other parameters has been given as well.

Derivation of individual response functions for nitrogen

The derivation of individual response functions for nitrogen is divided in:

- (i) assessment of functions for so-called anchor point plant species
- (ii) derivation of assumed response functions for other plant species.

Plant species were classified in 6 classes on the basis of the following reactions to increased N concentrations by plants:

1. The curves for plant species are only upward (basis is no negative effects of N have been found, quantitatively or qualitatively. Example is grass-type plants in agriculture and animal pastry), divided in weak and strong dependence reactions if possible.
2. The curves for plant species are either only promoting with respect to nitrogen or both promoting and retarding, divided in weak and strong dependence response, according to:
 - i. strong upward and strong downward,
 - ii. strong upward and weak downward,
 - iii. weak upward and strong downward
 - iv. weak upward and weak downward. In practice, the last combination never occurs.

Weak implies that a large change in driver is needed to give a standard response. Strong response implies that there will be a large response to a small change in the driver.

The curve shape has been made comparable for all plant species classified in one of the 6 classes according to:

- (i) 2 log units input width in the response curve form 1-99%: strong effect
- (ii) 3 log units input width in the response curve form 1-99%: weak effect

For the retarding response curves no experimental shape is available outside of blueberries. Based on observation of physiological response curves to pH and water, a strong and weak symmetrical to the up-curve was adopted. The retarding effect is thought to originate from one or several of the following reasons:

- (i) High plant content of nitrogen makes the plant desirable as substrate for parasites and phytopathogenes such as:
 - a. Fungi
 - b. Bacteria
- (ii) High plant content of nitrogen make plants more palatable to
 - a. Browsing ungulates and grazing animals
 - b. Herbivorous insects
- (iii) High soil solution concentration of ammonium is toxic to plants
- (iv) Interaction between NH_4 and K may disturb K uptake

The strategy for response function parameterization is summarized in Figure A8.1 and was as follows:

- (i) Establishment of basic curve shapes for response curves from a relatively small set of data from agriculture and forestry.
- (ii) Fit of response curves for plant groups with minimum only one single data point, assuming the curve shape to be known. The curve shape is drawn up, and the coefficients adjusted until the best fit to the available points has been found
- (iii) Scaling of plant groups in between known responses, using generic knowledge and expert opinions from Swedish plant ecologists. The scaling for Sweden was done in three consecutive group modelling sessions, each three weeks apart. At each session the all curves were assessed and the full table was obtained at the third session. Participants at all sessions were Prof. Bengt Nihlgård, Prof. Lars Ericson and Prof. Harald Sverdrup, at single sessions Dr. Ingrid Stjernquist, Dr. Mats Svensson, Prof. Bjarni Sigurdsson, Dr. Salim Belyazid and Msc. Asrun Elmarsdottir participated.
- (iv) Field tests at 16 sites in Sweden and final tuning of parameters based on assessment of field test misfits was done by Dr. S. Belyazid, Prof. Bengt Nihlgård and Prof. Harald Sverdrup.

The actual values for critical N concentrations in soil solution are based on agricultural experiments agricultural crops such as wheat, barley, oats, sugar beet, rye, peas and clover, but also some tree species like Norway spruce, Scots Pine, Birch and Beech. This refers to plants in chambers in which N is sprayed on the crop using nozzles, while controlling the N, P and K concentration in the spray solution (Experiments published by the group of the late Prof. Torsten Ingestad. Some of the data is not available in published form). Furthermore, Nordin et al. (2005) and Strengbom et al. (2002) have field data on N concentrations in suction cups and lysimeters at experimental sites. From those experiments the shape of the curves is extracted (weak or strong response; 2 or 3 log units).

$$f(\text{N}) = f(\text{promote}) * f(\text{retard})$$

The adopted equation is for the promotive effect:

$$f(\text{promote}) = a_0 * [\text{N}]^w / (k(\text{promote}) + [\text{N}]^w)$$

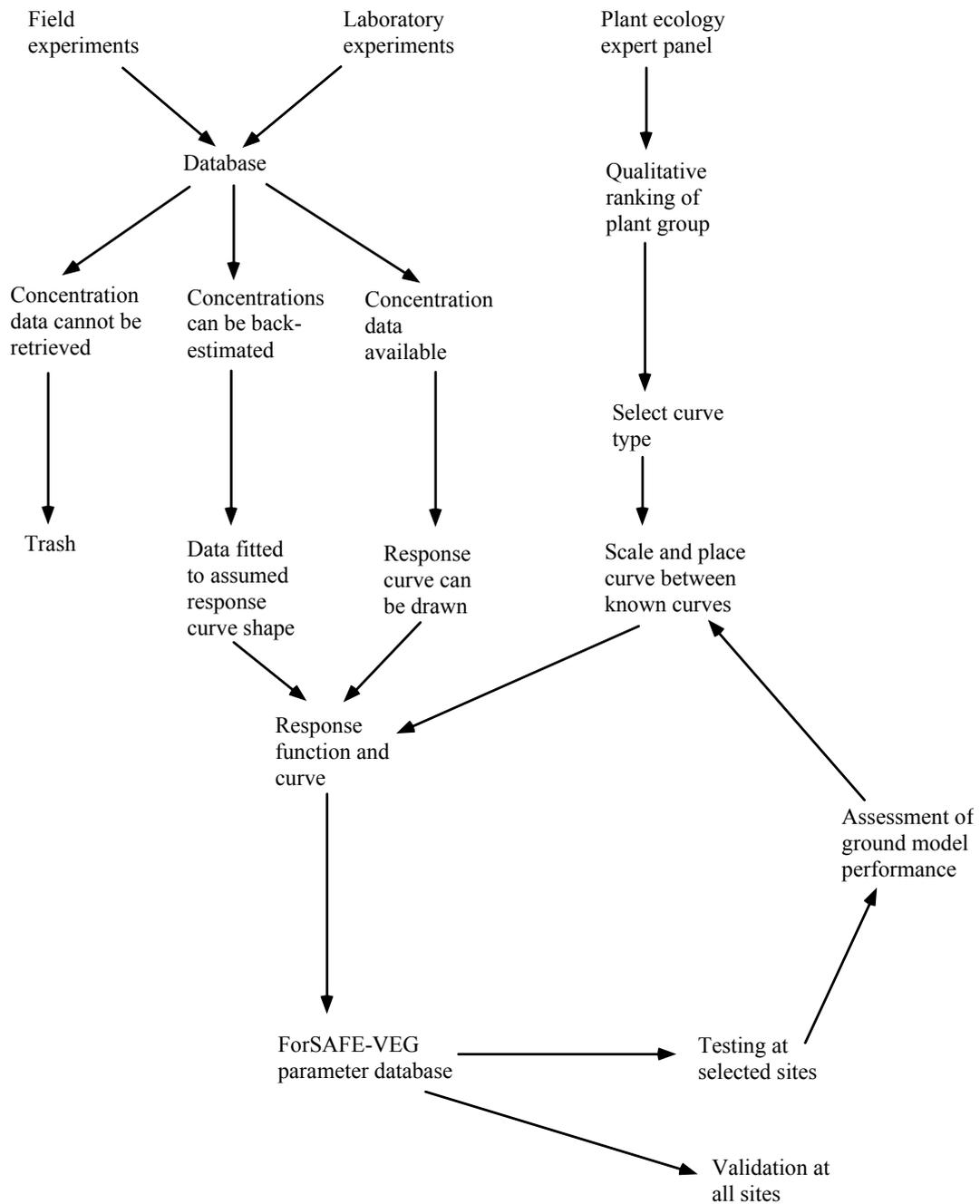


Figure A8.1 Flow chart for the parameterization process for the vegetation sub model parameter data base of FORSAFE-VEG.

The retarding effect is caused by high nitrogen content makes the plant desirable for predators and pathogens. The adopted equation was:

$$f(\text{retard}) = k(\text{retard}) / (k(\text{retard}) + [\text{N}]^V)$$

The per layer root weight weighted average of the response value is applied to the whole plant.. A lot of data available for crop plants like wheat, barley, oats, sugar beet, sorghum, rye, peas and clover was used. Some data are available for forest ecosystem plants, such as Swedish blueberries (*Vaccinium myrtillus*) and lingon berries (*Vaccinium vitis idea*) (Ellenberg et al., 1992; Strengbom et al., 2002; Nordin et al., 2005). Further some information on agrostis, gallium, trifolium, festuca, and poa are available in institute reports from the Ingestad research group at SLU, Alnarp (Ingestad & Lund, 1986; Falkengren-Grerup, 1992; Ingestad et al., 1994a, b; Hedlund & Hellgren, 1996; Ingestad et al., 1996; Zhu et al., 1999; Hellgren, 2003). Examples of results thus obtained are given in Figure A8.2.

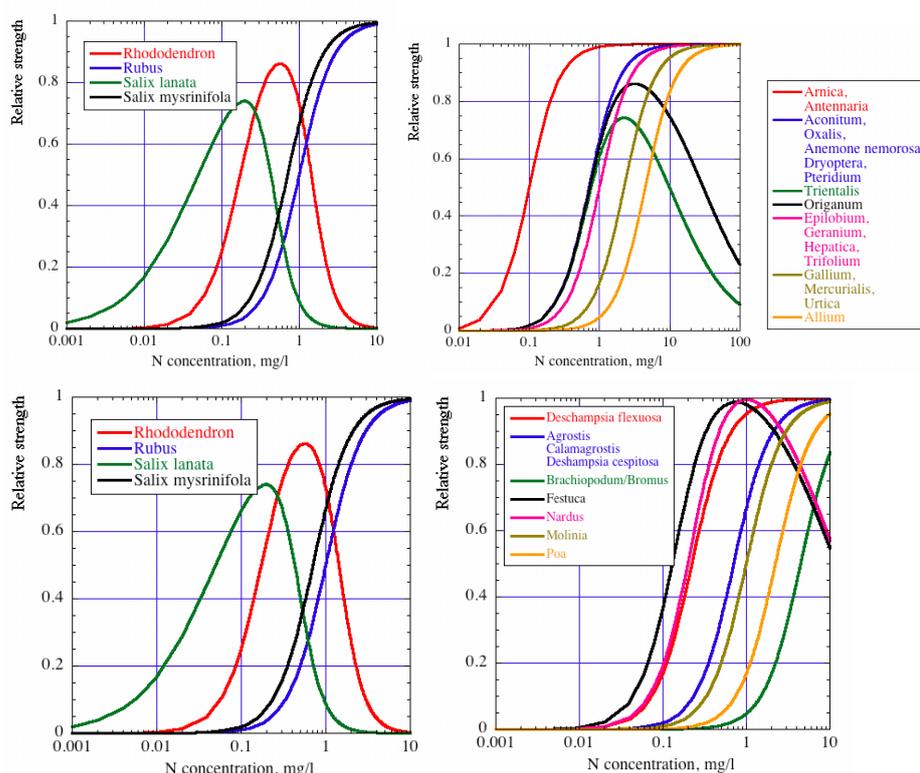


Figure A8.2 Diagrams showing the final response curves after fitting to response data and scaling. Retardation is included when quantitative data on retardation is available (Lichens, blueberries, lingonberries) or qualitative data (several of heather type, grasses and herbs.)

Derivation of individual response functions for all other influencing factors

As stated before, in ForSAFE-VEG, nitrogen is one of several factors that determine the competition success of ground vegetation. Below a very short outline of the parameterization for the other parameters has been given as well, starting with a summary for nitrogen, as presented above.

- **Nitrogen:** *Vaccinium myrtillus*, *Vaccinium vitis idea*, *Caluna*, *Poa*, *Festuca* and several crop plants such as pea, lupin, clover were used as anchor plants. Additional semi-quantitative data are available for some mosses from private archives of Prof. Emeritus Bengt Nihlgård and could be gleaned from

agricultural literature (Fridriksson & Sigurdsson, 1983; Ingestad et al., 1994a, b). Continually different species for the various anchor points are being added.

- **Water:** Hard scientific data is available for Norway spruce, Scots pine, White Birch, Potato, Poa grass and clover, which were used as anchor plants. As an example of a curve, Fig. A8.3 is presented. About 35 anchor plants were taken from the Nordic flora and gardening manuals (Ingestad et al., 1994a, b; Hedlund & Hellgren, 1996; Ingestad et al., 1996; Zhu et al., 1999).
- **Temperature:** Available for wheat and about 35 anchor plants taken from the Nordic flora and gardening manuals (Mossberg & Stenberg, 1997; Schul, 2000).
- **Acidity:** Complete data curves for several plants were available: Norway spruce, Scots pine, birch, oak, *calluna*, *vaccinium myrtillus*, *erica*, *agrostis*, *deschampsia*, *festuca*, *bromus*, *poa*, *nardus*, *carex*, *alopecurus*, *brachyopodium*, *lollium*, *origanum*, *allium*, *oxalis*, *gallium*, *trifolium*, *geum*, *stellaria*, and more adding up to about 35 of the plants on the list (Sverdrup & Warfvinge, 1993). Single data points are available for several others, including brachens and lichens.
- **Phosphorus:** A few data points are available for wheat, barley, potato, tomato and poa grasses.
- **Grazing plant preference:** Available for birch, Spruce, Pine and *agrostis* with respect to ungulents like moose (*alces alces*), and good expert knowledge is available with foresters, derived from field experience with forestry in areas with high moose populations.
- **Wind:** Qualitative field observation from Iceland and coastal Norway (Guyot, 1998). No published data available at all. The authors experience with the botany of the Norwegian high mountains, having spent 45 summers there.
- **Competition:** Is based on relative ranking, after group sessions with plant ecologists from Lund University, Umeå University, Institute für Angewandte Pflanzenökologie, Basel, and a formulation of the principles. Calibrated on very general patterns in coastal area habitats.
- **Delay time:** The delay time was estimated from field observations (No values have been really published explicitly, only indirectly to be seen in some sets of data). For plants where no data is available, estimated average life expectancy, alternatively 1/3 of maximum age observed. Data are available for Norway spruce (Generic forestry literature, data from Skogforsk Norway anecdotes by Per Nygaard), *caluna*, blueberry, *Erica*, semiempirical data by the author on *deschampsia* and *agrostis* and *cladonia* from the Frydalen Valley and Rondane National Park, Norway.

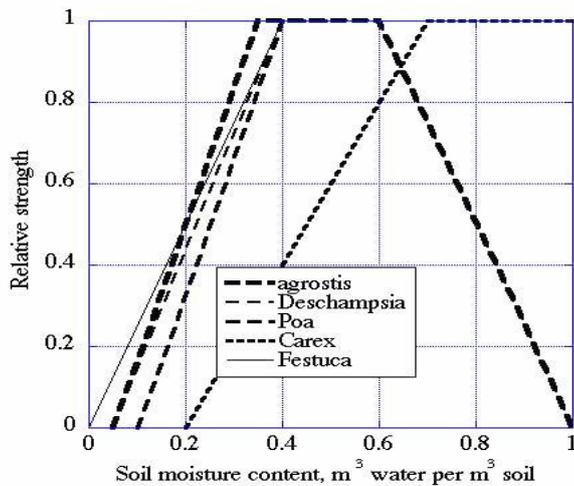


Fig A8.3. Example of water response curves for some grass species. The curve was based on setting the lowest moisture point, the highest and the onset of retardation. For grasses this is approximately known from watering recommendations

Grazing pressure from wild animals was calculated using the HELGE grazing model. The model is based on Forest stand populations simulations (juveniles, young trees, mature and old), total ground vegetation biomass (kg/ha) and Moose population (calf, cow, bull, old) dynamics. Hunting is included in the model. The model was built in the STELLA programming environment, and is available from the author.

For each of the identified plant groups, parameters were thus defined for all the specific response functions described above. Table A7.1 in Annex 7 summarises the parameterisation thus derived for Sweden.

Annex 9 Derivation of critical nitrogen concentration limits used in the present mapping manual

General approach

Critical nitrogen concentration limits presented in Warfvinge et al. (1992) and used (in adapted form) in the present mapping manual are said to be “based on preliminary experiences from the Swedish Forest Survey program”. Actually, the approach to derive these values was an inverse use of the critical mass balance approach, according to:

$$[N]_{(\text{crit})} = (\text{CL}(\text{N}) - N_{\text{up}} + N_{\text{im}} + N_{\text{de}}) / Q \quad (\text{A9.1})$$

where:

$[N]_{(\text{crit})}$	=	Critical N concentration ($\text{kg} \cdot \text{m}^{-3}$)
$\text{CL}(\text{N})$	=	Critical N load ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
N_{up}	=	Average net yearly uptake during a forest rotation ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
N_{im}	=	Long-term average net yearly immobilization ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
N_{de}	=	Denitrification flux at critical N load ($\text{kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)
Q	=	water flux ($\text{m}^3 \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$)

The values for CLN are based on a visual interpretation of maps of vegetation changes in Sweden in the period 1977-1987, using two digitized forest inventories. When vegetation changes were large, a CLN was set, using estimated N deposition values for the year 1987 (basis is the Swedish 1987 N deposition database). The implicit assumption is that (large) changes occurring in the period 1977-1987 is due to N deposition alone, whereas such changes may be corroborated with changes in acidity, climate, forest management change, wildlife grazing etc. Consequently, the approach is a purely empirical one and a more elaborated approach accounting for these impacts (see e.g. Annex 8) is favoured whenever reliable data are available.

Assessment of the empirical critical nitrogen loads

To derive values for CLN based on vegetation changes in Sweden in the period 1977-1987, a distinction was made in 6 classes of decreasing sensitivity to N deposition, as given in Table A9.1. Ultimately, a comparison was made of 5 maps in 1977 and 1987 with five vegetation types, namely Lichens, Cranberries, Blueberries (Division in class 2), Grasses (lumping of class 3 and 4) and Herbs (lumping of class 5 and 6). Critical N loads were derived on the basis of the N loads in areas with significant changes in the occupation of species, in terms of the occurrence in percentiles in grid squares of 50 km x 50 km (Swedish forest inventory grid squares) in the various maps. An example of one of these 5 maps (for lichens) is given in Figure A9.1. In this way, a critical N load was derived for the change from (i) lichens to cranberries (lingonberries), (ii) cranberries to blueberries, (iii) blueberries to grasses and (iv) grasses to herbs.

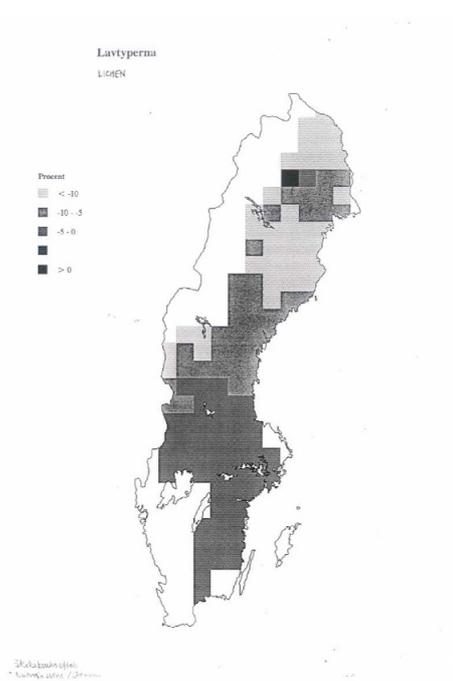


Figure A 9.1. Change in lichens cover between 1985 and 1997. Similar maps exist for *Vaccinium myrtillus*, *Vaccinium vitis-idaea*, thin leaved grasses, broadleaved grasses, Poor herbs, Rich herbs.

Table A9.1. Relative list of sensitivity for Swedish ground vegetation classifications, based on qualitative or semi-quantitative data as those exemplified in Figure A9.1 for field observations of lichens. Additional data were available from lichens species that exclusively grow on trees.

Sensitivity class	Forest vegetation type
1 Very high	Lichen moss type
2 High	Heather- lingeon type
3 Moderately high	Thin leaved grass types
4 Moderate	Broad leaved grass types
5 Low	Poor herbal types
6 Very low	Rich herbal types

Assessment of the various N flux terms and runoff

The various N flux terms and runoff were generated with data in the region of change where CL(N) was derived, using points from the Swedish critical loads database available by then (1987), having 1883 sites in terrestrial ecosystems in Sweden. The terms are therefore generalized to the same level as the generalized observation of a change on the regional level. An overview of the derived N flux terms and runoff in Eq. (A9.1) is presented below.

N uptake (N removal due to harvesting)

N removal due to harvesting is based on long term yields during a rotation period, available in forest inventories, and average N contents in stemwood

N-immobilization

N-immobilization is an on site constant, based on de Vries approach calibrated on data from Sverdrup-Ineson and Rosen

Denitrification

Denitrification is calculated according to the Sverdrup Ineson approach. Denitrification has been shown to follow Michaelis-Menten kinetics in laboratory experiments, where the rate depend upon the aqueous solution concentration of nitrate according to (Sverdrup and Ineson, unpublished):

$$N_{de} = (\max r_{min}, (k_0 \cdot (N_{dep} - N_u - N_{im}) / (K + (N_{dep} - N_u - N_{im}))) \cdot \text{sum}f_i \quad (\text{A9.1})$$

where N_{dep} is total nitrogen deposition, N_u is tree nitrogen uptake. $r_{min}=0.7 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ is a minimum denitrification rate under optimal conditions ($0.05 \text{ keq.ha}^{-1}.\text{yr}^{-1}$), the first order rate coefficient value is $k_0=25 \text{ kg N ha}^{-1}.\text{yr}^{-1}$ ($1.8 \text{ keq.ha}^{-1}.\text{yr}^{-1}$) and the Michaelis-Menten half rate saturation coefficient $K=40 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ ($2.86 \text{ keq.ha}^{-1}.\text{yr}^{-1}$). On the average immobilization in the region covered by our database is $N(im)=8 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$. This explains why relatively few sites at present leach nitrogen. k_0 is the rate coefficient k divided by the functional value of pH 5 in the non-normalised pH modifier and the functional value of the moisture modifier at $w=0.2$. Other kinetic expressions are also possible within the data set, but the accuracy of the compiled data was not considered to support differentiation arising from such small differences.

Runoff

Runoff is based on data from the Swedish Meteorological Institute.

Results and evaluation

Using this approach, broad critical N concentration limits were derived from the observed field data. The following changes are induced at the following approximate average soil solution concentrations in the rooting zone:

- lichens to cranberries : $0.2\text{-}0.4 \text{ mg N.l}^{-1}$
- cranberries to blueberries: $0.4\text{-}0.6 \text{ mg N.l}^{-1}$
- blueberries to grasses: $1\text{-}2 \text{ mg N.l}^{-1}$
- grasses to herbs: $3\text{-}5 \text{ mg N.l}^{-1}$

The borders of the areas with change was used to set a nitrogen deposition capable of making response. A check on the critical limit values was made by deriving critical N loads and their exceedances and comparing the exceedance values with the areas where relative large changes in plant species composition are taking place. There are several sources of error, as contributing to uncertainty and probably overestimation of the change-inducing concentration:

- (i) We ignored layering in the rooting zone in the back calculation, one mixed layer was assumed
- (ii) Uptake directly on leaves was ignored
- (iii) Exclusion of other factors is far from certain
 - a. Grazing and browsing by animals at the site
 - b. Climate effects at the site

- c. Forestry effects at the site
- d. Effects of acidity in the soil at the same time

It has to be stressed that the critical N concentrations thus derived are limited to the Nordic countries and based on a highly aggregated approach in deriving empirical critical N loads. Furthermore, there are many simplifying assumptions in deriving the critical N concentrations, but in broad lines, these values will lead to critical N loads that are applicable for the Nordic countries (at least comparable to the derived empirical critical N loads).