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Expert Workshop

Berne, 11–13 November 2002

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Swiss Agency for the Environment,
Forests and Landscape SAEFL

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Organization of the Workshop

The workshop was organized by the Swiss Agency for the Environment, Forests and Landscape (SAEFL).

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Preface

The impact of airborne nitrogen on sensitive ecosystems has already been addressed in a series of previous workshops held under the Convention on Long-range Transboundary Air Pollution covering the region of the United Nations Economic Commission for Europe (UNECE). Critical loads for nitrogen, derived from the scientific knowledge available at that time, were part of the effect-orientated approach chosen for the development of the Protocol on the Abatement of Acidification, Eutrophication and Ground-level Ozone, signed at Gothenburg (Sweden) in 1999. With this protocol, nitrogen measures were strengthened compared with earlier protocols and a multi-pollutant/multi-effect approach was adopted addressing simultaneously several pollutants and environmental effects in an integrated way. This was a substantial progress.

But we all knew that the Gothenburg Protocol was an interim step towards a sustainable air pollution control policy. There is significant potential and interest to further improve our approaches and to prepare a sound scientific and technical basis for the coming revision of the Gothenburg agreement.

This workshop is part of the preparatory phase of the planned revision of the Gothenburg protocol. The effect-oriented analysis of the Protocol has clearly shown that more efforts are needed to address - inter alia - the eutrophication problem. The scientific literature as well as international workshops and conferences on nitrogen have always highlighted that nitrogen is a very complex topic and that a single reactive nitrogen molecule can cascade through a wide variety of environmental systems and contribute to multiple sequential effects. Scientific experimental and field work can contribute to a better understanding of these processes and to improving our ability to assess how much a sensitive ecosystem can tolerate in the long-term. In this sense the empirical work, and the process understanding derived from it, are essential prerequisites for the development of process-oriented and dynamic ecosystem modelling.

The proceedings reflect the comprehensive assessment and update of the scientific knowledge on empirical critical loads for nitrogen prepared before the workshop and the results of the discussions carried out during the meeting. We are aware that there will always be gaps in knowledge hindering us to be final in our assessment. But since the work under the Convention is an iterative process, this should not hinder us from setting from time to time certain milestones. Such milestones are important for the political discussion on the need to further control emissions of air pollutants from all sources contributing to impacts on the environment over large areas. In addition, they can serve as a stimulus to further research.

I would like to express my sincere thanks to all the scientists involved for their efforts made before, during and after this workshop. I am especially grateful to the authors of the background documents, to the scientific reviewers and to the chairpersons and rapporteurs. I would also like to express my gratitude to the representatives of the Secretariat to the Convention for contributing over many years to the successful implementation of the effect-orientated activities within the Convention.

Gerhard Leutert

Swiss Agency for the Environment, Forests and Landscape
Head of the Air Pollution Control Division

Workshop Results

Workshop Summary

Workshop Summary

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I. Introduction

1. The Expert Workshop on Empirical Critical Loads for Nitrogen Deposition on (Semi-)natural Ecosystems took place in Berne, Switzerland, from 11-13 November 2002. The workshop was organized by the Swiss Agency for the Environment, Forests and Landscape (SAEFL).

2. The workshop was attended by 53 experts from the following Parties to the Convention: Austria, Denmark, Estonia, France, Germany, Netherlands, Norway, Sweden Switzerland and the United Kingdom. The International Cooperative Programme (ICP) Forests, ICP Waters, ICP Integrated Monitoring, ICP Mapping and Modelling, the Coordination Center for Effects (CCE) and the Secretariat to the Convention were represented.

3. Empirical critical loads for nitrogen had already been set at previous workshops under the Convention (Skokloster, 1988, Lökeberg, 1992, Geneva, 1995). A workshop on chemical criteria and critical limits held in March 2001 in York (EB.AIR/WG.1/2001/13) came to the conclusion that, on the basis of the availability of new scientific evidence for many nitrogen sensitive ecosystems, an update of empirical nutrient nitrogen critical loads was needed and should be evaluated and discussed at an expert workshop. In addition, more guidance should be given regarding the classification and mapping of nitrogen sensitive ecosystems and their corresponding empirical critical loads of nitrogen. The Berne workshop aimed at implementing these proposals.

II. Aims and Organization of the Workshop

4. The aims of the workshop were to:

(a) Evaluate, update or revise the empirical critical loads for nitrogen for (semi-)natural ecosystems set in the 1996 Manual on Mapping Critical Levels/Loads on the basis of the additional scientific information available for the period 1996-2002;
(b) Address the following broad classes of nitrogen sensitive ecosystems:
  - forest habitats;
  - heathlands, scrub and tundra habitats;
  - grasslands and tall forb habitats;
  - mires, bogs, fens and inland surface water habitats;
  - coastal and marine habitats;

(c) Evaluate appropriate criteria and ecological indicators for nitrogen effects to strengthen the scientific background for establishing cause-effect relationships and critical loads and to improve the assessment of ecosystem effects due to critical loads exceedances;
(d) Give guidance regarding the use of a harmonized classification of nitrogen sensitive ecosystems as a prerequisite for their uniform mapping in the UNECE region.

5. An international team of scientists (R. Bobbink, M. Ashmore, S. Braun, W. Flückiger, J. Hall, I. Van den Wyngaert) prepared detailed background information on the update of empirical critical loads for nitrogen and on the harmonized EUNIS (European Nature Information System) ecosystem classification. This information was reviewed by another team of scientists (U. Falkengren-Grerup, M. Hornung, J.G.M. Roelofs, M. Strandberg, S. Woodin) and made available to all participants one month before the workshop.

6. The workshop was opened by Mr. G. Leutert (Swiss Agency for the Environment, Forests and Landscape) who welcomed the participants on behalf of the host country and the organizers. Mr. R. Chrast from the secretariat to the Convention informed the participants of relevant ongoing activities under the Convention.

7. The contents of the background documents on empirical critical loads for nitrogen and on the EUNIS ecosystem classification were presented by the authors in an extended plenary session, chaired by Mr. J.-P. Hettelingh. In addition, information on nitrogen specific results from ICP Forests and ICP Integrated Monitoring and on the possibilities of multi-species models for the calculation of nitrogen critical loads was given during the plenary.

8. Detailed discussions of the background documents and the proposed empirical critical loads for nitrogen were carried out in three working groups:
(a) Working group on forest habitats (Chairpersons: U. Falkengren-Grerup/M. Hornung; Rapporteur: J. Strengbom);
(b) Working group on grasslands, fens and mires, inland surface waters, coastal and marine habitats (Chairman: A. Davison; Rapporteur: M. Strandberg);
(c) Working group on heathlands, scrub and tundra habitats, bogs (Chairman: J.G.M. Roelofs; Rapporteur: S. Woodin).

9. The outcome of the working group discussions was considered in several short plenary sessions. The results, conclusions and recommendations were discussed and summed up in a final plenary session, chaired by Mr. R. Bobbink.

III. Results, Conclusions and Recommendations

10. Based on observed changes in the structure and function of ecosystems, reported in European publications, empirical nitrogen critical loads were evaluated for specific receptor groups of natural and semi-natural ecosystems. Results from field addition experiments and mesocosm studies, from correlative or retrospective field studies, and, in a few cases, dynamic ecosystem modelling, were relevant in this respect.

11. Empirical nitrogen critical loads were agreed for a range of deposition values for each ecosystem class, because of: (i) real intra-ecosystem variation between different regions where an ecosystem has been investigated; (ii) the finite intervals between additions of nitrogen in experiments; and (iii) uncertainties in estimated total atmospheric deposition values, although these have been checked by local specialists on atmospheric nitrogen deposition. For every group of ecosystems, the empirical nitrogen critical loads were set with an indication of their reliability and of the effects to be expected in the case of exceedances.

12. The reliability of the nitrogen critical loads figures presented was indicated as follows:
- reliable ##: when a number of published papers of various studies showed comparable results;
- quite reliable #: when the results of some studies were comparable;
- expert judgement (#): when no empirical data were available for this type of ecosystem. For this, the nitrogen critical load was based upon expert judgement and knowledge of ecosystems which were likely to be comparable with this ecosystem.

13. To facilitate and harmonize the mapping procedure, the receptor groups of natural and semi-natural ecosystems were classified and ordered according to the EUNIS habitat classification for Europe (http://mrw.wallonie.be/dgrne/sibw/EUNIS). In general, the ecosystems were classified down to level 2 or 3 of the EUNIS hierarchy. The following habitats groups (with EUNIS level 1 code between brackets) were treated:
- Woodland and forests habitats (G)
- Heathland, scrub and tundra habitats (F)
- Grassland and tall forb habitats (E)
- Mire, bog and fen habitats (D)
- Inland surface water habitats (C)
- Coastal habitats (B)
- Marine habitats (A)

A limitation in using the many subcategories of the EUNIS classification was a lack of research and data on nitrogen impacts for those habitats. For forest ecosystems, it was at this moment only possible to set values for three broad EUNIS classes (G1, G3 & G4) with, however, some separation for grouping of forest types, such as coniferous from deciduous and boreal from temperate.

14. The summarized updated empirical critical loads for nitrogen (table 1) were agreed by consensus at the workshop. To facilitate the shift between ecosystem classifications, table 2 shows a comparison of the ecosystem classification used in the 1996 Manual on Mapping Critical Levels/Loads with the proposed 2002 classification according to EUNIS.

15. Fine resolution maps of sensitive ecosystems of high conservation value are needed for each country to map nitrogen critical loads for these systems. It was advised to use both the mass balance and empirically derived nitrogen critical loads for forest ecosystems and for other ecosystems for which the data were available. If the two approaches yielded different critical load values, the one with the lowest value should be used until the reasons for the difference have been clarified.

16. Some additional information was given on how to interpret the proposed ranges of critical load values in specific situations for an ecosystem. In the case of insufficient national data for specific (semi-)natural ecosystems, it is suggested to use the lower, middle or upper part of the proposed ranges of the nitrogen critical loads according to the general relationships between abiotic factors and critical loads for nitrogen as given in table 3.

IV. Gaps in Knowledge

17. Although considerable progress has been made since 1996 in the understanding of nitrogen impacts on several habitat groups, the following gaps in knowledge were recognized as most important:

- Research/data collection was required to establish a critical load for the following ecosystems: steppe grasslands, all Mediterranean vegetation types, wet-swamp forests, many mires & fens, several coastal habitats and high altitude systems;
- More research was needed for all EUNIS habitats with critical loads based on expert judgement or few research;
- Impacts of nitrogen enrichment in (sensitive) freshwater and shallow marine ecosystems needed further research and were sometimes overlooked;
- Additional effort was needed to allocate observed nitrogen effects to the appropriate EUNIS forest subtypes (levels 2 & 3);
- The EUNIS classification needed clarification/adjustment with respect to some grassland groups, Nordic bogs and mires and surface water habitats;
- The possible effects of the different deposited nitrogen species (NO_y or NH_x) were insufficiently known to allow differentiation between these nitrogen species when setting critical loads;
- In order to refine current critical loads, long-term (> 3 – 5 years) nitrogen addition experiments with high resolution of treatments between 5 and 50 kg N ha^{-1} yr^{-1} in regions with low background depositions or in mesocosms were needed.

18. In conclusion, it was crucial to understand the long-term effects of increased nitrogen deposition on ecosystem processes in a representative range of ecosystems. It was thus very important to quantify the effects of nitrogen loads by manipulation of nitrogen inputs in long-term ecosystem studies in unaffected and affected areas. Such data were essential to validate the set critical loads and to develop robust dynamic ecosystem models and / or multiple correlative species models, which were reliable enough to calculate critical loads for nitrogen deposition in (semi-)natural ecosystems and to predict (natural) recovery rates for nitrogen-affected systems.

Table 1. Overview of empirical critical loads for nitrogen deposition (kg N ha^{-1} yr^{-1}) to natural and semi-natural ecosystems. Classification of habitats according to EUNIS (except for forests). **## reliable; # quite reliable and (#) expert judgement**.

Ecosystem type	EUNIS-code	kg N $ha^{-1}yr^{-1}$	Reliability	Indication of exceedance
Forest habitats (G)				
Soil processes				
Deciduous and coniferous	-	10-15	#	Increased N mineralization, nitrification
Coniferous forests	-	10-15	##	Increased nitrate leaching
Deciduous forests	-	10-15	(#)	Increased nitrate leaching
Trees				
Deciduous and coniferous	-	15-20	#	Changed N/macro nutrients ratios, decreased P, K, Mg and increased N concentrations in foliar tissue
Temperate forests	-	15-20	(#)	Increased susceptibility to pathogens and pests, change in fungistatic phenolics
Mycorrhiza				
Temperate and boreal forests	-	10-20	(#)	Reduced sporocarp production, changed/reduced below-ground species composition
Ground vegetation				
Temperate and boreal forests	-	10-15	#	Changed species composition, increase of nitrophilous species, increased susceptibility to parasites
Lichens and algae				
Temperate and boreal forests	-	10-15	(#)	Increase of algae, decrease of lichens
Overall				
Temperate forests	-	10-20	#	Changes in soil processes, ground vegetation, mycorrhiza, increased risk of nutrient imbalances and susceptibility to parasites
Boreal forests	-	10-20	#	Changes in soil processes, ground vegetation, mycorrhiza, increased risk of nutrient imbalances and susceptibility to parasites
Heathland, scrub and tundra habitats (F)				
Tundra	F1	5-10[a]	#	Changes in biomass, physiological effects, changes in species composition in moss layer, decrease in lichens
Arctic, alpine and subalpine scrub habitats	F2	5-15[a]	(#)	Decline in lichens, mosses and evergreen shrubs
Northern wet heath	F4.11			
• ‘U’ *Calluna*-dominated wet heath (upland moorland)	F4.11	10-20[a]	(#)	Decreased heather dominance, decline in lichens and mosses
• ‘L’ *Erica tetralix* dominated wet heath	F4.11	10-25[a,b]	(#)	Transition heather to grass
Dry heaths	F4.2	10-20[a,b]	##	Transition heather to grass, decline in lichens

Grasslands and tall forb habitats (E)				
Sub-atlantic semi-dry calcareous grassland	E1.26	15-25	##	Increase tall grasses, decline in diversity, increased mineralization, N leaching
Non-mediterranean dry acid and neutral closed grassland	E1.7	10-20	#	Increase in graminoids, decline typical species
Inland dune pioneer grasslands	E1.94	10-20	(#)	Decrease in lichens, increase biomass
Inland dune siliceous grasslands	E1.95	10-20	(#)	Decrease in lichens, increase biomass, increased succession
Low and medium altitude hay meadows	E2.2	20-30	(#)	Increase in tall grasses, decrease in diversity
Mountain hay meadows	E2.3	10-20	(#)	Increase in nitrophilous graminoids, changes in diversity
Moist and wet oligotrophic grasslands	E3.5			
• *Molinia caerulea* meadows	E3.51	15-25	(#)	Increase in tall graminoids, decreased diversity, decrease of bryophytes
• Heath (*Juncus*) meadows and humid (*Nardus stricta*) swards	E3.52	10-20	#	Increase in tall graminoids, decreased diversity, decrease of bryophytes
Alpine and subalpine grasslands	E4.3 and E4.4	10-15	(#)	Increase in nitrophilous graminoids, biodiversity change
Moss and lichen dominated mountain summits	E4.2	5-10	#	Effects upon bryophytes or lichens
Mire, bog and fen habitats (D)				
Raised and blanket bogs	D1	5-10[a,c]	##	Change in species composition, N saturation of *Sphagnum*
Poor fens	D2.2 [d]	10-20	#	Increase sedges and vascular plants, negative effects on peat mosses
Rich fens	D4.1[e]	15-35	(#)	Increase tall graminoids, decrease diversity, decrease of characteristic mosses
Mountain rich fens	D4.2	15-25	(#)	Increase vascular plants, decrease bryophytes
Inland surface water habitats (C)				
Permanent oligotrophic waters	C1.1			
• Softwater lakes	C1.1	5-10	##	Isoetid species negatively affected
• Dune slack pools	C1.16	10-20	(#)	Increased biomass and rate of succession
Coastal habitat (B)				
Shifting coastal dunes	B1.3	10-20	(#)	Biomass increase, increase N leaching
Coastal stable dune grasslands	B1.4	10-20	#	Increase tall grasses, decrease prostrate plants, increased N leaching
Coastal dune heaths	B1.5	10-20	(#)	Increase plant production, increase N leaching, accelerated succession
Moist to wet dune slacks	B1.8	10-25	(#)	Increased biomass tall graminoids
Marine habitats (A)				
Pioneer and low-mid salt marshes	A2.64 and A2.65	30-40	(#)	Increase late-successional species, increase productivity

a) use towards high end of range at phosphorus limitation, and towards lower end if phosphorus is not limiting;

b) use towards high end of range when sod cutting has been practiced, use towards lower end of range with low intensity management;

c) use towards high end of range with high precipitation and towards low end of range with low precipitation;

d) for D2.1 (quaking fens and transition mires): use lower end of range (#) and for D2.3 (valley mires): use higher end of range (#);

e) for high latitude or N-limited systems: use lower end of range.

Table 2. Cross-comparison between the ecosystem classification used in the 2002 empirical N critical load setting (according to the EUNIS system) and the classification previously used (1996 Manual on Mapping Critical Levels/Loads); with n.d. = not distinguished.

Ecosystem classification 2002	EUNIS	Ecosystem classification 1996
Heathland, scrub and tundra habitats	**F**	**Heathlands**
Tundra	F1	n.d.
Arctic, alpine and subalpine scrub	F2	Arctic and Alpine heaths
Northern wet heaths		
• 'U' *Calluna* dominated wet heath	F4.11	Upland *Calluna* heath
• 'L' *Erica tetralix* dominated wet heath	F4.11	Lowland wet heathlands
Dry Heaths	F4.2	Lowland dry heathlands
Grasslands and tall forb habitats	**E**	**Species-rich grassland**
Sub-atlantic semi-dry calcareous grasslands	E1.26	Calcareous grasslands
Non-mediterranean dry acid and neutral closed grasslands	E1.7	Species-rich heaths and neutral acidic grasslands (partly)
Inland dune pioneer grasslands	E1.94	n.d.
Inland dune siliceous grasslands	E1.95	n.d.
Low and medium altitude hay meadows	E2.2	Neutral-acid grasslands (partly)
Mountain hay meadows	E2.3	Montane-subalpine grasslands
Moist and wet oligotrophic grasslands	E3.5	Neutral-acid grasslands (partly), Mesotrophic fens (partly)
• *Molinia caerulea* meadows	E3.51	n.d.
• Heath (*Juncus*) meadows and humid (*Nardus stricta*) swards	E3.52	n.d.
Alpine and subalpine grasslands	E4.3 and E4.4	Montane-subalpine grasslands (partly)
Moss and lichen dominated mountain summits	E4.2	n.d.
Mire, bog and fen habitats	**D**	**Wetlands**
Raised and blanket bogs	D1	Ombrotrophic bogs
Poor fens	D2.2	n.d.
Rich fens	D4.1	Mesotrophic fens
Montane rich fens	D4.2	n.d.
Inland surface water habitats	**C**	**Wetlands**
Permanent oligotrophic waters	C1.1	n.d
• Softwater lakes	C1.1	Shallow softwater bodies
• Dune slack pools	C1.16	n.d.
Coastal habitats	**B**	n.d
Shifting coastal dunes	B1.3	n.d.
Coastal stable dune grasslands	B1.4	Neutral-acid grasslands (partly)
Coastal dune heaths	B1.5	n.d.
Moist to wet dune slacks	B1.8	n.d
Marine habitats	**A**	n.d.
Pioneer and low-mid salt marshes	A2.64 and A2.65	n.d.

Table 3. Suggestions to use lower, middle or upper part of the set critical loads ranges for terrestrial ecosystems (excluding wetlands), if national data are insufficient.

Action	Temperature/ Frost period	Soil wetness	Base cation availability	P limitation	Management intensity
Move to lower part	COLD/LONG	DRY	LOW	N-LIMITED	LOW
Use middle part	INTERMED	NORMAL	INTERMED	UNKNOWN	USUAL
Move to higher part	HOT/NONE	WET	HIGH	P-LIMITED	HIGH

Reports from the Working Groups

Working group 1 – Forest habitats

Chairpersons: Ursula Falkengren-Grerup, Mike Hornung
Rapporteur: Joachim Strengbom

Introduction

The working group for forest habitat has provided recommendations for ranges of critical loads (CL) for the three EUNIS classes G1, G3 and G4. The EUNIS system allows more detailed subdivisions of forests but, as there was not enough available data (studies) for such subdivisions, it was felt that in general it was only possible to set a CL at the level of these three broad ecosystem types. However, in a few cases it was possible to assign separate CLs for groupings of forest types within G1, G3 and G4, e.g. coniferous versus deciduous, boreal versus temperate. Even within G1, G3 and G4 there are some types of forest, e.g. wet-swamp forests and Mediterranean forests, for which there is no reliable data on which to base CLs and they were therefore left out of the discussions. This implies that the given ranges for CL are not applicable for forest types belonging to these categories. The CLs for G1, G3 and G4 were set for a number of different compartments of the ecosystems (listed and specified below) and at the end the CL for these compartments was weighted together into one recommended CL for each of the three EUNIS classes.

The group began by discussing some general guidelines for the valuation of the studies that were used for setting the CL ranges. There was an agreement in the working group that the CLs should be set to protect all forests by protecting the most sensitive, i.e. the CL was set for the ecosystem that showed responses at the lowest level of N input. This implies that studies where no response was found for a specified indicator at a specified level of N input did not influence the range of recommended CL. It was assumed that in these latter cases the forest, and the specified indicator would respond given a longer period with N input above the CL or that changes had already occurred before the study was done. This concept is the same that has been used for N saturation and the risk of nitrate leaching. Today, we lack information of the time before such an ecosystem show responses to increased N input. Dynamic modelling on the responses to increased N deposition may be one way of predicting the time at which responses occur.

The group also stated that when data from N simulation experiments are used to set CLs, the background deposition at the experimental site need to be added to the doses used. This also implies that CL derived from an N addition experiment could by definition not be set to a lower value than the background deposition at the site. To set a CL below the background deposition mass-balance models or dynamic models should be used. It was also concluded that the best way of setting the background deposition for an experimental site would be by using the canopy budget model for throughfall (Ulrich, 1983; de Vries *et al.*, 2000). This model probably underestimates the true N deposition at a forest site, but gives a better value than ordinary throughfall measurements. For assessing N deposition over larger areas, the suggestion was to use EMEP modelled data.

Procedure for assessing the recommended CL

For each indicator, a range of CL was suggested in the background document. In most cases several studies were given as support for this range. Before setting a CL the working group evaluated the different studies and special attention was given to the following criteria:

- Is the CL suggested in accordance with the effects found in the studies?
- Has the background deposition been included and is the given value robust?
- Are there any important confounding factors such as S deposition, climatic variation along gradients etc.?
- Are there other studies that need to be included?

A range for the CL was then proposed. If the treatment effect at a given input was very large, the working group in some cases recommended a lower value of the CL than the addition responsible for the effect. In such cases the reliability of the CL was set to expert judgement (#). Moreover, the group did not discriminate between correlative and experimental studies, i.e. they were given equal credibility in the discussion of ranges for CL.

Indicators for which CL were discussed

Soil processes

Increased mineralisation and nitrification rates are strongly dependent on the deposition history. This may explain why some sites show little or no response to even large N additions. Thus it is important to know how far the system is from N saturation. There are however correlative studies that show large differences in both mineralisation and nitrification rate between regions with different N deposition. The recommendation is that CL for changed mineralisation and nitrification rates is set from the most sensitive system (soils), but the upper end of the range may be used for soils with higher pH, i.e. soils with high availability of macronutrients.

Recommended CL: 10-15 kg N ha^{-1} yr^{-1}
Reliability: #

For NH_4^+ accumulation there are some references cited in the background document (Roelofs *et al.*, 1985; Boxman *et al.*, 1988) that may not be representative for the response of European forests or not even the majority of forests in the Netherlands where the studies were conducted. Other studies from the Netherlands indicate a higher value for CL. In any case, as NH_4^+ accumulation is not specified as an indicator for CL, no CL was set on the basis of these studies.

Increased risk of nitrate leaching from coniferous forests is well documented in several studies, but the data for broad-leaved forest is less reliable. Thus, the group decided to set the reliability of the CL for coniferous forests to reliable, but only to expert judgement for broad-leafed forests.

Recommended CL (coniferous forests): 10-15 kg N ha^{-1} yr^{-1}
Reliability: ##

Recommended CL (broad-leafed forests): 10-15 kg N ha^{-1} yr^{-1}
Reliability: (#)

In the background document some studies reporting effects on decomposition rates following increased N input were cited. Because both the effects and direction of the response seem to vary between different studies and different experimental sites, it was concluded that the knowledge of effects on decomposition is too uncertain to support the setting of CLs for this indicator.

Minor comments to the soil chapter:

- Fig. 3.2 (p. 19 in Bobbink *et al,* (2002)) on the relationship between soil solution nitrate and throughfall should be updated with new data from ICP Forests (Wim de Vries will provide data).
- References from the NITREX work should be included for completeness, but the already quoted studies should be the base for the CL. These papers provide data on mineralisation, nitrification, litter decomposition and leaching for all NITREX sites, summary of NITREX sites and results of simulation experiments in USA, effects of N saturation and effects of inputs of reduced and oxidised N. The new studies are: Gundersen *et al.* (1998), Emmet (1999), Emmet *et al.* (1998) (Summary of NITREX sites) and Schleppi *et al.* (1999).

Tree growth

Although it was concluded that both increased and decreased tree growth could be a sign of ecosystem effects from increased N input, the conflicting results and the large uncertainties of the actual effects on tree growth made it impossible to set a CL for this indicator.

Trees (susceptibility to parasites)

The cited N manipulation experiments that reported increased susceptibility to parasites following increased N input were conducted in areas with rather high background deposition. The effect of the lowest N treatment was however large. Thus, the CL was set at a lower value than the dose (plus background deposition) that caused the effect, which resulted in reduced reliability (expert judgement).

Recommended CL (temperate forests): 15-20 kg N ha^{-1} yr^{1}
Reliability: (#).

The group concluded that there was insufficient data on which to judge the applicability of the CL to boreal forests.

Nutrient imbalances

A potential problem with the assessment of the impact of N deposition on nutrient imbalances is that there are sites in Europe where the natural availability of base cations is extremely low. At such sites even very small amounts of N deposition may result in large changes of ratios between N and macronutrients such as K^+ and Mg^{2+}. Data from ICP Forests could be used in order to see how common nutrient imbalances are, when the extreme sites have been excluded. An analysis is to be carried out before the end of 2002 of relevant data from the level II plots within the ICP Forests programme. Depending on the outcome of this new analysis, further adjustments of the CL range may be needed. It was concluded that the information on background deposition in some references cited in the background document (Nebe, 1991; Duquensay *et al.*, 2000; Hippeli and Branse, 1992) was not robust, and thus, the reported effects in these papers were not used as a base for the recommended CL range. Information from other studies were used in order to set the range of CL. This range was based on the effect on N to nutrient ratios and foliar N concentrations observed following N addition in manipulation experiments.

Recommended CL: 15-20 kg N ha^{-1} yr^{-1}
Reliability: #

Winter injury, physiological changes

The working group regarded the data support by the studies available as too weak to allow the recommendation of any CL based on these indices. Changes in N/nutrient ratios could be regarded as physiological changes of trees, but these effects are already included under nutrient imbalances.

Biodiversity

Mycorrhiza (temperate and boreal)

No distinction was made between above (sporocarp production) and below ground effects (mycelia). The two studies from North America (one on effects on sporocarp production and the other on effects on mycelia) that were cited in the background document were regarded as less important for two reasons. First, they were not conducted in Europe and second, there were uncertainties about how large the dry deposition at the investigated site was and thus, it was difficult to judge how large the actual total N deposition along the gradient studied was. From Strengbom *et al.* (2001) the effect from the lowest treatment level was very large and thus the lower end of the range was set to a lower value than the N addition responsible for the effect reported. In accordance with the procedure stated in the introduction, reliability is in such cases automatically set as expert judgement. The studies that suggested a lower CL were all correlative studies where confounding factors were believed to be of large importance for the presented results.

Recommended CL: 10-20 kg N ha^{-1} yr^{-1}
Reliability: (#)

Ground flora (temperate and boreal)

The recommended CL was based on both correlative and manipulation studies for boreal areas and only on correlative studies for temperate forests. The reported effects used were changes in species composition and increased number of nitrophilous species and increased susceptibility to parasites among vascular plants.

Recommended CL: 10-15 kg N ha^{-1} yr^{-1}
Reliability: #

Lichens and algae (temperate and boreal)

Large uncertainties about background deposition, confounding factors such as effects of S deposition and climatic differences in the correlative studies resulted in a recommendation of higher values for the CL than was suggested in the background document. Due to these uncertainties the reliability was set to an expert judgement.

Recommended CL: 10-15 kg N ha^{-1} yr^{-1}
Reliability: (#)

Effects on fauna

It was concluded that it was uncertain whether the effect on terrestrial snails, cited in the background document, was an effect derived from acidification (S deposition) or from N deposition. Thus, this study was not regarded as adequate support for recommending a CL for snails. Regarding effects on other parts of the fauna, data were found to be either insufficient or too weak in order to recommend any CL.

Overall recommendation

On the bases of the listed CL for the different indices presented above the working group recommend that a general CL for both boreal and temperate forests is set to **10-20 kg N ha^{-1} yr^{-1}, with the reliability of # (quite reliable).**

The recommended empirical critical loads for nitrogen deposition to forest habitats are summarized in table 1.

Table 10.2 in Bobbink *et al.* (2002) could be used to guide the choice of CL within the recommended range. National experts may be consulted on questions of whether or not specific criteria should be used.

Mass balance *vs.* empirically derived (biological effects) CLs

The two different approaches for establishing CL were briefly discussed during the meeting. The suggestion by the working group was that, whenever possible, both mass balance and biologically derived CL should be presented. If the two approaches give different values for CL, the one with the lowest value should be used until the reasons for the difference have been clarified.

Gaps in knowledge

Although a number of studies on the effects of N deposition in forest ecosystem have been published since the last update (1996) of the empirical critical loads, there is still a lack of data for several forest types. This is most obvious for Mediterranean forests, where no studies at all could be found. Also for wet forests information about effects of N deposition seem to be absent. In order to set up CLs that will protect European forests from effects of N deposition, data on responses from these ecosystems are needed. More data from manipulation experiments with low levels of N input (from areas with low background deposition) is also needed for several of the indicators for which the working group has made a recommendation of CL.

There is also a large gap in knowledge regarding the time span that could be expected before effects are seen in some forest types or to what extent some changes have already occurred and other changes may appear more slowly. Several manipulation experiments reported no detectable effects even though rather large amounts of N were added. Dynamic modelling may be one way of determining the time until effects occur while also increasing our understanding of the functioning of these forest types. There are also large uncertainties about how the forest ecosystems of Europe will respond to reduced deposition of N. At what level of N deposition can we expect recovery of the systems and how is the time to recovery affected by exceedances of the CL? Very few studies have addressed questions of biological recovery. Such questions are, however, important as they are related to how rapidly we need to reduce the N deposition in order to protect our ecosystems. Dynamic modelling of the recovery may be one way of increasing our knowledge about recovery and the functioning of the forest ecosystems.

There was also a brief discussion about effects of different forms of N deposition (NH_x and NO_y). There was an agreement about the potential differences in effects between different forms of N deposition, but data on this topic was too scarce to allow different CLs to be set for NH_x and NO_y. As there are large differences in which form of N is dominating the deposition in different areas of Europe and that changes in deposition pattern may occur in the future, increased data on this topic is needed.

Table 1: Overview of empirical critical loads for nitrogen deposition (kg N ha^{-1} yr^{-1}) to forest habitats. ## reliable; # quite reliable; (#) expert judgement.

	Critical load (kg N ha^{-1} yr^{-1})	**Indication of exceedance**
Soil processes	10-15 #	Increased mineralisation, nitrification
	10-15 ##	Nitrate leaching (coniferous forests)
	10-15 (#)	Nitrate leaching (broad-leafed forests)
Tree growth	-	
Sensitivity to pathogens (temperate forests)	15-20 (#)	Increased susceptibility to parasites, change concentration of fungistatic phenolics
Nutrient imbalances	15-20 #	Changed N/macro nutrients ratios, decreased P-, K-, Mg-concentration and increased N concentration in foliar tissue
Winter injuries, changed physiology	-	
Mycorrhiza (temperate and boreal forests)	10-20 (#)	Reduced sporocarp production, changed/reduced below-ground species composition
Ground flora (temperate and boreal forests)	10-15 #	Changed species composition, increase of nitrophilous species, increased susceptibility to parasites
Lichens and algae (temperate and boreal forests)	10-15 (#)	Increase of algae, decrease of lichens
Fauna	-	
Overall		
Temperate forests	10-20 #	Changes in soil processes, ground flora, mycorrhiza and increased risk of nutrient imbalances and susceptibility to parasites
Boreal forests	10-20 #	Changes in soil processes, ground flora, mycorrhiza and increased risk of nutrient imbalances and susceptibility to parasites

References (that were not included in the background document)

Bobbink, R., Ashmore, M., Braun, S., Flückiger, W., Van den Wyngaert, I.J.J., 2002. Empirical Nitrogen Critical Loads for Natural and Semi-natural Ecosystems: 2002 Update. Background Document for the Expert Workshop held under the UNECE Convention on Long-range Transboundary Air Pollution, Berne, Switzerland, 11-13 November 2002.

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Working Group 2 - Grasslands, fens, mires, inland surface waters, coastal and marine habitats

Chairman: Alan Davison
Rapporteur: Morten Strandberg

Introduction

The critical loads concept involves a reiterative process in which loads are revised in the light of new information. This is essential in order to make them reliable tools for mapping both loads and exceedances in Europe. On this occasion more ecosystem types were included, further benefiting the mapping efforts. The basis for the Working Group discussions was a background paper (Bobbink *et al.*, 2002) that reviewed expert knowledge gained since the last revision in 1995. The authors suggested tentative critical loads for a large number of ecosystems. The primary objective of Working Group 2 was to discuss, evaluate and eventually revise the empirical critical loads for nitrogen deposition suggested for grasslands, fens, mires, inland surface waters, coastal and marine habitats in the background paper. Because of differences in ecosystem definitions used by the Parties to the Convention this revision also included a harmonisation process based on a new classification system called EUNIS. Therefore a second objective was to ensure that the published data was allocated to the most appropriate EUNIS classes and that those classes were in accord with ecosystems that had previously been assigned critical loads.

The WG2 discussions focussed on:

- whether the review of studies was complete,
- whether studies were correctly used and cited,
- the weighting of critical loads when based on single studies,
- assessing the background deposition in field studies,
- whether ecosystem differentiation was sound and in accordance with EUNIS classification,
- if environmental and topographic conditions were taken into account in the assessment,
- whether the reliability was assessed correctly,
- the basis for and appropriateness of the range of CL,
- the selection of effect endpoints,
- the identification of gaps in knowledge and recommendations for future research,
- uncertainty handling,
- handling of time delay, i.e. delayed damage and delayed recovery.

The procedure of the group work was to take one ecosystem time at a time from the background paper and discuss it according to the above mentioned list. After that it was decided whether the CL should be revised or kept as suggested in the background document. The result of this work is summarized in Table 1.

Table 1. Proposal for Critical Loads for grasslands, fens, mires, inland surface waters, coastal and marine habitats. ## reliable; # quite reliable; (#) expert judgement

Ecosystem type	EUNIS code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Sub-atlantic semi-dry calcareous grassland	E1.26	15-25	##	Increase tall grasses, decline in diversity, increased mineralisation and N-leaching
Non-mediterranean dry and neutral closed grassland	E1.7	10-20	#	Increase in graminoids, decline of typical species
Inland dune pioneer grasslands	E1.94	10-20	(#)	Decrease in lichens, increase biomass
Inland dune siliceous grasslands	E1.95	10-20	(#)	Decrease in lichens, increase biomass, increased succesion
Low and medium altitude hay meadows	E2.2	20-30	(#)	Increase in tall grasses, decrease in diversity
Mountain hay meadows	E2.3	10-20	(#)	Increase in nitrophilous graminoids, changes in diversity
Moist and wet oligotrophic grasslands	E3.5			
• Molinia meadows	E3.51	15-25	(#)	Increase in tall graminoids, decreased diversity, decrease of bryophytes
• Nardus stricta swards	E3.52	10-20	#	
Alpine and sub-alpine grasslands	E4.3 & E4.4	10-15	(#)	Increase in nitrophilious graminoids, biodiversity change
Moss and lichen dominated mountain summits	E4.2	5-10	#	Effects upon bryophytes and lichens
Poor fens	D2.2	10-20	#	Increase sedges and vascular plants, negative effects on bryophytes
Rich fens	D4.1	15-25	(#)	Increase tall graminoids, decrease diversity, decrease of characteristic mosses
Mountain rich fens	D4.2	15-25	(#)	Increase vascular plants, decrease bryophytes
Oligotrophic lakes	C1.1	5-10	##	Isoetid species are negatively affected
Dune slack pools	C1.16	10-20	(#)	Increased biomass and rate of succession
Shifting coastal dunes	B1.3	10-20	(#)	Biomass increase, increase N-leaching
Coastal stable dune grasslands	B1.4	10-20	#	Increase tall grasses, decrease prostrate plants, increase N-leaching
Coastal dune heaths	B1.5	10-20	(#)	Increase plant production, increased N-leaching, accelerated succession
Moist to wet dune slacks	B1.8	10-25	(#)	Increased biomass of tall graminoids
Early succesional salt water marshes	A2.64 and A2.65	30-40	(#)	Increase late succesional species, increase productivity

The lower end of the CL range for dune slack pools was brought down to 10, partly to standardise with that for dune slacks, but also to harmonise with the upper limit for the oligotrophic coastal lakes which seem to differ only in their buffering capacity.

Knowledge gaps

- There is a need for new, long-term studies in which N is applied in areas with low N-deposition, at low rates, and with high frequency in order to properly simulate N deposition. Very few previous experiments fulfil these criteria. Such studies are prerequisites for the provision of knowledge concerning "delayed damage time" and "delayed recovery time".
- Data are sparse or lacking for several grassland types, especially mediterranean and eastern steppe types, for poor and rich fens, dune slack pools, dune heaths, inland dune and salt water marsh.
- The EUNIS classification needs clarification, especially some grassland types and Nordic mires. For WG2 the most problematic division was at the highest level based on "wetness": E1-E3.
- In view of emerging evidence that reduced N may be more damaging than oxidised N, comparative studies of effects of NH_y and NH_x are needed.

Conclusions and comments made by WG2

- Data are still needed to provide reliable ## empirical critical loads for the majority of ecosystems included in the WG2 list.
- Appropriate, valid data became available immediately prior to or during the workshop, for some grasslands and dune types.
- When the CL suggested in the background paper was lowered it was in many cases in order to protect bryophytes and lichens.
- Mesocosm studies eliminating the background deposition were of great value for setting empirical critical loads in areas with medium to high background N-deposition.
- It should be clarified how to reach a CL from a short-term addition study that produces clear effects within a range of say < 3 years. Studies of this kind were cited in Chapter 2 of the background paper.
- Table 10.2. in Bobbink et al. (2002) needs further discussion/revision. WG2 recommends that management, precipitation, oceanity – continentality and temperature should be considered for inclusion. Above all, the rationale underlying the modifiers should be explained.

References

Bobbink, R., Ashmore, M., Braun, S., Flückiger, W., Van den Wyngaert, I.J.J., 2002. Empirical Nitrogen Critical Loads for Natural and Semi-natural Ecosystems: 2002 Update. Background Document for the Expert Workshop held under the UNECE Convention on Long-range Transboundary Air Pollution, Berne, Switzerland, 11-13 November 2002.

Working Group 3 - Heathlands, scrub and tundra habitats, bogs

Chairman: Jan Roelofs
Rapporteur: Sarah Woodin

Introduction

This report summarises the recommendations of the working group for the critical loads for the effects of excess nitrogen on heathland, scrub and tundra habitats and bogs, providing supporting evidence for cases in which the recommendation differs from that initially given in the Background Document (Bobbink et al., 2002). The CLs recommended are all ranges (Table 1), and for some habitats specific modifiers determining which end of a range the CL for a specific site should be set are stated (Table 2). These modifiers should always be noted in conjunction with the CL estimates. It is suggested that the modifiers will be of greatest use to conservation practitioners and others wishing to determine critical loads for particular sites of interest: in some cases they may be less applicable in CL mapping.

The reliability of each CL estimate is stated (Table 1) and justified. There was considerable discussion regarding the interpretation of experimental data in the setting of CLs. The limitations of experiments are taken into account in assigning CL reliabilities, and a summary of major concerns is provided.

Recommendations are based on published data, or data which are appended to the Background Document. Other unpublished data and anecdotal evidence were cited during discussions; these were not taken into account in decisions regarding the actual setting of CLs, but nevertheless often corroborated decisions made.

The initial step in the evaluation of the CL for each habitat was consideration of the appropriateness of the EUNIS class and the way in which the ecosystem classification previously used for empirical CLs mapped onto it. Some modifications were made and are explained.

The following text summarises evidence considered in reaching the Working Group Recommendations, which are presented in Tables 1 & 2, particularly where those recommendations differ from the Background Document.

F1 Tundra

The EUNIS classification is appropriate and requires no further sub-division. The critical load is judged to be quite reliable despite being based on only one experiment. The justifications for this are (i) the experiment included three different plant communities, (ii) background deposition is very low (<1 kg N ha^{-1} yr^{-1}) (iii) the N treatment (10 kg ha^{-1} yr^{-1}) was within the critical load range (iv) effects were seen rapidly (within 3 years) and are very persistent. Subsequent unpublished data (Woodin *et al.*) from the experiment confirm the occurrence and persistence (7 years after the 3 year treatment period) of effects.

Inclusion of a phosphorus treatment in the experiment gave rise to unequivocal evidence of co-limitation by N and P, with greater responses to N when P was also applied. The CL is thus likely to be higher in a system limited by P than one which is not. Since P limitation can occur in all the habitats listed in Table 1, it is recommended that P availability is used as a modifier of CL, within the recommended range for the particular habitat, as stated in Table 2.

F2 Arctic, alpine and sub-alpine scrub

The EUNIS classification is appropriate.

The CL is mainly based on extrapolation from the tundra study and is therefore expert judgement. The CL is recommended as 5-15 kg N ha^{-1} yr^{-1}, the upper end of the range has not been reduced to 10 as for tundra as there is no supporting evidence for this for alpine systems. A study in the Dovre mountains in Norway, at 1000-1400 m a.s.l. should be considered in further evaluation of this CL. Three years fertilisation of a *Betula nana* dominated community with 12 and 61 kg N ha^{-1} yr^{-1} (background 2-4 kg N ha^{-1} yr^{-1}) appeared to have no effect on plant growth and the added N was not accounted for in the plant or soil pools (Paal *et al.*, 1996). Similarly the only response to 7, 35 and 70 kg N ha^{-1} yr^{-1} over three years in an oligotrophic lichen community appeared to be a negative treatment effect on *Cladonia coccifera* (Möls *et al.*, 2001). However, the analysis in this paper does not enable clear application of the data to CLs. Unpublished 10 year data from the lichen community study show negative effects on lichens at 35 but not at 7 kg N ha^{-1} yr^{-1}. The Working Group recommend that clearer evaluation of the experiment and the data is required to enable application to the CL.

F4.11 Northern wet heath

The wet heaths of both the uplands and lowlands in the UK and The Netherlands (and other comparable countries) fall within this EUNIS habitat type. Since upland, *Calluna* dominated wet heaths and lowland, *Erica tetralix* dominated wet heaths were previously assigned different critical loads, based on different evidence, it was recommended that this habitat distinction be retained. There is no clear altitudinal cut-off distinguishing these habitats, so the primary distinction should be based on species dominance. Advice to the mapping community on how to map each (which may be based on altitude) should be provided at a national level.

F4.11 "U" *Calluna* dominated wet heath (≈ upland)

CL is recommended to be 10-20 kg N ha^{-1} yr^{-1} because there is no justification for a higher CL than for dry heaths (which have a reliable estimate) as the dominant species and the indicators of exceedence (particularly lichen decline) are the same. This is expert judgement, being based on extrapolation, as the only available experimental evidence is based on total N inputs (background + treatment) significantly above the CL range.

In order to increase reliability of the CL it was proposed that the relationship between foliar N:P ratio and N deposition be used (following Kirkham, 2001), with N:P ratios above the critical range 14-16:1 being taken to indicate CL exceedence. However the R^2 of this relationship in Kirkham (2001) was only 0.1. In addition the CS 2000 *Calluna* data (unpublished) do not show the expected increase in N:P ratio with increasing N deposition, and data from a national *Hylocomium splendens* survey (Woodin & Sullivan, 2001) show N:P ratio to have a weaker correlation with N deposition than [N] alone. It is also likely that where N deposition increases mineralisation rate, P availability may also be increased and thus the assumption implicit in the argument does not hold.

F4.12 "L" *Erica tetralix* dominated wet heath (≈ lowland)

CL for this habitat is based on a model which includes sod (turf) cutting on a 50 yr interval (and grazing). Work on a model adapted for UK lowland (dry) heathlands (without sod cutting) has demonstrated how habitat management practices may affect the CL. For example, a change from low intensity mowing (above-ground biomass removal only) to high intensity mowing (with biomass and litter removal) increases the critical load by 5 kg N ha^{-1} yr^{-1}. Sod cutting, which also removes soil, could increase the CL further. Hence an upper CL of 25 kg N ha^{-1} yr^{-1} is recommended for sites subject to sod cutting. The CL for sites which are not sod cut should not be set at this upper end of the range.

The lower end of the CL range is recommended to be 10 kg N ha^{-1} yr^{-1}, as for *Calluna* dominated heaths (wet and dry). Comparison of the likely relative sensitivity of *Erica* and *Calluna* to N did not give rise to any clear outcome enabling differentiation of the lower end of the CL ranges.

F4.12 Southern wet heath

There are no data for habitats matching this EUNIS classification and thus no CL can be recommended.

F4.2 Dry heaths

The EUNIS classification is appropriate.

The recommended CL is 10-20 kg N ha^{-1} yr^{-1}. The upper value of 25 kg N ha^{-1} yr^{-1} in the Background Document was a typographical error. The lower value is reduced due to new measurements (over 2 years) of background deposition at the key experimental site in Surrey, UK (NO_3^- & NH_4^+ by bulk deposition; NO_2/NH_3 by diffusion tube; aerosol/HNO_3-N not included). These data are unpublished, but a report is included in the Background Document. Total bulk and dry deposition is measured at 8 kg N ha^{-1} yr^{-1}. Significant effects are seen with additions of 8 kg N ha^{-1} yr^{-1}, and thus at a total load of 16 kg N ha^{-1} yr^{-1}. The CL is suggested to be considerably below this total, the sensitivity of the system which has already been demonstrated being confirmed by lack of recovery of the fungal community during 6 years after cessation of treatment application (unpublished data). On the basis of this experiment, evidence of decline of lichens in The Netherlands (Roelofs, pers. comm.), modelling (including the effect of different forms of management) and expert judgement, the recommendation is for a CL lower limit of 10 kg N ha^{-1} yr^{-1}. This estimate is reliable.

D1 Raised and blanket bogs (including bog pools)

The EUNIS classification is appropriate.

The recommended CL of 5-10 kg N ha^{-1} yr^{-1} will protect the most sensitive systems. This low range is based in large part on the relationship between tissue N in *Sphagnum* and N deposition (Background Document Fig. 6.2). However, the significance of the increase in tissue N below the point of saturation for either *Sphagnum* growth or pore water [N] is not clear. Anecdotal evidence suggests that in the UK and Norway many such bogs receiving more than 5 kg N ha^{-1} yr^{-1} are healthy. Expert judgement, based on observations that response to N is less in wetter areas of a bog, is that bogs receiving high effective precipitation (eg N&W UK, Norway) are less sensitive to N than those in drier areas (eg Netherlands, Sweden). Precipitation is thus recommended to be a modifying factor to be taken into account when assigning CLs to individual bog sites.

Limitations on interpretation and application of CL values

1. Treatment application frequency: Treatments applied rather infrequently are suggested to have less effect than more frequent applications, and this must be taken into account in interpretation of experiments. The ideal is for real-time watering, but this is rarely achieved (particularly in winter when watering apparatus may freeze).

2. Treatment solution concentration: There is concern that the high concentrations of solutes applied in some experiments may themselves cause damage, independently of the effects of the total load of N being applied over time. This is sometimes not easily avoided, and if natural precipitation is low it may be inappropriate to reduce solute concentration by application in much larger volumes.

3. NH_y/NO_x differential effects: CLs are set for total N deposition but there is increasing evidence of NH_y having greater effects than NO_x, particularly on bryophytes, herb species within species rich heathlands and soft water macrophytes. Most experiments which have been used to define CLs have used NH_y, or a mixture of the two forms, and so the worst case scenario has been allowed for. However, there is a gap in our knowledge and we need more information before we can apply differential ion effects to our interpretation and application of CLs.

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Bobbink, R., Ashmore, M., Braun, S., Flückiger, W., Van den Wyngaert, I.J.J., 2002. Empirical Nitrogen Critical Loads for Natural and Semi-natural Ecosystems: 2002 Update. Background Document for the Expert Workshop held under the UNECE Convention on Long-range Transboundary Air Pollution, Berne, Switzerland, 11-13 November 2002.

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Table 1: Summary Recommendations: Nitrogen Critical Loads for Heath, scrub, tundra & bog.
reliable; #quite reliable; (#) expert judgement

EUNIS code	*Habitat*	*kg N* ha^{-1} yr^{-1}	*Reliability*	*Indicators of exceedence*	*Modifying factors (see Table 2)*
F1	Tundra	5-10	#	Change in biomass, physiological effects and changes in species composition in moss layer, decrease in lichens	P
F2	Arctic, alpine & subalpine scrub	5-15	(#)	Decline in lichens, mosses and evergreen shrubs	P
F4.11	Northern wet heath				P
	F4.11 “U” Calluna dominated wet heath	10-20	(#)	Decreased heather dominance, decline in lichens and mosses	Management, P
	F4.11 “L” Erica tetralix dom. wet heath	10-25	(#)	Transition heather to grass	Management, P
F4.12	Southern wet heath	no data	-	-	-
F4.2	Dry heath	10-20	##	Transition heather to grass, decline in lichens	Management, P
D1	Raised and blanket bogs	5-10	##	Change in species composition, N saturation of *Sphagnum*	Precipitation, P

Table 2: Factors modifying the critical load of a habitat within the recommended range

Factor	*Habitat*	*Indicator for CL toward high end of range*	*Indicator for CL toward low end of range*
Management[1]	F 4.11 "L" Northern wet heaths Erica dominated F4.2 Dry heath	Sod (turf) cutting practiced	No sod (turf) cutting, and low intensity management
Precipitation	D1 Raised and blanket bog	High precipitation	Low precipitation
Phosphorus status[2]	All in Table 1	P limitation	P not limiting

[1] Grazing and burning of Northern wet heath, Calluna dominated will affect response to N and should be taken into account locally

[2] This modifier should not be applied if the sole indicator of exceedence in a habitat is loss of lower plant species due entirely to the direct effect of N toxicity.

Background Documents

The main chapters of the background documents were presented by the authors during the plenary of the workshop.

Empirical nitrogen critical loads for natural and semi-natural ecosystems: 2002 update

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1. Introduction

The emissions of ammonia (NH_3) and nitrogen oxides (NO_x) have strongly increased in Europe in the second half of the 20th century. Ammonia is volatilized from intensive agricultural systems, such as dairy farming and intensive animal husbandry, whereas nitrogen oxides originate mainly from burning of fossil fuel by traffic and industry. Because of short- and long-range transport of these nitrogenous compounds, atmospheric nitrogen (N) deposition has clearly increased in many natural and semi-natural ecosystems across the world. Areas with high atmospheric nitrogen deposition are nowadays central and western Europe, eastern USA and, since the 1990s, Eastern Asia (e.g. Galloway and Cowling, 2002). Atmospheric N deposition rates of 20-60 kg N ha^{-1} yr^{-1} have been observed in non-forest ecosystems in Western Europe, whereas in forest stands in Europe or the USA values between 20-100 kg ha^{-1} yr^{-1} have been reached, instead of the estimated background inputs of 1-3 kg N ha^{-1} yr^{-1} in the early 1900s (e.g. Galloway, 1995; Asman *et al.,* 1998; Fowler, 2002).

The availability of nutrients is one of the most important abiotic factors which determines the plant species composition in ecosystems Nitrogen is the limiting nutrient for plant growth in many natural and semi-natural ecosystems, especially of oligotrophic and mesotrophic habitats. Most of the plant species from such conditions are adapted to nutrient-poor conditions, and can only survive or compete successfully on soils with low nitrogen availability (e.g. Tamm, 1991; Bobbink *et al.,* 1998; Aerts and Chapin, 2000). In addition, the nitrogen cycle in ecosystems is complex and strongly regulated by biological and microbiological processes, and it is thus likely that many changes can occur in plant growth, inter-specific relationships and soil-based processes as a result of increased deposition of air-borne nitrogen pollutants.

The series of events which occurs when nitrogen inputs increase in an area with originally low background deposition rates is highly complex. Many ecological processes interact and operate at different temporal and spatial scales. The severity of the impacts of atmospheric nitrogen deposition depends on a number of factors, of which the most important are: (1) the duration and total amount of the increased inputs; (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) the past and present land use or management. Acid neutralising capacity (ANC), soil nutrient availability, and soil factors which influence the nitrification potential, nitrogen immobilisation and denitrification rates, are especially important. As a consequence, high variations in sensitivity to atmospheric nitrogen deposition have been observed between and within different natural and semi-natural ecosystems.

Despite this highly diverse sequence of events, the following main effect "categories" can be recognised. A schematic overview of the possible sequence of events is given in Figure 1.1. We emphasise that this is a rather simplified outline of possible events, which will not necessarily be followed in any specific ecosystem.

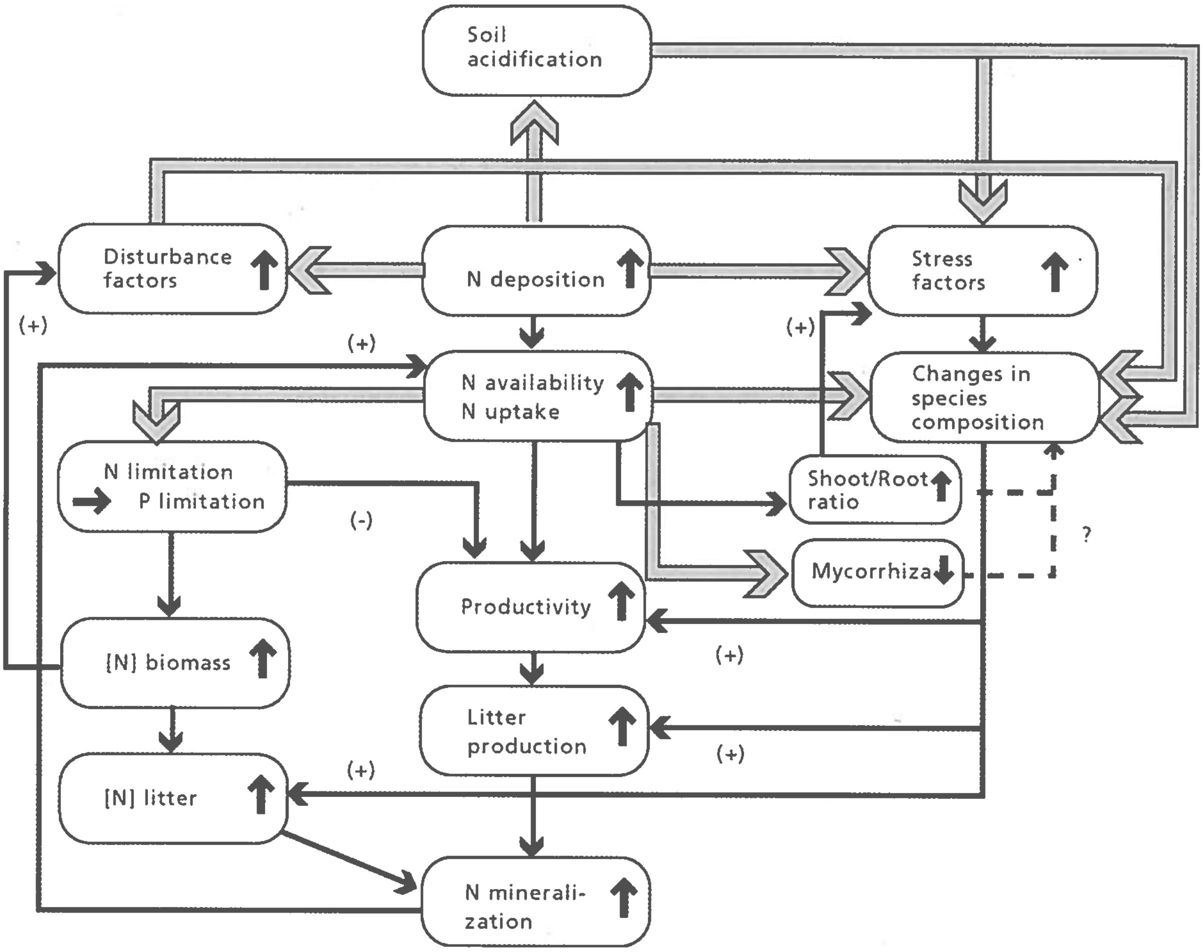

Figure 1.1. Scheme of the main impacts of enhanced nitrogen deposition on ecosystems. ↑ indicates increase; ↓ decrease; small arrow: effect will occur in the short term (< 5 yrs); wide arrow: indicates long-term effect. (+): positive feedback; (-) negative feedback (Bobbink and Lamers, 2002). In this figure stress is considered to occur when external constraints limit the rate of dry matter production of (part of) the vegetation, whereas disturbance consists of mechanisms which affect plant biomass by causing its partial or total destruction.

(a) Direct toxicity of nitrogen gases and aerosols to individual species

An important effect of dry and wet nitrogen deposition can be direct toxicity to the above-ground parts of individual plants. These effects have been mostly studied for crops and young trees, but studies with native herbaceous or dwarf shrub species in open top chambers (OTC) have also demonstrated changes in physiology and reductions in growth at high concentrations of airborne N pollutants (e.g. Pearson and Stewart, 1993). Direct toxicity effects of pollutant gases are not treated in this background document, as they are part of the critical level approach. Direct effects of wet deposited nitrogen on bryophytes and lichens are, however, considered, especially in organic soil systems with high abundance of these groups.

(b) Accumulation of nitrogen compounds, resulting in changes of species composition

Enhanced nitrogen loads result in an increase in the availability of inorganic nitrogen in the topsoil in the short term. In the first step this leads to an increase in plant productivity in nitrogen-limited vegetation and thus to higher annual litter production. Because of this, nitrogen mineralisation will gradually increase, which may cause an extra increase in plant productivity. This is a positive feedback, because higher nitrogen mineralisation gives higher nitrogen uptake, etc. In the longer term, competitive exclusion of characteristic species of oligotrophic or mesotrophic habitats by relatively fast-growing nitrophilous species is to be expected. The rate of nitrogen cycling in the ecosystem is thus clearly enhanced in this situation, although the response time to enhanced nitrogen inputs can be long in this respect in highly organic soils with their high C:N ratio, or, perhaps in any soil with large potential nitrogen sinks. When the nitrogen deficiencies in the ecosystem are no longer limiting, plant growth becomes restricted by other resources, such as phosphorus (P). In this situation, the productivity of the vegetation will not further increase. Nitrogen concentrations in the plants will however tend to increase, which may affect the palatability of the vegetation for herbivores or the sensitivity to pathogens. In addition, the first-stage decomposability of litter becomes higher, because of the decreased C:N ratios in the litter. Finally, the ecosystem becomes "nitrogen-saturated", which leads to an increased nitrogen (e.g. nitrate) leaching from the soil to the deeper groundwater.

(c) Long-term negative effect of ammonium and ammonia

In many regions with a relatively high rate of nitrogen deposition, a very high proportion of the deposited nitrogen originates from ammonia and ammonium (e.g. Asman *et al.,* 1998; Fowler, 2002). This may cause a change in the dominant nitrogen form in the soil from nitrate to ammonium, especially in habitats with moderately acidic conditions (pH: 4.5-6.5). The response of sensitive plant species can be significantly affected by this change. Species of calcareous or somewhat acidic soils are able to use nitrate, or a combination of nitrate and ammonium, as the nitrogen source, whereas early studies showed that species of acid habitats generally use ammonium (e.g. Gigon and Rorison, 1972; Kinzel, 1982), because at least some of these plants do not have nitrate reductase (Ellenberg, 1996). Recent laboratory and field studies demonstrated that most forest understory species are favoured when both ammonium and nitrate can be taken up (Falkengren-Grerup, 1998; Olsson and Falkengren-Grerup, 2000). One of the impacts of increased ammonium uptake is the reduced uptake of base cations (K^+, Ca^{2+} and Mg^{2+}). Ultimately this can lead to severe nutritional imbalances, which are considered to be the main factor in the decline in tree growth in areas with high ammonia/ammonium deposition (Nihlgård, 1985; Van Dijk *et al.,* 1990; see also section 3.3.2 of this background document). High ammonium concentrations in the soil or water layer are also toxic to the roots of sensitive plant species, causing very poor root development, and finally, inhibition of shoot growth. Strong evidence exists that several endangered grassland and heathland species are very intolerant to high ammonium concentrations and high NH_4^+/NO_3^- ratios (Roelofs *et al.,* 1996; De Graaf *et al.,* 1998).

(d) Soil-mediated effects of acidification

Soil acidification, caused by inputs of acidifying compounds, is characterized by a wide variety of long-term effects. It is defined as the loss of buffering capacity (ANC) and may lead to a decrease in pH. Decreases in pH are dependent on the buffering capacity of the soil (e.g. Ulrich, 1983; 1991). Acidifying inputs, deposited on calcareous soils, will at first not give a change in acidity. In these soils, which overly limestones, HCO_3^- and Ca^{2+} ions leach from the system because of the acidifying pollutants, but the pH remains the same until most calcium carbonate has been depleted. In soils dominated by silicate minerals (pH 6.5-4.5) buffering is taken over by cation exchange processes of the soil adsorption complexes (clay minerals and humus particles). In this situation, protons are exchanged with Ca^{2+} and Mg^{2+} of the adsorption complex, and these cations are leached from the soil together with anions, especially nitrate. Because of the restricted capacity of this buffering system, soil pH will soon start to decrease, together with losses of several base cations. In mineral soils with a large cation exchange capacity and high base saturation, this buffering may, however, hold for several hundred years, even at relatively high inputs. At low pH (<5.0), clay minerals are broken down and hydrous oxides of several metals are dissolved. This causes a strong increase of the concentration of toxic Al^{3+} and other metals in the soil solution. As a consequence of the decrease in pH, nitrification is strongly hampered or even completely absent in most soils. This may lead to accumulation of ammonium, whereas nitrate decreases to almost zero at these or lower pH values (e.g. Roelofs *et al.,* 1985). In addition, the decomposition rate of organic material in the soil is lower in these acidified soils, which leads to increased accumulation of litter (e.g. Van Breemen *et al.,* 1982; Ulrich, 1983; 1991). As a result of this complex of changes, growth of plant species and the species composition of the vegetation can be seriously affected; acid-resistant plant species will gradually become dominant, and several species typical of intermediate pH disappear.

(e) Increased susceptibility to secondary stress and disturbance factors such as drought, frost, pathogens or herbivores.

The sensitivity of plants to stress or disturbance factors may be significantly affected by deposition of nitrogen pollutants. The resistance to plant pathogens can be lowered because of a lower vitality of individual plants as a consequence of the different impacts of pollutants, or by decreased formation of secondary compounds such as phenolics, whereas increased organic nitrogen contents of plants can also result in increased herbivory. Furthermore, nitrogen-related changes in plant physiology, biomass allocation (root/shoot ratios) and mycorrhizal infection can also influence the sensitivity of plant species to drought or frost stress.

Under the Gothenburg Protocol, significant decreases in emissions of nitrogen oxides and smaller decreases in emissions of reduced nitrogen are expected by 2010. The issue of how quickly the changes described above can be reversed if nitrogen deposition falls, has thus become an important issue (e.g. Galloway and Cowling, 2002). Recent experimental evidence, and practical field experience in ecosystem restoration, suggests that, once the process of altered species composition and increased nitrogen mineralisation described in (b) above has occurred, recovery of the original vegetation may occur only over very long timescales, or with very active management intervention to decrease nitrogen status and cycling (e.g. Roelofs *et al.,* 1996). This emphasises the need for caution in setting critical loads at which these major changes in vegetation composition and nitrogen cycling do not occur.

Aims of the document

Within the Convention on Long-range Transboundary Air Pollution covering the UNECE region, procedures have been developed to set and to map critical loads for airborne nitrogen deposition. Both the steady-state mass balance method and the empirical approach have been used within this Convention. Empirical critical loads of nitrogen for natural and semi-natural terrestrial ecosystems and wetland ecosystems were firstly presented in a background document for the 1992 workshop on critical loads held under the Convention at Lökeberg (Sweden) (Bobbink *et al.*, 1992). After detailed discussion before and during the meeting, the proposed values were set at that meeting (Grennfelt and Thörnelöf, 1992). Additional information from the period 1992-1995 was evaluated and summarised in an updated background paper and published as Annex III in the Convention's Manual on Methodologies and Criteria for Mapping Critical Levels/Loads (Bobbink *et al.*, 1996). The updated nitrogen critical loads were discussed and set by full consensus at the December 1995 expert meeting held under the Convention in Geneva (Switzerland). They were also used for the development of the second edition of the Air Quality Guidelines for Europe of the World Health Organisation Regional Office for Europe (WHO, 2000). More recently, it was recognised (workshop at York held under the Convention, 2001) that considerable new insights into, and data on, the impacts of nitrogen deposition on natural and semi-natural vegetation have become available since the compilation of the last background paper in the mid-1990s. On the basis of the availability of new scientific evidence for many nitrogen-sensitive ecosystems an update of the background material used for the 1996 Convention's Mapping Manual is thus clearly needed.

The aims of this background document are:

- to collect and to evaluate the new scientific data from the period 1996 to 2002;
- to formulate an extended and updated table of empirical nitrogen critical loads for semi-natural and natural ecosystems;
- to update the Convention's Mapping Manual for empirical nitrogen critical loads;
- to provide more guidance regarding the classification and mapping of nitrogen-sensitive ecosystems.

In this background document, new information from the period 1996-2002 on the impacts of increased nitrogen deposition on the structure and function of natural and semi-natural ecosystems is given. The information is summarised by groups of ecosystems in more or less the same order as in the 1996 document. The approach used to set empirical critical loads is described in Chapter 2, while the information and proposals for updated nitrogen critical loads by groups of ecosystems is presented in Chapters 3 to 9 (from forest and woodland habitats to marine habitats). Finally, a fully updated table of empirical nitrogen critical loads is presented in the last chapter of this document, Chapter 10.

2. Updating procedure for empirical nitrogen critical loads

2.1 Updating procedure

Based on observed changes in the structure and function of ecosystems, reported in a range of publications, empirical critical loads for nitrogen were evaluated for specific receptor groups of natural and semi-natural ecosystems in both 1992 and 1996 (Bobbink *et al.,* 1992a; 1996). In this updating procedure we have used a similar *"empirical approach"* as for the earlier background documents. For this purpose, we have firstly collected, as completely as possible, European publications on the effects of nitrogen in natural and semi-natural ecosystems from the period 1996 to mid 2002. We used peer-reviewed publications, book chapters, nationally published papers and "grey" reports of institutes or organisations, if available by request. In principle, only European studies have been used as the basis for the assessment of nitrogen critical loads. However, when no or very few studies were available, non-European (mostly USA) literature has been used to make an expert judgement as reliable as possible.

Studies providing insights into ecosystem reactions to an increase in nitrogen input or availability have been conducted for a variety of reasons. This has resulted in many different designs, from correlative or retrospective field studies, experimental studies in pots and mesocosms to field addition experiments. In addition, dynamic ecosystem modelling is also relevant in this respect. We especially use statistically and biologically **significant** outcomes of **field addition experiments** and **mesocosm studies** for the assessment of empirical nitrogen critical loads. Only studies which have independent nitrogen treatments and realistic nitrogen loads and durations (below 100 kg N ha^{-1} yr^{-1}; more than 1 yr) have been used for the updating and setting of the critical load values. Studies with higher nitrogen additions or shorter experimental periods have only been interpreted with respect to the possible nitrogen limitation or sensitivity of the system. The methods used in these studies have been carefully scrutinised to identify factors related to the experimental design or data analysis, which may constrain their use in assessing critical loads. This includes evaluation of the precision of the estimated values of background deposition at the experimental site.

In addition, the results from correlative or retrospective field studies have been used as the basis for estimates of critical loads, but only as additional evidence either to complement the results from experimental nitrogen addition studies, or as an indication for expert judgement. If available, the outcome of dynamic ecosystem models has served to fill gaps where empirical data of appropriate time scales are missing, or to provide additional insight into underlying mechanisms, which are difficult to incorporate in experimental studies, such as increased frequencies of pests and diseases. In general, pot or microcosm studies have not been accepted for the setting of critical loads, except for bryophyte layer studies. However, the outcome of these studies is in some selected cases used as an indication of the nitrogen sensitivity of the most important plant species of an ecosystem.

2.2 Ecosystem classification

In this background document, the groups of natural and semi-natural ecosystems have now been re-classified and ordered according the EUNIS habitat classification for Europe (Davies and Moss, 2002). For a general description of the EUNIS classification and an introduction to its use with respect to nitrogen critical loads, see the separate background document of Hall *et al.* (2003). In general, the ecosystems described in this document have been classified down to level 2 or 3 of the EUNIS hierarchy and the EUNIS-code is given in the text and tables in brackets, e.g. raised and blanket bogs **(D1)**. Because of the use of the EUNIS classification, some ecosystems have been treated in different sections to those used in Bobbink *et al.* (1996). Specifically:

- soft water lakes and permanent dune pools are treated in a separate Chapter 7 (inland surface water habitats; C);
- the short items on dune grasslands, dune heaths and slacks are now described in a separate Chapter 8 on coastal habitats (B);
- salt marshes are discussed in a short section (Chapter 9) on marine habitats (A).

Generally, a good agreement exists between the vegetation classification used by Bobbink *et al.* (1996) and the EUNIS classification which is now adopted, although in some cases the EUNIS classification is somewhat incomplete (e.g. no gradient from soft to hard water in class C). However, the EUNIS classification of grasslands and tall forb habitats (E) has had considerable consequences for the structure of the grassland chapter. An overview of the old and new classification has been given in Table 2.1 to facilitate the shift to this newly used EUNIS classification.

Furthermore, it has to be mentioned that for forest ecosystems the text has been structured as in the 1996 background document, and thus not structured as in EUNIS. However, in the introduction of Chapter 3 the classification used in this document has been compared with the forest classification of EUNIS (forests and woodland habitats; G). As before, studies based on pure plantation stands are not accepted in the forest section, because the nitrogen critical loads of these intensively used systems are obtained via the steady-state mass balance method (Mapping Manual, 1996). Lichens and bryophytes have been, as before, incorporated in the sections of the ecosystems where they are found as an integral part of their biodiversity.

Table 2.1. Cross-comparison between the ecosystem classification used in the 2002 empirical N critical load setting (according to the EUNIS system) and the classification previously used (Bobbink *et al.*, 1996); with n.d. = not distinguished.

Ecosystem classification 2002	EUNIS	Ecosystem classification 1996
Heathland, scrub and tundra habitats	**F**	**Heathlands**
Tundra	F1	n.d.
Arctic, alpine and subalpine scrub	F2	Arctic and Alpine heaths
Northern wet heaths		
•'U' *Calluna* dominated wet heath	F4.11	Upland *Calluna* heath
•'L' *Erica tetralix* dominated wet heath	F4.11	Lowland wet heathlands
Dry Heaths	F4.2	Lowland dry heathlands
Grasslands and tall forb habitats	**E**	**Species-rich grassland**
Sub-atlantic semi-dry calcareous grasslands	E1.26	Calcareous grasslands
Non-mediterranean dry acid and neutral closed grasslands	E1.7	Species-rich heaths and neutral acidic grasslands (partly)
Inland dune pioneer grasslands	E1.94	n.d.
Inland dune siliceous grasslands	E1.95	n.d.
Low and medium altitude hay meadows	E2.2	Neutral-acid grasslands (partly)
Mountain hay meadows	E2.3	Montane-subalpine grasslands
Moist and wet oligotrophic grasslands	E3.5	Neutral-acid grasslands (partly), Mesotrophic fens (partly)
• *Molinia caerulea* meadows	E3.51	n.d.
• Heath (*Juncus*) meadows and humid (*Nardus stricta*) swards	E3.52	n.d.
Alpine and subalpine grasslands	E4.3 and E4.4	Montane-subalpine grasslands (partly)
Moss and lichen dominated mountain summits	E4.2	n.d.
Mire, bog and fen habitats	**D**	**Wetlands**
Raised and blanket bogs	D1	Ombrotrophic bogs
Poor fens	D2.2	n.d.
Rich fens	D4.1	Mesotrophic fens
Montane rich fens	D4.2	n.d.
Inland surface water habitats	**C**	**Wetlands**
Permanent oligotrophic waters	C1.1	n.d
• Softwater lakes	C1.1	Shallow softwater bodies
• Dune slack pools	C1.16	n.d.
Coastal habitats	**B**	n.d
Shifting coastal dunes	B1.3	n.d.
Coastal stable dune grasslands	B1.4	Neutral-acid grasslands (partly)
Coastal dune heaths	B1.5	n.d.
Moist to wet dune slacks	B1.8	n.d
Marine habitats	**A**	n.d.
Pioneer and low-mid salt marshes	A2.64 and A2.65	n.d.

Ranges and reliability

As in the 1992 and 1996 documents, the empirical nitrogen critical loads have been established within a range for each ecosystem class, because of: (i) real intra-ecosystem variation between different regions and sites where an ecosystem has been investigated; (ii) the intervals between experimental additions of nitrogen; and (iii) uncertainties in presented total atmospheric deposition values, although the latter have been checked by local specialists on atmospheric nitrogen deposition. In chapter 10 some additional information has been given on how to interprete this range in specific situations for an ecosystem. For every group of ecosystems (Chapter 3-9), the updated empirical nitrogen critical loads are, as before, summarized in a table, together with an indication of exceedance and of their reliability.

The reliability of the presented nitrogen critical loads figures is indicated as in the 1996 document (Bobbink *et al.,* 1996):

- reliable ##: when a number of published papers of various studies show comparable results;
- quite reliable #: when the results of some studies are comparable;
- expert judgement (#): when no data are available for this type of ecosystem. The nitrogen critical load is then based upon expert judgement and knowledge of ecosystems, which are likely to be more or less comparable with this ecosystem.

3. Effects of nitrogen inputs in forest habitats (EUNIS class G)

3.1 Introduction

Empirical critical loads were established for trees (deciduous and coniferous), for ground flora in forest stands, and for epiphytic lichens and algae, on acidic and on calcareous soils by Bobbink *et al.* (1996). The introduction of the EUNIS classification emphasizes the dominant tree species, soil hydrology and management practices more than soil chemistry. Furthermore, according to the review of new data, the differentiation into calcareous and acidic forests seems not to be necessary except for acidity related processes, as a range of changes such as effects on growth, nutrition, parasite infestation and changes in ground flora have been found at similar nitrogen loads. Critical load values can be set for EUNIS class G1 (broad-leafed woodland), and G3 (coniferous woodland), with the latter being divided into boreal and temperate, but often without clear differentiation between these classes and subgroups. When the critical load is the same for broad-leafed and coniferous woodland, values can also be applied to G4 (mixed deciduous and coniferous woodland). Moreover, a differentiation according to hydrology is not possible because of missing results for fluvial, riparian or swamp woodland. There are no studies available for G2 (broad-leafed evergreen). Highly artificial forest plantations (G1.C, G1.D, G3.F and G4.F) are excluded from setting empirical critical loads.

Forest ecosystems consist of different compartments which may be affected differently by increased nitrogen deposition. The soil may be acidified or eutrophied, both processes having consequences for microbiology, soil fauna and vegetation. Nutrition and growth of trees are changed by acidification and/or eutrophication. This may alter their resistance to abiotic and biotic stress factors. The species composition of the ground vegetation and of the mycorrhizal fungi may also be affected, with nitrogen demanding or acid tolerant species increasing and other species decreasing. Thus, the biodiversity may be altered by nitrogen deposition.

3.2 Effects on soil processes

The soil plays an important role in mediating nitrogen effects to the whole forest ecosystem. The following processes are important:

a) soil acidification, due to nitrification of ammonium and to leaching of nitrate. This process leads to accelerated leaching of base cations and, in poorly buffered soils, to increased dissolution of aluminium, which can damage fine root development and mycorrhiza, and thus reduce nutrient uptake (Ulrich, 1983; Ritter, 1990).

b) eutrophication. A surplus of nitrate, originating from deposition or enhanced nitrification due to accumulated nitrogen in the soil, will lead to eutrophication. Field-based ^{15}N studies demonstrate that a large proportion of incoming nitrogen (11-56%) is retained in the forest floor through biotic and abiotic processes within the first two years (Tietema *et al.*, 1998; Emmet *et al.*, 1998). An accumulation of ammonium at the ion exchanger may happen in areas with high NH_4^+-deposition (Roelofs *et al.*, 1985; Van Dijk and Roelofs, 1988; Schulze *et al.*, 1989; Boxman *et al.*, 1991), although ammonium is usually not detectable in soil solution as shown by the data from the ICP forest plots (De Vries *et al.*, 2003).

3.2.1 Soil acidification

Soil acidification per se is only briefly reviewed in this document, because critical loads for acidity, set on the basis of base cation to aluminium ratio and tree growth, are well established (e.g. Nilsson and Grennfelt, 1988; Sverdrup and Warfvinge, 1993). However, the significance of nitrogen compounds in acidification is increasing in parallel to sulphur reduction. In Western Europe, their contribution increased from 53% (1990) to 72% (1999). Regions with high acid loads have a high proportion of nitrogen and the significance of nitrogen in them is increasing (all data from Tarrason *et al.*, 2001).

3.2.2 Mineralization, nitrification, NO_3^--leaching, NH_4^+ accumulation

Nitrogen mineralization and nitrification may both be stimulated by nitrogen deposition. In a field study in 600 deciduous forests in four geographically separate regions of southern Sweden, nitrogen mineralization and nitrification were by far the highest in the region with the highest nitrogen deposition (17 kg N ha^{-1} yr^{-1}), especially in the most acid soils. Soil nitrogen mineralization rate was increased by 40-80%, nitrification rate increased by 20-90% and C:N ratio decreased by 10-25% compared to the region with 7-10 kg N ha^{-1} yr^{-1} (Falkengren-Grerup *et al.*, 1998; Falkengren-Grerup and Diekmann, 2003). Differences were also found between 7 and 10 kg N ha^{-1} yr^{-1} (Fig. 3.1). The effects depend on the deposition history. NITREX experiments, e.g., did not find effects on mineralization (Emmet, 1999), and McNulty *et al.* (1996) observed the highest nitrogen mineralization potential in control plots of a spruce-fir stand at high elevation but the net nitrification potential was highest in nitrogen fertilized plots (15.7 kg N ha^{-1} yr^{-1}).

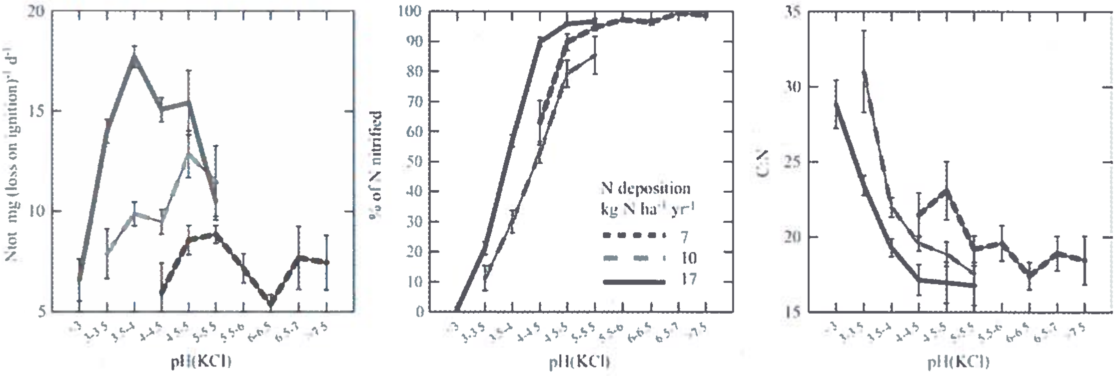

Figure 3.1. Nitrogen mineralization, nitrification and C:N ratio in 10 pH classes and three regions with varying nitrogen deposition. Means ± SE. From Falkengren-Grerup and Diekmann, 2003.

Moreover, organic forms of nitrogen, i.e. free amino acids, serve as important plant nitrogen sources in boreal forests (Näsholm *et al.*, 1998; Nordin *et al.*, 2001) and nitrogen deposition can disrupt the ratio between organic and mineral nitrogen supply in such soils.

In acid forest soils which have little or no nitrification, ammonium is accumulating (Van Dijk and Roelofs, 1988). As a reliable critical load, 10-15 kg NH_4^+-N ha^{-1} yr^{-1} has been proposed to prevent critical NH_4^+ accumulation and nutrient imbalances caused by high ratios between NH_4^+ and K, Ca or Mg (Roelofs *et al.*, 1985; Boxman *et al.*, 1988) although usually no NH_4^+ can be detected in the soil solution (De Vries *et al.*, 2003). However, in a large number of cases, NH_4^+ is nitrified and may leach together with deposited NO_3^- from the soil. Various investigations in deciduous and coniferous stands demonstrate that if nitrogen loads in throughfall exceed >12-15 kg N ha^{-1} yr^{-1}, NO_3^- leaching increases exponentially (Westling, 1991; Kölling and Neustifter, 1997; Gundersen *et al.*, 1998a; Nilsson *et al.*, 1998). NO_3^- leaching in coniferous stands is higher at sites with low C/N ratio in the organic layer (Dise *et al.*, 1998; Gundersen *et al.*, 1998a; De Vries *et al.*, 2001; 2002). In broad-leafed stands, the relationship between soil solution NO_3^- concentration and throughfall is steeper than in conifer stands (Fig. 3.2) indicating that NO_3^- concentration in soil solution under deciduous forests will respond much more to deposition changes than under conifers. The difference in relationship between soil solution nitrate and throughfall nitrogen between broad-leafed and conifers could be due to a higher potential nitrogen retention capacity in the forest floor or in the conifer trees because of a higher potential growth rate. The former has to be avoided because of subsequent negative effects on nutrition, and the latter will only work as long as nothing else becomes limiting to conifer growth. Therefore, the different relationship does not necessarily imply a differentiation between nitrogen critical loads for broad-leafed and coniferous forests but it has consequences for setting the reliability of the critical load.

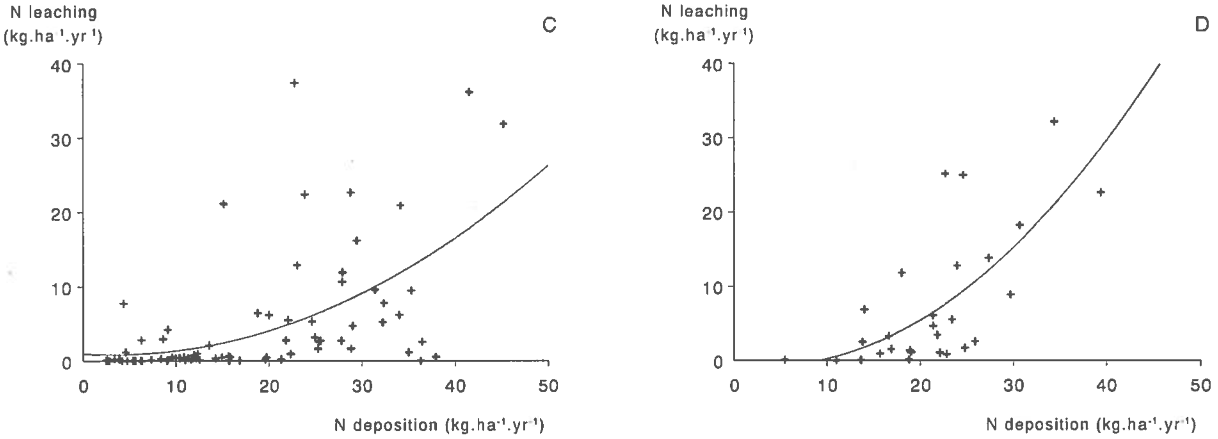

Figure 3.2. Scatter plots of the leaching of total nitrogen against the total nitrogen deposition for conifers (C) and deciduous trees (D) with a polynomial fit through the data (data from ICP Forests, De Vries *et al.*, 2003).

3.2.3 Litter decomposition

The effect of increased nitrogen deposition on litter decomposition seems to depend on the lignin concentration of the substrate (Carreiro *et al.*, 2000) and on the decomposition stage (Magill and Aber, 1998). The activity of phenol oxidases which decompose lignin is decreased by nitrogen while cellulase activity is stimulated. Thus, the decomposition of lignin-rich litter and of the late decomposition stages with lignin enrichment is delayed. Significant effects of experimental NH_4NO_3 application on litter decomposition of dogwood (*Cornus florida*) and oak (*Quercus rubra*) were found at nitrogen applications of 20 kg N ha^{-1} yr^{-1}, with an atmospheric deposition of 10 kg N ha^{-1} yr^{-1}. Mean litter residence time of oak litter was increased from 3.4 years to 4.0

years (20 kg N ha^{-1} yr^{-1}) and 4.5 years (80 kg N ha^{-1} yr^{-1}), respectively (Carreiro *et al*., 2000). In contrast, Gundersen (1998) found no response of decomposition rates of spruce litter to an addition of 25 kg N ha^{-1} yr^{-1} (atmospheric nitrogen deposition 15-20 kg). The observed responses may lead to increased accumulation of soil organic matter in stands with a higher nitrogen deposition (Magill and Aber, 1998), causing a fixation of base cations in this organic matter and thus enhancing the risk of soil acidification. Decomposition is going on until the remaining organic matter reaches a limit value at which total mass loss virtually stops. At such a stage the remaining litter is close to that of stabilized soil organic matter. In 106 long-term studies on litter decomposition, enclosing 21 litter types, limit values were significantly negatively related to nitrogen concentrations. The higher the nitrogen concentration in the litter (i.e. the lower the C/N ratio), the more organic matter was left when the litter reached its limit value (Berg and Meentemeyer, 2002).

In summary, the available data on soil processes suggest a critical load of **10-15 kg N ha^{-1} yr^{-1}** for **mineralization** and **nitrification** (**quite reliable**), for **nitrate leaching** in **coniferous forests** (**reliable**) and for **nitrate leaching** in **broad-leafed forests** (**expert judgement**). For litter decomposition, the exisitng data did not allow to define a separate critical load.

3.3 Effects of nitrogen deposition on growth, nutrition, physiology and parasite attacks of trees

The growth of a vast majority of the forest tree species in the Northern hemisphere was until recently limited by nitrogen (Tamm, 1991). An increase in the supply of any essential nutrient, including nitrogen, will stimulate tree growth as long as growth is not limited by other factors; the initial impact of enhanced nitrogen deposition will, therefore, most often be a fertilizer effect. However, chronic nitrogen deposition may result in 'nitrogen saturation', when enhanced nitrogen inputs no longer stimulate tree growth, but start to disrupt ecosystem structure and function (Agren, 1983; Aber *et al*., 1989; Tamm, 1991).

3.3.1 Growth

Growth of above-ground biomass

In temperate European regions, forest growth has increased in the past 40-50 years (Spiecker *et al*., 1996). Recently a pan-European research project concluded that the increased nitrogen deposition is the main cause for the observed increases in forest growth (EFI, 2002). There are many publications showing that nitrogen fertilization increased growth or had no effect (e.g. Alcubilla *et al*., 1976; Spiecker, 1991; Nilsson and Wiklund, 1992; Emmet, 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. The largest growth dataset comes from southern Norway, where data from increment cores of >31,000 spruce forest plots were analysed for the time period 1954-1996. Increases in growth during 1960-1970, followed by a subsequent decline in the 1980-1990, were observed in the plots with modelled wet nitrogen deposition >15-25 kg N ha^{-1} yr^{-1}. A growth decrease started in plots with wet nitrogen deposition of 7-15 kg N ha^{-1} yr^{-1} in the 1990s. No change was observed in plots with wet nitrogen deposition <7 kg N ha^{-1} yr^{-1} (Fig. 3.3; Nellemann and Thomsen, 2001). However, nitrogen deposition might be higher as dry nitrogen deposition was not included in the model. Also, an additional acidification effect of codepositing sulfur cannot be excluded. This pattern of initial growth stimulation, followed by subsequent growth decline, was also observed in a nitrogen addition experiment in a young beech stand in Switzerland on calcareous soil with a modelled atmospheric deposition of 16 kg N ha^{-1} yr^{-1}. Five years after the start of the experiment, shoot growth was significantly increased by addition of 10 kg N ha^{-1} yr^{-1}, not by higher nitrogen loads. After 8 years, there was still a growth stimulation at 10 kg, but a significant growth decrease at 160 kg N ha^{-1} yr^{-1}. In the 9th year, growth stimulation no longer occurred, but a significant growth decrease was found at 80 and 160 kg N ha^{-1} yr^{-1}. On acidic soil, a growth decrease was only observed at 160 kg N ha^{-1} yr^{-1} (Braun and Flückiger, 2002). It was suggested that the reason for a changed response to nitrogen addition is the limitation of other nutrients such as phosphorus or base cations (Flückiger and Braun, 1999b).

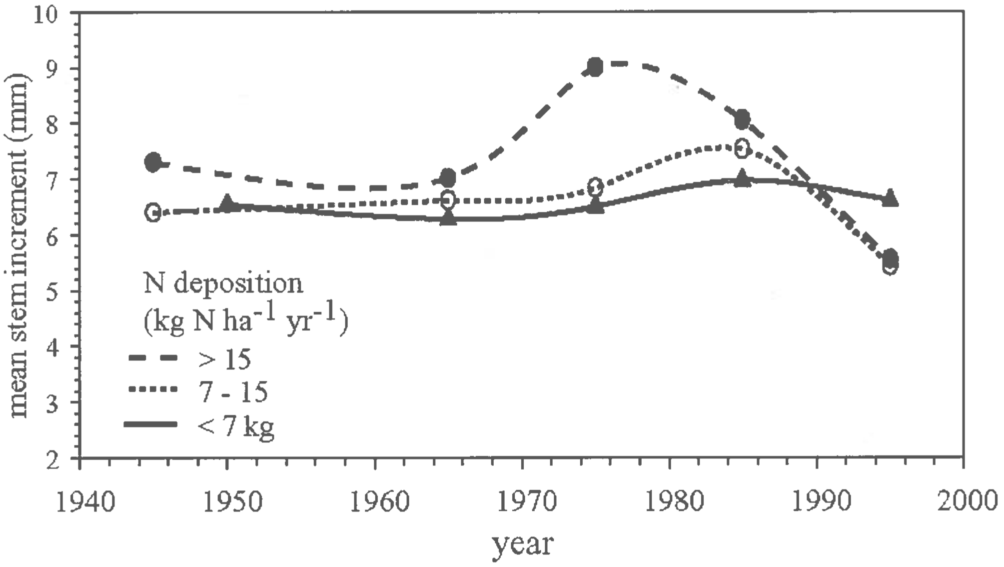

Figure 3.3. Stem increment of spruce in Norway from 31,606 increment cores grouped according to modelled wet nitrogen deposition. Growth increase in the highest deposition class as well as the decrease in the two highest classes are significant at $p<0.01$ (Nellemann and Thomsen, 2001).

A growth reduction by nitrogen was also shown in the NITREX experimental plot in the Netherlands where ambient nitrogen deposition was reduced from 56 to 4 kg N ha^{-1} yr^{-1}; trees in the roof clean condition grew better than in the roof control (Boxman *et al.*, 1998). In southern Sweden, a positive relationship between nitrogen concentration and branch growth of Norway spruce was found in trees which were younger than 40 years, whereas older trees seemed to be limited by K and/or phosphorus (Thelin, 2000).

Root growth

Nitrogen may also stimulate root growth (Zöttl, 1964; Heinsdorf and Schulzke, 1969; Persson, 1980) in ecosystems which are not nitrogen saturated. However, some studies showed that root growth is inhibited by excessive nitrogen supply, and there is strong evidence for increased nitrogen deposition causing reduced fine root biomass and root length. Boxman *et al.* (1995), Murach and Parth (1999) and Persson and Ahlström (2002) reported increasing root biomass and root vitality in Scots pine, Douglas fir and Norway spruce respectively when trees were protected from nitrogen deposition in the NITREX roof experiments. The corresponding treatments were a reduction of nitrogen deposition from 56 to 4 kg N ha^{-1} yr^{-1} in Ysselsteyn (Scots pine) (Gundersen *et al.*, 1998b), from 36 to <5 kg N ha^{-1} yr^{-1} in Solling (Douglas fir, Norway spruce) and from 13 to <4 kg N ha^{-1} yr^{-1} in Gårdsjön (Norway spruce). Total fine root biomass of Norway spruce saplings decreased significantly when NO_3^- + NH_4^+ in soil water was >2 mg N l^{-1} (Fig. 3.4; Matzner and Murach, 1995). From the relationship between nitrogen deposition and NO_3^- concentration published by De Vries *et al.* (2001), this concentration can be attributed in coniferous stands to an average nitrogen throughfall load of 25 (range 13-33) kg N ha^{-1} yr^{-1}.

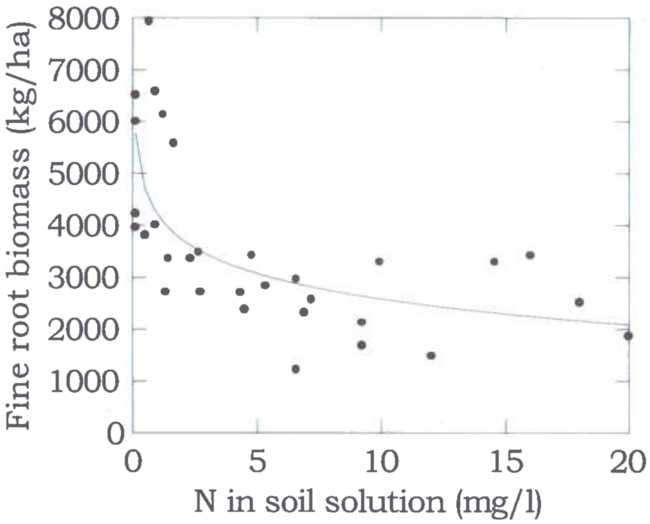

Figure 3.4. Fine root biomass of spruce in relation to nitrogen in soil solution (Matzner and Murach, 1995).

3.3.2 Nutrition of trees

Increased nitrogen deposition may change the nutrition of trees by increasing the nitrogen concentration in the foliage. Usually, P concentrations and - depending on the soil - also K and/or Mg concentrations decrease simultaneously. Thus, the ratios between nitrogen on the one hand and P, K and Mg on the other hand tend to increase. These changes have been shown in both field observations and experimental nitrogen additions. A decrease of P and Mg concentration in tree foliage may be caused by reduced uptake as a result of Al toxicity and hence soil acidification (Haynes, 1982; Godbold, 1991), as a result of competition with NH_4^+ in the soil (Roelofs *et al.*, 1985), as a result of nitrogen impacts on mycorrhizal fungi (see section 3.4), or by a decreased supply rate of nutrients other than nitrogen due to leaching and nitrogen induced high growth rates depleting the resource (Thelin *et al.*, 1998).

Rising nitrogen concentrations in the needles of various conifer species in Europe were reported in recent decades (Van den Burg, 1990; Sauter, 1991). Nitrogen concentrations in needles of Scots pine from the ICP forest plots (De Vries *et al.*, 2003) and of Norway spruce from Switzerland (Flückiger and Braun, 1998) were found to be correlated with nitrogen deposition. Based on a critical level of nitrogen concentration of 18 mg N g^{-1} in foliage, a critical nitrogen load for pine of 14 kg N ha^{-1} yr^{-1} and near 20 kg N ha^{-1} yr^{-1} for spruce was suggested by De Vries *et al.* (2003). In northeastern France, foliar mineral concentrations in 118 beech stands were compared between 1969-1971 and 1996-1997. Between the two sampling periods, foliar nitrogen concentrations increased by 12% whereas a decrease was observed for P (-23%), K (-6%), Mg (-38%) and Ca (-16%). The N/P ratio increased by 42%, N/K by 19% and N/Mg by 77%. The positive nitrogen trend did not depend on soil type, with similar trends on calcareous or acid soils. The atmospheric nitrogen deposition was estimated to be between 20-30 kg N ha^{-1} yr^{-1} (Duquesnay *et al.*, 2000). In Switzerland, foliar P concentration in beech (49 plots) decreased between 1984 and 1999 by 26%, leading to an increase in N/P ratios from 18.1 (1984) to 25.4 (1999) (+40%), in N/K ratios from 2.58 to 3.66 (+42%) and of N/Mg ratios from 16.3 to 18.3 (+12%) – as in the French study independently from the soil type. At the same time, P concentrations in Norway spruce (18 plots) dropped by 15%, with N/P ratios increasing from 14.0 to 16.3. Modelled nitrogen deposition was in the range of 15-35 kg N ha^{-1} yr^{-1} (Flückiger and Braun, 1998). In pine forests in Brandenburg and in spruce forests in the Mittelgebirge (Germany), nitrogen concentration in foliage increased, and Mg decreased, significantly within 25 years, between 1964 and 1988 (approx. nitrogen deposition of 15-25 kg N ha^{-1} yr^{-1}) (Nebe, 1991; Hippeli and Branse, 1992). In Sweden, addition of nitrogen enhanced nutritional imbalances in a 120 year old *Fagus* forest (Balsberg-Påhlsson, 1992). Mohren *et al.* (1986) and Houdijk and Roelofs (1993) found significantly lower phosphorus supply, or phosphorus deficiency, in conifers in forests exposed to raised nitrogen deposition in the Netherlands. Nihlgård and Olsson (1993) also report an increase of the N/P ratio in conifer needles during recent decades in South Sweden. There appeared to be a positive relationship between nutrient imbalances (increasing N/K and N/Cu) and deposition of nitrogen and sulphur (Thelin *et al.*, 1998). Phosphorus and nitrogen are well known to be limiting nutrients in forests (Johnson and Taylor, 1989). However, the increasing nitrogen deposition during the last decades could have aggravated phosphorus deficiency in forest ecosystems low in phosphorus supply (Tamm, 1991).

Experimental nitrogen addition to saplings of beech and Norway spruce in young stands on both acidic and calcareous soils induced nutrient imbalances and deficiencies, signficant 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 (Fig. 3.5) whereas on calcareous soil K and P became limiting (Flückiger and Braun, 1999a). 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).

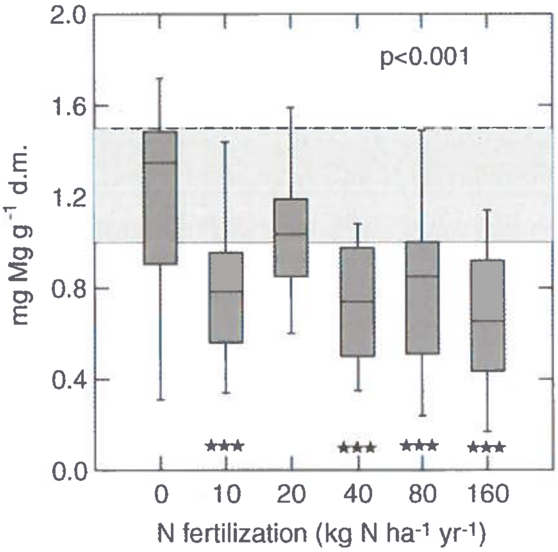

Figure 3.5. Magnesium concentration in beech leaves in a nitrogen fertilization experiment on acid soil. Significant differences to control are indicated with *** $p<0.001$, overall linear regression $p<0.001$ (Flükkiger and Braun, 1999b). Grey field: range for optimum nutrient concentration after Stefan *et al.* (1997).

Insight into the possible impact of nitrogen on foliar nutrition can also be gained from results from 109 level II plots of ICP Forests with information on both the chemical composition of the foliage and the total nitrogen input by deposition. In plots with "adequate ratios" of K, Ca, Mg or P to N, the median nitrogen deposition was 9.6 kg N ha^{-1} yr^{-1}, whereas in plots with unbalanced nutrition it was 21 kg N ha^{-1} yr^{-1} (De Vries *et al.*, 2003, Tab. 3.1). The results, however, depend on the critical nutrient ratios used for the evaluation (Flückiger and Braun, 2003).

Table 3.1. Ranges in nitrogen deposition (kg N ha^{-1} yr^{-1}) at 109 Intensive Monitoring plots of ICP Forests in Europe with a balanced and unbalanced ratio of the macronutrients P, K, Ca or Mg to N. (De Vries *et al.*, 2003).

	Unbalanced nutrition				**Balanced nutrition**			
Element	Nr of plots	*Median*	*5% Quantil*	*95% Quantil*	Nr of plots	*Median*	*5% Quantil*	*95% Quantil*
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

With high nitrogen inputs, the concentrations of organic nitrogen in needles can increase to high levels above the optimum range (Van Dijk and Roelofs, 1988; De Kam *et al.*, 1991). Nitrogen-rich free amino acids, especially arginine, significantly increased in the needles with high nitrogen concentration (Hällgren and Näsholm, 1988; Van Dijk and Roelofs, 1988; Näsholm and Ericsson, 1990; Pietilä *et al.*, 1991; Balsberg-Påhlsson, 1992). Arginine concentrations in foliage are suggested to be a sensitive indicator for nitrogen input (Edfast *et al.*, 1990; Huhn and Schulz, 1996). In Sweden, arginine concentrations of coniferous foliage >5 μmol g dw^{-1} has been linked to forest stands susceptible to NO_3^- leaching (Näsholm *et al.*, 1997), because arginine accumulation in trees corresponds to decreased uptake rates of NH_4^+, leaving NH_4^+ ions available for nitrification followed by NO_3^- leaching (Näsholm *et al.*, 1997). In permanent observation plots in Sweden, arginine concentrations in Norway spruce were higher in areas receiving high inputs of nitrogen (Ericsson *et al.*, 1995). In Swiss plots, arginine concentrations in spruce foliage were strongly correlated with modelled nitrogen deposition in the range of 14-37 kg N ha^{-1} yr^{-1} (Quiring *et al.*, 1997). By decreasing nitrogen deposition in the NITREX roof experiment, arginine concentrations in needles of Scots pine significantly decreased (Boxman and Van Dijk, 1994; Boxman *et al.*, 1995). Similarily, halted nitrogen additions resulted in a rapid decrease in arginine in Scots pine growing in central Sweden (Edfast *et al.*, 1996) and northern Sweden (Quist *et al.*, 1999).

Considering all demonstrated effects of nitrogen deposition on the **nutritional status** of trees, a **reliable** critical load for nitrogen for both **deciduous** and **coniferous temperate forests** of **15-20 kg N ha^{-1} yr^{-1}** can be recommended. Calcareous and acidic soils may differ in the type of nutrient imbalance but not fundamentally in the sensitivity of the response.

3.3.3 Tree physiology

Winter injury

Winter injury may be caused either by low temperatures or by drought stress. Whereas sensitivity to low temperatures is usually decreased by nitrogen addition (DeHayes *et al.*, 1989; Klein *et al.*, 1989; L'Hirondelle *et al.*, 1992), sensitivity to frost droughts seems to be increased. For example, long term nitrogen fertilization (0, 15.7, 19.8, 25.6, 31.4 kg N ha^{-1} yr^{-1}, atmospheric bulk precipitation 5.4 kg N ha^{-1} yr^{-1}) from 1988 to 1995/96 in the eastern US increased winter injury on montane red spruce (*Picea rubens*) foliage significantly at ≥15.7 kg N ha^{-1} yr^{-1}, although cold tolerance was not affected and dehardening was decreased by the nitrogen treatment (Perkins *et al.*, 2000). Moreover, in field fertilization experiments it is often observed that tree growth starts earlier in the season, which may increase damage by late frost.

Drought tolerance

The effects of nitrogen on fine roots of trees and on mycorrhiza may have consequences for drought tolerance. Indeed, the vitality of many tree species in the regions with high nitrogen deposition in the Netherlands was particularly low in the dry years in the mid 1980s, but vitality increased again in the following normal years (Heij *et al.*, 1991). 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 at ≥ 25 kg N ha^{-1} yr^{-1} and the leaf surface per tree at ≥ 50 kg N ha^{-1} yr^{-1} 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 at ≥ 100 kg N ha^{-1} yr^{-1}). It also led to increased leaf necroses after a drought period (significant at ≥ 200 kg N ha^{-1} yr^{-1}) (Flückiger and Braun, 1999b). Planted beech saplings in another nitrogen fertilization experiment showed significantly increased leaf necroses after short drought periods in the second and third treatment year at ≥20 kg N ha^{-1} yr^{-1} (Fig. 3.6; Thomas *et al.*, 2002).

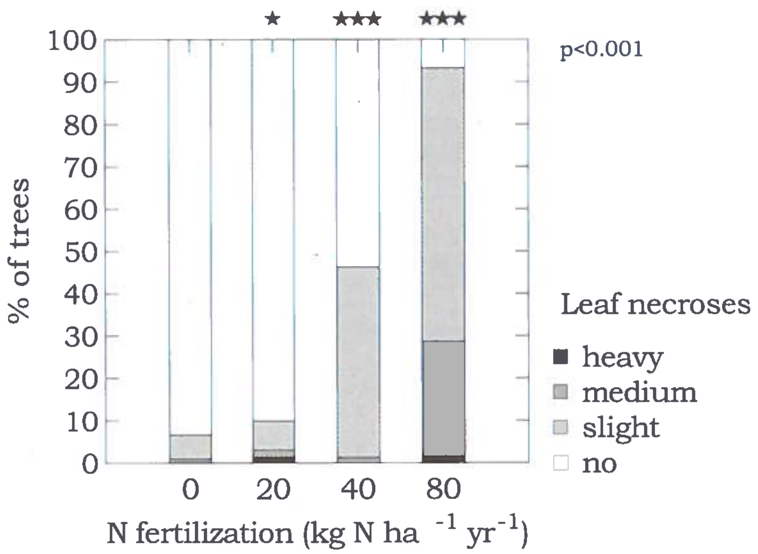

Figure 3.6. Leaf necroses in beech caused by drought in a nitrogen addition experiment. Differences to control significant at *p<0.05, ***p<0.001, general linear trend p<0.001 (Thomas *et al.*, 2002).

Root physiology, roots and soil properties

In a field fertilization experiment in Switzerland, starch concentration in fine roots of beech was significantly decreased at ≥ 20 kg N ha^{-1} yr^{-1} after 8 years of nitrogen application (Hiltbrunner *et al.*, 2001). Uprooting of mature beech in forest observation plots in Switzerland by the storm 'Lothar' in 1999 was positively correlated with nitrogen concentration in the foliage (atmospheric N-deposition 15-35 kg N ha^{-1} yr^{-1}), and with soils showing a base saturation ≤ 40% (Braun *et al.*, 2003).

Mycorrhizal fungi are crucial for nutrient and water uptake of forest trees and for resistance of roots against pathogens. Their species diversity, the formation of extramatrical mycelium and the production of fruitbodies are affected by nitrogen deposition. These effects are discussed in section 3.4.1.

3.3.4 Parasites

With increasing nitrogen deposition, the susceptibility to fungal pathogens and attacks by herbivores can change. This is probably due to altered concentrations of phenolic compounds and soluble nitrogen compounds such as free amino acids (Huber, 1980; McClure, 1980; Bolsinger and Flückiger, 1989; Balsberg-Påhlsson, 1992). A fertilization experiment with only 10 kg N ha^{-1} yr^{-1} in two sites (atmospheric deposition 12 and 20 kg N ha^{-1} yr^{-1}) was enough to alter the concentrations of fungistatic phenolic compounds in fine roots of young beech and spruce after seven years of nitrogen treatment, with most of them decreasing (Tomova *et al.*, submitted). In the same experiment, two fungistatic phenolic compounds in the leaves were negatively correlated with the N/P ratio in leaves. The total amount of phenolic compounds in *Fagus* leaves in a 120-year stand in southern Sweden decreased by more than 30% after fertilization with ca. 45 kg N ha^{-1} yr^{-1} over 4 years, compared with the control (Balsberg-Påhlsson, 1992).

Fungal pathogens

In the Netherlands, an epidemic outbreak of the pathogenic fungi *Brunchorstia pineae* and *Sphaeropsis sapinea* on *Pinus nigra* in coniferous forests was observed in the 1980s especially in the south-eastern part of the Netherlands with high levels of air-borne nitrogen deposition (Roelofs *et al.*, 1985). Attacked trees in the infested stand had significantly higher foliar nitrogen concentrations, occurring together with higher soil ammonium levels, than uninfected trees. Most of the additional nitrogen in the needles of the affected stands was stored as free amino acids, especially arginine, but proline concentrations were also increased in the infected trees, indicating an enhanced degree of water stress (Van Dijk *et al.*, 1992). Both high nitrogen supply and water stress increased the susceptibility of trees to attacks from *Sphaeropsis sapinea* (Blodgett *et al.*, 1997). High potassium supply at the same time reduced the development of necroses (De Kam *et al.*, 1991).

In permanent beech observation plots in Switzerland (modelled nitrogen deposition 15-35 kg N ha^{-1} yr^{-1}), as well as in nitrogen fertilization experiments, a significant positive correlation was found between the nitrogen/potassium ratio in leaves and the necroses caused by the beech cancer *Nectria ditissima* (Flückiger *et al.*, 1986; Flückiger and Braun, 1998). Beech bark lesions caused by the fungal pathogen *Nectria coccinea* var. *faginata* were found at 25 of 48 studied sites in Scania (South Sweden) and were more frequent at sites with higher nitrogen deposition (20-25 kg N ha^{-1} yr^{-1}) compared to sites with lower nitrogen deposition (15-20 kg N ha^{-1} yr^{-1}) (Westling *et al.*, 1992). Bark lesions were positively correlated to the occurrence of the beech scale *Cryptococcus fagisuga* and algal cover of *Lecanara fagisuga* (Jönsson, 1998).

In two nitrogen fertilization experiments with 0, 10, 20, 40, 80 and 160 kg N ha^{-1} yr^{-1} on acid and on calcareous soils, respectively, the damage of young beech by the pathogenic fungi *Apiognomonia errabunda* and *Phomopsis* sp., was significantly increased at treatments $\geq$10 kg N ha^{-1} yr^{-1} (atmospheric nitrogen deposition 20 and 16 kg N ha^{-1} yr^{-1}, respectively). There was also a strong positive correlation of the extent of twig necroses with N/P and N/K ratios in leaves (Flückiger and Braun, 1999a). The effect was similar on acidic and on calcareous soils.

Effects of nitrogen on the pathogen and insect infestation have also been observed in understorey shrubs. They are treated in chapter 3.4.3.

Insect pests

Most insects which have been found to react to nitrogen fertilization feed by sucking. The infestation of beech aphid *Phyllaphis fagi* in a nitrogen fertilization experiment increased significantly with increasing nitrogen concentration in leaves and N/P ratios respectively (Flückiger and Braun, 1998). The attack of beech nuts in Swiss permanent observation plots by the tortricid *Cydia amplana* also increased significantly with increasing N/P ratio in leaves (N-deposition 15-60 kg N ha^{-1} yr^{-1}) (Braun and Flückiger, 2002).

The occurrence of insect damage to pine needles in permanent observation plots in the UK was found to be positively correlated with modelled nitrogen deposition (range 7-22 kg N ha^{-1} yr^{-1}), but only within Scotland itself. This relationship was associated with a negative relationship between the years of needles retained and modelled nitrogen deposition (NEGTAP, 2001).

Whereas the better performance of the insects listed above may be explained by a better nutrition with soluble nitrogen compounds, the situation is more complicated in the case of the red-black pine bug *Haematoloma dorsatum*, originating from the Mediterranean region, which causes severe needle damage to pine stands in the Netherlands. The nymphs of the bugs suck on the roots or basal stem parts of *Deschampsia flexuosa* exclusively. Only the adults cause damage in the trees. *D. flexuosa* seems to be crucial for nymphal development, because it is a wintergreen grass and hence suitable for sucking in wintertime and early spring. The abundance of this grass in pine forests is thereby an important ecological factor for pest development of *H. dorsatum* (Moraal, 1996). It has been shown by various authors that the abundance of *D. (Avenella) flexuosa* increases significantly when nitrogen deposition is >10-15 kg N ha^{-1} yr^{-1} (see section 3.4).

Considering the **detrimental effects of pathogens and insect pests** upon tree health with increasing nitrogen deposition, a critical load for nitrogen of **15-20 kg N ha^{-1} yr^{-1}** is recommended (**expert judgement**). However, there is not enough data available to differentiate between forest types. No data are available for boreal forests; thus no separate critical load can be defined.

3.4 Effects of nitrogen deposition on biodiversity of forests

3.4.1 Effects on macrofungi and mycorrhiza

Nitrogen deposition can influence fruit body formation by ectomycorrhizal fungi, the production and distribution of the extramatrical mycelium in the soil and the formation of ectomycorrhiza. Available data from long-term nitrogen deposition studies indicate that the most prominent effects might be those which are discernible above-ground (i.e. on the formation of fruit bodies) (Wallenda and Kottke, 1998). 'Generalist' species, forming a symbiosis with a wide range of tree species, seem to be less affected by increased nitrogen availability than 'specialist' species. *Laccaria, Paxillus, Theleophora, Scleroderma* and *Lactarius* are examples of the less sensitive group whereas *Tricholoma, Cortinarius* and *Suillus* are more sensitive (Arnolds, 1991). Thus, *Paxillus involutus*, *Lactarius rufus*, *Laccaria bicolor* increased formation of fruitbodies after nitrogen fertilization with up to 240 kg N ha^{-1} yr^{-1} (Hora, 1959; Laiho, 1970; Ohenoja, 1988) while 35 kg N ha^{-1} yr^{-1} was sufficient to reduce fruit body production of most mycorrhizal species, especially *Cortinarius* spp. and *Russula* spp. (Brandrud, 1995). The easily cultivated species which are used in culture experiments are adapted to higher nitrogen concentrations which makes it difficult to derive a critical load from this type of experiment (Wallenda and Kottke, 1998).

There are numerous reports of decreases in species diversity and abundance of mycorrhizal fungi in forests. In the Netherlands, the average number of ectomycorrhizal species declined significantly between 1912-1954 and 1973-1983 from 71 to 38, while saprophytic and parasitic fungi colonizing wood increased from 38 to 50 (Arnolds, 1985; 1991). Similar observations were made by Rücker and Peer (1988) in forests of the Salzburg region (Austria). Data collected in 1937 showed 110 and in 1987 48 species of ectomycorrhizal fungi, while the species number of saprophytes and parasites colonizing wood increased from 17 to 19. Grosse-Branckmann and Grosse-

Branckmann (1978) compared the occurrence of sporocarps in the Darmstadt area of Germany between 1970 and 1976 with the data collected between 1918 and 1942. From the 236 species found in the first period only 137 were found in the second period, which corresponds to a loss of 99 species including many ectomycorrhizal fungi. Termorshuizen and Schaffers (1987) found a negative correlation between the total nitrogen input in mature *Pinus sylvestris* stands in the Netherlands and the abundance of fruit bodies of ectomycorrhizal fungi. Schlechte (1986) compared two *Picea abies* sites in the Göttingen area. He found a negative relationship between nitrogen deposition and ectomycorrhizal species: at the site with nitrogen deposition of 23 kg N ha^{-1} yr^{-1}, 85 basidiomycetes including 21 ectomycorrhiza (25%) were found, at the site with 42 kg N ha^{-1} yr^{-1} 55 basidiomycetes including 3 ectomycorrhiza (5%). In a gradient study from <1 to up to 18 kg N ha^{-1} yr^{-1} bulk deposition in the neighbourhood of an industrial ammonia production facility which had operated for almost 30 years, sporocarps of 14 mycorrhizal fungi species were found at the six plots with highest nitrogen load compared to 144 mycorrhizal species at the six plots with lowest nitrogen load (Lilleskov *et al.*, 2001). However, there is no information about the importance of dry deposition at the investigated site, and thus it is difficult to use these data to derive a critical load.

Reduced fruit body production of mycorrhizal fungi was also observed in experiments. Rühling and Tyler (1991) found that within 3 to 4 years after application of 60 and 180 kg N ha^{-1} yr^{-1} as NH_4NO_3 (atmospheric deposition 15-20 kg N ha^{-1} yr^{-1}) almost all mycorrhizal species ceased fruitbody production. Brandrud (1995) observed in a *Picea abies* stand in the Swedish NITREX site a decrease in fruit body production of mycorrhizal species at a nitrogen application of 35 kg N ha^{-1} yr^{-1} (NH_4NO_3) within 1.5 years. Termorshuizen (1990) applied 0, 30 and 60 kg N ha^{-1} yr^{-1} to young *Pinus sylvestris* stands as $(NH_4)_2SO_4$ or NH_4NO_3. In general, fruit body production was more negatively influenced by the higher ammonium level than by the ammonium-nitrate mixture. The mycorrhizal frequency and the number of mycorrhiza per unit of soil volume were not affected. It was concluded that fruit body production is much more sensitive to nitrogen enrichment than mycorrhizal formation. Effects on fruit body formation may be long lasting even though the nitrogen deposition is reduced or terminated. In an old forest fertilization experiment in northern Sweden the fruit body production of nitrogen-sensitive mycorrhizal fungi such as *Russula* species was still lower in formerly fertilized plots than in controls nearly 50 years after the nitrogen additions was terminated (Strengbom *et al.*, 2001).

The effect of nitrogen additions on the belowground ectomycorrhizal (EM) community may be less dramatic than that on fruit bodies (Wallenda and Kottke, 1998) and several studies stress the lack of correspondence between the abundance of EM fruit bodies and EM root tips (Dahlberg, 2001; Lilleskov *et al.*, 2002). The effect of reduced diversity and reduced external EM biomass may both have a substantial effect on nutrient uptake.

Several studies report responses in belowground EM communities after nitrogen addition, especially in long-term experiments. In a pot experiment with seedlings of *Pinus sylvestris* and added mycelium of *Paxillus involutus* or *Suillus bovinus* the total number of mycorrhiza per plant dry weight decreased significantly with 50 kg N ha^{-1} yr^{-1} ($(NH_4)_2SO_4$ or $NaNO_3$) (Termorshuizen, 1990). Erland and Taylor (2001) used a gradient of low deposition in northern Europe to higher deposition in southern Europe. There were no apparent negative effects on EM fungal diversity in beech forests. EM root tips in spruce forests appeared to be more sensitive to high levels of nitrogen deposition and the diversity of root morphotypes decreased from north to south. Interestingly, the proportion of species that can take up organic nitrogen declined as mineral nitrogen availability increased. Lilleskov *et al.* (2002) found a similar reduction in belowground richness of EM species with increasing nitrogen deposition in a gradient in Alaska. The authors hypothesize that N-efficient species under nitrogen limiting conditions are followed by species with best function in nutrient rich soils and thereafter by P-efficient species under high nitrogen conditions. In two similar 60 year old *Picea abies* forests in south Sweden with different rates of nitrogen deposition (Vedby with 14-15 kg N ha^{-1} yr^{-1} and Skrylle with 24-29 kg N ha^{-1} yr^{-1}), the level of mycorrhizal colonization was almost 100%, but the total number of mycorrhiza was 30-42% higher at the low N-deposition site. Data show 13 ectomycorrhizal taxa at Vedby and 7 at Skrylle. The total number of mycorrhizal roots was also significantly lower at Skrylle than at Vedby. Six morphotypes were distinguished at Vedby and four at Skrylle (Erland *et al.*, 1999). Wöllecke *et al.* (1999) investigated the number of mycorrhizal morphotypes in a stand with low nitrogen pollution (bulk deposition 1985-1988 10-20, 1996 7.6 kg N ha^{-1} yr^{-1}, average NH_3 concentration during 1996-1998 0.5 $\mu g/m^3$) and with high nitrogen pollution (1985-1988 >35 kg, 1996 13.5 kg N ha^{-1} yr^{-1}, NH_3 concentration 1996-1998 10.3 $\mu g/m^3$). They found 18 mycorrhizal morphotypes in the low nitrogen site and 9 mycorrhizal morphotypes and low mycorrhizal frequency at the high nitrogen site.

Considering the observed nitrogen effects on **mycorrhizal fungi**, a critical nitrogen load of **10-20 kg N ha^{-1} yr^{-1}** is recommended (**expert judgment**).

3.4.2 Effects on ground-living and epiphytic lichens and algae

In the Netherlands, the forest vegetation of one Scots pine stand in the central part of the country with a deposition of approx. 20 kg N ha^{-1} yr^{-1} was investigated in 1958 and then re-investigated in 1981 when the deposition was approx. 40 kg N ha^{-1} yr^{-1}. During this period, all lichens disappeared (Dirkse and Van Dobben, 1989). This could be also a sulphur effect, but results from fertilization experiments in northern Sweden with low deposition of both nitrogen and sulphur shows that all *Cladina* species had disappeared following 28 years of nitrogen addition (34 kg N ha^{-1} yr^{-1}), while they were still present in the unfertilized control plots (Strengbom *et al.*, 2001). This indicates that nitrogen deposition alone may be responsible for observed reductions in the abundance of ground-living lichens. Bråkenhielm and Quinghong (1995) found along a geographical gradient with 14 plots in Sweden a significantly increased weighted mean sensitivity (WMS) of lichens from south to north and from west to east. WMS drops with >6-8 kg N ha^{-1} yr^{-1}.

In Atlantic oakwoods in Scotland and NW England, epiphytic lichens such as *Lobaria pulmonaria* and *L. amplissima* have been shown by Mitchell *et al.* (2003) to be effective indicators for nitrogen pollution. Comparing sites with low nitrogen deposition (9.6-17.6 kg N ha^{-1} yr^{-1}) and high nitrogen deposition (11.2-53 kg N ha^{-1} yr^{-1}), these epiphytes were only found at low nitrogen sites, suggesting a critical load of 11-18 kg N ha^{-1} yr^{-1}.

Ca. 10 percent of all lichen species in the world have blue green algae as the photobiont (Insarova *et al.*. 1992). These blue-green algae lichens are negatively affected by acidity as well as by nitrogen. In an international survey from the Netherlands via Denmark to Sweden, the decline of these lichens with blue-green algae correlated significantly with nitrogen deposition rates above 5-10 kg N ha^{-1} yr^{-1} (Göransson, 1990). However, there is still little information about the effect of nitrogen on blue-green algae lichens which may be the most sensitive component of some forest ecosystems and thus determine the critical load for these systems (Hallingbäck, 1991). The negative effects on lichens may instead be an indirect effect from competition with nitrogen-favoured vascular plants (Cornelissen *et al.*, 2001). Dahlmann *et al.* (2002) found that among about 500 lichens which both have green algae and cyanobacterias as their photobiont most species seem to be rather resistant towards even high nitrogen additions.

Green algae, especially such of the genus *Pleurococcus* (syn. *Protococcus*, *Desmococcus*) are strongly stimulated by enhanced nitrogen deposition. They cover outdoor surfaces which are not subject to frequent desiccation in regions with high nitrogen deposition, above 10-15 kg N ha^{-1} yr^{-1} (Bobbink *et al.*, 1996). Results from the Swedish Environment Monitoring Programme and from Norway show that these algae, mainly *Pleurococcus viridis*, do not colonize spruce needles in regions with a throughfall nitrogen deposition lower than 5 kg N ha^{-1} yr^{-1} (Bråkenhielm, 1990; Thomsen, 1992). However, the results may be confounded by sulfur and temperature. In a 10 year monitoring period in Sweden, patterns for sensitive lichens were opposite to those observed for algae. WMS (weighted mean sensitivity) of lichens and algal colony thickness (15 areas) were positively correlated with growing season, nitrogen and sulphur deposition. The colonization rate demonstrated a clear trend along a geographical gradient decreasing from south to north and from west to east, showing an increasing algal cover with nitrogen deposition >5-8 kg N ha^{-1} yr^{-1} (Bråkenhielm and Quinghong, 1995). Poikolainen *et al.* (1998) investigated the abundance of the epiphytic lichen *Scoliciosporum chlorococcum* on conifers on 3009 permanent Finnish forest plots between 1985 and 1995. There was an increasing abundance of this lichen where nitrogen deposition reached ≥3-4 kg N ha^{-1} yr^{-1}. However, climate factors might be involved as well.

The data for this **freeliving algae and epiphytic lichens** suggest a **reliable** critical load of **10-15 kg N ha^{-1} yr^{-1}**. All data presented refer to **boreal** forests, and there is no information availabe to set the critical load for temperate forests.

3.4.3 Effects on forest ground vegetation

There is a large number of observations showing an increase in abundance of nitrophilous species in forests, either as time series or within a nitrogen deposition gradient. These species include *Galeopsis tetrahit*, *Rubus idaeus*, *R. fruticosus*, *Deschampsia flexuosa*, *Molinia caerulea*, *Poa trivialis*, *Milium effusum*, *Urtica dioica*, *Epilobium angustifolium*, *Galium aparine*, *Aegopodium podagraria*, *Sambucus nigra*, *Stellaria media*, *S. nemorum*, *S. holostea*, *Dryopteris filix mas*, *D. dilatata* and *D. carthusiana*. In parallel, the average N indicator number (after Ellenberg, 1988) has been shown to rise. A lot of endangered species are plants with a low N indicator value (Ellenberg, 1985).

A large representative evaluation of species occurrence at 2000 sampling sites covering ca. 90% of the Dutch forests revealed in the mid 1980s that, with a nitrogen deposition of approx. 40 kg N ha^{-1} yr^{-1}, nitrophilous species such as *Galeopsis tetrahit*, *Rubus* spp, *Deschampsia flexuosa*, *Dryopteris carthusiana*, *Molinia caerulea*, *Poa trivialis* and *Urtica dioica* were among the 40 most common plants in forest ecosystems (Dirkse and Van Dobben, 1989; Dirkse, 1993). 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 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 (1986; 1995) 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}.

The influence of atmospheric nitrogen deposition on mixed oak forest vegetation along a deposition gradient from Skåne (14-20 kg N ha^{-1} yr^{-1}) via eastern Småland and Öland (7-11 kg N ha^{-1} yr^{-1}) to lake Mälaren (6-9 kg N ha^{-1} yr^{-1}) was investigated by Brunet *et al.* (1998). 20 of 30 field layer species that are most closely associated with high nitrogen deposition, increased in frequency in areas with high nitrogen deposition during recent decades. This group of field layer species includes many species generally considered as nitrophilous but also several acid tolerant species. A site index calculated on Ellenberg N-values was higher in the most exposed regions except on soils with pH <3.5, where soil acidity was probably more important, and with pH >5.0 (Falkengren-Grerup and Diekmann, 2003). The number of species was about 20% lower in the most exposed regions within the same soil pH interval. The change in vegetation between 1983 and 1993 was large enough to give significantly higher Ellenberg N-values in 1993 in both Skåne and Småland (Diekmann *et al.*, 1999).

In Germany, too, an increasing abundance of nitrophilous species (such as *Dryopteris dilatata, D. carthusiana, Rubus idaeus, R. fruticosus, Milium effusum, Deschampsia (Avenella) flexuosa, Urtica dioica, Epilobium angustifolium*) in mixed fir/spruce and in Scots pine forests is reported, where nitrogen deposition is approx. 15-30 kg N ha^{-1} yr^{-1} (Rodenkirchen, 1992; Kraft *et al.*, 2000). However, changes in management practice may also affect these changes. In forest sites in Germany with earlier litter raking, the average nitrogen indicator values after Ellenberg (Ellenberg, 1988) increased by 0.6 units, in forest sites without litter raking by 0.3 units (Rodenkirchen, 1992). In Switzerland, a significantly increased abundance of nitrophilous species in 17 of 18 forest sites was observed in two regions (northern Switzerland, modelled N-deposition for 1995 20-30 kg N ha^{-1} yr^{-1}, and Geneva region, nitrogen deposition 15-20 kg N ha^{-1} yr^{-1}) between 1938/1942/1947 and 1984/1985 (Kuhn *et al.*, 1987). At 37 forest sites in the central plateau of Switzerland, the comparison between two surveys of the ground vegetation between 1940/1965 and 1998 revealed a decreased frequency of 241 species and an increased frequency of 44 species, some of them typical nitrophilous species such as *Rubus fruticosus, R. caesius, Dryopteris dilatata, D. filix mas, Sambucus nigra, Hedera helix* and *Urtica dioica*. The nitrogen deposition in this region is approx. 30-40 kg N ha^{-1} yr^{-1} (Walther and Grundmann, 2001). In Belgium also, Lameire *et al.* (2000) found a significant increase of nitrogen indicator species in mixed deciduous forests during the past 20 years - between 1977/80 and 1997/98 - such as *Dryopteris dilatata, Galeopsis tetrahit, Hedera helix, Moehringia trinervia, Sambucus nigra, Stellaria media* and *Urtica dioica* (approx. nitrogen deposition 25-30 kg N ha^{-1} yr^{-1}). In a gradient study, the species composition of ground flora along a 500 m transect from livestock buildings showed marked changes within 30 m downwind. *Deschampsia flexuosa, Holcus lanatus, Rubus idaeus* and *Urtica dioica* were abundant close to livestock units and their percentage cover decreased rapidly with distance from source. In this study, 15-20 kg N ha^{-1} yr^{-1} was proposed to protect the ground flora (Pitcairn *et al.*, 1998).

At two sites near Nancy (Haye forest) in northeastern France on calcareous soils with beech, oak and hornbeam, 278 floristic surveys from 1972 were reexamined in 1991 (139 at each site). A significant increase in nitrogen demanding and acidophilous species was observed. In site 1 (carré permanente), 100% of the plots showed an eutrophication and 79% an acidification, in site 2 (Villey), 85% of the plots showed an eutrophication and 74% an acidification (nitrogen deposition approx. 15-20 kg N ha^{-1} yr^{-1}) (Bost, 1991). In 1990, Thimonier *et al.* (1992) resampled at the intersections of a systematic grid covering a mixed hardwood forest in the Amance forest (Menthe et Moselle) in the Lorraine Plaine, northeastern France, 221 floristic plots sampled the first time in 1971 and 1972. Three main features were noted: a) an increase in the frequency of nitrogen-demanding species, b) an increase in Ellenberg's ecological N value and c) an enrichment of the site revealed by trophic level indices. Thinning does not appear to be responsible for the changes in the trophic level observed. While the N values increased from 4.89 to 5.16, the light values changed from 4.82 to 4.7. Annual nitrogen inputs in Amance forest were assessed to be 20-30 kg N ha^{-1} yr^{-1} in the beginning of the 1970s.

At 47 vegetation plots in the Villey forest near Nancy of the Lorraine Plain, northeast France, species were first sampled in 1972 and resampled in 1991. The mean number of species per sample increased from 32 to 39 during this period and the frequency of nitrophilous species increased throughout the forest. On acidic soils, acidophilous species increased in frequency. Repetition of soil analysis at the same time interval on nearby plots revealed a strong increase in nitrogen content. Although the stands have been converted from coppice to high forest, it is suggested that atmospheric nitrogen deposition is the most probable cause of eutrophication. Eutrophication was found equally on calcareous and on acidic soils (nitrogen deposition approx. 20 kg N ha^{-1} yr^{-1}) (Thimonier *et al.*, 1994).

Although *D. flexuosa* is not a typical nitrophilous species, it shows a positive reaction to nitrogen. After 5 years of treatment, 5 kg N ha^{-1} yr^{-1} was enough to increase the abundance of this grass species in a coniferous forest in Sweden (Fig. 3.7; Kellner and Redbo-Torstensson, 1995).

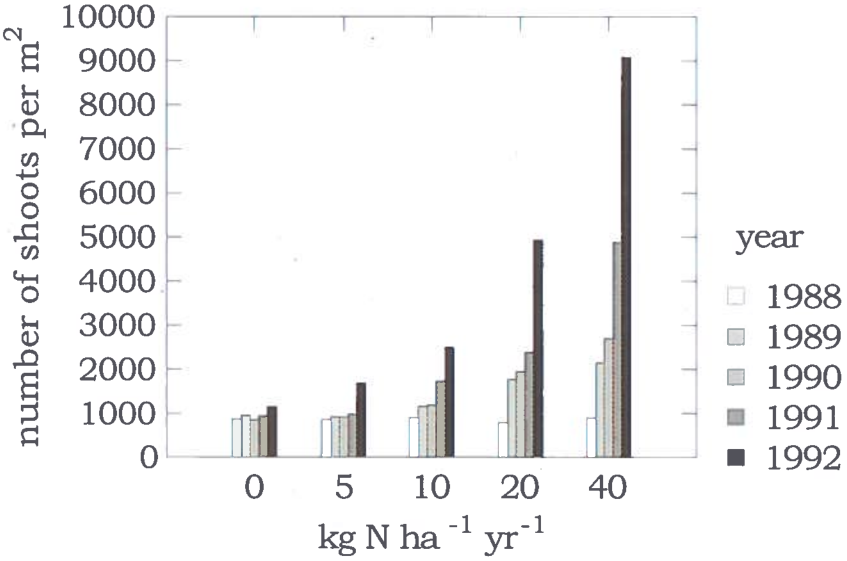

Figure 3.7. Number of shoots of *Deschampsia flexuosa* vs. nitrogen supply (kg N ha^{-1} yr^{-1}) from 1988 to 1992 in coniferous forest ground vegetation in Söderhamm, central Sweden (Kellner and Redbo-Torstensson, 1995).

In a field survey in 557 coniferous stands in Sweden, the occurrence of *Vaccinium myrtillus, V. vitis-idaea* and *Deschampsia flexuosa* was investigated. In areas with increased nitrogen deposition (≥ 6 kg N ha^{-1} yr^{-1}), *V. myrtillus* was less frequent and more susceptible to the leaf pathogen *Valdensia heterodoxa* than in areas with lower levels of nitrogen deposition. The occurrence of *V. vitis idaea* was also strongly negatively correlated with increasing nitrogen deposition but no such trend was observed for *D. flexuosa* (Strengbom *et al.*, 2003). These relationships were also shown experimentally. In a nitrogen fertilization experiment with 0, 12.5 and 50 kg N ha^{-1} yr^{-1} (atmospheric deposition 2-3 kg N ha^{-1} yr^{-1}), the abundance of *D. flexuosa* increased significantly after 3 years with 12.5 kg N ha^{-1} yr^{-1} while the abundance of *V. myrtillus* decreased.. Disease incidence by *Valdensia heterodoxa* was more than twice as high in plots receiving 12.5 kg N ha^{-1} yr^{-1} and more than three times as high in plots receiving 50 kg N ha^{-1} yr^{-1} compared to controls (Fig. 3.8). The abundance of the mildew fungus *Podosphaera myrtillina* on *Vaccinium* was increased ten-fold by 25 kg N ha^{-1} yr^{-1} compared to 0.5 kg N ha^{-1} yr^{-1}. As a consequence, *Vaccinium* density decreased, and *Deschampsia* cover increased (Strengbom *et al.*, 2002). In addition, shoots of *Vaccinium myrtillis* were significantly more damaged by larvae such as the moth *Orgyia antrana* after addition of 12.5 kg N ha^{-1} yr^{-1} in the first year of treatment (Nordin *et al.*, 1998). On the other hand, nitrogen fertilization with 30 kg N ha^{-1} yr^{-1} (atmospheric nitrogen deposition 12 kg N ha^{-1} yr^{-1}) did not change the ground vegetation in a montane forest within three years (Schleppi *et al.*, 1999).

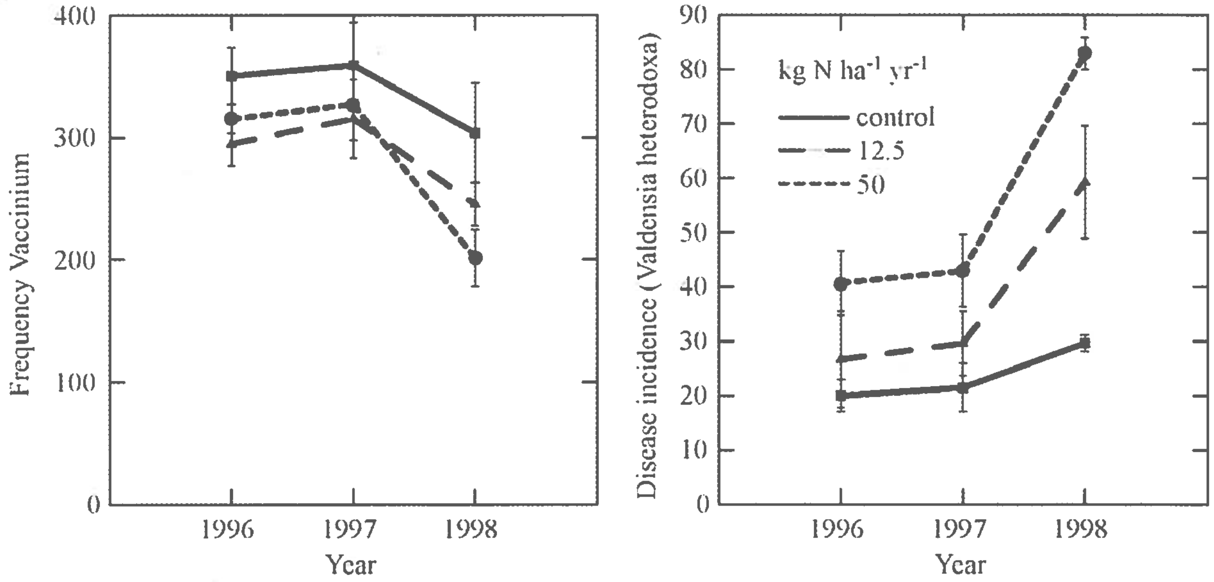

Figure 3.8. Development of *Vaccinium myrtillus* density (left) and of disease incidence on *Vaccinium* by *Valdensia heterodoxa* (right) in a nitrogen addition experiment in a boreal forest in North Sweden (Strengbom *et al.*, 2002).

Understorey species in deciduous forests respond to nitrogen deposition by higher growth rate, a more broad-leafed anatomy and higher nitrogen concentration and nitrate reductase activity in the tissue (Falkengren-Grerup and Diekmann, 2003).

Bryophytes depend directly on wet and dry deposition of nitrogen and are therefore considered to be highly sensitive even to small changes in the supply. Addition of nitrogen doses between 12.5 and 50 kg N ha^{-1} to coniferous forest in northern Sweden caused arginine concentrations of *Pleurozium schreberi* and *Dicranum majus* to increase (Nordin *et al.*, 1998). This indicates that the mosses were not able to respond to nitrogen additions by increased growth, and instead nitrogen was accumulated in the form of arginine. High amino acid concentrations may be harmful to bryophytes, and correlated to reductions in length growth of *Sphagnum* (Nordin and Gunnarsson, 2000). Some bryophytes are very sensitive to nitrogen deposition, while others are not. Effects of nitrogen on bryophytes were studied in a nitrogen field fertilization experiment in two southern Swedish Scots pine stands (Lisselbo and Norrliden). *Brachythecium oedipodium*, *B. reflexum* and *B. starkei* increased significantly at fertilization levels up to 60 kg N ha^{-1} yr^{-1}. At higher doses these species tended to decline. In contrast, *Hylocomium splendens* and *Pleurozium schreberi* declined strongly at doses of 30 to 60 kg N ha^{-1} yr^{-1} (Dirkse and Martakis, 1992). However, the biomass of the mosses *Pleurozium schreberi* and *Dicranum polysetum* was reduced by 60% and 78%, respectively, when the mosses were fertilized with only 25 and 30 kg N ha^{-1} yr^{-1}, respectively, after four years in a spruce stand in Finland with a low atmospheric deposition rate (Mäkipää, 1998). In two pine forest stands in central and northern Sweden (122 plots in total) which had been treated with N, P, K, lime and sulphuric acid for 15 years, nitrogen additions (20 and 60 kg N ha^{-1} yr^{-1}) had by far the strongest effect causing a shift in dominance from lichens and bryophytes such as *Pleurozium schreberi* and *Hylocomium splendens* and Ericaceae towards *Deschampsia flexuosa* and ruderal species (Van Dobben *et al.*, 1999). In an old fertilization experiment in northern Sweden the abundance of *H. splendens* was still lower in formerly nitrogen-treated plots nearly 50 years after termination of nitrogen additions, whereas nitrogen favoured bryophytes such as *B. reflexum* and *Plagiothecium denticulatum* were more abundant in formerly nitrogen-treated plots than in control plots. This shows that the negative effects on bryophyte communities may be long lasting although the nitrogen input is reduced and there are good opportunities for colonization from the surrounding non-affected vegetation (Strengbom *et al.*, 2001).

In an Atlantic oakwood stand in Scotland and NW England, the epiphytic brypophytes *Plagiochila atlantica* and *P. spinalosa* appear to be sensitive to nitrogen pollution. These epiphytic species were strongly associated with sheltered sites and low nitrogen deposition (9.6-17.6 kg N ha^{-1} yr^{-1}), compared to high nitrogen sites with 11.2-53 kg N ha^{-1} yr^{-1} (Mitchell *et al.*, 2003).

Most studies presented above suggest a **reliable** critical load of **10-15 kg N ha^{-1} yr^{-1}** for the protection of **ground flora** in **temperate forests**. The lack of low-dose experiments, uncertainties about background deposition and confounding factors in gradient studies do not allow to set a lower critical nitrogen load for **boreal forests** at the moment. Hence, the recommended critical load for boreal forests is set as an **expert judgment** to **10-15 kg N ha^{-1} yr^{-1}**.

3.4.4 Fauna

Soil fauna

The effect of enhanced nitrogen input on soil fauna is a serious gap in knowledge and needs more attention in further research. The abundance of *Nematoda, Oligochaeta* and microarthropodes, especially *Collembola*, had increased in some studies, but decreased in others after application of high doses of nitrogen fertilizers (>150 kg N ha^{-1} yr^{-1}) (Abrahamsen and Thompson, 1979; Huhta *et al.*, 1983; Vilkamaa and Huhta, 1986). A one-time application of 100 kg N ha^{-1} yr^{-1} (as NH_4NO_3) in the Fontères state forest in Eastern Brittany, France, produced an effect on soil microorganisms which was still significant after 23 years. There were decreases in *Oribatida, Camarida, Collembola, Symphyta* (small *Myriapoda*) and *Pseudoscorpionida*; the atmospheric deposition at this site was estimated as 10-20 kg N ha^{-1} yr^{-1} (Deleporte and Tillier, 1999). A reduction in the nitrogen deposition in a *Pinus sylvestris* stand (NITREX site Ysselsteyn) to preindustrial levels increased the species diversity of microarthropodes due to a decreased dominance of some species (Boxman *et al.*, 1995). A significant decrease by 66% of the abundance of earthworms was observed after seven years of fertilization in a young beech stand with 20 kg N ha^{-1} yr^{-1} (atmospheric deposition 12 kg N ha^{-1} yr^{-1}) (Flückiger and Braun, 1999b). Simultaneously, the pH of the upper soil layer (30 cm) decreased from 3.7 to 3.5. In Sweden, a significant decrease of snails over 14-46 years in areas with nitrogen deposition of 15-25 kg N ha^{-1} yr^{-1} was observed, while in areas with nitrogen deposition of 3-6 kg N ha^{-1} yr^{-1} no significant changes were found (Gärdenfors *et al.*, 1995). However, in the same area there was a sulfur deposition gradient (soil acidification) as well. Hence, no critical nitrogen load can be defined based on these data.

Mammals

Foliage of fertilized spruce seedlings (*Picea sitchensis*) was most palatable to Orkney voles, *Microtus arvalis orcadensis*. Foliage had higher levels of nitrogen (1.6-1.7% compared with 0.9% in control) and lower levels of phenolics (2.6%, control 5.1%). Whereas the fertilization of the spruce seedlings contained NPK and cannot be quantified in weight per unit area, a second experiment with *Calluna* fertilized with 52 kg N ha^{-1} yr^{-1} gave similar results (Hartley *et al.*, 1995). In southern Sweden, nitrogen concentrations and browsing of young Norway spruce was compared in two clear cut sites with different nitrogen deposition (Asa, N-deposition approx. 12-18 kg, and Tönnersjöheden, N-deposition approx. 18-29 kg N ha^{-1} yr^{-1}). At the site with higher nitrogen deposition (Tönnersjöheden), the seedlings had darker green needles and higher nitrogen concentrations in foliage. Frequency of browsed seedlings increased with increasing color and hence with increasing nitrogen concentration in the needles. Browsing damage was significantly higher at Tönnersjöheden compared to Asa. Vegetation cover in the clear cuts and density of roe deer around the clearcut were not different at the two sites (Bergquist and Örlander, 1998a; 1998b). However, the present data are not strong enough to set a critical nitrogen load for the fauna at the moment.

3.5 Summary of empirical critical loads for forests

3.5.1 Effects on soil processes

Considering the results presented in par. 3.2., the critical load for soil processes such as mineralization, nitrification, NH_4^+ accumulation, NO_3^- leaching can be set at 10-15 kg N ha^{-1} yr^{-1} for boreal and temperate forests.

3.5.2 Effects on trees

Growth: Growth effects to nitrogen may be positive, neutral or negative depending on the ecosystem status. Because nutrition changes will precede growth changes, no separate critical load is set for growth.

Nutrition: Long term observations in France, Sweden and Switzerland show a significant increase of N/P, N/K and/or N/Mg ratio in the foliage of beech and Norway spruce with nitrogen deposition between 20-35 kg N ha^{-1} yr^{-1}. Fertilization experiments also demonstrate a significant change in the N/P or N/Mg ratios when nitrogen fertilization is $\geq$10 kg N ha^{-1} yr^{-1} (atmospheric nitrogen deposition 12-20 kg N ha^{-1} yr^{-1}). Data from ICP Forests (De Vries *et al.*, 2003) show a median nitrogen deposition of 9.6 kg N ha^{-1} yr^{-1} for forest plots with a balanced nutrition and of 21 kg N ha^{-1} yr^{-1} for plots with an unbalanced nutrition. Hence, a nutrition based critical load of 15-20 kg N ha^{-1} yr^{-1} for coniferous and deciduous trees can be proposed and considered as quite reliable.

Physiology: The physiological data presented here are too weak to set a separate critical load for physiology.

Parasites: In a long-term fertilization study in young stands, beech showed significantly increased attacks by *Apiognomonia errabunda* and *Phomopsis* sp. with a nitrogen fertilization of ≥10 kg N ha^{-1} yr^{-1} (atmospheric deposition 20 and 16 kg N ha^{-1} yr^{-1}, respectively). A critical load of 15-20 kg N ha^{-1} yr^{-1} can be set as an expert judgment because the effect at the treatment of 10 kg N ha^{-1} yr^{-1} (+ 21 kg N ha^{-1} yr^{-1} atmospheric deposition) was quite strong. In 1996, no critical load was set on the basis of parasite sensitivity.

3.5.3 *Effects on mycorrhizal fungi and ground flora*

Mycorrhizal fungi

Strengbom *et al.* (2001) found large effects of a nitrogen fertilization with 34 kg N ha^{-1} yr^{-1} on sporocarp abundance even 9 years after termination of the treatment. This study was the basis to set a critical load of 10-20 kg N ha^{-1} yr^{-1} as an expert judgment. Also, the tendency of decreased growth of mycelia by ectomycorrhizal fungi in oak forest soils in a nitrogen deposition gradient (9-17 kg N ha^{-1} yr^{-1}) in southern Sweden (L.O. Nilsson, pers. comm.) support the proposed range. In 1996, the critical load set on the basis of effects on mycorrhiza (together with ground flora) was 7-20 kg N ha^{-1} yr^{-1} for coniferous and 10-20 kg N ha^{-1} yr^{-1} for deciduous forests.

Ground-living and epiphytic lichens and algae

The gradient studies by Göransson (1990), Bråkenhielm and Quinghong (1995) on epiphytic algae and of Poikolainen *et al.* (1998) on lichens which would support a low critical load have uncertainties on dry deposition and on climate and sulfur as confounding factors. Therefore, a critical load can be set to 10-15 kg N ha^{-1} a^{-1} as an expert judgment. This is higher than the critical load of 5-10 kg N ha^{-1} yr^{-1} set in 1996.

Ground vegetation

There is a large number of observations showing an increase in abundance of nitrophilous species either as time series or within a nitrogen deposition gradient. In boreal forests, changes of ground vegetation, e.g. decreasing *Vaccinium myrtillus*, occurred above ≥6 kg N ha^{-1} yr^{-1} (Strengbom *et al.*, 2003) and increased growth of *Deschampsia flexuosa* with ≥5 kg N ha^{-1} yr^{-1} (Kellner and Redbo-Torstensson, 1995). *Vaccinium myrtillus* showed an increased attack by parasites when nitrogen fertilization was ≥12.5 kg N ha^{-1} yr^{-1} (Strengbom *et al.*, 2002; Nordin *et al.*, 1998). Bryophyte species are also sensitive; the biomass of the mosses *Pleurozium schreberi* and *Dicranum polysetum* fertilized with 25 and 30 kg N ha^{-1} yr^{-1} respectively was reduced by 60% and 78% respectively after four years (Mäkipää, 1998). In temperate forests, many reports show an increasing frequency of nitrophilous species such as *Dryopteris filix-mas, Rubus idaeus, Epilobium angustifolium, Urtica dioica, Sambucus nigra* and others with nitrogen deposition >15-25 kg N ha^{-1} yr^{-1} in the long term. Hence, a critical nitrogen load on the basis of ground vegetation changes of 10-15 kg N ha^{-1} yr^{-1} is quite reliable. Together with effects on mycorrhiza, this critical load was set in 1996 to 7-20 kg N ha^{-1} yr^{-1} for acidic coniferous forests, to 10-20 kg N ha^{-1} yr^{-1} for acidic deciduous forests, to 15-20 kg N ha^{-1} yr^{-1} for calcareous forests and to 7-15 kg N ha^{-1} yr^{-1} for acidic unmanaged forests. Table 3.2. summarises these values. Given the range of responses found above these loads in field experiments and in field observations, we can consider **10-20 kg N ha^{-1} yr^{-1}** as overall empirical nitrogen critical load for both **temperate** and **boreal forests** as **quite reliable**.

Table 3.2. Empirical critical loads for the effects of excess nitrogen upon trees and different forest ecosystems. ## reliable; # quite reliable and (#) expert judgement.

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Forest habitats (G)				
Soil processes				
Deciduous & coniferous	-	10-15	#	Increased N mineralization, nitrification
Coniferous forests	-	10-15	##	Increased nitrate leaching
Deciduous forests	-	10-15	(#)	Increased nitrate leaching
Trees				
Deciduous & coniferous	-	15-20	#	Changed N/macro nutrients ratios, decreased P, K, Mg and increased N concentrations in foliar tissue
Temperate forests	-	15-20	(#)	Increased susceptibility to pathogens and pests, change in fungistatic phenolics
Mycorrhiza				
Temperate and boreal forests	-	10-20	(#)	Reduced sporocarp production, changed/reduced below-ground species composition
Ground vegetation				
Temperate and boreal forests	-	10-15	#	Changed species composition, increase of nitrophilous species, increased susceptibility to parasites
Lichens and algae				
Temperate and boreal forests	-	10-15	(#)	Increase of algae, decrease of lichens
Overall				
Temperate forests	-	10-20	#	Changes in soil processes, ground vegetation mycorrhiza and increased risk of nutrient imbalances and susceptibility to parasites
Boreal forests	-	10-20	#	Changes in soil processes, ground vegetation mycorrhiza and increased risk of nutrient imbalances and susceptibility to parasites, increase in free algae

4. Effects of nitrogen deposition on heathland, scrub and tundra habitats (EUNIS class F)

4.1 Introduction

Heathlands have for a long time played an important part in the West European landscape. Various types of plant communities have been described as heath, but the term is applied here to plant communities where the dominant life form is that of the small-leaved dwarf-shrubs forming a canopy of 1 m or less above soil surface. Grasses and forbs may form discontinuous strata, and there is frequently a ground layer of mosses or lichens (Gimingham *et al.*, 1979; De Smidt, 1979). Heathlands are classified together with scrub and tundra habitats in EUNIS class F. This category includes all dry or seasonally wet inland vegetation (cover > 30%) that is dominated by (dwarf-) shrubs or scrub (Davies and Moss, 2002). In part of these systems, the vegetation is determined by climate, and succession towards woodland is inhibited by drought, low temperature or length of the frost period (e.g., F1 and F2). In contrast, the extensive inland lowland dwarf-shrub heathlands in sub-Atlantic Europe are certainly man-made, although they have existed for several centuries. In these heaths the development towards woodland has been prevented by mowing, burning, sheep grazing or sod removal. They are widely dominated by some *Ericaceae*, especially *Calluna vulgaris* in the dry and *Erica tetralix* in the wet heathlands, or *Erica cinerea* in the western atlantic heathlands (Gimingham *et al.,* 1979). These communities are found on nutrient-poor mineral soils with a low pH (3.5-4.5), which makes them sensitive both to eutrophication and to acidification effects of increased nitrogen deposition. Because of their conservational importance, many lowland heathlands have become nature reserves in recent years.

In previous versions of the empirical critical load approach, dwarf-shrub heathlands were divided into five categories according to broad differences in habitat: (1) dry heathlands; (2) wet heathlands; (3) species-rich heaths and acidic grasslands; (4) upland *Calluna* moorland; and (5) arctic-alpine heathlands (Bobbink *et al.*, 1992a; 1996). Following the EUNIS habitat classification, in this document we distinguish the following classes. Tundra **(F1)** which is combined with arctic, alpine and subalpine scrub habitats **(F2)** and replaces (5) arctic-alpine heathlands. Temperate shrub habitats (**F4**) are divided into wet (**F4.1**) and dry (**F4.2**) heaths._Category F4.2 correspond to (1) dry heathlands in the former classification. In view of their functional differences, wet heaths are divided on a climatic basis into northern (**F4.11**) and southern (**F4.12**) wet heaths. For the latter, no data are available to assign a critical load. Northern wet heaths were assigned a different critical load if dominated by *Calluna vulgaris*, corresponding broadly to (4) upland *Calluna* moorlands in the previous division, and if dominated by *Erica tetralix*, which broadly correspond to the original (2) wet heathlands category. In contrast to previous versions, coastal dune heaths are treated in coastal habitats (Chapter 8; B1.5). The section (3) species-rich heaths and acidic grasslands is now (at least partly) included in the grassland section (Chapter 5; E1.7 & E3.5)). For other EUNIS categories in class F, including other heathland types, no data are available to determine an empirical nitrogen critical load.

4.2 Tundra (F1) and Arctic, alpine and subalpine scrub habitats (F2)

Alpine and arctic habitats have many ecological characteristics in common, although the climatic conditions are more severe in the arctic regions than in most alpine regions. The growing season is short, temperatures are low, winds are frequent and strong, and the distribution of plant communities depends on the distribution of snow during winter and spring. Most alpine and all arctic zones are influenced by frost activity or solifluction. Decomposition of organic matter and nutrient cycling are slow, and the low nutrient availability limits primary production (Robinson and Wookey, 1997). Despite the constraints, there are a number of plant species growing on tundra in the arctic and sub-arctic, including low shrubs, sedges, reindeer mosses and other lichens (crustose and foliose), bryophytes (mosses and liverworts), tussock grasses, and approximately 400 herb species.

In classifying these communities under the EUNIS system, it is necessary to distinguish between tundra (class F1) and arctic, alpine and subalpine scrub habitats (class F2). The key feature within EUNIS which distinguishes tundra is the presence of permafrost, which prevents root penetration and often keeps the ground waterlogged in summer. Hence, in assessing the need for critical load revision within the EUNIS system it is important to assign studies to these two distinct classes and to assess whether differentiated critical loads may be required.

4.2.1 Tundra (F1)

Significant increases in the atmospheric deposition of nitrogen in arctic regions have been reported during recent decades (Woodin, 1997). Plant growth and distribution in the Arctic are strongly constrained by the duration of the snow-free period, low air and soil temperatures, low soil moisture and low soil nutrient availability. Thus, anthropogenic sources of nitrogen may have direct and indirect effects on this ecosystem impacting on the local flora and fauna, and the economy and culture of indigenous populations (Press *et al.*, 1998a).

There have been numerous field manipulation studies with nutrients in tundra ecosystems; however, most have involved NPK fertilizer additions (e.g. Robinson *et al.*, 1998; Press *et al.*, 1998b; Schmidt *et al.*, 2000) or single large applications of nitrogen (e.g. Shaver and Chapin, 1995). Typical of the latter is the experiment of Henry *et al.* (1986), who examined the effect of a single nitrogen addition of 0, 50 and 250 kg ha^{-1} yr^{-1} to three Canadian tundra communities; wet sedge meadow, a mosaic *Cassiope tetragona* heath, and a dry-mesic *Dryas integrifolia* heath. Although a range of vegetation responses were observed in the third year after the addition of 250 kg N ha^{-1} yr^{-1}, no significant effects were observed after the application of 50 kg N ha^{-1} yr^{-1}.

In contrast, in a study of tundra ecosystems with a similar species complement, significant effects have been reported at much lower deposition rates. The experimental site was established in mixed tundra heath near Svalbard in 1991. Plots were located in each of three tundra heath vegetation types, which received factorial combinations of nitrogen (10 and 50 kg N ha^{-1} yr^{-1}) and phosphorus (5 kg P ha^{-1} yr^{-1}) as 4-5 applications in the summer period. Plots dominated by *Dryas octopetala* were treated from 1991 to 1998, those dominated by *Salix polaris* from 1991 to 1997 and those dominated by *Cassiope tetragona* from 1991 to 1993 only. Current atmospheric deposition was not measured, but is expected to be very low. Baddeley *et al.* (1994) reported early responses to the nitrogen treatments. *Salix polaris* had increased levels of foliar nitrogen, increased leaf biomass and increased photosynthetic rate. *Cassiope tetragona* showed no response to nitrogen addition, whilst *Dryas octopetala* showed an intermediate response of the measured variables.

More recently, Gordon *et al.* (2001) re-examined the impacts of nitrogen on these plots, with particular attention to the bryophyte communities. Overall bryophyte cover was unaffected by increased nitrogen supply, although this was a net result of individual species showing different responses. For example, *Polytrichum juniperinum* increased its cover whilst *Dicranum scoparium* cover was reduced. Tissue nitrogen content increased with increasing supply, demonstrating its close coupling with atmospheric inputs. Importantly, a number of significant persistent effects were observed at 10 kg N ha^{-1} yr^{-1}. Nitrate reductase activity was inhibited in *Polytrichum juniperinum*, suggesting nitrogen saturation, and the potential for inorganic nitrogen to pass through the bryophyte layer, thus becoming available for soil microbes and higher plants. The nitrogen addition also increased the proportion of green bryophyte shoots to a small extent, thus apparently increasing potential total bryophyte productivity. The increased 'greenness' of the bryophyte cover in the fertilised *Cassiope* heath plots was independent of phosphorus treatment and was found five years after nitrogen additions had ceased. This suggests that the added nitrogen is still being held within the bryophyte layer of the *Cassiop*e heath and implies that any reversal of the effect will be slow. This lack of recovery demonstrates the potential for long-term ecological change caused by even small amounts of nitrogen deposition (Gordon *et al.*, 2001).

Co-limitation by nitrogen and phosphorus was clearly demonstrated by this study. For example, with, but not without, added phosphorus, 10 kg N ha^{-1} yr^{-1} significantly decreased total lichen cover, increased the cover of *Polytrichum juniperinum* and changed vegetation composition in the *Dryas* plots. Hence critical loads for tundra ecosystems may be dependent on phosphorus availability and may be influenced by increased nutrient availability from global warming. Long-term responses of tundra vegetation to NPK fertiliser treatment show in most cases an effect of nutrient addition, with indication of both nitrogen and phosphorus limitation. Shaver *et al.* (1998) suggest that mechanistic analysis of the results of fertilization experiments can identify factors associated with either nitrogen or phosphorus limitation. They suggest that wet tundra sites are more likely to be phosphorus limited than moist sites with a thinner peat layer, while dry tundra deserts are primarily nitrogen limited. Cornelissen *et al.* (2001) examined relationships between macrolichen and vascular plant abundance across arctic vegetation ecosystem experiments which manipulated factors such as temperature and nutrient availability and concluded that negative correlations were greater at milder sites with a greater above-ground biomass, where increased shading and litter production is likely to negatively affect the lichens. Hence, it may be possible to identify those tundra communities for which low critical loads are required to prevent changes in species composition.

Summary

Despite the limited number of long-term experiments, a clear picture is now emerging of the potential impact of long-term nitrogen deposition on tundra ecosystems. Ecosystem response to nitrogen has been observed at deposition rates as low as 10 kg N ha^{-1} yr^{-1}. To a large extent, however, the response to atmospheric nitrogen within the tundra ecosystem may well depend on other factors, such as phosphorus status and temperature. Expert judgement was used by Bobbink *et al.* (1996), in the absence of empirical evidence, to propose a critical load for arctic and alpine heaths that was lower than that for lowland heathland and was set at 5-15 kg N ha^{-1} yr^{-1}. Since then, many more experimental manipulation studies have been reported but very few use nitrogen at deposition rates which are useful in defining critical loads.

Nevertheless, the tundra study showing significant effects of adding 10 kg N ha^{-1} yr^{-1} to a low current atmospheric deposition provides strong support for a lower critical load, especially since significant effects were observed rapidly and do not appear to be readily reversible. We therefore propose a critical load for **tundra (F1)** of **5-10 kg N ha^{-1} yr^{-1}**, as **quite reliable.** Although this classification as quite reliable is based on only one experiment, it can be justified because: (i) the experiment involved three different plant communities; (ii) background deposition was very low (<1 kg N ha^{-1} yr^{-1}); (iii) the N treatment (10 kg N ha^{-1} yr^{-1}) was within the critical load range; (iv) the effects were seen rapidly and are very persistent; and (v) subsequent unpublished results confirm the persistence of the effects.

Inclusion of a phosphorus treatment in this experiment produced unequivocal evidence of co-limitation by N and P, with greater responses to N when phosphorus was also applied. Phosphorus supply is thus identified as an important modifier of the critical load, with higher critical loads applying to systems, which are limited by P, and lower critical loads to systems which are not phosphorus limited.

4.2.2 Arctic, alpine and subalpine scrub habitats (F2)

In contrast to tundra, there is no new empirical evidence to support a revised critical load for arctic, alpine and subalpine scrub habitats, with most relevant studies involving either NPK fertilization or very high levels of nitrogen deposition (e.g. Theodose and Bowman, 1997). However, there are two Norwegian studies which have been reported since 1995 which are relevant.

A study in the Dovre mountains in Norway, at 1000-1400 m a.s.l., investigated the effects of three years fertilization of a *Betula nana* dominated community with 12 and 61 kg N ha^{-1} yr^{-1}, at a site with an estimated background deposition of 2-4 kg N ha^{-1} yr^{-1} (Paal *et al.*, 1996). There was no significant effect on plant growth, and no evidence of increased N contents in vegetation or soils. In a second study, two lichen-dominated communities were investigated, one low-alpine and the other middle-alpine. After three years of application of 7, 35 and 70 kg N ha^{-1} yr^{-1}, there was evidence of a negative response of the lichens *Cladonia coccifera*, *Cladonia mitis*, and *Cetraria nivalis* (Mols *et al.*, 2001). However, the statistical analysis presented in this paper does not allow the threshold deposition for effects to be identified. Some indication of a possible threshold is given by unpublished results for ten years of treatment of these plots, which suggest that significant negative effects on lichen cover are found at 35 kg N ha^{-1} yr^{-1}, but not at 7 kg N ha^{-1} yr^{-1}.

Thus, in contrast to tundra, there is no new evidence to support a change in the previous critical load for arctic and alpine heaths of 5-15 kg N ha^{-1} yr^{-1}, as recommended by Bobbink *et al.* (1996). Given the lack of supporting evidence for any change for alpine systems, we recommend a critical load of **5-15 kg N ha^{-1} yr^{-1}**, on the basis of **expert judgement** for **arctic, alpine and subalpine scrub habitats (F2)** .

4.3 Temperate shrub habitats: wet and dry heaths (F4.1 and F4.2)

As discussed in section 4.1, both wet and dry heathlands have to be placed within the class F4 (temperate shrub heathland), because of their climatic condition and their dominance by ericoid shrubs. The use of level 3 of the EUNIS system allows the class to be split into wet heaths (**F4.1**), which are damp and characterized by peat soils, and dry heaths (**F4.2**). Both upland *Calluna* moorlands and lowland wet heaths dominated by *Erica tetralix*, fall within the category of 'northern' wet heaths (class **F4.11**). However, since these communities are clearly ecologically different and assigned with different critical loads, it is important that this habitat distinction is retained. Since there is no altitude-based cut-off which can be recommended to distinguish the two habitats, the primary criterion must be based on species dominance. National experts will need therefore to provide advice to the mapping community on how to map the two habitats.

4.3.1 'U' Calluna-dominated wet heath (upland moorland) (F4.11)

It was suggested by Bobbink *et al.* (1996) that the *Calluna vulgaris* heaths of the upland areas of Britain and other mountainous parts of Europe are expected to be sensitive to nitrogen deposition. These acidic communities are characterized by a dominance of dwarf shrubs (in particular *C. vulgaris*), a high abundance of bryophyte species and peaty soils. Furthermore, the abiotic conditions are colder and wetter than in lowland heathlands (F4.2). The effects of nitrogen deposition on upland *C. vulgaris* heaths have been studied in the UK using both field surveys and experiments. The critical load of 10-20 kg N ha^{-1} yr^{-1} recommended by Bobbink *et al.* (1996) for this community was based on three types of evidence from these UK studies, which are considered in turn below in the light of more recent literature.

(a) Effects on growth and species composition

The longest *C. vulgaris* moorland nitrogen manipulation experiment is in North Wales where experimental plots were established in May 1989 on an area of moorland at an altitude of 470 m a.s.l.. This site has been estimated to receive a current atmospheric nitrogen deposition of approximately 20 kg N ha^{-1} yr^{-1}, although more recent assessments suggest that the deposition is somewhat higher. Additions of nitrogen in the form of ammonium nitrate at doses of 0, 40, 80 and 120 kg N ha^{-1} yr^{-1} have been applied since 1989 at monthly intervals and continue to date (Lee and Caporn, 1998; Carroll *et al.*, 1999). The period from 1989 to 1993 was characterized by apparently beneficial effects of nitrogen on *C. vulgaris* in terms of improvements in shoot growth, nitrogen concentration and flowering, with no indication that the dose applied exceeded the capacity of the plants for uptake and subsequent growth. The following three years of the study, however, showed a much reduced effect of the treatment on shoot extension, and no clear dose response to increasing nitrogen inputs. The 1996 data in particular showed no effect of nitrogen on shoot extension at all (Carroll *et al.*, 1999). One interpretation of the *C. vulgaris* growth responses in this experiment is that addition of N has accelerated the natural *C. vulgaris* cycle, with earlier ageing and opening of the canopy in the highest treatment plots. This is the only moorland study of such a duration and the results clearly demonstrate the need for caution in interpreting the growth increases commonly reported in response to nitrogen addition to *C. vulgaris* over periods of 1-5 years.

One factor causing increased canopy opening may be greater winter damage to *C. vulgaris* shoots. Detailed experimental studies of frost tolerance in *C. vulgaris* shoots collected in the early years of the study (1989-1994) demonstrated that nitrogen addition actually improved frost tolerance in autumn (Caporn *et al.*, 1994). However, field surveys in 1996 and 1998 clearly showed large increases in 'winter browning' of heather shoots, most notably in the 80 and 120 kg N ha^{-1} yr^{-1} treatments (Fig. 4.1) (Carroll *et al.*, 1999; Lee *et al.*, 2000). This damage mechanism may be the result of low temperature desiccation, rather than frost damage.

In the earlier years, both bryophytes and lichens had disappeared from below the *C. vulgaris* canopy in all nitrogen addition treatments, although *Vaccinium myrtillus* maintained its cover (Carroll *et al.*, 1999). It is not certain, however, whether this was a response to direct effects of nitrogen addition or through changes in *C. vulgaris* canopy architecture and increased litter production resulting in reduced light penetration. A later survey, after ten years of treatment, showed not only that bryophytes have returned to the nitrogen treated plots, but that their cover actually increased with increasing nitrogen deposition, probably as a result of the accelerated ageing and opening up of the *C. vulgaris* canopy (Lee *et al.*, 2000). However, no such response was found for lichen species. These results highlight the need to establish more precisely whether elevated nitrogen deposition is having a direct and/or indirect impact on bryophyte and lichen populations, especially as it has been suggested that nitrogen deposition is partly responsible for the decline of certain mosses and lichen species in upland *C. vulgaris* heath over the past decades (Pitcairn *et al.*, 1995).

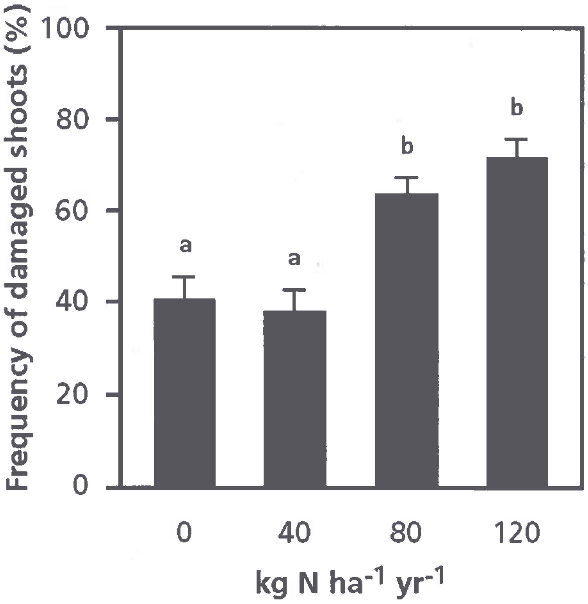

Figure 4.1. Effects of 7 years of ammonium nitrate additions (kg N ha^{-1} yr^{-1}) on the frequency (means ± SE) of winter damaged *C. vulgaris* shoots in northern wet heath (F4.11) in N. Wales (Carroll *et al.*, 1999). Colums sharing a letter are not significantly different.

A destructive harvest of the site took place in March 2000, prior to a controlled burn. Subsequent regrowth, primarily from stem bases, has been significantly lower in the higher nitrogen addition treatments, which may reflect the more mature status of *C. vulgaris* in the higher nitrogen addition plots (Lee and Caporn, 2001). Pilkington *et al.* (submitted) have completed a nitrogen budget for the site based on the harvest data. Significant increases in green tissue, wood and litter biomass, and litter depth were found in all nitrogen treatments. Although most of the added nitrogen in the 40 kg N ha^{-1} yr^{-1} treatment was found in green tissue and litter, increasing amounts of added nitrogen were found in the peat horizons in the higher nitrogen treatments.

Loss of *C. vulgaris* moorland in a number of UK areas has occurred as a result of invasion by bracken. Werkman and Callaghan (1996) undertook a three-year study in the northern Pennines (estimated current atmospheric deposition: 15 kg N ha^{-1} yr^{-1}) to examine the effect of nitrogen addition (50 kg N ha^{-1} yr^{-1}) to a series of plots set up on either heather or bracken-dominated areas, or at the heather-bracken interface. There was little effect on stands consisting entirely of heather or bracken, but at the heather-bracken boundary there was a trend for enhanced bracken growth and poorer heather growth. Anderson and Hetherington (1999) showed that decomposition rates of mixed litters were faster than those of litter of either of the two species alone, and also increased to a greater extent in response to nitrogen addition. These results, however, are in contrast to the findings of Gordon *et al.* (1999), who transplanted turf taken from a site in Scotland into experimental plots and added 50 kg N ha^{-1} yr^{-1} over three years, as part of a number of other treatments. This level of nitrogen addition increased the leading shoot length of heather from the first year of treatment onwards, and also increased shoot growth in spring, implying earlier budbreak. In contrast, the response of bracken was slow, showing no above-ground growth stimulation until the third year of treatment when there was a small, but significant, increase in canopy height. The responses of a mix of the two species were also examined in this three-year factorial experiment, but, unfortunately, yet not published (S.J. Woodin, pers.

comm.). Furthermore, both experiments show that factors such as drought and temperature play an important role in modifying responses to nitrogen. For example, the faster and larger shoot growth stimulation of heather than bracken by nitrogen, at least over 3 years, may be advantageous when water is not limiting but may be clearly disadvantageous when drought occurs (Gordon *et al.*, 1999).

These upland *C. vulgaris* heaths are actively managed, and this may be an important factor in their response to nitrogen. Increased frequencies of burn might compensate for the effects of acceleration of the *C. vulgaris* cycle in response to increased nitrogen deposition demonstrated by Carroll *et al.* (1999). Grazing pressure is also a significant factor; for example, Alonso *et al.* (2001) described experiments on the interacting effects of nutrient and grazing regimes on competition between *C. vulgaris* and two contrasting grass species on Scottish moorland sites. Although these experiments involved application of NPK, and hence cannot be used to estimate critical loads of nitrogen, their broader implications are important. Only where grazing levels are high and gaps are formed in the *C. vulgaris* canopy is the unpalatable and shade intolerant *Nardus stricta* likely to replace *C. vulgaris*; in contrast, the more shade tolerant and more palatable *Deschampsia flexuosa* is more likely to compete effectively with *C. vulgaris* at low grazing pressures. In general terms, active management of *C. vulgaris* moorlands may reduce the impact of increased nitrogen deposition and be associated with higher critical loads.

(b) Effects on shoot nutrient content

It has been shown in several studies that the nitrogen contents in moorland *C. vulgaris* and in bryophytes was significantly higher in areas with higher nitrogen inputs (>10-15 kg N ha^{-1} yr^{-1}), and increased in both the heather and mosses, compared with measurements of historical plant material (Pitcairn *et al.*, 1995). The increase in nitrogen content of *C. vulgaris* with nitrogen deposition has also been clearly demonstrated in experimental studies with a range of nitrogen treatments (Leith *et al.*, 2001; Carroll *et al.*, 1999). Increases in shoot nitrogen content have been linked in other studies to increased performance of winter moth (*Operophtera brumata*), outbreaks of which have led to extensive damage to heather moorland in Scotland over the last decade (Kerslake *et al.*, 1998).

In the long-term experiment in North Wales, analysis of foliar nutrient content between 1989 and 1992 showed increased nitrogen concentration with increasing nitrogen dose, and measurements taken in 1996 still showed a significant increase in response to nitrogen addition (Carroll *et al.*, 1999). A significant, but smaller absolute, effect was apparent in the two highest treatments in the 2000 harvest (Pilkington *et al.*, submitted). Interestingly, N:P ratios were clearly increased as a result of the treatments by 1996, with values of 16:1 in the controls compared with 23:1 in the plots with the highest nitrogen treatment (Carroll *et al.*, 1999). Carroll *et al.* (1999) compared these values with the critical threshold of N:P ratio of 14-16:1 proposed by Koerselman and Meuleman (1996) to indicate a switch from nitrogen to phosphorus limitation, and suggested that the induction of phosphorus limitation might explain the loss of response of shoot extension to nitrogen which occurred at this stage of the experiment. This interpretation is supported by evidence of increased phosphomonoesterase activity in peat and root surface (Johnson *et al.*, 1998; Lee *et al.*, 2000).

This observation is consistent with the study of Kirkham (2001), who sampled *C. vulgaris* shoots from a number of sites in England and Wales and analysed these for nitrogen and phosphorus content, and found a significant positive correlation between NO_x, but not total nitrogen, deposition and N:P ratio. The *C. vulgaris* N:P ratios were above 16:1 at about half the sampled sites, suggesting that nitrogen deposition has changed a substantial proportion of *Calluna*-dominated uplands in the UK from nitrogen-limited ecosystems into phosphorus-limited ones. Kirkham (2001) suggested that this could favour species such as *Molinia* that are better adapted to phosphorus limitation, although this would not necessarily be the case if increased nitrogen deposition increased mineralisation rates. The relationship between deposition and N:P ratio could be used to estimate a critical deposition threshold for this switch. However, the R^2 value for the relationship between NO_x deposition and N:P ratio reported by Kirkham (2001) was only 0.1, and analysis of other unpublished UK datasets, such as that from the Countryside Survey 2000 (S.A. Power, pers. comm.) does not show the expected increase in N:P ratio with increasing N deposition. Furthermore, data from a national survey of *Hylocomium splendens* in the UK show a weaker relationship between N deposition and N:P ratio than for N content (S.J. Woodin, pers.comm.).

(c) Effects on soils and root characteristics

C. vulgaris roots characteristically exhibit a substantial degree of ericoid mycorrhizal infection (Yesmin *et al.*, 1996), which is important for the degradation of complex organic substances in order to give plants access to nitrogen sources, which would be otherwise unavailable. The nitrogen addition experiment in North Wales has consistently shown little effect on mycorrhizal infection levels (Caporn *et al.*, 1995; Lee *et al.*, 2000), using either the ergosterol method or visual assessment. However, Yesmin *et al.* (1996) reported a negative correlation between nitrogen deposition and mycorrhizal infection rate at five remote Scottish sites with total deposition in the range 2-10 kg N ha^{-1} yr^{-1}, while a separate greenhouse study showed a small but significant decrease in infection rate when deposition rates were increased from 12 to 24 kg N ha^{-1} yr^{-1} for one year. The reasons for the differences in findings between the two studies are unclear and could relate either to methodological differences or the nature of the sites.

Soil studies at the North Wales site have found a number of responses with nitrogen addition. Leaching rates, although showing a small response to the higher nitrogen addition treatments, account for only a very small percentage of the added nitrogen. Significant increases in weight and total nitrogen content of *C. vulgaris* litter, collected beneath the canopy, were observed at 80 and 120 kg N ha^{-1} yr^{-1} , but increases in litter production were also observed at values as low as 40 kg N ha^{-1} yr^{-1} (Carroll *et al.*, 1999). There was also evidence of increased mineralization and decreased C:N ratio of litter and rhizosphere peat with increasing nitrogen treatment, although the peat effect was not found below 2 cm (Lee and Caporn, 2001). Overall, the results imply a high retention of the added nitrate and ammonium, probably through microbial immobilisation; Johnson *et al.* (1998) demonstrated that the long-term application of nitrogen at the site increased soil microbial biomass.

Summary

The long-term experimental study of a moorland ecosystem has significantly increased understanding of the key long-term mechanisms associated with vegetation change. However, the lack of studies at relatively low rates of nitrogen deposition is a major barrier to accurate definition of critical loads, which in turn may be highly dependent on management practices, while correlation studies based on a limited number of field sites are difficult to interpret with any confidence. It is recommended that the critical load of **10-20 kg N ha^{-1} yr^{-1}** for **northern wet heath dominated by *Calluna vulgaris* (F4.11)** as proposed by Bobbink *et al.* (1996), is retained. This is because there is no clear justification for a higher critical load than for dry heaths (which have a reliable estimate), as the dominant species, the indicators of exceedance, and the modifying factors are comparable. This is **expert judgement**, based on extrapolation, as the only available experimental evidence uses total N inputs (background plus treatments) significantly above the range of the proposed critical load.

4.3.2 'L' Erica tetralix dominated wet heath (lowland)

The west European lowland heathlands of wet habitats are dominated by the dwarf-shrub *Erica tetralix* (Ellenberg, 1988) and classified in EUNIS as northern wet heath (F4.11). The lowland wet-heathland communities are generally richer in plant species than dry heaths. In recent decades a drastic change in species composition of Dutch wet heathlands has been observed. Nowadays, many wet heathlands, which were originally dominated by *E. tetralix*, have become mono-stands of the grass *Molinia caerulea*. Together with *E. tetralix*, almost all of the rare plant species have disappeared from the system. It has been hypothesized that this change has been caused by atmospheric nitrogen eutrophication. Competition experiments using wet heath turfs have clearly shown that *M. caerulea* is a better competitor than *E. tetralix* at high nitrogen availability; after two years of application of nitrogen (150 kg N ha^{-1} yr^{-1}) the relative competitive strength of *M. caerulea* compared with *E. tetralix* was doubled (Berendse and Aerts, 1984). A 3-year field experiment with nitrogen application in Dutch lowland wet heathland (ca. 160 kg N ha^{-1} yr^{-1}) also indicated that *M. caerulea* is able to outcompete *E. tetralix* at high nitrogen availabilities (Aerts and Berendse, 1988). In contrast to the competitive relations between *C. vulgaris* and grass species, *M. caerulea* can outcompete *E. tetralix* without opening of the dwarf-shrub canopy. This difference is caused by the lower canopy of *E. tetralix* (25-35 cm), compared with *C. vulgaris* and the tall growth form of *M. caerulea*, which can overgrow and shade *E. tetralix* if enough nitrogen is available. It is in this respect also important that heather beetle plagues do not occur in wet heathlands, and until now no frost damage has been observed in this community.

Furthermore, it has been demonstrated in many Dutch wet heathlands that the accumulation of litter and humus has led to increased nitrogen mineralization (100-130 kg N ha^{-1} yr^{-1}) (Berendse *et al.*, 1987). In the first 10 years after sod removal, the annual nitrogen mineralization is very low, but afterwards it increases strongly to the above-mentioned high figures. This accumulation of nitrogen can be strongly influenced by increased deposition, because soil nitrogen leaching from wet heathlands is extremely low (Berendse, 1990). The observed nitrogen availabilities are high enough to change *E. tetralix*-dominated wet heathlands into mono-stands of *M. caerulea*.

Berendse (1988) developed a wet-heathland model to simulate carbon and nitrogen dynamics during secondary succession. He incorporated in this model the competitive relationships between *E. tetralix* and *M. caerulea*, the litter production from both species, soil nitrogen accumulation and mineralization, leaching, atmospheric nitrogen deposition and sheep grazing. He simulated the development of lowland wet-heathland after sod removal, because almost all of the Dutch communities are already strongly dominated by *M. caerulea* and it is impossible to expect changes in this situation without drastic management. Using the biomass of *M. caerulea* with respect to *E. tetralix* as an indicator, his results suggested 17-22 kg N kg ha^{-1} yr^{-1} as critical load for the transition of lowland wet-heath into a grass-dominated sward. This was the value recommended by Bobbink *et al.* (1996) as the critical load.

No new evidence has been published since 1995 with respect to the critical load for *Erica tetralix* dominated wet heaths. Because of the lack of natural variation in the modelling approach, the previous range of 17-22 kg N ha^{-1} yr^{-1} was too narrow. In particular, the model of Berendse (1990) is based on management using sod (turf) cutting every 50 years, together with grazing. Allchin *et al.* (2001) applied a similar model to UK dry heaths, and found that typical management regimes in the UK, such as mowing every 15 years, gave a threshold for changes in species composition which was about 10 kg N ha^{-1} yr^{-1} lower than for sod cutting. A similar effect of management regimes is to be expected in wet heaths. Thus, while the upper end of the critical load range may remain comparable to that proposed by Bobbink *et al.* (1996), based on the model with sod cutting, the lower end of the critical load range should be reduced to account for the effects of nitrogen under less intensive management regimes. There is no clear evidence of a differential response of *Calluna vulgaris* and *Erica tetralix* to simulated nitrogen deposition, and S. Smart (pers. comm.) found comparable spatial relationships between change in cover and nitrogen deposition in the UK for the two species. Therefore, it is recommended that the lower end of the critical load range should be the same as that for *Calluna*-dominated wet and dry heaths, i.e. 10 kg N ha^{-1} yr^{-1}.

In summary, a critical load range of **10-25 kg N ha^{-1} yr^{-1}** is recommended for ***Erica tetralix*-dominated northern wet heath (F4.11)**. This is **expert judgement**, as the effects of management and the validity of comparisons between *Calluna* and *Erica tetralix* are not supported by experimental data. The upper end of the range should be applied to systems with high intensity management, such as sod-cutting, and the lower end of the range to systems with low intensity management.

4.3.3 Dry heaths (F4.2; mostly Sub-Atlantic Calluna-Genista heaths F4.22)

(a) Introduction

Despite the conservation efforts and the adapted management in nature reserves, many lowland heaths (F4.2) have become dominated by grass species in Western Europe. An evaluation, using aerial photographs, has for instance demonstrated that more than 35% of Dutch heaths have been altered into grassland in the 1980s (Van Kootwijk and Van der Voet, 1989). It has been suggested that the strong increase in atmospheric nitrogen deposition might be a significant factor in the observed transition to grasslands. Similar, although not completely consistent, patterns were found in the UK over the past 20 to 50 years. Pitcairn *et al.* (1991) assessed changes in abundance of *Calluna vulgaris* at three heaths in East Anglia over the past decades. All three heaths showed a decline in *C. vulgaris* and an increase in grasses; the authors concluded that increases in nitrogen deposition (to ca. 30-40 kg N ha^{-1} yr^{-1}) was at least partly responsible for the changes, but also noted that the management had changed, too. A wider assessment of heathlands in SE England showed that in some cases *C. vulgaris* had declined and subsequently been invaded by grasses while other areas where still dominated by dwarf shrubs (Marrs, 1993). Although a move away from traditional management practices such as grazing, burning or sod cutting may be partly responsible, the decline in British heathlands has sometimes been linked with the steadily increasing levels of nitrogen deposition in the past 30 years. Furthermore it has been hypothesized that, besides important changes in land use, increased nitrogen deposition is an additional cause of the decline of heaths in the southern parts of the Nordic countries (e.g. Fremstad, 1992; Tybirk *et al.*, 1995.). Unpublished Norwegian studies indicated that *C. vulgaris* dominated heaths have been invaded by grasses after death of *C. vulgaris*, which seems to correlate with frost injury and plagues of the heather beetle, a species recently found in SW Norway and expanding its territory (Hansen, 1991; Fremstad, 1992).

Despite this range of circumstantial evidence and indications, competition experiments in the Netherlands have shown a significant effect of nitrogen addition between *C. vulgaris* and grass species only in young (low stature and cover) heaths (Heil and Diemont, 1983; Roelofs, 1986; Heil and Bruggink, 1987; Aerts *et al.*, 1990). A combination of mesocosm, field and modelling studies across Europe has made it clear over the last decade that the observed changes can only be explained by increased nitrogen deposition as part of a complicated and interacting sequence of events at different time scales, rather than by a simple change in competitive strength.

(b) Plant productivity and nutrient limitation

In nitrogen-limited systems, one of the first effects of increased nitrogen availability through atmospheric deposition is an increase in biomass production of the vegetation (e.g. see Fig. 1.1). Many studies indeed show increased plant productivity of dwarf shrubs after experimental nitrogen enrichment in dry heathlands in several NW European countries (e.g. Heil and Diemont, 1983; Van der Eerden *et al.*, 1991; Aerts and Heil, 1993; Power *et al.*, 1995; Lee and Caporn, 2001). This clearly indicates that most of these lowland dry heath ecosystems are primarily limited by nitrogen, although some inland dry heaths are limited by phosphorus (Riis-Nielsen, 1997; Nielsen *et al.*, 2000). An illustrative example of the growth stimulation of *C. vulgaris* has been found in a field experiment in Surrey, UK. It was set up in 1989 to assess the long-term impacts of realistic nitrogen loads on a lowland dry heath (F4.22) in southern Britain (Uren, 1992; Uren *et al.*, 1997; Power *et al.*, 1995; 1998a; 2001). After seven years, application of ammonium sulphate (7.7 & 15.4 kg N ha^{-1} yr^{-1}, current atmospheric deposition 8 kg N ha^{-1} yr^{-1}; Power and Barker, 2003) has not resulted in any negative effects upon *C. vulgaris*. Indeed, a significant stimulation of flower production, shoot density, and litter production occurred (15.4 kg N ha^{-1} yr^{-1}) and the canopy in the highest nitrogen treatment was 50% taller than in the control plots after 6 or 7 years (Power *et al.*, 1995; 1998a). The increased shoot growth in the nitrogen-treated vegetation is not reflected in root growth, and an increased shoot:root ratio has been inferred. Shoot nitrogen concentrations increased somewhat, too, but only significantly at the start of the experiment (yrs 1 and 3) or in current year's shoots. Clearly, this site remained nitrogen-limited over the whole experimental period of 7 years despite the additional nitrogen input of 7.7 or 15.4 kg N ha^{-1} yr^{-1}. This is consistent with the observation that the studied dry heath site in Surrey is probably one of the most nutrient-limited lowland heaths in the UK (Power *et al.*, 1998a). However, similar growth stimulation of C. *vulgaris* has been found in a dry lowland heath in Cheshire, UK and the Netherlands. Nitrogen was applied as ammonium nitrate (0, 20, 60 and 120 kg N ha^{-1} yr^{-1}) in Cheshire since 1996; the current atmospheric deposition at this site is estimated to be 20 kg N ha^{-1} yr^{-1}. Shoot growth and flowering in *C. vulgaris* clearly increased at the two highest nitrogen addition rates within two years; after 5 years of nitrogen addition these effects were maintained in terms of canopy density, while the canopy height of *C. vulgaris* was ca. 20 cm higher (Cawley, 2001; Lee and Caporn, 2001). Several nutrient addition experiments have shown nitrogen-limited plant growth in the Netherlands (see Bobbink *et al.* (1992a) for overview).

c) Nitrogen accumulation and mineralization

During secondary dry heath succession there is an increase in the amount of organic material and nitrogen in the soil (Chapman *et al.*, 1975; Gimingham *et al.*, 1979). The accumulation of organic matter and of nitrogen has been quantified after sod removal in dry heaths in the Netherlands by Berendse (1990). He found a large increase in plant biomass, soil organic matter and total nitrogen storage in the first 20-30 years of succession. Regression analysis suggested an annual increase of nitrogen in the system of ca. 33 kg N ha^{-1} yr^{-1}. These values are in good agreement with the measured nitrogen inputs in Dutch heathlands (Bobbink *et al.*, 1992b). The build up of organic matter in the soil after sod removal, which removed almost all of the soil surface organic matter, is likely to be accelerated by the enhanced biomass and litter production of the dwarf-shrubs caused by the extra nitrogen inputs. Furthermore, nitrogen accumulation will be accelerated. Using destructive harvesting in 1996, 7 years after the start of the Surrey-UK experiment, Power *et al.* (1998a) determined the nitrogen budget of the control and nitrogen enriched plots. Approximately 15 and 24% of the added nitrogen had been accumulated in plant material for the low and high nitrogen treatments respectively, while another 10 and 14% of the added nitrogen was found in the litter layer. Almost no nitrogen is lost from these systems, as ammonium immobilization in the soil is high and hardly any leaching losses to deeper layers have been measured in Dutch, British or Danish dry non-coastal heaths (De Boer, 1989; Van der Maas, 1990; Power *et al.*, 1998a; Kristensen and McCarty, 1999; Kristensen, 2001; Nielsen *et al.*, 2000). The nitrogen cycling in dry heaths is thus very tight and hardly any nitrogen leached to deeper soil layers, even at high nitrogen inputs, as has also observed in nitrogen manipulation studies in upland *C. vulgaris* moorland (par. 4.3.1; F4.11). Indeed, the nitrogen content of the soil (upper 10 cm) significantly increased from 35.0 g N m^{-2} to 45.5 g N m^{-2} after 7 years of 1.54 g N m^{-2} yr^{-1} addition in the Surrey dry heath, and nitrogen leaching remained very low, too (Power *et al.*, 1998a/b; Barker, 2001). At the Cheshire site, Caporn *et al.* (2002) reported leaching only at 120 kg N ha^{-1} yr^{-1} addition, constituting less than 10% of the added nitrogen. Only after severe damage of the *C. vulgaris* canopy by heather beetles, significant nitrogen leaching to the subsoil has been observed at lower rates of deposition (Van der Maas, 1990; Nielsen *et al.*, 2000).

The accumulation of nitrogen in the soil and litter layers affects the rates of decomposition and soil nitrogen mineralization. Power *et al.* (1998a) found that adding only 15.4 kg N ha^{-1} yr^{-1} resulted in faster rates of cotton strip degradation, a clear indication of stimulated decomposer activity. Furthermore, the estimated time for incorporation of litter into the soil humus pool, based on measurements of annual litter production and the size of the litter pool, decreased from 8.6 years in controls to 6.3 years in low nitrogen plots (7.7 kg N ha^{-1} yr^{-1} added) and 6.1 years in high nitrogen plots (15.4 kg N ha^{-1} yr^{-1} added) (Power *et al.*, 1998a). During secondary succession after sod cutting in the Netherlands, under high nitrogen loads, Berendse (1990) found that soil nitrogen mineralization was low in the first 10 years (ca. 10 kg N ha^{-1} yr^{-1}), but strongly increased in the next 20 years to 50-110 kg N ha^{-1} yr^{-1} with increased nitrogen content in the soil. This suggests a threshold for the amount of nitrogen immobilisation in the system, above which net nitrogen mineralization occurs. Thus, apart from direct nitrogen inputs from the atmosphere, soil nitrogen availability will also finally increase because of the almost complete retention of extra nitrogen in the system. Eventually, this will alleviate the nitrogen limitation to plant production and may lead to a shift to more nitrophilous species. The strong increase in nitrogen availability after an accumulation period of 1-2 decades may affect the competitive interactions within the systems, not only for (potential) dominant species, but also for subordinate vascular species. For lichens and mosses, direct effects might be also of importance in this respect.

(d) Changes in species composition

Competition experiments in containers and in the field have clearly demonstrated an important effect of increased nitrogen availability on the competitive interactions between *C. vulgaris* and grasses in the early phase of secondary succession in dry lowland heath. After experimental nitrogen additions (0,7 and 28 kg N ha^{-1} yr^{-1}) during 4 years in the 1980s in the Netherlands (with additional atmospheric deposition ca. 25-35 N ha^{-1} yr^{-1}), grasses (*Festuca ovina*) strongly outcompeted *C. vulgaris*, if the total cover of the vegetation was still low at the start of the experiment (e.g. Heil and Diemont, 1983). However, *C. vulgaris* clearly is a better competitor than grass species in mature heath vegetation even at high nitrogen loads if its canopy remains closed (Aerts *et al.*, 1990; Aerts, 1993).

Understorey species, especially the typical and frequent lichen and moss species, can be negatively affected by the increased growth of the vascular species. It has been found that after 7 years of nitrogen addition at Surrey (UK) the cover of the lichens and the lichen diversity (*Cladonia* spp; *Parmelia*) significantly decreased after additions of 7.7 & 15.4 kg N ha^{-1} yr^{-1} (current atmospheric deposition 8 kg N ha^{-1} yr^{-1}) (Barker, 2001; Fig. 4.2). Because of the weekly application of the extra nitrogen, this decline is certainly not caused by the direct toxic effects of nitrogen, but probably by increased shading through the greater canopy density of *C. vulgaris*. This has been confirmed by experimental removal of the shoots, which caused rapid recovery of the lichens (Barker, 2001). The same has been found for the moss understorey in the Cheshire-UK nitrogen experiment. The cover of *Hypnum* spp., a nitrophilous moss, was also considerably lower after 4 or 5 years of nitrogen additions (60 & 120 kg N ha^{-1} yr^{-1}) (Lee and Caporn, 2001). Recently, a strong decline in the lichen biomass (especially *Cladonia*) has been measured after 3 ½ year of addition of nitrogen (0, 20, 40 and 80 kg N ha^{-1} yr^{-1}) to a dry *C. vulgaris* heath at Clara, Ireland (current atmospheric 8-10 kg N ha^{-1} yr^{-1}). The cover of lichens has already been significantly reduced to ca. 10%, compared with ca. 30% in the control vegetation in the treatment with 20 kg N ha^{-1} yr^{-1}, while in the high nitrogen plots the lichens almost disappeared (Tomassen *et al.*, 2002).

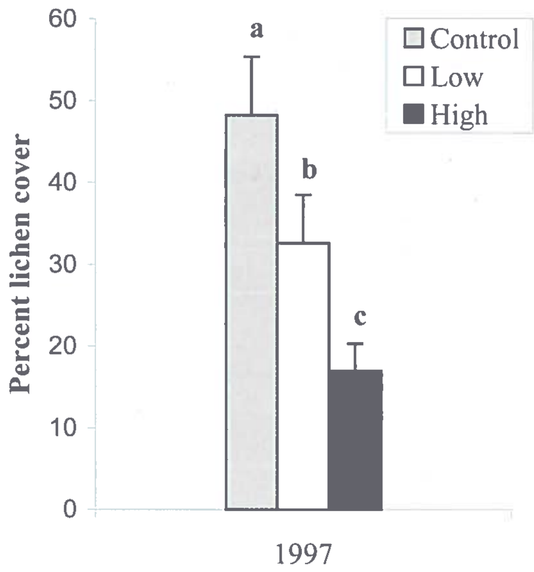

Figure 4.2. Lichen cover (%) after 7 years of addition of nitrogen on dry heath (F4.2) at Thursley, Surrey (UK) with low N (7.7 kg N ha^{-1} yr^{-1}) and high N (15.4 kg N ha^{-1} yr^{-1}) artificial raining (redrawn from Barker, 2001).

When the productivity of *C. vulgaris* is no longer primarily limited by nitrogen, the growth response will be less or absent, and luxury consumption will lead to increased nitrogen concentrations in the plant. In a phosphorus-limited heathland in Denmark (current atmospheric deposition 18 kg N ha^{-1} yr^{-1}), nitrogen addition of 15, 35 and 70 kg N ha^{-1} yr^{-1} as ammonium nitrate increased shoot nitrogen concentrations significantly within 2 years of the start of the experiment (Johansson, 2000), while there was no significant growth response of the dwarf shrubs (Riis-Nielsen, 1997). Correlative field studies in low and high nitrogen inputs areas and experimental applications of nitrogen to heathlands have shown increased concentrations of nitrogen in the (green) parts of *C. vulgaris* at high nitrogen fluxes (e.g. Heil and Bruggink, 1987; Bobbink and Heil, 1993; Pitcairn *et al.*, 1995). In forests, a clear relation between increased foliar nitrogen concentrations and pathogenic attacks as well as sensitivity secondary stresses has been established (see chapter 3). Similarly, experiments have at least partly indicated an increased risk to canopy damage for *C. vulgaris* under increased nitrogen availability. Heather beetle outbreaks and nitrogen-induced secondary stresses (especially winter injury and drought) are probably the main processes, which can open the canopy, and are thus crucial for the dramatic shift in species composition observed in lowland heath in high nitrogen load regions.

(e) Opening of the C. vulgaris canopy: heather beetle outbreaks

Outbreaks of heather beetle (*Lochmaea suturalis)*, a chrysomelid beetle, can occur frequently in dry lowland heaths. It forages exclusively on the green parts of *C. vulgaris*. Insect herbivory is generally affected by the nutritive value of the plant material, with nitrogen concentration especially important (e.g. Crawley, 1983). Outbreaks of the beetle can lead to the opening of closed *C. vulgaris* canopy over large areas, greatly reducing light interception (Berdowski, 1987; 1993) and leading to enhanced growth of understorey grasses, such as *Deschampsia flexuosa* or *Molinia caerulea*. Brunsting and Heil (1985) found that the growth of the larvae increased after foraging on the leaves of *C. vulgaris* with higher nitrogen concentrations in a rearing experiment. After field additions of ammonium sulphate in a roofed heath experiment, total number or biomass of the 1st stage larvae of the beetle was not affected by the treatments, but the development of subsequent larval stages was significantly accelerated (Van der Eerden *et al.*, 1990). Larval growth rates and adult weight of heather beetles were found to be significantly higher when these insects were reared on C. *vulgaris* plants, which were collected in the British lowland heath experiment, after seven years of relatively low nitrogen addition (Power *et al.*, 1998b).

Heather beetle larvae were also cultivated on shoots of *C. vulgaris* taken from plants, which had been fumigated with ammonia in open top chambers (12 months; 4-105 $\mu g\ m^{-3}$) (Van der Eerden *et al.*, 1991). After 7 days, both the mass and development rate of the larvae were clearly increased at higher ammonia concentrations. In addition, the growth of heather beetles instars was also significantly stimulated on *C. vulgaris* from UK heaths after ammonia fumigation, with relatively high concentrations, probably caused by the enhanced nitrogen concentrations in the plant material (Uren, 1992). It has also been demonstrated that increased nitrogen deposition to upland *C. vulgaris* stands stimulated larval development and growth rate of winter moth (*Operophtera brumata*) (Kerslake *et al.*, 1998). It is thus likely that the frequency and the intensity of insect outbreaks can be stimulated by increased atmospheric nitrogen loads. This is supported by the observations of Blankwaardt (1977), who reported that from 1915 onwards heather beetle outbreaks occurred at ca. 20 year intervals in the Netherlands, whereas in the last 15 years of the observation period, the interval has been less than 8 years. In addition, it was observed that *C. vulgaris* plants were more severely damaged in nitrogen-fertilised vegetation during a heather beetle outbreak, both in the Netherlands (Heil and Diemont, 1983), in Denmark (Tybirk *et al.*, 1995; Riis-Nielsen, 1997) and in the UK (Lee and Caporn, 2001). It is thus likely that enhanced atmospheric nitrogen deposition influences the frequency and severity of beetle outbreaks, although the exact controlling processes need further quantification.

(f) Opening of the C. vulgaris canopy: secundary stresses

It has been shown that frost sensitivity increased in some tree species with increasing concentrations of air pollutants (e.g. Aronsson, 1980; Dueck *et al.*, 1991). This increased susceptibility is sometimes correlated with the enhanced nitrogen concentrations in the leaves or needles. Impacts of nitrogen deposition on the frost sensitivity of *C. vulgaris* could be possible, and it is suggested that the observed die-back of the *C. vulgaris* shoots in the successive severe winters of the mid-1980s in the Netherlands was, at least partly, caused by increased winter injury.

Van der Eerden *et al.* (1990) studied the effects of ammonium sulphate and ammonia upon frost sensitivity in *C. vulgaris.* Fumigation with ammonia of *C. vulgaris* plants in open-top chambers over 4-7 month periods (100 $\mu g\ m^{-3}$) revealed that frost sensitivity was not affected in autumn (September or November), but in February, just before growth started, frost injury increased significantly at -12 °C (Van der Eerden *et al.*, 1991). A similar study in open-top chambers during the winter period in England also demonstrated increased frost sensitivity of heather plants, but the applied concentrations of ammonia were very high (140-280 $\mu g\ m^{-3}$) (Uren, 1992). Van der Eerden *et al.* (1991) studied the frost sensitivity of *C. vulgaris* vegetation artificially sprayed with different levels of ammonium sulphate (3-91 kg N $ha^{-1}\ yr^{-1}$). After 5 months the frost sensitivity of *C. vulgaris* increased slightly, although significantly, compared with the control in vegetation treated with the highest level of ammonium sulphate (400 µmol l^{-1}; 91 kg N ha^{-1} yr^{-1}). The sensitivity decreased again two months later and no significant effects of the ammonium sulphate application upon frost hardiness were measured at that time. All these studies used high levels of nitrogen deposition; however, the only study of the effects of low levels of nitrogen addition (7.7 & 15.4 kg N ha^{-1} yr^{-1}) on frost sensitivity of *C. vulgaris* showed no significant effects after seven years of treatment (Power *et al.*, 1998b). Hence, the significance of increased frost sensitivity at ambient nitrogen loads is very uncertain and application of ammonium nitrate may even reduce frost sensitivity.

Summer 'browning' of the *C. vulgaris* canopies was frequently seen in dry summers in the 1980s in the Netherlands - the decade with the highest nitrogen loads. This suggested that nitrogen enrichment stimulated the sensitivity of *C. vulgaris* to periods of drought, probably by reduced root growth with respect to the development of the shoot or by a decrease in mycorrhizal infection. These effects might have major implications for the capacity of plants to deal with water or nutrient stress. The partitioning of biomass is very plastic and mostly determined by nutrient availability and light intensity (e.g. Brouwer, 1962; 1983). It has been shown that most plant species studied allocated more biomass to the shoots than to the roots at higher nutrient concentrations (e.g. Poorter and Nagel, 2000). This also held for *C. vulgaris* and other heathland species in a pot experiment over two years; its root weight ratio (RWR) significantly decreased with increasing nitrogen additions (Aerts *et al.*, 1991). However, field validation of these lower root to shoot ratios as a response to nitrogen enrichment is scarce. An indication of its importance was observed after long-term nitrogen applications in the Surrey-UK dry lowland heathland (Power *et al.*, 1998a). They found a small reduction in root to shoot ratio after seven years of nitrogen addition (15.4 kg N ha^{-1} yr^{-1}). Van der Eerden *et al.* (1991) applied artificial rain containing ammonium sulphate (3, 6,11, 23, 45 and 91 kg N ha^{-1} yr^{-1}) during two growing seasons to a dry heath (Assel, the Netherlands), under a partial roof. Measurements of plant characteristic demonstrated decreases in the root:shoot ratios in both *C. vulgaris* and *Deschampsia flexuosa*. Power *et al.* (1998b) also found higher water losses from *C. vulgaris* plants from the nitrogen-treated vegetation, compared with the control situation, but no differences in water potential of the shoots. Hence the hypothesis that increased nitrogen deposition might lead to severe growth reduction or even local 'browning' and die-back of this species in cases of severe drought episodes is still mostly speculative. Recently, a summer drought period (May-September) was experimentally implied under roofs to half of the plots in the second year of the nitrogen-enriched lowland heath plots in the Cheshire-UK study (Cawley *et al.*, 1998; Lee and Caporn, 2001). The high nitrogen treatments (60 & 120 kg N ha^{-1} yr^{-1}) clearly increased the vulnerability of *C. vulgaris* to this drought stress. The cover of *D. flexuosa* significantly increased in these water-stressed plots, but this was partly attributable to a heather beetle outbreak, which was more severe in the droughted plots. However, *C. vulgaris* cover has partly recovered by 2000 from the 1997 drought period. This experiment is the only one to indicate the potential importance of the interaction between nitrogen enrichment and long dry periods.

Besides the changes in root to shoot ratios, ericoid mycorrhizal infection of the roots of the heathers could also be influenced by an increase in nitrogen load. However, few studies have been conducted and the outcomes are highly variable, as for AM mycorrhizal infection (Aerts and Bobbink, 1999). Some studies on the effects of increased nitrogen availability on ericoid mycorrhizal infection of *C. vulgaris* root showed no effects (Johansson, 2000), in others nitrogen inputs stimulated the infection of mycorrhizas (Caporn *et al.*, 1995), whereas other studies showed restricted infection after nitrogen treatment (Yesmin *et al.*, 1996). At this moment the importance of this phenomenon in the decline of *C. vulgaris* and the shift to grass dominance is not at all clear. It is obvious that the sensitivity of *C. vulgaris* to drought stress might be increased by the shift in root:shoot ratio and that grasses might profit from this damage to the heather canopy, but the precise importance of this process has to be clarified under long-term nitrogen applications with repeated drought episodes.

(g) Modelling

Computer models offer a tool to assess the impacts of nitrogen deposition on dry heathlands over decades with varying levels of secondary stresses and under different management regimes. Allchin *et al.* (2001) modified the CALLUNA model originally developed by Heil and Bobbink (1993) for UK lowland heaths and examined the effects of different management regimes. Under the less intensive current UK management systems, compared with traditional sod-cutting used in the Netherlands, the critical load for invasion of grass species was reduced to 10 kg N ha^{-1} yr^{-1}. This implies that a reduction in the lower limit of the critical load range may be needed to cover all the management regimes used for dry heaths.

Summary

The impacts of increased nitrogen inputs to dry inland heaths (F4.2) are complex and occur at different time scales. Firstly, increased nitrogen availability stimulates biomass and litter production of *C. vulgaris* in most situations. Nitrogen is strongly retained in the system, gradually leading to higher nitrogen mineralization rates in the soil. However, the species remains the stronger competitor with respect to grasses, even at very high nitrogen availability, if the canopy is not opened. The shift from dwarf shrub to grass dominance is clearly triggered by opening of the canopy caused by heather beetle attacks, winter injury or drought. After decline of the *C. vulgaris* shoots, grasses quickly profit from the increased light intensity, together with the high nitrogen availability because of nitrogen accumulation. Within a few years, this may lead to a drastic increase in grass cover. Because of the stochastic behaviour of several processes (e.g. heather beetle outbreaks, winter injury and drought) and the many long-term processes, which interact with them, it is very difficult to clarify experimentally all these relationships without long-term (10-20 years) and large-scale experiments.

It is finally concluded that it is necessary to modify the established critical load of 15-20 kg N ha^{-1} yr^{-1} for dry heaths which was recommended by Bobbink *et al.* (1992a; 1996). A key factor in this decision is the range of new evidence of impacts from the Surrey (UK) long-term experiment, which has applied relatively low rates of nitrogen deposition. Of particular importance is a re-evaluation of the background deposition at this site by Power and Barker (2003), who based their estimate on site measurements of NO_2 and NH_3 concentrations and of wet deposition of nitrate and ammonium. This provided a site-specific estimate of 8 kg N ha^{-1} yr^{-1}, rather than the previous estimate of 10-15 kg N ha^{-1} yr^{-1}. However, neither estimate accounts for the input from aerosol sources and nitric acid, which may add another 4 kg Nha^{-1} yr^{-1} at this site (M. Sutton, pers. comm.). Significant effects were observed in this experiment at application rates of 7.7 kg

N ha^{-1} yr^{-1}, i.e. at a total deposition of 16-20 kg N ha^{-1} yr^{-1}, the previously recommended critical load range for dry heaths.

It is therefore recommended that the lower end of the critical load range is lowered from 15 kg N ha^{-1} yr^{-1} to 10 kg N ha^{-1} yr^{-1}. This value, based primarily on a long-term field experiment, is also supported by the results of simulation modelling using low intensity management regimes. Hence the new recommended **critical load of 10-20 kg N ha^{-1} yr^{-1}** can be classified as **reliable**. It should be stated that most nitrogen-addition studies have been done in a subcategory of dry heaths (F4.22), but it seems reasonable that this critical load can be applied across F4.2. However, it is advisable to use the low end of the range in lichen-rich dry heaths, as shown in the Surrey-UK study. The relative importance of phosphorus availability in some dry heath areas, and habitat management, as modifiers of dry heath response to increased nitrogen deposition have to be quantified. However, in general terms, the lower end of the range should be used with systems which lave less intensive management and which are not phosphorus-limited.

Table 4.1. Empirical critical loads for the effects of excess nitrogen on heathland, scrub and tundra habitats (F). ## reliable, # quite reliable and (#) expert judgement.

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Tundra	F1	5-10^{a}	#	Changes in biomass, physiological effects, changes in species composition in moss layer, decrease in lichens
Arctic, alpine and subalpine scrub habitats	F2	5-15^{a}	(#)	Decline in lichens mosses and evergreen shrubs
Northern wet heath	F4.11			
• 'U' *Calluna*-dominated wet heath (upland moorland)	F4.11	10-20^{a}	(#)	Decreased heather dominance, decline in lichens and mosses
• 'L' *Erica tetralix* dominated wet heath (lowland)	F4.11	10-25^{a,b}	(#)	Transition heather to grass
Dry heaths	F4.2	10-20^{a,b}	##	Transition heather to grass, decline in lichens

[a)] use towards high end of range at phosphorus limitation, and towards lower end if phosphorus is not limiting;
[b)] use towards high end of range when sod cutting has been practiced, use towards lower end of range with low intensity management.

5. Effects of nitrogen inputs in grassland and tall forb habitats (EUNIS class E)

5.1 Introduction

An overwhelming range of grassland ecosystems (EUNIS class E) are found across Europe, from very dry to wet habitats, acid to alkaline conditions, inland saline soils, heavy metal vegetation or under very different climatic regimes (e.g. Ellenberg, 1988; Davies and Moss, 2002). Only a restricted part of these grasslands is of natural origin (e.g. dry steppe grasslands, alpine grasslands), while most of these habitats are covered by semi-natural vegetation. Traditional agricultural use or management is thus an important ecological factor in the structure and function of these grassland systems. These grasslands have long been an important part of the European landscape and contain many rare and endangered plant and animal species; a number of them have been set aside as nature reserves in several European countries (e.g. Ellenberg, 1988; Woodin and Farmer, 1993). Semi-natural, conservational grasslands are generally nutrient-poor, because of the long agricultural use with low levels of manure and with removal of plant parts by grazing or hay making. The vegetation is characterized by many species of low stature and of nutrient-poor soil status (Ellenberg jr., 1988). However, some semi-natural meadow communities of high nature conservation value, particularly those on deep alluvial soils in river flood plains subject to periodic inundation or inputs of farmyard manure, can be moderately fertile with soil macro-nutrients levels at the higher end of the spectrum covering species-rich grasslands. These are likely to have a higher proportion of relatively fast-growing specimes than for example oligotrophic acid or calcareous grasslands. To maintain high species diversity, artificial fertilisers have, of course, to be avoided. It is thus to be expected that several of these species-rich grasslands, especially of oligotrophic or mesotrophic soils, can be affected by increased atmospheric nitrogen inputs (e.g. Wellburn, 1988; Liljelund and Torstensson, 1988; Ellenberg jr., 1988). Moreover, some of the most species-rich grasslands are found under weakly buffered or almost neutral conditions, which make them sensitive to acidification and very sensitive to negative impacts of ammonium accumulation in case of high deposition of reduced nitrogen.

Empirical nitrogen critical loads were established for a restricted range of grassland habitats, i.e. calcareous and neutral to acidic grasslands, in the two previous reports on empirical nitrogen critical loads because of the lack of studies in almost all other grassland ecosystems. Very nutrient-poor acid grasslands were combined with species-rich heaths in the heathland section (Bobbink *et al.,* 1992a; 1996). In this report, the EUNIS-habitat cataloguing has been fully adopted, leading to some rearrangement of the former classification. Grassland and tall forb habitats (EUNIS class E), which are dry or only seasonally wet (with the water table at or above ground level for less than half of the year) have a vegetation cover greater than 30%. The dominant part of the vegetation is grasses and other non-woody plants, including moss-, lichen-, fern- and sedge-dominated communities. The first level of division with relevance for the empirical approach, is based on soil water availability (dry-mesic-wet grasslands). Most of the nitrogen effect studies (both those used to set previous critical loads, as well as those from the 1996-2002 period) have been done in ecosystems which have to be classified as dry grasslands (**E1**). This subcategory is also, unfortunately one of the few, with a subsequent and necessary division into base-rich systems (perennial calcareous grassland and basic steppes, **E1.2**) and into basedeficient grassland systems (non-Mediterranean dry acid and neutral grasslands, **E1.7**) (Davies and Moss, 2002). Thus these categories are more or less comparable with the 1996-groupings of calcareous grasslands (which are all dry grasslands) and the acidic/neutral grasslands (with some exceptions and additions, discussed in section 5.3). However, for the other sub-categories of E1, there is no relevant information.

The impacts of nitrogen inputs are, furthermore, only studied in a very small section of the other major EUNIS types (E2-E7). Some information exists for mesic grasslands (**E2**), which are especially classified on the basis of the present land use or management, and wet grasslands (**E3**). The latter group is split into oligotrophic wet grasslands (**E3.5**) and mesotrophic to eutrophic wet grasslands (**E3.4**). As this last division is of major importance for the setting of the critical loads, we tried to determine the critical load value for all grasslands at the third hierarchy level of EUNIS, despite the low overall number of experimental studies for mesic and wet grasslands. A new category within the grassland section is the impacts of nitrogen deposition in alpine and subalpine grasslands (**E4**).

5.2 Dry grasslands (E1)

5.2.1 Perennial calcareous grassland and basic steppes (E1.2)

Calcareous grasslands (EUNIS code E1.2) are communities on limestone, which are wide-spread in the hilly and mountainous regions of Western and Central Europe. Subsoils consist of different kinds of limestone with high contents of calcium carbonate (>90%), covered by shallow well-buffered rendzina soils low in phosphorus and nitrogen (A/C-profiles; pH of the top soil: 7-8 with calcium carbonate content of ca. 10%). Plant productivity is low and 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. Wolkinger and 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).

A gradual increase of one grass species (*Brachypodium pinnatum*) has been observed by the Nature Conservation Agencies in Dutch calcareous grasslands in the late 1970s/early 1980s, although the management (hay making in autumn) had not changed since the mid 1950s. It has been hypothesized that the increased atmospheric deposition of nitrogen (from 10-15 kg N ha^{-1} yr^{-1} in the 1950s to 25-35 kg N ha^{-1} yr^{-1} in the 1980s) caused this drastic change in vegetation composition (Bobbink and Willems, 1987). The effects of nitrogen enrichment have, therefore, been investigated in two field experiments in the Netherlands (Bobbink *et al.,* 1988; Bobbink, 1991). Application of ammonium nitrate (50-100 kg N ha^{-1} yr^{-1} + ambient load of 30-35 kg N ha^{-1} yr^{-1}) over 3 years resulted in a drastic increase of the grass *B. pinnatum* and a strong reduction in species diversity (including several Dutch Red List species), caused by the change in vertical structure of the grassland vegetation (Fig. 5.1). *B. pinnatum* proves to have a very efficient nitrogen acquisition and a very efficient withdrawal from its senescent shoots into its well-developed rhizome system. It benefits from the extra nitrogen redistributed to the below-ground rhizomes by enhanced growth in the next spring. In this way *B. pinnatum* strongly monopolized (>75%) the nitrogen storage in both the above-ground and below-ground compartments of the vegetation with increasing nitrogen availability (Bobbink *et al.*, 1988; 1989; De Kroon and Bobbink, 1997). Besides this decrease in phanerogamic plant species, many characteristic lichens and mosses have disappeared in recent years from the calcareous grasslands (During and Willems, 1986). This is partly caused by the (indirect) effects of extra nitrogen inputs, as experimentally shown by Van Tooren *et al.* (1990). Until now, data on the effects of nitrogen eutrophication on the species-rich fauna of calcareous grassland are not available.

Nitrogen cycling and accumulation in calcareous grassland can be significantly influenced by two major outputs: (i) leaching from the soil, and (ii) removal with management regimes.

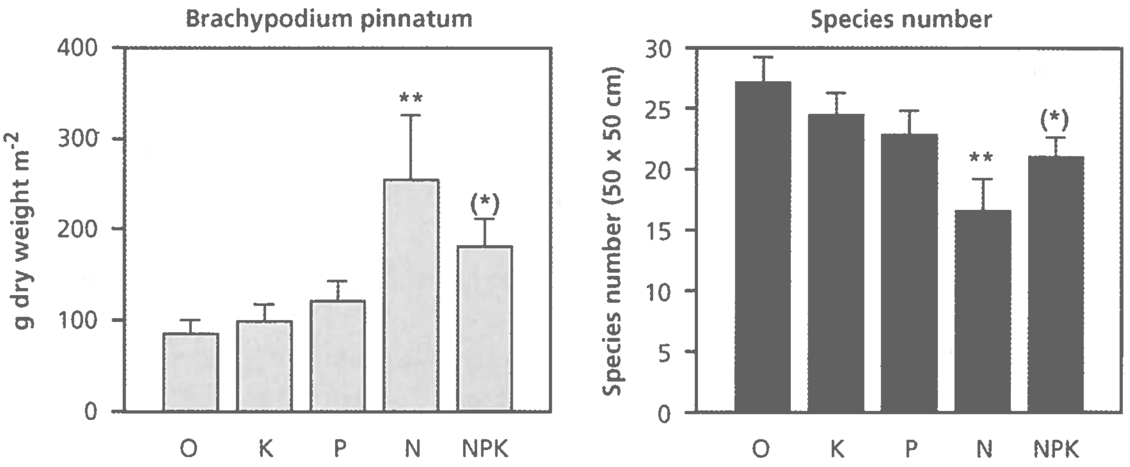

Figure 5.1. Above-ground biomass of *Brachypodium pinnatum* (g m^{-2}) and number of vascular plant species (per 50 x 50 cm) in Dutch perennial calcareous grassland (E1.2) after three years of nitrogen application as ammonium nitrate (kg N ha^{-1} yr^{-1}) (adapted from Bobbink, 1991).

Nitrogen losses by denitrification in dry calcareous grasslands are low (<1 kg N ha^{-1} yr^{-1}) (e.g. Mosier *et al.*, 1981). Ammonium and nitrate leaching has been studied in Dutch calcareous grasslands by Van Dam (1990). The observed nitrogen leaching from untreated vegetation is very low (0.7 kg N ha^{-1} yr^{-1}), and only 2% of total atmospheric nitrogen deposition. After two-weekly spraying of ammonium sulphate for 2 years (50 kg N ha^{-1} yr^{-1}), nitrogen leaching significantly increased to 3.5 kg N ha^{-1} yr^{-1}, although this figure is also only a small proportion (4%) of the total nitrogen inputs (Van Dam, 1990). It is thus evident that the studied calcareous grassland ecosystems almost completely retain nitrogen in the system, caused by a combination of enhanced plant uptake (Bobbink *et al.*, 1988; Bobbink, 1991) and increased immobilization in the soil organic matter (Van Dam *et al.*, 1990).

The most important output of nitrogen from calcareous grasslands is by exploitation or management. From the 1950s to mid 1980s, almost all of the calcareous grasslands in the Netherlands were mown in autumn with removal of the hay. The annual nitrogen removal in the hay varies slightly between years and sites, but in general between 17-22 kg N ha^{-1} is removed from the system with the usual management (Bobbink, 1991; Bobbink and Willems, 1991). Legume species (*Fabaceae*) also occur in calcareous vegetation, and provide an additional nitrogen input with the nitrogen-fixing microorganisms in their root nodules (ca. 5 kg N ha^{-1} yr^{-1}). The nitrogen mass balance of Dutch calcareous grasslands was summarized in the Lökeberg paper (Bobbink *et al.*, 1992a), and a critical nitrogen load was estimated with a steady-state mass balance model (e.g. De Vries, 1994), because of the lack of long-term addition experiments with low nitrogen loads. Assuming a critical long-term immobilization rate for N of 0-6 kg N ha^{-1} yr^{-1}, the critical nitrogen load can be derived by adding the nitrogen fluxes due to net uptake, denitrification and leaching, corrected for the nitrogen input by fixation. In this way, 15-25 kg N ha^{-1} yr^{-1} has been established in Lökeberg as nitrogen critical load. These values are in close accordance with the results found by Neitzke (1998; 2001) in a calcareous grassland in the East Eifel in southwest Germany. In a gradient of nutrient enrichment extending from an agricultural field, she found that nitrogen mineralization explained by far the highest variation in species composition and species degradation from the nutrient enriched border zone to the intact central chalk grassland (principal components analysis, correlation of phosphorus only with 3rd axis). Comparing the soil nitrogen mineralization rates of the undisturbed chalk grasslands and the plots with significantly altered species composition, she found that *B. pinnatum* had increased, and species diversity had decreased when nitrogen mineralization increased from 6-10 kg N ha^{-1} yr^{-1} in the unaffected parts to 35-55 kg N ha^{-1} yr^{-1} in the area bordering the agricultural fields (Neitzke, 1998; 2001).

In calcareous grassland in England, addition of nitrogen hardly stimulated dominance of grasses (Smith *et al.*, 1971; Jeffrey and Pigott, 1973). In these studies, with application of 50-100 kg N ha^{-1} yr^{-1} and a further addition of phosphorus, a strong dominance of the grasses *Festuca rubra*, *F. ovina* or *Agrostis stolonifera* was observed. However, *B. pinnatum* or *Bromus erectus*, the most frequent species in continental calcareous grassland, were absent from these British sites, so the data are not comparable in this respect. Following a survey of data from a number of conservation sites in southern England, Pitcairn *et al.* (1991) concluded that *B. pinnatum* had expanded in the UK during the last century. They considered that much of the early spread could be attributed to a decline in grazing pressure but that more recent increases in the grass had, in some cases, taken place despite grazing or mowing, and could be related to nitrogen inputs. This was partly confirmed by Hewins and Ling (1998), who observed lower plant diversity and more aggressive grass species in calcareous grassland in those parts of the Cotswold (UK), receiving higher ammonium concentrations. A retrospective study of a heavily grazed chalk grassland at Parsonage Downs (UK) has, however, shown no substantial change in species composition over the twenty years between 1970 and 1990, a period when nitrogen deposition is thought to have increased to 15-20 kg N ha^{-1} yr^{-1} (Wells *et al.*, 1993). *B. pinnatum* was present in the sward but had not expanded as in the Dutch grasslands, but these field data from British calcareous grasslands are still consistent with the range of the 1992 critical load. In a study with chalk grassland species in mesocosms (South Downs, southeast England) the biomass of *B. pinnatum* significantly increased after 2 years of nitrogen addition (≥70 kg N ha^{-1} yr^{-1}), even under a 8-weekly summer defoliation regime (Bryant, 1998). Unkovich *et al.* (1998) also found nitrogen limitation in their field study in Wytham, UK. These authors have found more than a doubling of herbage production with 6-weekly additions of 11.5 kg N ha^{-1} (total nitrogen addition of almost 60 kg N ha^{-1}) independent of nitrogen form (ammonium or nitrate) and no response to phosphorus or any other nutrient. Adding the nitrogen in the form of ammonium sulphate or ammonium nitrate increased nitrogen concentrations significantly compared with the controls and plots where nitrogen was added as potassium nitrate. Plants took up more than 40% of added nitrogen, but still most of the added nitrogen (almost 60%) was immobilised in the soil (Unkovich *et al.*, 1998).

The long-term effects of ammonium nitrate additions (35, 70 and 140 kg N ha^{-1} yr^{-1} since 1990) have been studied in a calcareous grassland on carboniferous limestone in Derbyshire in the UK (E1.26) (Morecroft *et al.*, 1994; Carroll *et al.*, 1997; Lee and Caporn, 1999; Carroll *et al.*, 2000). Within the first 4 years, they did not observe a significant decline in species composition of vascular plants caused by nitrogen. From the sixth year on, there has been a marked and significant dose-related decline in vascular plant cover with increased nitrogen addition (Carroll *et al.*, 1997; Lee and Caporn, 1999; Carroll *et al.*, 2003). In 1999 there was an increase in grass cover, and a strong dose-related decrease in herbs, legumes, geophytes and monocarpic plants from the lowest nitrogen treatment on (35 kg N ha^{-1} yr^{-1}). The species that were negatively affected by nitrogen included a range of typical subordinate species of calcareous grasslands, while on the other hand *Hypochaeris radicata* was able to invade in nitrogen-enriched plots only (Carroll *et al.*, 1997, Lee and Caporn, 1999). Significant changes in bryophyte species composition have also been observed in response to nitrogen addition, many of which reflect an acidification effect. Vegetation responses were overall slow, and significance for individual species cover was found mainly at higher levels of nitrogen addition, though the same trends could be witnessed from the lowest treatment on. Recent results from a labelling experiment with mesocosms from this site suggest an effect of long-term nitrogen application on increasing carbon allocation to shoots and decreasing it to roots. An additional nitrogen x phosphorus factorial field experiment, which was started in 1995 (35-140 kg N ha^{-1} yr^{-1}) at the same site, showed similar effects on species composition for the sole nitrogen treatment in the sixth growing season (Lee and Caporn, 2001).

In contrast to the slow vegetation responses, shoot nitrogen concentrations, nitrate reductase activities, and soil nitrogen mineralization and nitrification rates significantly increased during the first years with enhanced inputs of nitrogen ($\geq$70 kg N ha^{-1} yr^{-1}) (Morecroft *et al.*, 1994; Carroll *et al.*, 1997). By 1995 soil pH values also had decreased significantly in response to increased nitrogen ($\geq$70 kg N ha^{-1} yr^{-1}), most probably as a consequence of enhanced nitrification rates with its associated proton production. This acidification effect was stronger if ammonium sulphate (only included at the highest treatment (140 kg N ha^{-1} yr^{-1})) was added. Soil microbial activity was not significantly affected by ammonium nitrate additions but phosphomonoesterase activity increased significantly with the addition of ammonium sulphate (Johnson *et al.*, 1998). Despite high additions of nitrogen, analyses of the soil for total nitrogen gave no indication for nitrogen accumulation in these thin soils. As the site is strongly phosphorus limited (Carroll *et al.*, 1997), the vegetation could not take up extra nitrogen to a level that could explain the lack of nitrogen accumulation in the soil. Calculating the annual budget on the basis of nitrogen mineralization, fixation and addition as inputs and plant uptake as output, there is a substantial amount of nitrogen missing from the system. This calculated missing nitrogen has most probably leached out, thereby exceeding the accepted leaching losses for these kind of grassland systems ($\geq$ 2 kg N ha^{-1} yr^{-1}) already at an addition of 35 kg N ha^{-1} yr^{-1}.

The current nitrogen deposition in this area of the UK is estimated to be (19)-25/30 kg N ha^{-1} yr^{-1}. Thus, it is not possible to accurately estimate a critical load from the nitrogen additions used. For that reason, a reduction experiment in greenhouse conditions has been set up with mesocosms from a neighbouring calcareous grassland site (2, 10, 20 and 55 kg N ha^{-1} yr^{-1} added as ammonium nitrate) (Jones and Ashenden, 2000). Two years after the initiation of this experiment, very few effects of differential nitrogen loadings on the vegetation were evident. However, in view of the strong phosphorus limitation, it may take a much longer time for any effect to become significant. Leaching of nitrate, however, was already found in both winters since the start of the experiment in all treatments receiving 10 kg N ha^{-1} yr^{-1} or more, but an indication of a clear dose-related effect was only observed in the second winter (Jones and Ashenden, 2000).

In 1996 the values from the Lökeberg paper were adapted to 15-35 kg N ha^{-1} yr^{-1} to allow for the lack in vegetation response in case of phosphorus limitation in combination with adequate management. It was suggested to use the low end of the range for nitrogen-limited, and the high end of the range for phosporus-limited calcareous grasslands (Bobbink *et al.*, 1996). However, since 1996 it has become clear that even phosphorus-limited systems may be vulnerable to species change in response to increased nitrogen loading, as illustrated in the UK long-term experiment. In phosphorus-limited systems, however, the general decrease in plant cover suggests some kind of toxicity effects, instead of the increased light competition found in Dutch chalk grasslands, as most important mechanism to species loss (Carroll *et al.*, 1997). Additionally, the lack in growth response and associated nitrogen uptake in phosphorus-limited conditions lead to increased nitrogen leaching, as nitrification rates are high in these well-buffered systems. Thus, there is no longer evidence to allow a higher critical load for phosphorus-limited systems.

Simulations with a dynamic ecosystem model for calcareous grassland have also confirmed the original Lökeberg-critical load. This calcareous grassland model describes the biomass, nitrogen and light dynamics at different levels of nitrogen deposition (Vergeer and Heil, 1998). As an indicator for the effects of enhanced nitrogen the proportion of tall grass (>50%) is used. In the model the growth and competitive interaction between tall grass (i.e. in this case *B. pinnatum*) and the other three functional groups (other graminoids, perennial forbs, annual forbs), the litter production, decomposition and nitrogen mineralization, soil nitrogen leaching, removal with management, nitrogen fixation by legumes and atmospheric nitrogen deposition are incorporated. Simulations with different nitrogen loads deposition has shown a strong increase in tall grass above 20 kg N ha^{-1} yr^{-1} (Mouissie and Heil, 1999). Using the percentage of tall grass as an indicator, the outcome of this model clearly affirm 15-25 kg N ha^{-1} yr^{-1} as a critical load for the transition of a species-rich vegetation into a grass-dominated (>50%) sward within 10-15 years. In view of all this evidence, the **critical load** is set back at the level of the Lökeberg paper, i.e. **15-25 kg N ha^{-1} yr^{-1}**, and qualified as **reliable**.

Increased nitrogen availablity is probably of major importance in a number of European calcareous grasslands (E1.2). In nitrogen-limited calcareous grasslands, an increased availability of nitrogen is indicated by an enhanced growth of some 'tall' grasses, especially of stress-tolerant species which have a slightly higher potential growth rate and efficient nitrogen utilization. The original species composition, obviously influences which of the grass species will increase at enhanced nitrogen inputs. The nitrogen retention in the system is very high with hardly any leaching and nitrogen mineralization could be enhanced by nitrogen inputs. On the other hand, in phosphorus-limited conditions, vegetation responses are slow and loss of species is most probably associated with some kind of negative effects of the changed soil conditions. Nitrogen mineralization and nitrification are increased, and in these nitrogen-saturated systems with shallow soils this will most likely result in high leaching losses. Most data from calcareous grasslands are from studies conducted in temperate, sub-atlantic calcareous grasslands (E1.26), and sites with relatively high atmospheric nitrogen deposition. There is a need for more studies from continental regions, and for experiments with low nitrogen doses from sites with low atmospheric deposition.

5.2.2 Non-Mediterranean dry acid and neutral closed grassland (E1.7)

The EUNIS class E1.7 groups all dry grasslands on acid and neutral soils with closed vegetation, in Atlantic or sub-Atlantic regions. This includes *Nardus stricta* swards (E1.71) that were combined with species-rich heaths in Bobbink *et al.* (1996), as well as *Agrostis-Festuca* (E1.72), *Deschampsia flexuosa* (E1.73), *Calamagrostis epigejos* (E1.74) and *Carex arenaria* (E1.75) grasslands, which all were classified together as neutral-acidic to acid grasslands in Bobbink *et al.* (1996). The critical load for dry species-rich *Nardus stricta* swards has been set to 10-15 kg N ha^{-1} yr^{-1}, and this was based mainly on the observation that rare species disappeared from these grasslands before high and dense growing grasses started to dominate over the vegetation (Bobbink *et al.*, 1996). These endangered species are extremely sensitive to acidification and ammonium accumulation (Roelofs *et al.*, 1996; De Graaf *et al.*, 1998). The input of acidifying nitrogenous deposition easily decreases the ANC and later on soil-pH in these grasslands with their weakly buffered soils in the cation exchange range, while deposited ammonium starts to accumulate once the pH significantly hampers nitrification (pH<4.5). Thus for these systems, species changes and loss of diversity are mainly associated with soil acidification by nitrogen inputs, and the critical load for acidity is most probably below the critical load for nitrogen eutrophication effects.

Tomassen *et al.* (1999) have added nitrogen (1, 5, 10, 15, 20, 40, 60 or 80 kg N ha^{-1} yr^{-1} as ammonium nitrate) to planted mesocosms ("pristine" sand taken from 5-25 cm depth of sod-cut *Nardus* grassland) in greenhouse conditions. The mesocosms had received clean water for 15 weeks before treatments were initiated. Within two years they found a gradual increase in biomass of *Danthonia decumbens* and *Deschampsia flexuosa* with increasing nitrogen between the lowest nitrogen level and addition of 20 kg N ha^{-1} yr^{-1}. However, there was a much stronger effect on both biomass and cover of grasses (*D. decumbens* and *D. flexuosa*) between 20 and 40 kg N ha^{-1} yr^{-1}. Nitrate concentrations in the soil pore water were strongly increased by nitrogen additions of 40, 60 and 80 kg N ha^{-1} yr^{-1}, and this indication for high nitrification is in accordance with the observation that the system was still buffered (no change in soil pH).

Berlin *et al.* (2000) compared species composition of grasslands between 1965 (background deposition 5-7 kg N ha^{-1} yr^{-1}) and 1990 (10-15 kg N ha^{-1} yr^{-1}) in 8 semi-permanent plots in Smaland, Southern Sweden, from which they could ascertain that no changes in the tradional management (summer mowing and aftermath grazing, no fertilizer use) had occurred during that period. They found very little changes on species richness and diversity, and no directional change in species composition. The only indications for the effects of increased nitrogen inputs were a significant increase in graminoid cover at the expense of forb cover and a decrease in number of forb species. As among forbs, rosette species decreased most, there may be an indication for increased light competition. Additionally, experimental nitrogen applications were carried out for 3 years in an *Agrostis capillaris* grassland and a *Festuca ovina* grassland (both E1.72) differing in initial fertility in the just-mentioned province of southern Sweden. This resulted in an increased above-ground biomass as well as proportional more graminoid biomass with an addition of 19 kg N ha^{-1} yr^{-1} for the *Festuca ovina* grassland, with lowest initial productivity (atmospheric load 13 kg N ha^{-1} yr^{-1}) (Fig. 5.2). No significant response to addition of 37 kg N ha^{-1} yr^{-1} for the *Agrostis capillaris* grassland (with current atmospheric deposition of 15 kg N ha^{-1} yr^{-1}) was found within 3 years (Berlin, 1998).

The effects of nitrogen additions (35-70-140 kg N ha^{-1} yr^{-1} as ammonium nitrate) have been studied in a *Festuca-Agrostis-Galium* grassland (E1.72) in Derbyshire in the UK (current atmospheric deposition (19-25/30 kg N ha^{-1} yr^{-1}) (Morecroft *et al.*, 1994; Carroll *et al.*, 1997; 2000; Johnson *et al.*, 1999; Lee *et al.*, 2000). The experiment was initiated in 1990 and is still running up to day. Already during the second year of treatment a significant decline of bryophytes (especially *Rhytidiadelphus*) was found in all nitrogen treatments. This may partly have been an artefact of the relatively high nitrogen concentrations of the nutrient solution that was applied, but still indicates a high sensitivity to nitrogen. Moreover, *Rhytidiadelphus* stems collected from the plots showed an increasingly higher nitrogen concentration with nitrogen addition (Carroll *et al.*, 1997; 2000). Despite the fact that shoot nitrogen concentrations, nitrate reductase activities and soil nitrogen mineralization rates clearly increased with enhanced inputs of nitrogen within three year (≥35 kg N ha^{-1} yr^{-1}), this did not significantly alter cover of higher plant species, diversity or species composition during the first 4 years (Morecroft *et al.*, 1994; Carroll *et al.*, 1997). From 1995 (sixth year of treament) on, there was a clear tendency to decreased overall cover of the vegetation as well as decreased herb cover with increasing nitrogen addition. By 1999, the vegetation had became more grassy in the high nitrogen treatments (Carroll *et al.*, 1997, Lee *et al.*, 2000). In 1995 an additional experiment with a factorial nitrogen x phosphorus design was initiated with nitrogen additions of 35 and 140 kg N ha^{-1} yr^{-1}. The results from the nitrogen only treatments corroborate the results found in the longer term experiment, with a clear decrease of *Festuca ovina*, *Luzula campestris* and *Potentilla erecta*, and an increase of *Nardus stricta* (Lee and Caporn, 2001). This experiment also confirmed that these grasslands were strongly limited by phosphorus rather than by nitrogen (Lee and Caporn, 2001). As found in the phosphorus-limited calcareous grassland (section 5.2.1), nitrogen accumulation was not significant for any of the treatments, and calculated nitrogen leaching was significant at addition rates ≥35 kg N ha^{-1} yr^{-1}.

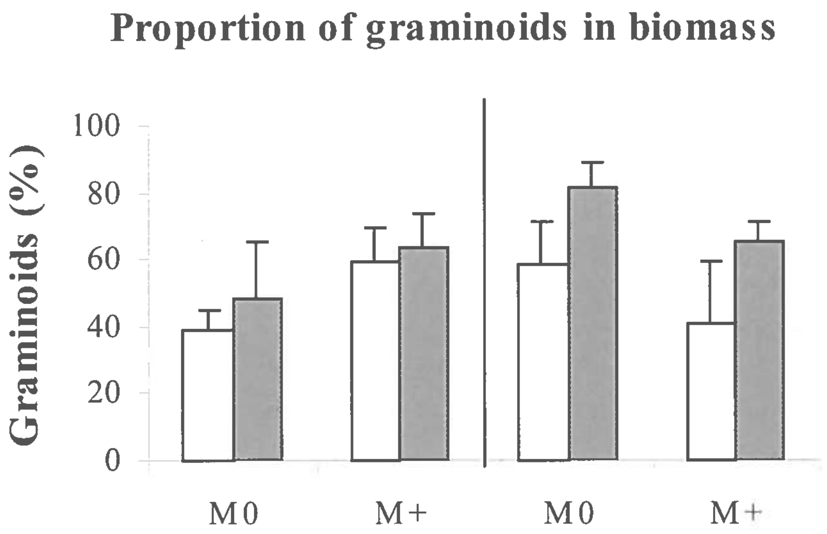

Figure 5.2. Proportion of graminoids after 3 years of adding 19 kg N ha^{-1} yr^{-1} in two Non-Mediterranean dry acid closed grassland sites (E1.7) in S. Sweden (Berlin, 1998). Hatched colums: with nitrogen addition; M0 = normal mowing; M+ = more intensive management; left part = *Agrostis* grassland (fertile site); right part: *Festuca ovina* site (less fertile site).

Phosphomonoesterase activity in the soil of the long-term experiment was enhanced already at the lowest nitrogen addition (35 kg N ha^{-1} yr^{-1}) (Carroll *et al.*, 1997). Additionally, microcosm studies with soils from this field experiment (after 7 years of nitrogen addition) showed increased root-surface phosphomonoesterase activity on the roots of *Agrostis stolonifera* seedlings already from 35 kg N ha^{-1} yr^{-1} on. Thus the increased nitrogen addition eventually also affected the phosphorus budget/economy in these severely phosphorus-limited system. A similar experiment with soils that had received nitrogen addition for only one and a half years showed no effect, indicating that the effect was the result of long-term changes in the soil (Johnson *et al.*, 1999).

These studies indicate that many of these dry acid grasslands are also sensitive to nitrogen eutrophication. In nitrogen-limited systems, there is evidence for a significant increase in grass dominance leading to increased competition and exclusion of smaller species like rosette forbs from 15-20 kg N ha^{-1} yr^{-1} or higher at relatively low atmospheric nitrogen deposition (e.g. Berlin *et al.*, 2000; Tomassen *et al.*, 1999). Most probably, this is enhanced by a positive feedback to litter quality and soil nitrogen mineralization, but this has not been studied yet. In phosphorus-limited systems, the response of the vascular vegetation is much slower, but there is a clear and rapid effect on bryophytes (Carroll *et al.*, 2000). On top of these effects, soil nitrogen and phosphorus economy is significantly impacted, which among other effects most probably results in significant leaching of nitrogen (Carroll *et al.*, 1997; Johnson *et al.*, 1999).

Taking into account that the UK results may be influenced by the already high deposition (19-25/30 kg N ha^{-1} yr^{-1}), and the effects found in Sweden, the critical load for **dry acid grasslands**

lands is set at **10-20 kg N ha^{-1} yr^{-1}** and qualified as **quite reliable**. There is clearly a need for more field studies in different countries, especially experiments with lower nitrogen addition in sites with low atmospheric deposition.

5.2.3 Non-Mediterranean dry acid and neutral open grassland, including inland dune grassland (E1.9)

The EUNIS class E1.9 contains all dry grasslands on acid and neutral soils with open vegetation, in non-Mediterranean regions. Grasslands of this class were not treated in Bobbink *et al.* (1996), because of lack of data. However, inland dune grasslands (E1.94 & E1.95), both pioneer and more mature communities, are also part of this EUNIS class. These acidic inland dune grasslands are in species composition and ecological functioning comparable with coastal (grey) dune grasslands (see par. 8.2.1 for details). Because of this similarity, the critical load for **inland dune pioneer (E1.94)** and **inland dune siliceous grasslands (E1.95)** has been set at the same level as that of coastal grey dune grasslands (**10-20 kg N ha^{-1} yr^{-1}**). This estimation is completely done on the basis of **expert judgement** and there is thus a significant need for research in these systems.

5.3 Mesic grasslands (E2)

5.3.1 Low and medium altitude hay meadows (E2.2)

Grasslands situated at low or medium altitude and with a management of hay making (E2.2) are the only class of mesic grasslands for which field experiments with nitrogen only treatments in realistic doses have been conducted. The Park Grass experiment at Rothamsted, UK, has been in existence since 1856 (Williams, 1978; Dodd *et al.*, 1994). Nitrogen has been applied as ammonium sulphate or sodium nitrate (48 kg N ha^{-1} yr^{-1}) to plots in this mesic low altitude hay meadow (E2.2). In nitrogen-treated plots, the vegetation has become dominated by a few grasses such as *Alopecurus pratensis*, *Arrhenatherum elatius*, *Holcus lanatus* or *Agrostis* ssp. Species diversity was correlated negatively with total biomass and soil acidity. Ammonium sulphate, through its acidifying effects, has reduced the diversity of higher plant and bryophyte species significantly more than other nitrogen forms (Goulding *et al.*, 1998; Virtanen *et al.*, 2000). In addition, Beltman and Barendreft (2002) clearly showed an increase in biomass and a 25% reduction in species diversity after long-term (>4 yrs) additions of nitrogen (100 kg N ha^{-1} yr^{-1}) in a hay meadow along the river Rhine in the Netherlands. Formerly, these grasslands were combined with other neutral-acidic grasslands (e.g. dune grasslands, wet oligotrophic stands) and the critical load has been set at 20-30 kg N ha^{-1} yr^{-1} and considered as quite reliable (Bobbink *et al.*, 1996). Because of the split of this group of grasslands under the EUNIS system, and the very few studies with low doses in E2.2, we suggest the critical load for **low and medium altitude hay meadows (E2.2)** is set at **20-30 kg N ha^{-1} yr^{-1}**, based upon **expert judgement**, because of the hay management which yearly removes nitrogen from the system. There is thus a strong need for field addition studies in different countries, especially in regions with low atmospheric deposition.

5.3.2 Mountain hay meadows (E2.3)

Many semi-natural grassland types occur in the montain regions across Europe, containing many rare and endangered plant and animal species (e.g. Ellenberg, 1996). It is important to emphasize the effects of nitrogen eutrophication in these mountain grasslands (E2.3), too, because nitrogen deposition has certainly increased in mountainous regions in central Europe (e.g. Hesterberg *et al.*, 1996; Rihm, 1996). However, relevant studies are not available to establish a critical load with any reliability for these grasslands, although it may be expected that: (i) several of these grassland types are sensitive to nitrogen (Ellenberg, 1996), and (ii) the critical load for nitrogen is probably lower than for lowland dry or mesic grasslands, because of the shorter vegetation period or the generally poorer soils. The effects of nitrogen in montane grasslands has been identified as a major gap in knowledge (Bobbink *et al.,* 1992a; Grennfelt and Thörnelöf, 1992; Hornung *et al.*, 1995). Unfortunately, this gap still exists and the critical load (**10-20 kg N ha^{-1} yr^{-1}**) for **mountain hay meadows (E2.3)** is based upon **expert judgement** only (Tab. 5.1).

5.4 Seasonally wet and wet grasslands (E3)

5.4.1 Moist or wet oligotrophic grassland (E3.5)

Moist or wet oligotrophic grassland habitats (E3.5) are characterized by oligotrophic and moist to wet peaty soil conditions. This class consists mostly of hay meadows under original agricultural management and are, especially, rich in typical plant and animal species. Two subcategories have been distinguished in EUNIS, namely (i) *Molinia caerulea* meadows (E3.51; “litter meadows” or “fen meadow”) and, (ii) heath meadows and humid *Nardus stricta* swards (E3.52). Because of their long traditional land-use with low additional inputs of nutrients, it is likely that these grassland communities are sensitive to extra nutrient inputs. Several fertilization experiments have demonstrated both nitrogen, phosphorus or even co-limitation of these elements in these wet oligotrophic grasslands (e.g. Vermeer, 1986; Egloff, 1987; Spink *et al.*, 1998; Van Duren *et al.*, 1998; Olde Venterink *et al.*, 2001). In the case of nitrogen limitation, productivity of the grasses, especially the dominant *M. caerulea*, had increased and species diversity declined (e.g. Vermeer, 1986). However, almost all of the studies performed in moist or wet oligotrophic grasslands have been done with high to very high loads (>100 kg N ha^{-1} yr^{-1}) or on a too short time span with respect to the critical load approach.

Fortunately, the impacts of nitrogen additions on species richness have been quantified in flower-rich, oligotrophic wet hay meadows (E3.51) in Somerset (UK) (Mountford *et al.*, 1994; Tallowin *et al.*, 1994; Kirkham *et al.*, 1996). Nitrogen additions of 25 kg N ha^{-1} yr^{-1} or higher (with estimated additional atmospheric load of 15-25 kg N ha^{-1} yr^{-1}) for six years significantly reduced the number of species, while several grasses increased in dominance (*Lolium perenne*, *H. lanatus* and *Bromus hordeaceus*). The number of forbs, characteristic of these old meadows, declined sharply and some, for example, *Cirsium dissectum*, *Lychnis flos-cuculi* and *Lotus pedunculatus* disappeared from nitrogen-treated plots.

More recently, a nitrogen manipulation experiment has examined the effects of ammonium sulphate (10 or 20 kg N ha^{-1} yr^{-1}) or sodium nitrate (20 kg N ha^{-1} yr^{-1} only) addition to an upland heath meadow in Wales (E3.52) (Emmett *et al.*, 2001). This site, at 600 m a.s.l., was overgrazed in the 1970s and 1980s, resulting in degradation of *Calluna* moorland to a sward dominated by *Nardus stricta*, *Vaccimium myrtilis* and *Festuca* (E3.52). The treatments were applied to paddocks which had different rates of experimental sheep grazing from 1989 to the start of the experiment in 1997. After four years of treatment, relatively small effects were observed on the vascular plants, although there was evidence of increased cover of *Festuca* cover in the lightly grazed paddock, and greater frost injury (in nitrate treatment only) to *Vaccinium*, which may be linked to earlier budbreak in the spring. The lack of response observed in palatable grasses in the heavily grazed paddock may reflect selective grazing by sheep. However, there was a clear decline in moss cover in response to increased nitrogen addition, with *Rhytidiadelphus loreus* and *Pleurozium schreberi* being the most response species, and of lichen cover in the lightly grazed paddocks.

Measurements of soil water chemistry at this site showed significant leaching losses in the control plots which represented 25% of inorganic inputs, and which were increased from 5 to 7 kg N ha^{-1} yr^{-1} in the 20 kg N ha^{-1} yr^{-1} treatment. These high leaching rates suggested that the site is already above the critical load for nitrogen as a nutrient. In the ammonium treatments only, there was also increased base cation leaching and decreased pH. However, there were no significant treatment effects on mineralization or nitrification rates. Root bioassay experiments with *Nardus* indicated relatively low overall rates of nitrogen uptake and relatively high rates of phosphorus uptake overall, suggesting the site is phosphorus rather than nitrogen-limited, although phosphorus addition experiments do not support this conclusion (Emmett *et al.*, 2001).

The current atmospheric deposition at this site is estimated to be 20 kg N ha^{-1} yr^{-1}. To assess the impacts of lower deposition rates, Jones and Ashenden (2000) applied a range of deposition rates above and below the site estimate (2, 10, 20 and 55 kg N ha^{-1} yr^{-1} as ammonium nitrate) to mesocosms taken from the site in a greenhouse experiment. To assess possible interactions with grazing pressure, simulated clipping was also applied at three levels. Within 1-2 years, there were strong effects of nitrogen treatments below 20 kg N ha^{-1} yr^{-1} in increasing the cover of certain moss and lichen species, but only with heavy clipping, presumably because of the lower competition from vascular plants. Subsequent data (Jones *et al.*, 2002a) show the emergence of different optima for bryophyte species – that for *Racomitrium lanuginosum* and *Polytrichum juniperinum* lying below 10 kg N ha^{-1} yr^{-1}, while that for *Hypnum jutlandicum* lies around 20 kg N ha^{-1} yr^{-1} (Fig. 5.3). Although the results show an increase in fine grass cover and a decrease of *Nardus* with increasing nitrogen addition, these effects occur primarily between 20 and 55 kg N ha^{-1} yr^{-1}. In contrast to the calcareous mesocosms in the parallel experiment and the results from the field site, there was no evidence of effects of nitrate leaching in the first two years of the experiment (Jones *et al.*, 2002a).

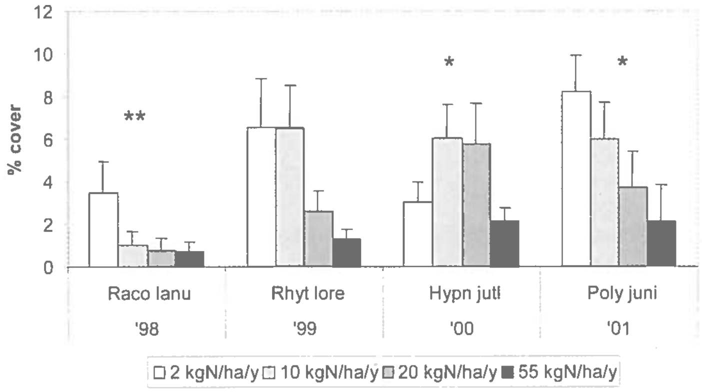

Figure 5.3. Cover of moss species in *Nardus stricta* grassland (E3.52) mesocosms exposed since 1997 to four nitrogen treatments (2, 10, 20, and 55 kg N ha^{-1} yr^{-1}; from left to right). The mosses are *Racomitrium lanuginosum, Rhytidiadelphus loreus, Hypnum jutlandicum* and *Polytrichum juniperinum*. The year denotes when the optimum nitrogen level for each species became apparent (Jones *et al.*, 2002a).

It is concluded that several moist or wet oligotrophic grasslands (E3.5) of high conservational value are sensitive to nitrogen eutrophication. Increases in dominant grasses and decreases in diversity have been observed with increased levels of nitrogen inputs. Furthermore, interactions with phosphorus limitation are obvious. The study of a degraded upland heath meadow (E3.52) provides evidence of response of bryophyte cover to relatively low levels of N deposition, and suggests increases in leaching and acidification above 20 kg N ha^{-1} yr^{-1}. In view of these UK studies, the critical load of **moist to wet oligotrophic grasslands** is set at **10-20 kg N ha^{-1} yr^{-1}** for ***Nardus stricta* swards (E3.52)** and considered as **quite reliable**. However, the critical load of the hardly studied ***Molinia caerulea* meadows (E3.51)** is estimated as somewhat higher (**15-25 kg N ha^{-1} yr^{-1}**), but till present only based upon expert judgement.

5.5 Alpine and subalpine grasslands (E4)

5.5.1 Alpine and subalpine acid or calcareous grasslands (E4.3 & E4.4)

Many natural grassland types occur in the alpine and subalpine regions of European mountains, containing a large proportion of the biodiversity of that area. It is important to emphasize the effects of nitrogen inputs in these grasslands, too, because nitrogen deposition starts to increase in these remote regions. However, the impacts of nitrogen deposition are hardly studied in these grasslands, although it may be expected that; (i) several of these grassland types are sensitive to nitrogen and (ii) critical load for nitrogen is probably lower (see table 10.2) than for lowland dry grasslands (E1), because of the very nutrient-poor and thin soils. The impacts of nitrogen in alpine and subalpine grasslands has been identified as a major gap in knowledge (Hornung *et al.*, 1995; Bobbink *et al.,* 1996). Only one experiment with low-dose nitrogen treatments has been performed in this important group of grasslands. It has been shown that an alpine grassland in Switzerland significantly responded to nitrogen additions (4 years; 20 kg N ha^{-1} yr^{-1}); the biomass of the vegetation and the leaf area index (LAI) were both doubled or more after nitrogen application compared with the control situation. Futhermore, no effects of enhanced CO_2 were found (Körner, 1999). This experiment clearly showed the sensitivity of alpine and subalpine grassland to nitrogen, but data are too scarse to establish a quite reliable critical load. The critical load for **alpine and subalpine acid** or **calcareous grasslands (E4.3 and E4.4)** is thus still based upon **expert judgement** and set as **10-15 kg N ha^{-1} yr^{-1}**.

5.5.2 Moss and lichen dominated mountain summits, ridges and exposed slopes (E4.2)

Within the EUNIS system, an important sub-category of alpine and subalpine grasslands (E4) are communities without extensive snow cover which are dominated by moss and lichen species. This constitutes class E4.2. Since these communities are nutrient limited and many moss and lichen species are highly responsive to increased nitrogen deposition, it is likely that they are sensitive and should be assigned a low critical load. However, the only substantive evidence to support a specific critical load is for *Racomitrium* heath, which is found on mountain summits in Britain and in montane areas of arctic and sub-arctic zones.

In the UK, there has been a serious decline in the distribution of *Racomitrium* heath in recent decades, with replacement by grass communities. Increasing rates of nitrogen deposition may be one of the main factors involved in this deterioration of *Racomitrium* heath (Thompson and Baddeley, 1991), although changes in grazing pressure are also likely to be a significant factor. However, only recently has evidence from experimental manipulation studies become available to support this interpretation. Pearce and Van der Wal (2002) set up an experiment in the north-east Scottish Highlands within montane *Racomitrium lanuginosum-Carex bigelowii* heath in which plots on the summit were subject to experimentally increased low (10 kg N ha^{-1} yr^{-1}) and high (40 kg N ha^{-1} yr^{-1}) nitrogen input in both forms (NO_3^- and NH_4^+) for a period of two summer seasons. Current deposition was estimated to be 15-18 kg N ha^{-1} yr^{-1}. *Racomitrium* was shown to be extremely sensitive to even the low rates of nitrogen deposition, responding with a raised tissue nitrogen content, a severe inhibition of nitrate reductase activity, increased leakage of potassium from the shoots and shoot growth rates which were less than 50% of the control. Of all the measurements, only nitrate reductase showed a clear distinction between NO_3^- and NH_4^+ application. After only two growing seasons, Pearce and Van der Wal (2002) also demonstrated how quickly *Racomitrium* is replaced by graminoid species; *Racomitrium* cover in the low treatment was reduced by 31%, while graminoid cover increased by 57%. The results may reflect both a toxic effect and increased competition for light from graminoids, which can utilise the excess nitrogen.

The high sensitivity of *Racomitrium lanuginosum* to nitrogen deposition is supported by the results of a glasshouse experiment using monoliths taken from a subalpine/montane grassland in central Wales, in which nitrogen deposition and simulated grazing have been manipulated over 4 years (Jones *et al.*, 2002a). The nitrogen treatments applied were both above and below the 20 kg N ha^{-1} yr^{-1} at the source field site. *Racomitrium* only had a significant presence in monoliths subjected to heavy simulated grazing, confirming its sensitivity to competition for light by grasses. In these monoliths, there was a significant effect of nitrogen application within one year, with the cover of *Racomitrium* falling from 10% at 2 kg N ha^{-1} yr^{-1} to 2% or less at 10 kg N ha^{-1} yr^{-1} and above. However, a three-year manipulation study by Jónsdóttir *et al.* (1995), in which low levels of nitrogen addition (4 kg N ha^{-1} yr^{-1}) were applied to a *Racomitrium-Carex* moss-heath in Iceland with a background deposition of about 2 kg N ha^{-1} yr^{-1} over a period of three years, found only small non-significant increases in *Racomitrium* growth and shoot density. These were associated with increases in nitrogen concentrations when reduced, but not oxidised, nitrogen was added. It was suggested that the small response to the low deposition rates used in this experiment was associated with growth limitation by other factors.

Although experimental evidence is limited to three studies with contrasting results, the fact that large and rapid changes in growth or cover have been observed in two of these studies at deposition rates of 10 kg N ha^{-1} yr^{-1} or lower suggests that a critical load for **moss and lichen dominated mountain summits, ridges and exposed slopes (E4.2)** should be set at **5-10 kg N ha^{-1} yr^{-1}** and described as **quite reliable**. A summary of the empirical critical loads for grassland and tall forb habitats is given in table 5.1.

Table 5.1. Empirical critical loads for the effects of excess nitrogen upon grassland and tall forb habitats (EUNIS class **E**). ## reliable, # quite reliable and (#) expert judgement.

Ecosystem type	**EUNIS-code**	**kg N ha^{-1} yr^{-1}**	**Reliability**	**Indication of exceedance**
Sub-atlantic semi-dry calcareous grassland	E1.26	15-25	##	Increase tall grasses, decline in diversity, increased mineralization, N leaching
Non-mediterranean dry acid and neutral closed grassland	E1.7	10-20	#	Increase in graminoids, decline typical species
Inland dune pioneer grasslands	E1.94	10-20	(#)	Decrease in lichens, increase biomass
Inland dune siliceous grasslands	E1.95	10-20	(#)	Decrease in lichens, increase biomass, increased succession
Low and medium altitude hay meadows	E2.2	20-30	(#)	Increase in tall grasses, decrease in diversity
Mountain hay meadows	E2.3	10-20	(#)	Increase in nitrophilous graminoids, changes in diversity
Moist and wet oligotrophic grasslands	E3.5			
• *Molinia caerulea* meadows	E3.51	15-25	(#)	Increase in tall graminoids, decreased diversity, decrease of bryophytes
• Heath (*Juncus*) meadows and humid (*Nardus stricta*) swards	E3.52	10-20	#	Increase in tall graminoids, decreased diversity, decrease of bryophytes
Alpine and subalpine grasslands	E4.3 and E4.4	10-15	(#)	Increase in nitrophilous graminoids, biodiversity change
Moss and lichen dominated mountain summits	E4.2	5-10	#	Effects upon bryophytes or lichens

6. Effects of nitrogen deposition on mire, bog and fen habitats (EUNIS class D)

6.1 Introduction

The EUNIS class D includes a wide range of wetland systems that have their water table at or above soil level for at least half of the year, dominated by either herbaceous or ericoid vegetation (Davies and Moss, 2002). Nutrient budgets in wetland ecosystems are characterized by inputs and outputs of nutrients via groundwater and surface water, and tightly linked with local hydrology. The extent to which these systems receive and lose nutrients with in- and out-flowing water determines for a large part their sensitivity to excess nitrogen. Very little or little effects of nitrogen enrichment is to be expected in several open wetland systems, like (riverine) reed marshes or sedge beds (EUNIS D5 and D6) (e.g. Morris, 1991). A larger impact of atmospheric nitrogen deposition is expected in systems with a closed nitrogen cycle. This is, of course, especially clear in the case of ombrotrophic raised bogs, which receive all their major nutrients from the atmosphere.

Several wetland types (e.g. D1, D2, D4) are characterized by an (partly) incomplete decomposition of the plant litter, resulting in peat formation (Mitsch and Gosselink, 2000; Malmer *et al.*, in press). The capacity for peat formation is a major component in the nutrient balance of these wetland systems, and is in many situations strongly linked to the presence of peat mosses (*Sphagnum* spp.) (Van Breemen, 1995). These peat mosses are capable of efficiently accumulating nitrogen supplied to the moss surface (e.g Clymo, 1970; Woodin, 1986). Their litter decays much slower than that of other plants. During the slow decomposition of litter (residence time of carbon up to 50-100 years) in the top layer of the peat, nitrogen is conserved in the peat. During peat accumulation, mineral storage actively removes nitrogen and other nutrients from the moss layer and rhizosphere, and is thus not available for plant life (e.g. Mitsch and Gosselink, 2000; Malmer and Wallén, in press).

In recent years, the concern over high atmospheric CO_2 levels has generated an increasing attention to bogs. Bogs and peatlands are large reservoirs of carbon, and the sequestration of carbon in peat depends on the ratio between primary production and decomposition of plant, mainly bryophyte, material (Risager, 1998; Mitsch and Gosselink, 2000). Both increased carbon and nitrogen availability may increase primary production in pristine (nutrient-poor) conditions, while higher availability of carbon and nitrogen should have opposing effects on the C:N ratio of litter and thus hypothetically also on peat decomposition rates. However, the peat mosses (*Sphagnum* spp.), like mosses in other systems (e.g. grasslands, forests), have proved to be sensitive to increasing nitrogen availability, and react with decreased growth to high nitrogen doses. Thus, the interactions between carbon and nitrogen under expected global change are still not clear.

This sensitivity of peat mosses may also have implications for other systems that are characterized by a large bryophyte component, but which lack reliable evidence for a critical load up to date.

6.2 Raised and blanket bogs (D1)

a) Introduction

Ombrotrophic (raised) bogs, which receive all their nutrients from the atmosphere, are particularly sensitive to airborne nitrogen loads. These bogs are systems of acidic, wet areas and are very common in the boreal and temperate parts of Europe. Because of the abiotic conditions, decomposition rates are slow, favouring the development of peat. In Western Europe and high northern latitudes typical plant species include bog-mosses (*Sphagnum* species), sedges (*Carex*, *Eriophorum*) and heathers (*Andromeda*, *Calluna*, and *Erica*). Within the EUNIS system, these communities are represented by class **D1** (raised and blanket bogs) for which the criterion is that the water supply is continuously or primarily rainwater. This is entirely consistent with the original description of these plant communities by Bobbink *et al.* (1996). For the purpose of critical load definition, there is no basis on which to differentiate raised and blanket bogs.

b) Effects on moss growth and species composition

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 and 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}. Several studies in British bogs have shown that increased supplies of nitrogen are rapidly absorbed and utilized by bog-mosses (*Sphagnum*), reflecting the importance of nitrogen as a nutrient and its scarcity in unpolluted regions. The high nitrogen loadings are, however, supraoptimal for the growth of many characteristic *Sphagnum* species, as demonstated by restricted development in growth experiments and transplantation studies between clean and polluted locations. In areas with high nitrogen loads such as the Pennines the growth of *Sphagnum* is in general lower than in unpolluted areas (Lee and Studholme, 1992). After transplantation of *Sphagnum* from an 'unpolluted' site to a bog in the southern Pennines, a rapid increase in nitrogen content from ca. 12 to 20 mg g^{-1} dry weight was observed (Press *et al.*, 1988). Furthermore, a large increase in nitrogen-containing amino acids (arginine) in the shoots of these bog-mosses was found after application of nitrogen, indicating a nutritional imbalance of the species.

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).

The importance of taking ambient atmospheric deposition into account when interpreting the results of field manipulation experiments is also demonstrated by Berendse *et al.* (2001), who analysed the results of the Bog Ecosystem Research Initiative (BERI), with sites in Sweden, Finland, the Netherlands and Switzerland. They found that adding 30 kg N ha^{-1} yr^{-1} for 3 growing seasons (50 kg N ha^{-1} yr^{-1} in the Netherlands) increased *Sphagnum* nitrogen concentrations at all sites. However, this treatment decreased *Sphagnum* production only at the two sites with the highest atmospheric deposition (Switzerland and the Netherlands, with 18 and 39 kg N ha^{-1} yr^{-1} atmospheric deposition, respectively), and these sites also had the highest nitrogen concentrations at the start of the experiment.

In another Swedish study (Gunnarsson and Rydin, 2000; Nordin and Gunnarsson, 2000), a three-year field manipulation experiment used a range of nitrogen addition rates at two mires with differing current atmospheric loads. N:P ratios and experimental results confirmed the idea that *Sphagnum* growth at the site in northern Sweden, with very low ambient deposition, was nitrogen limited, while at the southern site it was phosphorus limited (Gunnarsson and Rydin, 2000). After two years, the experimental addition of nitrogen increased free amino acid concentrations in *Sphagnum* capitula, whereas it decreased *Sphagnum* extension growth at deposition rates of 30 kg N ha^{-1} yr^{-1} and above. At these low nitrogen deposition rates (lower than 7-11 kg N ha^{-1} yr^{-1}), however, there was no correlation observed between *Sphagnum* total amino-acid nitrogen concentrations and growth rates (Nordin and Gunnarsson, 2000). After three years of treatment, biomass production and length increment decreased, while capitulum dry mass increased, with increasing nitrogen addition. This experiment included a treatment of 10 kg N ha^{-1} yr^{-1}, but the biomass and length responses showed a steady decline from the control treatment, especially in areas dominated by *Sphagnum rubellum*. Thus, even if the control and 10 kg N ha^{-1} yr^{-1} treatments are not statistically distinguished, these data suggest a threshold for effects on this bog community below 10 kg N ha^{-1} yr^{-1}.

Differential effects on the growth of different *Sphagnum* species have also been demonstrated in several experimental studies. For example, in a recent controlled environment experiment, Risager (1998) examined the growth responses of *Sphagnum fallax* to different forms of nitrogen (NO_3^-, NH_4^+ or NH_4NO_3) at low addition rates (0, 5, 10 and 20 kg N ha^{-1} yr^{-1}). Growth of *S. fallax* was significantly stimulated by application of NH_4^+ (especially with 5 and 10 kg NH_4-N ha^{-1} yr^{-1}), but this growth stimulation was, logically, not reflected in the tissue nitrogen concentration. Sole NO_3^- additions did not influence the growth of this species in this study. In addition, she investigated the responses of *S. magellanicum* after addition of NH_4NO_3-N in a comparable study with respect to the nitrogen levels, but without the differentiation between reduced or oxidised nitrogen. S. *magellanicum* showed no increase in length, but addition of nitrogen decreased the production density of capitula. In contrast to *S. fallax*, the tissue nitrogen concentration of *S. magellanicum* increased with increasing addition of nitrogen. In both cases, uptake of NO_3^- was considerably lower than for NH_4^+. Risager (1998) concluded that the form of nitrogen is also important in species response, and that increased nitrogen availability may cause shifts in species composition in favour of *S. fallax*.

A German nitrogen addition experiment has shown such effects on species composition in an established *Sphagnum* community (Lütke Twenhöven, 1992) in the field. In this two year study, the *Sphagnum* species responded in different ways to the nitrogen type applied, depending on micro-habitat (soil moisture content). *S. fallax* was significantly promoted by the addition of both nitrate and ammonium (10 kg N ha^{-1} yr^{-1} with an estimated ambient atmospheric deposition of 5 kg N ha^{-1} yr^{-1}) in bog hollows, but only partly on the bog lawns. This resulted in *S. fallax* outcompeting *S. magellanicum* in the hollows and, if water supply was sufficient, also on the lawns. However, on the hummocks in the bog, nitrate and, to a lesser extent, ammonium reduced the growth of both these species. The results of Gunnarsson and Rydin (2000) also suggest that lawn communities are less vulnerable to increased nitrogen deposition than are hummock communities. Because of the differences in vegetation structure, the rate of supply of nitrogen to a hummock community dominated by dwarf shrubs is about 40% greater than to a lawn community (Bobbink *et al.*, 1992b; Malmer and Wallén, 1999).

The importance of competition between moss species is also indicated by the recent study of Mitchell *et al.* (2002) at rather higher rates of nitrogen deposition. This experiment examined the effect of addition of 30 kg N ha^{-1} yr^{-1} *in situ* to a cutover bog in the Jura mountains of Switzerland, where ambient deposition is estimated to be 15 kg N ha^{-1} yr^{-1}. The normal pattern of succession in restoration of these sites is that keystone species such as the moss *Polytrichum strictum* create favourable microenvironments for establishment of *S. fallax* and thence of the typical bog vegetation. However, three years of study showed a cumulative differential height growth in response to added nitrogen deposition between overgrowing *P. strictum* and *Sphagnum* capitula, which might prevent this typical regeneration process in central European bogs. Thus, biomass production of *P. strictum* was almost doubled because of nitrogen addition, whereas production of *S. fallax* decreased with almost 50% in the study period.

c) Effects on competition between mosses and vascular species

A national survey of Danish ombrotrophic bogs has shown a decline of the original bog vegetation together with an increase of more nitrogen demanding grass species (such as *Molinia caerulea* and *Deschampsia flexuosa*) and trees (such as *Betula pubescens*) in areas with wet ammonium (NH_4^+) deposition loads higher than 10-15 kg N ha^{-1} yr^{-1} (Aaby, 1994), together with dry ammonia (NH_3) deposition, which is rapidly deposited to ombrotrophic mires. An increased growth of trees, particularly *Pinus sylvestris,* has also been demonstrated for many ombrotrophic sites in southern Sweden (Åberg, 1992; Gunnarsson *et al.,* 2002), as well as a reduced *Sphagnum* cover (Malmer and Wallén, 1999). For a bog in southern Sweden in a region with an ambient nitrogen deposition of 7-9 kg N ha^{-1} yr^{-1}, Gunnarsson *et al.* (2002) showed that the total number of species (vascular plants + mosses) per plot did not change much from 1954 to 1997. However, there were large changes in species composition, which were indicative particularly of a drier mire surface and an increased availability of nitrogen. The increase in trees may also have triggered further changes in the plant cover. Also, on a mire in central Sweden in a region with lower nitrogen deposition, species numbers decreased over a similar period (Gunnarsson *et al.,* 2000). However, this decrease was mainly found in sites with pH>5.0. On sites with lower pH, as at the site in southern Sweden, species numbers hardly decreased at all.

Such observations suggest that enhanced nitrogen deposition can influence the competitive relationships between mosses and vascular plants in nutrient-deficient vegetation such as bogs. For light the competitive interactions between *Sphagnum* and vascular plants are symmetric, while for mineral nutrients they are partly asymmetric, as *Sphagnum* mosses rely on atmospheric supply only while the vascular plants also rely on mineralization (Malmer *et al.,* 1994). Thus, in a field experiment in southern Sweden (ambient N deposition rate 7-9 kg ha^{-1} yr^{-1}) a supply of N and P (20 and 4 kg ha^{-1} yr^{-1}, respectively) only affected the growth of the mosses if it was supplied to the surface, and only affected the vascular plants if supplied below the moss surface. The mosses were obviously nitrogen saturated and the target vascular plant, *Narthecium ossifragum*, had a N/P-ratio as high as about 40 (Malmer *et al.*, in press).

These responses were also demonstrated experimentally by Heijmans *et al.* (2001), who studied the effect of added nitrogen deposition (50 kg N ha^{-1} yr^{-1}) at an ambient atmospheric deposition of the same magnitude on peat monoliths taken from a mire in the northern Netherlands. The N:P ratio in the mosses indicated phosphorus limitation, which corresponded with the observed lack in growth response to nitrogen addition. The mosses were still able to capture a large part of the deposited nitrogen (Heijmans, 2000) and three years after the start of the experiment, all species showed increased nitrogen concentrations. The mosses showed decreased length increment, but no changes in dry matter production, indicating an effect on moss morphology (Heijmans *et al.*, 2001). The fertilized mesocosms showed a significantly higher biomass production of vascular plants, but only the cover of *Vaccinium oxycoccus* was significantly increased, due to high experimental variability. There was a negative relationship between litter (also increased by nitrogen) and vascular plant cover on one hand and *Sphagnum* on the other hand (Heijmans *et al.*, 2001), suggesting that *Sphagnum* growth might also be limited by increased shading.

The significance of competition for light on the response of *Sphagnum* was also demonstrated by the study of Hogg *et al.* (1995) in a small valley mire near York in the UK. The growth of the mosses *Sphagnum palustre* and *Sphagnum fimbriatum* was reduced by 50% by the addition over two years of 12 kg N ha^{-1} yr^{-1}; a rate which was probably comparable to ambient deposition. Where *Sphagnum* was growing poorly and the dominant grass species *Molinia caerulea* was abundant, adding nitrogen had no effect, but cutting *M. caerulea* in the summer was beneficial to *Sphagnum*, re-invigorating its growth.

The effects of the supply of extra nitrogen on the population ecology of *Drosera rotundifolia* were studied in a 4-year fertilization experiment in Swedish ombrotrophic bogs using a range of deposition rates (Redbo-Torstensson, 1994). It was demonstrated that experimental applications of 10 kg N ha^{-1} yr^{-1} (as NH_4NO_3 at an ambient deposition of 5 kg N ha^{-1} yr^{-1}) significantly reduced the survivorship of the plants after 4 years, while additions of 20 and 40 kg N ha^{-1} yr^{-1} had this effect after one year, and also negatively affected flowering after two years (Fig. 6.1). The decrease in the density of the population of the characteristic bog species *Drosera* was associated with increased density of tall species such as *Eriophorum* and *Andromeda*, which resulted in enhanced competition for light.

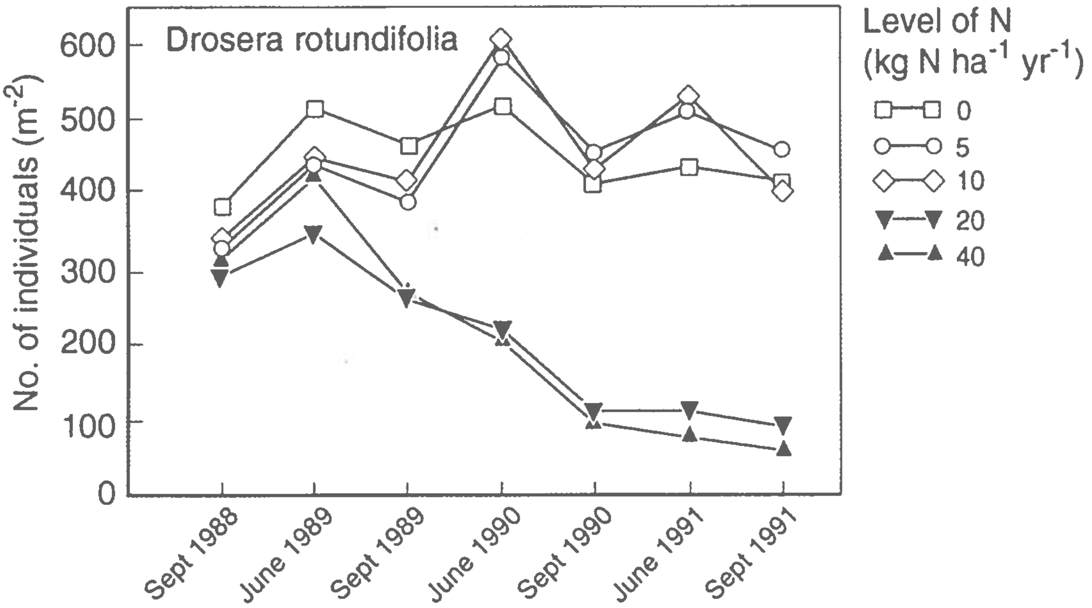

Figure 6.1. Numbers of individuals of *Drosera rotundifolia* in an ombrotrophic raised bog (D1) near Stockholm (Sweden) during 4 years of nitrogen additions (Redbo-Torstensson, 1994).

d) Effects on peat and peat water chemistry as evidence of nitrogen saturation

The influence of nitrogen deposition entering the moss layer on nitrogen dynamics in the underlying peat was investigated in experiments on a raised bog, the Moidach More, in the north-east of Scotland (Williams *et al.*, 1999b; Williams and Silcock, 2000). The bryophytes *Sphagnum capillifolium* and *Sphagnum recurvum* were chosen because they colonise contrasting sites; *S. capillifolium* is a hummock-forming species whereas *S. recurvum* occupies hollows and pools. The addition of 30 kg N ha^{-1} yr^{-1} increased the total nitrogen concentration in the mosses. Dissolved organic nitrogen in the moss water associated with *S. capillifolium* was proportional to the quantity of added nitrogen, suggesting an accumulation of amino acids in the mosses receiving nitrogen. This relationship was undetectable with *S. recurvum*, but this may be a consequence of its pool habitat, where surface waters in winter could dilute the water associated with the moss. Three seasons of nitrogen fertilization (30 kg N ha^{-1} yr^{-1}) had few effects on phosphorus cycling in the peat layer. The main effect was an increase in the C:P ratio of DOP (Williams and Silcock, 2001).

However, the difficulties of generalising these experimental findings is indicated by Williams *et al.* (1999a), who reported the effects of adding a range of nitrogen loads to *Sphagnum magellanicum* carpets at four contrasting sites, in Finland, Estonia and France, in addition to the Moidach More site in the UK. These experiments were only of four months duration, and hence of little value in terms of assessing critical loads, but showed clear contrasts in response between the sites, in terms of growth, increases in nitrogen content of the mosses, and in ammonium concentrations in the peat at 30 cm depth. The results suggest that both phosphorus limitation and a deep water table may decrease the retention of deposited inorganic nitrogen, and Williams *et al.* (1999a) suggest that peat drainage may reduce retention of deposited nitrogen by *Sphagnum* carpets.

The significance of nitrogen saturation of the *Sphagnum* layer, and increased availability of nitrogen in peat and in peat waters, was investigated by Tomassen *et al.* (2000; 2002), who treated transplanted turfs from an ombrotrophic floating bog in the Netherlands with a range of nitrogen deposition rates in the laboratory over 3 years, and also added *Betula pubescens* and *Molinia caerulea* plants. Tomassen *et al.* (2000) hypothesised that the birch and grass would only respond positively if nitrogen concentrations in the peat water below the *Sphagnum* mat increased. There was a significant and sometimes large increase in nitrogen-rich amino acid concentrations in the plant material of *Sphagnum recurvum* in the treatments above 5 kg N ha^{-1} yr^{-1}. However, after three years significantly increased ammonium concentrations in peat water were only measured in the 40 kg N ha^{-1} yr^{-1} treatment (ca. 25 µmol l^{-1}; a common value in Dutch ombrotrophic bogs), compared with 5-10 µmol l^{-1} in all other treatments, because of the very high nitrogen retention in the peat mosses. Nevertheless, after two years, there was evidence of increased birch extension growth with increasing nitrogen addition, and after 3 years, total biomass of both *Betula pubescens* and *M. caerulea* plants was significantly higher in the turfs which received 40 kg N ha^{-1} yr^{-1} (Tomassen *et al.*, 2002).

Lamers *et al.* (2000) used field data from Ireland and the Netherlands to produce an assessment of the capacity of the moss layer to restrict the amount of nitrogen deposition reaching the rhizospere and hence enhancing the growth of competitive graminoids and other species (Fig. 6.2). The estimates were based on field data for *Sphagnum* from sites covering a range of values of N deposition. Below 10 kg N ha^{-1} yr^{-1}, Lamers *et al.* (2000) suggest that growth stimulation would absorb the additional nitrogen inputs, while between 10-20 kg N ha^{-1} yr^{-1}, an increase in nitrogen content of the *Sphagnum*, primarily from accumulation of amino acids, might assimilate the further increase in nitrogen deposition. Other nutrients, especially phosphorus, may however become secondarily limiting to plant growth when nitrogen inputs reach a threshold (Aerts *et al.*, 1992). Above 20 kg N ha^{-1} yr^{-1}, it is suggested that this 'natural filter' would fail, and that nitrogen availability in the rhizosphere would increase, as also observed in high deposition regions (Fig. 6.2). It is important to note that Lamers *et al.* (2000) estimate total deposition as twice measured bulk deposition in constructing Figure 6.2. However, at the remote sites with low deposition, dry deposition is likely to be very low, and this method would significantly overestimate the actual deposition inputs; this would thus reduce the threshold nitrogen deposition rate for response in this figure.

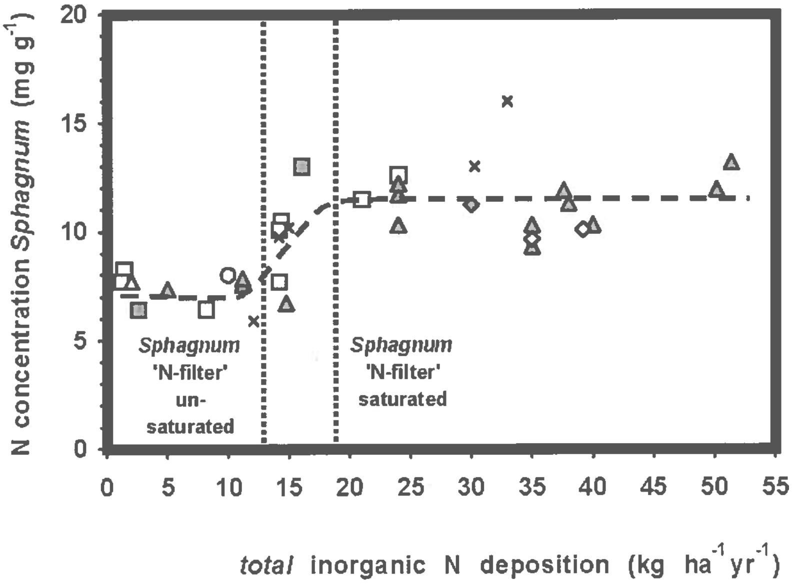

Figure 6.2. The nitrogen concentration (mg g dry weight^{-1}) of raised bog (D1) *Sphagnum* species (apical parts) in Europe and USA in relation to total atmospheric nitrogen inputs (estimated as twice wet deposition) (Lamers *et al.*, 2000).

e) Effects on decomposition and nutrient cycling

Aerts *et al.* (1992) concluded, based on the studies on the two Swedish sites discussed above, that high atmospheric nitrogen supply may affect the carbon balance of ombrotrophic bogs, because productivity under these circumstances is phosphorus, rather than nitrogen limited, but decomposition is probably increased by the high nitrogen loads. This hypothesis is supported by the results of the short-term experiments of Williams *et al.* (1999a) and Williams and Silcock (2000) (for details, see above). The addition of 30 kg N ha^{-1} yr^{-1} resulted in a decrease in peat C:N ratios (Williams *et al.*, 1999a), which could enhance rates of first-stage organic matter decomposition and nitrogen mineralization (Aerts *et al.*, 1992). Furthermore, additions of nitrogen to cores of *S. capillifolium* and *S. recurvum*, although captured mainly by the moss, had significant effects on the C and N values of the microbial biomass in the underlying peat (Williams and Silcock, 2000).

However, a study of *Sphagnum* peat in Swedish ombrotrophic bogs along a gradient of nitrogen deposition (Hogg *et al.*, 1994) indicated that the decomposition rate of *Sphagnum* peat was more influenced by the phosphorus content of the material, than by nitrogen. This finding that phosporus content is significant is consistent with the results of a further study on the same two Swedish sites by Aerts *et al.* (2001), which involved 4 years of fertilization with 5 or 10 kg N ha^{-1} yr^{-1}. These treatments had no significant effects at either site on potential decay rates, measured by taking litter formed in the first 3 years of the experiment and monitoring time-courses of CO_2 evolution in the laboratory. There was a significant relationship between potential decay rate and litter nutrient concentrations, but the effects of litter nitrogen and phosphorus content were comparable, and these relationships were primarily governed by the differences between the sites rather than treatments. Moreover, Tomassen *et al.* (2002) determined the decay rates, by measuring both CH_4 and CO_2 evolution, of the peat turfs, which had been treated with 8 levels of nitrogen deposition during 3 years (see previous section). Despite significant differences in C:N ratios, carbon mineralization rates were not at all affected by nitrogen treatments.

In summary, these results do indicate that effects of elevated nitrogen deposition on *Sphagnum* growth and litter chemistry, and on microbial biomass, are likely to affect the decomposition process, but this process is certainly not only regulated by the C:N ratio of the peat. However, further evidence is clearly necessary to properly evaluate the long-term effects of enhanced nitrogen supply on the decomposition of *Sphagnum* peat and nutrient cycling.

Summary

Even though there are a limited number of really long-term (above 3-4 years) nitrogen manipulation experiments in bog ecosystems, a clearer picture is now emerging of the potential impact of elevated nitrogen deposition on bog habitats. Responses to nitrogen addition as low as 10 kg N ha^{-1} yr^{-1} have been observed in a number of bog species (in terms of survivorship, flowering, and density). Bryophyte species, in particular *Sphagnum* spp., appear to be susceptible to the rise in anthropogenic nitrogen pollution, showing a decline in favour of grass and other competitive species, changes in competition between *Sphagnum* species, and changes in physiological and biochemical characteristics. The concept of a limited capacity for retention of nitrogen deposition in the moss layer, above which nitrogen availability in the rhizosphere increases, offers a tool for assessment of factors which may modify the critical loads. Initial estimates using this method are consistent with a long-term response threshold of 10 kg N ha^{-1} yr^{-1}.

The critical load previously recommended by Bobbink *et al.* (1996) (5-10 kg N ha^{-1} yr^{-1}) was based on a considerable body of field and experimental evidence and hence was judged to be quite reliable. Further studies since then in general provide results, which further support this value and it is therefore proposed that the critical load for **bog ecosystems (D1)** remains at **5-10 kg N ha^{-1} yr^{-1}, and** that this critical load is now described as **reliable**. Expert judgement, based on observations that response to nitrogen is less in wetter areas of a bog, is that bogs receiving high effective precipititation (e.g. N&W UK, Norway) are less sensitive to nitrogen than those in drier areas (e.g. the Netherlands, Sweden). Precipitation is thus, besides phosphorus limitation, recommended to be a modifying factor to be taken into account when assigning critical loads to individual sites. However, additional long-term studies with enhanced nitrogen deposition (both nitrogen oxides and ammonia/ammonium) are necessary to further validate this figure, especially in relation to the long-term effects upon nitrogen and carbon mineralization, and to provide a stronger basis for identifying variation in response within this range.

6.3 Valley mires, poor fens and transition mires (D2)

Valley mires, poor fens and transition mires all have permanently waterlogged soils, with ground water just below or at the soil surface. This water supply is rather poor in base-cations, leading to an acidic system, where peat mosses, but also calcifuge small sedges and some brown moss communities, dominate the vegetation. The distinction between valley mires, poor fens and transition mires is made on the basis of water level and water origin, and may have some implication for the critical load. However, the scarcity of studies does not allow a further distinction, and at present the little information that is available comes mainly from poor-fen systems (D2.2). Based on the generalisation from Morris (1991) on the link between nitrogen sensitivity and hydrology (see introduction par 6.1), it is then to be expected that valley mires are slightly less sensitive, and quaking bogs and transition mires more sensitive to excess nitrogen than poor fens.

In the northern part of Belgium, with a high deposition (about 30 kg N ha^{-1} yr^{-1}), Van Duren *et al.* (1997) found that biomass production in a poor fen was still mostly nitrogen limited. Adding 200 kg N ha^{-1} yr^{-1} for 3 years almost doubled biomass at cutting, and stimulated mainly the growth of *Filipendula ulmaria*, a high growing forb (Van Duren *et al.,* 1997). In a more realistic experiment in northern Sweden, Granberg *et al.* (2001) have simulated increased nitrogen deposition by spraying treatments of 15 and 30 kg N ha^{-1} yr^{-1} for three growing seasons, at a ambient deposition of 2 kg N ha^{-1} yr^{-1}. The authors were interested in possible effects of climate change on CH_4 emission, and they expected (and confirmed) sedges to be an important pathway for CH_4 release into the atmosphere. Similar to effects found in ombrotrophic bogs (see section 6.2), sedge cover was significantly increased with an increasing amount of nitrogen addition (Granberg *et al.*, 2001). Unfortunately, the regression analysis used did not permit a distinction between effects found at 15 or at 30 kg N ha^{-1} yr^{-1}. When sedge cover was accounted for, the effect of nitrogen addition on CH_4 emission changed over the two years of addition from non-significant in the first year (1995) to a significant negative effect during the last year (1997). This cumulative effect was probably linked to the significant accumulation of total nitrogen in the upper 5 (15 kg N ha^{-1} yr^{-1} treatment) or 10 (30 kg N ha^{-1} yr^{-1} treatment) cm of the soil (Granberg *et al.*, 2001). This accumulation corroborated the results found by Francez and Loiseau (1999), who added 5 kg N ha^{-1} yr^{-1} labelled with ^{15}N to a poor fen in Côte de Braveix, Central France (background deposition 10 kg N ha^{-1} yr^{-1}). All nitrogen, added in June or August, remained in the system until October. Most of this nitrogen (55-65%) had accumulated in the *Sphagnum* layer. The upper peat layer between 0-10 cm accumulated about 15-30% of added N, and in all other compartments less than 5% of the added nitrogen was retrieved (Francez and Loiseau, 1999). These results indicate that *Sphagna* have a similar function as a nitrogen filter in poor fens as in ombrotrophic bogs (Lamers *et al.*, 2000). Microcosm studies with *Sphagnum magellanicum*, a poor fen *Sphagnum* in Scandinavia, have shown significant negative effect of nitrogen additions (30 kg N ha^{-1} yr^{-1}) on concentrations of nutrients like phosphorus, potassium and calcium in mosses after 3 months (Jauhiainen *et al.*, 1998b).

These studies indicate that a critical load of 20-35 kg N ha^{-1} yr^{-1} for mesotrophic fens (Bobbink *et al.*, 1996) is too high for poor fens. Experimental addition in the field of 15 and 30 kg N ha^{-1} yr^{-1}, with a background deposition of 2 kg N ha^{-1} yr^{-1}, caused significant changes in species composition (increase in sedge cover) within three years. Furthermore, there are strong indications that at least some *Sphagnum* species found in poor fens are very sensitive to increased nitrogen. Therefore, we propose a **new** critical load of **10-20 kg N ha^{-1} yr^{-1}** for **poor fens (D2.2)** as **quite reliable** and recommend the initiation of field experiments with realistic doses of nitrogen addition. As long as field studies are lacking, we propose to use the **higher range** of this critical load interval for **valley mires (D2.1)** and the **lower range** of this interval for **quaking bogs and transition mires (D2.3)** and urgently recommend field studies in these systems.

6.4 Base-rich fens (D4)

Like poor fens, rich fens have developed on permanently waterlogged soils, but in these systems there is a base-rich, nutrient-poor, often calcareous water supply buffering the system. They are largely occupied by calciphile small sedges and brown moss communities. Despite the fact that rich fens are the habitat of a range of specialized and rare species, very few field experiments have been conducted with ecologically relevant doses of nitrogen enrichment to determine the effects of increased nitrogen deposition. Previous work has set the critical load for mesotrophic fen ecosystems at 20-35 kg N ha^{-1} yr^{-1} (Bobbink *et al.*, 1996). This was based mainly on nutrient budget studies done in rich fens (D4.1) in the Netherlands by Koerselman and Verhoeven (1992), assuming that the input of nitrogen should be counterbalanced by the output of nitrogen through usual management (mowing). Exceedance of this critical load results in a considerable increase of tall graminoids and a subsequent decrease in diversity of the subordinate plant species (Vermeer, 1986; Verhoeven and Schmitz, 1991). Though in some fens the strong phosphorus limitation inhibits changes in diversity with increased nitrogen, it is expected that such a situation leads to increased losses of inorganic nitrogen to the surface or ground water, thus leading to a similar critical load (Bobbink *et al.*, 1996).

Between 1996 and 2002, some additional field experiments have been conducted, but these were all with (very) high nitrogen additions (>100 kg N ha^{-1} yr^{-1}) (e.g. Beltman *et al.*, 1996; Boeye *et al.*, 1997; Wassen *et al.*, 1998). Five locations, in Belgium, Ireland and Poland, were used in factorial fertilization experiments with both nitrogen and phosphorus. Two sites (in the north-eastern part of Belgium and in Poland) gave clear evidence of nitrogen as most important growth-limiting factor, while the other three sites were strongly limited by phosphorus (Beltman *et al.*, 1996; Boeye *et al.*, 1997; Wassen *et al.*, 1998). Furthermore, the biogeochemistry and cycling of nutrients was recently studied in rich fens in a region with typical high atmospheric nitrogen loads (the Netherlands) and in a region with very low nitrogen deposition (northern Japan). This study demonstrated that nitrogen mineralization was much higher in the Dutch rich fens, than in the Japanese ones, despite the fact that the sites in the Netherland were managed by mowing (Nohara *et al.*, 2002). This clearly indicates the acceleration of the cycling of nitrogen in Dutch rich fens under high atmospheric nitrogen inputs (25-40 kg N ha^{-1} yr^{-1}; mostly as ammonia/ammonium) since the early 1980s.

Microcosm experiments with bryophytes (*Sphagnum wanstorfii*) or vascular plants (*Carex rostrata*) collected from rich fens of higher latitudes indicate sensitivity to ammonium nitrate additions of 30 kg N ha^{-1} yr^{-1} and higher. Within three months, changes were measurable in moss growth as well as in nitrogen concentration and the concentration of free amino acids of both mosses and vascular plants, both of which are clear indications for nitrogen enrichment effects (Jauhiainen, 1998a; Saarinen, 1998). Though the conditions in these experiments were too artificial to justify by themselves a decrease of the present critical load, they do indicate that the moss species in particular, which are an important and prominent part of biodiversity in rich fens, may be very sensitive to increased nitrogen deposition.

Preliminary results after the two first years of long-term nitrogen addition in a rich fen in Central Ireland (ambient load 7-10 kg N ha^{-1} yr^{-1}) clearly indicate that the typical species-rich brown moss vegetation is affected by the form of nitrogen enrichment (Paulissen *et al.*, 2003). Nitrate application (50 kg N ha^{-1} yr^{-1}) significantly stimulated phosphomonoesterase (PMEase) activity in the typical brown moss *Scorpidium revolvens*, whereas ammonium addition (50 kg N ha^{-1} yr^{-1}) did the same in the invasive *Sphagnum contortum* within a two-year experimental period (Fig. 6.3). PMEase is an enzyme that enables plants to take up organically bound phosphates that are otherwise unavailable. This enables the plant to increase phosphorus uptake under conditions of high nitrogen availability. This suggests that, in the longer term, *Sphagnum* will profit of high ammonium deposition (comparable with Dutch levels), while brown mosses are negatively affected by it. The first results of the changes in moss cover suggest an increase in *Sphagnum* and a decline in brown mosses after two years of ammonium addition, but this experiment will be continued in the near future to validate this conclusion.

Bergamini and Pauli (2001) studied the effects of nitrogen enrichment on bryophytes in montane rich-fens (D4.2). These authors added 100 kg N ha^{-1} yr^{-1} to 18 different montane rich-fens (background deposition between 15 and 30 kg N ha^{-1} yr^{-1}) for 2 years, and found a significant 30% increase in vascular plant plant biomass and a significant 39% decrease in bryophyte biomass. Using vascular plant biomass as a covariable, they showed that nitrogen had a direct negative effect on bryophyte biomass, excluding the effects through increased competition by vascular plants (Bergamini and Pauli, 2001; Pauli *et al.*, 2002). Despite the clear effects on bryophyte biomass, species heterogeneity between the sites was too high to find any significant changes in bryophyte community composition (Bergamini and Pauli, 2001). Although the nitrogen additions in this experiment are too high to set a critical load, they are an indication that montane rich fens are also sensitive to nitrogen.

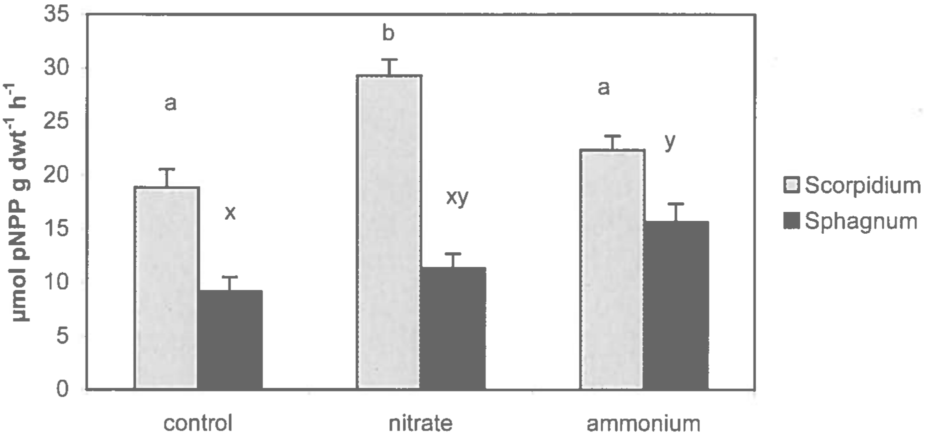

Figure 6.3. Phosphomonoesterase activity (mean + SEM, n=5) of *Scorpidium revolvens* and *Sphagnum contortum*, after two years of nitrogen addition in Scragh Bog (C. Ireland). Colums sharing a letter are not significantly different (p<0.05) (from Paulissen *et al.*, 2003).

The **critical load** for **rich fens (D4.1)** thus becomes **15-35 kg N ha^{-1} yr^{-1}**, with the higher end of the range recommended for managed or non-nitrogen-limited systems, and the lower end of the range recommended for **nitrogen-limited systems** but also for **all high latitude systems** and **montane rich-fens (D4.2) (15-25 kg N ha^{-1} yr^{-1}) (expert judgement)**. The latter have longer frost periods, on average a colder climate, and more shallow soils, which are all factors that increase sensitivity to nitrogen (Bobbink *et al.*, 1996). Long-term fertilization experiments with ecologically relevant additions of nitrogen, both in temperate regions, in northern countries and in montane fens should increase the reliability of these figures. An overview of the empirical nitrogen critical loads for mire, bogs and fen habitats is presented in table 6.1.

Table 6.1. Empirical critical loads for the effects of excess nitrogen upon different mire, bog and fen habitats (Eunis class **E**). ## reliable, # quite reliable and (#) expert judgement.

Ecosystem type	**EUNIS-code**	**kg N ha^{-1} yr^{-1}**	**Reliability**	**Indication of exceedance**
Raised and blanket bogs	D1[a,b]	5-10	##	Increase vascular plants, decrease bryophytes, altered growth and species composition of mosses, increased N in peat and peat water
Poor fens	D2.2 [c]	10-20	#	Increase sedges and vascular plants, negative effects on peat mosses
Rich fens	D4.1[d]	15-35	(#)	Increase tall graminoids, decrease diversity
Montane rich fens	D4.2	15-25	(#)	Increase vascular plants, decrease bryophytes

[a)] use towards high end of range at phosphorus limitation, and towards lower end if phosphorus is not limiting;

[b)] use towards high end of range with high precipitation and towards low end of range with low precipitation;

[c)] For D2.1 (quaking fens and transition mires): use lower end of range (#)
For D2.3 (valley mires): use higher end of range (#)

[d)] For high latitude or nitrogen-limited systems: use lower end of range (#)

7. Effects of nitrogen deposition on inland surface water habitats (EUNIS class C)

7.1 Introduction

In this chapter, the effects of atmospheric nitrogen deposition on freshwater ecosystems (inland surface water habitats; EUNIS class C) are evaluated. Inland surface water habitats are non-coastal open fresh or brackish waterbodies (e.g. lakes and pools, rivers, streams and springs), including their littoral zones. Class C, however, also includes dune slack pools (Davies and Moss, 2002). Previously, the consequences of nitrogen enrichment upon soft water lakes were treated in the wetland section (Bobbink *et al*., 1996). However, in the EUNIS system, these ecosystems are classified in standing surface waters as class 1.1: permanent oligotrophic lakes, pools and ponds. They are thus treated in this background document in a separate chapter. Surprisingly, the subdivision of standing waters is only based on the trophic status of the waters, ignoring the important gradient in alkalinity (very soft to hard water), which is at least partly independent of the nutrient status of the water (e.g. Moss, 1988). This chapter summarises field and experimental evidence to establish critical loads for nitrogen deposition with respect to eutrophication or adverse ammonium effects. The acidifying effects of airborne nitrogen compounds to surface waters are only briefly summarized, as critical loads for acidity, including nitrogen, are well established for most waters (Henriksen, 1988; Kämäri *et al.,* 1992; Henriksen and Posch, 2001).

7.2 Permanent oligotrophic lakes, pools and ponds (C1)

7.2.1 Permanent oligotrophic lakes, pools and ponds (softwater lakes; C1.1)

In the lowlands of Western Europe, many soft waters are found on sandy soils, which are poor in calcium carbonate or almost devoid of it. The waters are poorly buffered and the concentrations of calcium in the water layer are very low; they are shallow and fully mixed water bodies, with periodically fluctuating water levels, and are mainly fed by rain water, and thus oligotrophic. In the EUNIS approach, these waters have to be classified as part of permanent oligotrophic lakes, pools and ponds (C1.1). These softwater ecosystems are characterized by plant communities from the phytosociological alliance LITTORELLETEA (Schoof-van Pelt, 1973; Wittig, 1982; Arts, 1990; 2002) and have been well studied with respect to the effects of nitrogen deposition. The stands of these communities are characterized by the presence of rare and endangered isoetids (with the growth form of *Isoetes*), such as *Littorella uniflora*, *Lobelia dortmanna*, *Isoetes lacustris*, *I. echinospora*, *Echinodorus* species, *Luronium natans* and many other softwater macrophytes. These soft waters are nowadays almost all within nature reserves and have become very rare in Western Europe. This decline may be illustrated by the fact that *Littorella uniflora* was known from more than 230 sites in the Netherlands in the early 1950s, of which only ca. 40 still existed at the end of the 1980s. Furthermore, a strong decline in amphibians has been observed in these soft waters (Leuven *et al*., 1986).

The effects of nitrogen pollutants on these soft waters have been intensively studied in the Netherlands both in field surveys and experimental studies. Field observations in ca. 70 soft waters (with well-developed isoetid vegetation in the 1950s) showed that the waters in which these macrophytes were still abundant in the early 1980s were poorly buffered (alkalinity 50-500 meq l^{-1}), circumneutral (pH 5-6) and very poor in nitrogen (Roelofs, 1983; Arts *et al.*, 1990). The softwater sites where these plant species had disappeared could be divided into two groups. In 12 of the 53 softwater sites eutrophication, resulting from inlet of enriched water, seemed to be the cause of the decline. In this group of non-acidified waters plant species such as *Lemna minor* had become dominant. High concentrations of phosphate and ammonium were measured in the sediment in these waters. In some of the larger of the water bodies no macrophytes were found at all as a result of dense plankton bloom. In the second group of lakes and pools (41 out of 53) another development had taken place: the isoetid species were replaced by dense stands of *Juncus bulbosus* or aquatic mosses such as *Sphagnum cuspidatum* or *Drepanocladus fluitans*. This clearly indicates acidification of these soft waters in recent decades, probably caused by enhanced atmospheric deposition. In the same field study it was shown that the nitrogen levels of the water layer were higher in ecosystems where the natural vegetation had disappeared, compared with ecosystems where the isoetid stands were still present (Roelofs, 1983). This strongly suggests the detrimental effects of atmospheric nitrogen deposition in these softwater lakes.

A number of ecophysiological studies have revealed the importance of (i) inorganic carbon status of the water as a result of intermediate levels of alkalinity, and (ii) low nitrogen concentrations, for the growth of the endangered isoetid macrophytes. Furthermore, almost all of the typical softwater plants had a relatively low potential growth rate. Increased acidity and higher concentrations of ammonium in the water layer clearly stimulated the development of *Juncus bulbosus* and submerged mosses such as *Sphagnum* and *Drepanoclado*s species (Roelofs *et al.*, 1984; Den Hartog, 1986). It has also been shown in cultivation experiments that the nitrogen species involved (ammonium or nitrate) differentially influenced the growth of the studied species of water plants. Almost all of the characteristic softwater isoetids developed better with nitrate instead of ammonium addition, whereas *Juncus bulbosus* and aquatic mosses (*Sphagnum* and *Drepanocladus*) were clearly stimulated by ammonium nutrition (Schuurkes *et al.*, 1986). The importance of ammonium for the growth of these aquatic mosses is also reported by Glime (1992).

The effects of atmospheric deposition have been studied in softwater mesocosms during a 2-year treatment with different artificial rainwaters. Acidification due to sulphuric acid, without nitrogen inputs, has not resulted in an increased mass growth of *Juncus bulbosus* and a diverse isoetid vegetation remains present. However, after increasing the nitrogen concentration in the precipitation (19 kg N ha^{-1} yr^{-1} or higher as ammonium sulphate; control <2 kg N ha^{-1} yr^{-1}), similar changes in floristic composition as under field conditions have been observed: a dramatic increase in dominance of *Juncus bulbosus*, of submerged aquatic mosses and of *Agrostis canina* (Schuurkes *et al.,* 1987). These results demonstrate that the observed changes occurred because of the effects of ammonium sulphate deposition, leading to both eutrophication and acidification. The increased levels of ammonium in the system stimulated directly the growth of plants such as *Juncus bulbosus*, whereas the surplus of the extra ammonium will be nitrified in these waters (pH>4.0). During this nitrification process H^+-ions are produced, which increase the acidity of the system. The results of this study clearly demonstrated that large changes already occurred after 2-year treatment with 19 kg N ha^{-1} yr^{-1} or higher. In addition, it has been observed that the strongest decline in the species composition of the macrophytes of the Dutch softwater communities has coincided with areas with atmospheric nitrogen loads of ca. 10-13 kg N ha^{-1} yr^{-1} (Arts, 1990). Furthermore, Brouwer *et al.* (1997) showed that after 10 years of clean-rain treatment the recovery in the same softwater mesocosms, which had been treated with ammonium sulphate, is only partial and the vegetation is still dominated by *Juncus* and *Molinia* .

The nitrogen critical load for these **permanent oligotrophic waters** (**shallow softwater bodies, C1.1**) was set in 1996 at **5-10 kg N ha^{-1} yr^{-1}** (Bobbink *et al.*, 1996). Since 1996 no new experimental evidence has been published on the impacts of nitrogen deposition in softwater lakes, which necessitate a change of the set critical load. A clear support to the established critical load is the outcome of a dynamic ecosystem model of small soft water lakes in the Netherlands, including sediment-water interactions and different groups of water plants (Wortelboer, 1998). Simulations with this model over a 30-year period confirmed the empirically derived nitrogen critical load (Albers *et al.*, 2001), which is thus to be considered as **reliable**.

7.2.2 Permanent oligotrophic lakes, pools and ponds (C1.16; dune slack pools)

Another category in the EUNIS class of permanent oligotrophic waters are dune slack pools (C1.16). These relatively small, but permanent clear water pools are found in the coastal dune areas across Europe. Despite their location in the coastal dunes, they are classified in inland surface water habitats in EUNIS. The water layer is nutrient poor and these dune slack pools are mostly insensitive to acidification, because of their higher alkalinity (>1000 μeq l^{-1}), and characterized by clear water and a diverse submerged macrophyte vegetation (e.g. *Potamogeton* spp., *Chara* spp. and littoral isoetids). Eutrophication in these dune slack pools is more likely caused by atmospheric inputs or by high density of waterfowl than by the inflow of enriched surface water, because of their hydrological isolation. Very few experimental data exist on the sensitivity of these dune slack pools with respect to critical load setting, despite the generally well-known nitrogen limitation of dune slack wetlands (Lammerts and Grootjans, 1997). However, the impacts of atmospheric nitrogen deposition have been quantified in dune slack pool mesocosms (diameter ca. 2 m) during a 2-year treatment with different nitrogen loads (1, 20, 40 and 120 kg N ha^{-1} yr^{-1}) (Brouwer *et al.*, 1996). No acidification of the water layer was found within 2 years, but total biomass of water plants and helophytes strongly increased above 20 kg N ha^{-1} yr^{-1}. Nitrogen additions have clearly caused an increased succession rate in these dune slack mesocoms, leading to more helophytes and less open water. This phenomenom is also observed in many dune slack pools in the Netherlands, but experimental field studies are missing. Therefore, we propose a new critical load of **10-20 kg N ha^{-1} yr^{-1}** for **dune slack pools (C1.16)** as expert judgement and recommend the initiation of long-term field experiments with realistic doses of nitrogen additions to these and other shallow oligotrophic waters, especially in regions with a low current deposition.

7.3 Effects of nitrogen deposition on lakes and streams (C1-partly- and C2)

There is ample evidence that an increase of acidic and acidifying compounds in atmospheric deposition had resulted in recent acidification of lakes and streams in geologically sensitive regions of Scandinavia, western Europe, Canada and the USA (e.g. Hultberg, 1988; Muniz, 1991). This acidification is characterized by a decrease in pH and ANC, and by increases in concentrations of sulphate, aluminium, and sometimes nitrate and ammonium. It has already been shown since the 1970s with different approaches (field surveys, laboratory studies, whole-lake experiments) that these changes had dramatic consequences for plant and animal species (macrofauna, fishes) and for the functioning of these aquatic ecosystems. However, due to the strong reduction of sulphur deposition in the last decades, a (partial) recovery from acidification in very sensitive waters has been observed in both North America and Europe (e.g. Stoddard *et al.*, 1999).

The critical loads of acidity for aquatic ecosystems have been published under the Convention on Long-range Transboundary Air Pollution in 1988, 1992 and 1996 based on steady state water chemistry models. These recent models incorporate both acidity from both sulphur and nitrogen deposition and critical loads are calculated depending on (i) base cation deposition, (ii) internal alkalinity production or base cation concentrations and (iii) nitrate leaching from the water system. The calculated critical loads are thus site specific (sensitive geological regions or not) and also depend on the local hydrology and precipitation. For full details see Henriksen (1988), Kämäri *et al.* (1992) and Henriksen and Posch (2001). The critical loads for acidifying effects of nitrogen compounds, based upon nitrogen as the only cause of acidification, for the most sensitive lakes and streams are:

	kg N ha^{-1} yr^{-1}	
Scandinavian waters (C1.1)	1.4-4.2	(Henriksen, 1988; Kämäri *et al.*, 1992)
Alpine lakes (C1.1)	3.5-6.1	(Marchetto *et al.*, 1994)
Humic moorland pools (C1.4)	3.5-4.5	(Schuurkens *et al.*, 1987; Van Dam and Buskens, 1993)

In many areas with moderate to high alkalinity in the waters and/or high base-cation deposition, the values of the critical loads for acidity from nitrogen deposition are much higher than those for the sensitive fresh waters. At this moment the possible effects of eutrophication by ammonia/ammonium or nitrate deposition are hardly incorporated in the establishment of critical loads of nitrogen for waters, except for permanent oligotrophic water bodies (C1.1; see section 7.2). This is because primary production in almost all surface waters is limited by phosporus, and thus nitrogen enrichment has been considered unimportant in this respect (e.g. Moss, 1988). This certainly holds for those aquatic ecosystems, which are sensitive to acidification, for which the critical loads for acidifying effects (such as those listed above) are very low and thus certainly provide enough protection for the effects of nitrogen eutrophication.

It is, however, likely that some inland surface waters, which are relatively well buffered against acidification, are sensitive to nitrogen eutrophication, especially in shallow oligotrophic or mesotrophic clear water habitats, including their littoral zone (Kämäri *et al.*, 1992). The effects of nitrogen eutrophication (including reduced nitrogen forms) in these waters need further research, especially in areas with high phosphorus inputs, and should be incorporated into critical loads for nitrogen in future. At this moment, it is not possible to set critical loads of nitrogen eutrophication for these kinds of surface waters. The recommended critical loads of nitrogen with respect to eutrophication or adverse ammonia/ammonium effects to inlands surface water habitats (C1.1 & C1.16) are summarized in the following table (Table 7.1).

Table 7.1. Empirical critical loads for the effects of excess nitrogen upon surface water habitats (C1)*. ## reliable, # quite reliable and (#) expert judgement.

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Softwater lakes (permanent oligotrophic waters)	C1.1	5-10	##	Isoetid species negatively affected
Dune slack pools (permanent oligotrophic waters)	C1.16	10-20	(#)	Increased biomass and rate of succession

*: for waters sensitive to acidification, use also critical loads for acidity.

8. Effects of nitrogen deposition on coastal habitats (EUNIS class B)

8.1 Introduction

In this chapter the impacts of atmospheric nitrogen deposition on coastal habitats (EUNIS class B) are evaluated with respect to the setting of nitrogen critical loads. Coastal habitats are those above the spring high tide limit (or above mean water level in non-tidal waters) occupying coastal features and characterized by their proximity to the sea. They include coastal dunes (dry grasslands, wet to moist dune slacks and wooded dunes), beaches and cliffs (Davies and Moss, 2002). Dune slack pools are considered in the previous chapter, because of their position in EUNIS class C (inland surface waters), although a consistency between critical loads for dune slacks and dune slack pools is clearly important in practical applications. The first division within class B in EUNIS is based upon underlying substrates, i.e. sand, shingle or rock, but data to support proposals for critical loads are only available for sand substrates. Bobbink *et al.* (1996) considered the consequences of increased atmospheric nitrogen loads on coastal grasslands and heaths briefly in the grassland and heathland sections, when relevant data were available. In this background document, separate critical loads for nitrogen deposition are set for coastal dune and sand habitats (B1), using these older studies and updating the values with new information.

8.2 Coastal dune and sand habitats (B1)

Many dune ecosystems in the coastal areas of Europe are almost of natural origin and rich in typical plant and animal life. They are found on sandy, nutrient-poor soils, and considered to be sensitive to the impacts of eutrophication and acidification (e.g. Ellenberg, 1988; Wellburn, 1988; De Vries *et al.*, 1994). Evidence exists with respect to the setting of nitrogen critical loads in coastal dune and sand habitats only for some EUNIS categories, especially shifting coastal dunes (**B1.3**), coastal stable dune grasslands (**B1.4**), coastal dune heaths (**B1.5**) and moist and wet dune slacks (**B1.8**).

8.2.1 Coastal stable dune grasslands (grey dunes) and shifting coastal dunes (B1.4 & B1.3)

A number of species-rich stable dune grasslands occur across the dunes of Europe. However, tall grasses have increased in many Dutch dry dune grasslands over recent decades, a period with higher nitrogen loading (20-30 kg N ha^{-1} yr^{-1}). In the primarily non-calcareous dunes, *Ammophila arenaria* is now the dominant species, whereas in more calcareous areas *Calamagrostis epigejos, Elymus repens* and *Elymus athericus* dominate (Kooijman and De Haan, 1995). Because of reduced light penetration through the tall grass canopies formed by these species, the development of several prostrate species has been reduced and management is necessary to maintain its diversity. In the past, tall graminoids were usually not dominant on these low nutrient sandy dune soils in the Netherlands. A recent survey in dry dune grasslands along the Dutch coast revealed that non-calcareous, iron-poor dry dune ecosystems are nitrogen-limited, but that in calcareous, iron-rich dunes there is co-limitation of nitrogen and phosphorus. (Kooijman *et al.*, 1998; Kooijman and Besse, 2002). Kooijman *et al.* (1998) concluded that atmospheric nitrogen deposition may cause tall grass dominance encroachment in non-calcareous dunes, but probably only accelerates the process in calcareous dune grasslands. Yet, a strong negative correlation between the percentage open dunes and total nitrogen deposition, especially above 15 kg N ha^{-1} yr^{-1}, has recently been found in both Dutch dune regions (Van Hinsberg and Van der Hoek, 2003). The hypothesis that the present dominance of tall grasses and increased rate of succession in the Netherlands might be a result of the increased atmospheric nitrogen deposition, is also supported by the fact that in many coastal areas of Britain, with relatively low nitrogen deposition (ca. 10 kg N ha^{-1} yr^{-1}), coastal stable dune grasslands are still species-rich (Jones *et al.*, 2002b)

Additional field evidence is now available from a survey in the coastal dune areas across England and Wales in a region with much lower nitrogen depositions than the Netherlands (Jones *et al.*, 2002d; 2003). Eleven sand dunes sites were surveyed with a range of atmospheric nitrogen inputs from 10-30 kg N ha^{-1} yr^{-1}. In the mobile and semi-fixed dunes there were significant positive relationships between nitrogen inputs and biomass or sward heights. In the stable dune grasslands, combined nitrogen loads showed a negative relationship with species density and, less clearly, a positive correlation with sward height. In addition, a strong positive relationship between dissolved organic nitrogen in groundwater and nitrogen deposition in these stable grasslands was found. In general, the mentioned effects start to become apparent in the range 10-20 kg N ha^{-1} yr^{-1} and higher.

The effects of nutrients in dry dune grasslands (calcium carbonate 1%) on sandy soils were experimentally studied at Braunton Burrows (Devon, UK) by Willis (1963). Nutrients were applied for two years and complete NPK fertilization strongly stimulated the growth of grasses, such as *Festuca rubra*, *Poa pratensis* and *Agrostis stolonifera,* which significantly reduced the abundance of many small plants such as prostrate phanerogamic species, mosses and lichens. The impacts of different combinations of nitrogen, phosphorus and potassium were also investigated and nitrogen (>100 kg N ha^{-1} yr^{-1}) proved to be more limiting for plant growth than phosphorus. Although the changes in the vegetation were clearly less profound than after complete fertilization, reduction of species numbers (especially annual species, lichens and mosses) was observed in the treatments with nitrogen (Willis, 1963). Boorman and Fuller (1982) examined the effects of nutrient additions on the species composition of rabbit grazed dune grassland in Norfolk, UK over five years. The grazing prevented *F. rubra* becoming dominant, but several species (especially annuals, mosses and lichens) declined, while two species (*Carex arenaria* and *Calystegia soldanella*) increased in all treatments containing nitrogen (80 kg N ha^{-1} yr^{-1}). No evidence was found in this study for reduced diversity in +PK-plots. In a one-year experiment with additions of nitrogen (20, 40, 80 and 160 kg N ha^{-1} yr^{-1}; atmospheric load 15 kg N ha^{-1} yr^{-1}) or phosphorus, the above-ground biomass of a stable dune grassland at the Dutch Wadden island of Schiermonnikoog proved to be strongly nitrogen limited; the biomass significantly increased above additions of 40 kg N ha^{-1} yr^{-1} in this 30-year old stable dune grassland (Olff *et al.*, 1993). The effects of nitrogen have also been studied during two years on the coastal dunes of Iceland (probably EUNIS B1.3, shifting coastal dunes) in a region with very low atmospheric nitrogen deposition (Greipsson and Davy, 1997). The number of flowering spikes and total seed weight of the tall dune grass *Leymus arenarius* strongly increased within one year after application of 50 or 100 kg N ha^{-1}. In summary, these short-term experiments with relatively high nitrogen applications clearly indicate the importance of nitrogen limitation in several shifting dunes and stable dune grasslands, but in some cases co-limitation with phosphorus has been observed, or rabbit grazing has prevented dominance of tall grasses.

Only two nitrogen addition experiments have been carried out in stable dune grasslands with the specific objective to examine the effects of nitrogen deposition. The effects of nitrogen additions and the interaction with rabbit grazing were investigated in a factorial design in two coastal stable dune grasslands, one calcareous and one partly decalcified, in the Netherlands by Ten Harkel and Van der Meulen (1995) and Ten Harkel *et al.* (1998). After 4 years of nitrogen additions (25 kg N ha^{-1} yr^{-1} as ammonium nitrate pellets, twice a year; current deposition ca. 23 kg N ha^{-1} yr^{-1}) there were no significant changes in species composition in either the grazed or the ungrazed situation. Exclusion of grazing by rabbits and horses, through the use of enclosures, resulted in graminoid dominance (*Festuca rubra*, *F. ovina* and *Poa pratensis*), especially where nitrogen additions were made, suggesting that grazing can prevent grass dominance in stable dune grasslands (Ten Harkel and Van der Meulen, 1995). Because of the high, direct leaching losses from the added pellets, the plots were watered fortnightly with ammonium sulphate (50 kg N ha^{-1} yr^{-1}) in the last one and a half year of the experiment. In this period, it had become obvious that ≥80% of the total nitrogen inputs in the nitrogen-treated vegetation had leached from the soil as nitrate due to nitrification at the calcareous sites, while 70% of nitrogen input leached even from the control grazed plots at these sites. At the partly decalcified site, nitrate leaching was substantially smaller, but still between 7-40 % (4-11 kg N ha^{-1} yr^{-1}) of the experimental nitrogen inputs (Ten Harkel *et al.*, 1998). It is thus clear that in these stable dune grasslands experimental nitrogen enrichment had no effect on species composition. This may be related to a shift to phosphorus limitation after the long period of high atmospheric nitrogen inputs in the Netherlands and the relatively high rabbit grazing pressure in that period. The high nitrogen leaching from the control vegetation is also a strong indication of nitrogen saturation of these dune grasslands, probably because of the high nitrogen deposition rates (>20 kg N ha^{-1} yr^{-1}) over several decades.

Nitrogen manipulation experiments in stable dune grasslands have not been carried out to date in regions with low nitrogen deposition. The effects of elevated nitrogen loads starting at natural background deposition (<5 kg N ha^{-1} yr^{-1}) have, however, been studied during 2 to 3 years in dry dune calcareous grassland mesocosms (1x1 m) (Tomassen *et al.*, 1999; Van den Berg *et al.*, 2003). After a pre-treatment period of 2 months with clean rain water that removed the excess of nitrate in the soil, nitrogen was added as ammonium nitrate (1, 5, 10, 15, 20, 40, 60 and 80 kg N ha^{-1} yr^{-1}) and supplied twice a week. The effects on soil pore water chemistry and on two characteristic graminoid species and two endangered herbaceous species have been followed. Within one year of nitrogen additions, a clear difference was found in the amount of algae growing on the surface of the sand. The amounts of green algae increased in the treatments between 10 and 20 kg N ha^{-1} yr^{-1}, but the difference with the two lowest treatments was especially distinct above 20 kg N ha^{-1} yr^{-1}. The strong increase of algae on the soil top layer due to nitrogen deposition may have important implications as they prevent sand drift due to wind action. 'Blowouts' are important for renewed vegetation succession, and biodiversity will decrease when young successional stages decline (Tomassen *et al.*, 1999). The concentrations of nitrate in the soil pore water have shown a strong seasonal fluctuation. During the first winter period an increase of nitrate was measured for the treatments ≥40 kg N ha^{-1} yr^{-1}. During spring the nitrate concentration rapidly decreased. During the second and third winter, an increase in nitrate could only be observed in the highest nitrogen addition level. The ammonium concentration remained constantly very low (<5 µmol l^{-1}).

Differences in plant growth have been observed after one year and have become obvious after two years or three years. The total cover of the vegetation increased with elevated nitrogen inputs. This increase could almost completely be attributed to *Calamagrostis epigejos*. The shoot and root biomass of this tall grass species increased with higher nitrogen additions, significantly above 20 or 15 kg N ha^{-1} yr^{-1}, respectively (Fig 8.1).

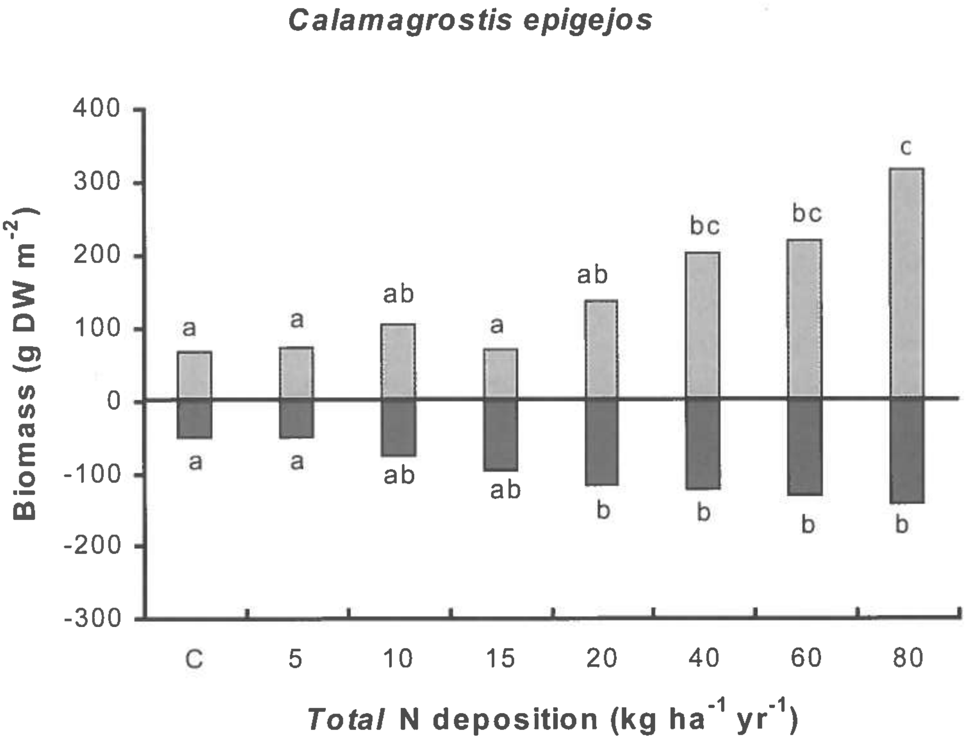

Figure 8.1. Above- and belowground biomass (g m^{-2}) of the aggressive dune grass *Calamagrostis epigejos* after 2 years of nitrogen application in coastal stable dune grassland (B1.4) mesocosms (Tomassen *et al.*, 1999). In the control treatment (C) nitrogen input was ca. 1 kg N ha^{-1} yr^{-1}.

No clear effects of elevated nitrogen deposition rates on the two herbaceous species *Galium verum* and *Carlina vulgaris* was measured within two years, although in the third year, the number and weight of *G. verum* decreased above 40 kg N ha^{-1} yr^{-1} (Van den Berg *et al.*, 2003). After two years of treatment, the total amount of nitrogen stored in the vegetation has been strongly elevated due to increased nitrogen deposition (Tomassen *et al.*, 1999). These results are consistent with those of Mohd-Said (1999), who exposed mesocosms to a range of nitrogen deposition rates in the range 2-55 kg N ha^{-1} yr^{-1}. At 10 kg N ha^{-1} yr^{-1} and above, the cover of the grasses *Festuca rubra* and *Dactylis glomerata* increased, while there was no change in cover of herb species, including *Galium verum*. These mesocosm experiments with hardly any background nitrogen load (1 kg N ha^{-1} yr^{-1}) have demonstrated that even in calcareous dune grasslands nitrogen loads

$\geq$15 kg N ha^{-1} yr^{-1} changed the algae and grass cover while high loads (when not grazed) led to dominance of graminoid species within a few years.

Previously, stable dune grasslands have been considered in the grassland section (neutral and acid grasslands) with a critical load (quite reliable) of 20-30 kg N ha^{-1} yr^{-1} (Bobbink *et al.*, 1996). Based upon the observed effects over a relatively short period in the mesocosm and field study (grass increase $\geq$ 15 kg N ha^{-1} yr^{-1}; strong nitrate leaching with 25-50 kg N ha^{-1} yr^{-1}), it is realistic to set the nitrogen critical load for **coastal stable dune grasslands (B1.4)** and **shifting coastal dunes (B1.3)** at **10-20 kg N ha^{-1} yr^{-1} (quite reliable**, and **expert judgement,** respectively). Although the impacts of nitrogen additions have now been studied in some coastal dune grasslands, especially in the Netherlands, 'long-term' studies with realistic nitrogen loads and low current deposition are strongly needed to validate this critical load value.

8.2.2 Coastal dune heaths (B1.5)

Besides dry dune grasslands, heathland vegetation is also present in the coastal dunes in north-west Europe (Gimingham *et al.*, 1979; Ellenberg, 1988). These natural coastal dune heaths are mostly dominated by the typical dwarf shrub *Empetrum nigrum*, while *Calluna vulgaris* is less common. Their podsolized soils are less developed than under inland heaths (Chapter 6) and poor in essential nutrients. Bobbink *et al.* (1996) did not consider critical loads separately for this type of heath, but incorporated it in the lowland dry heathland section. Within EUNIS, coastal heaths (B1.5) are classified as a subdivision of coastal dune and sand habitats.

Only one nitrogen manipulation experiment has been performed in coastal heaths, namely within the Danish HEATH experiment (Riis-Nielsen, 1997; Nielsen *et al.*, 2000), in a coastal heath at Lodbjerg, Denmark. This coastal heath, dominated by *Empetrum nigrum* and *Ammophila arenaria*, is present in a ca. 250-300 years old-dune area at the coast of Jutland, with relatively low current deposition (13 kg N ha^{-1} yr^{-1}) (Nielsen *et al.*, 2000). A high-frequency, low-dose application of ammonium nitrate (0, 15, 35 and 70 kg N ha^{-1} yr^{-1}) was made over 2 years. As a response to nitrogen additions, the vascular plant species increased in cover, whereas the lichens and bryophytes showed a very slight and insignificant decrease within 2 years (Riis-Nielsen, 1997). *Hypnum cupressiforme*, a typical heathland moss, declined linearly with the applications of nitrogen, whereas the cover of *Empetrum* and *Carex arenaria* increased linearly. Thus, plant productivity in this coastal heath system was obviously controlled by nitrogen limitation. However, drought effects upon *Empetrum* were clearly not related to nitrogen inputs (Tybirk *et al.*, 2000). The leaching of both nitrate and ammonium has also been quantified in this coastal Danish heath. In the control and plots with 15 kg N ha^{-1} yr^{-1}, no or hardly any nitrogen has leached to the subsoil. However, with higher N additions, especially 70 kg N ha^{-1} yr^{-1}, a considerable part of the nitrogen has leached as nitrate from the B horizon, accompanied by aluminium, leading to acidification of the soil (Nielsen *et al.*, 2000). Johansson (2000) found no effects of nitrogen additions on ericoid mycorrhizal infection of *Calluna vulgaris* in this coastal heath experiment. It is concluded that coastal heaths are affected by moderate nitrogen additions ($\geq$35-70 kg N ha^{-1} yr^{-1}), but because of the short experimental period at only one site, it is difficult to set a critical load for this ecosystem type. However, the leaching data indicate a higher sensitivity of coastal heaths than inland dry heaths to extra nitrogen inputs. This is in accordance with the shallow organic layer in these coastal heaths. Moreover, these dune heaths are generally unmanaged, and thus have low nitrogen removal from the system. Because of this, the empirical nitrogen critical load for **coastal heath (B1.5)** has been established at **10-20 kg N ha^{-1} yr^{-1} (expert judgement)**, a lower value than that for inland dry heaths. It is evident that more long-term information is needed to verify this value.

8.2.3 Coastal moist to wet dune slacks (B1.8)

Moist to wet dune slacks (EUNIS B1.8), of primary or secondary origin, are hot spots of plant diversity in the sandy dune areas around Europe. They are characterized by typical graminoids (sedges, rushes and grasses), together with many rare, basiphilous forb species. Groundwater level is at or above soil level in winter, whereas in the growing season the ground water level is considerably lower in these dune slacks. Nowadays, many dune slacks have become part of nature reserves, and these are sometimes maintained via management as hay making or sod cutting (e.g. Ellenberg, 1988; Lammerts and Grootjans, 1997; Davies and Moss, 2002). Because of their isolation in the landscape and their successional position, they mostly receive nutrients via atmospheric inputs. In addition, they are very sensitive to desiccation, because of groundwater extraction.

The limitation of dune slack vegetation by nutrients has been the topic of several studies (e.g. Willis, 1963; Olff *et al.*, 1993) and has been recently reviewed by Lammerts and Grootjans (1997). Factorial fertilization experiments have shown that in almost all studied moist to wet, primary or secondary dune slacks in the UK, USA or the Netherlands, the above-ground biomass is limited by nitrogen availability. Primary phosphorus limitation was found only once, in a dune slack where sod cutting had been applied very recently. Single nitrogen additions have led to increased dominance of *Carex* and *Juncus* species, as well of tall grasses as *Agrostis stolonifera* and *Calamagrostis epigejos*. In some studies, typical forb species had declined in such situations (for an overview and the references, see Lammerts and Grootjans, 1997). Unfortunately, none of the studies have been done with low nitrogen additions (<100 N kg ha^{-1} yr^{-1}; >1 year), and thus they are not adequate for setting a nitrogen critical load. In addition, a preliminairy field study in the UK showed no differences in dune slack vegetation attributable to N inputs between the sites where these communities occurred (range: 10-16 kg N ha^{-1} yr^{-1}) (Jones *et al.*, 2003). It is, however, clear that most of these dune slacks are sensitive to nitrogen enrichment. The critical load for these **moist to wet dune slacks (B1.8)**, is estimated by **expert judgement** as **15-25 kg N ha^{-1} yr^{-1}**, because of the resemblance of these slacks with wet, oligotrophic grasslands and base-rich fens. Of course, field validation of this value is certainly necessary. The empirical nitrogen critical loads for coastal habitats (B) are summarized in the following table (Table 8.1).

Table 8.1. Empirical critical loads for the effects of excess nitrogen upon coastal habitats (B) . ## reliable, # quite reliable and (#) expert judgement

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Shifting coastal dunes	B1.3	10-20	(#)	Biomass increase, increase N leaching
Coastal stable dune grasslands	B1.4	10-20	#	Increase tall grasses, decrease prostrate plants, increased N leaching
Coastal dune heaths	B1.5	10-20	(#)	Increase plant production, increase N leaching, accelerated succession
Moist to wet dune slacks	B1.8	10-25	(#)	Increased biomass tall graminoids

9. Effects of nitrogen deposition on marine habitats (EUNIS class A)

9.1 Introduction

Marine habitats, as defined by EUNIS class A, are distinguished from other ecosystems by their direct connection to the ocean. Most of these systems are either not covered by plants or fully aquatic, and therefore out of the scope of this paper. Salt marshes were considered by Bobbink *et al.* (1996) in the wetland section, although at that time it was not considered necessary to set a critical load for these systems with very open nutrient cycles. However, recent evidence has indicated the need to reconsider this.

9.2 Coastal salt marshes and saline reedbeds (A2.6)

Salt marshes have developed on the shores of sheltered coasts or in estuaries, and are typically intertidal, i.e. they are located between lowest and highest tide, and are periodically covered by salt water. They are characterized by an open nutrient cycle, receiving large amounts of nutrients from surface water, and exporting similarly large amounts of nutrients through surface water and denitrification (for nitrogen). This has led to the conclusion, in accordance with Morris (1991), that these systems would not be vulnerable to effects of increased atmospheric nitrogen deposition, at least not at current deposition rates. However, nitrogen limitation has been demonstrated in salt marshes at the island of Schiermonnikoog, the Netherlands (Kiehl *et al.,* 1997) and in Norfolk, UK (Jefferies and Perkins, 1977). During salt marsh succession, nitrogen accumulates in organic material, and nitrogen mineralization increases with marsh age as shown by Olff *et al.* (1993) and Van Wijnen *et al.* (1999). This accumulation of nitrogen is considered as a major driving force behind succession, as competition for nutrients is replaced by competition for light.

Van Wijnen and Bakker (1999) added 50 kg N ha^{-1} yr^{-1} for three years to a 15-year old salt marsh (EUNIS code A2.64) and a 100-year old salt marsh (EUNIS code A2.63) in the Netherlands (current atmospheric deposition 15-20 kg N ha^{-1} yr^{-1}). Biomass increased significantly from the first growing season on in the young salt marsh, and continued to be higher in all three years (Fig 9.1). However, in the studied older salt marsh the addition of 50 kg N ha^{-1} yr^{-1} had no significant effect on biomass, although the response to a much higher nitrogen application (250 kg N ha^{-1} yr^{-1}) showed that the vegetation was at least partly nitrogen limited (Van Wijnen and Bakker, 1999). Fertilization increased biomass of late-successional species and decreased the floristic differences between the young and old marshes. However, these species composition responses were measured only in the combined high nitrogen (250 kg N ha^{-1} yr^{-1}) and high phosphorus treatment, compared with the control situation. Thus the effect of nitrogen on species composition could not be isolated from the effect of phosphorus. However, as the effects of added phosphorus on biomass were either non significant or quite small compared with the effects of nitrogen, there is a clear indication that increased nitrogen availability does increase the rate of succession. As the successional age of these salt marshes is an important determinant of their quality as staging areas for Brent and Barnacle geese (*Branta bernicla* and *Branta leucopsis* respectively) (Bakker, 1985), increases in nitrogen deposition might decrease the surface of early successional vegetation on the marsh and thereby the foraging area that is suitable for these migratory birds.

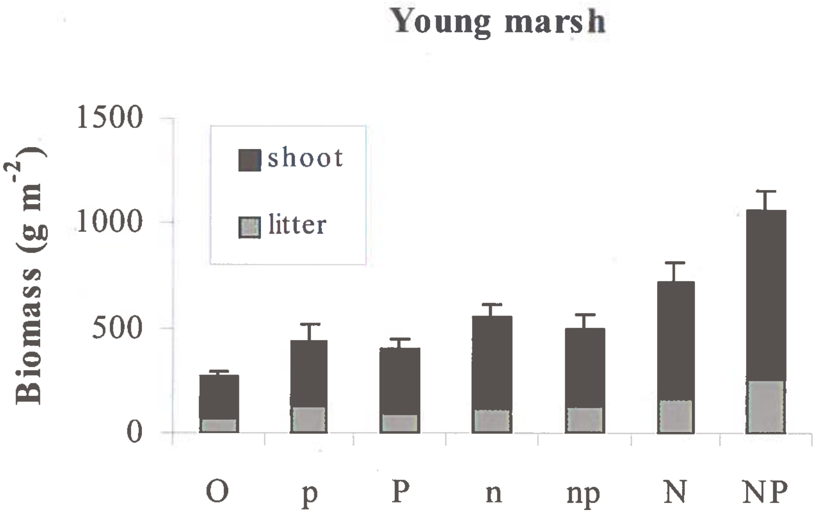

Figure 9.1. Above-ground biomass of young salt marsh vegetation (A2.64) in the Netherlands after 1-year addition of differential nutrients; small n = 50 kg N ha^{-1} yr^{-1}, capital N = 250 kg N ha^{-1} yr^{-1}, small p = 20 kg P ha^{-1} yr^{-1}, capital P = 100 kg P ha^{-1} yr^{-1} (adapted from Van Wijnen and Bakker, 1999).

During primary succession nitrogen accumulates in organic material in the soil of the salt marsh. The increasing nitrogen availability is one of the main driving forces for succession. Increased nitrogen deposition will accelerate this natural process, but because it does not affect the accretion rate of salt marshes, this will result in a net loss of salt marshes of a low successional age (EUNIS codes A2.65 and A2.64). Addition of 50 kg N ha^{-1} yr^{-1} to a current deposition of 15-20 kg N ha^{-1} yr^{-1} significantly increased biomass production each year over a 3-year period. Furthermore, no change in species composition or diversity was observed in salt marshes of NW Germany and the Netherlands, despite the present atmospheric N loads of 15-25 kg N ha^{-1} yr^{-1}. Therefore we propose as expert judgement a critical load for **salt marsh systems (A2.6)** of **30-40 kg N ha^{-1} yr^{-1}**. However, field experiments with lower nitrogen additions over an equal or preferably longer period of time are necessary to improve the reliability for this estimate (Table 9.1).

Table 9.1. Empirical critical loads for the effects of excess nitrogen upon marine habitats (A). ## reliable, # quite reliable and (#) expert judgement.

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Early successional salt marsh	A2.64 and A2.65	30-40	(#)	Increase late-successional species, increase productivity

10. Conclusions and gaps in knowledge

Within the Convention on Long-range Transboundary Air Pollution covering the UNECE region, procedures have been developed to set and to map critical loads for airborne nitrogen deposition. Both the steady-state mass balance method and the empirical approach have been used within this Convention. Based on observed changes in the structure and function of ecosystems, reported in a range of publications, empirical critical loads for nitrogen were evaluated for specific receptor groups of natural and semi-natural ecosystems in both 1992 and 1996 (Bobbink *et al.*, 1992a; 1996). They were also used for the development of the second edition of the Air Quality Guidelines for Europe of the World Health Organisation Regional Office for Europe (WHO, 2000). More recently, it was recognised that considerable new insights into, and data on, the impacts of nitrogen deposition on natural and semi-natural ecosytems vegetation have become available since the compilation of the last background paper in the mid 1990s.

In this updating procedure we have used a similar *"empirical approach"* as for the earlier background documents (see Chapter 2). For this purpose, we have firstly collected, as completely as possible, European publications on the effects of nitrogen in natural and semi-natural ecosystems from the period 1996 to mid 2002. We used peer-reviewed publications, book chapters, nationally published papers and "grey" reports of institutes or organisations, if available by request. In this background document, the groups of natural and semi-natural ecosystems have been reclassified (table 2.1) to facilitate the mapping procedure and ordered according the EUNIS habitat classification for Europe (Davies and Moss, 2002; see the separate background document of Hall *et al.*, 2003 for details). In Chapter 3 to 9 the effects of nitrogen enrichment have been evaluated per EUNIS class and the empirical nitrogen critical loads updated with newly published evidence. The empirical critical loads for nitrogen have been reformulated (if necessary) and are summarized in separate tables per chapter. The text of this background document was discussed and optimized at the November 2002 expert workshop held in Berne (Switzerland) under the Convention on Long-range Transboundary Air Pollution. At the end of this meeting, the updated list of empirical N critical loads for natural and semi-natural ecosystems was approved by full consensus (table 10.1).

Table 10.1. Overview of empirical critical loads for nitrogen deposition (kg N ha^{-1} yr^{-1}) to natural and semi-natural group of ecosystems. Classification of habitats according to EUNIS (except for forests). **## reliable; # quite reliable and (#) expert judgement.**

Ecosystem type	EUNIS-code	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Forest habitats (G)				
Soil processes				
Deciduous and coniferous	-	10-15	#	Increased N mineralization, nitrification
Coniferous forests	-	10-15	##	Increased nitrate leaching
Deciduous forests	-	10-15	(#)	Increased nitrate leaching
Trees				
Deciduous and coniferous	-	15-20	#	Changed N/macro nutrients ratios, decreased P, K, Mg and increased N concentrations in foliar tissue
Temperate forests	-	15-20	(#)	Increased susceptibility to pathogens and pests, change in fungistatic phenolics
Mycorrhiza				
Temperate and boreal forests	-	10-20	(#)	Reduced sporocarp production, changed/reduced below-ground species composition
Ground vegetation				
Temperate and boreal forests	-	10-15	#	Changed species composition, increase of nitrophilous species, increased susceptibility to parasites
Lichens and algae				
Temperate and boreal forests	-	10-15	(#)	Increase of algae, decrease of lichens
Overall				
Temperate forests	-	10-20	#	Changes in soil processes, ground vegetation, mycorrhiza, increased risk of nutrient imbalances and susceptibility to parasites
Boreal forests	-	10-20	#	Changes in soil processes, ground vegetation, mycorrhiza, increased risk of nutrient imbalances and susceptibility to parasites
Heathland, scrub and tundra habitats (F)				
Tundra	F1	5-10^{a}	#	Changes in biomass, physiological effects, changes in species composition in moss layer, decrease in lichens
Arctic, alpine and sub-alpine scrub habitats	F2	5-15^{a}	(#)	Decline in lichens, mosses and evergreen shrubs
Northern wet heath	F4.11			
• 'U' *Calluna*-dominated wet heath (upland moorland)	F4.11	10-20^{a}	(#)	Decreased heather dominance, decline in lichens and mosses
• 'L' *Erica tetralix* dominated wet heath	F4.11	10-25^{a,b}	(#)	Transition heather to grass
Dry heaths	F4.2	10-20^{a,b}	##	Transition heather to grass, decline in lichens

Grasslands and tall forb habitats (E)				
Sub-atlantic semi-dry calcareous grassland	E1.26	15-25	##	Increase tall grasses, decline in diversity, increased mineralization, N leaching
Non-mediterranean dry acid and neutral closed grassland	E1.7	10-20	#	Increase in graminoids, decline typical species
Inland dune pioneer grasslands	E1.94	10-20	(#)	Decrease in lichens, increase biomass
Inland dune siliceous grasslands	E1.95	10-20	(#)	Decrease in lichens, increase biomass, increased succession
Low and medium altitude hay meadows	E2.2	20-30	(#)	Increase in tall grasses, decrease in diversity
Mountain hay meadows	E2.3	10-20	(#)	Increase in nitrophilous graminoids, changes in diversity
Moist and wet oligotrophic grasslands	E3.5			
• *Molinia caerulea* meadows	E3.51	15-25	(#)	Increase in tall graminoids, decreased diversity, decrease of bryophytes
• Heath (*Juncus*) meadows and humid (*Nardus stricta*) swards	E3.52	10-20	#	Increase in tall graminoids, decreased diversity, decrease of bryophytes
Alpine and subalpine grasslands	E4.3 and E4.4	10-15	(#)	Increase in nitrophilous graminoids, biodiversity change
Moss and lichen dominated mountain summits	E4.2	5-10	#	Effects upon bryophytes or lichens
Mire, bog and fen habitats (D)				
Raised and blanket bogs	D1	5-10[a,c]	##	Change in species composition, N saturation of *Sphagnum*
Poor fens	D2.2[d]	10-20	#	Increase sedges and vascular plants, negative effects on peat mosses
Rich fens	D4.1[e]	15-35	(#)	Increase tall graminoids, decrease diversity, decrease of characteristic mosses
Mountain rich fens	D4.2	15-25	(#)	Increase vascular plants, decrease bryophytes
Inland surface water habitats (C)				
Permanent oligotrophic waters	C1.1			
• Softwater lakes	C1.1	5-10	##	Isoetid species negatively affected
• Dune slack pools	C1.16	10-20	(#)	Increased biomass and rate of succession
Coastal habitat (B)				
Shifting coastal dunes	B1.3	10-20	(#)	Biomass increase, increase N leaching
Coastal stable dune grasslands	B1.4	10-20	#	Increase tall grasses, decrease prostrate plants, increased N leaching
Coastal dune heaths	B1.5	10-20	(#)	Increase plant production, increase N leaching, accelerated succession
Moist to wet dune slacks	B1.8	10-25	(#)	Increased biomass tall graminoids
Marine habitats (A)				
Pioneer and low-mid salt marshes	A2.64 and A2.65	30-40	(#)	Increase late-successional species, increase productivity

[a] use towards high end of range at phosphorus limitation, and towards lower end if phosphorus is not limiting;

[b] use towards high end of range when sod cutting has been practiced, use towards lower end of range with low intensity management;

[c] use towards high end of range with high precipitation and towards low end of range with low precipitation;

[d] for D2.1 (quaking fens and transition mires): use lower end of range (#) and for D2.3 (valley mires): use higher end of range (#);

[e] for high latitude or N-limited systems: use lower end of range.

Furthermore, it is suggested to the different countries, where insufficient national data for specific national ecosystems are available, to use the lower, middle or upper part of the ranges of the empirical nitrogen critical loads for (semi-)natural ecosystem groups according to the general relationships between abiotic factors and critical loads for nitrogen as given in table 10.2.

Table 10.2. Suggestions to use lower, middle or upper part of the set critical loads of terrestrial ecosystems (excluding wetlands), if national data are insufficient.

Action	Temperature/ Frost period	Soil wetness	Base cation availability	P limitation	Management intensity
Move to lower part	COLD/LONG	DRY	LOW	N-LIMITED	LOW
Use middle part	INTERMED	NORMAL	INTERMED	UNKNOWN	USUAL
Move to higher part	HOT/NONE	WET	HIGH	P-LIMITED	HIGH

Most of the earth's biodiversity is present in semi-natural and natural ecosystems. It is thus crucial to control the atmospheric nitrogen loads, in order to prevent negative effects on these semi-natural and natural systems. Fine resolution maps of the sensitive ecosystems of high conservation value are needed per country to map nitrogen critical loads for these systems. It is advised to use both the mass balance and empirically derived nitrogen critical loads for forest ecosystems and other ecosystems for which data needed for the application of steady state models is available. If the two approaches yield different values, the one with the lowest values should be used until the background for this difference has been clarified.

Countries are advised to identify those receptor ecosystems of high sensitivity within the mentioned EUNIS classification relating to their individual interest. Effort should be directed to produce fine resolution maps of sensitive ecosystems of high conservation value. At this moment the empirical nitrogen critical loads have been set in values of total atmospheric nitrogen (kg N ha^{-1} yr^{-1}). More information is needed on the relative effects of oxidized and reduced nitrogen deposition. It was emphasized during the last two expert meetings held under the Convention that there is increasing evidence of NH_x having greater impact than NO_y. Particularly, bryophytes and lichens in a number of ecosystems, and several, mostly weakly buffered, ecosystems of EUNIS class F, E, C and B are (probably) more sensitive to deposition of reduced N. It is, however, at present not possible to set critical loads for both forms of N, separately.

Serious gaps in knowledge exist on the effects of enhanced nitrogen deposition (NO_y & NH_x) on semi-natural and natural ecosystems, although considerably progress has been made in several habitat groups from 1996 to 2002. The following gaps in knowledge have been recognized as most important:

- research/data collection is required to establish a critical load for the following ecosystems: steppe grasslands, all Mediterranean vegetation types, wet-swamp forests, many mire & fens, several coastal habitats and high altitude systems;
- more research is needed in all distinguished EUNIS items with expert judgement or few research;

- impacts of nitrogen enrichment in (sensitive) freshwater and shallow marine ecosystems needs further research and are sometimes overlooked;
- additional effort is needed to allocated observed nitrogen effects to the appropriate EUNIS forest subtypes (division 2 & 3);
- the EUNIS classification needs clarification/adjustment with respect to some grasslands groups, Nordic bogs and mires and surface water habitats;
- the possible differential effects of the deposited nitrogen species (NO_y or NH_x) are insufficiently known to make a differentation between these nitrogen species for critical load establishment;
- in order to refine current critical loads, long-term (>3-5 yrs) nitrogen addition experiments with a high resolution of treatments between 5 and 50 kg N ha^{-1} yr^{-1} at low background regions or in mesocosms are useful. This would increase the certainty of deriving critical loads when the lowest treatment level considerably exceeds the critical load.

In conclusion, it is crucial to understand the long-term effects of increased nitrogen deposition on ecosystem processes in a representative range of ecosystems. It is thus very important to quantify the effects of nitrogen loads by manipulation of nitrogen inputs in long-term ecosystem studies in unaffected and affected areas. These data are essential to validate the set critical loads and to develop robust dynamic ecosystem models and/or multiple correlative species models, which are reliable enough to calculate critical loads for nitrogen deposition in (semi-)natural ecosystems and to predict (natural) recovery rates for nitrogen-affected systems.

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Harmonisation of ecosystem definitions using the EUNIS habitat classification

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Introduction and background

In 2000 the UK National Focal Centre for Critical Loads Modelling and Mapping carried out a study as a "contribution in kind" for the International Cooperative Programme on Modelling and Mapping (ICPMM), to harmonise the definitions of ecosystems for which countries calculate and submit national critical loads data (Hall, 2001). The study showed that countries identify and map their sensitive ecosystems from a variety of data sources, ranging from land cover maps to aerial photographs or survey data. No information had previously been collated on the methods and data used to define these sensitive ecosystems; countries submitted an ecosystem name with their critical loads data to the Coordination Centre for Effects (CCE) and one made the assumption that all ecosystems with the same name represented similar ecosystems. A method to harmonise the definitions of ecosystems was required to enable comparisons of critical loads data and maps for similar ecosystems in different countries. The method selected and subsequently adopted by the CCE and the ICPMM, was EUNIS: the EUropean Nature Information System (Davies & Moss, 1999 and 2002). EUNIS is an hierarchical habitat classification system which can be used to provide a common framework for consistently defining and classifying ecosystems for current and future critical loads work. The EUNIS system has been developed for use across Europe and includes links to the habitats under Annex 1 of the EU Habitats Directive, enabling the effects of critical loads exceedance to be examined at the European scale for habitats of particular importance under the Directive.

Overview of EUNIS

EUNIS was developed at the Centre for Ecology and Hydrology (Monks Wood) for the European Environment Agency's (EEA) European Topic Centre on Nature Protection and Biodiversity, as a pan-European tool of the EEA. It is a successor to both the CORINE and Palaearctic habitat classifications and uses a common language and a common framework with links to other classifications, including the CORINE Land Cover Map (Davies & Moss, 2002). Relationships have also been derived between the EUNIS classes and the habitats listed in Annex 1 of the EU Habitats Directive (Moss & Davies, 2002). In addition, where appropriate, links to the broad and priority habitats of the UK Biodiversity Action Plan (BAP) are provided with the online (see below) habitat descriptions.

EUNIS is an hierarchical classification based upon ten major (level 1) habitat classes (Table 1), which are then sub-divided into more specific habitats (levels 2 and 3). The level 1 classes and the classes below them at levels 2 and 3, are identified from "criteria diagrams" which provide clear criteria for each division. The criteria diagram for defining the ten level 1 habitat classes is shown in figure 1, and figures 2 and 3 show the criteria diagrams for defining the level 2 and 3 classes for some of the heathland, scrub and tundra habitats. The diagrams should only be used in conjunction with the detailed explanatory notes which are available on the web site. Further lower level units (ie, below level 3) are drawn from the habitat units and codes of other classifications, namely the Palearctic, BioMar (marine areas around the UK and Ireland), Mediterranean and Baltic marine habitat classifications. EUNIS can therefore be applied at different levels of complexity and is relatively easy for the non-expert to use.

Table 1: The ten level 1 EUNIS habitats

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

The full hierarchical structure of the EUNIS classification, together with habitat descriptions, criteria diagrams for each level division and downloadable documents and reports, is available on the EUNIS web site:
http://mrw.wallonie.be/dgrne/sibw/EUNIS/home.html

The application of EUNIS in critical loads work

Since the proposal and agreement to use EUNIS, National Focal Centres (NFCs) have been encouraged to look at the EUNIS web site and consider its use in assigning habitat codes to their critical loads data. Current and future calls for data from the CCE will request that countries supply appropriate EUNIS habitat codes for the data being submitted. The EUNIS system enables individual countries to assign habitat codes to their critical loads data at an appropriate level depending on the amount of habitat information they hold, for example, E1 for dry grasslands, or E1.7 for dry acid and neutral closed grasslands. As the system includes classes for the different regions of Europe, from arctic and alpine to temperate and Mediterranean, it can be applied by many countries. There are also possibilities for the extension and further development of EUNIS and the authors welcome feedback via the web site.

EUNIS provides a framework for defining and classifying habitats and the inclusion of consistently named habitats with submitted critical loads data will enable the CCE to map critical loads at the European scale for individual habitat types and carry out cross-country comparisons. This will also provide the means for the CCE and the Task Force on Integrated Assessment Modelling to examine the impacts of different abatement strategies on critical loads exceedance by habitat type. This could be particularly important for considering habitats under Annex 1 of the EU Habitats Directive.

The application of EUNIS to the revision of empirical nitrogen critical loads for natural and semi-natural ecosystems

In preparing the background document on updating empirical nitrogen critical loads for natural and semi-natural ecosystems (Bobbink et al, 2002), the authors agreed to use EUNIS as a general framework for the habitats being considered. In reviewing the literature and assigning critical loads, they have grouped the habitats according to seven of the ten level 1 EUNIS categories. Within each category, ranges of critical load values have been assigned to specific habitats where there is sufficient evidence in the literature to do so. For each of these habitats, a EUNIS habitat code has also been assigned that represents the habitat on which the data are based. Ranges of critical load values have been assigned to a total of 31 EUNIS habitat classes. The EUNIS descriptions of these classes are given in Appendix 1. In most cases level 3 habitat classes have been used. However, the woodland habitats have only been defined to level 2 (ie, coniferous woodland, deciduous woodland), whereas for some of the heathland, grassland, water and marine habitats, critical loads have been applied to the more specific level 4 habitats. Figures 4 to10 represent simplified criteria diagrams for all 31 classes, showing only those criteria defining the habitats used.

When applying the revised critical loads to habitats, NFCs will need to consider:

- The criteria and habitat descriptions defining the EUNIS classes for which critical loads have been set.
- If they have sufficient data to adequately map the distributions of all the habitats for which critical loads have been set; some sensitive habitats may only occupy very small areas at the national scale.
- How to select the critical load value for mapping (and exceedance calculation) purposes from within the range given, using the additional guidance provided in Chapter 10 of Bobbink *et al.* (2002).
- In the case of EUNIS habitat codes for woodlands, ensure it is clear to the CCE whether the empirical critical load values submitted refer to effects on trees or effects on ground flora, to enable accurate comparison of results across Europe. This may require modification to the habitat codes used, for example GT1 for broadleaved deciduous trees and GF1 for ground flora under broadleaved deciduous woodland.

The ranges of critical loads values by EUNIS habitat can also be used for site-specific assessments as well as national mapping activities. However, it is then even more important that the evidence given by Bobbink *et al.* (2002) and the EUNIS criteria and habitat descriptions are studied carefully. Although the descriptions of the EUNIS habitats (Appendix 1) often include species lists, it must be remembered that the review and assignment of ranges of critical load values (Bobbink *et al.*, 2002) is based on communities, not individual species. The workshop groups need to critically review these descriptions to ensure they conform to the types of community that these community-level critical loads apply.

The authors compiling the revision of empirical nitrogen critical loads (Bobbink *et al.*, 2002) reported a good agreement between the previous habitat categories of Bobbink *et al.* (1996) and the EUNIS habitat classification now adopted. The main concerns raised by the authors and workshop participants related to:

- A need for further clarification of the wet, mesic and dry grassland categories.
- Acid grasslands in Wales (UK) being classified as dry grasslands.
- No distinction between acid and calcareous grasslands in the mesic grassland category.
- The sub-division of class C1 (surface standing waters) being based on trophic status and excluding an alkalinity gradient.
- The division of the mire, bog and fen category (D) being based on the water source for the habitat.
- The classification of dune slacks as part of the Coastal Class B, while dune slack pools, that are found in coastal dunes, are classified under Class C (Inland Surface Waters).

The key limitation preventing the allocation of critical loads to nitrogen-sensitive EUNIS habitats other than those in Bobbink *et al.* 2002, is a lack of research and data for those habitats.

Conclusions

The EUNIS habitat classification:

- Provides a method of assigning a consistent set of habitat codes to the habitats for which countries calculate critical loads.
- Has proved to be a useful tool for defining and classifying habitats for the revision of empirical nitrogen critical loads.
- Can be related to the Annex 1 habitats of the EU Habitats Directive, so an assessment of the sensitivity and impacts of acidification and eutrophication can be related to habitats acknowledged to be important at the European scale.

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APPENDIX 1: EUNIS habitats to which empirical nitrogen critical loads have been assigned by Bobbink et al. (2002)

The EUNIS habitat descriptions below, quoted from the EUNIS web site, often include species lists. It must be remembered that the review and assignment of ranges of nutrient nitrogen critical load values (Bobbink *et al.*, 2002) is based on communities, not individual species. At the end of this document there is a table showing the relationships between the EUNIS classes given here and some commonly used land cover and habitat classifications.

WOODLAND AND FOREST HABITATS (G)

Habitats are defined as: natural stands of area greater than 0.5ha and crown cover greater than 10% and tree height greater than 5m; natural stands of area less than 0.5ha and crown cover greater than 10% and tree height greater than 5m with more or less natural ground flora (i.e. not heavily influenced by man through management or damage); plantations of area greater than 0.5ha and crown cover greater than 10% and tree height greater than 5m.

G1 Broadleaved deciduous woodland

Broadleaved woodland is defined as wooded land on which more than 75% of the tree crown cover consists of broadleaved species (based on FAO definition).

G3 Coniferous woodland

Coniferous woodland is defined as wooded land on which more than 75% of the tree crown cover consists of coniferous species (based on FAO definition).

G4 Mixed broadleaved and coniferous woodland

Mixed woodland is defined as wooded land on which neither coniferous, nor broadleaved species account for more than 75% of the crown cover.
This code can be assigned if the empirical nitrogen critical loads for both broadleaved and coniferous woodland are the same and the habitat is not dominated by one woodland type.

HEATHLAND, SCRUB & TUNDRA HABITATS (F)

F1 Tundra

Communities of grasses and shrubs developed over permafrost. They are limited to Greenland, Spitzbergen and northern Russia. Communities sometimes composed of the same species in oro-boreal areas remote from the main permafrost region, such as those of continental Fennoscandia and Iceland are listed under units 36 and 31.4. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001))

F2 Arctic, alpine and sub-alpine scrub

Habitats characterised by their temperature regime, they may operate at a variety of geographical and altitudinal scales: cold to cold-temperate climate in the arctic, alpine or subalpine zones.

F4.11 Northern wet heaths

Wet heaths of the northern Atlantic and sub-Atlantic domaine and of the boreal enclaves in the southern Atlantic and sub-Atlantic domaine. They mostly comprise wet and humid heaths of anmoors, gleypodzols and thin peats of the British Isles, the Germano-Baltic plains and neighbouring Hercynian hills, northern and northwestern France, southern Scandinavia, dominated by *Erica tetralix* or *Calluna vulgaris*, sometimes in association with *Scirpus cespitosus* or *Molinia caerulea*, usually accompanied by non-peat building sphagna, in particular, *Sphagnum compactum*, *Sphagnum molle*, *Sphagnum tenellum*. They also include peaty heaths of more southern regions, in western France, the periphery of the Paris Basin and northwestern Iberia, mostly developed in the central, most humid, part of depressions, often in contact with somewhat less humid formations of unit 31.12. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

F4.11 "U" Calluna-dominated wet heaths

Sub-class of F4.11 agreed upon at Berne workshop.

F4.11 "L" Erica-dominated wet heaths

Sub-class of F4.11 agreed upon at Berne workshop.

F4.2 Dry heaths

Xerophile or meso-xerophile heaths on siliceous, podsolic soils in moist Atlantic and sub-Atlantic climates of the plains and low mountains of Western and Central Europe. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

GRASSLAND AND TALL FORB HABITATS (E)

E1.26 Sub-Atlantic semi-dry calcareous grasslands

More or less mesophile, closed formations dominated by perennial, tuft-forming grasses, colonizing relatively deep, mostly calcareous soils in the sub-Atlantic domaine of the *Quercion pubescenti-petraeae* and its northern irradiations and in the sub-Mediterranean mountains of the Italian peninsula, with *Bromus erectus*, *Brachypodium pinnatum*, *Koeleria pyramidata*, *Festuca guestfalica*, *Festuca lemanii*, *Avenula pubescens*, *Sesleria albicans*, *Briza media*, *Carex caryophyllea*, *Carex flacca*, *Gentianella germanica*, *Gentianella ciliata*, *Gentiana cruciata*, *Trifolium montanum*, *Lotus corniculatus*, *Ononis repens*, *Medicago lupulina*, *Ranunculus bulbosus*, *Sanguisorba minor*, *Cirsium acaule*, *Euphrasia stricta*, *Dianthus deltoides*, *Potentilla neumanniana* (*Potentilla tabernaemontani*, *Potentilla verna*), *Anthyllis vulneraria*, *Galium verum*, *Euphorbia brittingeri* (*Euphorbia verrucosa*), *Hippocrepis comosa*, *Helianthemum nummularium*, *Thymus praecox*, *Salvia pratensis*, *Linum catharticum*, *Scabiosa columbaria*, *Centaurea scabiosa*, *Carlina vulgaris*, *Viola hirta*, *Plantago media*, *Primula veris* and numerous orchids such as *Coeloglossum viride*, *Ophrys apifera*, *Ophrys holosericea*, *Ophrys insectifera*, *Ophrys sphegodes*, *Aceras anthropophorum*, *Himantoglossum hircinum*, *Anacamptis pyramidalis*, *Orchis morio*, *Orchis ustulata*, *Orchis militaris*, *Orchis simia*, *Gymnadenia conopsea*, *Platanthera chlorantha*, *Herminium monorchis*, *Dactylorhiza fuchsii*. Generally species-rich, these communities may be overwhelmed by the highly social *Brachypodium pinnatum*. Their range extends from the British Isles, Denmark, the Low Countries and northern Germany to the Cantabric range, the Pyrenees, Catalonia, the southern Alps and the Central Apennines, extending east to the Bohemian Quadrangle, beyond which they are replaced by the vicariant formations of the *Cirsio-Brachypodion*, to the Wienerwald, Styria and Illyria. Forming a bridge between the Mediterranean region and thermophile sites to the north, they can be identified by their high representation of Mediterranean species in the north and of Euro-Siberian ones in the south. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E1.7 Non-Mediterranean dry acid and neutral closed grassland

Closed, dry or mesophile, perennial grasslands occupying acid soils in Atlantic or sub-Atlantic lowland, collinar and montane regions of northern Europe, middle Europe and western Iberia, with *Nardus stricta*, *Festuca filiformis* (*Festuca tenuifolia*), *Festuca ovina*, *Festuca rubra*, *Agrostis capillaris*, *Danthonia decumbens*, *Anthoxanthum odoratum*, *Deschampsia flexuosa*, *Poa angustifolia*, *Galium saxatile*, *Polygala vulgaris*, *Viola canina*, *Meum athamanticum*, *Arnica montana*, *Centaurea nigra*, *Dianthus deltoides*, *Gentianella campestris*, *Chamaespartium sagittale*, *Jasione laevis*, *Potentilla erecta*, *Carex pilulifera*. Any of the grasses listed can dominate or codominate distinctive facies; *Calamagrostis epigejos* or *Carex arenaria* also can invade and dominate some formations. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E1.94 Inland dune pioneer grassland

Formations of unstable Germano-Baltic fluvioglacial inland sands with *Corynephorus canescens*, *Carex arenaria*, *Spergula morisonii*, *Teesdalia nudicaulis* and carpets of fruticose lichens (*Cladonia*, *Cetraria*). Communities of Jutland are rich in *Ammophila arenaria*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E1.95 Inland dune siliceous grassland

Grasslands of more stabilized Germano-Baltic fluvioglacial inland dune systems with *Agrostis spp.* and *Corynephorus canescens* or other acidophilous grasses. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E2.2 Low and medium altitude hay meadows

Mesophile hay meadows of low altitudes of Europe, fertilized and well-drained, with *Arrhenatherum elatius*, *Trisetum flavescens*, *Anthriscus sylvestris*, *Heracleum sphondylium*, *Daucus carota*, *Crepis biennis*, *Knautia arvensis*, *Leucanthemum vulgare*, *Pimpinella major*, *Trifolium dubium*, *Geranium pratense*; they are most characteristic of the Euro-Siberian zone, but extend to Atlantic Iberia, the Cordillera Central and Montseny, to the Apennines and to the supramediterranean zone of Greece. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E2.3 Mountain hay meadows

Often species-rich hay meadows of the montane and subalpine levels of higher mountains of the western Palaearctic deciduous forest zone. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E3.5 Moist or wet oligotrophic grasslands

Humid grasslands on soils very poor in nutrients of the deciduous forest and steppe zones of Eurasia. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E3.51 Molinia caerulea meadows and related communities (Purple moorgrass meadows and related communities)

Humid grasslands of soils poor in nutrients, unfertilized and with a fluctuating water level, of Western Europe, south to northwestern Iberia, of Northern Europe, of Central Europe, locally of western Eastern Europe, dominated by *Molinia caerulea*, with *Succisa pratensis*, *Deschampsia cespitosa*, *Potentilla erecta*, *Allium angulosum*, *Allium suaveolens*, *Stachys officinalis* (*Betonica officinalis*), *Cirsium dissectum*, *Cirsium tuberosum*, *Dianthus superbus*, *Trollius europaeus*, *Galium boreale*, *Gentiana asclepiadea*, *Gentiana pneumonanthe*, *Gladiolus palustris*, *Silaum silaus*, *Selinum carvifolia*, *Inula salicina*, *Iris sibirica*, *Laserpitium prutenicum*, *Lathyrus pannonicus*, *Tetragonolobus maritimus*, *Serratula tinctoria*, *Carex tomentosa*, *Carex panicea*, *Carex pallescens*, *Parnassia palustris*, *Platanthera bifolia*, *Colchicum autumnale*, *Ophioglossum vulgatum*, *Dactylorhiza maculata*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E3.52 Heath Juncus meadows and humid Nardus stricta swards (Heath rush meadows and humid mat-grass swards)

Humid, often peaty or semi-peaty swards of middle Europe, southwest to northwestern Iberia and east to Lithuania, with *Nardus stricta*, *Juncus squarrosus*, *Festuca ovina*, *Gentiana pneumonanthe*, *Pedicularis sylvatica*, *Scirpus cespitosus* and sometimes *Sphagnum spp.* (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E4.2 - Moss and lichen dominated mountain summits, ridges and exposed slopes

Relatively snow-free exposed summits, slopes and ridges dominated by mosses and lichens.

E4.3 Acid alpine and subalpine grassland

Alpine and subalpine grasslands developed over crystalline rocks and other lime-deficient substrates or on decalcified soils of the boreal mountains, including the Caledonian mountains of Scotland, and of the western Alpids and their satellite Hercynian ranges, with *Armeria alpina*, *Armeria alliacea* (*Armeria montana*), *Euphrasia minima*, *Gentiana alpina*, *Geum montanum*, *Juncus trifidus*, *Lychnis alpina*, *Pedicularis pyrenaica*, *Phyteuma hemisphaericum*, *Pulsatilla alpina* ssp. *sulphurea*, *Ranunculus pyrenaeus*, *Sempervivum montanum*, *Botrychium lunaria*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

E4.4 Calciphilous alpine and subalpine grassland

Alpine and subalpine grasslands of base-rich soils of the western Alpids, very locally, of their satellite Jurassian or Hercynian ranges, and of western Palaearctic boreal mountains, with *Dryas octopetala*, *Gentiana nivalis*, *Gentiana campestris*, *Alchemilla hoppeana*, *Alchemilla conjuncta*, *Alchemilla flabellata*, *Anthyllis vulneraria*, *Astragalus alpinus*, *Aster alpinus*, *Draba aizoides*, *Globularia nudicaulis*, *Helianthemum nummularium* ssp. *grandiflorum*, *Helianthemum oelandicum* ssp. *alpestre*, *Pulsatilla alpina* ssp. *alpina*, *Phyteuma orbiculare*, *Astrantia major*, *Polygala alpestris*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

MIRE, BOG AND FEN HABITATS (D)

D1 Raised and blanket bogs

Blanket bogs follow but do not depend upon the topography, developing on flat or gently sloping ground with poor surface drainage. Raised bogs are dependent upon the topography for their initial development, forming over depressions or on slopes. Raised bogs include a number of topographic types such as saddle and sloping bogs. They completely or primarily depend on rainwater (ombrogenous) as their water source.

D2.2 - Poor fens

Topogenous or soligenous valley, basin or spring mire systems fed by waters poor in bases. As in the rich fens, the water level is at or near the surface of the substratum and peat formation is infra-aquatic. The mire communities themselves, dominated by small sedges and brown mosses or sphagna, belong to the *Caricetalia fuscae*, but, in large fen systems, they are accompanied by acidocline wet grasslands (*Molinietalia caeruleae*), large sedge beds (*Magnocaricion*) and reed or related communities (*Phragmition*). Sphagnum hummocks (unit 51.11) form locally and transition mires (unit 54.5) or aquatic (unit 22.3), amphibian (unit 22.2) and spring (unit 54.1) communities colonize small depressions. Thus, codes from all the above categories are used in conjunction with the ones below to completely describe the fen. The subdivisions listed here are, in any case, understood to include, besides strict mire communities, their transitions to humid grasslands, and groupings phytosociologically affiliated with *Molinion* associations, but rich in species of the *Caricion fuscae*, provided they are integrated in a fen system (somewhat paralleling the *Junco acutiflori-Caricetalia nigrae* of Rameau *et al.*, 1989). Acidic fen communities also occur on small surfaces or within mosaics in other ecosystems, in particular in typical humid grasslands (37), humid woodlands and thickets (44), decalcified dune slacks (unit 16.3) and spring systems (unit 54.1). Their presence can be indicated by codes from this unit used in conjunction with the relevant main codes. Characteristic species of acidic mire communities are *Carex canescens*, *Carex echinata*, *Carex nigra*, *Eriophorum angustifolium*, *Eriophorum scheuchzeri*, *Scirpus cespitosus*, *Juncus filiformis*, *Agrostis canina*, *Viola palustris*, *Cardamine pratensis*, *Ranunculus flammula* and the mosses *Calliergon sarmentosum*, *Calliergon stramineum*, *Calliergon cuspidatum*, *Drepanocladus exannulatus*, *Drepanocladus fluitans*, *Sphagnum recurvum*, *Sphagnum auritum*, *Sphagnum cuspidatum*, *Sphagnum subsecundum*, *Sphagnum apiculatum*, *Sphagnum papillosum*, *Sphagnum russowii*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001))

D4.1 Rich fens, including eutrophic tall-herb fens and calcareous flushes and soaks

Wetlands of the Palaearctic region mostly or largely occupied by peat- or tufa-producing small sedge and brown moss communities developed on soils permanently waterlogged, with a soligenous or topogenous base-rich, nutrient-poor, often calcareous water supply, and with the water table at, or slightly above or below, the substratum. Peat formation, when it occurs, is infra-aquatic. Calciphile small sedges and other Cyperaceae usually dominate the mire communities, which belong to the *Caricion davallianae*, characterized by a usually prominent ""brown moss"" carpet formed by *Campylium stellatum*, *Drepanocladus intermedius*, *Drepanocladus revolvens*, *Cratoneuron commutatum*, *Acrocladium cuspidatum*, *Ctenidium molluscum*, *Fissidens adianthoides*, *Bryum pseudotriquetrum* and others, a grasslike growth of *Schoenus nigricans*, *Schoenus ferrugineus*, *Eriophorum latifolium*, *Carex davalliana*, *Carex flava*, *Carex lepidocarpa*, *Carex hostiana*, *Carex panicea*, *Juncus subnodulosus*, *Scirpus cespitosus*, *Eleocharis quinqueflora*, and a very rich flora including *Tofieldia calyculata*, *Dactylorhiza incarnata*, *Dactylorhiza traunsteineri*, *Dactylorhiza traunsteinerioides*, *Dactylorhiza russowii*, *Dactylorhiza majalis* ssp. *brevifolia*, *Dactylorhiza cruenta*, *Liparis loeselii*, *Herminium monorchis*, *Epipactis palustris*, *Pinguicula vulgaris*, *Pedicularis sceptrum-carolinum*, *Primula farinosa*, *Swertia perennis*. Wet grasslands (*Molinietalia caerulaea*, 37), tall sedge beds (*Magnocaricion*, 53.2), reed formations (*Phragmition*, 53.1), fen sedge beds (*Cladietum mariscae*, 53.3), may form part of the fen system, with communities related to transition mires (54.5, 54.6) and amphibious or aquatic vegetation (22.3, 22.4) or spring communities (54.1) developing in depressions. The subunits below, which can, alone or in combination, and together with codes selected from the categories just mentioned, specify the composition of the fen, are understood to include the mire communities *sensu stricto* (*Caricion davallianae*), their transition to the *Molinion*, and assemblages that, although they may be phytosociologically referrable to alkaline *Molinion* associations, contain a large representation of the *Caricion davallianae* species listed, in addition to being integrated in the fen system; this somewhat parallels the definition of an integrated class *Molinio-Caricetalia davallianae* in Rameau *et al.*, 1989. Outside of rich fen systems, fen communities can occur on small surfaces in dune slack systems (16.3), in transition mires (54.5), in wet grasslands (37), on tufa cones (54.121) and in a few other situations. The codes below can be used, in conjunction with the principal code relevant, to signal their presence. Rich fens are exceptionally endowed with spectacular, specialized, strictly restricted species. They are among the habitats that have undergone the most serious decline. They are essentially extinct in several regions and gravely endangered in most. A very few large systems remain, in particular in pre-Alpine Bavaria, in the Italian pre-Alps,in collinar and montane eastern France, in northeastern Germany, in the coastal marshes of northern France, in southeastern and northern England, in Wales and in Ireland. " (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001))

D4.2 - Basic mountain flushes and streamsides, with a rich arctic-montane flora

Rare Alpine, peri-Alpine, northern British and periarctic pioneer communities colonizing gravelly, sandy, stony, sometimes somewhat argilous or peaty, calcareous sedimentary substrates soaked by cold water, in moraines and on the edge of springs, rivulets, glacial torrents of the alpine or subalpine levels, or on alluvial sands of pure, cold, slow-flowing rivers and calm backwaters. The highly characteristic constituents, with a boreoarctic or glacial relict distribution, are *Carex bicolor*, *Carex microglochin*, *Carex maritima*, *Carex atrofusca*, *Carex vaginata*, *Kobresia simpliciuscula*, *Scirpus pumilus*, *Juncus arcticus*, *Juncus alpinoarticulatus*, *Juncus castaneus*, *Juncus triglumis*, *Typha minima*, *Typha lugdunensis*, *Typha shuttleworthii*, *Tofieldia pusilla*; they are often accompanied by *Carex davalliana*, *Carex dioica*, *Carex capillaris*, *Carex panicea*, *Carex nigra*, *Blysmus compressus*, *Eleocharis quinqueflora*, *Scirpus cespitosus*, *Primula farinosa*, *Equisetum variegatum*, *Drepanocladus intermedius*, *Campylium stellatum*. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

INLAND SURFACE WATER HABITATS (C)

C1.1 Permanent oligotrophic lakes, ponds and pools

Oligotrophic waters, of low nutrient status, usually on hard, acid rock with high oxygen concentration in the hypolimnion.

C1.16 Dune-slack pools

Fresh-water aquatic communities of permanent dune-slack water bodies of the Palaearctic region. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

COASTAL HABITATS (B)

B1.3 Shifting coastal dunes

Mobile sands of the coasts of the boreal, nemoral, steppe, Mediterranean and warm-temperate humid zones, unvegetated or occupied by open grasslands; they may form tall dune ridges or, particularly along the Mediterranean and the Black Sea, be limited to a fairly flat upper beach, still subject in part to inundation. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

B1.4 Coastal stable dune grassland (grey dunes)

Fixed or semi-fixed dunes of the coasts of the boreal, nemoral, steppe, mediterranean and warm-temperate humid zones of the Palaearctic region, with the perennial grasslands, chamaephyte-dotted grasslands, forblands, subshrub or succulent communities that stabilize them and the therophyte communities that may occupy the grassland clearings. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).

B1.5 Coastal dune heaths

Dune heath - predominantly ericaceous dwarf shrubs.

B1.8 - Moist and wet dune slacks

Humid depressions of the coastal dunal systems of the Palaearctic region. The most important habitats are included in the subordinate units. If the divisions proposed are not sufficient, appropriate codes from units 22.4, 22.3, 54.2, 54.4, 53 can be used in conjunction with them. Humid dune-slacks are extremely rich and specialized habitats, very threatened by the lowering of water tables. (Source: Devillers, P., Devillers-Terschuren, J. and Vander Linden, C. (2001)).
Note that dune slack pools are categorised under C1.1.

MARINE HABITATS (A)

A2.6 Coastal saltmarshes and saline reedbeds

Angiosperm-dominated stands of vegetation, occurring on the extreme upper shore of sheltered coasts and periodically covered by high tides. The vegetation develops on a variety of sandy and muddy sediment types and may have admixtures of coarser material. The character of the saltmarsh communities is affected by height up the shore, resulting in a zonation pattern related to the degree or frequency of immersion in seawater. Saltmarsh vegetation is generally well studied; its classification is fully covered by the UK National Vegetation classification, where 26 types are defined (Rodwell, 2000).

A2.64 Low-mid saltmarshes

A2.65 Pioneer saltmarshes

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Devillers, P., Devillers-Terschuren, J. and Vander Linden, C., 2001

Pierre Devillers, Jean Devillers-Terschuren and Charles Vander Linden. Palaearctic Habitats. PHYSIS Data Base, 2001 update.

Rodwell, J. S., ed. 2000. British plant communities. Volume 5. Maritime communities and vegetation of open habitats. Cambridge, Cambridge University Press.

Table showing the relationship between the EUNIS habitat classes and (a) the classes of the CORINE land cover map; (b) the Palaearctic classification; (c) UK Biodiversity Action Plan Broad Habitats; (d) UK Biodiversity Action Plan Priority Habitats.

EUNIS class	CORINE land cover class	Palaearctic Classification*	UK Biodiversity Broad Habitat*	UK Biodiversity Priority Habitat*
G1 Broadleaved deciduous woodland	2.2.2 Fruit trees & berry plantations 3.1.1 Broadleaved forest	<41 Broadleaved deciduous forest #44 Temperate riverine & swamp forests & brush	#1 Broadleaved, mixed & yew woodland	
G3 Coniferous woodland	3.1.2 Coniferous forest	<42 Temperate coniferous forest	#2 Coniferous woodland	
G4 Mixed broadleaved & coniferous woodland	3.1.3 Mixed forest	=43 Temperate mixed forests	#1 Broadleaved, mixed & yew woodland	
F1 Tundra	3.3.3 Sparsely vegetated areas	=39 Tundra		
F2 Arctic, alpine & sub-alpine scrub	3.2.2 Moors & heathland 3.3.3 Sparsely vegetated areas			
F4.11 Northern wet heaths	3.2.2 Moors & heathland	=31.11 Northern wet heaths		
F4.2 Dry heaths	3.2.2 Moors & heathland	=31.2 European dry heaths		
E1.26 Sub-Atlantic semi-dry calcareous grasslands	3.2.1 Natural grassland	=34.32 Sub-Atlantic semi-dry calcareous grasslands		# Upland calcareous grassland
E1.7 Non-Mediterranean dry acid & neutral closed grassland	3.2.1 Natural grassland	=35.1 Atlantic closed acidophilous grasslands	>8 Acid grassland	# Lowland dry acid grassland
E1.94 Inland dune pioneer grassland	3.3.1 Beaches, dunes & sand plains	=64.11 Inland dune pioneer grassland		
E1.95 Inland dune siliceous grasslands	3.3.1 Beaches, dunes & sand plains	=64.12 Inland dune siliceous grasslands		
E2.2 Low & medium altitude hay meadows	3.2.1 Natural grassland	=38.2 Lowland & collinear hay meadows	>6 Neutral grassland	# Lowland meadows
E2.3 Mountain hay meadows	3.2.1 Natural grassland	=38.3 Mountain hay meadows	>6 Neutral grassland	# Upland hay meadows
E3.5 Moist or wet oligotrophic grasslands	3.2.1 Natural grassland	=37.3 Oligotrophic humid grasslands	>8 Acid grassland	

EUNIS class	CORINE land cover class	Palaearctic Classification*	UK Biodiversity Broad Habitat*	UK Biodiversity Priority Habitat*
E3.51 Purple moorgrass meadows and related communities	3.2.1 Natural grassland	=37.31 Purple moorgrass meadows and related communities		= Purple moor grass & rush pasture
E3.52 Heath rush meadows and humid mat-grass swards	3.2.1 Natural grassland	=37.32 Heath rush meadows and humid mat-grass swards		
E4.2 Moss & lichen dominated mountain summits	3.3.3 Sparsely vegetated areas		>15 Montane habitats	
E4.3 Acid alpine & sub-alpine grassland	3.2.1 Natural grassland	=36.3 Boreo-Alpic acidophilous alpine grasslands		
E4.4 Calciphilous alpine & sub-alpine grassland	3.2.1 Natural grassland	=36.4 Boreo-Alpic calciphilous alpine grasslands		
D1 Raised and blanket bogs	4.1.2 Peat bogs		=12 Bogs	
D2.2 Poor fens	4.1.1 Inland marshes	=54.4 Acidic fens		? Fens
D4.1 Rich fens	4.1.1 Inland marshes	=54.2 Rich fens		
D4.2 Basic mountain flushes	4.1.1 Inland marshes	=54.3 Arctoalpine riverine swards		
C1.1 Permanent oligotrophic lakes, ponds and pools	5.1.2 Water bodies	<22.15 Lime-rich oligo-mesotrophic water bodies <22.11 Lime-deficient oligotrophic water bodies		
C1.16 Dune-slack pools	3.3.1 Beaches, dunes & sand plains	=16.31 Dune-slack pools		
B1.3 Shifting coastal dunes	3.3.1 Beaches, dunes & sand plains	=16.21 Shifting dunes		
B1.4 Coastal stable dune grassland (grey dunes)	3.3.1 Beaches, dunes & sand plains	=16.22 Grey dunes		
B1.5 Coastal dune heaths	3.2.2 Moors & heathland	<16.24 Heather brown dunes <16.23 Crowberry brown dunes		
B1.8 Moist & wet dune slacks	3.3.1 Beaches, dunes & sand plains	=16.3 Humid dune slacks		
A2.64 Low-mid saltmarshes	4.2.1 Saltmarshes			
A2.65 Pioneer saltmarshes	4.2.1 Saltmarshes	<15.1 Annual salt pioneer swards <15.2 Perennial salt pioneer swards		

* The symbols denote the match between the classes of the different classifications:

= exact equivalence between classifications

> habitat in other system includes that in EUNIS

< habitat in EUNIS includes that in other system

partial overlap between classification units

? relationship now known

Figure 1

Note: Complex habitats may not readily be located as an entity, as they comprise a number of different habitat units. Complexes are listed under code X.

EUNIS Habitat Classification: criteria for Level 1

(number) refers to explanatory notes to the key (following page)

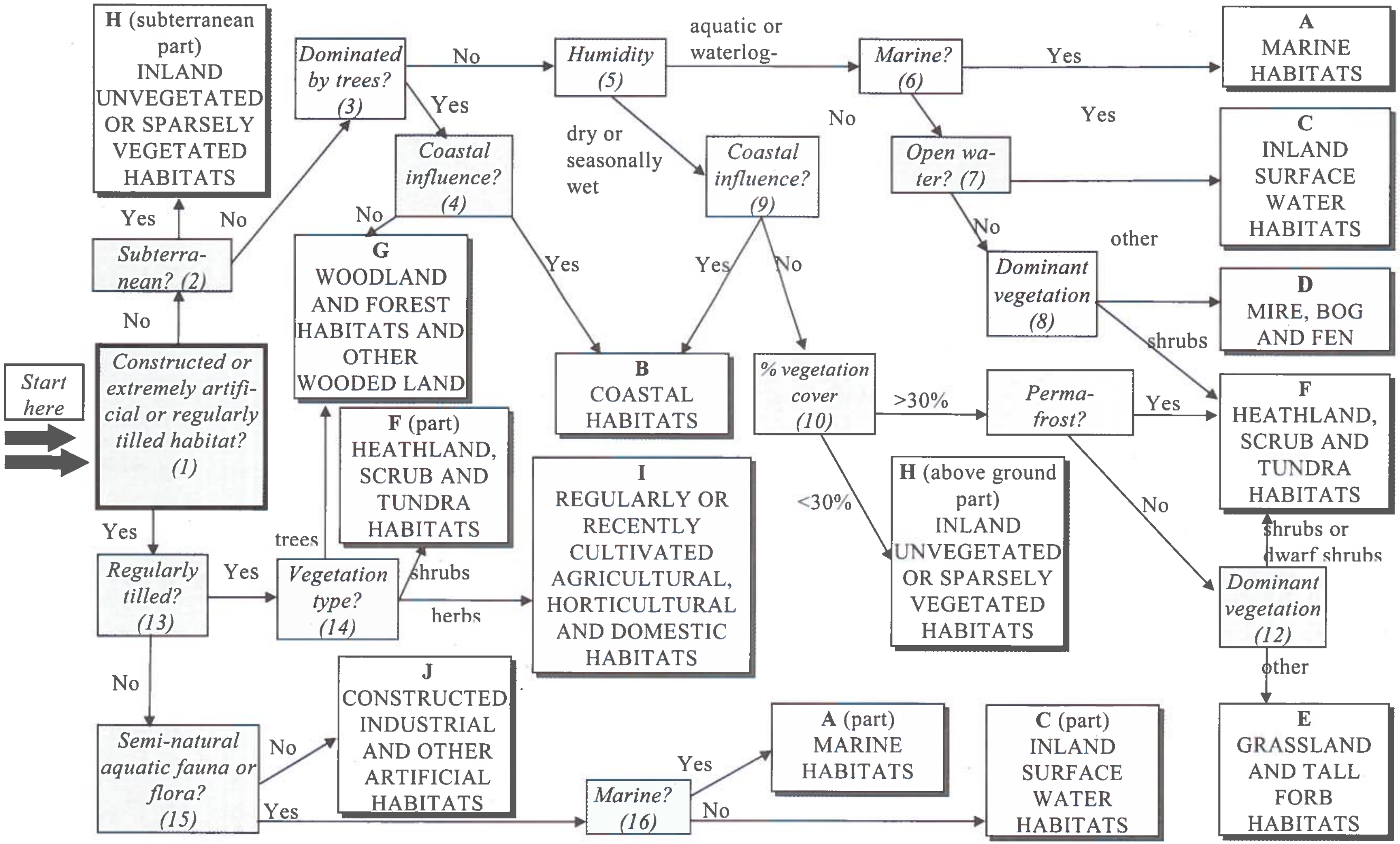


Figure 2

EUNIS Habitat Classification: criteria for heathland, scrub and tundra habitats to Level 2

Note that the key to Level 1 shows two pathways to reach habitat type F: these are recombined here.
(number) refers to explanatory notes to the key (following page)

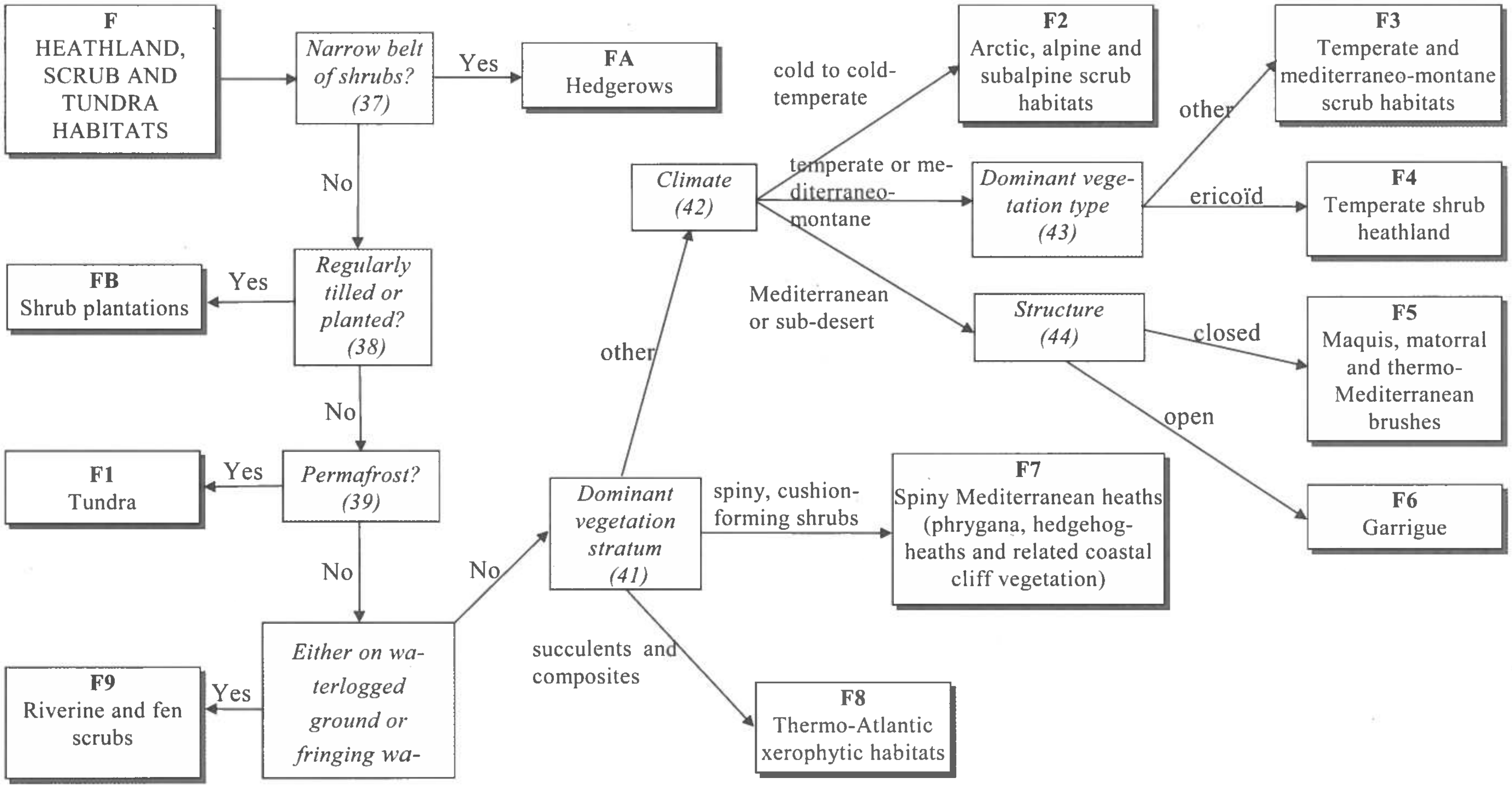


Figure 3

EUNIS Habitat Classification: criteria for temperate shrub heathland (F4) to Level 3

(number) refers to explanatory notes to the key

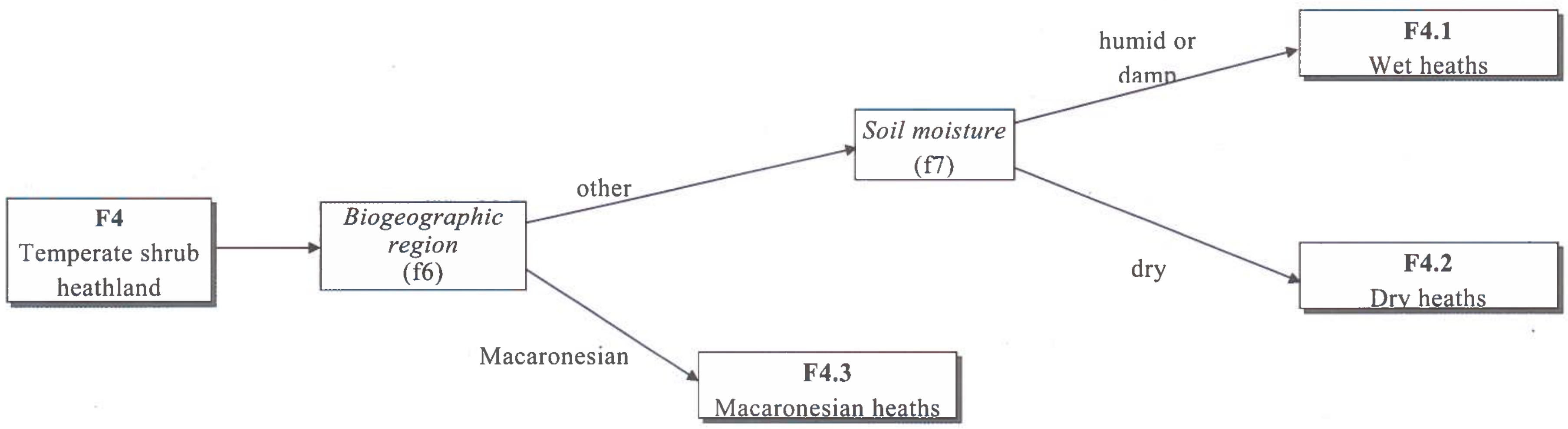


Figure 4

A. MARINE HABITATS

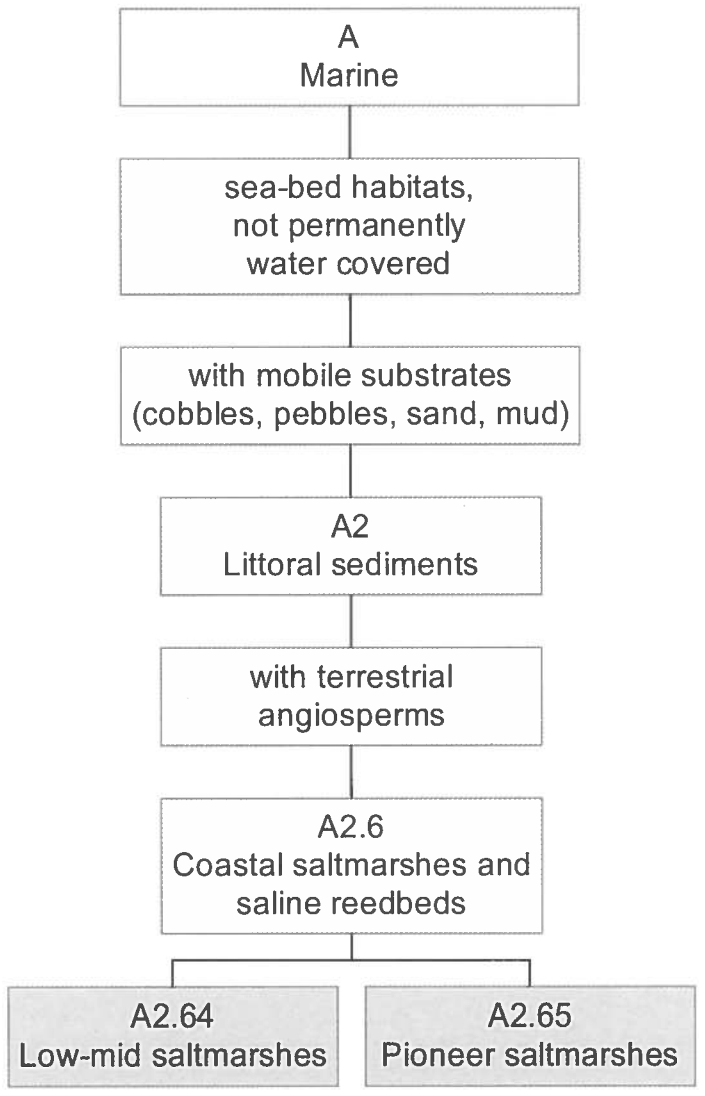


Figure 5

B. COASTAL HABITATS

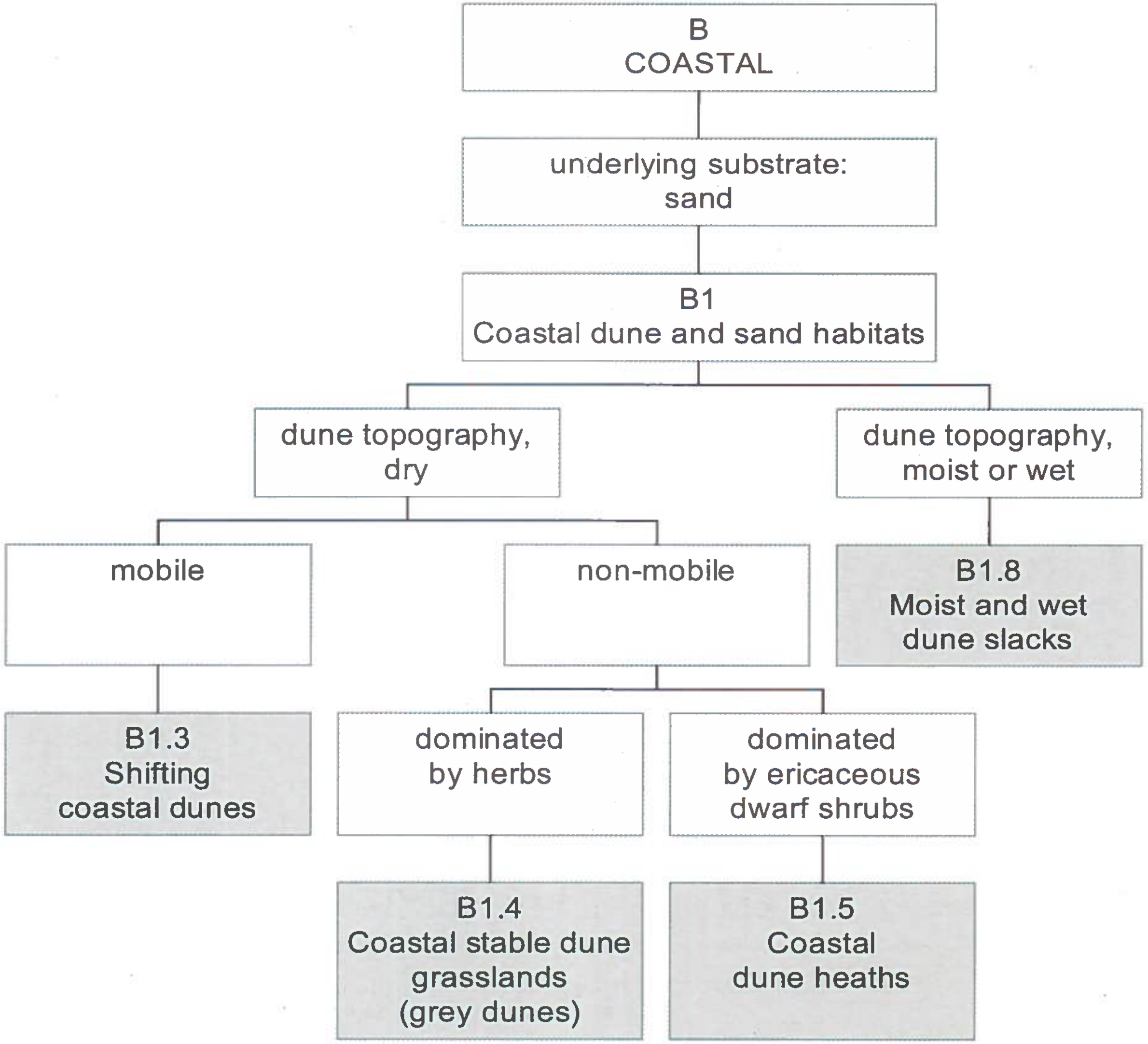


Figure 6

C. INLAND SURFACE WATER

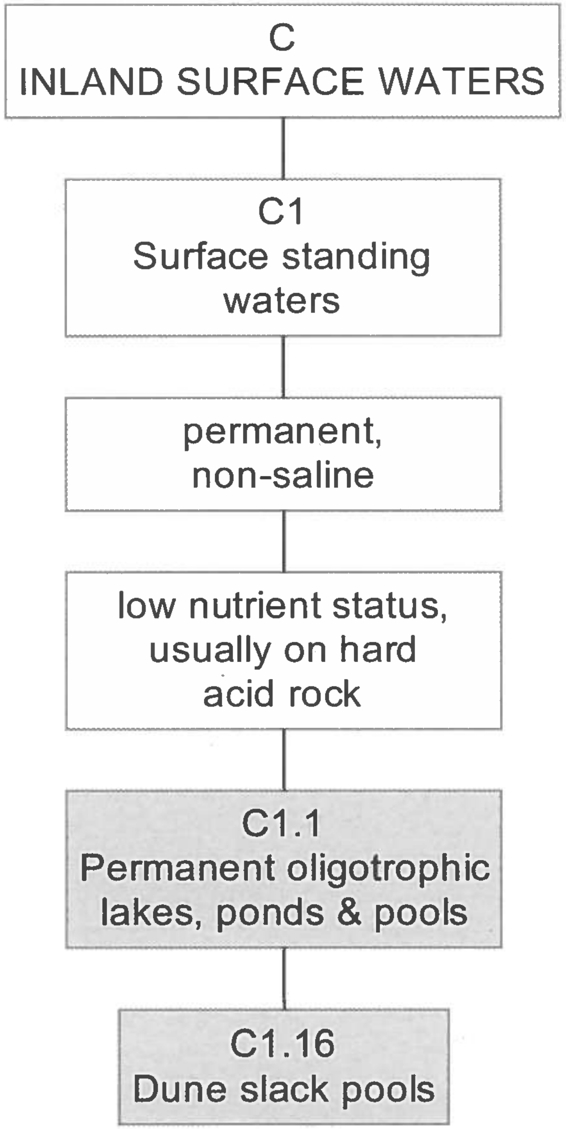


Figure 7

D. MIRE, BOG AND FEN HABITATS

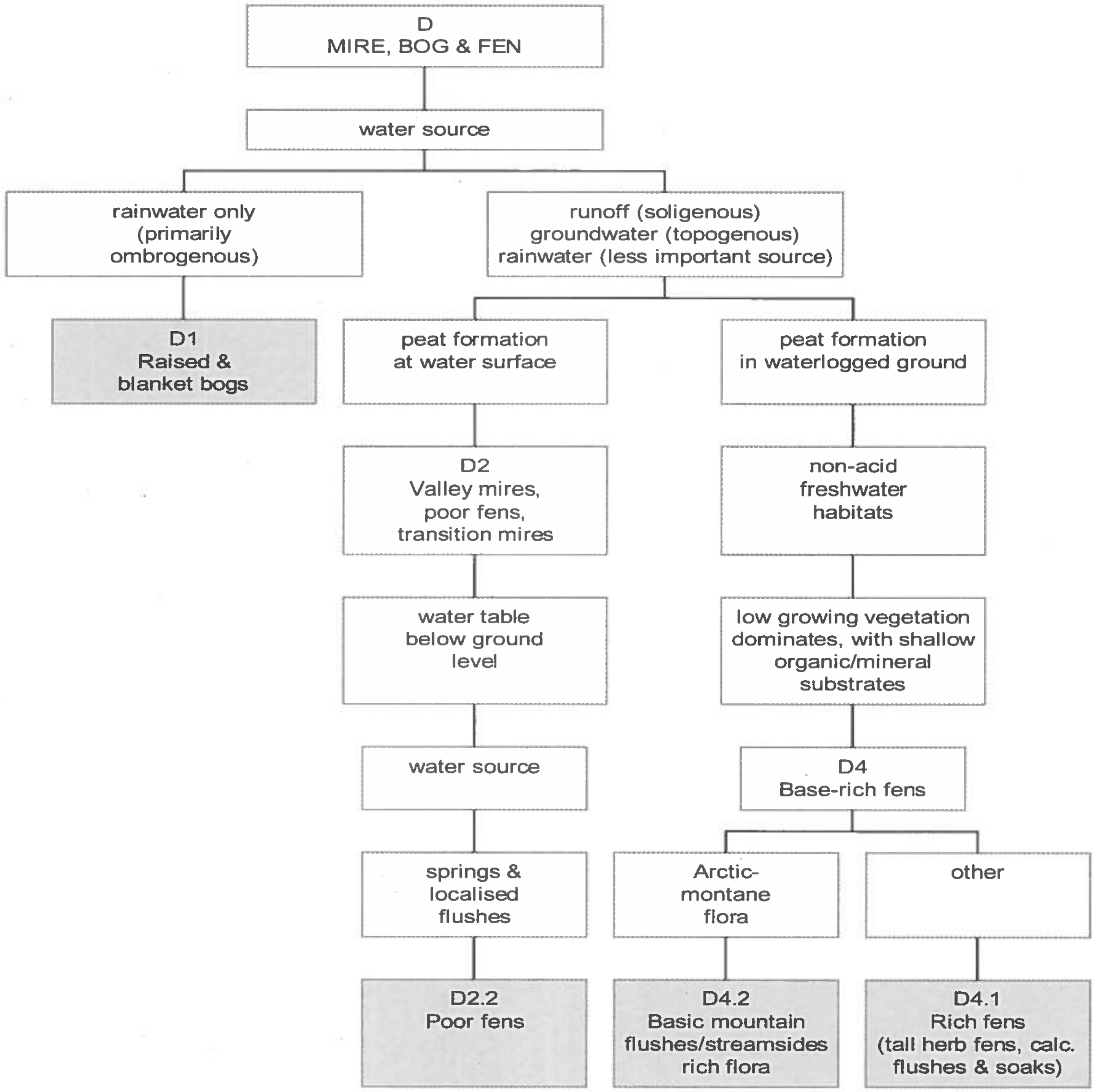


Figure 8

E. GRASSLAND AND TALL FORB HABITATS

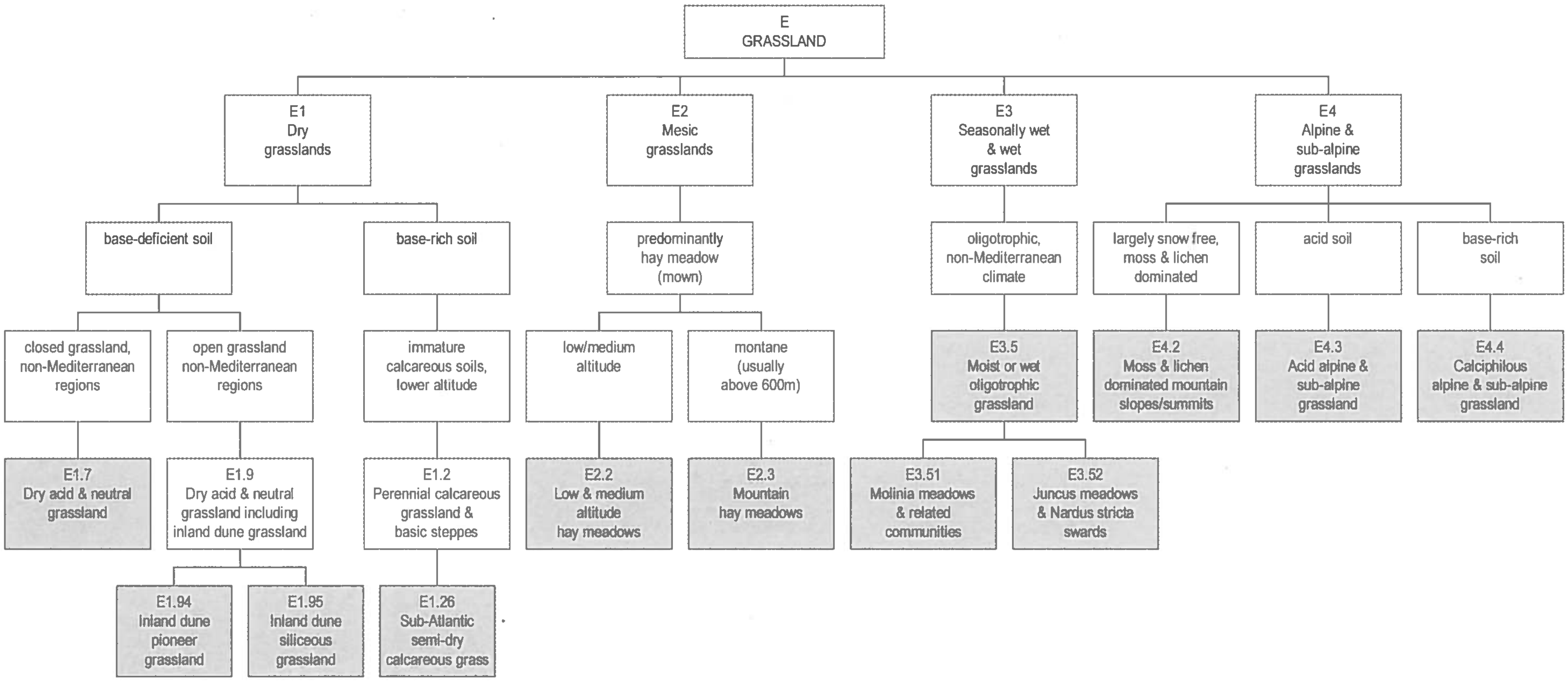


Figure 9

F. HEATHLAND, SCRUB AND TUNDRA HABITATS

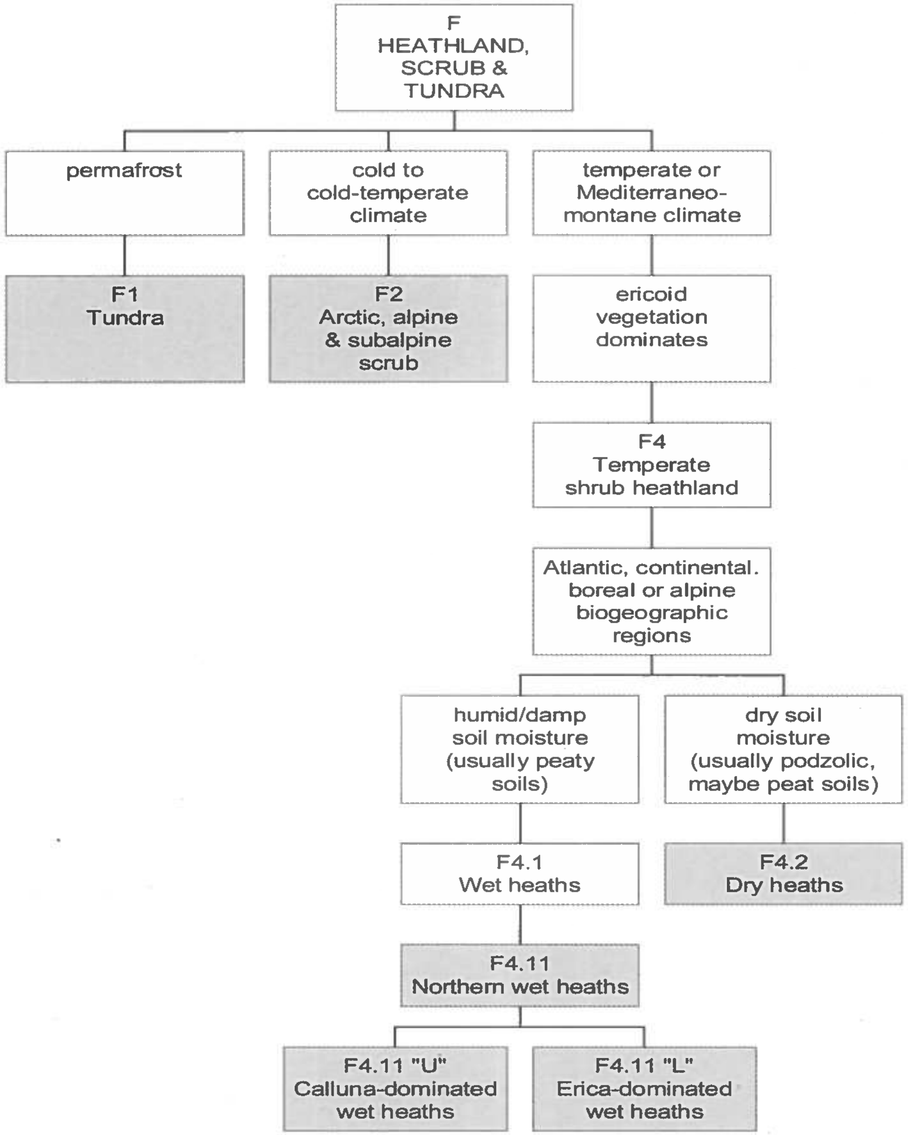
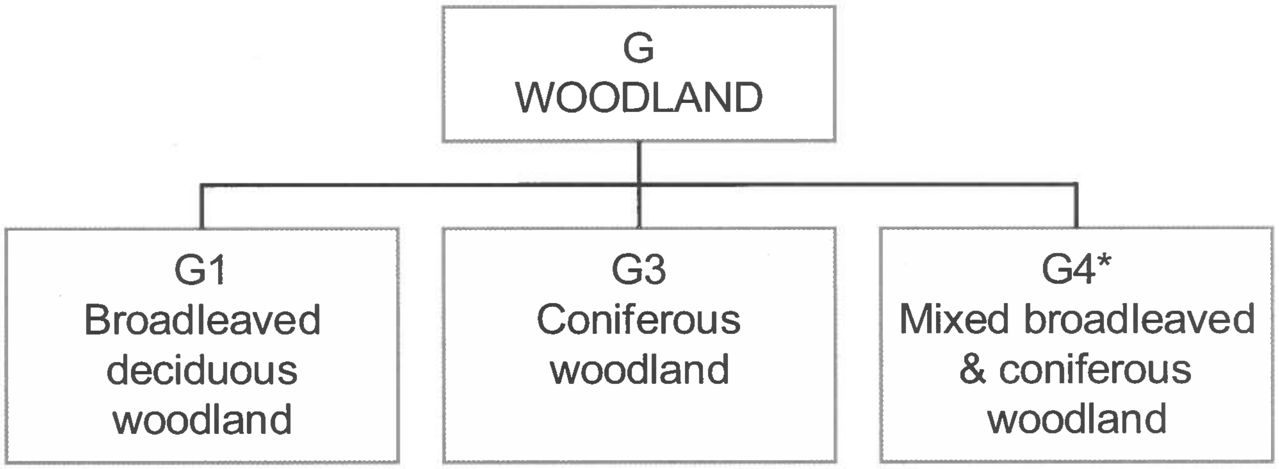


Figure 10

G. WOODLAND AND FOREST HABITATS



* This code can be assigned if the empirical nitrogen critical loads for both broadleaved and coniferous woodland are the same and the habitat is not dominated by one woodland

Plenary Presentations

Results on nitrogen impacts in the EC and UNECE ICP Forests Programme

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Since 1994, a 'Pan-European Programme for Intensive and Continuous Monitoring of Forest Ecosystems' has been implemented to gain a better understanding of the effects of air pollution and other stress factors on forests. At present 862 permanent observation plots for Intensive Monitoring of forest ecosystems have been selected (512 in the European Union and 350 in several non-EU countries). The Intensive Monitoring Programme includes the assessment of crown condition, increment and the chemical composition of foliage and soil on all plots, whereas atmospheric deposition, meteorological parameters, soil solution chemistry and ground vegetation composition are monitored at selected plots. These data are do allow an evaluation of the impacts of elevated atmospheric S and N compounds.

In this section, field evidence for those effects in these Intensive Monitoring plots is given. Results focus specifically on effects that are used in deriving critical loads, while concentrating on the effects of elevated nitrogen inputs, i.e.:

- Elevated N leaching (N saturation of forests),
- Release of Al and accumulation of NH_4 in soil that may disturb the nutrient uptake,
- Elevated N contents and N to base cation ratios in foliage that may cause stresses due to drought, frost, pests and diseases and nutritional imbalances
- Impacts on the species diversity of the ground vegetation

Although N is not the only substance inducing effects on forest ecosystems, it plays an important role in the multiple stresses forests experience, and therefore N is in the centre of the evaluations.

Elevated leaching of nitrogen

A first indication of adverse impacts of N inputs in forest ecosystems is elevated leaching of N (NO_3) that may cause acidification of ground and surface water. At more than 100 intensive monitoring plots, the input of and output of different N compounds (total N, NH_4 and NO_3) has been derived, using methods described in detail in De Vries et al. (2001). Results of the leaching of total N and NO_3 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} (Fig. 1A). The same is true for NO_3, that dominated the N leaching (Fig. 1B). These results are in accordance with those found by e.g. Dise et al. (1998a, b) and Gundersen et al. (1998), Actually, the latter authors found this result while using NO_3 leaching and throughfall N inputs. 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 (Fig. 1 A, B).

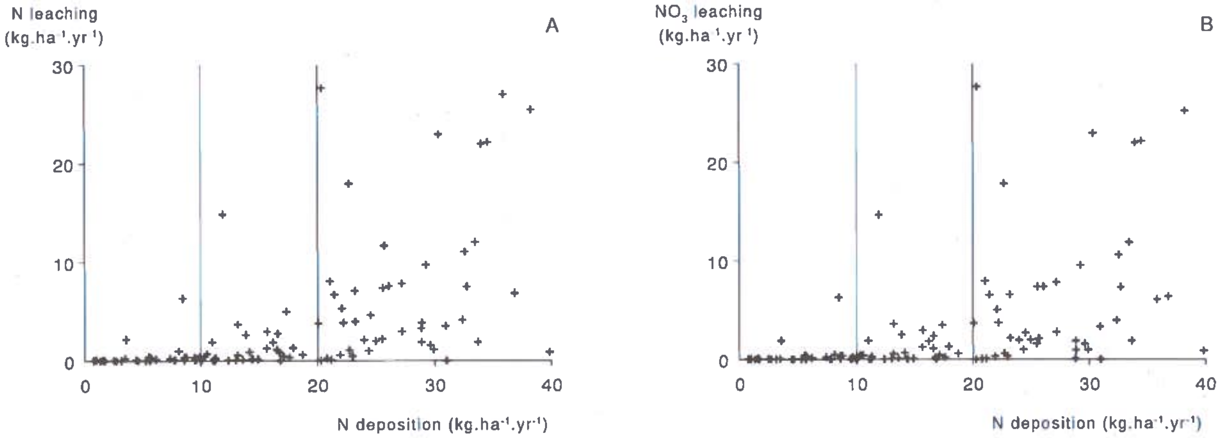


Figure 1. Scatter plots of the leaching of total N (A) and NO3 (B) against the total N deposition.

The latter situation indicates a clear disturbance in the N cycle in response to the elevated N input. NO_3 dominates N leaching. There are only two plots with very high NH_4 leaching rates, which is most likely due to the wet and acid circumstances that strongly limit nitrification. In summary, these data thus indicate a critical N load in relation to elevated N leaching of 10 kg.ha^{-1}.yr^{-1}. More information on the background of the plots and the calculations is given in De Vries et al. (2001).

There appears to be a clear distinction between N leaching from coniferous stands in response to N deposition as compared to deciduous stands (Fig 2). First of all, N depositon is generally lower on coniferous trees which often occur in the North with a relatively low N load. Furthermore, the increase in N leaching in response to elevated N loads (above 20 kg.ha^{-1}.yr^{-1}) appears to be larger for deciduous trees than for conifers (Compare Fig 2A and 2B).

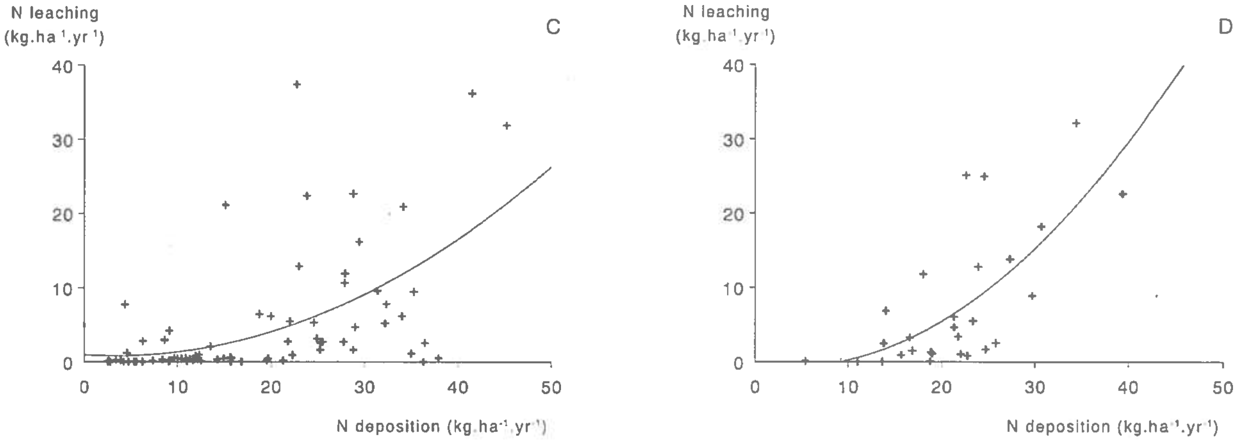


Figure 2. Scatter plots of the leaching of total N against the total N deposition for conifers (C) and deciduous trees (D) with a polynomial fit through the data

Soil acidification and ammonium accumulation

In acidic soils, atmospheric deposition of S and N compounds do lead to elevated Al concentrations, in response to elevated concentrations of sulphate (SO_4) and nitrate (NO_3), and also to accumulation of NH_4 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 NH_4 (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. 1997) may be the effect of increased N inputs.

Results of the concentrations of Al and of NH_4 in the soil solution and of ratios of those compounds to base cations in more than 100 Intensive Monitoring plots are given in Fig. 3. Results show a clear increase in Al concentration and in the ratio of Al to base cations going from the organic layer to the mineral soil (Fig 3A,B), whereas the reverse is true for the NH_4 concentration and the ratio of NH_4 to K (Fig 3 C, D).

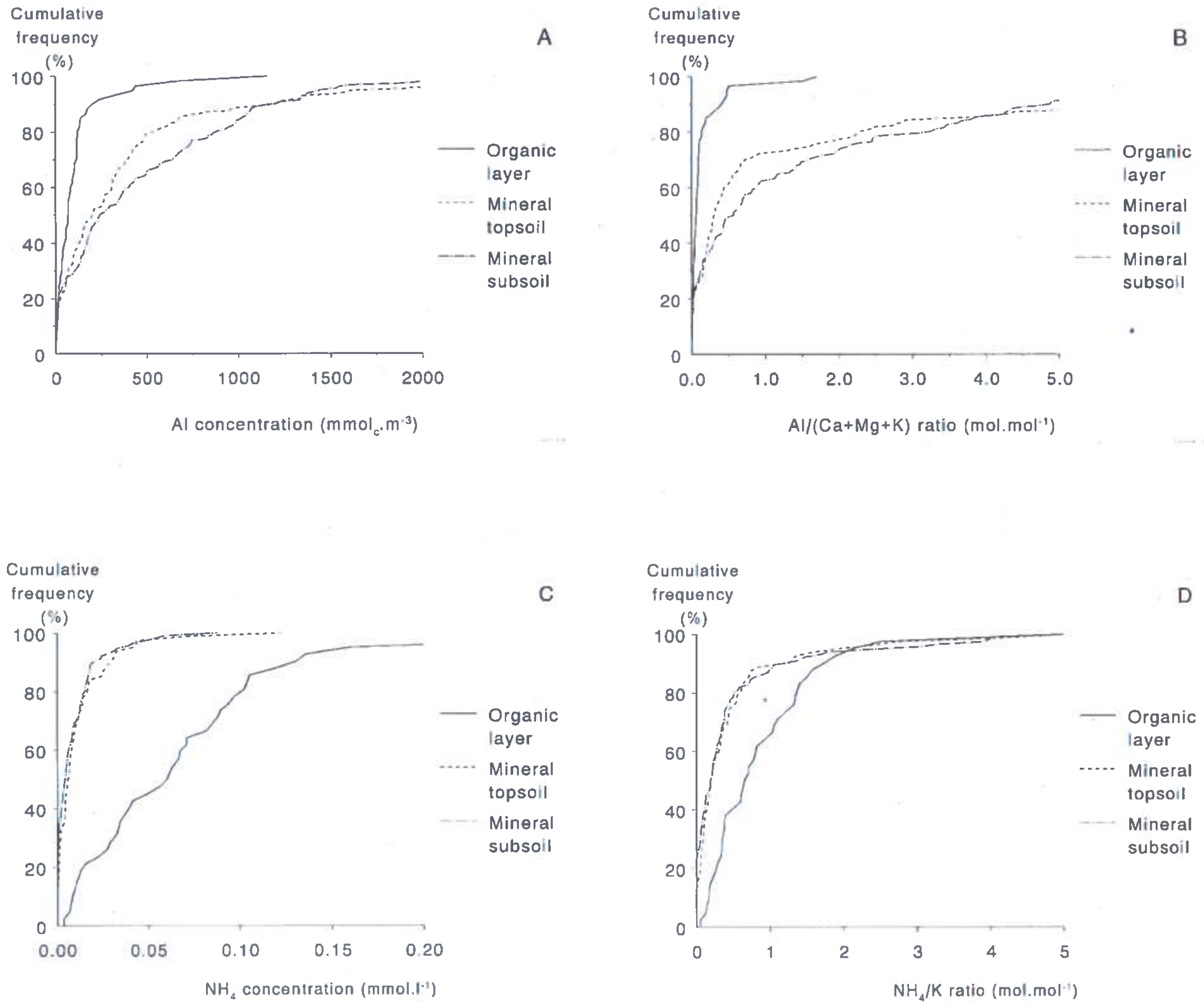


Figure 3. Cumulative frequency distributions of the concentrations of Al (A) and NH4 (C) and of the ratios of Al against the base cations Ca, Mg and K (B) and NH4 against K (D) in the organic layer, the mineral topsoil (0-20 cm) and the mineral subsoil (20-80 cm).

Differences between the mineral topsoil (0-20 cm) and the mineral subsoil (20-80 cm) are much smaller, specifically for NH_4. Although different numbers of samplers at different depths influence the results, it does indicate the large impact of Al release and nitrification in the mineral topsoil. Results show that critical molar Al/BC ratios of between 0.5-1.5, generally mentioned in literature (Sverdrup and Warfvinge, 1993) are regularly exceeded. The critical NH_4/K ratio of 5, mentioned in the literature (e.g Roelofs et al., 1985; Boxman et al., 1988), is hardly ever exceeded in the mineral soil. The same is true for the critical molar NH_4/Mg ratio of 5 (De Vries et al., 1999, 2000).

More detailed insight in the possible impact of acid deposition on Al release and of N deposition on NH_4 accumulation is given in Fig. 4. 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 Fig 4A. 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 Fig. 4B. Results do indicate that below an N deposition of approximately 10 $kg.ha^{-1}.yr^{-1}$, 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 $kg.ha^{-1}.yr^{-1}$. The results hardly indicate a clear critical load for N in relation to N accumulation, but one could use 25 $kg.ha^{-1}.yr^{-1}$ as a reasonable precautionary value.

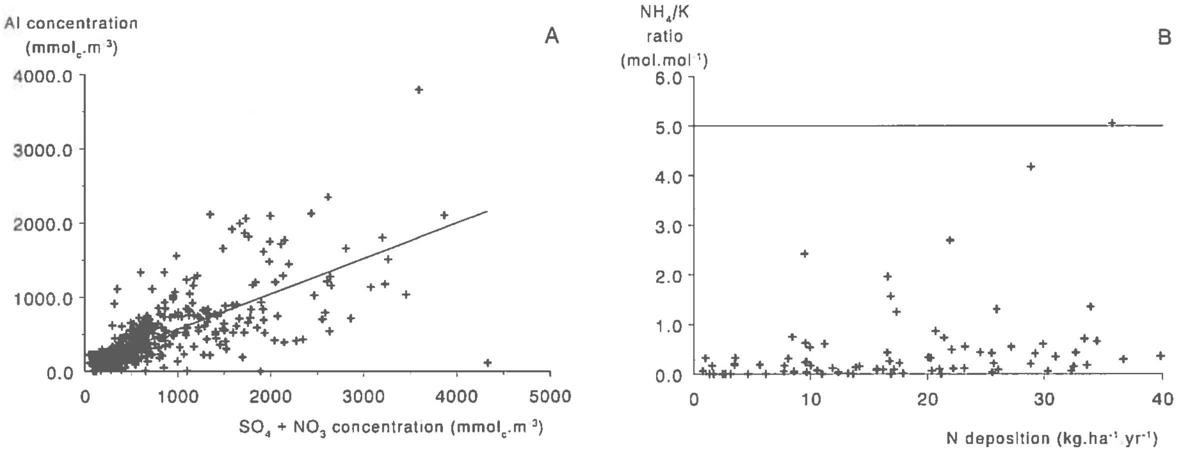


Figure 4. Scatter plots of the concentration of total Al against total SO4+NO3 in the subsoil of Intensive Monitoring plots with a pH < 4.5 (A) and of the NH4/Mg ratio in the mineral topsoil against the total N deposition (B). The solid line in A represents a regression line being equal to: $Al = -95 + 0.74\ (SO_4 + NO_3)\ \ (R^2 = 0.86)$.

Nutritional imbalances

An excess input of N may increase the N content in foliage, which in turn may cause an increased sensitivity to climatic factors, such as frost and drought (Aronsson 1980; De Visser, 1994) and diseases and plagues, such as attacks by fungi on (Roelofs et al. 1985, Van Dijk et al. 1992, Flückiger and Braun, 1998). In this context, a critical N content of 1.8% in needles has often been mentioned in the literature. The relation between N contents in first year needles of Scots pine and total N deposition at 68 Intensive Monitoring plots in Europe (Figure 5) indicating a critical N load of 20 $kg.ha^{-1}.yr^{-1}$ in view of this effect. Above this input level, N contents in foliage may exceed the critical N content of 1.8 % related to drought and frost stress.

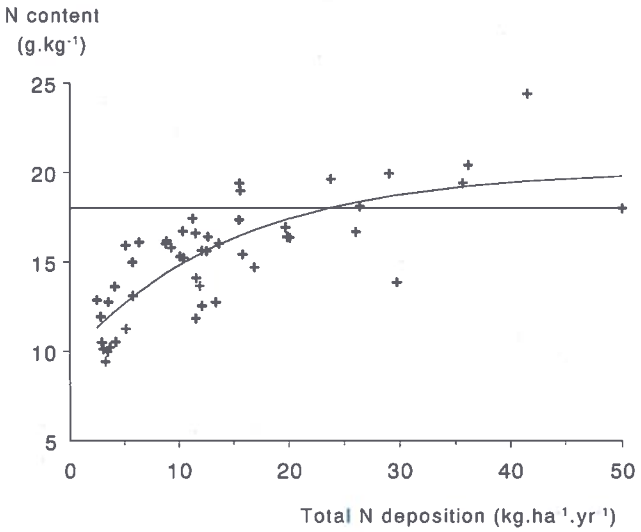


Figure 5. Relationship between N contents in first year needles of Scots pine and total N deposition at 68 plots in Europe.

An important impact of the high N contents in foliage is also a nutritional imbalance, i.e. deficiencies of the macro nutrients K, P, Mg and Ca relative to N in needles. Increased growth rate and elevated N concentrations in foliage may dilute the pool of other nutrients in absolute and/or relative terms. Insight in the possible impact of N deposition on a nutritional imbalance in foliage has been derived for more than 100 level II plots with information on both the chemical composition of the foliage and total N deposition. Ranges in N deposition at plots with a balanced and unbalanced ratio of the base cations K, Ca or Mg to N are given in Table 1. More information on the criteria related to balanced and unbalanced ratios is given in 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. 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 10-15 kg.ha^{-1}.yr^{-1} in view of tree nutrition.

Table 1. 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}							
	Imbalanced				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

Impacts on the species diversity of the ground vegetation

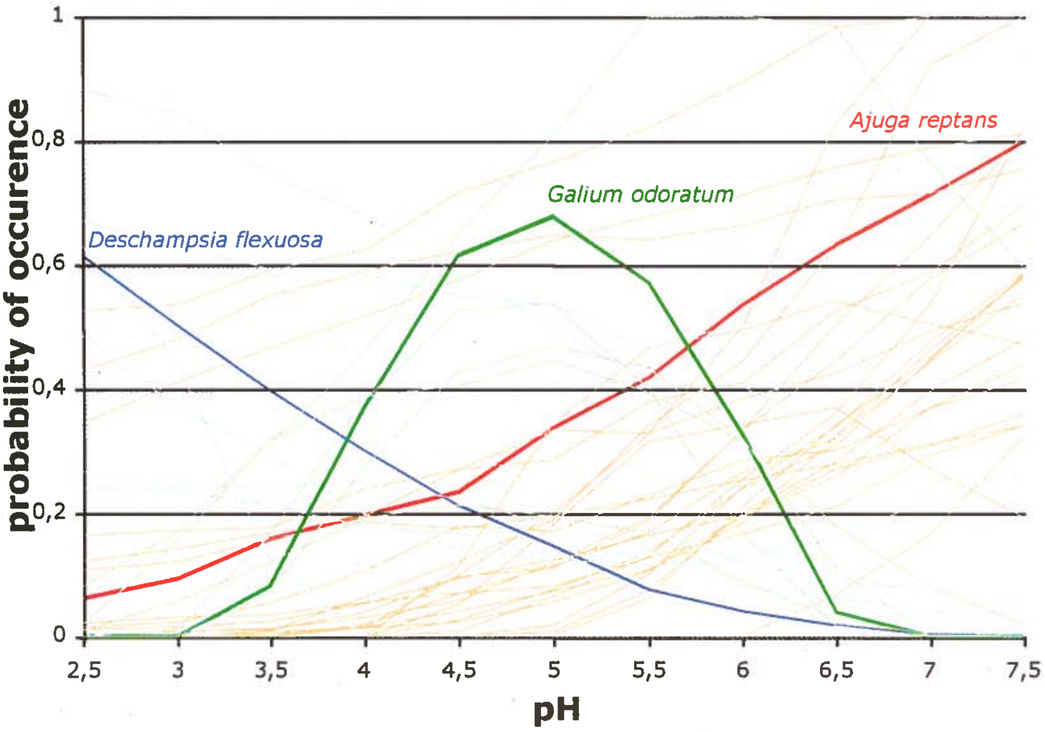
Relationships between species diversity of the ground vegetation and environmental factors were evaluated at 366 Intensive monitoring plots for which these combined data sets, including atmospheric deposition, were available. Included environmental factors were: (i) site characteristics, such as climate zone, altitude, tree species and soil type, (ii) climatic data, such as temperature and precipitation, (iii) chemical soil data, including element contents in the humus layer and the mineral topsoil both in soil and solution, and (iv) bulk deposition data of NH_4, NO_3, SO_4, Ca, Mg, K, Na and Cl.

Results of the analysis, as summarised in Table 2, show that 15% of the variation in the abundances of the various species occurring in the ground vegetation could be explained by the various environmental factors. The explained variance is almost exclusively due to the 'traditional' factors soil, climate and tree species, which contribute in approximately equal amounts to the fit of the model. Only a very small portion of the explained variance (0.6% out of 14% explained variance) is due to bulk deposition chemistry. Na and NO_3 are the only ions in bulk deposition that are in the significant model, and of these two only NO_3 is of anthropogenic origin. The explanation increases with 13% when country is included as an explicit predictor, but this only illustrates that part of the variation can be explained by differences in data assessment methods.

Table 2. Percentage explained variance of the species abundances that could be ascribed to the four main groups of variables

variable group	**Percentage explained variance**
Soil	5.8%
Climate	4.9%
Tree species	3.1%
Bulk deposition	0.6%
Total	**14.5%**

Relationships between the occurrence probability of individual species and environmental factors species were also investigated for 333 different species by drawing 10,000 value combinations for all predictor variables. This resulted in response curves expressing the probability of occurrence for different species at changing environmental factors. An example of such response curves for 36 selected species against soil pH is given in Fig 6.



Species that prevail in acid habitats (low pH) are: ***Deschampsia flexuosa,*** *Calluna vulgaris, Calamagrostis villosa, Vaccinium myrtillus, Vaccinium vitis-idaea, Picea abies, Sorbus aucuparia.*

Species that prevail in intermediate habitatss are: Galium odoratum, Melica uniflora, Anemone nemorosa, Veronica officinalis, Hedera helix, Carex sylvatica.

Species that prevail in alkaline habitats (high pH) are: ***Ajuga reptans,*** *Viola alba, Melittis melissophyllum, Dactylis glomerata, Sorbus domestica, Cardamine bulbifera, Silene italica, Digitalis lutea, Festuca heterophylla Daphne laureola, Cruciata glabra, Ruscus aculeatus, Carex flacca, Stachys officinalis, Rubus caesius, Poa nemoralis, Carpinus betulus, Mercurialis perennis, Solidago virgaurea, Rosa arvensis, Luzula forsteri, Rubus idaeus, Prunus spinosa, Rubus ulmifolius, Arum maculatum.*

Figure 6. Probability occurrence curves for 36 species demonstrating a considerable influence of pH in the organic soil layer at 366 Intensive Monitoring plots

Results show that most species have their optimum at more basic conditions whereas at acidified sites fewer and specifically adapted species will increase. This relation partly explains the decrease of species density at the more acidic sites in northern Europe, although there are also climatic factors that have to be taken into account. Based on the presented results dynamic models are foreseen to be developed in the coming years. They will allow simulations that predict changes in ground vegetation composition under changing environmental conditions. More information on the various results is given in De Vries et al., (2002).

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Nitrogen in the UNECE International Co-operative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems - ICP IM

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The ICP IM Programme

The ICP-IM programme focuses mainly on natural forest ecosystem. The basic objectives are to monitor the state and changes caused by air pollution without on site interference from land-use management. With an approach to enclose the total water balance and chemical budgets, causative explanations for induced effects should be clarified and elucidated. The ecosystem approach with input-output budgets relies often on catchment sites (Fig. 1). From the comprehensive determinations made, models are developed and tested. Bioindicator modelling is a priority task.

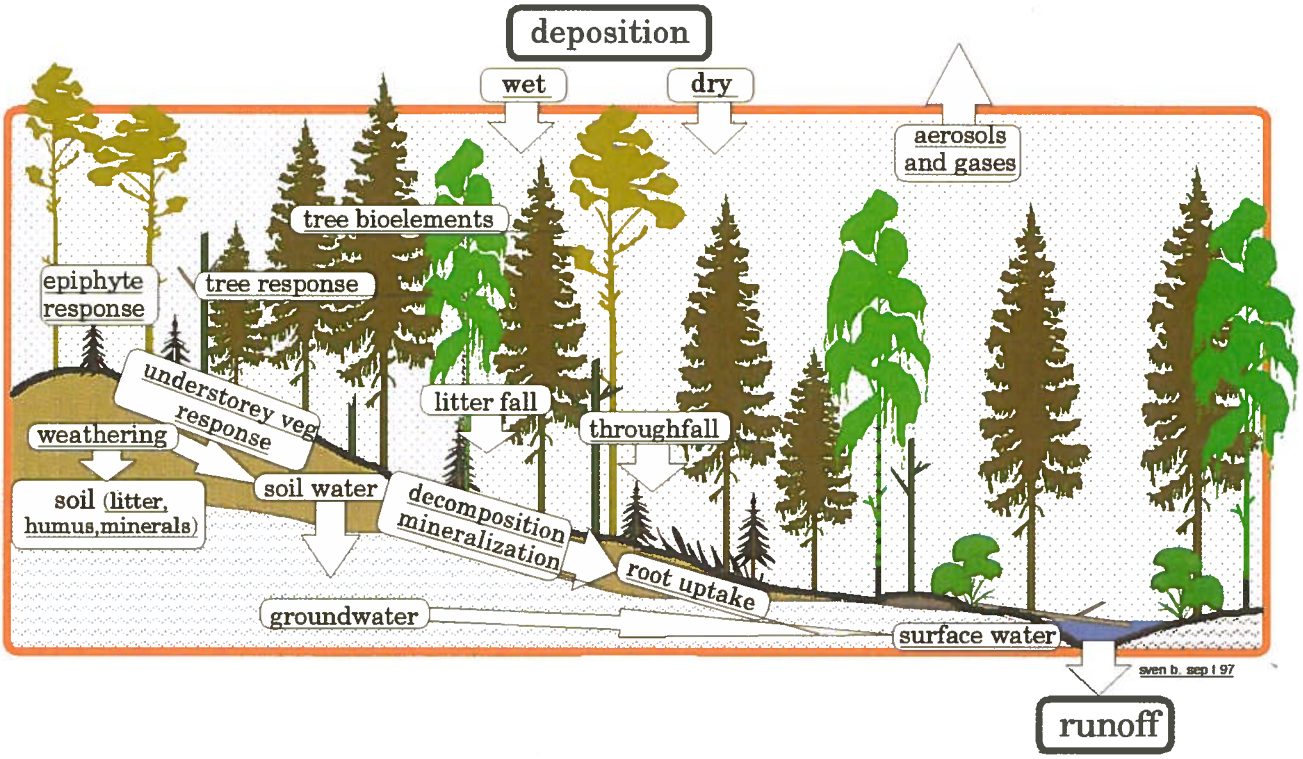

Figure 1. The catchment approach represented as an ideal slope with IM determinations from input to output including internal properties and processes (source of the figure: S. Bråkhielm).

The monitoring network presently covers the following twenty-one countries: Austria, Belarus, Canada, Czech Republic, Denmark, Estonia, Finland, Germany, Iceland, Ireland, Italy Latvia, Lithuania, the Netherlands, Norway, Poland, Portugal, Russia, Spain, Sweden and United Kingdom. All in total, integrated monitoring data is at present available from 70 mostly European sites. The number of sites with on-going data submission is, however, less than 50.

ICP IM Priority Work

Future and ongoing work priorities are:
* Heavy metals;
* Budgets and trends in sulphur, nitrogen and base cations;
* Dynamic modelling and related international work;
* Nitrogen loads and effects;
* Work and results on ground vegetation.

Expected coordination of work and cooperation with other ICPs, are particularly regarding dynamic modelling (all ICPs), cause-effect relationships in terrestrial systems (ICP Forests, ICP Vegetation), and surface waters (ICP Waters). A special report on cause-effect relationsships in forest ecosystems has been jointly accomplished with ICP-Forest.

Nitrogen in ICP IM

Nitrogen related work concerns several aspects among which should be mentioned acidification and related critical loads, eutrophication and biota influences, changes in soil pools and flows to surface water leaching. In nitrogen budgets, retention in specific sub-areas included in catchments is important. Relations between nitrogen soil processes and carbon sequestration, organic matter decomposition and nitrogen leaching are currently under study.

Fluxes and trends of N, S and base cations are lead Programme Centre. Priority in 2002-2003 is given to calculation of proton budgets, N leaching and C/N interactions. This work has strong links also to the CNTER project financed by the EU.

Results from ICP IM on input-output and proton (hydrogen ion) budget calculations show the importance of nitrogen in the acidification process. Such budgets are dependent on water turnover and methods to determine the water balance have been elaborated (Starr, 1999). Data from selected ICP IM sites were also included in a European study for evaluating the soil organic horizon CN-ratio as an indicator of nitrate leaching (Dise et al., 1998). New results regarding the calculation of fluxes and trends of S and N compounds are presented in Forsius et al. (2001).

At the IM sites, it has been observed a relation between N deposition and N leaching from the catchments. The already earlier found deposition threshold at about 9-10 kg N ha^{-1} $year^{-1}$ was confirmed (Fig. 2). Influences also relate to the forest soil CN-ratio, from which catchment outflows could be estimated at throughfalls of up to 30 kg N ha^{-1} $year^{-1}$ (Dise et al., 1998; Forsius et al., 2001).

However, an elevated leaching could be mitigated in certain soil types, i.e. wet soils such as mires and near-stream zones. In these areas retention of nitrogen is found to occur. The retention might have considerable effect on inorganic nitrogen, while organic nitrogen stays high or even increases.

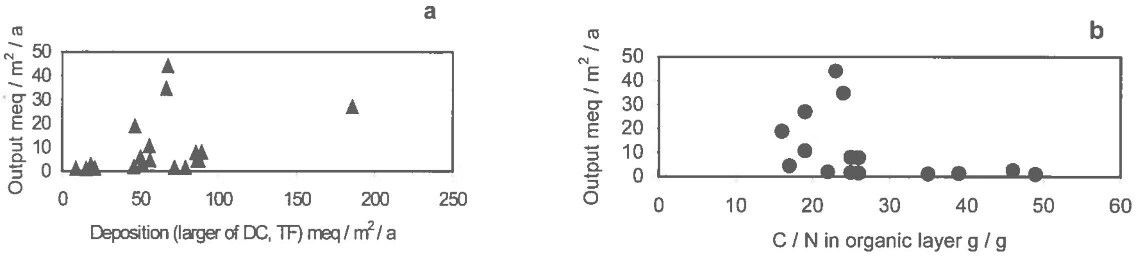

Figure 2. The relationship between N deposition and N output flux (2a), and C/N-ratio of the soil organic matter and N output flux (2b) at the ICP IM sites. The larger value of bulk deposition (DC) and throughfall deposition (TF) has been used for estimating N deposition.

Bioindication in ICP IM

Assessment of cause-effects relationships for biological data, particularly vegetation is led by the NFP of The Netherlands. A joint assessment of EU/ICP Forests Intensive Monitoring and ICP IM data is currently in progress. Effects of pollutant deposition are of central concern. Monitoring in the ICP IM programme has the intention to discover temporal changes in ground vegetation and relate these changes to environmental impacts. Early results show a fairly direct ground vegetation response on N deposition. However, later statistical evaluations, in the ICP work, could not show strong relationships, mainly because of the limited environmental gradients in the data sets. Lack of both chemical and biological information should be improved by more comprehensive reporting and the addition of more sites. In an extended study using ICP Forest data gradients in diversity were found over Europe but conditions related to pH and N showed very low environmental correlations.

A special investigation has been made concerning relations between nitrogen deposition and algae occurrence on spruce needles. For Swedish conditions the coverage of algae is higher in high N deposition regions.

ICP IM and Carbon Sequestration

National achievements regarding new topics, e.g. carbon-related issues, POPs, multi-effects evaluation, are currently discussed. The role of the international ICP IM network should be coordinated in relation to multi-effects evaluations and global change related topics. Further work should be reflected in the work plan beyond 2002.

Related to climate change and carbon sequestration, nitrogen is directly involved and estimation of accumulation or release depends on the CN-ratio in the soil. Possibly, the small changes in accumulation expected over, as short periods as the control related to the Kyoto agreement would be hard to detect by direct soil determinations. A combination with nitrogen determination and in relation to CN-ratios, estimations on the sequestration could be made.

Conclusions

It could be concluded that there are high emissions of nitrogen and this influence soil, water and biota. In the terrestrial ecosystem there is retention of nitrogen with effects on the vegetation and catchment budgets reveal elevated leaching of N. Concerning carbon turnover, nitrogen has a key role in the sequestration process.

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Possibilities and use of multi-species models for the calculation of critical loads for nitrogen for (semi-)natural ecosystems

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Introduction

The Gothenburg Protocol under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP addresses the problems of photochemical pollution, acidification and eutrophication for both human health and the environment. Therefore, this protocol is also called: *the multi-pollutant, multi-effect* protocol. Atmospheric transport causes deposition of NO_x, SO_x and NH_y emitted by traffic, agriculture and/or industry on natural and semi-natural areas. Soil and water chemistry in natural areas is changed by deposition of sulphur and nitrogen compounds. Species adapted to the abiotic conditions in their natural habitats are forced to cope with these changes. This is especially true for sessile organisms like plants. Changes in abiotic conditions will influence the competitive relationships among plant species. Soil acidity, nutrient availability and vegetation structure are, next to soil moisture content, the most important factors determining the competitive relationships and eventually plant species composition of terrestrial (semi-)natural ecosystems. Deposition of NO_x, NH_y and SO_x can affect these factors. Most plant species from oligotrophic and mesotrophic environments are adapted to grow in nitrogen-poor conditions and are out-competed by fast-growing species when the availability of nitrogen increases. An increase in nitrogen in the environment often leads to a decrease in many characteristic, and often rare, species and the increase in the abundance of a few others. In the Netherlands many plant species have become rare or extinct due to such changes (Bobbink *et al.*, 1998; Van Tol *et al.*, 1998), and loss of biodiversity is one of the risks of high levels of deposition of airborne nitrogen compounds.

In the background document for the expert workshop on empirical critical loads for nitrogen on (semi-)natural ecosystem (Bobbink et al., 2002) it is recognized that the severity of the impacts of atmospheric nitrogen deposition is depending on a number of factors. Most important factors are: (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. Here, we describe the additional possibilities that multi-species models in combination with a dynamic soil and vegetation succession model may have for calculating effects of air pollution. This is illustrated by describing the methods for calculating critical nitrogen loads for (semi-)natural ecosystems in the Netherlands with a dynamic soil model SMART (Kros, 2002) linked with a dynamic vegetation succession model SUMO (Wamelink *et al.* 2001) combined with a plant species oriented vegetation effect model MOVE (Latour and Reiling, 1993).

Methods

Dynamic models for critical load calculations

Dynamic models are often used to describe the complex effect chain from air pollution to nature (e.g. Cosby *et al.*, 1985; Groenenberg *et al.*, 1995; Van Oene and Berendse, 2001; Warvinge *et al.*, 1993). When coupled, the SMART2, SUMO2 and MOVE models can describe the influence of deposition on soil chemistry, vegetation succession and plant species composition. Each of the models will be shortly described here. Figure 1 shows the relationship between the models in terms of data exchange.

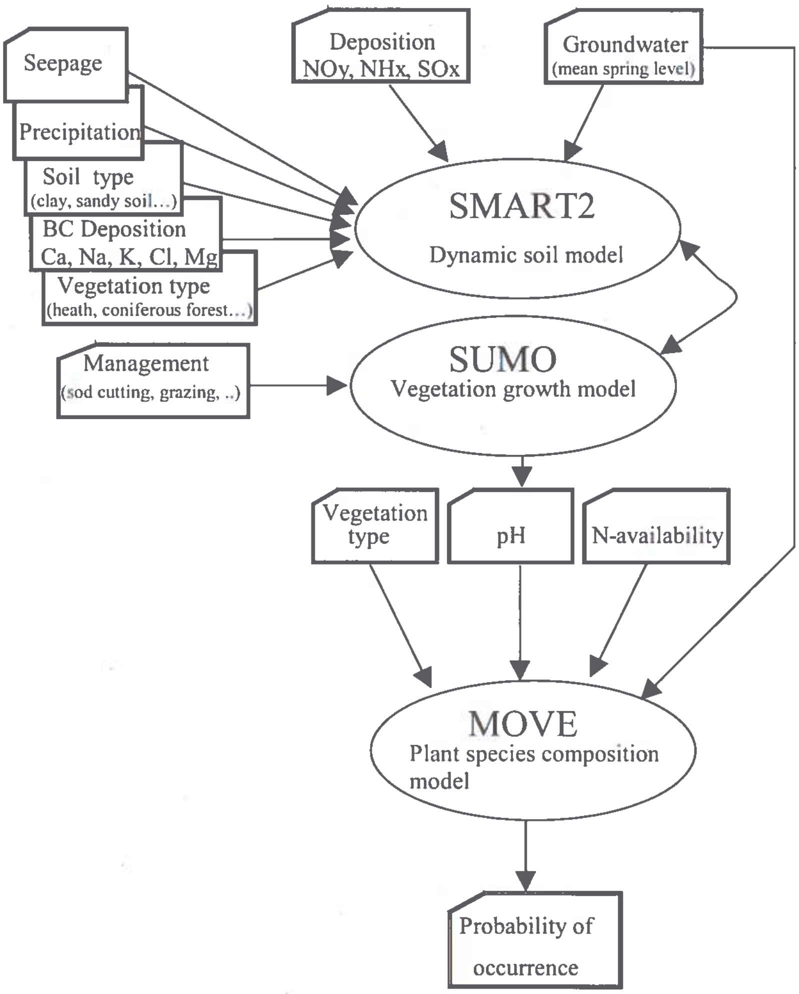

Figure 1. Data flow between SMART2, SUMO and MOVE.

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 (BC^{2+}), 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).

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. De Vries *et al.* (1998) provide justification for the various assumptions and simplifications.

SMART2 is parameterized for 7 soil types, 5 vegetation structure types and 5 groundwater levels (Table 1). The input data consists of maps on deposition, groundwater level, soil type and vegetation.

Table 1. Soil types, vegetation types and functional plant types in SMART2 and SUMO.

Soil types in SMART2	*Vegetation structure type in SMART2*	*Functional plant types in SUMO*
Sand-poor	*Spruce forest*	*Herbs and grasses*
Sand-rich	*Pine forest*	*Dwarf scrubs*
Sand-calcareous	*Deciduous forest*	*Scrubs*
Peat	*Heath*	*Pioneer trees*
Loess	*Grass*	*Climax trees*
Clay		
Clay-calcareous		

SUMO2

SUMO2 (Wamelink *et al.* 2001; Van Dobben *et al.*, 2002) is a dynamic vegetation model, describing the growth competition between 'functional plant types' e.g. herbs and grasses, dwarf shrubs, shrubs, and pioneer and climax trees (Table 1). The biomass is partitioned over three organs (root, stem, leaf) of the five 'functional types', competing for both light and nitrogen. The decay of light through the canopy is described with the Lambert-Beer equation. The highest functional type catches light first, and functional types with leaves at equal height compete for light according to their leaf biomass. Therefore plant height is also explicitly simulated as a function of biomass. Each functional type has three sources of nitrogen: internal recycling of nitrogen before litterfall, direct uptake from the atmosphere and uptake from the soil. In the soil the available nitrogen is distributed over the functional types according to the root biomass of these types. Biomass increment of each type is proportional to light interception, and related to nitrogen availability through a Michaelis-Menten equation. The carbon and nitrogen acquired are distributed among the organs. The uptake of nitrogen is limited by setting a maximum on the nitrogen content of the biomass; excess nitrogen remains in the soil and may be leached. The vegetation structure type is determined from the distribution of biomass over the functional types. Effects of nature management like mowing, thinning, clear-cutting, sod-cutting and grazing on soil chemistry and vegetation structure are also taken into account. The density of grazers is constant for domestic species and varies with limitations according to food availability for wild species. Grazers' faeces are returned to the soil organic-matter pool.

SUMO is parameterized for each functional type (e.g. biomass increment as a function of light interception, distribution of C and N among organs, height increment as a function of biomass increment, etc.). Twelve different tree species were distinguished. For grazers (15 species) the model was also parameterized at the species level.

MOVE

MOVE is a descriptive species-oriented vegetation model (Latour and Reiling, 1993) consisting of a set of regression functions relating to the probability of plant species occurrence to abiotic site conditions. The model gives a statistical description of the habitat preferences of plant species in terms of nitrogen availability, soil pH and groundwater level. It is based on Gaussian logistic regression analyses (Jongman *et al.*, 1987) of data on more than 100,000 vegetation releveés contained in an extensive database (Schaminee *et al.*, 1996). The regression functions can be visualized as bell-shaped optimum curves, representing species occurrence along a single abiotic gradient. Since MOVE focuses on more than one abiotic factor, the real curves are, in fact, multi-dimensional and bell-shaped. The abiotic factors are expressed as Ellenberg-indicator values (Ellenberg, 1991). The Ellenberg-indicator values were separately related to soil characteristics using a smaller data set (Ertsen *et al.*, 1998) in the next step. These relationships were used to link output of SMART2/SUMO to MOVE. Through this two-way approach, maximal use could be made of the availability of vegetation releveés; significant response curves could be calculated for over 1000 different plant species.

Calculation of Critical Limits and Critical Loads for Biodiversity

The calculation of critical loads using the SMART2/SUMO/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/SUMO, to calculate the critical loads at which the above critical limits were not exceeded (Van Hinsberg and Kros, 2001).

Critical limits

The critical limits are based on plant-species specific information on habitat preferences for nitrogen availability and soil pH, which is available in MOVE. 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.*, 1995). 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. The calculation procedure is described in Van Hinsberg and Kros (2001).

Critical loads

In calculating critical loads from these critical limits, the relationship between deposition level and abiotic site conditions are described with SMART2/SUMO. However, since different environmental factors might result in the same abiotic conditions in terms of soil pH and nitrogen availability, we simplified the models. First, the vegetation succession, as modelled by SUMO, was described with a simple logistic growth function as in earlier versions of the SMART model (Kros et al., 1995), describing biomass increment through time. For forest growth, the average growth for one rotation period was used. For heathland we used the average growth for a period of 30 year and for grassland the yearly production of a low productive grassland. Second, various environmental factors (like precipitation, BC deposition, groundwater quality, ratios between deposition of S and N and simulation horizon) were held constant in the calculation of critical loads. For the calculation of the critical loads for the nature target types, we derived regression equations to describe the model output (nitrogen availability and soil pH) as a simplified function of model input (deposition levels and seepage) for each combination of vegetation type, soil type and groundwater table. Nitrogen availability was related to nitrogen deposition, seepage, SO_x deposition, soil type and vegetation type. Seepage influences nitrogen availability by input of nitrogen or by influencing, for example, mineralization fluxes through effects on soil pH. Deposition of SO_x might also indirectly affect the nitrogen cycle by influencing soil pH. The correlation coefficients of the relationships between deposition and abiotic conditions were almost all higher than 0.95 and highly significant. Mean absolute differences between the original nitrogen availability, and soil pH in the data set, and the values which could be calculated with the regression functions, given the same model input (i.e 0.03 and 80 $mol.ha^{-1}.y^{-1}$ for soil pH and nitrogen availability, respectively). However, in some cases much larger differences were found.

Results

Critical limits

Critical limits were calculated on the basis of MOVE for both the Dutch nature conservation targets (Bal *et al.*, 1995) and vegetation classes (Braun-Blanquet, 1951). Direct validation of critical limits in terms of nitrogen availability is not possible since field measurements are not available (Wamelink *et al.*, 1998). At the level of vegetation classes the critical limits could be compared to measured or expected critical limits in terms of produced vegetation biomass, a factor highly correlated with nitrogen availability. Figure 2 shows the relationship between measured and calculated critical limit in terms of the biomass of different vegetation classes found in the Dutch dunes. The regression function is highly significant ($p<0.001$), although there is ample variation around the regression line ($R^2=0.69$).

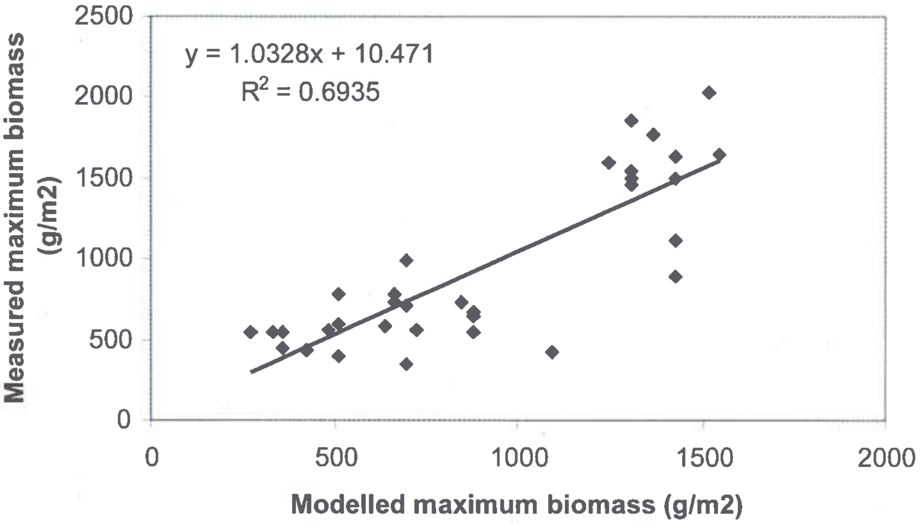

Figure 2. Relationship between modelled and measured maximum biomass levels for different dune vegetation types. Measurements are published in Koerselman et al., 1999.

Critical loads

Based on the critical limits national critical load maps for biodiversity were calculated using national maps of soil types, groundwater tables, vegetation structure types, seepage and a preliminary map of the nature targets (Van Hinsberg and De Vries, 2001). The resulting map of critical nitrogen deposition is shown in Figure 3. It must be noted that empirical critical loads are added to this map, since we were not able to calculate reasonable critical loads for all types of ecosystems (see discussion).

Figure 3 shows a highly spatial variability in critical loads. Nature targets are particularly vulnerable in the dunes and the sandy soils in the central, southern and eastern parts of the Netherlands. Nutrient-poor adapted ecosystems dominate these areas. The ecosystems pertaining to the clay soils in the western and northern part of the Netherlands are less vulnerable to high nitrogen deposition levels than the nature targets for the sandy soils.

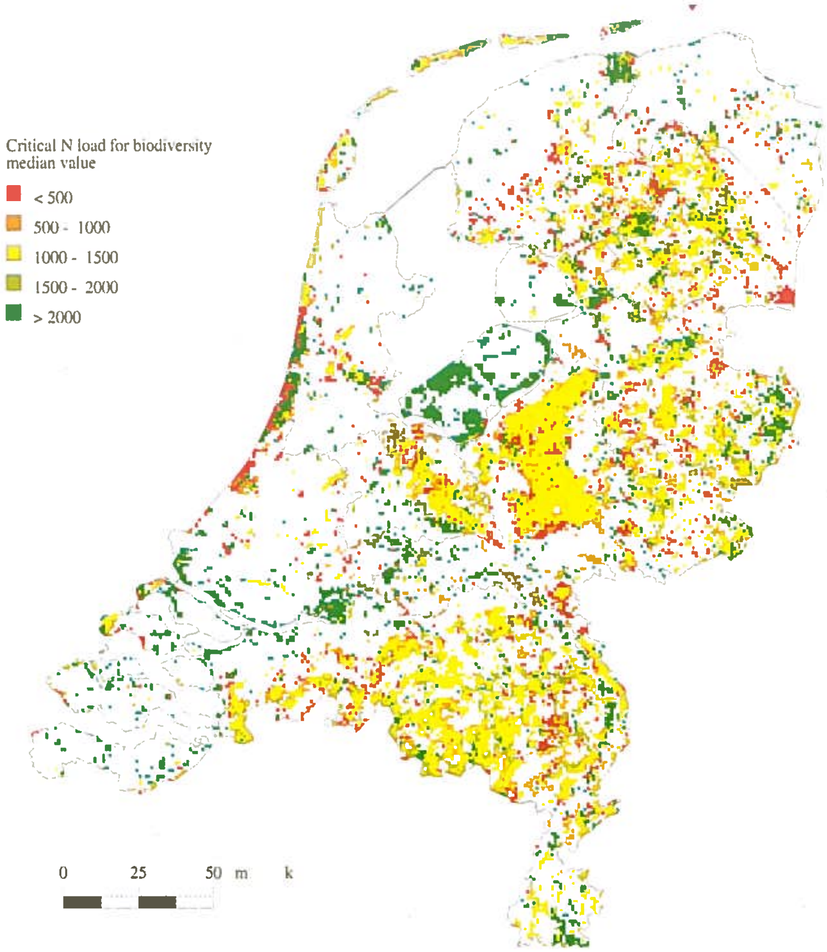

Figure 3. Geographical distribution of the critical nitrogen deposition for the protection of plant species composition of Dutch nature conservation targets (median critical loads in mol.ha^{-1}.yr^{-1} in 1x1 km grids).

Modelled critical loads often vary within a single EUNIS-ecosystem type at both the level of nature target types and vegetation classes (Table 2). Most critical loads from the broader defined ecosystem classes focus on protection of the most sensitive sub-classes distinguished within the larger classes and are therefor lower than the critical loads from narrower defined ecosystem classes. The empirical critical loads and the calculated critical loads correspond reasonably well, although results obtained were not similar for all ecosystems using both the empirical methods and the dynamic models. However, not for all ecosystems reasonable critical loads could be calculated with the dynamic models and empirical critical loads had to be used in the final critical load map (see also discussion).

Table 2. Empirical (Bobbink et al., 2002) and modelled critical loads (in kg.N.ha^{-1}*.yr*$^{-1}$*) for nature target types (Bal et al., 1995) and vegetation classes (*Braun-Blanquet, 1951). *The critical loads of nature target type refer to the median critical loads. The 5- and 95-percentile values are presented in parentheses. Critical loads for vegetation classes are derived from preliminary result of Schouwenberg et al., 2000. Al values are rounded off to the nearest 5kg.*

EUNIS Class	Empirical CL	Nature target	Modelled CL	Vegetation class	Modelled CL
Forest (G)	10-20	Forests on poor sandy soils	15-20 (5-30)	Fago-Quercetum	5-10
				Cladonio-Pinetum sylvestris	10-15
				Leucobryo-Pinetum	25-30
		Wet, calcareous forests	50-55 (35-60)	Carici elongatae Alnetum	30-35
				Pruno-Fraxinetum	30-35
		Forest on clay soils	45-50 (35-55)	Violo odoratae-Ulmetum	35-40
				Fraxino-Ulnetum	50-55
				Artemisio-Salicetum albae	55-60
				Irido-Salicetum albae	60-65
				Cardamino amarae-Salicetum albae	65-70
		Bog forest	5 (5-15)	Erico-Betuletum pubescentis	20-25
				Carici curtae-Betuletum pubescentis	20-25
				Carici elongatae Alnetum	30-35
		Lowland nutrient-rich peat forest	30-35 (25-50)	Carici elongatae Alnetum	30-35
				Thelypterido-Alnetum	35-40
		Non-calcareous dune forest	20-25 (5-35)	Fago-Quercetum	5-10
				Betulo-Quercetum roboris	5-10
				Cladonio-Pinetum sylvestris	10-15
		Calcareous dune forest	35 (35-40)	Betulo-Quercetum roboris	5-10
				Crataego-Betuletum pubescentis	50-55
Raised bogs (D1)	5-10	Bogs & wet heath	5 (5-15)	Erico-Shagnetum magellanici	5-10
				Sphagnetum cuspidato-obesi	10-15
				Sphagno-Rynchosporetum	10-15
				Caricetum limosae	15-20
				Carici curtae-Agrostietum caninae	25-30
Salt marsh (A2.64/65)	30-40	Salt and brackish marsh and grasslands	35 (35)	Trifolio fragiferi-Agrostietum stoloniferae	45-50
				Salicornietum dolichostachyae	45-50
				Puccinellietum maritimae	50-55
				Atriplici-Elytrigietum pungentis	55-60
				Suaedetum maritimae	60-65
Dune pioneer grasslands	10-20	Dry open-dune grasslands	5-10 (5-10)	Spergulo-Corynephoretum	5-10
				Violo-Corynephoretum	5-10
				Festuco-Galietum serpylli	5-10
				Phleo-Tortuletum ruraliformis	5-10
				Sileno- Tortuletum ruraliformis	10
Dry and neutral grasslands (E1.7)	10-20	Dry grasslands on sandy soils	5-10 (5-20)	Taraxaco-Galietum veri	5-10
				Violo-Corynephoretum	5-10
				Festuco-Galietum veri	5-10
				Phleo-Tortuletum ruraliformis	5-10
				Elymo-Ammophiletum	30-35
				Honckenyo-Agropyretum juncei	60-65
Semi-dry calcareous grasslands (E1.26)	15-25			Polygonato-Lithospermetum	20-25
				Echio-Verbascetum	30-35
				Balloto-Arctietum	60-65

Moist and wet oligotrophic grasslands (E3.5)	10-20	Moist nutrient-poor grasslands	5-10 (5-20)	Cirsio dissecti-Molinietum	15-20
				Cicendietum filiformis	20
				Crepio-Juncetum acutiflori	20-25
				Carici curtae-Agrostietum caninae	25-30
				Angelico-Cirsietum oleracei	45
				Ranunculo-Senecionetum	45
Coastal dune heaths (B1.5)	10-20	Dune heaths	-	Polypodio-Empetretum	10-15
				Dicrano-Juniperetum	20-25
				Pyrolo-Salicetuim	20-25
Dry heaths (F4.2)	10-20	Dry heaths	-	Spergulo-Corynephoretum	5-10
				Galio-hercynici-Festucetum ovinae	5-10
				Botrychio-Polygaletum	5-10
				Genisto anglicae-Callunetum	5-10

Discussion

The above-described methods were used to calculate the Dutch national critical load maps for nitrogen and acidity. Figure 4 shows the inverse cumulative frequency distribution of the calculated critical loads for nitrogen deposition alongside the distributions of the critical loads for protection for groundwater quality, tree growth and fens (see also Van Hinsberg and Kros, 2001). Protection of biodiversity (in oligotrophic softwater lakes and terrestrial vegetations) seems to require lower deposition levels than protection of groundwater quality and tree growth, which is in agreement with Bobbink et al., 2002. This finding stresses the need for critical loads for biodiversity. By using dynamic models the database of critical loads might be extended.

The current method has the advantage of linking information about species-specific habitat preferences directly to critical limits for ecosystems for which a clear national (and sometimes international) conservation policy can or has been defined. With the coupled dynamic models we are able to link biodiversity goals to deposition goals. Potentially, critical loads could also be calculated for the priority habitats of the EU Habitat Directive. Moreover, the models make it possible to use the same methodology for both calculating critical loads and integrated effect assessment studies. Through the use of dynamic effect modeling, insight could be gained into (i) the consequences of exceeding critical loads, (ii) the relationships between acidification and other threats, and (iii) the dynamic aspects of environmental effects. Moreover, the methods can also be applied to the evaluation of the effects on flora caused by acidification, eutrophication, groundwater depletion, habitat destruction and pollution by toxic substances, since the models are multi-stress models (Alkemade *et al.*, 1998). A multi-stress, and not single-stress, approach is followed since single-stress factors may have outcomes with confined reality when other stresses interfere. The multi-stress approach makes it possible to rank the different environmental threats, which might be of use in setting abatement priorities.

However, the current method has also its problems and reasonable critical loads could not be calculated for all relevant ecosystem types, (Table 2) and empirical critical loads for nitrogen are needed for validation and for obtaining a robust final critical load map as depicted in Figure 3. Several problems did occur due to (i) neglecting the vegetation succession, (ii) lack of data on nitrogen availability and (iii) simplifications of the used models.

Neglecting vegetation succession

Problems with calculating reasonable critical loads occurred mainly in heavily managed ecosystems like heathland and nature target types with broad abiotic preferences (like calcareous and non-calcareous dune grasslands). Calculation of critical loads with SUMO, incorporating the effects of nature management, instead of the used simple logistic growth function from earlier versions of the SMART model might offer a solution to this problem.
In cases were we now could not calculate a critical load, the lowest empirical critical load from similar ecosystems (Bobbink et al., 1996) was used in the critical load maps.

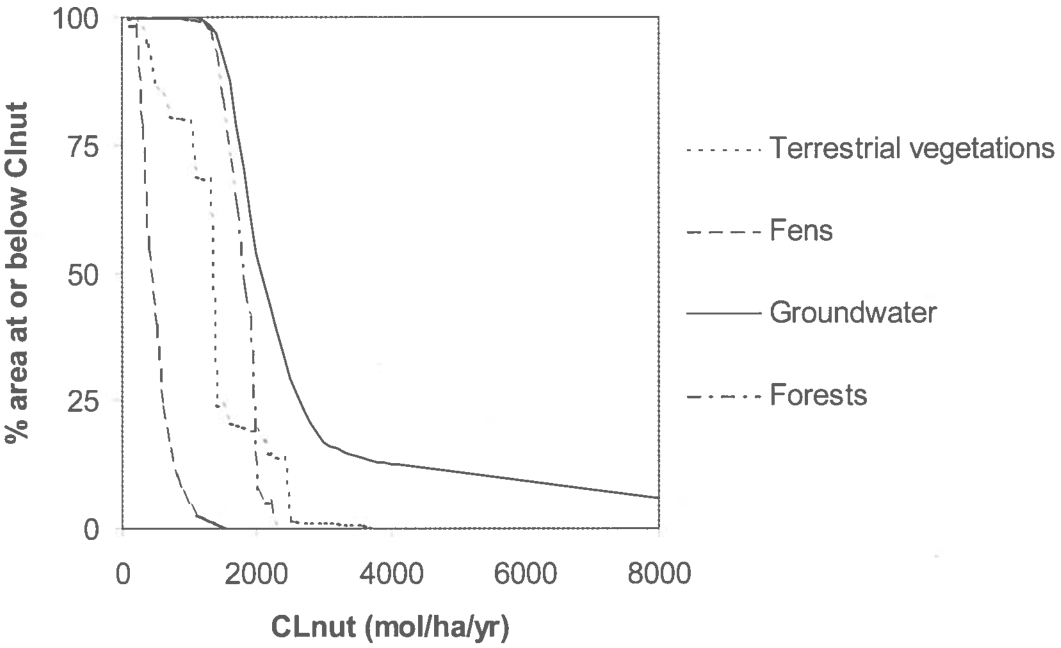

Figure 4. The inverse cumulative frequency distribution of the critical loads for nitrogen (in mol.ha^{-1}.yr^{-1}).

Lack of data on nitrogen availability

The current methods of calculating critical limits also have disadvantages. By using an extensive database of vegetation releveés we were able 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). However, due to the absence of measured abiotic variables in most of the releveés, it was only possible to use indirect estimates of the abiotic conditions. These estimates were calculated from the mean Ellenberg indicator values of plants within the releveés. An additional data set of vegetation releveés, in which abiotic conditions were also measured, was needed to link the averaged indicator values to abiotic conditions. With the help of this second data set we could 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, this extra step introduces additional errors in the calculation of critical limits and critical loads (Schouwenberg *et al.*, 1999; Wamelink *et al.*, 2002; Wamelink and Van Dobben, in press).

Simplifications of the used models

Linking the critical limits to critical loads required the use of relationships between deposition and nitrogen availability, and soil pH. To derive these relationships simplifications of the dynamic model and the underlying complex reality had to be made. The assumption of constant ratios of deposition components, BC deposition and seepage quality might have influenced the calculated critical loads. However, comparison between results of dynamic-effect modelling and calculated exceedance of critical loads showed no large differences in the percentage of protected area (Mol-Dijkstra *et al.*, 2001).

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Additional Documents - Working Group Presentations

Nitrogen deposition measurements and estimates

The importance of accurate background atmospheric deposition estimates in setting critical loads for nitrogen.

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Summary

1. Together with critical loads, estimates of atmospheric nitrogen (N) deposition form the basis of mapping the exceedance of critical loads. Reliable estimates of N deposition are, however, equally important in the primary studies used for setting empirical critical loads. In both experimental N addition studies and in observational surveys of biological change, the critical load derived depends at least partly on the robustness of the estimated background deposition at the study sites.
2. Most of the studies used to set empirical critical loads have been conducted with an emphasis on defining the biological effects. Conversely, the level of attention given to defining the atmospheric input is very variable, with studies often depending on deposition estimates of high uncertainty. Such differences can be used to weight the reliance on the different studies.
3. With experimental addition studies, background atmospheric deposition provides only a fraction of the total N deposition received by the ecosystem. However, for the lowest addition level at which effects are observed (which is used to derive the critical load), the background atmospheric contribution still frequently exceeds 50% of the total deposition.
4. For observational survey studies, the critical load value implied is, by definition, 100% dependent on the atmospheric deposition estimates used. In many cases, the smallest deposition at which effects are observed result from surveys rather than N addition studies. The estimated background N deposition is therefore one of the major sources of uncertainty in setting empirical critical loads.
5. Key issues include the underestimation of N deposition by 'throughfall' measurements, the need for ecosystem-specific deposition estimates and the interpretation of spatial correlations with deposition. In addition, because N in throughfall is less than total deposition, where N has been added experimentally under a woodland canopy, the N addition will be equivalent to a significantly larger atmospheric deposition. The issue of sub-grid spatial variability is also important: where sites are at mountain summits or immediately adjacent to agricultural land, grid average deposition values underestimate actual deposition. Conversely, where study sites are located in the centre of nature areas in an agricultural landscape, mapped values may overestimate actual deposition. Where N deposition in the primary studies is underestimated, this will lead to an underestimate of the critical load and *vice versa*.

I. Introduction

The development of methods to assess atmospheric nitrogen deposition has been discussed widely, including in the series of UNECE expert workshops underpinning the development and implementation of the Gothenburg Protocol. These include the workshops at Lökeberg (Lövblad and Erisman 1992), Göteborg (Lövblad et al. 1993; Duyzer and Fowler 1993, Sutton et al. 1993a), Grange-over-sands (Fowler 1995), as well as the Manual of the UNECE Programme on Modelling and Mapping (UNECE 1996, 2003). The emphasis of the discussions in relation to nitrogen deposition has been on generally improving the methods for modelling atmospheric inputs on a regional scale. These developments have helped improve the national and European air pollution models that provide the deposition estimates for maps of critical loads exceedance. While the emphasis has been on the development of models for regional mapping, it is important to recognize that deposition estimates are also of fundamental importance in the process of setting empirical critical loads for nitrogen. In this process, the primary studies investigating the effects of atmospheric N on ecosystems are used to gauge the level of deposition at which detectable impacts occur. The range of nitrogen deposition conditions necessary for this evaluation is generally either provided by:

a) Manipulating the level of N deposition at a given site, usually by the experimental addition of nitrogen over several years.
b) Comparing responses in areas of contrasting nitrogen deposition, often considering many sites at a regional level (e.g. several 100 km) or through local scale transects (e.g. < 2 km) of intense gradients in N deposition.
c) Spatial surveys examining the areas where change does or does not occur over a long period of time (e.g. several decades), related to the regional pattern of deposition at these sites.

From the perspective of uncertainties in atmospheric deposition, the experimental addition studies are in principle to be preferred, since a good estimate can usually be made of the amount of N added. The uncertainty in the background level of atmospheric deposition therefore makes a smaller contribution to the effects-level derived from the study. By contrast, in both the survey approaches, any estimate of the critical load derived becomes completely dependent on the robustness of the estimated atmospheric deposition levels. This therefore increases the overall uncertainty in the critical load derived from survey studies.

Given these factors, this paper considers the consequences of the different uncertainties in atmospheric deposition for the setting of empirical nitrogen critical loads. In the first part, the general principles are considered in relation to the likely direction of the different potential errors. Secondly, these principles are applied to a selection of the key studies identified in the detailed review of Bobbink et al. (2002), which was provided as a background document for the Bern Workshop. In this second part, tables are provided summarizing the basis of the deposition estimates of selected studies, and indications for their robustness and potential biases. The reference to these specific examples illustrates more clearly the consequences of the deposition uncertainties. Additionally, the aim is to provide transparency in the assessment of the different studies in relation to the level of atmospheric deposition at which effects were observed.

It is important to emphasize that the focus here is to consider the implications of uncertainty in deposition. Hence, estimates are provided here only as far as identifying the atmospheric deposition at which effects were detectable in the different studies. It is recognized that the use of these studies to set empirical critical loads also includes the application of expert judgement and interpretation of the significance of the biological responses. Therefore, no estimates of critical loads are provided here. By contrast, the information here has been used to help improve the robustness of the critical loads estimated by the different Working Groups of the Bern Workshop.

II. Uncertainties in Methods to estimate atmospheric N deposition

Atmospheric N deposition may be estimated by a wide range of techniques, which vary in level of detail and completeness. These include different measurement techniques, models and combinations of the two. A broad distinction can be made between methods to estimate wet deposition and dry deposition. Fuller descriptions of the different approaches are available in the reports from the previous UNECE workshops noted above. Here a brief description is provided that focuses on the sources of uncertainty in making site-based deposition estimates, and the consequences for the primary studies used to estimate critical loads.

II.1 Measurement of wet deposition

At its simplest, wet deposition may be estimated at a site by measuring the amount and composition of precipitation collected in an open funnel rain collector. Because such collectors potentially capture a small fraction of dry deposition, the samples are often known as 'bulk' deposition samplers. The term is perhaps misleading, however, as these measurements give values that are much less than the total deposition. In areas with very high concentrations of reactive gases, such as sulphur dioxide or ammonia, the dry deposition to collectors may provide a small positive error (Fowler and Cape 1984, Cape and Leith, 2002). For ammonia, the contribution of dry deposition on the collector to precipitation composition may also be important even at low gas concentrations. However, with this exception, the effect appears to be small due to the aerodynamic and surface properties of the collectors compared with vegetation. In the case of nitrogen, settling of particles onto collectors is not a great concern, due to the small size ranges in which ammonium and nitrate particles are mostly found, so that these are deposited only slowly compared with inputs to vegetated surfaces. Where there has been concern regarding the difference between wet and bulk deposition, measurements have been made using so called 'wet-only' precipitation collectors, which open only during a rain event. However, for water-soluble gases, deposition to the wetted gauge surface may still provide a positive bias even to 'wet-only' collectors (Cape and Leith, 2002).

A further important potential error concerns the accurate measurement of precipitation. Many devices used for collecting wet deposition are adequate to sample precipitation chemistry, but are not aerodynamically suited to measure accurately precipitation amounts (Erisman et al., 2003). This often leads to an underestimation of wet deposition. To account for this effect, the standard approach is to combine the measurement of precipitation chemistry (as volume-weighted concentration) with records of precipitation amount as measured by meteorological services (e.g. NEGTAP 2001). Corrections to account for the under-sampling of precipitation amount can be as large as 20% (Rodda and Smith, 1986). It should also be noted that annual wet deposition amounts vary significantly, so that accurate assessments require measurements over several years. Figure 1 demonstrates the inter-annual variability of precipitation amounts for five example locations across the UK. For these example sites, annual values may deviate by 27-39% (expressed as twice the coefficient of variation, representing 95% of the distribution) from the long-term mean. Taking a 5-year average reduces the deviations to 8-14%.

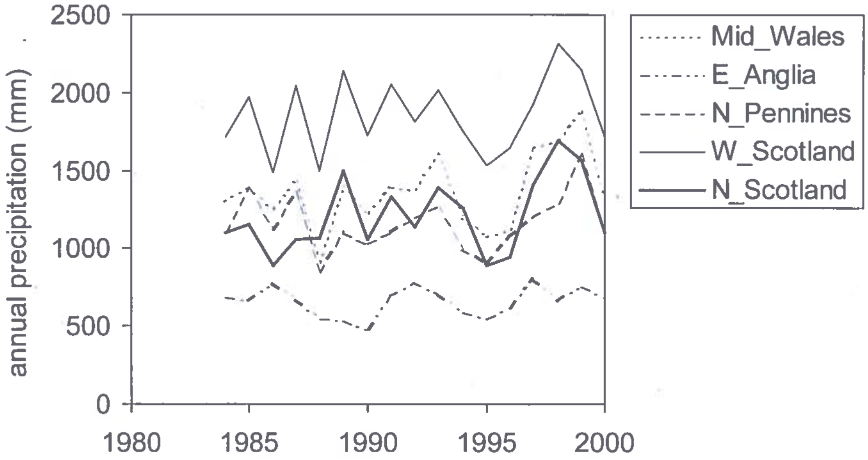

Figure 1: Inter-annual variability of precipitation amounts for five example sites located in different areas of the UK.

II.2 Measuring dry deposition

The measurement of dry deposition is much more complex than that of wet deposition. Given this complexity, continuous dry deposition measurements are available for only a few sites in Europe (e.g. Erisman et al. 1997, Fowler et al. 1998a). A range of micrometeorological measurement methods has been established to determine net fluxes of nitrogen species, which may include periods of both dry deposition and emission (Fowler et al. 2001, Sutton et al. 2000). Because of the complexity of these measurements, they are generally made at an intensive level (e.g. half hourly basis) in order to understand the governing biological, chemical and physical processes (e.g. Erisman and Wyers 1993, Hesterberg et al. 1996, Sutton et al. 1995b, 2001a), and from this understanding to develop models of the dry deposition process.

Recently, methods have been developed to allow low-cost sampling of gaseous and aerosol nitrogen concentrations over integrated periods of a week to a month (e.g. Sutton et al. 2001b). These have been applied together in gradient configuration sampling conditionally on atmospheric stability to provide low-cost estimates of monitored dry deposition for the first time (Fowler et al. 2001). The lower costs of this micrometeorological technique, referred to as the time-averaged gradient method (TAG), are such that it has now become feasible to apply it at selected sites where nitrogen manipulation experiments are conducted. While mains electricity is not required, the normal micrometeorological criteria including an extensive homogeneous site must still be met.

To date, however, extremely few experimental studies have actual measurements of dry deposition, and, where these are available, the deposition measurements were mostly made to understand deposition processes as the primary purpose, rather than provide support to N effects studies. With the advent of the TAG approach this may change in the future.

II.3 Measurement-based models of wet deposition

In areas with relatively flat terrain, robust maps of wet deposition may be constructed by interpolating site values of volume-weighted mean ion concentrations in precipitation, combined with measured precipitation amounts. By contrast, in hilly areas there is much more spatial variability, which leads to a major uncertainty in site-based wet deposition estimates. In the more oceanic parts of Europe where frontal rain dominates over convective rain there is a clear altitude enhancement of both precipitation amount and ion concentration. This is due to additional orographic cloud scavenging of gases and aerosols as the air passes into cloud, with the cloud droplets subsequently being incorporated in precipitation through the rain-seeding action of higher-level non-orgographic cloud. Models have been developed to account for this "seeder-feeder" effect (e.g. Fowler et al. 1991, Dore et al. 1992) allowing more robust grid average wet deposition to be calculated down to the 5 km scale (NEGTAP 2001). At the sub-grid level it is important to recognize that, while high altitude sites will often have increased wet deposition compared with the grid average, it is difficult to generalize site-based values in hill areas accurately because of the dependence of this effect on the interaction of wind direction with local and regional topography. The estimation of fine-scale wet deposition in hill areas requires the application of detailed process-based cloud chemistry-advection models for specific meteorological conditions (e.g. Bower et al. 1997).

In addition to increased wet deposition, it is important to note that high-altitude sites often receive a significant additional pollutant input from direct impaction of cloud droplets to the surface (Fowler et al. 1991). Similar physical principles of turbulent transfer apply as for dry deposition. Because of the importance of turbulence, cloud-water deposition is larger to forest compared with short vegetation. Although it is possible to monitor cloud-water chemistry at high elevation sites (e.g. Crossley et al. 1992), estimation of cloud deposition of N requires the application of deposition models, which may also be applied at a regional level (RGAR 1997, NEGTAP 2001, Gauger et al. 2003). As with estimates of wet deposition, where a grid estimate is applied for site-based measurements at a high-elevation site compared with the rest of the grid, then the actual cloud deposition of nitrogen may be substantially larger than the model estimate for the grid square. Conversely, the actual cloud deposition at a low-elevation site in such a grid square might be negligible, depending on the frequency of cloud immersion.

II.4 Measurement-based models of dry deposition

II.4.1. Model approaches

Recognizing the complexity and resources that are required to measure dry deposition continuously, a range of models has been developed to infer dry deposition from measured atmospheric concentrations of N species. The basis of the inferential technique is the application of Ohm's law to trace gas and particle transfer, whereby the total deposition flux (F_d), analogous to the current, is estimated from the product of the potential difference in concentrations (χ_{air}, $\chi_{surface}$) and the resistance to turbulent transfer (R). In the case of a pollutant that is always deposited and where $\chi_{surface} = 0$:

$$F_d = -\chi_{air}(z) / R_t(z) = -\chi_{air}(z) \, . \, V_d(z) \qquad (1)$$

where the deposition velocity, $V_d(z)$, is the reciprocal of the total resistance, $R_t(z)$, and z is a defined reference height above the zero-plane of the vegetation. The total resistance may be decomposed into three components:

$$R_t(z) = R_a(z) + R_b + R_c \qquad (2)$$

the turbulent aerodynamic resistance, $R_a(z)$, the quasi-laminar boundary layer resistance, R_b, and the resistance of the canopy, R_c, which is itself a function of several resistances for transfer to plant cuticles, the soil and through stomata (e.g. Fowler et al. 1991). While eqs. 1 and 2 demonstrate the approach at its simplest, there are many variants that become important for N species. For example, where ammonia emission occurs, such as with an "ammonia compensation point", $\chi_{surface}$ cannot be set to zero and modifications are required that allow bi-directional exchange (e.g. Sutton et al. 1995b, 2001, Nemitz et al. 2001).

Such models may be applied with varying levels of detail. In practical terms, the required input estimates are pollutant concentration at a defined reference height, local wind-speeds and surface roughness. At a more sophisticated level, account may be taken of local temporal effects of atmospheric stability, surface temperature, wetness, humidity and canopy structure.

In many cases, regional surveys of biological impact are compared against mapped deposition estimates. In these cases, it is essential to make the distinction between grid-average and receptor-specific dry deposition. Values of canopy roughness, stomatal resistance, and $\chi_{surface}$ differ widely between canopy types, with the result that $V_d(z)$ (or the effective deposition velocity in the case of bi-directional exchange) is substantially different between canopy types. This effect may give rise to rates of deposition of nitric acid being 50% larger for forest than short vegetation, while the differences for ammonia can be even larger. In comparing ecological responses with atmospheric nitrogen deposition, it is therefore essential to make the comparison with a robust estimate of ecosystem-specific dry deposition. Because of the generally lower net deposition rates to agricultural land (particularly for ammonia), the use of grid-average deposition values will tend to under-estimate nitrogen deposition for semi-natural ecosystems, and hence empirically derived critical loads. The extent of such errors depends on the local mix of different land uses in a grid-square. The differences are largest in areas with a mix of agricultural and non-agricultural and, and smallest where the grid-square is almost entirely made up of just one land-use type. The effect is illustrated in Figure 2, which shows a comparison of grid average total nitrogen deposition to Scotland with the receptor specific estimates for woodland. The difference is smallest in the areas of the north and west, which are dominated by upland moorland, and largest in the agricultural areas of central and eastern Scotland.

II.4.2. Uncertainties in the surface-atmosphere exchange coefficients

Although it is common practice at present to assume that V_d is independent of concentration, it is well established that this does not apply for all pollutants for all concentrations. In the case of particle dry deposition, the exchange depends largely on physical processes, so that it is reasonable to assume that the process is not concentration-dependent. By contrast, the exchange of nitrogen containing gases is linked to chemical equilibria on canopy surfaces and is therefore expected to be concentration-dependent. Ammonia provides an example with limitations to deposition at both high and low concentrations: at low concentrations, the ammonia compensation point may become larger than the atmospheric concentration, leading to periods of emission from the canopy thereby reducing net dry deposition. While the ammonia compensation point is much smaller for semi-natural ecosystems than for agricultural land, it may still affect the net flux. For example, from long-term continuous half-hourly measurements of ammonia exchange with a moor in southern Scotland (Fowler et al. 1998a, Flechard and Fowler 1998), the sum of measured half-hour dry deposition values was found to be 3 kg N $ha^{-1}yr^{-1}$. However, emission events also occurred and accounted for 0.5 kg N ha^{-1} yr^{-1}. This resulted in a net measured ammonia dry deposition flux of 2.5 kg N ha^{-1} yr^{-1}.

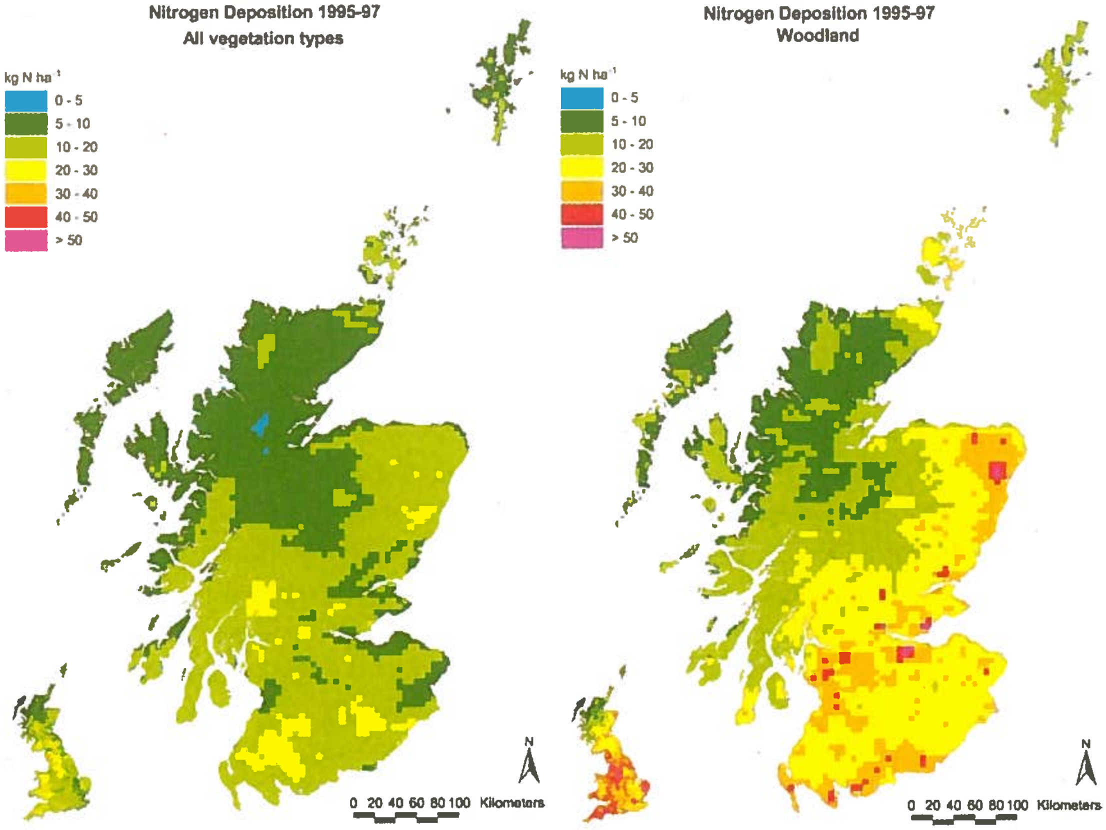

Figure 2: Comparison of estimated average total nitrogen deposition to 5 km grid-squares in Scotland with the deposition received by woodland where present. A substantial difference is seen between the grid-average and receptor-specific estimates. The spatial patterns of deposition across the UK are shown inset (Fowler et al. 2002).

The process-level flux measurements have shown that for semi-natural ecosystems, much of the ammonia is dry deposited to leaf surfaces. This may be enhanced through the parallel deposition of acid species such as sulphur dioxide (Sutton et al. 1993a, Erisman and Wyers 1993). Conversely, immediately adjacent to sources with very high ammonia concentrations, the cuticular uptake may be partly saturated, resulting in smaller V_d than would be estimated at more usual concentrations (Sutton et al. 1993b; Fowler et al. 1998b).

II.4.3. Uncertainties in the air pollutant concentrations

When inferring dry deposition fluxes using equation (1) or its variants, the estimated pollutant air concentration may be an equally large source of uncertainty as the exchange coefficients. Relevant components include the gases ammonia (NH_3), nitric acid (HNO_3) and nitrogen oxides (NO_x), as well as ammonium (NH_4^+) and nitrate (NO_3^-) containing aerosols. In many situations measured concentrations of each of these components may not be available at a study site, and it therefore is necessary to estimate concentrations from models or measurements in the region. In these cases a key distinction may be made between the primary emitted pollutants (NO_x and NH_3) and secondary pollutants (HNO_3, NO_3^-, NH_4^+), the latter being formed by atmospheric reaction of the primary pollutants. Strictly, most emission of NO_x occurs as NO, with NO_2 being a secondary pollutant, but in this case the conversion process is rather rapid, on a time-scale of tens of minutes. Most of the emissions of both NO_x and NH_3 are from transport and agriculture, both of which both of which represent ground level sources. As a consequence, concentrations change rapidly with distance from source, varying hugely at a local level in the rural environment. It is therefore particularly important to make measurements of the concentrations of these primary pollutant gases at key sites where experiments into the effects of nitrogen deposition are conducted. By contrast, the secondary pollutants HNO_3, NO_3^- and NH_4^+ are formed more slowly throughout the atmospheric boundary layer, with the result that concentrations are much less spatially variable. In most cases, a robust estimate of the concentration of these species may be made by interpolation from national monitoring stations, the nearest of which might be 10-50 km distant (e.g. Sutton et al. 2001c).

Local spatial variability for the primary gases is a major source of uncertainty where site-based measurements are not available. Even where modelled estimates are available, for example at a 5 km grid-resolution, there may be as much or more variability within a grid-square as between different grid-squares across a country. An example of this is provided for dry deposition of ammonia in an area of Switzerland in Figure 3 using the model of Rihm (1996) and Rihm and Kurz (2001). In this approach an empirical relationship is established between NH_3 emissions estimated at a 100 m resolution and NH_3 concentrations. NH_3 deposition is calculated from concentration by applying landuse specific deposition velocities. The comparison of the grid estimates at 1 km with 10 km averages shows that, in the centre of semi-natural areas, the 10 km deposition average gives a significant overestimate of the deposition modelled at a 1 km level. Using the low-resolution (10 km) estimates would therefore tend to over-estimate deposition in the centre of a semi-natural area. Conversely, the edges of such natural areas, particularly in agricultural landscapes, are expected to receive substantially more deposition than estimated by lower resolution models. This is further demonstrated by the modelling study of Dragosits et al. (2002), which estimated the landscape level variability of ammonia deposition at a 50 m grid resolution. In some cases, atmospheric ammonia dry deposition to semi-natural land immediately adjacent to farmland was over 5 times larger than at the centre of such a semi-natural area.

Figure 3: Comparison of high- and low-resolution ammonia deposition (kg N ha^{-1} yr^{-1}) estimates for a 100 km x 100 km area of Switzerland (Rihm, 2002, pers. comm.). On the left, estimates are shown at 1 km resolution, which allows a distinction between agricultural and semi-natural areas, with much smaller NH_3 concentrations and depositions over the latter. On the right, averages calculated from the 1 km data are shown at a 10 km resolution, which demonstrate over-estimates of deposition in the semi-natural areas. The EMEP 50 km grid is shown as the diagonal dashed lines.

II.5 Atmospheric transport and deposition modelling.

Atmospheric transport models provide a mass-consistent description of emission, transport and deposition processes. As such, they have been developed to assess the total deposition of air pollutants, including both wet and dry deposition. Under the Convention on Long-Range Transboundary Air Pollution, the European scale estimation of deposition is provided by the Lagrangian and Eulerian models of the European Monitoring and Evaluation Program (EMEP, e.g. Barrett and Berge 1996, Tarrason and Schaug 1999). In addition to estimating atmospheric deposition, a primary purpose of these models it to quantify source-receptor matrices of the origins and fate of air pollutants between different European countries.

In operating at the European scale, the grid resolution of the EMEP models is rather coarse, at 150 km for the Lagrangian model and 50 km for the new Eulerian model. While this resolution is sufficient for the quantification of the transboundary transport, it does provide a key limitation in the estimation of deposition received by sensitive ecosystems for comparison with critical loads. This can be clearly seen from consideration of Figure 3 the area of which covers 100 km x 100 km. Further work is required in order to deal with such spatial-scale issues for the assessment of critical loads exceedance (Spranger et al. 2001). However, the spatial variability at a local level means that the EMEP estimates are a very uncertain reference for estimating deposition to sites in the primary studies used to establish empirical critical loads.

A further limitation of the EMEP models is the extent to which differences in the dry deposition rate of N containing air pollutants are treated. For example, although a preliminary assessment has been made with the EMEP Lagrangian model of the bi-directional exchange of ammonia and its land-use dependence (Sorteberg and Hov 1996), this effect has yet to be included in the operational EMEP models. Further work to improve the EMEP dry deposition scheme is ongoing and should be complete for the review of the Gothenburg Protocol in 2004.

A key advantage of atmospheric transport models is that the primary input data is spatially disaggregated emissions information for each pollutant of concern. While this implies that the model is highly sensitive to uncertainties in the modelled emission data, the emissions can be estimated at a spatially consistent scale, and for regional studies at a high resolution. This allows atmospheric transport and chemistry models of such as FRAME in the UK (Fournier et al. 2002) or EUTREND/OPS in the Netherlands and Germany (van Jaarsveld 1995; Gauger et al. 2003) to be run with a 5 km grid resolution, while in Switzerland the simple empirical model of Rihm (Figure 3), utilizes NH_3 emission estimates at a 100 km level. Such a high degree of spatial resolution can only be achieved by modelling emissions, demonstrating the importance of this approach in estimating locally-relevant N deposition across different countries and regions of Europe. However, the dependence of the models on modelled emissions requires validation or calibration of the model estimates based on measured atmospheric N concentrations and deposition (e.g. NEGTAP 2001).

A wide range of models and approaches is applied to estimate N deposition for different European countries. For example, the FRAME model applies an ecosystem-specific dependence of ammonia dry deposition in a Lagrangian model with simplified statistical meteorology for the British Isles (Fournier et al. 2002). An advantage of the model is that the chemical interactions with atmospheric emissions of NO_x and SO_2 are treated using a multi-layer scheme, allowing explicit treatment of vertical diffusion in the atmosphere. In FRAME, the transboundary import to the country is provided by a related model, TERN, which is run at the European scale and used to initialise trajectories.

A similar approach is followed in the application of the EUTREND/OPS model in the Netherlands and Germany for estimating the dry deposition of NH_x, using high resolution emission, land use, and statistical meteorological data. In Germany this is combined with high resolution NO_y dry deposition estimates based on high resolution land use data and interpolated, low resolution EMEP/DEHM concentration fields and ECMWF modelled meteorological data, as well as interpolated wet deposition measurements. The final product is a national 500 m x 500 m total nitrogen deposition map, which can be directly linked to critical load maps (Gauger et al. 2003).

An example of a rather different national approach is that adopted by the MATCH model of Lagner et al. (1996), and Robertson et al. (1999) for Sweden. In this approach, measurements are assimilated into the model estimates. Gridded national emissions at a resolution of 11 km x 11 km are incorporated into a Eulerian dispersion model to calculate daily air concentrations. Monitored ground level air concentrations for the same periods are then subtracted, with the difference assumed to result from non-Swedish emissions. Because the non-Swedish contribution originates from long-range dispersion, this is assumed to be well-mixed, which allows the non-Swedish contribution to be interpolated between measurement stations. A dry-deposition module is applied in the Eulerian dispersion model to calculate deposition from Swedish sources, and the same module is applied to the concentrations estimated to derive from non-Swedish sources (Robertson et al. 1999, Persson and Bergström 2001).

The models described above are illustrative of the range of approaches currently being used. The model concepts differ significantly, while even for focused process descriptions (such as dry deposition) there are substantial differences in approaches and parametrization. A full critical evaluation of such differences is obviously a major task, and would require the models to be compared for a common area (e.g. a given country where all necessary input data are available) or against a common reference (e.g. the EMEP model). While such an inter-comparison is necessary, it is evident that the uncertainties of these approaches propagate to the setting of empirical critical loads. Different national models are applied as the reference deposition estimates in the primary effects studies across Europe: N-effects studies using national models that underestimate deposition will provide the smallest estimates of no-effect levels, while effects studies using national deposition models which overestimate deposition will provide higher estimates of no-effect levels. As a result, the binding studies used to set critical loads may be linked to models that provide lower estimates of deposition than other models. This uncertainty highlights the need for caution in using correlations between spatial survey data and regionally mapped deposition estimates as the only or primary source of information for setting a critical load.

III. Practical concerns in estimating atmospheric deposition to the plots of N effects experiments.

III.1 Accounting for the different components of deposition.

It can be seen from the above analysis that the many components of nitrogen deposition make assessment of total inputs a complex activity. It is therefore essential that all components of atmospheric input are included. In particular, estimates of wet deposition are significantly less than total deposition, making it essential to include estimates of nitrogen dry deposition. Similarly, as noted in Section II, the dry deposition inputs should be estimated for the specific receiving ecosystem under investigation.

The availability of estimates for each of the major components of nitrogen deposition is also of significant interest when interpreting the experimental results. The most important distinctions to make are between wet and dry deposition and between oxidized and reduced nitrogen deposition, since the relative effects of each may differ. In some instances, reduced nitrogen deposition may be more toxic than oxidized nitrogen, while oxidized nitrogen inputs may be more liable to be leached from the soil. Conversely, a larger fraction of inputs as dry deposition may enter plant foliage (e.g. by stomatal uptake, or cuticular absorption), while a substantial fraction of wet deposition inputs are washed directly to the soil. Hence differential effects of wet and dry deposition may be expected on plants and soils. Most experiments to date have addressed the input of N without consideration of the relative impacts of these different forms. Progress in this area requires that studies consider both the total inputs and each of the components, as well as compare the effect of the different nitrogen forms. In due course this may allow the establishment of a critical loads function accounting for the relative inputs of different nitrogen forms.

In estimating the background deposition to an experimental treatment, it is important to note that this may be modified by the experimental treatment itself. Firstly, high inputs of nitrogen are expected to lead to some release back to the atmosphere as ammonia emission, due to saturation of the ammonia compensation point (Sutton et al. 1995a; Hill et al. 2001). However, this represents a biological response of the system due to N saturation and as such does not appear to be significant at values at or less than empirically derived critical loads. A second example is where exclusion treatments are used to reduce atmospheric nitrogen inputs, such as the placing of roofs under a forest canopy above the soil or completely enclosing plants in unventilated chambers. It should not necessarily be assumed that the atmospheric input has been set to zero, though it becomes a challenge to quantify the input accurately. With experiments were a roof is placed above the soil to intercept throughfall a substantial amount of N still enters the tree foliage directly; with chamber exclusion experiments, the air exchange necessary to maintain CO_2 levels may also permit some dry deposition of N species.

Finally, caution is required in examining effects of nitrogen deposition where experimental treatments are used to apply N inputs at low doses (as kg N ha^{-1} yr^{-1}), but with this being applied on only a few occasions. In this case, any biological effects observed may rather be a result of the short-term high N concentrations supplied (as µg N m^{-3} in gas phase or mg N l^{-1} in aqueous phase), which would not normally occur with natural atmospheric inputs. While it is recognized that atmospheric inputs are not constant, experiments sometimes apply N with an intermittency that leads to concentrations being substantially in excess of those occurring with naturally enhanced N deposition.

III.2 Assessment of nitrogen deposition based on throughfall measurement

Recognizing the difficulties of quantifying dry deposition, an alternative empirical approach has been to measure the composition of throughfall under a vegetation canopy. Throughfall is the term applied to the water that is washed to ground off a canopy following precipitation. At its simplest, the concept is that any dry deposition remains on leaf surfaces to be washed off by precipitation, such that throughfall represents the sum of wet and dry deposition. Throughfall is typically measured by funnels or gutters placed under a forest canopy. This approach has been widely applied to estimate sulphur deposition to forests in Scandanavia, but it has been known for many years to provide substantial errors in estimating atmospheric N inputs (Lövblad et al. 1992). In the case of nitrogen, because significant fractions of both the wet and dry deposition are absorbed by the canopy, the N flux in throughfall is substantially less than the total deposition. The fraction absorbed also appears to be a function of the nitrogen supply and extent of N saturation in the canopy, with the largest canopy uptake occurring for N limited canopies. Hence in clean locations, the N flux in throughfall is usually even less than the wet deposition! (Canopy uptake of deposited N > dry deposition of N).

An example demonstrating the importance of canopy uptake of nitrogen is provided by an experiment that artificially added N in mists above a Sitka spruce canopy of around 15 years old (Cape et al., 2001). An estimate of background deposition was provided by nearby monitoring, while measurements of throughfall chemistry in gutters underneath the canopy allowed the relationships between atmospheric deposition and throughfall to be calculated. An example of the results for nitrate are shown in Figure 4, which indicates clearly how the measured NO_3^- in throughfall is much less than that applied experimentally. If these results are combined with a similar pattern for NH_4^+ in throughfall (*TF*) from this experiment ($TF(NH_4^+) = 0.69\, Applied(NH_4^+) - 4.16$, $r = 0.97$, meq m^{-2} in 4 months), and an estimated background dry deposition of 5.5 kg N ha^{-1} yr^{-1} added, a combined relationship is found of:

$$\textit{Total N deposition} = 1.48\ TF\ (NO_3^- + NH_4^+) + 11\ (\text{kg N ha}^{-1}\ \text{yr}^{-1}). \qquad (3)$$

The canopy in question was rapidly growing which may partly account for the substantial canopy uptake rates. However, this illustrates the substantial deviation possible between atmospheric N deposition and N in throughfall. In terms of uncertainties, for a slow growing mature forest, the relationship might be as low as:

$$\textit{Total N deposition} = 1.25\ TF\ (NO_3^- + NH_4^+) + 5\ (\text{kg N ha}^{-1}\ \text{yr}^{-1}). \qquad (4)$$

For another Scots Pine woodland in S. Scotland (Theobald et al. 2001) the relationship between estimated deposition and throughfall was found to fit well within these bounds. Wet deposition was measured at 10.1 kg N ha^{-1} yr^{-1}, with dry deposition estimated at around 7-12 kg N ha^{-1} yr^{-1}. This implies that total deposition to the site is around 17-22 kg N ha^{-1} yr^{-1}. The measured NH_4^+ and NO_3^- in throughfall was less than in wet deposition at 7.6 kg N ha^{-1} yr^{-1}. Applying the relationships above (equations 3 and 4), based on the throughfall estimate, total N deposition would be estimated at 14-22 kg N ha^{-1} yr^{-1}, which is close to that estimated at the site from wet and dry deposition.

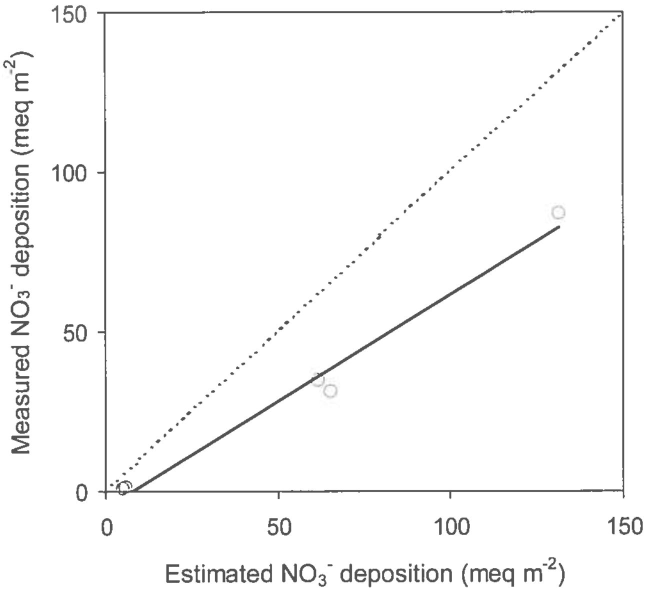

Figure 4: Relationship between estimated nitrate deposition applied in experimental treatments (not including estimated background dry deposition) and nitrate measured in throughfall under a 15 year old canopy of Sitka spruce in Scotland (Cape et al. 2001). $NO_3^-{}_{measured} = 0.66\ NO_3^-{}_{estimated} - 4.77$, $r = 0.99$.

In principle, canopy budget models (e.g. Draaijers et al. 1996, de Vries et al. 2001), can be applied to derive an improved estimate of total N deposition from throughfall measurements. In these approaches, base-cation fluxes in wet deposition and throughfall are also determined, and it is assumed that the canopy leaching of these components balances the canopy uptake of nitrogen. This approach provides a correction to increase the N deposition estimates based on throughfall measurements (e.g. de Vries et al. 2001, 2002). However, there is concern that this approach still leads to some underestimation of total atmospheric N inputs (e.g. Gauger et al. 2003). Where feasible, approaches that are based on direct flux measurements or inferential modelling are considered the most robust.

Given the underestimation of N deposition by throughfall, experiments using throughfall as the estimate of N deposition are expected to underestimate the critical load. Such experiments therefore need careful interpretation, and where possible correction to estimate the actual total deposition, if they are to be applied to estimate empirical N critical loads.

The canopy exchange of N revealed by the above comparison of throughfall and total N deposition also has important implications for experimental studies on forest ground flora. In most studies of this kind, nitrogen is added directly to the ground flora, under the main tree canopy (e.g. Nordin et al. 1998, Strengbom et al. 2002). The consequence of this direct addition, is that the tree canopy has no opportunity to absorb the N before it reaches the understory vegetation, as would occur with atmospheric inputs. Therefore, the dose of N applied in these studies should be related to the (larger) atmospheric N input that would be required to provide a throughfall flux of the magnitude of the N applied. In practice, similar equations of the form of equations (3) and (4) may be used. Hence a direct N treatment to ground vegetation of e.g. 12.5 kg N ha^{-1} yr^{-1} (Strengbom et al. 2001), might be equated to an atmospheric N input of 21 - 30 kg N ha^{-1} yr^{-1}.

III.3 Organic N in precipitation and dry deposition.

Organic nitrogen in precipitation, and as a component of dry deposition, has been ignored in the Critical Loads debate. Recent measurements and critical reviews have concluded that organic N may contribute on average 30% of total N in precipitation (Neff et al.,2002; Cornell et al., 2003). Because the contribution of organic N has been ignored in both setting Critical Loads and in measuring deposition, it may be that explicit inclusion of organic N would raise both Critical Loads and deposition values by a similar amount and have little effect on exceedances. However, little is known about the chemical composition or origin of organic N, whether it is exchanged in plant canopies, or whether it can be directly utilised by plants. It is therefore impossible at this stage to attempt to control emissions and deposition. However, as deposition of inorganic N decreases, organic N as an additional source of N to ecosystems may become an important component of the background deposition at a sensitive site.

III.4 Importance of local variability and spatial uncertainties

The importance of local variability in atmospheric N deposition was highlighted in Section 2, both in relation to wet and dry deposition processes. In assessing the deposition of nitrogen to experimental sites, it is therefore important to consider both aspects. Two examples from the UK may serve to illustrate this. In both cases, the uncertainty in deposition would have consequences for the estimation of empirical critical loads.

Pearce and van der Val (2002) applied additional nitrogen in the form of ammonium or nitrate to a mountain heath characterized by Woolly Hair moss in the Grampian Mountains of Scotland. On the basis of the mapped estimates of RGAR (1997), with a 20 km x 20 km grid resolution, they estimated that the background N deposition to their site was 12 kg N ha^{-1} yr^{-1}. While this approximate estimate may be sufficient for the original purpose of their study of studying biological effects with additional N (+10 and +40 kg N ha^{-1} yr^{-1}), the exact value of the background deposition becomes more critical if the results of their study should be applied to help set an empirical critical load for this habitat. In this location, concentrations of NO_x and NH_3 are established as being very small, and wet and cloud deposition dominate the total nitrogen inputs. As noted in Section 2, these inputs are highly dependent on altitude, so that the relative height of the study site to the grid-square is critical. The experiments were made on the exposed mountain summit of Glas Maol at 1068 m, which would be significantly higher than the 20 km grid average. This is at least part of the reason why more recent mapped estimates of deposition at a finer scale (5 km grid) suggest a larger background N deposition of 15.3 kg N ha^{-1} yr^{-1} (NEGTAP 2001). Accounting for the additional increase in cloud and wet deposition at the mountain summit itself, the actual inorganic N deposition to this site is probably more than 18-20 kg N ha^{-1} yr^{-1}.

In another experiment Uren et al. (1997) and Power et al. (1998, 2001) have applied a very low dose of nitrogen over several years to a lowland heathland at Thursley in south-east England. In this example, the regional topography is rather flat so a significant local variability in wet deposition is not expected. By contrast, the heathland represents a relict habitat existing within an area of mixed agricultural, rural and sub-urban land use. A substantial local variability in NH_3 and NO_x concentrations and dry deposition in the vicinity of this site is therefore expected. The 5 km resolution mapped N deposition to this site is 21 kg N ha^{-1} yr^{-1} (NEGTAP 2001). However, here the actual deposition to the site may be less than the mapped estimates, since the experimental site is in the centre of a nature reserve and is probably more than the average distance away from sources in the grid square. A combination of local monitoring data and mapped estimates allows a simple site-specific estimate of N deposition to this site to be made, as shown in Table 1. While the component inputs are calculated in Table 1 using a rather simple methodology using estimated deposition velocities, they demonstrate the importance of site-based data. The total deposition is estimated at around 14 kg N ha^{-1} yr^{-1}, which is substantially less than the mapped estimate. This is largely because the local NH_3 concentrations (0.4 μg m^{-3}) are much less than the 5 km grid mapped estimates, which would suggest an NH_3 concentration of 2.2 μg m^{-3}. This difference is fully consistent with the expected sub-grid variability of NH_3, with much lower concentrations in the centre of semi-natural areas compared with over adjacent farmland (Dragosits et al. 2002). It should be noted that measurements of N wet deposition were made at the Thursley site, but not used in the present estimate. This was because of the uncertainty in estimated precipitation amounts (Section II.1) and the need for long-term estimates over several years. In this instance the wet deposition estimates from the national monitoring network were preferred, especially since the locally flat topography would indicate little local spatial variability of long-term wet deposition estimates.

Table 1: Estimated contribution of the different components of nitrogen deposition to a lowland heathland at Thursley, south-east England. The site has been used by Power et al. (1998, 2001) to study the effects of nitrogen deposition.

Component	Concentration in air (μg m^{-3})	Deposition velocity (mm s^{-1})	Nitrogen Deposition (kg ha^{-1} y^{-1})
Gaseous NO_2	7.1 ^{a}	1	0.68
Gaseous NH_3	0.4^{b}	15	1.56
Gaseous HNO_3	1.4^{c}	25	2.45
Aerosol NH_4^+	1.3^{b}	2	0.64
Aerosol NO_3^-	2.9^{c}	3	0.62
Wet dep NH_4^+	-	-	5.18^{d}
Wet dep NO_3^-	-	-	3.36^{d}
Total			**14.5**

a, On site measurements (S. Power, pers. comm.).

b, National Ammonia Monitoring Network, measurements of NH_3 on site and interpolation from nearest sites measuring NH_4^+ aerosol (Sutton et al. 2001c; www.nbu.ac.uk/cara).

c. Interpolation from nearest sites of the National Nitric Acid Monitoring Network (Sutton et al. 2001d; www.nbu.ac.uk/cara).

d. NEGTAP (2001) based on interpolated volume weighted precipitation chemistry and mapped precipitation amounts mapped on a 5 km grid. (On site measurements for 22 months (Power, pers. comm.) suggest that the total N wet deposition could be even lower at 5.6 rather than 8.5 kg N ha^{-1} yr^{-1}).

III.5. Application of spatial surveys to assess significant deposition levels.

Spatial surveys form an important method to derive information about the effects of atmospheric nitrogen deposition. Where the biological signal can be clearly related to atmospheric N, the data can provide direct local or regional evidence of environmental impacts. Significant caution is, however, needed in using studies of this kind to set empirical critical loads. The primary uncertainty is that any critical load derived is completely dependent on the accuracy of the background atmospheric deposition estimates used. Depending on the methods used, the uncertainties may be more than +/- 50%. Particular caution is therefore required in interpreting the quality of the deposition estimates used.

Additional concerns centre on the interpretation of spatial correlations, and the level at which the response may be considered statistically significant. For example, several authors have identified a correlation across N. Scandinavia between the occurrence of epiphytic green algae and nitrogen deposition (Gőransson 1990, Thomsen 1992, Bråkenhielm and Quinghong 1995, Poikolainen et al. 1998). All of these studies are, however, dependent only on spatial correlations with estimated atmospheric

N deposition. However, the distribution of N deposition in Scandinavia is also correlated with several other environmental factors. This was recognized by Bråkenhielm and Quinghong (1995), who noted that higher spatial correlations of epiphytic algal occurrence occurred with both sulphur deposition ($R^2 = 0.67$) and growing season ($R^2 = 0.64$) than with nitrogen ($R^2 = 0.52$).

IV. Uncertainties in the deposition estimates of example studies.

Based on the analysis of issues outlined above it is possible to screen studies of N effects regarding the robustness of their atmospheric deposition estimates. This in turn feeds into their usefulness in estimating empirical critical loads for nitrogen (Bobbink et al. 2002), although in some cases likely approximate corrections to the deposition estimates may be made. As an example, this prioritisation of studies is considered here using studies for forest ecosystems that have been collated by Bobbink et al. (2002), with results summarized in Table 2A-E. As noted earlier, the point of the present review is explicitly *not* to estimate critical loads; this has been done by the various Working Groups of the Bern Workshop. However, the information on deposition provides a basis for interpretation and screening of studies in the estimation of empirical critical loads.

From Table 2, it is clear that each experimental or survey study must be evaluated individually. While the deposition estimates of some studies are rather robust, for others (e.g. Nebe 1991, Hippeli and Branse 1992, Duquesnay et al. 2000) the deposition estimates are too uncertain to allow the studies to be applied to estimate empirical critical loads.

Table 2A: Uncertainties in the deposition estimates of example studies of potential use for estimating empirical critical loads for nitrogen: tree nutrition and physiology.

Study	Effects observed at (kg N $ha^{-1}yr^{-1}$)	Deposition estimate basis	Contribution from atmos. deposition (i.e.sensitivity)	Possible biases and uncertainties	Robustness of deposition level for setting Crit Load	Possible deposition at which effects detected*
Duquesnay et al. 2000	20-30 (long term nutrient change)	early estimate of just wet deposition (Aussenac et al. 1972)	100%	-Regional estimate only - Inadequate dry deposition estimate	Unusable	(20-50) Unknown
Flückiger and Braun (1998); Braun et al. (2002)	15-35 (correlation with nutrient change; correlation with storm damage)	Regional model (BUWAL 1994; Rihm, pers. com)	100%	-correlation: so significant effects level depends on regression confidence limits	Medium-robust	25
Hippeli & Branse (1992); Nebe (1991)	15-25 (long term nutrient change)	Estimated only (no measurements reported)	100%	-Deposition values appear unrealistically low; -very high S deposition.	Not usable	unknown
Flückiger and Braun (1999)	22-40 (nutrient imbalance)	Regional model BUWAL (1994): 12-20 + Addition: 10-20	~50%	Only 4-6 yrs of treatment	Medium-robust	30 (22-40)
Quiring et al. (1997)	14-37 (correlation with leaf arginine levels)	Regional model (BUWAL 1994; Rihm, pers. com)	100%	-correlation: so significant effects level depends on regression confidence limits	Medium-robust	25

*This value should not be confused with the critical load, which includes consideration of various biological factors and experimental treatment periods.

Table 2B: Uncertainties in the deposition estimates of studies of potential use for estimating empirical critical loads for nitrogen: Studies on mycorrhizae and forest soils.

Study	Effects observed at (kg N ha^{-1} yr^{-1})	Deposition estimate basis	Contribn from atmos deposition (sensitivity)	Possible biases and uncertainties	Robustness of deposition level for setting Crit Load	Possible deposition load at which effects detected*
Lilleskov et al. 2001,2002	1-18, reduced mycorrhizal diversity at most polluted site cf. to cleanest (Alaska)	Bulk deposition **only** (Whytemare et al. 1997); local transect away from industrial NH_3 production factory. Historic N input was 5 x larger	100%	No account taken of dry deposition (large underest. varying with distance from source e.g. ~0-40 kg N ha^{-1} yr^{-1})	Very uncertain (unusable)	10-50? (largely unknown). Historic values larger
Erland et al. 1999	24-29, compared with 14-15 in two different areas: Reduced mycorrhiza occurrence	Not stated by the paper	100%	-Possible underestimate of deposition. -Spatial correlation with S deposition	Uncertain	24-29 (uncertainty could be 20-35)
Wöllecke et al. (1999)	>35 compared with 10-20. (reduced mycorrhizal diversity)	Bulk deposition estimates. NH_3 concs also measured at 10 μg m^{-3} compared with 0.5 μg m^{-3}.	100%	Deposition much reduced in later years. But may be residual effects. Additional input from NH_3 dry deposition could be >30 kg N ha^{-1} yr^{-1}	Very uncertain	>50?
Falkengren Grerup & Diekmann 2002	Average estim. deposition of three regions in Sweden: 7, 10 & 17. (change in mineraliz'n nitrification & C:N ratio)	Combination assimilation model (Lagner et al. 1996) 20 km resolution with forest specific estimates	100%	-low spatial resolution and regional groupings (so likely dep is both under- & over-estimated) -Spatial patterns correlated to S dep.	Uncertain-Moderate	10-15 (larger if high local variability in deposn)
DeVries et al. (2001)	12-15 (increase in nitrate leaching)	Throughfall, (with canopy exchange correction from base cations)	100%	- Throughfall estimate improved by canopy exchange correction, but underestimate still expected.	Very uncertain	Less than 20-30** depending on extent of base-cation correction

*This value should not be confused with the critical load, which includes consideration of various biological factors and experimental treatment periods.

** Values based on approximate correction of raw throughfall estimates to deposition (eqns 3,4). Actual deposition may be lower, depending on the completeness of the base-cation throughfall correction.

Table 2C: Uncertainties in the deposition estimates of studies of potential use for estimating empirical critical loads for nitrogen: Studies forest growth and health.

Study	Effects observed at (kg N ha^{-1} yr^{-1})	Deposition estimate basis	Contribn from atmos deposition (sensitivity)	Possible biases and uncertainties	Robustness of deposition level for setting Crit Load	Possible deposition load at which effects detected*
Nellemann and Thomsen 2001	7-15 (growth reduction)	Wet deposition in 1990 only; source of estimate unclear	100%	No dry deposition inc. (likely underest by 3-10 kg) -Grouped deposition estimates. -Acidification and climate stress also implicated	Very uncertain	10-25 (very uncertain)
Braun and Flückiger 2002	26 (growth stimulation)	BUWAL model: 16 Exptl addition: 10	62%	Lower uncertainty due to experimental addition	Medium	26
Matzner and Murach 1995	(2 mg N /l in soil solution) Estimated as equivalent to 25 kg N ha^{-1} yr^{-1}. (Reduction in fine root biomass)	Throughfall (De Vries et al. 2001, Bobbink et al. 2002)	100%	Underestimation of deposition from throughfall	Very uncertain	Less than 35-48** depending on extent of base-cation correction

*This value should not be confused with the critical load, which includes consideration of various biological factors and experimental treatment periods.

** Values based on approximate correction of raw throughfall estimates to deposition (eqns 3,4). Actual deposition may be lower, depending on the completeness of the base-cation throughfall correction.

Table 2D: Uncertainties in the deposition estimates of studies of potential use for estimating empirical critical loads for nitrogen: Forest ground living and epiphytic lichens, bryophytes and algae.

Study	Effects observed at (kg N ha^{-1}yr^{-1})	Deposition estimate basis	Contribution from atmos. deposition (i.e.sensitivity)	Possible biases and uncertainties	Robustness of deposition level for setting Crit Load	Possible deposition at which effects detected*
Strengbom et al. 2001	34 (lichen loss)	Single annual N addition for 28 yrs, direct to understory vegn Background dep not included.	Probably >5-10 kg N (>25-35%)	Need to include background N dep. - Because of canopy exchange N added below the tree canopy equiv to a larger atmos N input.	-Medium if N inputs corrected for background and below canopy application. - Note single annual N doses may alter responses	45-62**
Bråkenhielm and Quinghong (1995)	3-25 (spatial correlation to N dep with decrease in lichen diversity; increased *Pleuro - coccus* algae on tree bark). No temporal change.	Modelled deposition (Granat 1990, plus Lövlad et al. (1992) implied.	100%	-spatial correlation with N, but higher correlations S deposition and growing season. -grid estimated deposition results in local uncertainty	Uncertain	Unclear; Authors concluded causality with N was not certain
Poikolainen et al. (1998)**	3-4 (occur'nce of *Scoliciosporum* and *Pleurococcus* algae increased), based on spatial data in Finnland	Appears to be based on throughfall	100%	-Deposition values low; Significant underest by throughfall. -Spatial differences also reflect climate and S deposition.	Uncertain	6-12. Not clear that N is causal
Mitchell et al. (2002)	15-18 (change in epiphytic bryophyte spp in Atlantic Oakwood	18 is based on modelled dep to woodlands. 15 is refined estimate from site based measnts.	100%	- Modelled estimate a slight overestimate. - spatial differences also reflect S deposition	Medium-good	15

*This value should not be confused with the critical load, which includes consideration of various biological factors and experimental treatment periods.

** Equivalent deposition range calculated using eqns 3, 4 for 47 yr old *Pinus sylvestris* plantation to account for ground application of nitrogen.

Table 2E: Uncertainties in the deposition estimates of studies of potential use for estimating empirical critical loads for nitrogen: woodland ground flora.

Study	Effects observed at (kg N $ha^{-1}yr^{-1}$)	Deposition estimate basis	Contribn from atmos. deposition (sensitivity)	Possible biases and uncertainties	Robustness of deposition level for setting Crit Load	Possible deposition at which effects detected*
Falkengren-Grerup (1986, 1995)	7-11 and 11-17 (ground flora spp change), temporal change for forests in S. Sweden	MATCH assimilation deposition model (Lagner et al. 1996)	100%	- Response was also spatially correlated with S dep of Lövblad et al. (1992) - grouped deposition classes may mask substantial variability in deposition	Moderate	~10
Rosen et al. (1992) / Bobbink et al. 1996.	7-11 (increase in *D. flexuosa*) temporal change informally correlated with N	Based on Lövblad et al. (1992) regional model (used non-ecosyst -specific values)	100%	- Ecosystem specific dep may be 10-20 kg N ha^{-1} yr^{-1} - Also spatially correlated with S deposition	Uncertain	10-20
Kellner and Redbo-Torstensson (1995)	5 (increase in *D. flexuosa* following N addition to ground layer for 5 yrs) (Sweden)	Background dep not considered (prob at least 5 kg N ha^{-1} yr^{-1}), plus addition of N.	~60%	- Background dep should be included. - Addition was made under the canopy, so implies equiv larger atmos N input. - 5 yrs exposure	Very uncertain	15-23 ** (accounting for bg dep& tf effect)
Nordin et al. (1998), Strengbom et al. (2002)	16 (arginine levels in bryophytes increased & increased pest attack of ground flora following expertl addition; Sweden)	Background dep estimated at 3.5 kg N ha^{-1} yr^{-1} (regional model)). Plus 1 to 3 yrs addition of 12.5 kg N ha^{-1} as single annual application.	~75%	-Background dep from Lövblad et al. (1992a), appears non-ecosystem specific value used -Addition was made under the canopy, so equiv to larger atmos N deposition. - Some effects after only 1 yr exposure, but single annual doses, so may be conc. effects.	Very uncertain	24-33**
Pitcairn et al. (1998)	15-20 (species changes along local transect near a farm) (Scotland)	Site-based measurements and modelling.	100%	-Uncertainty of NH_3 cuticular saturation affects dry deposition model estimates	Moderate-Robust	15-20

*This value should not be confused with the critical load, which includes consideration of various biological factors and experimental treatment periods.

** Equivalent deposition range calculated using eqns 3, 4 for ground application of nitrogen.

V. Conclusions

This review has highlighted how deposition estimates are not just necessary to estimate critical loads exceedance, but are also essential to set empirical critical loads for nitrogen. The conclusions of experimental studies and observational surveys are found to be highly dependent on the background atmospheric input to study sites. The critical loads suggested by observational surveys are by definition 100% sensitive to the background atmospheric deposition estimates, while in such studies spatial or temporal correlations with other factors, such as S deposition or climate, may also obscure the results. The most robust data for the assessment of empirical critical loads are therefore provided by experimental N addition studies. However, even here at the lowest treatment levels showing effects, the contribution of background atmospheric deposition to the total is often significant (>50%). A particular concern applies to experiments where N is added directly under a tree canopy. Here the same canopy interactions that cause N in throughfall to be substantially less than total N deposition also affect the interpretation of results: the N applied under a tree canopy is equivalent to a significantly larger atmospheric deposition than the treatment applied.

The understanding of these factors is useful to prioritise the various ecological studies in terms of the robustness of deposition estimates and in some cases to allow corrected estimates of deposition to be made. This information on N deposition estimation has fed into to the discussions of the Bern Working Groups, which will have increased the robustness of the empirical critical loads derived by the groups.

At the same time, there is a continuing need for the research community studying the ecological effects of N deposition to be more aware of the importance of reliable atmospheric inputs. For studies that are to be applied to help set critical loads for nitrogen, the definition of atmospheric input is often a larger uncertainty than the description of biological effects. In order to increase awareness of the importance of deposition estimates, the various approaches for quantifying atmospheric N inputs have been summarized, highlighting the uncertainties. It is hoped that this will encourage a more robust assessment of atmospheric N inputs in future studies.

Finally, it should be noted that most studies still only consider the effects of total nitrogen deposition. There is still significant work to be done to consider the extent to which the effects in some experiments are a result of high N concentrations (in air or artificial precipitation) rather than N deposition, as well as to consider the relative impacts of NO_y vs NH_x. Substantial differences may be expected and these will have major consequences for the development of N abatement strategies. In due course, a target might be set to establish different empirical critical loads for both wet vs dry deposition, as well as for NO_y vs NH_x deposition.

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Deposition measurements at Thursley Common Heathland Nature Reserve

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Background

Since 1989, a long term nitrogen manipulation experiment has been ongoing at Thursley Common, a National Nature Reserve in Surrey, England (Grid Reference SU 910404). The aims of this study include the quantification of biological and chemical responses of lowland heathland to elevated inputs of nitrogen, and assessment of the speed of recovery following the cessation of experimental nitrogen inputs. A further aim has been to establish the role that habitat management plays in determining plant and microbial response to nitrogen.

Additions of ammonium sulphate were made to heathland plots at rates of 0, 7.7 and 15.4 kg N ha^{-1} yr^{-1}, from 1989-1996. No further additions were made after 1996, although the rate of ecosystem recovery has been subsequently monitored. Regular measurement of plant and, more recently, microbial parameters have been carried out throughout the experimental period and have revealed effects on growth, chemistry, phenology and nutrient cycling. Many of these effects were significant even at the lowest rate of nitrogen addition, suggesting that lowland heathland may be very sensitive to nitrogen inputs. However, in order to determine the deposition threshold at which effects are seen, measurement of background inputs is required. To this end, on site measurements of bulk deposition together with gaseous NH_3 and NO_2 concentrations were carried out over a two year period from 1998 to 2000. In addition, NH_3 concentrations have also been measured by the Centre for Ecology & Hydrology (CEH), as part of the National Ammonia Diffusion Tube Survey, from 2000-2002.

Measurement methodology

Gaseous NO_2 concentrations were measured using passive diffusion tubes (Willem's badges), following the method outlined by Willems (1993). Two replicate badges were placed in each of two locations close to experimental plots, at a height of 1.5m, giving a total of four replicates for each measurement occasion. In addition, two blanks were prepared (kept sealed and refrigerated) for each measurement period. Badges were exposed for a month in the field and then brought back to the laboratory for analysis using HPLC. Measurements were repeated monthly from April 1998 to July 2000.

Ammonia concentrations were measured on site during the same period using a second set of Willem's badges. However, the improved accuracy, at low concentrations, of alpha samplers used in the CEH NH_3 monitoring network, compared with Willem's badges (Tang *et al.*, 2001) indicate that data obtained on site using alpha samplers is more representative of background concentrations. For this reason, mean monthly concentrations of NH_3, measured using three replicate alpha samplers, at a height of 1.5 m, have been used for the period June 2000 to November 2002. A measurement protocol for alpha samplers is outlined in Tang *et al.* (2001).

Ammonium and nitrate concentrations in bulk deposition were measured using two replicate rainfall collectors, consisting of a collection vessel and a funnel. Acetonitrile was added to vessels to prevent microbial activity. Samples were analysed for NO_3 and NH_4 using HPLC. Rainfall quantity and NO_3/NH_4 concentrations were measured monthly from August 1998 to May 2000 and a 12 month mean deposition rate calculated for the 22 month period. For both dry and wet deposited compounds, inputs are expressed as kg nitrogen ha^{-1} yr^{-1}.

Deposition data

Table 1 summarises mean monthly concentrations and annual deposition inputs to Thursley Common over the sampling period. Deposition velocities of 0.00077 m s^{-1} and 0.01905 m s^{-1} have been used for NO_2 and NH_3 respectively, and are derived from the model described in Smith *et al.* (2000).

	Mean conc. ($\mu g\ m^{-3}$)	Mean annual deposition (kg N ha^{-1} yr^{-1})
Dry NO_2	7.05	0.52
Dry NH_3	0.40	1.98
Wet (NO_3+NH_4)	-	5.55
Total (wet + dry)		8.05

Table 1. On site deposition measurements at Thursley Common NNR.

During the measurement period, background nitrogen inputs were predominantly in the wet deposited form, representing almost 70% of the total deposition to the site. Low ammonia concentrations reflect the absence of any agricultural influence in the surrounding area. Whilst it is recognised that the measurements carried out on site do not include nitric acid inputs, the total measured deposition of 8 kg N ha^{-1} yr^{-1} indicates that the site is receiving below average inputs for UK heathland in general, and lower deposition rates than modelled estimates for the relevant UK grid square in particular.

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Forest habitats

Impact of atmospheric nitrogen deposition on epiphytes in Atlantic Oakwoods.

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Introduction

Lichens and bryophytes have been shown to be effective indicators of acidic pollution (Adams & Preston, 1990 Farmer, 1991). However, there have been few studies on the effects of atmospheric nitrogen deposition on bryophytes and lichens, especially epiphytic bryophyte species. Most studies have concentrated on either the lichen or bryophyte community, but in practice epiphytic communities are intermixed and it is important to assess them both. The present study has for the first time assessed the effects of atmospheric nitrogen (N) deposition on the epiphytes of Atlantic Oakwoods that may be very sensitive to changes in N supply.

In Europe, Atlantic Oakwoods are largely restricted to the oceanic western coastal fringes of the British Isles. Atlantic Oakwoods are particularly rich in bryophytes, ferns and lichens, and can be described as 'Temperate Rainforests'. As scattered remnants of a previously more extensive ancient woodland type, the remaining woods have an internationally recognised conservation value and are identified as a habitat of particular importance in the EU Habitats Directive (incorporated under habitat 91A0). According to the EUNIS classification, the group would fall into category G1.83. The bryophyte and lichen communities are one of the reasons for the establishment of an oakwood Biodiversity Action Plan (BAP) (UK Biodiversity Group, 1995) and air pollution is listed here as a possible factor affecting the quality of the habitat.

The aims of this study were to estimate the exposure of the epiphytes to atmospheric N (concentrations, deposition, stemflow) and to quantify the effects of atmospheric nitrogen deposition on epiphytes in Atlantic Oakwoods in terms of species composition.

Methods

The study sites

Seven comparable study sites were chosen to represent a range of N deposition levels (Fig. 1). There were six sites in Scotland ranging from Loch Maree in north-west Scotland to Wood of Cree in Dumfrieshire with a seventh site at Borrowdale in the Lake District, England. All the sites were mature oakwoods with trees with girths generally larger than 1 m. The woods contained a mixture of oak (*Quercus petraea* and *Q. robur* and presumed *hybrid Q. x rosacea*), rowan (*Sorbus aucuparia*), hazel (*Corylus avellana*), birch (*Betula pendula* and *B. pubescens*) and ash (*Fraxinus excelsior*). The survey work and measurements of atmospheric N inputs was confined to oak trees. The sites and monitoring equipment were established in February 2001.

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However, the outbreak of foot and mouth disease in March 2001 meant that sampling had to be abandoned until August 2001 when monthly sampling resumed until February 2002.

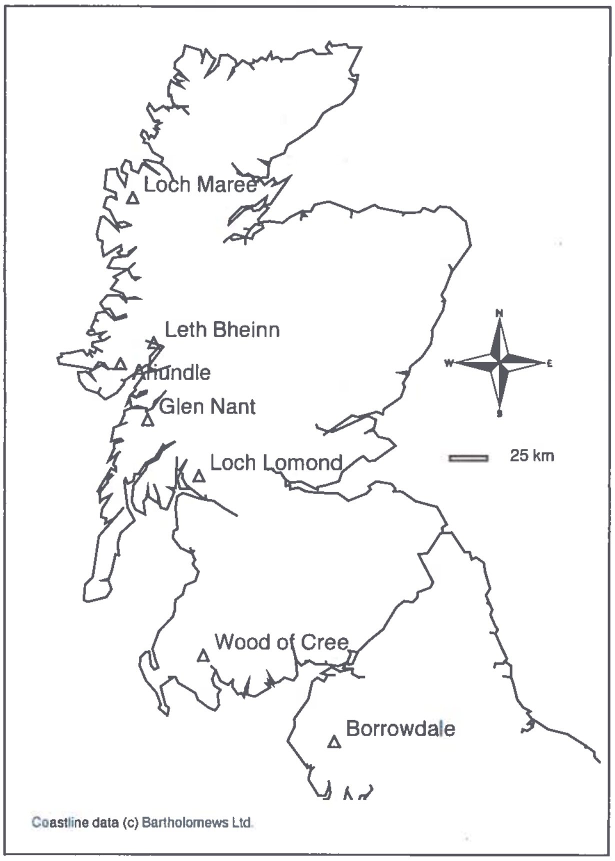

Figure 1: Location of the Atlantic Oakwood study-sites in Scotland and N.W. England.

Monitoring of nitrogen concentrations and deposition

At each site, 4 trees were randomly selected for stemflow collection. Plastic tubing was placed around the tree at a height of 1.5 m above the ground. Rectangular slots along the tube allowed water to flow into the tube. The tube was attached to the tree at a slight angle to enable the water to flow down the tube and into collection vessels placed at the bottom of the tree. The tubing around the tree was sealed to the tree using non-toxic silicone sealant. A rim of sealant was also made around the outside edge of the tube so all water was channelled into the tubing. The sealant was tested to ensure that it did not leak any N. All the water passed through a gauze bag to exclude any invertebrates and plant debris before entering the collecting vessel. Rainfall was collected in an open area, i.e. not under the tree canopy, within the sites. Thymol was added to both stemflow and rainfall collection vessels to inhibit microbial growth. The volume of stemflow and rainfall was recorded monthly and samples taken for analysis. The monthly water samples from both the rainfall and stemflow were analysed for nitrate and ammonium.

Monthly air NH_3 concentrations were measured at each site by high sensitivity passive sampling with the CEH ALPHA (Adapted Low cost Passive High Absorption) samplers (Tang *et al.*, 2001). The ALPHA samplers were attached on a post 1.5 m above the ground. Three replicate samples were used each month at each site in order to give a reliable estimation of the air concentration of ammonia.

Epiphytic species survey

A systematic survey of the epiphytes was conducted using a quadrat sampling procedure similar to that of (Bates & Brown, 1981; Bates, 1992). At each site 20 randomly selected trees were sampled. For each tree the percentage cover of all species was recorded in a 10 x 40 cm quadrat at 1.5 m above the ground. Independent quadrats were recorded on the north, east, south and west aspects of the trunk. Nomenclature follows Smith (1978) for mosses, Paton (1999) for liverworts and Dobson (2000) for lichens. The girth of the tree at 1.3 m above ground was measured. The survey work was carried out between April and September 2002.

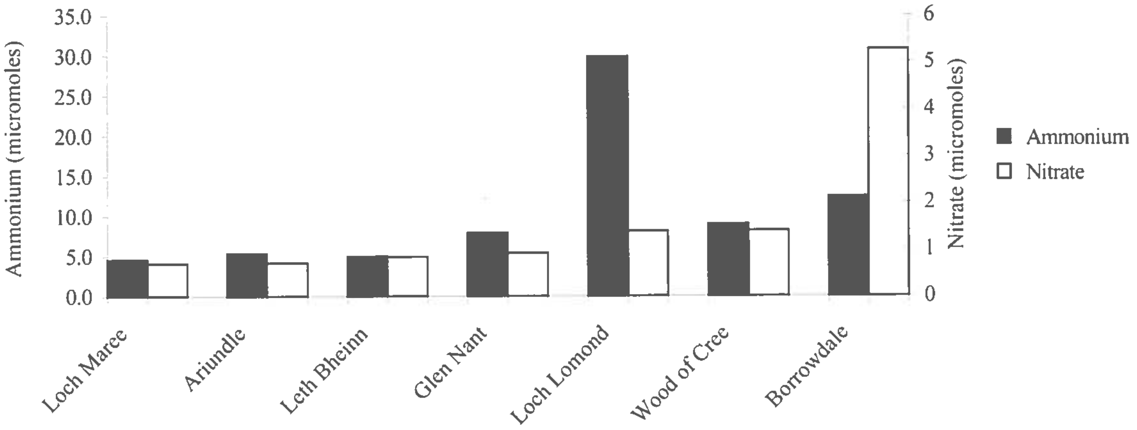

Figure 2: Volume weighted concentration of ammonium and nitrate in the stemflow at the seven sites.

Results

Measurements of nitrogen exposure

Volume weighted mean concentrations of NH_4^+ in stemflow (Fig 2) ranged from 4.6 μmol l^{-1} at Loch Maree to 12.4 μmol l^{-1} at Borrowdale, while the parallel concentration of NO_3^- were lower, ranging 0.7 μmol l^{-1} at Loch Maree to 5.3 μmol l^{-1} at Borrowdale. The total stemflow flux passing the surveyed epiphytes over the 7 months ranged from 3-12 mmol NH_4^+/tree and 0-5 mmol NO_3^-/tree.

Loch Maree, Ariundle, Leth Bheinn and Glen Nant had lower air NH_3 concentrations (monthly values in the range 0-0.08 μg N m^{-3}) than Loch Lomond, Wood of Cree and Borrowdale (0.07-0.32 μg N m^{-3}) (Fig. 3). Air NH_3 measurements varied monthly with the highest measurements occurring during the late summer and the lowest measurements occurring during the winter. Overall, epiphytes at Loch Maree, Glen Nant and Ariundle received the lowest N deposition and those at Borrowdale the highest.

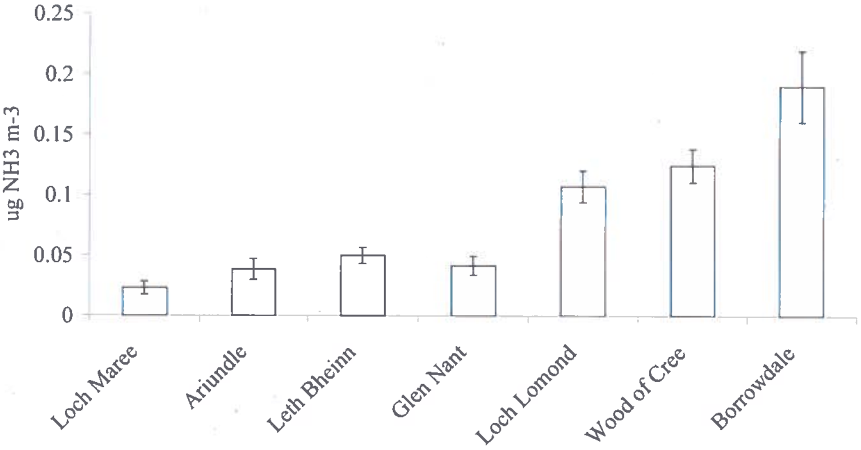

Figure 3: Mean atmospheric ammonia concentrations at the seven sites. Error bars show 1 standard errors of monthly values.

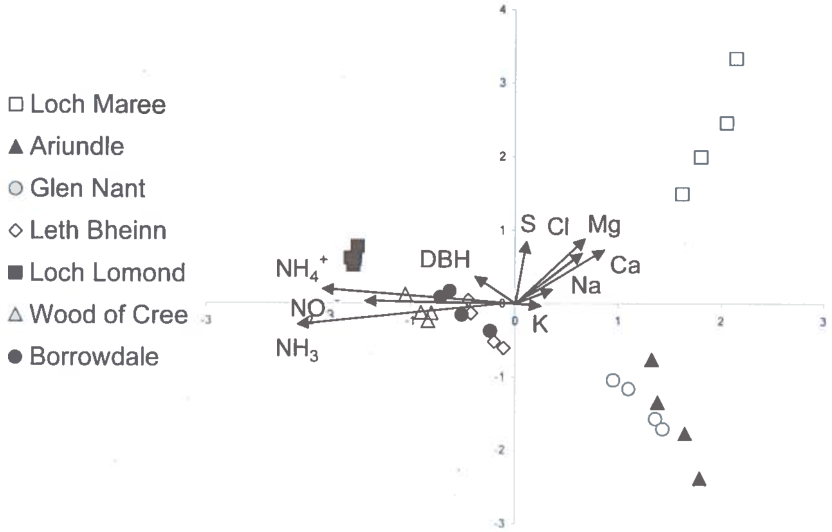

Figure 4: Site ordination diagram of survey results using Canonical Correspondence Analysis. The quadrat survey results at sites are considered in relation to environmental variables NH_4^+, NO_3^-, S, Cl^-, Na^+, Mg^{2+}, Ca^{2+} and K^+, weighted volume concentration in stemflow over 7 months; average NH_3 m^{-3} concentration in the air over the 7 months; DBH (diameter at breast height of tree). Mean position of north, south, east and west quadrats are plotted for each site.

The stemflow, wet deposition and ammonia concentration results were used in the analysis of the epiphyte survey results. In addition, total atmospheric N deposition to the sites were calculated. Initial estimates were provided from mapped wet deposition (based on long term precipitation records and volume weighted chemistry) and forest specific dry deposition according to Smith (2000). In addition, revised estimates were made by correcting the model estimates according to the locally measured wet deposition of N and NH_3 concentrations.

Epiphytic community

Initial analyses of the survey data have been carried out using Canonical Correspondence Analysis (CCA) (Ter Braak & Smilauer, 1988) which related the species to the following environmental variables: the weighted volume concentration in stemflow over 7 months of NH_4^+, NO_3^-, S, Cl^-, Na^+, Mg^{2+}, Ca^{2+} and K^+; average NH_3 concentration in the air over the 7 months; DBH (diameter at breast height). The first axis of the ordination diagram (Fig. 4) was negatively correlated with increasing nitrogen (NH_4^+ & NO_3^- in stem flow and air NH_3 concentration). The ordination diagrams split the sites into 2 groups. First, those occurring along the positive end of the first axis and at the low end of the nitrogen vectors; Ariundle, Loch Maree, Glen Nant, which were sites receiving low levels of nitrogen deposition. The second group of sites occurred at the negative end of the first axis and at the high end of the N vectors: Borrowdale, Loch Lomond, Wood of Cree, Leth Bheinn), which were sites receiving a high level of nitrogen deposition. The second axis is considered to represent the effects of exposure to the Altlantic as it separates Loch Maree from the other low N sites. The Loch Maree site is far more exposed and situated on the edge of large loch, while the other low N sites are more sheltered resulting in a difference in species composition. This is consistent with the observed differences in base cation deposition, which derives from marine sources.

The species ordination (Fig. 5) separated the species into 3 groups. First, those species associated with low N deposition and sheltered sites: *Plagiochila atlantica Plagiochila spinulosa, Frullania tamarisci, Parmelia laevigata* and *Platismatia glauca*. Second, those species associated with low N deposition, but able to grow in more exposed conditions: *Lobaria pulmonaria, Lobaria amplissima, Pertusaria hymenea* and *Parmelia glabratula subsp glabratula.* Third, those species able to grow in higher levels of N deposition: *Bryoria fuscescens, Usnea subfloridana, Hypnum cupressiforme, Hypnum mammillatum, Parmelia saxatilis, Chrysothrix candelaris* and *Hypogymnia physodes*.

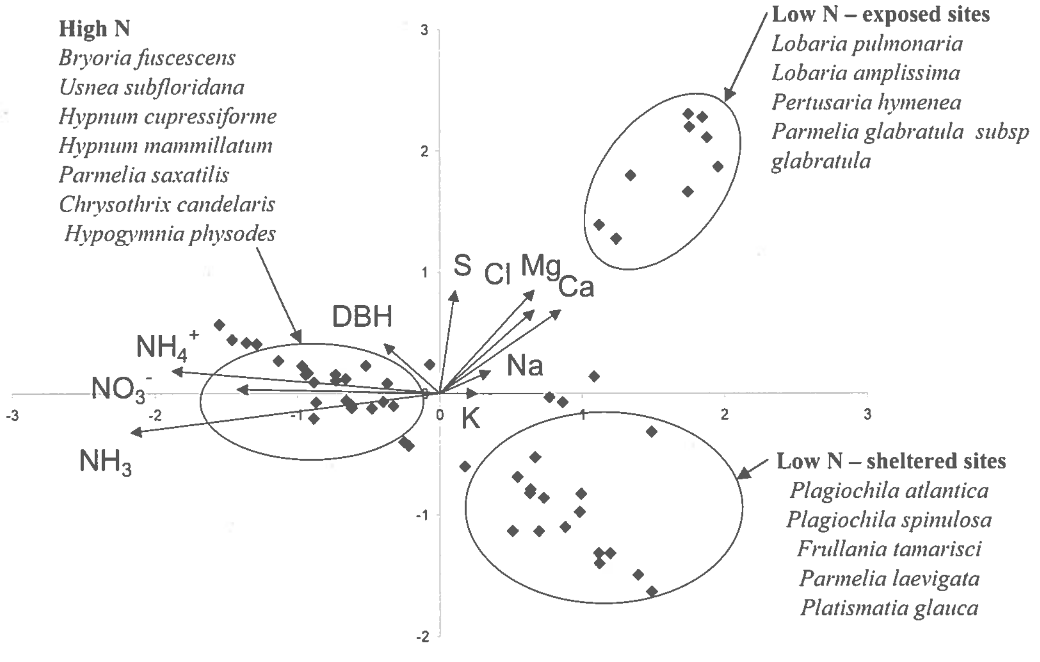

Figure 5: Species ordination diagram of survey results using Canonical Correspondence Analysis. The environmental variables are described in Figure 4.

Discussion

The results clearly show differences in the community composition of the epiphytic species at the different sites. These differences in community composition are consistent with differences in nitrogen deposition received at the sites. It is important to consider, however, whether the differences could also be due to climatic differences. The greatest climatic variable effecting epiphytes is rainfall, however all sites were chosen to receive at least 1400 mm of rain a year. There is a north-south temperature gradient, however, which may have some effect on species composition.

The sites clearly split into two groups: low N sites and high N sites. On the basis of national UK monitoring data and models, the total atmospheric input of nitrogen (NH_4^+ and NO_3^- in rain and dry deposition of NH_3^+, NO_2 and HNO_3) is mapped at spatial resolution of 5 km (Smith *et al.*, 2000; NEGTAP, 2001). These models suggest that the low N sites in this study (Loch Maree, Ariundle and Glen Nant) receive less than 18 kg N ha^{-1} y^{-1} (range 9.6 – 17.6 kg N ha^{-1} yr^{-1}) and the high N sites (Leth Bheinn, Loch Lomond, Wood of Cree and Borrowdale) receive more than 11 kg ha^{-1} yr^{-1} (range 11.2 to 53 kg N ha^{-1} yr^{-1}). The mean rainfall at the low N sites was 1843 mm and at the high N sites was 2073 mm, which were not significantly different. Preliminary refinement of these models based on site ammonia concentration and wet deposition measurements suggests that the distinction between the low- and high-N sites is actually slightly lower at around 15 kg N ha^{-1} yr^{-1}. While further analysis is continuing with these data, it becomes apparent that the best estimate of N deposition to the sites is not based directly on the local measurements. This is because deposition is temporally variable, and the measurements were only made in this study for 7 months. For the NH_3 concentrations, results were found to be temporally representative, and also only make a small contribution to the total N deposition for most of these sites. By contrast, there were substantial deviations from average precipitation at the sites for 2001, introducing uncertainty into the site-based wet deposition estimates. Further refinement of these estimates therefore needs to consider the temporal variability of wet deposition measured at these sites in relation to that observed at the long-term wet deposition monitoring sites.

While recognizing the uncertainties in the deposition estimates, the results here can be used to estimate a critical load for nitrogen deposition impacts on the epiphytic flora of Atlantic oak woods. Based on the distinction between sites from sites from the CCA (Fig. 4), and using both methods for estimating N deposition, suggests an empirical critical load for N deposition of 11-18 kg N ha^{-1} yr^{-1}. Key epiphyte species of Atlantic oakwoods that appear to be sensitive to N deposition include *Plagliochila atlantica, P. spinulosa, Lobaria pulmonaria* and *L. amplissima.*

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Critical limits for nutrient concentrations and ratios for forest trees – a comment

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Many approaches have been made to get a better understanding of adequate ranges for nutrient concentrations and ratios of woody plants in respect of growth, cold hardiness and drought resistance as well as susceptibility of plants to parasite attacks. To define critical nutrient limits, nutrient ratios are more useful than nutrient concentrations. The ratios not only reflect nutrient balance but are also less affected by growth dilution, concentrations of nonstructural carbohydrates and aging processes than nutrient concentrations. They are often derived from adequate concentration ranges by dividing the lower end of one nutrient by the higher end of the other nutrient and vice versa. This procedure is, however, unsuitable if the concentration ranges are already very large as it has been the case in a recent evaluation by ICP Forests (UNECE 2000). A starting point is certainly the mean of the concentration range, but the ends of the range have to be adjusted according to physiological criteria. The N/Mg ratio recommended by ICP Forest e.g. extends to values associated with symptoms of Mg deficiency as shown by Rodenkirchen (1998).

A large number of references on nutrient concentrations and ratios has been compiled by Van den Burg (1985) and Van den Burg (1990). A list of publications giving numbers on ratios is compiled in table 1. Summarising these publications on ratios and deriving ratios from the publications on concentration ranges, averages for the lower and upper end of normal/adequate ranges were calculated. They result in the recommended ranges given in table 2 (Altherr & Evers 1974; Altherr *et al.* 1974; Blok *et al.* 1972; Bonneau 1973; Bonneau 1986; Bonneau 1988; CERAFER 1971; Etter 1969; Fiedler & Höhne 1984; Foerst *et al.* 1987; Gussone 1964; Gussone 1978; Gussone *et al.* 1985; Höhne 1978; Hunger & Marschner 1987; Hüttl 1987; Hüttl 1988; Hüttl 1991; Ingestad 1979; Járó 1967; Landmann *et al.* 1987; Le Tacon & Millier 1970; Le Tacon & Toutain 1973; Linder 1990; Lyr *et al.* 1967; Kaupenjohann *et al.* 1987; Krauss *et al.* 1986; Materna 1965; Nihlgård 1986; Ovington 1956; Rehfuess 1983; Rodin & Bazilevich 1967; Silfverberg 1980; Silva Ferraz 1985; Stefan 1983; Stefan 1989; Strebel 1959; Strebel 1960; Summerer 1987; Tamm 1953; Van den Burg 1988; van den Driessche 1991; Wehrmann 1963; Weissen *et al.* 1988; Wenninger 1986; Wittich 1958; Zech *et al.* 1983; Zöttl 1964; Zöttl & Hüttl 1985; Zöttl & Hüttl 1986).

Tab. 1: Compilation of literature data for optimum/adequate nutrient ratios

Reference	tree species	N/P	N/K	N/Mg
Binns *et al.* 1983	broadleaved trees	10-20	2-4	
Bonneau 1988	various trees	10-15		<17.5
Strebel 1960	Picea abies (adult stands)	5.6-12.0		
Fiedler *et al.* 1973		8.0-8.3		7.2-8.6
Aldinger 1987		6.2-10		
Fiedler & Höhne 1987		7.1-7.7	1.7-1.9	
Hüttl 1991		6-12	1-3	8-30
Ingestad 1979	Picea abies (seedlings)	6.25	2	20

Tab. 2: Recommended adequate/normal nutrient ratios for Norway spruce and beech

		Recommended range (median values from lower and upper end)	Number of publications
Picea abies	N/P	7.0-12.0	37
	N/K	1.9-3.6	34
	N/Mg	10.3-20.0	31
Fagus sylvatica	N/P	10.0-17.1	14
	N/K	2.1-3.8	14
	N/Mg	10.4-21.5	9

Comparing the above recommended adequate nutrient ratios from literature and the nutrient ratios given by Stefan *et al.* (1997) and ICP Forests (UNECE 2000) as listed in table 3, respectively, there are remarkable differences. Using the first or the latter has severe consequences for the interpretation of the nutrient status of the European permanent observation plots. The evaluation on the basis of the nutrient ratios proposed by ICP Forests results in only 13 of 102 European permanent observation plots with nutrient imbalances (de Vries, pers. comm.). With the use of compiled and recommended literature data, there are 57 plots with nutrient imbalances (de Vries *et al.* 2003). It is unlikely that after 40 years of increased N deposition (Goulding & Blake 1993; Braun *et al.* 1996) only a few forest sites with nutrient imbalances can be found even in the highly polluted regions of Central Europe. Nutrient status and ratios changed drastically in Central Europe during the past decades (table 4).

Tab. 3: Nutrient ratios for Norway spruce and beech after ICP Forests (UNECE 2000)

		ICP Forests
Picea abies	N/P	6-17
	N/K	1.3-4.9
	N/Mg	8.0-28.3
Fagus sylvatica	N/P	10.6-25
	N/K	1.8-5.0
	N/Mg	12.0-25.0

Tab. 4: Comparison of historical with recent nutrient ratios

Species	Reference	region	number of plots		First evaluation		second evaluation	
					year	ratio	year	ratio
Fagus sylvatica	Duquesnay *et al.* 2000	Northeastern France	188	N/P	1969-71	16.2	1996-97	23.5
				N/K	1969-71	2.33	1996-97	2.80
				N/Mg	1969-71	14.03	1996-97	25.33
	Flückiger & Braun 1998	Switzerland	45	N/P	1984/85	18.1	1999	25.4
				N/K	1984/85	2.58	1999	3.66
				N/Mg	1984	16.3	1999	18.3
Picea abies	Hüttl 1990	Germany, Black Forest		N/K	1961	1.2-2.5	1983	2.0-6.8
				N/Mg	1975	8-13	1983	9-56
	Flückiger & Braun 1998	Switzerland	18	N/P	1984/85	13.5	1999	16.5
				N/K	1984/85	3.43	1999	3.71
				N/Mg	1984/85	17.1	1999	15.5

There is a number of studies showing that woody plants are significantly more susceptible to parasite attacks when nutrient ratios are not balanced:

Pine (Pinus sylvestris)	with N/K ratios >3.5 (ICP Forests adequate ratio for pine 1.2-4.9) showed distinct increased infestation by the fungal pathogen *Lophodermium pinastri* (Brüning 1964)
Oak leaves	with N/K 3.96 showed increased infestation by 17% with scales (*Eulecanium carni* und *E. rufulum*) compared to leaves with N/K 2.65 (Brüning & Uebel 1968). (ICP Forests adequate ratio for oak 1.5-5.0)
Apple trees	with N/K >2.5 showed significantly increased infestation by scarf (*Venturia inaequalis*) (Chaboussou 1972)
Beech	(adult stands) in permanent observation plots with N/K >3.7 showed significantly more dead branches caused by the pathogenic fungus *Nectria ditissima* (Flückiger *et al.* 1986)
Beech saplings	in a N fertilization experiment (pots) with N/P ≥17 showed significantly increased infestation by the leaf aphid *Phyllaphis fagi* (Flückiger & Braun 1994)
Beech	in a N fertilization experiment (afforestation) with N/K ≥4 showed increased shoot dieback caused by the pathogenic fungus *Apiognomonia errabunda* (Flückiger & Braun 1999)
Norway spruce	in a N fertilization experiment (afforestation) with N/K ≥4 showed increased infestation by the pathogenic fungi *Botrytis cinerea* and *Sclerophoma pithyophila* (Flückiger & Braun 1999).
Beech	in a N fertilization afforestation with N/P >15 showed lower concentrations of fungistatic phenolic compounds in leaves such as coumaric acid and hydroxyacetophenone (Tomova *et al.* 2002)

Maybe one of the most accurate method to determine nutrient imbalances is the DRIS system (Diagnosis and Recommendation Integrated System) that includes physiological, soil and environmental parameters (Beaufils 1971; Beaufils 1973; Sumner 1977a; Sumner 1977b; Sumner 1978; Timmer & Armstrong 1988).

In the context of setting empirical critical loads for N on the basis of limits for nutrient concentrations and ratios and the observed detrimental effects by parasites on woody plants with wide N/P, N/K and N/Mg ratios it is recommended to use adequate ranges of nutrient ratios which minimize secondary risks.

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Mire, bog and fen habitats

Differential effects of nitrate and ammonium enrichment on base-rich fen vegetation: preliminary results from Scragh Bog, Central Ireland

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1. Introduction

Base-rich fens are mire systems that are characterized by nutrient-poor, but base-rich conditions at circumneutral pH (Van Wirdum, 1991; Kooijman, 1993). Pristine base-rich fens are species-rich communities (Schaminée *et al.*, 1995) and small sedges and typical bryophytes called "brown mosses" are characteristic of these ecosystems (Kooijman, 1993).

The species composition of the moss layer of virtually all Dutch base-rich fens has drastically changed in recent decades. Most typical brown moss species, such as *Scorpidium scorpioides, S. cossonii* and *Scorpidium revolvens*, have strongly declined or disappeared from the Netherlands (Kooijman, 1993). Thus, many of these brown mosses are now red-list species (Verhoeven and Bobbink, 2001). Parallel to this, *Sphagnum* and *Polytrichum* spp. have rapidly become dominant in many Dutch base-rich fens (Kooijman 1993; Beltman *et al.*, 1995; Schaminée *et al.*, 1995). Many former base-rich fens have now turned into species-poor fens with an interstitial water pH of about 4.

It has been suggested that a combination of factors may cause the decline of brown mosses and the rapid increase of *Sphagnum* and *Polytrichum* in base-rich fens (Schaminée *et al.*, 1995 and references therein). Firstly, abandoned management of ditches and shallow drainage channels has led to reduced inflow of base-rich water into the fen vegetation. Secondly, mechanized mowing may have compressed the soil, thereby enhancing (acidic) rainwater influence at the mire surface. Thirdly, N and P enrichment has been found to enhance the observed changes in the vegetation of base-rich fens (Kooijman, 1993). In this respect, atmospheric deposition of nitrogen has been suggested to play an important role (Beltman *et al.*, 1995).

Few field studies have focused on the effects of increased nitrogen deposition on base-rich fens, when compared to other mire types such as raised bogs. Empirical critical loads for the effects of excess nitrogen deposition on base-rich fens have been set at 15-35 kg ha^{-1} yr^{-1} (Bobbink *et al.* 2003). These authors mention increase of tall graminoids and a general decrease in diversity as indicators of exceedance of these loads. However, it is insufficiently known to what extent increased nitrogen deposition has caused the drastic shifts that have been observed in the moss layer of Dutch base-rich fens.

In particular, studies that distinguish between the effects of ammonium and nitrate enrichment are rare. Yet, it is interesting to make this distinction. Total nitrogen deposition is relatively high in the Netherlands, and at least 70% of it consists of ammonium. Ammonium deposition was especially high (30 kg NH_4-N ha^{-1} yr^{-1}, locally even higher) in the 1980s and early 1990s (Bobbink *et al.*, 1992; Erisman, 1992). Interestingly, this was the period when reports of rapid *Sphagnum* and *Polytrichum* expansion became abundant (cf. Kooijman, 1993; Beltman *et al.*, 1995).

High nitrification rates can be expected in base-rich fens at circumneutral pH, and declining brown moss species may be sensitive to increased atmospheric ammonium input. Short-term greenhouse experiments, in which realistic nitrogen concentrations were applied, have shown that growth of the threatened brown moss *Scorpidium scorpioides* is reduced when grown on ammonium or ammonium nitrate instead of nitrate only. On the other hand, the expanding species *Sphagnum squarrosum* and *Polytrichum commune* are indifferent to nitrogen form (Paulissen *et al.*, unpublished).

To study the impact of increased nitrogen deposition on base-rich fen mosses *in situ*, we set up a nitrogen addition experiment in a fen in central Ireland. This is a relatively unpolluted area with an estimated background deposition of 7-10 kg N ha^{-1} yr^{-1} (cf. Aherne and Farrell, 2002). We added 50 kg N ha^{-1} yr^{-1} as ammonium or nitrate in artificial rain.

This paper presents the preliminary results after two years of nitrogen additions (covering three growing seasons). Both physiological and growth response data are presented. Given the scarcity of long-term effect studies, we intend to continue the field experiment described here for 2-3 more years.

2. Materials and Methods

2.1. Study site

Scragh Bog is situated 6 km north of Mullingar, county Westmeath, Ireland (53° 35'N 7° 22'W). It is a base-rich valley fen (EUNIS D4.1) surrounded by eskers, glacial meltwater deposits which are intensively used as pastureland (O'Connell, 1980, 1981). The fen is supposedly fed by springs and drained by a single stream at its north end. The groundwater that feeds the fen is calcareous, with circumneutral pH. Occasionally, during the winter season, the fen is flooded, with water levels up to 50 cm above the soil surface. The fen is partially fed by runoff and shallow groundwater originating from the surrounding pastureland, which has created a marginal buffer zone containing relatively eutraphentic and high-productive species. However, the major (central) part of the fen is characterized by a typical base-rich fen vegetation and oligotrophic conditions, with very low water and soil concentrations of N and P (Beltman *et al.*, 2002).

2.2. Experimental set-up

Starting in June 2000, we carried out a fertilization experiment in the central, pristine, part of the fen. The experiment had a randomized block design and consisted of 40 x 70 cm plots within natural bryophyte gradients of *Scorpidium revolvens* to *Sphagnum contortum* and *Sphagnum subnitens*. The plots were fertilized twice per year (in spring and in summer or early autumn) by applying an artificial rain solution. The treatments were: control (water only), nitrate (50 kg N ha^{-1} yr^{-1} as $NaNO_3$) and ammonium (50 kg N ha^{-1} yr^{-1} as NH_4Cl), with 5 replicates per treatment. We observed no detrimental effects of the concentrated nutrient solutions (0.5 l per plot) on the vegetation.

The cover of individual bryophyte species was measured in June 2000 (initial situation) and August 2002, using the point-intercept method (Jonasson, 1988). This was done using a 30 x 60 cm frame with a permanent grid of 189 points. The frame was put in the same position in 2000 and 2002 and the level position of the grid frame was checked before the measurements started.

In August 2002, *Scorpidium revolvens* and *Sphagnum contortum* shoots were collected from every plot and transported to the lab in a cool box. The next day, phosphomonoesterase (PMEase) activity was measured following Turner *et al.* (2001), with slight modifications to the method. The substrate (*p*-nitrophenyl phosphate) concentration in the assay was 100 μmol l^{-1} and the assay pH was 5.5.

Data were statistically analyzed using analysis of variance (SPSS 8.0 for Windows) in combination with Tukey post-hoc tests ($\alpha = 0.05$). A two-way anova was performed on the PMEase activity data, which yielded a significant interaction term species x nitrogen form. Therefore, separate one-way anovas were performed for each species with nitrogen form as the sole factor. With respect to the percentage cover data, one-way anovas were run for each of the four bryophyte categories (Figure 2).

3. Results

After two years of artificial rain application, phosphomonoesterase (PMEase) activity of *Scorpidium revolvens* and *Sphagnum contortum* was determined (Figure 1). PMEase is an enzyme that enables plants to take up organically bound phosphates that are otherwise unavailable. This enables the plant to increase P uptake under conditions of high nitrogen deposition. PMEase activity of *Scorpidium revolvens* was significantly increased by nitrate, but not by ammonium addition. *Sphagnum contortum* showed the opposite pattern: ammonium enrichment significantly stimulated PMEase activity, while nitrate addition led to intermediate enzyme activity (Figure 1). Results from a factorial N and P addition trial, situated next to the experiment described here, suggested that the growth of *Scorpidium* was P limited (data not shown). Our results suggest that *Scorpidium revolvens* is able to avoid P limitation under conditions of increased nitrate deposition, but not under increased ammonium deposition. On the other hand, *Sphagnum contortum* can cope better with increased ammonium deposition, while PMEase activity is also (but less clearly) stimulated by increased nitrate deposition.

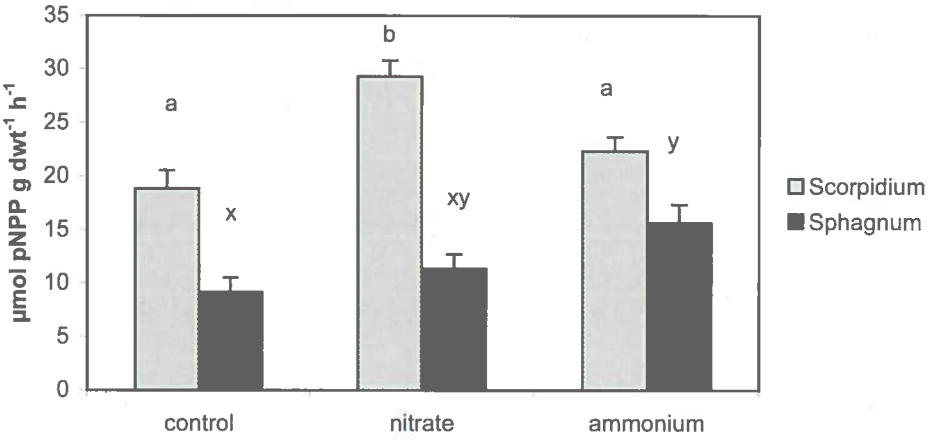

Figure 1. PMEase activity (mean + SEM, n=5) of *Scorpidium revolvens* and *Sphagnum contortum*, after two years of nitrogen addition. Letters indicate significant differences ($p<0.05$).

Two years of nitrate or ammonium addition had not led to significant shifts in percentage cover in the bryophyte layer (Figure 2), although a tendency towards better growth of *Sphagna* (*Sphagnum contortum* and *S. subnitens*) under increased ammonium addition was observed. However, prolonged exposure of *Sphagna* to high nitrogen deposition may eventually reduce their growth (Gunnarsson and Rydin, 2000). *Scorpidium* showed a decrease in cover in all treatments, which we attribute to the normal (but relatively slow) succession pattern towards a *Sphagnum*-dominated poor fen. Figure 2 suggests that *Scorpidium* and other non-*Sphagnum* bryophytes (mostly brown moss species) were, already on a short term, negatively affected by increased ammonium deposition.

Brown mosses like *Scorpidium revolvens* are typical bryophytes of base-rich fens and as such are optimally adapted to circumneutral pH. Under such conditions, nitrification rates are likely to be high and ambient ammonium concentrations will be low. The results presented here suggest that brown mosses are sensitive to ammonium deposition in the range of Dutch levels. On the other hand, *Sphagna* (even relatively basophilic species like *Sphagnum contortum* and *S. subnitens*) usually occur in more acidic habitats where nitrate/ammonium ratios are likely to be low. Therefore, they may be more tolerant to ammonium, even at very high deposition levels, like in the Netherlands. The PMEase data (Figure 1) suggest that *Sphagnum contortum* even profits from increased ammonium deposition, at least on the shorter term.

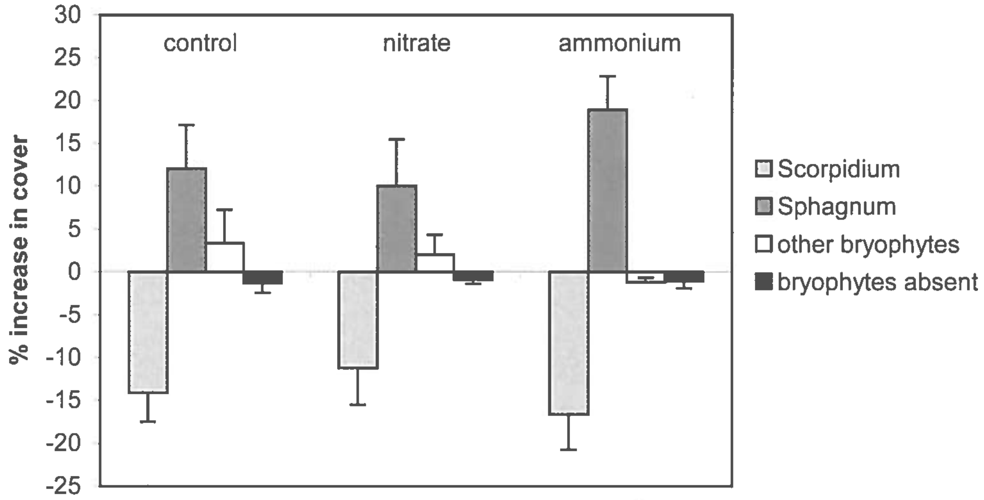

Figure 2. Change in percentage cover (means + SEM, n=5) of bryophytes (including points where bryophytes are absent) over a two-year period of nitrogen addition. "*Scorpidium*" includes *Scorpidium revolvens* and *S. cossonii*; "*Sphagnum*" includes *Sphagnum contortum* and *S. subnitens*. The "other bryophytes" group is dominated by non-*Scorpidium* brown mosses.

4. Conclusions

The first results of a long-term nitrogen addition experiment in a base-rich fen in Ireland clearly indicate that the typical species-rich brown moss vegetation is affected by the form of nitrogen enrichment. Nitrate application (50 kg N ha^{-1} yr^{-1}) significantly stimulated PMEase activity in the typical brown moss *Scorpidium revolvens*, whereas ammonium addition (50 kg N ha^{-1} yr^{-1}) did the same in *Sphagnum contortum* within a two-year experimental period. This suggests that, in the longer term, *Sphagnum* will profit from high ammonium deposition (comparable to Dutch levels), while brown mosses are negatively affected by it. The first results of the changes in species cover suggest an increase in *Sphagnum* and a decline in brown mosses after two years of ammonium addition, but we will continue this experiment in the near future to validate this preliminary conclusion. Finally, our results suggest that high ammonium deposition may, at least partly, cause the observed shifts in the bryophyte layer of Dutch base-rich fens.

5. Acknowledgements

We thank Dúchas (Irish Natural Heritage Service) for permission to carry out the field experiments in Scragh Bog. The PMEase assay was carried out at the Microbiology Department, National University of Ireland, Galway. A part of the fieldwork was funded by the Schure-Beijerinck-Popping Fonds (Royal Netherlands Academy of Arts and Sciences).

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Coastal habitats

Field survey linking soil, vegetation and groundwater parameters in UK sand dunes with atmospheric nitrogen deposition.

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Eleven sand dune sites around England and Wales were selected for a study based on the modelled atmospheric N deposition to these sites. They were chosen to represent a range of N inputs on the western, southern and eastern coasts of England and Wales, on both calcareous and acidic sites.

At each site, diffusion tubes were installed to measure dry deposition of NO_2 and NH_3 as these were identified as the least reliable component of the modelled N deposition data. The field survey was carried out in April/May 2002 and sample points were taken in mobile, semi-fixed, fixed and established dune grasslands and in dune slacks (where they occurred) at each site. Soil parameters measured included pH, Loss On Ignition, depth of organic layer, bulk density, available N (by single KCl extraction), total N, C:N ratio, available P (by Olsen-P extraction). Vegetation parameters included species composition, species diversity, biomass, sward height and tissue N. Groundwater samples were analysed for NH_4, NO_3, Dissolved organic N (DON), pH, and alkalinity.

The results from the NO_2 diffusion tubes showed a strong correlation with the modelled data. However, the NH_3 tubes showed a much poorer correlation with the modelled data and this is of crucial importance in interpretation of results as NH_3 is the dominant source of atmospheric N at these sites. Although the diffusion tubes were only exposed for 2 x 1 month periods, there was a much greater correlation of moss tissue N with the diffusion tube data than with the modelled NH_3 data. Therefore total N deposition to each site was calculated using the modelled wet deposition and the measured dry deposition using deposition velocities of 1.13 mm/s for NO_2 and 10 mm/s for NH_3.

As a result of strong heterogeneity of the vegetation between sites and a desire to separate effects in different successional and hydrological habitats, the data were subdivided into 3 classes for analysis: Mobile and Semi-fixed dunes; Fixed dunes and Established dune grasslands; and Dune Slacks. The measured parameters were meaned at the site level for each habitat and correlated with calculated N deposition. Significant correlations are presented below.

Mobile and semi-fixed dunes

Biomass and sward height (Figures 1 and 2) both showed positive correlations with N deposition, this has implications for accelerated stabilisation and soil development, with the potential for reduced area of early successional habitats.

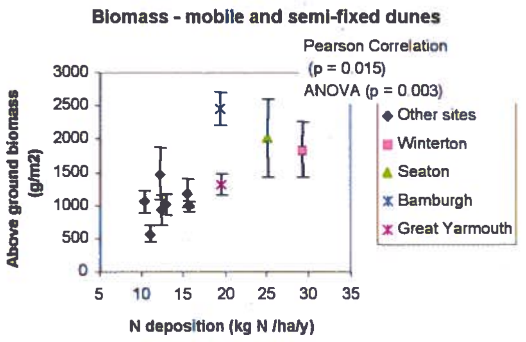


Figure 1. Above ground biomass plotted against N deposition for each site. Shifting dunes.

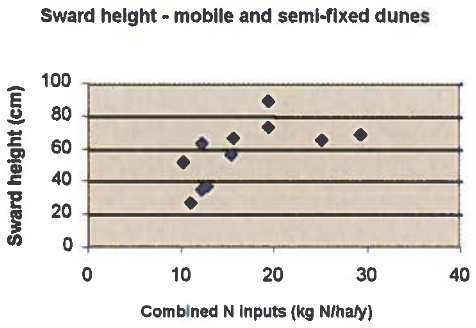


Figure 2. Sward height plotted against N deposition for each site. Shifting dunes.

Fixed dunes and established dune grassland

There was a negative correlation of N deposition with available N in the soil (Figure 3) and a corresponding positive correlation with soil C:N ratio (Figure 4). This relationship is opposite to that expected from established soils and was not due to a shift towards woody species at the high N sites. However, one possibility is that elevated N inputs may be slowing litter decomposition by inhibiting fungal activity and thus affecting N availability. This issue clearly needs further study. There was also a negative correlation with species diversity (Figure 5) in this habitat.

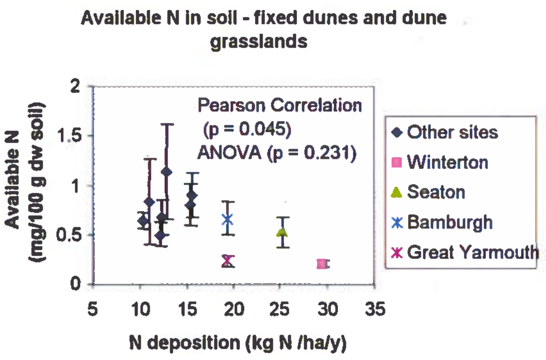


Figure 3. Available N in soil plotted against N deposition for each site. Stable dune grasslands.

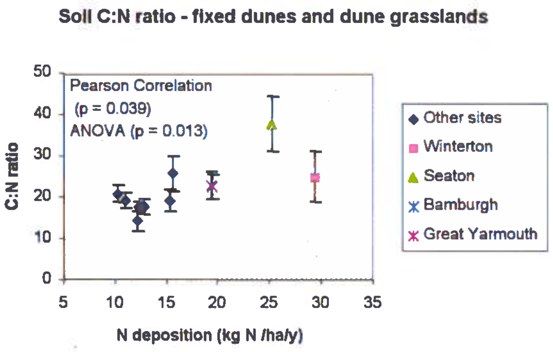


Figure 4. Soil C:N ratio plotted against N deposition for each site. Stable dune grasslands.

Dune slacks

There were no significant correlations of the measured parameters in the dune slacks. However, this habitat was poorly represented at the survey sites and tended to occur at the locations with low N inputs. Therefore, insufficient conclusions could be drawn as to the critical load for this habitat.

Groundwater

A positive relationship with dissolved organic N (DON) and atmospheric N inputs was obtained (Figure 6), and DON comprised 50 – 90 % of the total N. There was no relationship of NO_3 in the groundwater with N inputs.

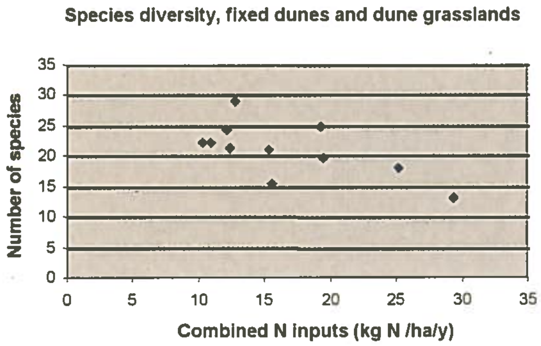


Figure 5. Species diversity plotted against N deposition for each site. Stable dune grasslands.

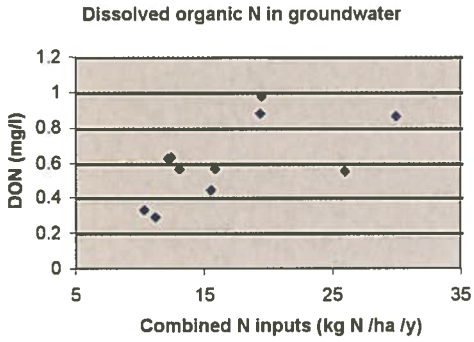
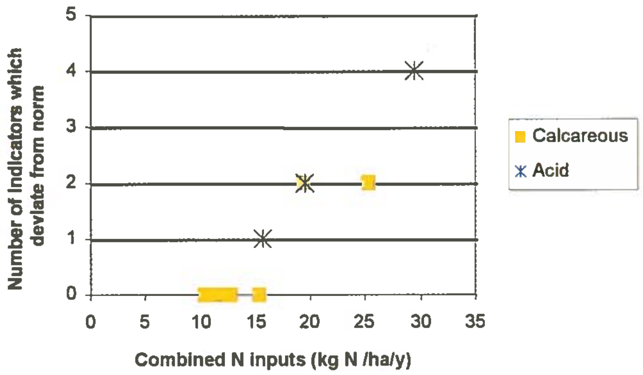


Figure 6. Dissolved organic N (DON) in groundwater plotted against N deposition for each site.

Overview

Where relationships were obtained between N inputs and the measured parameters, five sites could be identified as lying outside the main spread of data points for one or more of the parameters. If these are regarded as indicating exceedance of a critical load, and the number of exceeded parameters at a site is plotted against N inputs (Figure 7), we see that effects start to become apparent within the suggested CL range of 10 – 20 kg N for these habitats. It should be noted that the calculated deposition numbers for each site are likely to be overestimates, and therefore in reality, effects may occur at lower N deposition values. The results from this field survey suggest that the proposed CL range is suitable for UK stable dune grasslands, and can be extended to shifting dunes on the basis of the results above. However, we do not have enough information to suggest a critical load for dune slacks.

Figure 7. Plot of the number of 'CL exceeded' parameters against N deposition for each site. Predominantly acid and calcareous sites are differentiated.



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This work was conducted as part of a wider study on Critical Loads in sand dunes funded by the English and Welsh conservation agencies (English Nature and Countryside Council for Wales).

Biomonitoring

Plant ecological approaches to monitor the effects of N-deposition in the field

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Objective

Empirical critical loads for nitrogen have been established by the UNECE for different vegetation types and this system has now been refined using the EUNIS classification to be able to map in greater detail the regions in Europe where these are exceeded. However, the scientific background for establishing cause-effect relationships and testing the validity of critical loads needs to be strengthened and methods need to be developed to address the still ongoing adverse effects of nitrogen deposition and associated eutrophication in the field. Here we present three issues relating to nutrient gradient studies and information from comparative plant ecological analyses, which may serve in the future to develop practical monitoring approaches and eutrophication indicators on local, regional and national scales.

1. N-concentrations in plants rise with increasing N deposition

The study of leaf nitrogen concentrations along "fertility" gradients in the lee of large livestock buildings reveals significant effects of the deposition of NH_x (ammonia and ammonium). Within the same species, a steep decline in N-contents downwind from the emission source can be found, which may be used to indicate eutrophication effects on local scales (Fig. 1). Apart from N-contents, nutrient relations (e.g. N:P ratios) may be applied to address nutrient imbalances in plants. Other useful approaches identifying immission and deposition gradients relate to bark-biomonitoring (Spangenberg et al. 2002) and soil analyses in the lee of livestock farms (Formosa & Singh 2002).

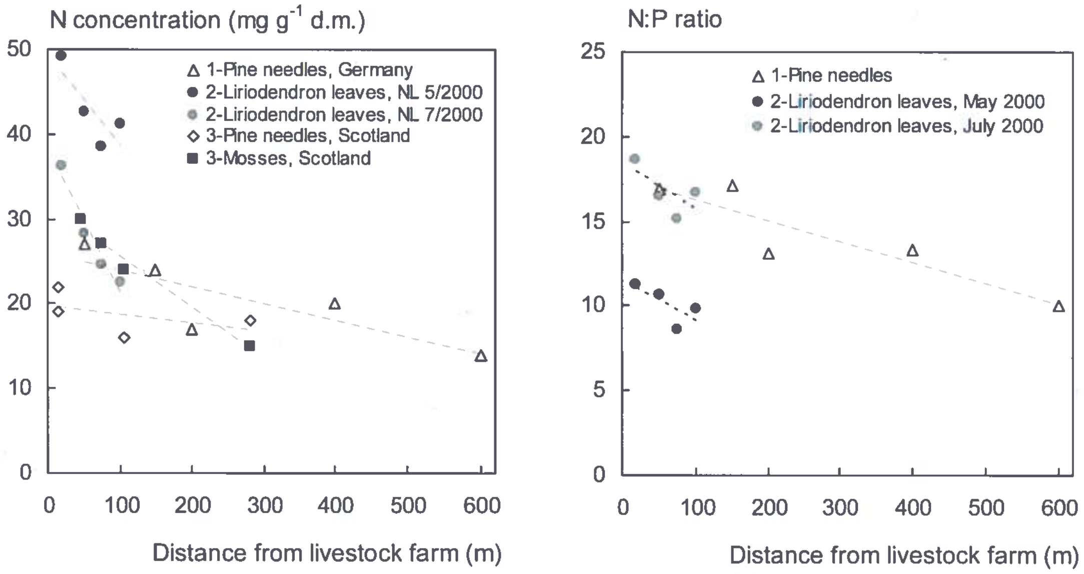

Figure 1: Examples of gradient studies in GB, NL and D using plants to mirror the effect of nitrogen deposition on nearby ecosystems. Response parameters are nitrogen concentrations (left) and N:P ratios (right) in various plant species cited from the literature; [1]- Kaupenjohann et al. (1989), [2]- van Dijk et al. (2001) and [3]- Pitcairn et al. (1998). Livestock sources in the three studies emitted about 5 Mg a^{-1} of ammonia. N:P ratios of >16 in soils and plants are regarded as critical with respect to deficits in P-nutrition.

N-analyses of plant material in regional (Mohr 1999; van der Eerden et al. 2000) and national programs (e.g. ICP Forests) have found wide application in recent years. However, edaphic and climatic factors have strong effects on the outcome of passive biomonitoring studies and it is not fully clear which N-levels and nutrient ratios may be tolerated to safeguard the health and functioning of ecosystems. Comparative plant ecological studies may be used to answer these questions as illustrated below.

2. N-concentrations in plants reflect the nutrient availability in the habitat

On average, plant species from fertile habitats have higher nutrient contents than species from infertile sites, which is indicated by a firm relationship between plant and soil nutrients. This is also confirmed by the positive relationship between N-contents in plant leaves of species and the Ellenberg ecological indicator values for nitrogen of these species (Fig. 2). Performing N analyses of plants with low and high Ellenberg N-values may thus in principle serve to identify the severity of eutrophication effects. If nitrogen contents exceed the *normal* range in a species with a low N-indicator value eutrophication is likely, whereas N-contents in the lower range found in species with high N-indicator values confirm that eutrophication is not yet pronounced. Ellenberg ecological indicator values have been used extensively in the past studying historical vegetation relevés to determine whether eutrophication has affected plant communities and whether mean (median) indicator values may be used to monitor environmental change throughout Europe (Diekmann et al. 1989, Bürger-Arndt 1994 and Hill et al. 2000). A similar approach has been performed on a local scale (fertility gradient) by Pitcairn et al. (2002). However, vegetation shifts lag years behind the actual effects of nitrogen deposition, while nutrient analyses may respond faster to nitrogen inputs. Like N-contents, N:P ratios may also be useful to monitor the effects of eutrophication on nutrient imbalances in vegetation.

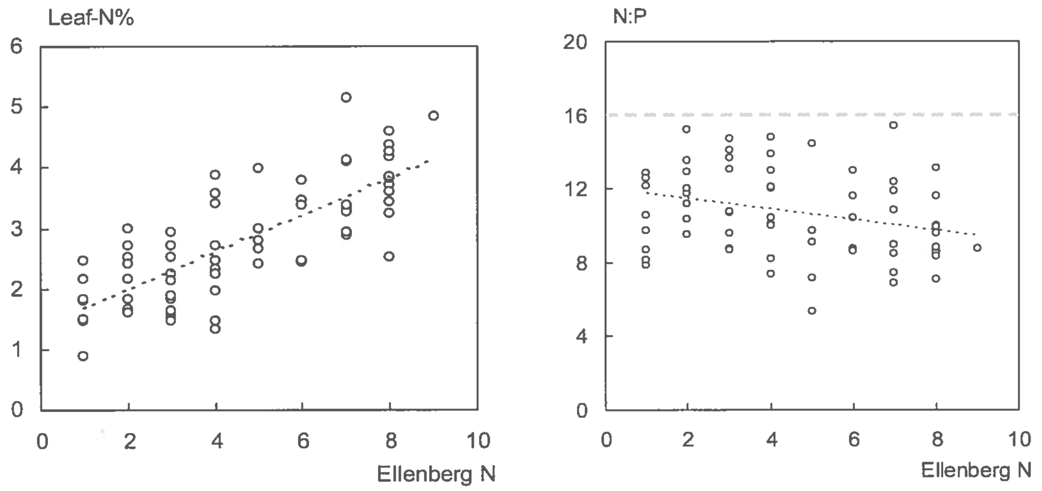

Figure 2: Relationship between Ellenberg-N-indicator values and average foliar N contents (left) and N:P ratios (right) in 78 European wild herbaceous species collected in the field. Data on N and P-concentrations from UCPE, University of Sheffield (internet) and indicator values after Ellenberg et al. (1992). Native nitrophilous species have N-contents of >2.5% N on the average, while plants from infertile soils have lower N-contents. Nitrophilous species tend to have lower N:P ratios than species adapted to infertile habitats. N:P values of all species remain below the critical threshold of 16 (dotted grey line), so it may be concluded that plant collections were made in N- rather than P-limited ecosystems.

3. Biomass production of plants reflects the nutrient status of the habitat

The entire physiology of plants is adapted to the specific habitat of plant species with fast growing species on the fertile (productive) sites and slow growing species on the infertile (unproductive) sites (Chapin 1980). This general notion is confirmed by the relationship between relative growth rates (RGR) of species and their N indicator values (Fig. 3), latter of which have been described by Schaffers & Sýkorá (2000) to be identical to "productivity values". The higher biomass production of plants due to N-deposition may thus be used as an eutrophication indicator. However, methods to study and monitor these effects should rely on active approaches rather than harvesting passive indicators grown on the natural substrates.

It is still matter of debate whether increased productivity of semi-natural ecosystems will automatically result in a lower biodiversity. Nevertheless, wide scale eutrophication has undoubtedly resulted in changes - not always in an impoverishment - of the regional floras with the spread of opportunistic, fast and tall growing nitrophytic ruderal and competitive plant species (R and C-strategists *sensu* Grime). However, with regard to forest species the prediction of the response of species to atmospheric nitrogen deposition using plant functional types and life history traits alone proved to be problematical (Diekman & Falkengren-Grerup 2002).

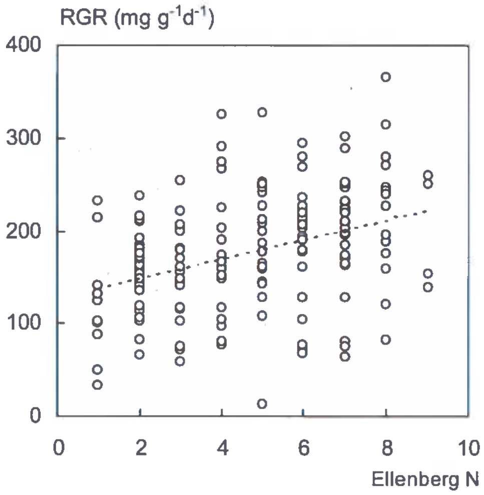

Figure 3: Relationship between Ellenberg N-indicator values and relative growth rates (RGR) of 203 European species reported in the literature. Data on RGR from laboratory and field measurements collated in own database and indicator values cited from Ellenberg et al. (1992). Nitrophilous species tend to have higher innate growth rates on the average, while slow growers are associated with infertile unproductive habitats.

Much information is available from ecosystem research on production and mineralization rates and this knowledge may be used to derive and refine critical loads. However, no general information exits on ecological stabilities of natural and semi-natural vegetation and how much nitrogen may be bound and cycled in the various vegetation types on the long term. While relative growth rates and N- and P- contents in forest species are generally lower than in ruderal species (see Fig. 4), these variables do not differ much between the groups of fens and bogs and heaths and grasslands. The lower growth rates and nutrient contents of latter groups confirm that N-deposition should be kept lower in these vegetations to avoid adverse effects on species composition and ecological services. Apart from differences in plant growth rates and storage of deposited N the variation in canopy area (LAI) between vegetation types accounts for differences in amounts of gaseous and particular NH_x scavenged. The performance of various studies has revealed that deposition velocities are highest in coniferous vegetation and lowest in grasslands. Other studies have confirmed that compensation points vary much in different vegetation types according to the average and actual nutrition status of an ecosystem, affecting the bi-directional fluxes of NH_x (uptake and emission).

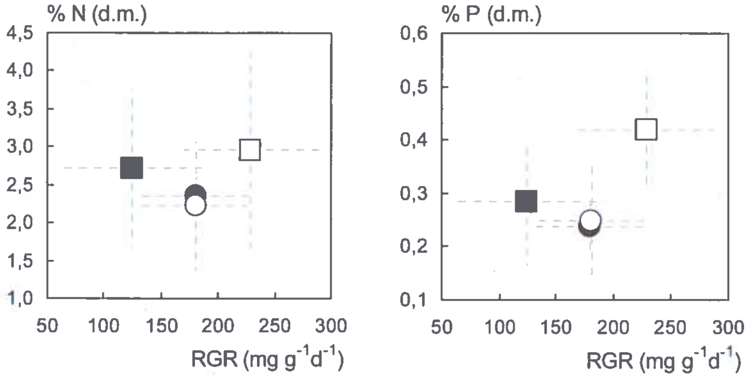

Figure 4: Relative growth rates and average concentrations of N (left) and P (right) in plant species of four ecological groups: vegetation of fens and bogs (●), herbaceous vegetation of disturbed sites (□), anthropo-zoogenous heaths and grasslands (○) and deciduous forests (■). Error bars indicate standard deviations of RGR, %N and %P. Data after Ellenberg sociological groups no. 1, 3, 5 and 8 and data on RGR, %N and %P from own data collection. Data availability ranged from 4 cases (fen and bog species) to 68 per group (ruderals).

Conclusions and suggestions

Ecosystem and ecological impact studies contribute to our understanding of how much nutrient deposition may be tolerated by different types of vegetation. Effect-related studies addressing the adverse effects of nitrogen deposition are still scarce to date; while a dense network on bulk sampling is operated throughout Europe (e.g. on ICP Forests level II sites). Latter method gives insight into the wet deposition loads, but it fails to explain how much nitrogen is actually scavenged by the vegetation as deposition velocities and compensation points greatly vary with the plant as the receptor. Applying plant ecological studies on wider scales than before may thus contribute to effect-related, receptor-oriented monitoring of the adverse effects of eutrophying compounds. We suggest using standardised experiments in pilot studies within European programs, e.g. ICP Vegetation and in the vicinity of emission sources using the response parameters nutrient concentrations, nutrient relations and biomass production. Further studies should also incorporate physiological indicators (e.g. those mentioned by Soares et al. 1995), morphological parameters as well as changes in competitive abilities, latter of which are important with respect to eutrophication mediated vegetation shifts.

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Modelling

The BERN Model: Bioindication for Ecosystem Regeneration within Natural conditions

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1. Introduction

At the last Critical Loads-Conference, which took place in Copenhagen in 1999, one point of critique was that, in the most cases, the Critical Load for air pollution was defined through the measurement of the soil chemistry parameters. The challenge was to include, in future models, more ecological indicators when exhibiting the environmental Cause-Effect Relationship and the determination of the Critical Load.

In order to determine ecosystematic connections, the BERN model was developed on the basis of empirical compilations, which took place within a region of Germany.

2. Background

The empirically determined reactions of plants and soil organisms to nutrient inputs (nitrogen eutrophication) and/or nutrient losses (leaching of base cations as a consequence of acidification) can only be interpreted reasonably where one determines the multiple correlations between carbon (C) to nitrogen (N) to base cations (Bc: sum of calcium [Ca], potassium [K] and magnesium [Mg]) considering water supply and temperature. The influence of phosphorus had been neglected in our model because the interaction with the plant composition could not be verified with evidence.

Both the "net primary productivity" of the plants and the "decomposing productivity" of the soil organisms may be an empirical measure and evaluation for the harmonious material and energy equilibrium. An anthropogenic dependent change in the nutrient household occurs as a result of exceedances of the Steady state-Critical Loads. Indicators of this change are manifest within the productivity of plants and soil organisms and are also characterized by the changes of the vegetation structure and conversion of humus structure. The Ellenberg (1991) system of "indicator value" for certain species represents values for the ecological optimum of the plant species. However, the system does not provide information on ecological niche widths of the species. Also, within this system, the mosses and lichens are missing as important indicators.

Plant communities exhibit the highest and most exact information content. The plant community structure is characterized not only through the presence or absence of plant species, but also through the abundance of the respective populations and the steadiness of plant distribution over the habitat and the degree of coverage over the soil surface. From community structure, one can draw more exact information about the soil parameters than from the average of the optimum values of the plant species based on Ellenberg (1991). On the other hand the Ellenberg´s optimum values were of great worth for the validation of our database by comparison.

In the presence of a harmonious equilibrium (of nutrients, water, and energy) a so-called natural plant community spontaneously settles in. Disharmonious nutrient conditions, caused by soil-chemical processes in reaction to anthropogenic eutrophication and/or acidification, result in the reduction of the vitality and the ecosystematic functional efficiency. Remaining behind after the extinction of the sensitive species are those species with very large ecological niche widths. Later, only "few-species-compositions" immigrate to the competition-free areas, which can no longer build into association structures. However, where nutrient input (e.g., with simultaneous nitrogen eutrophication and inputs of basic cations from flying ash from coal combustion) achieves a new harmonious nutrient equilibrium at a higher level, there will still be the death of the primarily existing dominant plant species, but many other plant species will migrate in, which altogether can develop a new natural ("potentially natural") plant community typical for the habitat. The same occurrence has been known to occur with the simultaneous acidification (with sulfur) and nitrogen deprivation.

The natural plant communities in the balanced habitats can be assigned directly to the scaled harmonious equilibrium classes. When analyzing the current condition of forest sites, one must use the bush and ground vegetation as an indicator for the condition of the site rather than the trees themselves. However, the dominant bush or ground species are also contained in the common community names.

3. Model concept

The conception of the model contains 3 stages:

- The determination of the primary-natural habitat parameters based on a) (quite) non changeable regular properties and b) easily changeable regular properties / allocation of typical vegetation communities to the classified regular habitat types;
- *The determination of the current conditions of the easily changeable soil parameters and its comparison with the original primary natural parameters of the regular habitat type;*
- The determination of the regeneration target and regeneration potential.

3.1. Habitat classification based on the regular site properties and allocation of the typical plant communities

The north German lowlands were chosen as the area of investigation for the purpose of a calibration and testing of the model.

The site classification is based on a combination of typified non changeable regular site properties. This combination includes:

- type of hydromorphy (degree of water saturation);
- type of climatic region;
- altitude;
- relief type;
- exposition type (degree of sun exposure);
- soil type group.

The area under investigation includes the habitat types of both the planar suboceanic and subcontinental regions. The soil types, which are present in the north German lowlands, were classified according to 33 groups of soil types. The forests, extensively used grassland, pastures and heaths including bogs and wet heaths were examined. For this area, 4840 vegetation and site observation relevees (OBERDORFER 1979; PASSARGE 1964; PASSARGE U. HOFMANN 1968; SCHLUTOW 1990-2002; SUCCOW U. JOOSTEN 2001) were evaluated.

The 130 plant communities recorded in Northern Germany can be allocated to regular site types.

For all regular site types, the typical natural (woodland) and semi-natural (grassland/fens/heath) vegetation communities (with their wild-spontaneous species endowment) are put into a database. To these vegetation communities have been assigned the regularly occurring dominant and character plant species. The following definitions apply:
Plant Community: A regularly arising combination of plant species with a regular structure at a habitat site. This consists of dominant species, character species and unstable species. The size of the habitat of a plant community is determined on the basis of the homogeneity (=steadiness) of the distribution of the dominance species and character species.
Dominance species: A species that covers more than 15 percent of the habitat's soil and appears with a steadiness (= uniformity of their distribution over the entire habitat of the community) of over 80 percent.
Character species: A species with a degree of soil coverage below 15 percent and with a steadiness exceeding 60 percent.
For all plant species retained in the database, the ecological niche widths are indicated in regard to the easily changeable site properties. These include the parameters Soil moisture, C/N-ratio and Base saturation.

If one understands the quantity of all parameter ranges that are suitable for a plant species as a soft, indistinct quantity (Fuzzy quantity), then any habitat parameter value is quantified regarding the species′ fitting as a figure between 0 and 1. This figure characterizes the degree of suitability between the respective habitat parameter value and the union of the possible habitats of a plant community. It also represents the quality of a habitat for the vitality of a certain plant species. It can be also interpreted as the possibility that a respective plant species lives in the habitat.

A habitat is defined by the regular site type and by the three easily changeable parameters. Hence, it can be manifested by a three-dimensional parameter vector containing the dimensions 1) base saturation, 2) C/N relationship and 3) moisture in a respective room of one regular site type.

The suitability of habitat x with a plant community is, therefore, a function of the definition quantity of R^3 and the value sample R. Because the possibility of a species living in a habitat is a non-measurable abstract value, one must agree upon the description of the suitability function. The Gaußian normal distribution commonly used for a representation of a productivity curve (in connection with the development of a habitat factor) has been adapted as follows:

$$\mu_a(x) = \max\left(1{,}1 \cdot e^{-\frac{(x-m_s)^2}{\left(\frac{2}{3}b_s\right)^2}} - 0{,}1;\, 0\right)$$

where
$\mu_a(x)$= suitability degree of a value x for a plant species
m_s = optimum position of a plant species on a given scale of a habitat parameter s
b_s = distance between the ecological optimum and pessimum of a plant species on a given scale of a habitat parameter s

Regarding the example of base saturation, figure 1 illustrates the suitability function of species (small blue and green lines) for a parameter analog to the suitability function. For each plant species, three fuzzy quantities have been established – each one respectively for one site parameter. The quantity of all suitable habitat values for a plant species results from the intercept quantity of these three fuzzy quantities. This means that the base saturation AND the C/N-ratio AND the moisture must fit this plant species.

The next step is, consequently, to determine the niche area of the whole community. Definitionally, the degree of suitability of a dominant species, which populates a community, must be evaluated with more gravity than the suitability of the character species to the same community. Therefore, from the suitability curves of all dominant species (small blue lines in figure 1), the smallest common range is formed (bold blue line). At the same time, from the suitability degrees of the character species (small green lines), the average is formed (bold green line). Summarized together, these curves supply the smallest common area (bold red line) for the minimum of dominant species and the average of the character species.

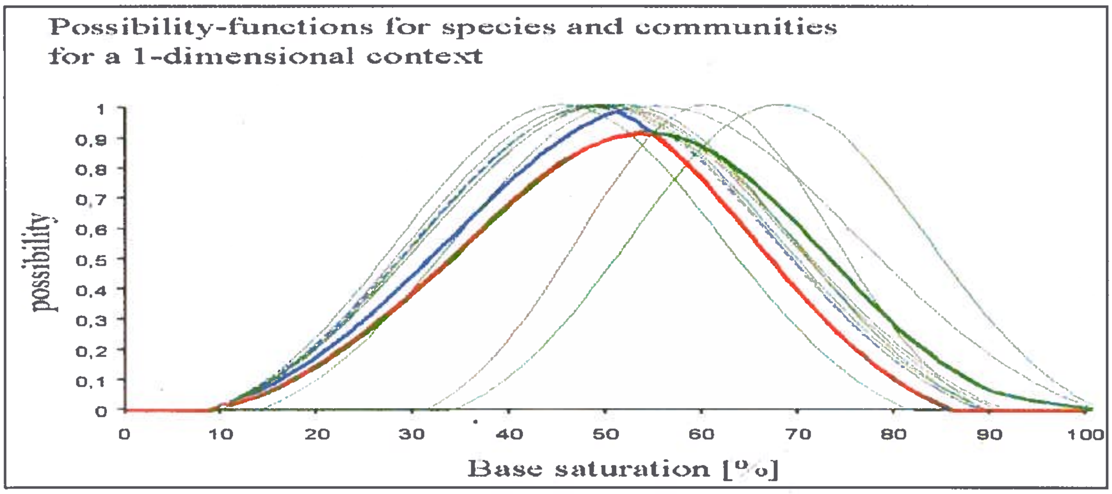


Fig. 1: The derivative of a community's suitability function (red line) through the construction of the minimum range, which is the consequence of the suitability function of 1) the dominant species (blue lines) and 2) the character species (green lines) - to a single site parameter (here, base saturation). The example community is the Lolio-Cynosuretum cristati.

Under the same procedure as was used to describe the base saturation derivative, the suitability curves are derived for the communities using the moisture degrees and the C/N relationship.

From the example of the habitat natural meadow that is situated in the planar-subcontinental region with plane relief and a groundwater distance of 6 dm - all typical semi-natural grassland communities can be indicated (fig. 2). Shown (graphically) is an obviously regular arrangement of the natural plant communities, which demarcates an indirect proportional connection between the base saturation and C/N in the natural equilibrium. This picture of the arrangement of the natural plant communities on an line of the harmonious equilibrium of base saturation to C/N-ratio repeats itself for all regular site types and all natural (or semi-natural) plant communities. Simultaneously, the overview of all natural plant communities of a regular site type shows the absence of the natural communities in the extremely disharmonious ranges, which are characterized by high nitrogen contents and low base content in the rooted soil (see white surfaces in fig. 2).

3.2 Determination of the actual habitat condition by bioindication and comparison with the primary natural habitat parameters

The locally occurring plant species in a homogeneous habitat are entered into a database. Additionally, the surveyed habitat's main parameters are obtained - soil type, type of hydromorphy, type of climatic region, type of relief, and sun exposure. This is done either by site measurements or from large-scale maps.

From the database of plant species suitability values, the common ranges occupied by the current species together (minimum suitability curves for all current species) were determined for the base saturation, the C/N relationship and for moisture. The current condition of the habitat is assumed to be the middle of the common range as showcased in fig. 2 on the example of the forest Klever Reichswald near the border with the Netherlands. Simultaneously, the model provides the primary natural plant community from the inputted characteristics. The modelling software now produces the distance (=deviation) between the primary natural condition and the actual current condition.

According to the definition of the Critical Loads (UBA 1996), the maximum stress loads are met (but not exceeded) as long as no changes in the structure or/and function of the vegetation are registered. Therefore the empirical Critical Load must result from a threshold value of the suitability function of the primary natural plant community.

A meaningful threshold value is the suitability degree of 0.5. At this value of the condition parameter (Critical Limit), the primary natural plant community has only the half possibility of its continued existence and only half its vitality. The threshold values of the primary natural plant community in fig. 2 lie within the limits of the border between the yellow and the green ranges.

The three-dimensional viewpoint of the plant community as a function of base saturation AND the C/N relationship AND the form of hydromorphy causes the threshold value of a community to always be a three-dimensional vector in the form of a threshold bowl. Simplified, the form of hydromorphy can be specified for a community as a constant (optimum), which then allows the threshold bowl to be illustrated as a two-dimensional relationship of the base saturation and the C/N relationship. The current exceedance over the threshold value corresponds to the shortest span from the actual condition to the nearest point on the threshold bowl (fig. 2).

To determine the deviation degrees, the rate of species loss is computed as a further parameter. The computation follows on the basis of the following definition:

Species Loss V_{akt} = portion of the non conforming species from the species list (A_{prim}) of the primary natural plant community of the uninfluenced habitat, (determined from the type of soil, type of hydromorphy, climate zone and local sun exposure) compared to the current species list (A_{akt}).

Similar to the determination of the degrees of deviation from the presently occurring plant species, one can obtain the degree of deviation from the current humus form in forests or fens (fig. 2).

3.3 Determination of the Recovery Target and the Current Regeneration Potential

If the actual condition lies not far from the primary natural condition (condition A in fig. 2), then, as a rule, populations of one or several 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 (condition B in fig. 2), then, as a rule, every dominant species and character species of the primary natural plant community are extinct. 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.

For the definition of the recovery target, one condition is the determination of the actually existing degree of remaining "naturalness" of the current species composition at the habitat in question.

The following definitions apply:

Degree of Naturalness N_{akt} = the proportion of actually occurring species (A_{akt}) of the list of dominant species (A_{pot}^{D}) of the natural plant community, to that which corresponds to the actual site parameters (indicated by actually observed species) that lie nearest the nature-identical harmonious equilibrium of BS and C/N. This nearest harmonious parameter combination should be the recovery target for the habitat. This target does not have to be identical with the primary uninfluenced condition.

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 (dominant and character species) at the recovery target A_{pot}. It is a function of the soil-chemical regenerative power and the propagation behavior of the potential dominant and character species.

Possible targets of recovery =

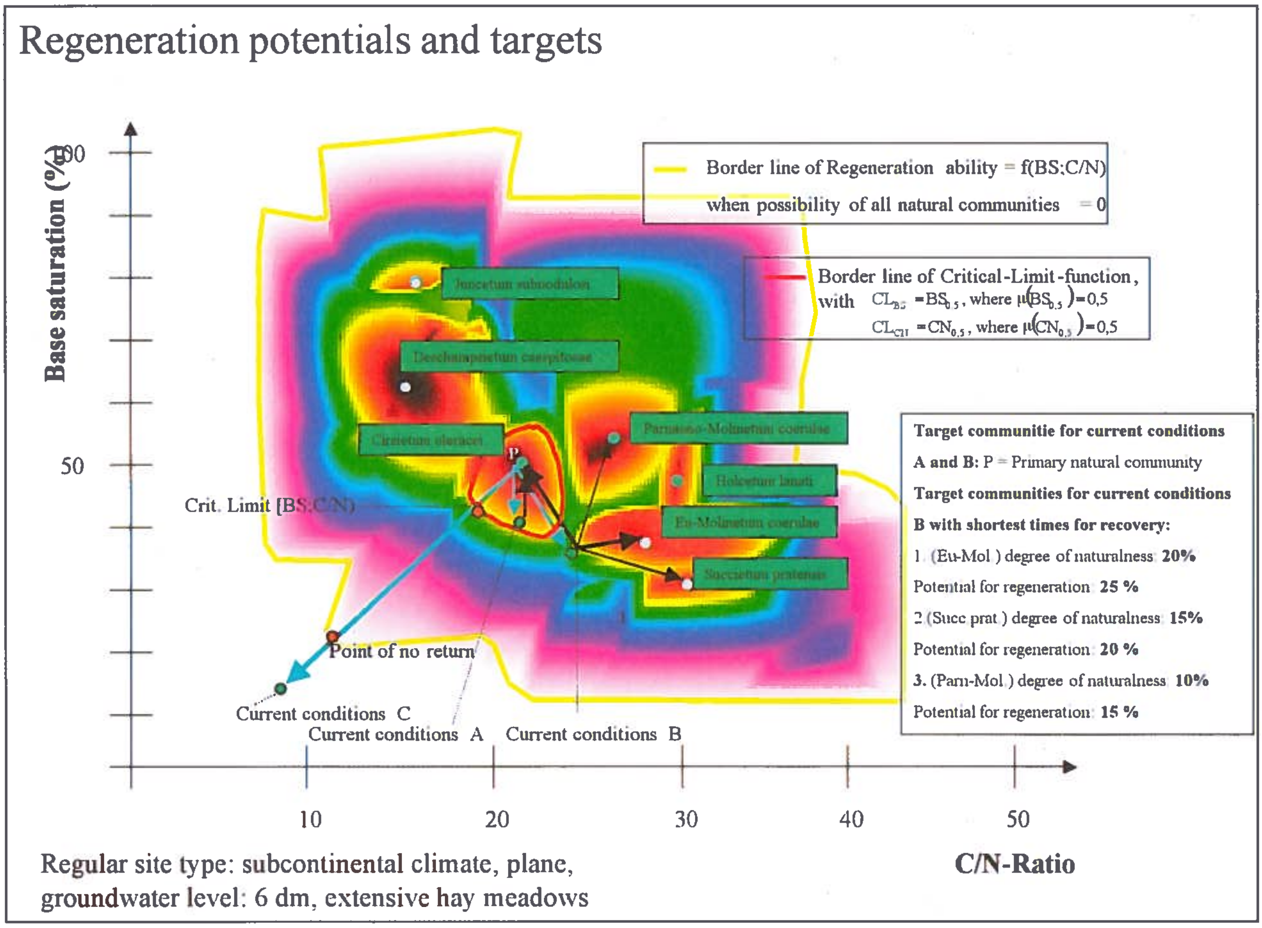
1. Target: (a slightly changed ecosystem): the primary natural habitat condition;
2. Target: (a highly changed ecosystem): the re-establishment of the nearest balanced nature-identical equilibrium of N, C and Bc, which is appropriate for the current (disharmonic) condition - the most quickly reached 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 (fig. 2). 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. Other targets with balanced natural-identical conditions are possible. Which, if either, of 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 fertilization could be a meaningful measure in a forest.

The recovery time is a function of the current presence of species, which can develop their population to a dominant position in potential natural plant communities. By this it means a function of the presence of a reproductively capable gene-potential for the development of a new plant community in or near of the regarded habitat.

In particular, where the concentration of base cations in the soil solution 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 aluminum 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. The border of regenerative power is exceeded irreversibly (condition C in fig. 2). These ecosystems are characterized by the absence of species, which could function in any potential natural plant community as either dominant species or as character species (degree of "Naturalness" = 0). Remaining species 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.



Target communitie for current conditions A and B: P = Primary natural community

Target communities for current conditions B with shortest times for recovery:

1. (Eu-Mol.) degree of naturalness: **20%**
Potential for regeneration: **25 %**
2. (Succ prat.) degree of naturalness: **15%**
Potential for regeneration: **20 %**
3. (Parn-Mol.) degree of naturalness: **10%**
Potential for regeneration: **15 %**

Regular site type: subcontinental climate, plane, groundwater level: 6 dm, extensive hay meadows

Fig. 2: (1) Distribution of the suitability maxima of all plant communities of the habitat type: natural meadow in the planar- subcontinental region with a plane relief, groundwater distance 6 dm (the sequence black-red-yellow-green-blue-purple corresponds to the reduction of the suitability degrees of the respective communities); (2) Assessment of the deviation between the current condition (A) after acidification influences and eutrophication and the former natural condition (P); (3) the line of the Critical Limit function; (4) Possible regeneration targets for a current habitat (B), which has more deviated from its primary natural condition; (5) Border line of regeneration ability and the "point of no return" for a current habitat (C), which is irreversible changed into extreme disharmonious condition

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 of BS in the upper mineral soil layer of less than 10 percent.

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

4. Derivative of empirical Critical Loads for nitrogen

In order to establish the connection between the threshold value for the C/N-ratio in the topsoil (determined here with the help of the BERN model and considering the threshold value of base saturation) and the threshold value for the nitrogen input, one must first clarify the question of whether there exists a significant dependency between the C/N relationship/Base-saturation and N-input. WOLFF AND RIECK (1997) point out in the 1996 German Forest Soil Report, which evaluated the results and condition of the country-wide forest soil investigation between 1987 and 1993, that both the average values of the N-supplies in the topsoil layers (median) and the medial C/N ratios do not show significant influence on the humus form. However, this statement does not apply to the respective extreme values (90th percentile of N-content, 10th percentile of the C/N ratios). In contrast to the average values, the extremes show a significant dependency to each other (r^2=0,74). A harmonious site type is indicated simultaneously by a natural plant community and a harmonious humus form. Therefore, the Critical limit function of the natural community could be valid also for the harmonious humus form in this site type respectively. In order to verify the thresholds for C/N-ratio the 10th percentile in connection with the 90th percentile of N-content of each humus form at the BZE-sampling points will be used.

Therefore, the following derivative describing the Critical Loads is legitimate.

The lowest acceptable C/N-ratio is determined, by definition, for each natural plant community at the point which exhibits the furthest disharmonic relationship of base saturation to C/N-ratio on the Critical limit function f(BS;C/N)=0,5. This point is determined by drawing a straight line from the point of optimum of the primary-natural community to the 0,0 point from base saturation and C/N-ratio. The intersection of this straight line with the Critical limit function ($C/N_{BS(crit)}$) is, hence, the extreme disharmonious condition, in which the primary-natural plant community is just able to exist, meaning it exists with half vitality. One can expect, therefore, that a significant connection between the Critical Loads for N and the $C/N_{BS(crit)}$ for each natural plant community is verifiable.

First, a statistical analysis was conducted with the empirical Critical Loads (= CL(N)emp) established from field observations of occurring species change and diversity loss. The matrix (which was set up in this workshop in Berne in November 2002) designates Critical Load-ranges to the habitat types of the ecosystems: semi-natural grassland, heaths, and fens (after EUNIS classification). In a additional matrix, these Critical Load-ranges can be further differentiated - with consideration of abiotic site parameters (BOBBINK ET AL., 2002). With the help of these two matrixes a clear allocation of discrete CL(N)emp to the natural plant communities of these ecological system types has been made possible. At the Berne workshop, no differentiation of the CL(N)emp for forest ecological systems was made. However, the components of the simple-mass-balance equation (NAGEL AND GREGOR 1999) were quantified for each specific ecosystem type.

The statistical analysis of the correlation from $C/N_{BS(crit)}$ to CL(N)emp and to CL(N)SMB is shown in the following table:

Coefficient of determination for correlation of parameter (r^2)	Grasslands, heaths, fens								Woodlands (forests)			
	total		sub continental		Sub oceanic		azonal wet		total	Sub continental	Sub oceanic	azonal wet
CL(N)emp	0.62		0.66		0.71		0.45					
$C/N_{BS(crit)}$		0.66		0.76		0.76		0.47	0.47	0.77	0.81	0.64
CL(N)SMB												

From this it can be concluded that both CL(N)emp and CL(N)SMB are equally well-suited for the quantification of the maximum Critical Load borders of natural plant communities. In fact, the coefficient of determination for the correlation of CL(N)emp =CL(N)SMB line also amounts to r^2=0.84.

5. Results, current State of the Work and Future Outlook

Presently for 130 regular site types which represent the German Northern Lowlands country-wide results are determined for:

- allocation of typical primary-natural plant communities (woodlands) and semi-natural communities (fens, heaths and grasslands);
- ecological Optima for all these plant communities regarding base saturation, C/N-ratio and hydromorphic type of the soil;
- threshold values for these parameters at the most disharmonious point on the Critical limit function line for each plant community (Critical Limit of C/N according to the critical BS);
- Critical Loads for N-eutrophication at the most disharmonious point on Critical limit function (empirical Critical Load N / Simple mass balance-Critical Load N).

The current model reflects a semi-dynamic approach because it represents actual conditions compared with the primary natural conditions before the influence of high chemical loads (the environmental condition from around 1950). In the future, however, there will also follow a coupling with the well-known dynamic models.

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Annexes

Annex 1: Acknowledgements

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Annex 2: Agenda of the Workshop

Sunday, 10 November 2002

Arrival of Participants

Monday, 11 November 2002

08.00 – 09.00	Registration
09.00 – 09.05	Welcome Address *Gerhard Leutert, Swiss Agency for the Environment, Forests and Landscape*
09.05 – 09.10	Organizational Information *Beat Achermann, Swiss Agency for the Environment, Forests and Landscape*
09.10 – 09.15	Status of the UNECE Convention on Long-range Transboundary Air Pollution *Radovan Chrast, Secretariat of the UNECE LRTAP Convention*

Presentation of scientific background information
Chairperson: Jean-Paul Hettelingh

09.15 – 09.40	Introduction to the background paper, methods to derive empirical critical loads for nitrogen *Roland Bobbink*
09.40 – 10.20	Empirical critical loads for nitrogen for forest habitats *Walter Flückiger, Sabine Braun*
10.20 – 10.40	*Coffee Break*
10.40 – 11.20	Empirical critical loads for nitrogen for grasslands and tall forb habitats, fens and mires, inland surface waters, coastal and marine habitats *Roland Bobbink, Isabel Van den Wyngaert*
11.20 – 12.00	Empirical critical loads for nitrogen for heathlands, scrub and tundra habitats, bogs *Mike Ashmore*
12.00 – 12.30	EUNIS ecosystem classification and ecosystem mapping *Jane Hall*
12.30 – 14.00	*Lunch (at the Conference Center)*
14.00 – 14.30	Nitrogen in the UNECE International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) *Thomas Haussmann, Wim de Vries*
14.30 – 14.45	Nitrogen in the UNECE International Co-operative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM) *Lars Lundin*
14.45 – 15.30	The use and possibilities of multispecies models for the calculation of critical loads for nitrogen for (semi-)natural ecosystems *Arjen van Hinsberg/Han van Dobben*
15.30 – 15.50	*Coffee Break*

15.50 – 17.20	Working Groups 1 – 3 Working Group 1: Forest habitats *Chairperson: Ursula Falkengren-Grerup; Co-chair: Mike Hornung* *Rapporteur: Joachim Strengbom* Working Group 2: Grasslands, fens and mires, inland surface waters, coastal and marine habitats *Chairperson: Alan Davison; Rapporteur: Morten Strandberg* Working Group 3: Heathlands, scrub and tundra habitats, bogs *Chairperson: Jan Roelofs; Rapporteur: Sarah Woodin*
18.15	*Departure for the Workshop Dinner*

Tuesday, 12 November 2002

09.00 – 10.15	Working Groups 1 – 3
10.15 – 10.45	*Coffee Break*
10.45 – 12.15	Working Groups 1 – 3
12.30 – 14.00	*Lunch (at the Conference Center)*
14.00 – 15.00	Plenary Session: Reporting of Working Groups *Chairperson: Mike Ashmore*
15.00 – 16.00	Working Groups 1 – 3
16.00 – 16.20	*Coffee Break*
16.20 – 17.20	Working Groups 1 – 3
Evening	*Dinner in the city of Berne (individual choice)*

Wednesday, 13 November 2002

09.00 – 10.00	Final Discussions in Working Groups 1 – 3
10.00 – 10.30	*Coffee Break*
10.30 – 12.15	Plenary Session Presentation of Working Group Conclusions and Recommendations, Final Discussion *Chairperson: Roland Bobbink*
12.15 – 12.30	Closure of the Workshop
12.30 – 13.30	*Lunch (at the Conference Center)*

Departure of Participants

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