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Approaches to modelling local nitrogen deposition and concentrations in the context of Natura 2000: Working group report

Pierre P. Cellier, M. M. Theobald, A. A. Bleeker, D. D. Bruce, L. M. L. M. Frohn, O. O. Hertel, R. R. Hill, E. E. Noordijk, M. M. Sharp, A. A. van Pul, et al.

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Nitrogen Deposition and Natura 2000

Science and Practice in Determining Environmental Impacts

Findings of a European workshop linking scientists, environmental managers and policy makers

cost

Nitrogen Deposition and Natura 2000: Science and Practice in Determining Environmental Impacts

Editors:

W.K. Hicks, C.P. Whitfield, W.J. Bealey and M.A. Sutton

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PREFACE

The evidence from across Europe, reviewed in this book, demonstrates that nitrogen deposition is a major threat to European biodiversity, especially in the Natura 2000 network, including sensitive habitats and species listed under the European Commission Habitats Directive (92/43/EEC). It documents the information presented and discussed at an international workshop on 'Natura 2000 and Nitrogen Deposition', held in Brussels in May 2009, to review new evidence of nitrogen impacts, develop best practices when conducting assessments, and recommend options for consideration in future policy development.

The Habitats Directive (92/43/EEC) is a cornerstone of Europe's nature conservation policy. It promotes the maintenance of biodiversity and requires Member States to take measures to maintain or restore natural habitats at a 'favourable conservation status'. The Directive establishes the Natura 2000 network with the aim of assuring the long-term survival of Europe's most valuable and threatened species and habitats. The provisions of the Directive require strict site protection measures, avoidance of deterioration and the introduction of a precautionary approach to permitting 'plans or projects' which may have a likely significant effect on a site. However, the impacts of nitrogen deposition on the Natura 2000 network, together with the associated impacts due to elevated atmospheric concentrations of ammonia (NH_3) and nitrogen oxides (NO_x), are often not addressed adequately or systematically. At present there is no common European approach for determining the impacts of nitrogen deposition on individual sites or on conservation status.

These issues provided the key challenges for the workshop: to develop best practices in environmental assessment and decision making, and to inform the needs for future policy development. The workshop compared case studies from different European countries and reviewed the scale of the nitrogen threat to Natura 2000 sites (and other important conservation sites with high biodiversity) linking the science and decision making at local to European scales.

This book represents current thinking in Europe for the assessment and protection of Natura 2000 sites receiving elevated nitrogen deposition inputs. It describes existing methodologies and also considers emerging issues. It is hoped that this book will provide a valuable resource for practitioners in nature conservation and environmental regulation; scientific experts in nitrogen deposition, impact detection and environmental assessments; and policy advisers and stakeholders in the fields of biodiversity, the Habitats Directive and nitrogen emissions management.

We take this opportunity to acknowledge the COST 729 programme, which took up the challenge of nitrogen deposition and Natura 2000 for its mid-term workshop, and funded the attendence of many of the participants. Similarly, we are grateful for the active contributions from the Nitrogen in Europe (NinE) programme of the European Science Foundation (ESF), and the NitroEurope Integrated Project supported by the 6th Framework Programme of the European Commission. The organizational aspects of the workshop were co-financed through the UK Joint Nature Conservation Committee (JNCC), the Countryside Council for Wales (CCW), the Stockholm Environment Institute (SEI) at the

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We would also like to thank the following Advisory Group members, who were unable to attend the workshop, for their advice: Benjamin Gimeno (CIEMAT, Spain), Jean-Paul Hettelingh (Coordination Centre for Effects, The Netherlands), Matti Johannson (UNECE, CLRTAP, Geneva) and Mike Ashmore (University of York, UK).

Finally, we particularly thank Martin Hens of INBO for welcoming all the workshop participants to Brussels and for hosting the workshop.

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GLOSSARY

"Annex I" – Habitat types listed in Annex I of the Habitats Directive that are identified to be most in need of conservation at a European level.

Source: Managing Natura 2000 sites, EC, 2000. Section 4.3.1.

"Annex II" – Species listed in Annex II of the Habitats Directive that are identified to be most in need of conservation at a European level.

Source: Managing Natura 2000 sites, EC, 2000. Section 4.3.1.

Biogeographic Region - The European Union has nine biogeographical regions, each with its own characteristic blend of vegetation, climate and geology. Working at the biogeographical level makes it easier to conserve species and habitat types under similar natural conditions across a suite of countries, irrespective of political and administrative boundaries.

Source: http://bd.eionet.europa.eu/activities/Natura_2000/chapter1

Birds Directive – The EC Council Directive on the conservation of wild birds (79/409/EEC). The Directive relates to all naturally occurring birds in the wild within the European Community and addresses the protection, management and control of these species and lays down rules for their exploitation. The provisions apply to birds, their eggs, nests and habitats.

Source: http://ec.europa.eu/environment/nature/legislation/birdsdirective/index_en.htm

Conservation Status – The sum of the influences acting on a natural habitat and its typical species that may affect its long-term natural distribution, structure and functions as well as the long-term survival of its typical species.

Source: http://ec.europa.eu/environment/nature/natura2000/management/docs/art6/provision_of_art6_en.pdf

Critical Levels – Concentrations of pollutants in the atmosphere above which direct adverse effects on receptors, such as human beings, plants, ecosystems or materials, may occur according to present knowledge.

Source: http://www.unece.org/env/lrtap/WorkingGroups/wge/definitions.htm

Critical Loads – 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.

Source: http://www.unece.org/env/lrtap/WorkingGroups/wge/definitions.htm

De minimis – A Latin expression meaning about minimal things. In a more formal legal sense "*de minimis non curat lex*" means something that is unworthy of the law's attention. In risk assessment it refers to a level of risk that is too small to be concerned with.

Source: http://dictionary.reference.com

Habitats Directive – The EC Council Directive on the conservation of natural habitats and of wild fauna and flora (92/43/EEC). The aim of the Directive is to contribute towards ensuring biodiversity through the conservation of natural habitats and of wild fauna and flora within the European Community. It requires Member States to take measures to maintain or restore at favourable conservation status, natural habitats and species of Community importance. In undertaking these measures Member States are required to take account of economic, social and cultural requirements and regional and local characteristics.

Source: http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm

Natura 2000 – An EU-wide network of nature protection areas established under the Habitats Directive. The aim of the network is to assure the long-term survival of Europe's most valuable and threatened species and habitats. It is comprised of Special Areas of Conservation (SAC) designated under the Habitats Directive, and also incorporates Special Protection Areas (SPAs) classified under the 1979 Birds Directive. Natura 2000 is not a system of strict nature reserves where all human activities are excluded. Whereas the network will certainly include nature reserves most of the land is likely to continue to be privately owned and the emphasis will be on ensuring that future management is sustainable, both ecologically and economically.

Source http://ec.europa.eu/environment/nature/natura2000/index_en.htm

Nitrogen Deposition – The input of reactive nitrogen species from the atmosphere to the biosphere in the form of rainfall (wet deposition), aerosols or gases (dry deposition). The pollutants that contribute to nitrogen deposition derive mainly from nitrogen oxides (NO_X) and ammonia (NH_3) emissions.

Plan or project – The Habitats Directive does not provide a definition of a plan or project. The EC has issued guidance that indicates a 'project' is anything that involves construction works, installations, schemes or other interventions that affect the natural surroundings or landscape of an area.

Source: http://ec.europa.eu/environment/nature/natura2000/management/docs/art6/provision_of_art6_en.pdf

Special Areas of Conservation (SACs) – Sites designated under the EC Habitats Directive. SACs are areas which have been identified as best representing the range and variety within the European Union of habitats and (non-bird) species listed on Annexes I and II to the Directive. Sites which have been adopted by the European Commission, but not yet formally designated by governments of Member States are known as Sites of Community Importance (SCIs). SACs, together with SPAs, form the Natura 2000 network.

Source: http://www.natura.org

Special Protection Areas (SPAs) – Sites classified under the Birds Directive to help protect and manage areas which are important for rare and vulnerable birds because they use them for breeding, feeding, wintering or migration. Source: http://www.natura.org

1

SUMMARY FOR POLICY MAKERS

Key Messages

Scientific evidence:

There is convincing evidence of nitrogen deposition impacts on (semi-) natural habitats across Europe, but there is also a high spatial variability in the magnitude of nitrogen deposition and its impacts on the Natura 2000 network. The latest scientific evidence:

- provides a clear picture of the scale of the threat from nitrogen deposition. For example, 60% of Natura 2000 sites exceed the critical load for nitrogen.
- demonstrates the different effects caused by the various forms of reactive nitrogen compounds, for example, ammonia being more damaging to ecosystems than nitrogen oxides and dry deposition of ammonia being more damaging than wet deposition;
- supports and strengthens the use of critical loads and critical levels as effects thresholds at the European and site levels.

Implications for the Habitats Directive

- Existing legislation controlling emissions to the air of reactive nitrogen compounds does not adequately, or systematically address the impacts of nitrogen on the Natura 2000 network, or the wider objectives of the Habitats Directive.
- A common approach to assessing nitrogen deposition impacts on individual Natura 2000 sites, and on the conservation status of habitats and species, is urgently needed to provide reliable information on the stock at risk, evidence of recovery, and potential for restoration. There is a need to standardise approaches for nitrogen impact assessment, building on established methods such as critical loads, for the 2013 reporting round under Article 17 of the Habitats Directive and for assessments under Article 6.
- Further reductions in nitrogen emissions are required to reduce their adverse impacts on the Natura 2000 network, supporting the case for more ambitious commitments in revision of the Gothenburg Protocol and National Emissions Ceilings Directive, including low emission techniques (e.g. Gothenburg Protocol Annex IX).
- Policy options that require further consideration include:
    - the strengthening of existing legislation and its implementation, including opportunities within the Common Agricultural Policy for reducing nitrogen impacts on the habitats and species listed under the Habitats Directive;
    - the development of spatial planning policies linking nitrogen and Natura 2000;
    - establishing a limit value for ammonia applicable over Natura 2000 sites;
    - incorporating the concept of ecosystem services into assessment techniques;
    - the establishment of a high-level target: “a long-term goal to ensure that 95% of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030”.

1.1 Introduction

Atmospheric nitrogen deposition represents a major threat to European biodiversity. Nitrogen emissions to the atmosphere have increased substantially over the 20th century, mainly in the form of ammonia from agriculture and nitrogen oxides from industry and transport. Following atmospheric dispersion and chemical transformation, these nitrogen forms are deposited across European landscapes, providing unplanned nitrogen inputs and adversely affecting many sensitive habitats.

The issue represents a serious challenge for the conservation of natural habitats and species under the Habitats Directive (92/43/EEC).

The Habitats Directive is a cornerstone of Europe's nature conservation policy. It promotes the maintenance of biodiversity and requires Member States to take measures to maintain or restore natural habitats at a favourable conservation status. The Directive established the Natura 2000 network with the aim to assure the long-term survival of Europe's most valuable and threatened species and habitats. These sites are afforded the highest degree of protection under European legislation: the provisions of the Directive require strict site protection measures and avoidance of deterioration and introduce a precautionary approach to permitting "plans or projects" which are likely to have significant effect on a site.

The Habitats Directive does not directly address nitrogen impacts and until now there has been no common European approach for determining the impacts of nitrogen deposition on individual sites or on conservation status. At the same time, the scale of pollution exposure suggests that there are widespread threats to the Natura 2000 network and, more widely, to conservation status due to the concentrations and deposition of reactive nitrogen species.

Noting these problems, this volume reports on a workshop organized to bring together scientists, environmental managers and policy makers to clarify the current understanding of the key issues. The workshop was held in Brussels in 2009 and addressed the different components of science, environmental management and future policy development needs. The overall workshop goal was to harmonize approaches for determining the impacts of atmospheric nitrogen deposition on Natura 2000 sites across Europe. The following conclusions and recommendations were discussed in plenary at the workshop.

1.2 General Conclusions

The workshop agreed that nitrogen deposition represents a major threat to European biodiversity, including sensitive habitats and species listed under the Habitats Directive. Many Annex I habitats are naturally adapted to low nitrogen supply, so that fertilization with nitrogen compounds from the atmosphere alters the natural ecological balance. This results in the loss of the most sensitive species, which are often a priority for protection, and their replacement by invasive species that prefer high rates of nitrogen supply. In addition, the evidence also points to a net loss in the overall number of species.

The workshop noted that both atmospheric nitrogen deposition and air concentrations of reactive nitrogen compounds were appropriate indicators of the scale of threat. The use of critical loads and critical levels, as effects thresholds for nitrogen deposition and air concentrations, respectively, have demonstrated their usefulness at the European and local site scales.

The workshop agreed that in many cases across Europe, nitrogen deposition and concentrations substantially exceed the critical loads and levels. Examples were presented of predicted and actual habitat change, demonstrating that this is a major current threat, implying serious management

challenges to achieving favourable conservation status and to prevent deterioration of Natura 2000 sites.

The working groups addressed the different components of science, environmental management and future policy development needs across a series of five themes. The working group conclusions for each theme, that were agreed in plenary at the workshop, are presented below.

1.3 Comparison of impact assessment approaches in the context of Habitats Directive 6.3 (Theme 1)

The Habitats Directive requires that all 'plans and projects' which are likely to have a significant effect on a Natura 2000 site have an appropriate assessment of the implications for the site. Subject to certain exemptions, the plans or projects can only be approved where they are shown to have no adverse effect on any Natura 2000 site. However, at present, there is no common approach for evaluating the effects of nitrogen deposition and concentrations on these sites. The workshop therefore reviewed the practices in use across Member States. A key challenge was how to handle the situation where local background levels of deposition (or concentrations), resulting from existing activities, already lead to exposure in excess of critical thresholds. In this case, the question was raised of how to define an acceptable additional pollution burden when, in principle, any further exposure will give rise to an increasing risk and magnitude of adverse impact.

The impact assessment and decision making approaches applied in the different Member States, for 'plans and projects' under Article 6(3) of the Habitats Directive, were found to be clearly influenced by national policy, national aspirations, and national court decisions. However, an examination of the different approaches identified a number of common components which were used to develop a 'best practice framework' relevant across Europe.

It is recommended that a staged approach is applied to the impact assessment, including: i) a relevance screen, ii) test of likely significant effect, iii) appropriate assessment and iv) final decision. Modelling predictions should be compared against the relevant critical loads and critical levels (applied at the Natura 2000 site scale).

It is recommended that assessment needs to consider 'in combination' effects. Therefore, the plan/project should be considered both alone and in combination with other plans and projects, as well as in the context of existing ambient air quality (and prevailing environmental conditions). An integrated management/assessment plan (at, for example, the province/region scale) could assist with this.

It is recommended that all relevant EU Directives and national regulations should be considered during the assessment, to ensure the requirements of the IPPC Directive, Nitrates Directive, Water Framework Directive, Environmental Impact Assessment (EIA) Directive etc, are considered alongside those of the Habitats Directive, allowing an integrated approach to be applied.

It was concluded that ongoing problematic issues include whether consideration of the spatial scale of impact, survey data, and/or application of *de minimis* criteria, in respect to the plan or project contribution, are appropriate. A Member State might choose to apply a *de minimis* criterion to allow new plans or projects in situations where the critical load/level is already exceeded. In the absence of any sound ecological justification for such a position, this would have to be a policy decision.

It was concluded that further work is required on the development and dissemination of a best practice approach, including the involvement of a larger number of Member States.

1.4 Assessing nitrogen impacts on conservation status (Theme 2)

The Habitats Directive requires Member States to provide an assessment of conservation status of habitat and species listed in the Annexes of the Directive every six years. At the highest level, 'favourable conservation status' is defined and there is a standardized approach as regards the parameters to assess and descriptive statements of condition (e.g. favourable, unfavourable, unknown). Across Europe nitrogen deposition is increasingly recognised as a major issue for biodiversity. However, there is currently no standardisation of methods to consider nitrogen deposition impacts on conservation status. There is a high likelihood that the scale of nitrogen deposition effects on conservation status of habitats and species is not being accurately reported.

The workshop compared experience between countries as a basis for investigating what might be considered best practice in the assessment of conservation status. Different approaches to assessing whether nitrogen deposition is a 'pressure' on the 'structure and function' of habitats or a 'threat' to the 'future prospects' were considered. These include critical loads exceedance, field survey and bioindicators. Limitations to implementation were considered, including financial and expertise requirements.

It was concluded that nitrogen deposition represents a major threat to semi-natural vegetation across Europe. There is widespread exceedance of critical loads for nutrient nitrogen and acidification and substantial field and experimental evidence of the impacts. Such responses threaten the achievement of favourable conservation status for a large number of Annex I habitats.

It was concluded that the impact of nitrogen deposition on conservation status should be explicitly considered in Article 17 reporting, and the results should inform air pollution policy development.

It was concluded that there is a need for a common methodology for assessing the threat from nitrogen deposition to conservation status to be developed for application across Europe. This requires an improved dialogue between air pollution and biodiversity communities, building on recent progress in this area such as the development of a nitrogen deposition indicator under the Streamlining European Biodiversity Indicators (SEBI) programme.

It is recommended that a harmonisation of the methodology for nitrogen deposition assessment in conservation status reporting is required.

It is recommended that the lists of pressures and threats used for Article 17 reporting of conservation status should include nitrogen deposition explicitly and be more clearly defined.

It was noted that there is a requirement for greater clarity in the definition of 'favourable conservation status' for different habitats or groups of habitats, particularly with respect to defining important elements of structure and function. It is recommended that a series of habitat working groups should be established between interested Member States to take this forward.

It is recommended that the Working Group on Effects (WGE) of the UNECE Convention on Long -range Transboundary Air Pollution (CLRTAP) and the Expert Group on Reporting under the Nature Directives should be brought together in order to develop a methodology for the assessment of nitrogen deposition impacts on conservation status. A two-tiered approach is recommended as the basis of further development:

- Tier 1: An assessment based on empirical critical loads for nutrient nitrogen deposited to sensitive Annex I habitats. This would build on the already established critical loads exceedance methodologies developed under the CLRTAP, but requires further development

to apply the concept consistently to Annex I habitats of the Habitats Directive and to recommend the most appropriate deposition data. It would enable identification of nitrogen deposition as a "threat to future prospects" and could also be used to help interpret species or biogeochemical based monitoring data in order to identify whether nitrogen deposition is a 'pressure to current structure and function'.

- Tier 2: Monitoring (likely to be non-mandatory) should be made up of biotic and abiotic variables to determine where nitrogen deposition is a significant pressure on structure and function. This would require agreement of abiotic and biotic variables/values relating to favourable conservation status and the production of a first set of European guidelines on this topic.

1.5 New science on the effects of nitrogen deposition and concentrations on Natura 2000 sites (Theme 3)

Actions to manage the Natura 2000 network and to assess conservation status must be based on a sound scientific understanding of how reactive nitrogen deposition causes impacts on sensitive habitats. The workshop reviewed the latest evidence to:

- provide a clear picture of the scale of threat from nitrogen deposition to the Natura 2000 network and to its conservation status;
- consider the relative effects of different nitrogen forms, including ammonia versus nitrogen oxides (especially as this relates to different polluting source sectors) and to dry versus wet deposition (as this relates to near source impacts versus long range transport);
- evaluate the critical loads and levels approach, and consider the role of other approaches, including indicators from site level measurements to the European scale;
- consider the potential to improve relationships between concentrations/dose and biodiversity loss, as well as the use of management practices to mitigate nitrogen impacts.

It was concluded that the latest science supports and strengthens the already established empirical critical loads approach, encouraging their use in environmental decision making.

The workshop concluded that there are no acceptable exceedances above a critical load or critical level. Discussions regarding "acceptable exceedances" are not a science issue and should be addressed at a policy level. In order to improve the situation, one should aim at reducing nitrogen deposition below the critical loads and levels.

New data have strengthened the view that it is important to consider different nitrogen forms when evaluating effects of nitrogen deposition. It was concluded that evidence of responses for the different nitrogen forms is consistent across ecosystems and species. Moreover, because the effects from nitrogen deposition differ between different nitrogen forms (dry/wet deposition and oxidized/reduced nitrogen) it is important to evaluate their effects independently. Hence several types of critical loads/levels for a particular habitat type are needed. For example, the critical level for ammonia may be well below the critical load set for total nitrogen deposition. Hence it is important that both critical loads and levels are used.

Important new data from Southern Europe have emerged over the last five years, for example, during the workshop results from experiments and surveys conducted in Portugal and Spain were presented. These should inform future revisions of critical loads for nitrogen.

The workshop concluded that improved conditions following reduction in nitrogen deposition are only relevant when nitrogen deposition is reduced below the critical load/level. Reduction of exceedance will only improve the situation in the sense that it reduces the risk of further worsening

of the effects. Information about the effects on recovery time following reductions below critical loads/levels is still largely lacking. Available data suggest that the rate of improvement will differ depending on type of function/species studied, and is often site specific.

It was concluded that management to reduce the impact of nitrogen deposition will only work in combination with reductions in nitrogen deposition and should not be seen as an alternative to reducing the nitrogen deposition. For semi-natural habitats, positive effects from reducing the nitrogen inputs will only be possible in combination with appropriate management.

The workshop agreed that there are important interactive effects between nitrogen deposition and climatic factors. Therefore a changing climate may also influence the effects of nitrogen deposition. Currently, the knowledge of such interactive effects, and how they may change with a changing climate is, however, poorly understood. The climatic factors most important for interactive effects with nitrogen are also the most uncertain in climate change modelling (e.g. precipitation), making predictions of future interactions between nitrogen deposition and climate change difficult.

It is recommended that future research should prioritize the assessment of relative impacts of different nitrogen forms in relation to critical thresholds and dose response relationships, the relationships between nitrogen dose and site- and landscape-level management practices as a basis for minimizing adverse effects on ecosystem integrity, and the quantification of the interactive effects between climate change and nitrogen deposition.

1.6 Approaches to modelling local nitrogen deposition and concentrations in the context of Natura 2000 (Theme 4)

Assessments of the threat of nitrogen to the Natura 2000 network are fundamentally dependent on the ability to model the pathway from emissions, though air chemistry to deposition. There are currently many atmospheric models available, and recent reviews (for ammonia) have considered these at both local and regional scales. The challenge of the present workshop was to address the effectiveness of such models for assessments in relation to the protection of Natura 2000 sites, including the different nitrogen forms, and consideration of relative contributions from short range, mesoscale and transboundary (international) atmospheric transport.

Key questions included, how well we can simulate measured air concentrations for comparison to critical level estimates, and to what extent ecosystem specific dry deposition rates are treated in models. Specific examples were considered of where models have been applied in existing case studies to investigate the relative contribution of emissions from different sources to nitrogen deposition and concentrations experienced at Natura 2000 sites.

It was noted that modelling assessment approaches differ widely from country to country, both in terms of the type of models used and the level of detail considered. In particular, two types of assessment can be used (source-based or receptor-based) and the workshop recommended the type used should be clearly specified in all assessments.

The workshop concluded that the uncertainty in concentration predictions by models is much smaller than the uncertainty in the deposition predictions. This has the practical implication that, from the perspective of the atmospheric modelling, assessments based on air concentrations will have less uncertainty than those based on atmospheric deposition.

The workshop noted that the emissions from fertiliser (including both inorganic mineral fertilizers and organic manures) when applied to land is not usually modelled in current assessments. This is

a major gap in current practice, given the substantial contribution to nitrogen deposition at many Natura 2000 sites from the nearby land application of fertilizers to agricultural land.

The workshop concluded that estimation of dry deposition of nitrogen compounds remains highly uncertain. In particular, uncertainty analysis for dry deposition is needed but remains a difficult task.

The workshop recommended that validation datasets for both concentration and deposition need to be developed and compiled in a form that can be made readily available for the purpose of model verification.

The workshop recommended that further development and testing of nitrogen dry deposition parameterisations are needed as a means to reduce uncertainties in assessing total nitrogen inputs to Natura 2000 sites. In particular, further assessment of ammonia canopy compensation points is needed for different habitat types. Overall, much more field deposition data are needed for model verification.

The workshop recommended that the emissions of ammonia to the atmosphere following fertiliser application (including both organic manures and mineral fertilizer) should be included in future environmental assessments of the impact of current and future activities on Natura 2000 sites.

It was recommended that a harmonised approach to uncertainty analysis for the models needs to be developed to aid the regulatory assessment of nitrogen emission, dispersion and deposition to sensitive habitats.

1.7 Options for future policy development to manage and mitigate the impacts of nitrogen deposition effects on the Natura 2000 network (Theme 5)

One of the motivations for the workshop was the perception that current practices to protect Natura 2000 from nitrogen deposition are far from optimal. While, in principle, the Habitats Directive affords the highest level of protection, much of the Natura network remains under threat. The workshop therefore reviewed the options for future policy development to better protect the Natura 2000 network. While the focus was on Natura 2000, the challenge was also viewed in the context of the wider aims of the Habitats Directive (inc. habitats/species outside of Natura 2000 sites) and other European biodiversity policy.

The workshop analyzed the current mechanisms by which the Habitats Directive affords protection to Natura 2000 sites, including the application of cross-compliance with other European Community legislation. It discussed the existence of potential loopholes, where certain polluting activities continue without formal review and assessment, including the relative roles of industrial, transport and agricultural emissions.

The workshop then reviewed a wide range of potential future options that could support Natura 2000 protection from nitrogen deposition, including: the strengthening of existing legislation, the application of spatial and land use-based policies, the role of ecosystem services, consideration of air quality objectives and local air quality management for the protection of Natura 2000 sites.

Overview of current situation with regard to nitrogen deposition impacts to Natura 2000 sites

Regarding the current policies and their adequacy for the protection of Natura 2000 sites from the threat of nitrogen deposition, the workshop concluded that:

- The Natura 2000 network remains under threat from atmospheric nitrogen deposition despite the Habitats Directive affording it a high level of protection.
- Atmospheric nitrogen deposition is a Europe-wide problem but with very high spatial variability in severity of impacts and a high variability in national policy responses.
- Natura 2000 sites are not routinely assessed for the risk of nitrogen deposition effects and present policies and /or their enforcement are not sufficient.
- A lack of awareness of the nitrogen threat is the main problem in many Member States.
- Ammonia emissions present the greatest policy challenge in Europe.
- There is currently insufficient linkage between biodiversity and air pollution policy development.
- Economic and conservation priorities clash particularly in countries with significant levels of nitrogen deposition.

Recommendations for policy development

The role of existing legislation
It was recommended by the working group that:

- Those Member States that have advanced policies integrating several legislative instruments could provide practical advice for other Member States.
- International agreements (National Emissions Ceilings (NEC) Directive and Gothenburg Protocol) should have a higher level of environmental ambition (especially for ammonia), in particular to improve protection at local scale.
- Exceedance of critical loads (including in Natura 2000 sites) should be more explicitly considered in optimization of abatement measures.
- Ammonia should be included in the Air Quality Directive (2008/50/EC) and there is potential for setting standards for annual mean concentrations of ammonia to protect ecosystems.
- The potential for 'cross compliance' of different legislative measures to address nitrogen deposition issues should be more actively promoted.
- All existing projects should be captured by Article 6.3 of the Habitat Directive.

Future options for protection of Natura 2000 sites
The working group discussions captured the following suggestions and recommendations:

- Legislation at both regional and local scales is needed, including measures to deal with within-country atmospheric transport.
- Policies and procedures should be considered that distinguish between the management of nitrogen oxides and ammonia, and to address the role of organic nitrogen compounds emitted to the atmosphere.
- It is recommended that new approaches are explored in future policy development to complement existing approaches to managing the nitrogen deposition threat in relation to Natura 2000 and the wider objectives of the Habitats Directive, including:

- Multi-media regional reactive nitrogen ceilings, limited by the most sensitive N_r species and effect, should be explored as a basis for further policy development. This approach could enable the optimization of all nitrogen emissions of a region in relation to the adverse impacts;
- Nitrogen reduction plans could include a long-term plan to attain critical loads on a regional level in countries with high levels of exceedance;
- Spatial Planning (operated at local and regional levels) can optimize the location of existing pollution sources to minimize the overall threats, exploiting where possible landscape structures to buffer impacts (including buffer zones and tree belts);
- Nitrogen impact assessment techniques should be further developed to take into account the objectives of the Habitats Directive more specifically;
- The Ecosystem Services concept may provide a holistic framework for examining the links between air pollution effects on ecosystems and human well-being.

• The following specific measures were recommended for further consideration:

- Improve ammonia coverage in the Integrated Pollution Prevention and Control (IPPC) Directive, i.e. include manure spreading, consider the current farm size thresholds and inclusion of cattle;
- Set strict emission limits and management obligations to encourage abatement technology development;
- Strategic Environmental Assessment (SEA) has a role to play at high level planning for pollution abatement;
- Develop and encourage non-technical measures (societal behaviour);
- ***A long term goal to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030***.

1.8 The way forward

The outcomes of the Workshop will be used to inform future research, environmental practice and policy development in relation to the threat of nitrogen deposition on European habitats. It was noted that there is currently no established framework for the harmonization of decision making approaches related to the threat of nitrogen deposition to the Natura 2000 network or on conservation status. Further effort is needed to consider how to develop such a framework in future.

The scientific outcomes, regulatory experience and policy options reported at the workshop will be considered for feeding into future plans at national, European and international scales. In particular, the messages will be fed into the Expert Group on Reporting under the Nature Directives, the UNECE Convention on Long Range Transboundary Air Pollution, through its subsidiary bodies (e.g. Working Group on Effects, Task Force on Reactive Nitrogen), and into work of the UN Convention on Biological Diversity on development of the nitrogen deposition indicator.

2

INTRODUCTION TO THE STRUCTURE OF THE WORKSHOP AND THIS VOLUME

2.1 The Context of the Workshop

Atmospheric nitrogen deposition represents a major threat to European biodiversity. Nitrogen emissions to the atmosphere have increased substantially over the 20th century, mainly as ammonia from agriculture and nitrogen oxides from industry and transport. Following atmospheric dispersion and chemical processing, these nitrogen forms are deposited across European landscapes, providing unplanned nitrogen inputs and adversely affecting many sensitive habitats.

The issue represents a serious challenge for the conservation of natural habitats and species under the Habitats Directive (92/43/EEC).

The Habitats Directive is a cornerstone of Europe's nature conservation policy. It promotes the maintenance of biodiversity and requires Member States to take measures to maintain or restore natural habitats at a favourable conservation status. The Directive establishes the Natura 2000 network with the aim to assure the long-term survival of Europe's most valuable and threatened species and habitats. These sites are afforded the highest degree of protection under European legislation: the provisions of the Directive require strict site protection measures and avoidance of deterioration. It introduces a precautionary approach to permitting "plans or projects" which are likely to have significant effect on a site.

Control of emissions to air of reactive nitrogen are regulated under several directives including the National Emissions Ceilings Directive (NECD, 2001/81/EC), the Large Combustion Plants Directive (LCPD, 2001/80/EC), the Air Quality Directive (AQD, 2008/50/EC) and the directive on Integrated Pollution Prevention and Control (IPPC, 96/61/EC). A range of other policies and legislation also influence emissions, such as the Nitrates Directive (91/676/EEC). However, the impacts of nitrogen deposition on the Natura 2000 network (and the habitat and species resource outside of the network), together with the associated impacts due to elevated concentrations of ammonia (NH_3) and nitrogen oxides (NO_x), are often not addressed adequately or systematically; this is despite the strong protection measures in place through the Habitats Directive.

The Habitats Directive does not directly address nitrogen impacts and until now there has been no common European approach for determining the impacts of nitrogen deposition on individual sites or on conservation status. At the same time, the scale of pollution exposure suggests that there are widespread threats to the Natura 2000 network and to conservation status more widely due to the concentrations and deposition of reactive nitrogen species.

Noting these problems, a workshop was organized to bring together scientists, environmental managers and policy makers to clarify the current understanding of the key issues. The workshop

was held in Brussels in 2009 and addressed the different components of science, environmental management and future policy development needs.

2.2 Outline of the workshop

The specific aims of the workshop were to:

- compare case studies of nitrogen (N) impacts on Natura 2000 sites from across Europe,
- compare national criteria for risk assessment between countries,
- develop clear messages that could improve assessment approaches,
- communicate the scale of the nitrogen threat to the Natura 2000 network,
- review the role of cross-compliance on managing Natura 2000 sites,
- link the science with decision making at local to European scales.

Taken together these aims contributed to the overall workshop goal: to harmonize approaches for determining the impacts of atmospheric nitrogen deposition on Natura 2000 sites and review the future policy options.

The workshop was structured into themes addressed by five Working Groups, supported in each case by a background document setting out the issues in detail and the challenges currently faced:

1 Comparison of impact assessment and decision making approaches to determine the nitrogen deposition impacts associated with plans and projects in the context of Habitats Directive Article 6.3 obligations (Bealey *et al*., this volume);

2 Comparison of approaches to assessing and reporting nitrogen deposition impacts on conservation status (Habitats Directive Article 17) and discussion of harmonising approaches for future reporting rounds (Whitfield and Strachan, this volume);

3 New science on the effects of nitrogen deposition and concentrations on Natura 2000 sites, including bio-indicators, effects of nitrogen form (e.g. reduced nitrogen, NH_x versus oxidized nitrogen, NO_y), and the relationships between critical thresholds and biodiversity loss (Nordin *et al*., this volume);

4 Approaches to modelling local nitrogen deposition and concentrations in the regulatory context of Natura 2000 (Hertel *et al*., this volume);

5 Options for future policy development to manage and mitigate the impacts of nitrogen deposition on the Natura 2000 network (Sutton *et al*., this volume, section 7.1).

Overall, the workshop encouraged links to be developed between the scientific basis of nitrogen deposition effects, regulatory practice and policy application. A graphical summary of the different themes and their relationships is shown in Figure 2.1. As part of the assessment, nitrogen effects were related to both atmospheric nitrogen deposition and atmospheric concentrations of reactive nitrogen compounds, including the use of critical loads and critical levels as effects thresholds.

The workshop was attended by 73 delegates from 13 countries: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Portugal, Spain, Sweden, Netherlands and the UK. The delegates were scientists, conservation practitioners and policy makers, including representatives from the European Commission DG Environment, and various Government Departments.

The country approaches and/or legislation reported in papers in this volume reflect the status at the time of the workshop, 2009. In some cases there may have been subsequent changes to policy or legislation, which are described where possible.

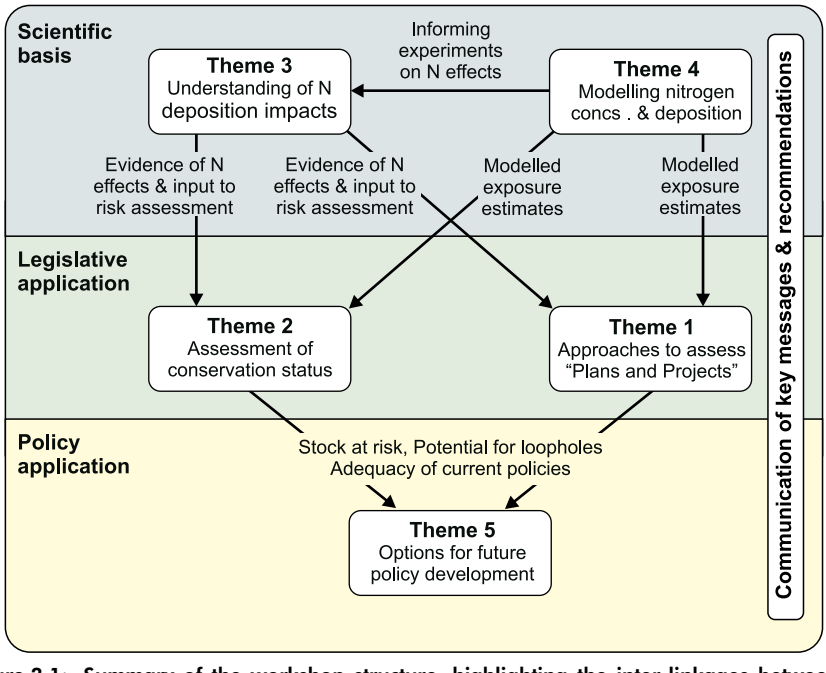

Figure 2.1: Summary of the workshop structure, highlighting the inter-linkages between the five main themes

3

APPROACHES TO ASSESSING THE IMPACT OF NEW PLANS AND PROJECTS ON NATURA 2000 SITES (THEME 1)

3.1 Background document

W.J. Bealey[1], A. Bleeker[2], T. Spranger[3], D. Bernotat[4], E. Buchwald[5] and M.A. Sutton[1]

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Summary

The Habitats Directive provides a high level of protection to the Natura 2000 network by taking a precautionary approach to permitting "plans or projects" which may have a likely significant effect on a site. Article 6.3 of the directive provides a mechanism by which plans and projects can only be permitted if they are shown to have no adverse effect on a Natura 2000 site.

Emissions of nitrogen are considered to be a significant threat to sensitive habitats across Europe. Many countries have adopted approaches to assessing these threats which include the use of critical load thresholds, the appraisal of the conservation objectives, and the determination of site specific conditions. These decisions include the need to understand and develop approaches for answering questions such as: what is a likely significant effect on the site; what is a significant contribution of a pollutant load to the site; and how to judge whether a project or plan will have an adverse effect on the integrity of a Natura 2000 site?

This background paper looks at Article 6 of the Habitats Directive focussing in particular on Article 6.3. An introduction to the requirements of Article 6.3 is given, followed by a consideration of the assessment of nitrogen deposition impacts in relation to these requirements. The paper compares the assessment and decision-making approaches taken by a number of EU Member States.

3.1.1 Introduction

The Habitats Directive (Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora) and the Birds Directive (Council Directive 79/409/EEC) provide a high level of protection to the Natura 2000 network by taking a precautionary approach to permitting. Plans and projects can only be permitted if they are shown to have no adverse effect on a Natura 2000 site, unless there is some form of overriding public interest why it should proceed.

While emphasis has been directed at reducing on-site activities, there is also a requirement for the assessment of off-site activities including the polluting effect of local and transboundary air

pollution sources. Emissions of nitrogen primarily from combustion and agricultural processes clearly present off-site pressures on the Natura 2000 network. Moreover, due to the proximity of the network to agricultural sources (both being present in the rural setting) nitrogen, particularly in the form of ammonia, contributes to widespread effects. Across the EU there is large exceedance of the critical load for nitrogen deposition for sensitive ecosystems. By 2020, 64 per cent of the natural ecosystem areas across the EU27 will be at risk from excessive nutrient N deposition (CCE, 2008).

3.1.2 The Habitats Directive

The provisions of the Habitats Directive require Member States to take measures to maintain or restore at favourable conservation status the natural habitats and species of Community importance. Additionally, Member States are obligated to designate the most suitable sites for these habitats and species under a network of sites across their respective countries. The Natura 2000 network is comprised of Special Areas of Conservation (SAC) designated under the Habitats Directive, and incorporates Special Protection Areas (SPAs) (classified under the 1979 Birds Directive). Together SACs and SPAs cover around 15 per cent of the territory of the EU. Under Article 6 of the Habitats Directive, Member States are required to establish the necessary conservation measures which correspond to the ecological requirements and conservation objectives of the site. These may be in the form of appropriate management plans or integration of other development plans, but essentially the deterioration of the habitats or species, including the disturbance of species, must be avoided. In addition, under Article 6.3 all plans and projects likely to affect a Natura 2000 site should be subjected to an assessment of the implications for the conservation objectives of the site. A plan or project can only be permitted after having ascertained that it will not adversely affect the integrity of the site concerned subject to the provisions of Article 6.4.

3.1.3 Article 6.3 and nitrogen deposition

> *Article 6.3 - Any plan or project not directly connected with or necessary to the management of the site but likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives. In the light of the conclusions of the assessment of the implications for the site and subject to the provisions of paragraph 4, the competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned and, if appropriate, after having obtained the opinion of the general public.*

Article 6.3 establishes the application of the precautionary principle for the first time for protected areas across Europe; that is, that projects can only be permitted if it has been ascertained that there will be no adverse effect on the integrity of the site. Projects may still be permitted if there are no alternative solutions, and there are imperative reasons of overriding public interest. In such cases compensation measures will be necessary to ensure the overall integrity of network of sites. Official guidance on Article 6.3 (European Commission, 2000) states that the geographical scope is not restricted to plans and projects which exclusively occur on a protected site ('on-site activities'), but they also target developments situated outside the site ('off-site activities'). Examples of on-site activities may include a highway intersecting a designated site or extraction of minerals. These represent actual physical damage to a site directly caused by the action of that activity.

Emissions of reactive nitrogen compounds from industrial and agricultural installations represent impacts from off-site activities. In respect of sources of nitrogen emissions, applications for permissions issued through various regulatory and planning instruments, give rise to a plan or project under the definition of the Directive. For example, an application for a permit under the IPPC Directive (Integrated Pollution Prevention and Control - EC Directive (96/61)).

In some cases, the sources may be many kilometres away (50-100 km) from the potentially affected site(s). The long-range transport potential of nitrogen pollutant species can trigger appropriate assessments where source and site are many kilometres from each other. In addition, localised impacts can also be important, for example local sources of ammonia from intensive agricultural units (<2 km). Furthermore, since these sources are usually located in rural areas, their potential for impacting a Natura 2000 site is more likely than in an industrial or urban area.

An overview of the requirements under Article 6.3 is given in the following sections:

'likely significant effect'

The first step is to consider whether the plan or project is likely to have a significant effect on a Natura 2000 site alone or in-combination. However, it is often hard to define what is **significant.** To assess a likely significant effect, the sites' conservation objectives and designated features should be considered. Finally the **likeliness** of a significant effect brings in the precautionary principle and an appropriate assessment should be carried out unless the likeliness of a significant effect(s) can be ruled out.

'subject to appropriate assessment'

For plans and projects that are likely to have a significant effect on a site, an appropriate assessment should be undertaken. The appropriate assessment should focus on the implications for the site in view of the site's conservation objectives. 'In combination' effects also need to be addressed in an assessment and account, needs to betaken of cumulative impacts (i.e. prevailing environmental conditions).

'not adversely affect the integrity of the site concerned'

The integrity of the site refers directly to the site's conservation objectives of the Annex I habitats or the Annex II species for which the site was designated (Annexes refer to the Habitats Directive). Integrity can be defined as: "*the ability of a site to maintain a coherent structure as a habitat or for supporting a complex of habitats and species*" (EC, 2000). The degradation of these features and their associated ecological functions would negatively affect the site's integrity. Assessments for sites designated as SPAs (Special Protection Areas - for birds) have to take into account the broad spectrum of habitats in which the protected bird nests, feeds or roosts.

The decision – compensation and overriding public interest.

Under Article 6.4 the competent authority (which will vary according to the type of plan or project and between Member States) is required to arrive at a conclusion regarding the consequences of the plan or project in relation to the integrity of the site concerned. If it is concluded that the plan or project would have no adverse effect, then the plan or project can proceed. If an appropriate assessment identifies that any activity cannot be proven to have no adverse effect, then the competent authority must refuse permission for the proposed plan or project.

In exceptional circumstances, a plan or project may still be allowed to go ahead, inspite of a negative assessment, provided there are no alternative solutions and the plan or project is considered to be of overriding public interest. In such cases, the Member State must take appropriate compensatory measures to ensure that the overall coherence of the Natura 2000 network is protected.

3.1.4 Comparison of approaches to Article 6.3 across the EU – country case studies

The approaches to Article 6.3 were compared across a number of EU countries. Comparisons were made between approaches taken in the UK, Germany, Netherlands and Denmark. These countries appear to have the most formalised procedures in respect of nitrogen deposition assessments required under Article 6.3. It was not possible to get details of the practices in other countries and

in some cases it is unclear how nitrogen deposition impacts from plans and projects are assessed, if at all. A full detailed approach for each of the countries that presented at the workshop is provided in Appendix 3.1, with the findings summarised in Table 3.1 below.

It is not surprising to find that most countries reviewed share some common approaches in the assessment of new/existing plans and projects and their impacts on Natura 2000 sites. Some key approaches are summarised below:

Site Relevant critical loads

Each country reviewed has carried out a process of linking designated features (habitats and species) and empirical critical loads for nitrogen. This has also included the assessment of whether a particular habitat/species is sensitive to nitrogen deposition. This approach is commonly used for determining likely significant effects and to assist with an assessment of potential effects on site integrity.

Distance parameter

Threshold distances are used by some countries as an initial step to identify relevant sources. This supports the screening process to exclude sources that are not going to impact on a particular Natura 2000 site. However, such distances take a rather different form between countries. In the UK, 10 and 15 km are generally used as distances that require screening assessment of individual activities regulated under the IPPC directive. In Denmark and the Netherlands, thresholds of 1 and 3 km are used for assessment of farm activities, though larger distances can apply in some circumstances.

Application of threshold factors

Critical loads and levels are typically used for comparing thresholds. They serve both to identify likely significant effects to a Natura site, and to determine whether an adverse effect will occur. There are a number of things to consider in assessing likely significant effects. The principle of what is a significant effect is defined by what is *de minimis* (trivial/inconsequential). In other words *de minimis* can be described as a process contribution that is small enough to be ignored For example, the <1 per cent contribution of a critical load/level (as used for some installations in the UK) could be seen as *de minimis* and having no significant effect as this represents 0.05 kg N $ha^{-1}yr^{-1}$ for the lowest empirical critical load (or 0.01 $\mu g/m^3$ for the lowest critical level for NH_3). However, there remains the question of what would be *de minimis* for the consideration of the cumulative effect of multiple projects. This presumably depends on the distribution of projects contributing to overall deposition (e.g. a few large combustion plants or many small farms).

In addition, there still needs to be a judgement on whether the plan or project is causing no adverse effect. This leads to the key question - what is an acceptable contribution? For Germany the extra nitrogen deposition for a project or plan has been set to 10 per cent of the critical load. This represents around 1 kg N $ha^{-1}yr^{-1}$ for a 'typical' critical load of 10 kg N $ha^{-1}yr^{-1}$ and is seen as within the precision of measurement. In the UK an acceptable process contribution of 20 per cent (in combination) of the critical level/load has been used in the assessment of impacts from existing installations from intensive livestock sector, but no per cent threshold has been set yet for generic application. The basis for choosing different per cent thresholds for different source types is one of the key areas that requires discussion. However, there are still numerous factors that influence these potential outcomes and decisions under the Habitats Directive should be based on the site-specific situation and should be precautionary. If there is any reasonable scientific doubt about there being no risk to the integrity of the Natura 2000 site it should not be possible to conclude that there is no adverse effect. This provides a challenge for the risk assessment process, since where critical loads and levels are already exceeded, it remains a matter of doubt for example, whether to apply a threshold of 10 per cent or 5 per cent etc.

Table 3.1: Comparison of approaches to the implementation of Article 6.3 of the Habitats Directive, in respect of nitrogen impacts, across four EU Member States. The table reflects the situation in mid-2009.

Questions	Denmark*	Germany	Netherlands	United Kingdom
Is/are distance criteria set to identify relevant sources?	Farm projects within buffer zones of 300 meters and 1000 meters from wet and dry heath/dunes, dry grassland, raised bogs and nutrient poor waters in Natura 2000 sites are evaluated. Larger projects need assessment regardless of distance if they can affect a Natura 2000 site.	For appropriate assessments there are at the moment no official distance criteria, because it always depends on the project type, the emissions and a case by case assessment. Independently, air pollution law prescribes that nitrogen deposition effects caused by new or to-be-expanded existing sources on sensitive areas within the evaluation area (generally one km for agricultural sources) have to be assessed if a likely significant effect is likely.	In the current procedure for the Netherlands no clear distance criteria are set. According to Dutch jurisprudence every source that can lead to a (further) exceedance of the critical load is relevant, regardless of the distance. However, for other nature areas (part of the National Ecological Network, but outside Natura 2000) ammonia sources outside a three km zone are considered to be not relevant.	The following criteria are used to screen for relevant sources. Any large combustion process within 15km of a European site. 10km for any other large industrial installation (including intensive farming) regulated under the Integrated Pollution Prevention and Control Directive. Reduced distances applied to smaller processes. A long-range assessment is also required for Large Combustion Plant.
Are critical loads/levels used at a site assessment level across the Natura network?	Yes, the UNECE critical loads have been translated into a national list of critical loads for all Natura 2000 habitat types. Critical levels have not been used.	Yes. Empirical critical loads have been used for the assessment of nitrogen deposition based on habitat type. But they are not directly used as levels for adverse effects (see below: " per cent contribution from project").	The critical loads for habitats have been assigned across the Natura 2000 network. This work included the assessment of habitat sensitivity to nitrogen deposition (van Dobben & van Hinsberg, 2008).	Critical loads have been assigned to designated features and mapped across the Natura network and compared with deposition values (Bealey *et al.,* 2007). Critical levels for ammonia have been assigned to Natura 2000 sites where potentially impacted by intensive farming installations.
Are Exclusion Zones used around sensitive Natura 2000 sites?	300 metres zone is prohibited for new farms and capped emissions for existing farms within this zone.	No	No	No

The situation in Denmark has changed since the workshop. For an update please see the Danish country report, Section 3.3.

Questions	Denmark*	Germany	Netherlands	United Kingdom
Does the assessment take into account multi-sources in-combination with each other?	Within 300-1000 metres the allowable extra contribution of deposited NH_3 from each farm is 0.3 kg N $ha^{-1}yr^{-1}$, for three or more farms (0.5 kg N $ha^{-1}yr^{-1}$ for two farms and 0.7 kg N $ha^{-1}yr^{-1}$ if there is only one farm)	In general yes, but there are still major methodological problems.	It is important to include all relevant activities to determine cumulative effects. This cumulative effect also takes into consideration any background deposition.	In combination effects (multi-sources) are taken into consideration.
Is location of interest feature and extent of impacts assessed?	No, same regulation is applied regardless of current state of the habitat, as deposition has to be lowered for habitats in a bad state in order for them to recover.	A concept of assessing the size of the affected area could be introduced for future guidance.	The location of a particular feature is taken into account as much as possible when assessing the impacts.	An assessment is made of the size of the site and the location of a particular feature in relation to the predicted pollution footprint.
Is a per cent contribution of nitrogen deposition from the project compared with critical loads/ levels?	No, per cent contribution is not used. Instead allowable extra contribution of deposited NH_3 from each farm is defined (see above).	For appropriate assessments, a project or plan contribution of 10 per cent of the critical load is tolerable even if the background (or background + the source) is already exceeded. This not applicable if the site is in unfavourable status caused by nitrogen inputs. These cases are assessed on a case by case basis.	At the moment no particular per cent nitrogen deposition in comparison with critical loads is taken into account.	Yes, likely significant effect based on a proportional contribution of the critical load or level. Intensive farming – 1 per cent-4 per cent, depending on how conservative the screening model used is. Other large IPPC Installations – 1 per cent.
Is the legal status of a designated site taken into consideration when comparing thresholds (e.g. Natura site vs a local nature reserve*)? **[* Appropriate Assessments are only carried out for Natura 2000 sites]**	Yes. Natura sites have more strict protection, but all oligotrophic lakes (type 3110), all raised bogs (type 7110+7120), and all large (>10 ha) heaths & grasslands also have buffer zones with similar protection even outside Natura 2000 sites.	Appropriate Assessments are only carried out for Natura 2000 sites. In general the Natura 2000 sites are protected the most strictly. The assessment of nitrogen deposition effects in German air pollution abatement law sets a mandatory target based on the critical load (x1). Other lower designation status sites can vary between x1 to x3 of the critical load.	These assessments are only carried out for Natura 2000 sites. For non-Natura sites, 'normal' Dutch legislation applies, taking into account emission ceiling zones around nature areas.	Yes – precautionary approach for Natura 2000 sites. For example, for ammonia impacts from (existing) intensive farming the allowable process contribution of the critical load or level is 20 per cent for SACs/SPAs, 50 per cent for SSSIs, 100 per cent for county wildlife sites.

Questions	Denmark*	Germany	Netherlands	United Kingdom
Are abiotic conditions taken into account?	Roughness of surface of the habitat and of the area between the source and the habitat is entered into calculation of the deposition contribution from a farm. In some cases hydrology, roughness of habitat or e.g. harvest of biomass are checked in order to better resolve local deposition and/or local critical loads.	The abiotic conditions that are important to a habitat or species are taken into account. The focus is on the most important abiotic condition(s). But until now which abiotic factors are relevant for the assessment of nitrogen deposition has not been determined.	The abiotic conditions that are important to the continued integrity of a habitat or species are identified. The initial focus is on the most limiting abiotic condition(s). Abiotic conditions include acidity, water content, salinity, nutrient availability, tolerance to flooding, groundwater level.	Any potential hazard from the proposal, which could affect the interest features are noted. This includes some 'abiotic' factors e.g. toxic contamination, nutrient enrichment, acidification, changes in salinity regime and changes in thermal regime.
Is the site assessed for current condition status? (favourable/ unfavourable)	The Annex 1 habitats are mapped including condition assessment. Habitats have protection whether or not they are in favourable condition.	A project contribution of 10 per cent of the critical load is not applicable if the habitats or species of a site are in unfavourable conservation status caused by nitrogen inputs. These cases are assessed on a case by case basis.	The present condition of the habitat or species is assessed.	The condition of the site is taken into account to a certain degree but it is recognised that current UK site Common Standards Monitoring (condition assessment) is not sensitive enough to detect and attribute air pollution effects (it was not designed for this). Questions asked include how long the project has been there, has there been any monitoring done on site and its relevance in relation to impact from the project.
Are long-range effects taken into account?	In general, total deposition and critical load exceedances are not used in the assessment. Yet, large sources at larger distances from a Natura site should also be included in the assessment of nitrogen deposition.	Yes – sources at larger distances from a Natura site are also included in the assessment of nitrogen deposition if there is a possible causal connection.	Yes – sources at larger distances from a Natura site are also included in the assessment of nitrogen deposition.	Yes – long-range contribution taken into account in determining background pollutant contributions. Long-range process contributions taken account of for major combustion processes beyond 15 km.

The need to deal with cumulative effects of multiple projects also remains a challenge for the decision-making process and Table 3.1 shows that there are a range of approaches in Europe. The appropriate assessment needs to consider in-combination effects as well as prevailing environmental conditions (European Commission, 2000). Plans and projects may not be significant on their own but their authorisation could lead to 'critical load exceedance creep'.

Alternatively, an approach could be to place limits on total deposition rather than a per cent contribution to a critical load. Denmark applies a threshold based deposition where any new agricultural 'installation' within 300-1000 metres of a Natura 2000 site is allowed an additional contribution of x kg N $ha^{-1}yr^{-1}$ to a sensitive Natura 2000 site depending on how many farms are involved (see Table 3.1).This is an interesting alternative and eliminates the use of critical loads/ levels (although they were considered in the derivation of the thresholds see Bjerregaard *et al*., this volume).

Conservation objectives and favourable status

For most countries, consideration is given to the conservation status of the site. Further additions of nitrogen are avoided when a site is deemed to be at unfavourable status, particularly when this is caused by nitrogen inputs. The Habitats Directive requires that judgements of 'likely significant effect' and 'no adverse effect' must be made in relation to the interest features for which the Natura site is designated, focusing on the conservation objectives of each feature. Country assessments often examine the ecological requirements a feature may have, looking at ecological function, sensitivities to nitrogen and the extent of impact across the site.

3.1.5 Conclusions and recommendations for workshop discussion topics

We have been able to present information in this paper for four European Member States, but the question may be asked how these regulatory practices compare in other Member States. The countries reviewed above are the most prominent in terms of guidance and practice in tackling the issue of atmospheric nitrogen deposition and ecosystem impacts.

It is clear from the country reviews that there are some key issues that are important in assessing impacts of nitrogen on the Natura 2000 network. Article 6.3 of the Habitats Directive brings to light a number of important challenges for assessing any plan or project impact on a Natura 2000 site. Recommended discussion topics for the workshop were:

1 What is a likely significant effect and how is it defined?

2 What is a significant contribution from a project or plan in relation to either a habitat critical load or an emission target?

3 What if the background is already exceeded? How much more additional nitrogen is seen as having no adverse impact on the integrity of a site?

4 How should in-combination (multi-source) effects be handled? For example, can *de minimis* values be set for the consideration of individual project contributions where the cumulative effect of many projects is being considered?

5 Is there sufficient knowledge within the scientific community on effects to be able to guide practitioners into making decisions on site integrity and what constitutes a likely significant effect?

6 Where are the relevant gaps in this scientific knowledge?

7 Are critical loads and levels fit for the purpose of site level assessments since they were originally developed for national risk assessments to inform the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP)?

8 What rules should apply for new plans or projects where background critical loads and levels are already exceeded? How should *de mimimis* be defined and cumulative (and in combination) effects be handled in this instance?

One theme running through the country reviews is that decisions often have to be made at the site specific level. Each site has its own set of ecological requirements and sensitivities.

9 Is there enough information at every site to be able to inform a regulator/site manager about these requirements when it comes to nitrogen deposition?

10 Is sufficient information available on conservation status to conduct an appropriate assessment for different Member States?

11 What would constitute an outline of 'best practice' in conducting such assessments, and what are the main limitations among the Member States to implementing this?

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Appendix 3.1: Country case studies

The following sections provide further detail of the approaches used by some Member States for assessing nitrogen deposition impacts on Natura 2000 sites in respect of Article 6(3). This reflects the situation in mid-2009, in some cases there has been further development of the approaches since the workshop.

United Kingdom – W.J. Bealey, Centre for Ecology and Hydrology, UK

In the UK the Habitats Directive is transposed into national law by means of the Conservation (Natural Habitats & c.) Regulations 1994 (as amended, known as the Habitats Regulations), including separate, but related regulations for the devolved regions Scotland and Northern Ireland.

In relation to assessing emissions of air pollutants from new plans and projects, including nitrogen emissions, the responsibility lies with a range of competent authorities depending on the relevant licensing regime. New plans or projects require planning permission which is often the responsibility of local planning authorities. In addition, polluting activities above a certain size will require a pollution control permit from the appropriate local or national regulator. The UK Habitats Regulations require that an assessment of the impact of the site for Habitat Directive purposes is carried out by the most appropriate authority.

This country review focuses on the application of Article 6.3 by the Environment Agency in England and Wales of applications for pollution control permits under the IPPC Directive (such as power stations and agricultural installations). For this, the Environment Agency and the statutory conservation agencies (Natural England and Countryside Council for Wales) have developed a staged risk assessment requiring increasing detail at each stage if effects have not been discounted, in line with the tests of the Habitat Regulations. The exact form of the assessment will depend on the characteristics of the industrial sector concerned. Furthermore, in this approach, critical loads and levels are instrumental to the assessment of nitrogen impacts from industrial and agriculture installations. The four stage process outlined below uses the concept of critical loads and levels to assess impacts on designated features making up a given Natura 2000 site.

Stage 1 – Identification of all 'relevant' permissions.

This initial stage has been set up to identify any projects or plans, which need further assessment, based on distance-based criteria from a designated site. These are:

- any application within the boundary of a Natura 2000 site ,
- any centrally dispatched coal or oil-fired power station within 15km of a Natura 2000 site,
- any other major installation (including intensive livestock farms) within 10km of a Natura 2000 site.

Additionally, long-range effects of major combustion processes should also be taken into account for a project or plan beyond 15km.

Stage 2 – Assessment of whether the permission is likely to have a significant effect (alone and/or in combination).

This is the key Stage in determining whether a project requires an appropriate assessment. Under the Habitat Regulations a likely significant effect is described as:

> *"..any effect that may reasonably be predicted as a consequence of a plan or project that may affect the conservation objectives of the features for which the site was designated, but excluding trivial or inconsequential effects."*

Stage 2 is based on source modelling to predict the process contribution of the concentration and deposition to the Natura 2000 site(s). This procedure acts as a screening process to separate out inconsequential sources. In the UK critical loads and levels are used as an 'environmental benchmark' to assess the potential impact on a site. The significance of the effect of an emission will depend on both the ambient (background) concentration/deposition at the site and the relative contribution of the process under consideration. Atmospheric dispersion models are often used to estimate the process contribution at the site. The critical loads/levels of the designated features of the site are established (Bealey *et al.,* 2007) and are then compared with the modelled process contribution and the background. This procedure is often described by the following equation:

$$\mathbf{PEC = PC + BC}$$

where PEC is the Predicted Environmental Concentration, PC is the Process Contribution and BC is the current background concentration (for Concentration you can also read Deposition PED, PD etc.).

The Environment Agency and conservation bodies have allocated an initial test threshold of between 1 and 4 per cent of a critical load or level depending on the industrial sector and how conservative the screening model used is. Therefore, a process contribution of less than one - 4 per cent of the critical load or level is seen as not significant, alone or in combination. If the PEC < 70 per cent of the critical load/level then there is also an assumption of no likely significant effect (even if PC > 1-4 per cent of critical load/level).

Stage 3 –Where 'a likely significant effect' on the site has been identified, undertake an appropriate assessment to determine adverse effect.

The outcome for an appropriate assessment, under Article 6.3, is to determine that there is no adverse effect on the integrity of the site concerned from the project or plan proposed. The assessment should be carried out in view of the site's Conservation Objectives. It should take into account uncertainties in the modelling and the critical load/levels and must clearly demonstrate how a specific impact on an interest feature then relates to the integrity of the interest feature and thus the site. There are however some general assumptions which the decision should be based upon, all of which rely on the basis of scientific uncertainty and what is a **significant/acceptable contribution:**

1 If the PEC < 100 per cent of the CL then there is an assumption of no adverse effect.

2 If the BC < CL, but a small PC leads to an exceedance then a decision should be made on the basis of local circumstances, taking into account the magnitude of exceedance, the likely ecological effect of exceedance on the features and site integrity, relative contributions from different sources (in combination) and whether the environmental criteria are likely to be met at some future date.

3 If the BC > CL and the PC will cause an additional small increase then, as above, the decision will have to be made on a case by case basis and on individual circumstances.

4 If the BC < CL, but the PC is significant and leads to an exceedance, then the application should be refused. The PC can be viewed as adding a significant additional risk to the site's integrity.

The decision as to when it can be concluded that there is no adverse effect on the integrity of the site will be a matter of judgement for the competent authority. However, in some circumstances, for example, intensive livestock farms, specific assessment criteria have been developed to enable decisions to be taken in a consistent manner when dealing with a large number of permit applications over a short period.

Stage 4 –Determination of the application

The appropriate assessment of the impacts of a plan or project on a site, provided for in Stage three enables the competent authorities to arrive at a conclusion whether the project or plan has an adverse effect on the integrity of the site. In cases where it is not possible to conclude no adverse effect on integrity, the competent authority has to consider if there are alternative ways that a conclusion of no adverse effect can be reached. For example, in some cases, permit conditions have been set to reduce emissions by a certain deadline. Failing this, further provisions of Article 6.4 would be considered e.g. overriding public interest. Article 6.4 of the Habitats Directive allows the competent authority to permit the project on reasons of overriding public interest, including those of a social or economic nature, which require the realisation of the plan or project in question. Under such circumstances compensatory measures should be taken.

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Bealey, W.J, Dore, A., Vieno, M. and Sutton, M.A. (2007) Source attribution and critical loads assessment for Special Areas of Conservation and Special Protection Areas in the UK. SNIFFER Project AQ02.

Netherlands – A. Bleeker, Energy Research Centre of the Netherlands (ECN)

The Habitats Directive is transposed into national law by means of the Nature Conservation Law (1998). The Nature Conservation Law makes it possible to address the Habitats Directive requirements, by means of maintenance plans for individual nature areas (under the Habitats Directive) and/or via specific environmental permit procedures for activities that potentially contribute to a decrease of the quality of the habitat or a significant disturbance of species. The responsibility for the implementation of these regulations lies with different competent authorities, but mainly with the local administrative level (as far as environmental permits are concerned).

The overall procedure with respect to the implementation of the Habitats Directive (and especially the procedure concerning the assessment of activities in the vicinity of nature areas) has been subject to much debate in the Netherlands over the last few years; especially the implementation of the Habitats Directive in relation to regulation of existing ammonia sources. This is mainly due to the fact that the existing regulation of ammonia emissions is not strictly effects based, while the Habitats Directive implementation definitely requires some sort of effects based approach. The method for dealing with nitrogen impacts, under the Habitats Directives, continues to be developed.

Because of the many problems that emerged during the implementation phase, a guidance document was developed describing possibilities of judging environmental permit applications in relation to existing and/or future activities in the context of the maintenance plans. This guidance document focuses mainly on nitrogen deposition and its purpose is to guide the legal authorities at a local, provincial and national level in the construction of maintenance plans. The basis for this guidance document originates from the recommendation from the so-called 'Task Force Trojan,' established by the Ministry of Agriculture, Nature and Food Quality (LNV), that concluded that for existing use, and possible future activities (where nitrogen deposition is involved), all factors that influence the attainment of the ecosystem targets need to be taken into account. Nitrogen deposition is only one of these factors.

The guidance document does not provide a complete solution for the overall process of judging environmental permits and therefore the legal authorities are responsible for making 'site specific' decisions, taking into account all relevant factors. In the following text, a further description of the guidance document is given in relation to the maintenance plans.

Role of the Habitats Directive maintenance plan

In the maintenance plan, the overall picture with respect to meeting the ecological targets is laid down and choices are made: which factors are the most important for meeting the targets; which measures are needed; what is the relation with existing use; what are the local conditions; what are the site specific objectives of the habitat types with regard to total area covered, exact locations and time (i.e.: how fast do we have to reach the targets)?

The maintenance plans give a better understanding about which activities are allowed and which activities are (without further conditions) not possible in relation to the targets.

Question to be answered:
The legal authorities are responsible for judging permit requests for individual situations with respect to potentially harmful (future) activities and to include (as much as possible) all the relevant factors. The following questions are important:

1. What are the targets for nitrogen deposition for the species and habitat types under protection and sensitive to nitrogen deposition?
Not all species and habitat types are equally sensitive to nitrogen deposition. An overview of the sensitivity of Habitat types is given in Van Dobben & Van Hinsberg (2008). If the species or habitat type is not sensitive to nitrogen deposition, a new activity that is being investigated can be permitted (unless other effects of the activity are not meeting the targets for the habitat types).

2. What is the location within the Natura 2000 site of these species and habitat types?
For judging the activities, it is important to know where the nitrogen sensitive habitat types and species are located within the site. This is important since nitrogen deposition can vary significantly between different locations in a site.

3. What is the present state for these species and habitat types?
The present state describes the condition of the habitat type or species. The relevant aspects of the local condition are described in the national 'Natura 2000 Profiles Document', which consists of detailed descriptions of the habitat types and species and their environmental requirements (Ministry of ANF, 2008). The quality of habitats and the threats in all Natura 2000 sites has been assessed to a certain extent (because this was important for setting the targets), but the legal authority has to collect further information. When this information is not available from e.g. the provincial authorities or conservation organisations, further ecological research is needed.

4. What are the abiotic conditions that are important for these species and habitat types and which (limiting) conditions determine the present state?
In the Profiles Document the 'ecological demands' describe the abiotic conditions needed for an optimal development of habitat types and species. The ecological demands look at the following abiotic conditions:

- acidity,
- water content,
- salinity,
- nutrient availability,

- tolerance for flooding,
- groundwater level.

Nitrogen deposition influences the abiotic conditions related to acidity and nutrient availability. Nitrogen deposition has an acidifying and nitrifying effect. Habitat types and species have demands with regards to different abiotic conditions. When judging existing use or future activities it is important to find out which abiotic conditions are important for the development of habitat types and/or species and which abiotic conditions are limiting with regard to realising the targets. This means: which abiotic conditions are important for the specific habitat types and species and need to be improved or maintained to reach the targets. In first instance the focus is on the most limiting abiotic condition(s). However, eventually all abiotic conditions that are limiting for the targets have to be optimal.

5. What is the prognosis for the development of the relevant abiotic conditions?
Based on an ecological analysis and recent developments of the abiotic conditions, a prognosis can be made for the future. This prognosis can be used for assessing existing or future activities and can be based on information about:

- recent or proposed measures on national or area scale,
- recent development of (economical) activities on national or area scale.

For the assessment also the timescale for reaching the targets for the nitrogen sensitive habitats and species is important. If the abiotic conditions of a habitat in a Natura 2000 siteare clearly improving and these improvements are sufficient with respect to reaching the targets, the effects of the (future) activities do not need to be judged as being significant. At the moment of permitting an activity no reasonable scientific doubt may exist about the positive effects occurring and that the extent of these permitted activities is thus not significant.

6. What is the effect of the (future) activities on the abiotic conditions?
Here only the effect of nitrogen deposition due to the (future) activity is of relevance. This amount of nitrogen deposition can be assessed by means of dispersion and deposition models. For the Dutch local situation the model “Aagrostacks” and OPS is used.

7. What are relevant activities in and near the Natura 2000 site and what is their cumulative effect?
When assessing the cumulative effect of different relevant activities, it is important to include all effects that have an influence on the different abiotic conditions relevant for the specific habitat type or species. The cumulative effect deals with both the additional negative effects of other nearby activities as well as the positive effects of mitigating measures.

When assessing the cumulative effect of nitrogen deposition, not only the deposition due to sources in or around the Habitat area has to be considered but also the background deposition. For the effect on the abiotic conditions it does not make a difference if the deposition is caused by a source located nearby or at larger distances from the nature area. It also does not make a difference if the deposition is caused by an agricultural source, industry, energy producer or traffic. The total amount of deposition is what is relevant and the effect it has on the nitrogen sensitive habitats or species.

The more complete the answers are to the questions above, the more likely it is that a decision can be made on whether or not a permit for new activity can be given.

Judging these seven points/questions in an integrated way is very important. The factors that are important for question six (effect of the activity) is different for each situation, but the final 'answer' also depends on e.g. the accumulation of effects of other activities (question 7).

8. What if the (future) activity doesn't meet the targets?
The outcome of the integrated investigation (based on answering the seven questions) can be that the (future) activities near a Natura 2000 site will result in non attainment of the targets for that site.

The legal authority than has the following options:

- Start discussing further conditions for the (future) activity. The applicant can be advised to take emission reduction measures, by which a sufficient nitrogen deposition can be achieved.
- Start discussing alternatives for the (future) activity. The applicant can be advised to start looking for an alternative, like e.g. move to another location.
- Take additional measures, enabling meeting the targets to be achieved despite the (future) activity. It should be monitored however, that these measures are indeed implemented.

If these options do not bring a solution, the plan or project cannot be permitted. In the case of existing activities the legal authorities can facilitate the moving of the activity, e.g. by subsidizing the relocation of farms.

References:

Van Dobben, H. and van Hinsberg, A. (2008) Overzicht van kritische depostiewaarden voor stikstof, toegepast op habitattypen en Natura 2000-gebieden. Alterra-rapport 1654. Alterra, Wageningen, Netherlands.

Denmark – E. Buchwald, Ministry of Environment

In Denmark two national regulations are relevant for assessing plans and projects regarding air pollution in relation to Article 6.3 of the Habitats Directive. One is a general regulation requiring appropriate assessment of all plans and projects which might significantly affect a Natura 2000 site. The other is a regulation dealing with livestock farms. Livestock production can only be established or enlarged/changed if the local authority grants permission. Permission may only be granted if the farm uses Best Available Technology for pollution control (BAT) and the authority ascertains that the plan/project will not adversely affect any Natura 2000 site.

In Denmark there is a list of critical loads of nitrogen deposition for all Natura 2000 habitat types (Annex I habitats) on the Ministry of Environment website. They are in line with the UN-ECE critical loads. This list together with assumptions and modelling of deposition to each type forms the basis for which selected habitats are included as vulnerable to ammonia in the regulations - see below. These habitats appear on existing maps of all nature areas in Denmark, and a map of them with buffer zones of 300 meter and 1000 meter is important in the evaluation of farm projects.

As part of the preparation for the upcoming Danish Natura 2000 plans, several studies have looked into the deposition of N compared to critical loads of the most vulnerable habitats. Nitrogen deposition in Denmark ranges from about 14 to about 25 kg N $ha^{-1}yr^{-1}$ modelled as a mean deposition in a 16 x 16 km grid. More detailed studies have revealed that many Natura 2000 sites have lower actual deposition than modelled, due to fewer farms and other local factors, whereas other sites have a higher deposition. Nevertheless, several habitats have problems with deposition exceeding the critical load in parts of Denmark.

The regulation on livestock farms includes many details including how to find out what are the thresholds in relation to adverse effects regarding ammonia, phosphorus and nitrate. Existing permissions to farms must be updated at least every 8-10 years in order to comply with the newest regulations and thresholds. In general, thresholds have become stricter over time, and there are plans to make them even stricter yet. Farms with three or fewer animals are not regulated.

The thresholds are set in a way that it can be assumed that no significant adverse effects on the integrity of Natura 2000 sites can be anticipated when keeping below them. In exceptional cases, the thresholds may not be strict enough, and in such cases, according to present legislation, the local authority may only permit the farm project on stricter conditions preventing adverse effects.

For ammonia, the following thresholds are listed in the regulation:

- Compared to BAT in 2005/2006 ammonia emissions must be 15 per cent lower in 2007, 20 per cent lower in 2008 and 25 per cent lower in 2009.
- Within 300 meters from habitat types vulnerable to ammonia/nitrogen deposition in Natura 2000 sites, new farms are not allowed, and emissions must not increase from existing farms.
- The vulnerable habitat types defined in the law include heath and dry grassland, raised bogs, nutrient poor lakes.
- Within 300 - 1000 meters from the vulnerable habitats in Natura 2000 sites, the allowable extra deposition of ammonia from a farm project is 0.7 kg N $ha^{-1}yr^{-1}$ if there are no other farms within one km, 0.5 kg N $ha^{-1}yr^{-1}$ if there is one other farm within one km and down to 0.3 kg N/ha if there are more than two other farms.

The Environmental Approval Act has recently been changed (2011) and now introduces a new concept for regulation of N-emissions in the neighbourhood of Natura 2000 sites. In the new regulation, the concept of buffer zones is abolished. The total allowable contribution from one livestock production unit is 0.7 kg N $ha^{-1}yr^{-1}$ if there are no other livestock farms within a certain distance of the applicant farm. If there is one other livestock farm within this distance, a total of 0.4 kg N $ha^{-1}yr^{-1}$ is allowed, and if there are two or more other livestock farms, a total of only 0.2 kg N $ha^{-1}yr^{-1}$ is allowed. The exclusion zone is proposed to be reduced from 300 m to 10 m. This implies that outside 10 m N-emissions are no longer capped and the establishment of new livestock farms is no longer prohibited, providing that the above requirements to total contribution are met. For phosphorus and nitrates all of Denmark has been mapped in relation to sensitive soils and sensitive Natura 2000 sites including marine sites. Depending on their location in Denmark farms must comply with thresholds for these issues also.

Germany - D. Bernotat, Federal Agency for Nature Conservation and Till Spranger, Federal Environment Agency

Nitrogen deposition in Natura 2000 sites is currently a high priority issue in Germany. In several court decisions regarding road projects the judges ruled that nitrogen deposition might lead to significant effects and therefore will likely affect the integrity of the site. Examples are the ruling of the Federal Court of Justice (BVerwG) on the Highway A 143 west bypass Halle (from January 17, 2007) and the highway A 44 Lichtenauer Hochland (from March 12, 2008). The Court also notes that there currently seem to be no generally accepted effect assessment standards, and that methods should be considered with regard to competence, impartiality and objectivity.

The Association of the German Länder´s nature conservation authorities (LANA) has therefore audited currently available approaches with a view to their possible applicability to the Appropriate Assessment.

Assessment of Nitrogen Deposition effects in German air pollution abatement law
The TA Luft (Technical Instruction Air), despite not being a law in legal terms, is used to directly implement source-related air pollution laws and regulations in Germany, e.g. for licensing newly built or extended air pollution sources. Section 4.8 states that "significant impediments" caused by nitrogen deposition due to new/extended sources have to be assessed - in practice in the ca. one km² surroundings of the source.

A consensus-oriented expert group mandated by the responsible Federal/Länder body (LAI) designed a methodology (which presently undergoes a two (three) year test phase mandated by the Conference of Federal and Länder Environment Ministers) which is based inter alia on critical loads: Total deposition (i.e. "background" deposition without the source plus the deposition diagnosed to be caused by the source) is compared to critical loads or a multiple (x) of critical loads.

The magnitude of the factor x (which characterises the "significance" of N deposition in the individual case) varies between one and three; it is determined by 1) the legal status of the area to be protected, and 2) the biochemical status (e.g. presence of N indicating species, pH, nitrate concentrations etc.) of the area to be protected.

For N sensitive protected areas (e.g. Natura 2000 sites), x = 1, i.e. for these areas, critical loads are used as the mandatory target value for total deposition. In addition, it is recommended to apply standard procedures within the nature conservancy law framework (see below).

The Länder have implemented the regulation in various ways, some as a standard procedure in present licensing cases, some only for ex-post analyses of cases where licenses have been issued.

Appropriate Assessment of Nitrogen Deposition effects regarding Article 6.3 HD in German nature conservation law

The nature conservation authorities (LANA) decided that the described approach, designed for licensing of air pollution sources, in its present form does not have sufficient explanatory power for the necessary assessment of nitrogen inputs into Natura 2000 sites in the context of Appropriate Assessments. The present procedure cannot meet the special requirements of the precautionary principle which is necessary for the protection of Natura 2000 sites under the Habitats Directive.

For Appropriate Assessments, the LANA recommends at the moment a guideline of the Brandenburg State Office for the Environment (Landesumweltamt Brandenburg, 2008). It also uses empirical critical loads for the assessment of nitrogen deposition in habitat types, but in a modified way: If critical loads of nitrogen are already exceeded - which happens in many parts of Germany - or will be reached by the project, the exceedance of the critical load would still be tolerable, if the additional load of the project is less than 10 per cent of the critical load.

There is an exception if a habitat or species is already in an unfavourable conservation status caused by nitrogen inputs. In this case an individual case by case decision is necessary, which in particular has to take into consideration whether the achievement of conservation objectives and the improvement of the situation may be at risk.

A revision of the concept in the future may particularly aim at further preventing a creeping deterioration due to cumulative effects of projects.

Furthermore, the concept could be improved by assessing the size of the affected area in relation to the total size of the Natura 2000 site. Moreover, it is suggested that this approach could be

integrated with the scientific standard / guideline of the Federal Agency for Nature Conservation dealing with permanent losses of habitat types in Natura 2000 sites (Lambrecht & Trautner, 2007, http://www.bfn.de/0316_ffhvp.html).

In addition, in 2009 a research project of Federal Highway Research Institute (BASt) was initiated to produce a guideline for the emissions of nitrogen along roads in the context of Appropriate Assessments.

References:

Landesumweltamt Brandenburg (2008) Vollzugshilfe zur Ermittlung erheblicher und irrelevanter Stoffeinträge in Natura 2000-Gebiete. Stand der Fortschreibung November 2008 (available at: http://www.mluv.brandenburg.de/cms/media.php/2338/vh2008e.pdf).

3.2 Working group report

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3.2.1 Conclusions and recommendations of group discussions

- It is recommended that a staged approach is applied to the impact assessment, including: i) a relevance screen, ii) test of likely significant effect, iii) appropriate assessment and iv) final decision. Modelling predictions should be compared against the relevant critical loads and critical levels (applied at the Natura 2000 site scale).
- It is recommended that assessment needs to consider 'in combination' effects. Therefore, the plan/project should be considered both alone and in combination with other plans and projects, as well as in the context of existing ambient air quality (and prevailing environmental conditions). An integrated management/assessment plan (at, for example, the province/region scale) could assist with this.
- It is recommended that all relevant EU Directives and national regulations should be considered during the assessment, to ensure the requirements of the IPPC Directive, Nitrates Directive, Water Framework Directive, EIA Directive etc, are considered alongside those of the Habitats Directive, allowing an integrated approach to be applied.
- It was concluded that ongoing problematic issues include whether consideration of the spatial scale of impact, survey data, and/or application of *de minimis* criteria, in respect to the plan or project contribution, are appropriate. A Member State might choose to apply a *de minimis* criterion to allow new plans or projects in situations where the critical load/level is already exceeded. In the absence of any sound ecological justification for such a position, this would have to be a policy decision.

- It was concluded that further work is required on the development and dissemination of a best practice approach, including the involvement of a larger number of Member States.

3.2.2 Introduction and discussion objectives

The Habitats Directive (Article 6.3) requires that all 'plans and projects' be assessed in relation to possible impacts on Natura 2000 sites and that, except where there are reasons of overriding public interest, the plans and projects can only be approved where they are shown to have no adverse effect on the integrity of a site. At present, however, there is no common approach across Europe for assessing the effects of reactive nitrogen concentrations and deposition resulting from such plans and projects. While some countries are pro-active in the mitigation of nitrogen emissions through various legislation, other countries in the EU are yet to develop guidance and mechanisms for dealing with the effects of nitrogen emitting sources.

The objectives for Working Group 1 were as follows:

1. To compare impact assessment and decision making approaches across Member States in determining the nitrogen deposition impacts of plans and projects in the context of the obligations under Habitats Directive Article 6.3.

2. To discuss what could be considered as a best practice approach to assessing nitrogen impacts on the Natura 2000 network.

3. To identify any particular problems associated with the implementation of the Habitats Directive in different countries.

In addition to these objectives, a number of specific questions were raised in the background document (Bealey *et al.*, this volume), which the group prioritised into five key questions and addressed under discussion of the second objective (above):

1. What is a likely significant effect and how is it defined?

2. What is a significant contribution from a 'plan or project' in relation to either a habitat's critical load or level?

3. If the background nitrogen deposition already exceeds the critical load, what rules should apply for new plans or projects? How much more additional nitrogen is seen as having an adverse effect on the integrity of a site?

4. How should sources in-combination be handled in the process?

5. What would constitute an outline of 'best practice' in conducting such an assessment?

Objective 1: A comparison of assessment approaches

The comparison of impact assessment approaches identified a number of differences but also some common factors (see Table 3.2). For example, all of the countries represented at the workshop use critical loads and/or critical levels in their impact assessments of nitrogen effects on Natura 2000 sites, either directly (UK, Germany & Netherlands) or indirectly as in Denmark where critical loads have been used to choose the most vulnerable habitats and in setting the deposition thresholds for new and existing farms.

Three of the countries compared employ assessment thresholds or allow small increases (*de minimis*) in nitrogen deposition in situations where the critical load is already exceeded (UK, Germany and partly Denmark). In the Netherlands, however, assessments currently work on the principle of allowing no net increase in nitrogen deposition (nitrogen additions are only acceptable where complementary actions are secured to reduce existing nitrogen deposition levels). This arose from a court case in the Netherlands that ruled that new and existing farms cannot be permitted when the critical load for a habitat is already exceeded (200802600/1/R2 and 200807857/1/R2). In Denmark a different approach is applied for agricultural sources of nitrogen, where increments of nitrogen deposition are allowed based on the number of farms within a 300 m -1 km radius from a sensitive habitat (see Section 3.3). In some cases, municipalities allow no net increase to Natura 2000 sites.

All countries use a staged process for handling plans or projects with an initial screening of plans or projects to assess the likelihood of a significant effect. In addition, at the site level, Annex 1 habitat features have been assessed for their sensitivity to nitrogen deposition and then allocated a relevant critical load. This assists the assessment process at the screening and the appropriate assessment level where the most sensitive features can be identified and any critical load exceedances evaluated.

Table 3.2: Comparison of nitrogen impact assessments in different Member States in 2009

	Dk	De	NL	UK
Staged assessment approach	✔	✔	✔	✔
Critical loads/levels applied	✔	✔	✔	✔
Buffer Zones Around Natura 2000 sites	✔	x	x	x
Multi-source, in-combination tests	✔	✔	✔	✔
Long Range/Short Range N deposition considered	✔	✔	✔	✔
Status/condition of Natura 2000 site considered	✔	✔	✔	✔
Size of impact (area) considered	x	✔	✔	(✔)
De-minimis thresholds applied/allowable increments	✔	✔	✔	✔

Further details on the approaches applied in individual countries are provided for the UK (see Appendix 3.1 and Russell *et al.*, this volume), Netherlands (see Appendix 3.1), Denmark (see Appendix 3.1 and Bjerregaard *et al.*, this volume) and Germany (see see Appendix 3.1 and Uhl, this volume). It is worth noting that the approaches adopted by some of the Member States have been significantly influenced by court decisions, while each approach also clearly reflects national policy and national goals.

Objective 2: Best practice approach for assessing N impacts to Natura 2000 sites.

A Staged Approach

Based on the countries' assessment approaches and the requirements of Article 6.3, the working group developed an assessment framework consisting of a number of discrete stages:

- Stage 1 – Relevance screen
- Stage 2 – Likely significant effect test
- Stage 3 – Appropriate Assessment
- Stage 4 – Final decision (i.e. can it be ascertained that the plan/project will not adversely affect the integrity of the site?).

Stage 1 is an initial screen to filter out those permissions that by virtue of their nature or location could not affect the interest features of the Natura 2000 site. Stage 2 and 3 mirror the terms set out in Article 6.3 of the Habitats Directive. The assessment includes an iterative process, as consideration should be given to potential conditions, restrictions or management measures that may be applied to the plan or project to enable a conclusion of no adverse effect on site integrity to be reached. Each stage is described in more detail in the guidance note in Appendix 3.2. The following sections report the discussions on some of the key issues by the working group:

Use of critical loads and levels

Critical loads and levels are useful as an environmental limit to be used in impact assessments. They can be applied at the site level, although they were originally developed for use in national/international risk assessments. Further refinement to make them more applicable to a specific habitat of an individual Natura 2000 site and its conservation objectives is, however, recommended. Such research has been carried out by many countries and other Member States may benefit from their approaches. These include:

- Research in the UK on defining 'site relevant' critical loads for the Natura 2000 network (Bealey *et al*., 2007).
- The Netherlands has defined critical loads for every Annex I habitat (Van Dobben & Van Hinsberg, 2008).
- In Denmark, critical loads have been allocated to all Annex I habitats. For precautionary reasons, the critical loads for Annex I habitats are normally placed at the lower end of the range. A more exact critical load may be assessed by collecting all existing information on N-sensitive plant species, mosses, lichens, soil, aerial photos etc., and, if necessary, field surveys. http://www.skovognatur.dk/NR/rdonlyres/78C70731-71A2-40B6-B611-2F1340CB922A/14951/Ammoniakmanual02122005.pdf
- Research in Germany on appropriate assessments has considered soil and vegetation at the site level to give expert judgement about more exact, site-specific values within the span of the empirical critical load values. Another approach that has been used in Germany

is the application of model outcomes (BERN vegetation model coupled to DECOMP biogeochemical model by ÖkoData).
- On-going developments carried out by the Coordination Centre for Effects (CCE) of the Convention of on Long-range Transboundary Air Pollution (LRTAP) (http://www.mnp.nl/en/themasites/cce/index.html)

Determining a 'Likely Significant Effect'
A 'likely significant effect' can be described as any effect that may reasonably be predicted as a consequence of a plan or project that may affect the conservation objectives of the features for which a site was designated. Here it should be underlined that the conservation objectives may require some improvement in state/quality, not just preservation of the current state/quality.

The ECJ Waddense ruling (http://curia.eu.int/en/content/juris/index_form.htm case number C-172/02'), said that where a plan or project not directly connected with or necessary to the management of a site is likely to undermine the site's conservation objectives, it must be considered likely to have a significant effect on that site.

Where an exceedance already occurs
Where local background levels of nitrogen are already in exceedance of the critical loads/levels, a policy decision may be required on how to interpret this. On one hand it can be argued that any further increase in nitrogen deposition would give rise to a greater risk to the site or worsen effects, and therefore a conclusion of 'no adverse effect' on site integrity cannot be reached (see Chapter 5). On the other hand, given the uncertainty in model predictions and the absence of critical loads specifically evaluated for many Annex I habitats, some countries argue that allowing a certain 'degree of tolerance' is acceptable (i.e the *de minimis* principle). It should be noted that critical loads have already been set with habitat management practices in mind so management options, such as grazing or mowing, should not be seen as a mitigation solution for critical load exceedance (R. Bobbink pers. comm.).

The In-Combination Test
Article 6.3 of the Habitat Directive specifies that plans or projects should be considered individually and in combination with other plans and projects. Article 6(3) does not explicitly define which other plans and projects are within the scope of the "in combination" provision. However, in the Natura 2000 guidance notes (European Commission, 2000a) the underlying intention of this combination provision is to take account of cumulative impacts, and these will often only occur over time. In that context, one can consider plans or projects which are completed; approved but uncompleted; or not yet proposed.

Many existing sources are often already part of the background deposition or concentration that are mapped by Member States. However, there are occasions where recent permitted sources are not taken into account in the background, and there are sources that may escape the modelling or mapping process.

Integrated Management Plans
The use of an 'integrated management plan' (IMP), an approach taken with many other activities affecting Natura 2000 sites, could represent an effective way to achieve a full consideration of in combination and cumulative effects. The plan, at a wider geographical scale (regional or by province), would integrate projects over time and space, and allow detailed consideration of cumulative effects.

Integrated management plans (as e.g. under art. 6(1) of Directive 92/43/EEC) offer the opportunity to unify all known, current and future projects in a given territory under one scheme. This means that instead of following Article 6(3) and 6(4) procedures for each individual site and project, one plans ahead (10 years is a common time frame) and conducts Article 6(3) and 6(4) procedures only once for the entire planning area and all its known projects. Experience with sites where economic activities overlap with nature protection shows that IMPs are best devised employing a bottom up approach. This means fixed goals for each indivisible sub-unit, contributing to overall targets.

The working group determined that IMPs have advantages and disadvantages:

Advantages:
- They bring planning and legal certainty.
- IMPs offer the possibility to take so called accompanying preventive measures, which are aimed at improving the overall conservation status of a managed site, thereby reducing the likelihood of significant effects and critical load exceedances.
- In principal, IMPs should also enable the integration of many forms of nitrogen, as IMPs consider a provincial/regional area in its entirety and not just individual Natura 2000 sites.

Disadvantages:
- Spontaneous projects cannot be integrated into the plan, once it has been approved. They still need individual Article 6(3) and 6(4) procedures.
- The relevant sites often transcend federal and national borders. There is yet no successful example of a trans-border IMP, however this may change in the future.
- Where critical loads are exceeded due to long-range transboundary air pollution, an IMP cannot be handled at a national scale.

An integrated approach would also ensure that the impact assessment seeks to consider and satisfy all relevant EU Directives and national regulations, e.g. the EIA-Directive, the IPPC-Directive, the Habitat Directive, the Bird Directive, the Water Frame Directive, the Nitrate Directive, and the SEA Directive (See Section 7.1, 7.2).

Objective 3 – implementation

It was recognised that the working group included representatives from a limited number of Member States and therefore wider consultation on this best practice approach is required. Further work is also needed to establish a mechanism for disseminating the approach. Finally, it was acknowledged that there are a number of issues raised that require further detailed discussion and resolution:

- Emissions to air may have effects over both long and short ranges. How to consider long range effects when assessing a source's impacts requires further consideration. The spatial variability of ammonia and its effects are complex, and understanding the relation between Natura 2000 site location and deposition is important. For example, while sites situated near an agriculture source may be vulnerable to dry deposition of ammonia, Natura 2000 sites in upland areas away from the source may be subject to wet deposition of ammonium ions from it.
- The extent of habitats or species population affected - for some features any area of impact could be judged as unacceptable, however it is perceivable that in other cases very small scale impacts may not be considered to affect the structure and function of the site. Site specific judgement may be influenced by extent of damage and extent of site. Some examples of this are provided in European Commission (2000b).

- Short term impacts - in the majority of cases, standards or benchmarks for the protection of vegetation or ecosystems are based on long-term exposures (annual means). However, short-term exposure to high concentrations can sometimes be significant (i.e. to lichens).
- How to assess cumulative effects, especially from a large number of relatively small plans or projects, and to consider effects that may be additive or synergistic.
- Use of site survey information and the role of monitoring. Monitoring has long been seen as a best practice in EIA, and is a requirement of the recently adopted directive on Strategic Environmental Assessment, but how does it fit with Habitats Directive (Article 6(3)) requirements?

References and further reading

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http://ec.europa.eu/environment/nature/natura2000/management/docs/art6/provision_of_art6_en.pdf).

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Appendix 3.2: Proposed guidance for determining the nitrogen deposition impacts of plans and projects, in the context of Article 6(3) of the Habitats Directive

This guidance provides a framework to assist with making robust, transparent and consistent decisions that meet the requirements of the Article 6(3) of the Habitats Directive. The impact assessment approach consists of four distinct stages (see Figures 3.1).

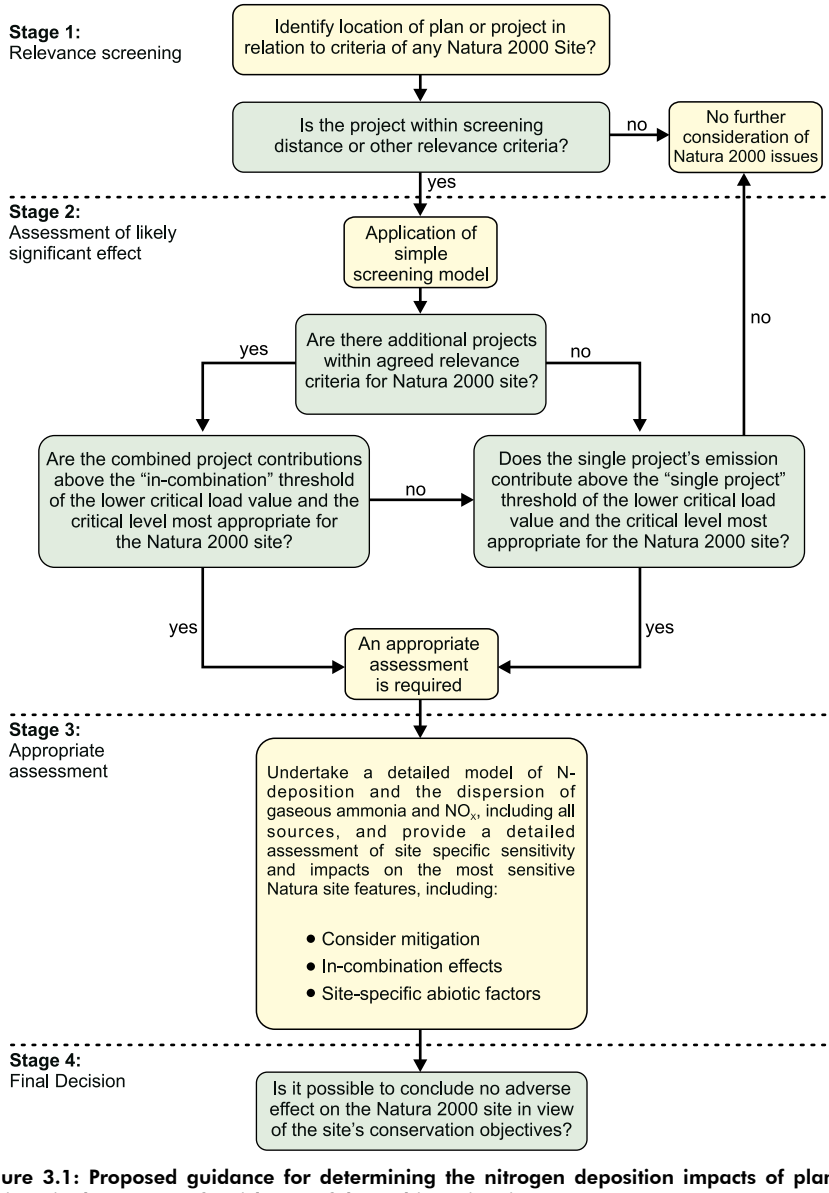

Figure 3.1: Proposed guidance for determining the nitrogen deposition impacts of plans and projects in the context of Article 6.3 of the Habitat Directive

3.3 Impact assessment and regulation of N-emissions from livestock farms in Denmark

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Summary

- In Denmark, screening according to the Environmental Impact Assessment (EIA) directive of all applications for establishment/extension/change of livestock production units, including impact assessment of N-deposition to Special Area of Conservation (SACs), was initiated in 2001.
- In 2007 this method was replaced by a new Environmental Approval Act for livestock farms.
- This law contains both specific regulation in buffer zones around selected habitats as well as general reduction demands and Best Available Technology (BAT) requirements.
- The municipalities have to carry out additional Natura 2000 impact assessment.

3.3.1 Introduction

In Denmark screening of all applications for establishment/extension/change of livestock production units according to the EIA directive was initiated in 2001. The screening included impact assessment of N-deposition to SACs and was carried out by the regional authorities (14 counties). By the end of 2006 the regional authorities were closed down as part of a municipal reform, and the local authorities were enlarged to comprise 98 municipalities.

Simultaneously, a new Environmental Approval Act for livestock farms was passed in 2007 (Ministry of Environment, 2007), replacing the former EIA-screening of livestock farms. The new law is a joint implementation of six EU directives: the EIA-Directive, the IPPC-Directive, the Habitat Directive, the Bird Directive, the Water Frame Directive, and the Nitrate Directive .

The Environmental Approval Act was amended in March 2009 to establish more exact criteria for impact assessment of Natura 2000 habitats and Annex IV species. Furthermore, a new concept for regulation of N-emissions in the neighbourhood of SACs has recently been introduced.

3.3.2 Screening of livestock farms according to the EIA Directive (2001-2006)

The EIA-screening carried out by the regional authorities from 2001-2006 included all livestock categories including pigs, poultry, dairy, mink and other farm animals. The impact assessment of N deposition comprised a staged approach, which might have been slightly different from county to county. The example below is from the former County of Aarhus:

a. Initial screening.

The increase in N-emissions from the farm unit in question is used to determine a modelled “0.1 kg consequence radius” which is the maximal distance from the farm with a risk of a >0.1 kg N $ha^{-1}yr^{-1}$ increase in deposition (ΔD). Further assessment was applied if Natura 2000 habitats or other very N-sensitive habitats were situated within this distance. A less restrictive ‘0.5 kg consequence radius’ was used to identify non-Natura 2000 nitrogen sensitive habitats for further assessment.

b. Actual assessment

1. The total contribution from each farm unit and the increase in N-deposition (ΔD) was calculated in spreadsheet including data on increase in N-emission, distance, wind data,

modelled dispersion factors and estimated surface roughness for both the habitat and the adjacent area.

2. Total Deposition (TD) was calculated by adding ΔD to the local background deposition.

3. In Denmark the UNECE CL ranges have been "translated" to all habitats occurring in Denmark. For precautionary reasons, the CLs of Natura 2000 habitats are normally placed in the lower end of the range. A more exact critical load may be estimated by collecting all existing information on N-sensitive plant species, mosses, lichens, soil, aerial photos etc., and, if necessary, field investigations.

c. Field sources (area sources)

The contribution of N-deposition from application of manure to neighbouring fields was calculated in a separate spreadsheet and added to the contribution from the point source (ΔD) and is assessed jointly according to the criteria above.

d. Decision step 1: Increment significant by itself

A $\Delta D \geq 0.5$ kg N $ha^{-1}yr^{-1}$ was considered to be a significant contribution to most N-sensitive habitats, whereas one kg N $ha^{-1}yr^{-1}$ or 10 per cent of background was used for less N-sensitive habitats. These thresholds were applied even if the critical load was not exceeded as moving towards the exceedance of the critical load was generally not accepted. This practise was to be seen both as some kind of "quota" where one farm is not allowed to use up all of the possibilities to increase production in a local area, and as a precautionary principle (which can be shown to be in accordance with the critical level approach, which has not been included in the Danish assessment so far).

e. Decision step two - In-combination effects

Contribution of ammonia from other nearby livestock farms was not calculated independently, but was considered to be included in the modelled background deposition.

1) If TD>CL: Any increment is considered to be significant to Natura 2000 habitats and to very N-sensitive non-Natura 2000 habitats. This practice was established after decisions by the Nature Protection Board of Appeal, where increments down to 0.09 kg N $ha^{-1}yr^{-1}$ to Natura 2000 habitats were judged to be significant. In practice, ΔD would then have to be 0.00 kg N $ha^{-1}yr^{-1}$, as results were given with a precision of two decimal points.

2) If TD≤CL and $\Delta D < 0.5$ kg N $ha^{-1}yr^{-1}$ a decision of no significant effect was made.

3.3.3 Regulation and impact assessment of N-emission from livestock farms (2007-2011)

The practise of EIA-screening was substituted by the Environmental Approval Act in 2007. This law contains both general reduction demands and BAT requirements as well as specific regulation in buffer zones around selected habitats. According to the law, environmental permits have to be renewed every 8-10 years, and accordingly it is estimated that the main part of livestock production units will have been subject to environmental assessment and approval within 10 years. Applications are entered in an electronic, web-based application system, where increments of N-deposition to the nearest habitats area calculated automatically (for more details on the modelling in Denmark see NERI report in Appendix 6.1).

a. The use of buffer zones in impact assessment of N-deposition

A selection of N-sensitive habitats automatically generate buffer zones of 300 m and 1000 m (Figure 3.2). These include most - but not all - of the habitat types with a critical load ≤10 kg N

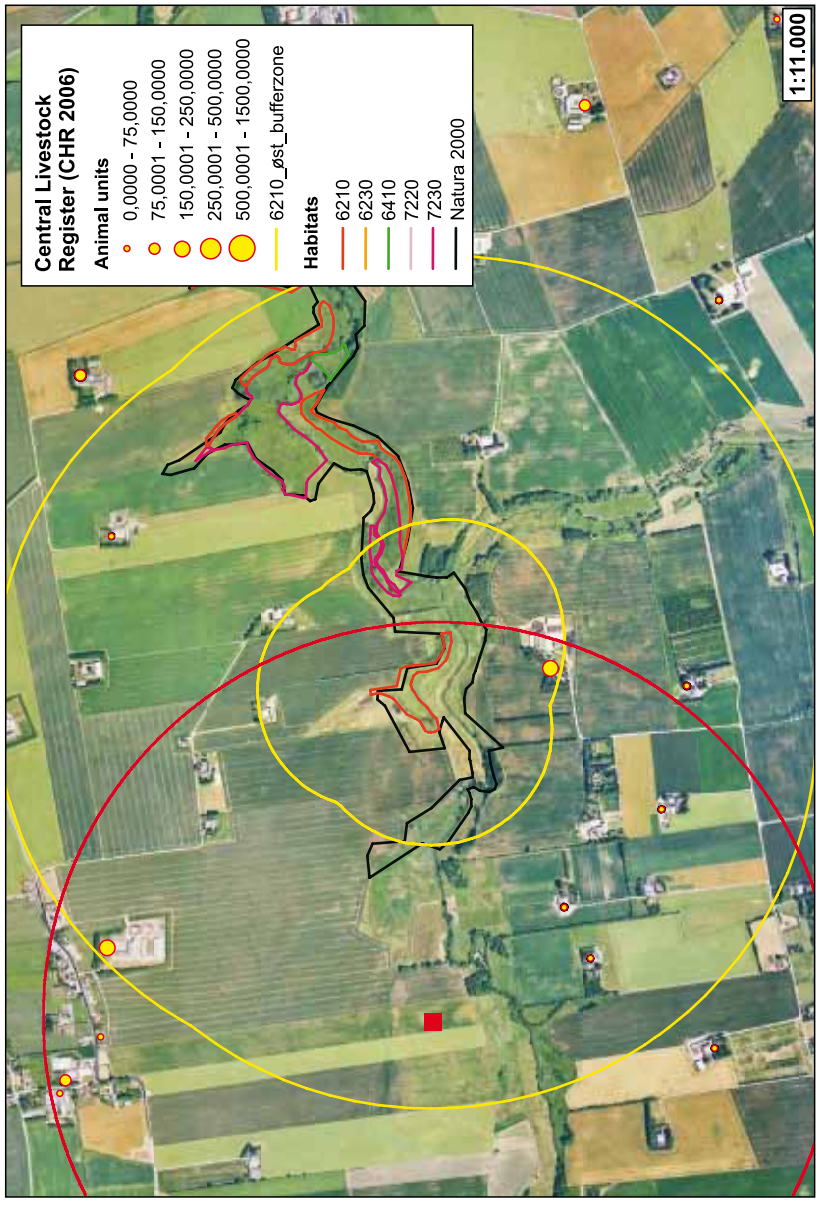

Figure 3.2: In this example, the applicant farm (red square) is situated within the 1000 m buffer zone (yellow line) around a calcareous grassland in Natura 2000 (but outside the 300 m buffer zone). As two other livestock farms with more than 75 animal units (AU) are situated <1000 m from the applicant farm (red line) and within the same buffer zone, the allowable increment is only 0.3 kg N $ha^{-1}yr^{-1}$.

$ha^{-1}yr^{-1}$ (low end of range): Raised bogs, oligotrophic mineral poor waters (code 3110), heaths > 2500 m^2, dry grassland > 2500 m^2, hard oligo-mesotrophic waters > 100 m^2 and dystrophic waters > 100 m^2 (codes 3140 and 3160) are designated as habitats sensitive to ammonia. Outside SAC only heaths > 10 ha and dry grassland > 2½ ha are considered. Within the buffer zones applications are met with special restrictions.

- 1) The 300 m buffer zone is an exclusion zone where establishment of new livestock farms with > 15 AU (animal units) is prohibited (for mink farms the threshold is three AU). Likewise increased emissions from existing point sources (housing and storage) is prohibited.
- 2) Within the 1000 m buffer zone the allowable increment to the nearest habitat is 0.7 kg N $ha^{-1}yr^{-1}$ if there are no other farms with > 75 AU within a distance of 1000 m from the applicant farm (Table 3.3). In-combination effects are handled by setting a limit of 0.5 kg N $ha^{-1}yr^{-1}$ if there is one other farm with > 75 AU within the buffer zone and <1000 m from the applicant farm, and only 0.3 kg N $ha^{-1}yr^{-1}$ if there are two or more other farms within this area.

b. Additional Natura 2000 impact assessment

Local authorities are responsible for preventing adverse effects to all designated habitats and species of a SAC as well as any habitat of Annex IV species. In each case, the municipality has to make an assessment and decide:

1. If buffer zone-regulations (like above) have to be employed to other sensitive habitat types than the ones mentioned above, such as forest habitat types, quaking bogs, oligotrophic/calcareous fens/meadows etc. – or occurrences below the size criteria mentioned above (3a).

2. If very large sources beyond the 1000 m buffer zone have to be regulated by the threshold values mentioned above.

3. If the general threshold values are strict enough to prevent adverse effects to designated Natura 2000 habitats or habitats with Annex IV species - otherwise the municipality is required to refuse the project.

Administration differs quite a lot among the 98 different municipalities, and there is not yet any generally approved approach for the impact assessment. It is unclear when effects after application of the general threshold values are so adverse that a project must be refused. In at least one case, a municipality has considered an increase of 0.1 kg N $ha^{-1}yr^{-1}$ to be a significant increment to a species rich alkaline fen in combination with the existing load above the critical load, and reason enough to refuse approval of the project. Future decisions from the Environmental Board of Appeal are expected to bring more clarity to setting the level of protection, but so far too few decisions on this matter have been made.

c. Impact assessment to non-Natura 2000 habitats

According to the Environmental Approval Act, heaths > 10 ha, grassland > 2½ ha, all raised bogs and oligotrophic waters (only Annex I habitat type 3110) outside Natura 2000 sites also automatically generate 300 m and 1000 m buffer zones. These are subject to the same thresholds as for Natura 2000 habitats (see a).

In addition, the local authorities are required to prevent adverse effects to N-sensitive habitats of Annex IV species (in Denmark amphibians and sand lizards are most relevant in this context)

by applying the buffer zone regulation. Likewise adverse effects to all protected nature areas comprised by the Danish Nature Protection Act have to be prevented. These include heaths, moors, dry grassland, meadows, fens, salt marches, bogs > 2500 m^2 and all lakes > 100 m^2.

As for the additional Natura 2000 impact assessment, administration among the municipalities differ quite a lot and is awaiting decisions from the Environmental Board of Appeal. A recent decision, though, has attached importance to 1) exceedance of the lower end of the critical load range of a species rich alkaline fen, and 2) an overall unfavourable conservation status of alkaline fens in Denmark according to the Article 17 reporting. In this specific case, increased deposition was > 0.8 kg N $ha^{-1}yr^{-1}$.

d. Field sources (area sources)

All fields belonging to a livestock production unit are contained in the environmental approval, and the municipality can decide on conditions for application of manure to the fields. The municipality can deny application of manure to specific fields situated close to sensitive habitats. There is no common procedure for conducting impact assessment of N-deposition from application of manure to fields. In some cases municipalities apply conditions in the approval like slurry injection of all applied manure or manure-free buffer zones bordering Natura 2000 sites.

General rules imply an obligation to inject slurry on grass fields and bare soil within the 1000 m buffer zone. By 2011 this obligation is due to be extended to apply outside the buffer zones as well.

In Denmark surface broadcasting of slurry manure has long been prohibited, instead band spreading (the most widespread method) and slurry injection is used. Manure has to be incorporated into bare soil within six hours to reduce N-emission, but since a very large part of the manure is spread on growing fields of winter wheat, which is by far the most common crop in Denmark, the main part is neither injected nor incorporated. Consequently a prominent spring peak in N-emissions from manure spreading is observed.

e. General N-emission reduction requirements

Apart from the specific impact assessment of livestock production units in the neighbourhood of sensitive habitats, all applicant farm projects with > 75 AU have to comply with general N-emission reduction demands. This general approach complies with the demands of the NEC Directive to reduce national emission, as a large number of farm units are not situated close to SACs and thus not covered by the specific regulation mentioned above.

Only a relatively small part of the total N-emission from a livestock farm is deposited close to the source, most is transformed into other nitrogen compounds and transported over longer distances. As such it contributes significantly to the general background deposition, and it is clear that in many cases it is not possible to eliminate critical load exceedances without considerable reductions of the background.

Weighed up against the environmentally most efficient housing system in 2005/06, ammonia emission from an applicant farm project with >75 AU has to be reduced by 25 per cent (this has been increased from 15 per cent in 2007 to 20 per cent in 2008). The environmentally most efficient housing system in 2005/06 is defined in an executive order for different types of livestock.

f. BAT requirements

Another means to reduce the background is the general demand for use of BAT (Best Available Technology), and all applications (projects >75 AU) must include an account for the use of BAT. This is seen as a very important – perhaps the most important - means of reducing the background

Table 3.3: Past and present regulation of livestock farms in Denmark

Habitat types	2009 level of protection	level of protection according to new regulation 2011
Ammonia sensitive Natura 2000 habitats, covered by present regulation	Max. increment in deposition of 0.3-0.7 kg N $ha^{-1}yr^{-1}$ in bufferzone II No increment of emission in bufferzone I.	Max. total deposition depending on number of animal units nearby*: 0.2 kg N $ha^{-1}yr^{-1}$ by > 1 animal unit 0.4 kg N $ha^{-1}yr^{-1}$ by > 1 animal units 0.7 kg N $ha^{-1}yr^{-1}$ by 0 animal units
Ammonia sensitive Natura 2000 habitats, not covered by present regulation	No present regulation, but the municipality has to make a specific assessment.	
Ammonia sensitive habitats outside SAC, covered by present regulation	Max. increment in deposition of 0.3-0.7 kg N $ha^{-1}yr^{-1}$ in bufferzone II No increment of emission in bufferzone I.	Max. total deposition 1.0 kg N $ha^{-1}yr^{-1}$.
Ammonia sensitive habitats and forests outside SAC, not covered by present regulation	No present regulation, but the municipality can make a specific assessment.	Max. increment in deposition of 1.0 kg N $ha^{-1}yr^{-1}$.

**** Defined as (cumulative model): no. units > 15 Animal Unit (AU) within 200 m + no. units > 45 AU within 200-300 m + no. units > 75 AU within 300-500 m + no. units > 150 AU within 500-1000 m + no. units > 500 AU contributing with > 0.3 kg N/ha beyond 1000 m***

deposition in Denmark and in several decisions from the Environmental Board of Appeal it is emphasized that the local authorities have to ensure that environmental permits contain conditions for maximum N-emissions equivalent to the level if BAT was used.

The applicant can choose between different BAT systems approved by the Environmental Protection Agency (EPA) including techniques for both dairy, pig and poultry farms. Among the approved techniques are adding of sulphuric acid to the slurry, air scrubbers with sulphuric acid, drying of poultry manure, systems with frequent cleaning out, cooling of slurry etc. Presently more BAT techniques are awaiting approval (published at the EPA home page www.mst.dk/Virksomhed_og_myndighed/Landbrug/BAT-blade.htm).

3.3.4 New regulation of livestock farms (2011)

The Environmental Approval Act has recently been changed and now introduces a new concept for regulation of N-emissions in the neighbourhood of Natura 2000 sites. Instead of a threshold for the increment, a threshold is applied for the total contribution to nearby sensitive Natura 2000 habitats from each production unit.

The total allowable contribution from one livestock production unit is 0.7 kg N $ha^{-1}yr^{-1}$ if there are no other livestock farms within a certain distance of the applicant farm. If there is one other livestock farm within this distance, a total of 0.4 kg N $ha^{-1}yr^{-1}$ is allowed, and if there are two or more other livestock farms, a total of only 0.2 kg N $ha^{-1}yr^{-1}$ is allowed. This implies that existing emission may have to be reduced if the threshold requirements are not met today. But the regulation is only enforced if an existing farm unit applies for enlargement or change.

In the new regulation, the concept of buffer zones is abolished. The exclusion zone is proposed to be reduced from 300 m to 10 m. This implies that outside 10 m N-emissions are no longer capped and the establishment of new livestock farms is no longer prohibited, providing that the above requirements to total contribution are met. One reason for this altered regulation is the need for establishment of new cattle farms in order to comply with demands of extensive

management of Natura 2000 sites. Abolishment of the 1000 m buffer zone leads to future assessment of all projects regardless of distance to the Natura 2000 site.

In relation to Natura 2000 sites, the thresholds for total contributions are going to apply to all N-sensitive Annex I habitats, including forests, alkaline and oligotrophic fens, quaking bogs etc. Outside Natura 2000 sites a threshold of a total contribution of one kg N $ha^{-1}yr^{-1}$ is introduced as a substitution to the buffer zone regulation, but here it still only applies to the selected habitats mentioned above (see section 3.3.3.c). Additionally, a decree on a threshold of one kg N $ha^{-1}yr^{-1}$ increment to certain other sensitive habitats outside Natura 2000 has been passed.

Regarding the general N-emission reduction demands, the proposal implies an increase to 30 per cent by 2011, but by 2012 reduction demands should be replaced by standardised BAT conditions in all permits (cf. section 3.3.3.e).

3.3.5 Results and discussion

According to the present regulation in Denmark a ***significant contribution*** is interpreted as a contribution which can be expected to cause change in an ecosystem ***by itself*** in the long term**.** The threshold of 0.7 kg N $ha^{-1}yr^{-1}$ increment from a single source is based on the consideration, that it is unlikely that long term effects of a habitat from a contribution below this size can be proven. The National Environmental Research Institute has stated, that an increment of about 0.6 kg N $ha^{-1}yr^{-1}$ is found to be the best statistical estimate for the threshold under which the contribution calculated with the present models is statistically equal to zero, and where no effect can be demonstrated in 95 per cent of the cases. On the other hand, it cannot be rejected that an effect may occur on very sensitive habitats like Natura 2000 habitats, i.e. certain parameters of the ecosystem may be found to change after many years of exposure. But it is hard to imagine that even long-term effects from increments of about 0.1 kg N $ha^{-1}yr^{-1}$ can be proven (NERI, 2005).

When considering in-combination effects these are presently handled by considering ammonia contributions from other livestock farms situated near the applicant farm project. When one other farm is present, the allowable extra contribution is lowered to 0.4 kg N $ha^{-1}yr^{-1}$, and with two or more farms to 0.2 kg N $ha^{-1}yr^{-1}$. In this respect 0.2 kg N $ha^{-1}yr^{-1}$ can be viewed as a *de minimis* (trivial/inconsequential) contribution.

Impact assessment of sensitive Natura 2000 habitats and other regionally or locally important habitats normally include evaluation of *critical load exceedances*, typically by using the municipal mean background deposition.

Nevertheless, in an executive order it is emphasized, that municipalities must refuse a project if a case-by-case assessment brings doubt to whether the buffer zone conditions are in line with the obligation to provide good conservation status of a Natura 2000 habitat. No cases on this matter have yet been tried in the Environmental Board of Appeal, but municipal administration is subject to wide-ranging variation on this point.

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3.4 Assessing impacts of nitrogen emissions on Natura 2000 sites in Germany

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Summary

- Nitrogen deposition has been recognized as a major threat to the conservation goals of Natura 2000 sites in Germany and has to be considered carefully within impact assessments.
- Critical loads have been accepted as an appropriate measure of sensitivity of protected Annex I habitats in respect to nitrogen deposition.
- No court decision and no expert convention has yet been made on how to treat project/plan contributions of nitrogen deposition where background deposition already equals or exceeds the critical loads. One approach that has been adopted in a number of cases advocates a value of 10 per cent of the critical loads (LUA Brandenburg, 2008), but is explicitly not intended to be used if conservation status is unfavourable due to nitrogen impacts.
- So far criteria are missing on how to classify conservation status with respect to nitrogen impacts.
- We propose an evaluation scheme that we think should be conservative enough to be accepted by the court under a variety of conditions as we experience them in our practice. In the case of background deposition exceeding critical loads it proposes thresholds for adverse effects on the integrity of Natura 2000 based on several criteria. In addition to the amount of project/plan contribution - in relation to critical loads – we consider the size of the area affected (based on an expert judgement, or guidance of acceptable area loss), and the site specific quality of the affected habitat.
- Another controversial issue has been the determination of background deposition, which frequently did not include dry deposition accurately. We briefly discuss the data provided by the UBA to overcome those problems.

3.4.1 Introduction

Nitrogen deposition is an important issue in Germany in the context of impact assessments under the Habitats Directive. Our consultancies are involved in a number of impact assessments on Natura 2000 sites in Germany, mainly for traffic projects and we have also been entrusted with basic methodological work that is needed to provide evaluations conforming to the requirements stated

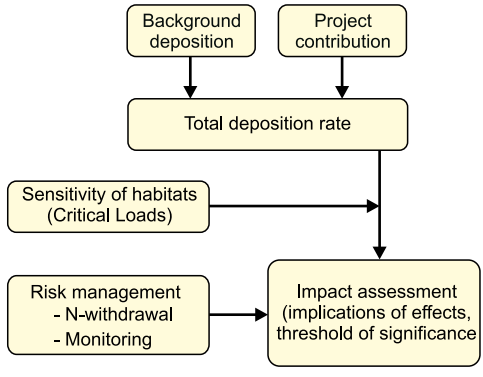

Figure 3.3: Scheme for the evaluation of effects caused by nitrogen deposition

by the courts. We are also watching very carefully the court decisions on this subject. Participation in this workshop allowed us to share the knowledge we had gained in this process with other European participants, and to widen our view on possible approaches as they can be found in the UK, Denmark and the Netherlands.

3.4.2 Aims and objectives

From this perspective we examine and provide knowledge on the following questions:

- How is nitrogen deposition considered within impact assessments under the Habitats Directive 6.3 in Germany?
- What are the controversial issues?
- How do we in practice try to resolve those issues?

A number of "key issues for discussion" as stated in the Bealey *et al.*, (this volume) proved to be an excellent guideline for answering those questions. They are restated below, followed by our remarks reflecting the situation in Germany.

3.4.3 Results and discussion

Impact assessments for Natura 2000 sites are conducted on a site specific basis, but the evaluation of effects caused by nitrogen deposition should essentially follow the scheme as depicted in Figure 3.3.

A quantitative estimate of nitrogen deposition is compared to the sensitivity of the habitat, usually expressed as critical loads. If adverse effects to the integrity of the site cannot be ruled out, the potential of mitigation and other measures will be explored. As stated by the court (Federal Administrative Court "Bundesverwaltungsgericht", judgement 17.01.07 – western motorway bypass of Halle), risk management is required to guarantee effectiveness of the measures and enduring absence of adverse effects.

How can background deposition be assessed reliably?
This question plays an important role in appropriate assessments. We do have good background values supplied by the UBA for the year 2004. They have a spatial resolution of one km² and are available online for nine different receptor types. So far, there are no prognostic values available, but work contracted by the UBA is in progress (research projects PAREST, MAPESI).

In the case of expansions more detailed background data are needed. Model calculations performed by B. Mohaupt-Jahr (UBA, pers. comm.) have shown for farms that it is appropriate to simply add locally dispersed contributions to the background data. The same holds true for roads.

What is a likely significant effect and how is it defined?
There are no general rules on how to handle a likely significant effect. As is stated in the background paper (Bealey et al., this volume) some federal states have adopted a regulatory proposal by the LAI (LAI, 2006) including a 4 kg N $ha^{-1}yr^{-1}$ threshold to screen for relevant installations. Since then, the threshold has been raised to 5 kg N $ha^{-1}yr^{-1}$, but at the same time a passage has been inserted restricting the scope of the paper: "it cannot be excluded, that further requirements might result e.g. from nature protection legislation". As far as assessments under the Habitats Directive are concerned, it has essentially been abandoned in favour of a 10 per cent rule: project contributions of less than 10 per cent of the critical loads may be considered as insignificant under certain conditions (see below). In our country the public dispute about assessing nitrogen deposition on SACs is often carried out in court. So if there are doubts if adverse effects on the integrity of the site cannot be ruled out, we have to assess the impacts of nitrogen deposition.

For significance, we consider effects that may in the long run compromise the ability of habitats as mentioned in the conservation objectives to stay at a favourable conservation status. As is stated by the European Commission, it would not be enough for a habitat to stay nominally in the same vegetation/habitat class, although there have been voices in favour of this interpretation.

What is a significant contribution from a project/plan in relation to either a habitats critical load or an emission target?
As long as a project's contribution does not lead to exceedance of critical loads, it is not considered to be significant in the sense of potentially exerting adverse effects on site integrity. If it does, its significance depends on the area affected. Indications on size have been given by court decisions: Thus exceedance of the critical load on 0.18 hectares of a *Molinia* meadow had been ruled as being significant in one court case. In the case of an Annex I - grassland, designated as priority Annex I habitat (*6120) with the characteristic species *Orchis morio*, even smaller areas were considered as potentially significant.

What if the background is already exceeded? How much more additional nitrogen is seen as having no adverse impact on the integrity of a site?
So far there has been - to our knowledge - no court decision for the case where critical loads are already exceeded by background deposition. The primary reason for this is that in the past background deposition had been determined without considering dry deposition, so background deposition was often underestimated. However, as described in Bealey *et al*., (this volume) by Till Spranger and Dirk Bernotat, there is a proposal by the State Office for the Environment Brandenburg (LUA Brandenburg, 2008) which can be applied to those cases.

Relevant thresholds have been set by the LUA Brandenburg to 10 per cent of the CL, in analogy to the German air pollution abatement law, where air concentrations of 30µg/m³ NO_x are accepted and an irrelevance value of 3 µg/m³ is stated (B. Hanisch, LUA Brandenburg, pers. comm.). Cases where the site has an unfavourable conservation status of C or less, and N-deposition might be a likely cause for degradation, are explicitly exempt. For these cases no threshold is indicated.

Unfortunately the mapping scheme used in Germany for collecting data on the SACs (in case of the "Grunddatenerfassung" - basic data collection) was not designed to appropriately detect effects of eutrophication. The conservation status thus does not necessarily reflect impairments or degradations caused by excessive nitrogen input (see also Whitfield *et al*., this volume). So in

essence one might conclude that there are no habitats at all with an unfavourable conservation status caused by nitrogen deposition. From a technical or scientific point of view however, one has to consider already existing damage in many cases, e.g. by subtle species displacements, which are recognised as a great threat to biodiversity, but are rather hard to detect. Furthermore, major changes in the ground vegetation of forests for example might not lead to an unfavourable conservation status, if the trees meet the standards of the guidance - even if their vitality has decreased (thus raising the amount of dead wood which, under a structural view, may be considered as contributing to favourable conservation status). Forests often are self stabilizing systems, in that it takes external impacts to make changes visible, that literally have been lying in the dark (as we say in German). Such impacts may become more likely though, e.g. nitrogen enriched foliage boosting insect damage, or faster growth making droughts induced by climate change more hazardous to tree health.

The first thing one can do is to rule out the possibility of existing adverse impacts caused by nitrogen deposition. For example nutrient-poor species are not always in the focus of conservation objectives (e.g. in the case of nutrient rich alder and willow forests 91E0), or there might be evidence that current management maintains habitats in a favourable conservation status. However, there will be cases where this is not appropriate. Currently, there is no way of neglecting the need for an irrelevance threshold (*de minimis*), otherwise we would have to assess the impact of one project on sites tens or more kilometres away from the road. (Once such a threshold is identified, one could imagine it to be implied in some other criterion, e.g. distance).

Another aspect must be emphasized: in analogy to the national regulations concerning air pollution abatement, the LUA Brandenburg considers a single receptor point as being relevant in terms of affected size. In the case of roads this may lead to rather random assessments: if a single 10 per cent-receptor point is hit, there is a significant effect, otherwise not. The overall effects of additive nitrogen loads however are rather low in intensity (at least in the case of traffic-borne emissions), but they often cover a large area.

In this situation we have proposed a method to refine that of the LUA Brandenburg. It starts out with 3 per cent of the critical load, often equal to about 0.3 kg N $ha^{-1}yr^{-1}$. This we consider to be clearly smaller than the amount of uncertainty that we are dealing with in respect to modelled background deposition or critical loads, and also very low in comparison to background deposition and fluctuations in the environment. In the case of forests it is in most cases less than 1 per cent of present background deposition. In the case of grasslands, for example, the chances are that fertilizer applications somewhere in the environment mask any conceivable effect of small deposition increments. On the other hand, speaking of impact assessments for roads, it is an amount that can be found several hundred meters away from the road, so it does have a true cumulative potential, and if very sensitive biotopes are concerned, it might make sense to consider such a threshold seriously. It is also not inappropriate, as there are analogies in the history of national air pollution abatement concerning human health, which also has a 3 per cent irrelevance threshold. Accumulation effects e.g. in the case of heavy metals have similar, although stricter thresholds of 2 per cent in the context of environmental impact assessments (UVPVwV, 1995).

Another level we consider in our approach is 5 per cent as an intermediate value. In the case of our 10 kg N $ha^{-1}yr^{-1}$ critical load example this would mean 0.5 kg, which could be rounded in the case of the critical loads (Bobbink *et al*., 2002) or the values the UBA database gives as background deposition. The upper level would be 10 per cent as proposed by the LUA Brandenburg (2008).

In our approach (FÖA, 2008) we have included two more criteria: the quality of the habitats concerned, in the sense of particular importance for the site and the size of the affected area.

Both criteria can, in concept, also be found in the convention on acceptable area losses, as proposed by Lambrecht & Trautner (2007), see Bealey *et al*., (this volume). For every German biotope covered by the Habitats Directive values of acceptable area losses have been determined. As a border criterion it was stated that the proportion of the habitat affected may be at most 1 per cent of the total area of the biotope within the protected site. (They gave supporting examples for 1 per cent being commonly considered as a minor proportion). They also recommended "qualitative-functional highlights", that should be considered regardless of their habitat type or affected size, with their function being of particular importance for the site (see Figure 3.4).

Although we think it might make sense to look at smaller intensities, we do not think that each small contribution has to be treated as a significant effect. As long as we do not overlook what is going on in the site concerned, i.e. in the case of small sites, the reasons for the 1 per cent proportion might support as well a spatial irrelevance threshold for small contributions below 10 per cent. If contributions are very small (less than 5 per cent of CL), it might be appropriate to consider effects dependant on the presence of functional highlights (otherwise the risk deems bearable). One thing, that is still unsatisfying about that approach however is significance being dependant on the size of the Natura 2000 site; particularly large sites may not be protected well enough by such a convention.

Another very appealing approach would be to extend the convention on partial function losses (Lambrecht & Trautner 2007, App. H). The rationale behind this is that the allowable affected area may be larger to a degree as the functions are only partially impaired. If one could tell, for example, what proportion of functions are impaired by an increment of one kg N $ha^{-1}yr^{-1}$, one could calculate the allowable affected area as:

Allowable affected area = Allowable area of loss * (100 / per cent of functional loss)

However, as was stated on this workshop (Nancy Dise, pers. Comm), dose-effect relations are currently not in sight. As discussed above, detecting the degree of existing effects is already hard to achieve.

Of course one could say that the effects of nitrogen deposition can hardly be as dramatic as a total loss. However, given the present degree of remaining uncertainty we think it is appropriate to apply the spatial threshold of total loss to impacts of 10 per cent or more, and at the same time to consider the effects and spatial extent of minor impacts.

How should in-combination (multi-source) effects be handled? For example, can de minimis values be set for the consideration of individual project contributions where the cumulative effect of many projects is being considered?
As far as the Habitats Directive is concerned, we have to consider all cumulative effects that we find. Since there is no centralized permit system in Germany though, some kind of threshold of significance is implicitly, and probably in most cases, rather unconsciously applied.

Apart from plans and projects in the sense of the Habitats Directive, which to our understanding have to be considered as contributing to the impact in question, we also have to deal with effects exerted by existing installations. We know that the LUA Brandenburg advocates a site register of all sources of impact, but this would require rigid concerted efforts.

Where are the relevant gaps in this scientific knowledge?
Commissioned by the Federal Highway Research Institute (BASt) we have started a research project comprising of methodological and scientific work. One goal is to continue the work

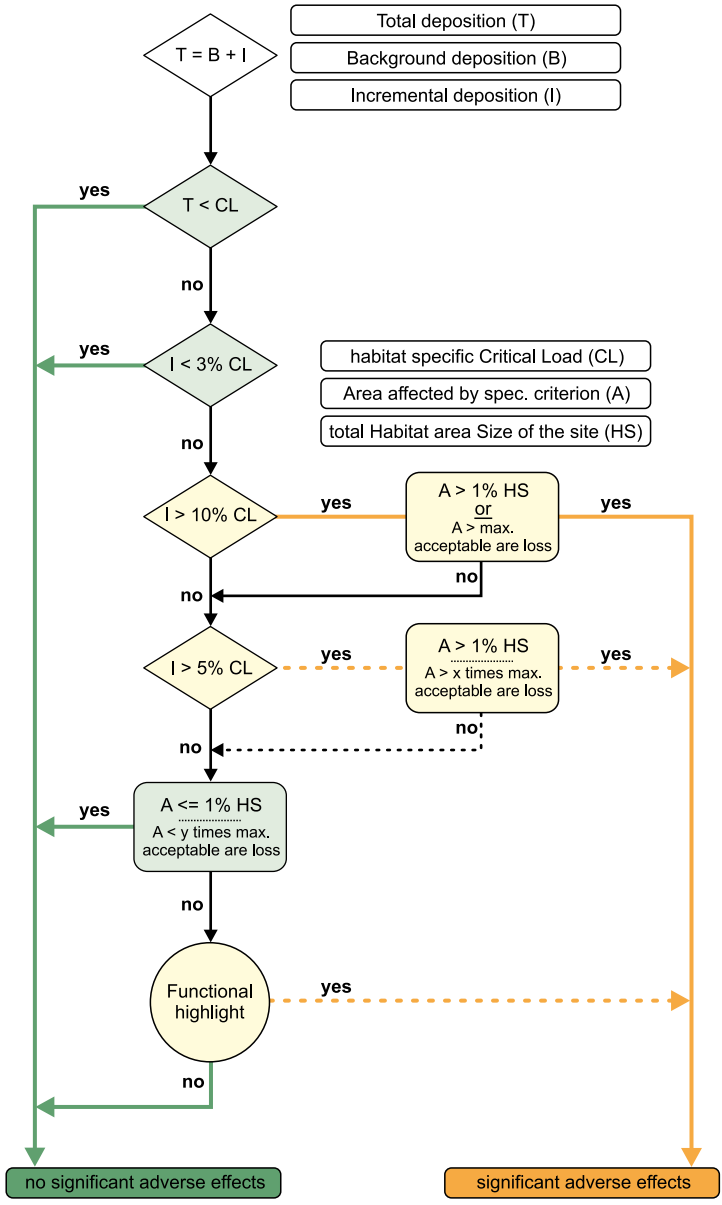

Figure 3.4: Proposed evaluation scheme in Germany

on the generally acceptable model as outlined above. Another goal is to differentiate critical loads within habitat classes where the Bern list only delivers general values for broad EUNIS classes. For example rather low critical loads are given for forest habitats in general. However, we are often confronted with Annex I habitat *91E0 forests, that are obviously quite eutrophic in nature. Accumulation effects are rather less likely due to undulations that certainly carry lots of nutrients with them, but should carry away project specific depositions. So the reasons

that led to critical loads for forest habitats might not apply to *91E0 forests. Similar problems arise in applying critical loads to water habitats.

Any 'higher resolution' (in terms of Annex I habitat types according to the Habitats Directive) of critical loads would be of great help. Of course we also would appreciate more guidance in the handling of ranges of critical loads. There are approaches to determine critical loads more specifically by using expert judgement (again by the LUA Brandenburg, 2007), as well as by modelling. The Dutch approach to systematically differentiate critical loads by a peer reviewed process (van Dobben & van Hinsberg, 2008) or in the UK by Bealey *et al*., (2007) may also be a model for further research on critical loads and its application in planning processes in Germany.

Are critical loads and levels fit for the purpose for site relevant assessments since they were originally developed for national risk assessments?
This is indeed a matter of controversy in Germany as well. There are approaches that say critical loads are not applicable in some cases or even in general. A discussion will be urgently needed in Germany to agree on accepted evaluation methods. Maybe a bundle of methods can be applied on a case by case basis. It would also be of great value if the scientific community could keep track of the discussion of the approaches that are used in impact assessments in practice.

From this point of view we had asked working group discussing theme:

- Under what circumstances are critical loads not appropriate to be applied as a measure for sensitivity of a habitat and hence as a measure for significance of possible effects?
- Under what circumstances can be stated, that present and past exceedances did not lead to an impairment of the habitat in question. Furthermore, under which circumstances may be concluded from the above, that critical loads do not have to be applied in an impact assessment (because there is no reason to believe that nitrogen-related damage will arise in the future)?

The answer, from a scientific point of view, was quite clear to both questions: critical loads are appropriate under most circumstances.

Another question might rather be subject to the formulation of appropriate conservation goals, but can still be quite obvious in the field. Therefore, would it make sense to switch to more realistic reference-states defining good conservation status if background deposition were very high (e.g. two or more times the critical loads)? In cases, where favourable conservation status of a site obviously is not linked to compliance with critical loads, or conservation objectives do not depend on reaching values below the critical load, one could imagine higher critical loads to be reasonable as a measure of sensitivity.

What rules should be applied for new plans or projects where background critical loads and levels are already exceeded? How should de mimimis be defined and cumulative (in combination) effects be handled in this instance?
As stated above, so far we do not know of any German court decision for the case, that critical loads had been exceeded already by background deposition. In accordance with indications given by the courts we tend to think that high background levels give reason to precautionary assessments. As long as incremental deposition caused by projects is overall rather small (as is the case with highways) it is appropriate just to look at the increment (take the view of the project), as described above. In other words, to our knowledge it is common sense in Germany, that high background depositions do not prescribe neither refusal of plans nor general permission. Under a technical point of view we think it could make sense to switch to more realistic reference states, if the project doesn't change the overall situation. A prerequisite would be that conservation goals do not demand

low nutrition status. Of course such a different reference state would still have to be guaranteed in the long run. In less severe cases we think it is reasonable to assume that conditions below critical loads can be attained within the next decades, so the reference state of the conservation objective should be the best one can conceive of.

What are some of the mitigation / compensatory measures that can, or are being applied across the EU? For example, mitigation of the effects with the use of tree shelter-belts have been used to capture N pollutant species. Are there other experiences of such landscape level mitigation / compensatory practices?
In the course of our work we have collected evidence (by empirical on-site tests), that grazing can be a way to remove additional nitrogen out of calcareous grassland (6510) and prairie grassland (6240), under certain conditions also out of dry heaths (4030), but not out of rocky habitats as 8230.

We are also confident that selective cutting of non-habitat tree species (in our cases mostly conifers) may result in forest habitats that can serve as compensatory habitats of the same type as the one that may become affected, and often in quite a short time.

In the past so called deposition protective plantings (mainly shrubs or smaller trees) close to highways had been accepted as mitigation measures, but they have relevant effects only in immediate vicinity.

3.4.4 Conclusions

- Within the common staged approach to impact assessments, nitrogen deposition plays an important role in Germany. Critical loads are widely accepted to be fit for the purpose of site relevant assessments, but there are no central regulations yet on how to evaluate projects contributions.
- Background values can be obtained by the UBA, and are combined with local models of dispersion. Further research projects (PAREST, MAPESI) will improve the estimates made available by the UBA for regional predictions as they are required for impact assessments.
- At present it is common sense that exceedance of the critical loads regularly means an adverse effect on the integrity of the site, if there had been no exceedance before.
- The question, how to deal with project contributions in the widely spread cases of existing exceedance of critical loads is less clear. In our work we have proposed an evaluation scheme that comprises and extends the less general scheme of the LUA Brandenburg. So far there has been no public discussion yet, and no court decision, on which approach will be accepted in the long run. This work remains to be finished, presumably within a research project of the BASt that will be conducted in the next two years, and maybe with the help of further court decisions.[1]

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1 During preparation of this volume there have indeed been court decisions confirming the assumptions expressed in this paper (BVerwG 9 B 28.09 10.11.09; BVerwG 9 A 5.08 14.04.10, http://bverwg.de/media/archive/8138.pdf)

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3.5 The approach taken by UK statutory agencies to assess nitrogen deposition impacts on Natura 2000 sites from 'plans and projects'.

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1 Natural England, UK
2 Countryside Council for Wales, UK
3 Environment Agency, UK
4 Scottish Environmental Protection Agency, UK
5 Northern Ireland Environment Agency, UK

Summary

- In the UK, a risk-based approach is used to assess the potential effects of atmospheric nitrogen deposition and concentrations on Natura 2000 sites arising from 'plans and projects', as required under Article 6(3) of the Habitats Directive.
- The assessment uses a staged approach and includes tests based on those in the Habitats Directive (Article 6(3) and 6(4)).
- Modelled pollutant concentration/deposition is compared with critical level(s) and critical load(s) allocated to each Natura 2000 site. Assessment thresholds are also applied, allowing a certain percentage of deposition above the critical load/concentration above the critical level
- Existing (background) pollutant concentration/deposition at the sites is also included in the assessment. A large number of Natura 2000 sites in the UK are predicted to be at risk from the harmful effects of atmospheric nitrogen (based on predicted critical load exceedance).
- The detailed assessment stage takes account of any additional site-specific information and considers the uncertainties within the assessment.
- As an example, the approach used for the assessment of the potential impacts of ammonia from (existing) intensive livestock units (pig/poultry farms) is described in more detail.

3.5.1 Introduction

This paper describes the approach applied in the UK, by the statutory environment agencies and conservation agencies, to assess potential impacts of reactive nitrogen concentration/deposition arising from 'plans and projects', as required under Article 6(3) of the Habitats Directive (92/43/EEC).

The Habitats Directive (Council Directive 92/43/EEC) has been transposed into UK legislation as The Conservation (Natural Habitats &c.) Regulations 1994 in England, Scotland and Wales, and as The Conservation (Natural Habitats &c.) Regulations (Northern Ireland) 1995 for Northern Ireland, all commonly referred to as the 'Habitats Regulations'. Since coming into force, there have been a number of amendments (e.g. 2004, 2007, 2010) to the Regulations that have been produced by the UK government both centrally and through the devolved institutions. The tests within the Habitats Regulations, in relation to 'plans and projects', closely mirror the tests in the Directive and require that permission can be granted only after it has been ascertained that the plan/project will have no adverse effect on the integrity of a Natura 2000 site; subject to certain provisions such as 'Overriding Public Interest'.

There are a large number of Natura 2000 sites in the UK, consisting of Special Areas of Conservation (SACs) and Special Protection Areas (SPAs). It is also Government policy to treat Ramsar sites (designated under the International Convention on Wetlands of International Importance) the same as SACs and SPAs with regards to the assessment of plans and projects. A large proportion of these sites are predicted to exceed their critical loads for nutrient nitrogen and/or acidity and are therefore considered to be at risk of significant harmful effects from air pollution.

In accordance with the Pollution, Prevention and Control (England and Wales) Regulations 2000, Pollution Prevention and Control (Scotland) Regulations 2000 and Pollution Prevention and Control Regulations (Northern Ireland) 2003, industrial installations must apply to the relevant pollution regulator (Environment Agency/Scottish Environmental Protection Agency (SEPA)/ Northern Ireland Environment Agency (NIEA)) for a permit to operate. In England and Wales these regulations recently became part of the Environmental Permitting Regulations 2007 (amended in 2010). Some categories of installations may also be regulated by local authorities. The application for a permit requires an assessment under the Habitats Regulations (i.e. it is a 'plan or project') which is undertaken by the competent authority, in this case the relevant pollution regulator. A planning application, e.g. an application to the local authority to build a new installation, will also require an assessment, in the this case the local authority usually the competent authority. The nature conservation agencies (Natural England/Countryside Council for Wales/Scottish Natural Heritage) are statutory consultees in both legislative processes (planning and pollution control). In Northern Ireland, the NIEA also has responsibility for nature conservation.

3.5.2 General assessment approach

To assess the potential nitrogen deposition impacts from a plan or project under the Habitats Regulations, a standard risk assessment procedure is applied. The risk assessment is carried out in a number of stages, which mirror the tests in Habitats Directive:

- Stage 1 – "Relevance screening" (distance based);
- Stage 2 – "Likely significant effect" test (modelling of process contribution to critical level/ load);
- Stage 3 – Appropriate Assessment - "No adverse effect" test (modelling of process contribution to critical level/load);
- Stage 4 – Determination.

The precise detail varies with the type of industrial/agricultural installation in question but the four stages are generally applied as follows: First, a distance screen (in many cases 10km from a Natura 2000 site) is applied to filter out any plans/projects that by virtue of their nature or location could not conceivably have an effect on the interest features of a Natura 2000 site. If it is deemed that the plan/project is not 'relevant' to any Natura 2000 sites the subsequent stages are not required. The second stage is a coarse screening stage, intended to identify those proposed plans and projects that require further assessment (an 'appropriate assessment'). A likely significant effect in this context is any effect that may reasonably be predicted as a consequence of a plan or project that may affect the conservation objectives of the features for which the site was designated, but excluding trivial or inconsequential effects. Potential impacts on all interest features of the Natura 2000 sites, identified at Stage 1, need to be assessed. The plan or project is assessed for 'likely significant effect' either alone or in combination with other plans or projects, and in the context of the prevailing environmental conditions. Prevailing environmental conditions include background/ diffuse pollution contributions to the site and the residual effects of plans and projects that have been completed/implemented.

If a likely significant effect is determined, an appropriate assessment is made of the implications for the Natura 2000 site, in view of that site's conservation objectives (Stage 3). Its purpose is to ascertain whether or not the proposal will have 'no adverse effect on the integrity of the Natura 2000 site'. Atmospheric dispersion models (such as ADMS and AERMOD) are generally used to estimate the 'process contribution' (potential /NH_3 concentrations and nitrogen deposition resulting from the installation) at the given Natura 2000 site. This is compared with the relevant environmental benchmarks (critical levels and loads) to assess the potential impacts on the designated features making up a given Natura 2000 site. At this stage, further consideration is given to the modelling

assumptions, location of designated features, sensitivity of the features, uncertainties within the assessment etc.

The UK regulatory and conservation agencies have developed a database of 'Site Relevant critical loads', whereby critical loads are assigned to each interest feature on each individual SAC and SPA, where possible (see Whitfield *et al*., 2010 in this volume, for more details). Information on 'background' (existing) nitrogen levels are usually derived from the UK Air Pollution Information System – www.apis.ac.uk (deposition at 5 km resolution, based on UK FRAME model; Singles *et al*., 1998) for each site.

If it cannot be concluded that there is no adverse effect on site integrity, having also taken into account any conditions, restrictions or mitigation measures that can be imposed on the plan/project, a process of determining alternative solutions and whether there is a case for applying for 'Overriding Public Interest' (OPI) is followed. If OPI is agreed (which is a decision for the Secretary of State) then compensatory measures (habitat) would need to be secured.

3.5.3 Assessment of Impacts of Ammonia from IPPC Intensive Livestock Installations

Evidence from modelling and monitoring studies has shown that very high concentrations and deposition of ammonia can occur around intensive livestock units. These have been associated with harmful impacts on semi-natural habitats, such as direct effects on sensitive species and changes in the compositions of the vegetation (Sutton *et al*., 2009).

The Pollution Prevention and Control Regulations 2000[2], Pollution Prevention and Control (Scotland) Regulations 2000 and Pollution Prevention and Control Regulations (Northern Ireland) 2003, required, for the first time, intensive livestock units above a certain size to apply for a permit. This requirement is applied to new pig/poultry units and also retrospectively to those already in operation. Permit applications also require an assessment under the Habitats Regulations.

In 2007, the regulators (Environment Agency/SEPA/NIEA) received over 1,000 permit applications from pig and poultry installations the vast majority of which were already in operation. To undertake an assessment under the Habitats Regulations, a distance criterion of 10km was used at Stage 1, so that a livestock unit was considered 'relevant' to all Natura 2000 sites (or Ramsar sites) within a 10km radius of the unit. At Stage 2 (the 'likely significant effect' screening), modelling of predicted ammonia concentrations was undertaken using simple assessment tools (Environment Agency Ammonia Screening Tool or Simple Calculation of Ammonia Impact Limits (http://www.scail.ceh.ac.uk/). The emission was calculated from the number of animal places multiplied by a standard emission factor. 'No likely significant effect' was concluded if the predicted concentration at the Natura 2000 site (resulting from the livestock unit) was equal to or less than a threshold of 4 per cent[3] of the appropriate critical level. If greater than 4 per cent, the unit proceeded to Stage 3 for further assessment. The assessment was based on the then newly revised ammonia critical levels for the protection of vegetation and ecosystems (1 $\mu g/m^3$ for lichens and mosses or 3 $\mu g/m^3$ for higher plants Cape *et al*., 2009).

2 The requirements have now been transposed into the Environmental Permitting Regulations 2007 in England and Wales.

3 These thresholds were determined by the environment agencies specifically for use in these circumstances (other numeric thresholds have been used with different installation types, pollutants, dispersion models etc).

Stage 3, the Appropriate Assessment, involved a more detailed assessment including: more detailed modelling, consideration of other 'background' sources, assessment of the installation's contribution and the consideration of site specific factors. An advanced dispersion model was used to predict the ammonia concentrations at the Natura 2000 site resulting from the livestock unit. At this point, the 4 per cent threshold was re-applied. If the predicted concentration was still greater than this, the assessment moved on to consider other sources of ammonia. The 'background' concentration of ammonia at the Natura site in question was identified from the Air Pollution Information System (www.apis.ac.uk) and added to the contribution from the livestock unit (to give a 'predicted environmental concentration'). If this total was less than the critical level then no further assessment was required. If greater, then further consideration was given to the farm contribution. If the process contribution (alone or in combination with that from any other neighbouring livestock units) was less than 20 per cent of the appropriate critical level, a conclusion of no adverse effect on site integrity was reached. If the process contribution was greater than 20 per cent[2] then a review of the data was undertaken.

The emission data were reviewed and checked with the operator, where appropriate, to confirm they matched the general operation of the farm. The application of the critical level was reviewed to ensure that the more stringent of the two critical levels was only applied to sites where sensitive lower plants (lichens and bryophytes) are considered key to ecosystem integrity. The location of the sensitive features relative to the predicted pollution 'footprint' was also considered. Other site specific information, e.g. site survey data, other sources of nitrogen for 'wet' sites were also identified and taken into account.

At Stage 4, if it was not possible to conclude 'no adverse effect on site integrity' based on the detailed assessment outlined above, the appropriate permit conditions were identified. The operators in England and Wales were required to produce an 'Emission Reduction Plan' and to implement actions to reduce ammonia, within specified timescales. In Northern Ireland, operators were required to assess/review whether standard ammonia emission factors were appropriate for their installation, and to submit proposals for reducing the impacts of ammonia emissions on the designated habitat(s). NIEA have also carried out monitoring of ammonia levels in the vicinity of the installations and the designated sites to establish actual air ammonia concentrations. In Scotland, SEPA has been piloting the development of a nitrogen bio-monitoring process in the vicinity of a number of intensive agriculture installations. Natural England and the Countryside Council for Wales have undertaken a series of site surveys in England and Wales, in support of the risk assessments. These looked for evidence of effects, consistent with ammonia/nitrogen impacts, on the designated features but also for other signals indicting high nitrogen conditions. Surveys included some or all of the following: a visual assessment, detailed quadrats of species abundance and cover, tree macro-lichen study, measurement of tissue nitrogen in mosses and soil nitrogen. The results were considered in relation to the site's conservation objectives. Where impacts consistent with the effect of ammonia on the site's conservation features were not found (despite the livestock unit being in operation for a significant period of time), permit conditions to reduce ammonia emissions from the installations(s) have subsequently been removed.

3.5.4 Conclusions

- In the UK, the tests within the Habitats Regulations (in relation to 'plans and projects') closely mirror the tests in the Habitats Directive, and require that permission can be granted only after it has been ascertained that the plan/project will have no adverse effect on the integrity of a Natura 2000 site (subject to certain provisions).
- A four stage risk assessment is applied in the assessment of nitrogen emissions from industrial sources, which also reflects the tests within the Habitats Directive and Habitats Regulations.

- Critical loads (for nutrient nitrogen and acid deposition) and critical levels (for ammonia and oxides of nitrogen) are applied at the site scale.
- After some initial screening stages (aided by various tools), detailed dispersion modelling is used to assess potential ecological impacts by comparing the predicted process contribution from the plan or project to the (site-relevant) critical loads and levels. Existing pollution levels at the Natura site(s) are also considered and assessment thresholds are applied.
- Site surveys have been conducted in some instances in support of the risk assessments, e.g. to look for impacts consistent with the effects of ammonia at sites close to (existing) intensive livestock units.
- Permit conditions have been imposed on some installations in order to reduce nitrogen emissions and enable a conclusion of 'no adverse effect' under the Habitats Regulations to be reached.

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3.6 Moninea Bog - Case study of atmospheric ammonia impacts on a Special Area of Conservation

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Summary
Moninea Bog is a lowland raised bog in Northern Ireland, designated as a Special Area of Conservation (SAC). The peatland flora typically supports many bog mosses, including the rare *Sphagnum pulchrum* and all three sundew species native to the British Isles. Farming activities take place around the bog, and questions were raised about the possible impact of ammonia emissions from a poultry farm directly to the north west. In response, following a site visit in January 2007, atmospheric ammonia was measured across the site, combined with measurements of nitrogen foliar bioindicators and the use of an atmospheric dispersion model. Taking the field observations, atmospheric measurements, modelling and bioindicators together, a clear picture emerged of a site under subsantial threat from atmospheric ammonia deposition. The combination of source- and receptor-oriented indicators coupled with a strong gradient in exposure 50-1000 m from the poultry farm provides for a robust approach to characterise these effects. This case-study graphically illustrates the nature of ammonia damage, showing how a short progamme of measurements and modelling can be used to support local decision making.

3.6.1 Background
Moninea Bog is a lowland raised bog in the west of Northern Ireland. The description of the Special Area of Conservation (SAC) notes that "*Moninea Bog is one of the best remaining examples of an active raised bog within the drumlin landscape that occurs across the southern counties of Northern Ireland. The peatland flora typically supports a high cover of bog-mosses, including the hummock-forming species* Sphagnum imbricatum *and* S. fuscum *and the nationally rare* S. pulchrum*. All three native British sundew species,* Drosera rotundifolia*,* D. anglica *and* D. intermedia*, are also present*" (JNCC, 2010).

The site became of interest from an air pollution perspective due the activities of a poultry farm directly to the north west of the bog. A number of units on the farm had been built and concern was raised that emissions of ammonia (NH_3) from the poultry farming activities might be causing adverse effects on the integrity of the SAC. The UK Centre for Ecology and Hydrology (CEH) therefore became involved to investigate whether there was evidence of ammonia damage, and, if so, whether this could be attributed to the poultry farming activities.

In planning such an assessment, we were able to draw on an extensive review of bioindicator methods for nitrogen deposition conducted by CEH with the support of the Joint Nature Conservation Committee (Sutton et al., 2004a and b'; Leith *et al*., 2005). In particular, that analysis had reviewed strategies for measurements in relation to what was termed the 'biomonitoring chain', the logical sequence of stages in biomonitoring from source to environmental effects (Sutton *et al*., 2005), as illustrated in Figure 3.5. The concept of the biomonitoring chain highlights how different monitoring methods may be selected to support an environmental assessment. Methods that are more 'source oriented' are naturally best for attributing their signal to particular emission sources, but typically give only indirect evidence of whether the biological environment is being adversely affected. By contrast, those that are 'receptor oriented' can have a much stronger link to the designated biological features of a nature conservation area, but typically have only an indirect connection to the pollution source, as other factors may also affect the designated features.

The biomonitoring chain envisages the range of possible stages for monitoring in the sequence from source to eventual biological effects. Thus a robust package of biomonitoring in any study can be envisaged as one that combines methods from several stages along the chain, including both source oriented, receptor oriented and intermediate methods (Sutton *et al.*, 2005).

From a practical perpsective resources are typically limiting in any particular study, and it is therefore important to note that some stages in the biomonitoring chain are easier to determine than others. In Figure 3.5, stages that are typically easier to assess are shown as darker shaded elipses, and it can be noted that these are conveniently distributed along the chain from source to ultimate effect.

These principles were applied in the observations and monitoring activities undertaken at Moninea Bog. With the available resources for such a study, it was not feasible to measure deposition fluxes or growth responses. Therefore, the asssessment focused on the following elements from along the chain:

- Monitoring of atmospheric ammonia concentrations,
- Measurement of nitrogen accumulation in indicator plant species,
- Assessment of visible injury to plants or other signals of nitrogen damage,
- Consideration of the status of particular target species/interest features.

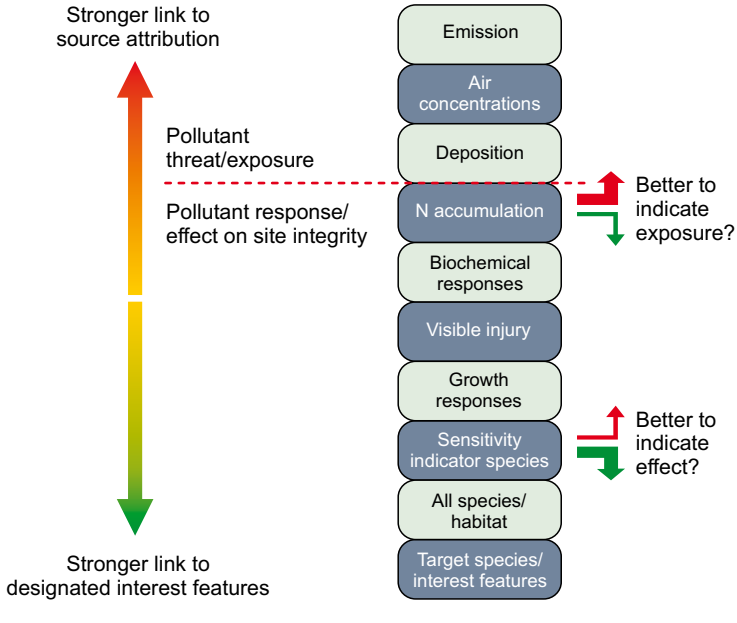

Figure 3.5. Overview of the "biomonitoring chain" showing how different indicator measurements may be ordered from pollutant source to ultimate pollutant impacts. Measurements closer to emission show a stronger link to source attribution, but weaker link to effects on designated interest features. Conversely, species-based measurements show a close link to the designated interest features, but a weak link to source attribution. A comprehensive robust programme of biomonitoring should therefore combine measurements distributed along the biomonitoring chain. Dark shaded elipses show the typical and most practicable approaches (Sutton *et al.*, 2005).

In implementing these observations and measurements, a transect was assessed with distance from the poultry farm, allowing quantification of the local profile from around 50 m to up to 1 km from the farm. In addition, in order to make a comparision with a clean reference site, observations and measurements were made approximately 40 km away at Loch Navar, where annual ammonia concentrations are typically 0.3-0.4 $\mu g\ m^{-3}$ (recorded as part of the long-term UK ammonia monitoring network (Sutton *et al*., 2001; Tang *et al*., 2001).

Finally, based on livestock numbers for the poultry management systems (including the existence of an open lagoon), estimates of ammonia emissions rates from the poultry farm were calculated, and used with nearby meteorological data to model the local dispersion of ammonia from the poultry farm, allowing comparision between modelled and measured ammonia concentrations.

Put together, these elements cover a broad range of the stages illustrated in Figure 3.5, representing a suitable package to assess the extent of local ammonia impacts on Moninea Bog.

3.6.2 Methods

Visual assessment: A visual assessment of the Moninea Bog site was made during January 2007. This consisted of an initial walk across the site from a location furthest from the poultry farm to areas on the bog closest to the farm. Based on the initial observations, the return walk was used to record photographic observations of damage indicators and to label plant samples for chemical analysis. The location of photographs and plant samples was recorded using a global positioning system, with the locations shown in Figure 3.6.

Plant nitrogen accumulation: Collected plant samples were dried and measured in the laboratory for chemical analysis using two methods, total foliar nitrogen concentration and foliar ammonium concentration (Sutton *et al*., 2004a; van Dijk *et al*., 2005, 2009). The use of these two complementary methods has been shown to be useful as the foliar ammonium indicator typically shows a much larger signal compared with total nitrogen, reflecting an increased availability of 'substrate nitrogen' in the plant system under situations of high nitrogen deposition.

Atmospheric ammonia: The locations for monitoring ammonia were identified during the site visit and subsequently established for measurements from February 2007. Air concentrations were measured using high sensitivity passive samplers ('ALPHA' samplers, limit of detectcion ~0.02 $\mu g\ m^{-3}$), exposed in triplicate, with the calibration of these samplers based on long term intercomparison with active denuder sampling (Tang *et al*., 2001).

Atmospheric modelling: The ADMS model was applied by the Northern Ireland Environment Agency (NIEA). This was run using local meteorological data using monthly wind statistitics to calculate average monthly ammonia concentrations, for the same periods as the ammonia monitoring to allow full comparison. In comparing the model outputs with the field measurements, some uncertainty is associated with assigning a representative local background concentration for the model estimates. For this purpose, the cleanest measured concentration at the site (~1 km distant from the farm), was considered to provide an upper estimate, since this may still, to some extent, be influenced by the poultry farm.

Since the time of the measurements in 2007, additional surveying has assessed the influence of ammonia on lichen community composition (cf. van Herk, 1999; Wolseley *et al*., 2006), though this is not reported here.

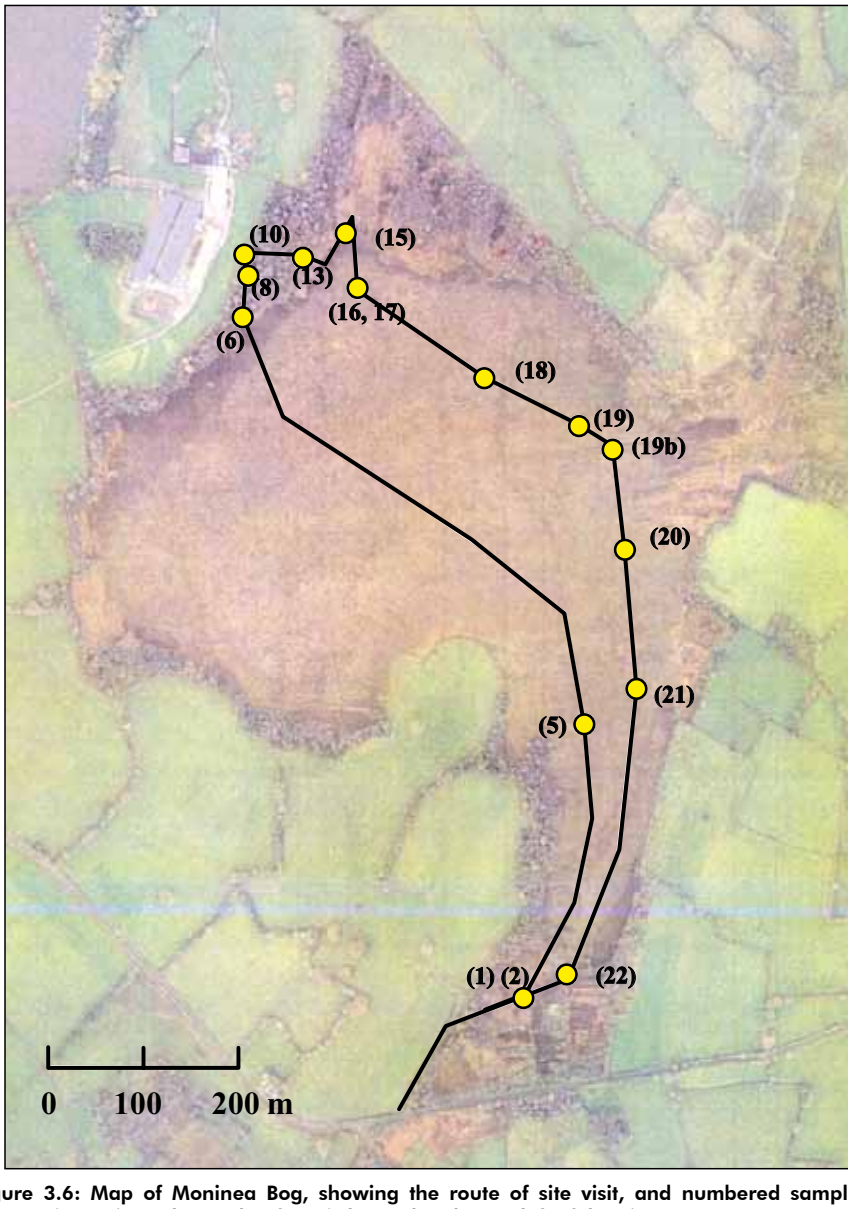

Figure 3.6: Map of Moninea Bog, showing the route of site visit, and numbered sampling/ observation points. The poultry farm is located at the top left of the picture.

3.6.3 Results

Visual assessment

At the northern edge of Moninea Bog the mixed decidous (mainly birch) woodland showed signs of extreme ammonia damage when compared with typical birch woodland in clean locations of Northern Ireland and northern Britain. In regard of the woodland ground flora, several mosses were present, but only in very limited amounts. For example, such a woodland at a clean location would be expected to have a rich bryophyte flora with species such as *Rhytidiadelphus* spp. present. At Moninea, a single small sample of *R. triquetrus* was observed at site 10, which was unusually green, suggesting a very high nitrogen level, (as also confirmed by a measured tissue nitrogen concentration of 4 per cent dry weight, see below).

For the trees, a number of gaps in the canopy were present. Although it was not possible to determine the cause of the tree decline from such a visual assessment, it was notable that bramble (*Rubus* spp.), ivy (*Hedera helix*) and Holly (*Ilex aquilifolium*) were flourishing, which species appear (from observations elswhere) to be characteristic of eutrophic conditions and insensitive to high ammonia concentrations.

One of the most graphic features of the woodland was the lack of the ephiphyte flora on the birch trees characteristic of clean locations. By contrast, on several trees, a thick algal slime had built up on the tree trunks, indicating an extreme level of eutrophication. This is illustrated in Figure 3.7, which contrasts the lichen and moss flora of a birch tree trunk characteristic of the clean reference location at Lough Navar, with an example tree in the woodland at Moninea Bog around 130 m east of the poultry farm (around sites 11-15). The contrast between these two trunks illustrates one of the strongest possible contrasts between clean and eutrophicated conditions in such woodlands.

Other matching signs of ammonia damage were seen to the vegetation of the open area of Moninea Bog. The most dramatic effects were in visible injury to lichen species, such as *Cladonia uncialis* and *Cladonia portentosa*, and to the bog mosses *Sphagnum* spp, which are particularly important for the peat building function of such sites. By contrast, because the survey was made in the winter season (January) it was not possible to evaluate whether there was ammonia damage to the sundew (*Drosera* spp) occurring on the site.

As an approximate indication, it was estimated that up to 200 m downwind (near site 17) of the poultry farm, the *Cladonia* and *Sphagnum* spp were more than 90 per cent eradicated or injured. At 400 m distant from the farm (near site 19) these species were estimated to be around 50 per cent eradicated or injured. The least injury was in the far south of Moninea Bog, 800-1000 m distant from the poultry farm. Here there was probably <10-20 per cent injury attributable to ammonia, and many apparently healthy *Cladonia* and *Sphagnum* specimens were found. However, even this area of Moninea Bog did not show the consistent vigour and health of the bog species present near Lough Navar.

Examples of ammonia damage to *Cladonia* spp are shown in Figure 3.8. In the left hand photograph, a hummock of *Cladonia uncialis* shows the bleaching that is characteristic of ammonia damage, as shown from the Whim Bog ammonia field release experimental study in Scotland (Sheppard *et al*., 2009). This bleaching (often accompanied by a slight pink colour) indicates where the algal symbiont appears to have been killed, and can be compared with a light bluish-green colour at the top of the photo, where the lichen is still alive. Following such an impact on the lichen, the lichen hummock eventually falls apart, leaving what can look like the 'dead bones' of the former lichen (see right side of Figure 3.8), which eventually decay into the peat.

Figure 3.7: Contrast between the epiphyte flora of a birch tree trunk at a clean location in northern Britain (left, 0.4 μg m^{-3} NH_3) and in the woodland on Moninea Bog (right, ~10 μg m^{-3} NH_3). The natural epiphyte flora of this area has in this case been replaced by a thick slime of algae.

© Left, Ian Leith; right, Mark Sutton

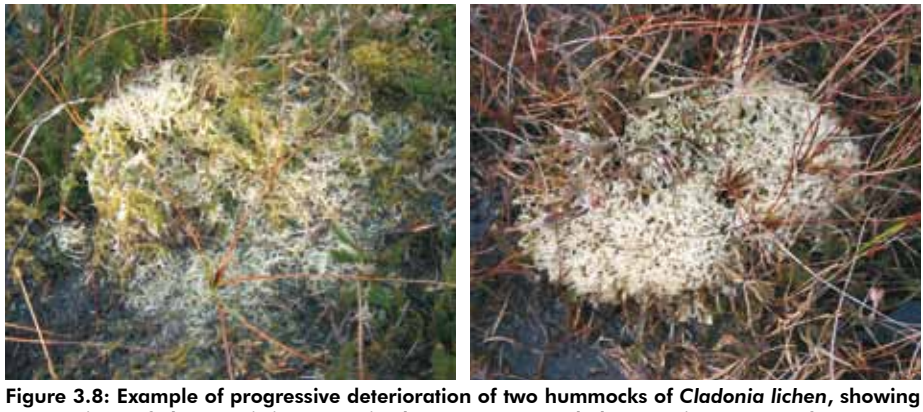

Figure 3.8: Example of progressive deterioration of two hummocks of *Cladonia lichen*, showing severe signs of characteristic ammonia damage, as recorded at Moninea Bog. Left: *Cladonia uncialis*, which is normally bluish (see at the top), is bleached over most of this specimen. Right: The eventual fate of this ammonia damage is illustrated by this specimen of *Cladonia portentosa*, where the lichen hummock falls apart, in this case becoming overgrown by a pleurocarpous moss.

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Ammonia damage to the *Sphagnum* bog mosses appears to proceed in a different way, and was clearly illustrated on Moninea bog. In the first instance, the high ammonia concentrations appear to favour algal growth over the surface of the *Sphagnum* plant. The main factors driving this require further elucidation, but the may be related to a combination of increased nitrogen availability from the ammonia and an increased pH of the moss surface due to ammonia exposure. As the algal slime develops, it appears that this smothers the *Sphagnum*, limiting gas exchange and photosynthesis, leading to eventual loss of integrity and death of the plant. This sequence is illustrated in Figure 3.9, which compares three specimens of *Sphagnum imbricatum* from site 18. The apparently healthy hummock on the left (based on visual assessment), is compared with a sample in the middle that shows the glistening coating of a developing algal slime (middle). Finally, on the right, the structure of the *Sphagnum* starts falling apart, leading to eventual decay and loss of this important peatland building element of the bog flora.

It is important to note that ammonia damage was not the only concern noted at Moninea Bog. In particular, broken fencing had allowed cattle to access and graze the bog, leading to physical damage (trampling of plants), as well as direct nutrient inputs through dung and urine. It was therefore important in making the field observations to ensure that the effects of grazing damage were distinct from those due to ammonia. In this respect, it was found to be fortuitous that the areas of most extensive grazing damage were to the south and east of Moninea Bog, at sites most distant from the poultry farm. In particular, it was found that the grazing damage was rather extreme where it occurred, but highly localized to areas of less than a few square metres (trampling) or less than 1 square metre (excretion). Thus the areas distant to the farm showed a clear distinction with patches of severe grazing damage, with apparently undamaged lichen and moss specimens growing immediately adjacent. Thus although, the grazing damage was easily visible, it did not in this instance present an all pervasive threat to the integrity of the bog ecosystem. By contrast, where the damage characteristic of ammonia was worst (closest to the poultry farm), this was pervasive, leading to a widely spread level of damage, representing a more significant threat to the integrity of the site. Figure 3.10 illustrates the localized impact of grazing on the bog, showing *Lolium perenne* grass (from the seed of adjacent fields) growing over a dung patch on the bog.

Plant nitrogen accumulation

The outcome of the plant nitrogen measurements is summarized in Figure 3.11. This shows the total foliar N concentration and the foliar ammonium concentration with distance from the north west edge of the Special Area of Conservation (SAC) near the poultry farm, as compared with the clean reference location (a peat bog near Lough Navar). The lowest foliar nitrogen and ammonium values were found at the reference location (plotted at an indicative 10000 m on the x-axis) at about 0.5-1 per cent N of dry weight and 0.1-0.7 μg NH_4 per g fresh weight, with the next lowest values at the Moninea site most distant from the farm. Close to the farm, values increased up to 4 per cent N of dry weight and 45 μg NH_4 per g fresh weight (the *Rhytidiadelphus triquetrus* specimen noted previously and in *Eurhynchium praelongum*). The increase in values closer to the poultry farm demonstrates how an additional source of nitrogen (in this case local ammonia dispersion) is leading to an accumulation of nitrogen in the plants, which can be expected from previous studies to be associated with an increased risk of adverse effects on the plant communities. This indicator therefore is reflective of its position midway along the biomonitoring chain, with a link to both the emissions and the eventual effects on species composition.

For the *Sphagnum imbricatum* specimens collected at site 18 (Figure 3.10), the tissue nitrogen concentration was also compared with the visual assessment of integrity. This is illustrated in Figure 3.12, which suggests that adversely affected speciments were associated with higher foliar nitrogen concentrations. Although statistics are not feasible for such a comparison, this points to the potential for further examination of the relationship between sample health in response to nitrogen

Figure 3.9: Example of progressive damage in the bog moss *Sphagnum imbricatum*, as observed at Moninea Bog. An apparently healthy specimen (shown left) is comparaed with a specimen showing algal invasion over the leaves (middle). On the right, severe algal invation has lead to a complete loss of integrity of the specimen (All samples from site 18). *© Ian Leith*

Figure 3.10: Illustration of grazing damage to Moninea Bog, where a localized patch of *Lolium perenne* has colonize a dung patch (site 4). Because of the localized nature of such damage, it was clearly distinguishable from the more pervasive effects characteristic of ammonia which were more severe closer to the poultry farm. *© Mark Sutton*

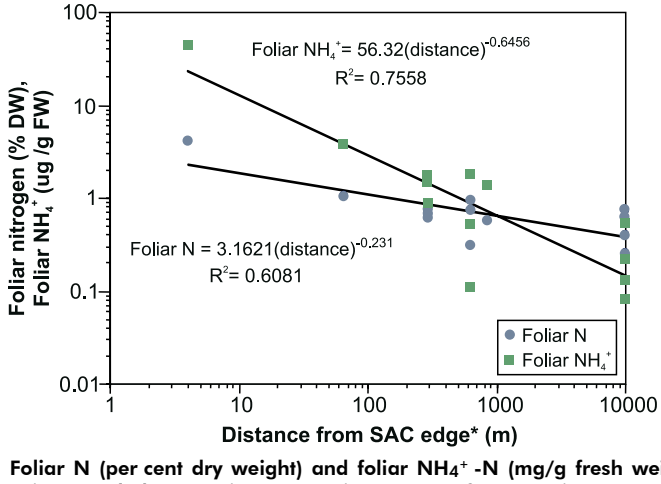

Figure 3.11: Foliar N (per cent dry weight) and foliar NH_4^+ -N (mg/g fresh weight) for moss and lichen species sampled on Moninea Bog. Distances are from a point 4 m north of Site 10. Samples from the clean reference location (Site 23) are plotted at an indicative 10000 m.

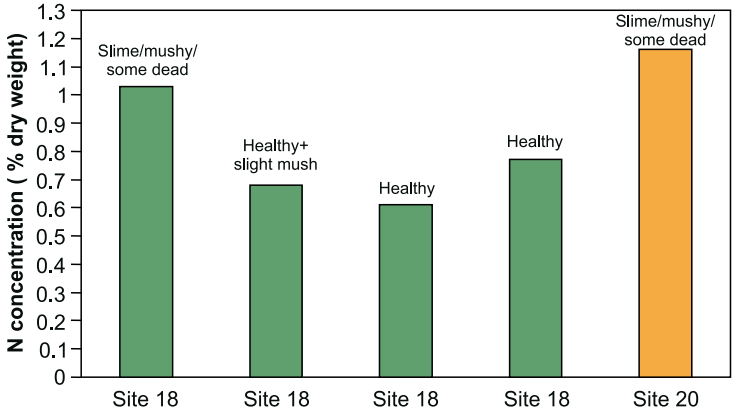

Figure 3.12: Comparison of foliar nitrogen contents of *Sphagnum imbricatum* specimens in different levels of condition from two sites on Moninea Bog (sites 18 and 20). There is an indication that damaged specimens (colonized with algae) have higher tissue N content.

deposition and the biodindicator value. For example, as samples become damaged, reduced growth rates may further augment the high nitrogen concentrations.

Atmospheric ammonia measurements and dispersion modelling

Example results from the monitoring of atmospheric ammonia concentrations are illustrated in Figure 3.13. By using triplicate measurements with the ALPHA samplers, robust estimates of ammonia concentration were provided, with coeficients of variation (standard deviation / mean) in the range 1 per cent to 4 per cent. For each of the three months illustrated (and for other subsequent months, not shown), the highest ammonia concentrations were recorded at the location closest to the poultry farm (14 to 34 $\mu g\ m^{-3}$), with the lowest concentrations at the site most distant to the farm (1-4 $\mu g\ m^{-3}$). These concentrations were substantially larger than those recorded at the clean reference site (Lough Navar) over the same period (0.2-0.4 $\mu g\ m^{3}$), indicating how even the most distant site was to some degree influenced by the poultry farm and other ammonia sources in the area, such as from adjacent fields grazed with cattle.

The local dispersion modelling conducted by NIEA was found to be fully consistent with the measured ammonia concentrations, as illustrated in the right side of Figure 3.13. The monthly variation could be partly explained due to differences in wind direction frequency between months. Apart from the model uncertainties for meteorology and ammonia emissions, an uncertainty of around 1 $\mu g\ m^{-3}$ applies to the assumption of background ammonia concentration for the model, though this has negligible influence on the comparison for locations close to the farm. Since the atmospheric dispersion modelling is based on estimated ammonia emission rates from the poultry farm and meteorology for each month, the comparison with measurements can be considered as extremely encouraging, indicating that the spatial pattern in measured ammonia concentrations are fully consistent with dispersion away from the poultry farm as a major local ammonia source.

3.6.4 Discussion and Conclusions

This case study at Moninea Bog shows how a package of methods applied from across the 'biomonitoring chain' can provide a robust demonstration of an ammonia threat to a Special Area of Conservation. Such a short-term assessment provides useful evidence to attribute changes to a particular driver. For example, two particular threats are noted here that appeared to be influencing

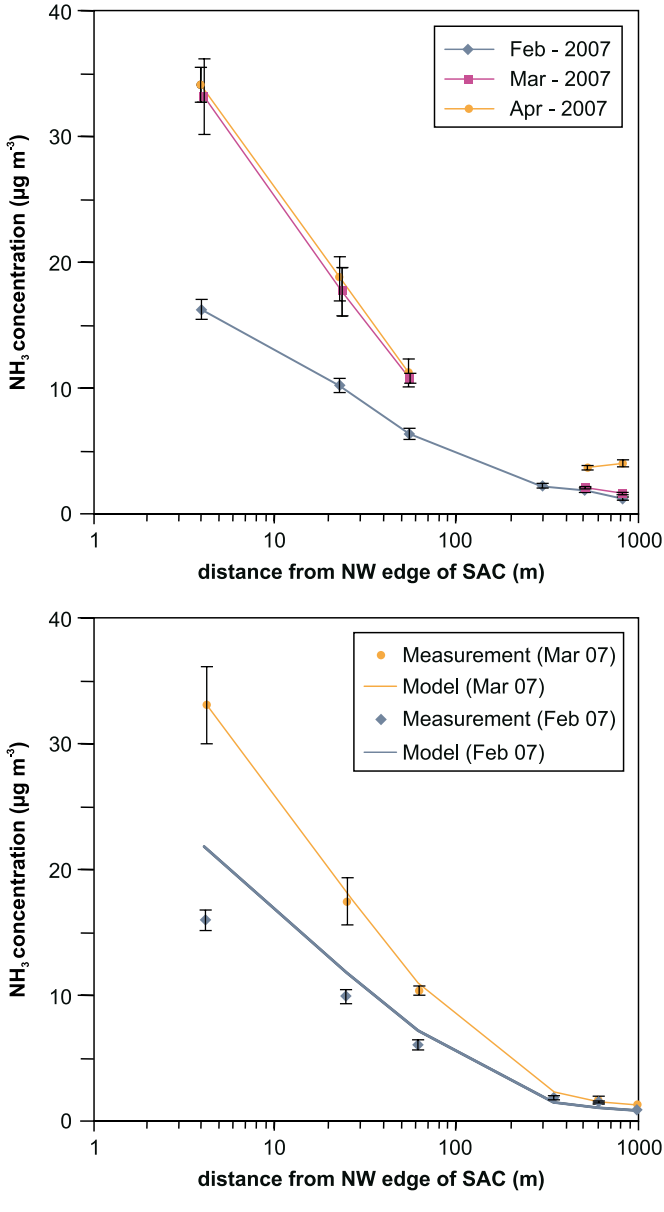

Figure 3.13: Atmospheric ammonia concentrations measured across Moninea Bog SAC for three months of 2007 (Measured at 1.5 m height above ground, showing +/-95 per cent confidence limits of the mean based on triplicate sampling). The bottom graph compares the measured concentrations with a simulation using the ADMS model for February and March 2007.

the integrity of Moninea Bog: grazing damage from cattle and the threat of ammonia deposition, especially from a nearby poultry farm.

Visual observations at the site allowed the grazing and ammonia threats to be clearly separated, which in this case was made easier by the fact that the worst grazing damage at the site was on locations most distant from the poultry farm. The grazing damage was found to be significant and easily identifiable (trampling, plants growing out of dung patches etc), but highly localized. By contrast, the ammonia damage was more pervasive, leading to wide scale damage, that was most extreme close to the farm, and which decreased with distance over the first km from the farm. This broader scale damage on the site can be attributed to ammonia from the poultry farm, because:

- the visible injury symptoms, both for mosses and lichens are characteristic of ammonia damage,
- the visible injury increased closer to the farm (representing higher ammonia concentrations).
- the plant nitrogen and ammonium concentrations were increased over the same range, proportionately to the exposure of ammonia, and consistent with other studies downwind of ammonia sources,
- the measured ammonia concentrations were substantially increased near the farm, decreasing with distance away from it, and
- these measured ammonia concentrations could be replicated by the a dispersion model, using emission factors for poultry management based on the local stocking rates.

Altogether, the range of indicators used, together with the availability of a clear transect of decreasing threat with distance from the farm, provide for a robust assessment of the site that establishes the link from source attribution, through chemical nitrogen accumulation, to eventual loss of integrity of the designated features.

Such an assessment, where effects can be attributed to a source, can also support long term monitoring activities. For example, ongoing monitoring of Moninea Bog by NIEA showed a 50 per cent loss of *Sphagnum* over a the period 2004-2007 for locations less than 400 m from the farm. On their own, such observations highlight a serious concern about a site, but can leave the questions of the causal threats unanswered. Using the approaches together, including the comparison with a clean refernece location, therefore provides a robust evidence-base on the cause and extent of concern, which can be used to inform local decision making.

The example of Moninea Bog provides a salulory less of how farming activities can have acute effects on the integrity of a Special Area of Conservation in the Natura 2000 network. The results are also consistent with the ammonia critical level (Cape *et al*., 2009; Sutton *et al*., 2009) of 1 μg m^{-3} for lichens bryophytes and habitats like peat bogs where these are essential to the ecosystem integrity. With concentrations, much larger than this at Moninea, it is not surprising that acute adverse effects were observed.

By contrast, the present assessment of Moninea Bog never set out to determine the extent to which the cleanest location the bog (to the south east, furthest from the farm) was under significant threat from ammonia. This becomes a harder question, given that the reference bog near Lough Navar was 40 km distant. However, based on a comparison of: a) visual assessment of *Cladonia* and *Sphagnum* spp between the two sites (with poorer condition on the same date for the cleanest location of Moninea), b) the difference in foliar nitrogen and ammonium concentrations (Figure 3.11), and c) the result that the concentration at around 850 m from the farm was in the range 1 to 4 μg m^{-3} (i.e., larger than the critical level), it seems most likely that even the cleanest location of Moninea was sufferering from chronic exposure to ammonia.

This case study illustrates an extreme case of ammonia exposure and damage to a Natura 2000 site. However, at the same time, it highlights the widespread nature of the ammonia threat to such ecosystems where lichens and bryophytes are essential to their integrity. As Hallsworth *et al*., (2010, 2011) demonstrate, the ammonia critical level is exceeded across more than 93 per cent of England, 68 per cent of Wales, 26 per cent of Scotland and 85 per cent of Northern Ireland (1 km resolution estimates), showing how widespread adverse effects can be expected.

Acknowledgements

Many of the methods and concepts used in this paper were developed with the support of the Joint Nature Conservation Committee and English Nature. The measurements and field observations at Moninea Bog were made by CEH staff with the support of Northern Ireland Environmental Protection Agency, who also kindly provided the model estimates using ADMS and exchanged the monthly ammonia samples. The model estimates illustrated here are those corrected by CEH for estimated local background ammonia concentration.

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4

ASSESSING NITROGEN IMPACTS ON CONSERVATION STATUS (THEME 2)

4.1 Background document

C.P. Whitfield[1] and I. Strachan[2]

1 Joint Nature Conservation Committee (JNCC), UK
2 Scottish Natural Heritage (SNH), UK

Summary

- The Habitats Directive requires Member States to take measures to maintain at, or restore to, favourable conservation status, the natural habitats and species of Community Importance. Member States are required to report on the implementation of the Directive every six years, including an assessment of conservation status (Article 17).
- Nitrogen deposition impacts are considered to be a significant threat to sensitive habitats across Europe. Therefore, it is necessary to understand the effects of nitrogen deposition on attaining favourable conservation status. In turn, this should inform air pollution policy development, helping to target it appropriately to account for the objectives of the Habitats Directive.
- For the last Article 17 reporting round, which covered 2001-2006, a number of Member States included an assessment of nitrogen deposition impacts based on an application of critical loads. Other Member States used evidence from field surveys or a combination of these alongside critical loads assessments. However, the detection and attribution of nitrogen deposition impacts is not straightforward, and the application of critical loads in this context also raises a number of challenging questions.
- This background paper identifies some of the key issues concerning the assessment of nitrogen impacts on conservation status. These were discussed at the workshop with a view to sharing experience and good practice, and with a forward look to improving methodologies and consistency in their application for the next reporting round in 2013 (as reported by Whitfield *et al*., this volume).

4.1.1 Introduction

The Habitats Directive (92/43/EEC) together with the Birds Directive (79/409/EEC) are the main drivers of Europe's nature conservation policy. The Habitats Directive promotes the maintenance of biodiversity and requires Member States to take measures to maintain or restore the threatened natural habitats and wild species listed in the Directive at favourable conservation status, introducing robust protection for those habitats and species of European importance.

The provisions of Article 17 of the Habitats Directive require Member States to produce a report every six years on the implementation of the Directive, including the assessment of **conservation**

status of all the relevant habitats and species listed in the Annexes of the Directive. The second report, which covered the period 2001-2006, included such assessments for the first time. The methodology for assessing the impacts of nitrogen deposition on conservation status is the subject of this paper.

Nitrogen deposition remains a threat to biodiversity across large areas of Europe (CCE, 2008). This concern is reflected in the incorporation of an indicator for nitrogen deposition under the Streamlining European Biodiversity Indicators 2010 (SEBI, 2010) programme (EEA, 2007), which helps measure progress towards the European target to halt the loss of biodiversity by 2010. Common assessment methods, such as critical loads, are already well established for use in European air pollution policy development. Critical load exceedance maps identify areas at risk from atmospheric nitrogen deposition. They show that a substantial area of semi-natural habitat in Europe exceeds the critical loads (see Figure 4.1).

Since the Habitats Directive is one of the priorities in European nature conservation policy, it is important to understand the risks from nitrogen deposition to achieving the Directive's objectives. An assessment of nitrogen deposition impacts on attaining favourable conservation status, based on a robust assessment approach, is essential to inform air pollution policy development and to ensure that it is targeted appropriately to help achieve the objectives of the Habitats Directive.

In this background paper we provide an introduction to the reporting of conservation status and consider how nitrogen deposition may impact on conservation status. We then provide a summary of the approaches taken by a selection of Member States to assess nitrogen deposition impacts on conservation status. An overview is then presented of the preliminary results from the most recent reporting round, in relation to the reporting of "air pollution" and "eutrophication" as a "pressure" or a "threat". Building on this experience and anticipating the next reporting round in 2013, we aim to begin to identify some key questions and challenges, concerning assessment methodology and procedures, which require further development to ensure a harmonized, robust and consistent approach between countries. Overall, the aim is to share experience and to open up discussion on the methods and mechanisms for future assessments.

4.1.2 An introduction to conservation status assessments

Background to reporting

The Habitats Directive requires Member States to report every six years on the conservation status of the habitats listed in Annex I and the species listed in Annexes II, IV and V of the Directive. The methodology for reporting conservation status is determined by the EC Habitats Committee. Supplementary guidelines were produced by the European Commission in collaboration with Member States (European Commission, 2006) to ensure that the reporting is done on a consistent and comparable basis. The reporting format requires a separate analysis for each habitat and species in each biogeographical region that a country covers.

Favourable Conservation Status (FCS) of a habitat is defined in Article 1(e) of the Directive as being when:

- its natural range, and areas it covers within that range, are stable or increasing; and,
- the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future; and,
- the conservation status of its typical species is favourable as defined in Article 1(I).

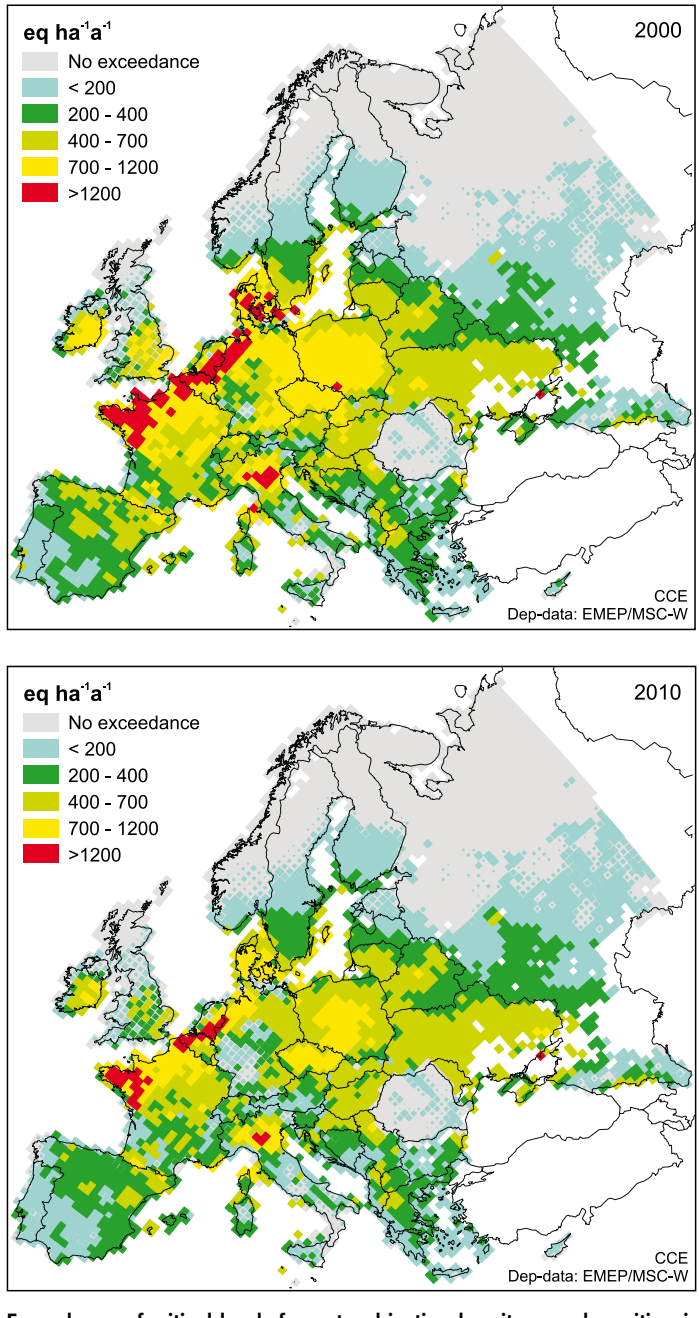

Figure 4.1: Exceedance of critical loads for eutrophication by nitrogen deposition in 2000 and 2010 under current legislation (courtesy of CCE, 2008).

FCS for a species is defined in Article 1(I) of the Directive as being when:

- population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats; and,
- the natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future; and,
- there is, and will probably continue to be, a sufficiently large habitat to maintain its population on a long-term basis.

In other words, in simple terms it can be described "as a situation where a habitat type or species is prospering (in both quality and extent/population) and with good prospects to do so in future as well" (European Commission, 2006).

The Commission guidance states that the range and area of the listed habitats, and the range and population of the listed species, should be at least maintained at their status when the Directive came into force or, where the status at that time was not viable in the long term, should be restored to a position where it would be viable. The six-yearly reports are intended to measure the effectiveness of the Directive in meeting its aims, which are essentially to secure favourable conservation status. The 2001-2006 report provides a baseline by which future assessments can be judged.

It is very important to recognise that the assessment of conservation status for a habitat or species should be made across the whole of its range, rather than being confined to Special Areas of Conservation (SAC) (which together with Special Areas of Protection make up the Natura 2000 network). The proportion of a feature which occurs within SACs will vary on a case by case basis and between countries and biogeographic areas. In many cases a substantial proportion occurs outside SACs in the 'wider countryside' or seas.

The Commission guidance stipulates four parameters for assessing the conservation status of habitats. These are:

- range,
- area,
- specific structures and functions including typical species,
- future prospects.

For species, the parameters are:

- range,
- population,
- habitat for the species,
- future prospects.

Each of these parameters is assessed as being in one of the following conditions: Favourable, Unfavourable-Inadequate, Unfavourable-Bad, or Unknown, according to agreed standards (European Commission, 2006). In addition to assessing the individual parameters referred to above, Member States are also required to make an overall assessment of the conservation status of each of the habitats and species following an agreed method. This overall assessment is determined by reference to the conclusions for the individual parameters, and, in general, reflects the least favourable of the individual parameter conclusions.

Taking nitrogen deposition into account

As stated above, for the conservation status of a habitat to be favourable, "the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future". Habitat structure and habitat function varies widely between different habitats, but it is clear that the various ecological processes essential for a habitat have to be present and functioning for the habitat to be considered to be at favourable conservation status (European Commission, 2006).

A large number of the habitats (and species, either directly or indirectly) listed under the Habitats Directive are sensitive and potentially vulnerable to atmospheric nitrogen deposition. Nitrogen deposition may cause changes to composition, often including a reduction in species richness and a loss of sensitive 'lower plants'; changes to soil microbial processes; changes to plant and soil biochemistry; increased susceptibility to abiotic stresses (such as winter injury) and biotic stresses (such as pests and pathogens); and it also contributes towards acidification (NEGTAP, 2001). It is clear that such impacts could adversely affect the "specific structures and functions" element of conservation status, as well as threatening the future prospects, for sensitive habitats and species. In some cases nitrogen deposition may also have affected the range of a habitat (through a change in species composition) or species.

Under the assessment of "specific structures and functions" for habitats, Member States are required to provide a list of the "main pressures" currently acting on each habitat. Similarly, for the "future prospects" assessment, future threats (to the range, extent, structures and functions) must be documented. The guidance (European Commission, 2006) provides an example under the notes for "future prospects" for defining "unfavourable-bad" (i.e. the habitat's prospects are bad, severe impact from threats expected; long-term viability not assured) as "under pressure from significant adverse influences, e.g. critical loads of pollution exceeded".

The EC guidance lists a suite of pressures and threats (European Commission, 1997) including "air pollution" (code 702). "Eutrophication" (code 952) is also listed separately, but in the context of biocenotic evolution (ecological succession). However, there is no guidance on the definitions of the listed pressures/threats, which are open to inconsistent interpretation, nor are there criteria for judging whether the severity of threat warrants its inclusion (but note that this is to be addressed by the EC Expert Group on Reporting). Presumably "air pollution" would be expected to include consideration of acidic and eutrophying deposition (and direct effects of the gases associated with these pollutant species) and ozone, in so far as an assessment is possible. However, the categories "eutrophication," "acidification," and "fertilisation" may also have been used to record the effects of nitrogen deposition.

As documented in the introduction to this paper, the most recent reporting round for Article 17 was 2007/8 and covered the period 2001-2006. The European Commission has produced a composite report providing an analysis of the results of the 2007 reporting round (European Commission, 2009). Prior to the publication of the composition report, the European Topic Centre (ETC) on Biodiversity provided the authors of this paper with a working draft copy of a database of the results of the conservation status assessments. This allowed an analysis across the EU, Member States and biogeographic regions of where "air pollution" and/or "eutrophication" was identified as a pressure or threat for each habitat assessment.

The main focus of this background paper is on the assessment for Annex I habitats, rather than species, since most information is available for these; and a comparison to other assessment tools, such as critical loads, is more straightforward. However, an assessment is also required for species.

4.1.3 Examples of the methodology used by a selection of Member States

Introduction

Member States were required to submit Article 17 reports, including conservation status assessments, in 2007. Each individual habitat and species assessment (by country and biogeographic region) is available on the ETC's website (http://biodiversity.eionet.europa.eu/article17). However, as stated earlier, the EC guidance (European Commission, 2006) on Article 17 reporting did not include guidance or criteria for identifying and assigning the main pressures and threats, and there was no obligation to provide details of the methodology used for such purposes. As a consequence, whilst it is possible to query the results of the individual habitat/species assessments, information on the generic approach to nitrogen deposition assessment is only available for a small number of countries (notably UK and Denmark) through the ETC website (http://rod.eionet.europa.eu/countrydeliveries?actDetailsId=269).

In an attempt to present an overview of approaches taken by different Member States, the authors requested details of the methodology used by a selection of Member States from members of the workshop advisory committee and other contacts. The summaries below reflect responses received. The paper describes methods of the UK, Denmark, Belgium, Austria, Germany and Netherlands. Contacts from some other countries have indicated that there was no explicit consideration of the issue (e.g. Czech Republic and Portugal). However, overall 18 Member States have reported air pollution as a threat or pressure for at least one habitat assessment (and likewise all 25 reported eutrophication, although this may include non-atmospheric inputs and discharges to water in marine and freshwater/wetland habitats).

Country Summaries

UK

The UK assessment of "specific structures and functions" for habitats was made based on the main pressures currently acting on the habitat, information on the habitat condition and, where relevant information was available, the status of typical species associated with the habitat.

Information on habitat condition from site condition monitoring formed a major component of the assessment. However, since the approaches used for site condition monitoring in the UK are largely based on fairly rapid visual assessment of key attributes of the habitat, it is acknowledged that this is not a sensitive tool for detecting and, in particular, attributing nitrogen deposition impacts (Williams, 2006). Therefore, a nitrogen deposition assessment, based on the use of empirical nutrient nitrogen critical loads and modelled nitrogen deposition from the UK models FRAME (Singles *et al*., 1998) and CBED (Smith *et al*., 2000), was also undertaken. This also has the advantage of providing a predictive approach for assessing 'future threats'. The methodology is reported in a technical annex to the UK's submission (http://www.jncc.gov.uk/pdf/FCS2007_techIII_airpollution.pdf) and in Whitfield (this volume), but a brief summary is given below.

The critical loads based assessment was carried out for Annex I habitats only. Species were excluded because of the difficulty in linking habitat-based critical loads to effects on individual species. Habitats judged not to be sensitive to nitrogen deposition (and acidification) impacts were also excluded from the assessment. In addition, habitats which could not be assigned a critical load (see later) were excluded.

The UK does not have nutrient nitrogen critical load maps for Annex I habitats, so existing critical loads information was adapted for the purposes of the conservation status assessments. The UK

was in a fortunate position having undertaken a substantial exercise to assign relevant critical loads to interest features on SACs known as Site Relevant Critical Loads (SRCL) (Bealey *et al*., 2007). Exceedance data for all sensitive Annex I habitats as they occur in SACs are therefore available. In this exercise, the 'relevant' critical loads were assigned to Annex I habitats where there is adequate equivalence with a EUNIS class for which critical loads have been assigned (UNECE, 2003). A few Annex I habitats which are potentially sensitive had to be excluded because there is not a habitat for which a critical load is set, which has sufficient equivalence with the Annex I habitat. This assignment of 'relevant' critical loads to Annex I habitats based on the EUNIS habitat classification is critical; it is a common theme amongst those countries which have used a critical loads based assessment for conservation status reporting, and will be considered in the workshop discussion.

However, the UK's SRCL exceedance data only provide information for the proportion of habitats which occur within SACs. To ensure the assessment adequately represented the risk to the whole Annex I habitat resource, a combined approach was used which drew on UK national critical loads exceedance mapping (Hall *et al*., 2003) in addition to the SRCL data. Difficulties with different habitat classifications, resolution of mapping and so on meant that only a qualitative assessment was possible.

Where 'relevant' critical loads are exceeded over a significant area for a particular habitat, air pollution was listed as a current "pressure" and future "threat" (future/foreseeable impacts). Any field evidence of impacts on the habitats, or other impacts information, was also used to inform whether air pollution would be listed as a current pressure or future threat. In practice, this was largely confined to coastal habitats, which were not well represented by the critical loads exceedance assessment, and freshwater habitats, for which there were no applicable critical loads.

Denmark

Denmark has established a new national monitoring programme (NOVANA) (Svendsen *et al*., 2005) which includes systematic monitoring of terrestrial habitats (and species). This aims not only to provide information on status and trends, but also to provide insight into natural and anthropogenic pressures in order to inform management. For each Annex I habitat, a set of measurable indicators of favourable conservation status has been developed. These define favourable biological status for the habitat type in question and what physical-chemical conditions are required for this favourable status to be maintained. The programme is not only designed to detect any changes in conservation status, but also to give answers as to why the changes have happened. The programme combines intensive and extensive monitoring. The intensive monitoring will elucidate cause-effect relationships between trends, pressures and conservation status. The extensive monitoring provides representative data at a national scale. Some of the parameters measured between the two are the same, but the frequency is lower in the extensive monitoring.

A number of the indicators relate to nutrient effects because of the established concern over eutrophication. These typically include nitrogen deposition (which should not exceed the relevant critical load (based on UNECE, 2003)), C:N ratio in soil, tissue N content and pH, as well as species composition parameters. Relevant empirical critical loads have been assigned to each Annex I habitat based on equivalence between habitat types. Further information is given in Nielsen (this volume).

In the Article 17 report, Denmark reported "unknown" future prospects for forest habitats, because the positive effects of better pollution control, nature and forest restoration/protection might outweigh the negative effects of air pollution within the next 20-30 years. However, it is recognised that there is uncertainty concerning this and little quantification of the true extent of critical load exceedance of forest habitats. As a result air pollution has not been listed as a pressure/threat on the forest habitat types in the Danish Article 17 report.

Netherlands

There is no specific documentation within the Netherlands' Article 17 submission in respect of the approach for N deposition assessment. However, the results have shown that nitrogen deposition is a pressure and threat for several habitat types. This was based on a scientific report providing an approach for assessing nitrogen deposition impacts in Natura 2000 areas (Van Dobben and Van Hinsberg, 2008), which was subsequently adopted by the Dutch government (Dick Bal, pers comm.) (see also, Van Hinsberg and Van Dobben, this volume).

Van Dobben and Van Hinsberg (2008) provide a basis for setting critical loads for all Annex I habitat types based on a phased application of empirical critical loads for nutrient nitrogen (UNECE, 2003), model results and expert opinion:

- *Phase 1*. The Annex I habitat is compared to the habitat types (based on EUNIS habitat classification) for which empirical critical loads have been set (UNECE, 2003). There are two possible outcomes (a) the Annex I habitat is equivalent to, is part of, or sufficiently resembles a habitat type defined under EUNIS for which a critical load range is set (referred to as "UN type"); or (b) the Annex I type does not resemble, or does not sufficiently resemble a UN type.
- *Phase 2*. The result from Phase 1 needs to be further refined (a) (i.e. value set within range) or estimated (b). As far as possible this is done on the basis of model results (from the SMART2 model). Where there are no sufficiently reliable model results a Phase 3 is required.
- *Phase 3*. This uses expert opinion to set the critical load (and indicates uncertainty) on the absence of reliable estimates from the model.

Austria

The conservation status assessments in Austria were undertaken by nine separate States. There was no common countrywide approach to reporting "air pollution" or "eutrophication" pressures or threats across a range of habitat types. These assessments were done exclusively by expert knowledge for all species and habitats (Thomas Dirnböck, pers. comm.).

Germany

Germany has not directly used critical loads, as such, for Article 17 reporting, but nitrogen deposition and eutrophication play an important role for assessing conservation status, being taken into account mainly in the assessment of structure and function, including typical species, via a series of evaluation matrices for every habitat/species that were negotiated with experts and the Federal Länder in order to ensure at least within Germany a homogenous approach of the 16 Federal States (Länder) (Axel Ssymank, pers. comm.). Germany also has recently published a national guideline (VDI, 2008), which is aimed at identifying and monitoring unwanted N-eutrophication effects (also with regards Article 6.3). However, it is unclear whether or how this has been used to inform the nitrogen assessment for Article 17 reporting (Jürgen Franzaring pers. comm.).

Portugal

There is only one record of air pollution and two records of eutrophication as a pressure/threat on Portuguese habitats. These relate to grasslands. It was not possible to find any reports specific to this subject from the Portuguese Institute for Nature Conservation and Biodiversity (ICBN) or through direct contact, so the underlying assessment is unknown presently. However, the view of some of the Portuguese scientific research community is that the impact of nitrogen on biodiversity is not a priority subject for conservation biology and management, in the ICBN. Thus, nitrogen deposition was unlikely to have been considered in habitat conservation status reporting. However, there is more widespread concern from Portuguese scientists regarding nitrogen (particularly ammonia)

deposition impacts on biodiversity (Cristina Branquinho, pers. comm.). A range of publications document the use of lichens as biomonitors and the impacts on epiphytic lichen communities (Pinho *et al*., 2008 and 2008).

Belgium

The Article 17 reporting for Belgium has been conducted separately for the Atlantic and Continental biogeographical regions in Belgium. The Research Institute for Nature and Forest (INBO) was responsible for the conservation status assessments of habitats and species in the Atlantic region of Belgium, which encompasses nearly the whole of Flanders.

In Flanders, reports on nitrogen deposition and critical load exceedance in a number of ecosystems (forests, grassland, heathland) are published annually (see www.milieurapport.be, www.natuurindicatoren.be). These reports are based on modelled deposition rates (1 km^2 spatial resolution, OPS-model) and on a geographically distributed set of point locations for which 'exact' critical load values are available. 'Exact' means that detailed soil profile information and vegetation characteristics have been taken into account to determine the part of the critical load range to apply for each of these points.

For the Article 17 reporting, a somewhat more empirical and simplified approach was used to assess the pressures and threats from nitrogen deposition. For each Annex I habitat type, a single empirical critical load for nutrient nitrogen was put forward, based on critical load literature and expert judgement. This critical load value was compared to average nitrogen deposition rates during the period 2001–2006 . Hence, spatial variation was not accounted for in N deposition or for differences in critical loads between locations or between Natura-2000 sites.

Habitat types for which the average 2001–2006 deposition exceeded their critical load where identified. For these types, fertilisation ('120') and air pollution ('702') were listed among the main pressures and as threat in the habitat assessment. Subsequently, the conservation status at biogeographical level regarding both 'specific structures and functions' and 'future prospects' was scored as either inadequate (U1) or bad (U2), depending on other pressures and threats.

Although roughly in line with common practice among Member States, INBO is aware that this pragmatic approach should be refined and improved for future conservation status assessments. INBO is currently looking into ways to improve the spatial resolution of model-based assessments and to complement this approach with measurements of N enrichment effects (cause-effect monitoring).

4.1.4 Illustrations of the results from the 2007 Article 17 report

The preliminary results from the 2001-2006 conservation status assessments, amounting to some 2771 habitat records, have been provided by the ETC. This has allowed an analysis across Member States and biogeographic regions of when "air pollution" and "eutrophication" have been identified as a pressure or threat for each habitat assessment.

The tables below provide an illustration of some potential outputs from the dataset. However, interpretation of the results should be made with caution: different methodologies have been used (as presented in Section 3); the use of pressure/threat categories "air pollution" and "eutrophication" appear to have been used variably between countries; and some countries made no assessment of the impacts of nitrogen deposition (whether because of no evidence/concern of nitrogen deposition impacts or because of no methodology, is not usually clear). Therefore, the results do not necessarily give an accurate representation of nitrogen deposition impacts on conservation status across the European Union. No comparison has been made with other pressures or threats as there is no

Table 4.1: Proportion (per cent) of records (habitat/biogeographic region/country) which record air pollution (code 702) or eutrophication (code 952) as a pressure or threat in Article 17 reporting for 20001-2006.

Broad Habitat Class	Pressure (per cent)	Threat (per cent)	Total number of records
Marine, coastal and halophytic habitats	25	25	351
Coastal sand dunes and continental dunes	36	37	258
Freshwater habitats	37	40	362
Temperate heath and scrub	30	31	134
Sclerophyllous scrub (matorral)	10	10	116
Natural and semi-natural grassland formations	27	29	416
Raised bogs and mires and fens	36	37	275
Rocky habitats and caves	18	19	276
Forests	21	22	583

guidance on prioritisation or weighting the relative importance (see comment in Section 2: this is to be addressed by the EC Expert Group).

Table 4.1 presents the proportion of records per broad habitat class across all Member States which have listed air pollution or eutrophication as a pressure to structure and function or as a threat to the future viability of the habitat. It is important to note that results also reflect other sources of eutrophication (and other nutrients for example phosphates) as well as atmospheric nitrogen deposition, particularly for habitats dominated by water and land-based sources such as marine, coastal and halophytic habitats and freshwater habitats.

Table 4.2 presents, by country, the proportion of habitat assessments for four broad habitat classes (as defined under Annex I of the Habitats Directive) which report air pollution or eutrophication as a pressure. These four broad habitat classes have been selected for illustration as they will tend to be dominated by atmospheric inputs (but not exclusively) of reactive nitrogen. These results can be compared to an estimate of risk from nutrient nitrogen deposition for each country, based on critical load exceedance in 2000 (EMEP domain) (CCE, 2008). The critical loads data incorporate all "natural ecosystem" area (as used by CCE, 2008), and care should be taken when comparing these with the columns presenting Article 17 assessment results which are presented as a proportion of the number of records per country which identify air pollution/eutrophication as a pressure (i.e. are illustrative of sensitivity and vulnerability) and are not illustrative of area. However, the table usefully shows that there are a number of countries where critical loads are exceeded over a substantial proportion of natural habitat, but where there are no records of air pollution or eutrophication being listed as a pressure (or threat – data not shown).

4.1.5 Identification and discussion of key issues

Introduction

In this paper, we have provided an introduction to conservation status reporting and have attempted, in so far that it has been possible, to provide examples of the methods used by a selection of countries to assess whether nitrogen deposition is a 'pressure' or 'threat', as well as an illustration

Table 4.2: Proportion (per cent) of assessment records for each Member State's Article 17 reports for 2001-2006, which show air pollution (code 702) or eutrophication (code 952) as a pressure for the broad habitat classes: forests; temperate heath and scrub; natural and semi-natural grassland formations; raised bogs and mires and fens.

Final column shows per cent of natural ecosystem area at risk of eutrophication based on critical loads exceedance in 2000 (CCE, 2008), this figure is not directly comparable with previous columns which show per cent of records not of area.

Country	Code	Proportion of assessments (per cent) showing air pollution or eutrophication as a pressure				Per cent 'natural ecosystem' area exceeding nutrient N CL in 2000
		Forests	Temperate heath and scrub	Natural and semi-natural grassland formations	Raised bogs and mires and fens	
Austria	AT	69 (32)	50 (4)	21 (24)	33 (15)	100
Belgium	BE	50 (20)	100 (4)	40 (15)	54 (13)	100
Bulgaria	BG	No data	No data	No data	No data	94
Cyprus	CY	0	0	25 (4)	0	68
Czech Republic	CZ	72 (25)	71 (7)	71 (21)	56 (9)	100
Germany	DE	86 (36)	33 (9)	23 (30)	64 (22)	84
Denmark	DK	0	100 (4)	44 (9)	85 (13)	100
Estonia	EE	0	0	0	13 (8)	67
Greece	EL	7 (27)	0	0	0	98
Spain	ES	6 (53)	0	6 (33)	47 (19)	95
Finland	FI	6 (17)	0	0	6 (16)	47
France	FR	2 (62)	16 (19)	30 (46)	44 (27)	98
Hungary	HU	0	0	0	20 (5)	100
Ireland	IE	0	33 (3)	0	0	88
Italy	IT	0	0	0	0	69
Lithuania	LT	15 (13)	0	0	0	100
Luxembourg	LU	0	100 (1)	29 (7)	100 (3)	100
Latvia	LV	11 (9)	0	0	14 (7)	99
Malta	MT	0	0	0	0	No data
Netherlands	NL	100 (7)	100 (2)	75 (8)	100 (7)	94
Poland	PL	28 (25)	25 (8)	37 (19)	0	100
Portugal	PT	0	0	19 (16)	0	97
Romania	RO	No data	No data	No data	No data	19
Sweden	SE	26 (35)	100 (8)	100 (31)	65 (23)	56
Slovenia	SI	0	0	11 (18)	0	98
Slovakia	SK	0	0	0	11 (9)	100
United Kingdom	UK	91 (11)	83 (6)	78 (9)	67 (9)	26*

**** UK figure is considered an underestimate (see Hicks et al., 2008) and national estimate is 61 per cent (Hall pers comm.). Number of assessment records is shown in parenthesis.***

of the results. However, in this collation, and in our own work for the UK's assessment of nitrogen deposition impacts, a number of issues and challenges have become apparent.

In this section we attempt to identify and summarise some of the key issues and challenges to assessing nitrogen deposition impacts on conservation status. These were discussed at the workshop and are reported in Whitfield *et al*., (this volume). A set of questions discussed at the workshop is given in Appendix 4.1.

Field evidence and confidence in attribution

Since historic/cumulative nitrogen deposition impacts should be evident in the current condition of habitats and their range and extent, consideration of the impacts is, in theory, implicit in conservation status assessments which are based on field surveys and monitoring. However, unless field sampling techniques are designed explicitly to do so, and are sufficiently representative to be scaled up, it is difficult to attribute nitrogen deposition effects and this can lead to significant under-reporting, or the reliance on risk assessment approaches such as critical loads. Nitrogen deposition impacts are particularly challenging to attribute because of the interplay between pollution impacts, management and abiotic and biotic stresses. Whilst there may be examples of some well researched sites where nitrogen deposition impacts can clearly be demonstrated and attributed, scaling this up to country level reporting and subsequently the biogeographic region is difficult. This leads to the question as to how confident we need to be to record something as a pressure or a threat and ultimately to engender a policy response?

Denmark specifically includes a range of nitrogen biomonitoring measures in conservation objectives and undertakes monitoring of these as part of representative sampling across habitats. This represents the most rigorous approach (on the basis of reports available at the time of writing) to assessing nitrogen deposition impacts on conservation status. However, there remain questions regarding the robustness of biomonitoring methods (Sutton *et al*., 2004; Leith *et al*., 2005; and see Nordin *et al*., this volume), in addition to significant resource implications if they were to be widely applied.

Two key topics for discussion are therefore (a) interpreting field evidence and the attribution of nitrogen deposition, and (b) use of bioindicators.

Use of critical loads

A number of countries have used critical loads exceedance mapping (with various adaptations) as a basis for assessing whether nitrogen deposition is a current pressure or future threat. This is unsurprising, and advantageous, since critical loads are an established tool (i.e. under the Convention on Long-Range Transboundary Air Pollution) and used routinely in European air pollution policy development. However, there are a number of issues concerning the application of critical loads and exceedance estimates. For example:

- They are a risk assessment tool and do not provide actual evidence of impacts (conversely this is useful for predictions of threats to future viability). There needs to be good confidence in the relationship between exceedance and effects on conservation status (e.g. structure and function, viability) of sensitive habitats and at present this is variable.
- Critical loads need to be assigned to Annex I habitats, since they are currently based on the EUNIS habitat classification. Many habitats will not have a 'relevant' critical load, others have a very weak equivalence with the habitats for which critical loads are set (which are often a lower EUNIS level). Furthermore, the research underpinning the 'relevant' critical load may be poorly indicative of impacts on a specific Annex I habitat.

- Countries' mapping of habitats for critical loads assessments may not correspond well with Annex I habitat mapping.
- Deposition modelling resolution varies and may not be appropriate for habitat/site level reporting.
- Critical loads are difficult to apply to species as they are habitat based and the relationship between habitat level responses and effects on species is complex.
- Dynamic models for nitrogen deposition impacts are under development and have been used by some countries to refine critical loads for Annex I habitats (and subsequently inform conservation status assessments). Their potential for a wider application in conservation status assessments should be discussed.

Defining impacts on structure and function and viability

For the conservation status of a habitat to be favourable, the assessment must show that "the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future". There is little guidance on the definition of structure and function. Nitrogen deposition potentially represents a pressure to this parameter, but the mechanisms for this need articulating. Furthermore, the relevance and appropriateness of bioindicators (including biochemical measures) and critical loads exceedance, as measures of impact of nitrogen deposition on this parameter, need to be considered.

The Directive defines when the conservation status of a habitat is to be considered as favourable. It requires that the range and area of the habitat should be at least maintained at their status when the Directive came into force or, where the status at that time was not viable in the long term, should be restored to a position where it would be viable.

Since there may have been significant changes in plant communities and species distribution prior to the Directive in areas exposed historically to high deposition, it is interesting to consider the requirement for recovery. This raises the question of what the objectives for recovery should be in order to fulfil the Directive's aims. It is unlikely that there is scope for highly aspirational targets in relation to conservation status i.e. a return to a former pristine state. Therefore, perhaps a more relevant question to consider is that of demonstrating the further/continuing risks to habitat viability. Understanding the impacts on habitat viability against a generally improving background deposition is an important consideration.

However, it is important to consider potential for recovery in the context of the 'future prospects' parameter. How should a declining background nitrogen deposition be accounted for even when critical load exceedance remains over large areas?

A further question to consider is whether there is cross linkage between conservation status assessment, and effects on structure and function, and consideration of ecosystem services and this will be considered in Theme 5 (see Section 7).

Definitions of threat and pressure

In the assessments of conservation status, Member States were required to list the main pressures and threats from a list in the EC guidance. However, for the 2007 reporting round there was no guidance as to how to judge which are the 'main' pressures and threats (i.e. how to prioritise), nor any on the definitions themselves. It is apparent that two categories, those of "air pollution" and "eutrophication", have been used in respect of nitrogen deposition impacts (and possibly "acidification" and "fertilisation"). However, eutrophication is also commonly used with respect to water quality issues. It is therefore difficult to untangle the various sources of nitrogen inputs and compare results, thus limiting the degree of analysis which is possible. Looking forward to the

next reporting round this is clearly an area which could be improved. This is being addressed for the next reporting round.

4.1.6 Conclusions

The results presented above illustrate that 'air pollution' or 'eutrophication' have been recorded as a pressure or threat on a significant number of habitat assessments across Europe. It is not possible to undertake a detailed analysis of this and examine the relative importance of specific pathways of pollutant inputs (e.g. for eutrophication this may be water, land-based or atmospheric inputs), or to compare to other pressures and threats, and thus draw out many useful conclusions. However, a focus on habitats which are only vulnerable to atmospheric inputs supports the case that nitrogen deposition is an important pressure to habitat structure and function and a threat to future prospects.

The Habitats Directive is a cornerstone of European biodiversity legislation. A robust assessment of the effects of nitrogen deposition on conservation status is necessary. In turn, this can be used as a driver for air pollution policy development and mitigation. Because of the transboundary nature of air pollution and the active policy agenda on this issue in the European Union, it would be reasonable to advocate that a consistent methodology for assessing nitrogen deposition impacts on conservation status be agreed and implemented.

There are common assessment tools such as critical loads, used for example in impact analysis and optimisation under the Convention on Long-Range Transboundary Air Pollution and the National Emissions Ceilings Directive. However, there is a need to strengthen the collaboration and establish a common set of objectives between the different communities working on nitrogen impacts assessment. It is recommended that the possibility of further work on improving/developing the use of critical loads, in the context of conservation status assessments, is explored.

This paper has presented the methodologies used by some Member States for assessing the effects of nitrogen deposition on conservation status. 18 Member States reported that 'air pollution' was a pressure or threat in at least one habitat assessment (and all 25 reported "eutrophication" for at least one habitat record). However, it was difficult to get access to information on the approaches that different countries used for this assessment. Despite large critical load exceedance, in many countries only a small proportion of sensitive habitats, or some cases none, were recorded as being affected by nitrogen deposition. This raises the question as to whether it reflects a low level of recognition of the pressure in many countries, or whether it reflects that the effects, which are evident from scientific research and indicated by critical loads exceedance maps, are not widely detected, and/or attributed, in the field at the broad scale.

In the previous section, we identified some of the issues and challenges concerning the assessment of nitrogen deposition on conservation status. Looking ahead to the next reporting round in 2013, the aim of the workshop session is to agree a focussed list of issues/challenges, to explore how they may be addressed and to provide recommendations for taking this forward, including how it could feed into the current review and improvement of the reporting guidance. This will include discussing scientific questions (for example, regarding field evidence and application of critical loads) and also exploring the mechanisms/routes for delivery and the potential organisations involved.

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Appendix 4.1 – List of questions discussed at the workshop

Priority questions in ***bold***

1. ***Why is an assessment necessary of nitrogen deposition as a 'pressure' or 'threat' in the assessment of conservation status under Article 17? Is a common approach across Member States needed?***

2. Pressures and threats list – definitions and recommendations. Capturing other air pollutants?

3. Evidence from survey and monitoring including:

a. How does N deposition effect habitat structure and function, and habitat viability.

b. How to measure/assess from field evidence:

   i. scaling from site to habitat/broad scale;

   ii. monitoring/surveillance approaches;

   iii. attribution of N as a causal factor (versus other multiple drivers);

   iv. use of bioindicators: 'exposure' indicators; 'effect' indicators – linking response to habitat structure and function or viability.

4. Use of critical loads including:

a. Relationship to structure and function.
b. Assignment to Annex I habitats – methods and challenges.
c. Habitat mapping issues.
d. Resolution of deposition mapping – suitable?
e. What proportion of habitat area needs to be exceeded to trigger inclusion as a significant pressure or threat?
f. What extent/proportion of exceedance is needed to trigger conclusion of unfavourable?
g. Assignment of critical loads to species – methods and challenges
h. Dynamic modelling – what potential does it offer. Development requirements.

5. Recovery (including level of ambition) and viability

6. Declining emissions/deposition – what does this mean for future prospects judgements?

7. What should be the approach for species listed in Annex II, IV and V of the Directive?

8. ***What is the process for developing this approach and subsequent guidance?***

4.2 Working group report

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4.2.1 Conclusions and recommendations of group discussions

It was concluded that nitrogen deposition represents a major threat to semi-natural vegetation across Europe. There is widespread exceedance of critical loads for nutrient nitrogen and acidification and substantial field and experimental evidence of the impacts. Such responses threaten the achievement of favourable conservation status for a large number of Annex I habitats.

It was concluded that the impact of nitrogen deposition on conservation status should be explicitly considered in Article 17 reporting, and the results should inform air pollution policy development.

It was concluded that there is a need for a common methodology for assessing the threat from nitrogen deposition to conservation status to be developed for application across Europe. This requires an improved dialogue between air pollution and biodiversity communities, building on recent progress in this area such as the development of a nitrogen deposition indicator under the Streamlining European Biodiversity Indicators (SEBI) programme.

It is recommended that a harmonisation of the methodology for nitrogen deposition assessment in conservation status reporting is required.

It is recommended that the lists of pressures and threats used for Article 17 reporting of conservation status should include nitrogen deposition explicitly and be more clearly defined.

It was noted that there is a requirement for greater clarity in the definition of 'favourable conservation status' for different habitats or groups of habitats, particularly with respect to defining important elements of structure and function. It is recommended that a series of habitat working groups should be established between interested Member States to take this forward.

It is recommended that the Working Group on Effects (WGE) of the UNECE Convention on Long -Range Transboundary Air Pollution (CLRTAP) and the Expert Group on Reporting under the Nature Directives should be brought together in order to develop a methodology for the assessment of nitrogen deposition impacts on conservation status. A two-tiered approach is recommended as the basis of further development:

- Tier 1: An assessment based on empirical critical loads for nutrient nitrogen deposited to sensitive Annex I habitats. This would build on the already established critical loads exceedance methodologies developed under the CLRTAP, but requires further development to apply the concept consistently to Annex I habitats of the Habitats Directive and to recommend the most appropriate deposition data. It would enable identification of nitrogen deposition as a "threat to future prospects" and also be used to help interpret species or biogeochemical based monitoring data in order to identify whether nitrogen deposition is a 'pressure to current structure and function'.
- Tier 2: Monitoring (likely to be non-mandatory) should be made up of biotic and abiotic variables to determine where nitrogen deposition is a significant pressure on structure and function. This would require agreement of abiotic and biotic variables/values relating to favourable conservation status and the production of a first set of European guidelines on this topic.

4.2.2 Introduction to structure of discussions

The Habitats Directive requires Member States to take measures to maintain at, or restore to, favourable conservation status, the natural habitats and species of Community Importance. Member States are required to report on conservation status every six years. This requires an assessment in respect of the range, area, structure and function and future prospects of habitats (including 'typical species'), and includes a consideration of the pressures and threats to their conservation status. Assessing the pressure or threat of nitrogen deposition impacts on conservation status is important if it is to inform air pollution policy development and to ensure that it is targeted appropriately to help achieve the objectives of the Habitats Directive. Therefore, the overarching objectives of Working Group 2 were to share experience and good practice with respect to the approaches taken in the 2007 reporting round and to discuss harmonising approaches and recommended methods for future reporting rounds.

At the start of the meeting, the members of the group agreed a list of questions or issues that were to be discussed during the meeting. An introduction to the issues and the context can be found in Whitfield and Strachan (this volume). The questions are listed below and those in bold were seen as a priority for discussion:

a. **Why is an assessment necessary of nitrogen deposition as a 'pressure' or 'threat' in the assessment of conservation status under Article 17? Should there be a common approach across Member States?**

b. Pressures and threats list – definitions and recommendations.

c. **Using evidence from survey and monitoring.**

d. **Use of critical loads/levels.**

e. **Recovery (including level of ambition) and viability.**

f. Declining emissions/deposition – what does this mean for future prospects judgements?

g. What should be the approach for species listed in Annex II, IV and V of the Directive?

h. **What is the process for developing this approach and subsequent guidance?**

To lead into the main discussion, group members were given the opportunity to give presentations concerning the approach taken in their country for Article 17 reporting, or to present the methods and results of research or monitoring which potentially could inform the assessments in the future. Some of the presentations are published as papers in this volume and their content is not recorded in this report of the meeting. However, some of the points raised and conclusions from the presentations are reflected in the discussion of the questions.

4.2.3 Highlights of discussion and views expressed

Why is an assessment of nitrogen deposition as a 'pressure' or 'threat' in the assessment of conservation status under Article 17 necessary? Should there be a common approach across Member States?

Nitrogen deposition represents a major threat to semi-natural habitats across Europe. It is agreed by scientists and national authorities that widespread eutrophication is responsible for ecological change and the loss of important taxa. Large areas of semi-natural/natural ecosystems including Natura 2000 sites, exceed the critical load for nutrient nitrogen and acidification (CCE, 2008). A number of recent studies have shown a decline in species richness of habitats related to nitrogen deposition (for example, Stevens *et al*., 2004; Maskell *et al*, 2010, Van Hinsberg *et al*., 2008, Dupre *et al.,* 2009). Many natural habitats and species of conservation interest thrive in nutrient poor habitats (e.g. BfN 1998, Bunce *et al*., 1999, Ellenberg *et al*., 2001, Preston *et al*., 2002, Braithwaite *et al*., 2006, EEA 2007; Dias *et al*., this volume) and conservation of many keystone species relies on reducing nitrogen loads on the long term. Nitrogen deposition rates are predicted to remain high in Europe for the coming decades.

The Habitats Directive is a cornerstone of European biodiversity policy, it is important that the effects of nitrogen deposition are considered in relation to the Directive's objectives and this should influence the development of air pollution policy. However, if there is an under-reporting of the scale and threat of nitrogen deposition, this will serve to weaken the drivers for further policy in this area.

A common approach is necessary to provide a consistent and comparable assessment across Europe. However, in establishing guidelines, it must be recognised that Member States are likely to have access to different levels of capacity with respect to assessment tools.

There are several bodies working on air pollution impacts in Europe, including the Working Group on Effects (WGE) of the Convention on Long-Range Transboundary Air Pollution (CLRTAP) and work undertaken for the review of the National Emissions Ceilings Directive. There has been some coming-together of groups for the development of indicators of nitrogen deposition impacts e.g. through the Streamlining European Biodiversity Indicators (SEBI) 2010 programme (EEA, 2007). There is recognition within CLRTAP that improved links to biodiversity research are needed, and the 19th workshop of the Coordination Centre for Effects (CCE) recently recommended that the WGE explore the applicability of a European scale of indicators for damage to biodiversity. Nonetheless there needs to be an improved dialogue between the air pollution communities and those working on biodiversity policy such as the Habitats Directive, in order to ensure the evidence of nitrogen deposition impacts is fully considered in relation to assessments of conservation status.

List of Pressures and threats – definitions and recommendations
There was not sufficient time to discuss this in detail and provide recommendations on categories and definitions. It was noted that there is currently no guidance on the use of the categories and their definitions, but that this is being reviewed. It was evident that Member States had used different categories to represent nitrogen deposition in the 2007 reporting round. Nitrogen deposition could potentially be recorded under 'air pollution', 'eutrophication', 'acidification', or 'fertilisation'. There was also concern that there was no way to prioritise or scale the various pressures or threats. The consensus of the group was that there needs to be a single category to which nitrogen deposition is clearly attributable. It is important that the categorisation can be used to inform subsequent policy response, and this needs to be borne in mind when combining pressures/threats (i.e. nitrogen deposition needs to be explicit).

Using evidence from survey and monitoring
Nitrogen deposition can affect the structure and function of sensitive habitats. Excessive inputs may impact sensitive species directly; may change species composition through altering competitive interactions or the soil chemical environment; and may increase susceptibility to other abiotic/biotic stresses such as disease and herbivory. Direct toxic effects are greater from dry deposition than from wet nitrogen deposition, and from reduced than from oxidised forms of nitrogen for the same relative dose. The different types of deposition also affect competitive interactions, since different plant species are adapted to using nitrate, ammonium or dissolved organic nitrogen. A major effect of increased N deposition is to increase the growth of competitive, tall-growing species, and hence species most at risk are those requiring high levels of light at ground level such as short-growing plants and associated invertebrates, other nutrients and water stress may play an important role (sections 5.1 and 5.2).

The group discussed a number of challenges regarding how to interpret evidence from surveys and monitoring in terms of identifying impacts of nitrogen deposition on conservation status. This ranged from the difficulty of attributing nitrogen effects from species-based monitoring, applying biomonitoring techniques and scaling up site-specific measurements to habitat level assessments.

One of the difficulties in defining the assessment and any monitoring that may be required is the need for more clarity on the definition of favourable conservation status for different habitats or groups of habitats. In particular, what are the important elements of structure and function? Nitrogen deposition impacts could then be related to these conditions if they were defined. However, it was highlighted that this element had previously been left open to Member States. It was recommended

that a series of habitat-specific working groups should be established between interested Member States to take this forward.

A range of bio-indicators are available. The topic was discussed by Working Group 3 (Bobbink and Hettlelingh, 2010) and so individual methods were not discussed in detail in Working Group 2. It was agreed that when considering the use of bio-indicators and results from vegetation surveys it is necessary to differentiate whether the method is indicative of 'exposure' (which might help to refine deposition/concentration estimates in relation to the critical load/level or to verify exposure estimates in spatially variable areas) or 'response' (in terms of elements of structure and function). With respect to 'response' it was agreed that interpreting cause and effect from species level changes is extremely hard because of multiple drivers. Biogeochemical responses may have more potential with regards to attribution, but the relationship between the abiotic and biotic variables and conservation status is poorly defined. However, Denmark have defined this for various habitats and response variables and they are monitored through the NOVANA network (Svendsen *et al*., 2005).

It is difficult to prescribe response variables and monitoring requirements because of a lack of a definition of structure and function for different habitats, and because the approach taken in Member States is variable. Any bio-indicators should be practical, simple, robust, specific and cost-effective. Sampling and vegetation relevées need to be representative and with sufficient statistical power to detect and attribute pressures. Interpreting species responses, which can be influenced by many factors, is difficult, therefore biogeochemical measurements are also recommended (particularly when considering the broad scale effects where there are multiple sources of nitrogen). However, requiring additional mandatory sampling of nitrogen response variables is unlikely to be accepted by Member States.

Use of critical loads exceedance

Critical loads (and levels for concentrations) are established tools for air pollution policy development (i.e. under the CLRTAP). The CCE (CCE, 2008) maps critical loads and exceedance across European countries. Each of the Member States which undertook an explicit assessment of nitrogen deposition impacts on conservation status in the 2007 reporting round had used critical loads, in some form, as part of their assessment methodology. The working group agreed that critical loads and critical loads exceedance should be an element of any recommended methodology. The group discussed some of the issues and questions regarding the use of critical loads (e.g. as raised by Whitfield and Strachan, this volume) with respect to assessing nitrogen deposition as a pressure on current structure and function and a threat to future prospects, as follows.

a. Relationship to structure and function. Critical loads exceedance mapping is a risk assessment tool to identify areas or habitats where there may be adverse effects from nitrogen deposition at some point. The principal focus of Working Group 2 was eutrophication effects. However, it was noted that nitrogen deposition also contributes to acidification and some studies (e.g. Dupre, 2009) have identified pH and nitrogen deposition as the main variables explaining declines in species richness. It was agreed by the group that critical loads were a suitable assessment tool for predicting effects on the functioning of habitats. There are a number of uncertainties in critical loads and estimates of exceedance. One significant limitation is the lack of temporal definition. Empirical nutrient nitrogen critical loads are protective for 20-30 years and this is consistent with the timeframe for the 'future prospects' parameter (European Commission, 2006). Exceedance of the critical load infers that damage will occur at some point, but does not indicate when. In the case of nitrogen deposition, effects may have occurred before the implementation of the Habitats Directive, particularly in high deposition areas. Thus

problems can arise with regard to a reference year. The group were confident in the application of critical loads exceedance as a tool to identify nitrogen deposition as a threat to 'future prospects' (see recommendations below concerning improving the application of critical loads to Annex I habitats). Information on critical loads exceedance may also be a useful aid to interpreting species or biogeochemical responses. However, there was less confidence in the application of critical loads exceedance in respect of assessing effects on current structure and function, in the absence of corroborative evidence from species composition/structure data and/or biogeochemical indicators. For example, if the species composition and structure of a habitat are both judged to be favourable, is critical load exceedance sufficient to turn assessment for the parameter "structure and function" to unfavourable-inadequate or unfavourable-bad?

b. Assignment to Annex I habitats – methods and challenges There was general concern that countries currently use different methods to generate nitrogen critical loads (e.g. calculated (mass balance), empirical and dynamic models) as set out in the CCE Mapping Manual (UBA, 2004). It was recommended that the empirical critical loads (UNECE, 2003) are the most applicable in respect of effects on structure and function, since many are set on the basis of species change end points or ecosystem processes *vs.* critical nitrogen concentration in soil for calculated nitrogen critical loads.

However, empirical critical loads are set for habitats classified under EUNIS and this requires a conversion to Annex I habitats. There are examples from the UK (Bealey *et al*., 2007), Netherlands (see van Hinsberg and van Dobben, this volume), Denmark (Svendsen *et al*., 2005) and the German Land of Brandenburg (LUA, 2008). This conversion introduces possible inconsistencies. Furthermore, some Annex I habitats do not have a relationship with any of the EUNIS classes for which a critical load is set and so it is not possible to assign a critical load. Ideally empirical critical loads should be developed with favourable conservation status as an endpoint.

The group recommended that the proposed review of critical loads includes the assignment of empirical critical loads for nutrient nitrogen to Annex I habitats with the concept of favourable conservation status as an 'end-point'. This could include the following steps:

- Identify which Annex I habitats are potentially sensitive to nitrogen enrichment (from nitrogen deposition).
- Assign 'relevant' empirical nitrogen critical load to Annex I habitat types.
- Provide information on the confidence in this allocation where based on comparison of EUNIS and Annex I habitat classifications.
- Where it is not possible to assign a critical load provide recommendations on sensitivity if possible.
- Provide further guidance on applying the modifying factors to help steer which part of the range might be more applicable for certain conditions/locations/management regimes.

An update of the UNECE 2003 empirical critical loads for nutrient nitrogen has subsequently been conducted (Bobbink and Hettelingh, 2011).

c. Habitat mapping issues. Digital mapping of Annex I habitat distribution is required in order to map associated empirical critical loads for nitrogen and exceedance. However, currently these are not available for all countries. The Explanatory Notes (European Commission, 2006) recommend using grid based data, typically at 10 km scale to estimate

range and these could be used to map critical loads. It was reinforced that the concept of conservation status is not confined to Natura 2000 sites since the habitat resource outside the sites may make a significant contribution to the overall status (the extent of which varies on a case-by-case basis). This aspect would need further consideration when refining the approach for mapping critical loads for Annex I habitats.

d. Resolution of deposition mapping – suitability. There was concern over the resolution of deposition mapping (EMEP = 50x50km grid; but 10x10km grid under development) which misses much of the sub-grid variation of nitrogen emissions and deposition and the patchy distribution of Natura 2000 sites. This is particularly important with dry deposition of ammonia and for assessments at the site or habitat scale. Because of this relatively low resolution, results may differ to those from national-scale higher resolution models which provide a more representative estimate of exceedance at a more appropriate resolution (e.g. 5x5km, 250x250m). This may lead to an under- or overestimate of the habitat area exceeded when using deposition from the EMEP model.

e. What proportion of habitat area needs to be exceeded to trigger inclusion as a significant pressure or threat? And what extent/proportion of exceedance is needed to trigger a conclusion of 'unfavourable'? It was agreed that guidance on this would be necessary, but no recommendations were made at the meeting. It was noted that the guidance for the 2007 reporting round (European Commission, 2006) stipulated the "unfavourable-bad is where more than 25 per cent of the areas of the habitat is unfavourable as regards its specific structures and functions (including typical species)". An example is given as "by discontinuation of former management, or is under pressure from significant adverse influences, e.g. critical loads of pollution exceeded".

f. Assignment of critical loads to species – methods and challenges. Not discussed due to time constraints.

g. Dynamic modelling – what potential does it offer. National Focal Centres are being encouraged to use dynamic models by the CCE. Netherlands have used them to refine/develop critical loads for Annex I habitats (see van Hinsberg and van Dobben, this volume) and such an approach was supported by the group. There was some discussion over the requirement to develop end points directly related to impacts on a habitat's species (or specifically 'typical species') (see Rowe, *et al.*, this volume) and how these could be defined. Models predicting environmental suitability for plant species are available (de Vries, *et al.*, 2010), but to make use of such forecasts it is necessary to define which species are important for the conservation status of each Annex I habitat. The Habitats Directive Interpretation Manual (European Commission, 2007) lists characteristic species for different habitat types, but these are not necessarily the most appropriate to use in many cases. Member States had been required to identify typical species for each habitat at a national/regional level as part of the 2006 Article 17 reporting, based on guidance in the Explanatory Notes (European Commission, 2006). This had not been straightforward to do and there was a risk of circularity in their use. There was debate as to whether it is necessary to predict effects on species 'end points', raising the question of how to define which species (or species attributes such as cover) are critical to an Annex I habitat, and by inference its conservation status; or whether a biogeochemical measure such as C:N is sufficient, as this represents effects on ecosystem processes, which are an instrumental part of the concept of structure and function.

Recovery and viability. Since nitrogen deposition in many parts of Europe has exceeded critical loads for many decades, it may be expected that changes have occurred to habitats, for example with a loss of sensitive species, prior to the Directive coming into force. This led to the question of what the ambitions for recovery should be in the context of the objectives of the Directive. It was agreed that any assessment of recovery or ongoing effects on viability needs to define a desired state. This could define a habitat in terms of a desired species composition or biogeochemical status and that would be the basis of defining whether the status is favourable. Whilst in high deposition areas some sensitive elements may have been lost and the objective (in the context of the Habitats Directive requirements) may not be for their recovery, it may still be possible to show that ongoing high nitrogen inputs are affecting the ecosystem functioning by failing to sustain the low nutrient conditions essential for the supporting processes of a viable Annex I habitat.

Declining emissions/deposition – what does this mean for 'future prospects' judgements?
There was not time to discuss this in detail as priority was given to other questions. Discussions led on from those on recovery and viability and in relation to future prospects and also in relation to critical loads (see Section 5.2). Even where deposition is decreasing, it still poses a risk of harmful effects where it exceeds the critical loads. Furthermore, even where deposition has fallen below the critical loads, this does not mean that habitats will have recovered (see Figure 4.2 and associated text for fuller explanation of this point).

It was noted that in Spain emissions of some pollutants, for example ammonia, have been increasing over recent years and in Portugal ammonia emissions are stable. Therefore, it is not the case that there is reduction in reactive nitrogen emissions across all countries.

What should be the approach for species?
There was not time to discuss an approach for assessing nitrogen deposition impacts on conservation status of species (Annex II, IV and V). There was a consensus that it should be considered, but is likely to be more complex than for habitats, which should be the first priority.

What is the process for developing this approach and subsequent guidance?
Having considered the potential use of field evidence, monitoring, critical loads/dynamic modelling, and issues surrounding the timing of impacts, the group discussed how this could fit into a framework for conservation status assessments.

The recommendations were for a two-tiered system:

Tier 1:An assessment based on empirical critical loads for nutrient nitrogen applied to sensitive Annex I habitats. This would build on the already established critical loads exceedance methodologies developed under the CLRTAP, but requires further development to apply the concept consistently to Annex I habitats of the Habitats Directive and to recommend the most appropriate deposition data. It would enable identification of nitrogen deposition as a "threat to future prospects" and also be used to help interpret species or biogeochemical based monitoring data in order identify whether nitrogen deposition is a 'pressure to current structure and function'.

Tier 2: Monitoring (likely to be non-mandatory) should be made of biotic/abiotic variables to determine where nitrogen deposition is a significant pressure on structure and function. This would require agreement of abiotic and biotic variables/values relating to favourable conservation status and the production of a first set of European guidelines on this topic.

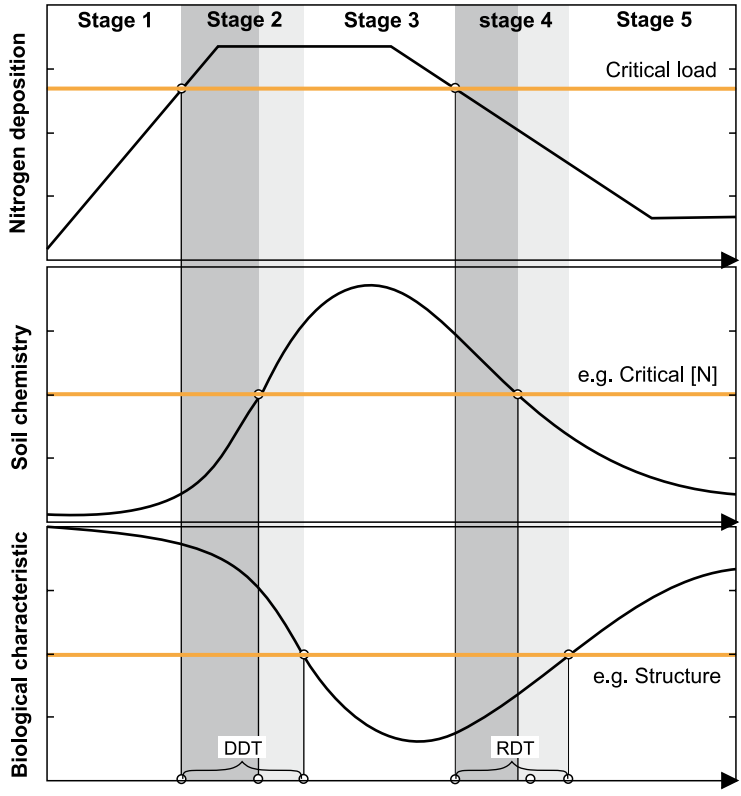

Figure 4.2: Possible stages of influence of N-deposition on conservation status. Adapted from CCE[1] [DDT = Damage Delay Time, RDT = Recovery Delay Time]

1 http://www.mnp.nl/en/themasites/cce/methods_and_models/dynamic-modeling/index.html

The discussion and conclusions of the group are explained below. Figure 4.2 is provided to support this and help with the conceptual understanding.

Five stages can be distinguished (see Table 4.3):

1 Nitrogen deposition is and has been below the critical load for a long period. Nitrogen deposition related chemical and biological variables (e.g. litter and soil C/N-ratio, nitrogen availability in the soil, presence of typical nitrogen-sensitive species) associated with favourable conservation status, are not influenced by deposition. As long as deposition stays below the critical load, the habitat is, with respect to nitrogen deposition, in favourable conservation status.

2 Deposition is above the critical load, but the critical chemical and biological variables are not yet violated. However, due to exceedance of the critical load the chemical and biological conditions are changing. The occurring changes are within the natural range of favourable conservation status, but risk of future negative effects on conservation status (i.e. changes in vegetation structure and functioning) are present. We call the time between the first exceedance of the critical load and first violation of the critical criteria the Damage Delay Time (DDT). This stage can be subdivided into two sub-stages: a

Table 4.3: Suggested framework for conservation status assessment

Stage	Critical load exceed-ance	Chemical deterior-ation	Biological deterior-ation	Conservation status with respect to nitrogen deposition	Measures Required
1	No	No	No	Favourable	Normal management
2a	Yes	Starting	No	Risk for future effects	Reduction in deposition
2b	Yes	Starting	Starting	Risk for effect in the near future	Reduction in deposition
3	Yes	Yes	Yes	Unfavourable	Reduction in deposition and/or removal of nitrogen required
4a	No	Ending	Yes	Unfavourable, with (good) prospects for the future	Further reduction in deposition and/or removal of nitrogen would speed up recovery
4b	No	Ending	Ending	Unfavourable with (good) prospects for the near future	Further reduction in deposition and restoration would speed up recovery
5	No	No	No	favourable	Normal management

stage where chemical changes occur and a stage where biological changes can also be observed. Besides the monitoring of any future changes, measures have to be taken to avoid damage to favourable conservation status in the future.

3 The deposition is above critical load and both the critical chemical and biological criteria associated with favourable conservation status are violated. Further measures have to be taken to avoid a (further) deterioration of the ecosystem.

4 Deposition is below the critical load, but the chemical and biological criteria are still violated due to the earlier exceedance. Recovery has not yet occurred. We call the time between the first non-exceedance of the critical load and the subsequent non-violation of both criteria the Recovery Delay Time (RDT). Like Stage 2 this stage can also be subdivided. Recovery Delay Time can be shortened by reduction of deposition below critical load or restoration management.

5 This stage is similar to Stage 1. Deposition is below the critical load and both criteria are no longer violated. Only at this stage can one begin to speak of full ecosystem recovery, although retention of nitrogen within the ecosystem and dispersal limitations may mean that sensitive species do not return for some time.

As shown above, risks and effects of nitrogen deposition can be best shown if information is available from monitoring deposition (and therefore having an accurate measure of exposure and critical load exceedance) and monitoring relevant abiotic and biotic variables (i.e. response: this would need to be over a representative sample of the habitat). Relevant abiotic variables are those variables influenced by nitrogen deposition and related to favourable conservation status. The guidelines[1] for reporting on the monitoring and modelling of air pollution effects name some of the relevant variables like total soil N, total carbon / N ratio, available N content and

1 http://www.unece.org/env/documents/2008/EB/EB/ece.eb.air.2008.11.e.pdf

pH for terrestrial ecosystems. However, whilst pH is a good indicator of acidification, there is less consensus on biogeochemical indicators of eutrophication. Bulk soil N and C/N may lag behind changes to vegetation processes. Numerous measures of soil available N and plant tissue assays have been proposed as indicators of nitrogen exposure. For aquatic ecosystems, variables such as nitrate concentration, acid neutralizing capacity (ANC), pH, alkalinity, aluminium concentration ([Al]) and total organic carbon (TOC) are relevant. Several countries, such as Denmark (Svendsen *et al*., 2005), Germany (VDI, 2008) and Netherlands (van Hinsberg and van Dobben, this volume), have already listed variables and have established methods to determine the adverse effects of nitrogen deposition in various ecosystems and in some cases defined critical values for favourable conservation status. Ideally, there would be collaboration between the Habitats Directive community (i.e. currently the Expert Group for Reporting under the Nature Directives) and the WGE of CLRTAP to agree on abiotic and biotic variables and values relating to favourable conservation status in the different EU biogeographic regions. Monitoring the difference between current values and desired values can be used to determine the influence of nitrogen deposition. In many European countries, such information is available for aquatic ecosystems. Such values have been used to compute habitat-specific critical loads by the use of dynamic ecosystem modelling. Examples of such approaches have been described by De Vries *et al*., (2007) and are used in the Netherlands. Biological monitoring is also very important for determining conservation status and the influence of nitrogen deposition on that status. In each Member State, habitats have to be described in terms of typical species and vegetation composition and structure. Since each species occurs in its own environmental niche, the occurrence of species gives information on the abiotic and biotic conditions. Some species can for example only be found in nutrient poor conditions in open vegetation (e.g. some nitrogen sensitive mosses or nitrogen sensitive herbs in dune grasslands). However, whilst much has been done within the European Vegetation Survey (http://www.iavs.org/part_groups_euroveg.asp) to define characteristic species of habitats and many national lists of habitats and their species composition exist, a clear guideline on how it can be used for nitrogen assessment is missing. Monitoring changes in species occurrence, frequency and cover (vegetation relevées) can, in combination with abiotic monitoring, give important information on changes caused by nutrient enrichment over time. However, this is complicated by interactions with other factors, such as management, and in practice it is often very difficult to attribute the cause or the relative importance of multiple factors. Furthermore, there may be cases where management is 'holding the line' i.e. suppressing nitrogen impacts.

Monitoring is expensive and time consuming and often not (yet) available. At the same time the number of experts who are able to determine the keystone taxa is decreasing. Without information on nitrogen deposition and its effects on Annex I habitats it was recommended that estimated critical load exceedance should be used as a first indicator as to whether to record nitrogen deposition as a threat to conservation status. This is especially true in those situations where critical load exceedances are high and exceedances have occurred for several years.

It was agreed that there were outstanding areas which need further development before a more specified approach can be recommended. Nitrogen deposition should be included explicitly in the lists of 'pressures' and 'threats'. Recommendations regarding the review of nutrient nitrogen empirical critical loads and their application to Annex I features were made and have since largely been addressed in Bobbink and Hettelingh, 2011. It was thought that such an exercise is relatively straight-forward and achievable. Furthermore, it has been identified that there should be a collaboration between the Habitats Directive community and the WGE to agree abiotic and biotic variables/values relating to favourable conservation status. Member States would then be free to use these in a monitoring programme and/or for development of dynamic models. These recommendations should be highlighted in the workshop synthesis report and workshop

summary information and raised with the Commission and relevant groups including the Expert Group on Reporting under the Nature Directives developing guidance for the 2013 reporting round.

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4.3 Setting critical loads for Dutch Natura 2000 sites using empirical information and dynamic modelling

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Summary

- In the Netherlands, critical loads were set for all Annex I habitat types using both internationally accepted empirical critical loads and dynamic ecosystem models.
- Habitat specific critical loads could be modelled using information on chemical soil conditions and plant species composition in habitats at favourable conservation status.
- The calculated critical loads have been adopted by the Dutch Government, and used for calculating threats to biodiversity and risks of significant negative effects on conservation status.

4.3.1 Introduction

Nitrogen deposition levels in the Netherlands are among the highest in Europe. Assessments have indicated that high deposition levels have negatively affected Dutch flora and fauna. For example, research using field data has shown that significant negative correlations between nitrogen deposition and occurrence of protected bird, plant and butterfly species exist (Van Hinsberg, *et al*., 2008). This paper describes how current scientific knowledge was used to derive critical loads of nitrogen deposition for Annex I habitat types in the Netherlands. The term 'critical load' here refers to the level of nitrogen deposition above which the risk of significant damage to the quality of a habitat type cannot be excluded. This is in close accordance which the internationally accepted definition, namely, 'a quantitative estimate of 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 & Grennfelt, 1988). The use of critical loads in assessing risks for significant damage is very straightforward. A nitrogen critical load can be compared to the current or future nitrogen deposition in order to obtain insight into the threats for eutrophication and/or acidification. If the atmospheric deposition at a location is higher than the critical load for a specific, existing (or desired) habitat type, then there is a clear risk of significant negative effects. In other words, the conservation objectives may not be achieved. The greater the critical load exceedance, and the longer the duration of this exceedance, the higher the risk of undesirable negative effects on habitats. Exceedances of the critical load of nitrogen have been used in European pollution abatement policy for defining emission-reduction targets, namely, in the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP Convention) and the European Union (National Emission Ceilings Directive). The exceedance of critical loads of nitrogen is also used as an indicator for risk of biodiversity loss by the European Environment Agency (EEA, 2007).

4.3.2 Aims and objectives

In 2008, a study was conducted for setting habitat specific critical loads for all Annex I habitat types which occur in the Netherlands, based on the latest scientific knowledge on thresholds and conservation status.

4.3.3 Results and discussion

Within the LRTAP Convention covering the UNECE region, procedures have been developed to set and map critical loads for airborne nitrogen deposition. Based on the UNECE mapping manual of the International Cooperative Programme on Modelling and Mapping and recent scientific

publications (e.g. De Vries *et al*., 2010) on dynamic modelling, the following methods were identified as being important for setting critical loads:

- *The empirical method.* Empirical critical deposition loads have been published for international use, most recently on the basis of a workshop, held in 2010 to review critical loads under the UN-ECE Convention on Long-range Transboundary Air Pollution (Bobbink and Hettelingh, 2011) . This source contains critical loads derived from field experiments in combination with indication of the inputs. Based on available information, this method yielded ranges of critical loads for 21 broadly defined ecosystem types (based on the EUNIS classification system). The ranges describe the variation in critical loads due to variation within the EUNIS ecosystem type, variation caused by abiotic conditions which vary across locations and uncertainties. Furthermore, when data are insufficient for setting critical load values for specific ecosystems, it is suggested to use the lower, middle or upper part of the ranges, depending on general relationships between abiotic conditions and critical loads (Table 4.4).
- *Dynamic ecosystem modelling*. Ecosystem models can be used for calculating critical loads (De Vries *et al*., 2010). Results from the SMART2 model are available for the majority of vegetation types in the Netherlands (Van Dobben *et al*., 2006). In addition, for some ecosystems, specific models have been developed (AquAcid for heathland pools, *Calluna* for dry heaths).
- *Expert judgement*. In addition to empirical evidence, expert judgement has been used for setting critical load.As described in Bobbink *et al*., (2003), information on abiotic conditions can be used for setting specific critical loads for sub-ecosystems within the broad EUNIS ecosystem types for which critical load ranges are known (Table 4.4). In the Netherlands it

Table 4.4: General relations between abiotic conditions and critical loads, which can be used for deciding whether the lower, middle or upper part of a critical load range is applicable for a particular ecosystem type *After Bobbink et al., 2003 **

Action	Temperature/ Frost period	Soil wetness	Base cations availability	P limitation	Management Intensity
Use lower part	Cold/Long	Dry	Low	N-limited	Low
Use middle part	Intermediate	Normal	Intermediate	Unknown	Usual
Use higher part	Hot/None	Wet	High	P-limited	High

** updated in 2010 (Bobbink et al. 2010)*

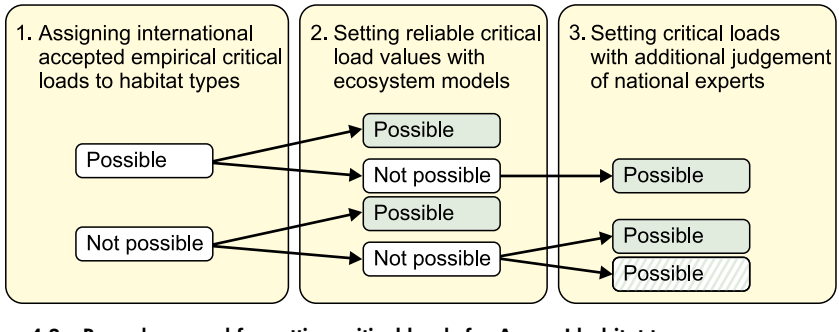

Figure 4.3: Procedure used for setting critical loads for Annex I habitat types.
In most cases, reliable critical loads could be set after steps 1 and 2 (green boxes). In some cases, additional judgement from national experts was used for setting reliable or quite reliable critical loads (green boxes), or even to give a best possible estimate (striped box).

was attempted to derive unique critical load values per Annex I habitat type within the ranges per EUNIS type as in Bobbink *et al*., (2003) by combining the empirical and simulation approaches, and using expert knowledge where necessary (Van Dobben & Van Hinsberg, 2008). The procedure that was used is depicted in Figure 4.3.

The procedure yielded critical loads for most Annex I habitat types. In most of the cases, empirical ranges and/or reliable model estimates were present, and critical loads could be set based on published information. In about 70 per cent of all the habitats, models yielded critical load values within the given empirical range. In those cases where simulated critical loads were outside the empirical range, the critical load was set to the nearest extreme of the empirical values. The difference between the middle of the empirical range and the modelled values per habitat type was on average less than 1 kg $ha^{-1}yr^{-1}$.

4.3.4 Conclusions and discussion

In most of the cases, empirical ranges and/or reliable model estimates were present and critical loads could be set for the vast majority of habitat types.

The set critical loads can be used for calculating exceedances, which indicate future threats or present negative effects on habitat structure and/or functioning. Monitoring data on nitrogen-related abiotic and/or biotic conditions are needed to show whether negative effects are already occurring.

The detailed habitat specific critical loads can only be used together with detailed maps of habitat occurrences and detailed deposition maps.

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4.4 Monitoring terrestrial habitat types in Denmark

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Summary

- The terrestrial programme, which started in 2004, includes a systematic and representative monitoring of terrestrial habitat types as well as species. This presentation will focus on concepts for monitoring of habitat nature types. The habitats are exposed to a large number of pressures like eutrophication, change in land use, fragmentation, climate change etc. Some of these changes are anthropogenic, others are attributable to natural development, but are, nonetheless, influenced by man. The effects of such pressures are reflected in the overall criteria defined in the directive - area, structure and function of the habitats. The 28 semi-natural habitat types and 10 forest types included in the present programme consist of approximately 1200 randomly selected monitoring stations within Natura 2000 areas as well as areas outside of Natura 2000 in order to give a representative picture of the conservation status. Each station has 20-60 sampling points defined by UTM coordinates.
- To understand the causes behind observed changes in the habitats, it is necessary to combine both biotic and abiotic criteria. The ambition of the monitoring programme is to "bridge the gap" between traditional biodiversity monitoring and monitoring of cause indicators.
- The favourable conservation status is defined on the basis of a number of threshold values which have proven important in ecological research and other monitoring programmes. The selected parameters are: coverage of the vegetation (point intercept method), lichen/moss ratio, nitrogen content in lichens, mosses and shoots of dwarf scrubs (reflector of short-time changes), C/N ratio and pH of the upper organic soil horizons (long-term impacts). Nitrate is measured in some water, dependent terrestrial habitat types (e.g. rich fens, raised bogs and springs) together with monitoring of water tables. As often as possible, a value, or interval, that identifies a favourable conservation status, is attached to each of these parameters – mainly based on literature values.
- The conservation status of a specific habitat depends on many parameters, and some of these are measured in the monitoring programme. The aggregation of the measured indicators into an overall assessment of the conservation status of the habitant is a non-trivial task. However, research on different multi-criteria approaches to assess conservation status has been initiated.

4.4.1 Aims and objectives

In the Natura 2000 areas, Member States must undertake surveillance of the conservation status and take the appropriate steps to maintain or restore favourable conservation status of the habitats. The overall concept behind the "favourable conservation status" (FCS) comprise criteria for area, structure and function. The national task is to translate these general criteria to operational criteria

Table 4.5: Criteria for favourable conservation status on local/site level for the habitat type 2130 – the grey dunes. Indicators marked with (P) are pressure indicators.

Type 2130	Property	Unit of measurement	Criteria	Comments
Area	Area (hectares)	Number of hectares	Stable or increasing	
Structure and function	Naturally low nutrient level	Nitrogen deposition (kg/N/ hectare/year)	Not exceeding the critical load	The critical load 10-20 kg/N/ hectare/ year, UNECE 2003
	Naturally low nutrient level	Nitrogen content (mg/g) in Cladonia portentosa. Damages on foliage leaf are observed by N >8 mg/g and by N=13 mg/m lichen are dying	Within the natural range of the habitat type in Denmark. Stable or improving	Should be less than 6 mg/g. level in countries without N-load 2-4 mg/g, in Denmark 5.3-9.6 mg/g, lowest in Western Jutland, highest in Mid-Jutland
	Acidity	pH	The pH must be stable and not considerably lower than the natural acidity of the locality.	If no historical information is available, the natural pH can be predicted by model
	(P) Mechanical impact	Proportion of area influenced by wear and tear from e.g. tourism	Stable or decreasing	Should not exceed 10 per cent
	Open, herbal dominated vegetation	Coverage of non-indigenous trees and bushes	Stable or decreasing	Overgrowth is partly due to seed-pressure from plantations and invasive species. Mountain pine, dune pine, Norway spruce and Japanese rose should be removed
	Cryptogams	Lichen/moss-ratio in grey dune	Within the natural range of the habitat type in Denmark. Stable or improving	Should be higher than 3:1. The grey dune is characteristic of a rich lichen flora. The criterion is preliminary, but studies have shown that eutrophication is increasing the proportion of mosses
	Species composition of plants	Deviation from the species composition of this habitat type in reference condition	The deviation is within the expected variation of the natural habitat type in Denmark	The species composition is a diversity indicator of changes in the environment factors
Characteristic species	Population of characteristic species	Index of populations of characteristic species present	Long-term maintenance on a stable or increasing level	Register by species, e.g. using the DAFOR scale. Variations are natural. In special cases declines may be acceptable / targeted.

and to implement surveillance of the habitats and species in order to follow the direction of the conservation status – which should be stable or improving (Søgaard *et al*., 2007).

No efforts have been made to reach a common European agreement on how to define FCS for the different habitat-types and species. In Denmark, the suggested criteria are an important background for monitoring, planning, and managing nature, and for carrying out assessments of potential setbacks or disturbances to the quality of the habitat within the specific areas. Table 4.5 is an example of criteria for habitat-type 2130; the grey dune heath.

We have defined structural features to include vegetation structure and composition, i.e. spatial distribution, age structure and biomass. Functional features include processes related to nutrient content and cycling. The starting point in defining FCS has been to list the different types of pressures affecting the different semi-natural habitat-types and forests. Natural and semi-natural habitats in Denmark are exposed to a large number of pressures like eutrophication, change in land use, fragmentation of habitat, drainage and invasive species. Some of these causes are anthropogenic; others are attributable to natural development. The effects of such pressures are reflected in the structure and function of the individual ecosystems/natural habitats, including the size of the nutrient pools, water table, etc. Terrestrial natural habitat monitoring aims not only to provide information about status and trends, but also to provide insight into both the natural and the anthropogenic pressures that is necessary in order to be able to carry out appropriate management.

4.4.2 Criteria of favourable conservation status

Criteria relevant indicators/properties for the habitat type in question with sets of specific values or intervals needed to be fulfilled to obtain favourable conservation status. These habitat type-specific criteria are based on the following:

- should form the basis for development of adequate monitoring leading to assessment of the conservation status
- should be scientifically based, biologically relevant and lead to the wanted state of conservation, i.e. support the goal for the monitoring
- should be simple and easy to understand, i.e. based on scientific justifiable simplifications
- should be operational, quantitative, objective and reproducible
- should be transparent – robust and precise
- should be sensitive enough to detect changes within a short time span
- should be thoroughly tested
- should be able to come up with a diagnosis as well as a prognosis of the conservation status of the habitat type

4.4.3 Discussion and some results

Table 4.5 is an example of criteria of favourable conservation status for habitat no. 2130 – "the grey (and green) dunes". Examples of the definition of favourable conservation status for a selected number of habitat types can be found in - http://www.dmu.dk/Udgivelser/Faglige+rapporter/Nr.+600-649/Abstracts/FR647.htm.

The Danish monitoring programme comprises about 1200 monitoring randomly selected stations, each having 20-60 random sampling points defined by UTM co-ordinates. An overview of parameters collected in sampling points and the 5-m circle is seen in Table 4.6 and in Figure 4.4.

Monitoring is carried out on the basis of recommended methods, called "technical instructions", that provide detailed instructions as to how the specific parameters in the criteria for favourable conservation status and the conservation objectives shall be monitored. The Agency for Spatial and

Figure 4.4: Data are sampled in a pinpoint frame supplied with data from a surrounding 5 metre circle. *© K. E. Nielsen*

Environmental Planning are carrying out the practical monitoring programme and the results are reported to the Topic Centre on Terrestrial Nature and Biodiversity placed within NERI, University of Aarhus. Together with the Agency, the Topic Centre is currently developing a database to store all data collected through the programme. The Topic Centre is responsible for quality control and aggregation of data for assessment of conservation status and reporting the results to national and regional authorities.

In most cases, the transition between the different habitat-types is gradual and only one method, or "technical instructions," has been developed for all habitat-types, including specific elements for the forest types, fen types etc. The fundamental object in the monitoring are is sample plot and the surrounding 5 m circle. Table 4.6 and Figure 4.4 show which element is investigated in the two categories.

Table 4.6: Parameter which are measured in sample points and in the surrounding circle.

Observations in the sample plots	Observations in the 5 m circle:
Cover of species	Frequency of species
Supplemental species	Vegetation height
pH in soil/water	Pct. cover of woody species
Conductivity	pct. flooding
C/N - ratio	pct. gaps in vegetation
Phosphorous	pct. cover of invasive species
Nitrate in soil/water	pct. cover of herbivori
N in shoots, mosses and lichens	pct. hollows in bog structure

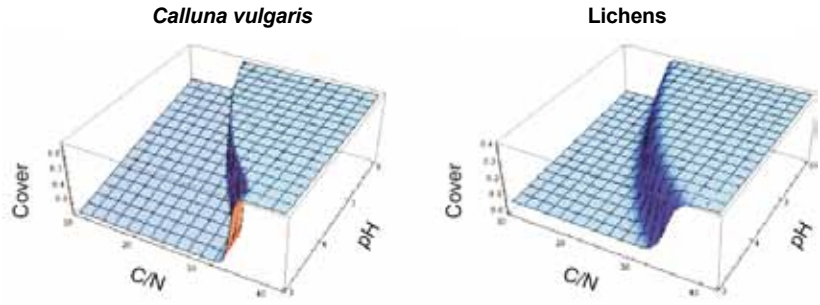

Figure 4.5: Effect of pH and C/N on cover of *Calluna vulgaris* and lichens on dry dune heathland. The majority of data are around pH 4. The favourable conservation status for C:N ratio in the organic topsoil is > 30 which is clearly seen as a threshold

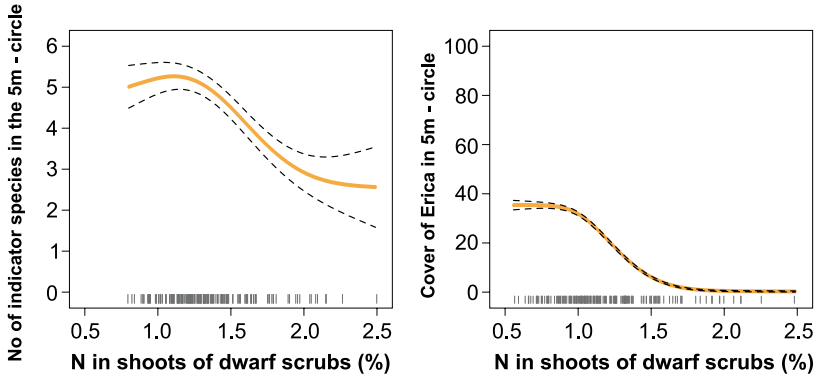

Figure 4.6: Nitrogen content in per cent in young shoots of *Erica tetralix* on wet heath – habitat no. 4010. The uncertainties of the regression model are shown by the upper and lower limit of the curve.

Figure 4.4 shows the field work in a wet dune slack. Information on species cover is sampled by the point-intercept method, i.e. counting the species touched by a thin pin transferred vertically through the vegetation in a grid formed by equidistantly crossed threads extended on a pin-point frame placed over the vegetation. The pinpoint frame is 50 x 50 cm with 16-grid points. The pinpoint method was chosen because data are robust and objective (Damgaard, 2008). The main objective of vegetation monitoring is to follow increases and decreases of the more dominant species over decades. A 5 m circle is made around the sample plot. In the 5 m circle, supplementary species and the cover of trees is noted.

As examples of the kind of results, which are achieved in habitat monitoring, Figure 4.5 shows the effect of pH and C/N on cover of *Calluna vulgaris* and lichens on dry dune heathland. The majority of data are around pH 4. The favourable conservation status for CN ratio in the organic topsoil is > 30. A spectacular threshold around the value of 30 supports the chosen value for the heathland habitat-type.

The left part of figure 4.6 shows the relation between the number of indicator species for wet heath (no 4010) and nitrogen content in the new shoots of *Erica tetralix*. The right figure shows the relation between nitrogen content in the shoots and cover of *Erica tetralix*.

Often, in many ecological connections, the relations between cause and effects are not linear. The examples shown in Figure 4.6 suggest a relation between cover of a dwarf scrub; number of indicator species and nitrogen content in shoots. The steeper the slopes, the bigger the importance of the indicator "nitrogen in shoots" have for the variation in number of indicator species. A lack of, or a weak relation, between biotic and abiotic indicators can, however, be due to time delay between cause and effect in relation to changes in species compositions. Changes in nitrogen content or changes in biomass of the vegetation are among the first indicators to react opposed to changes in species compositions.

4.4.4 Conclusion

The purpose of the proposed criteria is to make a first attempt to define favourable conservation status. Upon these criteria, a systematic national monitoring programme comprising a selected number of habitat-types is developed. The criteria are preliminary and will be adjusted from time to time, as data will be reported from the ongoing monitoring and the knowledge increases.

A value outside the acceptable limits/value should, then, act as a trigger for restoration of a given location. A monitoring programme should not only be designed to detect any changes in conservation status for species and habitats, but also to give answers as to why the changes have happened involving habitat-related parameters. Within the work of reducing the effects of the transboundary air-pollution and to achieve nitrogen deposition below the critical load, (Løkke *et al*, 1996) points to the lack of well-defined biological criteria. Combining elements from the monitoring of forest ecosystems with elements from monitoring of biodiversity, the concept behind the Danish model seeks to "bridge the gap" between traditional biodiversity monitoring and monitoring of effects on air-pollution. The choice of criteria must reflect the ability of diagnosis as well as prognosis.

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4.5 Assessment of nitrogen deposition impacts in support of conservation status assessments in the UK

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Summary

- In the UK, an assessment of nitrogen deposition effects on conservation status of Annex I habitats was undertaken for the 2007 Article 17 reporting round.
- The assessment was based on critical load exceedance. Relevant empirical nutrient nitrogen critical loads were applied to sensitive Annex I habitats, where possible. An assessment of critical load exceedance was then based on a combined approach using data on exceedance of the Annex I habitat resource within Special Areas of Conservation and national mapping of critical load exceedance based on a broader habitat classification.
- The critical loads assessment informed whether to list "air pollution" as a pressure to "structure and function" or a threat to "future prospects" of each habitat. Field evidence and expert judgement were also taken into account.
- Air pollution was listed as a pressure and threat for a large proportion of sensitive Annex I habitats and contributed to a conclusion of unfavourable status for many of these habitats.
- Further method development is required and it is recommended that a robust and consistent approach needs to be developed and adopted across Member States.

4.5.1 Introduction

This paper describes the approach to assessing nitrogen deposition impacts on the conservation status of Annex I habitats, in support of the second report by the UK under Article 17 of the Habitats Directive. A full account of the UK's methodology for assessing conservation status and the results are available at http://www.jncc.gov.uk/page-4060.

The UK assessment of the "specific structures and functions" parameter of conservation status for habitats was made based on the main pressures currently acting on the habitat (including nitrogen deposition), information on the habitat condition and, where relevant information was available, the status of typical species associated with the habitat.

Information on habitat condition from site condition monitoring formed a major component of the assessment. However, since the approaches used for site condition monitoring in the UK are largely based on a fairly rapid visual assessment of key attributes of the habitat, it is acknowledged that this is not a sensitive tool for detecting and, in particular, attributing nitrogen deposition impacts (Williams, 2006). Therefore, a nitrogen deposition assessment, based on the use of empirical nutrient nitrogen critical loads and modelled nitrogen deposition was also undertaken. Additionally, this has the advantage of providing a predictive approach for assessing 'future threats'. The methodology and results are described in this paper and reported more fully in a technical annex to the UK's submission, along with a description of the main uncertainties (http://www.jncc.gov.uk/pdf/FCS2007_techIII_airpollution.pdf). This nitrogen deposition assessment was combined with an acidification assessment and supplemented by evidence of air pollution impacts were available to provide an overall judgement as to whether "air pollution" (category 702; European Commission, 1997) would be listed as a threat or pressure.

4.5.2 Method

The critical loads based assessment was carried out for Annex I habitats only. Species were excluded because of the difficulty in linking habitat-based critical loads to effects on individual species.

Habitats judged not to be sensitive to nitrogen impacts were also excluded from the assessment. In addition, habitats which could not be assigned a critical load were excluded.

The UK does not have nutrient nitrogen critical load maps for Annex I habitats, so existing critical loads resources were adapted for the purposes of the conservation status assessments. These consisted of "Site Relevant critical loads" exceedance data for Special Areas of Conservation (SAC) and national critical loads exceedance maps produced by the UK National Focal Centre. Both of these are based on empirical critical loads for nutrient nitrogen (UNECE, 2003). The datasets are described in more detail below, followed by an explanation of how they were used to inform the overall assessment of nitrogen deposition impacts on Annex I habitats.

Site Relevant critical loads

The UK regulatory and conservation agencies have developed a database of 'Site Relevant critical loads' (SRCL). This database is used for assessments under Article 6.3 as described in Masters *et al*., this volume (Russel *et al*., this volume). Relevant critical loads are assigned to interest features on SACs and information provided on deposition (at 5 km resolution, based on UK FRAME model; Singles *et al*., 1998) to each site, attributed to different sources or source sectors (Bealey *et al*., 2007). Exceedance data for all sensitive Annex I habitats as they occur in SACs are therefore available. In this exercise, critical loads are assigned to sensitive Annex I habitats where there is adequate equivalence with a EUNIS class for which a critical load has been assigned (UNECE, 2003) (see http://www.jncc.gov.uk/page-1425). For the purpose of exceedance estimates, it is assumed that each Annex I habitat covers the whole of each SAC for which it is designated, which in practice is unlikely since many sites include a number of Annex I and Annex II features. It is important to note that some Annex I habitats are well matched to EUNIS habitats for which critical loads are assigned and there can be a lot of confidence in the critical load assigned. For others, this is much more tenuous. Although the Annex I habitats may nest within particular EUNIS classes (at level 2) they are often only a small part, and not necessarily a representative subset, of the wider EUNIS classification. This represents a significant uncertainty. A few Annex I habitats which are potentially sensitive had to be excluded from the assessment because there is not a EUNIS habitat for which a critical load is set, which has sufficient equivalence with the Annex I habitat.

National maps of nutrient nitrogen critical load exceedance

The UK's SRCL database only provides information for the proportion of habitats which occur within SACs. To ensure the assessment adequately represented the risk to the whole Annex I habitat resource, including that outside of SACs, the assessment also drew on UK national critical loads exceedance mapping (Hall *et al*., 2003) in addition to the SRCL data.

National critical loads maps are produced for Broad Habitats defined under the UK Biodiversity Action Plan according the method described in Hall *et al*., (2003). Deposition is modelled at 5km resolution using the CBED method (Smith *et al*., 2000) to provide exceedance estimates based on 2002-04 data for this assessment. The Broad Habitats are related to EUNIS categories in order to assign the appropriate critical load, in same way as described above for Annex I habitats. Since Annex I habitats are only a component of the wider BAP broad habitat classification, the broad habitat distribution maps and Annex I distribution maps were compared as part of the assessment.

Assessment Procedure

The individual habitat assessments were based on a combination of exceedance of SRCLs and national critical loads exceedance mapping. Where the extent of the habitat is primarily within the SAC series, it is reliable to base the assessment solely on the SAC exceedance data. Where not, more emphasis has to be put on the national critical loads mapping. There are a number of limitations and uncertainties with both these approaches including the habitat mapping; distribution of habitats

within SACs; deposition modelling and the relationships between the various habitat classifications. In practice, these differences are unlikely to affect the overall outcome of individual assessments.

In order to use the two datasets to derive a conclusion for whether nitrogen deposition should be included as a pressure or threat, the following questions were addressed for each habitat:

- Is the Annex I habitat sensitive to atmospheric inputs of nutrient nitrogen or acidity?
- Is there an appropriate critical load i.e.
  - is there a reasonable equivalence between the EUNIS habitats, for which critical loads are set, and the Annex I habitat? If so,
  - is the research upon which critical loads are based representative of potential impacts on the Annex I habitat?
- What is the exceedance of nutrient nitrogen critical loads of the habitat within the SAC series?
- What is the extent of the habitat which occurs in the SAC series (per cent)?
- How does the distribution of the Annex I habitat compare with the distribution of the relevant BAP Broad Habitat mapped for critical load exceedance if applicable.
- What is the critical load exceedance of the relevant Broad Habitat.

The judgement is then based on critical load exceedance for SACs and for relevant BAP habitats, but qualified by the level of certainty in the above steps. The assessment is based on national modelling of deposition and provides a national overview. Some Annex I habitats which are not identified as 'at risk' at a national level may still be under threat on a local/site specific basis.

Where 'relevant' critical loads are exceeded over a significant area for a particular Annex I habitat, air pollution was listed as a current "pressure" and future "threat" (future/foreseeable impacts). Any field evidence of impacts on the habitats, or other impacts information, was also used to inform whether air pollution would be listed as a current pressure or future threat. In practice, this was largely confined to coastal habitats, which were not well represented by the critical loads exceedance assessment, and freshwater habitats, for which there were no applicable critical loads.

4.5.3 Results and discussion

It was possible to undertake a nutrient nitrogen critical loads assessment for 51 Annex I habitats out of a total of 87 (77 excluding Gibraltar) habitats for which the UK has to report under Article 17. This means that the habitats were sensitive and a 'relevant' critical load could be assigned. Air pollution was recorded as pressure and threat in the assessments for 33 of the 51 habitats based on critical load exceedance. Air pollution was also listed for a further 20 habitats, based on expert judgement including, where available, field-based evidence.

This shows the severity of the threat that nitrogen deposition poses in the UK and is consistent with national critical loads reporting, survey and experimental evidence (NEGTAP, 2001).

In this context, the application of critical loads represents a risk assessment to identify the areas and habitats 'at risk' from nutrient nitrogen deposition. This is an appropriate tool for judging the future prospects parameter of conservation status, particularly where the exceedance can be calculated for future emissions based on implementation of currently agreed legislation. However, given the uncertainties and the definition and purpose of critical loads, the approach on its own, without field evidence, is less robust in terms of current pressure on structure and function. It cannot prove there is actually currently biological or biogeochemical 'damage' to a habitat area just that this will occur at some point.

Table 4.7: Results of the air pollution assessment (incorporating nitrogen deposition) for UK conservation status reporting in 2007: the proportion of assessments which record air pollution (code 702) as a pressure for the Annex I broad habitat classes. Number of assessment records per broad habitat class is shown in parenthesis. This includes the 10 habitats in the Gibraltar report which forms part of the UK's territorial boundary and the UK Article 17 submission, but for which a specific assessment of air pollution has not been undertaken.

Annex I Broad Habitat Class	Proportion of assessments which record Air Pollution as a threat or pressure
Marine, coastal and halophytic habitats	6 (17)
Coastal sand dunes and continental dunes	85 (13)
Freshwater habitats	88 (8)
Temperate heath and scrub	83 (6)
Sclerophyllous scrub (matorral)	25 (4)
Natural and semi-natural grassland formations	78 (9)
Raised bogs and mires and fens	67 (9)
Rocky habitats and caves	50 (10)
Forests	91 (11)

Therefore, ideally the critical loads assessment would be combined with representative field-based evidence of effects on the habitat structure and function. At the time of the assessment for the second round of reporting for Article 17 this was not available in the UK in a form that could be readily used. These principles are discussed in Whitfield *et al*., (this volume).

Air pollution was included as a pressure and a threat for a large number of the Annex I conservation status assessments in the UK and contributed to a conclusion of unfavourable status for many of these habitats. However, on no occasion did it tip the balance of an assessment outcome from favourable to unfavourable, for the structure and function parameter, because there were also other factors contributing to this. As a result, the qualitative approach using critical loads was fit for purpose and to identify nutrient nitrogen deposition as an important pressure/threat. In future, if other pressures/threats to structure and function are addressed, there may be a greater scrutiny on the air pollution assessment and a more quantitative approach may be required.

Air pollution was one of a large number of pressures and threats listed for UK habitats. The scale of each and their relative priority or importance is difficult to judge and cannot be established from the assessments.

4.5.4 Conclusions

In the UK Article 17 report; 'air pollution' (incorporating nitrogen deposition) was listed as pressure to the current structures and functions, or a threat to future prospects, of 53 Annex I habitats out of a total of 87 records.

The results show a widespread risk from nitrogen deposition and this is consistent with other evidence from the UK (NEGTAP, 2001). It should provide a strong policy driver for targeted reductions in emissions of nitrogen pollutants.

The nitrogen assessment method was based on critical load exceedance using two datasets: site relevant critical loads and national exceedance maps. The method was fit for purpose: to highlight the importance of nitrogen deposition.

However, a number of questions about the approach and the uncertainties remain. These are consistent with those reported in Whitfield *et al*., (this volume) and a robust and consistent approach needs to be developed and adopted across Member States to report on the range and extent of nitrogen deposition impacts under Article 17 of the Habitats Directive.

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5

NEW SCIENCE ON THE EFFECTS OF NITROGEN DEPOSITION AND CONCENTRATIONS ON NATURA 2000 SITES (THEME 3)

5.1 Background document

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Summary

This background paper summarizes established and new science on the effects of nitrogen (N) deposition on ecosystems and considers the potential for improved assessment of N deposition impacts on Natura 2000 sites. The key aspects covered are N deposition effects on biodiversity and on biogeochemistry, links to ecosystem services, the importance of N form, N deposition indicators, management practices and ecosystem reversibility following decreased N input.

The paper shows that:

- Evidence of N impacts on different vegetation types in Europe exists, but that it is important that it is translated meaningfully to the target habitats listed under the Habitats Directive. Evidence for N deposition effects to important ecosystem services, such as carbon sequestration, also exists but the cause and effect relationships underlying them are often complex and not sufficiently understood.
- Chemical N form can influence both the rate of ecosystem change, and the extent of impacts on the short and long-term. Evidence is presented for ammonia causing detrimental plant physiological effects, probably on the majority of species, whilst ammonium and nitrate effects will depend on plant species present.
- Plant species composition as well as plant biochemical parameters may be useful bioindicators for assessment of N deposition effects in Natura 2000 sites, however "baseline" data are mostly not available for rare species. Ecosystem specific indicators, that are predictive of further damage, rather than a consequence of already existing adverse effects (i.e., early warning indicators) are needed.
- Site level management practices can be useful to reduce the impact if N deposition but they will certainly not be able to mitigate all the impacts of enhanced N deposition and enhanced

N concentration on Natura 2000 habitats. More knowledge is needed to better understand where and if management intervention is appropriate to mitigate N effects.
- Studies on the reversibility of N impacts show that some ecosystem parameters may revert quickly, while other components may show strong inertia. In some cases reversion to the original state may however be impossible.
- Climatic factors interfere with ecosystem effects of N deposition. It is clear that climate both can emphasize and mitigate effects of N deposition. Current climate and expected climatic changes must be included in assessments and predictions of N deposition effects on ecosystems.

The aim is to provide a broad picture of scientific advancement within the field of N deposition research, and to provide a starting point for workshop discussions. Workshop discussions addressed the relevance of new science in assessing N deposition impacts on Natura 2000 sites, and identified when there is potential to make management adjustments to mitigate such effects.

5.1.1 Introduction

In response to rising world demand for food and energy anthropogenic nitrogen (N) emissions are now around the same order of magnitude as N input from natural sources, which means that the N pool available to terrestrial organisms has more than doubled in less than a century (Vitousek *et al*., 1997). The sources and sinks for biologically reactive N have become decoupled, as N released into the atmosphere from agricultural sources and combustion processes is subject to short and long range atmospheric transport (Galloway *et al*., 2008). Biologically, reactive N can be redistributed from emission "hot-spots" (i.e. agricultural and densely populated regions) to remote regions with undisturbed ecosystems naturally adapted to very low N inputs and availability.

Nitrogen is the second most important plant nutrient behind carbon and the productivity of terrestrial ecosystems is most often limited by the N supply (Tamm, 1991). Hence, increasing N deposition will be expected to exert a large impact on ecosystem biodiversity, biogeochemical cycling of N and ecosystem functioning and service provision. Nitrogen loads to European ecosystems have increased substantially over the last century. At the same time as N deposition loads have increased, substantial alterations in land-use have taken place. In addition, there are on-going climatic changes. Thus, it is difficult to estimate exactly how important N deposition *per se* has been for ecosystem changes in Europe. In many cases useful historic data on ecosystem structure and function prior to the time period of substantial N deposition do not exist. Nevertheless, experimental manipulation studies along with extensive environmental monitoring efforts suggest that N deposition effects on many habitat types have been substantial (see for example Bobbink *et al*., 1992, 1996, 1998, 2003, 2009).

In this background paper we discuss current understanding and recent scientific findings on N deposition effects on European ecosystems in relation to the requirements of implementing the Habitats Directive. In particular, how to assess potential for significant effects affecting the integrity of a site for the assessment of 'plans and projects' under Article 6.3 (Bealey *et al*., this volume) and assess whether N deposition is a 'pressure' or a 'threat' to conservation status under Article 17 (Whitfield and Strachan, this volume). Key considerations discussed below are:

- How does N deposition affect habitat structure and function of different habitat types?
- How does current scientific understanding map onto Annex 1 habitats?
- Is the chemical form of N deposition (reduced N versus oxidized N) or type of deposition (wet versus dry) important?
- What potential is there for the use of bioindicators to assess N deposition effects in Natura 2000 habitats?

- How reversible are N deposition effects?
- What is the potential for the use of on-site management practices for maintaining favourable status?

5.1.2 Effects of N deposition on structure and function of different habitat types

The concept of empirical critical load

The relationship between pollutant dose and resultant environmental effect forms the basis for the critical load concept. The 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" (Nilsson and Grennfelt, 1988).

Empirical critical N loads are based on field evidence with respect to increased N inputs. In Europe, empirical critical loads have been assessed since the early 1990s for use within the Convention on Long-Range Transboundary Air Pollution (LRTAP) for impacts on biodiversity in natural and semi-natural systems in Europe (Bobbink *et al*., 1992, Bobbink *et al*., 1996, Bobbink *et al*., 2003). 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) mineralization rate, decomposition or N leaching (functioning). The effects are evaluated for specific receptor groups of ecosystems.

Studies providing insights into ecosystem reactions to an increase in N input or availability have been conducted for a variety of reasons. This has resulted in many different designs, from correlative or retrospective field studies, experimental studies in pots and mesocosms to field addition experiments. Statistically and biologically significant outcomes of field addition experiments and mesocosm studies have been used for the assessment of empirical N critic al loads. Changes in species abundance, composition and/or diversity is used as the dominant indicator. Only studies which have independent N treatments and realistic N loads (below 100 kg N $ha^{-1}y^{-1}$) and two years or more duration have been used for the setting of the critical load values. The methods used in these studies have been carefully scrutinised to identify factors related to the experimental design or data analysis, which may constrain their use in assessing critical loads. This includes evaluation of the precision of the estimated values of background N deposition at the experimental site. In addition, the results from correlative or retrospective field studies have been used as the basis for estimates of critical loads, but only as additional evidence either to complement of the results from experimental N addition studies, or as an indication for expert judgement. Empirical nutrient nitrogen critical loads are given in Bobbink et al., (2003); an update has subsequently been conducted (Bobbink and Hettelingh, 2011).

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. However, a key question in their use to support policy development 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 of European N addition experiments in grasslands, wetlands, (sub) Arctic and alpine vegetation showed a clear negative-log relationship between exceedance of empirical N critical loads and plant species richness (Bobbink *et al*., 2009). Hence, although there are methodological limitations and scientific uncertainties in the methods used to derive empirical critical loads, exceedance of these values is clearly linked to reduced plant species richness in a broad range of European ecosystems.

Examples of N induced biodiversity loss for some European ecosystems

In a recent synthesis article Bobbink *et al*., (2009) described N deposition effects on terrestrial plant diversity across a latitudinal range of main categories of ecosystems. In this workshop background paper, we briefly summarize the N deposition effects on plant diversity according to this article for some of the ecosystem types represented in the Natura 2000 biogeographical regions.

Arctic and Alpine ecosystems

Common responses to increased N input in arctic and alpine ecosystems include decreased abundance of bryophytes and lichens and increased growth of graminoids. In the harshest habitats (polar deserts and arctic heaths) plant growth is often co-limited by N and P, and increased N input *per se* has in short-term studies not been found to cause significant vegetation changes.

Boreal forest

Increased N input causes major changes in ground vegetation species composition, but often no decline in overall plant species richness. Bryophyte, lichen and dwarf-shrub species are sensitive to elevated N inputs, while many graminoids and herbs with faster growth rates and higher N demand benefit and proliferate. Changes in biotic interactions (increased pathogen damage to plants), or physical disturbance of the ecosystem (for example tree harvest) may reinforce N induced vegetation changes.

Temperate forest

The vegetation response to increased N input in temperate forests include an initial increase in plant cover, a decrease in richness due to loss of N efficient species, a decrease in species evenness from increasing dominance of few nitrophilic species and loss of diversity due to decreases in species richness and evenness.

Temperate heathlands and grasslands

Most of these ecosystems in Europe have evolved under long-term, low-intensity agricultural management. Continued management is thus a prerequisite to sustain them. Natural temperate grasslands (steppe or prairie) with no tree growth due to climatic constraints are relatively rare in Europe. For heathlands in central Europe and UK dwarf-shrub growth is enhanced by elevated N inputs, while bryophytes and lichens often are negatively affected. Biotic, abiotic and climate stresses (for example herbivore damage to dwarf-shrubs and winter desiccation) and some management regimes may however trigger vegetation change from dwarf-shrub to grass dominance under high N input. For grasslands N induced species loss has been observed with more detrimental effects on rare than on common species. For acidic grasslands in the UK, species loss has been shown to occur as a function of cumulative N deposition.

Mediterranean vegetation

N effects on Mediterranean vegetation in Europe have been very little studied. One Italian study supported by several Californian ones, indicate that invasive grasses increase with N input and as a consequence species richness of native vegetation declines. In addition studies have shown that Mediterranean lichen communities are very sensitive to N deposition and major shifts in lichen communities occur at relatively low N input.

Such changes need to be considered in further detail at the workshop in relation to the target habitats listed under the Habitats Directive.

N induced ecosystem functionality changes

For many European ecosystem types, studies have concluded that N deposition results in loss of species richness. Species loss may lead to changes or even loss of key ecosystem functions

and the ability of ecosystems to provide valuable ecosystem services. Due to the assumption of more effective utilization of available ecosystem niches at high than at low biodiversity, a positive relation between species richness and ecosystem functionality has been proposed (van Ruijven *et al*., 2005, Fornara and Tilman, 2009). Important ecosystem functions that may be affected by N deposition effects on biodiversity include productivity, carbon sequestration, N cycling and N retention. There is therefore potential for the consideration of positive and negative N impacts on ecosystem services provided by Natura 2000 sites to further promote the importance of habitat preservation in policy development (see Hicks *et al*., this volume, section 7.9). However, the cause and effect relationships underlying important ecosystem services are often complex and not sufficiently understood as discussed below.

Peatland ecosystems provide an example of how species replacement, resulting from N deposition, may alter ecosystem functionality. On a global scale peatlands store huge amounts of carbon and usually function as active carbon sinks. However, several studies have indicated that the carbon sequestration capacity of ombrotrophic bog ecosystems decreases when subjected to elevated N inputs. Plant growth on ombrotrophic bogs is under low N deposition strictly N limited, as the ecosystem only receives water and nutrients from precipitation. Raised N deposition has a negative impact on the *Sphagnum* (peatmoss) productivity (see for example Gunnarsson and Rydin, 2000). In addition, increased N input may make *Sphagnum* shoots more easily decomposable (Limpens and Berendse 2003, Bragazza *et al*., 2006). It has been suggested that reduced polyphenol concentration may contribute to increased *Sphagnum* decomposability under high N input (Bragazza *et al*., 2006, Bragazza and Freeman, 2007). Nitrogen input also causes vegetation shifts from bogs dominated by *Sphagnum* to domination by vascular plants (mainly *Cyperaceae* and *Ericaceae* species; Gunnarsson *et al*., 2002, 2008, Wiedermann *et al*., 2007, 2009a, Heijmans *et al*., 2008). This shift can have several effects. Cyperaceae and Ericaceae species usually have higher growth rates and nutrient demands and are more easily decomposed than *Sphagnum* (Limpens and Berendse 2003, Breeuwer *et al*., 2008). Increased abundance of vascular species may also cause the groundwater table on 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.

5.1.3 Nitrogen deposition interferes with ecosystem biogeochemistry

Is the chemical form and type of N deposition important for ecosystem response?

Deposition of reactive N (all species except for unreactive N_2 gas) occurs in several chemical forms. Nitric oxide and nitrogen dioxide (collectively termed NO_x) are eventually oxidized to form nitrate (NO_3-) in aerosols as well as gaseous nitric acid (HNO_3). The combination of oxidized N forms (collectively NO_y) originates from combustion processes (using fossil or bio-fuels) and can be transported long distances in the atmosphere. Farmyard manure and emissions from intensive animal rearing units are the main emission sources of ammonia (NH_3) which forms ammonium (NH_4^+) in aerosols and precipitation. Organic N forms occur mainly in the form of amine N (R-NH_2). Reduced N forms (collectively NH_x) are generally transported more regionally/locally than NO_y Atmospheric N inputs (in the form of NO_y and/or NH_x) to an ecosystem can occur both via wet (with precipitation in the form of rain, cloud and snow) and dry (with particle or gaseous) deposition.

Currently, all forms of reactive N deposition are treated as equal with regard to their ecosystem effects when using the critical loads approach. However, their chemical and physical properties and their spatial and temporal deposition are very different. The deposition of wet, dry, oxidized and reduced N species for Natura 2000 sites and their individual effects are therefore a key consideration. The different chemical forms of reactive N have different effects on plant and soil properties and in the following we have briefly summarized the main differences. Some of these differences

between pollutant form may be treated by the 'critical levels' approach for air concentration effects, however, there may also be differences in impacts indeposition between N forms.

Ammonia emitted from farmyards can occur in potentially phytotoxic concentrations (Krupa, 2003). It deposits as a 'dry' gas, which is alkaline and highly reactive, and is taken up by plants through open stomata, directly into leaves in response to physical and chemical laws rather than biological demand. By contrast the deposition of ammonium and nitrate is in solution, as these ions are 'washed out' of the atmosphere in precipitation, be it rain, cloud or snow. For higher plants ionic concentrations in precipitation are rarely phytotoxic, with minimal uptake through the cuticle, although much higher concentration exposure can occur as a result of exposure to polluted cloudwater. Most ionic N is instead absorbed from the soil via plant roots or mycorrhizal fungi and is thereby subject to biological control as higher plants have physiological mechanisms regulating their root N uptake (Miller *et al*., 2008). For lower plants (bryophytes and lichens), factors influencing uptake of the different N forms are less well understood. Their high surface to volume ratio, the lack of cuticle and low (acidic) tissue surface pH will enhance ammonia deposition and uptake (Jones *et al*., 2007). Also all ionic N forms in solution are efficiently taken up by bryophytes and lichens (Dahlman *et al*., 2004, Turetsky 2006, Forsum *et al*., 2006), although there are studies indicating that ammonium is more readily absorbed than nitrate (Dahlman *et al*., 2004, Nordin *et al*., 2006, Palmqvist and Dahlman 2006, Wiedermann *et al*., 2009b). It has been shown that mosses subjected to high N input (which accumulate abnormally high internal N concentrations) still do not down-regulate N uptake (Forsum *et al*., 2006). In the long term (> 30 years of elevated N input) there is some evidence that at least *Sphagnum* mosses may be able to adjust their N uptake to high N supply (Press *et al*., 1986, Limpens and Berendse 2003, Wiedermann *et al*., 2009b).

In soils, ammonium, due to its positive charge, can accumulate adsorbed to minerals and organic matter. Hence it may compete with other cations (like for example potassium, K^+) that are important plant nutrients, for uptake by roots (Marschner, 1995). If not taken up by plant roots or soil microbes, ammonium can be nitrified, a soil acidifying process which can also increase the risk of plant root damage from elevated Al^{3+} toxicity, in mineral soils (see references in van den Berg *et al*., 2008). Nitrate is negatively charged and does not accumulate in soils, rather, if not taken up by plants or soil microbes, it will be leached into water courses taking with it base cations (the mobile anion effect) or it can denitrified to N_2 and/or N_2O, potentially adding to the greenhouse effect (N_2O is 298 times more potent than CO_2 as a greenhouse gas). Plant and soil microbe utilisation of nitrate can increase soil pH, unless base cations are lost through the mobile anion effect.

It should not be overlooked however, that our different ecosystems and the biota they sustain have evolved to deal with the properties associated with the different N forms which are inextricably linked to soil pH and other inherent soil chemical properties. Plant communities characteristic of acid conditions tend to 'prefer' ammonium while those inhabiting more alkaline soils are better adapted to use nitrate (Gigon and Rorison, 1972). The composition of N deposition in precipitation can change the balance of reduced to oxidised N in the soil solution, decoupling it from pH, as well as providing a supply of N for foliar uptake. This means that ecosystems that have evolved on more alkaline nitrate dominated soils may now be challenged by the deposition of ammonium and vice-versa.

Evidence of effects of the different N forms on ecosystem form and function

Field N manipulations offer the most objective approach to separating the effects of the different N forms, especially if the treatment scenario is coupled to real world environmental drivers like precipitation and wind direction. In a globally unique experiment, ammonia, ammonium and nitrate have been applied since 2002 to an ombrotrophic bog, Whim, in the Scottish Borders, with both historically and currently low (in UK terms) ambient N deposition. The treatments realistically

simulate deposition conditions for gaseous ammonia and wet ammonium (NH_4Cl) and nitrate ($NaNO_3$), using high application frequencies coupled to meteorology and low ionic concentrations (maximum 4mM) at a range of N doses (+ 8 kg N $ha^{-1}y^{-1}$ to 56 kg N $ha^{-1}y^{-1}$). Measured ammonia concentrations along the release transect have been converted to deposition using a model based on findings from carefully controlled flux chamber studies with the same bog vegetation (Jones *et al*., 2007, Cape *et al*., 2008). This experiment has provided confirmatory evidence that effects observed in controlled experiments (see Krupa, 2003) can be replicated in the field.

Ammonia was shown to be the most damaging N form, effects occurred faster and thus at lower accumulated N doses than with wet deposited oxidised or reduced N (Sheppard *et al*., 2008). Sensitive plants (e.g. *Calluna vulgaris*, *Sphagnum capillifolium* and *Cladonia portentosa*) have a significantly lower tolerance threshold to N deposited as ammonia compared to ionic N deposited in precipitation. Similar N doses as ammonium or nitrate increased the growth of *Calluna* and had no adverse effects on its tolerance of abiotic or biotic stress to date (Sheppard *et al*., 2008). Exposure to ammonia caused acute responses, probably indicative of higher exposure concentrations, than typically occur with rain or cloud water ion uptake, whereas, the effects of wet deposition were of a less severe but more chronic nature. The effects of oxidised versus reduced N in precipitation were restricted to the bryophytes and lichens: *Sphagnum capillifolium*, *Hypnum jutlandicum* and *Pleurozium schreberi* all showed that ammonium deposition increased N concentrations significantly more than nitrate at higher doses > 24 kg N $ha^{-1}y^{-1}$ when accumulation became exponential. In *S. capillifolium* the resulting high concentrations of soluble toxic ammonium at 56 kg N $ha^{-1}y^{-1}$ has contributed to reduced growth, loss of cover and breakdown of the *S. capitulum.*

N form may also affect soil mediated plant N responses. In a mesocosm study, the acidification effect associated with the nitrification of ammonium which resulted in high soil solution concentrations of toxic metal ions, together with the potential for ammonium adsorption on soil cation exchange sites and reduced base cation uptake, were all seen as contributory factors causing the decline in sensitive species seen with high ammonium inputs (van den Berg *et al.,* 2008). By contrast detrimental effects of nitrate additions appear to be associated with the growth promoting effects of nitrate additions on some selected plants and thereby increased competition. Studies have demonstrated that many N conservative dwarf-shrubs and herbs have only limited capacity to utilize nitrate (Chapin *et al*., 1993, Kronzucker *et al*., 1997, Nordin *et al*., 2001). In contrast, plant species adapted to N rich habitats (some of them invasive), often exhibit high capacities to take up nitrate (Nordin *et al*., 2001, 2006). In the context of increasing graminoid cover in response to nitrate rather than ammonium, Pearce and van der Wal (2002) recorded the opposite, with slightly more graminoids occurring with ammonium on a Racomitrium heath. However, in this situation all the plants preferred ammonium and the response was explained by the occurrence of nutrient leakage from Racomitrium in response to ammonium toxicity.

5.1.4 Indicators of N deposition

To be able to evaluate N deposition effects in Natura 2000 sites and be able to adjust the management of affected sites, easily identified indicators of N deposition appear useful. If we are to protect rare species and ecosystems and maintain ecosystem function and services it will be important to establish relationships between changes in soil chemistry, plant metabolites and species composition. There is a need to have a clear remit for N indicators, whether they are to indicate N effects already brought about by N, or provide an early warning of potential effects. Indicators of empirical critical loads for N have been discussed during previous workshops (see Lokke *et al*., 2000 for a report from a critical load workshop held in Copenhagen in 1999). Also in the UK there have been a series of detailed reports evaluating ecosystem characteristics that could serve as bioindicators of elevated N effects (Sutton *et al*., 2004, Leith *et al*., 2005).

In the following, we briefly summarize information on some potential N indicators;

- N indices for plant species have been suggested as one indicator of N deposition on vegetation. The idea is that by defining species according to their N requirements, one can assess the N status of a habitat by an inventory of its flora. The most frequently used index is Ellenberg's indicator values that have been assigned to a great number of European vascular, bryophyte and lichen species. The Ellenberg index characterizes a species according to a range of variables of which soil fertility at the site where the species is normally found is one of the more important. Another index is FNIS that characterize a species according to its occurrence in relation to soil ammonification and nitrification (Diekmann and Falkengren-Grerup, 1998). A limitation with both these indices is that they are developed explicitly for temperate ecosystems. The Ellenberg index is not specific to N as it denotes soil fertility (including all soil nutrients) rather than just N availability. Also it is only possible to assess changes that have already occurred, and the monitoring results cannot predict future changes.
- Another bioindicator that may be useful for assessing effects of N deposition on vegetation is measuring amino acid concentrations of plant tissues (Pitcairn *et al*., 2003, Wiedermann *et al*., 2009b). According to this idea elevated amino acid concentrations in tissues of a plant would denote that N uptake exceeded the plants' capacity to convert N to growth. This would indicate a risk for other species (with a better capacity to convert N to growth) to take over the habitat. An advantage with the method would be that instead of just assessing changes that already occurred, predictions of future changes may be possible to make as amino acid accumulation in plant tissues is an immediate response to N enrichment preceding any vegetational changes (Nordin *et al*., 1998).
- Total tissue N per cent may be a simpler measure of plant N accumulation than amino acid N concentrations. However, Sheppard *et al*., (2008) found that statistically significant increases in the N concentration in Calluna shoots did not correlate with loss of cover, unless the increase in N per cent was large, as happened when the N deposition was in the form of ammonia. Obviously, by only measuring plant tissue N per cent it is difficult to evaluate the size of the signal and scale of threat N deposition poses to an ecosystem. Although there exists a large literature on the effects of N on plant tissue N per cent there is no central database or major compilation of available data in Europe. Nevertheless, foliar N thresholds may be established for different ecosystem indicator plants which could be calibrated to supply a metric for predicting ecosystem sensitivity.
- The most sensitive N indicators, especially when the N form is ammonia, have been shown to be epiphytic lichen communities. Changes in the proportion of acidophiles to nitrophiles can indicate increasing exposure to ammonia. This suggests that this lichen community response could provide a reliable robust indicator for enhanced ammonia concentrations which is also relatively cheap once field workers have been trained in identification (Leith *et al*., 2005, Wolseley *et al*., 2009). By contrast, such lichens appear to be more sensitive to ammonia than total N deposition, and while lichens may give an indication of the latter, more work is needed to distinguish between the ammonia and overall N effects.

Bioindicators of N deposition need to capture the chain of events (N accumulation cascading through the various ecosystem compartments) that ultimately leads to altered ecological structure and/or function of an ecosystem. Capturing this chain of events may require a cocktail of bioindicator tools rather one specific, as discussed in detail by Sutton *et al*., (2005), especially since other environmental factors and management can also influence indicators. Probably, a combination of the bioindicators presented above will best report on the state on Natura 2000 sites. Moreover, the applicability of the presented N deposition bioindicators has still to be tested for all vulnerable ecosystems. Likewise for many important plant species, we lack data for many threatened species,

i.e. their "baseline" state and have no estimate of acceptable variability, tolerance range or seasonal variability in the majority of bioindicators. In the UK and Europe, many similar issues have been considered with respect to freshwater ecosystems since 1970s. Research into methods for assessing the biological quality of running waters was initiated in response to the perceived need by scientists and water quality managers for a greater understanding of the ecology of running water sites and their macroinvertebrate commununities. This resulted in the development of RIVPACS (River InVertebrate Prediction And Classification Scheme) by the FBA and CEH. It worked on the basis of classifying unpolluted running water sites based on their macroinvertebrate fauna and determining the composition of a macroinvertebrate community at specific sites in response to physical and chemical features. This concept of assessing 'reference condition' (now based on 500 sites) and making comparisons with the observed fauna at sites of interest, directly influenced the drafting of the European Union Water Framework Directive (WFD) (European Commission, 2000). The core concept of the WFD, that an ecological status target is set for each site, is essentially derived from the RIVPACS type approach. A similar approach may be used for terrestrial habitats.

5.1.5 Reversibility of N deposition effects

A key conservation question is whether, and to what extent, N induced changes are reversible, if N deposition levels are reduced. Related to this, it needs to be known over what timescale any recovery might operate for different effects and receptor ecosystems. From a scientific point of view, we have a reasonable knowledge concerning effects of increased N input, while the reversibility of N-induced effects is largely unknown. A small, but growing literature dealing with this topic is, however, emerging (e.g. Strengbom *et al*., 2001, Mitchell *et al*., 2004, Power *et al*., 2006, Limpens and Heijmans 2008, Clark and Tilman, 2008). From such studies it is apparent that some ecosystem parameters may revert quickly, while other components may show strong inertia.

Although several studies have shown that N leakage or exchangeable N in the soil may return to control conditions within a few years following cessation of external N input (Bredemeier *et al*., 1998, Högberg *et al*., 2006, Oulehle *et al*., 2006), changes in plant species composition may be slow. Strengbom *et al*., (2001) found no, or only small signs of recovery in boreal ground vegetation nine years after cessation of N addition (c. 100 kg N ha^{-1} for 20 years). Nearly 50 years after cessation of N addition, the abundance of bryophytes sensitive to N addition was still lower (e.g., *Hylocomium splendens*), while bryophytes favoured by N addition were still higher (e.g., *Brachytecium* spp.) (Strengbom *et al*., 2001). In a study using controlled mesocosms, containing an aquatic habitat similar to that which can be found in shallow soft-water lakes, a two-year treatment with ammonium resulted in substantially altered plant species composition (Brouwer *et al*., 1997). Despite a 10-year treatment with clean rain water following the cessation of the ammonium treatment, only minor recovery of the plant species composition was observed. In grasslands, here exemplified by prairie system from North America, low levels of elevated N input (6 kg N $ha^{-1}y^{-1}$ above a background deposition of 4 kg N $ha^{-1}y^{-1}$) reduce species richness and alter relative abundances of plant species (Clark and Tilman, 2008). A decade after cessation of the N treatment, plant species richness (on plot level) had returned to control level, but the relative abundance of component species still differed. This suggest that in several habitat types, once altered by elevated N input, the species composition if reversible, requires substantial time to revert to a state comparable to that prevailing under low N input.

Other habitats, or important parts of habitats, may revert more quickly to the low N input state. For example tissue N concentration in *Sphagnum* species in raised bogs may return to pre-treatment concentrations within 15 months after cessation of N addition (Limpens and Heijmans, 2008). This suggests that, as long as high N input has not caused the peat forming *Sphagnum* species to die, the ecosystem service of peat accumulation may be restored rather quickly following reduced N input (Limpens and Heijmans, 2008). Similarly, in a reciprocal transplant experiment, Mitchell *et*

al., (2004) were able to show recovery of tissue N concentrations and growth rates within a year of transplanting ephiphytic bryophytes to a cleaner location.

In some cases reversion to the original state may however be impossible. Species may locally or regionally have become very rare or even gone extinct providing no propagule source for the original species. Moreover, new internal or external factors may have emerged in the ecosystem, to hinder reversion. For example, changed precipitation patterns have proven reversion of N altered plant communities difficult or impossible (Choi *et al.,* 2006). In addition, internal feed-backs on nutrient turnover-rates (Bowman and Steltzer 1998, Chen and Högberg 2006, Power *et al*., 2006), may have increased the persistence of the N induced state, and made new alternative states of the system possible (Suding *et al*., 2003).

5.1.6 The use of management practices for maintaining favourable status

Decreasing N deposition would, of course, be the preferred way to protect Natura 2000 sites from N induced ecosystem changes. However, management methods that remove N from a habitat can be useful in mitigating N deposition effects on ecosystems. For semi-natural habitats, such as grasslands and heathlands, which require an active management regime for their maintenance, intensive use of methods causing biomass removal by mowing or prescribed burning may at least partly mitigate N induced alterations (Mountford *et al*., 1996, Barker *et al*., 2004).

For Dutch calcareous grasslands Willems (2001) suggests that N deposition effects can be decreased by mowing in early August. The mowing suppresses the N favored grass *Brachypodium pinnatum* (Tor grass) and promotes the original species-rich grassland vegetation. Also for heathlands, originally dominated by *Calluna vulgaris*, active management involving prescribed burning and mowing can mitigate effects of N deposition (Barker *et al*., 2004). Active management may be a promising alternative for many semi-natural habitats, and often the N management can be incorporated in the management that is already imposed to maintain the conservational value of the habitat. For other habitats there is no, or very little, available information on management strategies mitigating effects of N enrichment. For forest ecosystems, timber harvest and/or forest fires can remove large quantities of N. If timber harvesting is not combined with burning of the clear-cut area, it is necessary to remove also needles/leaves and branches (beside the timber) to achieve a significant N removal from the ecosystem. Moreover, we should be reminded that the physical disturbance caused by timber harvest may under some circumstances reinforce N effects on the ground vegetation, thus counteracting the potential positive effect of removing N from the ecosystem (Strengbom and Nordin, 2008). In forest types where re-occurring forest fires have been part of a natural disturbance regime, prescribed burning seems like an efficient management strategy for mitigating effects of N deposition, as it both reduces the amount of N stored in the forest floor vegetation and in the uppermost humus layer while also restoring the natural disturbance regime. However, such an approach could be considered controversial, due to the need to more fully understand other interactions, including the fate and impact of the reactive N emitted in such fires.

It should be added that realistic site level management practices will certainly not be able to mitigate all the impacts of enhanced N deposition and enhanced N concentration on Natura 2000 habitats. For example, the loss of epiphytic flora would be very difficult to deal with by on site management practices.

5.1.7 How can we use current understanding of N impacts to protect Natura 2000 sites from N deposition?

As already concluded, the majority of European Natura 2000 sites are affected by historic and present patterns of N deposition. It is scientifically well established that the effects of N deposition on plant community structure and function depend on the ecosystem type and the size of the cumulative N deposition load. Bobbink *et al*., (2003) provides the most recent compilation of data indicating sizes of N loads causing significant changes in various ecosystem types. In addition, recent scientific findings point out that:

- Chemical N form can influence both the rate of ecosystem change and possibly even whether N impacts will occur, at least in the short-term.
- Where the N source is agricultural and local, the effects will be more damaging and occur at lower N doses, mainly due to detrimental plant physiological effects of ammonia.
- Because plant species vary in their ability to use nitrate, nitrate effects will depend on the plant species present, and the likely risk from species invasion.
- Climatic factors interfere with ecosystem effects of N deposition and it is clear that climate both can emphasize and mitigate effects of N deposition.
- Following decreased ecosystem N input, some ecosystem parameters may revert quickly, while other components may show strong inertia. In some cases reversion to the original state may be impossible.
- More knowledge is needed to better understand where and if management intervention is appropriate to mitigate N effects.

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5.2 Working group report

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5.2.1 Conclusions and recommendations of group discussions

- It was concluded that the latest science supports and strengthens the already established empirical critical loads approach, encouraging their use in environmental decision making.
- The workshop concluded that there are no acceptable exceedances above a critical load or critical level. Discussions regarding “acceptable exceedances” are not a science issue and should be addressed at a policy level. In order to improve the situation, one should aim at reducing nitrogen deposition below the critical loads and levels.

- New data have strengthened the view that it is important to consider different nitrogen forms when evaluating effects of nitrogen deposition. It was concluded that evidence of responses for the different nitrogen forms is consistent across ecosystems and species. Moreover, because the effects from nitrogen deposition differ between different nitrogen forms (dry/wet deposition and oxidized/reduced nitrogen) it is important to evaluate their effects independently. Hence several types of critical loads/levels for a particular habitat type are needed. For example, the critical level for ammonia may be well below the critical load set for total nitrogen deposition. Hence it is important that both critical loads and levels are used.
- Important new data from Southern Europe have emerged over the last five years, for example, during the workshop results from experiments and surveys conducted in Portugal and Spain were presented. These should inform future revisions of critical loads for nitrogen.
- The workshop concluded that improved conditions following reduction in nitrogen deposition are only relevant when nitrogen deposition is reduced below the critical load/level. Reduction of exceedance will only improve the situation in the sense that it reduces the risk of further worsening of the effects. Information about the effects on recovery time following reductions below critical loads/levels is still largely lacking. Available data suggest that the rate of improvement will differ depending on type of function/species studied, and is often site specific.
- It was concluded that management to reduce the impact of nitrogen deposition will only work in combination with reductions in nitrogen deposition and should not be seen as an alternative to reducing the nitrogen deposition. For semi-natural habitats, positive effects from reducing the nitrogen inputs will only be possible in combination with appropriate management.
- The workshop agreed that there are important interactive effects between nitrogen deposition and climatic factors. Therefore a changing climate may also influence the effects of nitrogen deposition. Currently, the knowledge of such interactive effects, and how they may change with a changing climate is, however, poorly understood. The climatic factors most important for interactive effects with nitrogen are also the most uncertain in climate change modelling (e.g. precipitation), making predictions of future interactions between nitrogen deposition and climate change difficult.
- It is recommended that future research should prioritize the assessment of relative impacts of different nitrogen forms in relation to critical thresholds and dose response relationships, the relationships between nitrogen dose and site- and landscape-level management practices as a basis for minimizing adverse effects on ecosystem integrity, and the quantification of the interactive effects between climate change and nitrogen deposition.

5.2.2 Introduction

Actions to manage the Natura 2000 network and to assess conservation status must be based on a sound scientific understanding of how reactive nitrogen deposition causes impacts on sensitive habitats. The working group reviewed the latest science on the effects of nitrogen (N) deposition and concentrations on Natura 2000 sites, including the use of bio-indicators, effects of N-form (e.g. NH_x vs NO_y) and the relationships between critical thresholds and biodiversity loss.

Nordin et al., (this volume) summarizes established and new science on the effects of nitrogen deposition on ecosystems and considers the potential for improved assessment of N deposition impacts on Natura 2000 sites. The working group discussion was organised around five key issues identified from Nordin *et al*., (this volume):

- How does N deposition affect the structure and function of different habitat types?
- Is the chemical form of N deposition important?

- What is the potential for use of on-site management for improving conservation status?
- Interactions between N deposition and climate and climate change.
- How reversible are N deposition effects?

Group members were given the opportunity to give presentations concerning the topic. Some of the points and conclusions from the presentations are referred to in this summary. However, more detailed descriptions can be found in the papers in this volume (see sections 5.3 to 5.11).

5.2.3 Highlights of discussion and views expressed

Key issue 1: How does N deposition affect the structure and function of different habitat types?

The impacts of nitrogen deposition on structure and function were summarised as:

- direct toxicity of gases and aerosols,
- eutrophication, resulting in changes of species composition (more nitrophytic species) and sometimes reduction in species richness,
- soil-mediated effects of acidification (more acid-resistant species),
- increased sensitivity to stresses and disturbances (drought, frost, pathogens, herbivores).

The impacts are very complex, have many interactions, and are working on different timescales. Discussion of this key issue centered on the following topics which had been highlighted by other working groups:

- What are the strengths and limitations of the critical load/level approach?
- What is the minimum detectable effect above a critical load/level?
- What indicators/biomonitors can be used?

Critical load/level

Empirical critical loads for nutrient nitrogen are based on total nitrogen deposition and do not consider different nitrogen forms separately. Conversely, there are separate critical levels for ammonia and oxides of nitrogen. Critical levels for NO_x (30 μgm^{-3}) were established in 1992 (UBA, 2004). Areas most at risk of exceedance of the NO_x critical level are those in or close to urban or industrial areas or close to major roads. Exceedance of ammonia critical levels is more widespread in Natura 2000 sites (which tend to be located in rural areas). New critical levels for ammonia were approved by the UNECE in 2007. They are considerably lower than the former critical level, and incorporate an element of long-term protection cf critical loads. The 'long-term' annual average critical level is now 1 $\mu g\ NH_3\ m^{-3}$ for lichens and bryophytes and ecosystems where lichens and bryophytes are a key part of the ecosystem integrity. The 'long-term' annual average critical level for higher plants (e.g. heathland, grassland and forest ground flora and their habitats) is 3 $\mu g\ NH_3\ m^{-3}$, with an uncertainty range of 2–4 $\mu g\ NH_3\ m^{-3}$. The monthly mean value is 23 $\mu g\ NH_3\ m^{-3}$ to address the possibility of high peak emissions during periods of manure spreading. The impact of peak concentrations of ammonia is not well researched. Other nitrogen forms were discussed by the group. No critical levels exist for HNO_2. Evidence was provided of effects on Scots pine at very low concentrations (Sakugawa and Cape, 2007), PAN and HNO_3.

The empirical N critical loads are based on results from published experiments or surveys (see Nordin et al., this volume) The most useful data are those derived from long-term experiments (5-10 years or longer) using realistic N doses, conducted in areas with low N-deposition, with good estimate of background deposition.

Critical loads encompass dry and wet deposition of several reduced and oxidized nitrogen compounds. However, in most monitoring programmes only inorganic nitrogen is measured. However, organic N contributes accounts for around 30 per cent of UK wet N deposition (Cape, pers comm.). Presently, the origin of the organic nitrogen is unknown. Although spatially associated with NH_4^+ in rain, its concentrations have a different seasonal pattern. Organic N is currently not included in the assessment of eutrophication and critical loads. Further knowledge of its origin and its potential importance are needed before it can be evaluated whether this N form should be included in assessment of eutrophication and critical loads.

There has not been a validation of critical loads/levels against effects on invertebrates. This may be important in a biodiversity perspective, because invertebrates are a species rich group that often significantly contribute to the overall biodiversity of a habitat.

Indicators/biomonitors
The working group concluded that the Annex 3 (Ecological indicators) from the report made at a workshop on critical load in 1999 in Copenhagen (Arbejdsrapport DMU nr. 121) remained relevant and this issue was only covered briefly during the discussion. This key issue was, however, indirectly covered in several presentations, as these included case studies with different types of organism. Presentations covered:

- The use of butterflies as indicators of nitrogen deposition impact (see Feest, this volume)
- Measurements of nitrogen content of mosses across Europe and comparison with modeled data (see Harmens *et al.*, this volume)
- Species richness in calcareous grassland and correlations with nitrogen deposition (see Alard *et al.,* this volume)
- Lichen functional diversity and nitrogen content (see Pinho *et al.,* this volume)
- C/N ratio as an indicator of N leaching (Forsius, pers comm.).

What is the minimum detectable effect above and below a critical load/level?
This question had been put to the group to inform discussions under Theme 1: assessment of impacts on Natura 2000 sites under Article 6.3 of the Habitats Directive (see Section 3.2). The working group agreed that the detection level is a matter of resource availability. More resources imply that detection levels can be improved. Better replicated experiments will increase the statistical power, which enables detection of smaller effects. When the critical load or level is exceeded, it was concluded by the group that discussing minimum detectable effects is irrelevant. Once the N input reaches the critical load/level any further increase will, by definition, lead to an increased risk of negative effects. The more the critical load/level is exceeded the greater the risk of negative effects. Because the critical load/level describes an increased probability of negative effects when the load/level is exceeded, the exact response among sites will vary. Furthermore, for an individual site, negative effects may occur even though the N deposition is below the critical load/level, while for another site no negative effects may be apparent when the critical load/level is exceeded. However, once the critical load/level is exceeded, 95 per cent of the sites will show negative effects. Acceptance of exceedance above the set critical loads/levels is thus a political rather than a scientific issue: Science can only provide the evidence to help inform policy makers' decisions. In order to ensure protection of Natura 2000 sites from elevated N input deposition needs to be less than the critical load. The question is therefore, how can you best achieve this? Data were presented from field studies with N-addition experiments conducted in grassland and arctic/alpine areas. The data show that the more the critical load is exceeded the greater is the reduction in species richness (Bobbink, 2008). Furthermore, the datasets suggest that the established value for the empirical critical loads remains well supported by recent experimental data.

The working group also agreed that the timescale, geographical and spatial dependence need to be considered when the likelihood of detecting effects of elevated N input are discussed. The empirical critical load/level concept is designed to protect an ecosystem over a time–period of ~ 30 years. This implies that there often is a time lag between when the critical load/level is exceeded and when the negative effects of the exceedance become detectable. Further, effects are not necessarily linear related to N deposition, as they may depend on interactions with other factors e.g. drought and pathogens. This implies that negative effects from exceedance of the critical load/level may not be seen until such an event/episode occurs, potentially a further time lag. The effect of N deposition will also depend on the history of the site (cumulated deposition, management history), which may cause variation in the response to exceedance of the critical load/level.

A potential problem with using the concept of critical loads for protecting Natura 2000 sites is that critical loads are not designed to protect individual species, but rather to ensure that the N input is below the level responsible for negative effects on the habitat/ecosystem. To ensure protection to all species and functions one possibility is to use the most N sensitive (according to current knowledge) of "characteristic" species or the most N sensitive functions of the habitat to define the critical load or level for a habitat. By ensuring that such target species/functions are protected from the negative effects of N deposition, other parts of the system will be automatically protected. It was also stated that it is important to consider both critical loads and levels, since even if the critical load (based on total N deposition) is not exceeded the critical level of e.g. ammonia may still be exceeded for highly sensitive species.

Key issue 2: Is the chemical form of N deposition important?

The working group focused on the following key questions:

- How do effects from reduced and oxidized N forms differ?
- What are the differences between wet and dry N deposition?

There is clear evidence that N effects depend on the form in which N is deposited (see Nordin et al., this volume; see Table 5.1). Dry deposition of gaseous ammonia, per unit N deposited, causes more damage than the equivalent amount of wet deposited ammonium, which again is more damaging than the equivalent dose of wet deposited nitrate in most instances. The effects of ammonium on sensitive lichens and mosses are more detrimental than those of nitrate. Effects of wet deposited ammonium and nitrate on higher plants depend strongly on soil pH at the site. Currently, there are insufficient data to establish separate critical loads for NO_y and NH_x. However, during the discussion it was noted, that the relative importance of NH_x compared to NO_y is increasing due to greater reduction of emissions of NO_x relative to NH_x.

In an experiment in Mediterranean ecosystems in Portugal, nitrogen was added as oxidized and reduced N in combination, or only in the reduced form (Dias *et al.*, this volume). After one year of N treatment, effects were seen on species richness for both types of treatment. Diversity, expressed as Shannon diversity index, increased when oxidized and reduced nitrogen was added in combination, but decreased when N was added in reduced form only. The effect of wet deposited ammonium and nitrate on higher plants depends strongly on soil pH and can be summarized as follows:

- mineral soil pH 4.5-5: (Ca/Al buffering range) uptake of NH_4^+ reduces pH leading to increased risk of Al-toxicity, and potentially base cation deficiency.
- soils above pH 5:Acidification effect of NH_4^+ is not so pronounced, the effect is rather a change in species composition through eutrophication. Competitive relationship between species will shift, resulting in changed composition and loss of species, e.g. such that are unable to exploit

Table 5.1: Characteristics of the different depositing nitrogen compounds (Sheppard *et al.*, this volume)

Ammonia (NH_3)	Ammonium (NH_4^+)	Nitrate (NO_3^-)
Deposits close to source	Naturally occurring in soil (mineralization)	No reports of direct damage to plants
Highly reactive and alkaline	Dominates in acid soils, as pH increases more will be nitrified	Effects soil mediated through eutrophication, acidification and competition
Effects most likely mediated above ground	Reduces growth and survival in sensitive species	
Effects are concentration driven	High concentration potentially toxic	
Close relationship between effects and proximity to sources	Soil mediated effects through acidification and eutrophication	

increased N availability due to nitrification. However, if nitrification of the ammonium fails in these soils, due to the acidity generated, ammonium can accumulate to toxic levels.

- acidic soils pH < 4: In general, plants have a greater tolerance to NH_4^+. However if N input is high, nitrification may occur (normally less important in such soils), which may induce changes in plant species composition as it will favor a relative few number of species with good capacity to utilize nitrate.
- Insufficient data are available for calcareous soils.

Key issue 3: How reversible are N deposition effects?

The working group focused on the following key questions:

- What is the baseline?
- How can we measure improvement (i.e. effects of reduced N input above and below critical loads/levels)?
- Time horizons for improvement?

Baseline

It is difficult to define a baseline for most of the sites within the Natura 2000 network. Many Natura 2000 sites have a long history of high N deposition and negative effects such as loss of species and changed species composition have probably already occurred, a long time ago. For many sites the full extent to which N deposition has affected species richness and diversity is unknown, and will probably never be fully understood. During the discussion it was concluded that it was impossible to reach a consensus on a common definition on what the baseline should be.

It is important to distinguish between the situation above and below the critical load/level. If the critical load/level is already exceeded, a reduced N input will result in a decreased risk of a worsening of the effects rather than recovery. Only when the N input is reduced to below the critical load/level will recovery in the real sense be possible. The available literature on improved conditions/recovery from N induced effect is limited (see Nordin *et al.*, this volume). It is evident that the rate of improvement will differ between different components of the ecosystem and differ between different sites depending on geographical location, climatic conditions, N deposition history, and in some cases also on site management, and management history.

A wide range of parameters may be used to assess improvement from lowered N deposition. Indicators that can be used are: increased species richness and increased occurrence of N sensitive

species, as well as recovery of the original species composition (in cases where this is known and species have not been lost) or changes into a composition that resembles more pristine conditions. Use of the Ellenberg index will be restricted to some areas in Europe and cannot be used for all Natura 2000 sites, as Ellenberg index values are lacking for many species outside central/Western Europe. Chemical characteristics can also be used to measure improvement. For example reduced N leakage, reduced N mineralization rate, reduced exchangeable soil N, reduced concentrations of N rich amino acids in plant tissue are all potential indicators of improved conditions.

Time horizon

If N input is reduced below the critical load/level, recovery time from direct N effects and soil mediated indirect N effects can be assumed to differ between ecosystems. When the affected species are still present, recovery from direct effects is, in general, assumed to occur faster than recovery from soil mediated effects. It is important to note that improvement from N induced effects under conditions with lowered N input may not necessarily show the same dose response relationship as when the N load was increasing.This implies that some N induced effects will not necessarily recover when the N deposition is restored to the 'original' level. In some situations improvement or recovery may be difficult, or not possible, because key functions or key species may have been lost from the habitat. The time horizon before improvements show depends on the type of effect (e.g. acidification, leaching, species composition etc.). In some cases the improvement will also be dependent on management regimes (e.g. moving, grazing). Some ecosystem parameters may show rather rapid improvement following reduction of the N input. For example, improved conditions for exchangeable soil N or N concentration in *Sphagnum* mosses growing on bogs can be rapid, whereas other parameters such as re-establishment of species or recovery of original species composition are slow processes.

Key issue 4: What is the potential for use of on-site management for improving conservation status?

During the discussion it was concluded that active management, that removes nitrogen from the system, should not be seen as an alternative to lowering the N deposition at a site. Likewise it was concluded that intensified management cannot justify increased N deposition at a site. It was also concluded that management strategies are more or less confined to semi-natural habitats, and for many Natura 2000 sites there are no available management strategies today that will help to improve the situation. It is important to distinguish between management for "restoration" and management as a means of maintaining function and form of semi-natural habitats such as heathland and grassland (i.e. non-climax vegetation types that are man-made). In such semi-natural habitats active management like mowing, burning and/or grazing is necessary to maintain the desired species composition and function of the system. Such management will result in removal of nitrogen, which helps the system to maintain its nitrogen limitation. For wetlands many habitats have been drained, and here restoration of the hydrological conditions may be a prerequisite before benefits can be expected to come from lowered N deposition. It was also noted that mismanagement of water is a very important threat to Mediterranean ecosystems (Tsiouris, pers. comm.).

Key issue 5: Interactions between N deposition and climate and climate change

The working group concluded that there are undoubtedly important interactions between N deposition and climate, and that a changed climate will interact with N deposition and these need to be addressed in discussions of the effects of N deposition. It was also concluded that there is a large gap in knowledge concerning such interactive effects. It is evident that N effects interact with climate factors such as drought, frost, precipitation and temperature. Hence, alteration of such climatic factors will likely alter the effect of N deposition. The climatic factors that probably are most important for interactive effect with N are also the factors that are most uncertain in climate change modeling (e.g. changed precipitation pattern), making predictions of future interactions

between N deposition and climate change difficult. Habitats already exposed to N deposition will, regardless of future climate, still be nitrogen rich systems. Such systems may be sensitive to establishment of invasive or new species.

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5.3 Defining a biodiversity damage metric and threshold using Habitat Directive criteria

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Summary

- Changes in the environmental suitability of a site for particular plant species in response to nitrogen (N) load can be predicted fairly objectively using model chains such as MAGIC-GBMOVE.
- Lists of positive and negative indicator plant species, such as those in UK Common Standards Monitoring Guidance, provide operational definitions of habitat quality and damage.
- Whilst N-sensitive species can provide early warnings of change, they may not be representative of the desirable features of the habitat.
- A metric of habitat quality is proposed, based on predicted environmental suitability for positive and negative indicator species.
- This metric allows assessment of the impact of N pollution on habitat quality as defined independently of the N effects research community.

5.3.1 Introduction

National emissions limits of reactive nitrogen (N) are established under the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) using the critical load approach to reduce damage to habitats. This approach requires operational definitions of a metric which can be used to assess habitat quality, and a threshold above which damage can be said to have occurred. Metrics

such as soil pH and soil total C/N ratio have been used, but it is increasingly recognised that changes in these abiotic measurements may not be synchronised with significant changes such as species loss and invasion. There is thus a need for damage definition criteria that are explicit and repeatable (Nicholson *et al*., 2009) and that can be related directly to habitat quality as defined by conservation experts.

The Habitats Directive aims to maintain biodiversity in Europe. It definines habitats and species of Community importance and provides protection measures. (EEC, 1992). Comparatively few species are listed in the annexes, and these are rather rare. By contrast, some of the habitats listed are extensive and thus the Annex I definitions and criteria are applicable to a larger set of sites. Member States are required to designate Special Areas of Conservation (SACs) to protect habitats and species listed in Annex I and II respectively (see Whitfield and Strachan, this volume), although responsibility for these habitats and species also extends beyond these areas. Assessments must be made of plans or projects likely to have a significant effect on special areas of conservation in view of their conservation objectives. For habitats, conservation status is defined as the sum of the influences that may affect long-term natural distribution, structure and functions as well as survival of typical species. However, structure and function are not easily defined without reference to particular species. Plant species assemblages and their phytosociological names are primary definitions for terrestrial habitats within the Habitats Directive. These classifications can usually be related to lists of typical species, for example using the National Vegetation Classification (Rodwell, 1991-2000) in the UK. The quality of vegetation on a site is typically assessed in terms of conformity to descriptions of a typical assemblage of species, the presence of rare or typical species, and/or the absence of untypical species. In the UK, assessment of the condition of interest features on special areas of conservation includes a standard procedure known as Common Standards Monitoring (CSM), which lists positive and negative indicator species for many habitats.

This emphasis on species is useful not only to derive concrete and measurable criteria for site assessment, but because responses of individual species to reactive N pollution can be predicted empirically. Niches for individual plant species have been defined in relation to environmental factors in models such as MOVE (Latour and Reiling, 1993; Wamelink *et al*., 2009), VEG (Belyazid *et al*., 2006) and GBMOVE (Smart *et al*., 2010). By solving these models with respect to time series of abiotic factors generated by process models, the likely occurrence of individual species can be derived for different scenarios (de Vries *et al*., 2010).

For environmental suitability predictions for a set of species to be interpreted in terms of conservation status, it is necessary to refer to lists of desirable and undesirable species. The loss of N-sensitive species is a major basis for defining empirical critical loads (Bobbink *et al*., 2003), and N-sensitive species are useful in providing early warnings of change. However, N-sensitive species may not be important components of a particular habitat, and there is a worrying circularity in defining damage from reactive nitrogen as damage to N-sensitive components. A more objective approach would be to use existing lists of desirable and undesirable species defined beyond the N effects research community. Desirable and undesirable species for particular habitats have already been identified in many cases. Where such lists are not yet available they could be derived from information about species scarcity such as the IUCN Red List (Mace at al., 2008), by indicator species analysis (e.g. Ejrnaes *et al*., 2004) and/or by consultation with habitat experts and the public.

An assessment of a site's conservation status can be derived from a list of desirable and undesirable species, if indicators are available of the actual or likely occurrence of these species on the site. Models derived from floristic survey data can predict likelihood of occurrence, but many other factors such as dispersal traits and local presence govern actual occurrence on a particular site so such predictions are best interpreted as the suitability of the site for the species. Different statistics

can be derived from site floristics, such as the presence / absence, abundance or frequency of each species. Such data could in principle be converted into a habitat quality metric using the approach advocated here, but this is beyond the scope of the current paper. Rather, we propose a method for weighting and aggregating predictions of environmental suitability for individual species, for use in interpreting the outputs from models of effects of N pollution on plant species. This simply uses weightings of +1 for species listed as positive indicators, -1 for species listed as negative indicators, and 0 for species not listed for a given habitat.

5.3.2 Aims and objectives

- To summarise habitat suitability predictions for a large set of species into a single metric of habitat quality.
- To base this metric on existing definitions of site condition.
- To explore the use of this metric in predicting habitat damage for critical load exceedance calculations.

5.3.3 Results and discussion

Historic deposition sequences of N and sulphur derived from the FRAME model (Fowler *et al*., 2005) and Gothenburg scenario projections of future deposition were used to generate time series of soil C/N ratio and pH for blanket bog at the Moor House long-term monitoring site, using the MAGIC model (Cosby *et al*., 2001) (Figure 5.11a). These time series of abiotic variables were used to calculate probabilities of occurrence for species regarded as positive (Figure 5.1b) and negative (Figure 5.1c) indicator species for UK blanket bog in the UK Common Standards Monitoring guidance (JNCC, 2006). An overall habitat quality index Q was calculated as

$$Q = \sum_{i=1}^{p} \frac{\left(\frac{P_i}{P\max_i} \right)}{p} - \sum_{j=1}^{n} \frac{\left(\frac{P_j}{P\max_j} \right)}{n}$$

where P_i is the probability of occurrence of positive indicator species i and P_j is the probability of occurrence of negative indicator species j (Figure 5.1d).

These results illustrate the potential of the approach. Even though species were not selected for inclusion in the indicator calculation on the basis of their nitrogen-sensitivity, there is a tendency for probability of occurrence of positive CSM indicators for this habitat to decline with an increase in N saturation. The probability of occurrence for negative indicator species did not clearly increase with an increase in N saturation, but nevertheless the overall habitat quality index showed a clear and continuing decline under this N emission scenario.

The proposed metric has the advantage of using the entire set of species named in previous condition assessment, reducing the danger of bias towards well-studied and charismatic species (Sitas *et al*., 2009). The metric will be influenced by the number of indicator species included, and it would therefore be useful to standardise the number of positive and negative indicator species. Another issue is the relative weighting of positive and negative indicators and other species; giving a greater weighting to more rare or typical positive indicators, and a more strongly negative weighting to particularly negative indicators, would likely give a more responsive metric. This advantage should be assessed against the increased subjectivity involved with assigning different weightings to different species.

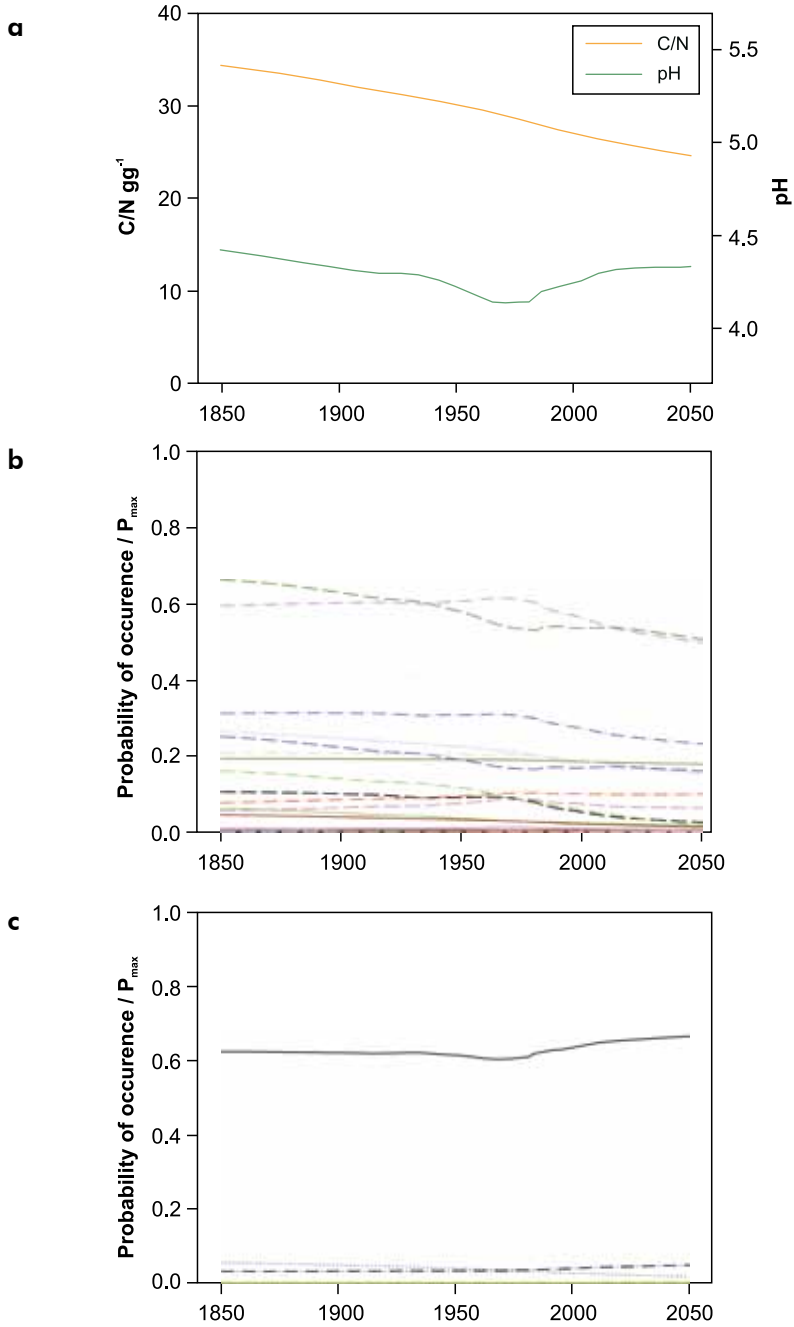

Figure 5.1: Simulated changes in blanket bog at Moor House long-term monitoring site, Cumbria, UK, under the Gothenberg emission scenario: (a) Soil pH and C/N ratio simulated using the MAGIC soil chemistry model; probabilities of occurrence for (b) positive and (c) negative Common Standards Monitoring indicator species for blanket bog obtained from the GBMOVE species niche model, rescaled to Pmax, the maximum probability of occurrence for the species (soil water content and canopy height were assumed to be constant).

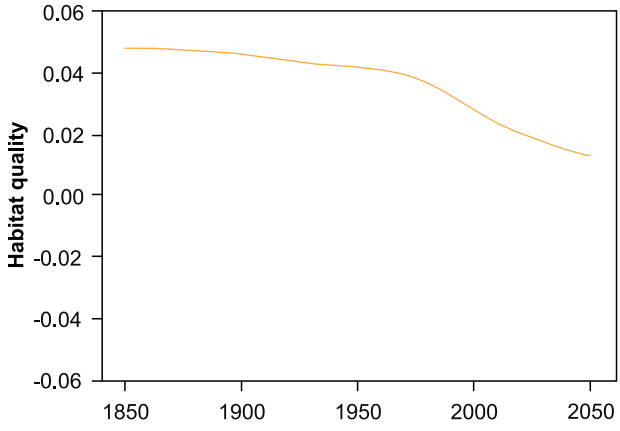

Figure 5.1 (d): Simulated changes in blanket bog at Moor House long-term monitoring site, Cumbria, UK, under the Gothenberg emission scenario: (d) overall habitat quality Q.

If it can be agreed that such a habitat quality index at least partially reflects conservation status, the next issue to be resolved is how to set a threshold level of Q below which the habitat can be said to be damaged. Empirical critical loads as currently defined (Bobbink *et al*., 2003) provide a basis for calculating threshold values for indicator species' probabilities of occurrence. In ongoing work, we aim to extend the approach outlined here to define threshold levels of habitat quality for different habitats.

5.3.4 Conclusions

- Operational definitions of conservation status and habitat quality can be derived from the occurrence of positive and negative indicator species as defined by conservation experts.
- Habitat suitability predictions for positive and negative indicator species can be summarised into a habitat quality metric.
- This habitat quality metric is a direct and useful link between N pollution and nature conservation policy mechanisms.

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5.4 Influence of nitrogen deposition on plant biodiversity at Natura 2000 sites in Spain

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Summary

- Worldwide scientific evidence has been accumulated showing that anthropogenic emissions of nitrogen compounds to the atmosphere are affecting terrestrial ecosystems, thus threatening the world's biodiversity.
- This problem may be relevant in Spain, where contrary to other European countries, NO_x emissions have significantly increased since 1990 until present.
- An occurrence index for nitrophilous species of herbaceous plants, mosses and lichens was calculated to check the potential influence of nitrogen emissions on plant biodiversity.
- Data from the latest eleven decades suggested an increasing trend in the reported occurrence and relative richness of nitrophilous species throughout Spain.
- The proposed methodology is useful for the detection of temporal and spatial changes in plant biodiversity, allowing the identification of potential threats to sensitive biotopes within the Natura 2000 Network.

5.4.1 Introduction

An acceleration of biodiversity loss has occurred worldwide, linked to the over-exploitation of natural resources, habitat destruction and climate change (Sala *et al*., 2000, Thuiller *et al*., 2005). Furthermore, with the intensification of agricultural and industrial activities over the past several decades, large amount of N compounds have been released to the atmosphere (Galloway *et al*, 2003), resulting in elevated N deposition in terrestrial and aquatic ecosystems (Asner *et al*., 2001; Matson *et al*., 2002). In this regard, several experimental studies have reported that nitrogen (N) enrichment reduces plant diversity, leading to the conclusion that anthropogenic N deposition is a threat to global biodiversity (Stevens *et al*., 2004; Davies *et al*., 2007; Xiankai *et al*., 2008).

One of the main actions that have been undertaken at the European level to contribute to the maintenance of biodiversity is the establishment of the Natura 2000 network. The network is composed of Special Areas of Conservation (SAC), designated under the Habitats Directive (Council Directive 92/43/EEC), and of Special Protection Areas (SPAs), designated under the Birds Directive (Council Directive 79/409/EEC). Spain has designated approximately 1400 SACs and 600 SPAs, representing almost 30 per cent of Spanish territory.

Since nitrogen enrichment has been considered one of the most important anthropogenic factors influencing ecosystem structure and function, the European Union has implemented several measures to abate the emissions of nitrogen compounds. As a result, the majority of EEA-32 countries have reported lower emissions of NO_x and NH_3 in 2007 compared to 1990. An opposite trend has occurred in Spain, as the emissions of these compounds increased by 19.9 per cent and 24.5 per cent, respectively, in the same period. This increasing trend determined that 2007 emissions of NO_x and NH_3 exceeded by 39 per cent and 20 per cent, respectively, the Spanish levels included in the National Emission Ceilings Directive (NECD).

In order to meet Spanish obligations with the NECD and Gothenburg Protocol Spain approved in December 2007 the second National Emission Reduction Programme (II PNRE) aiming to achieve significant reductions in NO_x and NH_3 emissions from 2010 onwards. According to this Programme, several measures are being implemented, including the development of new strategies for sustainable mobility, the reduction of NO_x emissions in stationary natural gas engines and the improvement of the national statistics in agriculture (MARM, 2009).

5.4.2 Aims and objectives

In Spain the number of studies dealing with long-term nitrogen deposition on a widespread area is almost negligible. Thus, it is very difficult to estimate whether the above-mentioned increasing trend in nitrogen emissions has resulted in the nitrogen enrichment of Spanish ecosystems.

Taking into account this lack of information, we aimed at assessing trends in nitrophilous plants and nitrogen emissions in the last decades by:

- Developing a large-scale approach to map and quantify temporal biodiversity changes related with nitrogen emissions.
- Calculating a bioindicator index from the occurrence records of nitrophilous herbaceous plants, mosses and lichens collected in Spain and currently indexed at the Global Biodiversity Information Facility (GBIF).
- Detecting the areas in Spain that may be at risk of recording structural changes in vegetation induced by nitrogen deposition (hotspots), including the Natura 2000 sites.

5.4.3 Material and methods

GBIF (www.gbif.org/) provides access to millions of scientific data records supplied by a wide range of institutions and organizations from all over the world. The strength of the data shared by the GBIF network (species occurrence records and names and classifications of organisms) relies on their potential to be represented geospatially, as geographical coordinates of the records are usually provided. In this work we have used the GBIF database to extract information about the spatial and temporal evolution of nitrophilous herbaceous plants (186 taxa), mosses (44 taxa) and lichens (78 taxa) in Spain from 1900 to 2008.

A nitrophilous index was calculated using over 750,000 occurrence records of plants that had been indexed at GBIF for the Iberian Peninsula by March, 2009. To construct this index, the annual number of records of nitrophilous taxa was obtained for the period 1900-2008. Next, the obtained value was divided by the total number of plant records documented every year, obtaining a nondimensional index representing the proportion of nitrophilous plants for each year of study.

These data, along with the latitude and longitude values, were used to elaborate nitrophilous index anomaly maps for the whole twentieth century. Data from recent years (1990-2008) were contrasted against the 1900-1989 baseline of previous records. The records were pooled into centered half-degree cells. Grid cells were coloured as a function of the nitrophilous index or the increase or decrease in percentage of nitrophilous plants with respect to the baseline.

The emission data model for NO_x and NH_3 for the period 1990-2006 was downloaded from the EMEP home page (www.emep.int/). In this case, longitude/latitude coordinates from EMEP database grid were also converted to centered square half-degree cells. The model's time precision is yearly, from 1990 onwards.

Finally, the index data were used to predict the effect on the SPAs and SACs included within the Natura 2000 network. The georeferenced site list was obtained from the EUNIS database

(www.eea.europa.eu/data-and-maps/data/natura-2000-eunis-database) and overlayed to the index anomaly maps, selecting the grid cells that included SPAs.

5.4.4 Results and discussion

The bioindicator index developed by the study showed a continuous increase of nitrophilous plants for the period 1900-2008 (Figure 5.2), thus suggesting a change in biodiversity composition related to nitrogen enrichment in ecosystems. This increase seemed to peak in the 1970 and 1980 decades, decreasing slightly in the last decade of the XXth century, and increasing again in the first decade of the XXIst century.

Both the N emission data model from EMEP and the more recent CHIMERE deposition model by Theobald *et al*., (2009) indicate an uneven distribution of anthropogenic N throughout Spain (Figure 5.3), with higher concentrations in the northeast, the northwest, and the south of the Peninsula.

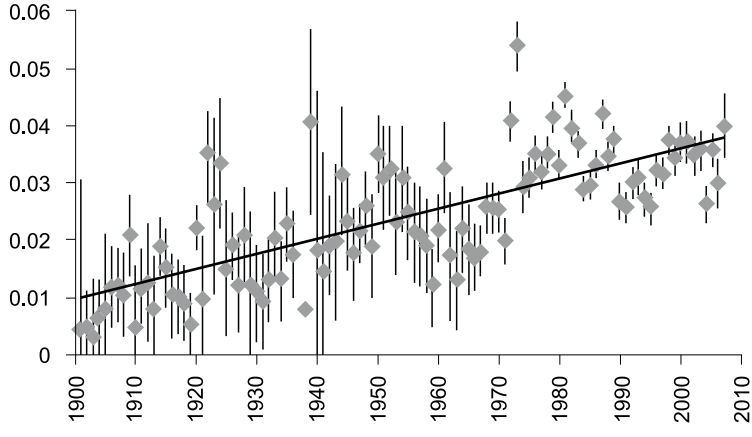

Figure 5.2: Variation of the plant nitrophilous index through 20th and 21st centuries in Spain

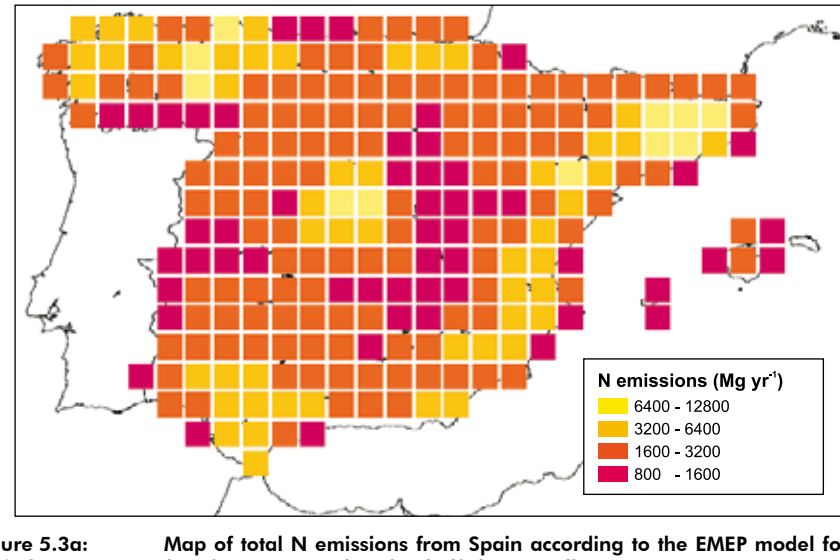

Figure 5.3a: Map of total N emissions from Spain according to the EMEP model for the period 1990-2006 (data from EMEP reduced to half-degree cell).

There are also disagreements. According to Theobald *et al*., most of N deposited in Spain is in the oxidized form, predominantly as nitric acid, as reduced nitrogen only contributes about 25 per cent to the total (in contrast with EMEP model ,which predicts a 52 per cent of reduced N deposition). Although the CHIMERE model has not been validated for N deposition (this would explain the different prediction of both models), these results can be used to qualitatively identify sites at risk, if not to predict absolute deposition rates.

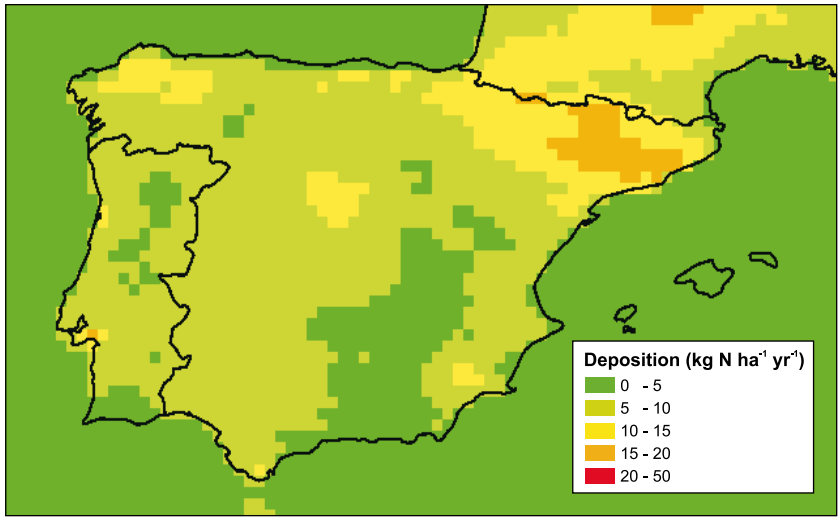

Figure 5.3 b: Map of predicted deposition within the modelling domain for oxidised and reduced nitrogen in the Iberian Peninsula (adapted from Theobald *et al*., 2009).

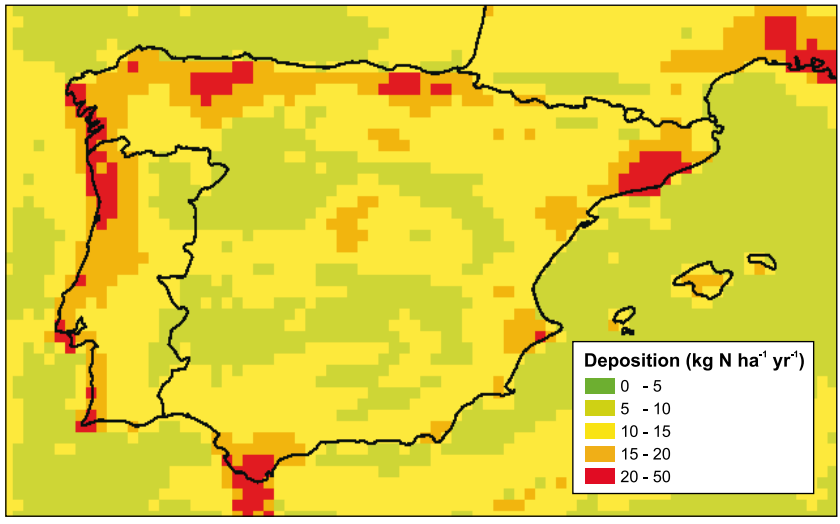

Figure 5.3 c: Map of predicted deposition within the modelling domain for oxidised and reduced nitrogen in the Iberian Peninsula (adapted from Theobald *et al*., 2009).

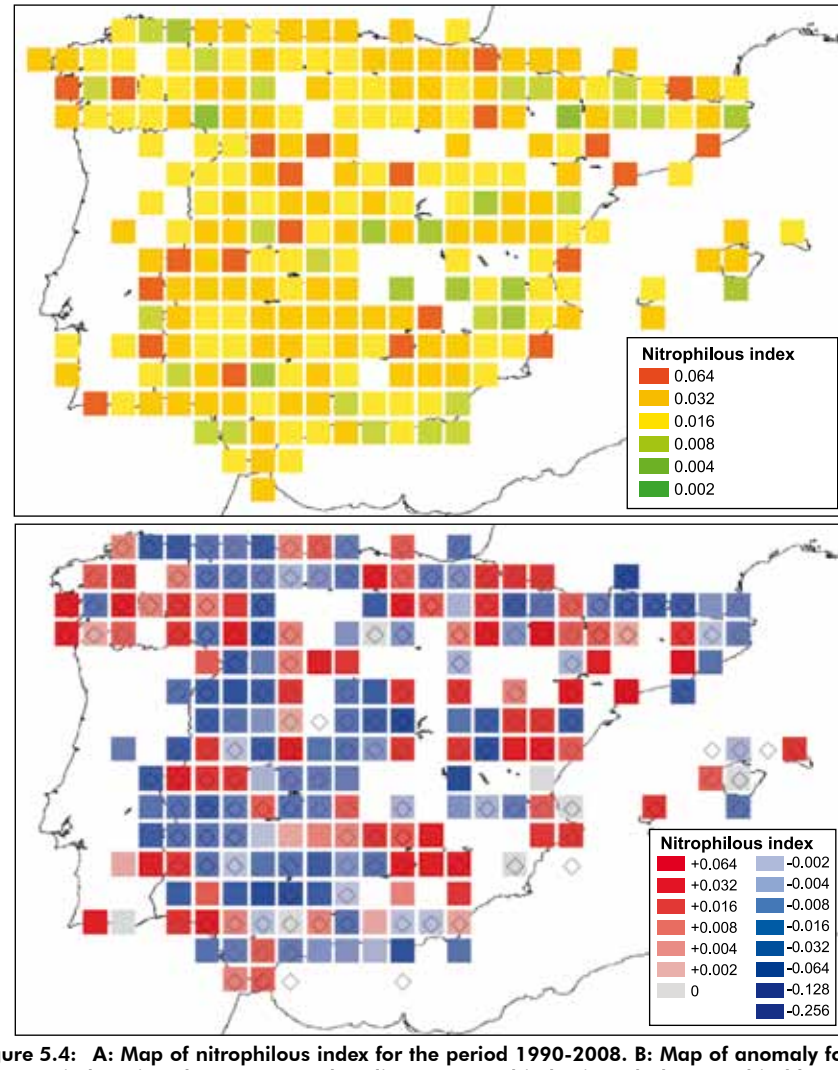

Figure 5.4: A: Map of nitrophilous index for the period 1990-2008. B: Map of anomaly for the same period against the 1900-1989 baseline. Increased index in red, decreased in blue. Half-degree cells containing one or more SPAs are marked with a diamond.

When the index was plotted, data indicated also an uneven distribution but did not seem to agree with the emission and deposition models (Figure 5.4). Therefore, the static value of the index would not appear to support the hypothesis of the effect of N on biodiversity.

However, when the index is plotted against the baseline of previous records (Figure 5.3b), there is a general agreement between the change in the occurrence index and the deposition models. Most increases in the index seem to occur in the northeast and northwest, and south of the Iberian Peninsula.

Therefore, these results seem to confirm that it is species richness (i.e. more nitrophilous species present as opposed to just the composition of existing species changing) that is affected by the changes in the N deposition.

Of the SPA and SAC site list in the Natura 2000 Network in Spain, 54 per cent experienced an increase in the nitrophilous index, whereas 44 per cent showed decrease and 2 per cent did not change. This suggests that a large number of sites would be suffering N-induced shifts in natural vegetation. Under these conditions, the Favorable Conservation Status (FCS) of a habitat is considered to be endangered, as the natural range of the species seems to be changing for the foreseeable future.

The proposed methodology could be used to derive empirical critical loads in the Spanish territory by relating both the time series of N deposition values and the temporal variations in structural changes in Spanish vegetation.

5.4.5 Conclusions

- The calculation of a bioindicator index from records of nitrophilous plants indexed at the GBIF has shown to be a useful method for the detection of temporal and spatial changes in plant biodiversity.
- A continuous increase of nitrophilous plants was detected in the Iberian Peninsula for the period 1900-2008, identifying numerous threats to biodiversity hotspots in the Natura 2000 Network.
- The obtained results suggest the need for developing more in-depth studies in order to confirm if the observed tendencies are related to nitrogen deposition.
- This methodology could be used in the future for the potential calculation of N critical loads.

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Data Sources
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5.5 Mosses as biomonitors of atmospheric nitrogen deposition - potential application at Natura 2000 sites

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Summary

- In 2005/6, carpet forming ectohydric mosses were tested for the first time as monitors of atmospheric N deposition at the European scale in 16 countries.
- The lowest total N concentrations in mosses were observed in northern Finland and northern parts of the UK, the highest concentrations were found in parts of Western, Central and Eastern Europe.
- The spatial distribution of the N concentration in mosses was similar to that of the total N deposition modelled by EMEP[1], except that the N deposition tended to be relatively lower in Eastern Europe.

1 Co-operative programme for monitoring and evalution of the long-range transmission of air pollutants in Europe

- The total N concentration in mosses showed the highest significant correlations with EMEP modelled deposition of different N forms, followed by urban and agricultural land use and population and livestock density.
- The total N concentration in mosses can potentially be used as a high spatial resolution tool in identifying Natura 2000 sites at risk from atmospheric N deposition.

5.5.1 Introduction

The European moss biomonitoring network was originally established in 1990 to estimate atmospheric heavy metal deposition at the European scale (Rühling, 1994). The moss technique is based on the fact that carpet forming ectohydric mosses obtain most trace elements and nutrients directly from precipitation and dry deposition with little uptake from the substrate. The technique provides a complementary, time-integrated measure of metal deposition from the atmosphere to terrestrial ecosystems. It is easier and cheaper than conventional precipitation analysis as it avoids the need for deploying large numbers of precipitation collectors with an associated long-term programme of routine sample collection and analysis. Therefore, a much higher sampling density can be achieved than with conventional precipitation analysis.

The European moss survey has been repeated at five-yearly intervals and has been coordinated since 2001 by the ICP Vegetation[2], a subsidiary body of the UNECE[3] Long-range Transboundary Air Pollution Convention (Harmens *et al*., 2008a; 2010). The most recent European moss survey was conducted in 2005/6; for the first time, 16 countries also determined the total N concentration in mosses at a total of almost 3,000 sites (Harmens *et al*., 2008b). A pilot study in selected Scandinavian countries had shown that there was a good linear relationship between the total N concentration in mosses and EMEP modelled atmospheric N deposition rates (Harmens *et al*., 2005). The potential of mosses as monitors of atmospheric N deposition and its limitations has been described in a number of studies (e.g. Solga *et al*., 2005; Pitcairn *et al*., 2006; Pesch *et al*., 2008; Salemaa *et al*., 2008; Zechmeister *et al*., 2008; Poikolainen *et al*., 2009). The 2005/6 survey was the first attempt to establish whether mosses can be used as biomonitors of atmospheric N deposition at the European scale.

5.5.2 Aims and objectives

- To establish whether mosses can be used as biomonitors of atmospheric N deposition at the European scale, including Natura 2000 sites.
- To provide, in the form of maps, spatial information on the distribution of total N concentrations in mosses across Europe and identify hotspots of N pollution.
- To identify factors contributing to the spatial variation of the total N concentration in mosses.

5.5.3 Results and discussion

Pleurozium schreberi (Brid.) Mitt was the most frequently sampled species, accounting for 41.3 per cent of the samples, followed by *Hylocomium splendens* (Hedw.) B.S.G. (19.0 per cent), *Hypnum cupressiforme* Hedw. (18.1 per cent), and *Pseudoscleropodium purum* (Hedw.) M. Fleisch (15.5 per cent). Other moss species constituted only 6.1 per cent of the mosses sampled. The sampling density varied between countries and in some countries mosses were only sampled in selected regions. Further details of applied methodologies have been described elsewhere (Harmens *et al*., 2008b; 2010).

2 The International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops

3 United Nations Economic Commission for Europe

The lowest total N concentrations in mosses were generally observed in northern Finland and northern parts of the UK (Figure 5.5a). In Finland there was a clear north-south gradient which continued into the Baltic States. In the UK, locally high concentrations were found in the Midlands and South-East. The highest concentrations were found in parts of Western, Central and Eastern Europe, in particular in Belgium, Germany, Slovakia, Slovenia and parts of Bulgaria and France. However, considerable regional variations in the total N concentration in mosses were observed in each country.

The spatial distribution of the N concentration in mosses was similar to the one of the total N deposition modelled by EMEP for 2004 (Figure 5.5b), except that the N deposition tended to be relatively lower in Eastern Europe. However, the relationship between total N concentration in mosses and modelled total N deposition, based on averaging all sampling site values within any one EMEP grid square, showed considerable scatter (Harmens *et al*., 2008b). Some of scatter can be explained by relating site-specific N concentrations in mosses with N depositions averaged per 50 x 50 km^2 EMEP grid. Actual deposition values vary considerably within each EMEP grid cell due to for example topography, vegetation, local pollution sources and climate. The apparent asymptotic relationship shows saturation of the total N in mosses above a N deposition rate of approximately 15 kg $ha^{-1}y^{-1}$. In Switzerland, however, the relationship was significantly linear (R^2 = 0.91) when based on measured site-specific bulk N deposition rates (Figure 5.6; Thöni *et al*., 2008). This suggests that the relationship is more robust when based on measured site-specific N deposition rather than modelled N deposition averaged over a larger area. There is a need to measure atmospheric N deposition at selected moss sampling sites in other countries too in order to further investigate the robustness of the relationship with total N concentration in mosses.

Bivariate Spearman rank correlation coefficients (rs) were computed to investigate the strength and direction of the statistical relationship between the total N concentrations in mosses and i) EMEP modelled N depositions and ii) additional factors that were expected to influence the total N concentration in mosses (Schröder *et al*., 2010; Table 5.2). Moderate correlation coefficients (i.e. $0.5 \leq rs < 0.7$) were observed for EMEP modelled N depositions ($0.55 \leq rs \leq 0.65$), independent of N form. Regarding regional land characteristics, the ratio of urban land uses in a radius of 100 km and agricultural land uses in a radius of 50 km around the monitoring sites (Corine Land Cover 2000 data) showed the highest correlations with the total N concentration in mosses (rs = 0.55 and rs = 0.54, respectively). Lower correlations ($rs < 0.5$) were observed for population and livestock density (Eurostat data), precipitation (Global Climate Dataset), distance to sea and altitude. In general, the total N concentration in mosses appears to mirror land use-related atmospheric N depositions.

In a pilot study in Germany, the moss sites where overlayed with the locations of Sites of Community Importance (SCI's). This allowed identifying SCI's with the highest total N concentration in mosses in Germany and therefore most at risk from high atmospheric N depositions (Figure 5.7). Such an analysis should be extended to other countries to establish whether the total N concentration in mosses can be used for identifying Natura 2000 sites at risk from high atmospheric N depositions.

Factors potentially influencing the relationship between total N deposition and N concentration in mosses require further investigation in order to establish the robustness of the application of mosses as biomonitors of atmospheric N deposition at Natura 2000 sites. Such factors include for example i) effects of N and climate on moss growth, ii) species-specific responses to N deposition, iii) the role of N speciation and iv) altitude (Harmens *et al*., 2008b). It might be that other N parameters, e.g. amino acid concentration or soluble ammonium concentration in mosses, are better indicators for potential effects of N deposition on Natura 2000 sites (Strengbom *et al*., this volume). The disadvantage of these measures is that they are more complicated to analyse and more expensive.

Table 5.2: Spearman rank correlation coefficients (rs) between N concentration in mosses and i) EMEP modelled deposition (50 x 50 km²) of different N forms and ii) other site-specific or regional characteristics. All coefficients were significant at p=0.001, except for livestock density with p=0.01.

EMEP deposition	r_s	Other predictors	r_s
Wet oxidised	0.65	Ratio urban land use (100km radius)	0.55
Total (wet + dry)	0.64	Ratio agricultural land use (50km radius)	0.53
Total wet	0.64	Population density	0.48
Dry oxidised	0.64	Livestock density	0.42
Wet reduced	0.62	Precipitation	0.25
Total dry	0.59	Distance to sea	0.25
Dry reduced	0.55	Altitude	-0.10

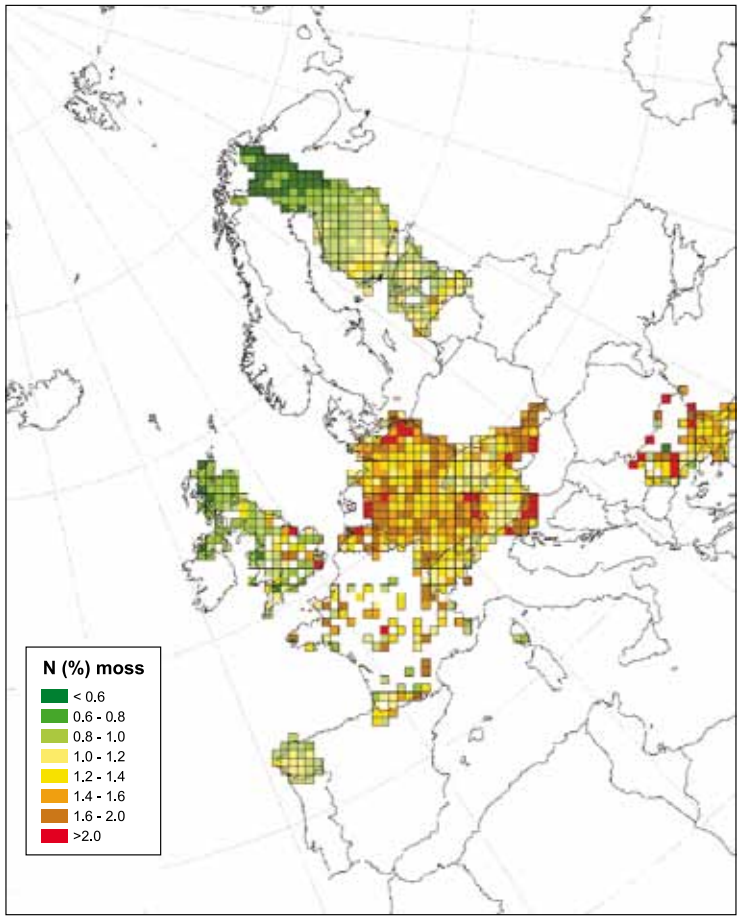

Figure 5.5a: Mean total N concentration in mosses per EMEP grid square in 2005/6

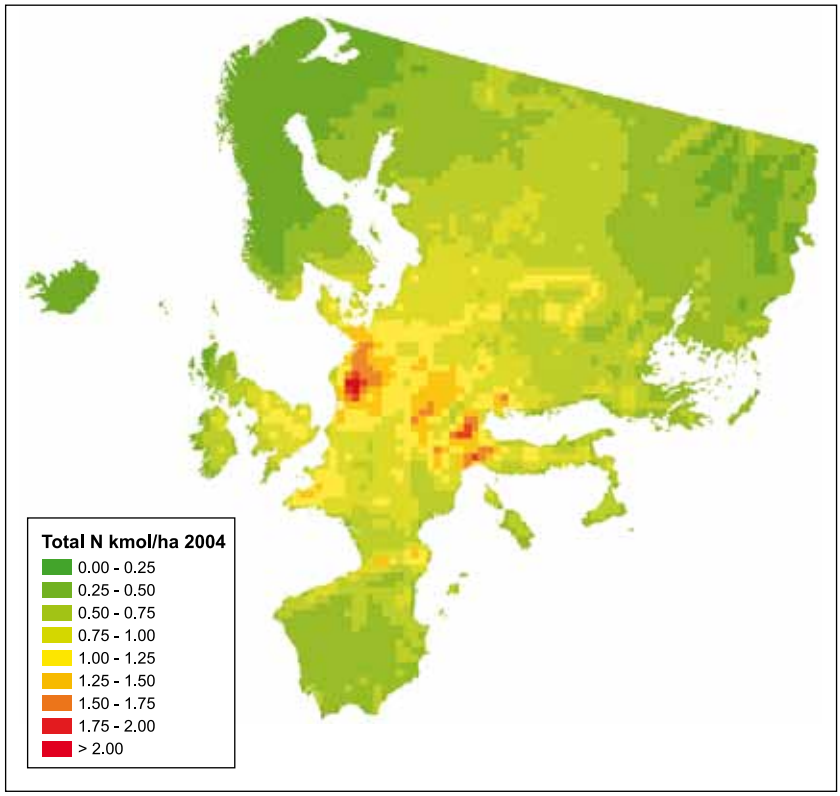

Figure 5.5b: N deposition per EMEP grid square in 2004

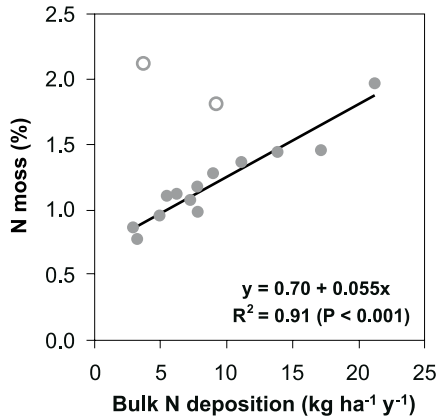

Figure 5.6: Relationship between measured bulk N depostion rate and N concentration in mosses at selected sites in Switzerland; the open symbols (samples possibly contaminated with faeces) were excluded from the regression (Thoni *et al.*, 2008)

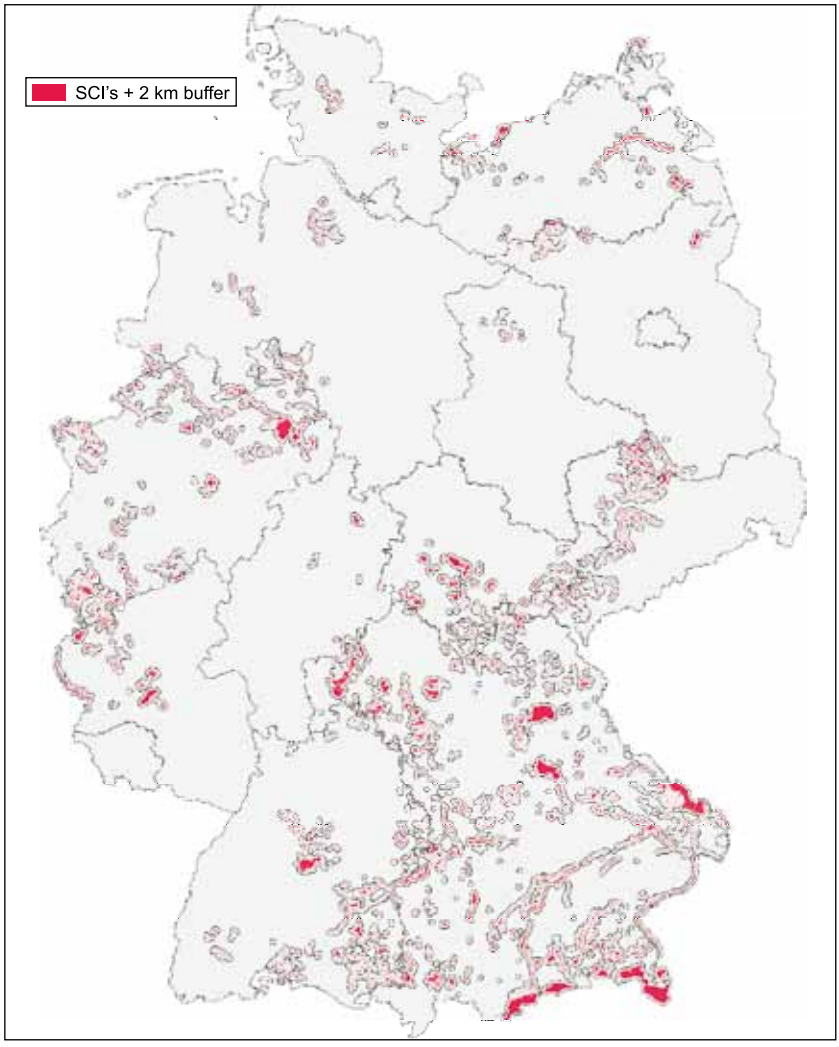

Figure 5.7: Sites of Community Importance (SCI's) in Germany with a total N concentration in mosses above the 75th percentile (1.36 per cent, calculated from the raster). The map of total N concentration in mosses was overlayed with the location of SCI's using a two km buffer around the SCI's. This maps identifies SCI's potentially most at risk from adverse effects of N deposition, however, SCI's might be at risk at concentrations below 1.36 per cent. Calculations by: W. Schröder, R. Pesch, M. Holy (University of Vechta, Germany), source of data on locations of SCI's: Federal Agency of Nature Conservation, Germany.

Most likely, a combination of the bioindicators/biomonitors will best describe the state on Natura 2000 sites (Nordin *et al*., this volume).

5.5.4 Conclusions

- The total N concentration in mosses can potentially be used in identifying Natura 2000 sites at risk from enhanced N deposition at a high spatial resolution.

- Measurements of site-specific N deposition and other site-specific characteristics are required to establish the robustness of the relationship between total N deposition and total N concentration in mosses.
- It is unclear yet whether the total N concentration in mosses can be used as early warning for N impacts on Natura 2000 habitats or as part of an integrated assessment of the state of habitats.

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5.6 Impact of nitrogen deposition on species richness of calcareous grasslands in Europe - some preliminary results

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Summary

- This paper seeks to determine whether N-deposition has a negative impact on the species richness of calcareous grasslands at a European wide scale.
- 100 calcareous grasslands across the Atlantic region of Europe were sampled in one season. Species composition and richness of vegetation communities were compared to some key environmental drivers (climate and N deposition) indirectly estimated via surrogates (latitude, longitude, N concentration in bryophyte tissue).
- There are marked differences in species composition across the calcareous grasslands of the Atlantic biogeographic zone within Europe. Contrasts in mean species richness between regions are also detectable at a European wide scale. These natural gradients may mask any footprint of N deposition on vegetation at a European-wide scale.

- For grasslands located along the western range of distribution, there are indications of a decline in species richness as N concentration in moss increases. This suggests that N deposition may be reducing biodiversity in calcareous grasslands at a wide scale, but that this impact can only be detected at the regional, rather than cross-European, level.
- Further research is needed to investigate the impact of N deposition on calcareous grasslands, particularly through the direct assessment of potential drivers as well as the characterisation of variations in species pools at the European scale.

5.6.1 Introduction

The increase of atmospheric deposition of nitrogen (N) in recent decades, due to fertilizer application and fuel consumption, represent nowadays a major threat for biodiversity in ecosystems (Langan 1999, Phoenix *et al*., 2006). In Europe, where this trend has been particularly documented, the effects of N deposition have been studied through empirical or experimental approaches (Bobbink 1998, Stevens *et al*., 2004). Among several effects, N deposition affects nutrient availability, which is a major driver of plant community composition and species richness (Tilman and Pacala, 1993). As a consequence, the impact of N deposition is of major concern for those species-rich ecosystems which are strictly associated with nutrient-poor soils such as Natura 2000 grasslands and heathlands. Recent comparative studies, based on either spatial gradients or on time series analyses, have shown clear evidences of an impact of N deposition on acidic grasslands, leading to a decrease of plant species richness and a loss of species associated with less fertile conditions (Dupré *et al*., 2009, Stevens *et al*., 2004). Such evidence is also found, at local or national level, for other species-rich habitats such as calcareous grasslands or heathlands (Maskell *et al*., 2009), or from experimental results (Bobbink, 1991, Willems and van Nieuwstadt, 1996). Whether such results are generally applicable is of particular importance because calcareous grasslands are of major interest for the conservation of biodiversity in Europe as they support communities of exceptional diversity and many rare and endangered species of plants, insects and birds (WalliesDeVries *et al*., 2002).

The lack of knowledge at a broader scale has motivated the BEGIN project (Biodiversity of European Grasslands – the Impact of Atmospheric Nitrogen Deposition) which seeks to determine whether N-deposition is impacting the species richness of grasslands on a European wide scale . Different approaches have been used to assess the loss of biodiversity associated with N deposition in acidic grasslands: historical analysis (Dupré *et al*., 2009), experimental and comparative surveys (Stevens *et al*., 2004, 2010). Another objective of BEGIN was to investigate whether a similar decrease in biodiversity is occurring in a contrasting grassland system. The most important grassland type across Europe in terms of biodiversity are the calcareous grasslands of the Mesobromion alliance (Koch, 1926). Compared to acid grasslands, these habitats have a much greater species richness and larger number of rare species, and are also presumed to be sensitive to N deposition through increasing nutrient availability. We thus hypothesised that N deposition may be significantly impacting these grasslands.

5.6.2 Aims and objectives

- We aim to determine whether any significant variability in plant species richness in calcareous grasslands across Western Europe could be detected and related to any regional-scale evaluation of N atmospheric deposition.
- In 2008, we surveyed 100 calcareous grasslands belonging to the Mesobromion alliance on a transect across the Atlantic biogeographic zone of Europe. Site selection was performed through a composition criterion (required presence of five species among a predefined list of target species) and a management criterion, in order to avoid abandoned grasslands. For each site, five 1 m × 1 m replicates were recorded. In each square meter, the cover of all occurring plant species (vascular plants and bryophytes) was visually estimated. Each site

is therefore characterised by a list of species with average abundance (calculated from the five quadrats) and an average richness per plot (n=5 replicates).

- In this preliminary study, we only used environmental surrogates to account for the major environmental drivers we identified. A major predictor of large scale species richness is latitude (Hillebrand, 2004). We used latitude and longitude as aggregate variables integrating distinct climate factors i.e. mainly temperature (North-South) and precipitation (West-East) gradients. Total nitrogen concentration in bryophytes was used as a surrogate to estimate total N deposition at a high resolution (Harmens *et al*., 2008). Because species-specific differences are expected, we considered only the sites where the same moss species (i.e. Ctenidium molluscum) was collected and analysed. This accounted for about half of the sampled sites (51 sites for the 100 sites of this study).
- We performed a correspondence analysis (CA) and a hierarchical clustering of the [100 sites x 161 species] data table in order to provide an ordination and classification of plant communities. The table was obtained after removal of species occurring in less than 5 per cent of the records in the initial table [100 sites x 225 spp]. Similar analysis was performed on the sub-set of 51 sites where some environmental surrogates were available. Simple regressions were performed, after data normality was tested (Shapiro-Wilk test), to assess correlations between environmental surrogates and species richness (i.e. mean species number for the five replicates) and species composition (floristic gradients from the CA). All analyses were performed with R free software (2007).

5.6.3 Results and discussion

The Correspondence Analysis shows that gradients in species composition are well correlated to the geographical distribution of the sites (Figure5.8a,b). Latitude and longitude are highly significantly correlated with respectively axis one (n=100, r^2=0.52, p<0.001), and axis two (r^2=0.63, p<0.001) of the CA. A hierarchical clustering (Ward method) performed on the output of this CA (Figure 5.8c) gives a very similar result to clusters defined on a national basis. Three main types of plant communities can be defined (Table 5.3), related to sites from France (FR), United Kingdom and Eire (UK+IRL), and Germany (GER). Sites from north central Europe (Belgium, Netherlands, Denmark) are distributed within these three clusters, while Norway sites form a specific sub-cluster.

Looking for pattern of variation of community species richness along these floristic/geographic gradients of the CA, we found no evidence of a correlation with species richness for any of the CA axes. However, when considering the different clusters from the CA, species richness appear significantly different between some regions (Figure 5.9 - one way ANOVA; F= 3.01, df=99, p<0.05). As we avoided abandoned sites, these differences could not be due to management contrasts but rather to differences in species pool size, depending on regional specificity (soil, climate, history).

Table 5.3: Some differential species in the four clusters identified in the calcareous grasslands of Atlantic Europe

Cluster	Region	Countries	Some differential species
CL1	E	GER, (B), (NL)	*Silene vulgaris, Inula conyza, Poa angustifolia*
(CL1bis)	E	Norway	*Viola canina, Deschampsia flexuosa, Alchemilla filicaulis*
CL2	SW	FR, (B)	*Teucrium montanum, Gaudinia fragilis, Seseli montanum, Thesium humifusum*
CL3	NW	UK, IRL, DK, (B), (NL)	*Carex humilis, Festuca arundinacea, Ranunculus repens*

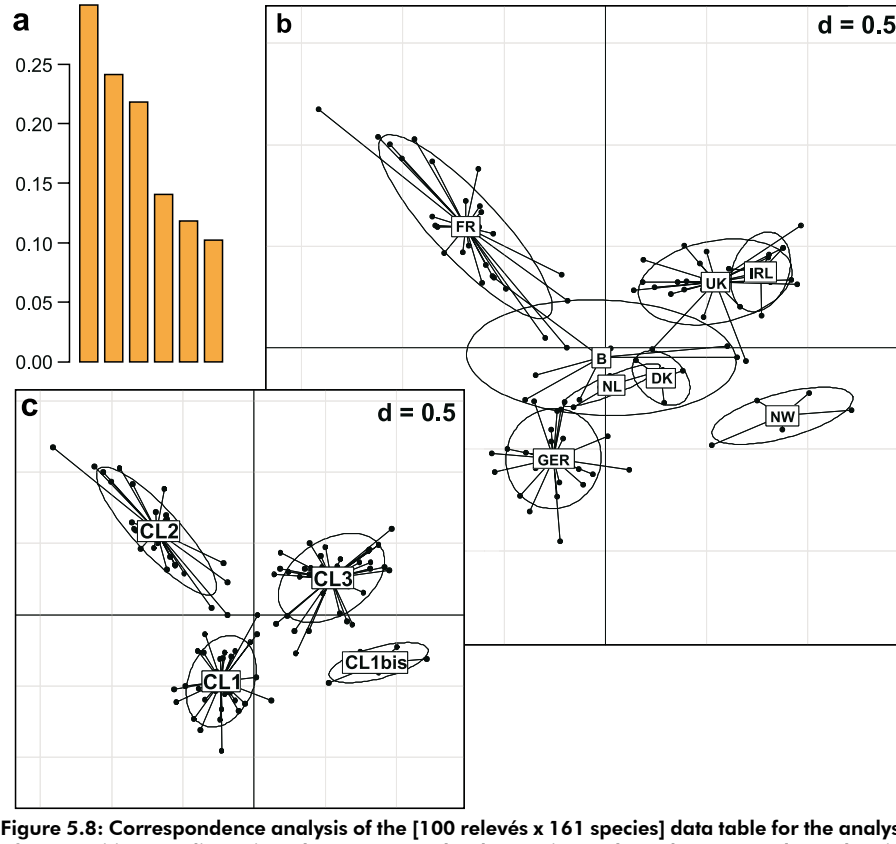

Figure 5.8: Correspondence analysis of the [100 relevés x 161 species] data table for the analysis of composition gradients in calcareous grasslands. a) Eigenvalues; b) F1×F2 plane showing distributions of relevés in the national surveys, c) F1×F2 plane with clusters performed from hierarchical clustering (Ward method) on relevés coordinates.

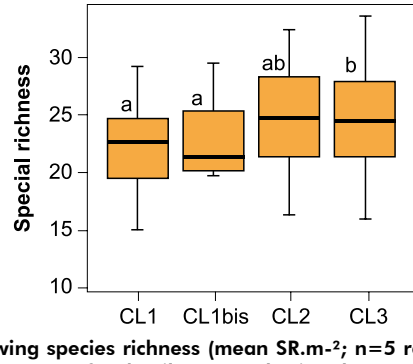

Figure 5.9: Box plots showing species richness (mean SR.m-²; n=5 replicates) variations in the four clusters from the CA. see text for details. Boxes sharing the same letter are not statistically different (P < 0.05, Tukey's HSD comparing all clusters).

We analysed the sub-set of 51 sites to test whether species composition gradients and species richness variations could be correlated to N deposition, estimated via the N surrogate (N per cent in the moss C. molluscum). The 51 sites were distributed in the three main clusters-regions: 27/27 sites of the South-West (SW) of Europe (CL2), 13/36 sites of the North-West (NW) of Europe (CL3) and 11/32 sites of the Est (E) of Europe (CL1). These sites were also regularly distributed along CA axes. We then performed a new Correspondence Analysis (CA2) on these 51 sites, to built floristic gradients on this specific data set. Patterns were similar to the first CA, the CA2 axes being even more correlated to latitude and longitude ($n=51$, $r^2=0.56$, $p<0.001$ for axis 1; $r^2=0.82$, $p<0.001$ for axis 2). N per cent in bryophyte tissue was correlated only with the axis three of this second correspondence analysis ($r^2=0.10$, $p<0.05$), suggesting at least that N deposition could be correlated to gradient of species composition in this data sub-set. We did not find any correlation between N surrogate and species richness of plant communities at a broad scale (Figure 5.10a; $n=51$, $p=0.18$). However, when we performed regressions at the regional level (Figure 5.10 b,c,d), correlation was significant for SW region (Cluster 2, $n=27$, $r^2=0.20$, $p<0.05$), while no trend was detected for other clusters. When significant, correlation shows a decline of species richness at the highest levels of N concentration (deposition). The processes responsible for this decline may be found in the effects of N enrichment, resulting in changes in vegetation structure and species interactions to the benefit of competitive tall grasses (Bobbink, 1991, Liancourt *et al*., 2005).

From our data, composition gradients in calcareous grasslands are marked at the European scale, even though we removed the less frequent species in the data set (i.e. with occurrence less than 5 per cent) which should attenuate contrasts between countries. This species turn-over is shown in phytosociological works (Royer 1985, Willems, 1982). Our survey confirms that climate gradients are likely to be the most important drivers of species turn-over in calcareous grasslands in Europe, as climate variables such as temperature and rainfall are known to be correlated with latitute and longitude (Ozenda 1994, Duckwoth *et al*., 2000). Similarly, our data suggest that regions in Europe could be characterised by species pools of different sizes. However, this has to be confirmed with species pool studies (e.g. Dupré, 2000), based on more complete phytosociological datasets.

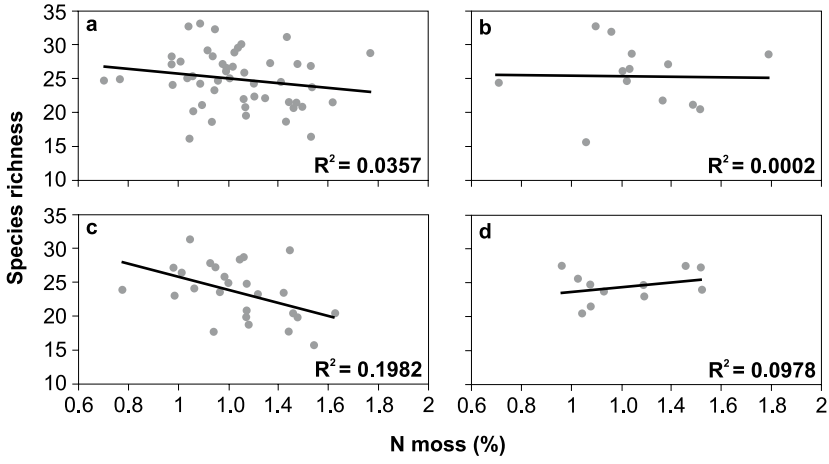

Figure 5.10: Regression plots between nitrogen concentration in the moss Ctenidium molluscum (N per cent dry weight) and species richness (mean SR.m-²; n=5 replicates) in calcareous grasslands according to different geographic ranges: a) subset covering the whole geographic range i.e. Atlantic Europe; n=51; b) subset from NW Atlantic Europe i.e. Cluster 3; c) subset from SW Atlantic Europe i.e. Cluster 2; subset from Eastern Atlantic Europe i.e. Cluster 1. See text for details.

Because of the strong climate-driven variation in species composition and richness, it is difficult to detect a separate signal of N deposition as a potential driver of calcareous grassland diversity on a cross-Europe scale. The use of N concentration in moss as a surrogate for N deposition can also introduce some potential artefacts. Besides differing among different species, this relationship might also depend on other factors such as N speciation, the ratio of wet/dry deposition in N deposition, and local climate (Harmens *et al*., 2008). Despite these limitations, there are some indications of an N-deposition signal on species composition at a European wide scale and on species richness for calcareous grasslands located at the western range of their distribution.

5.6.4 Conclusions

- There are strong gradients of species composition in calcareous grasslands in western Europe. Contrasts in mean species richness between regions are also detectable at a European-wide scale. These gradients and contrasts appear to be driven primarily by climate.
- Because of these strong environmental responses, it is difficult to detect a clear influence of N deposition on species richness and composition at the European scale.
- However, when filtering the data at the regional to national scale, there are indications of an N-deposition signal on species richness for grasslands sampled in western regions (Atlantic coast). If real, regressions suggest a fairly strong decline in diversity with increasing N deposition for these sites
- These intimations of a N impact on calcareous grassland diversity strongly point to a need for targeted research, particularly through the direct assessment of potential drivers as well as the characterisation of natural variations in species pools at the European scale.
- The above have strong implications for conservation and pollution mitigation actions for management of calcareous grasslands (Calciura and Spinelli, 2008).

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5.7 Nitrogen critical load and butterflies

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Summary

The biodiversity effect of nitrogen deposition is now well recorded and explored for vegetation but the effect on invertebrates representing most of biodiversity (by number of species) is unknown.

Biodiversity is defined and used as the product of a number of qualitative indices of a group of organisms (in this case butterflies).

Meta data analysis of seventeen years of survey data of butterflies in a wide range of habitats in the Netherlands is analysed along with nitrogen critical load exceedence (CLE) data calculated for the same survey sites.

The Netherlands is in recovery from high nitrogen CLE and yet nitrophilic species are still replacing nitrophobic species so the situation is complex due to the data still showing exceedence.

Species Richness is not indicative of nitrogen CLE. Other biodiversity indices have a better indication relationship with CLE.

The importance of nitrogen CLE in the set of biodiversity indicators accepted by the European Environment Agency in the SEBI 2010 assessment shows it to be a critical factor and related to butterfly populations.

5.7.1 Introduction

Almost all of the work on the ecological effect of nitrogen deposition has been related to plant biochemistry and distribution (in particular grasses, forbs, bryophytes and lichens). These habitat structural components do not in terms of species number represent more than a small part of biodiversity. Since the plant structural component effect of nitrogen deposition is now clearly established it becomes necessary to determine whether other components of biodiversity are also affected and whether they are more sensitive or less sensitive than the present set of bioindicators.

In terms of species number, ubiquity and sometimes biomass, invertebrates are the largest component of biodiversity. There are three mechanisms by which they might be affected by nitrogen deposition:

- They may be directly affected by the chemistry of the nitrogen (in particular that part due to NH+);
- They may be affected by a change in the food-plant chemistry. We know that, for example, some butterflies are very sensitive to the nutrient status of the plants upon which they lay their eggs;
- They may respond to a change in the habitat due to plant responses to nitrogen deposition.

The latter of these hypotheses is the basis of the paper by Wallis de Vries and van Swaay (2006) where they hypothesize that increased nitrogen-induced grass growth in the spring causes a slower increase in ground temperature and thus eggs and larvae are delayed in emergence from overwintering diapause.

Butterflies are good exemplar invertebrates since a) they have been the subject of extensive standardized recording in a number of countries b) they are popular c) they are obvious when flying and therefore relatively easy to record and d) they are found in some habitats that are clearly affected by nitrogen deposition. Difficulties result from their non-ubiquity and presence only in low numbers of individuals or species in some habitats.

5.7.2 Aims and objectives

The aim of this research was to triangulate butterfly biodiversity against nitrogen deposition to determine if indeed butterfly biodiversity was affected by nitrogen deposition. This (and the methodologies utilised) could then be a starting point for estimation of the effect of nitrogen deposition on a wider element of biodiversity than is currently established.

In order to understand more clearly the possible effect of nitrogen deposition on butterfly populations a refinement of the definition of biodiversity is used (Feest, 2006; Feest *et al*., 2010) to cover its functional qualities (Hooper *et al*., 2005) supplemented by an Ellenberg scale of nitrogen sensitivity (Oostermeijer & van Swaay, 1998), known as theSpecies Nitrogen Value Index (SNVI). The butterfly population biodiversity quality elements were calculated using the Fungib programme (ecosulis ltd.: https://www.ecosulis.co.uk) as follows:

1 Species Nitrogen Value Index (SNVI)
2 Species Conservation Value index (SCVI)
3 Species Richness
4 Simpson Biodiversity Index
5 Population Index
6 Biomass Index

For derivation of indices 2-6 see Feest (2006) and Feest *et al*., (2010).

These data were then statistically analysed for relationships with nitrogen CLE. The butterfly data were supplied by Chris van Swaay of de Vlinderstichting and the nitrogen CLE for the identified butterfly sites was supplied by Arjen van Hinsberg of MPN. Since the data set covered a spread of 17 years and butterfly data are subject to considerable annual variations the data were smoothed by aggregation into three groups representing six, six and five years.

The advantages of this approach are:

- Data produced can be easily tested statistically for significance
- Other groups e.g. bryophytes, beetles, spiders, macrofungi etc. could be assessed in the same way
- Baseline biodiversity quality indices/values can be set and revisited at any time in the future and tested for statistical significance of differences
- The taxonomy of butterflies is not difficult

5.7.3 Results and discussion

Table 5.4 gives the data for two habitat types in the Netherlands. The data show clear apparent trends for all indices except Species Richness.

To add statistical strength to this picture the data were further analysed by principle component analysis by habitat and year aggregation. Table 5.5 gives the result of the first two component axes for an example of Woodlands in the Netherlands for the years 2001-6. The picture revealed in this assessment was one typically found for other habitats and year aggregations.

Table 5.4: Typical results for the biodiversity of Dutch butterflies for consecutive six, six and five years. SNVI= Species Nitrogen Value Index; SR= Species Richness; SCVI= Species Conservation Value Index; nCLE= nitrogen critical load Exceedence

Habitat	SNVI	SR	SCVI	Population	Biomass	nCLE	n=
Grassland period 1	5.8	15.86	2.83	720	15698	1765	23
Grassland period 2	5.74	14.9	2.78	638	12778	641	57
Grassland period 3	6.22	16.48	2.72	527	11352	572	54
Heathland period 1	4.01	18.22	3.23	569	10407	1626	12
Heathland period 2	4.22	15.9	3.2	437	7423	1071	20

Table 5.5: Principlal Component Analysis of Woodland Butterflies 2001-2006. PC1= Principal Component 1; PC2= Principal Component 2. Other abbreviations as for Table 5.4.

	PC1(0.532)	PC2(0.247) (∑=0.779)
SNVI	0.281	0.415
Sp Rich	-0.490	0.040
Simpson	-0.406	-0.109
SCVI	-0.433	-0.298
Pop	-0.341	0.549
Biomass	-0.324	0.571
nCLE	-0.327	-0.313

In component axis one representing 53.2 per cent of the variation all indices have a similar strength and relationship with the exception of SNVI. Looking at the coarse data these will be seen to indicate that all of the indices are in decline except the one indicating the nitrophilic/nitrophobic balance of the butterfly population which is moving to a more nitrophilic status. Whilst this shows a clear relationship between CLE and other factors this is not very helpful since it fails to differentiate the indices. In component axis two representing 24.7 per cent of the variation a radically different picture emerges that allows some understanding of the mechanisms operating. Firstly, Species Richness is of low significance (nitrophobic species being replaced by nitrophilic); SNVI, population/biomass are associated (the more nitrophilic the species the larger the populations/ biomass) and Simpson, SCVI and CLE are associated, but less strongly (the rarity, CLE and evenness of populations are in decline). The analysis also reveals the problematic situation in the Netherlands where over the period of the surveys the nitrogen CLE has been in a steady decline throughout although still registering positive results of exceedance. Figure 5.11 shows the area of uncertainty that has prevailed during the survey period in that there is a recovery delay, as Nitrogen CLE declines, of unknown duration (whereas deposition has a much more immediate effect). It is hypothesized that this relates to slow vegetation change and differential colonization effects (Pöyry *et al*., 2009).

5.7.4 Conclusions

This paper shows that there is a clear relationship between nitrogen deposition and butterfly biodiversity quality and that this may be complex during a period of recovery of populations in response to reduced nitrogen deposition. This is important for future biodiversity protection since this work has not determined whether this effect extends to other invertebrate groups (or indeed other taxa such as macrofungi) and how sensitive butterflies are -are they more or less sensitive than plants?

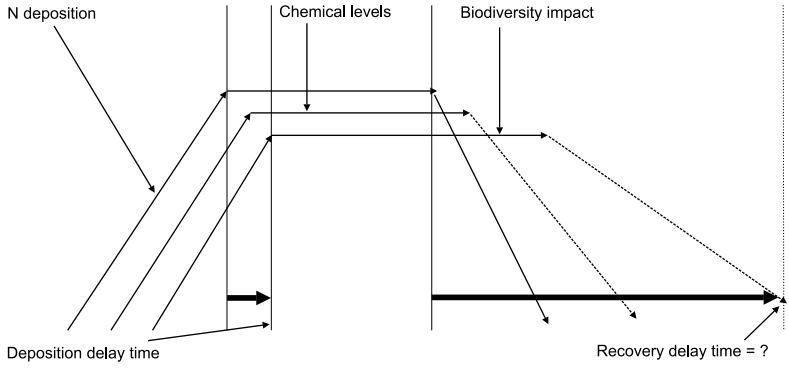

Figure 5.11: The disjunction between the impact nitrogen deposition, its effects and the recovery.

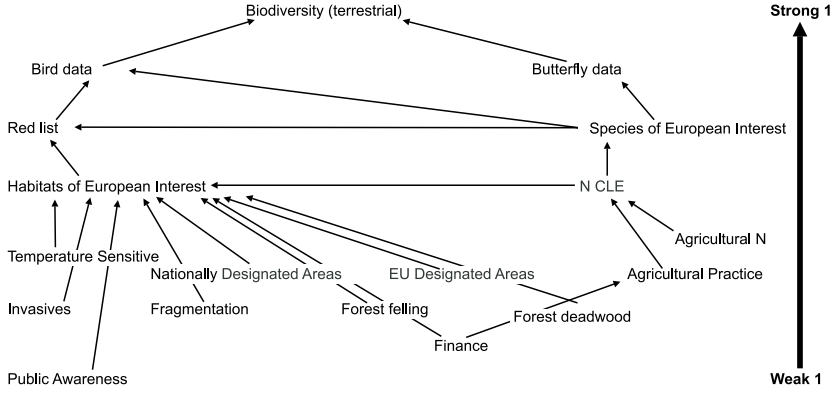

Figure 5.12: Linkages between the terrestrial SEBI 2010 indicators and their relationship with biodiversity

The overall success of biodiversity protection in the European Union is being registered by the Streamlining European Biodiversity Indicators 2010 (SEBI, 2010) process. SEBI 2010 has determined a set of 26 indicators and a putative relationship of the terrestrial indicators is presented in Figure 5.12. Note that only two groups of organisms representing organism group biodiversity are included in the scheme (birds and butterflies). It is also clear that two elements are of importance to the whole indicator set and they are: 1. Habitats of European Interest and 2. Nitrogen CLE. This work is the first that has validated the relationship of two of the SEBI 2010 indicators and shows the importance of nitrogen CLE for more than vegetation characteristics.

Acknowledgements:

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5.8 Selecting critical areas for monitoring the impact of ammonia on biodiversity

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Summary

- The main impact of atmospheric ammonia (NH_3) is known to occur at short distances (less than 500 m). Therefore, for successful development and implementation of policies aiming at preventing the impacts of NH_3 on biodiversity of Natura 2000 sites, it is necessary to make use of a high spatial resolution mapping. However the available information of NH_3 emission and deposition is clearly of an insufficient resolution for this purpose, at least in Portugal.
- Our objective was to provide a practical method for selecting critical areas for monitoring the impact of NH_3 in plant and animal biodiversity within Natura 2000 sites. Lichen functional-diversity is a good indicator of NH_3 impact at small spatial scales, when a single source of disturbance is present. However, because Natura 2000 sites are large areas and contain multiple sources of disturbance, lichen-functional diversity may not be the most appropriate indicator since it responds to other factors besides NH_3. Therefore, we propose to use total nitrogen concentration in lichens, [N], as a method for selecting critical areas of NH_3 impact in Natura 2000 sites.
- A first question that was addressed was: what are the [N] concentration in lichens reflecting? For that we tested if: i.) [N] was reflecting atmospheric NH_3 deposition (by relating it with agriculture land-use); ii.) was [N] reflecting NO_x (by relating it to industrial and urban land-uses).
- The [N] in lichens was shown to be very significantly related to agriculture areas and not to urban or industrial areas thus showing that N concentration in lichens is most probably reflecting the NH_3 emissions. In this way we propose here to apply the N concentration in

lichens as a detailed ecological indicator for selecting critical areas for the impact of NH_3 on biodiversity.

- Furthermore, we applied this indicator to two Natura 2000 sites by mapping [N] in lichens. By doing so we could select the critical areas for the assessment of the impact of atmospheric NH_3 deposition on plant diversity in Mediterranean Natura 2000 sites. Further studies on the impact of plant diversity can now be focus in high NH_3 deposition areas.

5.8.1 Introduction

In Mediterranean landscapes a large variety of land-cover types occur in small areas (Blondel and Aronson, 1999). Besides, areas dedicated to Nature Protection are surrounded by centuries old human-matrix. Among important sources of disturbance in Natura 2000 sites we can find agriculture. Those activities are major sources of atmospheric NH_3 in Europe (EPER, 2004; Galloway *et al*., 2003). Moreover, the deposition of N is related to biodiversity loss (Phoenix *et al*., 2006; Suding *et al*., 2005) and is considered not only a major threat to global biodiversity but also one of those threats that are expected to increase worldwide (SCBD, 2006). Thus, biodiversity within protected areas might be highly threatened by atmospheric NH_3. However in Portugal the spatial resolution of the available emission and deposition mapping is clearly insufficient to allow an adequate selection of the areas under the greater risk of NH_3 impact (Martins-Loução, this volume). Therefore the main goal of this work was to provide a practical method for selecting critical areas for monitoring the impact of NH_3 on biodiversity within Natura 2000 sites. To do so we used lichens, poikilohydric organisms resulting from the symbiosis of a fungus and a photosynthetic partner. Lichens are considered one of the most sensitive communities of organisms in the ecosystems, due to its particular physiological characteristics such as the absence of a protective cuticle. Lichens are the most sensitive group of organisms to N and its functional biodiversity changes with NH_3 atmospheric deposition (Pinho *et al*., 2009). However the application of lichen diversity in this case is problematic, since it responds to a series of other factors. In fact, lichens have been shown to be sensitive to a large number of factors including pollutants, with both human and natural origin and microclimate changes (Geiser and Neitlich, 2007; Giordani *et al*., 2002; Pinho *et al*., 2004; Pinho *et al*., 2008a) and therefore have been used as biomonitors of complex environmental changes such as habitat fragmentation, habitat stability and influence of forest management (Coxson and Stevenson, 2007; Edman *et al*., 2008; Nascimbene *et al*., 2007; Ranius *et al*., 2008). More specifically, we propose to use lichens as nitrogen biomonitors (Gaio-Oliveira *et al*., 2001). This is so because total N in lichens has been shown to increase in NH_3 rich areas (Pinho *et al*., 2008b) and increased N concentration in lichens and plant tissues has been associated to higher N availability (Adrizal *et al*., 2008; Boggs *et al*., 2005; Fluckiger and Braun, 1998; Gaio-Oliveira *et al*., 2001; Pocewicz *et al*., 2007).

5.8.2 Aims and objectives

- The aim of this work was to provide a method with high spatial resolution for selecting critical areas for monitoring the impact of NH_3 in plant and animal biodiversity within Natura 2000 sites.
- For that we proposed to use total N concentration in lichens. We firstly tested if [N] was related to the area occupied by agriculture (a source of atmospheric NH_3) and artificial zones (a source of NO_x). Secondly we mapped [N] within the Natura 2000 sites.

5.8.3 Results and discussion

We found that [N] in lichens can be used as an ecological indicator to reflect NH_3 deposition in Natura 2000 sites with a high spatial resolution. The hypothesis was that this indicator would be useful for mapping critical areas of potential impact of NH_3 pollution, and to test this we first determined if [N] in lichens was related to human activities in neighboring areas, considering agriculture (emitting NH_3), industry and traffic (emitting NO_2).

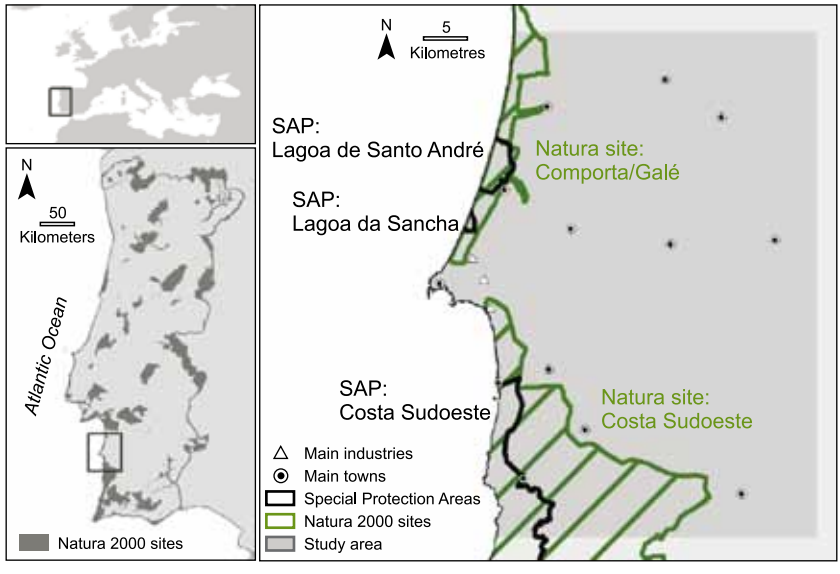

Figure 5.13: Location of Natura 2000 sites in Portugal and of the studied region in SW Portugal. The labeled Natura 2000 sites are the ones included in this work. Main industrial areas are part of Sines industrial complex and include a power-plant, an oil refinery and other petrochemical industries.

In small areas, with a single source of disturbance, lichen biodiversity has been shown to respond accurately to NH_3, leading to a change in lichen functional-diversity (Pinho *et al*., 2009). However, when dealing with Natura 2000 sites, normally occupying a regional area, we must consider the possible existence of multiple disturbance sources, many of which may be diffuse. In such areas, and more specifically in the studied area (Figure 5.13), nitrophytic and oligotrophic lichen species have been shown to respond to a large number of factors, including natural (such as the sea) and anthropogenic ones (such as industrial areas) (Pinho *et al*., 2008a; Pinho *et al*., 2008b). These may impede the use of biodiversity as an indicator of NH_3 pollution. In order to avoid the interference by other factors we used [N] in lichens to determine the critical areas of potential impact of nitrogen pollution, the usefulness of the use of nitrogen concentration as a biomonitoring tool having already been suggested by a preliminary work (Gaio-Oliveira *et al*., 2005). In this work, [N] was determined in the lichen species *Parmotrema hypoleucinum* (J.Steiner) Hale, collected from 104 cork-oak woodland sites. First we determined if [N] was related to neighboring sources of atmospheric nitrogen, by performing a local correlation analysis (Figure 5.14). We considered neighborhood areas around sampling sites, with radius ranging from 50 to 6400 m (Pinho *et al*., 2008a). A local analysis was made using these areas, by relating [N] and area of agriculture and artificial land-cover, considering sites located at less than 10 km distance. This local correlation was preferred to a regional analysis because NH_3 pollution is known to be of short-range (Pinho *et al*., 2009; Sutton *et al*., 1998). The local correlation analysis was plotted for the study area (Figure 5.14), showing many very significant correlations between [N] in lichens and the area occupied by annual agriculture (mainly rice fields and cereals), as well as heterogeneous and other permanent agriculture (mainly small farms and orchards) (Figure 5.14). The results also showed that there was no significant correlation between [N] in lichens and artificial areas (mainly roads, urban and industrial areas) (data not shown), excluding NO_x emissions as contributing to the [N] in lichens. Moreover the distance of influence, that for which the maximum correlation was observed (Pinho *et al*., 2008b), was found to be on average 1600 m for annual cultures and 1200 m for

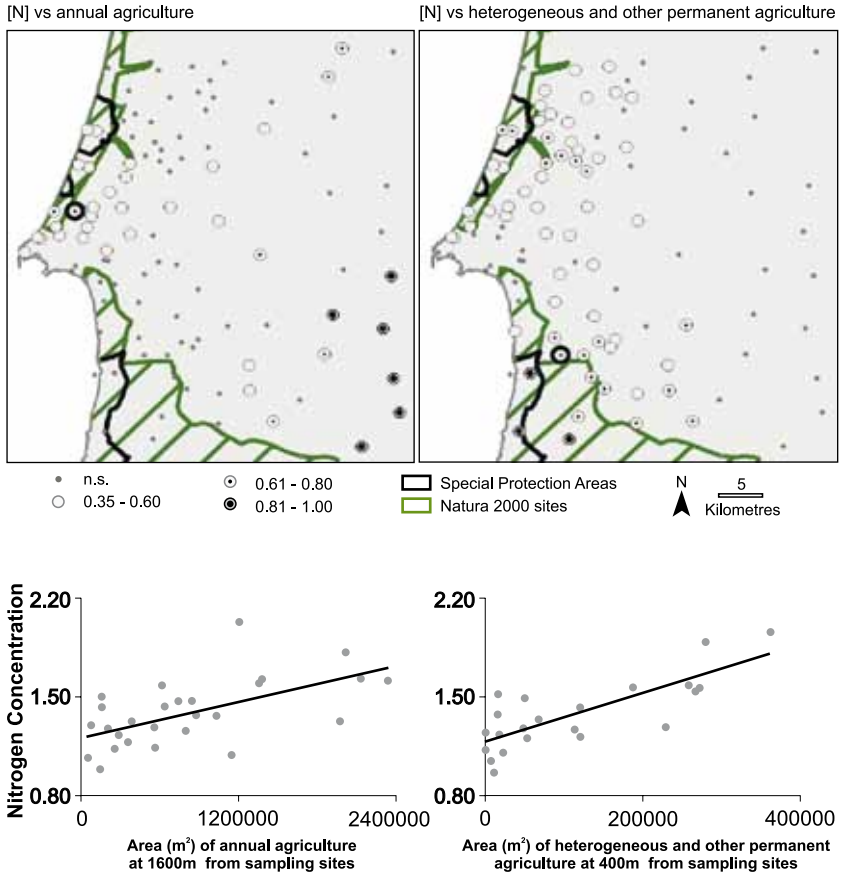

Figure 5.14: Local correlation analysis between [N] in lichens and: i.) neighboring annual agriculture (left) and ii.) neighboring heterogeneous and other permanent agriculture (right). ***This is the result of a moving window analysis that correlates two variables using as samples all sites at a distance of 10km from each sampling site. A significant correlation indicates that, within a 10km radius neighborhood, the two variables are significant correlated. In the maps not-significant correlations (n.s.) are marked with small dots, significant ones with larger circles. The magnitude of the correlation (R) is given on the legend, and varied between 0.35 and c. 1.00. The two biplots are example of correlation for two sites, marked with a darker symbol in the maps above. More details on this type of analysis can be found in (Pinho et al., 2008a).***

heterogeneous and other permanent agriculture (average values of all sites). This distance is in agreement with other studies that have shown that most nitrogen deposition and its effects on biodiversity occur less than 1 km from sources (Pinho *et al*., 2009; Sutton *et al*., 1998). Taken together, these results have shown that [N] in lichens can be used to map areas under the impact of agriculture NH_3, even if other sources of disturbance are present.

Once we had determined that agricultural areas are the most likely cause for increased nitrogen concentration in lichens, we mapped [N] in the study region. This variable was analyzed by geostatistical techniques (CERENA, 2000), namely variogram interpretation used to interpolate [N] for the region using ordinary kriging (ESRI, 2008). By focusing on the Natura 2000 sites (Figure 5.14) we could observe that it presents a patchy distribution, highlighting the short-range nature of nitrogen pollution. Moreover this mapping also provides an efficient way to map the critical areas probably affected by nitrogen pollution, and should be considered critical areas

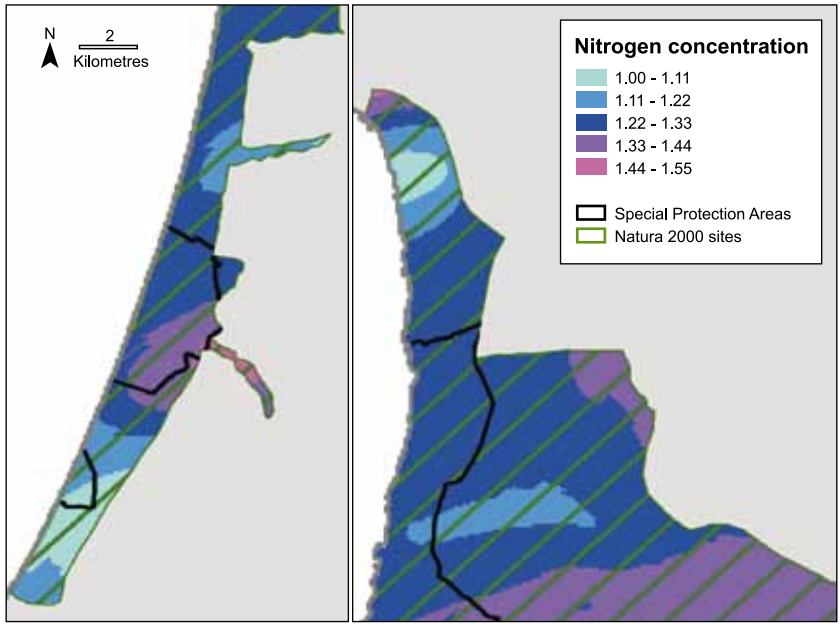

Figure 5.15: Mapping of the interpolation of Nitrogen concentration (per cent dry weight) measured in *Parmotrema hypoleucinum* within the two Natura 2000 sites located in the study area. The areas with the higher concentration are the ones under greater threat from N pollution, and are the ones that should be monitored for biodiversity loss.

for monitoring biodiversity. In this way we reduce cost focusing the plant and animal diversity monitoring in high impact NH_3 deposition areas.

In the Natura 2000 site "Costa Sudoeste" (Figure 5.13, right) most nitrogen is probably being emitted by vegetables, as well as by grain cultivation sites located in the nitrogen hot-spot areas (Figure 5.14, right). Of special concern is the Natura 2000 site "Comporta/Galé" (Figure 5.15, left), characterized by costal dunes habitats and costal lagoons. Up-stream of those lagoons there are rice cultures, known to be important sources of NH_3 (Yan *et al*., 2003). Rice fields are the probable source of N leading to accumulation in lichens (Figure 5.14). Within this Natura 2000 area there are two Special Protection Areas (Birds Directive), one of which, "Lagoa de Santo André", is located in the area with higher Nitrogen concentration (Figure 5.15). This area is likely to be under strong nitrogen-pollution and should be particularly monitored for its impacts.

Aiming in the future at establishing precise boundaries for the critical areas outlined in Figure 5.15, we aim at calibrating [N] in lichens with a legal-bounded variable, namely i.) loss of biodiversity (e.g. loss of endangered plant species) or ii.) exceedance of critical levels of measured NH_3. Within the legal limits the criteria for protection level given by the chosen boundaries is a matter of decision makers choice.

5.8.4 Conclusions

- Although lichen-diversity is a good ecological indicator of NH_3 impact on ecosystems, in areas with multiple sources of disturbance, it responds also to factors other than NH_3, such as industrial and urban pollution. Therefore in such areas lichen diversity is not a good indicator of atmospheric NH_3.

- Nitrogen concentration in lichens was shown to be highly significant related to agricultural areas, and not related to artificial ones, being therefore a good measure of NH_3 atmospheric deposition, even in areas with multiple disturbance sources.
- By mapping nitrogen concentration in lichens, we could provide criteria for selecting critical areas with potential risk for biodiversity from NH_3 pollution within Natura 2000 sites.

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5.9 Effects of increased N availability on biodiversity of Mediterranean-type ecosystems: a case study in a Natura 2000 site in Portugal.

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Summary

- Although Mediterranean-type ecosystems are biodiversity hotspots, very little is known about the effects of increased N availability in these systems;
- This paper describes an integrated field study on the effects of increased N availability in a Mediterranean-type ecosystem in a Natura 2000 site in Portugal;
- The ecosystem was highly N responsive: visible changes were seen within one year; N additions created new and distinct seasonal patterns of soil N availability; plant and soil bacterial diversity together with plant cover were increased;
- The effects of increased N availability appeared to depend on N form - plant evenness; N dose - plant species richness; and on both N form and dose - species cover and soil bacterial richness.

5.9.1 Introduction

Global biodiversity is changing at an unprecedented rate (Pimm *et al*., 1995) as a complex response to anthropogenic-derived changes at the global scale (Sala *et al*., 2000). The magnitude of this biodiversity change is so large that it constitutes a threat to the sustainability of human societies and natural systems (Galloway *et al*., 2008). But, what is biodiversity and how can we study it in order to preserve it? Biodiversity is a complex term that includes taxonomic, functional, spatial and temporal aspects of organism diversity, with species richness (the number of species) and evenness (their relative abundance) considered among the most important measures (Wilsey and Potvin, 2000). Worldwide, many have focused on biodiversity loss. Sala *et al*., (2000) developed biodiversity change scenarios in terrestrial ecosystems, ranking increased N deposition as its third (out of five) main driver. Subsequent works, inferred that N deposition constitutes a threat to biodiversity (Phoenix *et al*., 2006, Clarisse *et al*., 2009). Mediterranean-type ecosystems are biodiversity hotspots (Phoenix *et al*., 2006), and could be experiencing the greatest proportional change in biodiversity (Sala *et al*., 2000). However, very little is known about the effects of increased N availability in these systems (Phoenix *et al*., 2006). Apart from being nutrient-poor (Cruz *et al*., 2003, Cruz *et al*., 2008) Mediterranean-type ecosystems also have distinct seasonal resource availability (water, nutrients and temperature). Enhanced N availability is likely to create new patterns of N availability that will allow new species to appear and others to disappear. It is also possible that N form, especially ammonium, could influence the system's response. The seasonal differences in N availability mean that it will be difficult to extrapolate from northern European to Mediterranean ecosystems.

5.9.2 Aims and objectives

- To study the effects of short-term increased N availability in a Mediterranean-type ecosystem; and
- To understand the effect of N doses and forms in the biodiversity of above- and below-ground communities.

5.9.3 Methods

The study site (38º29' N - 9º 01' W) is in Serra da Arrábida in the Arrábida Natural Park, south of Lisbon, Portugal (a Natura 2000 site - PTCON0010 Arrábida/Espichel). N availability (dose and forms) at the site has been modified by the addition of 40 and 80 kg N $ha^{-1}yr^{-1}$ as NH_4NO_3 or 40 kg N-$NH_4^+ha^{-1}yr^{-1}$ (control plots are not fertilized) since January 2007. N is added in three equal applications throughout the year, correlating with distinct biological activities (spring, summer and middle autumn/winter). Each treatment has three replicates (400 m^2 experimental plots).

5.9.4 Results and discussion

Aboveground community change

The standing plant community (Eunis class F5.2 – Mediterranean maquis) is in an early stage of a post-fire succession. The fire occurred in 2003, four years before the beginning of the N additions. Assessment of the plant communities in two consecutive springs (the first and second springs after beginning N fertilization) identified 80 plant species belonging to 27 families (Table 5.6). At the beginning of the N manipulations (first spring) the plots were homogeneous for species richness, evenness and plant cover. However, after one year of N additions treatment differences were found (Table5.7):

1) Non-fertilized plots exhibited a decrease in species richness (Table 5.7) that is characteristic of communities in similar stages of succession in Mediterranean-type ecosystems (Thompson, 2005) whereas adding N appeared to have prevented the natural decrease in species richness. This N-induced effect was dose-dependent and form-independent.

2) Plant evenness decreased in all treatments (Table 5.7). If this trend is sustained, plant communities would become more uneven, a characteristic of natural ecosystems (Naeem, 2009). Ammonium fertilization caused the greatest decline in plant evenness, possibly due to low ammonium tolerance of some plant species. We would expect these plots to become dominated by plant species that are more tolerant to ammonium. Changes in plant evenness were N-form dependent and dose-independent.

In ecology, it is widely accepted that 'nutrient limitation' occurs when there are differences between fertilized and unfertilized samples (Vitousek and Howarth, 1991). Similarly, ammonium toxicity would occur when there are differences in response to the same N dose but supplied in different N forms. Since only fertilization with 40 Kg N-$NH_4NO_3ha^{-1}yr^{-1}$ led to an increase in plant cover relatively to the control (Table 5.7), it may again be related with plant ammonium toxicity (Güsewell, 2004). Throughout succession the predominant form of available N changes (Cruz *et al*., 2003). As a consequence, early successional species prefer nitrate while late successional are ammonium tolerant (Cruz *et al*., 2003, Kronzucker *et al*., 2003). The standing plant community is in an early phase of succession and therefore dominated by species less ammonium tolerant making it therefore more ammonium sensitive. If the site had been in a latter stage of succession the response to ammonium may well be different, with the effect being less detrimental (Cruz *et al*., 2003).

First-year effects of fertilization are often determined by the original dominant species' responses, but in following years, subordinate or even new species may reach dominance (Stöcklin *et al*., 1998). Dittrichia viscosa (L.) W. Greuter was the only plant species that significantly changed (increased) its cover in response to N additions (Table 5.6). However, there is growing literature suggesting that focusing on functional traits rather than species (McGill *et al*., 2006) is more relevant for ecosystem functioning (Naeem, 2009) and a more practical approach for biodiversity hotspots like Mediterranean-type ecosystems. Therefore, plant species were grouped (Table 5.6) according to their functionality (Barradas *et al*., 1999) or their common habitat. Increased N availability appears

Table 5.6: Grouping of plants according to their functionality or common habitat

Plant Group	Species	Family	Ambient N depostion	.+40kg $N\text{-}NH_4^+ha^{-1}yr^{-1}$	.+40kg $N\text{-}NH_4NO_3$ $ha^{-1}yr^{-1}$	.+80kg $N\text{-}NH_4NO_3$ $ha^{-1}yr^{-1}$
Summer Semi-deciduous	*Helichrysum stoechas*	*Asteraceae*	D			
	cf Halimium halimifolium	*Cistaceae*				
	Cistus crispus	*Cistaceae*			D	
	Cistus ladanifer	*Cistaceae*				
	Cistus monspeliensis	*Cistaceae*				A
	Cistus salvifolius	*Cistaceae*				
	Lavandula stoechas ssp lusieri	*Lamiaceae*				0
	Rosmarinus officinalis	*Lamiaceae*				
Ericaceous	*Calluna vulgaris*	*Ericaceae*				
	Erica arborea	*Ericaceae*				
	Erica scoparia	*Ericaceae*				
	Erica umbellata	*Ericaceae*				
Legume Shrubs	*Genista triacanthos*	*Fabaceae*				
	Ulex densus	*Fabaceae*				
Grasses	*Carex flacca*	*Cyperaceae*				
	Agrositis sp	*Poaceae*	A			
	Avenula sp	*Poaceae*	A			
	Brachypodium phoenicoides	*Poaceae*		0		
	Briza maxima	*Poaceae*			A	A
	Briza minima	*Poaceae*		D		D
	Briza minor	*Poaceae*			0	A
	cf Arrhenatherum album	*Poaceae*				D
	cf Dactylis glomerata ssp lusitania	*Poaceae*				A
	Poaceae	*Poaceae*	0			0
	Vulpia sp	*Poaceae*	D	D		D
Herbaceous maquia sp	*Apiaceae*	*Apiaceae*	A			A
	Asphodelus ramosus	*Asphodelaceae*	D			A
	Asteraceae	*Asteraceae*	D	D	A	
	Cynara sp	*Asteraceae*	D			
	Leontondon taraxacoides	*Asteraceae*				0
	Phagnalon saxatile	*Asteraceae*				D
	Pulicaria odora	*Asteraceae*			D	A
	Lithodora prostrata	*Boraginaceae*	0			A
	cf Anthyllis vulneraria	*Fabaceae*				A
	Fabaceae	*Fabaceae*				D
	Blackstonia perfoliata	*Gentianaceae*			D	
	Centaurium erythraea	*Gentianaceae*			D	
	Gladiolus illyricus ssp reuteri	*Iridaceae*		A	A	A
	Iris xiphium	*Iridaceae*				A
	Stachys arvensis	*Lamiaceae*	A	A	A	
	Liliaceae	*Liliaceae*	A		A	
	Urginea maritima	*Liliatae*		A		D
	cf Orobanche latisquama	*Orobancheae*	A			
	Anemone palmata	*Ranunculaceae*	A			
	Sanguisorba hybridia	*Rosaceae*			D	D
	Galium sp	*Rubeaceae*			A	A
	Rubia peregrina	*Rubeaceae*				A
	Rubeaceae	*Rubeaceae*				A
Ruderals	*Daucus carota*	*Apiaceae*			D	
	Carlina corymbosa	*Asteraceae*			A	
	cf Andryala ragusina	*Asteraceae*				
	cf Centaurea melitensis	*Asteraceae*	A			
	cf Evax pygmaea ssp ramosissima	*Asteraceae*				A
	cf Filago minima	*Asteraceae*		D	A	D
	Chrysanthemum coronarium	*Asteraceae*		A		
	Crepis capillaris	*Asteraceae*		A		
	Dittrichia viscosa	*Asteraceae*	a	b	b	b
	Galactites tomentosa	*Asteraceae*		D	D	D
	Matricaria recutita	*Asteraceae*	D	D	D	
	Picris echoides	*Asteraceae*		D		
	Senecio jacobaea	*Asteraceae*	A		D	
	Sonchus sp	*Asteraceae*	D		D	A
	Campanula rapunculus	*Campanulaceae*			D	
	Lotus sp.	*Fabaceae*	A	A		
	Trifolium sp.	*Fabaceae*	D		D	D
	Vicia sp	*Fabaceae*		A		
	Hypericum sp.	*Hypericaceae*				

Table 5.7: Estimation of bacterial diversity using the number and intensity of temperature gradient gel electrophoresis separated bands

Properties	Ambient N depostion	+40kg N-NH_4^+ $ha^{-1}yr^{-1}$	+40kg N-NH_4NO_3 $ha^{-1}yr^{-1}$	+80kg N-NH_4NO_3 $ha^{-1}yr^{-1}$
Vascular plants community				
Species richness in 2007 (S_{2007})	18.67 ± 1.2	18.00 ± 1.53	20.00 ± 1.53	20.33 ± 1.45
Species richness in 2008 (S_{2008})	16.33 ± 1.76	18.33 ± 1.3	20.33 ± 2.4	24.00 ± 2
Weighted $\Delta_{(S2008-S2007)}$ (per cent)	14 ± 6	2± 4	1 ± 9	16 ± 3
Weighted Gain (per cent)	. ± 5a	16 ± 4ab	15 ± 6ab	31 ± 4b
Evenness in 2007 (E_{H2007})	0.78 ± 0.01	0.84 ± 0.03	0.80 ± 0.01	0.81 ± 0.01
Evenness in 2008 (E_{H2008})	0.67 ± 0.03	0.61 ± 0.08	0.77 ± 0.03	0.77 ± 0.05
Weighted $\Delta_{(E2008-E2007)}$ (per cent)	-16 ± 4	-32 ± 15	-4± 4	-6 ± 8
Weighted Gain (per cent)	. ± 2ab	-17 ± 8a	12 ± 3b	8± 4b
Plant Cover in 2007 (per cent)	157.70 ± 18.27	141.37 ± 18.47	130.38 ± 9.67	156.03 ± 6.67
Plant Cover in 2008 (per cent)	219.06 ± 32.35	199.40 ± 8.29	257.37 ± 20.25	213.72 ± 16.94
Weighted $\Delta_{(per\ cent2008-\ per\ cent2007)}$ (per cent)	31 ± 17	35± 16	65 ± 13	31 ± 4
Weighted Gain (per cent)	.- ± 12ns	4 ± 12ns	34 ± 11ns	0 ± 8ns
Soil Bacterial Community				
Band richness in 2008 (S_{2008})	5.67 ± 0.33	8.33 ± 0.33	9.67 ± 0.33	7.67 ± 0.33
Absolute Gain (per cent)	. ± 24a	267 ± 24b	400 ± 24c	200± 24b
Band evenness in 2008 (S_{2008})	0.93 ± 0.01	0.93 ± 0	0.89 ± 0.01	0.90 ± 0.04
Absolute Gain (per cent)	.- ± 1ns	0± 0ns	-4 ± 1ns	-4 ± 2ns
Soil				
Soil (inrogN) (ppm) *	8.93 ± 1.05	9.45 ± 1.85	9.21 ± 2.25	11.51 ± 4.8
Yearly average (sd) **	7.18 (4.27)	10.26 (4.43)	12.13 (8.90)	18.74 (2.76)
Gain in annual mean (per cent)	.- ± 32a	308 ± 28b	495 ± 40b	1156± 14c
Gain in annual sd (per cent)	.- ± 37a	17± 26a	463± 147ab	697 ± 249b

to promote the appearance of new herbaceous maquia species and the maintenance of ruderals. Changes in plant cover were analysed on a plant group basis highlighting the effects of increased N availability (Figure 5.17). Plant groups could be viewed as: (i) benefiting from increased N availability - ruderals and herbaceous maquia species; (ii) benefiting from increased N availability as long as there was no ammonium toxicity - ericaceous, legume shrubs and grasses; and (iii) affected by increased N availability especially in the form of ammonium - summer semi-deciduous.

Effects on soil microorganisms

Aboveground communities may also be influenced by changes in soil microorganisms and vice-versa (Klironomos 2002, Brooker, 2006). Therefore, Temperature Gradient Gel Electrophoresis (TGGE) fingerprinting was applied to monitor the impact of N addition in the soil bacterial communities structure. Numerical analysis of TGGE profiles and the corresponding dendrogram (Figure 5.18) indicated that, for the N doses used, only one year of addition was needed to induce changes also in soil bacterial communities. The three main clusters observed in the dendrogram presented high similarity levels and included: (i) control plots and one of the 40 kg N-NH_4^+ha^{-1}yr^{-1} fertilized plots (84 per cent similarity), (ii) all the remaining 40 kg N ha^{-1}yr^{-1} and one of the 80 kg N-NH_4NO_3ha^{-1}yr^{-1} and (iii) the remaining two plots receiving 80 kg N-NH_4NO_3ha^{-1}yr^{-1} (both with 100 per cent similarity). These results suggest a community shift mainly in response to the amount of N added rather than to the N form. TGGE outputs also allowed the estimation of bacterial diversity using the number and intensity of the TGGE separated bands. Accordingly plots fertilized with 40 kg NH_4NO_3ha^{-1}yr^{-1} displayed the highest bacterial band richness, while non-fertilized plots had the lowest. The remaining treatments showed intermediate values (Table 5.7). These data

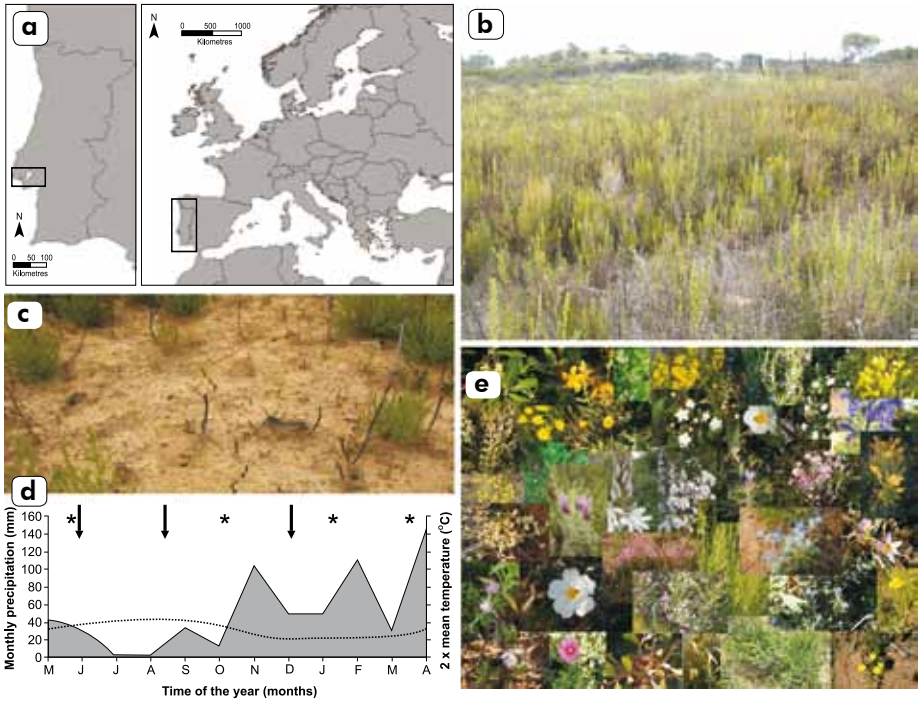

Figure 5.16: a) General location of the studied site; b) landscape view; c) surface soil view; d) mean monthly temperature (dotted line), total monthly precipitation (grey) and times of N additions (arrows) and soil sampling (*) from Spring 2007 to Spring 2008; and e) examples of some of the existing plant species at the study site. *Photos © T. Dias*

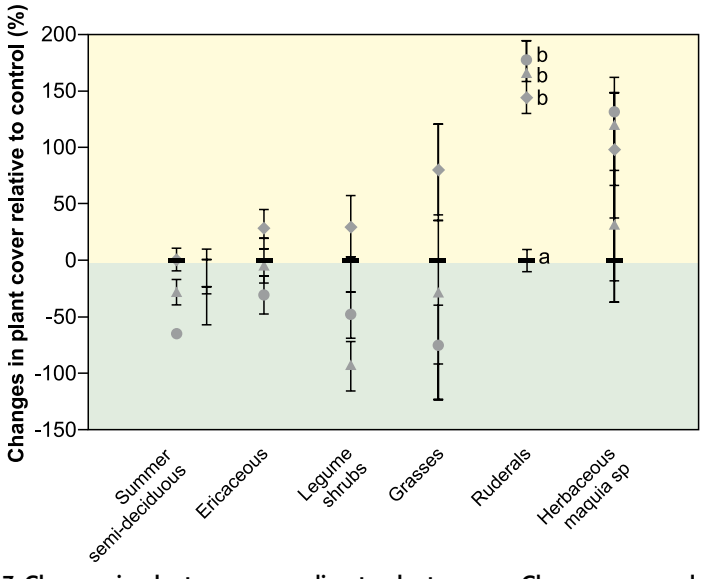

Figure 5.17 Changes in plant cover according to plant groups. Changes were calculated as in Table 5.6. Plant species were grouped according to (i) functional groups (Barradas *et al.*, 1999) - * - or (ii) its common habitat. Trees and evergreen sclerophyllous were not considered on this analysis because at this stage of succession they are not dominant and their study requires other methods. Horizontal bars: control; Triangle 40 kg N-NH_4^+; Diamonds- 40 N-NH_4NO_3; and Circles 80 N-$NH_4NO_3ha^{-1}yr^{-1}$. Different letters refer to statistically significant differences between treatments (ANOVA followed by a Bonferroni test). Dark grey shadows correspond to a decrease; light grey shadows correspond to an increase (n = 3 experimental plots per treatment).

support the previously invoked ammonium toxicity as a mechanism that could account for the lack of stimulation of plant cover (Table 5.7 and Figure 5.18) by increased N availability. Soil bacterial band richness depended on both N-form and dose. Bacterial band evenness showed no differences between treatments (Table 5.7).

Soil inorganic N availability

Because plants and soil biota evolved under specific nitrogenous environments they show preferences for specific patterns of N availability (Cruz *et al*., 2003, Gallardo *et al*., 2005, Cruz *et al*., 2008). Can the observed biotic changes be explained by changes in patterns of soil inorganic N availability? In spite of the levels of N applied to the system, soil inorganic N concentration, did not change between treatments after one year of fertilization (Table 5.7). Although, based on the seasonal means, there were significant changes in the annual pattern of soil inorganic N concentration: adding N significantly increased annual mean availability and annual variation. After fertilization, N accumulates in these soils until it is washed away by strong rain events, characteristic of the Mediterranean climate. The final value corresponds to the Mediterranean spring when strong rains (Figure 5.16) and intense biological activity occur (Sardans and Peñuelas, 2005), these factors combined explain why soil inorganic N concentrations were similar in all treatments. However more research is needed to identify periods of simultaneous increased soil N availability and biological activity (strongly limited by water) because biota (plants and microorganisms) that are active during these periods will directly ‘win’ or ‘lose’ from increased N availability.

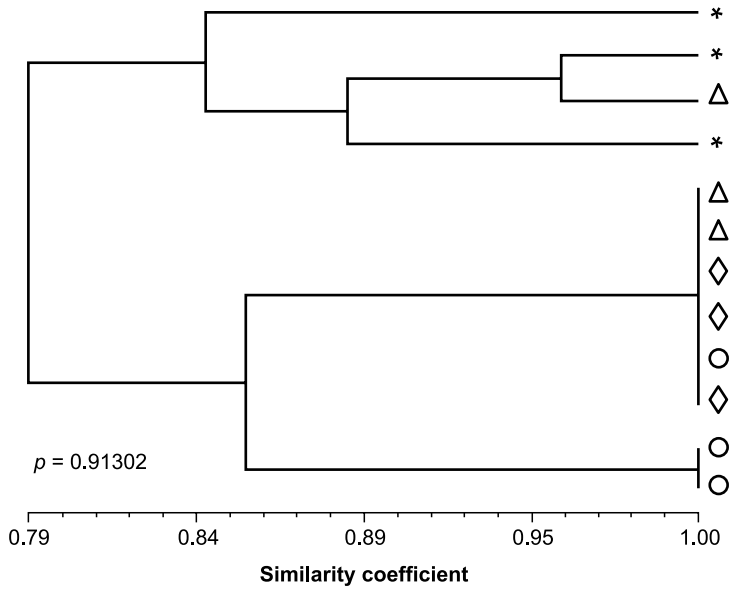

Figure 5.18: Dendogram obtained from numerical analysis of TGGE fingerprints of soil bacterial communities evaluated in Spring 2008. Asterisk: control; Triangle 40 kg N-NH_4^+; Diamonds- 40 N-NH_4NO_3; and Circles 80 N-$NH_4NO_3ha^{-1}yr^{-1}$.

Increased N availability in a Mediterranean-type ecosystem

In general our data suggest that short-term N fertilizations increase plant and soil bacterial diversity (richness and evenness), which seems to contradict most of the worldwide studies published so far (see Bobbink *et al*., 2010 for review). However, current knowledge suggests that the effects of N enrichment are dependent on the initial N status of the system: on highly productive sites, there is a potential for biodiversity loss and vice versa (Emmett 2007, Chalcraft *et al*., 2008). In fact, many studies have been performed in systems no longer N-limited, so that the stage of N-induced increases in biodiversity are not detectable. Data reported here refer to the initial changes in an ecosystem that has historically been subjected to low N deposition, with the initial increase in species richness probably representing an alleviation of the N limitation imposed on communities. Similar results have been reported for lichen community diversity in cork-oak woodland (Pinho *et al*., 2009) and other systems under similar circumstances (e.g. Calvo *et al*., 2007).

Adding N significantly changed the pattern of soil inorganic N availability. But the different N treatments appear to be differentially targeting distinct plant species (Table 5.6) and groups (Figure 5.17) and therefore distinct ecosystem functions (Hooper and Vitousek, 1997). These community changes may have been driven by altered patterns of soil inorganic N thus suggesting a key role for N in shaping Mediterranean-type ecosystems. Considering the reactivity of Mediterranean-type ecosystems to N, maintaining these systems within favourable conservation status constitutes a scientific, social and political challenge.

5.9.5 Conclusions

- The ecosystem was very responsive since only one year of N fertilizations was enough to induce ‘visible’ changes in both biotic and abiotic compartments: N fertilizations created new and distinct annual patterns of soil inorganic availability; N induced an increase in plant and soil bacterial biodiversity and plant cover.

- Ecosystem responses depended on the N-form (plant evenness), N-dose (plant richness) or both (plant cover and soil bacterial richness).

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5.10 All forms of reactive nitrogen deposition to Natura 2000 sites should not be treated equally: effects of wet versus dry and reduced versus oxidised nitrogen deposition.

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Summary

- Atmospheric nitrogen deposition occurs in several different forms, including wet deposition of ammonium and nitrate, and dry deposition of ammonia. Each of these inputs occurs

intermittently, according to patterns of precipitation, long range pollutant transport and local ammonia dispersion from point sources.

- Evidence is presented from a nitrogen manipulation study, undertaken using 'real world' treatment scenarios, on an ombrotrophic bog where the effects of gaseous ammonia are compared with wet deposited nitrogen, as ammonium or nitrate.
- Per unit N deposited, ammonia is found to be much more damaging to nitrogen sensitive plant species than wet deposited ammonium, which, in turn, is found to be more damaging than wet deposited nitrate.
- Damage is related to the likelihood of nitrogen accumulation in the plant tissue, which is greater with ammonia > ammonium > nitrate.
- Ammonia effects on lower plants are thought to be related to physiological damage associated with the intermittent high ammonia concentrations.
- Thresholds for damage effects from ammonia reduce logarithmically with the logarithm of time, indicating a memory effect.
- Ammonia damage to *Calluna vulgaris* appeared to be mediated indirectly through interaction with stress, winter desiccation, pests and pathogens.
- Wet ammonium deposition at N doses > 24 kg N $ha^{-1}y^{-1}$ significantly increases N accumulation in lower plants leading to reduced growth in the pleurocarpous mosses *Hypnum jutlandicum* and *Pleurozium schreberi.*
- By comparison no significant effects of nitrate have been detected except in *Sphagnum capillifolium*, which is sensitive to N dose.
- The results clearly demonstrate that the form of nitrogen deposition affects nitrogen impacts on an acid peatland ecosystem, with adverse effects per unit N input in the order: dry ammonia > wet ammonium > wet nitrate. These differences need to be recognized in the development of air pollution policies, although the order may vary with ecosystem type, particularly with respect to nitrate versus ammonium.

5.10.1 Introduction

The effects of enhanced nitrogen deposition on semi-natural systems: elevated N concentrations, increased incidence of pests, pathogens and stress (Bobbink *et al*., 2009) and reductions in species richness and change in function (Stevens *et al*., 2006) are well described. However, the contributions of the main forms of N deposition to these N responses are much less well understood, but may be crucial to predicting N impacts on Natura 2000 sites. A key question for conservation policy and regulation is, are all forms of N equally damaging per unit N deposited, and if not, what are the implications for biodiversity and ecosystem services?

This paper briefly summarises what is known about the N forms most likely to affect semi-natural ecosystems, i.e. ammonia as dry deposition and ammonium and nitrate in wet deposition. Field evidence of effects of these three forms, based on a seven year, ongoing experiment on an ombrotrophic bog (Whim Bog, Sheppard *et al*., 2004; Leith *et al*., 20004) is discussed in relation to the different effects of these three forms of nitrogen deposition. Observations from this experiment on Whim Bog are particularly relevant for Natura 2000 sites because they represent the only comparison between these three N forms on the same site and where the application of N is consistent with 'natural' N deposition.

It should be recognized that other forms of reactive nitrogen contribute to dry deposition, including nitric acid, nitrous acid, nitrogen oxides, organic nitrogen forms (including PAN, amines, etc) and nitrogen containing aerosol. Although there may be differential toxicity of these many different forms, in most cases each of these forms provides only a small additional contribution to total nitrogen deposition. The major individual forms are ammonia, wet deposited nitrate and wet deposited ammonium, as investigated here.

The aims of this paper are:

- to show that the effects of different N forms are not equal per unit N deposited to on an acid bog ecosystem and are species specific;
- to show that field effects are generally consistent with current understanding of the mode of action of the different N forms.

5.10.2 The Whim Bog experiment

Wet deposition exposure frequency is coupled to precipitation and concentrations are relatively low, though individual rain events can contain higher ammonium and nitrate concentrations. By contrast, gaseous ammonia may deposit with a much higher variation, especially in sites near point sources, such as intensive pig and poultry farms. The Whim experiment is designed to reproduce these differences in delivering the different N deposition inputs to the experimental treatments.

For the dry nitrogen treatment, the ammonia gas is released only in a 'downwind' direction from the source. In the free air release, ammonia is mixed with air and dispersed from a line source, providing an exponential concentration gradient declining down to ambient concentrations (0.3-0.4 $\mu g\ m^{-3}$) ~ 100 m from the source. The release simulates a range of ammonia concentrations typical of those measured downwind of agricultural sources, intermittent high concentrations (1000-2000 $\mu g\ m^{-3}$) for short (<h) periods with ambient concentrations dominating for ~85 per cent of the time (Leith *et al.,* 2004).

For the wet nitrogen treatment, salts of sodium nitrate or ammonium chloride, combined with rainwater, are used to provide wet deposited oxidised N and reduced N inputs, respectively (Sheppard *et al*., 2004). Ammonium sulphate salt was not used, because sulphate strongly influences soil pH and aluminium concentrations and can be phytotoxic (Sheppard *et al*., 1994; Silvertown *et al.,* 2006). Treatment was automated and all year round, except when the temperature fell below 0 ^{o}C. As with dry deposition, N was deposited at a range of concentrations, at least 100 times more concentrated than average rain concentrations, in > 100 events. The N deposition loads supplied 8, 24 or 56 kg N $ha^{-1}y^{-1}$, over and above the ambient 8 kg N $ha^{-1}y^{-1}$ deposition to 4 ~13 m^2 replicate plots (Sheppard *et al*., 2004b). These N doses were replicated at different distances along the ammonia transect (Cape *et al*., 2008).

5.10.3 Results and discussion

Observed effects of the different nitrogen forms relative to the damage they cause: dry versus wet deposition of nitrogen.

Ammonia is a highly reactive alkaline gas that deposits to acidic surfaces especially when wet. Acid ombrotrophic bogs support many lower plants that have large acidic surfaces, e.g. *Sphagnum* and *Cladonia* species (Jones, 2007). Such species have evolved to obtain their nutrients from atmospheric deposition, i.e. maximising their uptake surface (area to mass) rather than developing features to exclude ions/gases, such as cuticles.

Sphagnum capillifolium and *Cladonia portentosa* growing along the gaseous ammonia transect at Whim were both found to be highly sensitive to ammonia, with detrimental effects observed very quickly (< three months) in *C. portentosa*. The lichen was first bleached, then 'turned' green and slimey. Membrane damage was assumed based on the much lower potassium concentrations recorded in the apical parts of the lichens for the transect zone receiving > 24 kg N (Sheppard *et al*., 2004c).

Over subsequent years, the zone of species loss has extended to areas receiving > 8 kg NH_3-N $ha^{-1}y^{-1}$ of dry deposition, indicating that the impact of gaseous ammonia increases with time. It appears that continuous exposure gradually weakens sensitive plants, causing the damage threshold concentration to fall (Sheppard *et al*., 2009).

Equivalent N doses (56 kg N $ha^{-1}y^{-1}$) applied as either wet deposited ammonium or nitrate have caused limited damage, which has developed slowly over the years. Damage has been confined to small patches within a clump, rather than causing complete eradication of *C. portentosa* from the plot (Figures 5.19, 5.20).

The detrimental effects of gaseous ammonia on the bog moss *S. capillifolium* took longer to develop than the effects on sensitive lichens, i.e. required a larger accumulated NH_3-N dose and were restricted to the green pigmented form of the moss. (The red pigmented form of *S. capillifolium* is found to be less sensitive to ammonia.) The speed of response of *S. capillifolium* to wet N deposition was more similar to that seen with *C. portentosa*, except that detrimental effects were more pronounced with ammonium than with nitrate.

Implications of the form of nitrogen: dry versus wet deposition.

The implications from this field comparison are that sensitive lower plants can tolerate > 5 times the N load, when the N is deposited wet in precipitation than when it is supplied by ammonia dry deposition. One explanation for this may be related to the high gaseous concentrations associated with ammonia deposition, when the wind is blowing from an ammonia source. In the field in source regions, ammonia concentrations are locally highly variable (Asman *et al*., 1998) responding to wind direction, meteorological conditions and the distribution of ammonia sources. Also, due to its reactive nature, up to 50 per cent of ammonia emissions are deposited within a 5 km radius of their source (Sutton *et al*., 1998). In rural areas dominated by livestock production, sites downwind of ammonia sources will experience intermittent very high, ammonia concentrations, > 600 µg m^{-3} (Esther Vogt pers comm.). Such intermittent spikes in ammonia concentrations are characteristic of this pollutant, even in areas remote from sources. In the Whim free air release the average concentrations of these spikes can be estimated from the mean monthly concentration because the duration of the ammonia release is known. During the six years the experiment has been running the prevailing wind direction has permitted release for < 14 per cent of the month, implying that the ammonia concentrations during the exposure period to be on average seven times higher than the monthly mean air concentration measured with passive samplers, and potentially phytotoxic (Van der Erden, 1982; Krupa, 2003). Exchange of this gaseous ammonia concentration with plants is dominated by the solubility of ammonia in water, how these air concentrations compare with concentrations of ammonium in precipitation, wet deposition, have still to be demonstrated for the vegetation at Whim when ammonia is being released, but we expect them to significantly exceed those measured in rain.

Concentrations of ammonium and nitrate in precipitation are generally quite low: highly skewed to the low concentrations, median 20-25 µM max ~ 320 µM, even in cloudwater where they can be up to 10 times higher. Although high concentrations can occur, these tend to be restricted to occasional pollution events associated with long range transported air pollution. In the Whim manipulation study considerably higher concentrations, maximum 4000 µM, with a high application frequency are used to deliver the N dose, interspersed amongst the low N concentrations in rainfall. But, despite these relatively high concentrations, visible damage has been minimal, even to N sensitive species (Figure 5.19). Our results suggest that the effects of ammonia on vegetation may be mediated aboveground through foliar uptake of high concentrations that exceed the capacities, especially those of lower plants, to detoxify it. This hypothesis is supported by the significantly higher N concentrations (+>45 per cent) measured in *Calluna* shoots treated with ammonia, compared with

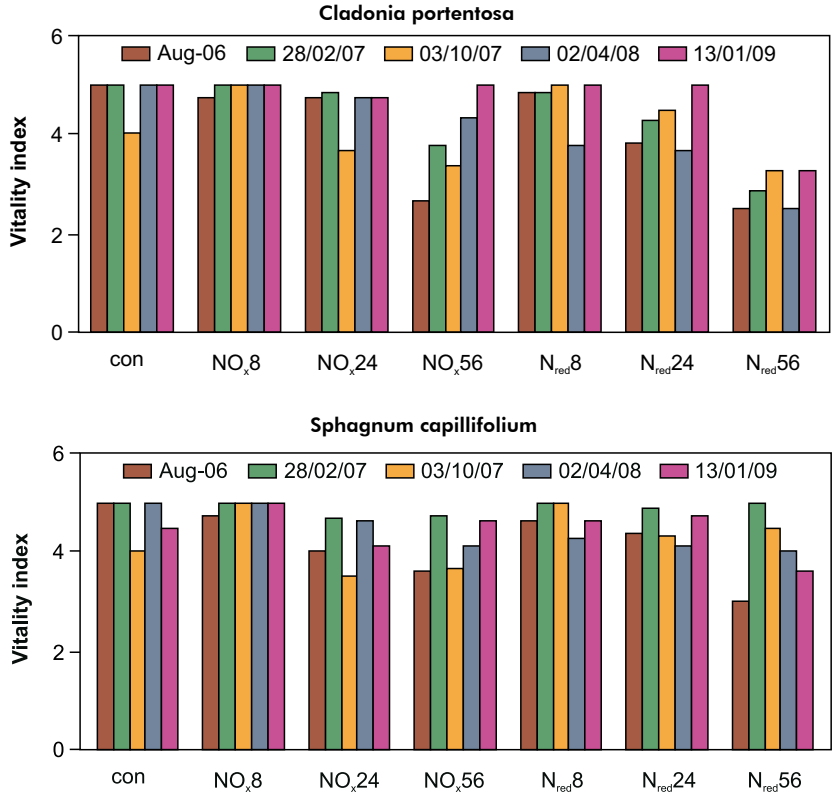

Figure 5.19: Vitality Indices, subjective scoring of visible damage where a maximum score of five constitutes healthy, for *Cladonia portentosa* and *Sphagnum capillifolium* treated with sodium nitrate (NO_x) or ammonium chloride (Nred) in rainfall at 8, 24 or 56 kg N $ha^{-1}y^{-1}$

those treated with the equivalent wet N deposition (+ 10-15 per cent) (Sheppard *et al*., 2008). In addition, effects may be partly mediated by pH interactions, where gaseous ammonia can increase the pH of leaf surfaces. The role of this mechanism remains to be explored in more detail.

As a gas, ammonia has more potential uptake pathways than wet deposited N, which is taken up through the roots, which are often mycorrhizal, and for some plants, e.g. *Calluna* also via the shoots (Bobbink *et al*., 1992). Gases are taken up through stomata in higher plants, when they are open. The stomatal pathway is controlled through a combination of physical and biological processes (see Dragosits *et al*., 2008) and is not as effectively regulated as root N uptake, which can be adjusted in relation to demand (see Sheppard and Wallander 2004a).

Loss of *Calluna* cover was recorded at > 10 kg N after five years and significant losses, > 85 per cent cover loss were observed when the annual NH_3-N deposition exceeded 24 kg N. The loss of *Calluna* cover coincided with an almost 100 per cent increase in N content and was accompanied by significantly increased damage from abiotic and biotic stress (Sheppard *et al*., 2008). The combination of ammonia exposure and drying winter conditions led to the loss of *Calluna* cover. These observations suggest that much of the damage/loss of heathlands in the Netherlands during the late 80s early 90s was caused by the dry deposited gaseous ammonia form of nitrogen. Exposure to ammonia can double foliar N status significantly increasing the likelihood of death from abiotic stressors such as drought, freezing injury and pests and pathogen attacks, as were recorded in the

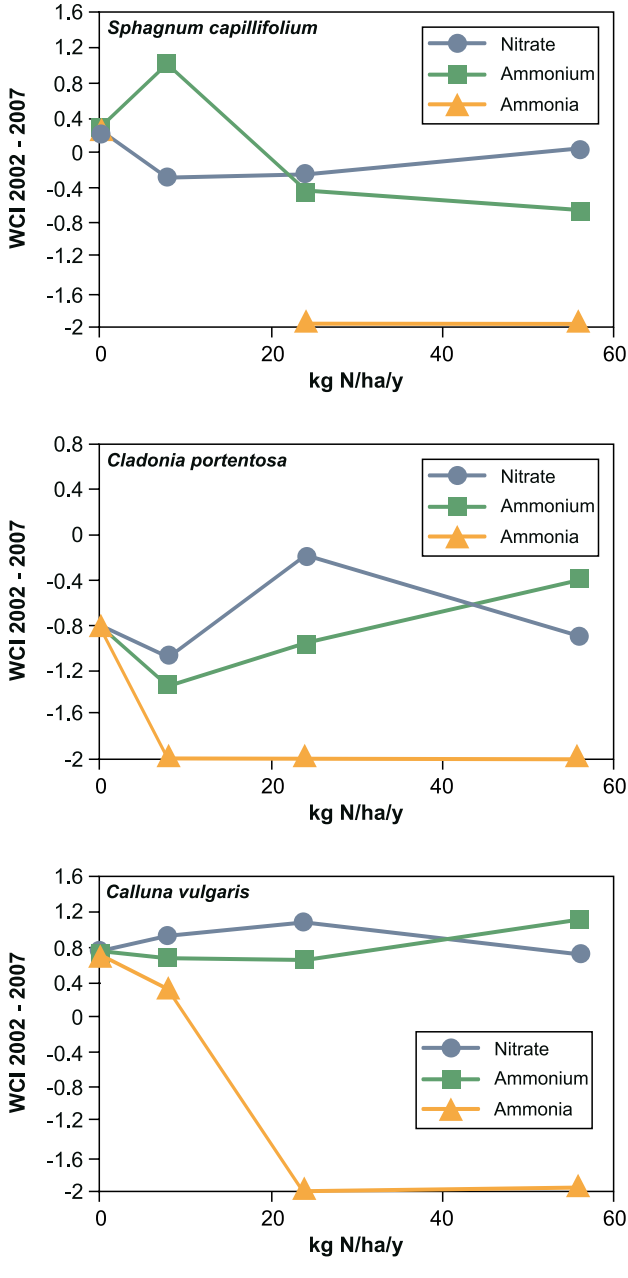

Figure 5.20a,b,c: Change in the cover index, cover after five years of treatment in relation to the start cover, of N sensitive species *Sphagum capillifolium*, *Cladonia portentosa* and *Calluna* over five years of treatment with nitrate, ammonium or ammonia

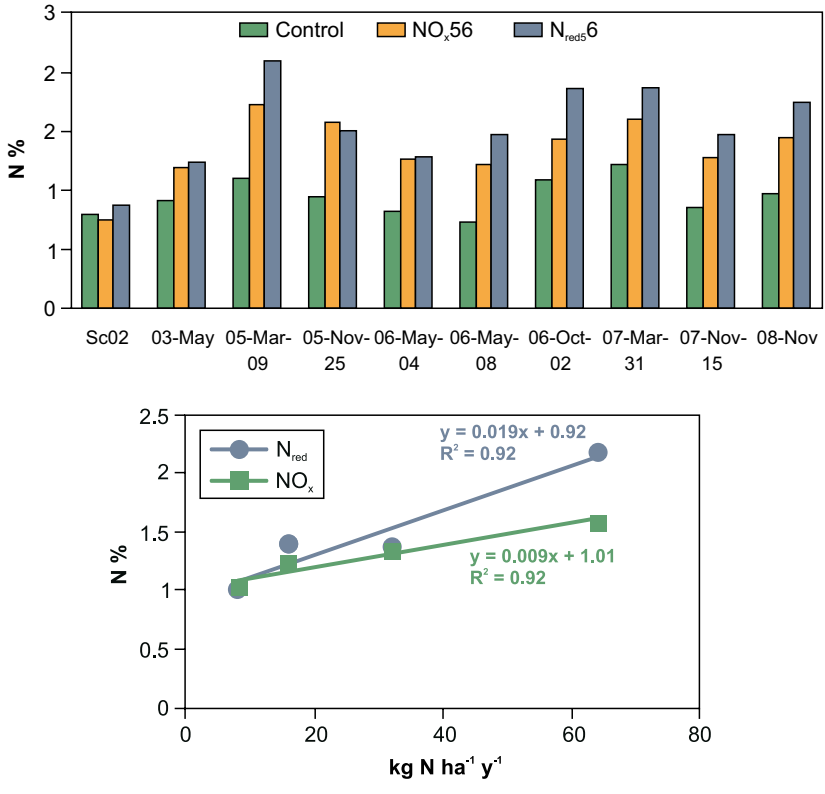

Figure 5.21: a) Nitrogen concentrations (per cent dry wt) in *Sphagnum capillifolium* treated with 56 kg N ha^{-1} of nitrate (NO_x) or ammonium (Nre) and the control between 2002 pre treatment and the end of 2008; and b) *Pleurozium schreberi* in March 2009 in relation to N dose in precipitation

Netherlands (van der Eerden *et al*., 1991). The potency of ammonia and its memory effects are now acknowledged in the revised critical levels (Sutton *et al.,* 2009). The *Cyperaceae* (*Eriphorum* spp.) and other ericoids on site, *Erica tetralix*, *Empetrum nigrum*, *Vaccinium oxycoccus* and *V. myrtillus* were not apparently sensitive to ammonia-N.

Differential effects of reduced versus oxidised nitrogen in precipitation

By contrast to the effect of gaseous ammonia in raising surface pH, accumulation of ammonium in plant tissue can lead to acidification, accompanied by toxicity, membrane damage and nutrient imbalance with negative impacts on growth (Krupa, 2003). In addition, foliar uptake of cations exceeds that of negatively charged anions such as nitrate, so that ammonium uptake from atmospheric deposition is thought to be more difficult to regulate. By contrast no such effects have been reported for nitrate, its assimilation is not associated with acidification (rather the reverse) and it can be accumulated safely in the vacuole. Little field evidence, by way of comparisons, exists to support the greater potential for ammonium to cause detrimental effects compared with nitrate. However, high ammonium concentrations in the soil or water layer have been shown to be toxic, restricting root development in sensitive plant species especially when the ratio of ammonium to nitrate is high (Roelofs *et al*.1996). Evidence from field surveys, in the UK at least, is confounded by the differences in range of reduced versus oxidised N deposition.

The Whim experiment on an ombrotrophic bog is the longest running, with the most realistic treatment scenario, to examine the significance of reduced versus oxidised nitrogen.

Significant changes in the cover of the major species in response to wet deposition of either ammonium or nitrate have not been detected after six years, although the vitality of N sensitive plant species does appear to be poorer with ammonium (Figure 5.20). Foliar N content was always found to be higher in the mosses *Sphagnum capillifolium* and *Pleurozium schreberi* (Figure 5.21) when treated with ammonium than with nitrate, although the effects were rarely significant below 24 kg N $ha^{-1}y^{-1}$. No significant enhancement, 'memory effect' has been detected in six years of per cent N data for *Sphagnum capillifolium*, although significant differences between oxidised and reduced N were found ($p<0.05$) (larger effect with ammonium than with nitrate).

Significant reductions in annual growth of -30 per cent were measured in *Hypnum jutlandicum* a common pleurocarpous moss in response to wet ammonium inputs of > 24 kg N $ha^{-1}y^{-1}$. Both N forms significantly ($p=0.1$) reduced plant mass in *Pleurozium schreberi* (-3 per cent with nitrate and -21 per cent with ammonium). In *Sphagnum capillifolium* both N forms also reduced growth and again the effects were more pronounced with ammonium at all N doses (Kivimaki *et al*., 2008). In *Hypnum jutlandicum* the N content was increased by +59 per cent with ammonium compared with by +29 per cent with nitrate, with similar increases being recorded for *Pleurozium schreberi* (+66 per cent and 31 per cent, respectively).

Both ammonium and nitrate increased the *Calluna* cover, only enhanceing the foliar N concentration by a modest 15 per cent, significant ($p < 0.05$). Exposure to equivalent wet N doses continues to enhance the growth of *Calluna*, helping to restrict the accumulation of N. No significant effects of N form were found.

In this manipulation experiment effects associated with wet deposited oxidised N remain barely detectable with respect to changes in vitality, growth and cover of the N sensitive species.

5.10.4 Conclusions

Our results from a field manipulation experiment of an acid peatland bog, comparing dry ammonia with wet nitrate and ammonium inputs confirm a clear pattern of much greater N accumulation in response to ammonium addition than to nitrate addition, with the largest accumulation and effects being observed following treatment with gaseous ammonia. The results suggest that the greater the accumulation of N the greater the risk of damage. Thus the implications for Natura 2000 sites are that habitats in the vicinity of point sources of ammonia are expected to be subject to much larger impacts per unit N deposited, than sites where the nitrogen is primarily received by wet deposition. These differences must be recognized in the development of policies to protect the Natura 2000 network from nitrogen deposition. In particular, there would be a case to locate ammonia emission point sources as far from sensitive Natura 2000 sites as possible. With respect to wet deposited N, ammonium is found to be more damaging than nitrate because it accumulates with toxic effects. However, on other soil types the form effects may differ, depending on soil chemistry.

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5.11 Evaluation of nitrogen indicators on ombrotrophic acid bogs: observations from a nitrogen manipulation study.

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Summary

- Among Natura 2000 ecosystems, ombrotrophic bogs/peatlands are likely to be one of the most sensitive to reactive N deposition.
- This paper briefly examines methods of evaluating species, growing on an ombrotrophic bog, to different N forms using a case study from Whim bog to illustrate peatland vegetation response to wet and dry N deposition.

- Vegetation cover data are used to examine typical N indicators in terms of species richness, diversity, functional group response, Ellenberg indicator values, species cover and National Vegetation Classification (NVC).
- In addition an ordination technique is applied to the data and principal response curves (PRC) generated to examine temporal community response to the N deposition.
- The N indicators tested here of species richness, diversity, Ellenberg indicator values and NVC were insensitive to the apparent effects of the N deposition treatments. This is despite significant responses at both the species and community levels detected using a weighted cover index and PRC.
- In addition responses varied with N form and particular functional groups e.g. species from the ericoids or *Sphagnum* show differential responses: *Calluna* responds negatively to ammonia whereas *Vaccinium myrtillus* and *Empetrum nigrum* have responded positively, the hummock forming *S.capillifolium* is negatively affected by N while the wetter loving *S. fallax* increased in response to N additions.
- The results emphasise that indicators represent a range of sensitivities and a considered approach to the choice of indicator needs to be applied with reference to the ecosystem under question and form of nitrogen applied.
- Future research efforts should concentrate on developing predictive indicators with sufficient sensitivity to show species level change that can be used to detect and predict loss of important peatland species or groups.

5.11.1 Introduction

In the UK, many of the remaining lowland ombrotrophic peat bogs are designated as Natura 2000 sites. Atmospheric reactive nitrogen (N) deposition, largely as a consequence of activities associated with the intensification of agriculture, threatens the sustainability of naturally nutrient poor ecosystems such as ombrotrophic peatlands (Bobbink *et al*., 2009). This is because the organisms that inhabit these ecosystems have evolved under nutrient limitation and, until recently, have been maintained in this state. The threat posed by N deposition can also be compounded by changes to land-use practices such as grazing and burning. Once effects are detected there are high labour costs associated with mitigating N deposition via removal and management. Globally, nitrogen deposition has been ranked as the third most important driver behind land use and climate change, but regionally, in northern temperate biomes, N deposition is expected to be the major driver of biodiversity change (Sala *et al*., 2000).

Evaluating the effects of enhanced N deposition is a complex area of study, but several indicators of the effects have been proposed. Ideally, indicators of enhanced N deposition need to be specific to N, robust, widely applicable and preferably cheap and simple to apply (Sutton *et al*., 2004). A further challenge is to identify sensitive approaches that reliably point as 'early warning' indicators towards the initial onset on nitrogen impacts, before changes become irreversible. More recently, the need to take account of N effects with respect to ecosystem functioning and the provision of ecosystem services has been widely recognised, raising the importance of ecosystem composition at the functional group and species levels.

In ombrotrophic peatlands, non-vascular plants, in particular *Sphagnum* mosses are crucial, being described as the 'engineers' of these ecosystems (van Breemen, 1995). This genus is arguably largely responsible for the worlds largest terrestrial carbon store mostly found in northern peatlands (Gorham, 1991). However, non-vascular species include mosses, lichens and liverworts, groups which are often overlooked in vegetation surveys. Yet globally, these groups are widespread, present in most major ecosystems, and make substantial contributions to biodiversity, biomass and biogeochemical cycling (Cornelissen *et al*., 2007).

Of the 38 main mire communities recognized in the UKs' National Vegetation Classification (NVC), 11 are specifically defined by occurrence of *Sphagnum* species (Rodwell, 1991). In a recent survey of 15 Scottish lowland raised bogs, we found that *Sphagnum* spp. dominated, with up to 11 different species, which together with other mosses and *Cladonia* spp., made up > 66 per cent of the species recorded, with vascular plant species being in the minority.

Non-vascular species tend to derive almost all their nutrients from atmospheric deposition. They have mostly evolved to maximise the uptake of these atmospherically derived nutrients. For example, they have relatively large surface area to mass, with no means of excluding or restricting uptake, i.e., they have no cuticle. This tight coupling to atmospheric deposition means these plants are often the most sensitive components of the vegetation and importantly their responses can provide a potential monitoring tool for following the effects of N deposition (Leith *et al*., 2005).

The question arises of how best to quantify the response of ombrotrophic peatland species in order to evaluate effects of reactive N deposition?

The most common approach is to look at total species richness; this simply refers to the number of species recorded in a given area, the greater the number the greater the richness. The approach is widely used in conservation studies to determine the sensitivity of ecosystems and their resident species. A long-term decrease in plant species richness in the UK since 1990 has also coincided with the decline in abundance of farmland butterflies and birds over the same period (Countryside Survey UK headline messages from 2007). However, the species richness approach ignores that some species may be more important than others i.e. fails to prioritise between species. With respect to conservation value, ecosystem function and sustainability, all plant species are not equal e.g. the case of *Sphagnum* spp. detailed above, this genera exerts a pivotal influence over both the existence and sustainability of peatland ecosystems.

Ellenberg N indicator values represent another approach that was initially considered as a general indicator of soil fertility rather than specifically N availability (Hill *et al*., 1999), but has been widely used to assess habitat change with respect to eutrophication (e.g., Countryside survey UK). The value of the Ellenberg approach is that it is easy to score, based on species occurrence and abundance, with the method relying on a derived 'average' species response based on the habitats in which the species is typically found.

This paper examines some of the tools available for assessing the impact of nitrogen in the context of an ombrotrophic bog, one of the ecosystems expected to show the greatest response to increasing nitrogen deposition in terms of change in biodiversity (Sala *et al*., 2000). We use a field nitrogen manipulation study (Whim Moss) established in 2002 (see this volume; Leith *et al*., 2004; Sheppard *et al*., 2004), comparing the three nitrogen forms most likely to affect ombrotrophic bogs (ammonia gas, wet deposited ammonium and nitrate). Specifically, we address the significance of our evaluation methods for the sustainability and functioning of a semi-natural ecosystem. We consider whether all species should be treated equally, and discuss how to evaluate species with respect to the functions they serve within an ecosystem.

Whim Moss in the Scottish Borders is a relatively acid site, pH ~3.6 in HO, and shows substantial differentiation in micro-habitat. In hollows, the water table is at the surface for most of the year, whereas in hummocks, which can be up to 0.5 m above the hollows, the water table almost never reaches the surface. The bog is not actively managed at present and is subject to some grazing by rabbits. The vegetation, classified as NVC M19 *Calluna vulgaris – Eriophorum vaginatum* blanket mire dominated by *Calluna vulgaris* (L.) Hull, *Eriophorum vaginatum* L., *Sphagnum capillifolium*

(Ehrh.) Hedw. and *Cladonia portentosa* (Dufour) Coem., which make up >85 per cent of the cover. The *Calluna* is either mature phase or in places degenerate. Permanent quadrats were established in the plots, with species cover recorded since 2002. We examine the data collected in terms of species (vascular and non-vascular), functional group and community responses, analysing in terms of richness and diversity using common statistical and multivariate ordination techniques. Changes in species cover were also evaluated through the computer programme ComKey (C. Legg, unpublished).

5.11.2 Results and discussion

Species richness and diversity

No differences in overall species richness were detected reflecting the widely observed trend for species swapping in response to reactive N deposition: N sensitive species are replaced by more N loving, N tolerant species, so that the overall richness remained unchanged.

Species diversity is generally a much more useful metric than species richness because abundance is included rather than just presence/absence. However, the Shannon Weiner Index, Equitability and Simpson's Index likewise remained unaffected by the N treatments at ~ 2.6, 0.55 and 0.8 respectively. The appearance and/or disappearance of species within any single area can be caused by a number of extraneous or inter-related factors. Both richness and diversity suffer from the loss of identity in their metric and thus it is not immediately obvious how changes in these metrics can be related to N deposition effects or to ecosystem function without reference to the actual species. This would suggest that the techniques may be somewhat redundant in low diversity systems.

Ellenberg Indicator Values

In response to the increases in N deposition in this study we might expect to observe an increase in the Ellenberg N indicator score. After five years of N additions equivalent to > five times the critical N load for ombrotrophic bogs, no change in Ellenberg N was detected. There may be three reasons for such an observation. Firstly Whim bog, like many such ombrotrophic bogs, is co-limited by P availability and not just N. Secondly, the Ellenberg indicator N values are derived from the species abundance data. In order for a significant difference to be detected a shift from dominance of, for example, species with an indicator value of one (indicator of extremely infertile sites) to those with a value of three (indicator of more or less infertile sites) would be required. However, given the species pool at Whim, it is more likely that dominance will shift between species that have similarly low Ellenberg N scores or bare peat (which by definition has no Ellenberg value) rather than a large shift to more fertile species. Lastly, there are likely to be temporal effects (see Principal Response Curves below) and it may be that not enough time has elapsed for a shift to more fertile species to have occurred.

We therefore urge caution in the use of Ellenberg indicator values for the detection of N deposition effects as, for ombrotrophic bogs at least, Ellenberg indicator values may not have sufficient sensitivity. Given the significant changes in species composition observed at the Whim site, our results point to the need to develop more suitable indicators for ombrotrophic bogs.

Change in cover of important peatland species, functional groups

As indicated in Sheppard *et al*., (this volume), exposure to ammonia resulted in large reductions in the cover of three major species/groups, *Calluna*, *S. capillifolium* and *Cladonia portentosa*, which are functionally, key components of this peatland. The effects of exposure to ammonia are captured in the photographs (Figs 5.22, 5.23) and quantified as change in cover after five years in relation to the pre-treatment cover (weighted cover index) (Figure 5.24), for the most common bog plants at Whim. Positive values indicate an increase in cover while negative values indicate cover has been

Figure 5.22: Photograph (taken on 27/08/09) looking downwind of the 60m ammonia free air release transect at Whim Bog, a lowland raised bog in the Scottish Borders. The photo shows the undamaged vegetation upwind of the 6m release pipe and the change in vegetation with the loss of *Calluna vulgaris*, *Sphagnum* spp. and *Cladonia portentosa* up to 24m downwind of the NH_3 release point, which is now dominated by *Eriophorum vaginatum*. *© Ian Leith*

Figure 5.23: Impacts of NH_3 on vegetation at Whim Bog showing death of *Calluna vulgaris,* blackening of *Eriophorum vaginatum* shoots and the loss of vegetation cover (mostly *Sphagnum* and lichen spp.) resulting in the exposure of bare peat. The droppings reflect the presence of rabbits across the site.

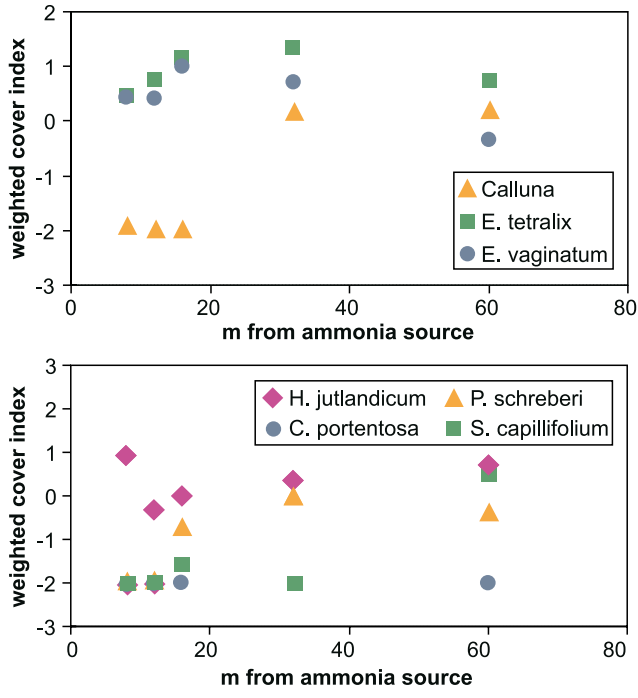

Figure 5.24 Change in the weighted cover index (value based on start value and after five years exposure) for vascular plants growing at different distances from the ammonia source and changes in the non vascular plant index. The exponential decrease in ammonia concentrations along the transect produces NH_3-N doses equivalent to the 56, 24 and 8 kg N $ha^{-1}y^{-1}$ wet deposition treatments at ~ 8-12m, 24m and 32-40m from the line source, depending on the year.

reduced, with -1 and -2 indicating a 67 per cent and 100 per cent cover loss (the latter indicating complete eradication).

If we examine functional groups, among the ericoids *Calluna* has disappeared close to the gaseous ammonia source, whereas *Erica tetralix L.* and *Empetrum nigrum L.* and *Vaccinium myrtillus L.* (data not shown) have increased in the area within 32 m of the ammonia source. Of the graminoids at Whim, *Eriophorum* spp, (Cyperaceae) are the main components. Graminoids are generally considered to be effective at utilising increased levels of N, so unsurprisingly, *E. vaginatum L.* has increased its cover. The cover of non-vascular plants has generally decreased, as expected, but there are big differences in sensitivity: among the pleurocarpous mosses *Hypnum jutlandicum* Holmen & Warncke has generally increased its cover whereas its' cohabiting species *Pleurozium schreberi* (Willd. ex Brid.) Mitt. has declined. Close to the ammonia source, *S. capillifolium*, and *C. portentosa* have completely disappeared.

The addition of wet deposited nitrate or ammonium has produced much more subtle changes in species cover (Figure 5.25). Responses to oxidised N, nitrate tend to be non-linear showing a positive response to N additions, < 24 kg N $ha^{-1}y^{-1}$ (total deposition, including background <32 kg N ha^{-1} yr-1) which continue after five years to alleviate N limitation in the ericoids, *Calluna* and *E. tetralix*. The higher N addition of oxidised nitrogen (56 kg N ha^{-1} yr-1) has reduced cover indicating the capacity of the vegetation to use N has been exceeded. *H. jutlandicum* has shown the most

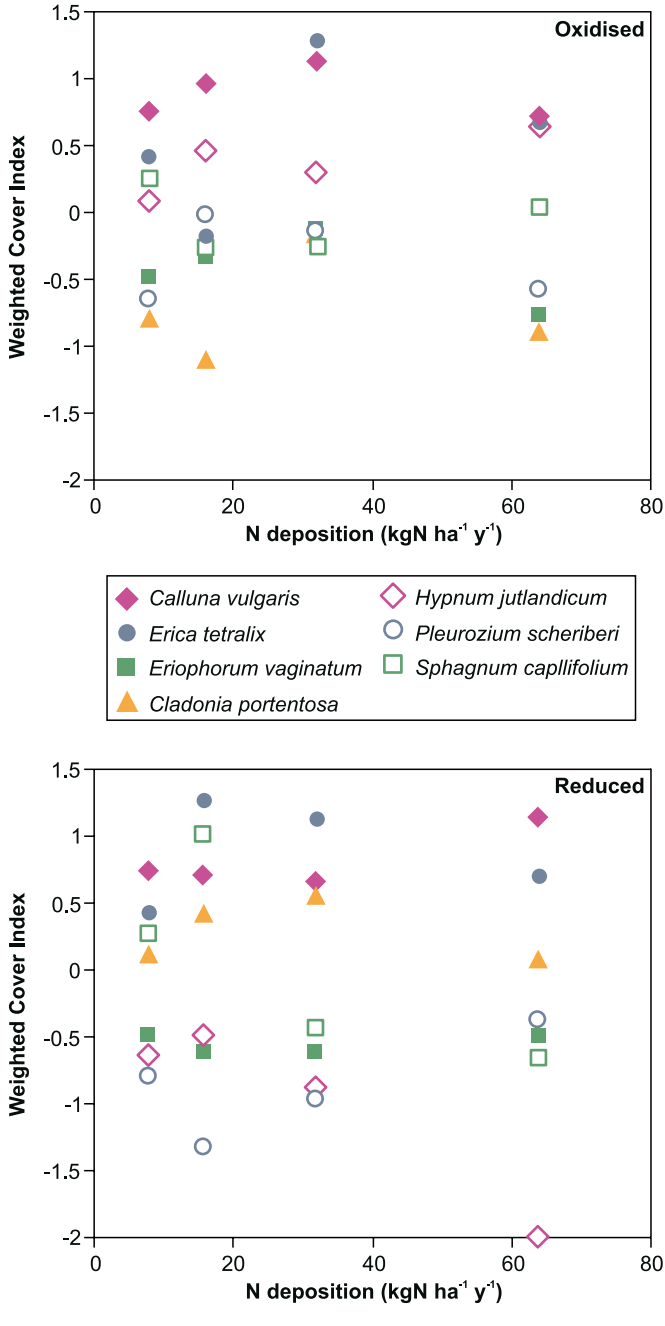

Figure 5.25: Change in the weighted cover index (value based on start value and after five years exposure) for vascular and non vascular plants treated with wet oxidised and wet reduced nitrogen as $NaNO_3$ or NH_4Cl, respectively. The species represented are the most common species growing at the experimental site an ombrotrophic bog, Whim in Southern Scotland. Values above the bold line represent an increase in cover and vice versa. Cover at 8 kg N $ha^{-1}y^{-1}$ represents the control response, receiving no additional N.

consistent positive response to oxidised N, irrespective of dose. Different responses were observed for reduced N (Figure 5.25). Calluna appears to show a preference for wet deposited reduced N over wet deposited oxidised N, which may reflect the low activity and inducibility of nitrate reductase in its foliage (Smirnoff *et al*., 1984), and the increased potential for soil acidification from ammonium deposition. *S. capillifolium* has shown consistent reductions in cover in response to both N forms, though not on the scale caused by equivalent N doses as ammonia. *E. vaginatum* has not responded positively to the wet deposited nitrogen treatments, probably reflecting the increase in *Calluna* cover. In the wet deposition treatments, the cover response appears to be mediated through

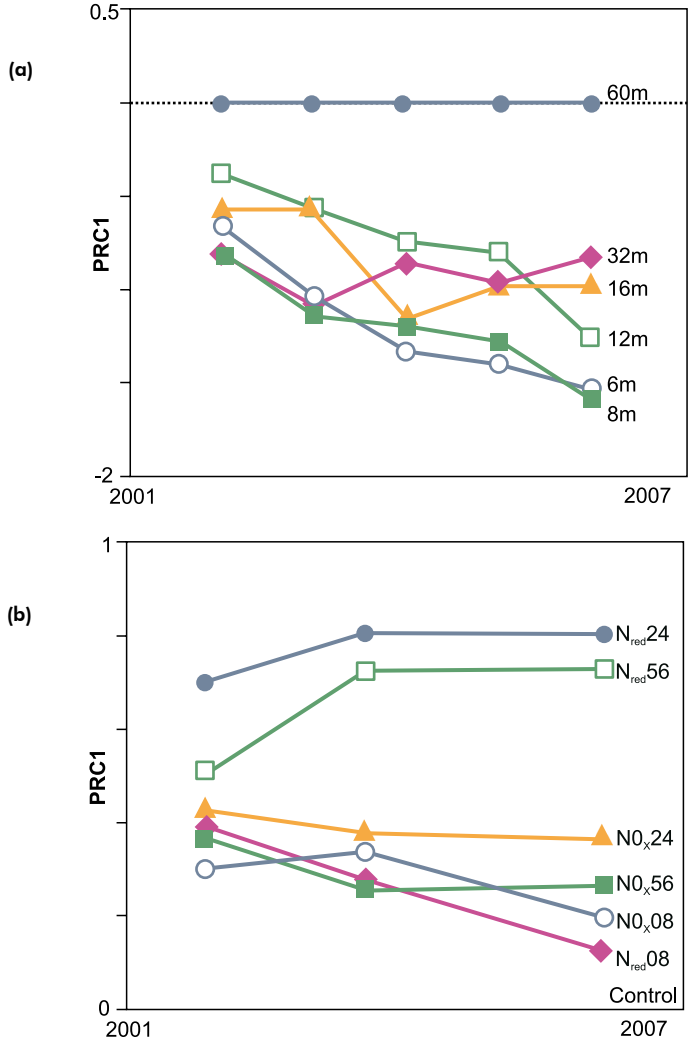

Figure 5.26: Principal Response Curves (PRC) based on redundancy analysis (RDA) of the vegetation cover data from dry and wet N deposition treatments at Whim Moss. (a) PRC for dry N treatment vegetation. (b) PRC for wet N treatment vegetation. The curves represent the temporal trajectory of community composition for each of the experimental treatments

competition rather than phytotoxicity. As expected, the most N-sensitive species are again the non-vascular plants, *C. portentosa and S. capillifolium*.

Anecdotal observations on non-vascular species that are less frequent at the field site suggest that *Cladonia rangiferina* (L.) Weber ex F.H.Wigg. and *C. chloropheae* (Flörke ex Sommerf.) Sprengel are less sensitive than *C. portentosa*. Similarly, among the less frequent *Sphagnum* species present, limited data suggest that *S. fallax* H. Klinggr. and S. *papillosum* Lindb. are much less N-sensitive than *S. capillifolium*.

Recording the cover of vascular and non-vascular plants from prior to treatment to the present has provided a significant insight into the response of some environmentally important plant species to N deposition, which is clearly visible in the changing character of the site (Figs 5.22 and 5.23). Such a detailed approach appears to work well for experiments, informing us of the relative sensitivities of different species, and enabling us to predict which species are most likely to be adversely affected by additional atmospheric N deposition. However, species composition/cover fails as an 'early warning indicator' because, by the time the effects are expressed, the damage has already been done.

National Vegetation Classification (NVC)

The National Vegetation Classification (NVC) is a standardised method for community classification in the UK. Interestingly it has also been used to demonstrate community level effects in relation to hydrological disturbance in fens (e.g. Fojt and Harding, 1995). However, no direct differences between the treatments at Whim were detected in terms of the NVC classification, either at community or sub-community levels. All treatments were most closely classified within M19, with the weighted match (the number of species in the sample that also occur in the community weighted according to species fidelity to a community) remaining above 88 per cent for both the wet and dry N treatments.

The lack of detection of community difference using the NVC may be due to several reasons, but is probably mostly related to scale. Firstly, the NVC describes community at a national level from a conglomerate of UK data, so that even when using 'typical' community data, the fit of any individual site is rarely exact. Secondly, the detection of change in the Whim experiment is at a local level from one community type, where differences are likely to be small in relation to differences between distinct communities. Finally, plot replication in expensive field manipulation studies on semi-natural communities can rarely match the scale and level of replication that can be achieved in vegetation surveys. At Whim, logistical reasons dictated the layout of treatment plots which meant that the initial species composition was not precisely replicated in each plot or quadrat. For many of the non- vascular plants occurrence was often limited to a few plots or a few treatments. In the case of species only occasionally occurring at the site, this reduces the quantitative value of individual species observations to anecdotal evidence. Rodwell (1991) suggests that 2 or 4 m^2 plots are appropriate for assessing mire NVC communities; thus NVC matching is probably inappropriate for the smaller experimental quadrats (total area of three per plot = 0.75 m^2) used here.

Principal Response Curves

Principal Response Curves analysis (PRC) is a means of analysing repeated measurement designs and testing and displaying optimal treatment effects that change across time (see van Brink and Ter Braak, 1999). It is based on a redundancy analysis (RDA) that is adjusted for changes across time in the control treatment. Thus the treatment effects are expressed as deviations from the control and PRC gives simple representation of how treatment effects develop over time at the community assemblage level. Interpretation of species response can be assessed with reference to the additional species score for the first RDA axis (shown to the right of Figures 5.26a and 5.26b).

Species with higher values are generally those that are increasing in abundance, those with negative values are decreasing with time, and species with near zero values show no response to the treatments. The significance of the PRC can also be tested using Monte Carlo permutation tests where the significance of the first RDA axis is tested. The dry manipulation was not tested for significance due to insufficient replication, but the first axis of the wet treatment RDA was significant ($p < 0.01$) showing apparent community assemblage change over time in response to N addition (Figures 5.24a,b); i.e. there are clear developments in the deviations from control assemblages over time. All the assemblages are seen to respond in the same direction for the dry treatment, although those closer to the ammonia source are the furthest from the control response. The response of the communities in relation to the wet treatments is dependent on the form of N, reduced or oxidised, and dose. It can also be seen that the species responses are similar to those discussed above and illustrated in Figure 5.25.

5.11.3 Conclusions

This paper has exposed a range of issues facing the science community interested in evaluating N effects on semi-natural systems, particularly those in which non-vascular species are pivotal to the delivery of key ecosystem services, e.g., the sustainability of our peatlands and their ability to sequester and store carbon. In particular it is recognized that:

- All species are not equal with regard to ecosystem function. In peatlands, indicators of N deposition in ombrotrophic ecosystems must account for non- vascular plants.
- Species richness and/or diversity suffer from the loss of species identity and thus attach no value to the role of specific species within the ecosystem, e.g. *Sphagnum* the ecosystem's engineers thus failing to convey the damaging effects of N impacts. They may also be inappropriate measures in species poor ecosystems.
- Ellenberg indicators are not direct N indicators and in peatlands can be insufficiently sensitive to pick up real change, especially in ecosystems which are generally less favourable for vascular plant growth, i.e. are acid and often waterlogged.
- Assessment of community-level N effects can be aided with the use of multivariate techniques such as PRC, which have the ability to assess both community assemblage change and species level information.
- Spatial scale is important when using national classification systems, such as the NVC; in the present study the NVC was not an appropriate tool, as significant species changes occurred, without altering the NVC category.
- Basing assessments of N deposition effects on species composition has the benefit of revealing the important consequences, i.e. it provides evidence to support N pollutant emission regulation.
- Until we can identify the plant/ecosystem equivalent of the 'canary', for conservation purposes, we need to focus the search for 'early warning indicators', on chemical indicators either within the plants themselves or in the soil.
- Research must be directed at identifying threshold values for plant chemical or physiological traits that respond to reactive N, indicating the likelihood of detrimental effects leading to loss of cover / species of conservation value (Sutton *et al*., 2004).

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6

APPROACHES TO MODELLING LOCAL NITROGEN DEPOSITION AND CONCENTRATIONS IN THE CONTEXT OF NATURA 2000 (THEME 4)

6.1 Background document

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Summary

- Transport-chemistry models are useful and well-used tools in the environmental assessment of impacts of atmospheric N on Natura 2000 sites.
- Impacts arise from two groups of reactive nitrogen compounds: reduced nitrogen compounds (NH_3 and its reaction product NH_4^+) and oxidised nitrogen compounds (NO, NO_2 and their reaction products). Often the contributions from these two groups are of the same magnitude, but close to intensive livestock farms, the contribution from NH_3 may dominate.
- Oxidised nitrogen compounds contribute to the long-range transport component of nitrogen deposition, but may for most practical purposes be disregarded in local scale assessments. Local scale assessments, therefore, concentrate on the assessment of NH_3 deposition from local agricultural sources.

6.1.1 Introduction

The impact of atmospheric nitrogen (N) on nature areas is both a local scale and a transboundary problem. These impacts can be the result of both deposition (wet and dry) and elevated ambient concentrations of reactive nitrogen (N_r) compounds. In some regions the contribution from (mainly) wet deposition of long-range transported aerosol-bound N_r compounds exceeds the critical loads of sensitive ecosystems. Close to intense agricultural areas the contribution from dry deposition of locally emitted ammonia (NH_3) may, however, equal or even exceed the nitrogen contribution from long-range transport.

Two types of assessment studies are carried out with respect to evaluating the impacts on Natura 2000 areas; the assessment of critical load and critical level exceedances, respectively. The model approaches for the two types of studies are similar, although the critical levels approach focuses on ammonia concentrations, whereas the critical load approach focuses on deposition (arising from dry and wet deposition of both reduced and oxidized N_r compounds in gas and particle phase). The assessment of atmospheric N_r deposition, therefore, needs to account for both local and long-range transport contributions to the nitrogen input to the nature area in question (Hertel *et al*., 2006a), whereas critical levels may be limited to considering only contributions from local sources of NH_3 or in some situations elevated nitrogen dioxide concentrations from local sources such as major roads (e.g. Cape *et al.,* 2004).

It should be noted that Natura 2000 areas vary considerably in size throughout Europe – with cross-sectons of a few hundreds of meters to many kilometres. Naturally, this has strong implications for which tools are applicable in the assessment of impacts of atmospheric N. An overview of the chemistry-transport modelling of NH_3 on local and regional scale has recently been presented by (Loubet *et al*., 2008) and (Van Pul *et al*., 2009). The focus in the current chapter is on describing the model approaches used in the assessment of local atmospheric N_r deposition. We limit our selves to describing the contribution from local sources within a distance of 20 km of a Natura 200 site.

Processes on the local scale

In the following section we will look at the physical and chemical processes that play a major role in the impacts of N on Natura 2000 areas, and therefore need to be included in assessment studies at the local scale (<20 km).

Transport and chemical transformation

Buildings and other obstacles may have significant influence on the initial downwind dispersion of NH_3 emitted by livestock houses. The landscape may also impose certain flow conditions that have to be accounted for and nearby vegetation may influence the generation of turbulence, thereby affecting the rate of dispersion.

Only fast chemical transformations can take place on a time scale of minutes to a few hours. Concerning N_r compounds, fast reactions of this kind applies to the conversion of NO to NO_2 through the reaction with ozone and the photo dissociation of NO_2 reforming NO. Further conversion of atmospheric NO_2 to HNO_3 through the reaction with hydroxyl radicals (OH) has a rate in the atmosphere of about 5 per cent per hour, which means that this reaction is negligible on the time scale of local dispersion. NH_3 reacts with acid gases and aerosols in the atmosphere to form aerosol-phase ammonium (NH_4^+); a reaction that used to be relatively fast. In the 1980s and 1990s the average conversion rates in Europe were estimated to be in the order of 30 per cent per hour, but due to the large reductions in European sulphur emissions leading to significant decrease in sulphate concentrations, the conversion rate is now believed to be of the order of 5 per cent per hour (Van Jaarsveld, 2004). This means that NH_3 now has an atmospheric lifetime of the order of a day, where it used to be of the order of 4 to 6 h. Thus, NH_3 to NH_4^+ conversion may, for most practical purposes, also be disregarded in local-scale modelling.

Wet and dry deposition

Precipitation events take place during a very short period of time. The residence time of local pollutants within the 20 km distance from the source is on the order of minutes to a couple of hours. This means that within this domain wet deposition of locally emitted pollution is generally negligible and it may for most purposes be disregarded in local scale modelling of nitrogen deposition (Hertel *et al*., 2006a). The contribution to the background deposition is, on the other hand, very significant.

It is common to assume a zero concentration at the ground in the computation of the dry deposition flux. This may, however, not apply over vegetation where the flux of NH_3 and NO_2 may be bi-directional (Schjørring *et al*., 1998a; Schjørring *et al*., 1998b). This bi-directional flux depends on the concentration in the air as well as in the plants, which changes during the season and is also dependent on management such as grass cutting or use of fertilizer. An important pathway for dry deposition of NH_3 is the uptake through stomata of plants, but it may also be absorbed through dew on the plants or through the thin water film on the leaf epidermis (Nemitz *et al*., 2004). Experimental studies have shown that co-deposition of NH_3 and SO_2 takes place, and that the ratio between atmospheric concentrations of these two compounds together with the humidity and temperature are important factors for the deposition of NH_3 (Neirynck *et al*., 2005). NO_2 may also be taken up by plants through the stomata. Although this deposition is not very fast (to a forest the dry deposition velocity is on the order of 2 to 4 mm/s), it may in some areas be of importance for overall N_r deposition (Wesely and Hicks, 2000). HNO_3 sticks to almost any surface and is therefore, quickly deposited. In addition, the dry deposition of peroxyacetyl nitrate (PAN), N_2O_5, and HONO may be of some importance for overall N_r deposition (Wesely and Hicks, 2000). As previously discussed, the formation of these NO_y compounds is too slow for local emissions to play a role in the atmospheric deposition of N_r.

Current state-of-the-art in modelling

Above we have discussed the processes that need to be accounted for in modelling of local-scale nitrogen deposition. Table 6.1 summarises the state of the art in process descriptions of currently applied models. For comparison modelling of long-range transport is also included in the table.

Table 6.1: State-of-the-art in LRT & local-scale modelling of N_r deposition (Hertel *et al*., 2006a).

Process	Long-range transport (LRT)	Local scale
Emissions	European 17 km x 17 km inventory. For parts of the domain detailed seasonal variation is included.	Single farm level with dynamic seasonal variation divided into contributions from livestock housing and manure application
Transport	Eulerian 2-way nested grid models with 17 km x 17 km or even 5 km x 5 km resolution in the inner nest.	In process studies, CFD models provide detailed information, e.g. flow around building obstacles.
Aerosol description	Dynamic aerosol-phase chemistry.	1st order transformation between gas phase NH_3 & aerosol phase NH_4^+.
Gas phase chemistry	Explicit chemical mechanisms including the most important species.	First-order transformations.
Wet deposition	Full wet phase chemistry in cloud and rain droplets and subsequent scavenging of rain droplets.	Not included.
Dry deposition	Resistance method incl. detailed seasonal variation & actual meteorology. Bi-directional flux parameterisation for NH_3.	Resistance method incl. detailed seasonal variation & actual meteorology. Bi-directional flux parameterisation for NH_3.

6.1.2 Methodologies and tools

The following section outlines the main features of model tools and systems applied in the literature. A review of literature reveals assessment studies from only a few European countries including United Kingdom, the Netherlands, France, Poland and Denmark. Concerning approaches in the calculations of atmospheric N_r impacts to nature areas, one can distinguish between the methods and tools that have been developed for:

- Screening purposes – to provide a crude and quick assessment of the potential environmental impact of atmospheric N_r impacts;
- Routine assessments e.g. by local authorities using methods based on nomograms and tables generated from model calculations of source-receptor relationships;
- High resolution assessments using all available information and applying detailed modelling systems.

Screening methods are based on simple parameters like size of farm and distance between sources and nature areas, and it provides crude, worst-case deposition estimates. Nomograms and tables are also somewhat crude tools for estimating the loads but, in compliance with the local legislation, in some countries they are applied by local authorities and consultants to assess impacts from single farms; this is the case e.g. for Denmark. Focus in this paper is, however, mainly on the more detailed modelling systems such as those listed in Table 6.2.

Table 6.2 contains references to three types of models applied in assessment of atmospheric nitrogen deposition:

- The Gaussian plume models are used in local scale modelling (limited to a distance of about 20km from the source) and assume a Gaussian distribution of the pollution concentration in both the vertical and horizontal direction. The simplest versions apply source depletion for describing dry deposition; whereas surface depletion provides a more correction description. First order chemistry is often applied.
- The Lagrangian type models may be applied for describing local to long-range transport. The model concept is based on following a single air parcel at a time along a transport route (a so-called trajectory). The difficulty of this type of model is the description of mixing of air masses and the uncertainties in determining the transport routes. Full chemistry and dry and wet deposition processes are usually implemented.
- Eulerian type models are based on fixed sets of grid cells and describing exchange of pollutants between these cells. In order to describe concentrations and depositions well a high resolution is necessary, but this costs substantial computer time. Horizontal advection may generally be well resolved whereas vertical mixing is less well described. Just as in the case of the Lagrangian models, full chemistry as well as dry and wet deposition processes are usually implemented.

The local legislation may be a powerful tool in reducing local atmospheric nitrogen loading of nature areas. These regulations may include use of buffer zone around sensitive nature areas, restrictions on number of animal units per farm etc. Trees have been discussed as a tool in landscape planning to reduce the impact of atmospheric NH_3 deposition, and local scale models have been used to evaluate the impact (Sutton *et al*., this volume). The efficiency of measures on a local scale depend largely on how much nitrogen is coming from local sources and how much from sources further away (often called background deposition). In the Netherlands, for instance, in general the background deposition to nature areas is dominant over the contribution by local sources. This makes the efficiency of local measures limited (van Pul *et al*, 2004).

Table 6.2: Models applied in local scale modelling of nitrogen deposition.

Name	Type of model	Process description	Refs
AERMOD	Gaussian plume model.	Uses hourly meteorological data to derive stability & dispersion parameters. Ground & elevated sources in simple & complex terrain. Dry deposition using source depletion & applying resistance algorithm. Wet deposition. using simple wash out ratios for gases & particulate matter (PM).	(US-EPA, 2003)
ADMS	Gaussian plume model.	Uses hourly meteorological data to derive stability & dispersion parameters. Ground and elevated sources in simple & complex terrain. Dry deposition using source depletion & resistance algorithm. Wet deposition using simple wash out ratios for gases & PM. Dry & wet deposition algorithms recently updated.	(Carruthers *et al.*, 1994)
FRAME	Lagrangian transport model.	Horizontal resolution of 5 x 5 km & 33 vertical layers. Boundary conditions from FRAME-Europe on 150 km x 150 km. Calibrated to interpolated measured deposition on cell basis over three most recent years. Dry deposition applying resistance method & 5 land use classes. Wet deposition applying scavenging ratios from EMEP model & a directional orographic model for enhanced precipitation generated by terrain.	(Singles et al., 1998)
CMAQ	Eulerian chemistry-transport model	Uses MM5 meteorological data, but other data may be applied. Carbon-Bond V (CB05) chemistry mechanisms incl. aerosol module from RPM. A variable resolution multi-layer model incl. sub-grid scale treatment of large point source plumes (PinG). Hourly dry deposition using resistance methods from RADM incl. seasonal variation & 11 land use types. Wet deposition using scavenging coefficients.	(Binkowski and Roselle, 2003)
LADD	Lagrangian multi-layer model	Uses the trajectory approach from FRAME. Horizontal resolution 50m x 50m & very high vertical resolution (lowest points defined at 0.25m; 0.5m; 1.0m & 2.0m above ground level. Computes dry deposition. of NH_3 applying resistance method. Wet deposition. not accounted for.	(Dragosits *et al.*, 2002; Sutton *et al.*, 1998)
DEHM	Eulerian chemistry-transport model	Uses MM5 meteorological data, but other data may be applied. Explicit chemistry mechanisms with 80 species & 200 reactions. Covers Northern Hemisphere on 150 x 150 km, nested grid 50 x 50 km for Europe, & 5 x 5 km for Denmark & close vicinity. Hourly dry deposition applying resistance method from EMEP model. Wet deposition applying in-cloud & below-cloud scavenging coefficients.	(Christensen, 1997; Frohn *et al.*, 2002a; Frohn *et al.*, 2002b)
OPS	Gaussian plume model on local scale & Lagrangian transport further away from source.	Uses statistical climatology derived from hourly meteorological data. Computations performed for climatological classes & hourly concentration & deposition derived from frequency. Chemistry using 1st order conversion. Dry deposition based on resistance method. Wet deposition as wash out & rain out using coefficients; rain out after plume penetrates cloud; thus at some distance from source.	(Colles *et al.*, 2004; Pul *et al.*, 2004; Van Jaarsveld, 2004)
OML-DEPOSITION	Gaussian plume model.	Uses hourly meteorological data to derive stability & dispersion parameters. Handles point & area sources in simple terrain applying variable horizontal resolution. (usually 400 x 400 m or 100 x 100 m). Dry deposition using surface depletion & resistance algorithm. No wet deposition or bi-directional flux.	(Hertel *et al.*, 2006b; Olesen *et al.*, 1992; Sommer *et al.*, 2009)

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6.2 Working group report

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6.2.1 Conclusions and recommendations of group discussions

- It was noted that modelling assessment approaches differ widely from country to country, both in terms of the type of models used and the level of detail considered. In particular, two types of assessment can be used (source-based or receptor-based) and the workshop recommended the type used should be clearly specified in all assessments.
- The workshop concluded that the uncertainty in concentration predictions by models is much smaller than the uncertainty in the deposition predictions. This has the practical implication that, from the perspective of the atmospheric modelling, assessments based on air concentrations will have less uncertainty than those based on atmospheric deposition.
- The workshop noted that the emissions from fertiliser (including both inorganic mineral fertilizers and organic manures) when applied to land is not usually modelled in current assessments. This is a major gap in current practice, given the substantial contribution to nitrogen deposition at many Natura 2000 sites from the nearby land application of fertilizers to agricultural land.
- The workshop concluded that estimation of dry deposition of nitrogen compounds remains highly uncertain. In particular, uncertainty analysis for dry deposition is needed but remains a difficult task.
- The workshop recommended that validation datasets for both concentration and deposition need to be developed and compiled in a form that can be made readily available for the purpose of model verification.
- The workshop recommended that further development and testing of nitrogen dry deposition parameterisations are needed as a means to reduce uncertainties in assessing total nitrogen inputs to Natura 2000 sites. In particular, further assessment of ammonia canopy compensation points is needed for different habitat types. Overall, much more field deposition data are needed for model verification.
- The workshop recommended that the emissions of ammonia to the atmosphere following fertiliser application (including both organic manures and mineral fertilizer) should be included in future environmental assessments of the impact of current and future activities on Natura 2000 sites.
- It was recommended that a harmonised approach to uncertainty analysis for the models needs to be developed to aid the regulatory assessment of nitrogen emission, dispersion and deposition to sensitive habitats.

6.2.2 Introduction to structure of discussions

Prior to the meeting, the members of the group agreed on a list of key questions that were to be discussed during the meeting. The key questions were:

- How comparable are modelling approaches of different European countries?
- What are the current uncertainties in the modelled estimates of concentrations and deposition?
- What model developments or data requirements are needed to reduce these uncertainties?
- How can model uncertainties be taken into account?
- How should model outputs be used/presented in impact assessments?

It was also concluded that there are many different models of varying complexity to predict concentrations and deposition rates and that the most suitable type of model should be chosen to carry out a specific task.

6.2.3 Key Question 1: How comparable are modelling approaches of different European countries?

The presentations highlighted the similarities and differences of the approaches used in the Netherlands and Denmark. In addition to these two countries, information was provided from the UK representatives on the assessment approach used there. Appendix 6.1 gives a more detailed overview of the process in these three countries.

An interesting point was made during the subsequent discussions, which was that assessments can be approached in two ways, depending on their objectives. One approach is to consider the impact of all relevant sources on one specified natural area (receptor-based) and the other is to consider the impact of a specified source on all relevant natural areas (source-based). The modelling methods may be different for the two different approaches. The following discussion focussed on the source-based approach only.

Although information was only available for three European Member States, there are considerable differences between the approaches. In the UK, different distances (2-10 km) are used to decide whether a source is likely to have an impact (screening distance) depending on the classification of the nature site. In Denmark, a screening distance of one km is used in all cases, whilst in the Netherlands screening distances are not applied. Different dispersion models are used in the different Member States. In the UK there is no prescribed regulatory model and assessments are usually carried out using either the ADMS or AERMOD models. In Denmark and the Netherlands national regulatory models (OML-Dep and AAgro-Stacks respectively) are used. The use of different models in different Member States potentially could lead to inconsistencies across Europe. Results were presented from an intercomparison of short-range dispersion models (LADD, ADMS 4, AERMOD and OPS-st), using four hypothetical emission scenarios (of different source types), that showed that, at least for concentrations, the predictions of the different regulatory models are similar (Theobald *et al*., 2010). The discussions also highlighted that different types of meteorological data are used for the assessments. For example, in the UK, five years of continuous data are normally used, whereas in Denmark, the tables based on OML-Dep model output were developed using data from 2005 and in the Netherlands hourly meteorological data from the year in question are used. One surprising difference between the approaches is that assessments in Denmark do not use ammonia concentration predictions or take into account background nitrogen deposition. The assessment only calculates the farm's additional deposition to a natural area. One surprising similarity between the different approaches is that none of the Member States model the ammonia concentrations or nitrogen deposition resulting from the application of organic or

mineral fertilisers. This is because it is difficult to get information on the exact timing and location of the applications. However, since these emissions may make a significant contribution to Member States' ammonia emissions, the group agreed that these emissions should be taken into account in assessments. The development of protocol to calculate these emissions should be a priority for future discussions.

6.2.4 Key Question 2: What are the current uncertainties in the modelled estimates of concentrations and deposition?

Uncertainty of deposition predictions is significantly larger than that of concentration predictions. This is because: a) deposition rates are usually calculated from modelled concentrations; b) deposition processes are less well understood/more complex than atmospheric dispersion processes, c) consequently deposition is modelled/parameterized in very different ways and d) deposition predictions are more difficult to validate than concentration predictions. The model validation data presented showed that ammonia concentrations predicted by regulatory models agree fairly well with measured concentrations. Although detailed uncertainty analyses have not been carried out for the models, the validation studies provide an estimate of average prediction uncertainty of approximately ± 20 per cent for concentrations (e.g., absolute fractional bias < 0.2), when using measured emission data and on-site meteorological data. This is of a similar order of magnitude to the uncertainty in national emission rates of ammonia in the UK (Misselbrook *et al*., 2000). Model uncertainty can be significantly greater than this, however, if emission factors are used instead of measured emissions or on-site meteorological data are not used, which is often the case for assessments. Specifying uncertainty estimates for nitrogen deposition is much more difficult due to lack of validation studies. For the Danish assessment approach an uncertainty estimate of +/- 35-70% was reported. Based on expert judgement the group agreed that uncertainties in deposition estimates of regulatory models lay in the range of ± 50-100 per cent. This uncertainty can be reduced if more detailed information on deposition characteristics (surface roughness, vegetation type) of the area is known. At present it is not possible to provide a more accurate estimate of uncertainty and the group concluded that more model uncertainty analyses are needed. Moreover a common approach in uncertainty assessment would be useful for model comparison across Member States.

6.2.5 Key Question 3: What model developments or data requirements are needed to reduce these uncertainties?

Since the uncertainty in modelled annual concentration predictions is of a similar order of magnitude as the emission data, model improvements are unlikely to reduce the prediction uncertainty significantly. If there is a need to model predictions at a higher temporal resolution (e.g. daily or hourly) then the temporal resolution of emission data will need to be increased where necessary. With regards to the uncertainty in deposition predictions there is still a lot that can be done. For some ecosystems the bi-directional exchange of ammonia with the atmosphere has a large influence on the net deposition rate and on ammonia concentrations further downwind and therefore models should include a canopy compensation point to model this. In order to validate the dry deposition parameterisation of models, it is necessary to have measurements of dry deposition rates. Very little data are currently available for this purpose and that that is available has been used in the formulation/calibration of model routines. Therefore, more measurements of dry deposition over a range of ecosystems are necessary to validate the models.

6.2.6 Key question 4: How can model uncertainties be taken into account?

When using model predictions for regulatory assessments, the values used must be accompanied with an estimate of model uncertainty, even if (as the previous discussion highlighted) this estimate is only based on expert judgement. An example would be an ammonia concentration prediction of

4 µg m^{-3} with an uncertainty estimate of +/- 20 per cent. This could be presented in an assessment as either:

4 ± 0.8 µg m^{-3} or

3.2 - 4.8 µg m^{-3}

The option to use is at the discretion of the assessor. If the uncertainty assessment is based on a published study then the reference should also be given.

6.2.7 Key Question 5: How should model outputs be used/presented in impact assessments?

Ideally (receptor-based) regulatory assessments should provide an estimate of concentration and deposition predictions for each source (within a screening distance) at locations of interest within the natural area. For each location the assessment should provide the following information (with uncertainties):

- Concentration contribution from each local source (NH_3 and NO_x)
- Dry deposition of reduced and oxidised nitrogen from each local source
- Background wet and dry deposition (split into reduced and oxidised)

Background data should come from national modelling/monitoring activities.

Provision of these data will enable an estimate of the concentrations (NH_3 and NO_x) and nitrogen deposition rates at the locations of interest, as well as the contribution from each source within the screening distance. These estimates can then be compared with critical levels and loads to assess impacts on the natural area.

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Appendix 6.1: Examples of the modelling assessment process in different countries

Denmark - L. M. Frohn and O. Hertel, National Environmental Research Institute, Aarhus University, Denmark

The Danish regulatory system is required by authorities to be as simple, transparent and robust as possible in order to secure a uniform scientific base for decision making within the municipality administration. National demands for approval of new husbandry activities (or expansion of existing activities) depend on the distance to nature areas included in the Danish VMPIII (Action Plan for the Aquatic Environment) agreement (like e.g. Natura 2000 areas). If any point source related to an application is located within 1000 meters (buffer zone 2) from a VMPIII area, then the additional nitrogen deposition to the nature area arising from the expansion applied for, must be estimated. If any point source is located within 300 meters (buffer zone 1) from a VMPIII area, the application is rejected. Within buffer zone II, the cut-off criteria for the additional nitrogen deposition are differentiated depending on the number of other farms with more than 75 animal units, located within a distance of 1000 m from the farm house of the applicant in order to account for the accumulated effect. The differentiation is set to:

- 0.3 kg N pr. hectar per anno (more than two farms)
- 0.5 kg N pr. hectar per anno (two farms)
- 0.7 kg N pr. hectar per anno (one or no farms)

The emissions of ammonia from application of manure, fertilizer as well as releases from crops are not taken into account in the system, since the calculation only deals with the ammonia loss from stables and storages in the husbandry production.

The atmospheric ammonia deposition is based on a set of "standard tables" corresponding to different combinations of intermediate land surface types (between the farm and the nature area) and nature area surface types. The standard tables give the atmospheric ammonia deposition with distance from the source of one kg emitted ammonia. The deposition is then scaled according to the actual emission and corrected for local meteorological conditions.

Based on the information supplied by the applicant, the deposition to a nature area is calculated as:

$$A(L) = E \times D(L) \times \frac{WF}{100} \times WK\ (WM)$$

Where A(L) is the annual deposition (kg N ha^{-1} $year^{-1}$), E is the emission from the point source (kg N $year^{-1}$), L is the distance from the source to the nature area, D is the standard deposition for the relevant combination of intermediate type of land surface and land surface of the nature area, WF is the wind frequency in the relevant wind sector, WM is the mean wind speed in the relevant wind sector and WK is the wind correction factor depending on WM.

Denmark has been divided into nine climatological regions based on data for wind speed and wind frequency provided by the Danish Meteorological Institute (Cappelen and Jørgensen, 2008). A wind correction is applied using tables generated from meteorological data representative for each specific region. The calculations may be performed applying roughness data for three different surface types that the applicant selects on basis of a guidance booklet with aerial photos.

The atmospheric N deposition to Danish land and sea surfaces are mapped within the national monitoring programme for water and nature (NOVANA). The background atmospheric N

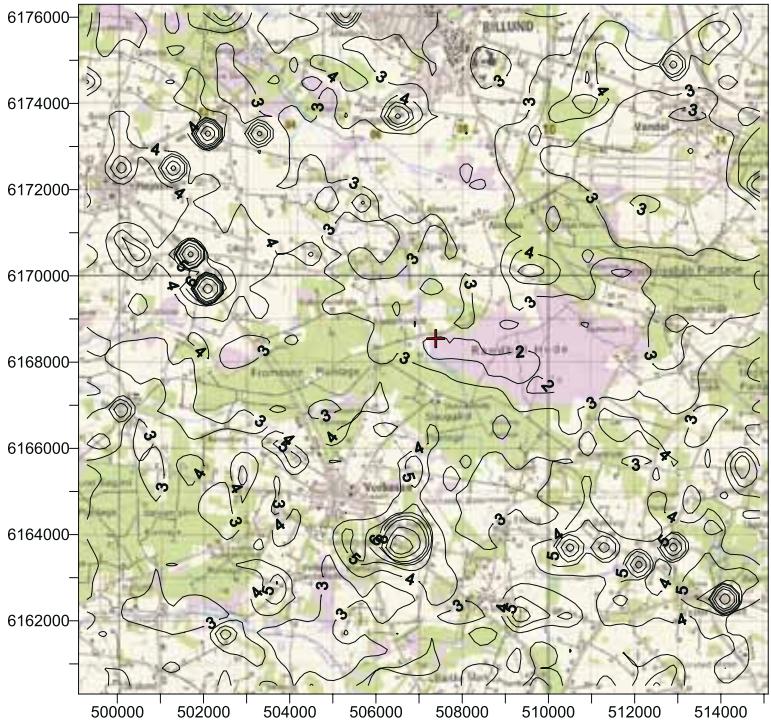

Figure 6.1: Modelled deposition (kg N ha^{-1} yr^{-1}) to a heath area in the western part of Denmark (Randbøl Heath) for the year 2006. Heathland and woodland are depicted in pink and green respectively. The modelled ammonia deposition is locally high around sources, e.g. farms (Ellerman *et al.*, 2007).

deposition at a resolution of 17 x 17 km is computed using the Danish Eulerian Hemispheric Model (DEHM), and for selected nature areas detailed mapping of NH3 deposition fluxes at a resolution of 400 x 400 m are calculated using the OML-DEP model (Ellermann *et al*., 2007). The resolution in the DEHM computations is soon to be increased to 5 x 5 km. The combination of DEHM and OML-DEP is termed DAMOS – the Danish Ammonia Modelling System (Hertel *et al.*, 2006a; Hertel *et al*., 2006b) and this system is applied in the NOVANA program and also in a number of assessment studies carried out for selected regions in Denmark.

A farm may include more than one point source, and information for all these are entered into the system, which then calculates the ammonia deposition from all the sources individually and in total. The point within the nature area with the largest total deposition, will be the starting point for the municipality administration, i.e. that is the point where the maximum atmospheric nitrogen deposition criteria must be complied with.

The model tool which has been used in the development of the regulatory system is the OML-Dep model (Olesen, 1995; Olesen *et al*., 1992), which is a Gaussian plume model with a setup that allows for multiple point and area sources of ammonia. Furthermore the model has been extended with a deposition module based on the dry deposition routine of the EMEP model (Simpson *et al*., 2003; Emberson *et al*., 2000). OML-Dep calculates dispersion and deposition of ammonia from local sources within the model domain.

For the generation of the standard tables, the model has been setup with a domain of 4 km x 4 km, a resolution between receptor points of 100 m x 100 m and a single point source located in the left part of the domain. The meteorological data are obtained from the MM5 model for the year 2005, but the wind direction has been fixed at 270º, corresponding to a wind from west. All other meteorological parameters vary in response to changes in atmospheric stability, radiation input etc.

The OML-Dep model is also used in the national monitoring of nature areas. In this context the model setup includes a domain which typically extends to around 16 km x 16 km with a 400 m by 400 m resolution. All ammonia point and area sources are included in the calculation and background concentrations of ammonia are obtained from calculations with the Danish Eulerian Hemispheric Model (DEHM) which covers the northern hemisphere (Christensen (1997), Frohn *et al*., 2001; Frohn *et al*., 2002). An example of calculated deposition of ammonia for a Natura 2000 area (Randbøl Heath) for the year 2006 is shown in Figure 6.1.

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Netherlands - M. van Zanten and A. van Pul (National Institute for Public Health and the Environment, Netherlands) and E. Noordijk (Netherlands Environmental Assessment Agency)

A formal regulatory method to assess nitrogen depositions to Natura 2000 areas is not yet established. However, such assessments will be based on model calculations. To estimate the total nitrogen deposition from multiple sources on one receptor point or on a receptor grid, the Operational Priority Substances (OPS) model is applied (van Jaarsveld, 2004). Currently, calculations are done on a 5 by 5 km grid (data available at http://www.mnp.nl/nl/themasites/gcn/kaarten/index.html) but in the near future this will be changed to 1 by 1 km. The OPS model is also applied for local scale contributions in a variety of situations. For regulatory purposes, however, the model AagroStacks is used to assess contributions from local sources.

The assessments may also include measurements to support the calculations. These measurements are available from the Measuring Ammonia in Nature (MAN) monitoring network. Started in 2005, this network currently includes more than four years of monthly mean ammonia air concentration data from 121 monitoring sites in 30 Natura 2000 areas. These data may allow local or regional adjustments of calculated concentrations/deposition rates.

Characteristics of the OPS-model

- Annual emission data from various sources (animal housing, manure application, foreign, traffic, etc) is used. Emission data are available on a 500 m by 500 m resolution from the Dutch Emission Registration (ER). Emissions from all other European countries are also included, although with decreasing spatial detail on with increasing distance from the Netherlands.

- The meteorological input consists of hourly measurements and is taken from stations of the Dutch meteorological institute (KNMI). On the basis of the average wind regime The Netherlands has been divided into six meteorological regions. For every location within such a region the same dataset, derived from several stations within the region, is used.

- Roughness length data are included. This is available from maps with a 25 by 25 m resolution for the whole of the Netherlands.

- Within OPS, nine land use classes are available. Information on the actual land use class to be taken is available from maps with a 250 by 250 m resolution for the whole of the Netherlands. When calculations are done for gridded receptor points the most dominant land use class in the grid cell is applied.

- Uncertainty in concentration and deposition flux calculations for a single receptor point is high (25 per cent and 100 per cent respectively). The largest uncertainty arises from the dry deposition estimates. For larger areas, this uncertainty is however considerably lower. This uncertainty is also largely reduced (typically halved) if more detailed information on deposition characteristics (roughness, vegetation type) of the area is known.

- The dry deposition module (DEPAC) is currently being updated to include a compensation point and an updated external resistance parameterisation. The latter is derived from ammonia dry deposition flux estimates above grassland. (Wichink Kruit *et al*., 2007).

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United Kingdom – M. Sharp, Scottish Agricultural College, Edinburgh, and D. Bruce, Northern Ireland Environment Agency, Belfast

Modelling assessments for farms in the UK are generally undertaken as part of the IPPC permitting process or as part of the planning process for the purposes of the Environmental Impact Assessment Directive (85/337/EEC). The regulatory process for undertaking assessments for IPPC permitting is more clearly set out than assessment processes used in EIA work.

Typical stages in the UK approach for IPPC permitted farms.

- Using National GIS databases, European sites (SAC, SPA and RAMSAR) within a 10 km radius of the farm are identified. Sites of Special Scientific Interest (SSSI – Areas of Special Scientific Interest (ASSI) in N Ireland) within a 5 km radius and other wildlife sites such as ancient woodland within a 2 km radius are also identified.

- Screening tools are then used to speed up the process and provide greater consistency to the permitting process. The tools used vary in different regions of the UK. Regulators in England and Wales and in Northern Ireland use a screening tool that uses generic emission factors and data provided in the IPPC application process applied to a generalised concentration/distance curve. Results are provided in the form of look up tables. In Scotland the SCAIL model (Theobald *et al.*, 2009) is used for screening purposes.

- In cases where screening tools indicate that a farm has the potential to impact on a sensitive site then further detailed modelling is carried out. This is undertaken using advanced 'new generation' dispersion models such as the UK ADMS 4.1 and the American AERMOD. ADMS 4.1 is typical in that it uses the boundary layer height $\boldsymbol{h}$ and the Monin-Obukhov length L_{MO} to describe the atmospheric boundary layer and using a skewed Gaussian concentration distribution calculates dispersion under convective conditions. The model is applicable up to 60 km downwind of the source and provides useful information at distances of up to 100 km. Distances of interest for farms are typically 100 – 5000 m.

- In order to set permit conditions regulators have undertaken modelling as part of the permit assessment process using standard modelling assumptions and annual average emission factors. There are limitations to this approach and where more detailed modelling is required this has then to be provided by the operator.

- Annual average emission factors derived from the BREF document and the UK inventory are usually used but other factors may be used provided they are backed up by appropriate peer-reviewed research studies.

- A number of years (e.g. five years) of hourly averaged meteorological data from the nearest UK meteorological station are used for detailed modelling purposes.

- The source and group data used depends to a large extent on site specific features and emission characteristics of the farm and use of appropriate methods is largely dependant on the experience of the modeller and their familiarity with complex agricultural sources. Emissions can be modelled as point, area, line, volume, or jet sources. Emissions from housing and from separate manure storage are usually modelled whereas emissions from

down stream operations such as slurry/manure spreading are not. Techniques for slurry/manure spreading to reduce ammonia emissions are however controlled by general rules.

- Increasingly in the UK, particularly in England and Wales, it is concentrations only that are being modelled for comparison with critical levels. This has advantages in that the operation is simpler and the uncertainties are reduced. It is the case that where very detailed assessments are required, critical loads are estimated and compared with background loads. Estimates are often made based on only two deposition velocities, one for short vegetation (0.02 m s^{-1}) and one appropriate for woodlands (0.03 m s^{-1}). Information on background levels and loads and estimates of exceedence are obtained from the UK Air Pollution Information System (APIS: www.apis.ac.uk). Wet deposition of ammonia is not modelled as it is not considered significant for short-range modelling (Loubet *et al*., 2009).

- A range of modelling assumptions may be used depending on the level of detail required. As far as possible these accurately reflect the situation on the farm, e.g. release heights, efflux velocities, temperature, location etc. Generally, terrain and buildings are not considered, but can be included if required. The addition of building effects means that sources have to be modelled as point sources and this is not always appropriate for farms. Usually the surface roughness length selected for the dispersion site is assumed to apply throughout the domain, a typical value for agriculture being 0.3 m. If the need arises, advanced models such as ADMS 4.1 have the facility to define a distribution of surface roughness over the domain.

- It is usual for model output to be plotted on to 1:10 000 or other appropriate scale maps. Additional specific points or transects across sensitive areas are included as required.

Typical stages in the approach for modelling developments as part of the EIA process are less clearly set out than the above process for IPPC permitting. However it is likely that the assessment process follows a similar pattern and uses the same assessment criteria although there may be greater variation in methodologies.

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6.3 The nitrogen load on Dutch Natura 2000 areas; local effects and strategies

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Summary

- To assess nitrogen levels on a local scale, an experimental version of the OPS model was combined with monitoring measurements from Natura 2000 areas.
- In a pilot study, expected emission trends were converted into future local deposition levels. These levels define the nitrogen loads which limit the future development of habitats. Such levels may alter due to local policies and measures.
- Within a specific Natura 2000 area, local patterns of nitrogen deposition were used for defining core areas where desired habitats can develop. Other areas could function as buffer zones to minimise influences from outside.

6.3.1 Introduction

Nitrogen deposition in the Netherlands shows considerable local variation. This is caused by the high deposition velocity of ammonia, in combination with a large number of clustered ammonia sources. Agriculture is the most important contributor, especially from dairy farming and intensive cattle breeding, but traffic and natural sources can also dominate patterns, locally.

Considering the high critical load exceedances for Dutch Natura 2000 sites, we study the effects of exceedance on nature quality and define options to mitigate negative effects. Because of the high local variability in nitrogen deposition, a sophisticated approach is required.

Detailed deposition maps of all Dutch Natura 2000 areas can only be derived through model calculations. Such a spatial detail on a national scale is beyond the reach of measurements alone. However, measurements credibly present the real situation on a measurement site, whereas model calculations may differ considerably from reality. Thus, to create realistic deposition maps, both model calculations and measurements have to be combined. Most of this work will be executed in the near future, some preliminary results are presented here.

6.3.2 Aims and objectives

- To assess present and future nitrogen pressure on a local scale for Natura 2000 areas. Model calculations and measurements have to be combined, to produce nitrogen deposition maps on a scale of 500 m × 500 m or 250 m × 250 m.
- To assess the consequences of elevated nitrogen levels for the individual Natura 2000 areas. For this purpose, the deposition maps will be combined with local flora monitoring data.
- To define local policy options for individual Natura 2000 areas. Point of departure is present local exceedances of critical nitrogen loads and their future trend, following national emission and deposition scenarios. Local circumstances and opportunities will be included to define options for local policies.

6.3.3 Results and discussion

Concentrations and depositions rates of ammonia were calculated by the Dutch OPS model (Van Jaarsveld, 2004). This statistical trajectory model includes dispersion, air transport, dry and wet deposition and chemical reactions. In the case of ammonia, the model was expanded to include specific effects, such as re-emission. Currently, this specific model version has an experimental status (not yet official). Several measurement campaigns and nine monitoring sites served to verify the model results for ammonia.

Average monthly ammonia concentrations were collected from the Measuring Ammonia in Nature areas (MAN) network (Stolk *et al*., 2009). This network focuses on those Natura 2000 areas which are vulnerable to nitrogen deposition. These areas have sandy soils and are located in the east and south of the Netherlands and along the coast (dunes).

The measurements were compared with site-specific model calculations. In general, model and measurement were in good agreement (Figure 6.2). In the southern areas with intensive cattle farming and in the large Veluwe heathlands, the difference, on average, was less than a few percent. The model showed a slight overestimation of about 10 per cent in the intensive cattle farming areas in the east, and an underestimation of the same magnitude in areas with relatively extensive agriculture. Only the calculations for areas along the coast seriously underestimated the measured concentrations. This underestimation is an object for further study.

The model will be used for calculating deposition maps for all Natura 2000 areas, on an appropriate scale of 500x500 or 250x250m. These maps have to be combined with the information from the measurements. An easy and direct way is a simple multiplication of the map with the quotient of measurement and model averaged over the natural area, separately, or the average of this quotient over a larger area, such as the intensive cattle farming area in the east. The latter method also allows for a correction of the maps for natural areas without monitoring sites. More sophisticated methods, however, will also be explored to combine measurement and model.

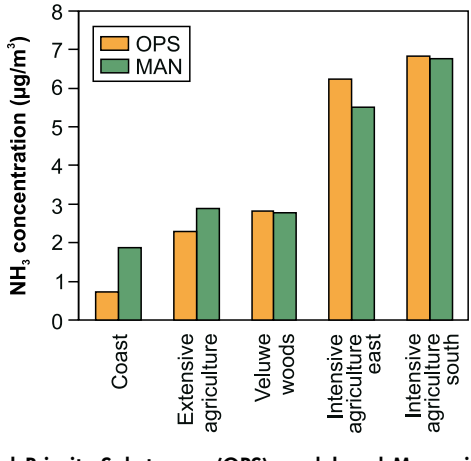

Figure 6.2: Operational Priority Substances (OPS) model and Measuring Ammonia in Nature (MAN) monitoring network values for ammonia concentration (μgm^{-3}), averaged over a number of Natura 2000 areas in the South and East of the Netherlands.

To gain insight into future trends, the maps of nitrogen deposition will be scaled to future local nitrogen loads and effects, by use of projected emission trends. On the basis of these projected deposition charts, local strategies can be derived to comply with EU directives. Together with general national efforts to decrease the nitrogen load, these local strategies may include the mitigation of local sources, the management of the ecosystems involved, and the spatial organisation of natural areas and their surroundings.

These maps will become available in the near future. At this moment, only some preliminary exercises are available. These were based on calculations with an older model version and rely more on the monitoring data. A crude example of this approach is available for the Groote Peel, a Natura 2000 area with heathland, grassland, pools and bogs, in the south of the Netherlands.

Nine monitoring sites were set up in the Groote Peel. The measured concentrations were combined with OPS calculations to calculate estimated deposition levels for each site. Together with expected emission trends (a low and high scenario), future deposition levels were also derived (Figure 6.3).

For these sites, the nitrogen deposition exceedance was calculated for several habitat types (Figure 6.4). This showed that critical loads for both heathland and bogs will still be exceeded in the future. On several more-favourable locations, future exceedance may be rather small if emissions follow the lower scenario. Thus, with proper additional measures, part of the area may develop into high-quality heathland. Other parts, with higher depositions, may serve as buffer areas between sensitive nature and agricultural land.

However, the Groote Peel was not put forward as a Natura 2000 area for its potential to as a heathland but as a peatland. Even within a low-emission scenario, the deposition will be much higher than required to develop healthy peatland. Within the coming decades, achieving this aim would require unrealistically extreme efforts and, therefore, may not be appropriate for this area.

Further insight is needed into the effects of nitrogen critical load exceedance on biological end points, to clarify the consequences of policy options. To gain such insight, exceedance maps will be combined with flora monitoring data. Prior to this step, dominant effects

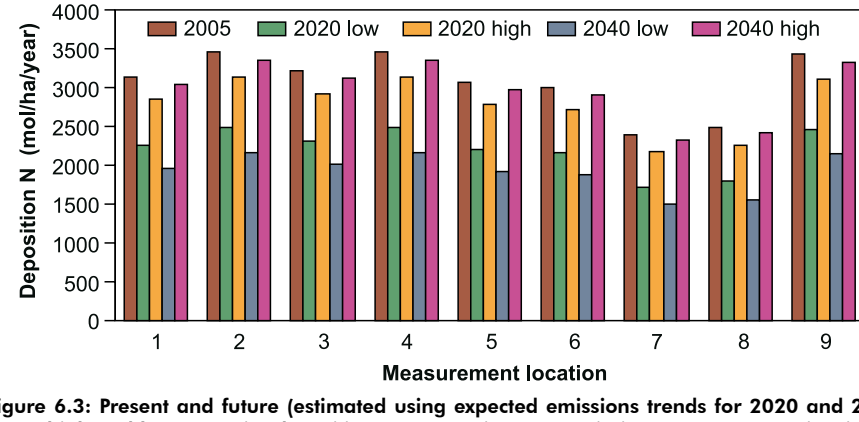

Figure 6.3: Present and future (estimated using expected emissions trends for 2020 and 2040 for as high and low scenario) deposition on Measuring Ammonia in Nature (MAN) -sites in the Natura 2000 area "Groote Peel".

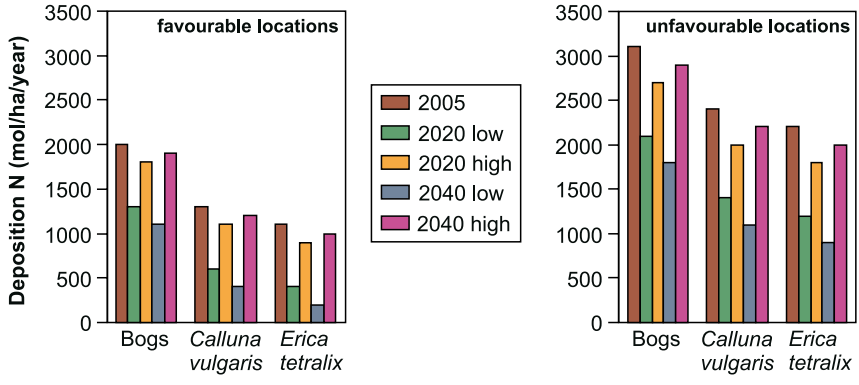

Figure 6.4: The exceedance of critical deposition levels in the Groote Peel, averaged over sites with low deposition (favourable) and with higher deposition (unfavourable) in 2005 and projected for 2020 and 2040.

of acidification and soil moisture content need to be excluded from the flora data by categorising the available data into sites with the same acidity and soil moisture content, but with differences in nitrogen load. This information can then be used for converting calculated nitrogen critical load exceedances into expected floral changes.

6.3.4 Conclusions

- To assess the present nitrogen load on Dutch Natura 2000 areas, a combination of model calculations and measurements is necessary.
- The new experimental version of the OPS model and the MAN monitoring network allowed a reliable estimate of ammonia concentrations on a very local scale. These concentrations were converted into local deposition levels, although results from this step are much more uncertain.
- Expected future deposition levels define which habitats may develop in a healthy way. Local patterns in the deposition may help in defining the core areas where these habitats may develop, and buffer areas that protect these habitats against influences from outside.

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Stolk, A.P., van Zanten, M.C., Noordijk, H., van Jaarsveld, J.A. and van Pul, W.A.J. (2009) Meetnet Ammoniak in Natuurgebieden. Meetresultaten 2005 – 2007. RIVM Report 860710001. Bilthoven, Netherlands.

7

CURRENT AND FUTURE POLICY OPTIONS FOR TACKLING NITROGEN DEPOSITION IMPACTS ON NATURA 2000 SITES (THEME 5)

7.1 Background document

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7.1.1 Introduction

Atmospheric nitrogen deposition represents a major anthropogenic threat to the 'Natura 2000' network and to the conservation status of habitats and species listed under the Habitats Directive. The Natura 2000 network has a central place in European conservation legislation, affording sites the highest degree of protection of any nature conservation areas under European law. Many of these habitats are naturally adapted to limited nitrogen supply, so that additional inputs can cause substantial changes in biogeochemistry and species composition. The importance of nitrogen as a key threat has been recognized through 'nitrogen deposition' being listed as one of the long-term indicators under the Convention on Biological Diversity, and, related to this, in the SEBI 2010 process of the European Environment Agency (Streamlining European Biodiversity Indicators for 2010; EEA, 2007).

In this background document, we briefly review the challenge of protecting the Natura 2000 network from nitrogen deposition, arguing that there is a need for further policy development, as well as improvement in the enforcement procedures. We then explore a range of possible policy options that could help address the concerns identified. It should be noted that the Habitats Directive uses the Natura 2000 network as part of its overall ambition to maintain and improve conservation status, including the occurrence of species outside of Natura 2000 sites. Here we deliberately focus on Natura 2000, as the flagship network with the highest degree of protection for conservation sites in the European Union. While not losing this focus, the present discussion should be seen in the context of these wider objectives.

The purpose of this document is to stimulate discussion for the COST 729 workshop. It is hoped that the ideas presented here will encourage additional suggestions. Together, these options can then be refined to provide a shortlist of approaches that merit in-depth investigation for future policy development and enforcement.

7.1.2 The nitrogen deposition threat and the need for further policy development to protect the Natura 2000 network.

The Natura 2000 network comprises all Special Areas of Conservation (SACs) and Special Protection Areas (SPAs), as designated under the Habitats Directive (92/43/EEC) and the Birds

Directive (79/409/EEC), with the Habitats Directive also including updated provisions for the protection of SPAs. In aiming to provide the highest degree of conservation protection, a precautionary approach is specified, as illustrated by Article 6.3 of the Habitats Directive:

> *Any plan or project not directly connected with or necessary to the management of the site but likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives.*
>
> *In the light of the conclusions of the assessment ..., the competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned and, if appropriate, after having obtained the opinion of the general public.*

For this purpose, a 'plan or project' is understood to be intended to mean any activity which might potentially have an adverse effect on the integrity of one or more SACs. Apart from exceptions outlined in Article 6.4 (in the case of no alternatives and of overriding public interest), the Habitats Directive thus, in principle, guarantees a high level of protection, particularly as it explicitly notes that multiple activities should also be assessed in regard of their combined effect on the sites.

Given this precautionary approach, it is therefore of interest to note that many SACs and SPAs remain under the threat of anthropogenic nitrogen deposition. For example, Figure 7.1a shows the estimated location of critical load exceedance for nutrient nitrogen across Europe. This is the amount by which estimated total nitrogen deposition is larger than the 'critical load', the estimated amount of deposition below which effects do not occur according to present knowledge. Critical load exceedance is the indicator used by the SEBI 2010 activity, for which values have been established using extensive analysis of field observations, experiments and models (e.g., Achermann and Bobbink, 2003; ICP Modelling and Mapping, 2004). Similarly, critical levels are used for NH_3 and NO_x, which are the air concentrations above which effects do occur according to present knowledge (ICP Modelling and Mapping, 2004; Sutton *et al*., 2009b). Wherever exceedance of either a critical load or critical level occurs, adverse impacts of nitrogen on Natura 2000 site integrity may be expected. Figure 7.1a refers to 2010, assuming that the existing commitments under the UNECE Gothenburg Protocol (UNECE, 1999) and the EU National Emissions Ceilings Directive (2001/81/EC) to reduce emissions of nitrogen oxides () and ammonia (NH_3) are met. From an international perspective, there is therefore a long way to go until adverse effects of nitrogen deposition on the Natura 2000 network can be avoided.

It is important to consider spatial scale when assessing the overall threat of nitrogen deposition to sensitive habitats. Thus Figure 7.1b shows the estimated pattern of critical loads exceedance for two example habitat types in the UK, based on national models. These maps illustrate the variation in sensitivity between habitat types (through differing values of critical loads) and the fact that the rates of nitrogen deposition are also dependent on land cover type (nitrogen deposition is largest to rough forest vegetation). While Figure 7.1b shows the regional patterns using 1 km estimates of critical loads and 5 km resolution estimates of nitrogen deposition, it still does not reveal the full extent of spatial variation. Reactive nitrogen emissions can occur in the rural environment, leading to gradients in atmospheric concentrations and deposition downwind of major roads (for NO_x and NH_3, Cape *et al*., 2004), and downwind of livestock farms (for NH_3, e.g. Dragosits *et al*., 2002 and other organic nitrogen compounds). Figure 7.1c illustrates the pattern of modelled critical load exceedance that may occur in a single 5 km grid-square in an agricultural landscape. Major gradients of nitrogen deposition occur with distance from ammonia sources, including manure spreading, grazing, farm buildings and manure stores. These spatial patterns are extremely important and can

help guide the search for nitrogen mitigation policies. In particular, they highlight two extremes to the nitrogen deposition problem:

- **Long range transport**, leading to well dispersed increases in N deposition, which only vary as a result of topographic effects on wet deposition, and on dry deposition of secondary gases (e.g., nitric acid) and secondary particulate matter.
- **Short range transport**, leading to locally enhanced increases in N deposition, which are extremely spatially variable, mainly as a result of gradients in air concentrations away from sources and ecosystem dependent rates of gaseous dry deposition (especially ammonia and to a lesser extent nitrogen oxides).

Rather different strategies are needed to combat these two extremes, though both are important in contributing to the nitrogen threat to Natura 2000 sites.

Of course, critical loads and levels exceedances only provide an indicator of the threat to sites. Nevertheless, in the case of empirical critical loads, the values have been derived from a combination of experiments and field observations where effects are seen in practice (e.g., Bobbink and Achermann, 2003). The result is that these maps give a good indication of the areas in Europe and the extent of spatial variability of where Natura 2000 sites can be considered under threat from nitrogen deposition.

Where a SAC or SPA is located in an area with exceedance of a critical load or level, it is therefore anticipated that adverse impacts on site integrity will follow. This may include both damage and

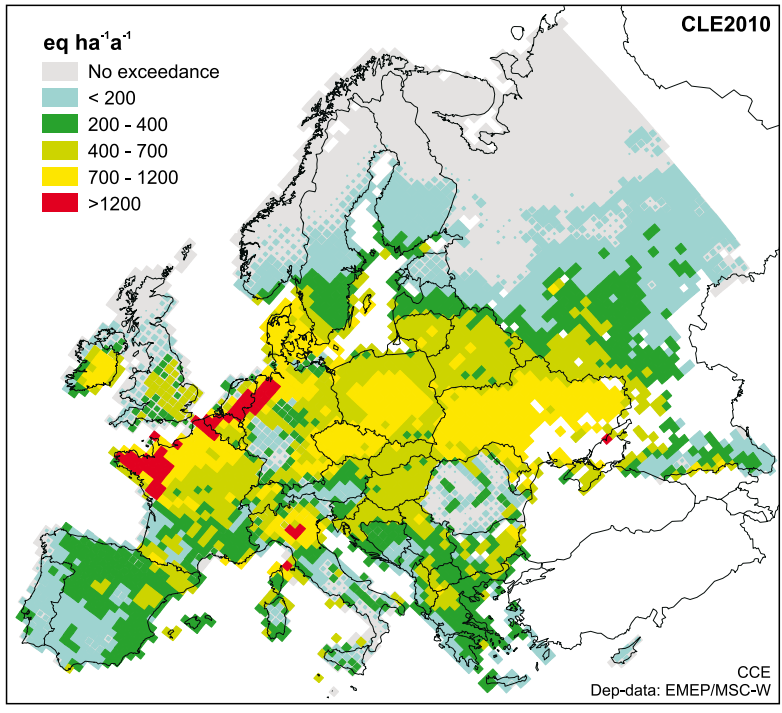

Figure 7.1a: Patterns of exceedance of the nutrient nitrogen critical load at different spatial scales: estimated exceedance across Europe in 2010 at 50 km resolution in response to total ammonia and nitrogen oxides emissions (Hettelingh *et al.*, CCE, 2008);

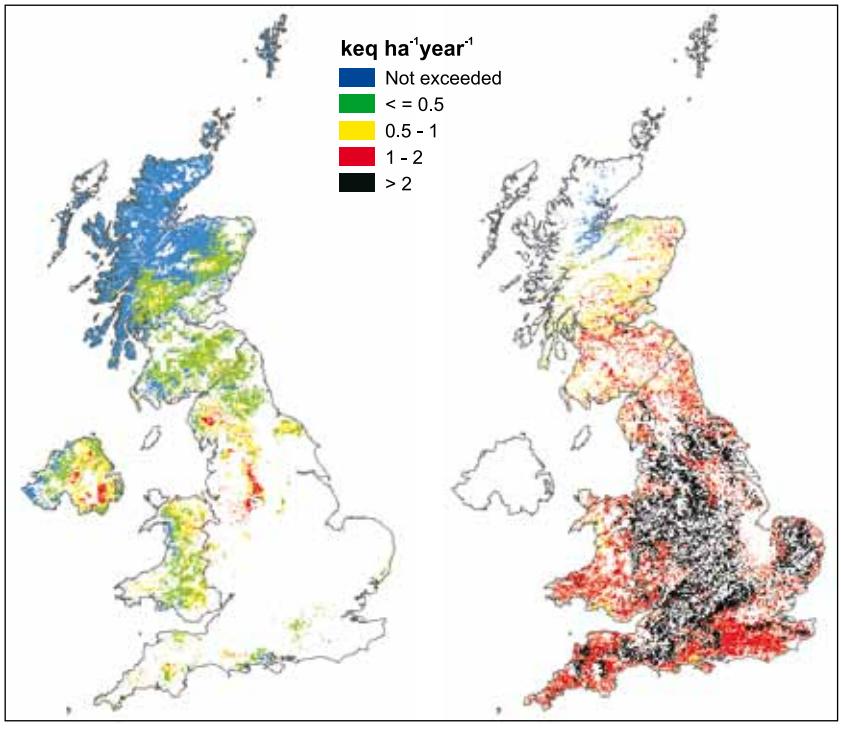

Figure 7.1b: Patterns of exceedance of the nutrient nitrogen critical load at different spatial scales: estimated exceedance across the UK at 1 km - 5 km resolution for two contrasting habitats: dwarf shrub heath (left) and managed broadleaf woodland (right) (for 2002-2004) (J. Hall, CEH);

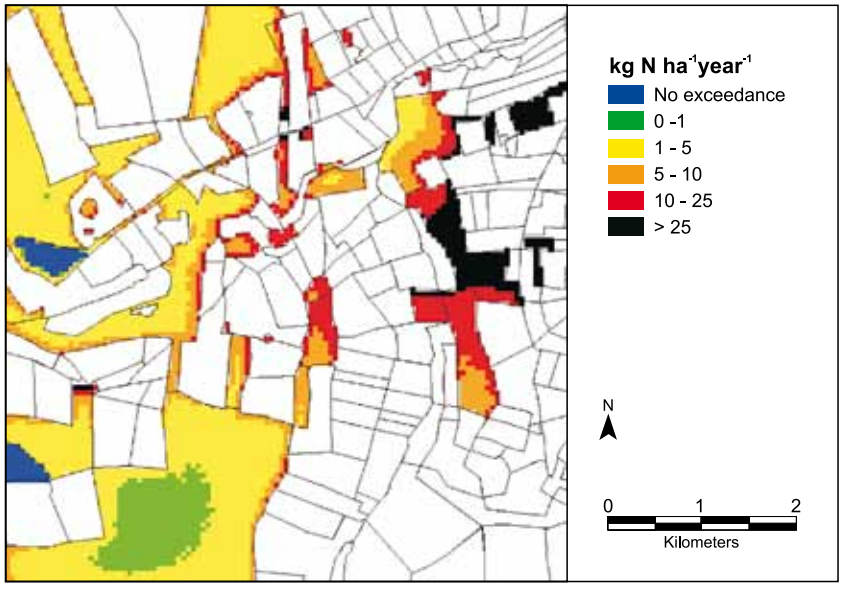

Figure 7.1c: Patterns of exceedance of the nutrient nitrogen critical load at different spatial scales: estimated exceedance across a landscape in central England at 50 m resolution in response to only dry deposition of ammonia (agricultural fields are shown in white; Dragosits *et al.*, 2002).

loss of nitrogen sensitive species communities, coupled with invasion by nitrogen loving species of lower conservation value. Examples of such changes include the loss of sensitive shrubs and wild flowers from heathlands and woodlands and their replacement by grasses (e.g., Pitcairn *et al*., 2002), loss of sensitive forbs from grasslands (Stevens *et al*., 2004) or the loss of sensitive lichens growing on trees trunks and their replacement by a few nitrogen loving species (van Herk, 1999; Wolseley *et al*., 2006; Sutton *et al*., 2009b).

Lichens are particularly sensitive to air pollution, and major changes can occur at low concentrations of ammonia. An extreme example of change appears in the paper on Moninea Bog (see Figure 3.7, this volume). These photos compares the trunk of a birch tree under clean conditions (0.4 $\mu g\ m^{-3}$ NH_3, Whim Bog, southern Scotland), with another tree growing on an SAC about 60 m downwind of a small poultry farm (15 $\mu g\ m^{-3}$ NH_3, Moninea Bog, Northern Ireland). In the latter case, the typical lichen community has been completely replaced by a thick green slime of free living algae. Such changes in species composition are replicated for many different plant groups, and can be accompanied by subsequent changes in associated animal communities.

The contrast between the high degree of protection afforded to Natura 2000 sites and the actual degree of critical load exceedances and current impacts might be considered as rather surprising. Over a decade after its adoption, it seems that the commitment to protect the Natura 2000 network has still to be met. There are a number of reasons for why nitrogen deposition is still a significant threat to Natura 2000 sites, and these apply on both on local and regional scales. For example:

- Article 6 (3) of the Habitats Directive can only meet its purpose where an appropriate assessment of a plan or project is carried out. However, in practice it requires other regulatory requirements to trigger such assessments when these are not located on a Natura 2000 site. Polluting activities that do not require any formal assessment therefore potentially constitute a loop-hole for protection of the Natura 2000 network (cf. Frost, 2004), i.e. plans and projects which are unregulated.
- Although required by the Directive, it is often difficult to consider all polluting activities in combination. Even when the polluting emissions in an area are known, it can be a major modelling challenge to consider all together. In addition, it is a point of debate whether the requirement is to consider a particular regulated source in combination with all other sources, or only to all other ***regulated*** sources.
- Nitrogen deposition results from both local and long-range sources. For example, deposition to remote tundra ecosystems is the result of long-range transport from Europe-wide nitrogen emissions. Such transboundary fluxes can only be reduced by international agreement, such as the NECD and the Gothenburg Protocol.

Presently, the goal of avoiding critical load exceedance over the whole Natura 2000 network therefore remains a long-term aspiration, even if the Habitats Directive implies an existing indirect legal commitment to reduce nitrogen deposition to the sustainable levels that would be necessary to achieve favourable conservation status.

In this context, there is an obvious need to investigate the future policy options that could strengthen the protection of the Natura 2000 network from nitrogen deposition. In the next section we first review the role of existing policies in supporting the implementation of the Habitats Directive as regards the threat of nitrogen deposition. In the subsequent sections we then explore several future options that could be developed, making the distinction between policies designed to protect from long-range transported air pollution and from those designed to protect from nearby air pollution sources. In practice, both elements are needed, with the priority depending on the location of individual Natura 2000 sites.

7.1.3 The role of existing legislation in protecting Natura 2000 sites from the impacts of atmospheric nitrogen deposition.

There are a large number of policy instruments that potentially interact with Natura 2000. In order to keep the focus, we here restrict the discussion to the main linkages. We consider the current status of each of the measures, and the potential for further development of each. The status of ongoing revisions is mentioned as far as it is known to the authors.

National Emissions Ceilings Directive (NECD, 2001/81/EC) and the UNECE Gothenburg Protocol.

The NECD provides for the EU implementation of the Gothenburg Protocol, with the focus on reducing transboundary impacts of air pollution. These instruments provide for national emissions ceilings of NO_x and NH_3 to reduce both acidification and eutrophication in sensitive ecosystems at the European scale. As the Gothenburg Protocol covers the UNECE, which has a much larger area than the EU, it has the advantage of also reducing reactive nitrogen import into the EU (and exports from the EU), as well as the transboundary fluxes between the EU Member States.

In addition to the national emissions ceilings, annexes in these instruments specify technologies that should be used to reduce both NO_x and NH_3 emissions, including various combustion and engine standards for NO_x, and a selection of mandatory measures to reduce ammonia emissions from agriculture. It should be noted that these texts represent the first time that Europe has set limits on ammonia emissions, and as such the ammonia ceilings are easily achievable for most countries. Both the Gothenburg Protocol and the NECD are being considered for future revision and the possible adoption of more ambitious targets (i.e., national ceilings) and requirements to adopt low emission technologies.

Although it is recognized in both instruments that the prime focus is on reducing transboundary transport and deposition, in practice it is difficult to separate deposition of local and transboundary origin. In general, a country reducing its emissions will be one of the largest beneficiaries of this action. On the other hand, the NECD and Gothenburg Protocol are not specifically designed to target the ***local*** reduction of emissions and environmental impacts. Thus, in meeting a national emission ceiling, it is still possible that source activities continue immediately adjacent to, and cause large local impacts on Natura 2000 sites.

Integrated Pollution Prevention and Control (IPPC, 96/61/EC and 2008/1/EC)

The EU Directive on Integrated Pollution Prevention and Control (IPPC) provides a contrasting emphasis to the NECD and Gothenburg Protocol. Rather, IPPC outlines a regulatory regime for an extensive list of specified industrial activities. Individual sources, described as 'installations' must obtain a permit to operate, based on the operation of Best Available Techniques (BAT) to reduce emissions.

The Directive is integrated to the extent that a wide range of emitted pollutants are specified, as well as noise, odour and losses to water. Many industrial activities are specified, which provides a means to reduce NO_x emissions. The main challenge in relation to nitrogen emissions has been the inclusion of agricultural emissions into such an 'industrial' regulatory regime for the first time. For this purpose, pig and poultry farms over certain size thresholds must operate according to BAT, which have been defined in extensive BAT Reference documentation (BREF, 2003). Currently, the thresholds are set at installations with more than 40,000 bird places for poultry, more than 2,000 fattening pigs or more than 750 sows.

As part of recent review of the IPPC directive, discussions have focused on possible lowering of these thresholds and inclusion of large cattle farms in the directive. For example, the body-mass

and nitrogen excretion rates between poultry types are very different, and it could be justified to have a more diverse set of thresholds, e.g. with lower thresholds for large birds like turkeys and higher thresholds for small birds like pullets. These differences are illustrated in Figure 7.3, which shows the estimated annual total nitrogen excretion for farm installations according to different animal numbers, as well as estimated rates of total ammonia emission. In this graph, bars are also shown for farm level values of N excretion and ammonia emission for cattle farms according to different size classes. For all three of the farm size thresholds indicated, overall N and ammonia emission is at least as large as the amounts for the existing IPPC thresholds. It may be noted that the ammonia values for cattle in Figure 7.3 are relatively smaller than those for overall nitrogen. This is because this graph is calculated for UK conditions, where it is assumed that cattle spend roughly half of the year outdoors, where ammonia emissions are much smaller than for housed livestock (which contribute to emissions through housing, manure storage and manure spreading).

For the livestock sector, a particularly strong emphasis was given to the consideration of ammonia emissions in the definition of BAT (BREF, 2003). In addition to requiring practices in animal houses, which have been clearly specified, BAT was also defined for the land spreading of pig and poultry slurries and solid manures. For example, the Technical Working Group (TWG) agreed that default use of a 'splash plate' spreader system (the reference method) did not constitute BAT (BREF, 2003). However, the TWG was unable to reach consensus on fully defining what BAT would be for these systems. For example, low emissions spreading techniques listed as Category 1 (well suited methods) by the UNECE (2001), such as band spreading and slurry injection, were not specified as being BAT, possibly because at that time (discussions up to 2002) countries had limited experience of these methods. Most focus was placed on discussion about the maximum time before applied manure should be incorporated for arable land.

In addition, the debate continues on the extent to which manures generated by IPPC regulated farms are considered in different Member States as regulated through their entire life cycle. It seems that the potential remains for manures generated on IPPC regulated farms to be passed to other landowners, where BAT measures would not be required. For example, this could include uncontrolled manure spreading to land (and the associated peak ammonia emissions) immediately adjacent to sensitive SAC habitats.

The debate on whether to extend IPPC to cattle appears to have focused on agreeing an acceptable number of permits across Europe, from which a farm size limit could be defined. This process led to a rather large farm size threshold for discussion (e.g., ~600 cattle). The result was that this would only address a small percentage of the cattle farms in Europe, and it has therefore been argued that such an approach would not be worth the benefits. Discussions are ongoing and there are further points that should be considered. Firstly, cattle are the main source of ammonia emissions in Europe. Thus, even if only 10 per cent of the European cattle herd were included in IPPC, the emissions regulated would be of the same order as that from pigs or poultry. Secondly, the IPPC regime introduces a regulatory framework, requiring review and assessment in relation to other environmental issues. This means that where there is an application for an IPPC permit for a farm located near to an SAC or SPA, it must be assessed in relation to the provisions of the Habitats Directive (Article 6.3). IPPC thus provides an important mechanism to ensure that the objectives of the Habitats Directive are met. At present, it seems that cattle farms often operate without a requirement for environmental impact assessment. Inclusion of the largest cattle farms would therefore ensure that such assessment could be made, supporting the Habitats Directive.

Environmental Impact Assessment Directive (97/11/EC)

The Environmental Impact Assessment Directive specifies conditions where environmental assessments of new plans and projects should be made, linking to planning policies in different

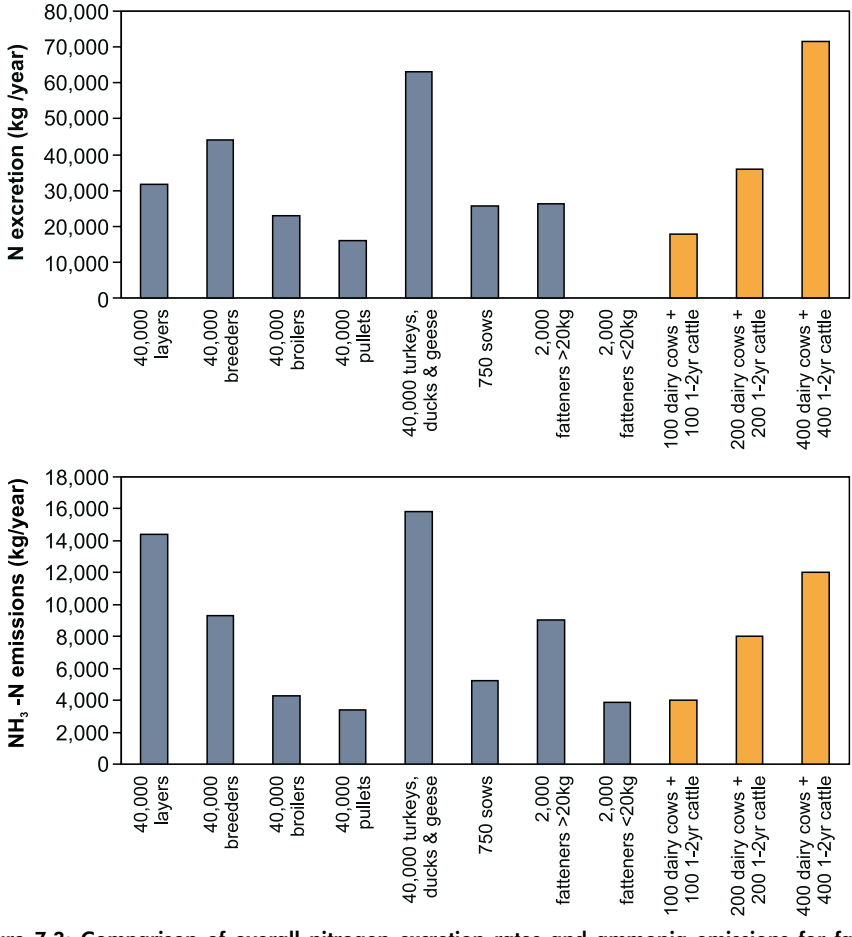

Figure 7.3: Comparison of overall nitrogen excretion rates and ammonia emissions for farm installations of different sizes according to numbers of different animals. The blue bars indicate current thresholds under IPPC, while the green bars indicate notional thresholds for cattle farms.

Member States. The EIA Directive includes a list of project categories that are subject to assessment (specified in the Directive Annex I), including oil refineries, power stations, motorways or express roads, widening of dual carriage ways of more than 10 km continuous length, waste disposal installations and quarries, open cast mining and peat extraction of over 150 hectares. It can be seen that many of these are relevant to ensure the assessment of NO_x emissions from combustion sources. The directive also includes thresholds for agriculture, 85,000 places for broilers, 60,000 places for hens, 3,000 places for production pigs (over 30 kg) and 900 places for sows. It is curious that the categories for animals broadly follow the IPPC directive, but with higher thresholds. Since assessment would already be required for IPPC installations, the intention of these higher thresholds is not clear.

The Directive also specifies a second list of activities (Annex II), for which assessments are required on a case-by-case basis according to thresholds to be set by Member States under the guidance of listed selection criteria (Annex III). The list includes many other small industries relevant for NO_x emissions. For ammonia, the list includes waste treatment plants, sludge deposition sites, projects

for the restructuring of rural land holdings and intensive livestock installations (where not included in Annex I). The selection criteria for Member States to identify projects requiring assessment (Annex III) includes the characteristics of the project in regard to pollution, cumulative effect with other projects and the environmental sensitivity of areas likely to be affected, including areas classified as protected under Member States' legislation (including the Habitats Directive; SPAs are specifically mentioned).

In principle, therefore, provisions are available in the EIA Directive requiring the assessment of effects of most projects causing NO_x and NH_3 emissions on SACs and SPAs. However, work is needed to evaluate the interpretation given to Annex II categories by Member States. In practice, it appears that many agricultural activities are not assessed in regard of their impact on Natura 2000 sites. In the UK this links to the idea that agricultural activities are in general not classed as 'development'. A more-clear enforcement of the requirement to conduct environmental impact assessments for Annex II listed agricultural activities could provide a lighter touch approach than the extension of the detailed regulatory regime of IPPC to include more farms. However, as Annex II allows Member States to set their own criteria, there remains the danger that many activities impacting on Natura 2000 sites would continue to operate without assessment.

Strategic Environmental Assessment (SEA) Directive (2001/42/EC)

The focus on the SEA Directive is the specification of environmental assessment for large scale plans and programmes. A list of conditions apply that require an EIA under this directive, including the requirement to inform other Member States of possible transboundary impacts of proposed plans or programmes.

Most importantly, the SEA Directive specifies that assessment should be made in relation to regional plans. Under Article 3, paragraph 2 is written:

> *"Subject to paragraph 3, an environmental assessment shall be carried out for all plans and programmes, (a) which are prepared for agriculture, forestry, fisheries, energy, industry, transport, waste management, water management, telecommunications, tourism, town and country planning or land use and which set the framework for future development consent of projects listed in Annexes I and II to Directive 85/337/EEC,…"*

Here it should be noted that the Annexes to Directive 85/337/EEC specify an extremely long list of categories including (under Annex II): "1. Agriculture (a) Projects for the restructuring of rural land holdings,… (e) Poultry-rearing installations (f) Pig-rearing installations." Cattle and arable farming activities are not specified, and no size thresholds are stated.

This directive therefore has the potential to review the impacts of nitrogen emissions more widely, including both NO_x emissions from roads and NH_3 emissions from agriculture. For example, where a regional plan specifies an area as being targeted for agricultural activities rather than urban or other development, then it could be argued that this choice should be assessed in relation to the protection of the Natura 2000 network. Such assessments are urgently needed, especially since the N deposition threat to many SACs and SPAs will result from the cumulative effect of many farms (inc. small farms) from the surrounding region.

Other national legislation

It would be a large task to summarize all the other national legislation that exists which is relevant to support implementation of the Habitats Directive. Nevertheless, it would be useful to list examples

during the workshop, in order to develop a fuller understanding of the variation between Member States.

In particular, as has been highlighted in the sections above, there appears to be a major loophole regarding the regulation and impact of ammonia emissions from agriculture on Natura 2000 sites. For example, under UK legislation, many agricultural activities are not considered part of 'development' legislation for the purposes of local planning policies. This may mean that a new animal house might be built or stocked without requiring planning permission, thereby avoiding assessment under the terms of the Habitats Directive.

Only in certain instances would such developments be assessed. For example, in the UK one public planning enquiry considered the siting of an agricultural dwelling in an area designated as 'green belt'. In such an area, only 'agricultural dwellings' would be allowed (pending the requirement obtain planning permission). However, to be accepted as an agricultural dwelling, the applicant had to demonstrate a viable agricultural business (in this case a poultry farm). In fact, the farming activity itself required no permission (it was below the IPPC threshold), even though the site was immediately adjacent to a sensitive heathland SAC. The inspector noted that there might be a loophole in the legislation, i.e. were it possible to conduct the farm business without an associated dwelling. However, he concluded that such a possible loophole did not apply in this instance, since the dwelling and the farm needed to be considered together, and thereby tested in relation to Article 6(3) of the Habitats Directive. Considering, in particular, the short distance to the SAC (around 10 m), the proposal was refused (Frost, 2004). This example highlights that there will be many other instances of agricultural activities that go untested in relation to the Habitats Directive.

7.1.4 Future options for protection of Natura 2000 sites from long-range transported nitrogen deposition

Here we consider the potential for other approaches that could reduce the nitrogen deposition impacts to Natura 2000 sites, firstly from long-range transported N deposition and secondly (in the following section), from locally transported deposition in source regions. We give particular attention to the role of agricultural sources, as the issue of most concern.

Revision of the Gothenburg Protocol and NECD

Both instruments are currently undergoing development work in preparation for their potential revision. The establishment of new, more ambitious national ceilings would result in an overall reduction in nitrogen deposition from both nitrogen oxides and ammonia emissions. It is worth comparing the progress already made in reducing the emissions of pollutants regulated under the Gothenburg Protocol. Figure 7.4 distinguishes between countries in the EU and other Parties in the UN-ECE area. For the EU, the baseline reductions are largest for SO_2 (72 per cent reduction) and NO_x (53 per cent reduction), and smallest for ammonia (7 per cent reduction). The gap between Baseline and the Maximum Reduction specified by measures included in the RAINS model (MRR) is also largest for ammonia, highlighting that the current commitments for this pollutant are the lightest of the different pollutants considered.

Figure 7.4 shows that there is considerable potential for further reduction of ammonia emissions under revision of the Gothenburg Protocol, which would result a substantial decrease in the threat to sensitive Natura 2000 sites. At present the degree of ambition, both in terms of the national ceilings and in the technical requirements, remains a topic for future discussion among the Parties to the Convention.

In addition to the benefits for the Natura 2000 network and Europe's natural environment as a whole, there would be substantial co-benefits from further reduction of nitrogen emissions under

the Gothenburg Protocol. Both nitrogen oxides and ammonia contribute to particulate matter formation, which leads to significant life shortening across Europe, through respiratory and other illnesses. In the case of agriculture, nitrogen lost from the farming system as ammonia represents a waste of fertilizer N inputs. Given the high costs of fertilizer nitrogen, their sensitivity to oil price changes, and the energy consumed in nitrogen fertilizer production (2 per cent of world energy consumption), saving nitrogen in the system has the potential to save farmers money, make them less at risk to fertilizer price changes, and reduce energy consumption. Many other co-benefits can be expected. For low emission manure spreading this can include: increased agronomic flexibility, more accurate delivery of manure to crops, more accurate avoidance of spreading adjacent to surfaces to be avoided (near water courses, near SACs etc) and a reduction in odour emissions (see discussion by Webb *et al*., 2009).

Interactions between other community legislation and Natura 2000

The targets of the Gothenburg Protocol and the NECD are set using a modelling optimization approach that aims to minimize environmental effects, including those on ecosystems as specified using Europe-wide maps of critical loads, such as that illustrated in Figure 7.1. By contrast, the legislative commitments of these instruments are set as the combination of required technologies (e.g., Gothenburg Protocol annexes) and the national emissions ceilings. There is currently no legal commitment in these instruments that is directly related to an ecosystem protection target.

As a large scale 'plan', it might be argued that the even revision of the NECD should be assessed under the SEA Directive, meaning that the implications of revision must be assessed explicitly in relation to the possible threat to the Natura 2000 network. Potentially this could lead to a circular position where only a revision that was sufficiently ambitious to protect the Natura 2000 network fully could be adopted, but that this would be, at the same time, too ambitious to be acceptable by Member States.

More constructively, such interactions should be considered in relation to directives considering other objectives. For example, it is understood that new European animal welfare legislation will require a change in animal housing, leading to a phasing out of the traditional 'tied stalls' for housing of cattle. This will require a change to more open animal houses allowing free animal movement, which is its core objective. However, it is also expected that this change will increase ammonia emissions, leading to an exacerbated threat to the Natura 2000 network. Presumably, through the requirements of the SEA directive, the impact on Natura 2000 should to be assessed. Subject to the conclusions of any such review, it might therefore be expected that any move from tied stall to open barn would be accompanied by the requirement to adopt techniques to ensure that overall ammonia emissions from each farm did not increase.

Development of an effects oriented goal for nitrogen exposure to Natura 2000 sites

In order to better protect the Natura 2000 network, there is a need for the legal commitments to be set directly in relation to environmental goals. Thus the NECD achieves a general reduction in emissions, but it does not relate closely to the commitment to protect the Natura 2000 network. For this purpose, critical loads (as already used by SEBI, 2010) and critical levels could be used to set a ***nitrogen target for Natura 2000 protection*** across Europe and for each Member State. Such a target could be expressed as:

> *"A long term goal to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030".*

The details would need debate, including the 95 per cent number and the target year, but the principle should be clear. It may be noted that this goal is phrased as the per cent number of designated sites,

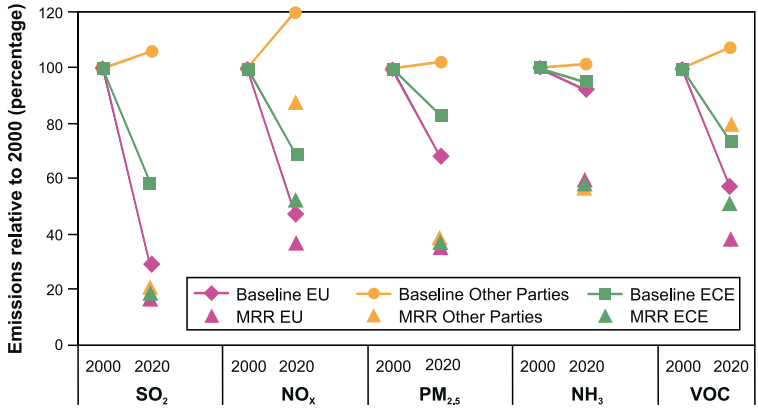

Figure 7.4: Comparison of baseline projections in emissions between 2000 and 2020 with the maximum rains reduction (MRR), which relates to full implementation of measures currently considered within the RAINS model. The distinction is made between countries of the EU (i.e. linked to NECD projections) and other parties to the UN-ECE Convention on Long Range Transboundary Air Pollution. The Gothenburg Protocol did not include explicit commitments for particulate matter ($PM_{2.5}$), but it is currently proposed to include this in the protocol revision (Amman, 2009, pers. comm.).

rather than the per cent area of the overall Natura 2000 network. This is important, since it could be argued that each SAC or SPA designation is of equal value to society. For example, a large SAC may occur in a very remote area, where there is no shortage of land, while a small SAC may occur as a priority for protection in a landscape under high human pressure. In an analysis for the UK presented by Hallsworth *et al*., (this volume; Hallsworth *et al*., 2011), it is shown that there is a tendency for small SACs to occur in the most polluted areas. Finally, loss of integrity over any part of an SAC may be considered as a threat to the integrity of the whole. For this reason, Hallsworth *et al*., (2011) calculated the per cent number of SACs where the critical level was exceeded over some part of the each SAC (Designation Weighted Index, DWI). They compared this with the total area of SACs exceeded in the country (Area Weighted Index, AWI). Using ammonia critical levels of 1 µg m^{-3} (ecosystems with relevant epiphytes) and 2 µg m^{-3} (precautionary value for higher plants on Natura 2000 sites), they concluded that 11 per cent and 1 per cent of the area of the UK Natura 2000 network (AWI) exceed the critical levels, respectively. By contrast, 59 per cent and 24 per cent of Natura 2000 sites (DWI) exceed the same critical levels (Hallsworth *et al*., 2011). The AWI approach did not provide an appropriately precautionary measure because of: a) the anti-correlation between NH_3 concentrations and area of each SAC and b) the failure to consider variation in NH_3 concentrations across SACs. These last points can be seen clearly in Figure 7.5.

Co-benefits of planting trees and other low-nitrogen biomass

A rather different regional scale approach to reduce impacts of reactive nitrogen deposition and concentrations on Natura 2000 sites is through the application of land use policies. For example, such policies are already discussed in relation to carbon sequestration under the Kyoto Protocol, i.e. allowing credit for increasing carbon sinks in planted forests (Article 3.3).

In the context of carbon sequestration, it has recently been discussed whether a certain amount of N deposition would be beneficial in increasing forest C uptake rates (Hogberg, 2007; Magnani *et al*., 2007, de Vries *et al*., 2008, Sutton *et al*., 2008). Of course it must be recognized that such potential benefits must be balanced against increases in nitrous oxide emissions and impacts on biodiversity, water quality etc. (De Schrijver *et al*., 2008).

In the present discussion, however, the point of interest is that increasing forest area will lead to a decrease in atmospheric nitrogen concentrations and deposition to other receptor ecosystems. The reason for this is that forest land (and other unfertilized tall biomass crops) scavenges nitrogen compounds (especially ammonia, nitric acid and particulate matter) through dry deposition more effectively than short, fertilized agricultural land. A larger area of woodland therefore results in faster removal of these compounds from the atmosphere to these surfaces, resulting in less being available for deposition elsewhere. Policies of extending forest area based on this principle therefore have the potential for substantial co-benefit between carbon and nitrogen impacts. The idea of urban forest plantations has also been considered in relation to its benefits for human health, through reducing particulate matter concentrations (McDonald *et al*., 2007).

Theobald *et al*., (2004) examined scenarios of forest planting in the UK, showing that these had potential to give significant reductions in ammonia deposition to existing forests and to other semi natural land, such as heathlands. However, they pointed out that the location of the forest plantings is important in this context, as these should be made in the areas with highest nitrogen emissions and deposition. Planting a forest in a remote area with very low nitrogen deposition would lead to little benefit. Such policies should also be considered in relation to their local implications, for example in the establishment of buffer-zones (Section 5.2).

It should be examined whether this link between carbon and nitrogen policies could be made at a European scale. For example, it should be considered whether the benefits of Article 3.3. forests under the Kyoto protocol could also be considered as 'nitrogen emission credits' under the terms of a revised NECD.

Patterns of societal behaviour

It should briefly be noted that the directives discussed focus mainly on technical changes, whereas the overall burden of nitrogen emissions is a result of a much wider set of societal choices. For example, the choices of individual European citizens determine their energy consumption (emissions through electricity generation), their annual vehicle mileage and (emissions from transport) and their consumption of animal products (NH_3 emissions from livestock agriculture). A great deal of effort is currently placed on educating the public in their energy and transport choices, particularly to reduce carbon footprints. In parallel, much more thinking needs to be done to consider how to optimize European dietary choices for both human health and the environmental consequences. Such societal changes have a huge potential to influence European scale emissions of reactive nitrogen, thereby affecting the transboundary transport and deposition of nitrogen to Natura 2000 sites.

7.1.5 Future options for protection of Natura 2000 sites from short-range transported nitrogen concentrations and deposition

While the above policy interactions have the potential to affect transboundary fluxes, they do not directly address the problems of short range transport to Natura 2000 sites in source areas, with these often being the sites under the most extreme threat. Options for further development include strengthening the links with cross-compliance in agriculture, spatial planning including buffer zones and the application of air concentration objectives and local air quality management for ecosystems.

Strengthening the cross-compliance links for Natura 2000

One of the principles of European agricultural financial support (i.e., the single farm payment system) is that the payments are made to farmers under the principle of cross compliance. This includes two requirements:

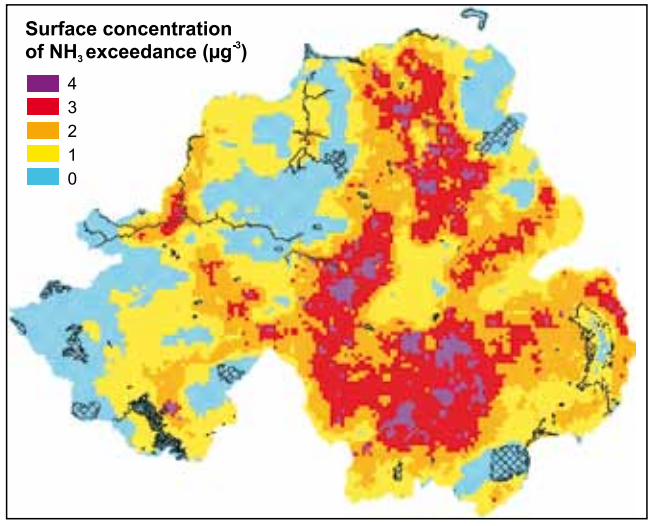

Figure 7.5: Spatial pattern of NH_3 concentration and the location of Special Areas of Conservation (SACs) in Northern Ireland (1 km resolution FRAME model estimates calibrated against UK measurement network). Although many of the largest SACs do not exceed the lowest critical level (1 μg m^{-3}), substantial exceedance is seen for the smaller sites. 22 per cent and 5 per cent of the area of SACs in Northern Ireland exceed the 1 and 2 μg m^{-3} critical levels, respectively (Area Weighted Index, AWI), however, 74 per cent and 42 per cent of the SACs exceed the same critical levels over part of their domain (at 1 km^2 resolution, Designation Weighted Index, DWI). The DWI is considered the legally correct approach under the terms of the Habitats Directive (Hallsworth *et al.*, 2009a,b). Moninea Bog is located ~2 km from the SW border (1-2 μg m^{-3}, area ~1 km^2).

- ***statutory management regime***: that farmers are in full compliance with existing legislation relating to their farm and the environment. For example, farmers need to comply with the Nitrate Directive, the Habitats Directive and any other relevant legislation. This requirement applies equally across the European Union.
- that farmers maintain land in ***good agricultural and environmental condition***, primarily relating to the condition of the farmland itself, but also with implications for off-site losses, e.g. avoidance of manure spreading adjacent to water courses. This requirement is delegated for each Member State to define.

The implication of cross-compliance is that, in principle, any farmer in receipt of a single farm payment should already have demonstrated that they have no adverse impact on Natura 2000 sites. In practice, it should be asked to what extent such links are currently made between different Member States. The impression is that, at present, this link is not adequately treated and that further guidance needs to be developed on: a) general rules for avoiding impacts on Natura 2000 sites through N concentrations and deposition, b) specification of suitable impact assessment approaches, including cost-effective methods applicable for small farms.

It is worth noting that, even under the previous system of agricultural area support payments, the principle of cross-compliance already applied. However, in practice the linkages seem to have been rarely enforced. This highlights the challenges involved in developing these linkages for the future.

Spatial planning, including buffer zones

Spatial planning has a significant role to play in reducing the impacts of nitrogen deposition and concentrations on the Natura 2000 network. In landscapes with large N emissions (source areas), the amount of N deposited to a sensitive site is very closely linked to the distance from major nearby emissions. This is for example, clearly shown for Northern Ireland (Figure 7.5), where the patterns of ammonia concentration (modelled at 1 km resolution) closely match to the mapped ammonia emissions.

In the Netherlands, policies were already established some years ago whereby manure from areas with high ammonia emissions was transported to areas with low emissions. Naturally, this resulted in an increase in ammonia concentrations in the cleaner areas, which caused some debate as to the benefits of the policy (see, Bleeker and Erisman, 1998). However, if the priority is to protect those areas most under threat and the other areas were established as a) less under threat and b) of lower priority for nature conservation, then the policy remains logical. If such policies should be considered more widely, a clear agreement on the relative priorities would need to be established from the outset. This poses a challenge for the wider objectives of the Habitats Directive, which seeks to maintain conservation status of habitats and species across Europe as a whole, including sites not designated as Natura 2000.

Local spatial planning policies, including the use of buffer zones have the potential to be much less controversial, and are already established for other effects, such as the use of buffer zones adjacent to water courses. In the case of nitrogen emissions to air, such buffer zones could be appropriate both for nitrogen oxides emissions from roads and for ammonia emissions from agriculture. Three aspects to such buffer zones should be considered:

- increasing the distance from the source, allowing greater dispersion before the air reaches the sensitive area, such as an SAC,
- increasing the dispersion between source and receptor, such as by planting tall rough vegetation, further diluting the pollutant before it reaches the sensitive area,
- encouraging deposition between the source and receptor, such as provided by planting tall vegetation as a buffer zone.

In practice, the first two benefits are expected to be most important for narrow buffer zones of a few 10s of metres. For the third benefit, planting a single row of trees would have a trivial effect in removing ammonia from the atmosphere, for which purpose wide tree belts of >100 m would be required (see Theobald *et al*., 2004, Loubet *et al*., 2009). As dry deposition rates for NO_x are very small, only the first two benefits would apply for this pollutant. Enhanced nitrogen deposition adjacent to major roads is due to both NO_x and NH_3 (Cape *et al*., 2004), due to catalytic converters increasing NH_3 emissions compared with cars without converters. Hence broad woodland plantings adjacent to roads could achieve all three benefits.

Dragosits *et al*., (2006) considered the potential for tree plantings to reduce nitrogen deposition to a landscape in the UK. For example, they showed how tree plantings both adjacent to farm sources and to the nature reserve sinks could lead to significant reductions in deposition (Figure 7.6). They also investigated the potential of other buffer zones, for example, the avoidance of manure spreading and urea application up to 100 m, 300 m and 500 m from the nature reserves. These scenarios led to smaller benefits, mainly because in their model scenario, overall emissions were dominated by farm point sources (including a large poultry farm). Such buffer zones would, however, have significantly reduced peak ammonia concentrations on the nature reserves immediately after manure spreading.

The same authors addressed the effect of location of the major point source. The scenarios shown in Figure 7.7 indicate that there are significant benefits, even if the farm is located 1 km further away from the reserve, to the west in this example. At a distance of three km, the farm makes a relatively minor contribution to deposition at this the nature reserve site, as shown by comparison of the scenario with the farm removed.

The use of buffer zones therefore has a high potential for further policy development to protect Natura 2000 sites from nitrogen deposition in source areas. In particular, the approach has the advantage that rather simple distance rules could be set for the avoidance of different sources, e.g., farm buildings or of manure spreading activities. For example, rules might be established that up to 300 m from a sensitive SAC (effectively at least one field distance), slurry and urea were not spread to agricultural land (or not unless a high abatement efficiency technique was applied, such as immediate ploughing in).

Air concentrations objectives and local air quality management for ecosystems.

Under the Air Quality Directive (AQD) (2008/50/EC), ambient air standards have been set for NO_x (expressed as NO_2), SO_2, O_3 and particulate matter, with the prime focus on protecting human health from air pollutant exposure in the urban and industrial environments. However, the directive also includes critical levels for SO_2, NO_x and O_3 set for the protection of vegetation.

A major tool that was widely used in previous air quality directives, and has been continued in the AQD is the establishment of objective concentrations linked to local Air Quality Plans, or local air quality management (LAQM). There is a requirement for local authorities to regularly review and assess air quality in their area against the standards and objectives prescribed in regulations.

When these objectives are not being achieved, or are not likely to be achieved within the relevant period an Air Quality Management Area (AQMA) must be designated. Once this area has been designated the local authority must develop a remedial Action Plan to improve air quality in that area. The local authority should define the boundaries of the AQMA, communicate the implications to the local community and statutory consultees and coordinate with neighbouring authorities regarding possible adjacent AQMAs.

Given the existing commitment under the Habitats Directive, such an approach would be applicable for the protection against ecological effects on Natura 2000 sites. For this purpose, existing critical levels for NO_x (ICP Modelling & Mapping, 2004) and NH_3 (UNECE, 2007; Sutton *et al*., 2009) could be used as the starting point for defining objective concentrations.

The actual values set for this purpose would presumably depend on the balance of ecological risk versus costs, as negotiated between the Member States. For the purpose of ecosystem protection, the main focus could be in relation to annual mean concentrations, based on monthly sampling (also ensuring that certain peak monthly concentrations are not exceeded). Since daily fluctuations in NO_2 and NH_3 are not considered important from an ecosystem perspective, this would reduce the costs of the measurements required, because passive sampling methods could be used (where shown to be reliable).The following approach might be taken:

- Establish NO_x and NH_3 concentration objectives that apply in air over the surface of Natura 2000 sites (e.g., measured at 1-2 m above ground). The main focus should be on annual values, but monthly averaged maxima should also apply.
- National modelling is used to assess whether the NO_2 or NH_3 objectives are exceeded over all or part of the domain of a Natura 2000 site.

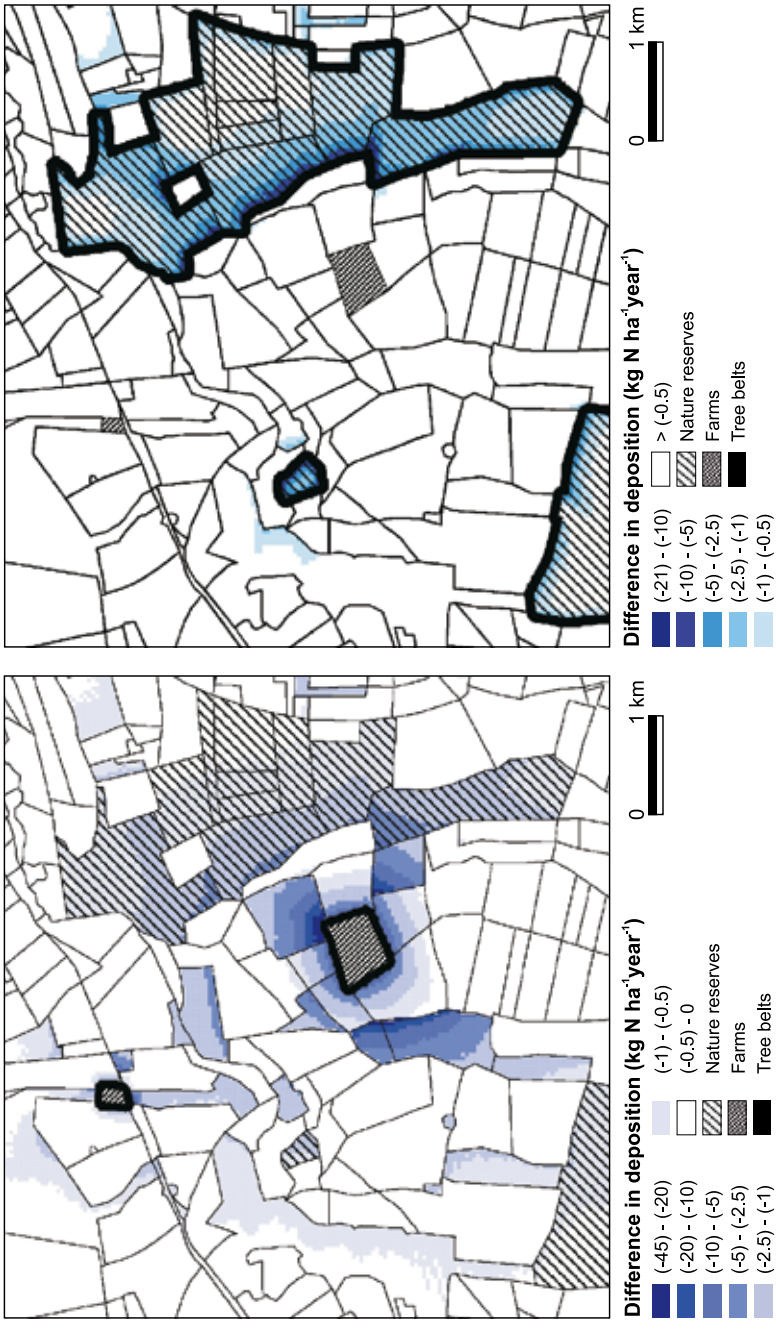

Figure 7.6: Landscape-level scenarios illustrating the effect of woodland buffer zones on atmospheric nitrogen deposition to three nature reserve areas. On the left, the scenario consisted of adding a 50 m wide buffer of trees planted around two farms (a small beef farm, and a large poultry farm). On the right, the scenario represented the adding of a 50 m wide buffer of trees around each of the nature areas. The maps show the reduction in N deposition compared with the base run. Both scenarios demonstrate significant benefits (Dragosits *et al.*, 2006).

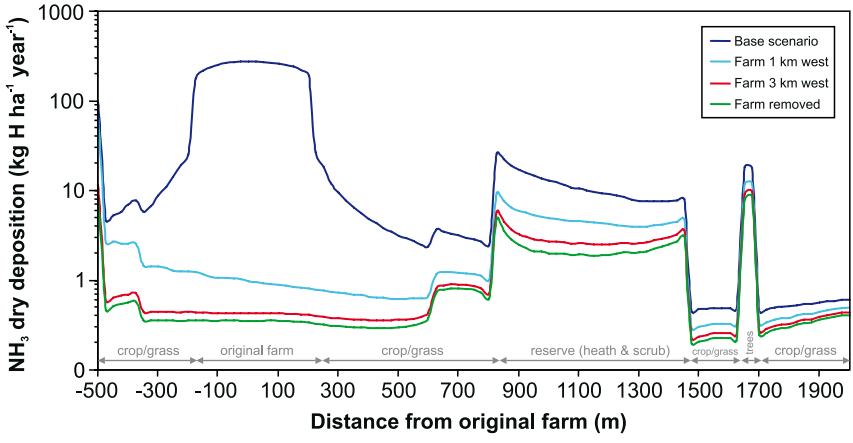

Figure 7.7: Modelled transect of atmospheric nitrogen deposition due to ammonia assuming a base situation and three scenarios related to the location of a major point-source livestock farm (Dragosits *et al.*, 2006). The reserve area could be considered as an SAC or SPA.

- Local screening models are applied to identify the locations on the Natura 2000 site that are most at risk of exceeding the NO_2 or NH_3 objectives.
- Atmospheric monitoring is conducted at the locations identified in c) for at least one year (using monthly sampling with robust passive sampling methods).
- If the objective concentrations are not exceeded, no action needs to be taken. If the objectives are exceeded, then a local management plan should be established that specifies a course of action by which they would be reduced.

Such an approach would necessarily need to be backed up by a clear set of legislative and voluntary tools to achieve the concentration objective values, and thereby reduce the impacts to Natura 2000 sites. The potential to link this to the existing Articles 6.1 and 6.2 of the Habitats directive should also be considered.

7.1.6 Conclusions

This review has identified that atmospheric nitrogen deposition and the associated concentrations of reactive nitrogen represent a significant threat to the Natura 2000 network. The evidence is that the application of existing policies is not currently adequate to protect these flagship sites for the protection of Europe's biodiversity. Many sites exceed critical levels and loads, with consequent adverse ecological effects.

It is concluded that the nature of the nitrogen deposition problem for biodiversity can be distinguished into: a) reducing long-range transboundary air pollution and b) reducing short-range pollution impacts in source areas. While policies addressing the first, e.g. NECD, will have some benefits for the second, they are not specifically targeted for this purpose, with the result that many local impacts can still be expected. To reduce the impacts on Natura 2000 sites in source areas requires a specific set of policies designed for this purpose.

In comparing NO_x and NH_3 emissions, it is clear that there is a much greater regulatory control over the NO_x emissions. This is reflected in a significant reduction in baseline estimates of European NO_x emissions over 2000-2020. By contrast, there has been hardly any reduction in NH_3 emissions, which mainly arise from agriculture. This difference is reflected in the current degree of attention to reducing NH_3 emissions in existing policies. Although requirements are included in both the

NECD and IPPC Directives, these represent the first such agreements, and consequently the current ambition levels are rather modest.

In regard of the impacts on Natura 2000 sites, the existing commitments of the Habitats Directive should afford a high level of protection. In practice, this intended degree of protection is not achieved, in particular, because many sources of NH_3 continue with little regulation.

At the regional scale, there is potential for more effective protection of the Natura 2000 network through revision of the NECD and the IPPC Directives. In addition, there is substantial scope for revision or more rigorous enforcement of the Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) Directives. For example, extending the provisions of the EIA Directive to include other farms could provide a light touch approach for these farms that would avoid the full regulatory regime of IPPC. Secondly, under the SEA Directive, the implications of regional plans on ammonia emissions need to be tested in relation to the terms of the Habitats Directive. Similarly, the effect of other proposed legislation (e.g., animal welfare legislation causing increased NH_3 emissions) needs to be tested in relation to the Habitats Directive.

New approaches that should be investigated include an effects-oriented goal for N effects on Natura 2000, the linking of carbon sequestration and nitrogen deposition benefits in forest planting policies (linking Kyoto Article 3.3 and NECD revision) and approaches that help foster reduced nitrogen consumption by European citizens. Approaches that include the assessment of ecosystems services, such as carbon sequestration, could highlight important positive and negative effects of nitrogen deposition on Natura 2000 sites that could provide an added incentive for actions to protect sites.

Much more effort needs to be given to managing the local impacts of nitrogen deposition and concentrations on Natura 2000 sites in source areas. This could include strengthening the enforcement of existing cross-compliance links between single-farm payments and impacts on Natura 2000 sites, coupled with the development local spatial planning measures, including guidance on buffer zones for atmospheric N deposition. Finally, substantial focus has been given to developing local air quality management under the EU Air Quality Directive, linked to human health protection. Currently, no such system is in place for ecosystem protection. A combination of establishing objective concentrations for NO_x and NH_3, together with a system of local air quality management for ecosystem protection would provide a suitable approach. By integrating ecosystem-level air quality management with some of the options mentioned above, a more rigorous approach could be developed that matches to the existing commitments under the Habitats Directive.

7.1.7 Key questions for discussion

- Have Natura 2000 sites been assessed for the risk of N deposition in your country?
- Are sufficient policies in place to protect Natura 2000 sites, and if so are they being adequately implemented and enforced?
- Do you see a need for further policy development in this area?
- To what extent do you agree that the procedures needed to protect from NO_x emissions are largely in place?
- Do you agree that the challenges to protect Natura 2000 sites from nitrogen deposition and concentrations are greatest for the impacts of agricultural ammonia emissions?
- To what extent do you think that existing legislation could be enforced more effectively to protect the Natura 2000 network?
- How important do you rate the usefulness of high level goals, e.g., "***A long term goal to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030***", as compared with the application local level policies?

- What are the other possible approaches that have not been discussed in this document?
- If you were to develop a package of measures to protect Natura 2000 sites from nitrogen deposition, what would you consider to be the most suitable elements?
- How might such a package be expected to differ when viewed from different viewpoints (scientific, administrative, policy, political, industry, conservation etc.)?
- How should such a package be considered in relation to wider objectives of the Habitats Directive to maintain Europe wide conservation status, including areas outside the territory designated as Natura 2000 sites?
- Would an assessment of ecosystem services provided by Natura 2000 sites be a help or a hindrance to policy development for their protection?

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7.2 Working group report

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7.2.1 Conclusions and recommendations of group discussions

Overview of current situation with regard to nitrogen deposition impacts to Natura 2000 sites

Regarding the current policies and their adequacy for the protection of Natura 2000 sites from the threat of nitrogen deposition, the workshop concluded that:

- The Natura 2000 network remains under threat from atmospheric nitrogen deposition despite the Habitats Directive affording it a high level of protection.
- Atmospheric nitrogen deposition is a Europe-wide problem, but with very high spatial variability in severity of impacts and a high variability in national policy responses.
- Natura 2000 sites are not routinely assessed for the risk of nitrogen deposition effects and present policies and /or their enforcement are not sufficient.
- A lack of awareness of the nitrogen threat is the main problem in many Member States.
- Ammonia emissions present the greatest policy challenge in Europe.
- There is currently insufficient linkage between biodiversity and air pollution policy development.
- Economic and conservation priorities clash particularly in countries with significant levels of nitrogen deposition.

Recommendations for policy development

The role of existing legislation
It was recommended by the working group that:

- Those Member States that have advanced policies integrating several legislative instruments could provide practical advice for other Member States.
- International agreements (NEC Directive and Gothenburg Protocol) should have a higher level of environmental ambition (especially for ammonia), in particular to improve protection at local scale.
- Exceedance of critical loads (including in Natura 2000 sites) should be more explicitly considered in optimization of abatement measures.
- Ammonia should be included in the Air Quality Directive (2008/50/EC) and there is potential for setting standards for annual mean concentrations of ammonia to protect ecosystems.
- The potential for 'cross compliance' of different legislative measures to address nitrogen deposition issues should be more actively promoted.
- All existing projects should be captured by Article 6.3 of the Habitat Directive.

Future options for protection of Natura 2000 sites
The working group discussions captured the following suggestions and recommendations:

- Legislation at both regional and local scales is needed, including measures to deal with within-country atmospheric transport.
- Policies and procedures should be considered that distinguish between the management of nitrogen oxides and ammonia, and to address the role of organic nitrogen compounds emitted to the atmosphere.
- It is recommended that new approaches are explored in future policy development to complement existing approaches to managing the nitrogen deposition threat in relation to Natura 2000 and the wider objectives of the Habitats Directive, including:

    - Multi-media regional reactive nitrogen ceilings, limited by the most sensitive N_r species and effect, should be explored as a basis for further policy development. This approach could enable the optimization of all nitrogen emissions of a region in relation to the adverse impacts;
    - Nitrogen reduction plans could include a long-term plan to attain critical loads on a regional level in countries with high levels of exceedance;
    - Spatial Planning (operated at local and regional levels) can optimize the location of existing pollution sources to minimize the overall threats, exploiting where possible landscape structures to buffer impacts (including buffer zones and tree belts);
    - Nitrogen impact assessment techniques should be further developed to take into account the objectives of the Habitats Directive more specifically;
    - The Ecosystem Services concept may provide a holistic framework for examining the links between air pollution effects on ecosystems and human well-being.

- The following specific measures were recommended for further consideration:

    - Improve ammonia coverage in the Integrated Pollution Prevention and Control (IPPC) Directive, i.e. include manure spreading, consider the current farm size thresholds and inclusion of cattle;
    - Set strict emission limits and management obligations to encourage abatement technology development;
    - Strategic Environmental Assessment (SEA) has a role to play at high level planning for pollution abatement;
    - Develop and encourage non-technical measures (societal behaviour);

– Consider establishing a high-level goal as part of a package of actions, for example "... ***to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030.***"

7.2.2 Introduction to the structure of discussions

The working group started its discussions by considering the key questions presented by Sutton *et al.,* (this volume, section 7.1). To facilitate structured discussions the working group decided to order key questions with similar themes into the following groups:

- Are sufficient policies in place? (Questions: 1, 2, 3 and 6 from Sutton *et al*, this volume, section 7.1)
- Do existing policies adequately cover oxidized and reduced nitrogen? (Questions: 4 and 5)
- Local versus regional policy, and the usefulness of an overall goal (Question: 7)
- New approaches (Question: 8)
- Most suitable approaches (Questions: 9, 10 and 11)
- Ecosystem services (Question: 12)

Working group members and members from other groups were invited to share their experience of nitrogen deposition and Natura 2000 network issues in their country of residence. These presentations are presented as supporting papers in Sections 7.3 to 7.7.

7.2.3 Highlights of discussion and views expressed

Are sufficient policies in place? (Questions: 1, 2, 3 and 6)

The working group agreed that there is various legislation and policy in place that can address nitrogen emissions and impacts across Europe but that it is not consistently applied in all Member States.

It was agreed that the most pronounced nitrogen deposition problems for Natura 2000 sites are in NW Europe. However, nitrogen deposition also affects biodiversity in other areas, and it is crucial that improvements to policy implementation are made in all areas with significant nitrogen deposition. "Significant" could be defined as "above the critical load".

Many EU Member States are active in implementing measures to protect Special Areas of Conservation (SAC) sites, designated under the EC Habitats Directive, and Special Protection Areas (SPAs), classified in accordance with the EC Directive on the Conservation of Wild Birds. The consensus view was that more ambition needs to be realised in respect of nitrogen deposition. Several ways of doing this were explored by the group:

- International agreements could have a higher level of environmental ambition to help reach local targets, and decision makers and polluters could be made more aware of the benefits. In this respect the National Emission Ceilings Directive (NECD) and the Gothenburg Protocol to the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP) are very important as countries are unlikely to be more ambitious than what is needed to fulfil the Directive/Protocol emission reduction obligations at national levels. The consensus was that National Emission Ceilings should be much more closely tied to the conservation status in the Member / Signatory States (Sutton *et al*., this volume, section 7.1).
- The policy process at regional level could be optimized by an ***ex post*** analysis to see how scenarios fulfil effects targets. This is being done within the CLRTAP in the Gothenburg Protocol revision process. The European Union should consider using and extending these methodologies to focus specifically on Natura 2000 sites (Sutton *et al*., this volume, section 7.1).

- The Integrated Pollution Prevention and Control (IPPC) Directive could be expanded, e.g. to include cattle and manure spreading, although this route may not be the most suitable for small farmers[1]. It was also suggested that the Environmental Impact Assessment (EIA) Directive could cover this without the full burden of IPPC approach. The IPPC Directive is a suitable tool for nitrogen management as it can introduce more efficient nitrogen use through Best Available Techniques (BAT) (see Section 3 of Sutton *et al*., this volume, section 7.1).
- Under the Air Quality Directive there is potential for setting standards for annual mean concentrations of NO_x and NH_3 to protect ecosystems (Sutton *et al*., this volume, section 7.1).
- The EU Environmental Liability Directive (2004/35/EC) that aims to 'establish a common framework for the prevention and remedying of environmental damage at a reasonable cost to society…' covers air pollution and could potentially be linked to the protection of Natura 2000 sites from N deposition.
- Existing projects are not always included in site assessments; both existing and future developments need to be assessed together.

As well as the value of individual measures, the importance of cross-compliance was highlighted, with an emphasis on assessing the willingness to apply existing or new measures and the potential for enforcement. For instance, the advantages of farms taking an integrated approach to applying legislative requirements is clear, but the mechanisms needed to do this efficiently are currently lacking. This situation could be improved by more integration of directives under DG Environment (e.g. Habitats Directive and NECD) such as integrated policy to lower background nitrogen deposition.

Oxidized versus reduced nitrogen (Questions: 4 and 5)

The working group recommended that increased emphasis be given to considering policies and procedures that distinguish the management of nitrogen oxides and ammonia:

- The procedures needed to protect sites from NO_x emissions are largely in place in many Member States. While this can be considered a success, it does not mean there is no need for further reduction in NO_x emissions;
- The challenges concerning agricultural ammonia emissions, which are under-regulated across most Member States, are much larger. In most cases, agricultural ammonia emissions are not assessed in relation to their impacts on the Natura 2000 sites;
- Agricultural activities are also thought to emit various organic nitrogen compounds to the atmosphere. These have seldom been assessed and represent a potentially significant additional threat to Natura 2000 sites that requires further quantification.

Local versus regional policy and the usefulness of an overall goal? (Question: 7)

This question was articulated as 'how important do you rate the usefulness of high level goals, e.g., "***A long term goal to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030***", as compared with the application of local level policies?'

There was consensus that this type of target would be a very useful high level policy goal (equivalent to the preamble of Gothenburg Protocol or NECD). It was also proposed, as discussed

1 Note: the IPPC Directive has meanwhile been recast as the Industrial Emission Directive. It includes neither cattle nor manure spreading, but it calls for a review by the European Commission by 31 December 2012 on the need for emission controls on these sources.

in the background document on N policies (Sutton *et al*., this volume, section 7.1), that any target should be defined as the per cent number of designated sites, rather than the per cent area of the overall Natura 2000 network (see Hallsworth *et al*., this volume). This is important, because each site contributes to conservation status of habitats and species listed under the directive.

New approaches (Question: 8)

The working group recommended that several new approaches are explored in future policy development to complement existing approaches to managing the nitrogen deposition threat in relation to Natura 2000 and the wider objectives of the Habitats Directive:

- Multi-media regional reactive nitrogen (N_r) ceilings, limited by the most sensitive nitrogen species and effect, should be explored as a basis for further policy development. This approach could enable the optimization of all nitrogen emissions of a region in relation to the adverse impacts.
- As already proposed a few years ago (see e.g. http://asta.ivl.se/Saltsjobaden3.htm, Conclusions of Group 5), N_r produced in an area enters the soil, water or air and it is therefore theoretically possible to set critical loads / limits based on the sensitivity of ecosystems affected by these fluxes. The use of thresholds gives the opportunity to spatially integrate the effects using modelling approaches potentially including fluxes of reduced and oxidized nitrogen to air, soils and water. Spatial scaling could be attempted linked to the thresholds but a major challenge would be to balance the sources and avoid double counting etc. The aim would be to operate the model at regional scale or even European scale but there needs to be a demonstration of whether the idea is implementable in the near future or long-term (and a road map defined).
- Nitrogen reduction plans: this could include a long-term plan to attain critical loads at a regional level including: (i) regional legislation; (ii) abatement techniques (BAT); (iii) autonomous development; (iv) trading permits (as considered already in the Netherlands). It should be noted that any trading permits should consider the spatial aspects of the ecological impacts.
- Spatial Planning: this may be operated at landscape and regional levels. The approach optimizes the location of existing pollution sources to minimize the overall threats, exploiting where possible landscape structures to buffer impacts.
    - The use of tree belts, and other buffering options, around habitats and sources were discussed. The group agreed that local spatial planning policies, including buffer zones, are a practical and usually uncontroversial way to tackle more local effects.
    - In addition, the group discussed how the Nitrates Directive could have important co-benefits for ammonia emission control in combination with spatial planning.
- Further development of nitrogen indicators: a number of indicators are available, but the policy message depends on their implementation. For example, it was shown that for ammonia critical level exceedance in the Natura 2000 network, the Area Weighted Index (AWI), underestimates the scale of threat compared with a Designation Weighted Index (DWI) (see Sutton *et al*., this volume, section 7.1).
- Ecosystem Services concept: This may provide a holistic framework for examining the links between air pollution effects on ecosystems and human well-being (see below and Hicks *et al*., this volume, section 7.9).

Most suitable approaches (Questions: 9, 10 and 11)

The discussions underlined the importance of flexible and integrated approaches, building on existing legislation such as the Air Quality, NEC, Nitrates, Water Framework, IPPC, EIA, SEA and Environmental Liability Directives. Denmark for instance has combined different legislation (see Dinesen and Bjerregaard, this volume). It was stressed that future options need to address nitrogen

import and export, as well as within-country transport. The role that non-technical measures (societal behaviour) could play was also highlighted.

An immediate step that could be taken was improving the ammonia coverage of the IPPC Directive by including manure spreading in IPPC, e.g. consider the current farm size thresholds and inclusion of cattle. The potential for the Environmental Impact Assessment (EIA) directive and other approaches to attain ammonia emission reductions without the full burden of the IPPC obligations was also noted. Attention was drawn to the current proposals for revision of the mandatory measures in the Gothenburg Protocol (Technical Annex IX), for which options are being provided by the CLRTAP Task Force on Reactive Nitrogen.

The SEA Directive could also have a role to play in high level planning for pollution abatement but that a permitting system may be required. Also regional plans should be tested in relation to Article 6.3 of the Habitats Directive.

In addition to these messages, the following specific measures were recommended for further consideration:

- Negotiate more ambitious ammonia ceilings under NECD and Gothenburg Protocol.
- Include ammonia in the Air Quality Directive (2008/50/EC).
- Set strict limits encourages abatement technology development.
- Include non-technical measures (societal behaviour).
- Consider establishing a high level goal as part of a package of actions to ensure that 95 per cent of Natura 2000 designated sites do not exceed critical loads or levels for reactive nitrogen compounds by 2030.

Ecosystem services (Question: 12)

The suitability of a using assessment approaches based on the concept of ecosystem services to provide a holistic assessment of nitrogen impacts in the environment was discussed (Hicks *et al.*, this volume, section 7.9). The potential for ecosystem services such as carbon sequestration and greenhouse gases fluxes (N_2O) to be valued using carbon equivalent pricing approaches was also discussed. It was clear that data availability is not sufficient to allow for quantification and economic valuation of the whole range of effects that nitrogen deposition can have on ecosystems. But there is potential for qualitative assessments to offer a framework for policy development where benefits and trade-offs of different policies can be compared. However, the scale and temporal aspects that need to be addressed are seen as key challenges.

7.2.4 Country presentations

A series of informal presentations were given to the working group on the approaches to protecting Natura 2000 sites from nitrogen deposition in different countries. The main issues discussed are described below and fuller descriptions are provided in Section 7.3 to 7.7.

The key issues raised were:

- A lack of awareness of the nitrogen threat is the main problem in some Member States. Many Member States have not assessed the risk to Natura 2000 or conservation status from nitrogen deposition, yet European critical loads exceedance maps show widespread exceedance.
- Some Member States see other more significant threats to their Natura 2000 sites which are a higher priority than nitrogen deposition, such as land-use change, fragmentation or fires.
- Some Member States are actively pursuing integrated approaches to nitrogen management;

- For other Member States, nitrogen emissions are thought to be very closely coupled with the economy and there are political sensitivities with limiting emissions despite many ecologically sensitive sites with critical loads already being exceeded and not fully recovered.
- For some of the more nitrogen polluted of the Member States, where critical loads are already exceeded, the focus may need to be on maintaining the status quo with a requirement to guarantee that existing and new sources do not lead to increases in nitrogen deposition to vulnerable ecosystems.
- Air pollution experts in Member States are often not linked effectively to conservation practitioners.
- Some Member States are limited in their capacity to reduce nitrogen deposition as over 50 per cent of their deposition can be imported, therefore international agreements, e.g. NECD and Gothenburg Protocol, are important.

7.3 Nitrogen deposition and Natura 2000 sites in Austria

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7.3.1 Introduction

Austria is a landlocked country in Central Europe covering an area of c. 84000 km^2. The eastern Alps cover two-thirds of Austria, making it the country with the largest share of the entire Alps. In the eastern part continental pannonian flatlands are typical. Accordingly,Habitats Directive habitats and species from the alpine and the continental biogeographic region can be found. The article 17 reporting in the year 2007 includes 66 habitats and 172 species listed in the annexes of the Habitats Directive (http://eea.eionet.europa.eu/Public/irc/eionet-circle/habitats-art17report).

Similar to many European countries, nitrogen deposition exceeds the critical loads at the majority of the area and particularly in forests (Umweltbundesamt, 2008). Effects of nitrogen deposition in natural and semi-natural ecosystems were found in the early nineties (Zukrigl *et al*., 1993) and later on (Hülber *et al*., 2008, Zechmeister *et al*., 2007, Dirnböck & Mirtl, 2009) but most studies focussed on forests. Very little knowledge exists regarding alpine habitats and other natural and semi-natural grasslands. Neither the current status of nitrogen deposition and related effects in the Natura 2000 network were assessed nor will the future monitoring of the conservation status include a "nitrogen component".

7.3.2 Trends of nitrogen deposition

Emission of nitrogen oxides decreased by 10.8 per cent, ammonia by 4.4 per cent between 1990 and 2008 (Umweltbundesamt, 2010). Whereas emission targets for ammonia (according to the NEC directive) were achieved, the emission of NO_x is still far above. Totally 46 per cent of the Austrian area is covered by forests and almost the entire forested area is exposed to a critical load exceedance regarding nitrogen. A preliminary assessment shows that the risk for adverse effects in non-forest habitats is also very high (Umweltbundesamt, 2008).

7.3.3 Agriculture and Natura 2000

The Natura 2000 network in Austria includes 220 single areas and 14 per cent of the total designated area is used for agriculture. Apart from this direct impact of agricultural practices on the conservation status, emissions of nitrogen may pose indirect negative effects in neighbouring Natura 2000 areas through short-range transport. This is particularly important in a country where

small scale farming is dominating in a very heterogeneous environment. As a result, conservation areas are almost always embedded in agricultural land.

Land management in Austria is mainly cattle farming, whereas crop farming is less important. Totally 187034 agricultural and forestry enterprises were managed in Austria in 2007. Since the year 1999, a reduction by 14 per cent was observed. The average size of farms is 18.9 hectares of arable land. Mountain farms are even smaller. In 2009 the Austrian cattle population amounted to about two million, pig population to three million, the sheep population to 350000.

In 2009 the number of subsidized organic farms rose to 20870, which is 15 per cent of all holdings. The share of organic farming area in the arable land is 18.5 per cent. Totally 73 per cent of all farms or 89 per cent of the total arable land participated in the Agri-environmental Programme (ÖPUL) in 2009. For the ÖPUL and the "Compensatory allowance for less-favoured areas" (2nd CAP pillar), 73 per cent of all subsidies (agriculture and forestry) were used (Lindner *et al*., 2010).

This increasing share of organic farming is most likely one of the major triggers towards decreasing ammonia emissions in Austria. Between 1990 and 2008 ammonia emissions decreased by 4.4 per cent (Umweltbundesamt, 2010).

The opening of the agricultural market after 2013 will likely reduce cattle numbers in Austria and strengthen the current ongoing loss of farmland to forestry. Further decrease of ammonia emissions from agriculture is thus likely.

7.3.3 Implications for Natura 2000 sites

Although the risk of adverse effects of excess N deposition in designated conservation areas and for endangered species is obvious, the problem is not currently recognized as a top priority issue in Austria. Firstly, air pollution experts in Austria are not linked effectively to conservation practitioners. Secondly, knowledge about the effects of N deposition in some important habitats is very rare. There are very few studies in alpine areas, especially in calcareous grasslands, and none in Austria. In particular, studies on short range impacts near farms are missing. As a result, and though Article 17 reporting included air pollution as a frequent pressure, the Article 11 monitoring scheme does not address the issue. There is a general need for a broader monitoring system that is effect-related because currently effect-monitoring is restricted to forests.

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7.4 Nitrogen deposition and Natura 2000 in Denmark

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7.4.1 Introduction

Status and trends

From 1989 to 2008 nitrogen deposition on land surfaces and marine waters in Denmark has decreased about 30 per cent and 28 per cent respectively (Ellermann *et al*., 2010). It has been a gradual decline with some variation between years due to changing weather conditions, and it is explained by a reduction in European as well as Danish emissions. Results from the National Monitoring Programme (NOVANA) of terrestrial habitats indicate a resulting significant decline in the nitrogen content of lichens on heaths and dunes, which receive all nitrogen from the air, although the level is still considered being too high (Bruus*t al*., 2010). A corresponding decline in the nitrogen content in shoots of dwarf shrubs is not seen which is interpreted as the nitrogen content in the soil still being high due to accumulation of deposited nitrogen from earlier years (Bruus *et al*., 2010).

The Danish emissions of ammonia peaked in the mid 1980s at about 114,000 tons and were reduced by about 30 per cent about twenty years later. About 98 per cent of these ammonia emissions come from agriculture especially livestock production. In Jutland the average nitrogen deposition in 2005 from agriculture was 61 per cent and on Zeeland 49 per cent and the contribution from Danish farms were about 41 per cent and 24 per cent respectively (Ellermann *et al*., 2007).

The nitrogen deposition in 2008 was via modelling based on pilot stations estimated by the National Environmental Research Institute (NERI) to be about 14 kg N $ha^{-1}yr^{-1}$ on land surfaces and 6.7 kg N $ha^{-1}yr^{-1}$ in marine areas (Ellermann *et al*., 2010).

Airborne nitrogen deposition plays a major role regarding impact on Natura 2000 habitat types and the Danish National Monitoring Programme (NOVANA) includes monitoring the annual deposition.

For marine waters the overall Danish contribution in 2008 range from about nine per cent to the North Sea to about 27 per cent to the Lillebaelt and up to 44 per cent to Limfjorden (Ellermann, 2010). For land surfaces the average deposition deriving from Danish sources is estimated to be about 36 per cent and this generally larger proportion is caused by livestock production locally and differences in surface characteristics. The regional variation is quite large with livestock production being responsible for 41-45 per cent of total deposition in mid and northern Jutland but only 21 per cent in the capital region.

Natura 2000 and favourable conservation status

Nutrients, especially nitrogen, are a threat to vulnerable habitat types in Denmark, and large amounts are emitted from livestock, industry, energy production and transport. Some nitrogen is deposited close to the source but wind can transport nitrogen some distances. Ammonia is usually transported 60-120 km and NO_x 400 km dependent on the weather, however, more than 1,000 km is also possible under dry conditions without rainfall (Ellermann *et al*., 2007). In connection with rain nitrogen will be washed down quite quickly. When nitrogen is deposited to natural habitats on land or water it leads to eutrofication of plant communities and habitats may eventually disappear due to changes in species composition. Increased levels of nitrogen deposition through many years have led to overloading of Danish ecosystems (Normander *et al*., 2009). Airborne deposition consists of wet deposition deposited by rain or dry deposition deposited by wind.

The aim of the Natura 2000 network is to maintain or restore favourable conservation status of the habitat types and species the areas have been designated for. A number of Annex I habitat types are sensitive to nitrogen deposition and the critical load for some of the habitats has been exceeded.

The most sensitive habitats such as raised bog and oligotrophic waters (which are very poor in nurients) have critical loads of 5-10 kg N $ha^{-1}yr^{-1}$ and other sensitive habitats such as certain heath and dunes habitats have critical loads of 10-20 kg N $ha^{-1}yr^{-1}$ (Ellermann *et al*., 2010). Hence the critical load for some habitats has been exceeded for a long period in Denmark as average airborne deposition of nitrogen exceeded 5 kg N $ha^{-1}yr^{-1}$ about 1910 and 10 kg N $ha^{-1}yr^{-1}$ about 1945 (Ellermann, 2007).

Government objectives

The overall aim for the government is to reduce nitrogen deposition and thereby protect sensitive nature and the biological diversity (Regeringen, 2009). Denmark has undertaken to reduce atmospheric nitrogen emissions by 2010 by 55 per cent in comparison to 1990. The government's Green Growth agreement of 2009 sets more stringent requirements with regard to the emission of ammonia in order to protect especially sensitive habitats from nitrogen (see discussion).

7.4.2 Nature conservation legislation and measures

Natura 2000 planning

As part of the Danish implementation of the Habitat and Bird Directives, Denmark has chosen to develop a management plan for each of her Natura 2000 sites as part of implementing article 4.4 of the Habitat Directive. The basic objective for the mangement plans is to provide for the maintenance or restoration of favourable conservations status as set out in article 6.1 and 6.2 of the directive. Thus 246 draft plans have in 2010 been submitted to a technical hearing with relevant municipalities. The plans cover 3,591 km^2 on land equivalent to about 8.4 per cent of the Danish land area and in total about 13,047 km^2 are covered by the Danish Natura 2000 network. The plans are legally binding and come with funding. The management plan is a framework plan and relevant municipalities, possibly in collaboration with state agency landowners were relevant, will design action plans to implement the overall plan.

In the long run, the plans aim to ensure the integrity of a site and a favourable conservation status for the habitat types and species for which the Natura 2000 sites have been designated. The plans generally regard nitrogen deposition as one of the major threats to e.g. dunes, oligotrophic waters, heath and scrub, dry grassland and meadows, raised bogs, mires and fens and decidious forests. Apart from opportunities in the action plans for implementing management activities such as removal of nitrogen by removal of plant material, sod cutting etc. regulation of nitrogen deposition is handled by the Environmental Approval Act for Livestock Production and follows a separate track in the Green Growth agreement.

Nature Conservation Act

The Nature Conservation Act provides the main legislative framework for nature conservation in Denmark. It includes general protection of habitats and specific regulatory powers for the protection of nature. Thus lakes over 100 m^2, water courses that have been designated as protected areas, heaths, bogs, moors, salt marshes, swamps, coastal meadows, grasslands of more than 2,500 m^2 are protected - so-called § 3 areas in the Conservation Act. Dispensation from the act is necessary if activities including e.g. a nitrogen source are considered established or extended and which may result in changing the § 3 areas. Moreover, it may be possible to prevent harmful activities if they constitute a threat to Natura 2000 habitats following § 19 f.

Agricultural policies and measures

In 2008 about 63 per cent of the total area of Denmark i.e. 27,330 km^2 was used for agriculture. This proportion is decreasing slightly. The primary agricultural sector produced 1.5 per cent of GDP in 2005, and has been in a steady decline since the 1960's. The adoption of intensive farming increased the average size of holdings from 16 ha in 1965 to about 55 ha in 2005, while the number of holdings decreased from about 200,000 to 46,000 during the same period. There was also an increase in the number of livestock (less cattle but more pigs), though the number of livestock units has been almost the same through these years (BLST, 2010).

According to the arable land Denmark has a regulation which set obligatory and fixed standards (a nitrogen quota) for the application of nitrogen from both livestock manure and chemical fertilizers. The nitrogen quota is set 10 per cent below the agronomic economical optimum. All farmers have to submit a fertilizer status account to the authorities every year. In order to control the information, the regulation also requires any company, who trades with fertilizers, to submit information on the annually delivery of nitrogen fertiliser to each farmer. To control the livestock production the authorities have a legal access to information from slaughterhouses and dairies about deliveries from each farm. In order to control the exchange between farmers, the farmers also are imposed to submit information about every exchange of nitrogen in fertilizer or manure between farmers.

Denmark also has legislation with fixed environmental standards regarding odour emission, ammonia emission, and surplus of phosphorus and leaching of nitrate, which should be met in connection with the approvals. Local authorities often set further demands with reference to the Habitat Directive or the Water Frame Directive, which in some cases have given long casework.

Impact assessment

Under the Planning Act the Danish Government has issued two orders, which implement the EU directives on EIA and SEA. All projects, plans and programmes that may have a significant effect on national and environmental values of national interest are subject to such assessments. Moreover, plans and projects in Natura 2000 areas are subject to an assessment regarding the habitats and species for which the areas are designated according to the rules for administration of the Natura 2000 sites and regarding deterioration or destruction of breeding sites or resting places of annex IV species set up in Executive Order No. 477, as amended (Ministry of Environment, 2004).

In addition to the general rules, livestock production units have since 2001 been subject to special legislation requiring impact assessment of nitrogen deposition to habitats. From 2001 to 2006 the impact assessment was carried out as part of the EIA screening (c. The EIA Directive). Since 2007 livestock production units with more than 15 animal units have been required to attain an environmental approval, whenever there are plans to establish, change or extend their production.

In addition to the impact assessment in relation to Natura 2000 the Danish legislation includes means to achieve a general reduction of nitrogen emission as part of the implementation of the IPPC and EIA directives. Beside that the environmental approvals of livestock production units are required to include the use of approved eco-efficient technologies (BAT). A more detailed description of the Danish impact assessment and regulation of livestock holdings is provided in Bjerregaard (in press.).

Discussion

In 2007 a new Environmental Approval Act for livestock farms was endorsed. With this act a set of criteria was established for securing that establishing, changing or extension of livestock holdings did not result in negative environmental impacts following the Habitat Directive and the act on Environmental Impact Assessment. As indicated earlier it is the expectation that most livestock holdings will be involved in this regulation in a 10 to 20 years time period because of the rapid development in the sector. This act contains both specific regulation in buffer zones around selected habitats as well as general reduction demands and BAT requirements. Moreover the municipalities have to carry out additional Natura 2000 impact assessment in cases were this is relevant.

The Green Growth agreement

The Danish government signed an agreement on Green Growth in 2009. The purpose of the agreement is to ensure that a high level of environmental, nature and climate protection goes hand in hand with modern and competitive agriculture and food industries. This is along-term plan defining environment and nature policies and the agriculture industry's growth condition. A total of DKK 13.5 billion is to be invested until 2015, which is about a 50 per cent increase in investments compared to previous initiatives. The agreement sets up new targets for general nitrogen depostion as well as regulation in relation to Natura 2000 habitats. In relation to sensitive Natura 2000 habitats the resulting target is a maximum total nitrogen contribution from each livestock unit of 0.2 to 0.7 kg N $ha^{-1}yr^{-1}$ depending on the number of livestock holdings in the particular area (Regeringen, 2009).

The wider countryside

The government intends to strengthen regulation of nitrogen deposition in the wider countryside, hence according to the Green Growth agreement the general regulations would be enhanced as well.

Thus BAT standard criteria for all holdings over 250 animal units and special BAT criteria for holdings of more than 500 animal units are being developed. For certain valuable and nitrogen sensitive nature areas outside Natura 2000 a permitted load from holdings would be up to a total load of 1 kg N $ha^{-1}yr^{-1}$ and for other nature the permitted load would be up to one additional kg N/ha. Moreover, Denmark has set up ceilings for emissions as obliged to by the Gothenburg protocol from 1999 and the NEC Directive (National Emission Ceilings).

Management

Studies have shown that large amounts of nitrogen accumulated in the soil on heath can be removed by sod cutting and other results show that hay mowing may remove as much as 40 to 180 kg N $ha^{-1}yr^{-1}$ (Damgaard *et al*., 2007). Removal is only applicable however on relatively level soils without stones, swampy patches, scattered woody plants and characteristic structures like tufts, tussocks, ant hills etc. For habitats such as raised bogs or decidious forests it is not an option. Such management interventions would probably be seen as a restoration approach or an additional effort in situations where reducing nitrogen deposition needs to be catalysed.

Challenges ahead

The Green Growth agreement is expected to limit the number of specific Natura 2000 impact assessments thus reducing the administrative burden on the part of the farmer as well as the municipality as well as being better at securing the sensitive Natura 2000 habitats. At the same time a strong structural development is going on and e.g. the number of farms is expected to be reduced by 50 per cent in the next 10 years. Thus because of the regulation there will be a tendency that the remaining farms will be located away from neighbours and vulnerable nature. As part of the air quality programme under the National Monitoring Programme (NOVANA) nitrogen deposition is monitored at local scale at various Natura 2000 habitat types around the country with specific reference to the Habitat Directive. The calculations include both wet and dry deposition in the 400 x 400 m grids i.e. the deposition on the targeted habitat types, which included heath, fens, meadows, dunes, raised bogs, deciduous forest and a few other habitat types to get information at as high a resolution as possible.

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7.5 Nitrogen deposition and Natura 2000 in Greece

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7.5.1 Introduction

The responsibility of the European countries towards the global community for the conservation of biodiversity is high and they have agreed to join efforts to conserve threatened species and habitats within their territories. In order to further promote their conservation, the Bern Convention on the conservation of European wildlife and natural habitats, the Birds Directive (79/409/EEC) and the Directive 92/43/EEC, commonly referred to as the "Habitats Directive", are the main tools at a European level.

Greece has a wide range of climate types, ranging from the semi-desert to the cold humid continental climate. The topography of the country is also complex. More than two third of the land is mountainous, there is a relatively high number of islands (Figure 7.8) and a lengthy shore line of about 15,000 km. These unique environmental features are reflected in the high number of plant and animal species, considerable number of which are endemic. Approximately 6,000 plant species have been identified and according to Legakis (2004), 23,130 land and freshwater animal species have been recorded in a variety of terrestrial and aquatic ecosystems. This diversity of ecosystems is crowded into a relatively small space (132,000 km^2).

Unfortunately this national heritage has not been adequately studied, evaluated and managed to date. The Habitats Directive aims at contributing to the preservation of biodiversity through the conservation and hopefully the restoration of the various types of natural habitats and species.

The identification of the Natura 2000 network in Greece has started in June 1994, with the execution of a project entitled "Inventory, Identification, Evaluation and Mapping of the Habitat types and Flora and Fauna species in Greece" (Dafis *et al*.1996).

7.5.2 Legal framework of protected areas and Natura 2000 Sites in Greece

A number of laws offer a direct or indirect protection of the Natura 2000 sites. Legislation is complex and covers a number of designated categories of protected areas in Greece. Most important legal framework arises from the forest legislation, the law for the environment and the Ramsar convention. According to forest legislation (L.D. 996/1971), a number of sites (or parts of them) have the status of 1) National Forest Parks, 2) Aesthetic Forests and 3) Natural Monuments. Designation categories defined in law for the environment (L. 1650/86) are: 1) Strict nature Reserves, 2) Nature Reserves, 3) National Parks, 4) Protected Natural Formations - Protected Landscapes, 5) Ecodevelopment Areas. Ten wetlands of international importance are designated under the Ramsar convention. The vast majority of the above areas are included in the Natura 2000 network. With regard to the Natura 2000 network, in Greece this is composed of 239 Sites of Community Importance (SCIs) and 163 Special Protection Areas (SPAs) according to the Birds Directive. Their surface area (excluding overlaps) comes to around 34.000 km^2 covering around 21 per cent of the land area of the country. The breakdown of the sites' areas, as presented in the EC Natura 2000 barometer, is shown in Table 7.1. These sites are of community interest and require the designation and proper management of special areas of conservation. Responsible for managing protected areas are management bodies (L. 2742/99).

Table 7.1: Sites of Community Importance and Special Protection Areas in Greece
European Commission Natura 2000 barometer, Natura 2000 newsletter no 26 (2)

	Number of sites	Total sites area (km²)	Terrestrial area (per cent)	Number of marine sites	Marine area (km²)
SCIs	239	27,641	16.4	102	5,998
SPAs	163	16,755	12.3	16	567

7.5.3 Threats to Natura 2000 Sites and nitrogen deposition in Greece

Various human activities might have adverse effects on valuable habitats and species. Human activity in Greece has resulted in three quarters of the wetlands having been destroyed in the past. Today, it is widely accepted that the main threats to Natura 2000 sites are: Forest fires, drainage and pollution of wetlands (eutrophication), road construction through sensitive ecosystems, overgrazing, illegal hunting and fishing, industrial pollution (water and air pollution), intensive agricultural practices and not regulated tourism. These threats have not been studied systematically and in relation to Natura 2000 sites in Greece. More specifically the nitrogen effects on these sites are largely unknown, although nitrogen deposition is a threat to biodiversity across large areas of Europe (CCE, 2008).

Research on nitrogen deposition by rain (Tables 7.2 and 7.3) has been conducted in urban areas of Greece (Figure 7.8).

Nitrate nitrogen deposition by rain was 1.2 to 5.9 kg $ha^{-1}yr^{-1}$, while the ammonia nitrogen was 1.5 to 11 kg $ha^{-1}yr^{-1}$. In recent years the highest deposition was measured in the area of Ptolemais city, because of the local lignite-burning plants operation. This N-deposition contributes to the fertilization of various terrestrial and aquatic ecosystems with unknown effects on species composition of the Natura 2000 sites in Greece. It has been reported that species composition of Greek grasslands was considerably affected by fertilizer application. N favours grasses and depresses legumes unless it is combined with P when a more balanced species composition is secured (Papanastasis and Koukoulakis, 1988). Addition of N increased community productivity and changed also species composition, especially in years when soil moisture was adequate (Mamolos *et al*., 1995). In a competition experiment the nitrophilous species *Bromus sterilis* was able to increase growth at increasing N-fertilizer level, at the expense of other species (Tsiouris and Marshall, 1998).

The wetlands in Greece are also threatened by various human activities taking place either in the water bodies or on their watersheds. Various agricultural activities e.g. application of agrochemicals, ploughing, burning plant residues etc., which take place on the watersheds are considered as one reason for non point pollution of the wetlands. The NO_3-N concentrations of the runoff water from experiment sites in the watersheds of two Ramsar wetlands (Prespa and Koronia) were higher than the NO_3-N concentrations in rain and stream water samples taken from the same watersheds (Tsiouris *et al*., 2002a and Tsiouris *et al*., 2002b).

7.5.4 Management tools

As stated in the 2nd national report on the implementation of the Habitats Directive (article 17 report), there are management plans and management bodies for some of the Natura 2000 sites. One comprehensive management plan has been adopted for the National Park of Shinias, but several others are in preparation. More particularly, according to the above report, there are 95 sites for which comprehensive management plans are in preparation and in 48 Natura 2000 sites management bodies have been established. In 203 sites there is not a comprehensive management plan, but nature conservation objectives have been included in the relevant territorial planning

Figure 7.8: Five cities of Greece, where nitrogen deposition by rain was studied.

Table 7.2: Rain Nitrogen (kg N $ha^{-1}yr^{-1}$) in two sites of Thessaloniki (East and West).
(Mourkides et al., 1981).

Site	Form	1977	1978	1979	1980
East	NO_3-N	1.24	1.65	2.69	2.12
West[1]	NO_3-N	1.38	4.16	3.14	2.34
East	NH_4-N	1.97	4.56	5.00	-
West	NH_4-N	4.81	11.13	5.67	-

1 West is industrial site.

Table 7.3: Nitrogen deposition (kg N $ha^{-1}yr^{-1}$) by rain in five cities of Greece.
(Tsikritsis, 2006).

Cities	Period	mm	NO_3-N	NH_4-N
Athens	1987-8	377	1.51	1.47
Patras	2000-1	678	1.83	1.56
Larissa	2001-2	424	1.91	1.99
Thessanloniki	1993-4	458	1.74	3.71
Ptolemais	1986-7	493	5.87	3.25

instruments as for example, designation for wildlife refugee, forest management plan, management of grazing etc.

In 72 sites, nature conservation objectives are not defined in a territorial planning instrument (nor in a comprehensive management plan), but other management instruments have been put in place as for example, application of agrienvironmental measures, management project through operational project "Environment", application of Life-Nature project etc.

7.5.5 Conservation measures

As stated in the 2nd national report on the implementation of the Habitats Directive (article 17 report), in Greece, the main statutory measure for the conservation of the Natura 2000 sites is their designation according to the existing national legislation. The core areas of National Forest Parks and the Natural Monuments are considered strictly protected and various activities like excavation, industrial activities, tree felling and destruction of plants, grazing and every construction in general except for those favouring nature conservation are prohibited. In the peripheral zones of National Forest Parks and in Aesthetic Forests, activities are regulated by the competent Forest Services, aiming to nature conservation. Wildlife Refugees (L. 2367/98), aim to the protection of the areas for feeding, wintering, breeding and rescuing of the species of wild fauna and flora. Within Wildlife Refugees, hunting, caption of species for reasons other than research, destruction of vegetated areas, sand removal, drainage of marshes, pollution and inclusion of the area in town planning is prohibited. According to the Law 1650/86 for the protection of the environment, for the designation of protected areas, a Specific Environmental Study (SES) is required. After its completion, the SES is approved and then, together with the draft legislation text for the designation of the area, it is opened to the public. Comments are incorporated and the legislative text is signed by the competent Ministers. If the draft legal text is a Presidential Decree, then it has to be checked by the High Court and then signed by the President of Democracy.

Under the designation act of each area, a number of restrictions for works and activities are determined; among them, restrictions and prohibitions in land use, in building and cutting of land into smaller pieces, in constructions, in implementation of agricultural, fishing, stock raising activities etc.

In general, in Strict Nature Reserves all activities are prohibited, except research and works for nature conservation. In Nature Reserves only research and some traditional activities are allowed.

Protection and management of the natural environment lies within a number of public services with overlapping responsibilities. Protected areas designated according to L. 1650/86 can be managed by various management schemes. The scheme applied till now regards the establishment of Management Bodies (L. 2742/99) consisted by a Managing Board of representatives of central ministries, regional, prefectural and local authorities, local stakeholders, NGOs and scientists. Managing Boards must be supported by scientific, technical and administrational personnel. The existent 27 management bodies have not yet in all cases engaged all the personnel needed for their proper operation.

Through spatial planning (Regional Spatial Plans, Specific Spatial Plan for the Renewable Sources of Energy), specific provisions have also been issued for the sites of the Natura 2000 network.

Management measures have also been applied by beneficiaries through projects supported at EC and at national level. Most important of them are Life-Nature projects.

Applied agro-environmental measures concern mainly organic farming, organic livestock farming, protection of nitrate vulnerable zones, protection of wetlands, extensive livestock farming, protection of traditional orchards, maintenance of local endangered breeds, maintenance of plant resources under threat of genetic erosion, promotion of farm practices for the protection of wild life, long-term set-aside, conversion of arable land to extensive pastures and preservation of hedgerows and terraces.

At cultivated areas, Codes of Good Agricultural Practice were implemented in all agro-environmental schemes, whereas Cross Compliance Requirements and additional measures, in accordance with the regulation 1782/03, are applied. In the freshwater environment, L. 1740/87 provides for the issuing of Presidential Decrees for the regulation of fisheries in inland waters. In general, regulations and restrictions are valid for the protection not only of fish but also of lobsters, shrimps, mussels, molluscs, shells etc. Coralligenous are protected through regulation of exploitation whereas fishing with trawlers is prohibited above posidonia meadows of Natura 2000. Midwater otter trawls and pelagic pair trawl are not allowed in Greece, whereas fishing with beach seines and trawlers is regulated (several prohibitions exist at local level as regards the distance from the coast and the period of fishing). Drift nets are prohibited since 1993 according to P.D. 40/93. Through L. 3409/05, recreational diving is regulated. Enforcing of legislation is monitored by the competent services of the Ministry of Merchant Marine.

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7.6 Nitrogen deposition and Natura 2000 in Portugal

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Summary

- The aim of this paper is to provide a general perspective on the current status of nitrogen related issues in Portugal. The focus is on the current science and practice in mainland Portugal.
- It is concluded that for the Natura 2000 network and Mediterranean type ecosystems, the current monitoring of atmospheric ammonia in Portugal is clearly insufficient for a suitable protection of Natura 2000 biodiversity.
- Three different integrative ecological indicators are considered for the assessment of the impact of N deposition on biodiversity: functional lichen diversity for evaluating the impact of atmospheric NH_3 in sensitive ecosystems, N in lichens as a first level for regional risk of N deposition impact and N in litter as the second level for assessing ecosystem functional response of N deposition.
- Considering the insufficient number of national monitoring stations, the Mediterranean landscape's peculiarities together with the N trade-offs, we recommend the use of ecological integrative indicators as innovative tools for risk analysis of N deposition, as well as assessment of biodiversity shifts at ecosystem level.

7.6.1 Introduction

The spatial resolution of NO_x and NH_3 measurements in Portugal

In Portugal there are two main institutions that may deal with nitrogen (N) emissions and deposition compliances and their effects on biodiversity that are, respectively: the Environmental Portuguese Agency-APA (www.apambiente.pt) and the Institute for Nature and Biodiversity Conservation-ICNB (www.icnb.pt).

APA is the national entity responsible for the overall coordination of the Portuguese inventory of air pollutants emissions. Air emission inventories in Portugal were initiated in 1989/1990 and first estimates of NO_x were made at this time. Only in 1992, under the CORINAIR90 and UNECE/EMEP report was NH_3 first included in the inventory. At present, emission factors for NO_x and NH_3 are determined from the available set of algorithms reported in EMEP/CORINAIR handbook (EMEP, 2002).

In Portugal there are approximately 70 air quality monitoring stations measuring NO_x permanently (http://www.qualar.org), located in urban, suburban and rural areas. Concerning this pollutant both in space and time the level of information is quite detailed.

The Convention on Long-Range Transboundary Air Pollution (LRTAP), requires NH_3 emissions to be reported in a spatial pattern following a 50km x 50km grid. However, at present, APA present the data according to the council level, which is more detailed. Despite this effort, the level of information at spatial dimension is still not adequate to characterise deposition at local scale, most effects occurring at less than 500 m from the source (Pinho *et al*., 2009). Moreover, there are no NH_3 monitoring stations at the national level and there are only two NH_4^+ monitoring stations in the country, one in the north and other in the south (http://www.meteo.pt/pt/ambiente/atmosfera/). Thus, the information concerning the air quality and the deposition of NH_3 is only based on statistical information and air deposition models not validated with NH_3 measurement. As we can see in Figure 7.9 most part of the NH_3 emissions are related to agricultural activities (41.5 per cent) or livestock production (39.3 per cent). Knowing that most of Portugal's 2000 Natura Network is located in rural areas with high agriculture and livestock activities, the assessment of the impact of NH_3 on biodiversity and ecosystem function is of high importance for Portugal. Moreover, the Global Strategy for Plant Conservation, that Portugal has also signed, emphasizes the need for capacity-building in order to enable the implementation of the targets for 2010 using a flexible framework within which national and regional actions are developed. Thus, there is a need to take the targets into consideration for monitoring and assessing progress of N deposition particularly on Natura 2000 sites.

7.6.2 Aims and objectives

- The aim of this paper is to evaluate the situation of Portugal in terms of monitoring assessments of N deposition particularly on Natura 2000 areas.
- Consideration of the use of integrative ecological indicators that reflect the NH_3 atmospheric deposition and the ecosystem response to N increase.
- Specifically scientifically based strategic and practical tools, to assess the potential for shifts in biodiversity in response to N deposition are considered to fulfill Target 3 of the Global Strategy of Plant Conservation within the Natura 2000 sites.

7.6.3 Results and discussion

The Portuguese climate and biogeography

Portugal is on the edge between Mediterranean and Atlantic-eurosiberian biogeographic regions. It presents a high patchiness of natural habitats, and it is unique in the Mediterranean context because of the Atlantic influence that produces higher levels of precipitation, and therefore the climate varies between humid and arid Mediterranean within a small area. This climate is associated with poor or very poor nutrient soils (Cruz *et al*., 2003; Cruz *et al*., 2008) some of them with low water retention. These environmental conditions have a great effect on vegetation dynamics and landscapes (Figure 7.10). In Portugal, natural conditions together with the long history of land use has produced a landscape dominated by thin, acid or slightly acid and oligotrophic soils, normally with an extensive woodland for wood production and agriculture use. This combination of ecological factors and of anthropogenic perturbation patterns led to a heterogeneous landscape.

The Portuguese Mediterranean type ecosystems

Most of Mediterranean Portuguese ecosystems are part of a mosaic-type landscape, shaped by diverse geomorphologic, climatic and human-induced factors (Blondel and Aronson, 1999; Palahi *et al*., 2008). In fact, human influence shaped most of the Mediterranean ecosystems over centuries of traditional land-use practices. For example, Montado, the dominant landscape in

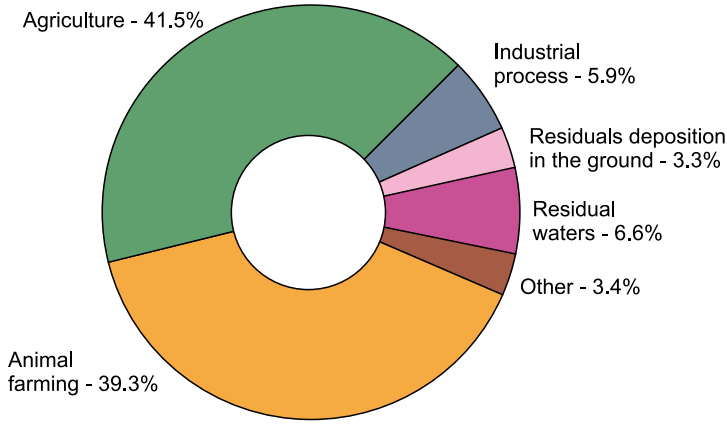

Figure 7.9: Relative proportion (per cent) of NH_3 emissions in 2005 following each activity sector (source APA, 2008).

the south is a unique agro-silvopastoral system found only in the Iberian Peninsula dominated by evergreen tree-species (cork *Quercus suber* and holm *Q. rotundifolia* oaks). This multi-use forest system combines, in a single space, forest harvesting, extensive livestock husbandry and intermittent cereal cultivation, together with the provision of mushrooms, aromatic plants, game and bees. This long history of human-nature interactions in a region identified as one of the 25 world hotspots of biodiversity (Mediterranean basin; see Myers *et al*., 2000) allowed species, many of which endemic and therefore of high conservation value, to co-evolve under traditional management practices. In modern times however this system faces degradation due to different type of threats, namely, intensive and extensive agriculture, agriculture abandonment, fires, different types of forest production, invasive species. All these have, in the short- and long-term a negative influence on biodiversity, threatening the extinction of many species and habitats.

Several authors (e.g. Sutherland *et al*., 2006) identified a large number of ecological questions with policy relevance related to nature conservation in humanized landscapes. These include the impact of farming, urban development, pollution, and conservation strategies. An enrichment in N of vegetation tissues (Pocewicz *et al*., 2007) and a change towards more nitrophytic flora (Willi *et al*., 2005) resulting from an increase in nitrogen deposition, mainly from ammonia emitted by farming activities (EPER, 2004; Galloway *et al*., 2003), is related to biodiversity loss (Bobbink *et al*., 2010; Phoenix *et al*., 2006; Suding *et al*., 2005). In fact, nitrogen deposition is considered not only a major threat to global biodiversity but also one of those threats that are expected to increase worldwide (SCBD, 2006).

Nevertheless, the impact of nitrogen on biodiversity is not a priority subject for our conservation biology governmental authority, ICNB (www.icnb.pt). Thus, N deposition is never considered as a threat/pressure in habitats status reporting, or as a factor for conservation management. Nevertheless, some protected Natura 2000 sites are located in areas where the NH_3 deposition is between 1 and 1.6 ton/km^2 (Figure 7.11). Despite the weak spatial resolution of the NH_3 emissions that this level of information can provide, it is important to notice that the Natura 2000 sites that are located in areas with high NH_3 deposition should be assessed as a priority for the impact on the biodiversity. Of those, the most problematic are the ones located near large urban areas or in the west central part of the country, where intensive agriculture practices take place (Figure 7.11). It is also interesting that low intensive agriculture practices and/or extensive

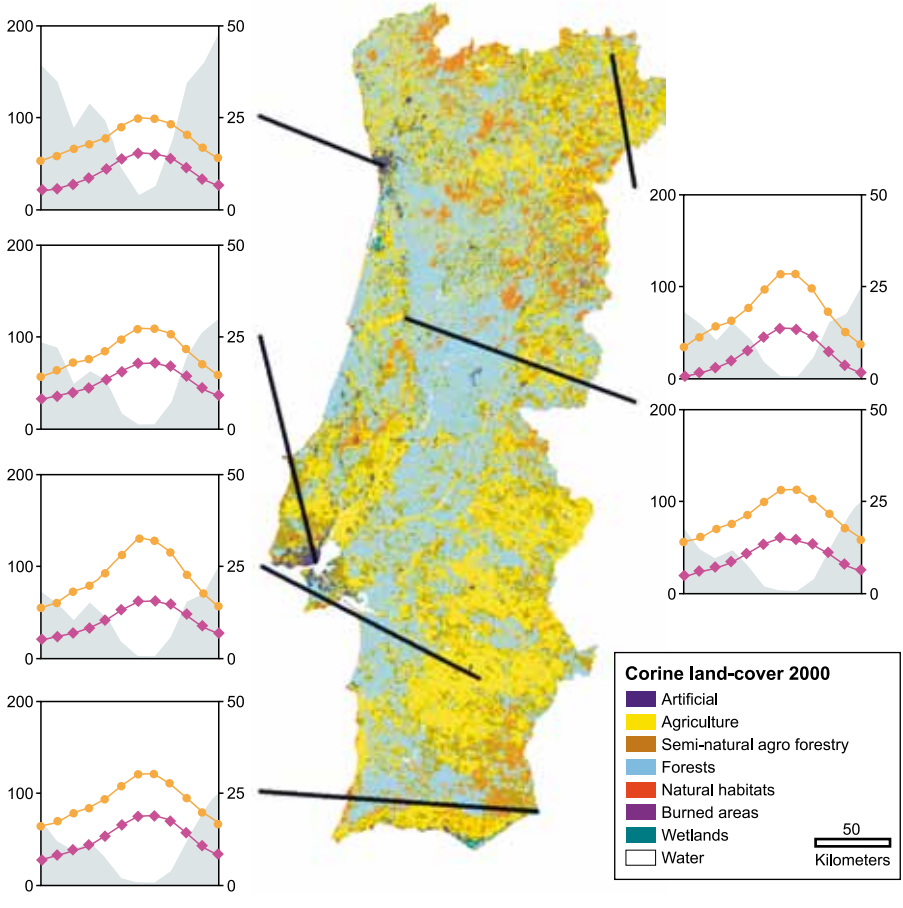

Figure 7.10: Map of the distribution of land-cover types in Portugal, adapted from Corine Land-Cover 2000. Note that the class "forest" includes Pinus and Eucalyptus plantations, oak forest as well as cork and holm-oak woodlands. Climatograms are shown for different areas in continental Portugal. Lower axis are months (from January do December), left axis monthly total precipitation (mm) represented by the filled shape, right axis monthly temperature (ºC) using averages of the maximums (triangles) and minimums (circles). Values are averages from 1971 to 2000, source IM (2009).

livestock production, associated with Montado ecosystems, that occur in the south part of the country, lead to medium levels of NH_3 emissions.

Use of lichens to determine critical areas for monitoring N impact N ecosystems

Because the available information on NH_3 emission is clearly at an insufficient spatial resolution to allow its use for assessing the impact of N in biodiversity, another approach must be considered. In order to assess the range of effects of NH_3 in natural ecosystems, that can be used for effective NH_3 mitigation policies (Dragosits *et al*., 2006) one can rely on two distinct approaches: (i) direct measurements of atmospheric NH_3 concentrations, which provide an estimate of dry NH_3-N deposition, but require intensive and costly operations; (ii) monitoring of effects on the biotic component. The latter approach should be carried out using groups of biota that are more sensitive to the pollutant of interest. Lichens have been reported as the most

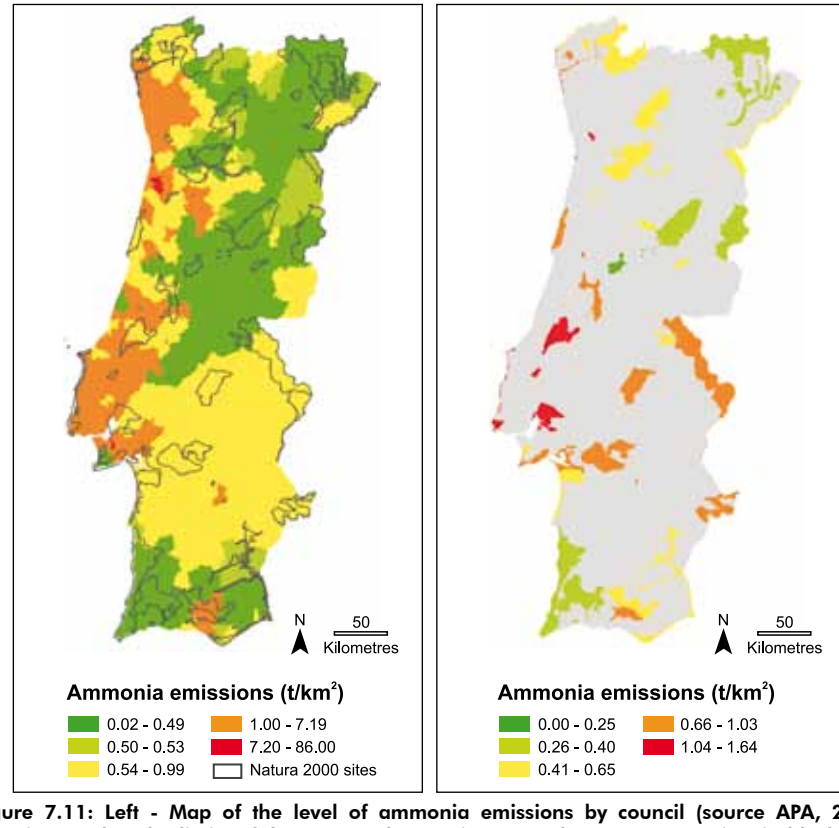

Figure 7.11: Left - Map of the level of ammonia emissions by council (source APA, 2008) superimposed to the limits of the protected areas in Portugal, Natura 2000 sites in black line. Right – Average value of the NH_3 emissions for the Natura 2000 network in Portugal based on NH_3 emissions information at the council level

sensitive group to NH_3 emissions (van Herk, 1999; Wolseley *et al*., 2006). Lichens are symbiotic organisms widely used as biomonitors of environmental changes (Pinho *et al*., 2004; 2008a, b). In fact, the information obtained from lichens compliments the information collected from chemical sampling, because lichens provide a biological perspective, integrated in the long-term on the effects of N. By examining changes in lichen communities, specifically by using lichen indicators based on nitrogen-tolerance, an estimate of atmospheric NH_3 critical levels was made for Portugal in the Montado ecosystem under Mediterranean climate (Pinho *et al*., 2009). The critical level found was between 1 and 2 $\mu g/m^3$, much lower than previous limits (eight $\mu g/m^3$) but in accordance with the concentrations found in other works using lichens (Cape *et al*., 2009; Wolseley *et al*., 2006).

However, although lichen diversity is a suitable tool for determining if ecosystems are affected by N pollution, its use at a landscape scale, e.g. within Natura 2000 areas, may be hampered by the fact that lichen diversity may respond to other environmental factors (Pinho *et al*., 2008b). Therefore, how to select critical areas for monitoring N polluted areas? Pinho *et al*., (this book) provided a practical method for selecting critical areas for monitoring the impact of NH_3 in plant biodiversity within Natura 2000 sites. There, it was shown that the concentration

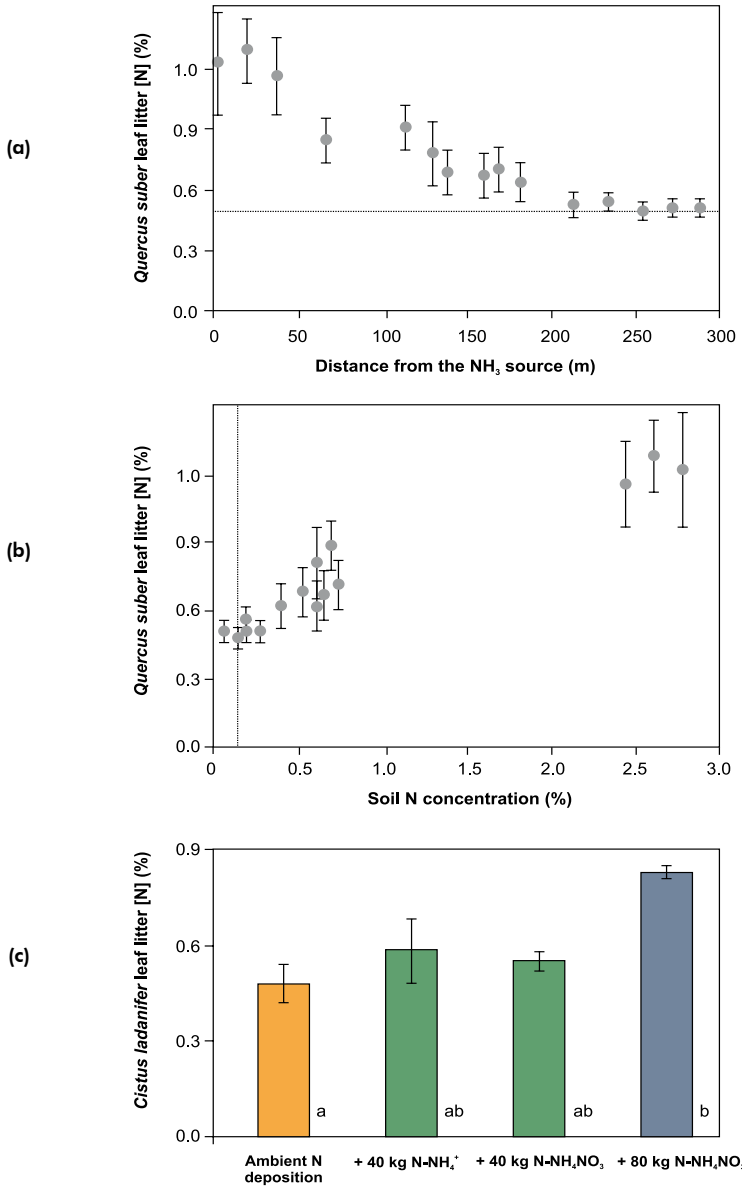

Figure 7.12a,b,c: Changes in leaf litter N concentration in response to increased N availability in two Natura 2000 sites in Portugal that correspond to different Mediterranean-type ecosystems: a) relation between leaf litter N (mainly Quercus suber leaves – collected in the four seasons in 2008) concentration and distance to a source point of ammonia (barn with 200 cows, Pinho *et al.*, 2009) in a cork oak system (values represent mean ± sd; n= 4 sampling points); b) relation between leaf litter N concentration (collected in summer 2008), and soil N concentration in the same cork oak system as in a); c) relation between leaf litter N concentration from Cistus ladanifer and N additions beginning in 2007 in a semi-natural Mediterranean Maquis (bars represent mean values ± se; N = 3 experimental plots per treatment).

of N in lichens was very significantly related to agriculture land-use and not to industrial and urban areas thus showing that N concentration in lichens is most probably reflecting the NH_3 emissions. In this way the authors proposed to apply the N concentration in lichens as a detailed ecological indicator for fulfilling the objective of selecting critical areas for the impact of NH_3 on biodiversity. The authors applied this indicator to two Portuguese Natura 2000 sites by mapping N concentration in lichens. By doing so, they select the critical areas for the assessment of the impact of atmospheric NH_3 deposition on plant diversity in Mediterranean Natura 2000 sites.

How can we monitor increased N availability in ecosystems

Mediterranean-type ecosystems are expected to be very responsive to increased N availability as increased N deposition constitutes a significant increase in the availability of a nutrient that limits the productivity of these systems (Cruz *et al*., 2003).

Dias *et al*., (this volume) provided evidence that Mediterranean-type ecosystems are highly N responsive, and that changes can be seen after one year of N additions. Increasing N availability leads to increased below and aboveground diversity (richness and evenness) and creates new and distinct seasonal patterns of soil N availability, which translates into changes in the nitrogen recycling in the ecosystem. Higher nitrogen availabilities change the chemical composition of plant and microbial biomass, affecting the remobilization processes in the plant. Therefore the litter produced under high nitrogen availability is enriched in N. Two parallel studies that are being conducted in distinct Natura 2000 habitats showed that N concentration of litter from the dominant plant species could be a good indicator of the N status of the site (Figure 7.12). One site is a cork oak field with a source point of ammonia (Pinho *et al*., 2009). Litter mainly corresponds to cork oak (*Quercus suber*) leaves. Litter N concentration decreased inversely with distance to the ammonia source, which was an important nitrogen input to the system (Figure 7.12a), and increased with increasing soil N concentrations (Figure 7.12b). The other site (PTCON0010 Arrábida/Espichel) is a Mediterranean Maquis dominated by Cistus ladanifer and has been submitted to N-manipulation since 2007 (Dias *et al*., this volume). Litter N concentration's dependence on the added N dose was evident (Figure 7.12c). Adding 40 Kg N $ha^{-1}yr^{-1}$ did not significantly affect the nitrogen concentration of the litter (relatively to the control), but the adding 80 Kg N $ha^{-1}yr^{-1}$ had a significant effect.

In Mediterranean ecosystems the high N use efficiency is related with a great nutrient remobilization capacity from old to new leaves. This decreases dramatically the nitrogen content of the litter and constrains decomposition, consequently altering the structure and activity of the microbial community. An alleviation of the nitrogen limitation to plant growth allows plants to increase their N content and to afford a decrease in internal N turnover. This changes litter quality, as well its decomposition rate and, consequently, the structure and activity of the microbial community. These small adjustments at individual and community level take place in different time scales. Internal resources and plant-microbe interactions may be some of the adjustments that induce changes in species composition in a larger time scale. Therefore, monitoring changes in litter N concentration may function as an integrative ecological parameter of the Mediterranean-type ecosystem's responses to high N inputs. For these systems, N concentration in litter can thus be considered a more integrative indicator that foliar N concentration, acting as a tool for evaluating N-induced biodiversity shifts.

7.6.4 Conclusions

- In the framework of Natura 2000 network and Mediterranean type ecosystems, we conclude that the current monitoring of atmospheric ammonia in Portugal is clearly insufficient for a suitable protection of biodiversity on Natura 2000 sites.

- Here we make use of an integrated framework for assessing Mediterranean Ecosystems responses to N availability: (i) nitrogen concentration in lichens was shown to be related to agriculture areas, and could therefore be used to map the areas at greater risk from N-deposition; (ii) in risk areas, lichen functional-diversity can be used to establish the ecosystem critical level for ammonia and (iii) by measuring N concentration on litter we could integrate the balance between the two compartments of the ecosystem, the below- and aboveground.
- Considering the insufficient number of national monitoring stations, the Mediterranean landscape's peculiarities together with the N trade-offs, we recommend the use of ecological integrative indicators as innovative tools for risk analysis of N deposition, as well as assessment of biodiversity shifts at ecosystem level.

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7.7 Challenges to reducing the threat of nitrogen deposition to the Natura 2000 network across the UK and Europe

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Summary

- While emissions of nitrogen compounds (oxides of nitrogen and ammonia) have decreased in the UK, there is evidence of only a small reduction in total nitrogen deposition over the last 20 years.

- Even with projected emission reductions factored in, critical loads for nitrogen deposition will still be exceeded at almost half of the UK's sensitive habitats in 2020. This clearly demonstrates the need for significant additional reduction in the emission of nitrogen compounds.

- The Habitats Directive requires a very high level of protection for habitats across Europe. However, it is widely accepted that presently nitrogen deposition impacts are not addressed consistently in relation to the requirements and objectives of the Directive.

- A number of countries have used critical load assessments to inform their reporting on the conservation status of habitats listed under the Directive. However, it is also recognised that there needs to be a much more robust and consistent approach taken to reporting nitrogen deposition impacts, linking critical loads and levels approaches to the protection of biodiversity.

- Where nitrogen deposition is a "pressure" or "threat" to the conservation status of habitats it should be identified in reporting by Member States under Article 17 of the Habitats Directive. The use of critical loads and levels should form an integral part of this assessment, to inform the determination of threat.

- While at EU level there is a high level objective to achieve no exceedance of critical loads, there is no timetable or trajectory in place to help deliver this. It is therefore vital that a number of high level and local air pollution initiatives are coordinated in a targeted manner to help achieve this.

- The UK has rigorously applied the requirements of Article 6.3 of the Habitats Directive in relation to nitrogen deposition. However, ambiguity in interpreting and defining an "adverse affect" remains an issue.

- There is a great deal of legislation at an EU level intended to offer a high level of environmental protection. This could be much better integrated at a UK level to deliver outcomes to significantly reduce the impacts of nitrogen deposition on habitats and species.

- The UK Air Quality Strategy (UK AQS) needs to be strengthened in order to deliver the requisite provision for the protection of sensitive habitats. At present ammonia is not covered by the UK AQS and should be incorporated and supported by a Nitrogen or Ammonia Strategy.

Background

This paper focuses on the issues relating to atmospheric nitrogen pollution and its threat to biodiversity. The main emphasis concerns impacts on Natura 2000 sites relating to the situation within the UK, but set in a European perspective.

A brief introduction to current and future air pollution threat is provided. Details of how nitrogen pollution is being addressed in relation to the protection of Natura 2000 sites are then discussed. Examination of how two regulated sectors (power stations and intensive livestock) provide reflections of how we tackled this in the UK and the issues still remaining. Finally, consideration is given to the various policy drivers that exist within the EU and UK to tackle nitrogen pollution. Suggestions are made on how existing measures could be better integrated and used to provide greater protection. The chapter does not intend to provide a comprehensive summary of the entire process, but draws upon the approach we adopted, with observations provided on the strengths and weaknesses encountered. The views expressed are therefore those of the author rather than the organisations detailed.

7.7.1 Introduction: Global and European context

Over the past 40 years the world population has more than doubled from approximately three billion to over six billion currently with projections for a global world population of over nine billion by 2050 (UNFPA, 2008). As a result of emissions arising from food production and combustion activities, global levels of nitrogen pollution will continue to rise. For the first time,

man made emissions of nitrogen compounds are now on a level comparable to, or exceeding, the releases from natural sources. This means the global pool of "available" nitrogen has doubled in less than a century (Galloway *et al*., 2008). Emissions of nitrogen can have local impacts (e.g. close to conurbations or intensive livestock production), and can also be carried long distances and contribute to transboundary impacts away from the point of origin. The result is that many sensitive ecosystems are exposed to rates of nitrogen deposition much larger than sustainable limits.

Emissions of nitrogen pollution are considered to pose a significant threat to sensitive habitats across Europe. An assessment method using "critical loads" is well established and has allowed us to report risk in an agreed and consistent manner. Critical loads are used to inform EU air pollution policy development, for example under the National Emissions Ceiling Directive (NECD, 2001), as well as the UNECE Convention on Long Range Transboundary Air pollution (LRTAP), incorporating the Gothenburg Protocol. A substantial area of semi-natural habitat in the UK exceeds its critical load for nitrogen and will continue to do so in 2020 (Hall *et al.*, 2006). The critical loads approach provides a very useful tool to support air pollution policy options. However, it tells us little about the areas on the ground, or the ecological interest of these areas, where this impact is predicted to occur.

More recently attempts have been made to use the critical loads approach in a more targeted manner to better understand the environmental outcomes of predicted exceedances. For example "Nitrogen Deposition" has been listed as a global threat to biodiversity and an indicator by the Convention on Biological Diversity (CBD). More recently, critical load Exceedance for nitrogen being agreed as the indicator as part of the Streamlining European 2010 Biodiversity Indicators (SEBI, 2010) programme (EEA, 2007).The critical loads approach allows the cross-over of an air pollution indicator, *per se* (i.e. a pressure) into the potential effects on biodiversity (i.e. and impact). This provides an opportunity to better understand the impacts of nitrogen deposition on biodiversity and to respond to emission reduction considerations in a much more informed manner. Although critical loads originated in Europe they are now more widely used, for example in the United States and parts of Asia.

7.7.2 UK air pollution trends and forecasts in the European context

By 2020 emissions of sulphur dioxide across Europe are predicted to have been reduced by approximately 90 per cent from 1980 levels. Emission of will have fallen by about 70 per cent from their peak around 1990. However, reductions in ammonia (NH_3) emission are predicted to be much more modest with only an estimated fall in European emissions of 40 per cent, by 2020 from 1990 levels predicted, unless further NH_3 emission control is implemented (ROTAP, 2011).

In the UK sulphur emissions have fallen by over 90 per cent from their peak in the 1970s. This is predominately down to a decline in heavy industry, sulphur reduction in vehicle fuel and the use of sulphur abatement on some power stations. There has also been a significant increase in gas fired power generation and a consequent reduction in coal use overall (despite increasing use of coal in the last few years).

The fall in NO_x emissions are much less pronounced, in comparison to SO_2, with a reduction of only 50 per cent since 1970. There are two main reasons for this. Firstly, the largest emission sector is transport. The EU has been tightening emission standards on new vehicles through various phased Euro standards. However, the "lab based" theoretical improvements have not translated into the real world situation in the UK. Higher vehicles number on the UK roads and the level of congestion means that the cars are performing worse in terms of national emissions than had been calculated.

The second reason for the lower fall in emissions in the UK relates to a lack of control on the power sector, the second largest source of emission. There has been a repeated failure to retrofit existing power plant with reduction technology (such as Selective Catalytic Reduction –SCR) used elsewhere in Europe) and apply this technology to all new power stations. This means that the relative contribution of power station , as a percentage of the total emission budget, has increased from about 19 per cent in 1999 to about 28 per cent today (NAEI, 2009). As a result of the UK is only likely to narrowly achieve its 2010 NEC Directive ceiling for NO_x.

Ammonia (NH_3) emissions have fallen even less than NO_x with only a 20% decrease since 1990 (the earliest date from which reliable NH_3 inventories exist in the UK). The main reason for this reduction is a gradual decline in animal numbers and in total use of mineral fertilizers. Although the Directive on Integrated Pollution Prevention and Control (IPPC, 96/61/EC) has applied to large pig and poultry farms, in the UK, since 2007, this is estimated to have had little impact on overall UK NH_3 emissions. The reason is that IPPC only applies to a relatively small fraction of the UK NH_3 source (cattle, fertilizers, small farms, other sources etc are not included). Until now, there has been little explicit action to implement the discretionary requirements of the Gothenburg Protocol (as listed in the Gothenburg Protocol Annex IX) to use low NH_3 emission methods in the cattle or fertilizer sectors.

Concentrations of sulphur dioxide in ambient air have fallen to levels where they no longer pose a threat to ecosystems. Generally this is also true for concentrations of NO_x, although there are some notable exceptions, for example close to major roads (Defra, 2007) and developments such as airports.

Despite the 50 per cent reduction in NO_x emissions, measurement of total nitrogen deposition (both oxidised and reduced) has not reduced significantly over the past 20 years, remaining at approximately 400kT pa throughout (ROTAP, 2011). Over the UK, concentrations of NH_3 have changed little over the past 10 years, with the exception of localised variability. Over the past 20 years the proportion of NH_3 to total nitrogen deposition has increased from 45-55 per cent (ROTAP, 2011).

At present, in the UK, approximately 60 per cent of all sensitive habitat area exceeds their critical load for nutrient nitrogen deposition (ROTAP, 2011). This figure will only decrease to approximately 50 per cent by 2020 unless further and substantial cuts are made in the emissions of NH_3 and NO_x.

7.7.3 The Habitats Directive in relation to nitrogen deposition

The Habitats Directive (92/43/EEC) provides a cornerstone for European nature conservation policy. It promotes the maintenance of biodiversity and requires Member States to maintain or restore the threatened natural habitats and wild species listed in the Directive at "favourable conservation status", introducing robust protection for those habitats and species of European importance.

Habitats Directive- Article 17

Every six years, Member States must report on the implementation of the Directive. Article 17 requires Member States to make an assessment of the conservation status of all relevant habitats and species listed in the annexes of the Directive. The 2006 reports include a list of "pressures" to the structure and functions of habitats or "threats" to future prospects. However, at present there is no category specifically for nitrogen deposition as there is under the overarching Convention on Biological Diversity or the SEBI 2010. So while nitrogen deposition is a well known and accepted pressure and threat, Member States were unable to report it explicitly during the 2007 Article 17 reporting round. This shortfall has been recognised and should be addressed by the

next reporting round. Examination of how Member States dealt with reporting under Article 17 was a key consideration of the COST 729 workshop and is reported elsewhere in this publication (Whitfield *et al*., this volume).

A number of Member States have annotated their reports with explicit reference to nitrogen deposition and used critical load exceedance as a method to assess air pollution or eutrophication as "current pressure" or "future threat". However, it is clear from the workshop that some countries such as Portugal and Austria have yet to recognise N deposition as a national issue, despite concerns of their scientists and the critical load maps showing exceedance (e.g. see Martins-Loução et al., and Dirnböck, this volume). This contrasts sharply with the position in the UK, where air pollution (including nitrogen deposition) was listed as a pressure to "the current structures and functions" or a "threat to future prospects" for 53 out of 87 Annex I habitats. Quite clearly there is a need to identify nitrogen deposition as a pressure or threat in reporting for all Member States under Article 17. It is recommended that critical loads and levels should form an integral part of this assessment (Whitfield *et al.*, this volume). This should recognise the need to better understand the consequences of critical load exceedance on those habitats involved.

As well as improving the reporting of nitrogen deposition impacts on conservation status for the Habitats Directive, there is obviously the need to minimise the nitrogen risk, in the most efficient and targeted manner at EU, national and local levels. Because of the risk that nitrogen deposition poses across Europe air pollution policy should be targeted to ensure that EU Member States can meet the Habitats Directive objectives.

Habitats Directive – Article 6.3

The Habitats Directive provides for a very high level of protection for the Natura 2000 network. It does this by requiring a considered and precautionary approach to allowing or authorising "plans or projects" that may have a significant effect on a site. Article 6.3 of the Directive provides a requirement under which plans or projects may only be permitted if it can be demonstrated that they will have no adverse affect on the integrity of a Natura 2000 site.

We have already recognised that nitrogen deposition is a significant threat to many Natura 2000 sites across the EU. Indeed from 2010, it is estimated that N deposition threatens the long term viability of about 70 per cent of the EU 27 natural area (CCE, 2008). Therefore, at the workshop it was very interesting to see how Article 6.3 was being applied across the EU (Bealey *et al*., this volume) to see if the required precautionary approach was being applied in a robust and consistent manner. It became apparent that there are major issues on how Member States interpret the provisions of Article 6.3, which affect the degree of protection afforded to the Natura 2000 sites.

In light of the workshop findings, it is worth exploring the wording of Article 6.3, to see the key areas where the environmental outcome may be influenced by interpretation of the text (as highlighted in bold below):

> *Article 6.3 – Any plan or project not directly connected with or necessary to the management of the site but* ***likely to have a significant effect*** *thereon, either individually or in combination with other plans or projects, shall be* ***subject to appropriate assessment*** *of its implications for the site in view of the site's conservation objectives. In the light of the conclusions of the assessment of the implications for the site and subject to the provisions of paragraph 4, the competent national authorities shall agree to the plan or project only after having ascertained that it* ***will not adversely affect the integrity of the site*** *concerned and, if appropriate, after having obtained the opinion of the general public.*

According to current practice, each Member State is left to define what they view to be a ***"significant effect"***. In the UK the term significant has caused some confusion and has been adopted to mean a 'process contribution' (of a plan or project) that is clearly identifiable at a site level (rather than the emission itself causing a significant effect). Pre-existing criteria, already used in UK pollution legislation, were adopted and once a plan or project was screened as being identifiable (significant) a further examination is required to gauge its impact (degree of significance) via an "***appropriate assessment***". This assessment was undertaken "in view of that site's conservation objectives". As we have seen, not all Member States have identified nitrogen deposition as a major threat and of those that have, (e.g. the U.K.) nitrogen deposition is not detailed systematically in the conservation objectives. Therefore, even where critical loads have been used in assessments under Article 6.3, it is still up to individual countries to define an assessment that is appropriate, because no common methodology exists to consider the impacts of nitrogen deposition.

Assuming an assessment has been carried out in a robust manner, there is still the final hurdle of defining what represents an acceptable level of critical load contribution from a plan or project. In other words, there is a difficulty in agreeing the threshold emission/deposition contribution that "***will not adversely affect the integrity of the site".*** It appears that, of all Member States, only Denmark and Germany have set out clear parameters of acceptable additional process contributions relating to additional nitrogen emissions from intensive livestock units that would not adversely affect an SAC or SPA (See e.g., Bealey *et al*.,this volume).

From this we can clearly see that Article 6.3 is open to interpretation as to the degree of precaution that individual Member States apply. This applies to their interpretation for each of significant, appropriate assessment and adverse affect.

Site-specific consideration will always be a key factor in assessing a plan or project in the light of the conservation objectives for a site. This allows some latitude in defining significance and the scope of the appropriate assessment. However, unless some generic guidance is provided (as already happens in Denmark and Germany), it is open to interpretation and possible abuse as to what level of process contribution will or will not cause an adverse effect. It is quite clear that linking a defined threshold or additional process contribution in terms of allowable critical level or critical load contribution for nitrogen is required to determine if the plan or project is acceptable as proposed.

In the UK, these issues of interpretation have vexed the application of Article 6.3 as applied through national regulations transposing the Directive. So although agreed frameworks have been established between competent authorities and the nature conservation agencies in terms of "significant effect" and what constitutes an "appropriate assessment", the process has struggled to agree a generic threshold for an "adverse affect". This is highlighted below in two examples: a) relating to the power station sector and the introduction of the IPPC Directive and, b) consideration between the UK environment agencies and the country conservation agencies of pig and poultry units, regulated under the IPPC Directive.

7.7.4 Assessment of key nitrogen source sectors

Assessment of UK Power Stations

During the 1970's and 1980's, pollution from UK power stations transformed the air chemistry of the UK and resulted in high levels of acid deposition across much of Northern Europe. The failure to tackle pollution from the power station sector led to Britain being termed the "dirty man" of Europe and brought the term acid rain into common parlance in the UK. Since that time, the decline in coal burn and EU legislation has led to significant reductions in emissions of sulphur

dioxide from the power station sector. In 2006, the major UK coal and oil fired power stations were reviewed under the 'Habitats Regulations' with assessments based on critical loads exceedance.

The UK Department for Environment Food and Rural Affairs (Defra) (Defra, 2007) predict that, by 2020, acid deposition will still be exceeded at almost 40 per cent of sensitive UK habitats with almost 50 per cent exceeding their critical load for nitrogen deposition. Overall, the largest regulated source of acid and nitrogen deposition to UK ecosystems is the Electricity Supply Industry (ESI), mainly through emissions from power stations. The critical load modelling studies concluded that the power stations were responsible for significant deposition at a number of Natura 2000 sites (Environment Agency, 2006). However, the UK environment agencies concluded that while significant, all the emissions from the ESI were not having an adverse impact on the integrity of sites. In Wales, the statutory conservation agency (the Countryside Council for Wales) rejected this conclusion and formally adopted an "agree to disagree" position with the Environment Agency.

Because there is no agreed threshold for defining 'adverse affect' in terms of critical load exceedance it is hard to argue definitively against the environment agencies' position. However, as the critical loads approach is being used as a risk based criteria for an appropriate assessment it can equally be argued that it should be incumbent on the 'competent authority' (i.e. the relevant environment agency) to define an acceptable process contribution on the basis of critical load. In identifying a risk it is essential to quantify at what point that risk is acceptable or unacceptable.

Another issue about using the critical loads as the risk based criteria for the assessment results is in the time taken to reduce deposition to reach critical load. It is planned that the ESI will be making significant emission cuts by 2016 under the provisions of the Large Combustion Plant Directive (LCPD) (LCPD, 2001). It is hard to quantify the risks involved in delaying emission cuts until 2016. The same issue remains at the highest level in terms of EU and UK air policy. For both, there is a high-level policy commitment to achieve no exceedance of critical loads or levels. However, without a timetable for this commitment, it is hard to set high-level emission reductions, for example within the National Emissions Ceilings (NEC) Directive (NECD, 2001), or to provide clear targets for the ESI which are needed to help deliver these commitments.

Assessment of the Intensive livestock Industry in the UK

Under the provisions of the IPPC Directive, large pig and poultry units are currently being authorised in the UK. Where these units are sited near to Natura 2000 sites and are judged to have a significant effect, they require an 'appropriate assessment' under the provisions of the Habitats Directive.

A number of studies from the early 1990's have demonstrated that ammonia emissions from these units can be many times the critical level and critical load for the receiving habitat (Sutton *et al*, 2009). Studies have shown that these emissions can cause substantial changes in vegetation structure and composition with the loss of sensitive lichens and forb species at the expense of nitrogen tolerant species such as grasses (Pitcairn *et al*., 2009). In part, the results of these studies supported the need to regulate large NH_3 sources under the IPPC Directive.

The site-level environmental assessments have closely followed the requirements of Article 6.3, with screening criteria being agreed along with the scope of the appropriate assessment. However, as with the power stations in the UK, the major issue under consideration is defining an acceptable process contribution, i.e. a contribution of the plan or project (in terms of critical load or critical level contribution) below which it can be concluded that the emission will not have an adverse impact on the integrity of the site.

The situation for the UK contrasts with the situation in Denmark where allowable additional emissions are defined in terms of kgN $ha^{-1}yr^{-1}$ and in Germany where any process contributing more than 10 per cent is deemed unacceptable (Bealey *et al*., this volume). It is agreed that control of pig and poultry units under IPPC will make only a modest reduction in overall UK ammonia emissions. However, locally control could have a profound effect by making significant reductions in critical load and critical level contribution to specified sensitive habitats.

These two examples demonstrate that while the UK has expended great effort and detail in transposing the Habitats Directive, there has been little air pollution environmental improvement beyond that which would have occurred as a result of other drivers, such as the Large Combustion Plant Directive. Yet in terms of environmental outcome, the Habitats Directive arguably provides a stronger requirement for protection than any other piece of legislation. *Therefore, despite the UK's intention to rigorous application of the Directive, there has been little significant environmental gain in terms of additional measures to tackle air pollution impacts.* This is largely a result of the difficulty faced by the relevant authorities to agree a process contribution that will not cause an adverse impact. Furthermore, regulated sources, while significant are often only a part of a wider diffuse nitrogen issue (see section 5.2).

7.7.5 Legislative and Policy framework to protect Natura 2000

European Union context

While it can be concluded that the Habitats Directive provides a requirement for robust protection, the elements that could deliver that protection are open to interpretation and therefore fragment its effectiveness. A cornerstone of both EU and UK legislation is to provide a high level of protection to man and the environment as a whole (IPPC Directive, 1996). However, in reality legislation is often narrowly scoped with limited environmental focus targeted at very specific outcomes.

A notable exception to this is the transposition of the Water Framework Directive (2000/60/EC) with its objective end point in the attainment of "good ecological status" of water bodies. Integrated action for the protection of water is thus much more advanced than that to protect terrestrial environments from air pollution. Such thinking has been used to protect shared resources through initiatives on rivers such as the Rhine and Danube through to regional measures to protect shared coastal resources (e.g. the Barcelona Convention (1976) and Oslo and Paris Commission (OSPAR) (1992)).

Nevertheless, there are cases where a more effective integration would be justified. For example, the 1991 Nitrates Directive (91/676/EEC) makes provision for the protection of water from nitrate pollution. However, the way it has been interpreted in the UK and several other European countries means that the primary focus has been on the protection of human health by targeting nitrate levels in groundwater used for potable extraction. This is despite the Directive also referring to the protection of nature conservation interests.

The closest we get to an integrated initiative to control air pollution is Directive 2008/50/EC on ambient air quality and cleaner air for Europe (CAFÉ) and the Air Quality Framework Directives(2008/50/EC). However, again the main thrust of CAFÉ, so far, has been the protection of human health. Nevertheless, in recent years there have been positive moves to link air pollution drivers with biodiversity outcomes. Within the UN-ECE Convention on Long-range Transboundary Air Pollution (CLRTAP), the critical loads scientific community are committed to establishing appropriate critical loads aimed at protecting biodiversity more specifically (CCE, 2008). Although this may be some way off, it will hopefully help influence meaningful emission control strategies

focused on biodiversity outcomes. This offers the opportunity to link ecological outcomes of a Directive directly to the emission reductions required for various Member States.

In December 2007, the European Environment Agency published a list of 26 indicators that will be used to monitor progress toward the objective of halting biodiversity loss by 2010 (EEA, 2007) under the Convention on Biological Diversity. The Streamlining European Biodiversity Indicators (SEBI, 2010) endorsed the use of critical load Exceedance (CLE) for Nitrogen as the indicator against which to mark progress toward the 2010 target.

The use of CLE within a convention to protect nature conservation interests represents a major cross-over between two key policy arenas. While this represents a significant development in using critical loads, it is recognised that much work will be required to develop biodiversity relevant critical loads.

With the critical loads and levels approach providing a suitable framework, there is now the need to develop more specific high-level goals of suitable ambition. Such goals should be broad in scope and combine a quantifiable target with a suitable time frame.

Transposition into UK regulations and practice

European legislation is transposed in the UK through a series of Acts and Regulations. Additional strategies provide mechanisms to deliver national objectives, for example the National Air Quality Strategy (Defra, 2007). At a European level there would appear to be sufficient drivers to deliver a high level of environmental protection. Some of the major EU policy instruments offer a range of opportunities to deliver this, for example the Habitats Directive, the Environmental liability Directive (2004/35/EC), the Environmental Impact Assessment Directive (85/337/EEC) Strategic Environmental Assessment (SEA) Directive (2001/42/EC) the Water Framework Directive (2000/60/EC) and the Thematic Strategy for soil protection and proposals for an EU Soil Framework Directive (Soil Framework Directive, 2006).

Individual Member States have some flexibility when transposing EU Directives into domestic legislation. While we have highlighted examples where interpretation has caused problems (e.g. in defining adverse affect), there are opportunities to use the directives to target domestic programmes to deliver a high level of environmental protection. For example, while cattle are not covered by the IPPC Directive, their emissions can have a significant impact on habitats. In Denmark the introduction of the EIA Directive has provided a lever to enact domestic measure to address ammonia emissions from cattle close to Natura 2000 sites (H. Bjerregaard, this volume). It is quite clear that before Member States go seeking new legislation, to protect wildlife, reinterpretation and better use of the provisions incumbent in the existing rafts of legislation is essential.

Within the UK, the main framework for delivering Air Quality outcomes is through the UK Air Quality Strategy (Defra, 2007). The target of the Strategy is to ensure compliance with EU air quality legislation, as well as setting national objectives. However, its main focus is on the protection of human health, where a number of air quality thresholds are mandatory. The few thresholds that are listed for ecosystems are much less prescriptive being termed "national objectives" relating to concentrations for SO_2 and that in large part are already met. As previously detailed, the UK is committed to the long term objective of non-exceedance of critical loads, but a targeted trajectory to support this is not currently covered in the AQS, or elsewhere. The absence of ammonia as a named air pollutant in the strategy is notable.

As a result the level of public awareness about potential impacts of ammonia to nature conservation is significantly lowered. So while large organisations like the UK environment agencies are aware of ammonia as an issue, other competent authorities such as local planning authorities may be poorly informed of the potential impact. This has come to light on a number of occasions when livestock units are developed, such as for cattle or for pigs and poultry but below the IPPC threshold. Such plans or projects often proceed without assessment, "off-the-radar", in relation to the terms of the Habitats Directive, while others may occasionally be identified if planning permission is for some other reason required (Frost, 2004). It is thus of concern that Natura 2000 sites may be more severely impacted by proximal livestock units not covered by the IPPC thresholds than other sites further from livestock units that are regulated under IPPC. While other legislative measures may need to be adopted to prevent this happening in future, there is a great urgency to raise the issue of ammonia in future development of the Air Quality Strategy.

As discussed, control of ammonia is a key priority in protecting sensitive habitats. For a number of years there have been indications from Defra (and its predecessors) that a separate strategy is required for ammonia. Both Joint Nature Conservation Committee and the Environment Agency have called for an ammonia strategy. However, given the current recognition of the wider issue of nitrogen impacts, it is probably now more intuitive to have a 'nitrogen strategy', within which NH_3 control would form a central component. This could ensure that all environmental media were consider in a holistic manner in order to prevent control of one sector leading to pollution swapping to a different environmental receptor. While control of nitrogen pollution from individual sources can be extended and strengthened, a major source of this pollution will come from a range of diffuse agricultural sources. Joined up approaches to tackle point-source and diffuse pollutants have been proposed before in the UK in relation to the water environment. For example, the environment agency 'Aquatic eutrophication strategy' in England and Wales (Environment Agency, 1999). A similar integrated approach considering impacts from all sources across all environmental media is urgently needed.

Over recent years a number of studies across the globe and northern Europe have examined a range of ammonia emission options from buffer areas to consideration of diet formulation to reduce ammonia emission (Sutton *et al*., this volume, section 7.1). NH_3 abatement techniques have also been trialled and applied in countries such as the Netherlands where frameworks exist in which these measures alongside other controls can be delivered within a common framework.

There are therefore an extensive range of options available to address the ammonia issue. An overarching national strategy is, however, needed as a basis to ensure that appropriate measures are put in place to ensure the protection of sensitive habitat sites.

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7.8 Quantifying the threat of atmospheric ammonia to UK Natura 2000 sites at 1 km resolution

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Summary

- High levels of atmospheric ammonia can damage sensitive ecosystems with critical levels having been set at 1 µg m^{-3} ammonia for most sensitive vegetation types (lichens, bryophytes) and 3(2-4) µg m^{-3} for sensitive communities of higher plants (heathlands, forests).
- We derive spatially detailed atmospheric-ammonia surface concentrations using a high resolution atmospheric transport model applied at 1 km grid resolution.
- We apply two types of indicator to quantify the threat of atmospheric ammonia to UK Natura 2000 sites, the flagship for biodiversity protection in the European Union. We show that a designation weighted indicator (DWI) provides a more precautionary and appropriate metric than the more traditional area weighted indicator (AWI). Overall, 69% of Natura 2000 sites in the sites in the UK exceed the critical level of 1 µg m^{-3} (DWI).

7.8.1 Introduction

High levels of atmospheric ammonia (NH_3) may cause adverse effects on the environment through a range of processes, including eutrophication effects on biodiversity, acidification of soils and particulate matter effects on human health (Erisman and Sutton, 2008). The magnitude of the ecological effects can be assessed by thresholds of atmospheric NH_3 concentrations, referred to as critical levels (Achermann and Bobbink, 2003; Sutton *et al*., 2009c). New critical levels (CLE) for assessing the effects of atmospheric ammonia on sensitive ecosystems (shown in Table 7.4) have recently been adopted by the United Nations Economic Commission for Europe for different habitats (UNECE 2007, Sutton *et al*., 2009b).

The FRAME atmospheric dispersion and deposition model (e.g., Fournier *et al*., 2005) was used to estimate surface air concentrations of ammonia at a spatial resolution of 1 km by 1 km (Figure 7.13a). By overlaying the air concentration data with the boundaries of the UK Natura 2000 sites (Figure 7.13b) in a Geographical Information System, a map of Critical level exceedance was derived (Figure 7.13c).

7.8.2 Aims and objectives

The new CLE estimates are particularly relevant for assessing ecological conditions under the terms of the Habitats Directive (Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora), and the associated Birds Directive (Council Directive 79/409/EEC on the conservation of wild birds). These seek to protect Europe's natural resources, especially the most seriously threatened habitats and species across Europe. The 'Natura 2000' network, to be implemented by all EU Member States, represents a flagship for biodiversity protection in the European Union. In this study, we have sought to investigate how the recently established CLEs could be used to develop indicators to assess the ammonia threat to the Natura 2000 network in the UK.

Two main types of indicator were investigated, to assess the threat of atmospheric ammonia concentrations on Natura 2000 sites in the UK:

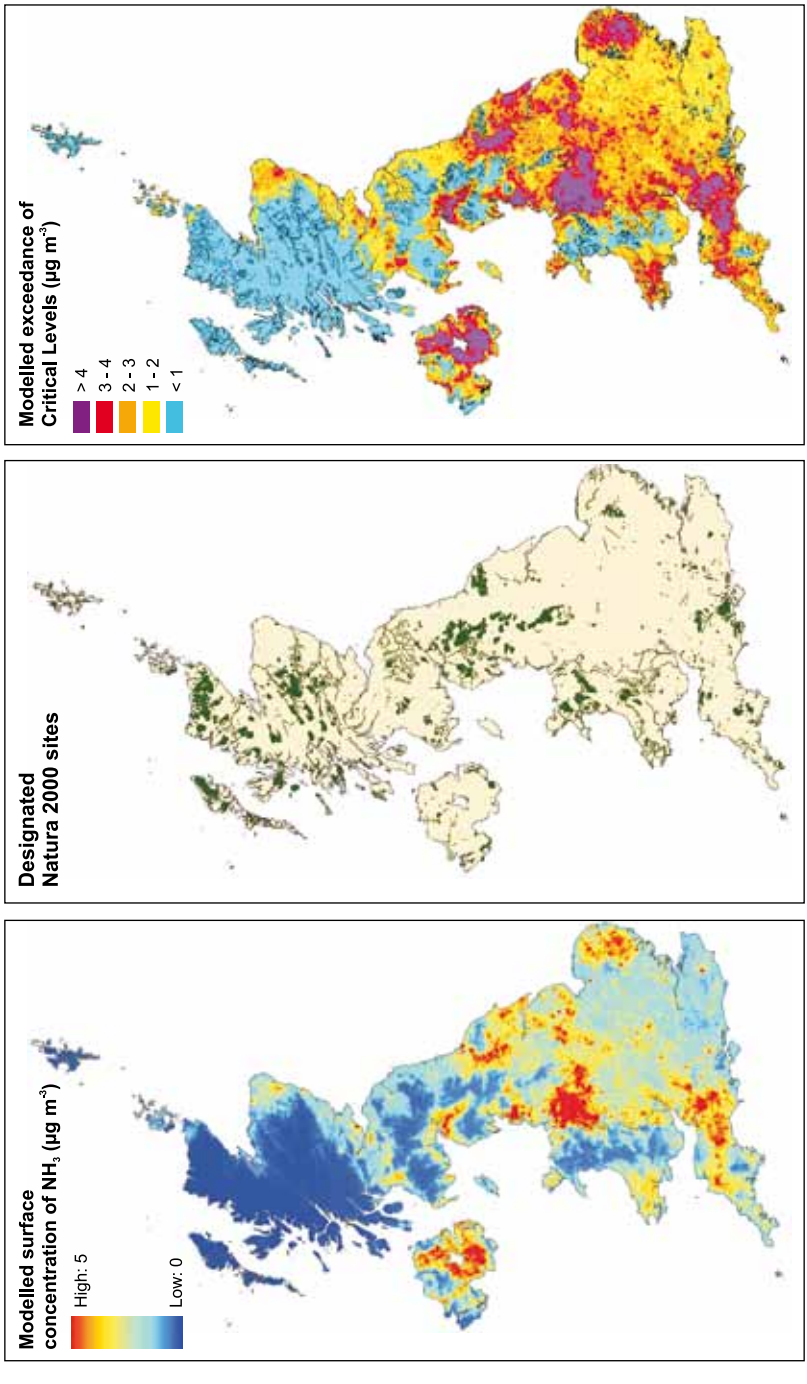

Figure 7.13: a) Modelled UK NH_3 concentrations at surface level (average height of air column 0-1 m above ground), b) Designated Natura 2000 sites in the UK, c) Modelled exceedance of critical levels for the UK.

Table 7.4: New critical levels (CLE) for effects of atmospheric ammonia on sensitive ecosystems adopted by UNECE (2007):

Receptor	Critical level ($\mu g\ NH_3\ m^{-3}$)
Lichens and bryophytes	1
Higher plants	3 (uncertainty range 2-4)

Table 7.5: Whole country area (in per cent) where critical levels of 1, 2 and 3 $\mu g\ NH_3\ m^{-3}$ are exceeded.

Critical level	England	Wales	Scotland	N. Ireland	UK
1 mg m^{-3}	93 per cent	68 per cent	26 per cent	85 per cent	69 per cent
2 mg m^{-3}	61 per cent	38 per cent	9 per cent	65 per cent	42 per cent
3 mg m^{-3}	27 per cent	14 per cent	2 per cent	43 per cent	19 per cent

Table 7.6: Number of UK Natura 2000 sites (DWI, in per cent) where critical levels of 1, 2 and 3 $\mu g\ NH_3\ m^{-3}$ are exceeded.

Critical level	England	Wales	Scotland	N. Ireland	UK
1 mg m^{-3}	98 per cent	92 per cent	34 per cent	85 per cent	68 per cent
2 mg m^{-3}	69 per cent	56 per cent	13 per cent	63 per cent	42 per cent
3 mg m^{-3}	32 per cent	26 per cent	6 per cent	34 per cent	19 per cent

Table 7.7: Area of UK Natura 2000 sites (AWI, in per cent) where critical levels of 1, 2 and 3 $\mu g\ NH_3\ m^{-3}$ are exceeded.

Critical level	England	Wales	Scotland	N. Ireland	UK
1 mg m^{-3}	44 per cent	16 per cent	3 per cent	39 per cent	22 per cent
2 mg m^{-3}	7 per cent	4 per cent	0.4 per cent	17 per cent	4 per cent
3 mg m^{-3}	2 per cent	1 per cent	0 per cent	4 per cent	1 per cent

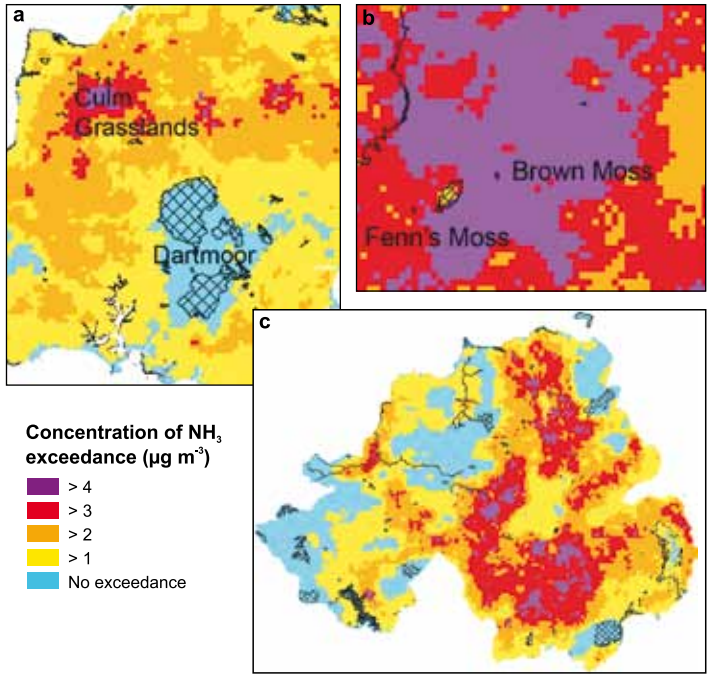

Figure 7.14: Spatial pattern of estimated critical level exceedance and the location of Natura 2000 sites. a) SW England: With the exception of small areas at the boundaries, most of Dartmoor is below the 1μg m^{-3} critical level. By contrast, the Culm Grasslands mostly exceed a critical level of 2 μg m^{-3}. 2b) W England: small SACs in agricultural regions, such as Fenn's Moss and Brown Moss (NW England) exceed the critical levels. 2c) Northern Ireland: Although many of the largest sites do not exceed the lowest critical level, substantial exceedance is seen for the smaller sites.

- Percentage area of Natura 2000 sites where the critical level is exceeded (Area Weighted Indicator AWI)
- Number of Natura 2000 sites where the critical level is exceeded (Designation Weighted Indicator DWI)

7.8.3 Results and discussion

Over the UK as a whole, the three critical levels of 1, 2 and 3 µg NH_3 m^{-3} are exceeded over 69 per cent, 42 per cent and 19 per cent of the land area, respectively (Table 7.5). The choice of indicator (AWI or DWI) used to estimate the stock-at-risk at UK 'Natura 2000' sites has a large impact on the outcome (as does the spatial resolution). Using the AWI we estimate that 11 per cent and one per cent area of the UK Natura network exceeds the CLE values of 1 and 2 µg NH_3 m^{-3}, respectively. By contrast, using the DWI, the equivalent exceedances are 59 per cent and 24 per cent. The highest regional exceedance (DWI, CLE 1 µg NH_3 m^{-3}) was calculated for England (92 per cent exceeded), and the lowest for Scotland (24 per cent exceeded). This is shown in Tables 7.6 and 7.7 and Figure 7.14. The Designation Weighted Indicator is more precautionary than the Area Weighted Indicator. It may be argued that the DWI is the most appropriate indicator since exceedance over any part of a Natura 2000 site represents a threat to the integrity of the whole site (Frost, 2004).

7.8.4 Conclusions

The choice of indicator used to estimate the stock-at-risk at UK Natura 2000 sites has a large impact on the outcome.

The Designation Weighted Indicator is more precautionary than the Area Weighted Indicator, as it may be argued that exceedance over any part of a Natura 2000 site represents a threat to the integrity of the whole site.

Small sites are often more at risk than larger sites, as they tend to occur in source areas with larger atmospheric NH_3 concentrations and dry deposition of N. They are also more likely to be overlooked when meeting policy targets with an Area Weighted Indicator, since their small area will not contribute significantly to the overall statistics.

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7.9 Beyond nitrogen critical loads – is there a role for the 'Ecosystem Services' approach?

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Summary

This paper considers the extent to which indicators of critical load exceedance capture the potential impacts of changes in nitrogen deposition on ecosystem services. It shows that there are significant links between nitrogen deposition and a large range of ecosystem services. There is potential for indicators to be adapted to provide more specific qualitative information for Natura 2000 sites of the implications of critical load exceedance for ecosystem services.

For the provision of ecosystems goods (e.g. food, fuel, fibre) and water quality and erosion regulation, it is likely that quite specific information can be provided on the effects of nitrogen deposition. For others, such as pollination and cultural services, the implications for ecosystem services are likely to depend on the specific changes in species composition that are found in specific habitats and sites. The issue of climate regulation has been identified as a critical ecosystem service, but this effect is not currently considered explicitly in setting critical loads, and given the complexity of the potential effects of nitrogen deposition on different greenhouse gas fluxes, it seems impractical to include this in any simple assessment of effects of critical load exceedance.

There are ecosystem services where exceedance of the established empirical critical load for nitrogen input can be a positive outcome, for example, increases in more nitrophilous species increasing productivity in certain grassland types and increased grass growth stabilising coastal dunes, and hence enhancing erosion regulation. An ecosystems approach would therefore have value in informing the prioritization of conservation management practices in areas with high nitrogen deposition, depending on the ecosystem service that is most valued at any particular site. However, given that the cause and effect relationships underlying important ecosystem services are often complex and not sufficiently understood, more data and research is needed to provide specific guidance on potential conservation priorities.

7.9.1 Introduction

The ecosystem service approach based on the work of the Millennium Ecosystem Assessment (MEA, 2005) and the Convention for Biological Diversity (CBD, 2004) is currently being considered across Europe as a potential means of more effective management of the environment. By comparison with the existing focus on critical thresholds, an ecosystems approach may offer important advantages for air quality management, such as:-

- a holistic assessment that considers the whole range of ecosystem services affected as a starting point;
- inclusion of regulating services, such as ecosystem controls on fluxes of pollutants in land-water-air systems, which are currently under-represented in European policy;
- identification of negative externalities, ancillary benefits and trade-offs of policy measures;
- insight into the full costs and benefits of policy measures.

Currently, evaluation of the benefits of measures to reduce nitrogen (N) deposition across Europe is based on critical loads, which are set to prevent 'significant harmful effects on specified elements of the environment'. These provide policy makers with values of ecological thresholds above which

adverse and potentially irreversible environment effects may occur. Critical loads for nitrogen are either calculated using a steady-state mass balance approach to determine, at steady state, the rate of deposition at which a critical chemical threshold for effects is exceeded or an empirical critical load has been set based on observed effects in the field and in long-term field experiments. Alongside these empirical critical loads, typical biological or chemical indicators of exceedance are provided for different habitats (Bobbink *et al.*, 2010). However, there is no explicit consideration of ecosystems services in setting critical loads and in identifying the implications of critical load exceedance. Therefore, a key question for the application of the ecosystem service approach is if and how this approach could be better integrated with the assessment and application of critical loads for sensitive habitats and sites.

This short paper addresses this question by considering the extent to which indicators of critical load exceedance provided for the users of this information capture the potential impacts of changes in N deposition on ecosystem services. Our analysis is based on a study on the feasibility of embedding an ecosystem services framework into air quality policy (Hicks *et al*., 2008) and an assessment of economic quantification of changes in ecosystem services caused by control of ammonia emissions by Smart *et al*., (in press), to which the reader is referred for more detail.

7.9.2 Results

The results of an initial scoping study by Hicks *et al*., (2008) to identify the presence of any significant links between ecosystem services and nitrogen deposition are summarized in Table 7.8. The results show significant potential links between N deposition and a large range of ecosystem services. The most important, and relatively well understood, positive (beneficial) changes to ecosystem services that could result from decreasing N deposition were related to air and water quality, species composition and climate regulation (i.e., decrease in greenhouse gas (GHG) emissions from soils). Important negative changes as a result of decreasing nitrogen deposition occurred where the fertilizing effect of nitrogen deposition had previously had a beneficial effect on harvested goods and carbon sequestration by vegetation and with specific agricultural management changes (e.g., changes in methods of slurry storage and application may lead to decreased ammonia emission, potentially at the expense of increased nitrate leaching from soils unless certain precautions are taken).

Table 7.9 provides a more detailed analysis of the effects of regional-scale N deposition on a selected range of the most important ecosystem services listed in Table 7.8. This study integrates data from two sources. Hicks *et al.*, (2008) conducted a qualitative assessment, based on expert judgement, of the impacts of nitrogen deposition on different UK broad habitat types, of which a selection (chosen to represent the Natura 2000 network) are shown in Table 7.9. Table 7.9 also shows the relevant EUNIS category for each UK Biodiversity Action Plan (BAP) broad habitat type, and lists the indicators of exceedance of the empirical critical load (Bobbink *et al*., 2010). On the basis of these indicators, we provide a qualitative assessment based on our own judgement, for each ecosystem service and habitat, as to whether any effect on ecosystem services could be deduced based on the indicators of exceedance for critical loads.

The results in Table 7.9 are considered briefly below for each ecosystem service in turn.

Provisioning Service: Ecosystem goods

Nitrogen deposition to nitrogen-limited ecosystems can cause a plant fertilisation effect leading to an increase in harvestable material, e.g. of crops (arable land), timber (woodlands), hay (grasslands). Some of the indicators of empirical critical load exceedance imply this effect (e.g., through an increase in tall grasses) but do not state it specifically. It is also important to note that N deposition at rates both above and below the critical load may be beneficial for ecosystem goods.

Table 7.8: Preliminary assessment of effects of nitrogen emissions on ecosystem services (after Hicks *et al.*, 2008)

Ecosystem Service	Effect of nitrogen emissions
1. Provisioning Services	
Ecosystem goods	Production of goods (e.g. food, fuel, fibre) can be increased and decreased.
Water quality	Acidification and eutrophication of surface waters can be caused by direct deposition or by leaching from terrestrial ecosystems.
Biochemical/genetics	Abundance of species can be reduced (or increased in certain circumstances) and community composition can be changed in both terrestrial and aquatic ecosystems.
2. Regulating services	
Air-quality regulation	The growth of trees and tall vegetation can be affected, altering their ability to remove air pollution, while NH_3 emissions contribute to formation of secondary particulates
Climate regulation	Carbon sequestration, methane fluxes and nitrous oxide production are all affected
Water regulation	Effects on peat creation and forest growth can affect water storage and interception.
Water purification	The capacity of wetlands to remove nutrients from water may be reduced by excess atmospheric inputs.
Natural hazard regulation	No significant direct effects
Pest regulation	No significant direct effects
Disease regulation	No significant direct effects
Pollination	Both vegetation composition and flowering intensity can be affected.
Erosion regulation	Increases and decreases in vegetation cover can be caused, leading to changes in rates of erosion
3. Supporting services*	
Soil formation	Detrimental effects can occur on peat formation, but successional change and soil formation can be enhanced in other soils
Primary production	Increase of biomass in N limited terrestrial and aquatic habitats
Nutrient cycling	Rates of soil mineralization can be increased and production of greenhouse gases and nitrate leaching can be enhanced Increased soil N accumulation can occur and may be associated with increased C sequestration
4. Cultural services	
Recreation and tourism	Large changes in terrestrial and aquatic species composition may affect field sports and ecotourism
Aesthetic	Significant if it is assumed that changes from the status quo (e.g. changes in species composition) are negative.
Educational	Reduction in species rich habitats as sites for study
Cultural heritage	Loss of iconic species

Note: Including Supporting Services can lead to double counting

Table 7.9: Impacts of nitrogen deposition on selected ecosystem services in UK Biodiovbersity Action Plan (BAP) Broad Habitat Types and typical indicators of empirical critical load exceedance for relevant EUNIS (European Nature Information System) habitat types (after Bobbink et al., 2010).

UK BAP Broad habitat type (Empirical critical load category in parenthesis)	Supra littoral sediment (Coastal Habitat (EUNIS B))	Broad-leaved, Mixed and Yew Woodland (Forest habitats (EUNIS G))	Acid Grassland (Grasslands and tall forb habitats (EUNIS E))	Calcareous Grassland (Grasslands and tall forb habitats (EUNIS E))	Dwarf Shrub Heath Heathland, scrub and tundra habitats (EUNIS F)	Bogs (Mire, bog and fen habitats (EUNIS D))
Typical indicators of empirical critical load exceedance (Relate to broad habitat type within EUNIS class.)	Increase tall graminoids, decrease prostrate plants, increased N leaching, soil acidification, loss of typical lichen species, accelerated succession	Changes in soil processes, nutrient imbalance, altered composition mycorrhiza and ground vegetation	Increase in gramin-oids, decline of typical species, decrease in total species richness	Increase in tall grasses, decline in diversity, increased mineral-ization, N leaching; surface acidification	Transition from heather to grass dominance, decline in lichens and mosses, changes in plant biochemistry, increased sensitivity to abiotic stress, increase in N leaching	Increase in vascular plants, altered growth and species compos-ition of bryophytes, increased N in peat and peat water
Provisioning Services Ecosystem goods (e.g. food, fibre, fuel)	+/+	+/0	+/+	+/+	+/0	?/+
Water quality	-/-	-/0	-/0	-/0	?/-	-/-
Regulating services Climate regulation	0/0	-/0	-/0	-/0	-/0	-/0
Pollination	0/?	-/0	-/?	-/?	?/?	0/+
Erosion Regulation	+/+	0/0	+/0	+/0	-/0	?/?
Cultural services Recreation and tourism	-/-	-/0	-/-	-/-	-/-	-/?
Cultural heritage	-/-	-/?	-/-	-/-	-/-	-/?

Scores are indicated as follows.
1. The symbol before the / indicates the evaluation of effects on ecosystem services based on Hicks et al., (2008) as follows : '+' potential positive or beneficial effect; '0'negligible effect; '-'potential negative or adverse effect; ?' gaps in evidence.
2. The symbol after the / indicates the possible effects on ecosystem services based on the indicators of critical load exceedance (Bobbink et al., 2010) as follows: + potential positive or beneficial effect can be inferred; - potential negative or adverse effect can be inferred; 0 no inferences can be drawn about effects on ecosystem service; ? some effect can be inferred but the direction is uncertain.

It should be noted that, in the case of oxidized N deposition from industrial and traffic emissions of NO_x, these simply represent additional N sources. By contrast, in the case of reduced N deposition, any productivity gains need to be weighed up against the reduction of agricultural productivity due to NH_3 losses from crop and livestock systems.

Provisioning Service: Water quality
Water quality can be directly and indirectly impacted by N deposition through both acidification and eutrophication. Nitrogen leaching, which is linked to acidification, is a common indicator of exceedance of critical loads in Table 7.9, but is not indicated in some cases (e.g., for acid grasslands) for which effects might be expected.

Regulating services: Climate regulation
None of the indicators of critical load exceedance specifically relates to climate regulation. This is likely to reflect the complex responses of habitats to N deposition and the need to consider several greenhouse gases. In addition to any effect on above-ground or below-ground carbon sequestration, increased N deposition generally causes higher rates of N_2O emission, an effect that becomes more pronounced as deposition rates increase (Skiba *et al.,* 1998). This effect will occur to some extent in all terrestrial habitats, but it is particularly important in arable and improved grassland areas, which are subject to direct fertilisation. Furthermore, nitrogen fertilisation effects are also known to suppress CH_4-oxidation in grasslands, forests and arable systems potentially causing increased concentrations of this potent greenhouse gas (Hutsch *et al*., 1993).

Hence, identifying the net effect of additional N on the greenhouse gas balance and hence climate regulation represents a major current research challenge (e.g. Sutton *et al.*, 2008; Smart *et al*., 2011), and it is difficult to quantify the trade-offs between changes in CO_2, N_2O and CH_4 fluxes for different habitats. These trades-offs become even more complex when the interactions between nitrogen emissions and secondary aerosol and ozone formation are considered. Hence, it is not realistic to consider this effect within the critical load exceedance context. Butterbach-Bahl *et al*., (2011) have made a first estimate of the net effect of nitrogen emissions on European radiative balance, highlighing the importance of addressing this interaction from the broadest possible perspective.

Regulating services: Pollination
Pollinators require islands of alternative flowers to provide food at specific times. N deposition can reduce patchiness of vegetation and change species composition, which may result in the loss of flowering species that may be crucial to particular pollinator species. When species composition remains unchanged, empirical data show both increases in flowering (e.g. in dwarf shrub heath) and decreases in flowering (e.g. in acid and calcareous grasslands) (RoTAP, 2011). Pollination ecosystem services are likely to be affected by critical load exceedance, but it is not possible to deduce the direction of change without further information on the specific changes in species composition. Furthermore, some effects (e.g., changes in and amount and timing of flowering) are not currently captured in critical loads.

Regulating services: Erosion regulation
Erosion is controlled by the presence or absence of vegetation and N deposition may change their abundance and occurrence. In early successional communities (e.g. sand dune systems (supra-littoral sediment)), nitrogen inputs may increase the growth of "sand holding" grass and sedge species and so reduce coastal erosion. Hence, in this case, critical load exceedance may be associated with a benefit for the ecosystem service. However, this effect is only clearly inferred by the indicators of exceedance in the supra littoral sediment habitat. In bogs and montane habitats,

N inputs may decrease moss, and lead to increased erosion, but the indicators of exceedance only refer to 'altered growth and species composition' of bryophytes, hence not clearly identified the potential benefits for erosion.

Cultural services (Recreational, Aesthetic, Educational, and Cultural)
The impact of N deposition on cultural services is highly subjective and difficult to define. For the purposes of Table 7.9, we assumed that all changes from currently defined BAP habitat types are negative, and this is the case where loss of typical species is included in the indicators of critical load exceedance. However, the impact of N on cultural heritage is likely to be particularly important where individual iconic species are under threat (e.g. insectivorous plants such as sundew; fruit bearing plants such as bilberries). However, the critical load indicators only refer to broad functional groups which cannot be used to infer effects on individual species.

7.9.3 Discussion

The results show that there is potential for the consideration of positive and negative nitrogen impacts on ecosystem services provided by Natura 2000, and other sites of conservation interest, to guide policy development for their protection. However, the cause and effect relationships underlying important ecosystem services are often complex and not sufficiently understood. The implication of our study is that factors that may have a likely significant effect on a site protected under the Habitats Directive may not always be sufficiently described by the current indicators of exceedance of critical loads. Table 7.9 is very preliminary, and can certainly be improved upon, but does illustrate the range of ecosystem services affected and that the link to indicators of exceedance of empirical critical loads is stronger for some ecosystem services than for others.

For empirical critical loads to prevent eutrophication, a range of adverse effects have been identified as potentially occurring when the critical load is exceeded (Bobbink *et al*.,2003 and 2010; see Table 7.9). While there is considerable variation between habitats, these effects can generally be characterised as one of three major classes of impacts:-

- Invasion of competitive, fast growing species
- Decreased plant species diversity or loss of characteristic species of the habitat,
- Increased nitrate leaching once the system reaches nitrogen saturation.

Each of these has a broad link to specific ecosystem services. However, the critical load approach does not consider the implications of loss of characteristic species or nitrate leaching in terms of specific ecosystem services – rather it is simply set to prevent these adverse effects. Furthermore, changes in primary production are treated quite differently under the ecosystems approach than under the critical load approach. Whereas the former sees this an increase in provisioning services, the latter sees this as an adverse effect because it is normally associated with increased cover of fast-growing species which will out-compete other valued species for the particular habitat. The balance between these two effects is habitat-specific – for most woodlands and grasslands, for example, primary production is a central ecosystem service, but for mires or sand dunes it is not. An ecosystems approach would therefore have implications for the prioritization of conservation management practices depending on the ecosystem service most valued at any particular site.

Nevertheless, it could be argued that the indicators of critical load exceedance could be readily adapted and clarified to provide more specific, albeit qualitative, information for Natura 2000 sites of the implications of critical load exceedance for ecosystem services. Actual quantification of these effects is another challenge, and is outside the scope of this brief paper. For ecosystem goods, water quality, and erosion regulation, it is likely that quite specific information can be provided. For others, such as pollination and cultural services, the implications for ecosystem services are

likely to depend on the specific changes in species composition that are found in specific habitats, and hence the rather general language of the indicators of exceedance would need to be made more specific if effects on ecosystem services were to be evaluated.

Finally, the issue of climate regulation is particularly challenging, although it has been identified as a critical ecosystem service. This effect is not considered explicitly in setting critical loads, and given the complexity of the potential effects on different greenhouse gas fluxes and nitrogen aerosol affects on radiative balance (Butterbach Bahl *et al*., 2011), it seems impractical to include this in any simple assessment of effects of critical load exceedance. This is especially because any positive or negative effects of N deposition on ecosystem radiative balance need to be weighed against the potential existence of an opposing effect elsewhere, such as that related to an accompanying loss of N from agriculture. The extent of such interactions points to the need to complement the development of the ecosystems approach with studies that integrate the multiple consequences of human activities.

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W.K. Hicks, C.P. Whitfield, W.J. Bealey and M.A. Sutton
Editors

Nitrogen Deposition and Natura 2000:

Science and practice in determining environmental impacts

Pollution from nitrogen and other nutrients is a continuing and growing threat to biodiversity in terrestrial, inland water and coastal ecosystems and may count among the most significant changes humans are making to ecosystems. This book, documenting the threat of nitrogen deposition to sites of high conservation status across Europe, is therefore a timely and important contribution to work on Target 8 of the Strategic Plan on Biodiversity 2011-2020: "By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity".

Ahmed Djoghlaf,
Executive Secretary,
Convention on Biological Diversity

In its 2010 long-term strategy, the LRTAP Convention aims at meeting emerging challenges in delivering a sustainable long-term balance between the effects of air pollution, climate change and biodiversity. Since nitrogen plays a pivotal role in all three areas, this book is timely in addressing this pollutant in a trans-disciplinary context. Given that more than 60% of Natura 2000 areas show exceedances of nitrogen critical loads, the integrative and innovative approaches covered in this book may help inspire further action.

Dr. Jean-Paul Hettelingh
Head, Coordination Centre for Effects (CCE)
Working Group on Effects, LRTAP Convention

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