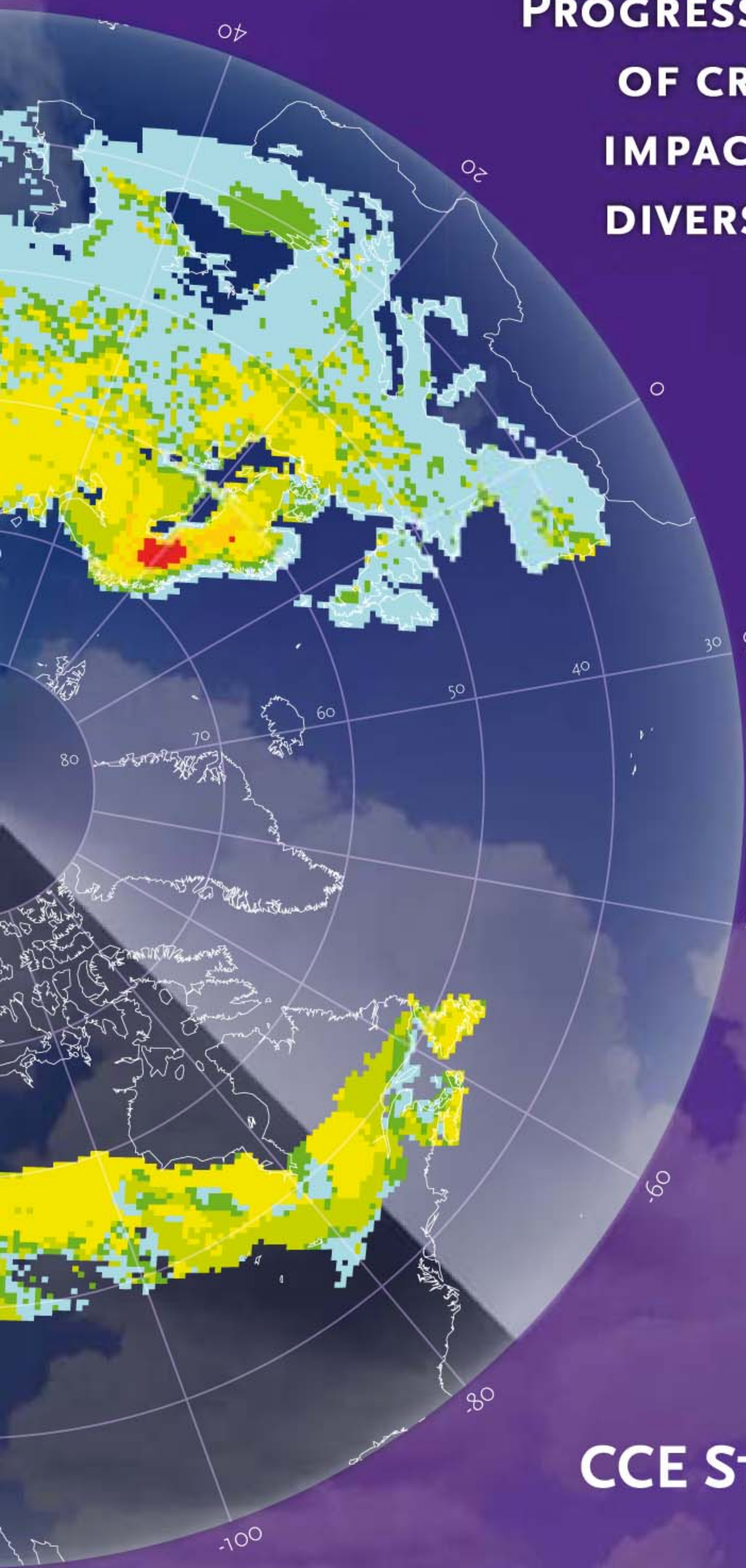
Progress in the modelling of critical thresholds, impacts to plant species diversity and ecosystem services in Europe



CCE Status Report 2009

Netherlands Environmental Assessment Agency

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J-P. Hettelingh, M. Posch, J. Slootweg (eds.)

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

Convention on Long-range Transboundary Air Pollution

ICP M&M Coordination Centre for Effects

Netherlands Environmental Assessment Agency

LRTAP Long-range Transboundary Air Pollution

Progress in the modelling of critical thresholds, impacts to plant species diversity and ecosystem services in Europe :
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Contents

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Context and Summary

Progress in the modelling of critical thresholds, impacts to plant species diversity and ecosystem services in Europe: CCE Status Report 2009

The CCE Status Report 2009 addresses the progress made in completing knowledge on effects of air pollution in Europe, with interactions involving the change in climate and biodiversity. It further strengthens the logic of the Executive Body at its 25th session (2007) with the request to the Working Group on Effects " to consider further quantification of policy relevant effect indicators such as biodiversity change, and to link them to integrated modelling work".

It is useful for understanding the objective of the CCE SR 2009 report and answers to the question on how to further develop the modelling and mapping work in support of integrated modelling, to put the call of the Executive Body in the context that air pollution and the change of climate and biodiversity are interrelated.

There is a well established awareness of interactions at the *starting points of the source-receptor chain*, between sources and emissions of greenhouse gas and air pollution. Growing policy attention in general, and in the LRTAP Convention in particular, is paid to the fact that some particles offset, and others reinforce, global warming, while fine airborne particles have an additional and well-established adverse effects on human health (see also www.ircg.org). Other source-related linkages are established between emissions of nitrous oxide with its ozone-depleting potential which, if the emissions are left unchecked, will further increase anthropogenic forcing of the climate system (see Ravishankara, 2009, *Science* 326:123-125).

Interactions between climate change and air pollution are also important for the *endpoints of the source-receptor chain*. A well-known example of such interaction is that between nitrogen deposition, which is excessive on most of European nature, and carbon sequestration. Elevated nitrogen deposition can lead to enhanced growth of forests, and thus to increased carbon sequestration, as long as the receptors are nitrogen limited (De Vries *et al.*, 2009, *Forest Ecology and Management* 258: 1814-1823). This direct link is of widely recognised importance to policies on greenhouse gas reduction to meet the target of 450 ppmv CO_2. However, the impact of nitrogen on carbon sequestration may be unfavourable for peatlands, even to the extent where it is not unlikely that existing peatland carbon stocks are released to the atmosphere.

Knowledge needs to be strengthened of synergies and antagonisms between endpoint related processes that are relevant for policies that wish to include nitrogen deposition levels as a measure to mitigate climate change. For this, also indirect linkages need to be understood. Such policies thus involve the requirement not to affect interactions between air quality and endpoints for soil chemistry, biodiversity and ecosystem services that sustain human welfare. Atmospheric pollution in general, and nitrogen deposition in particular cause adverse effects to biodiversity. Recently, Turner *et al.* (*Nature* 462:278-279, 19 November 2009) wrote "If human adaptation to climate change compromises biodiversity, then the loss of forests and other natural ecosystems will accelerate climate change, increasing the need for adaptation even as the planet's capacity to accommodate it diminishes". The adverse effects of air pollution on the provisioning, regulating and supporting services of nature can only amplify this risk.

Following the request by the Executive Body a *two track approach* has been adopted under the ICP Modelling and Mapping, converging in robust effect-based components for integrated assessment. *The first track* is model based. It complements the modelling of soil chemistry and critical loads with the dynamic modelling of the change in plant-species diversity. For this the CCE draws on the collaboration with Alterra (www.alterra.wur.nl) in a project co-funded by the Netherlands Environmental Protection Agency since 2007. Other work under the ICP M&M includes the development of the ForSAFE-veg model co-funded by the Swiss Federal Office for the Environment (BAFU), the Swedish Environmental Protection Agency (Naturvårdsverket) and the CCE.

The second track involves the regionalisation within the ICP M&M of empirical, site specific, knowledge about nitrogen effects on biodiversity.

Progress regarding both tracks was presented at the 18th CCE Workshop in Berne (21-23 April 2008) and the 19th CCE Workshop in Stockholm (11-13 May 2009).

The authors of the presentations at the Stockholm workshop kindly agreed to contribute to the present report.

The CCE Status Report 2009 consists of four parts. Parts 1 and 2 focus on modelling methodologies, while Parts 3 and 4 address empirical knowledge on the effects of nitrogen deposition.

Part 1 "*Integrated Assessment*" focuses on current and new effect-based contributions to integrated assessment modelling. Chapter 1 describes the analysis of the effects of emissions in 2000, 2020 and 2030 according to a draft baseline scenario, of which depositions were compiled by EMEP-CIAM in November 2009 and made available to the CCE. It includes an overview of the use of currently available methods and data under the ICP M&M for assessing the current state and future risk of acidification and eutrophication in European nature, including Natura2000 areas. Results show that the risk of eutrophication in particular is widespread in terms of critical load exceedance, expected time delays of recovery and, tentatively, changes of species richness. The draft baseline scenario was only used in Chapter 1. In other chapters use was made of the so-called Current Legislation scenario that has been more widely available over a longer period of time. Chapter 2 describes the current state according to the land cover map which has been compiled by the CCE in collaboration with the Stockholm Environment Institute (SEI) in support of the modelling of atmospheric dispersion by EMEP and effects under the Working Group on Effects. Chapter 3 illustrates the current capability of the Very Simple Dynamic (VSD) model to analyse recovery from acidification under climate change. Results confirm that climate change can reinforce recovery of acidified forest soils in Europe. Finally, Chapter 4, provides a quantification of the impacts of nitrogen deposition on forest ecosystem services, such as the regulation of climate, water quality and quantity and soil quality. It is shown that the reduction in nitrogen deposition since 1980 has had a positive effect on biodiversity, and on the quality of soil and water, and a negative effect on global warming.

Part 2 "*Dynamic Modelling*" consists of two chapters that address the progress in the development of dynamic modelling to capture the interaction between nitrogen deposition and the change of climate and biodiversity. Chapter 5 presents results of a pilot study showing that it is possible to simulate the relationship between nitrogen deposition and the ensuing composition of the plant community while keeping track of the change of climate and land use. Chapter 6 focuses on the extension of the Very Simple Dynamic (VSD) model, currently widely used by National Focal Centres, to include carbon and nitrogen dynamics. This extended model, termed 'VSD+', is designed to limit the data requirements needed by National Focal Centres to simulate vegetation responses to changes in nitrogen deposition.

Part 3 "*Options to further quantify effect indicators*" includes three chapters each addressing indicators that could be suitable for capturing the change in biodiversity under the LRTAP Convention in a way that is understandable for policy makers and the general public, and that could attach to air pollution the attention it deserves in biodiversity policy. Chapter 7 proposes to generalise the criteria behind the red-list species (rarity and decline) to include all species. In Chapter 8 a distance measure between physically and potentially available plant species is discussed as possible indicator for nitrogen-induced changes of plant species diversity. In Chapter 9 an approach is explored to interpret habitat suitability predictions using lists of predefined indicators that are rare and typical as well as indicators that are untypical and invasive of a habitat.

Part 4 "*Current knowledge on empirical critical loads*" addresses new knowledge that confirms the need to review and revise empirical critical loads that have been established in 2002. Empirical critical loads are used to reduce the uncertainty of model assessments of areas at risk of nitrogen deposition, while improving the understanding of the role of biology. An overview of the need for revision is provided in Chapter 10 while findings focussing on central and northern Europe are described in Chapters 11 and 12 respectively. A contribution addressing nitrogen effects in southern Europe is available as powerpoint presentation for the 19th CCE workshop (www.pbl.nl/cce).

As a result the CCE has started an international project in the autumn of 2009 in collaboration with the Swiss Federal Office for the Environment (BAFU) and the German Umweltbundesamt (UBA). Together with these international institutions, the Dutch Ministry of Housing, Spatial Planning and the Environment (VROM) and the National Institute for Public Health and the Environment (RIVM), the CCE organises in the UN international year of biodiversity 2010, a workshop under the LRTAP Convention entitled "Workshop for the review and revision of empirical critical loads and dose response relationships" (Noordwijkerhout, 23-25 June 2010) to finalize a comprehensive report to the 29th session of the Working Group on Effects under the LRTAP Convention.

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

Rapport in het kort

Vooruitgang in de modellering van kritische drempels, effecten op de rijkdom van plantensoorten en ecosysteemdiensten in Europa: CCE Status Rapport 2009.

Het CCE Status Rapport 2009 beschrijft de vooruitgang in de analyse van effecten van luchtverontreiniging en de interacties met de verandering van het klimaat en biodiversiteit. Het rapport past in de methodeontwikkeling die nodig is om te voldoen aan het verzoek van de *Executive Body* van de Conventie voor Grensoverschrijdende Luchtverontreiniging (LRTAP) tijdens zijn 25e zitting (2007) aan de *Working Group on Effects* om, voor gebruik in geïntegreerde modellering, meer beleidsrelevante indicatoren te ontwikkelen voor effecten op bijvoorbeeld de biodiversiteit.

Het CCE Status Rapport 2009 sluit aan bij de toenemende bewustwording bij beleidsmakers en wetenschappers dat uitputting van het bufferend vermogen van de natuur door luchtverontreiniging gevolgen kan hebben voor de verandering van, en terugkoppelingen met, klimaat en biodiversiteit. De Verenigde Naties hebben 2010 uitgeroepen tot jaar van de biodiversiteit. De analyse van de rol van – en effecten op de ecosysteemdiensten voor het menselijk welzijn krijgt daarbij toenemende aandacht, en zo ook in dit rapport.

Het rapport bestaat uit 4 delen waarin wetenschappers die betrokken zijn bij de *Working Group on Effects* bijdragen leveren op het gebied van de empirie en de modellering van (in-) directe effecten van stikstofdepositie op bodemprocessen en de diversiteit van plantensoorten in Europa. Deze bijdragen zijn een weergave van de wetenschappelijke voordrachten tijdens de 19e CCE workshop (Stockholm, 11-13 mei 2009).

Deel 1 "*Integrated Assessment*" gaat ondermeer in op nieuwe modelontwikkelingen ten behoeve van effect gerichte analyses in geïntegreerde modellering.

Hoofdstuk 1 beschrijft de analyse van effecten van emissies in 2000, 2020 en 2030 van een voorlopig referentiescenario waarvan de bijbehorende deposities, berekend door EMEP-CIAM, beschikbaar zijn gesteld aan het CCE in november 2009. Dit referentiescenario is uitsluitend in Hoofdstuk 1 toegepast. In ander hoofdstukken van dit rapport is gebruik gemaakt van het breder, en sinds langere tijd, beschikbare "Current Legislation Scenario". Resultaten laten zien dat het risico van vooral vermesting wijdverspreid is in de Europese natuur (waaronder Natura2000 gebieden) in termen van ondermeer de overschrijding van kritische drempels, uitgestelde schade en herstel, en verandering van de diversiteit van plantensoorten. Hoofdstuk 2 beschrijft de status van de Europese landbedekkingskaart die in samenwerking met het Stockholm Environment Instituut (SEI) is ontwikkeld en is toegesneden om binnen de LRTAP Conventie de ecosysteemeffecten te schatten van luchtverontreiniging. Hoofdstuk 3 illustreert de toepassing van het *Very Simple Dynamic* (VSD) model om het herstel van verzuringschade te simuleren bij verandering van het klimaat. Het vermoeden dat een stijgende temperatuur het herstel van verzuurde bosbodems kan bevorderen wordt bevestigd. Hoofdstuk 4, tenslotte, geeft een kwantificering van de gevolgen van stikstofdepositie voor de diensten van bosecosystemen zoals bijvoorbeeld de regulatie van klimaat, waterkwaliteit, waterkwantiteit en bodemkwaliteit. Er wordt aangetoond dat de vermindering van de stikstofdepositie op bosgebieden sinds 1980 een positief effect heeft voor de biodiversiteit, en de kwaliteit van bodem en water maar een negatief effect op de opwarming van de aarde.

Deel 2 "*Dynamic Modelling*" behandelt de vooruitgang in de ontwikkeling van de dynamische modellen van de interactie tussen stikstofdepositie en de verandering van klimaat en biodiversiteit. In Hoofdstuk 5 worden resultaten van een voorbeeldstudie gepresenteerd die aantonen dat het mogelijk is om de relatie na te bootsen tussen stikstofdepositie en de daarmee samenhangende plantensamenstelling, rekening houdend met de verandering van klimaat en landbedekking. Hoofdstuk 6 beschrijft de uitbreiding met de koolstof en stikstof dynamiek van het, bij vele Nationale Focal Centra toegepaste, VSD model. Dit uitgebreide model, genaamd 'VSD+', is zodanig ontworpen dat de databehoefte van Nationale Focale Centra beperkt blijft om op regionale schaal de vegetatie respons van stikstofdepositie te kunnen bepalen.

Deel 3 "*Options to further quantify effect indicators*" bestaat uit drie hoofdstukken die elk ingaan op alternatieve indicatoren om verandering van biodiversiteit onder de LRTAP Conventie te kunnen kwantificeren. Dit op een manier die begrijpelijk is voor het publiek, en waarbij risico's van luchtverontreiniging ook vanuit het biodiversiteitsbeleid voldoende aandacht kunnen krijgen. In Hoofdstuk 7 wordt voorgesteld om de criteria achter de IUCN rode lijst van bedreigde soorten te generaliseren tot alle soorten. In Hoofdstuk 8 wordt het verschil tussen feitelijk - en potentieel aanwezige plantensoorten bediscussieerd als maat voor de verandering van de diversiteit van plantensoorten door stikstof. In Hoofdstuk 9 wordt een methode beschreven waarbij de kans op het voortbestaan van een habitat worden bepaald aan de hand van zowel indicatorsoorten die zeldzaam en typisch zijn , als indicatorsoorten die invasief en a-typisch zijn voor een habitat.

Deel 4 "*Current knowledge on empirical critical loads*" geeft een overzicht van de nieuwe inzichten op basis waarvan het besluit wordt gerechtvaardigd om de empirische critical loads uit 2002 te herzien. Empirische critical loads worden gebruikt om de robuustheid van de gemodelleerde risico's van stikstofdepositie voor Europese natuurgebieden te vergroten, onder een verbeterd begrip van de rol van de biologie. Hoofdstuk 10 geeft een algemeen overzicht, waarna meer specifieke informatie voor Centraal – en Noord Europa in respectievelijk de hoofdstukken 11 en 12 wordt verschaft. Een bijdrage voor Zuid Europa is als PowerPoint presentatie voor de 19e CCE workshop (Stockholm, 11-13 mei 2009) beschikbaar op www.pbl.nl/cce. Als vervolg hierop is het CCE in de herfst van 2009 een international project gestart in samenwerking met het Zwitserse Bundes Ambt für Umwelt (BAFU) en het Duitse Umwelt Bundes Amt (UBA). In samenwerking met deze internationale instanties en met het Ministerie voor Volkshuisvesting, Ruimtelijke Ordening en Milieu (VROM) en het Rijksinstituut voor Volksgezondheid en Milieu (RIVM) organiseert het CCE in het VN jaar van de biodiversiteit 2010, een workshop onder auspiciën van de LRTAP Conventie getiteld "Workshop for the review and revision of empirical critical loads and dose response relationships" (Noordwijkerhout, 23-25 June 2010) waarna de resultaten zullen worden gerapporteerd aan de 29e vergadering van de Working Group on Effects onder de LRTAP Conventie.

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

Part 1 Integrated assessment

This Part focuses on current and new effect-based contributions to integrated assessment modelling. Chapter 1 describes the analysis of the effects of emissions in 2000, 2020 and 2030 according to a draft baseline scenario, of which depositions were compiled by EMEP-CIAM in November 2009 and made available to the CCE. It includes an overview of the use of currently available methods and data under the ICP M&M for assessing the current state and future risk of acidification and eutrophication in European nature, including Natura2000 areas. Results show that the risk of eutrophication in particular is widespread in terms of critical load exceedance, expected time delays of recovery and, tentatively, changes of species richness. The draft baseline scenario was only used in Chapter 1. In other chapters use was made of the so-called Current Legislation scenario that has been more widely available over a longer period of time. Chapter 2 describes the current state according to the land cover map which has been compiled by the CCE in collaboration with the Stockholm Environment Institute (SEI) in support of the modelling of atmospheric dispersion by EMEP and effects under the Working Group on Effects. Chapter 3 illustrates the current capability of the Very Simple Dynamic (VSD) model to analyse recovery from acidification under climate change. Results confirm that climate change can reinforce recovery of acidified forest soils in Europe. Finally, Chapter 4, provides a quantification of the impacts of nitrogen deposition on forest ecosystem services, such as the regulation of climate, water quality and quantity and soil quality. It is shown that the reduction in nitrogen deposition since 1980 has had a positive effect on biodiversity, and on the quality of soil and water, and a negative effect on global warming.

1

Effect-oriented assessment of the 2009 PRIMES BaseLine Scenario

Jean-Paul Hettelingh, Maximilian Posch, Jaap Slootweg

1.1 Introduction

This chapter describes results from an effect-oriented assessment of the so-called 2009 PRIMES BaseLine Scenario (PRIMES-BL2009). Depositions of sulphur and nitrogen compounds in EMEP grid cells were derived from this scenario to assess critical load exceedance and generate other environmental impact related results. The EMEP Centre for Integrated Assessment Modelling (CIAM) at the IIASA prepared the PRIMES-BL2009 scenario for the interim assessment of the European Consortium for Modelling Air Pollution and Climate Strategies (EC4MACS).

In summary[1], this scenario was based on 2000 and 2005 emission data, calibrated to national values as were reported to the EMEP/CEIP. Draft PRIMES energy projections were included, as developed in mid-2009 for the EU27, Macedonia, Croatia, and Turkey. Projections for Norway and Switzerland did not include impacts of the economic crisis; they originated from PRIMES 2008 model runs. Projections for Albania, Bosnia and Herzegovina, Moldova, Serbia and Montenegro were based on trends from the IEA World Energy Outlook 2009. Agricultural activities included national data as reported to EUROSTAT for 2005. Projections are based on trends estimated by the CAPRI model in September 2009. Regarding the legislation on air pollution, it was assumed the implementation of EU and national legislation (if stricter), plus the EU IPPC Directive for combustion sources and Euro IV on heavy-duty vehicles.

It was assumed that national legislation related to greenhouse gas emissions would be fully implemented in all countries. In addition, for EU27 Member States, it was assumed the implementation of the EU CAP reform and Directives on landfill, waste, Fluorocarbon gas, motor vehicles, and the Emission Trading Scheme (EU ETS) for controlling carbon dioxide emissions. The latter assumes adoption of mitigation options by ETS sectors at marginal costs, under carbon price levels of 15 €/t CO_2 in 2015, 20 €/t CO_2 in 2020, 25 €/t CO_2 in 2025, and 30 €/t CO_2 in 2030 (in 2008 prices).

1 Source: Chris Heyes of IIASA

1.2 Impact assessment methodology

Air pollution impacts on the environment vary in time and space. A number of phases can be distinguished whereby excessive exposure propagates changes in biodiversity, finally reaching an endpoint expressed as a risk to human welfare. Biodiversity is not, as many believe, only about rare flora and fauna. Biodiversity plays an important role in human welfare, including health. Biodiversity and human welfare are related through ecosystem services, including those of support (e.g. nutrient cycling, soil formation), provision (e.g. food, fresh water), regulation (e.g. climate regulation) and finally culture (e.g. education, recreation). The World Health Organization (WHO) prepared a report as a contribution to the Millennium Ecosystem Assessment to address the issue of ecosystems and human well-being (Corvalan et al. 2005). The report states that 'ecosystems are essential to human well-being and especially to human health – defined by the World Health Organization as a state of complete physical, mental and social well-being' (Corvalan et al. 2005, p.12).

Under the Effects Programme of the Convention on Long-range Transboundary Air Pollution (LRTAP) and EC4MACS, the objective of the CCE Environmental Impact Assessment (CCE-EIA) methodology is to establish protection levels against adverse effects of eutrophication and acidification of air pollution on ecosystems in European countries, thereby diminishing the risk of adverse effects to ecosystem services.

These protection levels are the 'critical loads', which were modelled and mapped for European ecosystems in support of air pollution abatement policies (see Hettelingh et al. 1995, 2001, 2007a), and under EC4MACS also include Natura

2000 areas and methods for including biodiversity indicators. Regarding uncertainty, a method was developed under EC4MACS which combines IPCC approaches with ensemble modelling (Hettelingh et al. 2007b).

Input data to compute critical loads were submitted by the network of collaborating National Focal Centres operating under the Effects Programme of the LRTAP Convention. For countries that do not collaborate in this CCE network, critical loads are derived from European soil and vegetation databases.

However, knowledge on biological effects of these exceedances is limited, as they vary as function of a number of elements including soil type, meteorology, land cover and species occurrence. This is especially important from the viewpoint of risks caused by excessive deposition of nitrogen. The risks due to nitrogen inputs to the environment are increasing and are well documented (Galloway et al. 2008). Despite the lack of operational vegetation models on a European scale, improved knowledge is urgently required on the relationship between excessive nitrogen deposition and effects on biodiversity in Europe. Therefore, CCE has extended the European application of computed critical loads to also include empirical critical loads of nutrient nitrogen. Empirical critical loads are not based on mathematical models but on nitrogen addition experiments, both in the field and in the laboratory (Achermann and Bobbink 2003). First results (included in this interim report) are now available, using tentative relationships between nitrogen doses and impacts on the species richness. Methods and models to quantify the dynamics of changes in vegetation caused by air pollution under climate change are currently being developed for use under the LRTAP Convention (see De Vries et al. 2007, Reinds et al. 2009).

A summary of the CCE-EIA methodology is illustrated in Figure 1.1 (also see Hettelingh et al. 2008a) Two kinds of critical loads, that is, computed and empirical, are available to identify European regions which are at risk according to emission reduction scenarios that are simulated with the GAINS model. The availability of two kinds of critical loads is used in a method called Ensemble Assessment of Impacts, to establish the likeliness of the distribution of exceedances over European ecosystems. The Ensemble Assessment of Impacts methodology was documented in the CCE Progress Report 2007 (Hettelingh et al. 2007b).

In addition, impacts of changes in exceedances to biological and bio-geochemical endpoints can be further analysed in terms of (a) the dynamics of geochemistry (using dynamic models), and (b) impacts to species richness (using dose–response curves). The latter two kinds of endpoint assessments can be used for extending application possibilities of the Ensemble Assessment of Impacts in GAINS scenario analysis of impacts (see Figure 1.1).

Figure 1-1 shows that environmental impacts of excess nitrogen deposition can be analysed in two ways. The first (upper pathway) is to compute exceedances of modelled critical loads for acidification and eutrophication, and to perform dynamic modelling on delay times as appropriate. For reasons of simplification, Figure 1.1 illustrates the case of analysing Damage Delay Times only. The second (lower pathway) focuses on the analysis of effects of nutrient nitrogen, illustrating how exceedances of empirical critical loads can be used in conjunction with the analysis of impacts on species richness. Empirical critical loads have been reported in Achermann and Bobbink (2003), while regionalisation to Europe is conducted by the CCE on the basis of the European land-cover map.

The remainder of this chapter presents a review of the impacts of the PRIMES-BL2009 scenario for the risk of acidification and eutrophication, following the scheme shown in Figure 1.1. For this, computed and dynamic modelling, as well as empirical critical loads and tentative dose–response curves, were applied to European ecosystems including Natura 2000 areas. When possible and appropriate, a distinction between forest, vegetation and surface water systems was made. Finally, the chapter presents an assessment of uncertainty.

1.3 The risks of acidification and eutrophication in Europe

An overview of ecosystem areas at risk of acidification and excessive nutrient N deposition in countries within the domain of EMEP is given in Table 1.1. Results were computed by using the 2008 critical load database (Hettelingh et al. 2008a). Depositions were based on the PRIMES-BL2009 scenario for 2000, 2020 and 2030.

Table 1.1 shows that the European area at risk of acidification decreases from 10% in 2000 to 3% by 2020 and 2030. In the EU27, the area at risk of acidification under the CLE decreases from 19% in 2000, to 5% by both 2020 and 2030. From Table 1-1 can be seen that the computed European area at risk of eutrophication decreases from 51% in 2000 to 35 and 34% by 2020 and 2030, respectively. In the EU27, the areas at risk of eutrophication are 74% in 2000, 56% by 2020 and 54% by 2030.

In addition to the ecosystem area at risk, Table 1.1 also lists Average Accumulated Exceedances (AAE), computed as the area weighted average of the difference between deposition and critical loads. The objective is to reduce the AAE to zero, from the point of achieving biodiversity protection in Europe from the risks of acidification and eutrophication. As long as the AAE stays above zero, damage is likely to occur at some point in the future.

From Table 1.1 can be seen that the PRIMES-BL2009 scenario does not meet the objective of a zero AAE. The AAE of critical loads for acidification in the EU27 was 105 eq $ha^{-1}yr^{-1}$ in 2000, which is reduced to 17 and 15 eq $ha^{-1}yr^{-1}$ by 2020 and 2030, respectively. For eutrophication, the AAE in EU27 countries was 331 eq $ha^{-1}yr^{-1}$ in 2000, and estimated at 152 and 144 for 2020 and 2030, respectively.

The continued exceedance of critical loads for acidification and eutrophication warranted a closer geographical view of the locations of these exceedances in Europe. Dynamic modelling was then applied to establish delay times of

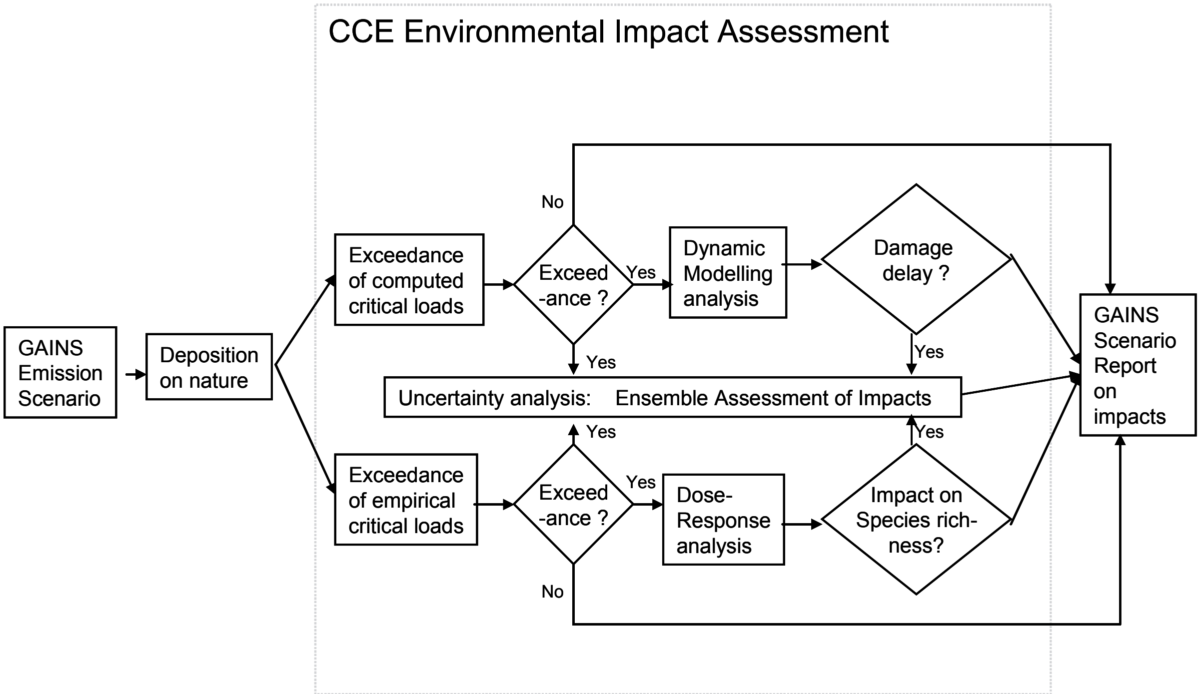

Figure 1.1 A simplified flowchart of the framework for the assessment of impacts of excess nitrogen deposition, in the context of integrated assessment modelling (e.g., the GAINS model) (Hettelingh et al. 2008a).

recovery – in regions with no exceedance – as well as damage for regions in which the AAE remains above zero. Results from these geographic analyses are presented in the following sections.

It should be noted that the protection levels achieved with the PRIMES-BL2009 scenario 2009 were generally higher than those presented in 2007, following Current Legislation (see Slootweg et al., 2007). For example, the total area at risk of acidification by 2020 was computed to be 6% using emissions under Current Legislation scenario, and 3% under the PRIMES-BL2009 scenario. For eutrophication, the risk is 42 and 35%, under the respective scenarios. The reason for the different outcomes is a difference between the scenarios in the spatial distribution and magnitudes of the emissions – and therefore also of depositions and of critical load exceedances – of sulphur and nitrogen compounds.

1.3.1 Risk of acidification to forests, vegetation and surface waters

Figure 1.2 shows the location of areas at risk and the magnitude of the exceedance of critical loads. Comparing the critical loads of all ecosystems (top row maps) in 2000 (left) to those by 2030 (right) shows that the area with high exceedances (red shaded areas) diminishes. The area changes from covering large areas in the United Kingdom, the Netherlands, Germany and Poland in 2000, to smaller areas remaining in the Netherlands and Poland by 2030. Areas with relatively low exceedances in 2000 (blue shaded) are shown to increase, also covering Germany, by 2030.

Note that not all countries with a National Focal Centre have submitted data for all ecosystem types. However, most of the countries that participated in the modelling and mapping of air pollution effects (see Hettelingh et al. 2008a) included forests. For countries which did not provide national data, critical loads were computed by the CCE using a background database (Posch et al. 2003a, Posch and Reinds 2005) for forests and (semi-)natural vegetation.

The location of the areas most at risk did not alter much, when we looked at forested areas only (second row of maps). Areas with (semi-)natural vegetation with relatively low exceedances (blue shading) by 2030, remain in some scattered grid cells in the United Kingdom, the Netherlands, Germany and France. Relatively low exceedances of critical loads for surface waters can be seen in the United Kingdom and northern European countries.

As illustrated in Figure 1.1, an exceedance requires the use of dynamic modelling (Posch et al., 2003b, 2007) to answer the question of how long it would take before chemical soil criteria are violated. This so-called 'Damage Delay Time' (DDT) may cover any time span, depending on the buffer capacity and other bio-geochemical conditions. Conversely (not illustrated in Figure 1.1), non-exceedance does not imply that recovery of the buffer capacity would be instantaneous. The 'Recovery Delay Time' (RDT) may also take a considerable time. This can be seen in many, but not all, lakes in northern Europe, where both the buffer capacity and fauna have started to recover due to decreasing acidification since the 1980s. Therefore, it was important to complement the analysis of critical load exceedance with an exploration of

Table 1.1 Percentage of ecosystem areas at risk of acidification (left) and eutrophication (right) for countries within the EMEP modelling domain, the EU27 in 2000, and for the PRIMES-BL2009 scenario for 2020 and 2030

	Acidification						Eutrophication					
	2000		2020		2030		2000		2020		2030	
Country	**AAE**	**Ex%**	**AAE**	**Ex%**	**AAE**	**Ex%**	**AAE**	**Ex%**	**AAE**	**Ex%**	**AAE**	**Ex%**
Albania	0	0	0	0	0	0	317	99	219	95	212	95
Austria	3	1	0	0	0	0	427	100	126	70	92	60
Belarus	52	18	4	4	4	4	382	100	257	94	260	94
Belgium	490	29	72	12	66	12	927	100	331	78	316	77
Bosnia & Herzegovina	46	13	0	0	0	0	260	88	114	68	101	65
Bulgaria	0	0	0	0	0	0	229	94	51	50	44	41
Croatia	26	4	2	2	2	2	513	100	276	97	254	97
Cyprus	0	0	0	0	0	0	120	68	125	68	158	69
Czech Republic	275	27	68	18	53	16	1058	100	648	100	601	100
Denmark	376	50	6	4	6	4	1084	100	532	100	542	100
Estonia	0	0	0	0	0	0	87	69	21	28	21	28
Finland	5	3	1	1	1	1	57	48	14	23	13	22
France	55	12	7	3	6	2	577	98	233	80	220	79
Germany	398	58	54	17	44	15	632	85	276	61	256	59
Greece	15	3	0	0	0	0	256	98	138	88	134	90
Hungary	113	23	6	3	5	3	540	100	301	99	273	94
Ireland	100	23	10	5	7	4	634	88	381	78	384	78
Italy	0	0	0	0	0	0	345	69	135	43	117	41
Latvia	40	19	3	3	3	3	269	99	136	90	133	89
Liechtenstein	150	52	0	0	0	0	590	100	311	100	274	100
Lithuania	211	34	70	29	67	29	489	100	365	100	363	100
Luxemburg	160	15	27	12	17	12	1091	100	610	99	577	99
Macedonia FYR	22	11	0	0	0	0	325	100	170	99	158	95
Moldova	0	0	0	0	0	0	413	100	356	98	361	98
Netherlands	2012	76	821	66	794	66	1427	93	844	84	839	84
Norway	47	16	8	6	7	6	30	21	5	7	5	7
Poland	665	77	143	35	124	33	738	100	471	98	455	97
Portugal	50	9	4	2	3	2	183	97	49	60	49	60
Romania	189	46	1	3	1	2	16	18	1	2	1	1
Russia (European part)	1	1	1	1	1	1	30	26	10	9	9	9
Serbia & Montenegro	44	16	0	0	0	0	291	96	161	83	154	81
Slovakia	103	16	10	7	6	6	672	100	370	100	329	99
Slovenia	39	7	0	0	0	0	363	98	61	59	38	43
Spain	19	3	0	0	0	0	342	95	170	87	170	87
Sweden	22	16	2	3	1	3	133	56	49	34	51	34
Switzerland	36	9	7	3	5	2	581	99	201	81	163	72
Ukraine	14	6	0	1	1	1	503	100	318	100	319	100
United Kingdom	246	39	24	12	20	11	139	25	36	14	35	14
EU27	**105**	**19**	**17**	**5**	**15**	**5**	**331**	**74**	**152**	**56**	**144**	**54**
All	53	10	8	3	7	3	184	51	86	35	82	34

the dynamics of the risk of acidification, as described in the following section.

1.3.2 Dynamics of the risk of acidification

Assuming that the depositions under the PRIMES-BL2009 scenario by 2020 remain constant in the future, we analysed the question about delay times of both recovery (RDT) and damage (DDT).

Figure 1.3 shows recovery delay times (using data from the background database) for the areas where critical loads will no longer be exceeded by 2020, according to the PRIMES-BL2009 scenario. The shadings in the left map indicate the minimum recovery year in each grid cell, that is, the year by which that no-longer-exceeded ecosystem recovers first. Areas that are shaded grey, blue or green indicate that at least one ecosystem will recover before or in the year corresponding to the colour, other ecosystems in such a grid cell may recover at later dates. Finally, white-shaded areas indicate that there is either no data or that geo-chemical recovery has already taken place for *all* ecosystems. The right map indicates the latest recovery delay time of an ecosystem in a grid cell. For example, in a yellow-shaded grid cell, there is at least one ecosystem that will recover between 2050 and 2100. Other ecosystems may exist in that grid cell, which recover earlier. Finally, pink-shaded grid cells imply that none of the ecosystems in that grid cell are expected to recover (either because exceedances remain or because they have already recovered).

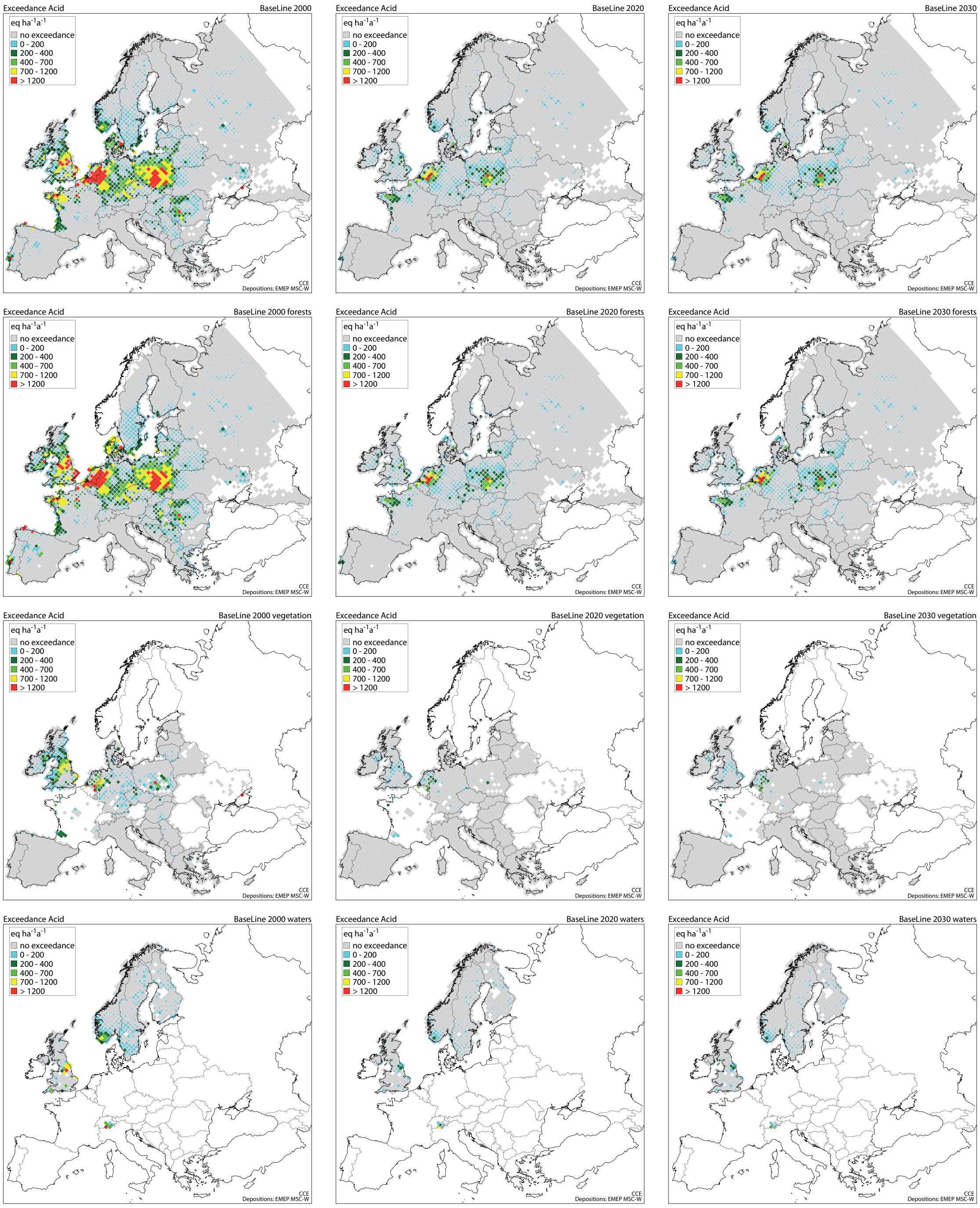

Figure 1.2 Exceedance of critical loads of acidification in 2000 (left), 2020 (middle) and 2030 (right) for all ecosystems (top row), forests (2nd row), vegetation (3rd row) and surface waters (bottom row). The size of a coloured grid cell is proportional to the fraction of the ecosystem area in the cell in which critical loads are exceeded.

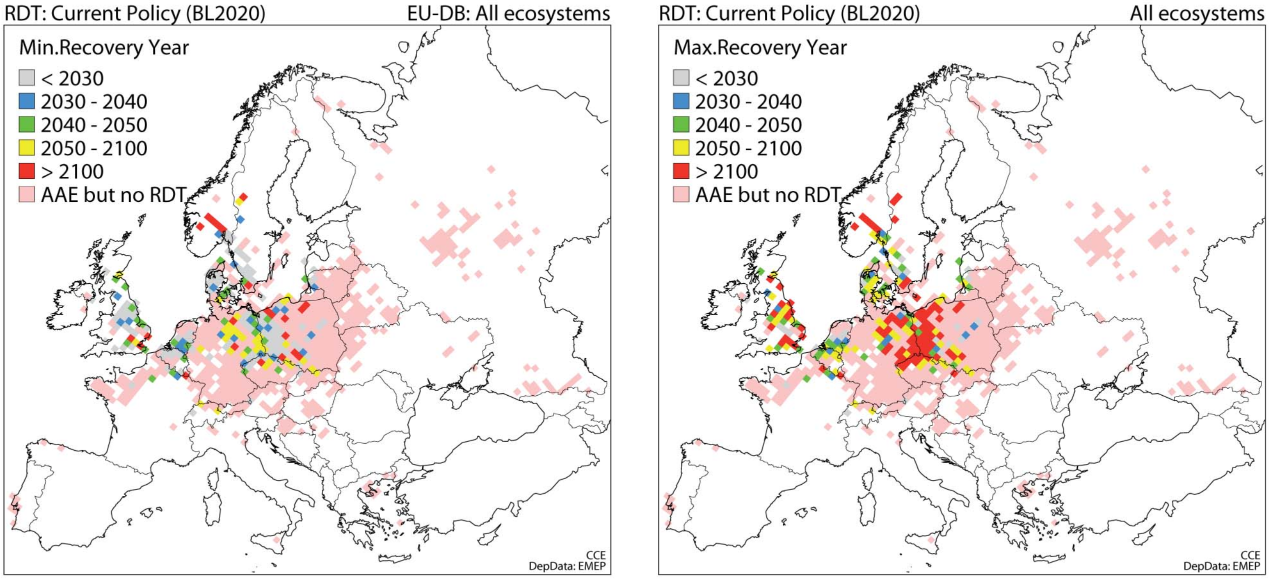

Figure 1.3 Recovery Delay Times for acidification, calculated for the earliest recovery year of an ecosystem in a grid cell (left) and the latest recovery year of an ecosystem in that grid cell. A red-shaded grid cell in the left map indicates that all ecosystems in that grid cell will recover after 2100. In the right map a red-shaded grid cell implies that at least one ecosystem in that grid cell will recover after 2100.

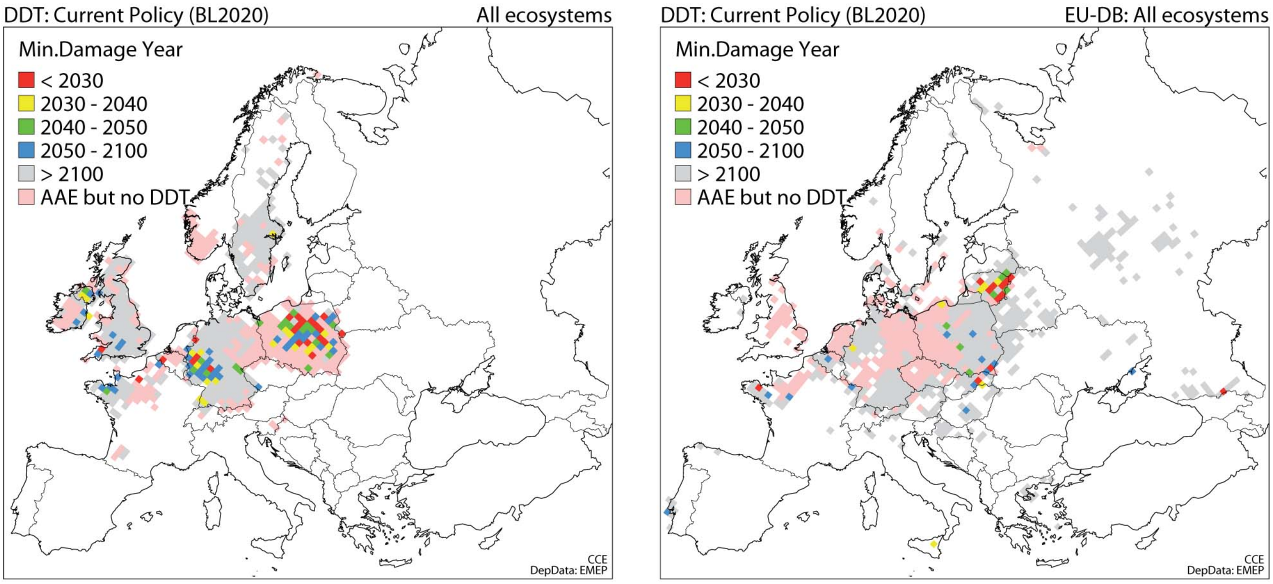

Figure 1.4 Damage Delay Times for acidification calculated using NFC data (left) and using the CCE background data base (right). A grid cell is shaded according to the earliest damage year (=violation of chemical criterion) for an ecosystem in that grid cell. Red shading implies that the earliest damage is expected before 2030.

Figure 1.4 shows the location and the earliest year by which at least one ecosystem in the respective EMEP grid cell will have its acidification chemical criterion violated.

The Damage Delay Time according to country submissions (left map in Figure 1.4) reveals that large areas in, for example, the United Kingdom and Germany show damage occurring after 2100 (grey shading). The background database computations are more pessimistic, indicating damage already occurred (pink shading) in these areas. Polish submissions indicated shorter minimum damage delay times than those computed with the background database.

Figures 1.3 and 1.4 show that ecosystems that may or may not recover are in the same grid cells in western and central Europe. This raises the question of what deposition would be required to ensure recovery of the most sensitive ecosystem in each grid cell. This deposition is called a 'target load' (Posch et al. 2003), which is lower than, or equal to the critical load of any ecosystem for which a target load exists. Target loads are calculated using dynamic models for a particular 'implementation year' (here: 2020), subject to the constraint that this deposition should yield recovery in a specified target year (> 2020). Figure 1.5 shows target loads for achieving recovery by 2030, 2050 and 2100. Target loads are low when

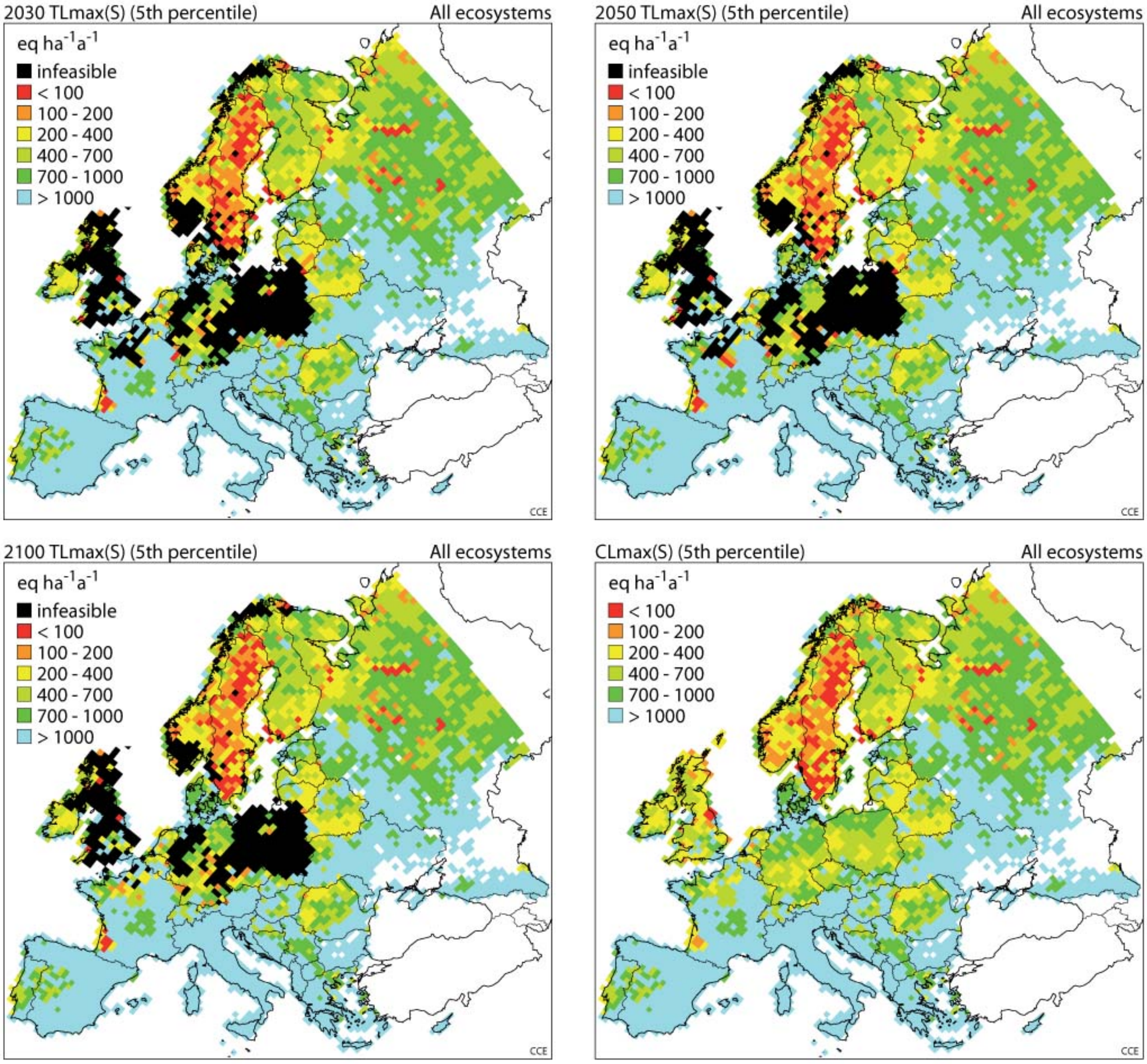

Figure 1.5 Target loads to achieve recovery of acidification in 95% of the ecosystems by 2030 (top left), 2050 (top right), 2100 (bottom left), in comparison to the critical loads map that would protect 95% of the ecosystems in the long run (bottom right).

recovery is required to occur early. For example, ecosystems in the northern central part of France require depositions below 100 eq $ha^{-1}yr^{-1}$ to achieve recovery by 2050. When recovery is only required to occur in 2100, target loads in that area can be between 400 and 700 eq $ha^{-1}yr^{-1}$. Figure 1.5 also shows that 95% of the ecosystems in a large area covering parts of the United Kingdom and central western Europe cannot recover before 2100.

1.3.3 Risk of nutrient nitrogen to biodiversity

As is shown in Figure 1.1, the risk of nutrient nitrogen can be analysed via its effects on soil chemistry (top pathway of Figure 1.1) using exceedances of computed critical loads (Figure 1.6) and dynamic modelling (Figures 1.9 and 1.10), or (bottom pathway) via exceedances of empirical critical loads (Figure 1.7) and impacts on plant species diversity (Figure 1.8).

From Figure 1.6 it can be seen that the areas with high risks to all ecosystems (red and yellow shadings) in 2000 are located in central and western Europe. Nitrogen depositions according to the PRIMES-BL2009 scenario would be hardly capable of reducing the area and magnitude of the exceedances in 2030, in comparison to 2000 (the red and yellow shaded areas are only slightly smaller). Critical loads for eutrophication have hardly been computed for surface waters.

Empirical critical loads of ecosystems were established as ranges, for classes that have been distinguished in the European Nature Information System (EUNIS). Empirical critical load ranges include values that may be higher than computed critical loads, which are mostly based on soil chemical information. This is one of the reasons why the exceedances that were calculated for critical loads for forests, vegetation and surface waters in Figure 1.7 are different from those in Figure 1.6.

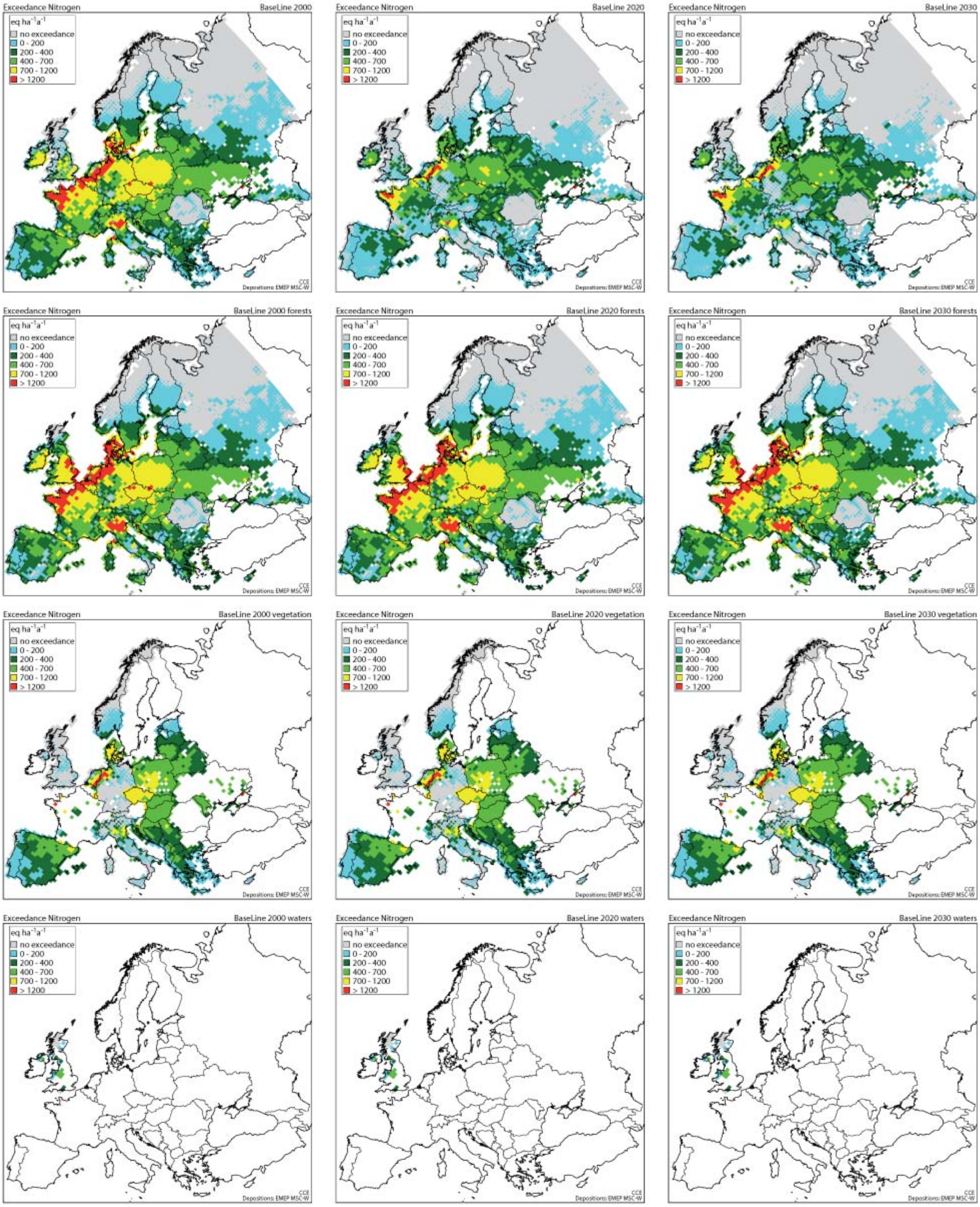

Figure 1.6 Exceedance of critical loads for eutrophication in 2000 (left), 2020 (middle) and 2030 (right) for all ecosystems (top row), forests (2nd row), vegetation (3rd row) and surface waters (bottom row). The size of a coloured grid cell is proportional to the fraction of the ecosystem area in the cell where critical loads are exceeded.

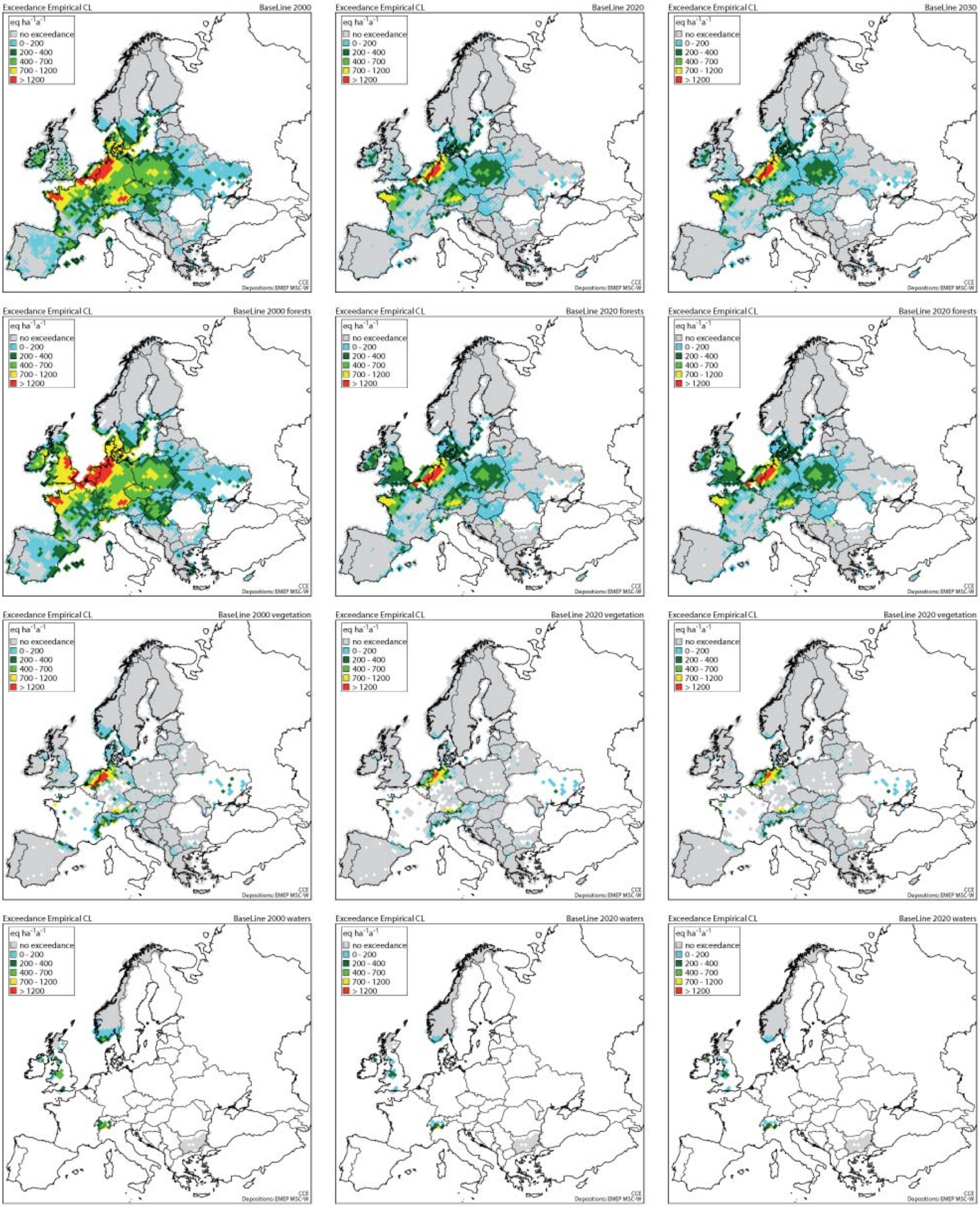

Figure 1.7 Exceedance of the minimum empirical critical loads in 2000 (left), 2020 (middle) and 2030 (right) for all ecosystems (top row), forests (2nd row), vegetation (3rd row) and surface waters (bottom row). The size of a coloured grid cell is proportional to the fraction of the ecosystem area in the cell where critical loads are exceeded.

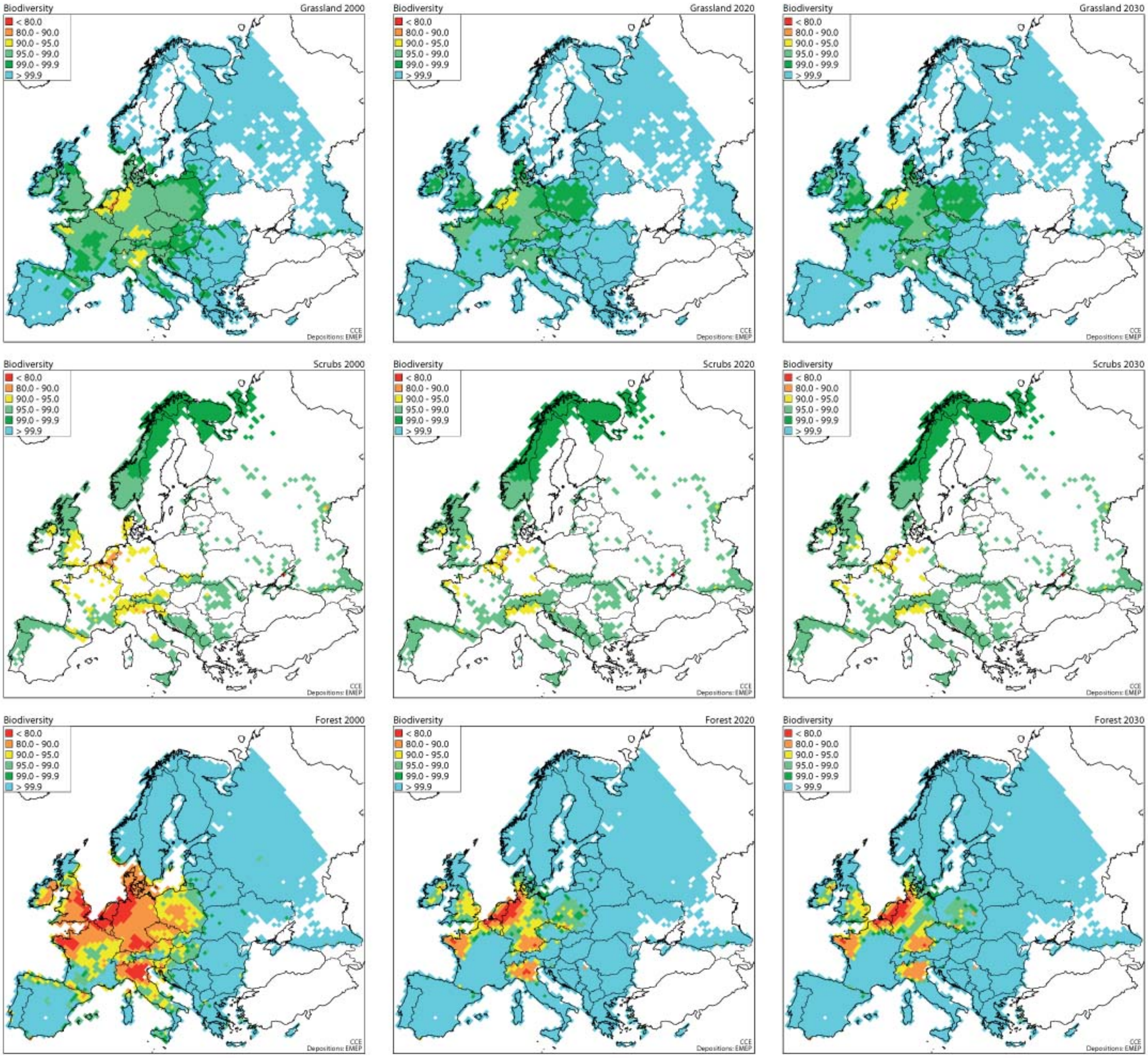

Figure 1.8 Percentage of species richness in 2000 (left), 2020 (middle) and 2030 (right) in grasslands (top), and scrubs (centre), and of species similarity in forests (bottom). Red-shaded areas indicate that the estimated biodiversity indicator percentages are lower than 80%, while green shadings indicate areas where this percentage is between 95 and 100%.

Nitrogen depositions cannot only be used for computing critical load exceedances, but also tentatively in dose–response relationships, established by Bobbink (2008), to assess impacts on species richness of (semi-natural) vegetation, and species similarity of forests (Hettelingh et al. 2008b). The result is shown in Figure 1.8.

Figure 1.8 does not show much difference between 2020 and 2030 in the pattern of the biodiversity indicator (species richness or similarity). Comparing species similarity in forests for 2030 (Figure 1.8, lower right map) with the exceedance of empirical critical loads (Figure 1.7, lower right map), showed that species similarity, for example in France, will be relatively high, although empirical critical loads will be exceeded. This could have many reasons, including the inappropriateness of extrapolating dose–response functions over Europe. To establish empirical critical loads, there is also a need to review the influence of background deposition.

1.3.4 Dynamics of the risk of nutrient nitrogen

Figure 1.9 shows the Damage Delay Time (DDT), that is, the year in which the concentration of nitrogen in the soil solution violates the critical limit for ecosystems. According to country submissions of dynamic modelling data (left map), large areas in the Netherlands and Germany include at least one ecosystem which will be damaged before 2030. Using the background database (right map) indicates that in many regions in Europe damage is already occurring, that is, the critical limit of nitrogen concentration is already being violated. The background database yields shorter Damage Delay Times, because a precautionary low critical limit of 0.3 mg N l^{-1} has been used throughout Europe.

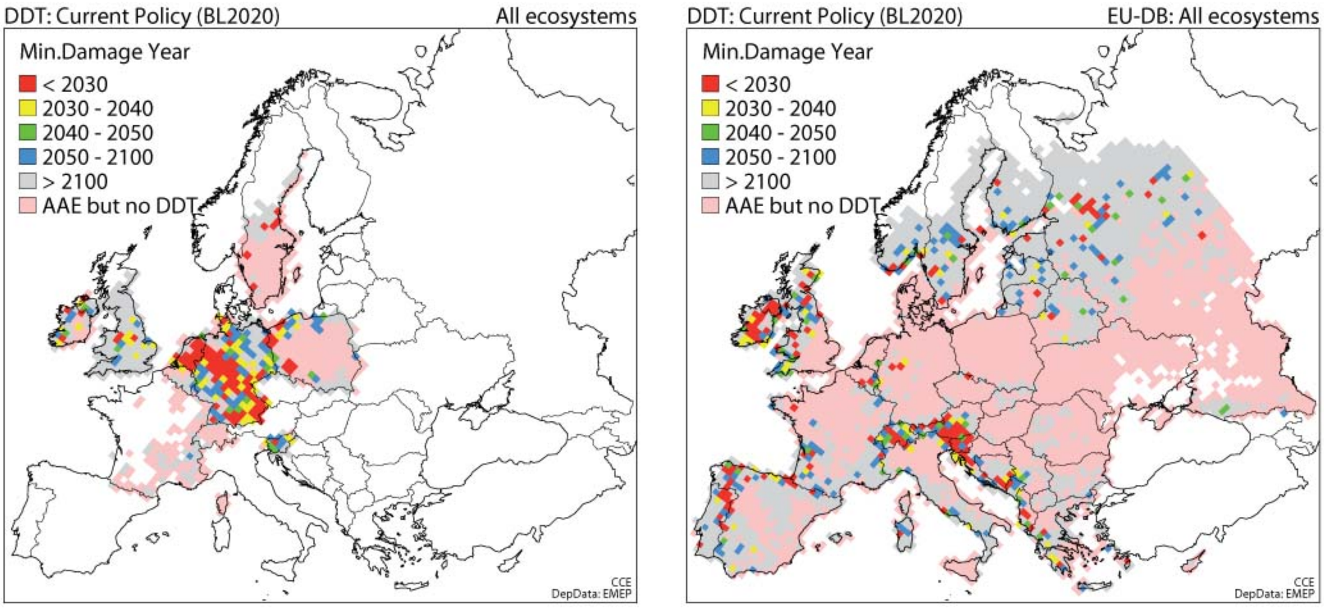

Figure 1.9 Damage Delay Times for eutrophication calculated by using data from country submissions (left), and by using CCE background data (right). A grid cell is shaded according to the earliest year of ecosystem damage in that grid cell. Red shading implies that the earliest possible damage will occur before 2030.

Figure 1.10 illustrates the nitrogen deposition that would be needed to achieve recovery of 95% of the ecosystems. In central and western Europe, rather wide areas require a nitrogen deposition between 400 and 1000 eq $ha^{-1}yr^{-1}$ by 2030, 2050 and 2100. For the Netherlands, no target loads were computed because no critical concentrations were submitted. The CCE does not use the background database for countries that submit only critical loads data and no (complete) dynamic modelling data set. Finally, note that target loads exist for all natural ecosystems in Europe, if recovery is not required before 2100.

1.4 Risk of acidification and eutrophication in Natura 2000 areas

As comprehensively described by Whitfield and Strachan (2009), the legal basis for the Natura 2000 network comes from the EU Birds Directive, which dates back to 1979, and the Habitats Directive from 1991 (EC 2009). The focus of the use of critical loads addresses effects on (plant species diversity and soil chemistry of) ecosystems, that is, for which the Habitats Directive can be more appropriate than the Birds Directive. The implementation of the Habitats Directive requires measures to be taken to maintain and restore the threatened natural habitats listed in the directive at 'Favourable Conservation Status' (FCS). The FCS of a habitat is defined as when (a) its natural range and areas within that range are stable or increasing, (b) the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist in the foreseeable future, and (c) the conservation status of its typical species is favourable, as defined in Article 1(i) which addresses FCS for a species. The last is summarised as when (i) population dynamics data indicate that the species is maintaining itself, (ii) the natural range of the species is not (likely) to be reduced in the foreseeable future, and (iii) there is a sufficiently large habitat to maintain its population on a long-term basis.

The focus of the impacts of air pollution on Natura 2000 areas assumes that the deposition of acidifying and eutrophying compounds is likely to affect FCS. An overview of approaches to assess and report nitrogen deposition impacts on conservation status in EU Member States can be found in Whitfield and Strachan (2009).

Figure 1.11 shows the exceedance of acid deposition and nitrogen deposition in Natura 2000 areas. For the latter, both computed and empirical critical loads were used. Natura 2000 areas are sensitive to acidification, in particular, in the Netherlands, Germany and Poland. The sensitivity to nitrogen deposition is more widespread, and also includes Natura 200 areas in southern France and Spain. Natura 2000 areas are not mapped for Ireland and Italy, because these countries did not submit data for their Natura 2000 areas. For the CCE, to use Natura 2000 data from the background database requires a change in current consensus between the CCE and NFCs, according to which the background database is only used for countries that did not submit any data.

Figure 1.12 shows the result from applying dynamic modelling to Natura 2000 areas, to answer the question of which deposition (target load) would be required to reach recovery in 2030. For acidity, the results are given in the right-hand maps (top), and for nutrient nitrogen (bottom). The results show that non-achievement[2] of acidity target loads is low (light blue), but evident, particularly in Germany. Non-achievement is highest in Natura 2000 areas in the south of the Netherlands, close to the border with Germany. Non-

2 Deposition which exceeds target loads is called 'non-achievement' or 'non-attainment'. The term 'exceedance' is reserved to the difference between deposition and critical loads.

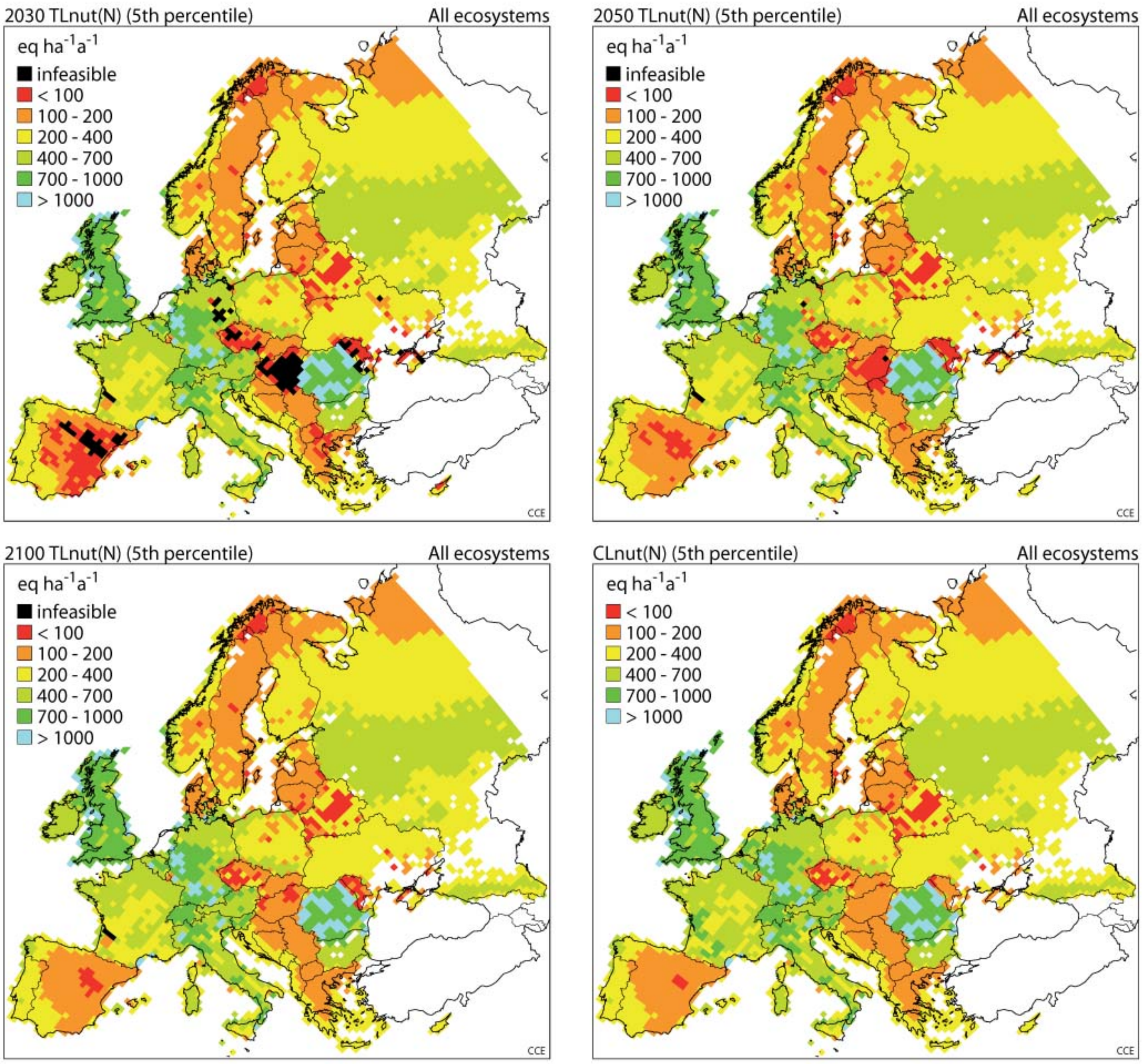

Figure 1.10 Target loads as of 2020 for achieving recovery of eutrophication in 95% of the ecosystems by 2030 (top left), 2050 (top right), 2100 (bottom left), in comparison to the critical loads map that would protect 95% of the ecosystems in the long run (bottom right).

achievement of nitrogen target loads in Natura 2000 areas is more widespread, mostly ranging from zero to 700 eq ha^{-1} yr^{-1}.

1.5 Uncertainty in assessments of impacts on biodiversity

The main aim of the critical load approach is the identification of the geographical location of an ecosystem of which the critical load is exceeded by atmospheric deposition. At the end of the day, it is the exceedance that matters, not the critical load as such. For the design of air pollution abatement policies it is important to know where (in Europe or in a country) adverse impacts can be expected to occur as a result of the dispersion of national emissions and resulting excessive regionalised depositions. Moreover, policy analysts also wish to know the magnitude of the exceedance, because it is assumed that an adverse effect may occur 'sooner' when the exceedance is higher. Therefore, when addressing ecosystem impacts, integrated assessment modellers and policy analysts are primarily interested in the likelihood of (the occurrence of) an exceedance, and its emission scenario-dependent trend.

Of course, we know that the uncertainty in exceedances depends on variables and data in the chain from emissions to depositions, and their spatial and temporal resolution. These include data and emission factors behind national emission reports, input data, meteorology and climate conditions behind atmospheric dispersion models and input data, soil-vegetation characteristics and modelling methods behind critical loads. Elements that are relevant to uncertainty analyses, in this context, have been conducted and reported under the LRTAP Convention (e.g., Hettelingh and Posch 1997, Suutari et al. 2001). An important element is that integrated

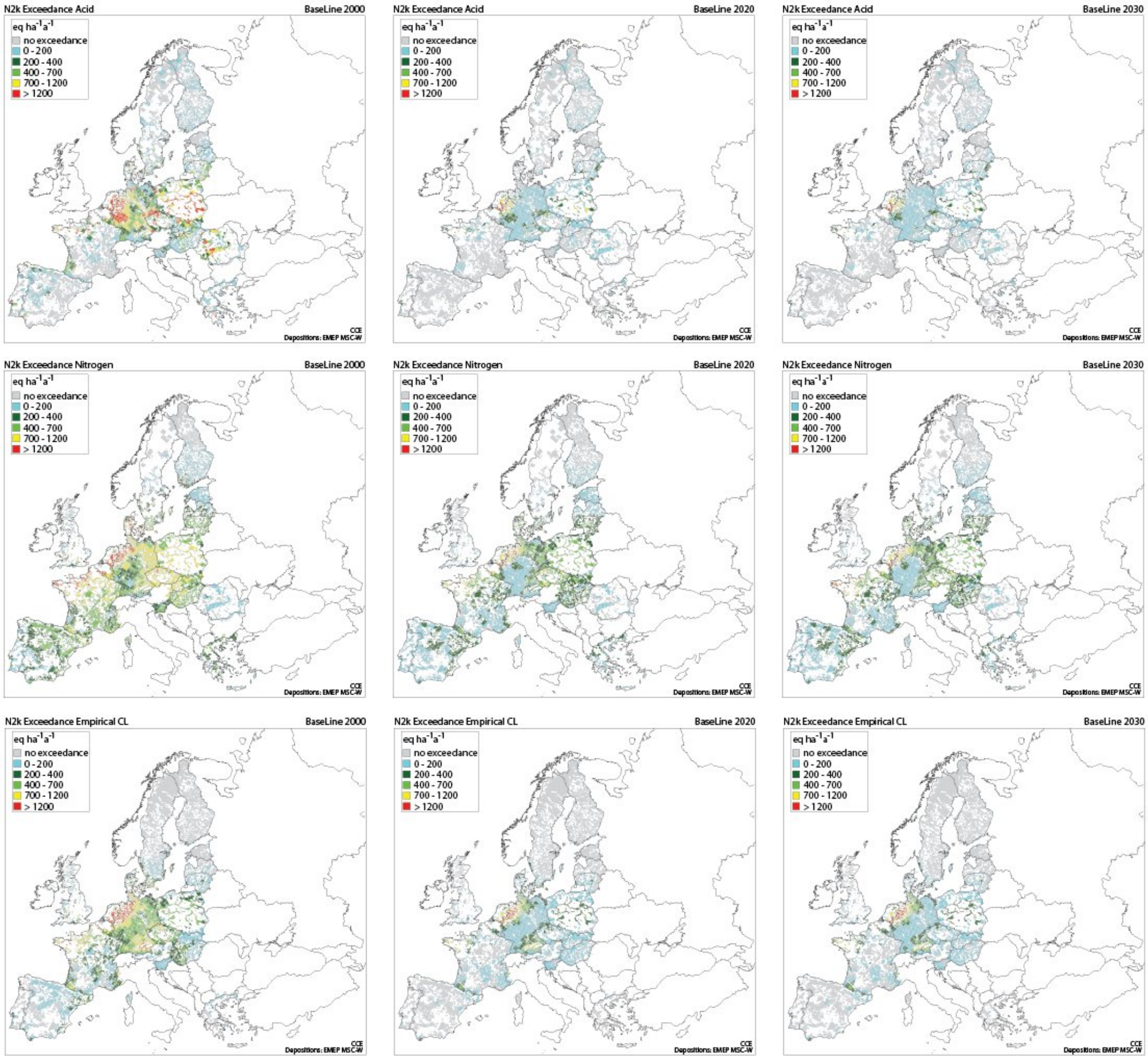

Figure 1.11 Exceedance, for Natura 2000 areas, of acidification (top), eutrophication (centre) and empirical critical loads (bottom) in 2000 (left), 2020 (centre) and 2030 (right).

assessment is concerned with scenario analysis, that is, with the change in important indicators when comparing one scenario to another. Model and data uncertainty, to a certain extent, is 'cancelled out' when comparing one scenario to another. More important is the uncertainty caused by false positives, for example, not taking climate change into account when actually one should. The impacts of climate change on target loads and delay times have recently been investigated in Reinds et al. (2009).

Ensemble Assessment of Impacts (EAI) is presented to explore the robustness of exceedances on a scale that ranges from 'exceptionally unlikely' to 'virtually certain', in analogy to the manner in which uncertainties are proposed to be addressed in the IPCC Fourth Assessment Report (IPCC 2005) (see also Hettelingh et al. 2007). The result is shown in Figure 1.13. The area where exceedance is 'virtually certain' (red shading) covered most of Europe in 2000. The area where exceedance is 'likely' will increase by 2030.

Once exceedance occurs, the Damage Delay Time is subject to the uncertainty in the chosen critical limit of the nitrogen concentration in soil solution. Figure 1.14 illustrates the variation in Damage Delay Time between reaching a critical limit of 0.3 mg N l^{-1} (left) and 3 mg N l^{-1}. The results show that areas in central Europe that include at least one ecosystem for which the Damage Delay Time is almost instantaneous, with a low critical limit (pink shaded in the left map), get some leeway until 2030, when a high critical limit is used (red shaded in the right map).

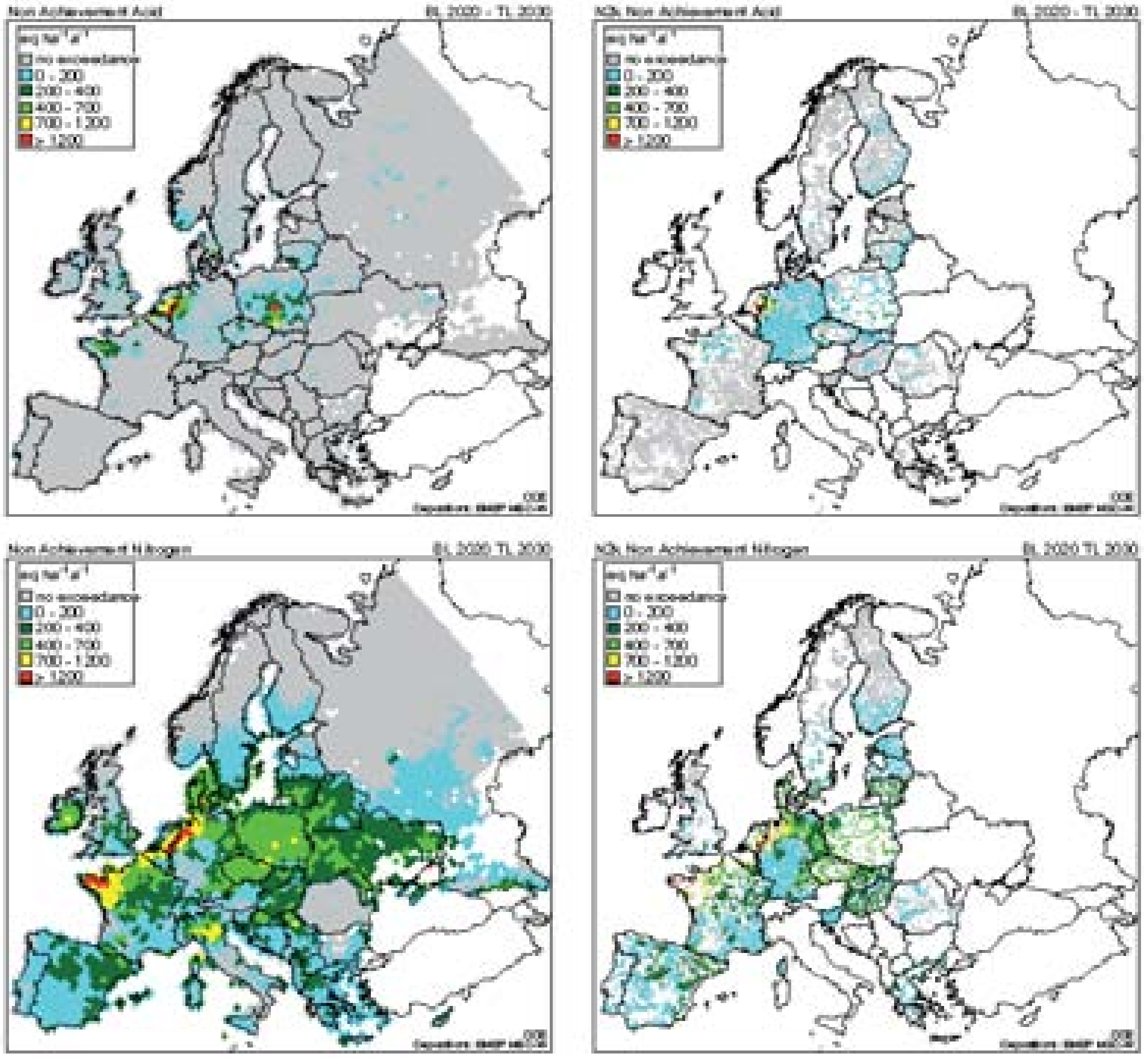

Figure 1.12 Non-achievement of 2030 target loads of acidification (top) and eutrophication (bottom) for all ecosystems (left), in comparison to Natura 2000 areas (right).

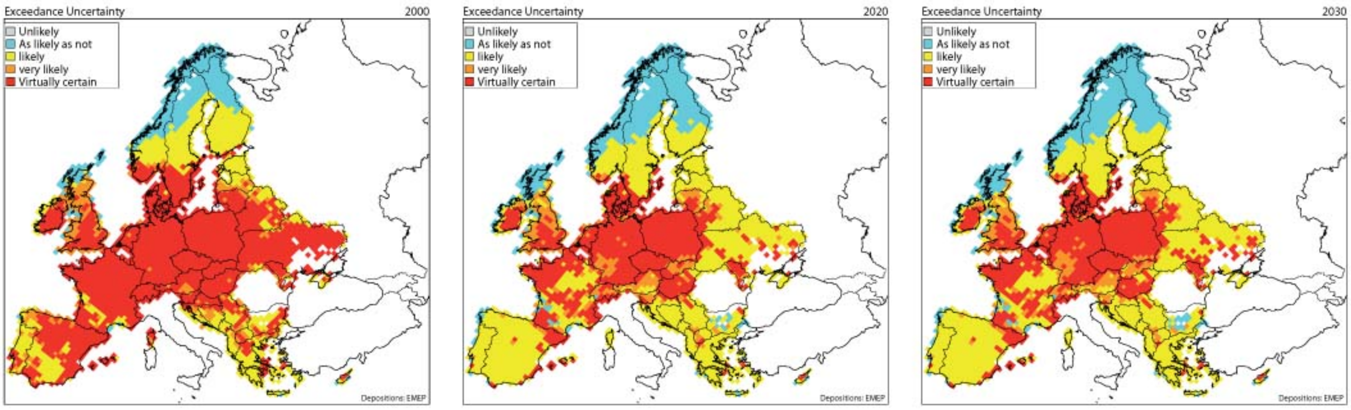

Figure 1.13 The likelihood that the Average Accumulated Exceedance (AAE) of nutrient nitrogen in an EMEP grid cell exceeds zero in 2000 (left), 2020 (middle) and 2030 (right), that is, that it contains at least one ecosystem for which the critical load of nutrient N is exceeded under the PRIMES-BL2009 scenario. Red shaded areas indicate that exceedance is 'virtually certain', whereas blue shaded areas indicate that exceedance is 'as likely as not' to occur.

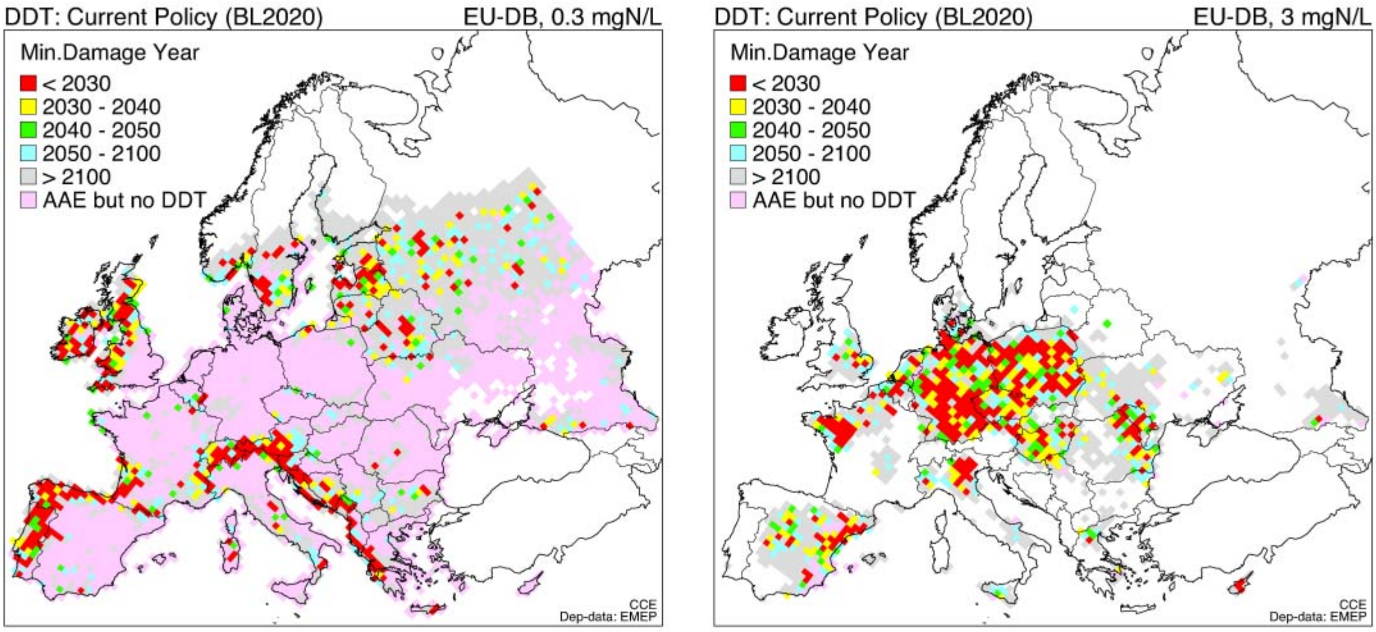

Figure 1.14 Damage Delay Times before the concentration of nutrient nitrogen in soil moisture exceeds 0.3 mg N l^{-1} (left) or 3 mg N l^{-1} (right). The chosen level of N concentration is of crucial importance to both the value of the critical load (higher with high concentrations) and time horizons of damage or recovery, and, therefore, to uncertainty.

1.6 Summary and conclusions

Depositions of sulphur and nitrogen compounds under the 2009 PRIMES BaseLine Scenario (PRIMES-BL2009) are based on patterns and magnitudes of emissions that are different from integrated assessments made in the past. This leads to a total European area at risk, under PRIMES-BL2009, which is smaller than obtained from results in 2007 under the then available ‘Current Legislation’ scenario.

The total European area at risk of acidification, under PRIMES-BL2009, is 3% by 2020 and 2030. While the total area at risk of acidification is small, recovery of some areas, in which critical loads are no longer exceeded, may take a long time for.

Eutrophication is a risk for 35 and 34% of the total European ecosystem area by 2020 and 2030, respectively. Following a method which combines an approach developed under the IPCC with ‘ensemble assessment’ (using more methods to analyse a single phenomenon), the robustness of the total European area at risk is confirmed.

The pertinent risks of eutrophication are confirmed by dynamic modelling, showing that it will be a challenge for many ecosystems to recover, and tentative assessments of the impact on plant species abundance and diversity. Computations are confirmed when the analysis is focused on Natura 2000 areas.

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2

Status of the harmonised European land cover map

Jaap Slootweg, Maximilian Posch, Allard Warrink

2.1 Introduction

During 2009 the harmonised LRTAP land cover map (Cinderby et al. 2007) was improved in collaboration with the Stockholm Environment Institute (SEI). This chapter describes these improvements, the resulting database and the implications for the European Background Database.

2.2 Update of the harmonised European land cover map

Updated land cover data was provided by the SEI on Austria, Italy, Luxembourg, Moldova, Portugal, Turkey and the Ukraine. Table 2.1 lists the European Nature Information System (EUNIS) classes and their coverage of the data set. Note that some new classes were identified, and many classes listed in De Bakker et al. (2007) don't occur in the harmonised land cover map. 'Arable land and market gardens' and 'Cultivated areas of gardens and parks' are no longer in the list, because they cannot be properly distinguished from 'Irrigated or non-irrigated arable land'.

The data provided by the SEI underwent the same technical steps as described in De Bakker et al. (2007). From these processed GIS data, a (huge) matrix (ASCII file) was exported for every European and some EECCA (Eastern Europe, the Caucasus and Central Asia) countries with a single (EUNIS) code for every 100×100 m² (i.e. 2,500 data points per *EMEP5* (5×5 km²) grid cell). Because of inaccuracies, a few data points have no value assigned to it. These '0 – pixels' were eliminated by (repeatedly) assigning them the dominant EUNIS class surrounding them. The aggregated dataset stores the number of 100×100 m² grid cells within each EMEP5 grid cell grouped according to EUNIS code.

The resolution of the database is 100 m, which makes it impossible to plot the data accurately due to the very modest size of CCE Status Reports. But the EMEP5-aggregated data set provides a resolution whereby individual pixel values do still show. Figure 2.1 shows the number of 100×100 m² grid cells of forests out of the possible 2,500.

Table 2.1 EUNIS classes and their coverage in the harmonised LRTAP land cover map.

EUNIS code	EUNIS description	Numeric code (E3)	Number of 100×100 m^2 grid cells
A	Marine habitats	1000	18
A1 or A2 without A2.5	Littoral rock and other hard substrata or Littoral sediment without Coastal saltmarshes and saline reedbeds	1102	325,070
A2.5	Coastal saltmarshes and saline reedbeds	1250	301,489
A3 or A4	Infralittoral rock and other hard substrata or Circalittoral rock and other hard substrata	1304	150,030
A3 or A4 or A5	Infralittoral rock and other hard substrata or Circalittoral rock and other hard substrata or Sublittoral rock	1349	174,407
A5	Sublittoral sediment	1500	54
B	Coastal habitats	2000	3,857,073
C1	Surface standing waters	3100	15,741,695
C1 or C2	Surface standing waters and surface running waters	3102	45,644,462
C2	Surface running waters	3200	1,906,137
C3	Littoral zone of inland surface waterbodies	3300	783,152
D1	Raised and blanket bogs	4100	7,894,818
D2 or D4	Valley mires, poor fens and transition mires or Base-rich fens and calcareous spring mires	4204	1,175,174
E1 without E1.2, E1.7, E1.8, E1.9, E1.A	Dry grasslands without Perrenial grasslands and basic steppes or Non-Mediterranean dry acid and neutral closed grassland or Non-Mediterranean dry acid and neutral closed grassland or Mediterranean dry acid and neutral closed grassland or Mediterranean dry acid and neutral open grassland	5109	509,283
E1.2	Perrenial grasslands and basic steppes	5120	65,872,200
E1.7 or E1.9	Non-Mediterranean dry acid and neutral closed grassland or Non-Mediterranean dry acid and neutral closed grassland	5179	45,482,750
E1.8 or E1.A	Mediterranean dry acid and neutral closed grassland or Mediterranean dry acid and neutral open grassland	5189	16,559,146
E2 without 2.3	Mesic grasslands without Mountain hay meadows	5209	131,234,216
E2.3	Mountain hay meadows	5230	8,527,801
E3	Seasonally wet and wet grasslands	5300	75,549,849
E4	Alpine and subalpine grasslands	5400	5,461,568
E5	Woodland fringes and clearings and tall forb stands	5500	23,862,969
F1	Tundra	6100	32,184,362
F2	Arctic, alpine and subalpine scrub	6200	6,999,695
F4	Temperate shrub heathland	6400	38,091,789
F5 or F6	Maquis, arborescent matorral and thermo-Mediterranean brushes or Garrigue	6506	12,303,313
F9	Riverine and fen scrubs	6900	359,039
G1	Broadleaved deciduous woodland	7100	77,352,602
G1.1	Riparian Salix, Alnus and Betula woodland	7101	9,232
G1.6	Fagus woodland	7106	5,750,505
G2	Broadleaved evergreen woodland	7200	9,398
G2.1	Mediterranean evergeeen [Quercus] woodland	7201	3,434,898
G3	Coniferous woodland	7300	130,382,924
G3.1	Abies and Picea woodland	7301	4,475,944
G3.2	Alpine Larix - Pinus cembra woodland	7302	3,296,956
G3.4	Pinus sylvestris woodland south of the taiga	7304	3,834,720
G3.6	Subalpine mediterranean Pinus woodland	7306	1,999,355
G4	Mixed deciduous and coniferous woodland	7400	145,907,408
G4.1	Mixed swamp woodland	7401	246,007
G4.2	Mixed taiga woodland with Betula	7402	2,686,294
G4.3	Mixed subtaiga-taiga woodland with acidiphilous Quercus	7403	348,580
G4.4	Mixed Pinus sylvestris - Betula woodland	7404	663,516
G4.6	Mixed Abies - Picea - Fagus woodland	7406	1,532,234
G4.7	Mixed Pinus sylvestris - acidiphilous Quercus woodland	7407	29,365
G4.B	Mixed mediterranean Pinus - thermophilous Quercus woodland	7411	87
G4.C	Mixed Pinus sylvestris - thermophilous Quercus woodland	7412	15,860
G4.E	Mixed mediterranean pine - evergreen oak woodland	7414	94,852
H	Inland vegetated or sparsely vegetated habitats	8000	138,420
H3	Inland cliffs, rock pavements and outcrops	8300	6,130,358
H4	Snow or ice-dominated habitats	8400	4,313,769
H5	Miscellaneous inland habitats with very sparse or no vegetation	8500	86,018,750
II	Irrigated arable land	9100	519,278,234
IN	Non-irrigated arable land	9200	1,278,061
J	Constructed, industrial and other artificial habitats	10000	24,548,627
Y	Unknown	25000	1,427,273

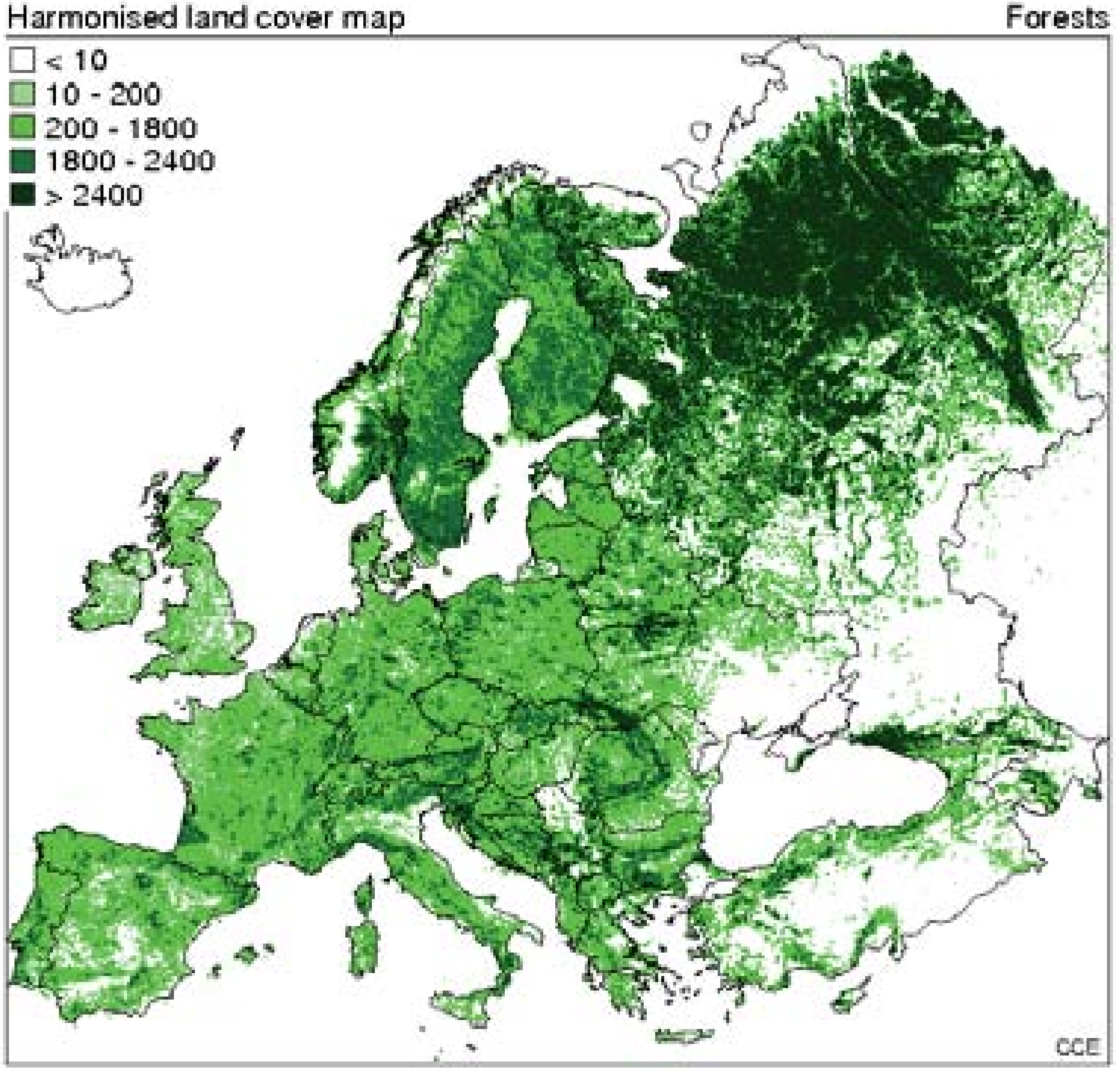

Figure 2.1 The number 100×100 m² grid cells of forest in each 5×5 km² grid cell.

2.3 European Background Database of critical loads

To complete Europe-wide critical load maps, the CCE uses the European Background Database (EU-DB) for gap-filling. This database uses the harmonised land cover database as vegetation data input. With the update of the land cover database, the critical loads of EU-DB were also updated. Most striking is the extension of the database to the east as can be seen in Figure 2.2, which shows the 5th percentile of CLnut(N) in the 'before' and the 'after' database. The figure further demonstrates that other changes are very small. The picky reader will also spot minor changes in Poland and Germany, for which land cover was not updated. These are caused by improvements in other input data of EU-DB.

2.4 Conclusions

An updated land cover database is available for Austria, Italy, Luxembourg, Moldova, Portugal, Turkey and the Ukraine. The updated land cover implies an update of the European Background Database of critical loads. For further use the database was also aggregated to a 5×5 km² grid, which was made available to EMEP and the EC4MACS consortium, primarily for dispersion modelling.

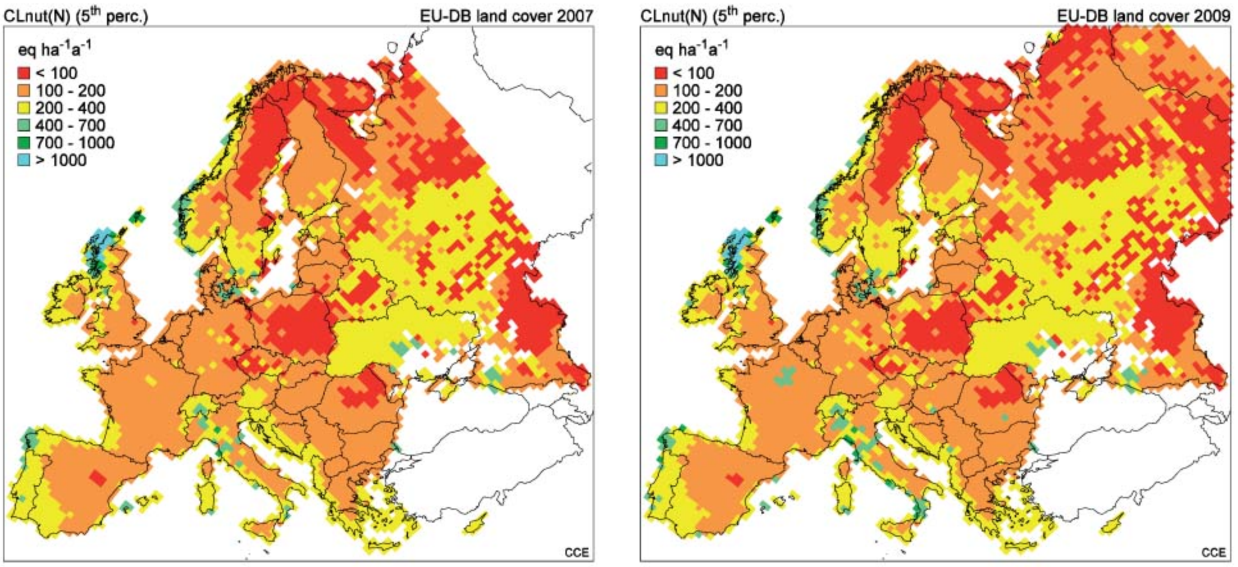

Figure 2.2 5th percentiles of CLnut(N) of the European background database before (left) and after (right) the update of the land cover database.

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3

Modelling recovery from soil acidification in European forests under climate change

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3.1 Introduction

It is now well established that the climate is changing and will continue to do so in the future (IPCC 2007). Recent studies have shown that climate change has accelerated over the last decades, consistent with the higher climate change projections (Rahmstorf et al. 2007). Nevertheless, until now, all Europe-wide studies on critical loads and dynamic modelling of acidification in forests have assumed a constant climate. Since European forest ecosystems have many functions, related to biodiversity, forest products, groundwater protection and carbon sequestration (Metzger et al. 2005), it is crucial to know the extent to which current air quality legislation protects them from acidification and eutrophication under climate change.

Climate change could both increase and decrease levels of acidity and N in forest soils. For example, higher temperatures lead to higher base cation weathering rates that buffer acidity. Also, increased drought stress leads to higher concentrations in soil solution and may lead to slower tree growth rates, causing less N removal from the soil. To gain insight into the combined effects of climate change (temperature, precipitation and radiation) and deposition reductions in N and S, we applied the Very Simple Dynamic (VSD) model (Posch and Reinds 2009), to a large number of forest receptors across Europe. We evaluated the resulting simulated soil solution chemistry relative to widely used critical limits (molar Al/Bc = 1 and ANC = 0 in soil solution). The relative importance of deposition reductions and climate change on modelled soil solution chemistry, for the 1990-2050 period, was assessed for combinations of two deposition and two climate scenarios (scenario analysis). Furthermore, the potential to recover within a specified period of time (target loads) and the time delay in recovery from an acidified state (recovery times) were evaluated for these scenario combinations.

3.2 Methods and data

Modelling and scenarios

The Very Simple Dynamic (VSD) model (Posch and Reinds 2009) was used to simulate the combined effects of acidification and climate change. This model has been widely used on a national and European level to simulate the acidification of soils in support of European air pollution control policies (Hettelingh et al. 2007). The VSD model simulates soil solution chemistry and soil nitrogen pools for natural and semi-natural ecosystems. The model consists of a set of mass balance equations that describe the soil input-output relationships of ions, and a set of equations that describe the rate-limited and equilibrium soil processes. A detailed description of the VSD model and its functionality is provided in Posch and Reinds (2009). For this study, the VSD model was adapted to incorporate effects of climate change by making model inputs and chemical processes temperature dependent. Temperature dependence was modelled for the bicarbonate equilibrium, H-Al equilibrium, base cation weathering, nutrient uptake and precipitation surplus. Details on how this temperature dependence was modelled, are provided in Reinds et al. (2009).

A scenario analysis was carried out using two deposition scenarios and two climate scenarios. For 2020, the current legislation (CLE) and the maximum (technically) feasible reductions (MFR) emission scenarios were used (Amann et al. 2007). From 2020 onwards, deposition was assumed constant (Figure 3.1).

Time series of temperature, precipitation and cloudiness were obtained from a high resolution European data base (Mitchell et al. 2004). Projections for 2001 to 2100 stem from the HADCM3 GCM model, and two sets of climate change scenarios were used, based on the IPCC SRES A1 and B2 scenarios (Nakícenovíc et al. 2001). The A1 scenario is based

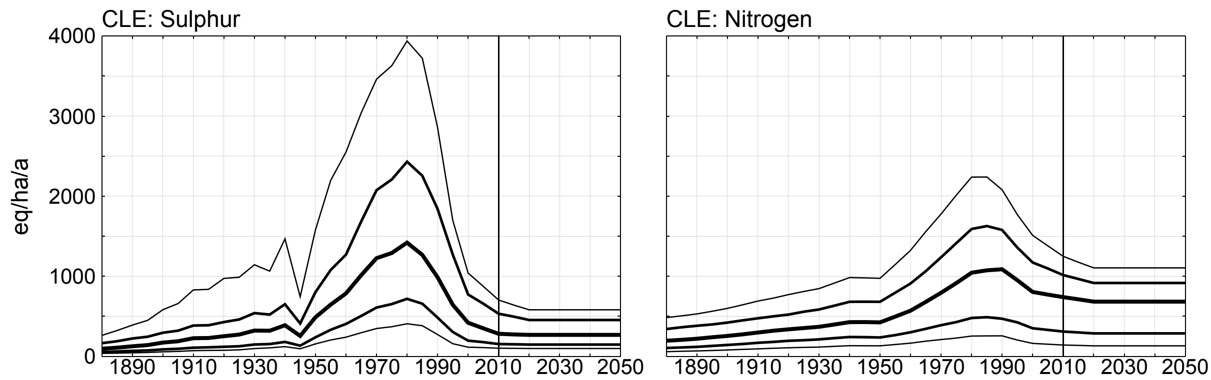

Figure 3.1 Temporal development of the 10th (lowest), 25th , 50th, 75th and 90th (highest curve) percentiles of the sulphur (left) and total nitrogen (right) deposition on the forest areas in Europe (after 2010 the CLE scenario is shown).

Table 3.1 Median values of $\log_{10}KAl_{ox}$, $\log_{10}KHBc$ and $\log_{10}KAlBc$for soils with the texture classes sand, clay and heavy clay, based on a calibration of the VSD model on 180 intensively monitored forest plots (Reinds et al. 2008b).

	$\log_{10}KAl_{ox}$	$\log_{10}KHBc$	$\log_{10}KAlBc$
Sand	8.87	3.39	0.528
Clay	8.58	3.79	−0.65
Heavy clay	8.41	3.54	−0.67

Table 3.2 Median values of base cation weathering for 6 weathering rate classes based on a calibration of the VSD model on 180 intensive monitored forest plots. Each weathering rate class is a combination of soil texture (5 classes) and parent material type (acid, intermediate, basic).

Weathering rate class	1	2	3	4	5	6
BCwe (eq.ha⁻¹m⁻¹.yr⁻¹)	730	1050[1]	1360	3100[2]	3100	1100

[1] No plots with weathering rate class 2 were in the set of calibration plot; the value is the average of classes 1 and 3.
[2] Values for weathering rate class 4 were set equal to those obtained for class 5.

on a future world with globalisation and rapid economic growth, low population growth, and the rapid introduction of new and more efficient technologies everywhere. The B2 scenario mimics a world in which the emphasis is on local solutions to economic, social, and environmental sustainability (Strengers et al. 2005). A reference climate set was created by computing the mean monthly temperature, precipitation and cloudiness for the 1961-1990 period. Future CO_2 air concentrations consistent with the above mentioned scenarios were obtained from Carter (2007). Next to the scenario analysis, target loads and delay times were computed; details on the methodology are provided in Posch and Reinds (2009) and Reinds et al. (2009).

Geographical data

Input data for the VSD model simulations consist of spatial information describing climatic variables, deposition of base cations, S and N, weathering of base cations, nutrient uptake, N transformations and soil properties, such as carbon pool and cation exchange capacity. These data were derived from combining maps and databases on soils, land cover and forest growth regions. A map with computational units (receptors) was created by overlaying maps on land cover, soils and forest growth (Reinds et al. 2008a). The resulting map with forest-soil combinations for Europe contains 414,000 units, covering at least 1 km², that were used in this study. Base cation deposition for Europe was taken from simulations with an atmospheric dispersion model for base cations (Van Loon et al. 2005). Historical N and S deposition data were taken from Schöpp et al.(2003), and scenarios of N and S deposition were obtained from the Eulerian atmospheric transport model of EMEP/MSC-W (Tarrasón et al. 2007).

Model parameterisation and initialisation

Values for weathering rates, cation exchange constants, the H-Al equilibrium constant, N-immobilisation fractions, and denitrification fractions, were obtained from a calibration of the VSD model on 180 intensively monitored sites in Europe (Reinds et al. 2008b). Results for cation exchange constants and the H-Al equilibrium constant were categorised by soil texture. Results are given in Table 3.1.

Weathering rates of the root zone were calibrated for six weathering rate classes (Table 3.2). It should be noted that the calibrated weathering rate of class 1 (sandy soils on acidic materials) is substantially higher than the default value given in (UBA 2004). Probably, the value in UBA (2004) is valid for very acid pure sandy soils, whereas most plots with texture class 1 in the set of calibration plots had a richer parent material.

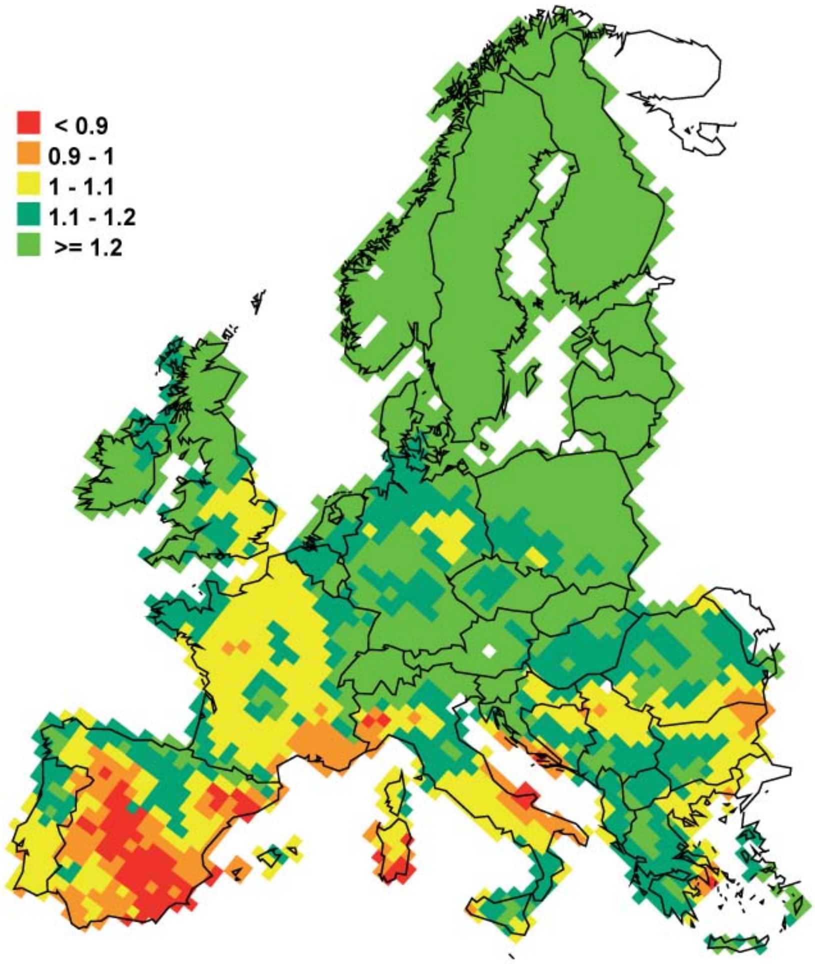

Figure 3.2 Growth under the A1 scenario, by 2050, divided by the reference growth from 1961 to 1990.

In the VSD model, N immobilisation is modelled as a function of the actual C/N ratio plus a constant immobilisation (Posch and Reinds, 2009). Because a meaningful simultaneous calibration of both processes was not feasible, we modelled N immobilisation according to Reinds et al. (2008b):

$$N_{im} = a \cdot \left(N_{dep} - N_{upt}\right) \quad (1)$$

where N_{im} is the nitrogen immobilised, N_{upt} is the net growth uptake, and *a* is an empirical constant. Parameter *a* was calibrated as a function of the measured C/N ratio in the top soil. Calibration showed that most of the incoming N is immobilised: median calibrated values for *a* were 0.83 for high C/N ratios (> 30), 0.76 for intermediate C/N ratios (20-30), and 0.71 for low C/N ratios (< 20). These fractions are the current immobilisation fractions. Some studies have shown that soils may become nitrogen saturated and, subsequently, N immobilisation will diminish (Aber et al. 1989, Aber et al. 2003). Thus, in this study, future N immobilisation in high N deposition areas may have been overestimated.

The VSD model was initialised using the calibrated model parameters. The model was run for the period from 1990 to 2050, to assess the effects of changes in deposition and climate on soil solution chemistry in Europe, for the two deposition scenarios (CLE and MFR) and the three climate scenarios (A1, B2, reference data from 1961 to 1990 (His)).

3.3 Results

Growth and uptake

To obtain insight into the effects of future climate on forest production, the combined effect of temperature, respiration and drought was computed for 2050, for both future climate scenarios, using the CLE scenario for N deposition. Under the A1 scenario, growth decreases by about 5% of the forested area compared to the reference period (especially in southern Europe), mainly caused by increased drought stress (Figure 3.2). In central and western Europe, growth is expected to increase by 10 to 20% and in Fennoscandia by more than 20%.

The difference between the A1 and B2 scenario is small. The higher temperatures in the A1 scenario, on average, lead to a 6% higher growth than in the B2 scenario. The decrease in N deposition under the MFR scenario somewhat reduces growth compared to that under the CLE scenario: the mean difference over all plots is limited to 10%. The higher growth in the A1 scenario mostly leads to a somewhat higher uptake of nutrients, and may thus cause lower nitrate concentrations in the soil, both for the CLE and the MFR deposition scenarios.

Scenario analyses: impact on soil solution chemistry

Future soil chemistry strongly depends on the scenario that is used (Table 3.3). For both the Al/Bc and ANC criteria, the area in which critical limits are exceeded is smaller under the MFR scenario. Under the CLE scenario, the area where ANC < 0 will continue to decrease up to the year 2020, but will remain

Table 3.3 The forested area (in %) where Al/Bc > 1 mol.mol^{-1}, ANC < 0 and [NO_3] > 0.3 and 3 mg N.l^{-1} for two emission scenarios (CLE and MFR) and two climate scenarios (A1, B2), as well as for the reference climate data from 1961 to 1990 (His), for the years as indicated.

Al/Bc	His,CLE	A1,CLE	B2,CLE	A1,MFR	B2,MFR
1990	4.17	4.17	4.17	4.17	4.17
2010	3.34	3.26	3.27	3.26	3.27
2030	2.60	2.37	2.40	1.25	1.27
2050	1.95	1.64	1.71	0.68	0.76
ANC					
1990	6.76	6.76	6.76	6.76	6.76
2010	5.31	5.25	5.25	5.25	5.25
2030	5.14	5.07	5.05	4.01	4.00
2050	5.14	4.97	4.97	3.53	3.56
NO_3 (0.3 mgN.l^{-1})					
1990	54.85	54.85	54.85	54.85	54.85
2010	43.69	40.00	39.73	40.00	39.73
2030	40.32	35.08	34.60	15.20	14.65
2050	40.86	33.31	33.06	12.39	12.20
NO_3 (3 mgN.l^{-1})					
1990	11.42	11.42	11.42	11.42	11.42
2010	2.29	1.86	1.80	1.86	1.80
2030	0.89	0.46	0.44	0.00	0.00
2050	1.01	0.22	0.21	0.00	0.00

almost constant at about 5% of the European forested area afterwards. The area with Al/BC > 1 will continue to decrease, over time, and will be < 1% by 2050, under the MFR scenario. Differences in future climate had little effect on the simulated trends in soil acidity. Only the trend in the area with Al/Bc > 1 was somewhat affected by climate: under the A1 scenario this area would become smaller than when using the B2 scenario, because of higher weathering rates and thus higher Bc concentrations. The mitigating effect on soil acidity of higher N uptake would be largely compensated by the higher uptake of base cations. Both climate scenarios lead to lower exceedances, compared to current climate (His), most likely because weathering increases due to higher temperatures. The ANC criterion is more stringent then the Al/Bc criterion (Table 3.3). This is consistent with a study on critical loads in Eurasia, that also showed the ANC=0 criterion to be more stringent (Reinds et al. 2008a). For nitrogen concentrations in soil solution, the critical limit for vegetation varies within Europe. According to De Vries et al. (2007), values vary between 0.3 mgN.l^{-1} for regions with sensitive vegetations (Scandinavia) to about 3 mgN.l^{-1} for western Europe; we applied both these limits to the whole of Europe to assess the range of potential N effects. Results show that, in the future, the area where N limits are exceeded diminishes, due to climate change, through enhanced uptake of N and as a result of lower N deposition, especially under the MFR scenario. The size of the area with exceedances strongly depends on the criterion chosen: the higher criterion leads to non-exceedance by 2030, under all scenarios, but when the strict N criterion is used, the area exceeded is still about 12% – even under the most favourable scenario.

The effect of the climate scenarios on nitrate concentrations and pH is limited, and the MFR scenario is much more effective in reducing nitrate concentrations than the CLE scenario (Figure 3.3). Under the MFR scenario, concentrations of N exceeding 3 mgN.l^{-1} do not occur by 2050, whereas they persist under the CLE scenario.

Target loads

Target loads were computed for both the Al/Bc =1 and the ANC=0 criterion, using the target years of 2030 and 2050. Under both climate scenarios, most (89–96%) of the European forested area is protected by maintaining current deposition or reducing it to critical loads (Table 3.4). For the remaining area, target loads exist almost everywhere; only for Al/Bc=1 in 0.05% of the area (about 54,000 ha) even zero deposition is not sufficient to attain the criterion by 2030.

The ANC=0 critical criterion is stricter than Al/Bc=1; for more than 11% of the forested area a target load smaller than the critical load is required as opposed to between 3 and 4% for Al/Bc=1. Target loads can be found in strongly acidified areas, such as in Poland, the Czech Republic and the Netherlands (Figure 3.4). Using ANC=0, this area also includes large parts of Germany, eastern United Kingdom and central Europe. In the rest of Europe, a reduction in deposition to the critical load is sufficient for attaining the desired soil chemical state in (or before) the target year.

Recovery delay

Recovery times (RT) were computed for the CLE and MFR scenarios, setting the year from which deposition levels remain constant at 2020. Using Al/Bc=1, more than 95% of European forests will be safe (i.e. the criterion is not violated) by 2030, even under the CLE scenario (see Table 3.4). In central Europe, a substantial percentage of forest areas have an Al/Bc > 1 by 2020, and a RT exists. In some areas, particularly in the Netherlands, Poland, the north of the Czech Republic and south-eastern Germany, no recovery occurs under the CLE scenario, because the critical load would still be exceeded; for these areas, there is no RT. Recovery under the MFR scenario is much more rapid than under the CLE scenario

Table 3.4 Percentage of forest area for the three target load cases for two criteria (Al/Bc=1, ANC=0) and two target years (2030 and 2050).

Cases	Al/Bc=1		ANC=0	
	2030	2050	2030	2050
1: TL=CL	96.00	96.73	88.65	88.72
2: TL < CL	3.95	3.27	11.35	11.28
3: TL infeasible	0.05	0	0	0

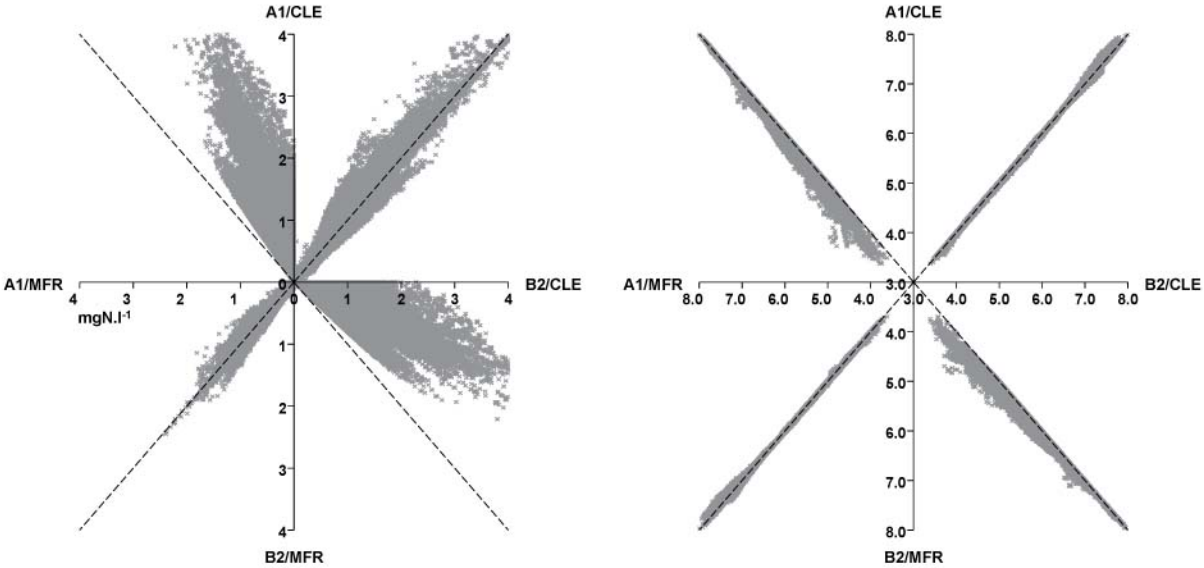

Figure 3.3 Simulated nitrate concentrations (mgN.l-1left) and pH (right) for the climate (A1 and B2) and deposition (CLE and MFR) scenarios, for 2050.

(Figure 3.5). While under the latter it often takes 40 to 50 years for ecosystems to recover, most systems recover within 20 years under the MFR scenario.

3.4 Conclusions

Recovery of most acidified forest soils in Europe can be achieved for both soil chemical criteria when depositions would be reduced, using current available technologies. Climate change reduces the area where chemical indicators were exceeded, because of higher weathering rates owing to higher temperatures and because of higher uptake of nitrogen. However, differences between the A1 and B2 scenario and the climate, for the 1970-2000 period, were small. Areas where critical nitrogen concentrations are exceeded can be reduced by implementing the MFR scenario. Changes in future climate further reduce this area, because of higher N uptake through enhanced growth. In principal, recovery of most acidified soils in Europe would be possible by 2050. Moreover, for most acidified areas in central Europe, target loads can be found that return the soils to the desired chemical state by 2050. For about 30% of the receptors where TL<CL, however, the target load is lower than the deposition under the MFR scenario, indicating that no recovery can be achieved by 2050 with current emission reduction techniques. The speed of recovery under the MFR scenario is much more rapid (mostly within 20 years) than under the CLE scenario, where recovery delay for Al/Bc takes 20 to 50 years or more. Results are consistent with a study carried out in Sweden, where a detailed multi-layer model was applied to about 600 forest sites (Sverdrup et al. 2005); the study concluded that the Gothenburg Protocol (CLE scenario) would lead to significant improvement in the acidification of forest soils in Sweden. Similarly, measurements and simulations at a highly acidified forest site in the Czech Republic showed that a fast recovery of pH and Al/Bc in the topsoil can be achieved with current emission reductions (Navrátil et al. 2007).

We modified the VSD model in such a way that formerly fixed processes became climate-dependent. However, the model was not explicitly developed to include (all) climate-sensitive processes and, therefore, final conclusions on the impact of climate change on soil acidification cannot be drawn from this study. For example, since the model assumes a closed nutrient cycling, we could not include effects of climate change either on litter composition and organic matter mineralisation, or on future N immobilisation. To account for such effects, the VSD model is currently extended to simulate also dynamics in carbon and nitrogen pools as a function of these processes.

All model results indicated that emission reductions are more important for the recovery of forest ecosystems than potential effects of climate change. Moreover, climate change has only limited effects on soil chemistry. There was a small but positive, synergistic effect of climate change, consistent with earlier studies (Wright and Jenkins 2001). Changes in climate will also affect tree species distribution over Europe (Leemans and Eickhout 2004); this effect was not taken into

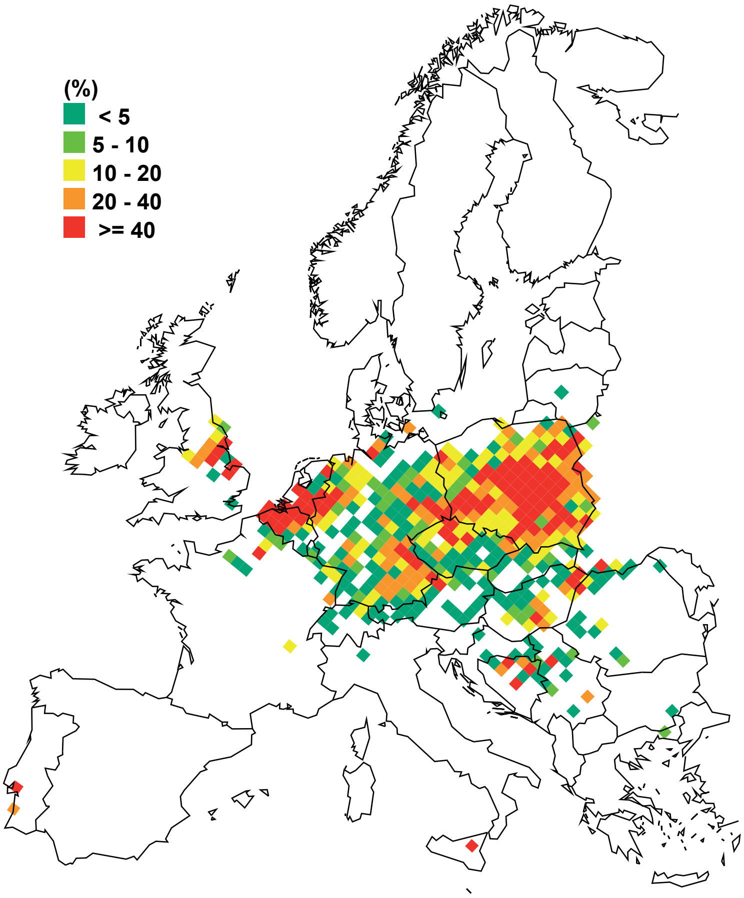

Figure 3.4 Percentage of forested area per EMEP grid cell with target loads < critical load for 2050 for Al/Bc=1.

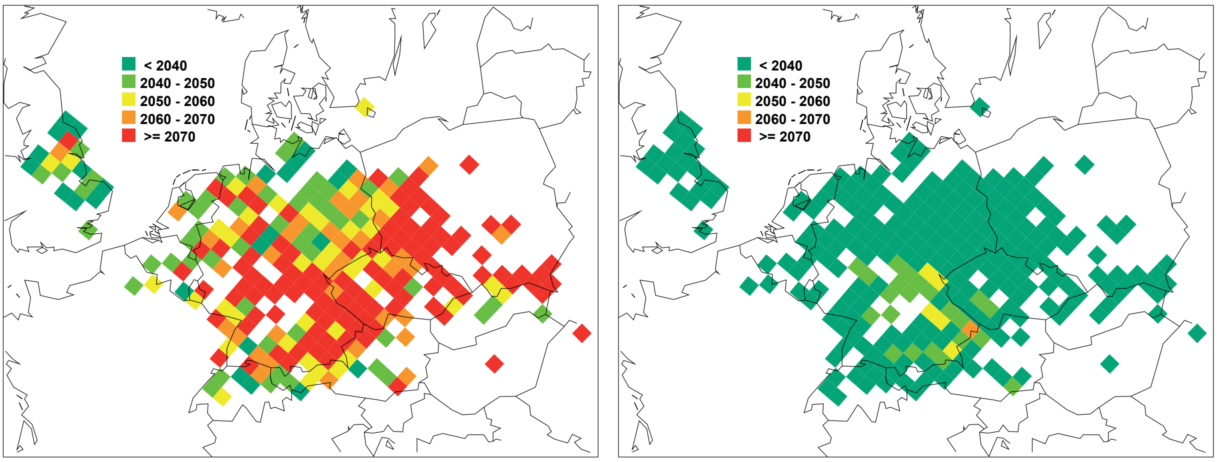

Figure 3.5 Average Recovery Times in central Europe for Al/Bc=1 per EMEP grid cell, under the CLE scenario (left) and the MFR scenario (right), for those receptors that have a recovery time.

account. Such land cover changes may influence the recovery of soils, if the change would occur in species with very different growth rates than current species.

Earlier studies found a strong synergy in simultaneously mitigating air pollution and climate change. This is the case because many of the traditional air pollutants and greenhouse gases have common sources, offering a cost-effective potential of concurrent improvements for both problems (Syri et al. 2001, Swart et al. 2004). Although this study applied a model with only a simple parameterisation of the climatic factors, we think that it indicated the appropriate directions and magnitude of the interactive effects.

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4

Quantifying relationships between N deposition and impacts on forest ecosystem services

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4.1 Introduction

The concept of ecosystem services

An increasing amount of information is being collected on the ecological and socio-economic value of goods and services provided by natural and semi-natural ecosystems, and their economic value. Earliest literature on valuation dates back to the mid-1960s and early 1970s (e.g. Helliwell 1969). More recently, there has been an almost exponential growth in publications on the benefits of natural ecosystems to human society (e.g., see Costanza et al. 1997, De Groot et al. 2002).

Inspired by De Groot et al. (2002), who grouped ecosystem services into four primary functions, the Millennium Ecosystem Assessment made a distinction between Provisioning services, Regulating services, Supporting services and Cultural services (Reid et al. 2005). Provisioning services are related to the products obtained from ecosystems, specifically food, fiber and wood/fuel, but also include the provision of, for example, fresh water. Regulating services refer to the regulation of, for instance, climate, flood control, water quality and disease control. These services are related to the impact of ecosystems on greenhouse gas exchange (climate), and on the buffering and filtering capacity of the soil affecting water and element fluxes. Supporting services relate to the capacity of natural and semi-natural ecosystems to regulate essential ecological processes and life support systems through biogeochemical cycles and other biospheric processes. These functions are indirectly related to the provisioning and regulatory functions as they affect many services that are of direct and indirect benefit to humans (such as clean air, water and soil).

Forest ecosystems provide a full suite of services that are vital to human health and livelihood. Important forest ecosystem services are the provision of an adequate soil and water quality, watershed services, carbon storage, and a habitat for a diversity of plants and wildlife. These services are also the basis for the 'Criteria and indicators for sustainable forest management' as adopted by the Ministerial Conference on the Protection of Forests in Europe. Healthy forests provide a host of watershed services, including water purification, groundwater and surface flow regulation, and erosion control. With their considerable potential for C sequestration, forests also constitute one of the most important elements of the global C cycle. Forests in Europe presently serve as an important C sink, mitigating approximately 10% of European induced CO_2 emissions. Furthermore, biodiversity levels are high in forest ecosystems, as these provide habitats for a wide range of animal and plant species. Forests will only continue to provide their indispensable ecological and economical benefits on the condition that they remain healthy, stable and sustainably managed.

Qualitative link between N deposition and ecosystem services in interaction with climate and other air pollutants

Nitrogen deposition affects, among other things, the following ecosystem services:

- Diversity of plant species in ecosystems (impact on habitat function for wild plants, reducing biological and genetic diversity; provisioning service).
- Primary production (provisioning service of wood/fiber and supporting services, as photosynthesis produces oxygen necessary for most living organisms) and carbon sequestration (climate regulating service).
- Water quality: acidity and leaching of nitrogen, aluminium and metals to groundwater and surface water (regulating service, i.e. clean soil and water).

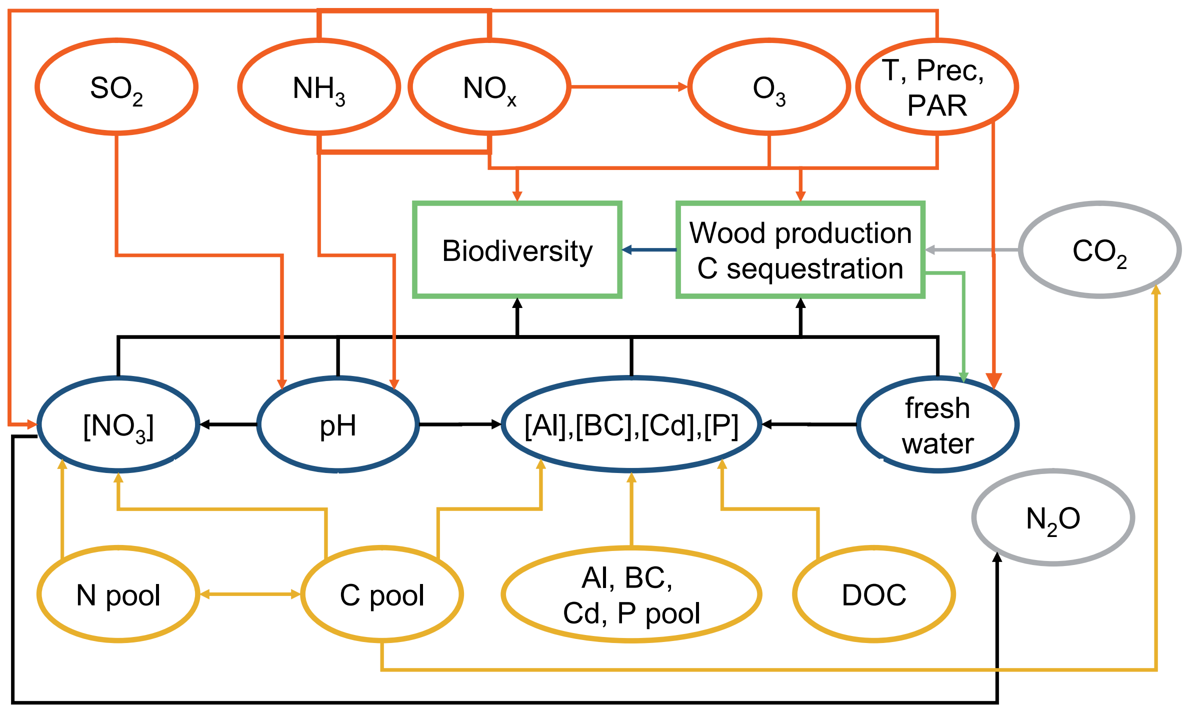

Figure 4.1 Relationships between air quality (red), biodiversity and forest growth (green), water quality (blue), soil quality (yellow) and greenhouse gas emissions (grey), accounting for climate change (red)(BC = base cations).

- Water (quantity): hydrological budgets and groundwater recharge (water regulating service).

Figure 4.1 shows the relationships between reactive N and ecosystem services, including the interaction with climate change effects and other air pollutants, such as sulphur (acidity) and ozone.

More general, forest ecosystem services are affected by climate change in interaction with changes in air quality, which may cause a loss to the future contribution of forests as C sinks and to the compositional, structural and functional biodiversity of forest ecosystems. The pressures to be considered in relation to ecosystem services are thus:

- climatic parameters related to water stress such as drought, temperature stress such as late frost, and extreme meteorological events such as hail and windstorms;
- air quality parameters, including deposition of N, and acidity and ozone exposure.

More information on the causal link between nitrogen deposition and ecosystem services is given in Table 4.1.

4.2 Quantification of the link between nitrogen deposition and ecosystem services

General approach

A quantitative link between N deposition and ecosystem services was made in view of:

- Biodiversity: linking (i) excessive N deposition to critical N loads, and (ii) excessive NH_3 air concentrations to critical limits for lichens and herbaceous plants
- Water quality regulation: linking excessive NO_3, Al and Cd concentrations from leachate in groundwater and surface water to critical limits
- Soil quality regulation: depletion of the pools of BC and Al
- Carbon sequestration and greenhouse gas emissions

We made runs with the VSD model for Europe for the 2000-2050 period, using EMEP-based deposition data and NO_x, NH_3 and SO_2. The following scenarios were evaluated:

- Using the 1980 deposition between 2000 and 2050 (1980 deposition)
- Deposition according to the Gothenburg protocol between 2000 and 2010 and constant thereafter (current legislation)
- We started with deposition data on 2000 and changed N deposition linearly to critical N loads for biodiversity ($CL_{nut}(N)$) and related ($CL_{max}(S)$) inputs towards 2010, and kept these values constant for the years thereafter (critical loads).

The excessive N deposition in relation to critical N loads, in view of biodiversity impacts, was assessed by simply taking EMEP model results on N deposition and comparing them with those in the official European critical load data base. The results from VSD model simulations for about 670,000 sites (forests and semi-natural vegetation covering 3,667,603 km²) were based on data in the background EU-DB. Results for forests have also been presented, separately (about 385,000 sites covering 2,303,808 km²).

Table 4.1 Relationships between N deposition and ecosystem services

Ecosystem services	Examples of nitrogen effects	Causal link with nitrogen deposition
Provisioning services		
Food/fiber, including *- Crops* *- Wild plants and animal products*	Increase in crop production Impacts on biodiversity (based products)	N deposition increases crop growth in N limited systems (low N fertilizer inputs) N induced eutrophication and soil acidification affects soil, plant and faunal species diversity and thereby biodiversity-based products
Timber/wood fuel	Increase in wood production	In N limited systems, nitrogen increases forest growth and wood production; in N saturated forests, N can induce mortality.
Natural medicines	Impacts on medicinal plants	N induced eutrophication and soil acidification affects plant species, but linkage to medicinal plants is largely unknown
Fresh water	Impacts on ground water recharge and drainage	N induced impacts on growth and plant species diversity also affect water uptake and thereby freshwater supply (see also water quantity regulation).
Regulating services		
Air quality regulation	Decline in air quality	Nitrogen deposition is correlated with increased concentrations of ammonia (NH_3), nitrogen oxides (NO_x), ozone (O_3) and particulate matter (PM10 and PM2.5), all affecting human health and ecosystems.
Climate regulation *Green house gas balance*	Increased carbon sequestration in forests Increased/decreased carbon sequestration in peat lands Increased N_2O production Decreased CH_4 consumption Increased O_3 production	In N limited systems, nitrogen deposition increases forest growth and related tree carbon sequestration, but can enhance mortality in some species. It also can cause an increased litterfall and reduced decomposition, leading to soil carbon sequestration. At low N deposition, additional atmospheric nitrogen deposition may stimulate net primary productivity. At high rates of N deposition, species composition changes lead to loss of peat land forming species and changed microbial activity causing degradation of peat lands Ecosystem losses as N_2O increase with increasing N loading Soil microbes decrease CH_4 consumption in response to increased NH_4 availability Increased production in tropospheric O_3 from interactions with NO_x and VOC emitted from ecosystems, which serves as GHG and can also inhibit CO_2 uptake through plant damage
Water quantity regulation	Increased/decreased runoff and ground water recharge Increased drought stress	Excess nitrogen may cause decreased runoff and ground water recharge due to increased water uptake (elevated growth) but also the reverse because of a lower LAI due to defoliation caused by e.g. pests/diseases. Recharge may in the long term also be affected by impacts on soil carbon content and soil biodiversity, affecting water retention in soil Excess nitrogen causes an increased need for water by an increased growth and an increased sensitivity for drought stress by an increase in the ratio of above versus below ground biomass
Water quality regulation (water purification)	Decline in ground water and surface water (drinking water) quality	Nitrogen eutrophication and N induced soil acidification increases NO_3, Cd and Al availability, leading to - NO_3, Cd and Al concentrations in groundwater and surface water exceeding drinking water quality criteria in view of human health effects. - Increased Al concentrations in acid sensitive surface waters resulting in the reduction or loss of fish (salmonid) populations and reduction of aquatic diversity at several trophic levels (acidification) - Fish dieback by algal blooms and anoxic zones (eutrophication). Eutrophication is also affected by silica and phosphorus in estuaries.
Soil quality regulation	Decrease in acidity buffer; change in soil structure	N induced soil acidification decreases the exchangeable pool of base cations, that may cause reduced forest growth and it decreases the readily available Al pool, affecting soil structure.
Pest/disease regulation	Increased human allergic diseases Increase in forest pests	Increasing N availability can stimulate greater pollen production, causing human allergic responses, such as hay fever, rhinitis and asthma. Increase in bark or foliar N concentrations can attract higher infestation rates, such as beech bark disease
Supporting services		
Nutrient cycling and primary production	Increases N inputs by litterfall and reduces soil biodiversity	N induced impacts on growth/litterfall and on soil biodiversity (soil mesofauna and bacteria composition) affects decomposition, nutrient mineralization and N immobilization and thereby nutrient cycling and primary production
Cultural services		
Cultural heritage values	Impacts on culturally significant species in historically important landscapes.	N deposition may change heathlands into grasslands, affecting historically important landscapes
Recreation and ecotourism	Impacts on recreation due to impacts on ecosystems	Nitrogen induces the increase in nitrophilic species like stinging nettles and algal blooms reducing recreational and aesthetic values of nature. Most extreme is closed beeches due to algal blooms due to N induced eutrophication in estuaries and coastal ecosystems

Table 4.2 Comparison of ranges in critical limits (HC_5 values, i.e. the hazardous concentration potentially affecting 5% of the population) for Cd related to ecotoxicological impacts on soil organisms, phytotoxic impacts on plants, impacts on aquatic organisms and drinking water limits.

HC_5 concentration ecotoxicity ($mg\ m^{-3}$)	HC_5 concentration phytotoxicity ($mg\ m^{-3}$)	HC_5 concentration surface waters ($mg\ m^{-3}$)	Drinking water limit ($mg\ m^{-3}$)
1.6	2.6	0.19	1

Table 4.3 Estimated ranges in the impact of 1 kg of N deposition on average annual CO_2, N_2O and CH_4 emissions and on their global warming potential (GWP) in CO_2 equivalents.

Greenhouse gas	N deposition impacts in kg $ha^{-1}yr^{-1}$ on CO_2-C , N_2O-N and CH_4-C	GWP (kg CO_2 equivalents $ha^{-1}yr^{-1}$)
CO_2-C	24.5±8.7 kg CO_2-C $ha^{-1}yr^{-1}$	-89.8 ± 32.0
N_2O-N	0.0087 ± 0.0025 kg N_2O-N $ha^{-1}yr^{-1}$	+4.0 ± 1.2
CH_4-C	-0.015±0.004 kg CH_4-C $ha^{-1}yr^{-1}$	+0.44 ± 0.12
Total		85.4± 33.3

Quantification and evaluation of N-deposition impacts on water quality

The water quality regulation function of forest ecosystems is negatively impacted by elevated nitrate concentrations and by the nitrate induced acidification leading to an increase in the concentrations of aluminium and heavy metals. The impacts on water quality and soil quality were derived from the VSD model, and implied that the water quality parameters refer to the soil solution draining to groundwater and surface water. We assessed Cd in solution on the basis of: (i) given values for the total Cd concentration in the soil, (ii) a relationship between reactive and total soil Cd contents, accounting for effects of soil organic matter and clay content, and (iii) a relationship between total dissolved Cd concentration and reactive soil Cd content, accounting for effects of soil organic matter, clay, DOC and pH (calculated with VSD). The impact of N deposition was evaluated by comparing concentrations for NO_3 Al and Cd (influenced by pH) with water quality criteria, such as:

- a limit for Al concentration in groundwater of 0.02 mmol Al l^{-1} in view of possible Alzheimer disease;
- a limit for NO_3 concentration in (i) groundwater in view of possible health impacts (25 mg NO_3 l^{-1} is target and 50 mg NO_3 l^{-1} is EU quality criterion), and (ii) surface water in view of eutrophication effects (we applied 2.2 mgN l^{-1}, being a water criterion used for the Netherlands);
- various limits for the Cd concentration in soil solution, as presented in Table 4.2, based on De Vries et al. (2007a).

Quantification and evaluation of N-deposition impacts on soil quality

The impact of N (and S) deposition on soil quality, in terms of BC and Al depletion, was also derived by running the VSD model. Depletion of the readily available Bc and Al pools was derived as total BC and Al soil release rates minus BC and Al weathering rates. BC depletion from the exchangeable base cation pool is directly given in VSD. Al depletion was derived by Al leaching, and Al weathering was calculated as twice the BC weathering. Results are given as annual average Bc and Al depletions in eq $ha^{-1}yr^{-1}$.

Quantification of N-deposition impacts on terrestrial green house gas emissions

The overall impact of N deposition on terrestrial ecosystems, as global warming potential, is mainly determined by the interactions between anthropogenic N deposition and CO_2, N_2O and CH_4. Elevated N deposition has three general effects. First, it causes an increase in C sequestration from increased wood production and accumulation of soil organic matter through an increase in litter, increased leaf/needle biomass production and a reduced decomposition of organic matter, depending on the stage of humus formation. Second, it causes an increase in N_2O emission due to an elevated nitrification and denitrification and, third, elevated N deposition causes reduced CH_4 uptake in well-drained soils. Various review studies have assessed the impacts of N deposition on the above greenhouse gases. For this study, we used the results of a recent meta-data analysis by Liu and Graever (2009) of studies on the GHG flux (CO_2, CH_4, N_2O) from N additions in multiple terrestrial and wetland ecosystems. In their analysis, sixty-eight publications that contained 208 observations across North and South America, Europe and Asia were included. We quantified the overall effect of N deposition on GHG fluxes on forests by using the average results from Liu and Graever (2009) on fertiliser-induced emission/uptake factors per kg N $ha^{-1}yr^{-1}$ in terms of CO_2 equivalents. The global warming potential approach was used, that is, 1 kg N_2O was assumed to equal 296 CO_2 equivalents and 1 kg CH_4 to equal 23 CO_2 equivalents (Ramaswamy 2001). Results are presented in Table 4.3.

The results from Liu and Graever (2009) for CO_2-C are comparable to a review paper by De Vries et al. (2009), which presented estimated ranges in carbon (C) sequestration per kg nitrogen (N) addition in above-ground biomass and in soil organic matter in forests. The results in this overview paper were based on (i) empirical relations between spatial patterns of carbon uptake and influencing environmental factors, including nitrogen deposition (forests only), (ii) ^{15}N field experiments, (iii) long-term low dose N-fertiliser experiments, and (iv) ecosystem models. Results indicated a total carbon sequestration range of 5 to 75 kg C/kg N deposition, with a most common range of 20 to 40 kg C/kg N (De Vries et al. 2009). The results from Liu and Graever

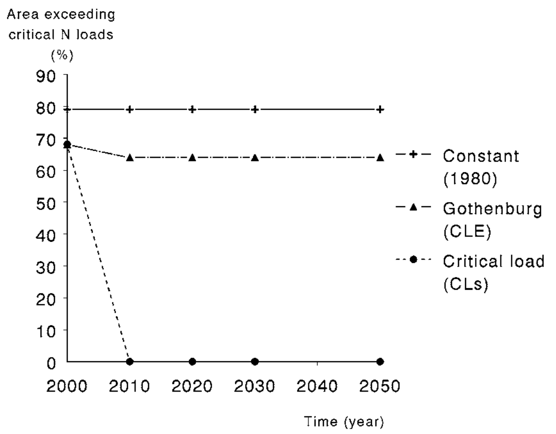

Figure 4.2 Trends in the exceedance in critical N loads, using N-deposition values of 1980, and according to the 'Current Legislation' scenario (CLE) and according to critical N loads.

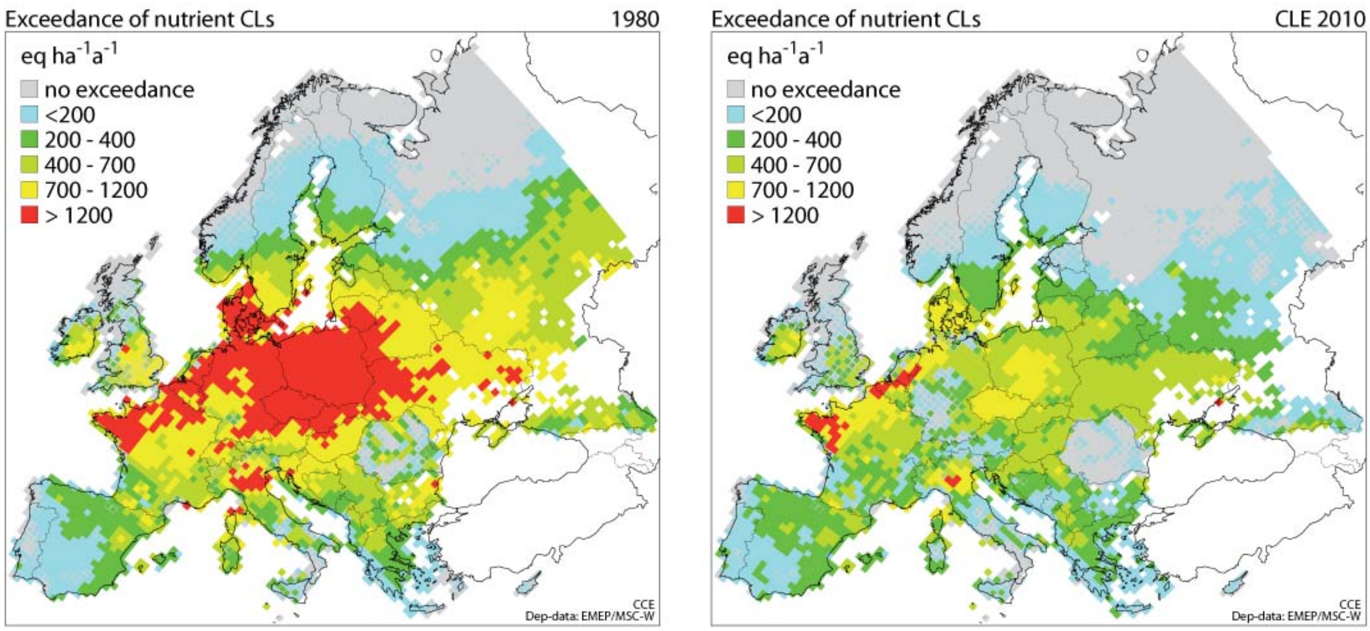

Figure 4.3 Geographic variation in the exceedance in critical N loads, using N-deposition values of 1980 (left), and the Current Legislation scenario (right).

(2009) for N_2O-N and CH_4-C showed that the impact of N deposition on N_2O emission was approximately 5% compared to CO_2 sequestration, whereas the impact on CH_4 uptake was negligible, in terms of global warming potential (Table 4.3). De Vries et al (2007b) found comparable results based on independent estimates of the emissions for the various greenhouse gases in response to N deposition.

4.3 Results

Exceedances of critical loads in relation to biodiversity impacts

Exceedances of critical loads according to the three scenarios are presented in Figure 4.2. By definition, the 'Critical Load' scenario (CLs) implies no exceedance in critical N loads after 2010. The impact of the 'Current Legislation' scenario (CLE) is relatively small, compared to the constant 1980 N deposition scenario, on the areas with exceeding critical loads (Figure 4.2), but the magnitude of the exceedance differs greatly, as shown in Figure 4.3.

N-deposition impacts on water quality

Figure 4.4 shows the results of the exceedances of the area exceeding a critical NO_3 concentration, either in groundwater (50 mg NO_3 l^{-1}) or surface water (2.2 mgN l^{-1}), according to the three scenarios. Unlike in the biodiversity assessment, results show that the area for which a critical NO_3 concentration is exceeded, is larger when comparing the 'Current Legislation' scenario (CLE) to constant the 1980 N deposition scenario, than when comparing the CLE scenario to the Critical Load scenario (CLs), particularly when the groundwater limit of 50 mg NO_3 l^{-1} is used as an additional criterion. This is in line with the much larger reduction in N loads between the 1980 N deposition scenario and the CLE scenario, compared to between the CLE and CLs scenarios. The magnitude of the exceedance in the critical NO_3 concentration also differs

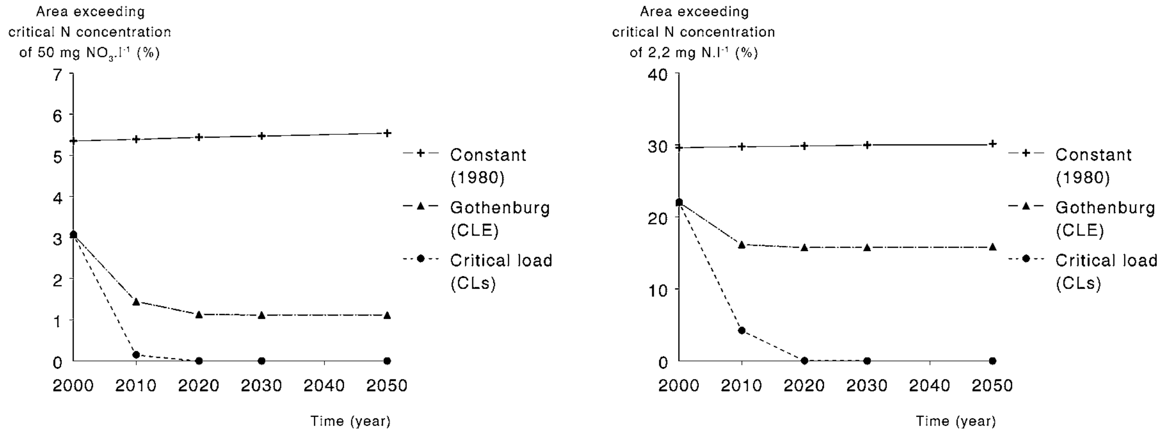

Figure 4.4 Trends in the exceedance of a critical NO_3 concentration of 50 mg NO_3 l^{-1} (left) and a critical N concentration of 2.2 mgN l^{-1} (right), using N-deposition values of 1980, and according to the CLE scenario, and according to critical N loads.

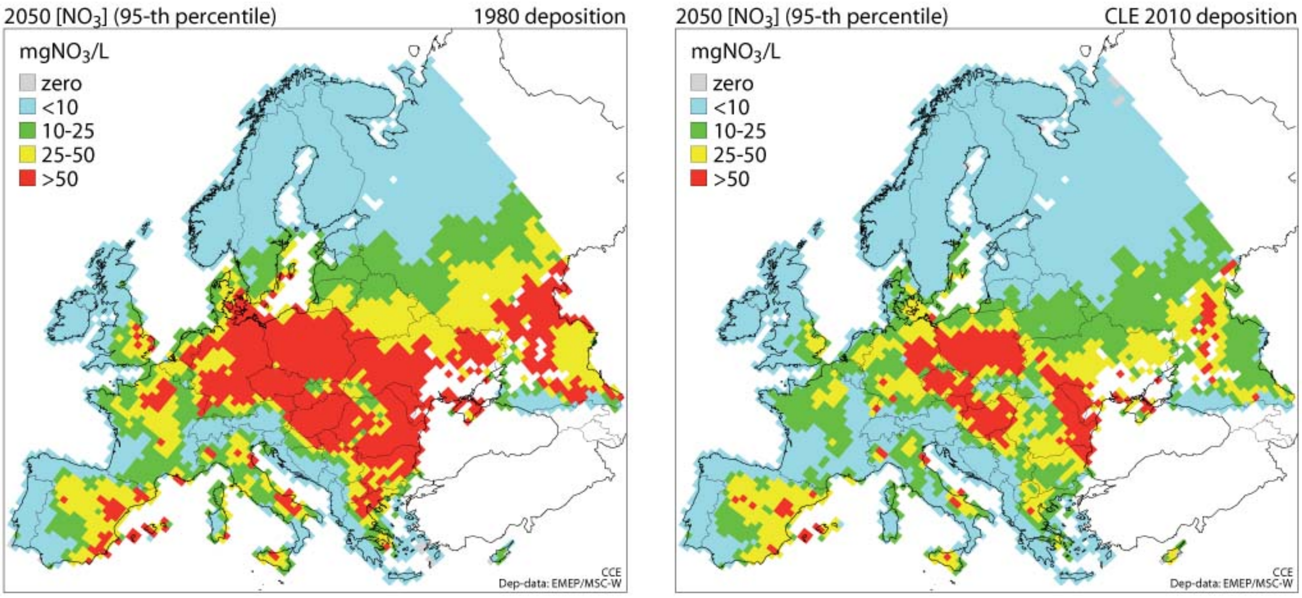

Figure 4.5 Geographic variation in the exceedance of critical NO_3 concentration of 50 mg NO3 l^{-1}, using N-deposition values of 1980 (left), and according to the CLE scenario (right).

greatly between the 1980 deposition and the CLE scenario, as shown in Figure 4.5.

Compared to CL(N) exceedance, the difference between the CLE and the constant 1980 N deposition scenario, on the areas with an exceeding critical Al concentration of 0.02 mmol Al l^{-1}, is much larger than the difference between the CLE and CLs scenarios (compare Figures 4.3 and 4.6). It is important to realise that the difference is not only caused by the impact of a reduction in the deposition of N, but also in that of S. The magnitude of the exceedance of the critical Al concentration also differs greatly between the 1980 N deposition scenario and the CLE scenario, as shown in Figure 4.7.

Exceedances of critical Cd concentration in soil solution, related to either the HC_5 concentration in surface waters (0.19 mg.m^{-3}) or those exceeding the drinking water limit (1mg.m^{-3}), for the three scenarios (Figure 4.8), show similar results as for Al, but even more extreme. The impact of the 'Current Legislation' scenario (CLE), compared to the constant 1980 N deposition scenario, on the areas with an exceeding critical Al concentration is here much larger than the difference between the CLE scenario and the 'Critical Load' scenario (CLs), which is barely discernible. This is in line with the much larger reduction in N and S loads between the 1980 N deposition and CLE scenarios, than between the CLE and CLs scenarios, thus having a much larger effect on the pH affecting Cd mobility. The magnitude of the exceedance of the various critical Cd concentrations also differs greatly between the 1980 N deposition scenario and the CLE scenario, as shown in Figure 4.9.

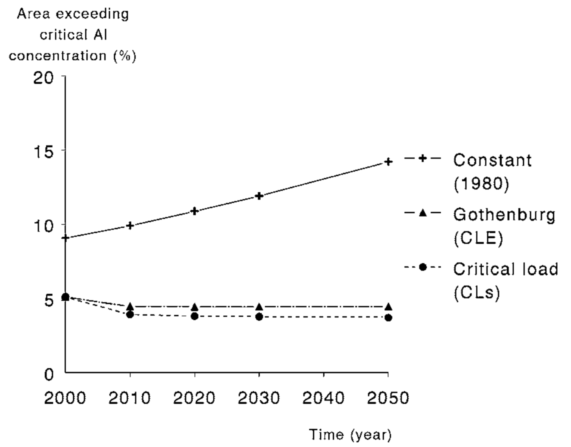

Figure 4.6 Trends in the exceedance in critical Al concentration of 0.02 mmol Al l⁻¹, using N deposition values of 1980, and according to the 'Current Legislation' scenario (CLE), and according to critical N loads.

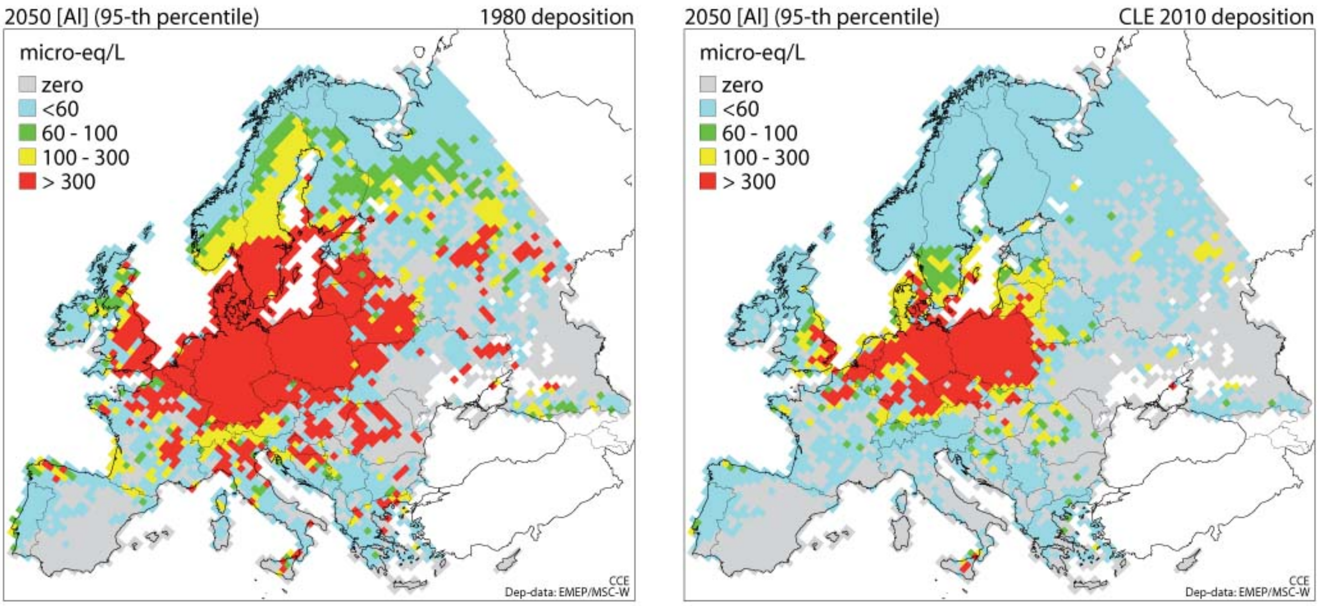

Figure 4.7 Geographic variation in the exceedance in critical Al concentration of 0.02 mmol Al l⁻¹, using N deposition values of 1980 (left), and according to the CLE scenario (right).

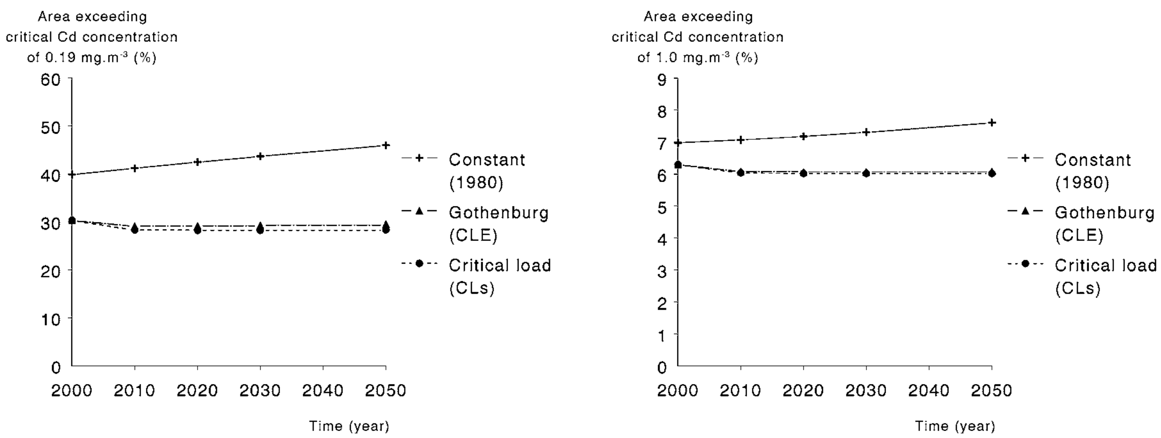

Figure 4.8 Trends in the exceedance in a critical Cd concentration of 0.19 mg m⁻³ (left) and a drinking water limit of 1.0 mg m⁻³ (right), using N deposition values of 1980, and according to the 'Current Legislation' scenario (CLE), and according to critical N loads.

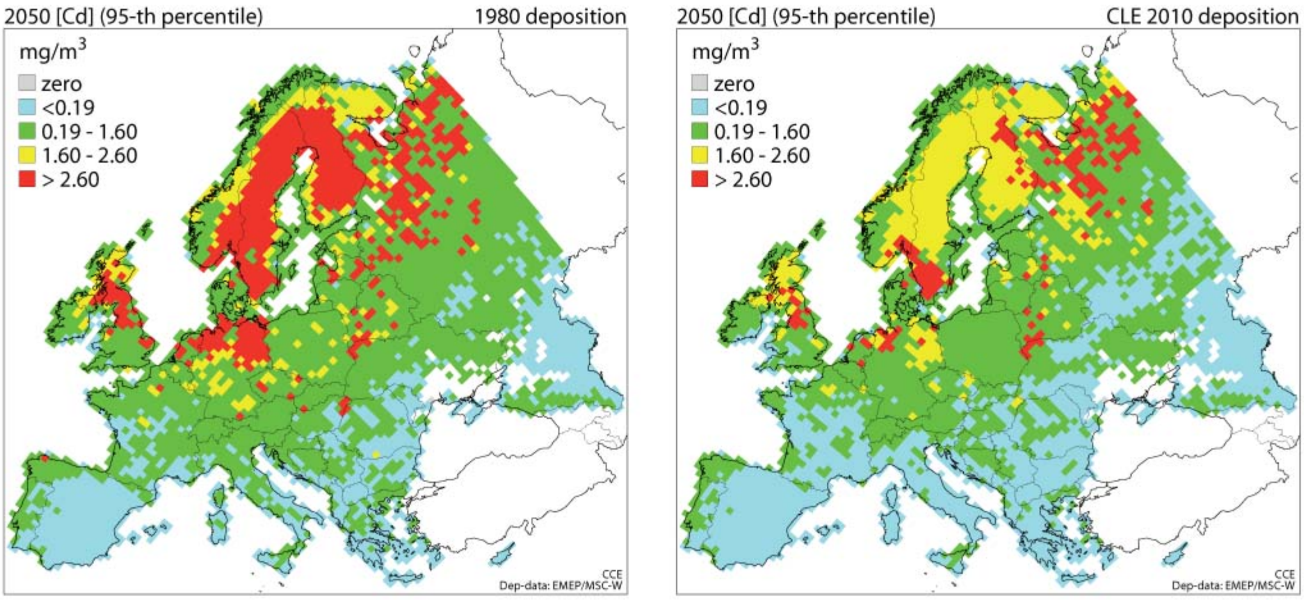

Figure 4.9 Geographic variation in the exceedance in critical Cd concentration of 0.19 mg m⁻³, using N deposition values of 1980 (left), and according to the CLE scenario (right).

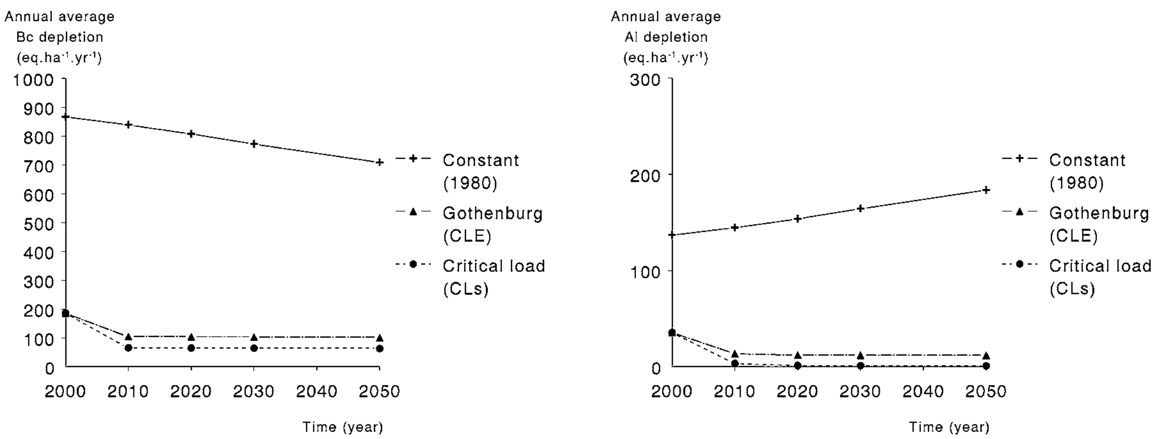

Figure 4.10 Trends in the annual average BC depletion (left) and Al depletion (right), using N deposition values of 1980, and according to the 'Current Legislation' scenario (CLE), and according to critical N loads.

N deposition impacts on soil quality

Results from the trends in average Bc and Al depletion according to the three scenarios (Figure 4.10) show again that the impact of the 'Current Legislation' scenario (CLE), compared to the constant 1980 N deposition scenario, is much larger than the difference between the CLE and CLs scenarios, in line with the much larger reductions in N and S loads affecting the depletion of both Bc and Al. The large difference in depletion between the 1980 N deposition scenario and the CLE scenario is further illustrated on a European scale in Figure 4.11. Results show that in the present situation (CLE), Al depletion is only an issue in central Europe, but Bc depletion is more widespread, although the rate of depletion has strongly diminished due to emission reductions (Figure 4.11).

N deposition impacts on terrestrial greenhouse gas emissions

Assuming an average impact of 85 kg CO_2 equivalents/ha/yr, per kg N (see Table 4.3), for a forested area of 162 million hectares, and considering an average increase of 2.8 kg N $ha^{-1}yr^{-1}$ in 2000, compared to 1960 levels, De Vries et al. (2007b) implied a reduction in annual GWP due to elevated N deposition of 38.56 Mt CO_2 equivalents.

Unlike the other ecosystem services (protected biodiversity, good soil structure, clean water), N deposition has a positive effect on the global warming potential (GWP) since the positive effect on the carbon sequestration potential of ecosystems overwhelms the negative effect of increased N_2O mission and reduced CH_4 oxidation (Table 4.3). The impact of N deposition differences on the GWP on a European scale has been estimated by assuming an average impact of 85 kg CO_2 equivalents $ha^{-1}yr^{-1}$ per kg N (see Table 4.3), for a forested area of 162 million hectare, and considering the average

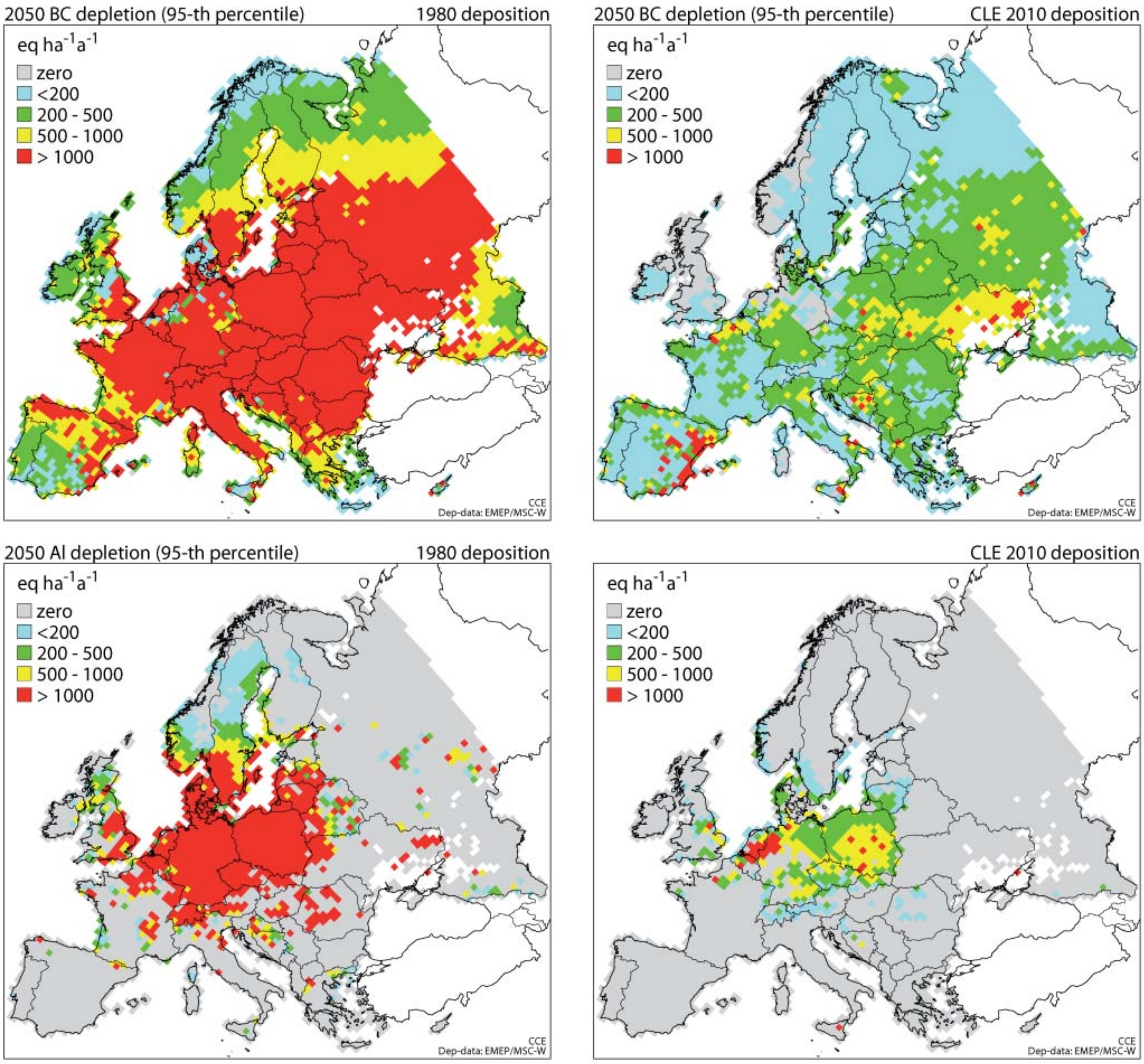

Figure 4.11 Geographic variation in BC depletion (top) and Al depletion (bottom), using N deposition values of 1980 (left), and according to the CLE scenario (right).

increase in N deposition in the 2000-2050 period, compared to the deposition in 1960 (reference situation), under the various scenarios. The geographic difference in N excess between two scenarios (1980 N deposition and CLE) is shown in Figure 4.12.

Taking an average N increase for the 2000-2050 period, compared to 1960 levels, of 4.5 kg N ha $^{-1}$yr^{-1} under the CLE scenario and multiplying this with the GWP impacts and forested area as mentioned above, implies a predicted N induced reduction in annual GWP of 62.0 Mt CO_2 equivalents. Using the 1980 N deposition scenario, the average N excess in the 2000-2050 period will be 9.6 kg N ha^{-1}yr^{-1} implying a reduction in annual GWP of 132.2 Mt CO_2, that is, an extra 70.2 Mt CO_2.

De Vries et al. (2007b) used this approach to assess the average GWP reduction in the 1960-2000 period, compared to 1960 levels. They used an average increase in N deposition of 2.8 kg N ha^{-1}yr^{-1} for 2000, compared to 1960 levels, implying a reduction in annual GWP due to elevated N deposition of 38.6 Mt CO_2 equivalents (De Vries et al. 2007b). Compared to an estimated total carbon CO_2 uptake by European forests of 100 Mt C, being equal to 366 Mt CO_2 equivalents, their result implied that N deposition increased the CO_2 uptake by approximately 10%. Therefore, for the 2000-2050 period, we estimated that the increase in CO_2 uptake, compared to 1960 levels, will be approximately 17% under the CLE scenario, but 36% under the 1980 N deposition scenario.

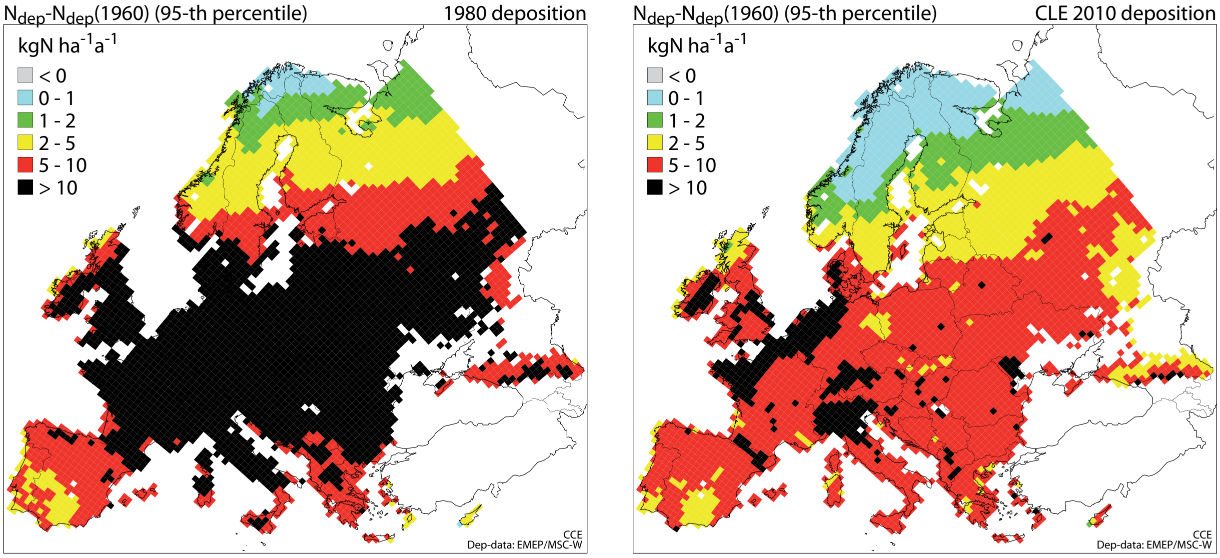

Figure 4.12 Geographic variation in excessive N deposition (average N deposition in the 2000-2050 period minus N deposition in 1960), using N deposition values of 1980 (left), and according to the CLE scenario (right).

Table 4.4 Predicted areas with exceeding critical N loads or critical limits related to water quality, in the year 2050, under three scenarios

Model output	**Critical limit**	**Area exceeding critical limits**		
		1980 N deposition	**CLE scenario**	**CLs scenario**
N loads	Critical N loads	79.0	64.0	0
NO_3	50 mg NO_3 l^{-1}	30.0	16.0	0
N concentration	2.2 mg N l^{-1}	5.2	1.0	0
Al concentration	0.02 mmol l^{-1}	14.0	4.4	3.7
Cd concentration	0.19 mg m^{-3}	46.0	30.0	28.0
	1.0 mg m^{-3}	7.6	6.1	6.0
	1.6 mg m^{-3}	5.4	4.1	4.1
	2.6 mg m^{-3}	3.3	1.0	1.0

4.4 Conclusions and Outlook

A summary table showing the effects of all scenarios for areas with exceeding critical loads or limits related to water quality for the year 2050 is given in Table 4.4. This study shows that where N deposition reduction measures have been taken since 1980, this has had strong positive effect on biodiversity in the areas with exceeding critical loads. In addition, there is a clear positive impact on water quality, decreasing in the direction of NO_3 concentration > Al concentration > Cd concentration. The impact on soil quality in terms of Al and Bc depletion is comparable to the Al concentration impact.

This decreasing impact is due to the fact that Bc and Al depletion, Al concentration and Cd concentration are also influenced by S deposition. The difference between the CLE and CLs scenarios is much less, except for biodiversity impacts, which are the result of making the N load equal to the critical load, at each location.

Unlike biodiversity, soil and water quality, N deposition reduction has a negative impact on the GWP. For the 2000-2050 period, we estimated that the increase in CO_2 uptake, compared to 1960 levels, is approximately 17% under the CLE scenario, but 36% under the 1980 N deposition scenario.

We aim to further develop an integrated VSD^+ model coupled to the database for growth by the European Forest Institute (EFI) by (actions to be carried out are given in italics):

- Assessing N and climate impacts on plant species diversity by (i) direct linkage of N deposition with empirical responses, (ii) coupling of VSD^+ with an empirical database linking pH and N availability with plant species occurrence and (iii) linking VSD^+ with VEG.
- Including impacts of ozone exposure, in combination with CO_2 fertilization, atmospheric deposition of nutrients (N, S, P, base cations) and climate reductions on growth and carbon sequestration.
- Calculating N_2O, NO_x and N_2 emissions and comparison of the results with the DNDC model and empirical relationships between N_2O emissions and environmental factors.
- Including impacts of climate change and acid deposition on metal concentrations in soil and soil solution through impacts on concentrations of dissolved organic carbon (DOC) and pH.

With respect to plant species diversity it allows us to assess the robustness of N deposition impacts on plant species diversity by comparison of:

- Empirical and computed critical N loads using stead-state VSD.
- Empirical N dose-response relationships with computed plant species diversity indicators in response to N deposition using the integrated model VSD+.

The integrated VSD+ model will allow us to assess the combined impact of air quality parameters, climatic parameters and nutrient inputs on various provisioning and regulating services, including (i) diversity of plant species, (ii) wood production and global warming potential and (iii) soil filtering/buffering affecting water quality. This is important for the support of policies in the field of air pollution, climate change and biodiversity in Europe.

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Part 2 Dynamic modelling

This Part consists of two chapters that address the progress in the development of dynamic modelling to capture the interaction between nitrogen deposition and the change of climate and biodiversity. Chapter 5 presents results of a pilot study showing that it is possible to simulate the relationship between nitrogen deposition and the ensuing composition of the plant community while keeping track of the change of climate and land use. Chapter 6 focuses on the extension of the Very Simple Dynamic (VSD) model, currently widely used by National Focal Centres, to include carbon and nitrogen dynamics. This extended model, termed 'VSD+', is designed to limit the data requirements needed by National Focal Centres to simulate vegetation responses to changes in nitrogen deposition.

5

Developing a method for estimating critical loads of nitrogen deposition under a changing climate, based on biological indicators

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5.1 Introduction

Under the LRTAP convention, the classic methods for setting critical loads of acidity have been successful in linking atmospheric deposition to ecosystem damage, and contributed decisively to the setting of lower emission targets for acidity. For nitrogen (N), however, emissions continue to be elevated over large parts of Europe, often even exceeding current critical loads. To date, two methods have been used for setting critical loads of nitrogen deposition for terrestrial ecosystems, namely the empirical critical loads and the dynamically modelled concentrations of nutrient nitrogen in soil solutions. The empirical critical loads are based on experimental or observed damage to plants at given nitrogen inputs, and by definition do not include other sources of reactive N nor other drivers which may alter the response of plants to N inputs. The dynamic modelling of nutrient N in soil solution measures excess N deposition by its effect on N concentrations in soil solution, in excess of a vegetation type-specific value defined by De Vries et al. (2007).

The assumption of steady state inherent in the classic critical loads methodology supposes unchanging environmental conditions in the future. This assumption is challenged by current and continuing changes in climate and land use, particularly given that the effects of climate and land use can confound the response of ecosystems to N deposition. This raises the need for a complementary approach to estimating critical loads of N deposition, directly based on biological indicators. The method presented in this study is meant to be such a complementary tool to integrate different simultaneous drivers, which could be the basis for assessing responses of terrestrial ecosystems to N deposition.

5.2 Aims of the study

The primary aim of the present study was to investigate the feasibility of estimating critical loads of N deposition, based on changes in the composition of plant communities. The study was divided into three steps: 1) the set up of a general, one-dimensional indicator to follow up changes in the ground vegetation, 2) to investigate the relative influence of different drivers (climate, deposition, and forest harvesting intensity) on soil chemistry and the composition of the ground vegetation community, and 3) to give a preliminary estimate of critical loads of N, based on changes in the plant community.

5.3 Method

5.3.1 The ForSAFE-Veg model

ForSAFE-Veg is a composite model, for which the ForSAFE model is the biogeochemical simulator platform, providing data for driving the vegetation composition simulator Veg (Belyazid, 2006). The ForSAFE model simulates the physical cycles of carbon, nitrogen, base cations, and water in a forest ecosystem, simultaneously simulating soil acidity and aluminium mobility. The ForSAFE model requires site-specific inputs on the physical properties of the soil (including mineralogy, hydrological parameters, density, depth and stratification), tree type, and time series for atmospheric deposition and climatic data (temperature, light, and precipitation). The model then gives monthly estimates of weathering rate, soil moisture, soil solution concentrations, uptake fluxes for Nitrogen and Base cations, litterfall, decomposition and mineralisation, and photosynthesis and growth rates. The Veg module then reads a set of five drivers (soil solution pH, Bc concentration, N concentrations, ground-level light, soil moisture) from the ForSAFE model, in addition to air temperature, and uses them for estimating the relative abundance of a set of indicator plants at the site. The result is a model chain that can link atmospheric deposition, climatic conditions and land-use changes to biogeochemical responses and plant community composition at site level.

5.3.2 Modelling the ground vegetation community

The composition of the ground vegetation community is simulated in the Veg module by distributing the available ground area (a hypothetical representative 1m²) over the plant species that would be able to establish at the site, given its abiotic conditions. For each species, a 'niche window' is estimated by the model. This niche window is the combined limits of N and Bc concentrations in the soil solution, soil solution pH, soil moisture, air temperature, and light intensity reaching the ground vegetation (light below tree canopy in case trees are present), within which a species could become established at a certain site. Usually, site conditions are favourable enough for several species to be present simultaneously. The model then calculates the relative ground area occupied by each species, depending on their vigour in response to site conditions and their respective competitiveness. The plant species compete by growing roots to different soil depths, and by shading other plants above ground. The root depth and shading height are given as inputs for each indicator plant species.

The model requires a list of indicator plant species, which is typically based on input from biologists and ecologists familiar with the ecosystem to be modelled. Firstly, a list of representative plant names is drawn up. For practicability, not all plant species are listed; only those that are most representative of the ecosystem, that are important to conservation or ecosystem services, or that have other traits of interest. Secondly, the responses of each plant species to N and Bc concentrations, soil solution pH, air temperature, soil moisture, shade tolerance and palatability (Table 5.1 and Table 5.2) are defined.

The parametric responses to drivers presented in Table 5.1 and Table 5.2 are defined in mathematical terms below. The nitrogen response combines a promotion function and a retardation function, as in Equation 1, where w_+ denotes the slope of the promotion section, k_+ the threshold of [N] where positive response starts, k_- the threshold [N] where the retardation slope starts, w_- the slope of the retardation function, and *a*0 is a normalising factor.

$$f(N) = a0 \times \frac{[N]^{w_+}}{k_+ + [N]^{w_+}} \times \frac{k_-}{k_- + [N]^{w_-}} \qquad (1)$$

The pH response is given in Equation 2, and is expressed as a function of [H+] in the soil solution.

$$f(pH) = \frac{1}{1 + k_{pH}\left[H^+\right]} \qquad (2)$$

The calcifugal retardation function is defined for a limited number of plants that show signs of decline when their uptake function is impaired by elevated Ca^{2+} concentrations, given in Equation 3

$$f(Bc) = \frac{1}{1 + k_{Ca}\left[Ca^{2+}\right]^2} \qquad (3)$$

The plant species' responses to soil moisture, air temperature and ground level photosynthetically active radiation (PAR) are extrapolated between thresholds as in Table 5.2. For each plant, a minimum soil moisture threshold is defined below which it cannot establish nor subsist, a moisture level optimal for the plant, and an upper limit above which the soil is too moist for it to be present. Soil moisture is given as the fraction of saturated pores in the soil, meaning that a soil moisture value of 1 corresponds to a fully water saturated soil. For air temperature, three values are defined, a minimum below which the plant cannot establish, an optimal value, and a maximum above which the physiology of the plant fails due to excessive heat. Temperature is given as the average annual air temperature. For the response to light only two values are needed; a lower limit, below which it is too dark for the plant to establish, and an optimal level, above which the plant receives sufficient light. The light driver is given as the average annual PAR in $\mu mol_{photon} \cdot m^{-2} \cdot sec^{-1}$.

Besides the parametric responses described above, each plant species has two competitive strategies. The first is shading above ground, and is expressed as the shading height of the plant species, that is, the elevation of the plant organ which is able to shade out other plants. The second is the root distribution over soil depth, which denotes the soil depth that the plant's roots can reach to have access to water and nutrients.

Appendix 1 shows the plant parameters table that was developed for Sweden, where forty-two indicator plant species were parameterised.

Table 5.1 Examples of parametric plant responses to soil solution drivers.

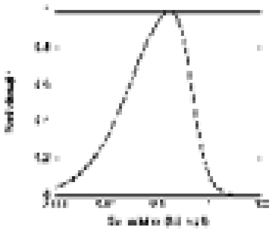
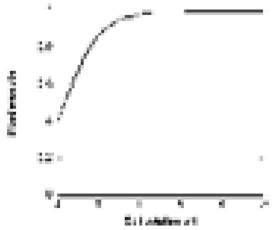
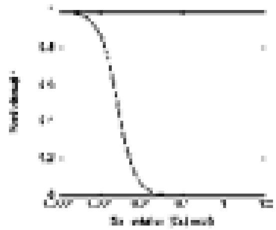
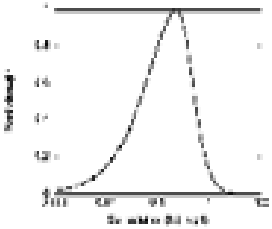
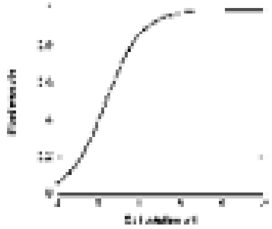
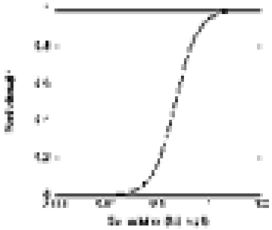
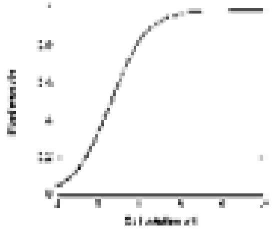
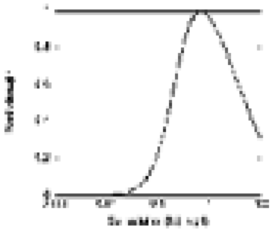
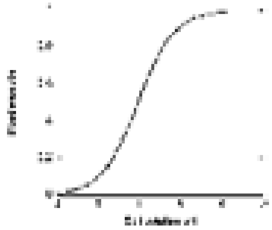
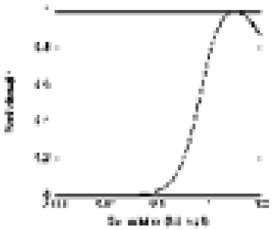
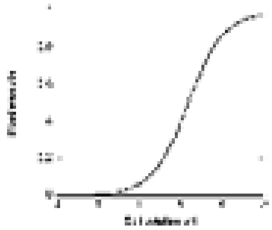
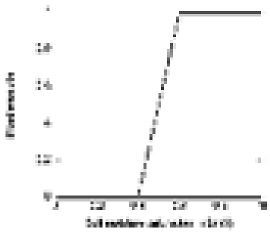
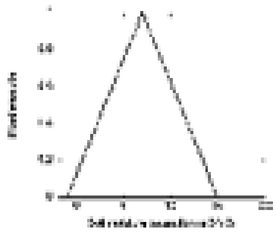
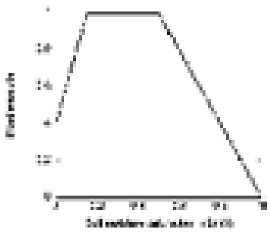
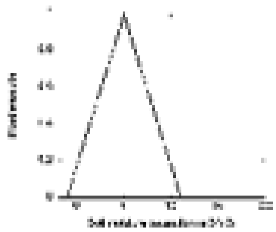
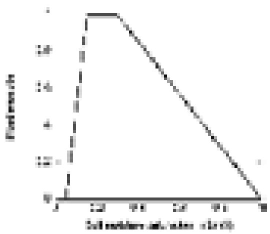
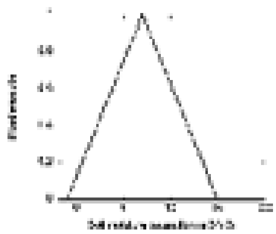
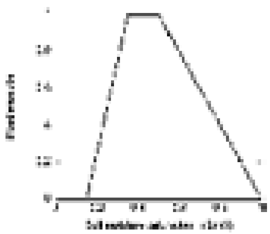
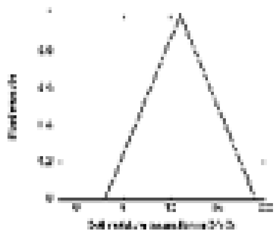
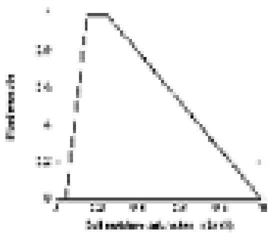
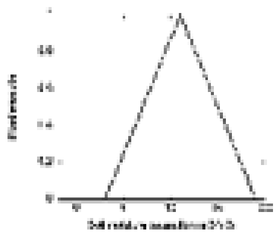
Indicator plant	Soil nitrogen	Soil pH	Cacifugicity retardation
Mosses			
Sphagnum mosses			
Dwarf shrubs			
Vaccinium myrtillus			No retardation
Grasses			
Deschampsia flexuosa			No retardation
Ferns			
Blechnum spicant			No retardation
Herbs			
Origanum vulgare			No retardation

Table 5.2 Examples of parametric plant responses to soil solution drivers.

Indicator plant	Abiotic niches and competition parameters		
	Soil moisture	**Air temperature**	**Shade tolerance**
Mosses			
Sphagnum mosses			
Dwarf shrubs			
Vaccinium myrtillus			
Grasses			
Deschampsia flexuosa			
Ferns			
Blechnum spicant			
Herbs			
Origanum vulgare			

5.3.3 Defining excessive change in a plant community composition

Three parameters are crucial to estimating whether a change in the composition of the ground vegetation due to N deposition is acceptable or not:

1. The reference population under a given reference deposition, against which eventual changes in the composition of the vegetation are evaluated.
2. The target population, that is, the segment of the ground vegetation community for which change is evaluated.
3. The limit of acceptable change, which is the magnitude of divergence of the plant community from the reference population beyond which change in the target population is unacceptable.

For the current round of method testing presented here, N deposition according to the Maximum Feasible Reductions (MFR) emission scenario is adopted as the reference N deposition, with the corresponding vegetation communities as the reference populations. The target population is defined as the entire plant community. A critical limit of 5% difference

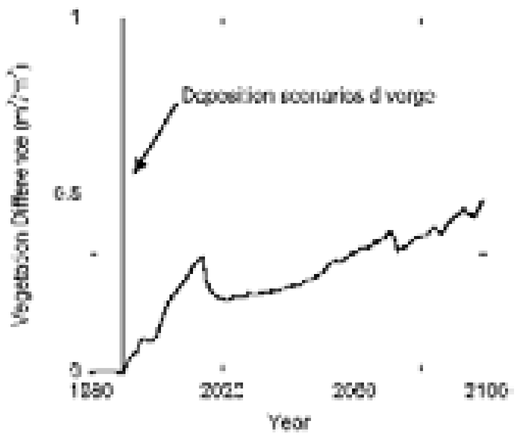


Figure 5.1 The vegetation difference over time expressed in difference in area size between the vegetation community under NITREX deposition and under MFR deposition at Gårdsjön.

is adopted, meaning that differences in area cover of plants species are acceptable up to a limit of 5% of the specific site area.

5.3.4 Defining a one-dimensional indicator for tracing changes in the composition of ground vegetation

Unlike classic chemical indicators, such as base saturation for acidity, or nitrogen concentrations for eutrophication, which are one-dimensional, using plant communities as indicators requires tracking multiple species simultaneously. The occurrences of multiple plant species need to be simplified into a single variable that can be tracked over time, and for which a limit can be set to identify excessive change. Another difference with the classic critical load method is that, instead of defining fixed critical values (ex. Bc/Al=1), using the plant community as indicator requires a steadily changing reference level, as plant populations are affected by other drivers than nitrogen deposition.

To solve this problem, the first step is distilling the relative occurrences of different plant species to area cover, and tracking the amount of ground area that varies between different scenarios of N deposition over time. In this way, it is possible to determine, over time, the magnitude of the difference in the plant community between two deposition scenarios, expressed as the difference in area size between a given scenario and the chosen reference scenario (Figure 5.1).

The next step is to translate the time dependent vegetation difference indicator in Figure 5.1 to a unique number, that can be used to define the deposition of N that keeps the vegetation community within an acceptable range. The vegetation difference changes over time, and some of the changes can be short lived (as with changes due to transient disturbances, such as forest cuttings). To even out the transient peaks and dips, the cumulative vegetation difference for the period when deposition scenarios differ is divided by the number of years for which this difference lasts, to produce a time-independent value called the Average Yearly Difference (AYD) and is given by the following equation

$$AYD = \frac{\sum_{1}^{yearsdiff} VegDiff}{yearsdiff}$$

Where *AYD* is the Average Yearly Difference ($m^2 \cdot m^{-2}$), *VegDiff* is the vegetation difference for each specific year ($m^2 \cdot m^{-2}$), and *yearsdiff* is the number of years for which the deposition scenarios differ.

5.3.5 Finding the critical loads of N deposition

The next step is to define a reference deposition which produces a reference population that is to be protected. For illustrational purposes, the MFR scenario is used here to provide a default reference deposition, meaning that the future plant community, that would be produced by the N deposition from the MFR scenario, will be used as a reference population. The definition of the reference population does not consider the effects of other drivers than N deposition, since other drivers (e.g., climate, land use) are assumed to be independent of the deposition.

Having selected a reference deposition, it is possible to produce a relationship between elevated theoretical N deposition levels and the corresponding AYD (Figure 5.2). The resulting curve can then be used to read a specific N deposition that corresponds to a chosen limit. The limit is the chosen level of vegetation difference, beyond which further difference is not acceptable, and is expressed in the same unit as the Average Yearly Difference (m^2/m^2).

5.4 Results

A set of simulations was carried out with the aim of identifying the contribution of climate, deposition and forest management to changes in soil chemistry and eventually the composition of the plant community. The simulations were performed on a set of 49 sites, and the discussion below refers to the median of the time trends of each of the discussed indicators.

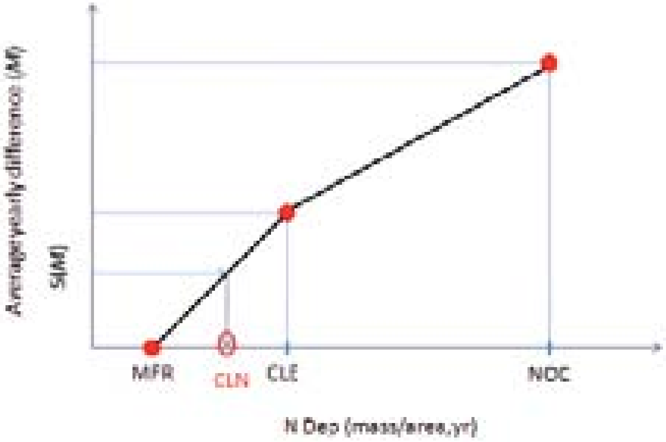


Figure 5.2 There is a positive relationship between N deposition beyond the reference deposition (the MFR scenario in this case) and the Average Yearly Difference.

Table 5.3 12 scenario combinations were used to investigate the potential effects of different climate, deposition and harvesting regimes on the simulated ecosystems.

Drivers	Scenarios		
Climate	Change scenario IS92a	No future change	
Deposition	CLE	MFR	NOC
Harvest	Stem-only harvesting	Whole-tree harvesting	

The three tested drivers, climate, deposition and harvest intensities, were listed into different scenarios according to Table 5.3. The two adopted climate scenarios were a future trend with no deviation from today's climate, and a future trend according to the IS92a scenario. For deposition, three scenarios were used according to the Maximum Feasible Reduction (MFR) scenario, the Current Legislation (CLE) scenario, and the No Control (NOC) scenario. The MFR and CLE scenarios diverged after the year 2010, while the NOC scenario remained at the 1980 level. Finally, two harvest scenarios were adopted, the first assuming conventional stem-only harvesting, and the second according to whole-tree harvesting, where between 80 and 90% of the above-ground wood and 50% of the foliage are harvested. Twelve simulations resulted from the possible combinations of the scenarios in Table 5.3.

5.4.1 Effects of climate, deposition and forest management on soil chemical indicators

The assumed base scenario combining a changing climate (assumed to be according to IS92a), deposition according to the CLE scenario, and stem-only harvesting, will lead to a steady but slow recovery in soil base saturation, over the coming 100 years (Figure 5.3). If deposition after 2010 would take place according to the MFR scenario, the simulations predict a steadily increasing improvement in base saturation (BS), which would be around 20% higher than if deposition remained at CLE scenario level. On the other hand, if deposition would remain at the NOC level from the 1980s, BS would be around half the expected level in 2100, compared to the CLE scenario. Whole-tree harvesting has a relatively negligible effect on BS. If the changing trend in climate would not continue, but would remain at today's levels, BS would be around 20% lower than under the effect of climate change. This is because a change in climate according to IS92a, in median values, would cause a doubling of the weathering rates.

The flux of N leaching has followed both the increase and subsequent decrease in atmospheric N deposition during the second half of the 1900s (Figure 5.4). After 2010, N leaching is expected to remain on a slightly positive trend, despite the fact that deposition remains fixed. The increase in N leaching is due to an increased release of N from decomposition and mineralization, an increase that appears to surpass the contribution of climate change to increased growth. If there would be no climate change in the future, N leaching would be lower by half of its predicted value for 2100 under climate change. Deposition under the MFR scenario would reduce N input to the ecosystems and cause a notable decline in N leaching. The NOC scenario, however, could lead to more than a doubling of N leaching.

5.4.2 Effects of climate, deposition and forest management drivers on ground vegetation composition

The ground vegetation community under MFR scenario deposition, climate change according to IS92a, and stem-only harvesting, is assumed to be the reference population against which plant communities according to other scenarios are compared. The CLE scenario, alone, would produce a negligible 4% difference in area cover with the plant cover under the MFR scenario. Deposition under the NOC scenario, however, would cause 20% of the ground cover to be different. Whole-tree harvesting has a negligible effect on the composition of ground vegetation. The most important driver affecting the composition of the ground vegetation

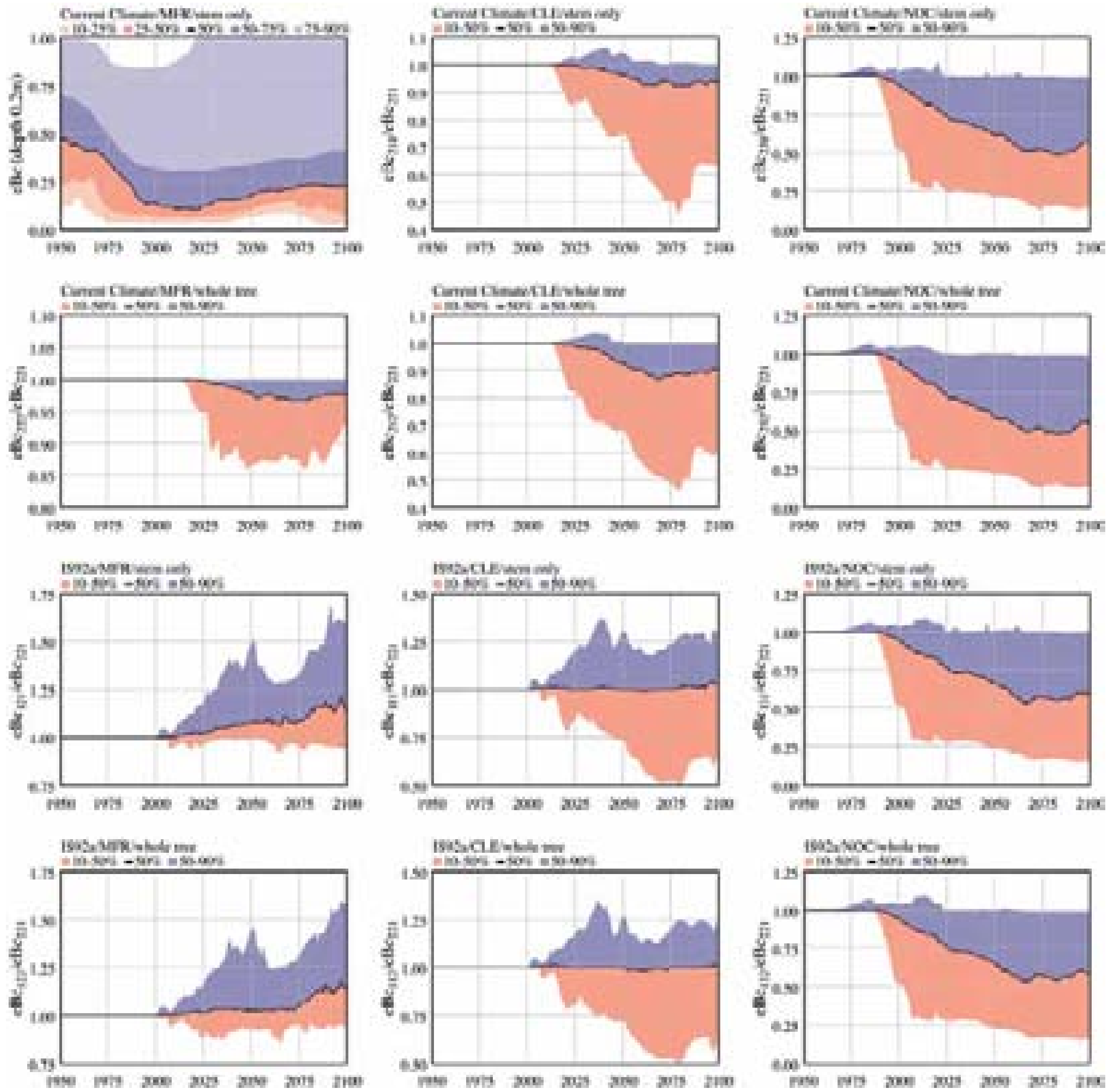


Figure 5.3 Median responses of soil BS at 49 sites to climate, deposition and harvesting intensities.

community is climate. The composition of ground cover would differ by around 40%, solely due to the difference between scenarios with and without climate change. The additional effect of N deposition under the NOC scenario is smaller when combined with the effect of climate (NOC, alone, causes a 20% difference in area cover, while NOC in combination with climate effects causes an additional 5% difference). This can be due to the fact that climate change itself contributes importantly to N mobilisation, thereby reducing the relative contribution of N deposition to N availability in the soil solution, and, accordingly, its effect on the composition of the plant communities.

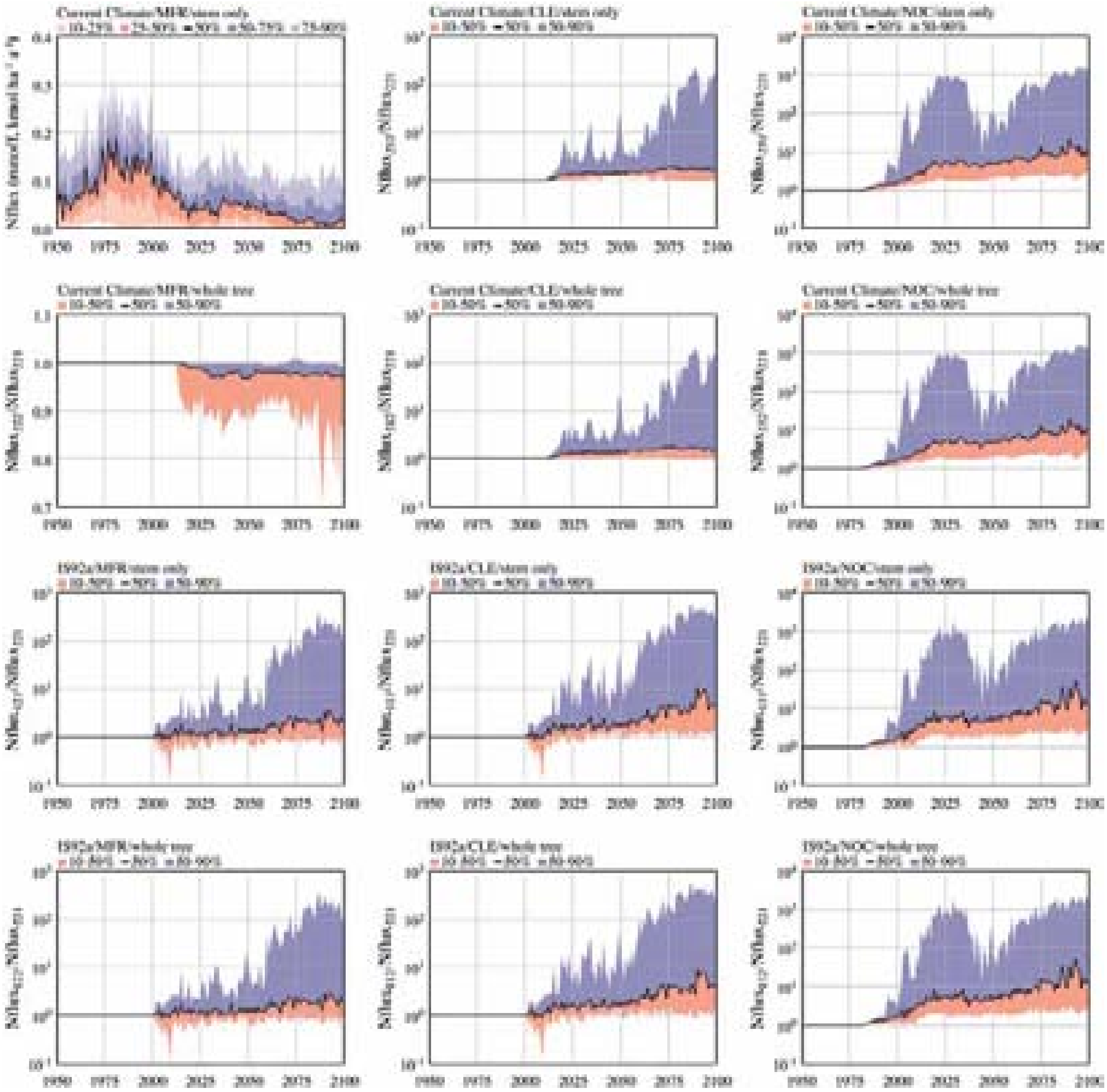


Figure 5.4 Median changes in N-leaching fluxes, and the relative deviations from the base scenario IS92a/CLE/Stem-only harvesting.

5.5 Estimating a Critical Load of Nitrogen (CLN)

The following assumptions were made:

1. The reference population is according to deposition under the MFR scenario.
2. The entire plant community is to be protected.
3. A limit of 5 $m^2 \cdot m^{-2}$ is taken as the critical limit of acceptable change.

This set of assumptions is detrimental to the CLN values that are produced. For sites with low deposition, the CLN values lie below the CLE scenario deposition, meaning that under the CLE scenario excessive change in the plant community would occur, compared to the reference deposition under the MFR scenario; and that sites with low deposition are relatively sensitive to N deposition. On the other hand, the estimated CLN values, based on the adopted assumptions, lie above CLE at sites with elevated MFR deposition. This means that at sites with high MFR deposition, current legislation (CLE) would cause a difference of less than 5$m^2 \cdot m^{-2}$, indicating that sites with elevated N deposition are less sensitive to small additional increases in N deposition. Deposition under the NOC scenario would cause more change than acceptable at all sites.

Adopting the MFR scenario as the reference for deposition values causes another distortion in the estimation of CLN values. The MFR scenario describes only future N deposition, but incorporates the historically accumulated N deposition from the 1900s. While future deposition declines under this scenario, the accumulated N within the ecosystem will remain there for a longer time, and the ground vegetation may not

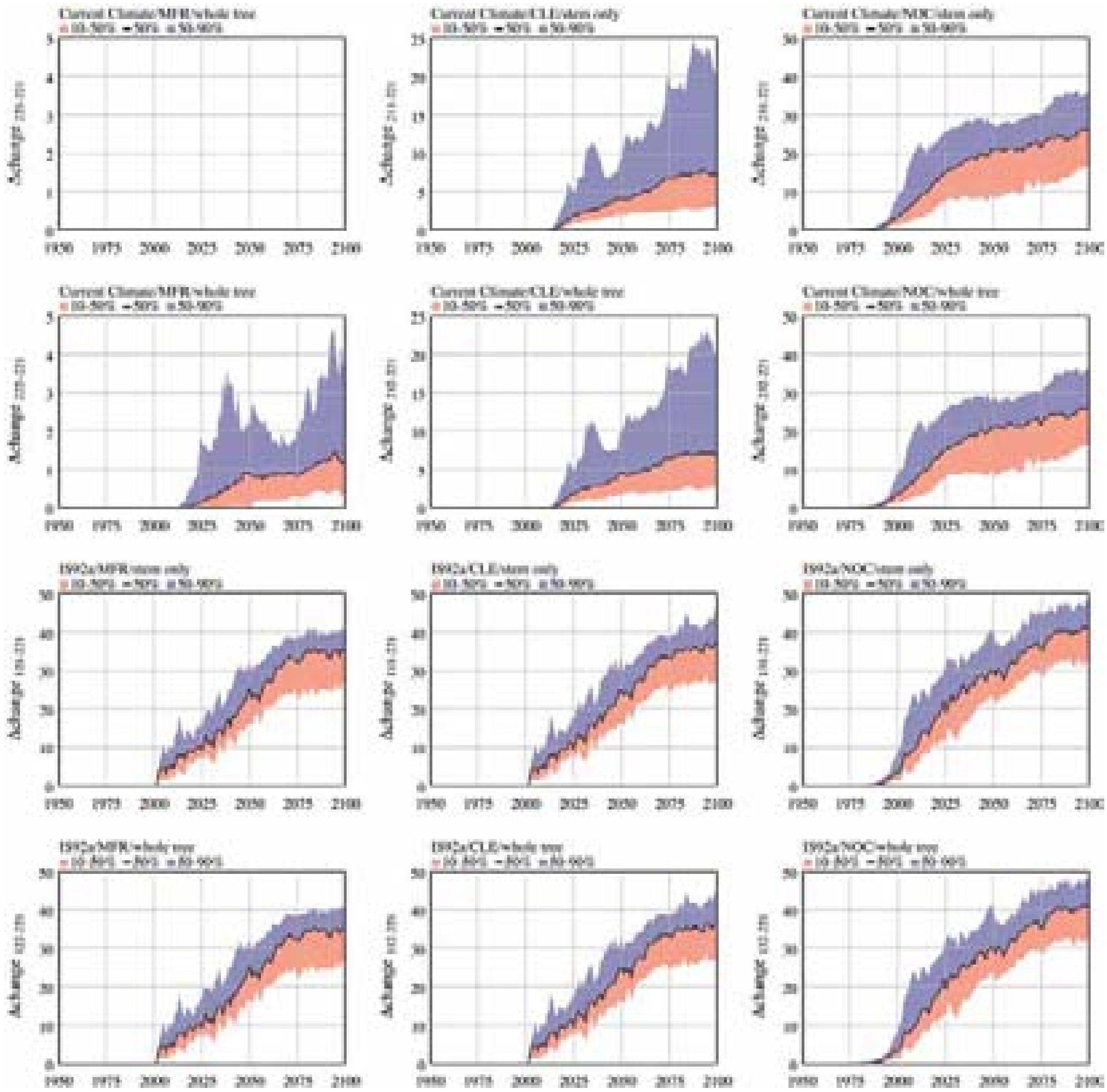


Figure 5.5 Median of differences in area cover of ground vegetation communities, due to different climate, deposition and harvesting intensity scenarios.

respond in tandem with the deposition reduction. There is a considerable delay between the response of the vegetation community and the reduction of N deposition, meaning that the future population under the MFR scenario will retain strong similarities with the population affected by the historically accumulated N deposition.

5.6 Conclusions

The pilot study presented here shows that it is possible to simulate the link between atmospheric deposition and the ensuing biological impact; in this case, the composition of the plant community, while at the same time keeping track of the concurrent effects of changes in climate and land use. The multiple driver analysis, presented above, indicates that climate change and atmospheric deposition have comparably-important effects on soil chemistry, and that these effects are not additive. The intensity of forest harvesting has a lesser effect on soil chemistry. However, the ground vegetation responds most strongly to the two climate scenarios tested, and only marginally to harvesting intensities. The difference in response between soil chemistry and plant community is due to the fact that plants are directly affected by temperature and moisture, in addition to soil chemistry. Therefore, it is essential that climate change is included in any future simulations of ecosystem responses to atmospheric deposition.

The presented methodology provides a tool for investigating the delay times governing plant communities' responses to change in environmental conditions. Because of this, it

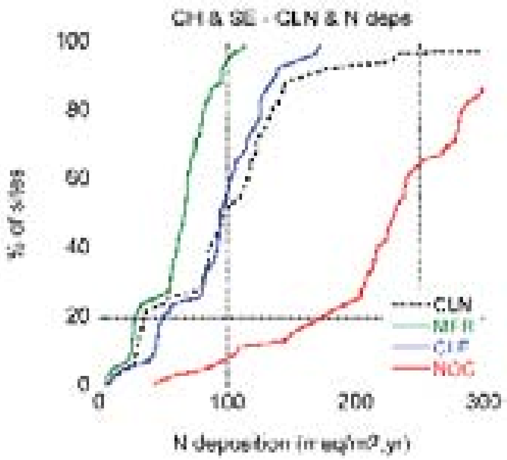


Figure 5.6 Cumulative distributions of N depositions and CLN values at 49 test sites.

has been shown that the adoption of the MFR scenario's deposition as the reference case, with its historically accumulated N during the elevated deposition period prior to emissions reductions, is not appropriate. Instead, a reference level should be used that allows the reference population to reach a stationary level, reflecting low ecosystem N loads.

The present simulations are limited to a set of specific sites. The method presented needs to be tested on a wider geographical scale, to investigate its implications on the estimation of critical loads on a level which is meaningful to the LRTAP Convention. However, because of the wide difference in ecosystems, the target plant populations and the protection limits need to be specified for different ecosystem types, depending on the ecosystem's characteristics of interest.

References

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Appendix

Vegetation parameterisation table for Sweden

	τ	[N]					[H⁺]			W			T			I		h	root		
Latin name	**years**	**a0**	**k+**	**w+**	**k-**	**w-**	**Kbc/al**	**kbc**	**kph**	**min**	**top**	**max**	**min**	**top**	**max**	**min**	**max**	**(m)**	**class**	**kP**	**kG**
Cladonia_lichen	20	1	0.01	1	0.003	3	0.07	0	1050	-0.2	0.05	0.25	-2.5	5.5	13.5	500	2500	0.05	0	0.1	0.7
Hylocomium_mosses	20	1	0.03	1	-	-	0.07	150000	1050	0.05	0.15	0.35	-1	7	15	100	2500	0.02	0	3	0
Mnium_mosses	20	1	0.3	2	-	-	0.4	0	6000	0.15	0.25	0.6	0	8	16	50	2500	0.02	0	3	0
Sphagnum_moss	20	1	0.03	1	0.1	3	0.01	150000	150	0.4	0.6	1	-1	7	15	100	2500	0.02	0	1	0
Calluna_vulgaris	30	1.4	0.2	1	3	3	0.2	0	3000	-0.25	0.15	0.4	-1	7	15	500	5000	0.25	2	1	0.7
Empetrum_nigrum	15	1.6	0.03	1	0.003	3	0.2	150000	3000	-0.2	0.1	0.4	-1.5	6.5	14	500	5000	0.1	1	1	0
Erica_tetralix	15	1.6	0.3	1	0.03	3	0.4	0	6000	0.2	0.35	0.6	0	8	16	1000	5000	0.15	1	1	0
Vaccinium_myrtillus	10	1.6	0.1	1	0.1	3	0.1	0	1500	-0.1	0.15	0.5	-1	5	11	100	2000	0.3	1	1	2.3
Vaccinium_vitis-idea	15	1.6	0.03	1	0.003	3	0.35	0	5250	-0.2	0.1	0.45	-1.5	4.5	10.5	500	4000	0.15	1	1	0.7
Agrostis_capillaries	10	1	0.5	2	-	-	0.2	0	3000	0.05	0.15	0.5	3	11	19	750	4000	0.25	2	3	2.3
Brachiopodium_pennatum	5	1	20	2	-	-	6	0	3500	0.1	0.2	0.35	3	11	19	1000	3500	0.5	1	3	9
Bromus_benekenii	5	1	20	2	-	-	12	0	180000	0.1	0.2	0.4	5	13	21	250	3000	0.6	2	30	9
Calamagrostis_arundinasius	5	1	0.5	2	-	-	1.8	0	20800	0.1	0.2	0.4	2	10	18	750	3500	0.5	2	3	0.67
Deschampsia_cespitosa	5	1	0.5	2	-	-	0.2	0	3000	0.15	0.35	0.6	3	11	19	1000	5000	0.35	2	3	0
Deschampsia_flexuosa	5	1	0.05	2	-	-	0.13	0	1950	0.05	0.15	0.3	-1	7	15	250	3000	0.2	2	3	2.3
Festuca_ovina	10	1.4	0.02	2	10	1	0.1	0	1500	-0.25	0.05	0.25	3	11	19	1500	5000	0.1	1	30	0.67
Milium_effusum	5	1	20	2	-	-	8	0	150000	0.15	0.45	0.6	5	15	20	250	3000	0.5	2	3	9
Molinia_caerulea	5	1	1	2	-	-	0.2	0	3000	0.2	0.3	0.45	5	13	21	1000	5500	0.4	2	30	2.3
Nardus_stricta	10	1.2	0.05	2	10	1	0.2	150000	3000	0.15	0.25	0.4	0	8	16	1500	5000	0.15	2	1	0
Poa_nemoralis	5	1	5	2	-	-	8	0	120000	0.05	0.1	0.2	2	10	20	1250	5000	0.4	2	3	9
Dryopteria_dilata_coll	20	1	0.5	2	-	-	2	0	30000	0.1	0.3	0.5	3	11	19	150	2500	0.4	2	1	2.3
Pteridium	20	1	0.5	2	-	-	12	0	180000	0.05	0.2	0.3	2	8	18	750	3250	0.5	2	1	0
Aconitum_lycoctonum	20	1	5	2	-	-	10	0	150000	0.25	0.55	0.9	2	6	10	1000	5000	1	2	1	0
Allium_ursinum	2	1	20	2	-	-	40	0	600000	0.25	0.2	0.6	4	12	20	250	5000	0.25	2	30	0
Anemone_nemorosa	10	1	0.5	2	-	-	0.8	0	12000	0.2	0.3	0.4	2	10	18	250	3500	0.15	1	3	2.3
Antennaria_diocia	5	1	0.01	2	-	-	0.1	0	1500	0.05	0.1	0.2	0	6	12	2000	5500	0.01	1	1	0
Arnica_montana	5	1	0.01	2	-	-	0.6	0	9000	0.05	0.1	0.2	7	15	20	2000	5500	0.01	1	1	0
Epilobium_augustifolium	5	1	1	2	-	-	2	0	30000	0.15	0.2	0.3	0	8	20	1750	5500	0.8	2	3	32
Galium_odorata	3	1	5	2	-	-	1.2	0	18000	0.15	0.25	0.4	3	11	19	250	3000	0.15	1	1	0.67
Geranium_sylvestrum	3	1	1	2	-	-	1.8	0	27000	0.15	0.25	0.4	2	10	14	500	3000	0.5	2	3	9
Hepatica_nobilis	20	1	1	2	-	-	8	0	120000	0.15	0.25	0.4	2	10	18	375	3000	0.5	1	3	0
Mercurialis_perennis	5	1	5	2	-	-	2	0	30000	0.1	0.25	0.4	5	15	20	1000	5000	0.5	1	1	0
Origanum_vulgare	20	1	0.5	2	30	1	10	0	150000	0.05	0.15	0.25	4	12	20	1500	6000	0.04	2	3	0.67
Oxalis_acetocella	2	1	0.5	2	-	-	0.2	0	3000	0.1	0.2	0.4	0	8	18	250	3000	0.05	1	1	0
Trientalis	2	1	0.5	2	10	1	0.2	0	3000	0.1	0.2	0.4	2	10	18	250	3000	0.15	1	1	0.67
Trifolium_repens	5	1	1	0	-	-	1.3	0	19500	0.2	0.35	0.4	5	15	25	1250	5500	0.3	2	1	32
Urtica_dioica	5	1	5	2	-	-	10	0	150000	0.15	0.25	0.45	2	10	20	500	5000	0.8	1	3	0
Norway spruce	100	1	0.3	2	30	1	0.33	0	5000	0.1	0.4	0.9	5	15	20	400	700	0.25	1	3	0.7
Sitka spruce	110	1	0.1	2	3	1	0.07	0	1050	0.15	0.45	0.9	2	12	17	600	700	0.25	1	1	0.7
Scots pine	150	1	1	2	100	1	0.28	0	4730	0.05	0.3	0.8	3	13	18	1200	2296	0.2	2	1	0.7
Larch	70	1	1	2	30	1	0.5	0	7500	0.05	0.25	0.8	6	16	25	400	700	0.2	2	3	0.7
Birch	60	1	1	2	100	1	0.25	0	4000	0.15	0.45	0.9	2	12	17	800	1600	0.2	2	1	9
Beech	120	1	3	2	300	1	0.22	0	3500	0.15	0.45	0.7	6	16	30	320	600	0.25	3	3	2.3
Oak	160	1	3	2	300	1	0.2	0	3000	0.05	0.3	0.7	6.5	16.5	35	600	800	0.2	3	3	9
Ash	80	1	1	2	-	-	0.25	0	4000	0.15	0.45	0.7	7	17	35	600	1600	0.25	2	3	9
Norway maple	80	1	3	2	-	-	0.25	0	4000	0.05	0.3	0.7	5.5	15.5	25	160	280	0.2	3	3	9
Myrica_qale	10	1	1	2	-	-	0.8	0	12000	0.25	0.35	0.6	3	7	18	1500	4000	0.6	2	1	0.67
Rhododendron_toment	10	1	0.03	2	-	-	0.2	150000	3000	0.25	0.35	0.5	-1	5	9	1000	3500	0.5	2	1	0
Rubus_idaeus	5	1	1	2	-	-	1	0	15000	0.15	0.25	0.4	2	10	18	1500	5000	0.8	2	3	9
Salix_lanata	30	1	0.5	1	0.1	3	1	0	9000	0.15	0.35	0.6	-2	2	6	1000	4000	1.2	3	1	2.3
Salix_myrsinifolia	30	1	0.5	2	-	-	0.5	0	9000	0.15	0.35	0.6	-1	5	11	1000	4000	1.2	3	1	9

6

Dynamic modelling of effects of deposition on carbon sequestration and nitrogen availability : VSD plus C and N dynamics (VSD+)

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6.1 Introduction

In recent years, there has been a shift in focus regarding the effects of atmospheric deposition. Initially, the main attention was on acid deposition and its effects on soil acidification and subsequent negative effects on vegetation and aquatic organisms through elevated aluminium levels. In addition, simple models and empirical data were used to assess the effect of nitrogen deposition on vegetation. Recently, also effects of nitrogen deposition on biodiversity, greenhouse gas emissions (notably N_2O), and carbon sequestration have become policy relevant (e.g., see Hettelingh et al. 2008, Millennium Ecosystem Assessment 2005, EEA 2007). In order to simulate or predict these effects of atmospheric deposition, model outputs are required, additional to those provided by current soil chemistry models. To calculate effects on biodiversity parameters, such as nitrogen availability, information about C/N ratios in the soil and NO_3 and NH_4 concentrations in the soil solution is needed.

This chapter first presents the current approach of dealing with C and N processes in the Very Simple Dynamic (VSD) model. Subsequently, it introduces the alterations to the current VSD model that allow the simulation of additional parameters and have led to the VSD+ model.

6.2 C and N in the VSD model

Currently in the VSD model, N immobilisation is the sum of two processes (Posch and Reinds 2009):

- a constant immobilisation ($N_{i,acc}$);
- and immobilisation dependent on the current C/N ratio of the soil ($N_{i,C/N}$)

In the latter process, no N is immobilised below a certain ratio (often C/N < 15) and immobilisation increases linearly with C/N ratio until a maximum (often at C/N ≥ 30), above which all available N (i.e. N deposition minus N uptake minus a minimum N leaching) is immobilised.

Carbon sequestration follows N immobilisation, again in two ways:

- for the constant N immobilisation, C sequestration is proportional to immobilisation using the current C/N ratio of the soil;
- the model includes an additional C sequestration that is proportional to the C/N dependent N immobilisation using a fixed C/N ratio (C/N_{seq}).

Figure 6.1 shows the effects of the above mentioned approach on the size of C and N pools and C/N ratio for a hypothetical, but realistic, scenario of N deposition. In the first (red line) calculations, there is only a C/N dependent N immobilisation and no C sequestration, that is, C/N_{seq} = 0. The second (blue line) calculations include a constant immobilisation and the last (green line) calculations also include C sequestration related to $N_{i,C/N}$.

All calculations show that, despite the decrease in N deposition, the C/N ratio will decrease down to a certain minimum. This minimum is either C/N_{min} of C/N_{seq}. In addition, the figure shows that the C and N pools will never decrease, because the model does not include C and N mineralization, only immobilisation. N mineralization is required to calculate

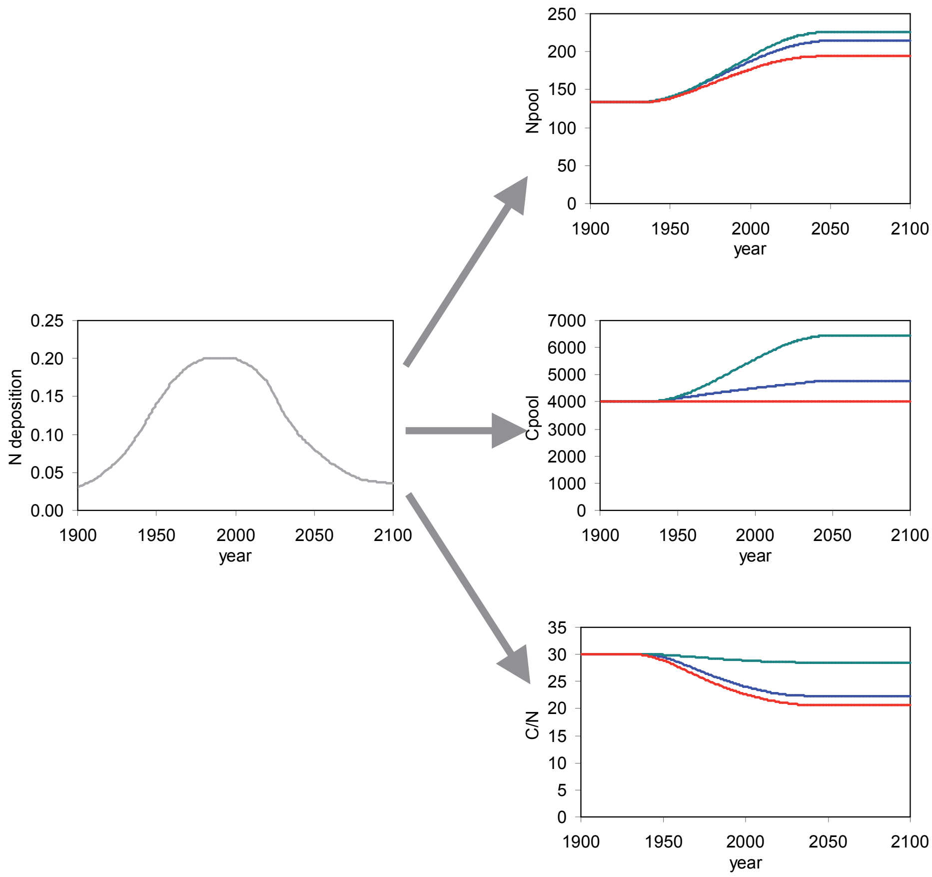

Figure 6.1 Effects of N deposition on calculated C pool, N pool and C/N ratio for VSD.

N availability, which might be used as an input for assessing plant species biodiversity. The absence of C mineralization prevents the model from being used to properly calculate C sequestration. Therefore, we extended the VSD model with C and N dynamics, which is presented in the next section.

6.3 Incorporating C and N dynamics in VSD

For the new C and N model (CNorg), we assumed soil organic matter to consists of four different pools, each with a fixed C/N ratio and a fixed turnover rate coefficient (Figure 6.2). Total soil organic matter is now the sum of all four pools, and the soil C/N ratio is the average of the C/N ratios of the individual pools, weighted by the size of the C pools. Litterfall and root turnover are distributed over the two fresh litter pools (easily degradable and recalcitrant fresh litter) depending on the C/N ratio of litterfall and root turnover.

The C/N ratio in litterfall is dependent on N deposition and type of vegetation. For this, we used measurements of deposition and N contents to define a relation between these two parameters (Figure 6.3; example for pine).

In addition to the new sub-model for C and N turnover, we also modified the way other N processes are calculated. Figure 6.4 gives an overview of all the N processes included in VSD+. The numbers indicate the sequence in which the different processes are modelled.

Compared to VSD, the following changes were made in the N processes:

- uptake of N is dependent on the vegetation growth and litterfall
- uptake of NH_4 is preferential over uptake of NO_3;
- nitrification is calculated as a rate-limiting process, and complete nitrification is no longer assumed, allowing for the calculation of NH_4 concentrations;
- both nitrification and denitrification are dependent on T, moisture and pH.

The next step in the development of VSD+ was the calibration of the new C and N model (CNorg). With this calibration we tried to obtain sound default values for the parameters of CNorg, in order to reduce the number of input parameters that should be provided by future users of the model. For the calibration, we applied a Bayesian calibration using data from the following five chronosequences:

- a spruce site in Sweden;
- a spruce site in Gejlvang, Denmark;
- a spruce site in Vestskoven, Denmark;
- an oak site in Vestskoven, Denmark;
- an oak site in Sellingen, the Netherlands.

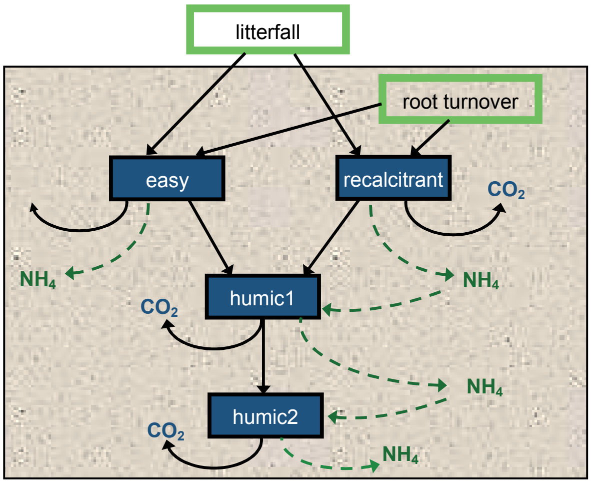

Figure 6.2 Schematic representation of a organic matter model.

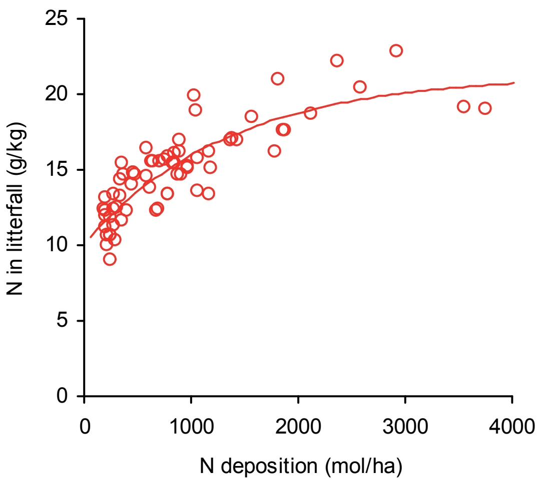

Figure 6.3 N content in litterfall of pine as function of N deposition; measurements (from: De Vries et al. 2000) are indicated by dots, and the line shows the regression equation used in VSD+.

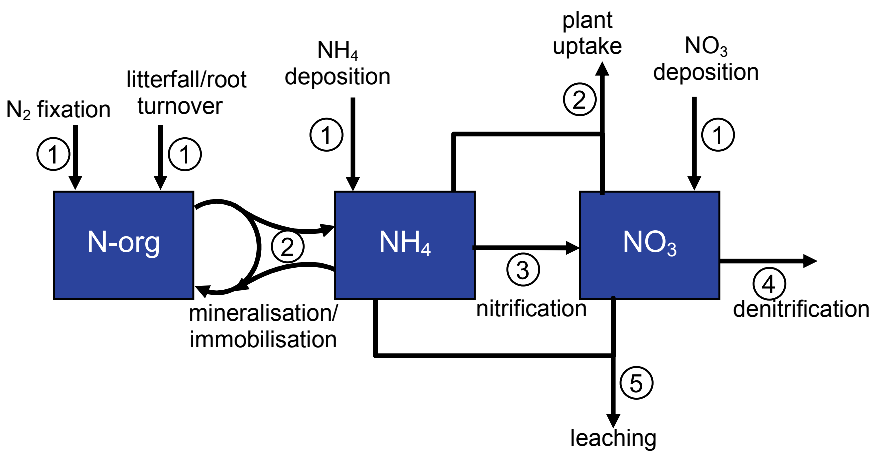

Figure 6.4 Schematic overview of N processes in VSD+.

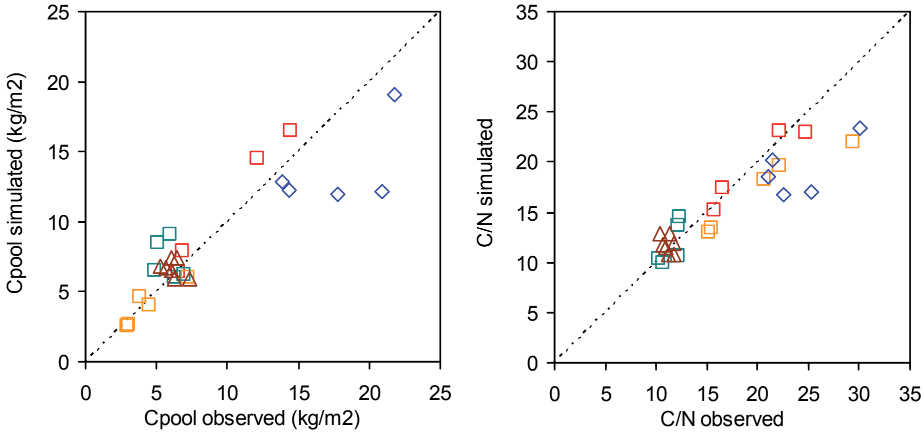

Figure 6.5 Observed and calculated values for C pool (left) and C/N ratio (right) for five chronosequences after calibration of the parameters of the C and N model.

Table 6.1 Calibrated values for the rate of mineralization (kmin) and C/N ratios of the C pools

C pool	k_{min} (yr^{-1})	C/N (g/g)
fresh easily degradable	8.7	17
fresh recalcitrant	0.06	295
humic I	1.0	9.5
humic II	0.0013	9.5

Table 6.2 Overview of new and obsolete input parameters for VSD+ (parameters with default values are marked grey).

	new input for VSD+	obsolete input
biomass cycling of nutrients	minimum N content of stems	net uptake of N, Ca, Mg, K
	maximum N content of stems	
	N availability above N_{stems} is constant	
	N availability below N_{stems} is constant	
	minimum N content in litterfall [a)]	
	maximum N content in litterfall [a)]	
	exponent to relate $N_{litterfall}$ to N deposition [a)]	
	Ca, Mg and K content of stems	
	N contents of root turnover	
vegetation	vegetation type	
	growth function (2, 3 or 4 parameters, depending on vegetation type)	
	ratio between litterfall and root turnover*a)*	
	age of vegetation	
mineralization, immobilisation	turnover rate constants of C pools (4)	constant N immobilisation rate
	fraction mineralization of C pools (3)	max. C/N for C/N dependent N immob.
	C/N ratios of C pools (4)	min. C/N for C/N dependent N immob.
	reduction of mineralization, due to moisture and temperature[b)]	C/N for C sequestration proportional to C/N dependent N immobilisation
	N fixation rate	
nitrification, denitrification	nitrification rate constant	fraction denitrification
	denitrification rate constant	
	reduction in (de)nitrification, due to moisture and temperature[b)]	
leaching		minimum N concentration in leaching

[a)] default values have been derived for the following vegetation types: spruce, pine, broadleaf softwood, broadleaf hardwood, evergreen broadleaf, shrubs, grassland, tundra/peat/heather.
[b)] values for reduction in mineralization, nitrification and denitrification have to be derived from soil temperature and moisture on a daily or weekly basis. A simple hydrological model shall be developed to derive these values.

In the calibration, calculated values for the C pool, N pool and C/N ratio were compared to observed values. Figure 6.5 shows the observed and calculated values of the C pool and the C/N ratio after calibration.

Table 6.1 shows the parameter values for the turnover rates and the C/N ratios of the C pools.

The calibrations showed that a single set of parameter values can be used, reasonably well, for describing C sequestration and evolution of C/N ratios for different forest sites with different tree species and deposition rates.

6.4 Application of the VSD+ model

Because new processes are included in the VSD+ model and the mathematical description of other processes has changed, the model requires several new parameters, while some other parameters from VSD have become obsolete (Table 6.2) The number of new parameters and for which no default values are available, is only slightly larger than the number of parameters that have become obsolete. The VSD+ model, thus, still requires a minimum set of input data and can, just as the original VSD model, be applied on regional and national scales.

A stand-alone single-site version of the VSD+ model with graphical user interface (called VSDp-Studio) was made – similar to the VSD-Studio model – to facilitate the use of the model.

6.5 Future developments

The VSD+ model has been developed to provide additional output parameters required to simulate biodiversity and vegetation responses to changes in N deposition. In the near future, the VSD+ model, therefore, will be linked to a model, or models, that describe these responses.

The VSD+ model will also be tested on several sites in Europe and, subsequently, be applied to some European regions. Finally, the steady-state (or stationary) solution(s) of the VSD+ model will be derived to enable the calculation of critical loads.

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Part 3 Options to further quantify effect indicators

Part 3 includes three chapters each addressing indicators that could be suitable for capturing the change in biodiversity under the LRTAP Convention in a way that is understandable for policy makers and the general public, and that could attach to air pollution the attention it deserves in biodiversity policy. Chapter 7 proposes to generalise the criteria behind the red-list species (rarity and decline) to include all species. In Chapter 8 a distance measure between physically and potentially available plant species is discussed as possible indicator for nitrogen-induced changes of plant species diversity. In Chapter 9 an approach is explored to interpret habitat suitability predictions using lists of predefined indicators that are rare and typical as well as indicators that are untypical and invasive of a habitat.

7

A Red-List-based biodiversity indicator and its application in model studies in the Netherlands

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7.1 Diversity indices

Diversity is an ecosystem property that is the subject of ongoing debate among scientists. Since the introduction of the term by Fisher et al. (1943), mathematical and ecological properties of diversity indices have been studied extensively. In its original sense, diversity is a strictly quantitative property of an ecosystem that can be derived from observations of species quantities using a well-defined mathematical procedure. In theory, this diversity has two components:

- the number of species: an ecosystem with more species has a higher diversity;
- the relative abundances of the species: an ecosystem that is completely dominated by a single species is less diverse than a system with the same number of species in about equal proportions. This property is usually referred to as 'evenness'.

Many diversity indices weight these two components to different degrees. It has been shown that all these indices can be generalised to a single formula (Baczkowski et al. 1997):

$$H(\alpha,\beta) = \sum_{i=1}^{s} p_i^{\alpha}(-\log p_i)^{\beta}$$

in which *s* is the number of species, p_i is the quantity (abundance) of species *i*, and α and β are parameters that determine the behaviour of the index.

There are two popular elaborations on the above formula:

- The Shannon-Weaver index = $H(1,1)$, which can be ecologically interpreted as one minus the chance to correctly guess the name of an individual taken randomly from the population;
- The Simpson index = $H(2,0)$, which can be ecologically interpreted as one minus the chance of two individuals taken randomly from the population belong to the same species.

Ever since they were first publicised, diversity indices have been the subject of debate among scientists. A common criticism, for example as expressed by Hurlbert (1971), is that, in spite of their mathematical elegance, the indices have little relation with what really happens inside the ecosystem. A concept that was especially popular in the 1950s (e.g. Odum 1953, Elton 1958) was that the diversity of an ecosystem was related to its stability, that is, its ability to withstand external changes. This hypothesis was also greatly criticised (e.g. May 1973, Van Dobben and Lowe-McConell 1975), as a sound theoretical framework is lacking and, in practice, its falsification is virtually impossible. Still, it may be correct, at least in a probabilistic sense and within ecosystems (McCann 2000).

7.2 Biodiversity

The term *bio*diversity started to appear around 1985, but became strongly burdened, politically, after the Rio Convention in 1992. In contrast to the diversity indices, which have strictly mathematical definitions, biodiversity is a rather loosely defined entity that is usually associated with ecosystems in a pristine state, containing numerous species with many interactions between them. Since the Rio Convention, the conservation or promotion of biodiversity has become a political aim, at many levels, from the local to the global scale. This created a challenge to ecologists to find practical implementations for this term, and countless proposals have been made during the past decades. A recent study, carried out by the European Environment Agency

Table 7.1 Main classification of biodiversity indicators of the SEBI project (EEA 2007). Bold = related to the critical load concept; italic = related to species.

1	***Abundance and distribution of selected species***
2	***Red List Index for European species***
3	*Species of European interest*
4	Ecosystem coverage
5	Habitats of European interest
6	Livestock genetic diversity
7	Nationally designated protected areas
8	Sites designated under the EU Habitats and Birds Directives
9	**Critical load exceedance for nitrogen**
10	*Invasive alien species in Europe*
11	*Occurrence of temperature-sensitive species*
12	Marine Trophic Index of European seas
13	Fragmentation of natural and semi-natural areas
14	Fragmentation of river systems
15	Nutrients in transitional, coastal and marine waters
16	Freshwater quality
17	Forest: growing stock, increment and fellings
18	Forest: deadwood
19	Agriculture: nitrogen balance
20	Agriculture: area under management practices supporting biodiversity
21	Fisheries: European commercial fish stocks
22	Aquaculture: effluent water quality from finfish farms
23	Ecological Footprint of European countries
24	Patent applications based on genetic resources
25	Financing biodiversity management
26	Public awareness

(EEA), enumerates 655 proposed biodiversity indicators (EEA 2003), which can be gathered in 26 main groups (EEA 2007). Even these main groups contain a wide range of definitions, based on biotic, abiotic, administrative and societal indicators (Table 7.1). A priori, the following demands can be made on a biodiversity indicator, for it to be useful in the present political and ecological context:

- Unlike in the older diversity indices, a biodiversity indicator should be based on ethical, economical or social rather than mathematical considerations;
- It should agree with conservational attitudes, that is, an ecosystem's numerical value should correspond with conservationists' intuitive or generally accepted appreciation of that system;
- It should be ecologically meaningful and quantifiable, on the basis of data that are already available or can be easily collected;
- It should be politically useful, that is, politicians should be able to 'press buttons' that influence it.

7.3 Biodiversity in the context of atmospheric deposition

Besides the above-mentioned considerations, a biodiversity indicator has to fulfil a number of extra demands to be useful as an indicator for effects of atmospheric deposition:

- It should have a relation with vegetation, that is, with vascular or non-vascular green plants, because critical load is nearly always determined on the basis of effects on vegetation;
- It should be possible to estimate the indicator on the basis of model outcomes, that is, on the basis of abiotic conditions, which is a necessity if one wants to make projections into the future.

For the construction of our index, we made the following a-priori, ethical considerations:

- The notion behind most of the political implementations of biodiversity is that the extinction of species is something to be prevented, in the first place. After all, this is the only really irreversible damage that can be done to the world's biological heritage;
- For a species to be threatened with extinction, it should be both rare and declining.

Rarity and decline are exactly the criteria of the Red List (IUCN 2003). The Red List categorises species as susceptible, vulnerable, endangered or critically endangered, according to the combination of rarity and decline. The exact quantitative criteria may differ per species group and geographical region. Table 7.2 gives the classification of plant species for the Netherlands, as an example.

In themselves, species on the Red List are no useful indicators for the construction of an index. By definition, they are rare, so they will be absent in most observational data and, therefore, would not be practical indicators for the state of a given ecosystem. Also, their abiotic requirements are generally poorly known for exactly the same reason, as argued by Rowe et al. (this volume). However, the criteria of the Red List (rarity and decline) can be generalised and applied to all species (Figure 7.1). Data on rarity and decline are available in national floristic databases. The idea behind this generalisation is that a vegetation is valued higher when

Table 7.2 Red List categories for vascular plants in the Netherlands. LR = low risk, SU = susceptible, VU = vulnerable, EN = endangered, CR = critical, EXT = extinct.

% decine	< 25%	SU	LR	LR	LR
	25% - 50%	VU	VU	VU	LR
	50% - 75%	EN	EN	VU	SU
	75% - 100%	CR	EN	VU	SU
	100%	EXT			
		< 1%	1% - 5%	5%-12%	> 12%
		% squares(actual)			

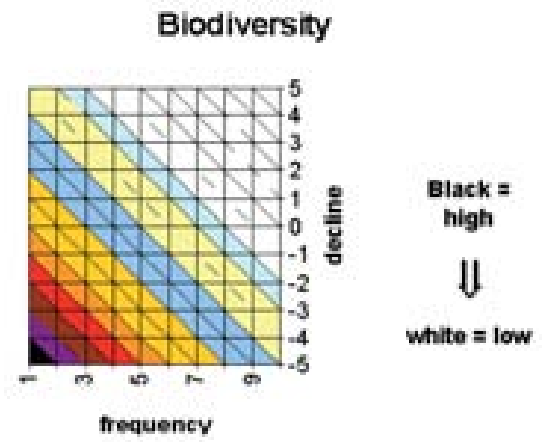

Figure 7.1 Schematic representation of dependence of biodiversity indicator on frequency and decline per species.

it contains more rare and/or declining species, and all the more if these species are rarer or more rapidly declining, even if no or only a few of the actual Red List species are present.

7.4 Elaboration of vascular plants in the Netherlands

In the Netherlands, flora is inventoried on the basis of 5×5 km grid squares. Species are divided into 10 rarity classes according to the logarithm of the number of grid squares in which they were present in the 1990-2000 period, and in five change classes according to the difference in rarity class in the period between 1900 and 1940 and between 1990 and 2000 (the five change classes represent differences in rarity of <-1, -1, 0, +1, >+1, respectively). The classes of both rarity and decline are expressed in numerical indices, per species, which are then added to form an index. Vegetation relevés (i.e., vegetation descriptions in terms of quantity per species) are used as the basis for evaluation; the following formula is used to combine the species' indices to a biodiversity score for the relevé:

$$H = \frac{1}{\log \max\{5, N\}} \sum_{i=1}^{N} Q_i \cdot S_i$$

in which H is the biodiversity score for the relevé, N is the number of species in the relevé, Q_i is the weighting factor for species i, and S_i is the index of species i. The weighting factor for each species is determined on the basis of its abundance, such that Q=1 for species present in very small quantities, Q=2 for species covering more than half of the plot, and

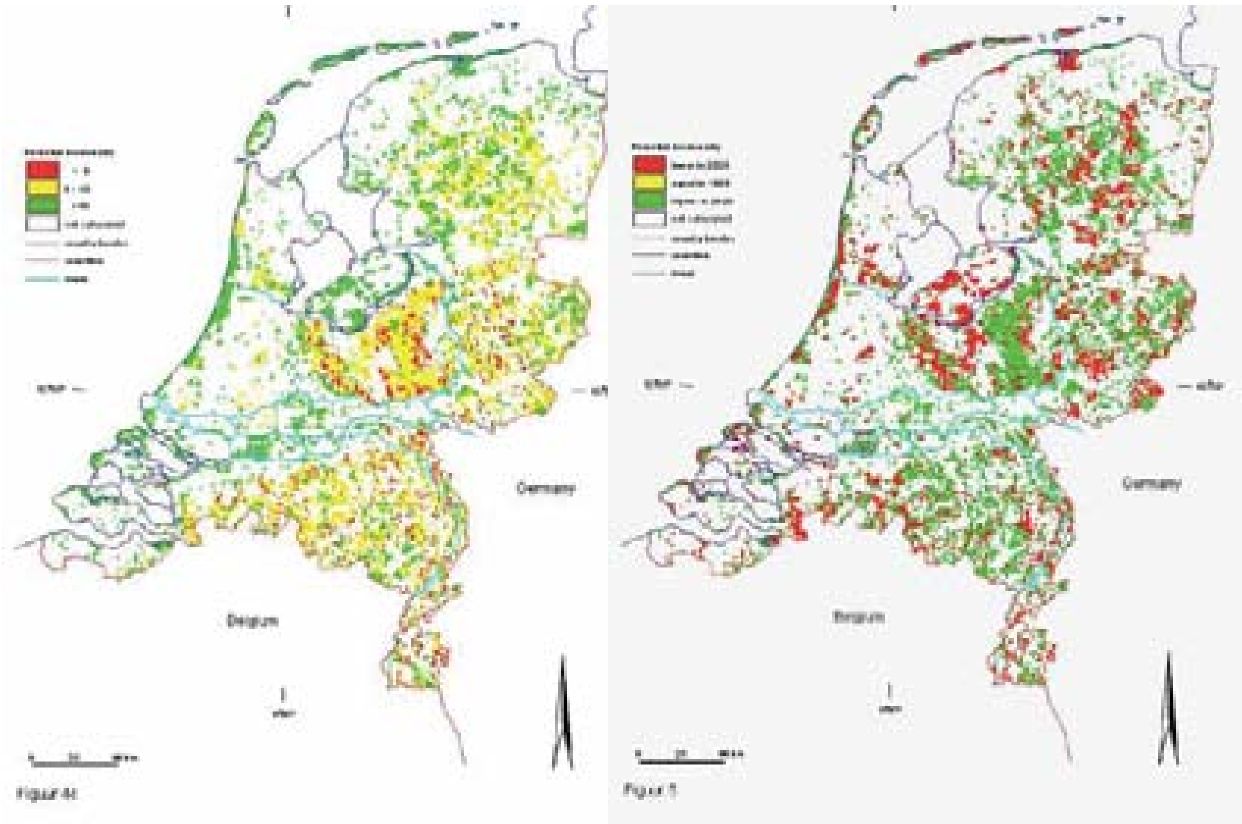

Figure 7.2 Potential botanical biodiversity inferred by the NTM model in 1995 (left), and change in 2020 (right).

logarithmically increasing with the species' abundance between >0 and 50% cover.

In this way, the final score of the relevé is mostly determined by the summed indices per species, with corrections for species present in large quantities (weighted more heavily) and species- poor relevés (score somewhat increased). Finally, *H* is square-root transformed to achieve a normal distribution.

The scores per relevé were validated by expert opinion, in two ways. First, a set of 55 relevés was assessed by vegetation experts, and their opinion was compared to the biodiversity scores derived from the above method. Second, a set of 160,000 relevés from all over the Netherlands was syntaxonomically identified and mean scores per syntaxon were determined. In addition, these mean scores were compared to expert opinion. In both cases, there appeared to be a good agreement.

7.5 Prediction of biodiversity in scenario studies

To be useful in policy-driven effect studies, a biodiversity indicator must be suitable for prediction in model systems. Models exist that predict probabilities of occurrence for individual species, and that can be linked to abiotic (atmosphere - soil) models ([GB]MOVE, Rowe et al. this volume, Latour et al. 1994). Such models, in turn, could be linked to the above procedure to produce an estimate for biodiversity under a given scenario. However, the uncertainty of these per-species models is very high (Wamelink et al. 2001). As the probability-per-species models are just statistical descriptions of the relation between abiotic conditions (usually pH and availabilities of water and nitrogen) and the occurrence of species, uncertainty can be reduced by skipping the species step and directly linking biodiversity to abiotic conditions. The NTM (Vegetation Evaluation Model) model is built on this idea. It is a regression model in which the above biodiversity index H is described as a function of vegetation structure type (grassland, heathland, deciduous forest, pine forest) and the abiotic conditions groundwater level, soil pH and nitrogen availability. The model uses spline functions to describe these relations, and includes all two- and three-way interactions (Wamelink et al. 2003). Uncertainty in model output can be assessed through a bootstrap procedure; percentages of explained variance are in the order of 40 to 60%.

The NTM model has proven a useful tool for the evaluation of economic scenarios (Wamelink et al. 2003). The soil model SMART2 can be used to produce spatial pictures of soil pH and nitrogen availability as a function of (changes in) nitrogen en sulphur deposition, and hydrological models and scenarios can produce pictures of future groundwater levels. These can be combined by the NTM model to produce projections of (potential) (botanical) biodiversity under given scenarios (Figure 7.2).

7.6 Discussion

The above described indicator fulfils the criterion that it can easily be used in connection with an abiotic process model, and, therefore, can be used to evaluate deposition scenarios. Still, the validation of biodiversity indicators is a difficult issue - by nature, they have a number of arbitrary decisions behind them. Nevertheless, our indicator can be validated against expert opinion, because it can also be applied to vegetation relevés. One might wonder why our quantification of biodiversity has such a strong relation with abiotic conditions. The answer is that under the present circumstances, decline of species is primarily due to unfavourable abiotic changes, in particular nitrogen enrichment, acidification and desiccation, rather than due to the direct death of individuals (e.g., by 'harvest') or the destruction of their biotope. In fact, under certain rare abiotic conditions (e.g., the combination nitrogen-poor and lime-rich) far more threatened species occur than under other, more common combinations. However, an important disclaimer is that the NTM model merely predicts *potential* biodiversity, that is, the suitability of a given biotope for threatened species; it does not predict their actual occurrence. Actual occurrence of threatened species depends on factors such as accessibility, presence of diaspores and site management.

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8

Assessment of the effects of top-soil changes on plant species diversity in forests, due to nitrogen deposition

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8.1 Introduction

Plant-available nitrogen has proved to be the most significant site factor differentiating vegetation in forests not influenced by ground water, within a given climatic region. This is the reason why atmospheric nitrogen inputs have been leading to rapid and large-area vegetation changes. Since the middle of the last century, anthropogenic pollution originating from agriculture, industry, and vehicular traffic has induced a rapid change in forest vegetation, within the space of only a few decades. The input of atmospheric nitrogen compounds has been identified as the main driving force of the observed vegetation change in various studies (Hofmann, 1972; Ellenberg, 1985; Hofmann *et al.*, 1990; Bücking, 1993; Bobbink *et al.*, 1998; Anders *et al.*, 2002; Jenssen and Hofmann, 2005).

International efforts based on the Geneva Convention on Long-Range Trans-boundary Air Pollution require effect indicators, also in respect of the protection target 'biodiversity of forest vegetation'. This chapter discusses possible indicators for nitrogen-induced changes in plant species diversity. Special attention will be given to the reference states and to the question of which deviations from these reference states may be harmful. Following Nilsson and Grennfelt (1988), a critical load is defined as 'a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge`. As a first step, our study aims at the derivation of critical limits with respect to the nitrogen-induced changes of top-soil parameters. Based on these critical limits, critical loads of nitrogen deposition can be derived by the application of biogeochemical process models relating regional N deposition to topsoil changes, quantitatively.

8.2 Method

Measuring plant species diversity

Plant-species diversity was measured by diversity profiles (Hill, 1973; Tóthmérész, 1995; Walker *et al.*, 2003; Jenssen, 2010)

$$H = \frac{1}{1-q}\ln\left(\sum_{i=1}^{S} p_i^q\right) \qquad (1)$$

with p_i denoting relative species abundances. Relative species abundances were estimated by the cover percentages D_i of species *i*:

$$p_i = \frac{D_i}{\sum_{j=1}^{S} D_j}$$

The parameter *q* weights relative abundance of species. Lower values of *q* give greater weight to rare species. For *q*=0 each species makes the same contribution to diversity *H* and Equation (1) yields logarithmic species numbers. Higher values of *p* give greater weight to more abundant species. Equation (1) with *q*=2 yields a diversity measure related to the well-known Simpson index (Simpson, 1949). As *q* approaches the limit one, it can be shown that

$$\lim_{q \to 1} H = -\sum_{i=1}^{S} p_i \ln p_i \qquad (2)$$

yields the well-known Shannon diversity (Shannon, 1948; Khinchin, 1957; Rényi, 1961, 1970). Shannon diversity mainly stresses medium abundances (relative abundances of the order *1/e*), whereas very small or very large p_i provide only minor contributions to this measure.

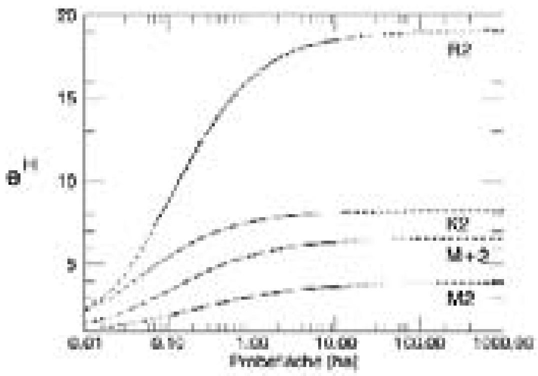

Figure 8.1 Saturation of Shannon diversity due to Equation (2) with increasing (accumulated) sample area for four different beech-forest ecosystems of east German lowlands growing on mesotrophic (M2), meso- to eutrophic (M+2), eutrophic (K2), and hypertrophic (R2) soils. The dashed sections of the curves were modelled using a hyperbolic saturation equation (Jenssen, 2007). Shannon diversities approach saturation values corresponding to the vegetation potentials of the sites.

Relating plant species diversity with ecosystem functioning

Relations between parameter *q*, spatial scale, and ecosystem functioning, were studied in previous work (Jenssen, 2010). Shannon or Simpson diversities are well suited to describe vegetation patterns that are strongly related to ecosystem processes, such as primary production, nutrient cycling, and water balance. The latter processes are mainly controlled by species in high and medium abundance. Low values of these diversity indices may indicate a strong limitation of resources, but also a high adaptation of a few species to the site conditions. These ecosystem processes are tightly related to plant-available energy, which is determined by local site factors (climate, soil, relief) and forest management.

Low values of parameters that give more weight to rare species have great importance to the adaptability of ecosystems. Rare and transient species provide the potential for development of new dominants and subordinates after ecosystem perturbations balance (Grime, 1998). The corresponding vegetation patterns can be observed on rather large spatial scales and are the result of a complex networking of many processes on a landscape scale.

Defining reference states

As a reference state of vegetation, we define the site potential for vegetation with the given kind of forest management prior to the observation of area covering and significant vegetation changes due to nitrogen deposition. Accumulating vegetation relevés of a particular ecosystem type that correspond to similar ecological constraints (plant-geographical region, climate and altitude, soil and relief, management) leads to a saturation of diversity measures, such as the Shannon diversity with increasing sample number *n* (Jenssen, 2007, 2010), (Figure 8.1):

$$\lim_{n\to\infty} H = H^O \qquad (3)$$

These saturation diversities correspond to distributions of plant species abundances

$$H(p_1,..,p_S) \xrightarrow[n\to\infty]{} H^O\left(p_1{}^O,..,p_S{}^O\right) \qquad (4)$$

which are used as a reference for species composition of plant assemblages that can be assigned to the particular ecosystem type. These reference distributions of plant species can be obtained from a sufficiently large number of sample relevés that are subject to the same site factors and to similar management conditions. Selecting relevés that have been unmanaged for long times would deliver species distributions that correspond to the so-called potential natural vegetation of a particular forest site (Tüxen, 1956). However, natural vegetation potentials are modified by forestry, which frequently grows other tree species than nature would do on a particular site. Hence, we are also led to consider these modified vegetation potentials, in order to assess the influence of nitrogen deposition on species diversity.

Measuring the distance to reference states

The 'deviation' of a plant assemblage from the vegetation potential of the corresponding forest site can be measured by the so-called Kullback information (Kullback, 1951):

$$K\left(p_1,..,p_S,p_1{}^O,..,p_S{}^O\right) = \sum_{i=1}^{S} p_i \ln\left(\frac{p_i}{p_i^O}\right) \qquad (5)$$

Kullback information measures the information gain when observing a particular vegetation state, compared to the knowledge about the vegetation potential of the site. Low values of Kullback information correspond to a high similarity between the actual vegetation state and the vegetation potential of the site.

As a second distance measure, the percentage similarity of species composition in relation to the vegetation potential of the site, will be used:

Table 8.1 Classes of endangerment of plant-species diversity in forests, with respect to quantitative effect indicators according to Equations (5) and (6).

Class	Per cent similarity SI		Kullback distance K		Number of protected species N
Not endangered	$SI>SI_{min}$ AND $\Delta SI<0,5*(100\%-SI_{min})$	AND	$K<K_{max}$ AND $\Delta K<0,5*K_{max}$	AND	$N>N_{min}$ AND $\Delta N<0,5*(N_{max}-N_{min})$
Potentially endangered	$SI>SI_{min}$ AND $\Delta SI>0,5*(100\%-SI_{min})$	OR	$K<K_{max}$ AND $\Delta K>0,5*K_{max}$	OR	$N>N_{min}$ AND $\Delta N>0,5*(N_{max}-N_{min})$
Significantly endangered	$SI<SI_{min}$ AND $\Delta SI<0,5*(100\%-SI_{min})$	OR	$K>K_{max}$ AND $\Delta K<0,5*K_{max}$	OR	$N<N_{min}$ AND $\Delta N<0,5*(N_{max}-N_{min})$
Extremely endangered	$SI<SI_{min}$ AND $\Delta SI>0,5*(100\%-SI_{min})$	OR	$K>K_{max}$ AND $\Delta K>0,5*K_{max}$	OR	$N<N_{min}$ AND $\Delta N>0,5*(N_{max}-N_{min})$

ΔSI Change in percentage similarity SI (Equation 6) between deposition-induced vegetation state and vegetation potential before 1965

SI_{min} Difference of mean and standard deviation of SI (Equation 6) of all vegetation relevés of the ecosystem type before 1965

ΔK Change of Kullback distance K (Equation 5) from deposition-induced vegetation state to vegetation potential before 1965

K_{max} Sum of mean and standard deviation of K (Equation 5) of all vegetation relevés of the ecosystem type before 1965

ΔN Change in the number of protected species per area unit (600 m²), between deposition-induced vegetation state and vegetation potential before 1965

N_{max} - N_{min} Twice the standard deviation of the number of protected species corresponding to the vegetation potential before 1965

$$SI(p,p^0) = \sum_{i=1}^{S} \min(p_i\ ,\ p_i^0)\cdot 100\% \qquad (6)$$

This measure can be interpreted as a modified Sörensen index, considering species abundances instead of presence / absence only.

These distance measures can be considered as indicators of ecosystem integrity and the maintenance of ecosystem functions. Furthermore, the species protected by law have to be considered as a legislative protection target. For that, the nitrogen-induced change in the number of protected species was considered as a third effect indicator.

Defining classes of endangerment

The investigated time series and results of ecosystem research were used to define different classes of endangerment (Table 8.1). Forests are considered to be not endangered, provided the following two conditions are met: 1) species composition must be within the site-specific amplitude (vegetation potential), and 2) there are no strong dynamics of species composition induced by N deposition. Forests are considered to be potentially endangered if there are strongly induced dynamics of vegetation without exceeding the site-specific amplitude. They are significantly endangered if species composition exceeds the site-specific amplitude, that is, if a change in ecosystem type takes place. Forests are extremely endangered if this vegetation change proceeds rapidly.

Deriving critical limits of top-soil changes

The approach is based on the thesis that C/N-ratio (ratio of organic carbon and total nitrogen) in combination with pH value (measured in n/10 KCl) proves to be the key parameter of chemical top-soil state determining vegetation composition in forests not influenced by groundwater or backwater. This thesis was confirmed in long-standing and comprehensive studies on central European forest vegetation (Hofmann, 1974; Anders *et al.*, 2002; Jenssen and Hofmann, 2005).

In order to relate the observed vegetation changes to the induced changes in top-soil state, the author developed an indicator model relating the abundances of plant species with the top-soil parameters C/N ratio and pH, value both referring to the upper five centimetres of humus layer and/or mineral soil (Jenssen, 2009). The model is based on 1,643 relevés of central European forests with data on C/N-ratio and pH (KCl) of top soil. These data provide information on top-soil state for more than 800 plant species. For 321 of them, probability density functions were modelled with a statistical significance of p=0.05 (Figure 8.2).

The indicator value of plant species frequently depends on the soil cover or the vegetation height. For many species, a significant increase in soil cover or growing up to a higher vegetation layer decreases the amplitude of the probable top-soil state (Figure 8.3). For that reason, probability density functions were modelled for different classes of soil cover or different vegetation layers if the indicator value was statistically significant (p=0.05).

The modelling of top-soil state corresponding to a specific vegetation relevé is obtained by multiplication of the probability density functions of the occurring plant species (Figure 8.5). The model fits to the data (1,643 vegetation plots) with high accuracy (coefficient of determination r^2=0.82 for C/N, r^2=0.69 for pH).It was applied to more than 60 time series and chrono-sequences of deposition-induced vegetation dynamics. The purpose was to derive critical limits of top-soil state and top-soil dynamics corresponding to the classes of endangerment given in Table 8.1. These critical limits are strongly differentiated with respect to dominating tree species and edaphic site conditions.

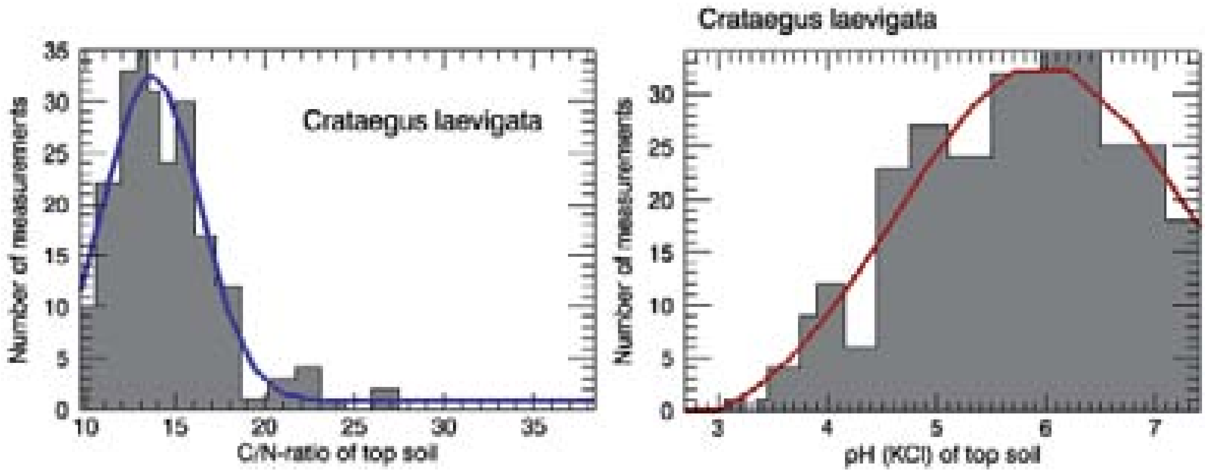

Figure 8.2 Histogram and derived probability density function of Crataegus laevigata versus top-soil state.

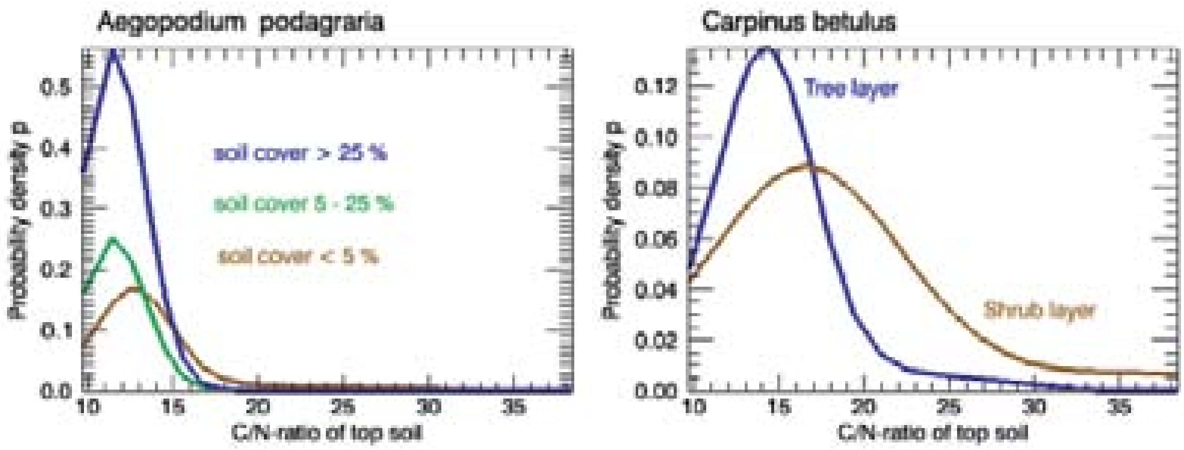

Figure 8.3 The indicator model considers soil cover (left) and vegetation layer (right) of plant species.

8.3 Data

The study is based on a database of forest vegetation of northern central Europe, comprising about 13,000 vegetation relevés, recorded mostly between 1950 and 2007 (Jenssen, 2007). For 1,643 of them, reliable measurement data of top-soil parameters C/N ratio and pH value (KCl) were recorded. Moreover, the reference states (site-specific vegetation potentials) were derived from vegetation relevés recorded before 1965. This restriction was made because significant and area-covering vegetation changes due to N deposition had been observed since the beginning of the 1970s (Ellenberg, 1985; Hofmann *et al.*, 1990; Bücking, 1993; Bobbink *et al.*, 1998). This observation coincides with the development of nitrogen emissions in Germany (Ulrich, 1989). Moreover, vegetation relevés spread over a relatively broad climatic range were summarised for deriving vegetation potentials. These vegetation potentials can be used as site-specific reference states under the condition of increased climatic variability.

The following time series of forests exposed to N deposition were analysed:

- Scots pine forests on oligotrophic sandy soils, exposed to
  - massive N deposition due to local emission sources in the 1980s, dynamics between 1967 and 2007
  - controlled massive N deposition in different combinations with other compounds (fertilisation experiments), dynamics between 1963 and 2002
  - strong N deposition due to local emission sources in the 1980s, dynamics between 1959 and 2006
  - weak N deposition due to local emission sources in the 1980s, dynamics between 1963 and 2004
  - far away from local emission sources, dynamics between 1935 and 2006
- Scots pine forests on mesotrophic sandy soils, exposed to
  - weak N deposition due to local emission sources in the 1980s, dynamics between 1964 and 2004
  - far away from local emission sources, dynamics between 1993 and 2004
- Scots pine forests on eutrophic loamy soils (calcareous), exposed to

Figure 8.4 Cladonio pinetum in Central Brandenburg. (Photo: G. Hofmann).

  - strong to massive N deposition due to local emission sources in the 1980s, dynamics between 1959 and 2006
  - N deposition originating from agricultural crops, dynamics between 1957 and 2004
- Oak forests on oligotrophic to mesotrophic sandy soils, exposed to
  - strong N deposition due to local emission sources in the 1980s
  - far away from local emission sources
- Beech forests on mesotrophic sandy soils
  - far away from local emission sources, developed from a Scots pine forest, dynamics between 1962 and 1998
- Beech forests on eutrophic loamy soils, exposed to
  - strong N deposition originating from agricultural crops, dynamics between 1962 and 1993
  - far away from local emission sources, dynamics between 1959 and 2004

The time series are documented by repeated vegetation records. Furthermore, data about deposition, forest growth, nutrient contents of needles and leaves, and chemical topsoil parameters were measured for a part of the investigated vegetation plots. These data were the basis for a comprehensive ecological interpretation of the observed vegetation dynamics (Anders *et al.*, 2002).

8.4 Results

The derivation of critical limits for top-soil changes due to N deposition is demonstrated here for one example in detail. The author refers to the forest ecosystem *Cladonio-Pinetum* that proves to be one of the most nutrient-poor forests occurring in north-east Germany. The tree layer is dominated by Scots pine (*Pinus sylvestris*) of stunted growth (Figure 8.4). The gappy ground vegetation is covered by shrubby lichens, such as *Cladonia arbuscula, Cl. rangiferina,* and *Cl. gracilis* mostly. Moreover, mosses, such as *Dicranum scoparium, Dicranum spurium, Pohlia nutans,* and *Ptilidium ciliare* can be observed. The area of this forest type was increased by the use of tree-litter and other kinds of soil degradation in the past. However, since the middle of the past century, the area of this forest type was strongly decreased due to increasing nutrient deposition. In particular, protected lichens were reduced in favour of widespread mosses and grasses.

In 1964, a long-term fertilisation experiment in Scots pine forests was established in different regions of the east-German lowlands including this forest type (Hippeli, 1970; Hofmann, 1972; Hippeli and Branse, 1992). For this chapter, the time series of vegetation of the fertilisation experiment were used, in order to reconstruct the development of top-soil parameters C/N-ratio and pH (KCl) with different scenarios of nutrient addition (Figure 8.5). Furthermore, the quantitative effect indicators of plant species diversity were

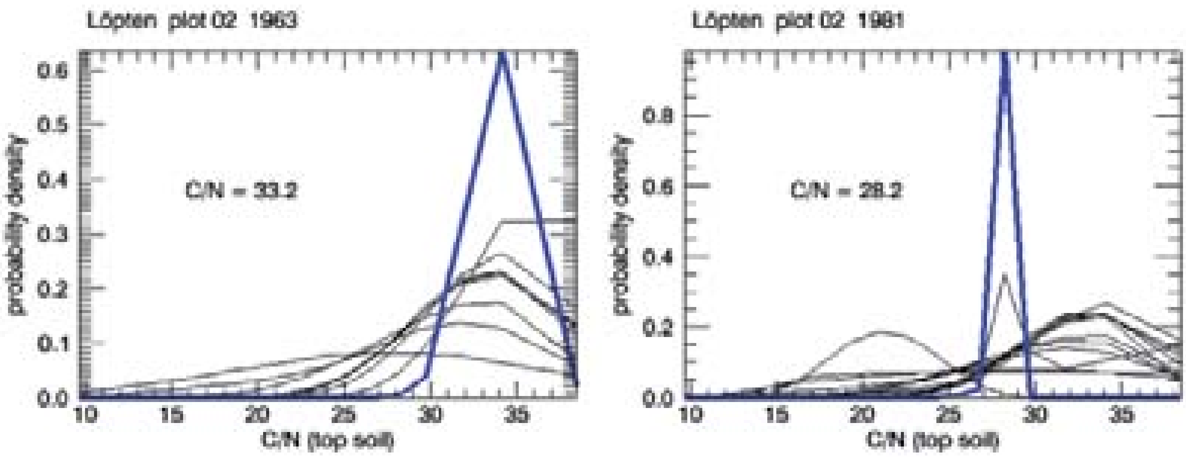

Figure 8.5 The chemical top-soil parameters were derived by multiplicative combining of probability density functions of the occurring plant species. The example shows the application of the indicator value to a Scots pine forest (Cladonio pinetum) on oligotrophic sandy soil with degraded top soil, in Central Brandenburg (plot 2 of fertilisation experiment Löpten, see Table 8.2) exposed to diffuse N deposition between 1963 and 1981.

Table 8.2 Development of a Cladonio-Pinetum on oligotrophic sandy soil with degraded top soil, subject to diffuse N deposition far away from local emission sources (fertilisation experiment Löpten in Central Brandenburg, 'zero'-plots without fertilisation). The C/N ratio is calculated from the vegetation composition by the indicator model. The classes of endangerment of plant-species diversity in forests are derived from Table 8.1.

	Plot 02			Plot 09			Plot 15		
	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:
1963	33.3	0	not	32.5	0	not	32.5	0	not
1966	33.1	0.2	not	32.1	0.4	not	31.8	0.7	not
1970	32.2	1.1	potentially	32.2	0.3	not	32.4	0.1	significantly
1981	28.2	5.1	potentially	28.1	4.4	significantly	28.2	4.3	significantly
1985	28.1	5.2	significantly	32.3	0.2	extremely			
2002	32.3	1.0	extremely	30.8	1.7	extremely	31.6	0.9	extremely

calculated over time, in order to derive different classes of endangerment as presented in Table 8.1. The results are presented in Tables 8.2 to 8.4. Furthermore, the indicator value was applied to 160 vegetation relevés of Scots pine forests on oligotrophic sandy soils of north-eastern Germany, recorded before 1965. The site-specific ecological amplitude of top-soil parameters turned out to be (mean value ± standard deviation):

C/N-ratio (0-5 cm): 31.7 ± 1.3

pH (KCl) (0-5 cm): 3.3 ± 0.1

The comparison of C/N-dynamics with the dynamics of species diversity (percent similarity, Kullback distance, number of protected species) and the derived classes of endangerment, shows some interesting results (Tables 8.2 to 8.4). Even without any fertilisation, and far away from local emission sources, there was an area covering impact of N deposition on forest vegetation beginning around 1970 and culminating in the 1980s. This impact decreased again after the decline in agriculture and industry, due to German reunification, as is indicated by the partial recovery of C/N-ratio in 2002 (Table 8.2). This result was confirmed by many other time series from the east-German lowlands, which provide striking evidence for area-covering impacts of nitrogen deposition on forest vegetation (Hofmann, 1987; Jenssen and Hofmann, 2005).

Furthermore, a considerable time delay can be observed between the emergence and disappearance of indicator plants that indicate top-soil change on the one hand, and the reorganisation of vegetation on the other hand (Tables 8.2 and 8.3). The shift of vegetation type takes place several years to decades after eutrophication, and recovery of the site was indicated by several plants. For the derivation of critical limits of top-soil parameters this time delay has to skipped, that is, the critical limits of C/N or pH have to be correlated with the classes of endangerment derived from diversity parameters measured later on (Table 8.5).

The effect on species diversity increases with the amount of nitrogen supply, as can be seen from Table 8.3. In particular, the additional input of calcium increases the nitrogen effect, considerably, and shortens the delay time of the effects (Table 8.4).

Table 8.5 presents the critical limits of C/N-ratio in Scots pine forests on oligotrophic sandy soils, which were derived from the site-specific ecological amplitude and the time series

Table 8.3 Development of a Cladonio-Pinetum on oligotrophic sandy soil with degraded top soil, subject to N-P-K-Mg-fertilisation (fertilisation experiment Löpten in Central Brandenburg). The fertiliser was added in two series (1964-1966: 360 kg N, 270 kg P_2O_5, 420 kg K_2O, 90 kg MgO per hectare; 1974-1976: 360 kg N, 300 kg P_2O_5, 300 kg K_2O per hectare). The C/N ratio is calculated from the vegetation composition by the indicator model. The classes of endangerment of plant-species diversity in forests are derived from Table 8.1.

	Plot 04			Plot 17			Plot 20		
	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:
1963	32.6	0	not	32.2	0	not	32.1	0	not
1966	27.2	5.4	significantly	30.6	1.6	significantly	31.6	0.5	potentially
1970	30.4	2.2	extremely	29.8	2.4	extremely	29.3	2.8	significantly
1985	24.8	7.8	extremely	26.2	6.0	extremely			
2002	30.0	2.6	extremely	31.6	0.6	extremely	30.3	1.8	extremely

Table 8.4 Development of a Cladonio-Pinetum on oligotrophic sandy soil with degraded top soil, subject to N-P-K-Mg-Ca-fertilisation (fertilisation experiment Löpten in Central Brandenburg). Additional to the fertilisation amounts presented in Table 8-3, also applied were 300 kg/ha $CaCO_3$ in 1964 and 2000 kg/ha $CaCO_3$ in 1974. The C/N ratio is calculated from the vegetation composition by the indicator model. The classes of endangerment of plant-species diversity in forests are derived from Table 8.1.

	Plot 03			Plot 14			Plot 12		
	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:	C/N	ΔC/N	endangered:
1963	32.4	0	not	32.8	0	not	31.9	0	not
1966	25.1	7.3	extremely	26.4	6.4	extremely	25.8	6.1	extremely
1970	29.1	3.3	extremely	23.0	9.8	extremely	25.8	6.1	extremely
1985	23.0	9.4	extremely	24.7	8.1	extremely			
2002	29.4	3.0	extremely	26.9	5.9	extremely	30.1	1.8	extremely

Table 8.5. Critical limits of C/N-ratio (top soil 0-5 cm) in Scots pine forests on oligotrophic sandy soils of the east-German lowlands, as derived from the results of this chapter and further data.

Class of endangerment	Deposition-induced change in top soil (humus layer)
Not endangered	C/N > 30 and ΔC/N < 0.5
Potentially endangered	C/N > 30 and ΔC/N > 0.5
Significantly endangered	C/N < 30 or ΔC/N > 1
Extremely endangered	C/N < 29 or ΔC/N > 2

presented in Tables 8.2 to 8.4 after skipping the time delay of nitrogen effects. These results were confirmed by further time series not explicitly presented here.

In the appendix, a series of critical limits for topsoil changes due to N deposition is presented for selected forest ecosystem types not influenced by groundwater or backwater. All the examples refer to the young-moraine regions of north-east Germany. Vegetation changes and important consequences for ecosystem integrity are sketched briefly. In some cases, further time series are presented in order to demonstrate the results.

8.5 Conclusions / Recommendations

The effects of N deposition on plant species diversity in forests are mainly determined by a narrowing of the C/N-ratio in the topsoil. In particular, heavy and long-standing N deposition may lead to 'disharmonious' topsoil states. In this case, the base saturation of topsoil is additionally changed. Hence, the topsoil parameters C/N-ratio and base saturation (or alternatively pH value) should be used for the parameterisation of models suited to assess and predict pollution-induced changes of species diversity. These parameters provide the interface between biogeochemical process models, on the one hand, and biodiversity assessment models on the other. Model coupling will lead to regionally differentiated predictions of nitrogen effects on plant-species diversity and to improved critical loads for N deposition.

Reference states for the assessment of vegetation changes are the vegetation potentials of different forest sites. Vegetation potentials are represented by species-abundance distributions corresponding to saturation diversities derived from a sufficiently large number of vegetation relevés with similar site conditions and ecological constraints. These site-specific reference states can be derived from suitable comprehensive databases. The assessment of nitrogen effects on diversity in central Europe requires the use of data that were recorded before 1970. Up to this time, the influence of nitrogen deposition on species diversity in forests can be assumed to have been restricted to the direct neighbourhood of a few local emission sources.

The Kullback distance and/or simple similarity indices (considering species abundances instead of presence / absence of species only) can be used to measure the deposition-induced deviation from the site-specific reference states. Kullback distance measures the deviation from the site potential in terms of Shannon diversity. Shannon diversity is proven to be a well-established diversity measure considering both evenness of species distributions and species richness merged into one parameter. In this way, different aspects of ecosystem integrity and ecosystem functioning can be considered when assessing the importance of nitrogen-induced changes in species diversity.

For this chapter, vegetation time series were used to derive different classes of endangerment of plant species diversity. From the results, we derived critical limits of topsoil parameters for a series of widespread forest ecosystem types in north-east Germany. Further work should be aimed at the derivation of reference states and critical limits relating to the harmful effects of N deposition on species diversity for the most important forest sites in all regions of Europe.

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Appendix

Scots pine forests on oligotrophic sandy soils

Vegetation:

Cladonio-Pinetum sylvestris, Festuco-Pinetum sylvestris, Calluno-Pinetum sylvestris, Vaccinio-Pinetum sylvestris

Dominating tree species: *Pinus sylvestris*

Protected species (600 m²): 2 – 8

Vegetation change with N deposition:

Decrease in lichens, mosses, and dwarf shrubs (*Cladonia-, Calluna-, Vaccinium* species), spreading of grasses (*Deschampsia flexuosa*)

Negative effects of N deposition on ecosystem integrity:

Impediment to natural regeneration, reduction in seepage water, loss of protected species (lichens, mosses) and of the natural habitat type

Table A8.1

Class of endangerment	Deposition-induced change in topsoil (humus layer)
Not endangered	C/N > 30 and ΔC/N < 0.5
Potentially endangered	C/N > 30 and ΔC/N > 0.5
Significantly endangered	C/N < 30 or ΔC/N > 1
Extremely endangered	C/N < 29 or ΔC/N > 2

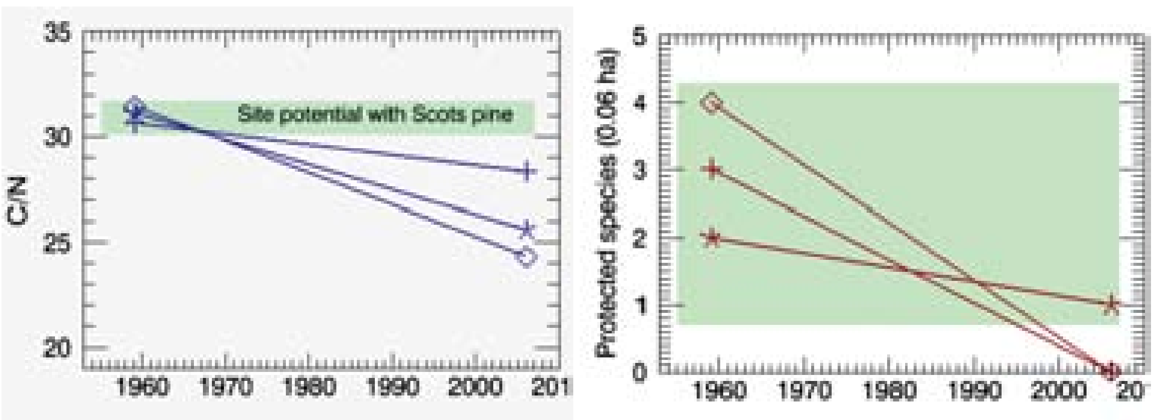

Figure A8.1 Change in C/N-ratio of topsoil (left, according to the indicator-value model) and decrease in protected species (right), for three investigation plots of a Scots pine forest on oligotrophic sandy soil (Calluno-Pinetum), near Schwedt / Oder in north-east Germany, with heavy N inputs between 1959 and 2006. The amplitude of the ecosystem type before 1965 is marked with a green colour.

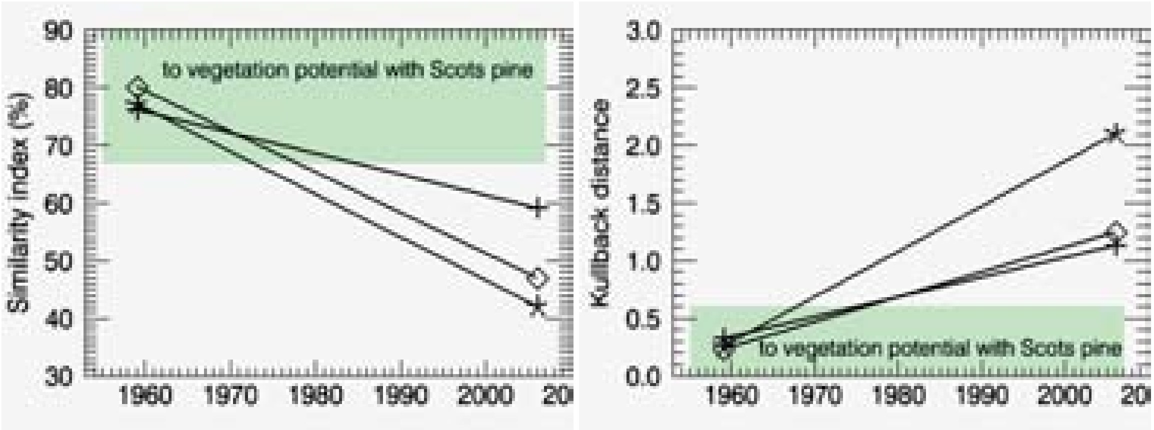

Figure A8.2 Similarity (left) and Kullback distance (right) with respect to the vegetation potential of the site with Scots pine before 1965 (investigation plots presented in Figure A1). The amplitude before 1965 is marked in green.

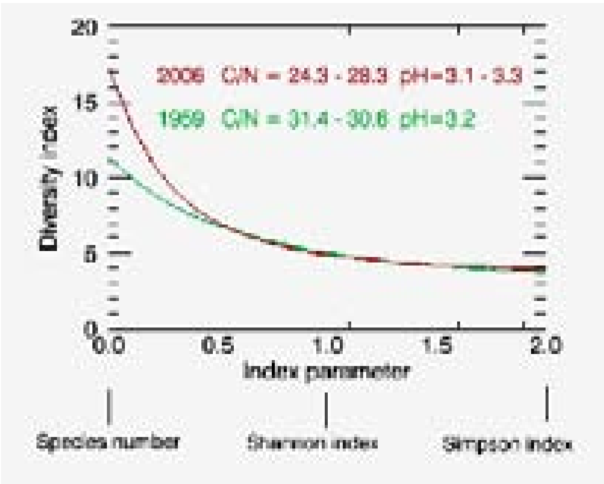

Figure A8.3 Change in the diversity profile between 1959 and 2006 under the influence of heavy N deposition (investigation plots presented in Figures A8.1 and A8.2). The initial dominance of Calluna vulgaris was replaced by the N-induced dominance of Deschampsia flexuosa. The vegetation change was accompanied by an increase in net-primary production of ground vegetation, a loss of the natural regeneration potential of Scots pine, and an increase of water stress. Total species number increased but rare lichens and mosses were reduced in favour of widespread nutrient-demanding species.

Secondary Scots pine forests on mesotrophic sandy soils

Vegetation: *Myrtillo-Culto-Pinetum sylvestris, Deschampsio-Culto-Pinetum sylvestris, Festuco-Culto-Pinetum sylvestris*

Dominating tree species: *Pinus sylvestris*

Protected species (600 m²): 0 – 3

Vegetation change with N deposition:
Decrease in mosses and dwarf shrubs (*Pleuozium-*, *Vaccinium* species), area-covering development of tall grasses (*Deschampsia flexuosa, Calamagrostis epigejos*), partially spreading of *Prunus serotina*

Negative effects of N deposition on ecosystem integrity:
Strong spreading of *Calamagrostis epigejos* leads to enhanced water stress for pine trees. N inputs initially lead to enhanced tree growth. With further N inputs, disharmonious nutritional states in pine needles are induced, leading to growth depression and increased tree mortality.

Table A8.2

Class of endangerment	Deposition-induced change in topsoil (humus layer)
Not endangered	C/N > 25 and ΔC/N < 1
Potentially endangered	C/N > 25 and ΔC/N > 1
Significantly endangered	C/N < 25 or ΔC/N > 2
Extremely endangered	C/N < 24 or ΔC/N > 4

Figure A8.4 Vegetation change in a Scots pine forest near Eberswalde (north-east Germany) on mesotrophic sandy soil, between 1970 and 2004. The forest was established on arable land at the beginning of the past century. The top soil was degraded until the 1970s. In the 1980s, the forest was subjected to heavy N inputs. At the beginning of the 1990s, the forest floor was completely dominated by Calamagrostis epigejos. After a decrease in N deposition, since 1993, a reversed development of vegetation was observed. Today, the ground vegetation again is dominated by Deschampsia flexuosa, which corresponds to the site-specific vegetation potential.

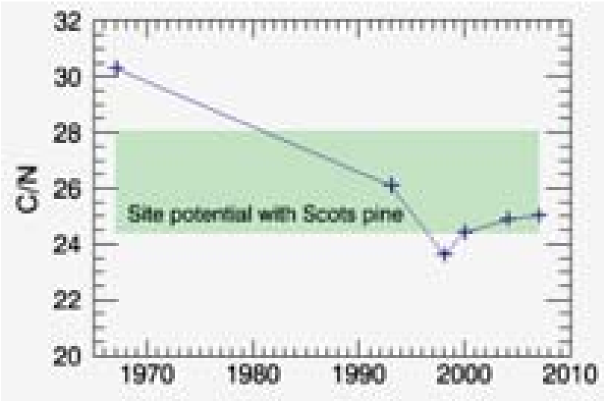

Figure A8.5 Change of C/N ratio of topsoil (according to the indicator-value model) in the Scots pine forest presented in Figure A8.4. The amplitude of the ecosystem type before 1965 is marked in green. The state at the end of the 1960s shows the degradation of the topsoil.

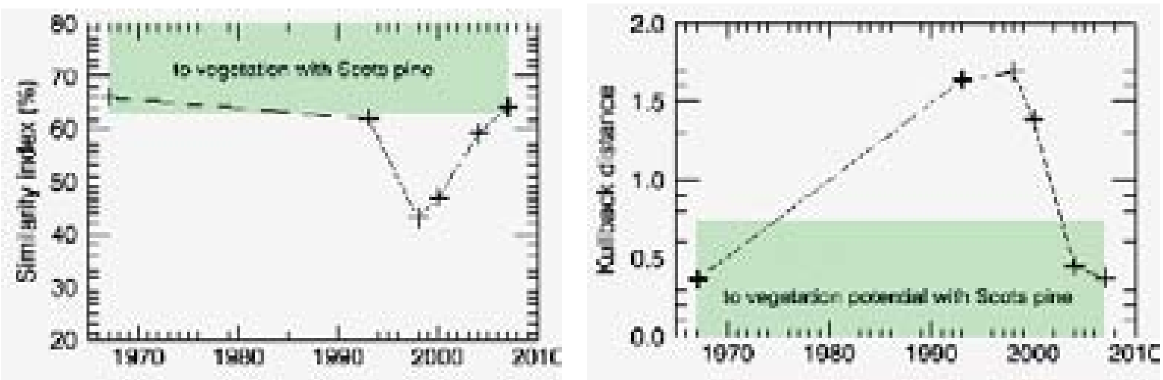

Figure A8.6 Similarity (left) and Kullback distance (right), with respect to the vegetation potential of the site with Scots pine, before 1965 (investigation plots presented in Figures A4 and A5). The amplitude before 1965 is marked in green. Vegetation dynamics display the resilient behaviour of topsoil state (Figure A8.5).

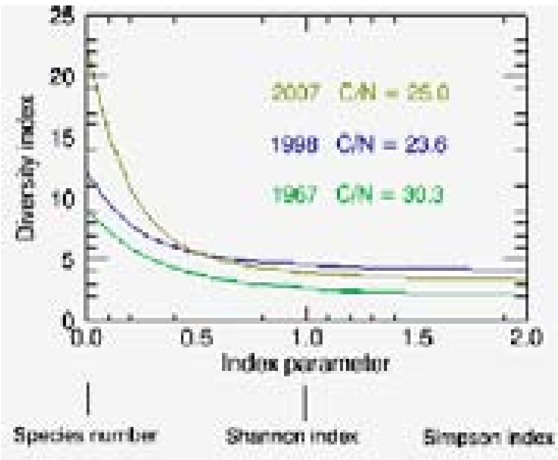

Figure A8.7 Change of the diversity profile between 1967 and 2007 (investigation plot presented in Figures A8.4 to A8.6). Heavy N inputs induced an overall increase in diversity, compared to the initial state with degraded topsoil. Ground vegetation had been dominated increasingly by Calamagrostis epigejos. Disharmonies in tree nutrition had led to increased tree mortality by the end the 1980s. With decreasing N deposition after 1993, the dominance of Calamagrostis epigejos has weakened again and species numbers increased significantly, due to the diminished competition of Calamagrostis epigejos. Nevertheless, nutritional disharmonies have been overcome and trees grow better now, compared to the 1960s.

Scots pine forests on mesotrophic sandy soils, sub-continental climate, arid slopes with southerly exposure

Vegetation:

Diantho-Pinetum sylvestris

Dominating tree species: *Pinus sylvestris*

Protected species (600 m^2): 7 – 16

Vegetation change with N deposition:

Decrease in, or loss of, protected species, such as *Carex caryophyllea, Carex supina, Potentilla incana, Dianthus arenarius, Koeleria glauca, Silene otites, Pulsatilla pratensis, Allium oleraceum, Briza media, Campanula sibirica, Helianthemum nummularium, Koeleria macrantha, Linum catharticum, Pimpinella nigra, Potentilla incana, Prunella grandiflora,* and *Trifolium montanum*. Growing dominance of the grass *Arrhenatherum elatius*.

Negative effects of N deposition on ecosystem integrity:

Destruction of rare habitats with a high number of protected species

Table A8.3

Class of endangerment	**Deposition-induced change in topsoil (humus layer and upper mineral soil down to 5 cm)**
Not endangered	C/N > 22 and ΔC/N < 1
Potentially endangered	C/N > 22 and ΔC/N > 1
Significantly endangered	C/N < 22 or ΔC/N > 2
Extremely endangered	C/N < 21 or ΔC/N > 4

Figure A8.8 Diantho-Pinetum sylvestris on an arid slope with southerly exposure near Schwedt / Oder (north-east Germany).

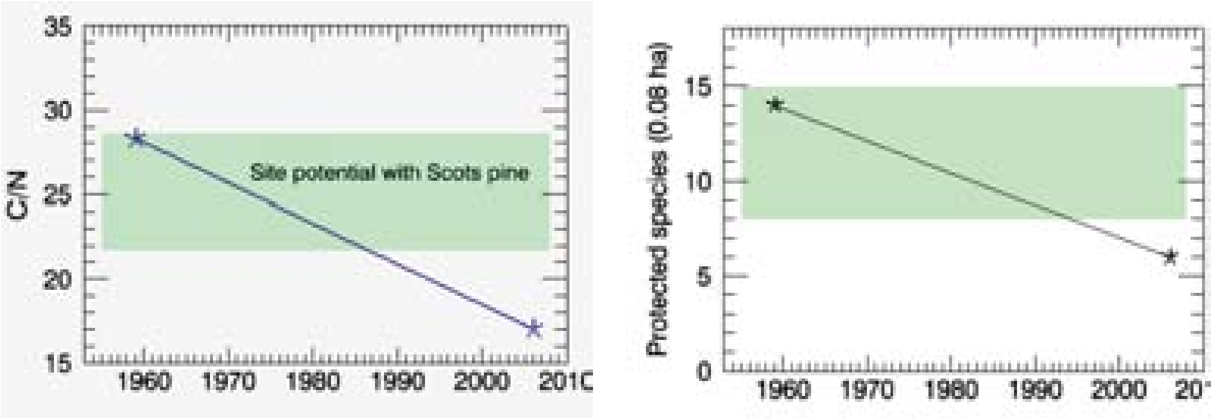

Figure A8.9 *Change in C/N ratio of topsoil (left, according to the indicator-value model) and decrease in protected plant species (right) in the Scots pine forest presented in Figure A8.8. The amplitude of the ecosystem type before 1965 is marked in green.*

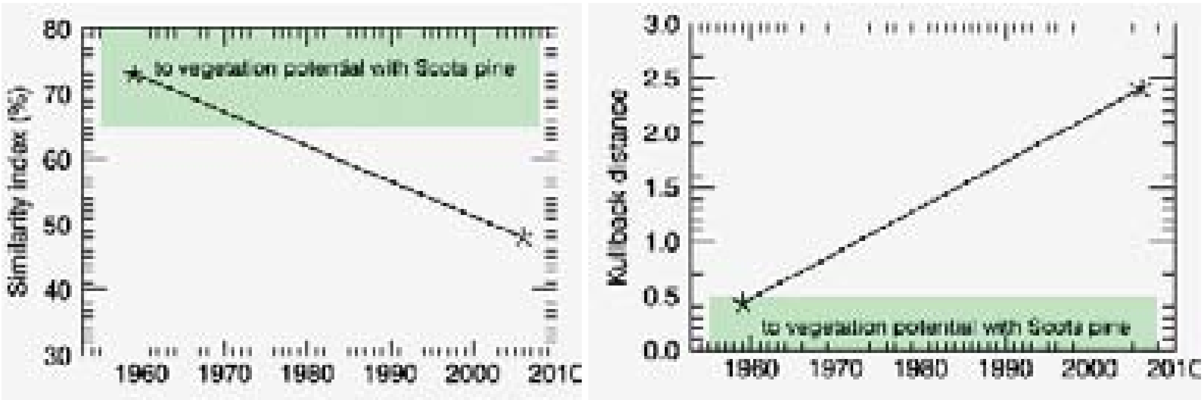

Figure A8.10 *Similarity (left) and Kullback distance (right), with respect to the vegetation potential of the site with Scots pine before 1965 (investigation plots presented in Figures A8.8 and A8.9). The amplitude before 1965 is marked in green.*

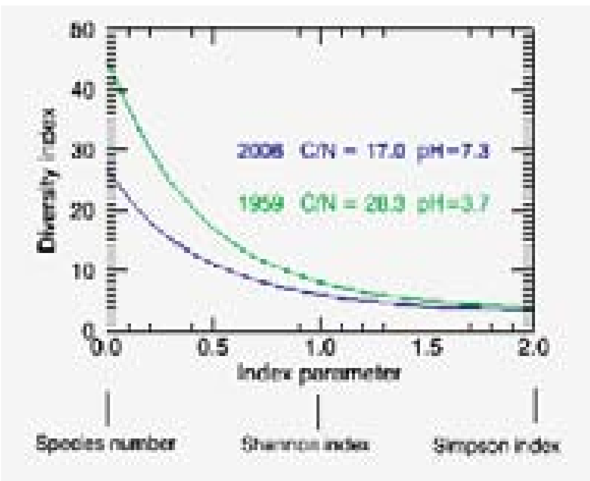

Figure A8.11 *Change of the diversity profile between 1967 and 2007 under the influence of heavy N deposition (investigation plot presented in Figures A8.8 to A8.10). The increasing dominance of Arrhenatherum elatius was coupled with a drastic decrease in species diversity. The specific species composition of this highly valuable habitat was completely destroyed.*

Oak forests on oligotrophic to mesotrophic sandy soils

Vegetation:
Deschampsio-Quercetum, Vaccinio-Quercetum

Dominating tree species: *Quercus robur, Quercus petraea*

Protected species (600 m²): 0 – 2

Vegetation change with N deposition:
Emerging N-indicator species, such as *Urtica dioica* and *Impatiens parviflora.* Decrease in, or loss of, low-nutrient demanding species, such as *Vaccinium* and *Calluna*.

Negative effects of N deposition on ecosystem integrity:
N deposition initially induces better growth in oak trees. However, long-standing and heavy N deposition leads to nutritional disharmonies in leaves, growth depression, and increased mortality.

Table A8.4

Class of endangerment	Deposition-induced change in topsoil (humus layer and mineral soil down to 5 cm)
Not endangered	C/N>20 and pH>3.1 and pH<4.0 and ΔC/N<0.8 and ΔpH<0.4
Potentially endangered	C/N>20 and pH>3.1 and pH<4.0 and (ΔC/N>0.8 or ΔpH>0.4)
Significantly endangered	C/N<20 or pH<3.1 or pH>4.0 or ΔC/N>1.5 or ΔpH>0.8
Extremely endangered	C/N<19 or pH<2.8 or pH>5.2 or ΔC/N>3.0 or ΔpH>1.2

Beech forests on oligotrophic and mesotrophic sandy soils

Vegetation:
Luzulo-Fagetum

Dominating tree species: *Fagus sylvatica*

Protected species (600 m²): 0 – 1

Vegetation change with N deposition:
Decreasing vitality of *Vaccinium myrtillus* (oligotrophic and arid sites), increase in *Deschampsia flexuosa* (oligotrophic sites), emergence and spreading of *Calamagrostis epigejos* and *Impatiens parviflora* (mesotrophic sites)

Negative effects of N deposition on ecosystem integrity:
N deposition initially induces better growth in beech trees. However, continuous and heavy N deposition leads to nutritional disharmonies in leaves, growth depressions, and increased mortality. Increased spreading of grasses reduces natural potential for regeneration.

Table A8.5

Class of endangerment	Deposition-induced change in topsoil (humus layer and mineral soil down to 5 cm)
Not endangered	C/N>17 and pH>3.1 and pH<4.5 and ΔC/N<0.8 and ΔpH<0.4
Potentially endangered	C/N>17 and pH>3.1 and pH<4.5 and (ΔC/N>0.8 or ΔpH>0.4)
Significantly endangered	C/N<17 or pH<3.1 or pH>4.5 or ΔC/N>1.5 or ΔpH>0.8
Extremely endangered	C/N<16 or pH<2.8 or pH>5.2 or ΔC/N>3.0 or ΔpH>1.2

Beech forests on eutrophic sandy soils

Vegetation:

Galio-Fagetum

Dominating tree species: *Fagus sylvatica*

Protected species (600 m²): 0 – 1

Vegetation change with N deposition:

Area-spreading of *Impatiens parviflora*, under heavy N deposition emergence, and spreading of *Padus serotina* and *Sambucus nigra*

Negative effects of N deposition on ecosystem integrity:

N deposition initially induces better growth in beech trees. However, continuous and heavy N deposition leads to nutritional disharmonies in leaves, growth depression and increased mortality. Increased spreading of large shrubs reduces the natural potential for regeneration.

Table A8.6

Class of endangerment	Deposition-induced change in topsoil (0-5 cm)
Not endangered	C/N>13.5 and pH>4.5 and pH<5.5 and ΔC/N<0.4 and ΔpH<0.3
Potentially endangered	C/N>13.5 and pH>4.5 and pH<5.5 and (ΔC/N>0.4 or ΔpH>0.3)
Significantly endangered	C/N<13.5 or pH<4.5 or pH>5.5 or ΔC/N>0.8 or ΔpH>0.6
Extremely endangered	C/N<16 or pH<2.8 or pH>5.2 or ΔC/N>1.5 or ΔpH>0.9

Figure A8.12 Heavy N deposition in beech forests on loamy soils leads to the development of large shrubs, replacing site-specific ground vegetation and interfering with the natural potential for regeneration.

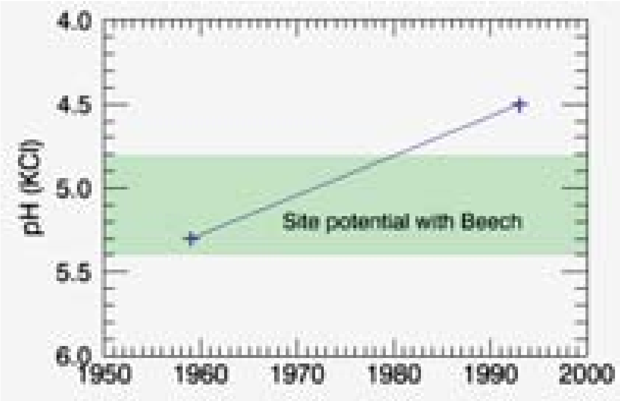

Figure A8.13. Change in pH value of topsoil (according to the indicator-value model) in a beech forest on loamy soil that was exposed to heavy N inputs from a neighbourhood agricultural crop (Uckermark, north-east Germany). The amplitude of the ecosystem type before 1965 is marked in green.

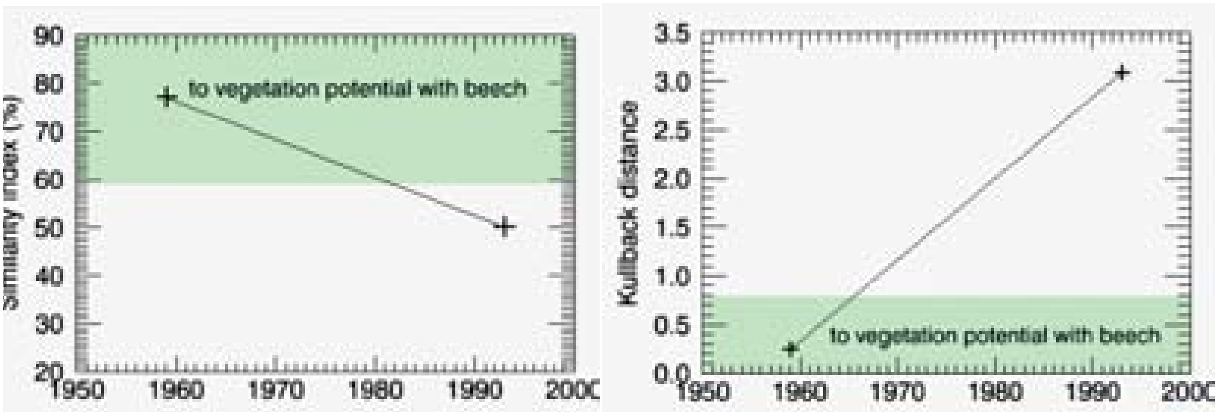

Figure A8.14 Similarity (left) and Kullback distance (right), with respect to the vegetation potential of the site with Scots pine before 1965 (investigation plot presented in Figure A8.13). The amplitude before 1965 is marked in green.

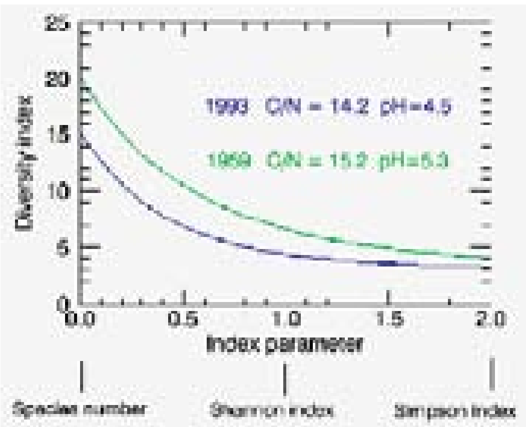

Figure A8.15 Decrease in plant-species diversity between 1959 and 1993 in a beech forest on loamy soil under the influence of heavy N deposition (investigation plot presented in Figures A8.13 and A8.14).

9

A single metric for defining biodiversity damage using Habitats Directive criteria

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Abstract
A quantitative metric of biodiversity value is required to define biodiversity damage and acceptable limits of change. We define a habitat quality index *HQ* to summarise effects on positive and negative indicator species as defined in biodiversity protection legislation. This allows appropriate interpretation of outputs from models that predict impacts of biogeochemical drivers on habitat suitability for multiple species. Since it uses the entire list of indicator species, the *HQ* index does not rely greatly on predictions of environmental suitability for rare species, which are less likely to be available than predictions for more common species. The sensitivity of the HQ index to pollutant and management scenarios is demonstrated.

9.1 Introduction

Atmospheric emission ceilings for acidifying and eutrophying pollutants are set using the concept of Critical Load (Hettelingh et al. 1995). This depends on the definition of a metric or indicator of ecosystem quality, and of a threshold along this metric beyond which the ecosystem is seen as damaged. Simple soil chemical metrics can be defined for acidification, such as pH or Ca/Al ratio, and effect of different pollutant loads on these metrics can be predicted using models such as VSD (Posch and Reinds 2009). Defining metrics and thresholds for N eutrophication is more difficult. Soil N measurements have their limitations (e.g., total soil N includes much non- or barely-reactive N, and soil mineral N can fluctuate rapidly), and effects on plants and plant competition may precede changes to soil N. Biodiversity protection is an increasingly important outcome for pollution control, and it is important that model outputs directly correspond to the needs of users (Pielke 2003). Thus ecosystem quality metrics and thresholds need to be defined in terms used within biodiversity protection legislation, that is, effects on species, species assemblages, and/or habitats.

Methods used for monitoring and assessing biodiversity provide a starting point for defining a habitat quality metric. Simple metrics of biodiversity can be derived from data on species presence and abundance, such as species richness (i.e. number of species) or indices of evenness. However, the number of typical species, in particular taxa, can be small in certain habitats (e.g. vascular plants in Atlantic heathlands). Hence, biodiversity metrics that weight species according to criteria such as typicality or rarity are more likely to reflect societal and legal definitions of biodiversity.

Two main elements of biodiversity protection legislation are relevant for the effects-based work under the LRTAP Convention: the Habitats Directive of the EU (EEC 1992), and the Convention on Biological Diversity (CBD 1992). Also relevant is the ‘red list’ of species of particular conservation concern that is maintained by the International Union for the Conservation of Nature (IUCN 2009). This list has been proposed as the basis for assessing progress towards the aims of the Convention on Biological Diversity (Butchart et al. 2005), and has also been suggested as a basis for defining habitat quality and damage in the Critical Loads approach (Van Dobben et al., this volume).

The Habitats Directive requires signatories to monitor the conservation status of certain habitats, listed in Annexe I to the directive, and certain species, listed in Annexe II. For habitats, conservation status is defined as the sum of the influences that may affect the long-term natural distribution, structure and functions of the habitat, as well as survival of typical species. The implementation of the Habitats Directive in the United Kingdom includes a standard procedure for site assessment, known as Common Standards Monitoring (CSM) guidance (JNCC 2006), which lists indicators of favourable condition for all Annexe I habitats. These include structural indicators, such as the cover proportion of dwarf shrubs and, for many habitats, lists of indicator species – those which are typical, and those which are untypical and invasive. In general, habitat structure is not easily defined without

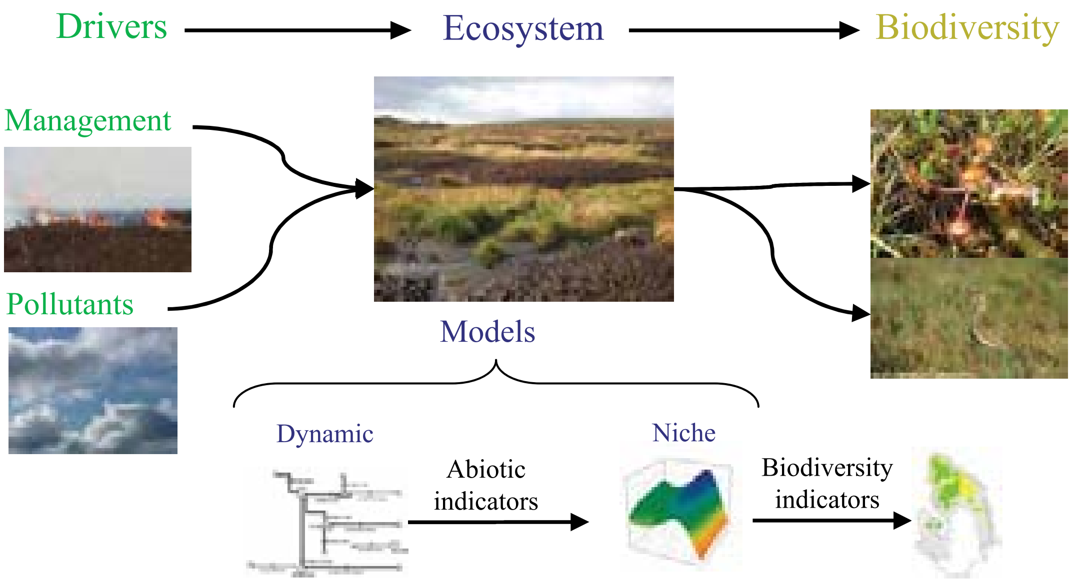

Figure 9.1 *Schema for predicting biodiversity change in response to environmental drivers.*

reference to species, and there is little consensus on which aspects of biogeochemical function are direct indicators of biodiversity value. Thus, the occurrence and abundance of particular species constitute the main operational definitions of conservation status.

Models are available that predict impacts of environmental drivers on many aspects of biodiversity change, including habitat structure (Terry et al. 2004), and occurrence of individual threatened species based on habitat preference or population dynamics. Ecosystem models vary greatly in their detail and complexity, and, hence, in the number of parameters required to set up the model for a particular site. For national or supra-national scale assessments, a pragmatic choice has to be made for models which can be run using the data likely to be available, which generally precludes the use of habitat-specific structural models or species-specific population models. The most widely applicable approach is to link models of element dynamics in soil and vegetation to models of the niche space for individual species, via a set of abiotic parameters that define this niche space (De Vries et al. 2010) (Figure 9.1).

The focus of this chapter is on interpretation of the results from such dynamic niche occupancy models. This is problematic for two reasons. Firstly, the models can generate habitat suitabilities for a large number of species, only some of which are relevant to a given habitat. Secondly, niche models are derived from empirical survey data, and thus are often poorly defined for precisely those rare species which are of most interest, since occurrence data are sparse for these species. For this publication, we explored an approach to interpreting habitat suitability predictions using pre-defined lists of positive and negative indicator species, and illustrated this approach using CSM indicator species lists defined for the United Kingdom. Positive indicator species, in this context, are those which are rare, typical for the habitat, or otherwise considered valuable by biodiversity specialists. Negative indicator species are those which are untypical for the habitat and invasive, and may include non-native species.

9.2 Methods

A blanket bog site at Moor House in northern England was chosen to illustrate the approach. Soil and soil solution data are available for this site, and changes to soil pH and C/N ratio have been simulated using the MAGIC model (Cosby et al. 2001, Smart et al. 2005). Deposition of N and S was projected to 2050, under the 2010 Current Legislation (CLE2010) scenario using the FRAME model (Dore et al. 2006). An extreme N deposition scenario was also simulated, to explore the sensitivity of the habitat quality score to N deposition, with an additional 50 kg N $ha^{-1}yr^{-1}$ that was assumed to be completely retained. Soil pH and C/N ratio dynamics were used as inputs to GBMOVE, a set of multiple logistic regression niche models for 1130 UK plant species (Smart et al., submitted). Soil water content and canopy height were assumed to be constant, at 1.5 g g^{-1} dry weight and 50 cm, respectively. The version of GBMOVE used did not include climatic variables, although a version incorporating climate drivers has been developed.

The training data for GBMOVE were species occurrences within large national datasets, hence the maximum probability of occurrence varies according to how common and widespread the species is. Model outputs were therefore normalised by dividing by the maximum probability of occurrence for each species. The outputs are best interpreted as current environmental suitability, since occurrence at a given site is also determined by presence in the local gene pool and by dispersal and extinction rates. A habitat quality score, *HQ*, was calculated as the mean suitability for positive indicator species minus the mean suitability for negative

Table 9.1 Positive and negative indicator plant species for blanket bog, as listed in UK Common Standards Monitoring guidance (JNCC 2006). Where genera were included, for example Erica spp., all species in the genus were included. Negative indicators were those mentioned in the Mandatory Attributes table for this habitat, that is, herbaceous species required to occur at < 1 % cover, and a selection of the woody species that are likely to invade this habitat.

1. Positive indicators				
Andromeda polifolia	*Erica cinerea*	*Racomitrium lanuginosum*	*Sphagnum imbricatum*	*Sphagnum squarrosum*
Arctostaphylos alpinus	*Erica tetralix*	*Rhynchospora alba*	*Sphagnum lindbergii*	*Sphagnum strictum*
Arctostaphylos uva-ursi	*Eriophorum angustifolium*	*Rubus chamaemorus*	*Sphagnum magellanicum*	*Sphagnum subnitens*
Betula nana	*Eriophorum vaginatum*	*Sphagnum auriculatum*	*Sphagnum molle*	*Sphagnum tenellum*
Calliergon cuspidatum	*Hylocomium splendens*	*Sphagnum balticum*	*Sphagnum palustre*	*Sphagnum teres*
Calluna vulgaris	*Hypnum cupressiforme*	*Sphagnum capillifolium*	*Sphagnum papillosum*	*Sphagnum warnstorfii*
Carex bigelowii	*Hypnum jutlandicum*	*Sphagnum compactum*	*Sphagnum platyphyllum*	*Trichophorum cespitosum*
Cornus suecica	*Menyanthes trifoliate*	*Sphagnum contortum*	*Sphagnum pulchrum*	*Vaccinium myrtillus*
Drosera intermedia	*Myrica gale*	*Sphagnum cuspidatum*	*Sphagnum quinquefarium*	*Vaccinium oxycoccus*
Drosera longifolia	*Narthecium ossifragum*	*Sphagnum fimbriatum*	*Sphagnum recurvum*	*Vaccinium vitis-idaea*
Drosera rotundifolia	*Plagiothecium undulatum*	*Sphagnum fuscum*	*Sphagnum riparium*	
Empetrum nigrum	*Pleurozium schreberi*	*Sphagnum girgensohnii*	*Sphagnum russowii*	
2. Negative indicators				
Agrostis capillaris	*Phragmites australis*	*Pinus sylvestris*	*Quercus petraea*	
Agrostis stolonifera	*Picea abies*	*Pteridium aquilinum*	*Ranunculus repens*	
Holcus lanatus	*Picea sitchensis*	*Quercus robur*	*Sorbus aucuparia*	

Table 9.2 Inputs for the GBMOVE plant species niche models, under different land use scenarios.

Scenario	Soil carbon %	Soil nitrogen %	Soil pH	Soil water %	Canopy height m
Business-as-usual	50.7[1]	FRAME→VSD outputs	FRAME→VSD outputs	87[2]	0.3[5]
Re-wetting	50.7[1]	FRAME→VSD outputs	FRAME→VSD outputs	95[3]	0.3[5]
Re-wilding	50.7[1]	FRAME→VSD outputs	FRAME→VSD outputs	95[3]	0.6[6]
Food security	50.7[1]	(FRAME + 30 kg N/ha/y) →VSD outputs	Limed to pH 5.5	74[4]	0.2[7]
Grouse shooting	50.7[1]	FRAME→VSD outputs	FRAME→VSD outputs	87[2]	0.2[7]

[1]mean % C in peat cores (0-15 cm) from Countryside Survey, n = 107. [2]mean; [3]maximum; [4]minimum % water in peat cores from Countryside Survey, n = 107. Canopy height estimates: [5]current; [6]extensified; [7]intensified management.

indicator species (Equation 1), where P_i and N_i are the probabilities of occurrence of the p positive indicators and n negative indicators for blanket bog habitat, respectively (Table 9.1), and $P_{max,i}$ and $N_{max,i}$ are the maximum probabilities of occurrence for the species. Species not included in the indicator lists were not used in this calculation.

$$HQ = \frac{1}{p}\sum_{i=1}^{p}\frac{P_i}{P_{max,i}} - \frac{1}{n}\sum_{i=1}^{p}\frac{N_i}{N_{max,i}} \quad (1)$$

Upscaling of the habitat quality method was illustrated using predictions of changes in soil N and pH at a 1 km² scale generated for the 2008 UK data submission to the CCE (Hall et al. 2008). The quality of blanket bog habitat across three Special Areas for Conservation (SACs; designated under the Habitats Directive as sites of particular importance for conservation) was assessed under five different land use scenarios (business-as-usual; re-wetting, i.e. blocking drains; re-wilding, i.e. blocking drains and removing grazing; food security, i.e. liming and fertilising to increase plant productivity; grouse shooting, and intensifying heather burning) were quantified as effects on the inputs for the GBMOVE models (Table 9.2). Climate inputs to GBMOVE were not varied. HQ was calculated only for 1 km² squares that had > 0.01 km² of bog habitats according to the UK Land Cover Map (Fuller et al. 2002).

9.3 Results

a) Habitat Quality at Moor House

Increased N pollution decreased environmental suitability for most blanket bog positive indicator species, and increased environmental suitability for some negative indicators, so there was a clear decline in the overall habitat quality score (Figure 9.2).

b) Upscaling

Changes in soil and vegetation under different land use scenarios had differential effects on different species (Figure 9.3). The habitat suitability for some positive blanket bog indicator species, such as *Drosera intermedia*, increased with the increase in pH and N availability in the Food Security

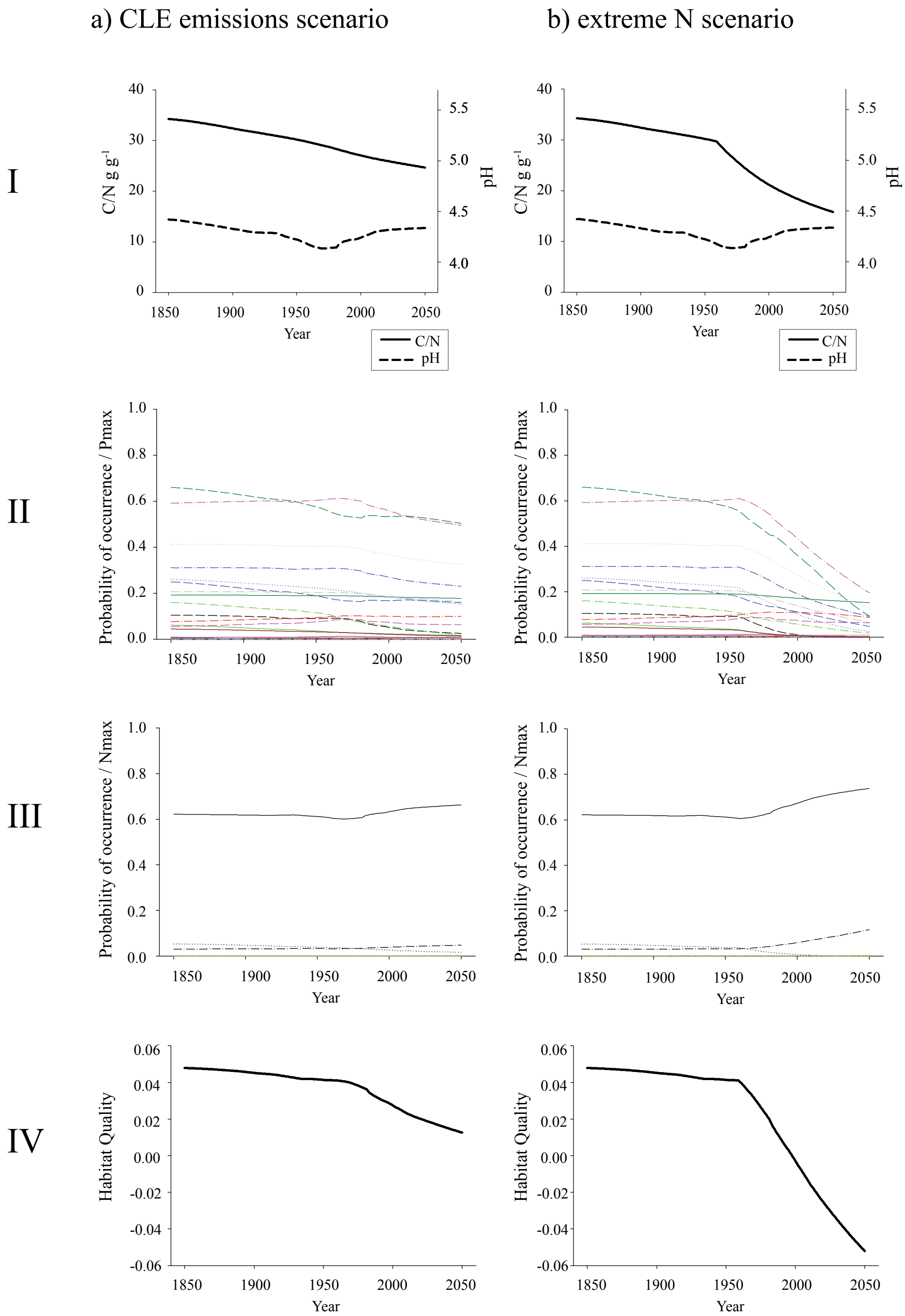

Figure 9.2 Simulated changes in blanket bog at Moor House long-term monitoring site, Cumbria, UK, under a) CLE emission scenario, and b) an extreme N addition scenario with an additional 50 kg N $ha^{-1} yr^{-1}$ from 1960. (I) Soil pH and C/N ratio simulated using the MAGIC soil chemistry model. (II) Probabilities of occurrence of positive indicator species for blanket bog, relative to maximum probability for the species. (III) Relative probabilities of occurrence of negative CSM indicator species for blanket bog. (IV) Overall habitat quality HQ (see equation 1).

scenario, but in general negative indicator species showed a greater increase in habitat suitability under this scenario.

The predicted distribution of HQ for blanket bog across the three SACs under five land use scenarios is illustrated in Figure 9.4. The HQ index appeared to be most sensitive to changes in water content, soil pH and N contents; changes in canopy height between the scenarios had relatively little effect. Thus, the largest decline in habitat quality was observed under the Food Security scenario, and there was little difference between Business-as-usual and Grouse Shooting, and between Re-wetting and Re-wilding scenarios. A full sensitivity analysis would be required to determine the key drivers of change in different habitats. Parameters used to represent differences among scenarios would also need to be

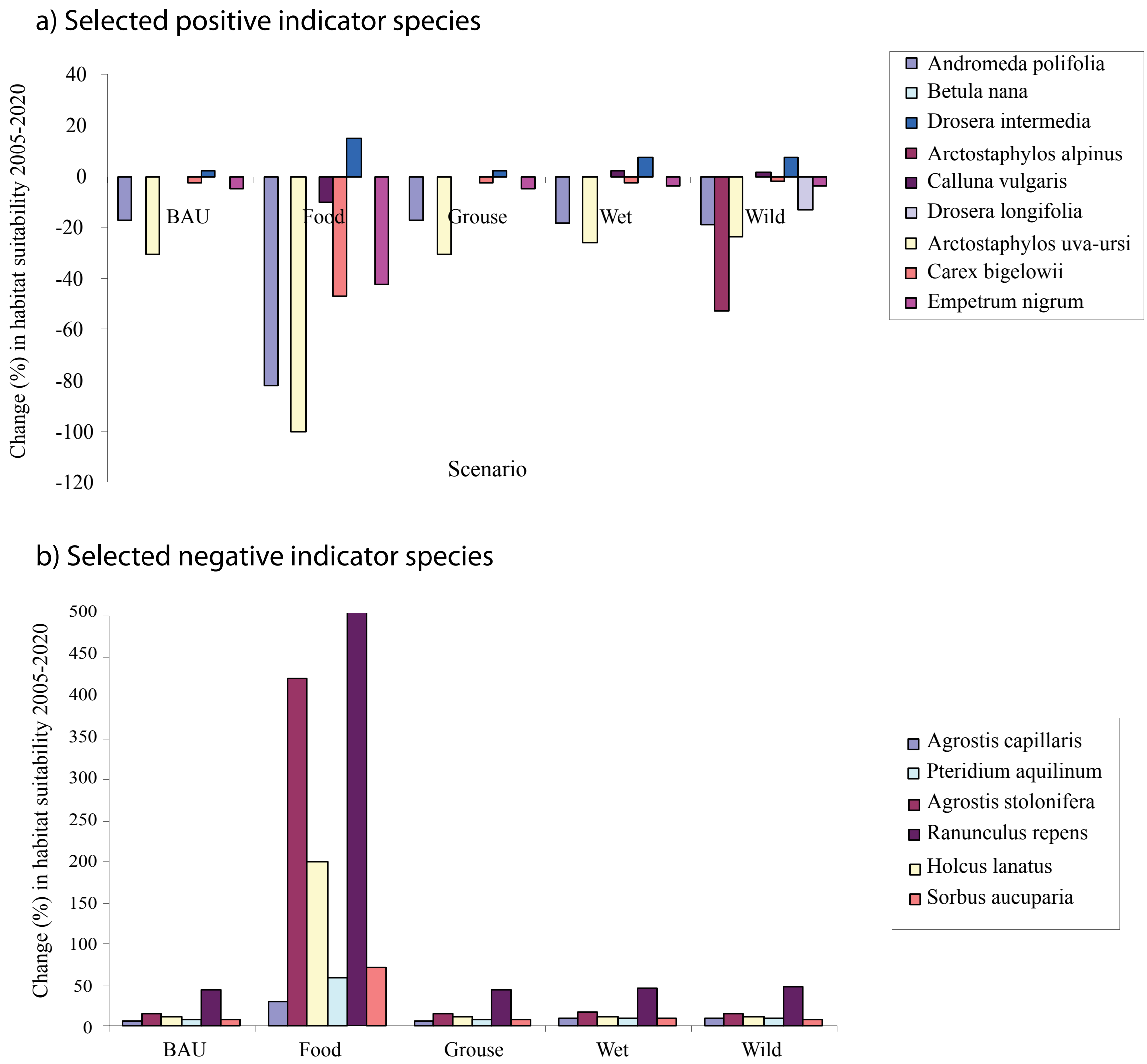

Figure 9.3 Changes in suitability for selected positive (a) and negative (b) indictor species as defined in the Common Standards Monitoring guidance for blanket bog, 2005–2020, under different land management scenarios: BAU = business-as-usual; Food = food security; Grouse = grouse economy; Wet = re-wetting; Wild = re-wilding. Mean changes across three peatland SACs: Migneint, Peak District, and Thorne and Hatfield Moors.

based on more empirical data to have full confidence in the results.

9.4 Discussion

The proposed habitat quality index provides a useful summary of effects on multiple plant species, using criteria defined by biodiversity specialists. A limited set of scenarios has been presented for illustration, but it would not be difficult to extend the approach to national scale datasets, such as those used in submissions to the CCE. The approach is also suitable for evaluating outputs from other model chains that predict environmental suitability from species occurrence, such as SMART-SUMO-MOVE and FORSAFE-VEG (De Vries et al. 2010).

Translating changes in environmental suitability for a set of species into an overall metric or index of biodiversity requires an evaluation of which species are desirable and which undesirable on a site. This necessarily involves subjective choices, although objective criteria, such as the rarity of species, can be used to support such choices. In particular, major differences in the conservation objectives for particular habitats make it difficult to compare the biodiversity value of one habitat with that of another. Once a target habitat has been defined it is usually possible to obtain lists of desirable and undesirable species, based on expert judgment. Using the entire set of indicator species reduces the danger of bias towards well-studied and charismatic species (Sitas et al. 2009).

Evaluations could also be derived from phyto-sociological lists, or from criteria such as rarity. Species lists based on rarity already exist, such as the Red List (IUCN 2009) or those in national implementations of the Convention on Biological Diversity, for example, Biodiversity Action Plans (UKBAP 2007). However, such lists contain only positive indicator species and so may not adequately reflect the risk to biodiversity posed by the invasion of habitats by untypical or abundant species. Also, defining niche models for rarer

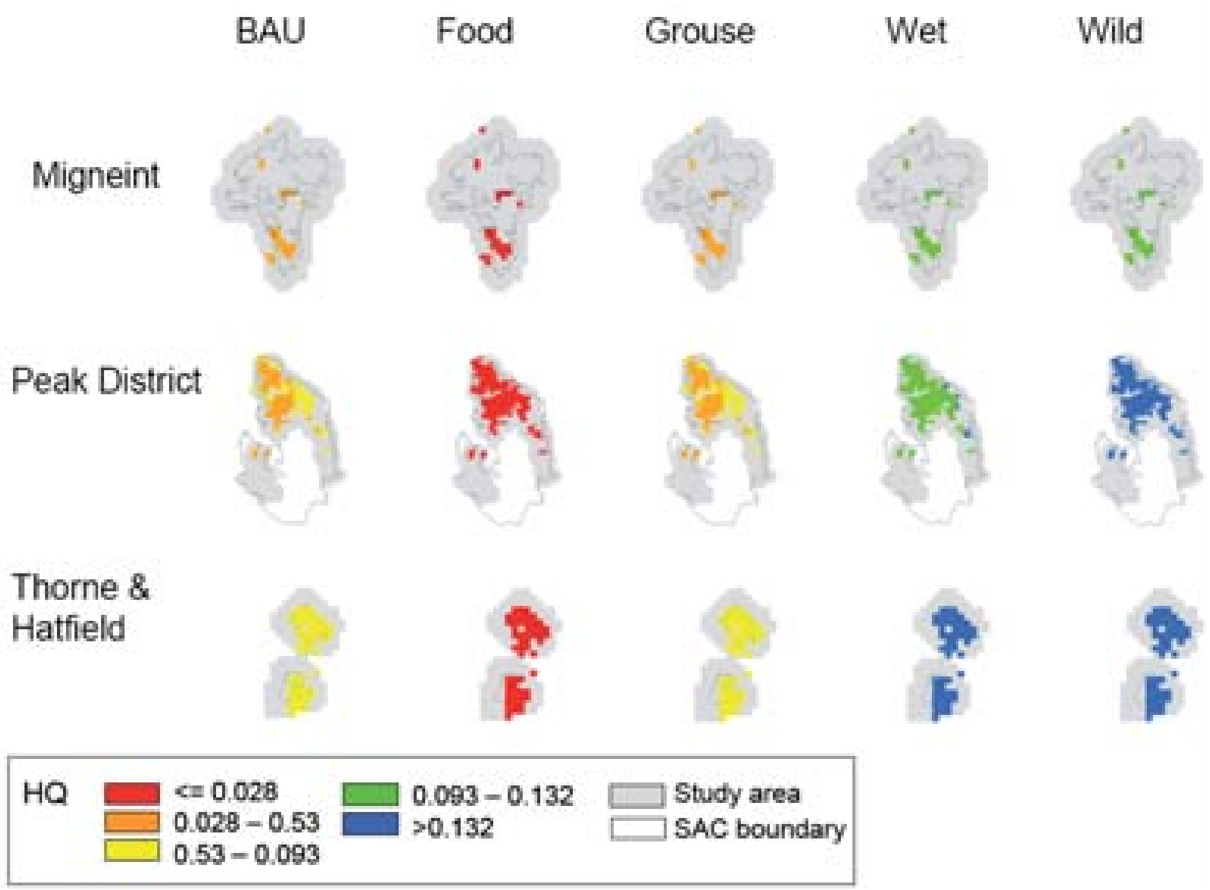

Figure 9.4 Distribution of a biodiversity metric based on suitability for blanket bog indicator species across three peatland SACs under different land-use scenarios. High scores indicate greater biodiversity value, that is, high environmental suitability for positive indicator species for blanket bog and/or low suitability for negative indicator species.

species would require targeted sampling, and may not be possible for the rarest species that occur on very few sites.

The proposed method for calculating a habitat quality index requires further testing before we can be fully confident that it reflects conservation value. In particular, the sensitivity of HQ to the numbers of positive and negative indicator species needs to be evaluated. The HQ index is calculated by subtracting one average proportion from another, and hence has a limited distribution close to zero; a transformation might be usefully applied.

The draft report from the CCE workshop in May 2009 (CCE 2009) requires visions on the state of a) the species present; b) the subset of species requiring protection; and c) acceptable limits for change. An implied assumption is that change to the biodiversity value of a set of species can be quantified, and the proposed HQ index provides a method to do precisely this. Further work would be required to identity an acceptable limit for change, but this limit could be based on existing empirical loads (Bobbink et al. 2003), or the forthcoming revisions to these. A major advantage of the approach is that species niche models, such as GBMOVE, are highly empirical, being based on very extensive survey datasets. Thus, the sensitivity of the HQ index to drivers, such as N and S deposition, is likely to closely reflect the true sensitivity of biodiversity value to these drivers.

9.5 Conclusions

- Changes in environmental suitability for individual species in response to pollutant and land use drivers can be predicted by linking dynamic models to niche models, as illustrated with the FRAME–VSD–GBMOVE model chain using data available at national scale.
- Translating species responses into biodiversity responses requires species evaluations in relation to conservation objectives.
- The proposed habitat quality index provides a quantitative summary of effects on multiple plant species.
- The approach could be extended to other taxa for which niche models are available or could be developed, such as birds and macroinvertebrates.

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Part 4 Current knowledge on empirical critical loads

Part 4 addresses new knowledge that confirms the need to review and revise empirical critical loads that have been established in 2002. Empirical critical loads are used to reduce the uncertainty of model assessments of areas at risk of nitrogen deposition, while improving the understanding of the role of biology. An overview of the need for revision is provided in Chapter 10 while findings focussing on central and northern Europe are described in Chapters 11 and 12 respectively. A contribution addressing nitrogen effects in southern Europe is available as powerpoint presentation for the 19th CCE workshop (www.pbl.nl/cce).

As a result the CCE has started an international project in the autumn of 2009 in collaboration with the Swiss Federal Office for the Environment (BAFU) and the German Umweltbundesamt (UBA). Together with these international institutions, the Dutch Ministry of Housing, Spatial Planning and the Environment (VROM) and the National Institute of Public Health and the Environment (RIVM), the CCE organises in the UN international year of biodiversity 2010, a workshop under the LRTAP Convention entitled "Workshop for the review and revision of empirical critical loads and dose response relationships" (Noordwijkerhout, 23-25 June 2010) to finalize a comprehensive report to the 29th session of the Working Group on Effects under the LRTAP Convention.

10

Empirical N critical loads for Europe: is an update and review needed?

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10.1 Introduction

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

The availability of nutrients is one of the most important abiotic factors which determine 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, Aerts and Chapin 2000). The series of events which occurs when N 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. As a consequence, high variations in sensitivity to atmospheric nitrogen deposition have been observed between different natural and semi-natural ecosystems. As a consequence, high variation in sensitivity to N deposition has been observed between different ecosystems. Despite this diverse sequence of events, the following main effects, the so-called 'mechanisms', can be recognised (Figure 10.1):

1. *Direct toxicity of nitrogen gases and aerosols to individual species* (e.g., Pearson and Stewart 1993). High concentrations in air have an adverse effect on the above-ground plant parts (physiology, growth) of individual plants. Such effects are only important at high air concentrations near large point sources;
2. *Accumulation of N compounds, resulting in higher N availabilities and changes in plant species interactions* (e.g., Bobbink et al. 1998). This ultimately leads to changes in species composition, plant diversity and N cycling. This effect chain can be highly influenced by other soil factors, such as P limitation;
3. *Soil-mediated effects of acidification* (e.g., Van Breemen et al. 1982, Ulrich 1983, 1991, De Vries et al. 2003). This long-term process, also caused by inputs of N compounds, leads to a lower soil pH, increased leaching of base cations, increased concentrations of potentially toxic metals (e.g. Al^{3+}), a decrease in nitrification and an accumulation of litter;
4. *Long-term negative effect of reduced–N forms (ammoniaand ammonium)* (e.g., Roelofs et al. 1996, Kleijn et al. 2008). Increased ammonium availability can be toxic to sensitive plant species, especially in habitats with nitrate as the dominant N form and originally hardly any ammonium. It causes very poor root and shoot development, especially in sensitive species from weakly buffered habitats (pH 4.5–6.5);
5. *Increased susceptibility to secondary stress and disturbance factors* (e.g., Flückiger et al. 2002). The resistance to plant pathogens and insect pests can be lowered because of lower vitality of the individuals as a consequence of N-deposition impacts, whereas increased N contents of plants can also result in increased herbivory. Furthermore, N-related changes in plant physiology, biomass allocation (root/shoot ratios) and mycorrhizal infection can also influence the susceptibility of plant species to drought or frost.

In conclusion, the most obvious effects of increased N deposition are significant changes in the nitrogen cycle, soil buffer capacity, vegetation composition, and in biodiversity. For more details, see Bobbink et al. (1998) and Bobbink et

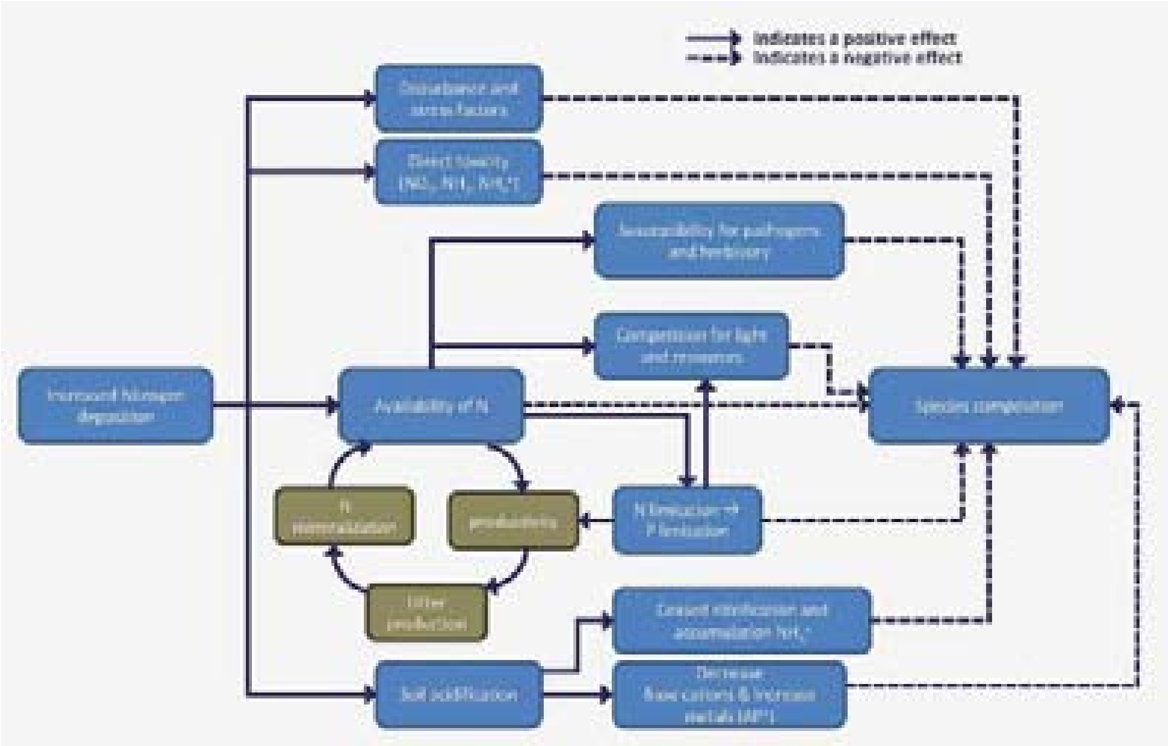

Figure 10.1 *Schematic of the main impacts of enhanced N deposition on ecosystems. Stress is considered to occur when external constraints limit the rate of dry matter production of the vegetation, whereas disturbance consists of mechanisms which affect plant biomass by causing its partial or total destruction.*

al. (2010). To control the negative effects of atmospheric N emissions, the critical load concept has been developed within the UNECE, since the mid-1980s.

10.2 Empirical N critical loads

Critical loads are defined as 'a quantitative estimate of an exposure to one or more pollutants, below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson and Grennfelt 1988). They are most commonly used in connection with deposition of atmospheric pollutants, particularly acidity and N, and define the maximum deposition flux that an ecosystem is able to sustain in the long term. Two approaches are currently used to define critical loads of N. The first, steady-state models, use observations or expert knowledge to determine chemical thresholds (e.g., N availability, N leaching, C/N ratio) in environmental media for effects in different ecosystems. Then, steady-state biogeochemical models are used to determine the deposition rate that results in this threshold value. In the second approach, empirical critical N loads are set based on field evidence.

Approach

Empirical critical N loads are fully based on observed changes in the structure and functioning of ecosystems, primarily in a) species abundance, composition and/or diversity (structure), or in b) N leaching, decomposition or mineralization rate (functioning). For an overview of indicators, see Løkke et al. (2000). The effects are evaluated for specific ecosystems. Statistically and biologically significant outcomes of field addition experiments and mesocosm studies have been used to quantify empirical critical loads. Only studies having independent N treatments with a duration of two years or more have been used. Especially data from long-term experiments in low-background areas are most useful to observe effects of N enrichment. However, since experimental studies have been conducted for a variety of reasons, their designs differ, and the methods used are carefully scrutinised to identify factors related to the experimental design or data analysis that may constrain their use. This includes evaluation of the accuracy of the estimated values of background N deposition at the experimental site. In addition, the results from correlative or retrospective field studies have been used, but only as additional evidence to support conclusions from experiments, or as a basis for expert judgement.

History

Empirical critical loads of nitrogen for natural and semi-natural ecosystems were firstly presented in a background document for the 1992 workshop on critical loads, held under the LRTAP 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 1992–1995 period was evaluated and summarised in an updated background paper and published as Annex III in the UNECE manual on

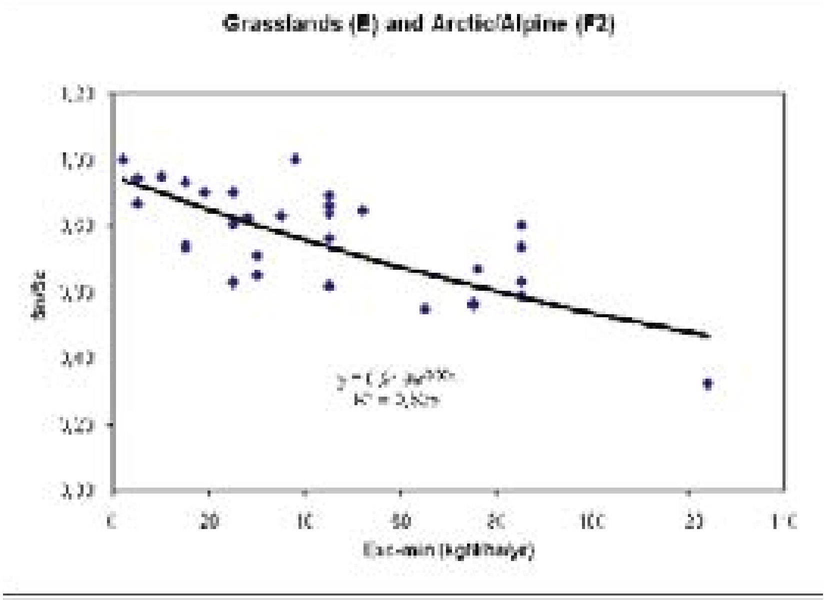

Figure 10.2 Relationship between the species richness ratio (SN/SC) and N exceedance (total load addition minus maximum value of critical load) on grassland (E) and artic/alpine habitats (F2).

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 UN/ECE 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 Organization's Regional Office for Europe (WHO 2000). Furthermore, the empirical critical loads for N deposition have been extensively reviewed and updated in 2001–2002 (Berne workshop; Achermann and Bobbink 2003). In that update, the classification of the receptor ecosystems has been brought in line with the EUNIS system (mostly level 3), and results from new N-impact studies in the 1996–2002 period were incorporated (Bobbink et al. 2003).

Critical loads of N can be compared to past, present or future deposition rates, in order to establish the amount of excess deposition, also called 'exceedance'. Exceedances of empirical critical loads have been used in European pollution abatement policy for defining emission reduction targets (Spranger et al. 2008). However, a key question in their use to support policy development (both in deriving national emission ceilings and for biodiversity protection through the UN Convention on Biological Diversity and the European Habitats Directive) is whether there is a link between the exceedance of critical N loads and effects on biodiversity, such as species richness. A recent synthesis of results from European N addition experiments showed a clear negative-log relationship between exceedance of empirical N critical loads and plant species richness, expressed as the ratio between the plant species richness in the N-added treatment and the control treatment (Bobbink 2008). Hence, although there are some limitations and scientific uncertainties in the empirical critical load approach, exceedance of the set European values is significantly linked to reduced plant species richness in a broad range of European ecosystems. This clearly demonstrates the usefulness of the approach as an indicator of the impacts of increased N loads on biodiversity (Figure 10.2).

10.3 Update and review of empirical N critical loads

More recently, it has been recognised (CCE workshops at Berne, 2008, and Stockholm, 2009) that considerable new insights into, and data on, the impacts of nitrogen deposition on natural and semi-natural vegetation in Europe have become available since the compilation of the last background document. Indeed, more than 6.5 years of new data have been published, especially new results became available in the following ecosystems: alpine and subarctic vegetation, coastal dunes, species-rich grasslands, ombrotrophic bogs, fens, Mediterranean communities, and boreal forests. Furthermore, more differentiation and quantification of empirical loads for European forest ecosystems is needed, because until now empirical N critical loads were only established for three main groups of forest systems. Finally, the guidance for the use of the specific modifying factors within the ranges of the empirical critical loads could be optimised, based on new results from several meta-studies and the new dynamic modelling approach. On the basis of this information, it was recently decided to start a new update (2009–2010), funded by the CCE, BAFU and UBA.

Aims of the 2009/10 update
The aims of the update and review are:

- to add new relevant information from studies (November 2002 to end of 2009) on the impacts of N on semi-natural and natural ecosystems in the existing empirical CLN database;
- to differentiate and quantify empirical N critical loads for more specific EUNIS forest ecosystem types;
- to write a new background document using the new data, based on the 2002 document;

Figure 10.3 *Schematic representation of the working procedure.*

- to evaluate the scientific data, including background N deposition, and to formulate an extended and updated table of the empirical N critical loads for Europe;
- to refine and provide more guidance on the use of the table with site-specific modifying factors (Table 10.2 in Bobbink et al. 2003);
- to synthesise the relationships between N exceedances and diversity at a European scale;
- to link, where appropriate, the empirical N critical loads according to the EUNIS classification with the Natura2000 Annex 1 habitats.

Working procedure

In this updating procedure, a similar '*empirical approach*' as for the earlier background documents will be used with the following phases: 1) data collection, 2) drafting of the different sections (per EUNIS class), 3) optimisation of the drafts after exchange between the contributing authors, 4) review of the second draft by external expert team, and 5) finalisation of the background document for the UNECE expert meeting (Figure 10.3). Following the expert meeting, the background document will be finalised after addition and incorporation of the comments of the participants of the meeting. Finally, a draft workshop summary report will be produced for formal use under the LRTAP Convention.

a) Data collection

European publications on the effects of nitrogen in natural and semi-natural ecosystems from the middle of 2002 to the present day will be collected, as completely as possible. Peer-reviewed publications, book chapters, papers published nationally, and 'grey' reports from institutes or organisations, can be used on request, if available. Relevant information on these studies will be put in an electronic database, including location, background deposition (if available), and EUNIS classification. If feasible, the Natura2000 habitat type will also be added.

b) Drafting of the section chapters

After all the data will have been collected, pre-drafts of the several sections (per EUNIS class) of the background document will be written, using the 2002 document as a starting point. If no new data are available, the 2002 text will be used. If some new data are found, the 2002 text will function as a base, but the text will be adapted. In a situation of many new results, the text will be completely rewritten. At the end of each section of a EUNIS class, a concluding table with critical loads will be given. The authors will pay much attention to the adding of Natura2000 habitats codes in the different subsections and final table, where possible.

c) Optimisation of the section drafts

All drafts of the different section chapters will be sent to the author team for discussion and review. The comments of the co-authors will be incorporated and corrected by the author of the specific section. The corrected drafts of the different sections will then be checked for consistency and integrated in the master document of the background document.

d) External review round

The second drafts of the section chapters will be send out to a team of international experts on the impacts of N in natural and semi-natural ecosystems. The team of reviewers will consist of experts from different parts of Europe (Nordic countries, western Europe, central/eastern Europe and southern Europe). Each chapter on a specific EUNIS class will be evaluated by at least two experts.

e) Finalisation of the background document

The comments from the review team on the second drafts will be incorporated in the text by the leading author, in tight cooperation with the author of the commented section. After a last check, the background document will be send to the participants of the UN/ECE CCE expert meeting on empirical N critical loads (June 2010, The Netherlands). The comments and additions of the participants will be used to finalise the definite table and background document for the formal revision procedure.

10.4 Concluding remarks

Empirical N critical loads for natural and semi-natural ecosystems have been recognised as an important indicator for the sensitivity of European ecosystems to the impacts of atmospheric N deposition (e.g., Bobbink et al. 2010). It became obvious that the exceedance of empirical N critical loads is significantly related to the reduction of plant species richness via a negative-exponential relationship. In addition, the last update and revision of the empirical N critical loads have been performed, thus, a new update and review is seriously needed, because many new evidence have been published since then. Therefore, the UNECE supports a new update and revision taking place from August 2009 to September 2010, with an expert meeting in June 2010.

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11

New results on critical loads of nitrogen in Central Europe

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11.1 Introduction

At the UNECE workshop on Empirical Critical Loads for Nitrogen, held in Berne, in 2002 (Achermann and Bobbink 2003), it was concluded that exposure-response curves with low levels of added or atmospheric nitrogen were needed. At the high N loads which are encountered in Central Europe this can be achieved either through long lasting N reduction, or through gradient studies with careful consideration of confounding factors, whereas in alpine regions, experiments with addition of low levels of N are meaningful. Therefore, one focus of this summary is on gradient studies in both non-forest and forest ecosystems. Another focus is on modifying factors for setting the nitrogen critical load. A discussion on these focal points is presented, based on data from N-addition experiments in areas with higher background deposition.

11.2 Non-forest ecosystems

Jones (2005) presented an illustrative example of how important a low background deposition is. He observed no response in the moss *Racomitrium lanuginosum* in an upland grassland to a N addition of 35 kg N $ha^{-1}yr^{-1}$, on top of an ambient N deposition of 20 kg N $ha^{-1}yr^{-1}$. However, N removal down to 10 or 0 kg N $ha^{-1}yr^{-1}$ raised the moss cover drastically. Overall, the moss cover decreased exponentially with increasing N loads, with strong responses already at 10 kg N ha^{-1} yr^{-1} (Figure 11.1). Stevens et al. (2004) described a relationship between N deposition and plant species richness for acid grasslands in Great Britain, with a decreased richness at around 20 kg N ha^{-1} yr^{-1}, compared to deposition values at around 6 to 10 kg N ha^{-1} yr^{-1}. The forbs were the most sensitive functional group (Stevens *et al.,* 2006). Although the relation-

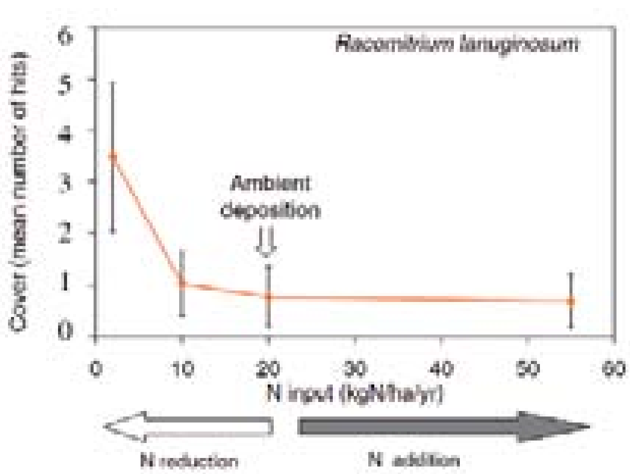

Figure 11.1 Response of the moss Racomitrium lanuginosum in an upland grassland to N addition and N removal (Jones 2005).

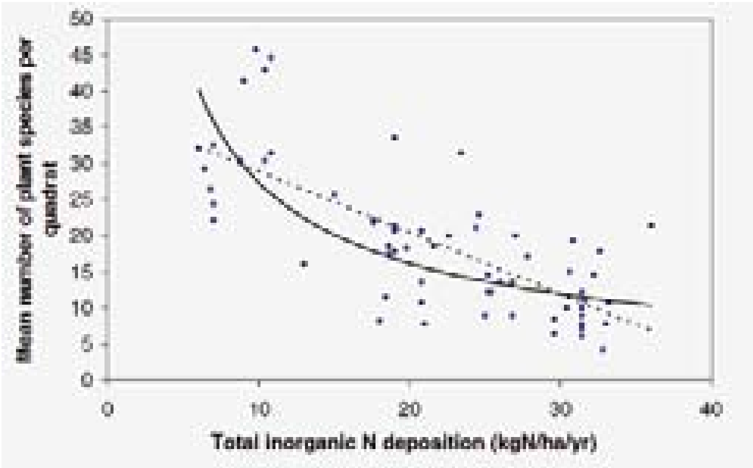

Figure 11.2 Mean number of plant species in an acid grassland, in relation to N deposition (Stevens et al. 2004 [dotted line], redrawn by Emmett 2007 [solid line]).

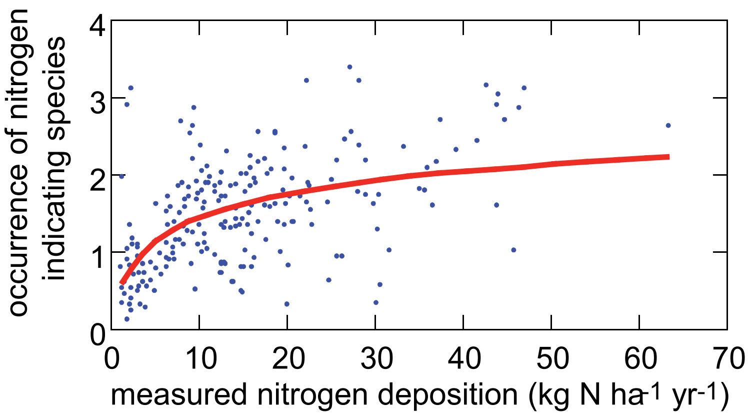

Figure 11.3 Relationship between the occurrence of nitrogen-indicating species and nitrogen deposition (redrawn from UNECE 2006).

ship was described as linear, Emmett (2007) suggested an exponential fit (Figure 11.2), similar to that in the study by Haddad et al. (2000) on plant species richness in a US grassland. These data show that the strongest response to N may occur at low levels, and that changes may already occur at 10 to 20 kg N ha^{-1}yr^{-1}. The study by Power et al. (2006) is not a gradient study, but describes recovery of *Calluna* from earlier N additions at rather low rates. A treatment of 15.4 kg N ha^{-1}yr^{-1} on top of a background deposition of 8 kg N ha^{-1} yr^{-1} produced significant effects on budburst of *Calluna,* even eight years after cessation of the treatment.

Two N-addition experiments in alpine grassland were carried out at low ambient deposition levels of 4 to 5 kg N ha^{-1}yr^{-1} (Bassin 2007, Hiltbrunner 2008). Both found significant increases in sedge species at N addition rates of 5 kg N ha^{-1}yr^{-1}. It cannot be determined whether these sensitive reactions reflected a higher sensitivity of alpine ecosystems, compared with lowland systems, or if they were the result of a low background deposition.

11.3 Forest ecosystems

Gradient studies

For forest ecosystems, gradient studies are available within the framework of ICP Forests. The occurrence of nitrogen-indicating species was correlated with measured nitrogen deposition, with the largest changes occurring between zero and 10 kg N ha^{-1}yr^{-1} (Figure 11.3, UNECE 2006). Nitrate leaching depended not only on N input, but also on C/N ratio of the forest floor (Gundersen et al. 1998). If data analysis was restricted to the plots with low C/N (<22), nitrate leaching would start at a N deposition rate of about 10 kg N ha^{-1}yr^{-1} (Figure 11.4, UNECE 2005).

Another gradient study was carried out from Swiss permanent forest observation plots. The cover of *Rubus fruticosus* was related to modelled N deposition (Flückiger and Braun 2004). Above a deposition rate of 20 to 25 kg N ha^{-1}yr^{-1}, the cover increased exponentially (Figure 11.5). This result was

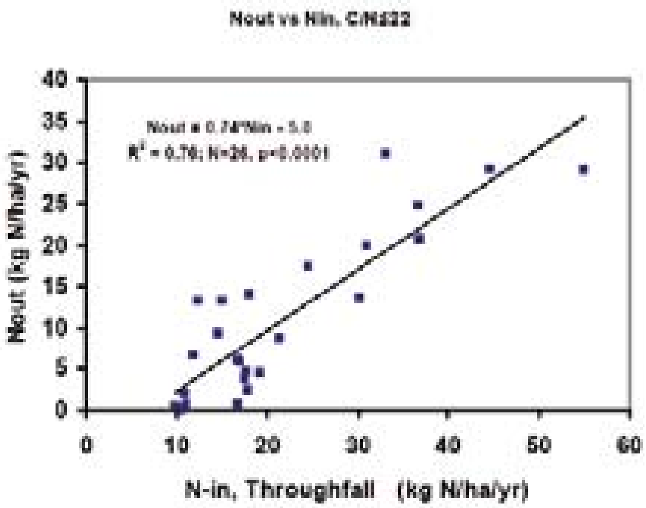

Figure 11.4 N leaching fluxes (kg N ha–1 y–1) against N input in throughfall, for sites with C:N < 22 (Source: UNECE 2005).

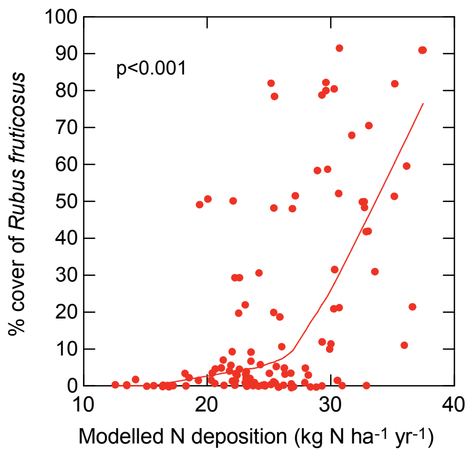

Figure 11.5 Cover of Rubus fruticosus in Swiss forest observation plots, in relation to modelled N deposition (Flückiger and Braun 2004).

Table 11.1 Site conditions of the N-addition experiments

Plot	Species	Altitude (m asl)	Geology	N deposition	Base saturation (%	C/N	Annual rainfall (mm)
Axalp	spruce	1700	limestone	13.8	100	18.5	1710
Hochwald	beech	670	limestone	15.3	100	21.1	1260
Lurengo	spruce	1600	gneiss	11.5	13	23.3	2020
Möhlin	beech	290	gravel	13.7	12	20.1	1060
Rötiboden	spruce	1580	granite	12.1	26	25.4	1850
Wengernalp	spruce	1880	limestone	9.2	23	21.3	1970
Zugerberg	beech	1000	till	19.6	12	18.5	2010

remarkable, as *Rubus fruticosus* does not have a high Ellenberg N indicator value.

How do site factors modify the critical load of nitrogen?

In the conclusions of the Berne workshop (Achermann and Bobbink 2003), it was recommended that a higher critical load for nitrogen may be used on soils with high base cation availability and on P-limited sites. This recommendation was tested using results from N-addition experiments and forest observation plots, both from Switzerland.

N-addition experiments (Swiss study)

N-addition experiments with tree saplings (beech and Norway spruce) were performed in Switzerland, in seven afforestation plots, on a variety of soil types and at different altitudes (Table 11.1, Flückiger and Braun 1999, Braun et al. 2009). In

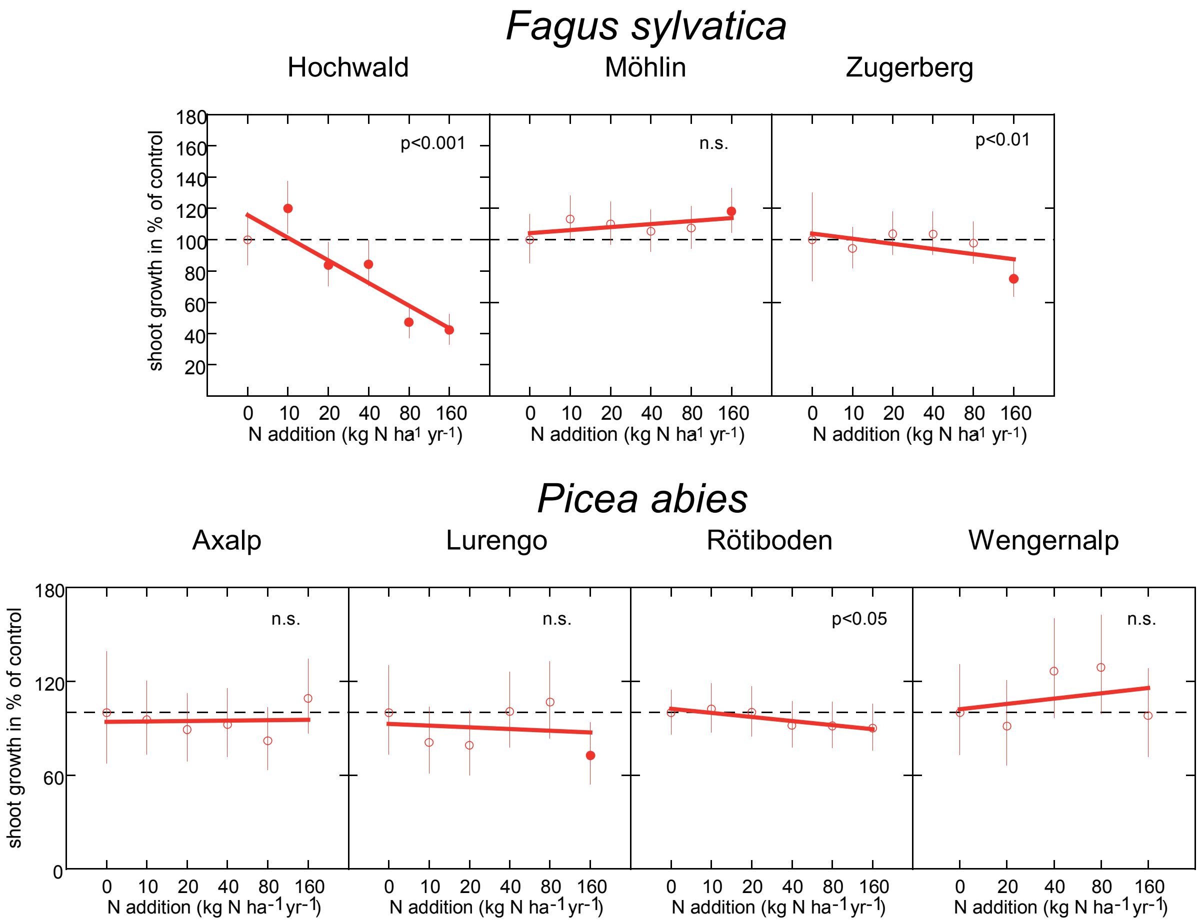

Figure 11.6 Average growth responses to N addition in 7 experimental plots in Switzerland, over 15 years of treatment (Braun et al. 2009). Bars: 95% confidence interval, filled points: significant differences to the control group.

three out of seven plots (two beech and one Norway spruce), growth decreased after the N addition; none showed a significant growth increase. The N addition caused significant decreases in foliar concentrations of phosphorus in all plots, of potassium in five plots and of magnesium in six plots. In one of the beech plots, growth reduction from N addition at the highest addition rate amounted to almost 40%. This plot was situated on lime, and the foliar concentrations of phosphorus and potassium were deficient, even in the controls. The N addition lowered these concentrations even further (Figures 11.7 and 11.8). At low foliar K concentrations, drought necroses increased drastically (Figure 11.9), suggesting that the K deficiency caused by the N addition impaired the physiological regulation of water loss in the plants. This was also expressed by a strongly decrease in efficient water use with increasing N addition rates (Figure 11.10).

The effects of N addition on parasite infestation have been described by Flückiger and Braun (1999). The most sensitive reactions were observed in the beech plots at Zugerberg and Hochwald, at N-addition rates of ≥ 10 kg N $ha^{-1}yr^{-1}$. Parasite infestation was also correlated with nutrient imbalances, such as the ratio between N and P, or between N and K.

The results suggest that nutrient imbalances are intensified by nitrogen input, and lead to an amplification of the N effect. Overall, more significant effects of N addition were found in beech than in Norway spruce, except for nutritional changes, which were similar in both tree species.

Field observations in Swiss permanent forest observation plots

In Swiss forest observation plots, the phosphorus concentration in the foliage decreased significantly between 1984 and 2007 (Figure 11.11; Braun et al. 2009). In 2007, values were found, on average, below the lower limit for normal nutrition for both beech and Norway spruce, suggesting a P limitation. This hypothesis was supported by a significant relation between P nutrition and stem increment (Figure 11.12). For beech, there was still a significant relationship between N deposition and stem increment, whereas in the case of Norway spruce this was only weak, with a trend of flattening off at N deposition rates of > 20 kg N $ha^{-1}yr^{-1}$. However, there was no relation between stem increment and foliar N concentrations, neither for beech nor Norway spruce.

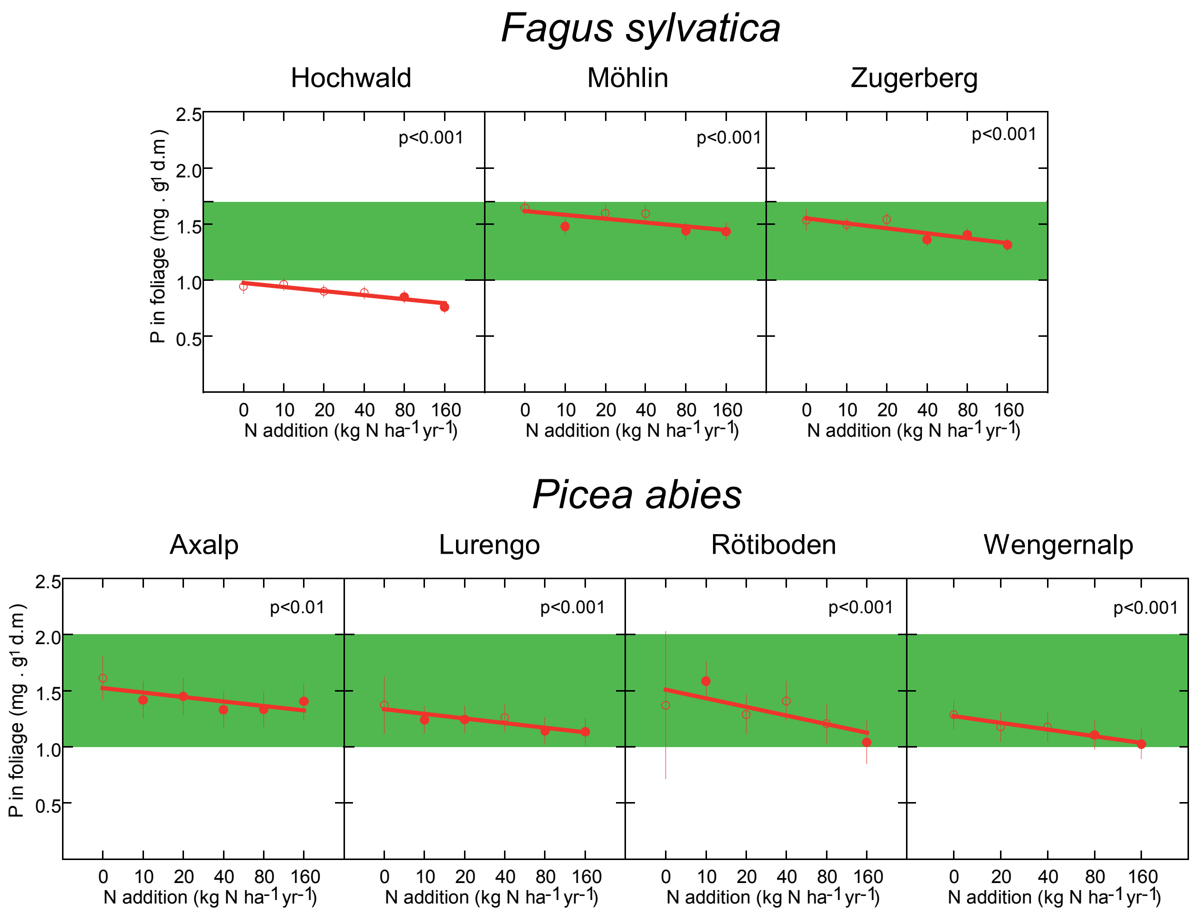

Figure 11.7 Foliar phosphorus concentration in the N-addition experiments (spruce: current needles). Average over 15 years of treatment. Bars: 95% confidence interval, filled points: significant differences to the control. Green fields: range for normal nutrition according to ICP Forests (Stefan et al. 1997).

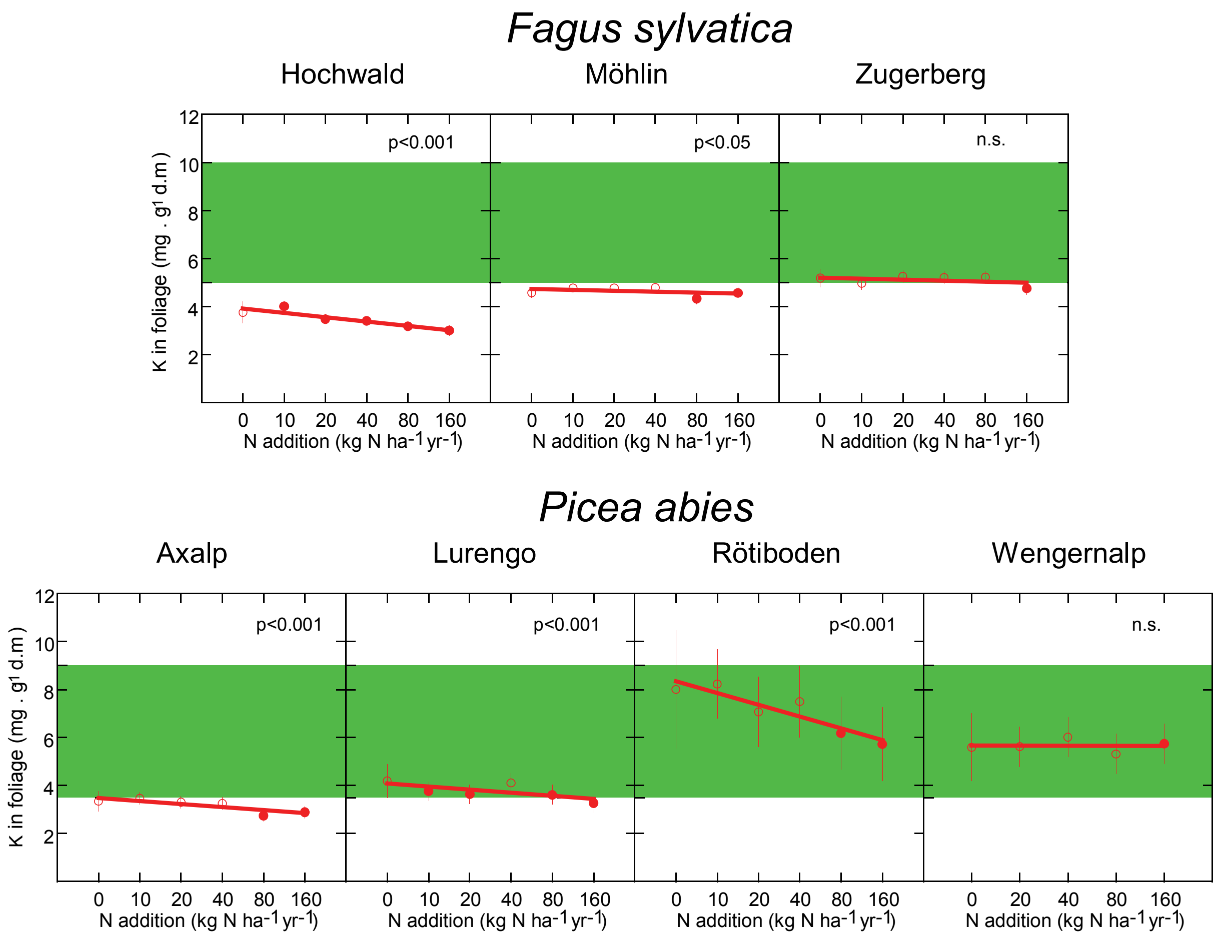

Figure 11.8 Foliar potassium concentration in the N-addition experiments. For further explanation, see Figure 11-7

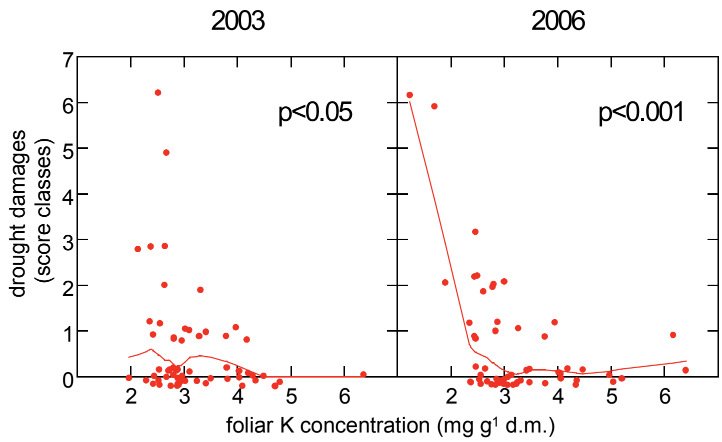

Figure 11.9 Drought necroses of beech in the N-addition experiment in the plot at Hochwald, for two years (2003 and 2006, after 12 and 15 years of treatment) in relation to foliar K concentrations.

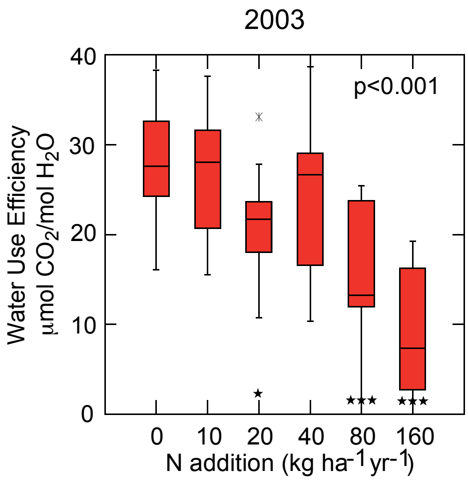

Figure 11.10 Water-use efficiency of beech in the N-addition experiment at Hochwald, after 12 years of treatment, calculated from $\delta_{13}C$ measurements in the leaves (analysis Rolf Siegwolf, PSI, Switzerland).

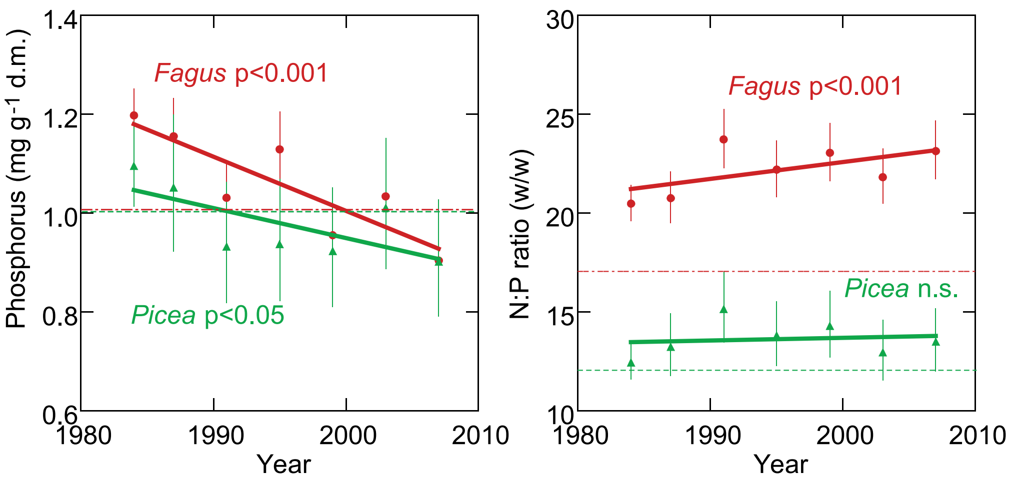

Figure 11.11 Development of the foliar phosphorus concentrations (left) and ratios between N and P (right) in forest observation plots of mature beech and Norway spruce. Dashed lines: lower limit of normal nutritional range. Values have been corrected for age trend. Bars = 95% confidence intervals of plot medians (Braun et al. 2009).

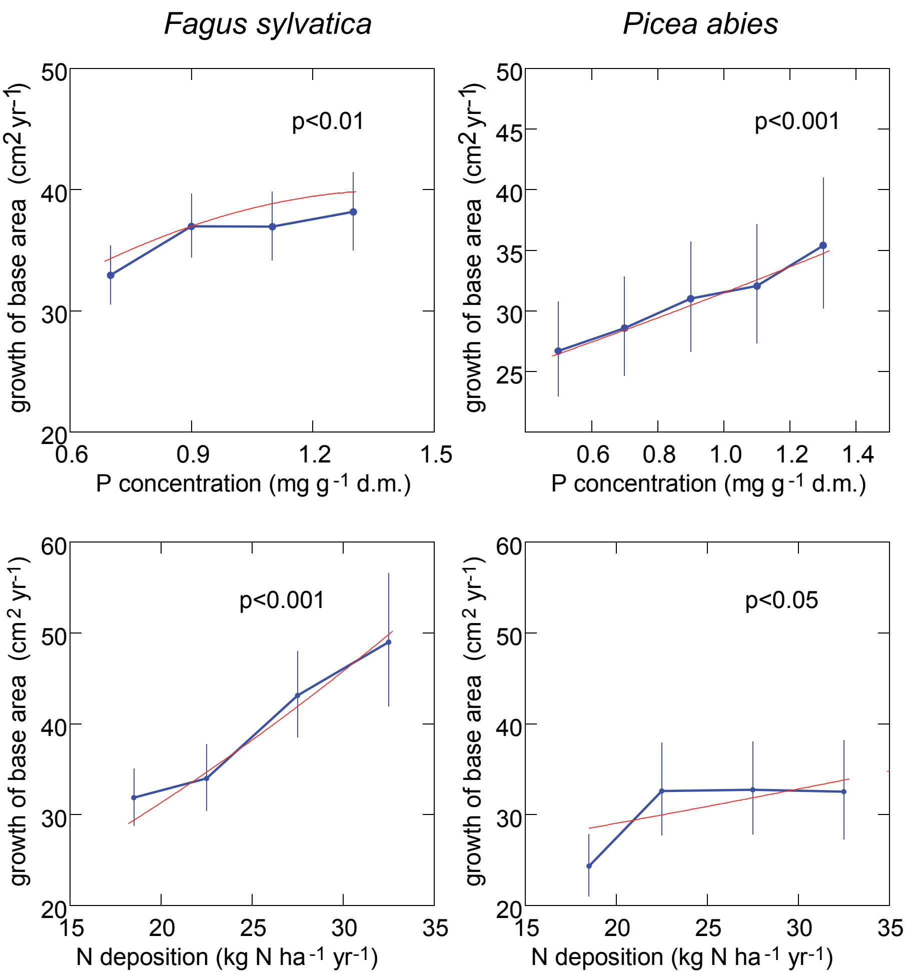

Figure 11.12 Relation between foliar P concentration (upper two graphs) or N deposition (lower two graphs) and stem increment of individual trees, for beech (left) and Norway spruce (right). The dataset comprised 2296 observations for beech and 1253 for Norway spruce, within 6 assessment periods. Bars = 95% confidence interval (mixed regression, multivariate with age and altitude), red line: fitted regression line (Braun et al. 2009).

11.4 Conclusions and recommendations

The exposure-response studies from non-forest ecosystems suggested that changes may occur at low N loads and that the response may level off at higher loads. This stresses the necessity to either perform experiments at low background deposition or to use gradient studies for setting the critical load. The vegetation studies suggested a critical load of ≤ 10 kg N $ha^{-1}yr^{-1}$ for lowland and upland grassland and of ≤5 kg N $ha^{-1}yr^{-1}$ for alpine grasslands.

The gradient studies from ICP Forests showed also changes occurring at an N deposition of around 10 kg N $ha^{-1}yr^{-1}$. According to Emmett (2007), leaching of nitrate is a rather late stage of N saturation, preceded by changes in ground vegetation. Thus, when leaching of nitrate from forest soils is observed at N loads of ≥10 kg N $ha^{-1}yr^{-1}$, the critical load of N for forests should not be higher than this.

In the conclusions from the Berne workshop, a higher empirical critical load for nitrogen was suggested for sites with high base cation availability and phosphorus limitation. The data from the Swiss experiments did not support this recommendation. Trees in the plot with the highest base saturation and the strongest P limitation even reacted most sensitively to N addition. Thus, it is suggested to combine base cation availability and P limitation into a single factor called 'nutrient deficiency' or 'nutrient imbalance'. Nutrient deficiency or imbalance will decrease the critical limit, while adequate or balanced nutrition will cause an increase (both ranges and ratios should be considered), although it should be noted that nitrogen leads to changes in nutrient limitations. The results from the Swiss permanent observation plots suggest that P limitation is an increasing reality in mature forests.

11.5 Acknowledgements

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12

New results on critical loads of nitrogen in Northern Europe

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In Northern Europe, boreal forests and wetlands, and subarctic and arctic tundras, are the dominant vegetation types. Plant growth in these ecosystems is restricted by short growing seasons, cold temperatures, and low nutrient supply. Current loads of atmospheric N deposition to Northern Europe are relatively low (generally < 6 kg N ha^{-1} yr^{-1} for boreal regions and < 3 kg N ha^{-1} yr^{-1} for arctic regions). This chapter presents recent empirical evidence that even these relatively low N deposition rates have the potential to change plant species composition, diversity and ecosystem functioning.

In both boreal and arctic ecosystems, bottom-layer vegetation is often dominated by bryophytes. Bryophytes efficiently retain N added by wet and dry deposition and are, therefore, considered to be highly sensitive to airborne N pollutants. Bryophyte responses to N addition are species specific. The feather moss *Hylocomium splendens* appears to be more sensitive to N enrichment than several other common boreal bryophytes (Salemaa et al. 2008), and moss decline has been observed at N input rates of c. 10 kg N ha^{-1} yr^{-1} (Nordin et al. 2009). The total cover of bryophytes may, however, not necessarily decline (Kochy and Brakenhielm 2008), as N favoured species, such as *Brachythecium* spp. and *Plagiothecium* spp., instead increase in abundance (Strengbom and Nordin 2008).

Nitrogen induced bryophyte decline has been demonstrated also for tundra ecosystems. In many cases, the responsible mechanism has been an increased cover of vascular plants, shading the bottom-layer vegetation (Cornelissen et al. 2001, Nilsson et al. 2002, Soudzilovskaia and Onipchenko 2005). For polar deserts with large areas of bare ground, it has been demonstrated that N addition in combination with P addition, strongly increases vascular plant cover, while sole N addition causes less pronounced effects (Madan et al. 2007).

For northern wetlands, N input ≥ c. 8 kg N ha^{-1} yr^{-1} may cause increased cover by vascular plants (mainly *Cyperaceae* and *Ericaceae* species) and decreased *Sphagnum* spp. cover (Gunnarsson et al. 2008, Wiedermann et al. 2009). This shift can have several effects on ecosystem function. *Cyperaceae* and *Ericaceae* species usually have higher growth rates and nutrient demands and decompose more easily than *Sphagnum* spp. Increased abundance of vascular species may also cause the groundwater table of bogs to lower. Taken together, these N induced alterations of plant species composition and chemistry are likely to reduce the ability of bogs to sequester carbon at elevated N inputs (Gunnarsson et al. 2008).

For vascular vegetation in Northern European ecosystems, recent reports demonstrate considerable vegetation changes at N input rates exceeding 5 to 10 kg N ha^{-1} yr^{-1}. In boreal spruce forest, damage to the dominant understory dwarf shrub *Vaccinium myrtillus* from pathogens has been found to increase in response to experimental N inputs of 12 kg N ha^{-1} yr^{-1} (Strengbom et al. 2002, Nordin et al. 2006). In a natural gradient of N deposition from north to south Sweden, pathogen damage to the *V. myrtillus* became more frequent as N deposition exceeded c. 6 kg N ha^{-1} yr^{-1} (Strengbom et al. 2003). Pathogen damage to *V. myrtillus* occurs in well-defined patches of the shrub canopy. In such patches, the shrubs become leafless early in the growing season, and more fast-growing plant species (mainly the graminoid *Deschampsia flexuosa*) proliferate from the increased N supply in combination with increased availability of light. Moreover, understory vegetation in forested ecosystems is largely influenced by successional changes in tree canopy light transmission (Gilliam 2006), and to separate N deposition effects on the vegetation from successional effects may sometimes be difficult (Kochy and Brakenhielm 2008). Hence, N induced vegetation changes in forest understory may sometimes lessen or diminish as the tree canopy closes, and it remains to be studied whether they reappear if the tree canopy is disrupted and the light availability suddenly increases.

For alpine and arctic grasslands, N inputs of c. 10 kg N ha^{-1} yr^{-1} cause significant plant biomass increase. While sedges may benefit more from N addition than grasses and forbs, species unresponsive to N still do not decline, but maintain their productivity (Bowman et al. 2006, Bassin et al. 2007). In plant communities with no overstory canopy, the unlimited supply of light may allow N favoured species to increase their productivity, without a concomitant decrease of species not favoured by N.

Although many effects of N deposition to ecosystems can be related to the quantity of N deposited, the chemical form of the deposited N may influence the ecosystem response to N deposition. In soils of cold climate ecosystems, slow N mineralization rates result in the dissolved N pool directly available for plant uptake being dominated by organic N forms (such as amino acids) and/or NH_4^+, while NO_3^- hardly occurs (Kielland 1995, Nordin et al. 2001, Jones and Kielland 2002, McFarland et al. 2002, Nordin et al. 2004). Airborne N deposited over these ecosystems consists of more or less equal portions of NH_4^+ and NO_3^-, and, in coastal areas, NO_3^- can even be the dominant N form. Many slow-growing plant species adapted to cold climate ecosystems have only limited capacity to utilise NO_3^- (Chapin et al. 1993, Nordin et al. 2001). In contrast, plant species adapted to N-rich habitats, often exhibit high capacities to take up NO_3^-, but only limited capacity to take up organic N (Bowman and Steltzer 1998, Nordin et al. 2001, Nordin et al. 2006). Only few studies have made an effort to separate effects of adding different N forms. For example, it has been demonstrated that N induced graminoid expansion in boreal forest is larger if NO_3^- is added than if NH_4^+ is added (Nordin et al. 2006).

12.1 Conclusions

In Northern European ecosystems where the N deposition historically has been low, even relatively small (5 to 10 kg N ha^{-1} yr^{-1}) long-term (>5years) increases in N deposition can result in unwanted changes in plant diversity, as well as in ecosystem function (see also Bobbink et al. 2010). For forested ecosystems, more knowledge is needed particularly to separate N deposition effects from natural forest successional changes, and to differ between NH_4^+ and NO_3^- effects. More knowledge is also needed about long-term effects on plant diversity and ecosystem function from very low N additions (at least < 8 kg N ha^{-1} yr^{-1}).

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Progress in the modelling of critical thresholds, impacts to plant species diversity and ecosystem services in Europe: CCE Status Report 2009

The CCE Status Report 2009 demonstrates that air pollution and the change of climate and biodiversity are interrelated. The report focuses on modelling methodologies and empirical knowledge addressing effects of nitrogen deposition. It shows that the risk of eutrophication is widespread in terms of critical load exceedance, expected time delays of recovery as well as, tentatively, changes of species richness. It confirms that climate change can reinforce recovery of acidified forest soils in Europe. It asserts that the reduction of nitrogen deposition since 1980 had a positive effect on biodiversity, and on the quality of soil and water, but a negative effect on global warming. It illustrates that it is possible to simulate the relationship between nitrogen deposition and the ensuing composition of the plant community while keeping track of the change of climate and land use. The report underpins the need for a review and revision of European nitrogen empirical critical loads and dose-response relationships in the UN year of biodiversity 2010. Finally, it proposes a number of indicators and modelling methodologies that have the capability of providing effect based support to policies that are aimed at mitigating air pollution and the change of biodiversity and climate in an integrated manner.

The Coordination Centre for Effects (CCE) is responsible for the development of methodologies and databases of the ICP-Modelling and Mapping in support of the effect oriented work under the LRTAP Convention