

# PLANT SPECIES RICHNESS AND THE EXCEEDANCE OF EMPIRICAL NITROGEN CRITICAL LOADS: AN INVENTORY



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## Chapter 1. Introduction

The emissions of ammonia (NH<sub>3</sub>) and nitrogen oxides (NO<sub>x</sub>) have strongly increased in the second half of the 20<sup>th</sup> century. Ammonia is volatilized from intensive agricultural systems, such as dairy farming and intensive animal husbandry, whereas nitrogen oxides originate mainly from burning of fossil fuel by traffic and industry. Because of short- and long-range transport of these nitrogenous compounds, atmospheric nitrogen (N) deposition has clearly increased in many natural and semi-natural ecosystems across the world. Areas with high atmospheric nitrogen deposition are nowadays central and western Europe, eastern USA and, since the 1990s, Eastern Asia (e.g. Galloway & Cowling 2002). Atmospheric N deposition rates of 20 - 60 kg N ha<sup>-1</sup> yr<sup>-1</sup> have been observed in non-forest ecosystems in Western Europe, whereas in forest stands in Europe or the USA values between 20 -100 kg ha<sup>-1</sup> yr<sup>-1</sup> have been reached, instead of the estimated background inputs of 1-2 kg N ha<sup>-1</sup> yr<sup>-1</sup> in the early 1900s.

The availability of nutrients is one of the most important abiotic factors which determine the plant species composition in ecosystems. Nitrogen is the limiting nutrient for plant growth in many natural and semi-natural ecosystems, especially of oligotrophic and mesotrophic habitats. Most of the plant species from such conditions are adapted to nutrient-poor conditions, and can only survive or compete successfully on soils with low nitrogen availability (e.g. Tamm 1991; Aerts & Chapin 2000). In addition, the nitrogen cycle in ecosystems is complex and strongly regulated by biological and microbiological processes, and it is thus likely that many changes can occur in plant growth, inter-specific relationships and soil-based processes as a result of increased deposition of air-borne N pollutants. The series of events which occurs when N inputs increase in an area with originally low background deposition rates is highly complex. Many ecological processes interact and operate at different temporal and spatial scales. As a consequence, high variations in sensitivity to atmospheric nitrogen deposition have been observed between different natural and semi-natural ecosystems. Despite this diverse sequence of events, the most obvious effects of increased N deposition are significant **changes in vegetation composition and diversity** and in **nitrogen cycling** (see Bobbink *et al.* 1998 and Bobbink & Lamers 2002 for more details).

Within the UN/ECE Convention on Long-range Transboundary Air Pollution, empirical procedures have been developed to set critical loads for atmospheric N deposition. Based on observed changes in the structure and function of ecosystems, reported in a range of publications, empirical N critical loads have been established for European natural and semi-natural ecosystems in 1992, 1996 and extensively updated in 2002 (Achermann & Bobbink 2003). They have also been used for the development of the second edition of the Air Quality Guidelines for Europe of the World Health Organisation Regional Office for Europe (WHO, 2000). It is clear that the exceedance of the set empirical N critical loads should be prevented for protection of the structure and function of natural and semi-natural ecosystems. At present, the causal relationship between the amount of exceedance of these N critical loads and the biodiversity of these systems is hardly quantified.

### *Aims of this study*

The central aim of this study is to reveal the empirical relationship between the amount of exceedance of the N critical loads in major world ecosystems and the loss of biodiversity. Because almost only data are available for plant life, this inventory study is restricted to **plant species richness**. An overview of the studied vegetation types and the approach used to obtain the data and to quantify the relationship between N critical loads and plant species richness is described in **Chapter 2** (Methods). The outcome of this inventory is presented per world vegetation type in **Chapter 3** (Results), with expert judgement if possible. The results are also shortly discussed in this section and preliminary conclusions are presented. Concluding remarks are given in **Chapter 4** of this report. Finally, the quoted references of this inventory study are listed.

## Chapter 2. Methods

### *Data collection*

The basis of this study was the extended database, prepared for the empirical N critical load studies in 1992, 1996 and 2003. For a full list of references, see the original background documents of the UN/ECE-studies (Bobbink *et al.* 1992; 1996; 2003). International bibliographic systems (BIOSIS; SCI-Web of Science) were, moreover, used to obtain additional information on non-European ecosystems (if present), and the most recent studies, not covered before (September 2002 – October 2003). Peer-reviewed publications, book chapters, nationally published papers and “grey” reports of institutes (if available) were incorporated. Studies providing insights into ecosystem reactions to an increase in nitrogen input or availability have been conducted for a variety of reasons. This has resulted in many different designs, from correlative or retrospective field studies, experimental studies in pots and mesocosms to field addition experiments. In this study I only used the outcome of field addition experiments, which have **independent, sole nitrogen treatments** (in general below 30 g N m<sup>-2</sup> yr<sup>-1</sup>; at least 2 years). Thus, all results of fertilization experiments with joint applications of nutrients were not useable. Furthermore, the papers were selected to have **quantitative data** on plant species richness (number of plant species per plot) or full species description per plot and ordered per world vegetation type (see below). Per used study data were averaged per treatment per investigated site (varying between 1 – n) to avoid pseudo-replication and overrepresentation.

### *World Vegetation types*

Many natural and semi-natural ecosystems are present across the globe. In this report, the following world vegetation types (a.o. Archibold 1995) were recognized:

#### Zonal:

- Arctic and alpine vegetation
- Boreal coniferous forests
- Temperate forests
- Temperate grasslands
- Temperate semi-natural grasslands
- Heaths and scrub vegetation
- Mediterranean vegetation
- Arid vegetation (deserts)
- Tropical savannas
- Tropical forests

#### Azonal:

- Terrestrial wetlands
- Coastal salt vegetation.

Unfortunately, no or hardly any experimental field data on the effects of experimental nitrogen enrichment were available for the last four of the zonal vegetation types, and the last azonal vegetation types.

## Data analyses

All useable field N addition studies were classified according to one of the mentioned world vegetation types. Number of plant species per control plot or N-treated plot were obtained from the papers, or calculated (if possible) from presented species tables. The used experimental N load (in  $\text{g N m}^{-2} \text{ yr}^{-1}$ ) were, of course, also noted. To prevent the effects of different plots sizes used in the studies of a specific world vegetation type (e.g. boreal coniferous forest or temperate semi-natural grasslands), the following species richness ratio (per N treatment) has been calculated:

Species richness ratio:  $S_N / S_C$

with:  $S_N$  = species number in N-treated situation  
 $S_C$  = species number in control situation

If this ratio is 1, the number of species in the N-added species equals that of the control situation, above 1 the number of species after N addition is higher than in the control vegetation. If the value of this ratio is below 1, the species number after N enrichment is lower than in the control situation. This species richness ratio is relatively simple to interpret: a ratio of 0.75 indicates 25 % reduction of the species number after N treatment compared with the control situation. The species richness ratio was in few cases estimated, when only increases – decreases of most species were given, but full lists or species numbers were missing. The available data on the species richness ratio were ordered per world vegetation type (see before) and analysed with 1x1 regression methods (including line fit procedures) against the N exceedance, i.e. the applied N load minus the mean empirical N critical load for the specific vegetation type (Achermann & Bobbink 2003). In all figures, the smallest subdivision of the x-axis equals the mean critical load of that vegetation type.

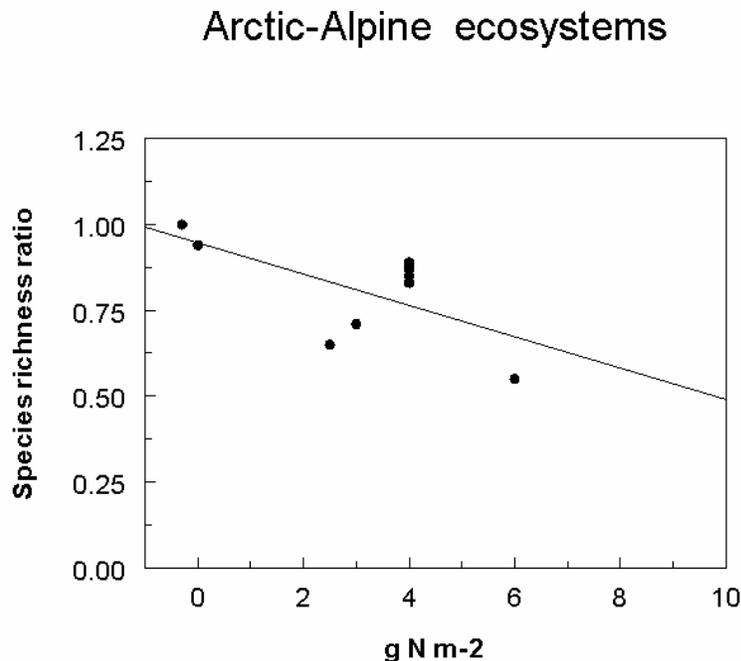
The changes in species composition of the understory vegetation in boreal forests could be analysed in detail, because of the availability of a full digital data set of long-term N fertilization experiments in Swedish forests (kindly provided by Dr. Han W. van Dobben, Alterra, Wageningen, The Netherlands). Besides plot species richness, the similarity index of Sorensen could be calculated with MVSP 3.0 based upon presence and absence of species in the experimental plots in comparison with the respective control situation (7 locations across boreal Sweden,  $n=12$ ). This index quantifies the (dis)similarity between two vegetation samples in a simple scale between 0 – 1, with 1 is full similarity (all species common) and 0 is no similarity (no common species). This approach was used because it was too arbitrary to classify all observed plant species into characteristic or non-characteristic species for the studied boreal forests.

### Chapter 3. Results

In this chapter the outcome of this study are presented per world ecosystem type, in the same order as given in the previous section. If the number of data per ecosystem type was too restricted to perform a regression analysis (in general below  $n=6$ ), only a short indication of the sensitivity of the specific ecosystem type is given, and expert judgment has been used to suggest an “hypothetical relationship” (if possible).

#### *Arctic-alpine vegetation*

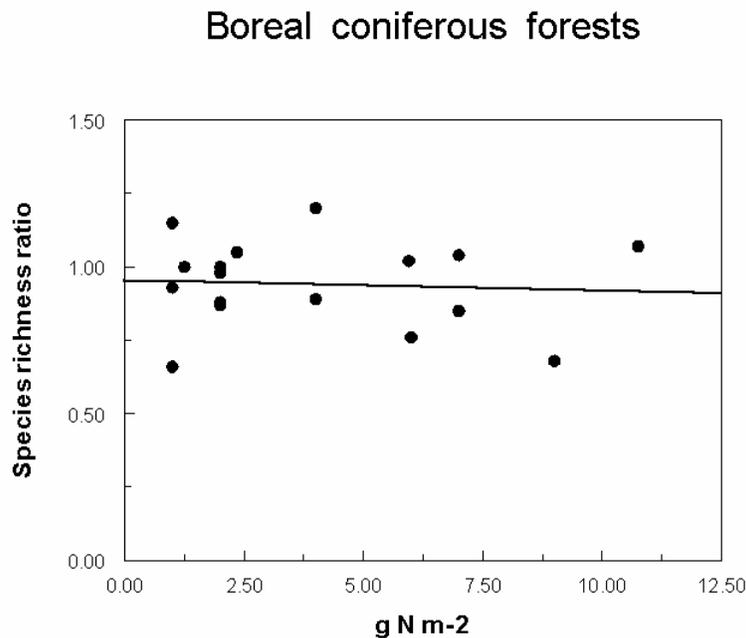
In general, plant productivity of arctic and alpine ecosystems is highly influenced by N, of course within the specific climatic constraints. Because of the remoteness of most of these ecosystems and thus (still) low atmospheric N inputs, possible impacts of N additions on the vegetation got only recently attention. Fortunately, results from relatively long-term studies in arctic heaths, alpine heaths, alpine tundra and alpine grasslands could be used to get a first impression of the relation between the amount of N exceedance and the species richness ratio in arctic-alpine vegetation (Lüdi 1959; Gordon *et al.* 2000, Nilsson *et al.* 2002; Fremstad *et al.* in press). These results showed an almost significant negative linear relationship between exceedance of the N critical loads and the species richness ratio ( $r=0.64$ ;  $p < 0.10$ ) (Fig. 1; App. 1). This suggests a 20-25 % reduction of plant species richness at an exceedance of  $3 \text{ g N m}^{-2} \text{ yr}^{-1}$ . However, the data set is rather restricted at this moment and should be extended in near future to obtain a more sure regression.



**Figure 1.** Relationship between the species richness ratio ( $S_N/S_C$ ) and N exceedance (addition minus mean critical load) in arctic-alpine vegetation (4 countries;  $Y = -0.046X + 0.948$ ;  $r^2 = 0.41$   $p < 0.10$ ;  $n=9$ ).

### *Boreal coniferous forests*

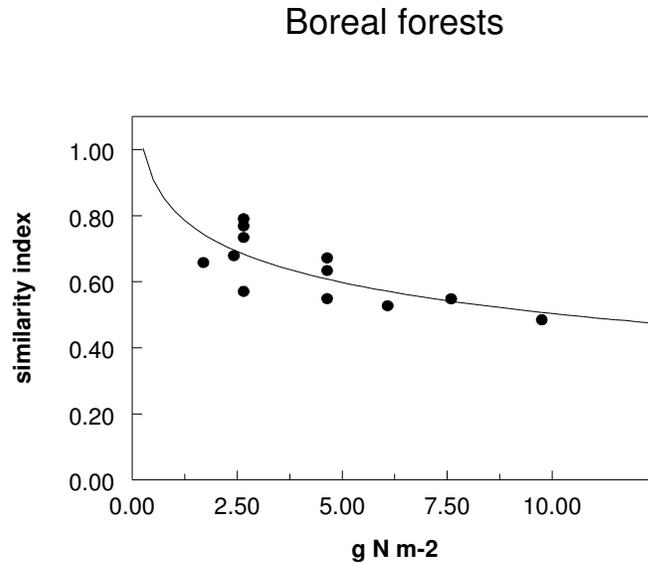
Boreal coniferous forests are probably the most large forest zone of the present World vegetation. Numerous studies have been performed with respect to the influence of acidic deposition. In the last decades the number of studies on the impacts of N deposition increased, but mostly the consequences for tree growth and soil have been determined. In the last 2 decades the impacts on the understory vegetation got more attention, but the number of published studies with species number per plot or full vegetation lists are, however, scarce and fully restricted to the Scandinavian countries (Van Dobben 1993; Kellner 1993; Mäkipää 1998; Van Dobben *et al.* 1999; Shrindo & Okland 2002). Because of the possibility to analyse the original data set of Van Dobben (n=12), the number of data points became rather “large” (n=18). The results for this world vegetation type clearly showed that the species richness ratio was not influenced at all by the amount of N exceedance (Fig. 2), although many studies reported a sharp decline in lichen diversity, a change in bryophyte composition and lower abundance of typical dwarf shrubs. This results reveal that this decline in typical species is counterbalanced by increases in invasive or non-characteristic species of more nutrient-rich habitats.



**Figure 2.** Relationship between the species richness ratio ( $S_N/S_C$ ) and N exceedance (N addition minus mean critical load) in boreal coniferous forests (3 countries; n= 18; not significant).

The data, previously shown, demonstrated that plant species richness is a too restricted indicator for the N effects upon understory vegetation in these boreal forests. However, the detailed comparison of the species composition of the understory in Swedish boreal forests (7 locations, n=12), showed that the similarity of the N treated vegetation significantly decreased (log curve) with the amount of N exceedance Fig. 3). These results indicates that

ca. 30 % of the species has changed with an exceedance of  $2.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ , where as above  $5 \text{ g N m}^{-2} \text{ yr}^{-1}$  45 – 50 % of the species composition is different. Since the species richness did not change (Fig. 2), this means that the original species have been replaced by others!

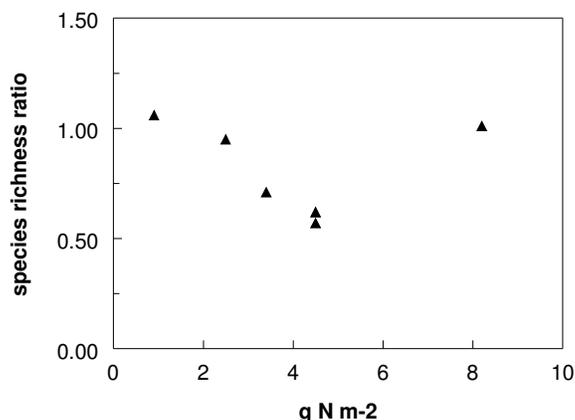


**Figure 3.** Sorensen's similarity index of N-treated understory vegetation compared to the control vegetation in Swedish boreal forests (7 locations;  $Y = 0.815 - 0.135 \ln(X)$ ,  $r^2 = 0.52$ ,  $p < 0.01$ ;  $n=12$ ) against the N exceedance (N addition minus mean critical load).

### *Temperate forests*

Despite the large number of studies of atmospheric deposition in temperate deciduous and coniferous forests, the effects of experimental N enrichment on the species richness of the understory are poorly studied or not published with full information, although many retrospective or correlative studies suggested changes in the understory vegetation. The outcome from two studies in the USA (both coniferous and deciduous) and from two in Europe (both coniferous and deciduous) resulted in only 6 data-points (Hallbäck & Zhang 1998; Pitcairn *et al.* 1998; Thomas *et al.* 1999; Corbin *et al.* 2003). No significant relationship was found in this (too restricted) data set; this might be caused by the outlier at the highest N addition: this is the only result from a short-term experiment (3 years), all other results are from long-term (> 10 years) studies (Fig. 4). Excluding this outlier showed a clear negative linear trend ( $r^2 = 0.946$ ;  $p < 0.05$ ), with ca. 40% reduction at  $4 \text{ g N m}^{-2} \text{ yr}^{-1}$  exceedance (but based upon few results!). This line might be used as testable relationship for this vegetation type in future, but, more studies are clearly needed to validate these results. It seems, however, more appropriate to use also the established relationship for the changes in the understory in boreal forests (Fig. 3) for these forests till enough data are available for temperate forests.

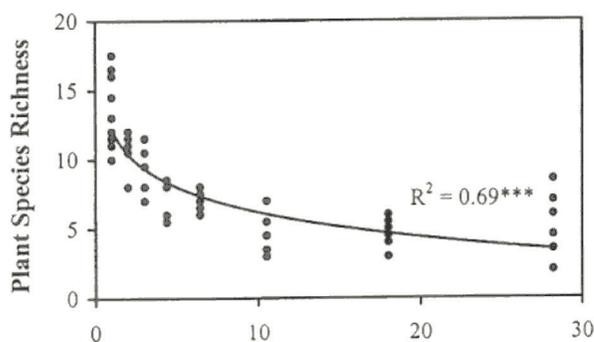
## Temperate Forests



**Figure 4.** Relationship between the species richness ratio ( $S_N/S_C$ ) and exceedance (N addition minus mean critical load) in temperate forests (4 countries;  $n=6$ ; not significant).

### *Temperate grasslands*

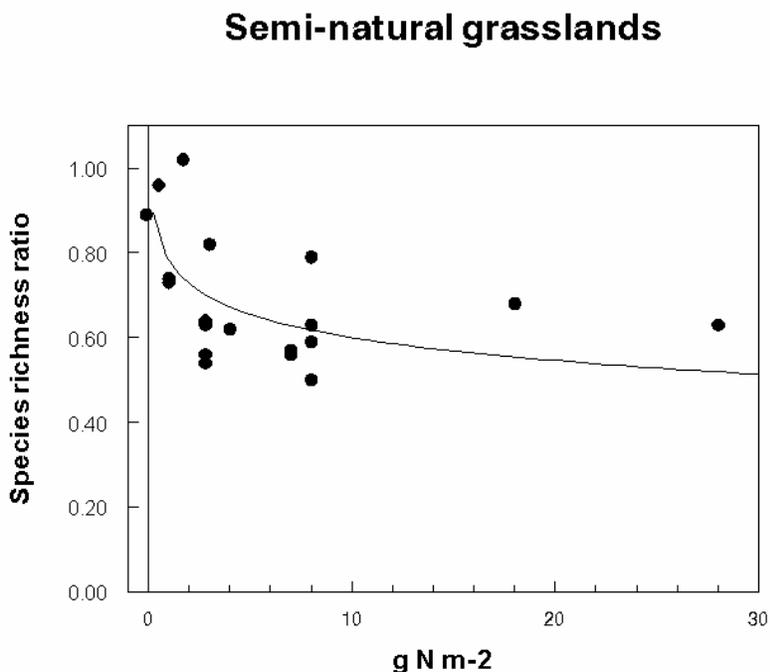
This important vegetation type (also called steppe or prairie) occurred in the temperate continental regions with no natural tree growth because of climatic constraints (dry conditions). Only in Cedar Creek (Minnesota; USA) the effects of sole N additions have been extensively and with a long duration (10 – 14 yrs) studied in three old fields (young – old) on former prairie rangeland and in one natural prairie vegetation by the group of Prof. D. Tilman. This work clearly revealed the major importance of N additions for the structure and function of this temperate grassland, but no separate data have been published on the species composition or species richness of the natural prairie-grassland site. Fortunately, the oldest field has only been in agricultural use till 1935 and is nowadays more or less comparable in species composition with the natural prairie vegetation. Haddad *et al.* (2000) have published the relationship between N additions and plant species richness number in this temperate grassland. They clearly found a negative log relation between species richness and N additions (Fig. 5). Please note, that the fitted curve strongly resembled the relationship demonstrated for semi-natural temperate grasslands in this study (next paragraph)!



**Figure 5.** Plant species richness against N additions (g N m<sup>-2</sup> yr<sup>-1</sup>; not corrected for critical loads) in field C in Minnesota, USA (Haddad *et al.* 2000).

### Temperate semi-natural grasslands

The effects of nutrients on semi-natural (species-rich) grasslands have been studied in many European and some non-European countries. The literature study on the effects of sole N treatments on plant species richness revealed the largest regression data set of this report (n=21)! Results included studies under both dry and wet conditions, or with very different soil pH (acid – calcareous) across Europe (5 countries) and the USA (Williams 1976; Bobbink 1991; Mountford *et al.* 1994; Berlin 1998; Foster & Gross 1998; Joyce 2001; Beltman & Barendregt 2002; Jacquemin *et al.* 2003). A significant negative relationship (log fit) between species richness ratio and N exceedances was found for these mixture temperate, semi-natural grasslands (Fig. 6). These data clearly demonstrated that a reduction of 30-40 % of the species richness occurred within an exceedance of the empirical N critical loads of ca. 5 g N m<sup>-2</sup> yr<sup>-1</sup>. In addition, the negative log curve is in clear accordance with that found in the N addition study in more or less natural prairie in the USA (Fig. 5).



**Figure 6.** Relation between the species richness ratio ( $S_N/S_C$ ) and exceedance (N addition minus mean critical load) in semi-natural grasslands (6 countries; n=21;  $Y = 0.782 - 0.079 \ln(X)$ ,  $r^2 = 0.38$   $p < 0.001$ ).

A remarkable outlier has been excluded from this regression analysis; in the British Park Grass plots 100 years of nitrate addition (exceedance 2.8 g N m<sup>-2</sup> yr<sup>-1</sup>) caused a reduction of the species richness ratio of 45 %, whereas sole ammonium (identical exceedance) led to a drastic reduction of almost 90 %, probably caused by severe soil acidification, in combination with high soil N (Williams 1976). It is thus of importance to incorporate acidification effects separately in future, if possible, especially in areas with high inputs of reduced N pollutants.

### *Heaths and temperate shrublands*

Both lowland and upland *Calluna* heaths have been intensively studied with respect to N deposition. Significant effects of N addition on the growth and development of *Calluna* have been found, besides as opening of the dwarf-shrub canopy because of enhanced heather beetle attacks or winter damage. In some cases, clear shifts between dwarf shrubs and grasses have been demonstrated, besides impacts on lichens and bryophytes, clearly the most sensitive parts of these heaths (e.g. Bobbink *et al.* 2003). Only very few studies studied the effects on plant species richness or presented full species lists of this characteristic vegetation type, probably because of its species-poor nature. Because of this, an experimental relationship between N exceedances and species richness can not be constructed for heaths. Based upon expert judgement it is advised to use the relationship as found for the understorey of boreal coniferous forests (Fig. 3), because of the similarity in species composition (dwarf shrubs, grasses, lichens and bryophytes) between the understorey of this forests and heaths.

### *An azonal intermezzo: terrestrial wetlands*

Many types of terrestrial wetlands occur across the globe, from swamps and reed beds to fens and bogs. In general, it is assumed that several oligotrophic and mesotrophic wetlands can be sensitive to increases of N loads. Recently, the effects of 3-4 years of N additions have been quantified in ombrotrophic bogs. Significant changes in the cover of peat mosses (*Sphagna*), increases in graminoids or *Betula* spp have been found, but no data on species richness were given. In addition, changes in the moss layer and sedge biomass were reported in poor and rich fens studies under high N fertilization, but only one study presented data on the species richness of these fens (Bergami & Pauli 2001; Pauli *et al.* 2002). After two years of 7.5 - 8 g N m<sup>-2</sup> yr<sup>-1</sup> exceedance, they did not find a significant effect on vascular species richness, but a 9 % reduction of bryophytes species richness. It is thus clear that no regression between N exceedances and species richness can be given for terrestrial wetlands. Because of the structural similarity of these oligo- to mesotrophic wetlands with semi-natural grasslands, it is suggested to use the quantified relationship of this system, because of the precautionary principle.

### *Mediterranean vegetation*

Mediterranean vegetation is characterised by evergreen shrubs and sclerophyll trees, forming typical shrublands or low forest stands. These communities have been adapted to the distinctive climatic conditions, with summer drought and cool moist winters (Archibold 1995). Some studies recently reported increased N loads in Mediterranean vegetation in Spain or California (Padgett & Allen 1999; Sanz *et al.* 2000), but the effects of experimentally increased N loads on species richness have not been quantified till present. At this stage it is thus not possible to present a quantitative relationship between species richness of Mediterranean vegetation and increased N loads, and even a judgement based upon only expert knowledge seems unrealistic at this moment.

### *Arid vegetation (deserts)*

No data were found between increased N availabilities and plant species richness in hot or cold deserts. It is, however, likely that these desert ecosystems are not sensitive to increased N loads, and, in addition, they are only present in regions with very low N deposition. Quantification of the N impacts in these systems thus not possible, but also not necessary for this world vegetation type, because of its location in regions with very low background N deposition.

### *Tropical savannas*

No data on the effects of realistic N additions on the plant species richness or species composition of tropical savannas could be found in literature, although some studies revealed nutrient limitation (N and/or P) of the tropical Savanna vegetation (Bennett & Adams 2001; Snyman 2002). Based upon expert judgement, it could be suggested to use the negative log-fitted curve as quantified for temperate (semi-natural) grasslands, because of the similarity in grass structure, although the (possible) effects upon tree species number are not incorporated.

### *Tropical forests*

It is generally assumed that wet lowland tropical forests (including rain forests) are limited by P because of their very old weathered soils, whereas wet montane tropical forests are most likely limited by N (e.g. Tanner *et al.* 1998). The influence of N or P inputs on dry (seasonal) tropical forests is not known, but probably very low. It is thus likely that wet montane forests are most sensitive to N deposition in the tropics. Osterlag & Verville (2002) applied 10 g N m<sup>-2</sup> yr<sup>-1</sup> to a relative young and old wet montane forest on Hawaii and found an significant increase of non-native invaders in the youngest stand. This may indicate that species composition can be influenced by increased N loads, but no data are present to quantify the aimed relation between N exceedance and plant species richness for these tropical forests. At this stage it is unrealistic to present a relationship between species richness in tropical forests and increased N loads, even based upon expert judgement only.

## **Chapter 4 Concluding remarks**

Within the UN/ECE Convention on Long-range Transboundary Air Pollution, empirical procedures were developed to set critical loads for atmospheric N deposition. Based on observed changes in the structure and function of ecosystems, empirical N critical loads have been established for European natural and semi-natural ecosystems (e.g. Achermann & Bobbink 2003). It seems clear that the exceedance of the set empirical N critical loads should be prevented for protection of the structure and function of natural and semi-natural ecosystems. At present, the causal relationship between the amount of exceedance of these N critical loads and the biodiversity of these systems was hardly quantified. The central aim of this inventory study was to reveal this relationship for major world ecosystems. As expected, this study became restricted to plant species richness, because experimentally quantified data for fauna is extremely rare.

The effects of experimentally enhanced on plant species richness was expressed as the species richness ratio, i.e. the ratio between the number of plants in the N-treated situation and in the control situation ( $S_n/S_c$ ; see page 8). Only for semi-natural grasslands and arctic/alpine ecosystems (almost significant) a negative relationship between the exceedance of the empirical N loads and the species richness ratio was found. Remarkably, the quantified relationship between N exceedance across a range of semi-natural grasslands in Europa strongly reflected the observations in temperate grassland in the USA (Haddad *et al.* 2000). In both studies, a significant negative log function was the best fit for the observed data points, indicating the possible soundness of the observed relations. For temperate forests, only an indication for a negative relationship between N exceedance and species richness was found, probably biased by the small number of observations. As in semi-natural grasslands, the number of studies in boreal coniferous forests was large enough to come to a quantitative relationship between species richness and exceedance. The species richness ratio was clearly not related with increasing N exceedances (Fig. 2). However, because of the availability of a full data set, it became after comparing the similarity of the species composition in the N-treated situation with the control vegetation, that a significant negative log relationship could be demonstrated (Fig. 3). This clearly suggest that typical species may be replaced by “invasive” species, without changing the overall species richness. It is suggested that the **number of characteristic plant species** is a much better indication for plant diversity in a given group of ecosystems with respect to environmental stresses, but it was in this framework not possible to analyze almost all publications in this way; this clearly needs much more effort and, in many situations, the availability of the original data sets. Finally, it became obvious that for many world vegetation types (e.g. subtropical, tropical, Mediterranean and deserts) hardly or no data were useable, but even for some (relatively) well-studied systems (heaths and bogs) the effects upon species richness (or number of characteristic species) was hardly published.

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