umweltbundesamt

EFFECTS OF NITROGEN AND SULPHUR DEPOSITION ON FORESTS AND FOREST BIODIVERSITY

Austrian Integrated Monitoring Zöbelboden

Thomas Dirnböck, Michael Mirtl,
Stefan Dullinger, Maria-Theresia Grabner,
Peter Hochrathner, Karl Hülber,
Gerhard Karrer, Ingrid Kleinbauer,
Wolfgang Mayer, Johannes Peterseil,
Veronika Pfefferkorn-Dellali,
Friedrich Reimoser, Susanne Reimoser,
Roman Türk, Wolfgang Willner,
Harald Zechmeister

REPORT
REP-0077

Vienna, 2007

Project management

Michael Mirtl, Umweltbundesamt

Authors

Thomas Dirnböck, Umweltbundesamt
Michael Mirtl, Umweltbundesamt
Stefan Dullinger, VINCA – Vienna Institute for Nature Conservation & Analysis
Maria-Theresia Grabner, Umweltbundesamt
Peter Hochrathner, ORCHIS – Technisches Büro für Biologie
Karl Hülber, VINCA – Vienna Institute for Nature Conservation & Analysis
Gerhard Karrer, Universität für Bodenkultur
Ingrid Kleinbauer, VINCA – Vienna Institute for Nature Conservation & Analysis
Wolfgang Mayer, Universität Salzburg
Johannes Peterseil, Umweltbundesamt
Veronika Pfefferkorn-Dellali, Universität Salzburg
Friedrich Reimoser, Veterinärmedizinische Universität Wien
Susanne Reimoser, Veterinärmedizinische Universität Wien
Roman Türk, Universität Salzburg
Wolfgang Willner, VINCA – Vienna Institute for Nature Conservation & Analysis
Harald Zechmeister, Universität Wien

Translation

Thomas Dirnböck, Umweltbundesamt
Brigitte Read, Umweltbundesamt

Lektorat

Brigitte Read, Umweltbundesamt

Layout and typesetting

Ute Kutschera, Umweltbundesamt

Title photograph

Michael Mirtl, Umweltbundesamt

For further information about the publications of the Umweltbundesamt please go to:
http://www.umweltbundesamt.at/

Imprint

Owner and Editor: Umweltbundesamt GmbH
Spittelauer Lände 5, 1090 Vienna/Austria

Printed by: Umweltbundesamt GmbH

Printed on recycled paper.

ISBN 3-85457-874-1

TABLE OF CONTENTS

EXECUTIVE SUMMARY 5

KURZFASSUNG 7

EXECUTIVE SUMMARY FOR POLICY MAKERS 9

Relevant political initiatives 9

Trends of Sulphur and Nitrogen deposition 9

Effects of Nitrogen and Sulphur on biodiversity 10

1 INTRODUCTION 13

2 UNECE ICP-INTEGRATED MONITORING 15

3 THE AUSTRIAN IM SITE 17

3.1 Study area description 17

3.2 Spatio-temporal data 18

4 NITROGEN AND SULPHUR DEPOSITION 21

4.1 Measurements and models 21

4.2 Spatial distribution and temporal trends 22

4.3 Confounding environmental and human factors 25

4.3.1 Climate 25

4.3.2 Large herbivores 27

4.3.3 Forest and management 28

4.3.4 Forest tree layer changes 28

5 EFFECTS ON SOILS AND BIOINDICATORS 32

5.1 Soil changes 32

5.1.1 Results 32

5.1.2 Conclusion 35

5.2 Forest floor vegetation 35

5.2.1 Results 35

5.2.2 Conclusion 37

5.3 Bryophytes 37

5.3.1 Results 37

5.3.2 Conclusion 39

5.4 Lichens 39

5.4.1 Results 40

5.4.2 Conclusion 42

5.5 Birds 42
5.5.1 Results 42
5.5.2 Conclusion 42

6 EFFECTS ON BIODIVERSITY 43
6.1 Results 43
6.2 Conclusion 47

7 SYNTHESIS AND FUTURE SCENARIOS 48

8 REFERENCES 50

9 APPENDIX 55
9.1 Methodological details 55
9.1.1 Soil 55
9.1.2 Forest floor vegetation 56
9.1.3 Bryophytes 57
9.1.4 Lichens 58
9.1.5 Birds 58
9.1.6 Effects on biodiversity 59

EXECUTIVE SUMMARY

Nitrogen (N) and sulphur (S) emissions increased dramatically during the second half of the 20th century causing excess deposition of N and S in natural and semi-natural ecosystems. Both excess N and S deposition have a wide array of detrimental effects in forest ecosystems and their biodiversity. Unlike N emissions, which remain at a level which is too high, S emissions have been abated successfully in Europe through internationally ratified protocols (UNECE/CLRTAP) and related measures.

In the Austrian forest ecosystems, critical loads – the threshold for long term deposition – for deteriorating effects (eutrophication) of N are exceeded particularly in areas with elevated precipitation and thus high deposition. Deposition of S decreased substantially during the last 30 years so that N deposition is to date the predominant agent for acidification. Against this background, the UNECE-ICP Integrated Monitoring (IM) site „Zöbelboden" can serve as a reference for current N and S effects in montane, calcareous forest ecosystems. Within a catchment of 90 ha, air pollutants, climate and a wide array of bioindicators are measured in the long term across ecosystem compartments. The N deposition at the ICP IM site is as much as twice the critical load of deteriorating eutrophication effects in temperate deciduous forests. On the other hand, S deposition has decreased substantially in the area. In this report, we focus specifically on the combined effect of eutrophication and acidification through airborne N and the release from acidification through airborne S between 1992 and 2005. We analyse long-term observational data on soil chemical characteristics and bioindicators: forest floor vegetation, bryophytes, lichens, and birds. In addition, biodiversity and its changes are assessed across these organisms.

The soils responded to the long-term trends in the deposition of airborne pollutants. Excess N deposition caused soil eutrophication and the decrease of S deposition resulted in a significant, but soil-specific, recovery from acidification.

The detected trends of soil properties were not unambiguously reflected in changes of forest floor vegetation. Though nitrophilous species increased in abundance, the overall change of the species composition of forest floor vegetation is only weak. However, the weak response predominates under intermediate site conditions, which may mask significant changes on oligotrophic sites. As a likely long-term consequence, oligotrophic species will probably disappear from the regional species pool due to the erosion of oligotrophic sites.

Terrestrial and epiphytic bryophyte species remained rather stable in their overall abundance. The observed changes, which are restricted to single species, can only to some extent be attributed to effects of airborne N and S pollution. The bryophyte communities as a whole however did not show directional changes attributable to the observed amounts of N deposition and the decrease of S deposition. However, the spatial distribution of bryophyte communities is related to different deposition regimes of airborne N and S, which indicates long-term chronic effects.

Long-term airborne N and S deposition had a significant impact on epiphytic lichens. The overall abundance of epiphytic lichens and those of sensitive species decreased. Some sensitive species even became extinct. Since some new species colonised at the permanent plots this did not lead to an overall decrease of lichen diversity. Lichen communities show a deteriorating development in response to air pollution and are becoming increasingly dissimilar from communities typical of clean air conditions. The impact from airborne pollution increased continuously from 1993 to 2005. Currently, all plots are affected. The plots indicating low impact decreased and those

indicating medium and strong impact increased. Though acid deposition through S decreased, epiphytic lichens did not recover. Additionally, lichens show clear signs of eutrophication due to excess N deposition.

The relatively short-term data on changes in the abundance and distribution of birds does not allow clear conclusions about effect related trends.

In general, biodiversity declined from the beginning of the 1990s until the year 2005. With only one exception – epiphytic lichens – this trend is consistent for species numbers of the entire study area, and for mean species numbers at the scale of permanent plots. Biodiversity is controlled by a range of interlinked factors with air pollution as a significantly negative contributor. Tree layer diversity is one of the main factors controlling the diversity of the entire forest ecosystems. Linkages also exist between different trophic levels, such as the tree and forest floor diversity, which favours the coexistence of a high diversity of bird species. Though long-term trends of cross-taxon biodiversity were observed, these trends should be interpreted with caution due to their low significance. Compared with the pronounced variability of diversity between permanent plots, the overall temporal trends are rather weak. What needs thus to be further observed in future is if the decline of biodiversity continues and to which extent air pollution can be made responsible.

The present study stresses the importance of the simultaneous measurement of all these influences together with a wide array of bioindicators. The application of single indicators may lead to wrong conclusions about ecosystem changes. Though many of the above factors were dynamic and thus co-influenced the behaviour of bioindicators, we were able to identify the single effect of airborne N and S deposition. It can be concluded from these results, and from other studies, that continuous eutrophication takes place in large parts of the Northern Limestone Alps in Austria where airborne pollution levels are elevated. If N emission is not abated efficiently biodiversity loss and effects on ecosystem functions is a likely future scenario. Even with the prospect of an eventual reduction of N emissions, ecosystem changes will still occur due to the time-lag inherent in recovery processes. The missing recovery from acidification of some bioindicators in response to the decreased levels of airborne S pollution exemplifies the time-lags which can be expected.

Notwithstanding the urgency of the eutrophication issue for biodiversity, programmes for the assessment of measures to reduce emissions at the European and the national level in response to the CBD 2010 target – the halt of biodiversity loss by 2010 – have just been started. The initiative ”Streamlining European 2010 Biodiversity Indicators” (SEBI2010) is making an important step towards linking Pan-European’s clean air (UNECE) and biodiversity policy. An efficient implementation of European initiatives at the national scale would be achieved through the Austrian biodiversity monitoring scheme (MOBI). A national-scale assessment of the effects of N deposition in natural and semi-natural ecosystems would be a first step towards setting up targeted abatement measures.

KURZFASSUNG

Emissionen von Stickstoff- und Schwefelverbindungen haben in der zweiten Hälfte des 20. Jahrhunderts drastisch zugenommen. Dies führte zu überhöhten Einträgen in natürliche und naturnahe Ökosysteme. Beide Stoffe zeigen eine Reihe von schädlichen Wirkungen auf Ökosysteme und die biologische Vielfalt. Auf Basis international ratifizierter Protokolle (UNECE/CLRTAP) konnten Schwefelemissionen in Europa erfolgreich reduziert werden. Im Gegensatz dazu sind Stickstoffemissionen immer noch zu hoch.

In den österreichischen Wäldern werden die "Critical Loads" (kritische, langfristige Stoffeinträge mit schädlicher Wirkung) in Bezug auf Nährstoffanreicherung durch Stickstoff vor allem in Regionen mit hohen Niederschlagsmengen überschritten. Saure Einträge durch Schwefel haben in den letzten 30 Jahren stark abgenommen, so dass heute vor allem Stickstoffdeposition zur Versauerung beiträgt. In Anbetracht dieser Situation dient der UNECE-ICP Integrated Monitoring Standort „Zöbelboden" als Referenzsystem für montane Karbonatwälder. Im 90 ha umfassenden Untersuchungsgebiet werden Luftschadstoffe, Klima und eine Reihe von Bioindikatoren über lange Zeiträume gemessen. Die Auswertungen zeigen, dass Critical Loads für Stickstoffeinträge um bis zum Doppelten überschritten werden. Saure Einträge durch Schwefel haben stark abgenommen. Der vorliegende Report bewertet die Effekte von Nährstoffanreicherung und Versauerung durch Stickstoffeinträge bei gleichzeitiger Abnahme saurer Einträge durch Schwefel im Zeitabschnitt von 1992 bis 2005. Dazu werden Langzeitdatenreihen zum Bodenchemismus und Bioindikatoren wie Waldbodenvegetation, Moose, Flechten und Vögel herangezogen. Zusätzlich wird die Vielfalt der verschiedenen Organismengruppen und ihr zeitlicher Trend bewertet.

Die Böden des Untersuchungsgebietes zeigen eindeutige Veränderungen aufgrund von Luftschadstoffeinträgen. Die überhöhten Stickstoffeinträge führen zu Nährstoffanreicherung, die verringerten Schwefeleinträge zu einer gewissen bodenspezifischen Erholung von der vorhergehenden Versauerung.

Die Änderungen des Bodenchemismus spiegeln sich nur undeutlich in den Änderungen der Waldbodenvegetation wider. Obwohl Stickstoffzeiger zunahmen, kann in der Artenzusammensetzung der Waldbodenvegetation nur eine schwache Reaktion nachgewiesen werden. Der schwache Trend betrifft vor allem die, in Bezug auf ihre Nährstoffverhältnisse, durchschnittlichen Standorte. Die Bodenvegetation auf nährstoffarmen Standorten reagiert hingegen stärker. Es erscheint daher wahrscheinlich, dass die Arten der nährstoffarmen Standorte langfristig gefährdet sind.

Die Häufigkeit der verschiedenen Moosarten ist eher stabil. Nennenswerte Änderungen betreffen nur einige wenige Arten und dürfen nur teilweise den Effekten von Stickstoff- und Schwefeleinträgen zugewiesen werden. Die zeitlichen, in den meisten Fällen ungerichteten, Änderungen der Artenkombinationen sind natürliche Populationsschwankungen. Allerdings hängt die Ausprägung der Moosgesellschaften sehr wohl von der Eintragsmenge von Stickstoff und Schwefel ab. Hier scheint es sich um einen langfristigen, zum größten Teil bereits abgelaufenen Effekt von Luftschadstoffen zu handeln.

Stickstoff- und Schwefeleinträge üben einen starken negativen Einfluss auf die epiphytische Flechtenvegetation aus. Die Häufigkeit von Flechten nimmt insgesamt ab, wobei sensible Arten stärker abnehmen und einige wenige völlig aussterben. Die Vielfalt an Flechten nimmt jedoch statistisch nicht ab, da neue Arten hinzukommen. Auch die Flechtengesellschaften verändern sich klar durch Luftschadstoffe und sind

immer weniger mit typischen Flechtengesellschaften von Reinluftgebieten vergleichbar. Die Stärke der Immissionswirkung nimmt von 1993 bis 2005 zu. Heute sind alle Dauerbeobachtungsflächen negativ beeinflusst, nur mehr wenige schwach belastet und die Schadklassen mit mittlerer und starker Einwirkung zunehmend. Die nachlassenden sauren Schwefeleinträge führen somit zu keiner Erholung der epiphytischen Flechten. Flechten zeigen darüber hinaus eindeutige Schädigungen durch Nährstoffanreicherung aufgrund von Stickstoffeinträgen.

Die relativ kurze Datenreihe zur Häufigkeit und Verteilung von Vögeln erlaubt noch keine Schlussfolgerungen zu wirkungsbezogenen Trends.

Die Vielfalt der untersuchten Indikatorarten nimmt seit dem Beginn der 90er Jahre bis 2005 stetig ab. Dieser Trend ist mit der Ausnahme von epiphytischen Flechten in der gesamten Artenzahl des Untersuchungsgebietes und der Diversität pro Dauerfläche abgebildet. Einträge von Stickstoff und Schwefel haben negative Folgen für die biologische Vielfalt. Sie sind aber nur ein Glied in der komplexen Wirkungskette bestimmender Faktoren der biologischen Vielfalt. Die Vielfalt der Baumschicht bestimmt zu einem hohen Maß die Vielfalt der Wälder. Zudem sind trophische Ebenen miteinander verknüpft, so dass beispielsweise die Vielfalt der Vegetation jene der Vögel mitbestimmt. Da der Trend im Vergleich zur natürlichen Variabilität im Gesamtgebiet schwach ist, lassen nur längere Datenzeitreihen gesicherte Schlussfolgerungen zu den wirkungsspezifischen Effekten von Luftschadstoffen auf die biologische Vielfalt im Gebiet zu.

Die vorliegende Ökosystemstudie verdeutlicht die Wichtigkeit der simultanen Messungen der bestimmenden Wirkungsfaktoren und einer Reihe von Bioindikatoren. Unter Verwendung nur einzelner Indikatoren könnten leicht falsche Schlussfolgerungen zu Ökosystemveränderungen gezogen werden. Obwohl viele Faktoren einer zeitlichen Änderung unterliegen, können die singulären Effekte von Stickstoff- und Schwefeleinträgen in das Waldökosystem des ICP-Integrated Monitoring Standorts „Zöbelboden“ nachgewiesen werden. Auf Basis unserer Ergebnisse und vieler anderer Studien kann angenommen werden, dass die Waldökosysteme der Nördlichen Kalkalpen Österreichs durch die andauernden überhöhten Einträge einer Nährstoffanreicherung ausgesetzt sind. Werden die hohen Stickstoffemissionen nicht erfolgreich reduziert, ist mit einem Verlust an biologischer Vielfalt und Auswirkungen auf die Ökosystemfunktionen zu rechnen. Aufgrund der typisch langen Reaktionszeiten werden auch bei einer Reduktion von Stickstoffeinträgen Ökosystemverbesserungen erst verzögert eintreten.

Ungeachtet der Dringlichkeit des Problems von Stickstoffeinträgen für die biologische Vielfalt haben politische Initiativen zur Bewertung von Emissionreduktionsmaßnahmen gerade erst begonnen. Die wesentliche umweltpolitische Grundlage dafür ist das „2010 Ziel zur signifikanten Reduktion des Verlustes biologischer Vielfalt der CBD (UN Convention on Biological Diveristy)“. Das Pan-Europäische Programm “Streamlining European 2010 Bio-diversity Indicators” (SEBI2010) versucht durch die Entwicklung von passenden Indikatoren die internationalen umweltpolitischen Werkzeuge der Luftreinhaltung mit jenen zum Schutz der biologischen Vielfalt zu koppeln. In Österreich könnten diese Initiativen vor allem durch ein Österreichisches Biodiversitätsmonitoring umgesetzt werden. Eine Bewertung von Effekten überhöhter Stickstoffeinträge in die natürlichen und naturnahen Ökosysteme in Österreich ist ein erster Schritt für zielgerichtete umweltpolitische Maßnahmen.

EXECUTIVE SUMMARY FOR POLICY MAKERS

Relevant political initiatives

A recent UN study on global biodiversity and the services it provides to human society (MEA 2003) highlights that most of these services are declining, both in the EU and globally. The study also states that this decline could be reversed with substantial changes in policy and practice. The policy framework of air pollution and its impact on biodiversity rests upon two major pillars, the Convention on Biodiversity (CBD) and the Convention of Long-Range Transboundary Air Pollution (CLRTAP).

European air pollution policy

Europe's air pollution policy, particularly with regard to N and S emissions, is based on the UNECE CLRTAP (www.unece.org/env/lrtap) of 1979. Emission ceilings for SO_2, NO_x, NH_3 were set by the 1999 Gothenburg Protocol to abate acidification, eutrophication and ground-level ozone. With regard to N and S Critical Loads for detrimental effects for natural and semi-natural ecosystems were defined. Once the protocol is fully implemented, Europe's S emissions should be cut by at least 63%, its NO_x emissions by 41%, and its ammonia emissions by 17% compared to 1990. According to the Austrian legislation, critical levels of N and S are set by the Air Quality Protection Act (IG-L; Federal Legal Gazette I 115/1997). Furthermore, the EU Directive on National Emission Ceilings sets national limits of emission of acidifying substances which were transposed into Austrian legislation in 2003 (Federal Act on National Emission Ceilings (EG-L; Federal Legal Gazette I No 34/2003)).

European biodiversity policy

In the international arena, the EU's focus has been on strengthening the Convention on Biological Diversity (CBD). At the EU level biodiversity objectives are integrated in a wide range of environmental and sector policies. An EC Biodiversity Strategy was adopted in 1998 and related Action Plans in 2001. In 2006 a Road Map was adopted by the member states of the EU to halt the loss of biodiversity by 2010. One target specifies measures to reduce pollutant pressures on terrestrial and freshwater biodiversity by 2010. Most member states have also developed, or are developing, strategy plans. In Austria a strategy plan for the implementation of the CBD was developed in 1998 (www.biodiv.at/chm). The EU has made significant commitments with regard to the international target – the 2010 target – to reduce biodiversity loss. EU heads of state or government agreed in 2001 "to halt the decline of biodiversity by 2010" and to „restore habitats and natural systems" (http://eur-lex.europa.eu/LexUriServ/site/en/com/2006/com2006_0216en01.pdf). The Austrian strategies plan is now being updated with reference to the 2010 target.

Trends of Sulphur and Nitrogen deposition

Successful reduction of Sulphure emissions in response to political measures

Emissions of SO_2 were successfully reduced in Europe (EMEP 2006). S levels are continuously decreasing and first signals of a recovery from acidification of soils have been observed (FORSIUS et al. 2001, SLIGGERS & KAKEBEEKE 2004, BFH 2006, KLEEMOLA & FORSIUS 2006). In Austria, emission reductions of S and deposition in forest ecosystems follow this decreasing trend. By the year 2004, a reduction of 61% compared to 1990 was achieved. (www.umweltbundesamt.at/umweltschutz/luft). Acid deposition could be reduced to below critical loads for almost all forests in Austria (SMIDT & OBERSTEINER, submitted).

Emission of Nitrogen is far beyond the political targets

Emission reductions of total N are far from reaching the targets set by the CLRTAP (EMEP 2006). On the contrary, N emissions and thus deposition in natural and semi-natural ecosystems remain too high (FORSIUS et al. 2001, SLIGGERS & KAKEBEEKE 2004, BFH 2005, KLEEMOLA & FORSIUS 2006). In Austria, current NO_x emissions are above those in the year 1990 (7% in 2004) with traffic as the main source (www.umweltbundesamt.at/umweltschutz/luft). NH_3 emissions from agriculture were reduced in Austria so that current emissions almost meet the targets (7% reduction in the period 1990-2004, www.umweltbundesamt.at/umweltschutz/luft). The situation regarding the exceedance of critical loads for eutrophication effects in Austria is controversial. Although standard critical load modelling shows exceedances for 97% of the Austrian forest area (UMWELTBUNDESAMT 2005), deposition measurements at ICP forest level II plots show that exceedances are limited to only a few sites (SMIDT & OBERSTEINER submitted). The main reason for such divergent results is that the way deposition paths of airborne pollutants in forests are considered for measurements is different from that of modelling. This points to a need for standardization. Nevertheless, ICP forest level II sites from the Northern Limestone Alps also show elevated measured N deposition, which is in line with the data from the ICP IM site Zöbelboden.

Effects of Nitrogen and Sulphur on biodiversity

Effects of excess N on ecosystems are far more problematic than S to date

Owing to the current trends of emissions, effects of excess N on ecosystems are far more problematic than S effects. Numerous examples underpin the negative impact excess N depositions have on biodiversity. The global assessment study, the Millennium Ecosystem Assessment, which was launched by the UN, states that eutrophication through N deposition will be one of the three major threats for biodiversity besides land use and climate change (MEA 2005). Notwithstanding the urgency of the eutrophication problem, programmes for the assessment of measures taken at the European and the national level in response to the 2010 target have just been started. The initiative "Streamlining European 2010 Bio-diversity Indicators" (SEBI2010), which was initiated by the European Environment Agency (EEA), UNEP, and the European Centre for Nature Conservation as well as experts from a wide range of institutes are developing policy-relevant indicators for the needs of the 2010 target (http://biodiversity-chm.eea.europa.eu/information/indicator/F1090245995). One of a series of headline indicators is nitrogen deposition. Exceedance of critical loads for eutrophication effects in sensitive ecosystems will be used and calculations should be done through the ICP Mapping and Modelling activities of the CLRTAP. With its implementation, there would be a much better link between European's clean air and biodiversity policies than there has been so far. The Austrian biodiversity monitoring initiative (MOBI) also suggests N deposition as a pressure indicator (http://umwelt.lebensministerium.at/article/articleview/48562/1/6914/).

No clear picture about the impact of excess N on biodiversity is available for Europe

Though a series of initiatives exists, no clear picture about the impact of excess N on biodiversity is currently available, neither for Europe nor at the national scale. Some information is provided by the International Cooperation Programmes (ICP) of the CLRTAP (e.g. from ICP forest BFH 2006). Assessments of long-term ICP forest data at the national scale were done for example in Italy (FERRETTI et al. 2006) and Germany (AUGUSTIN et al. 2005). Long-term deposition trends in forest ecosystems (ICP forest level II) throughout Austria were analysed recently by (SMIDT &

OBERSTEINER, submitted) but an analysis of the effect of airborne pollution on biodiversity is still lacking. This is due to the fact that long-term monitoring data suitable for assessing changes is very scarce.

The results from the ICP IM site Zöbelboden provides a reference for montane calcareous forests

The ICP IM site Zöbelboden is a reference site for montane calcareous forests of the Northern Limestone Alps. Here, deposition above the site specific critical load caused clear signs of eutrophication and the decrease of S deposition has led to some, but not at strong, recovery from acidification. N emissions have to be lowered substantially in order to combat biodiversity loss in the Northern Limestone Alps of Austria. The present study also shows the complex relationships between N and S deposition which are specific to site conditions. This should be taken into account when developing indicators. The different indicators exhibit fast (e.g. chemical soil conditions), slow (e.g. forest floor vegetation) and controversial (e.g. soil pH) changes depending on soil and forest type. The trends of diversity can be different when different taxa and/or indices are taken into account. In addition, change of diversity is scale dependent. Regional scale diversity may still decline even though the change in the mean plot diversity is insignificant. When establishing monitoring schemes, several bioindicators should be used and the focus should not only be the plot scale but also larger areas and regions.

1 INTRODUCTION

Airborne Nitrogen and Sulphur have detrimental effects in ecosystems

Nitrogen (N) and sulphur (S) emissions have increased dramatically during the second half of the 20th century causing excess deposition of N and S in natural and semi-natural ecosystems (BOUWMAN et al. 2002, ERISMAN et al. 2002, GALLOWAY et al. 2004). Both excess N and S deposition have a wide array of detrimental effects in ecosystems. Chronic N excess disrupts the nutrient balance in soils, increases the susceptibility to parasite attacks, increases the emissions of nitrogenous greenhouse gases from the soil, and elevates nitrate loss to groundwater (FENN et al. 1998, FLÜCKIGER & BRAUN 1998, ERISMAN & DEVRIES 2000, ABER et al. 2001, HERMAN et al. 2001). Significant changes of soils, lichen, bryophyte and vascular plant species composition in response to chronic N deposition were found in many ecosystems (BOBBINK et al. 2003). N excess has a negative impact on biological diversity and, as N emissions are predicted to increase globally during the decades to come, will be a major threat for biodiversity in future (BOUWMAN et al. 2002, MEA 2005; SUDING et al. 2005, PHOENIX et al. 2006). Increased S deposition, causing soil acidification, was recognised as a major environmental problem in the 1970s (BOUWMAN et al. 2002). As for N, detrimental effects on living organisms were found in freshwater and terrestrial ecosystems (VANDOBBEN et al. 1999, ASHENDEN 2002, LEGGE & KRUPA 2002, PAGE et al. 2004). Unlike N emissions, S emissions were successfully abated in Europe through internationally ratified protocols (UN-ECE/CLRTAP) and related measures. S levels have thus been decreasing continuously over the last 25 years (FORSIUS et al. 2001, SLIGGERS & KAKEBEEKE 2004, BFH 2005, BFH 2006, EMEP 2006, KLEEMOLA & FORSIUS 2006, ROGORA et al. 2006). How ecosystems may recover from acidification is largely unknown (BEIER et al. 2003).

Nitrogen deposition is a major threat for biodiversity

The Austrian UNECE Integrated Monitoring site Zöbelboden

In Austria the intensive ecosystem monitoring site „Zöbelboden" was established in 1992 as part of the UNECE International Co-operative Programme „Integrated Monitoring of Air Pollution Effects on Ecosystems" (ICP-IM) (www.umweltbundesamt.at/im). The site also forms part of the Austrian Long-term Ecosystem Research Network (LTER-Austria). Within a catchment of 90 ha the EMEP standard set of air pollutants, climate, a wide array of bioindicators and other parameters are measured in the long term across ecosystem compartments. The estimated total deposition at the Austrian ICP-IM site – including wet, cloud, fog and dry deposition – ranged between 28–43 kg $N*ha^{-1}*yr^{-1}$ and 10–18 kg SO_4-$S*ha^{-1}*yr^{-1}$ in the years between 1999 and 2002 for different forest types (KALINA & ZAMBO 2003). The N deposition is thus as much as twice the critical load – the threshold for long term deposition – of deteriorating eutrophication effects in temperate deciduous forests (BOBBINK et al. 2003). Following broader trends of emission reduction, S deposition has decreased substantially in the area over the last decade. Though acid deposition is less harmful in calcareous soils, topsoil acidification can still occur.

Objectives of the report

In Austria's forest ecosystems, critical loads for deteriorating effects (eutrophication) of N are exceeded particularly in areas with elevated precipitation. The current levels of airborne S are thought to be negligible for acidification but high levels of N acids play an important role (UMWELTBUNDESAMT 2005, SMIDT & OBERSTEINER submitted). Against this background, the ICP-IM site „Zöbelboden" serves as a reference for current N and S effects in montane, calcareous forest ecosystems. In this report, we focus specifically on the combined effect of eutrophication and acidification through airborne N and the release from acidification through airborne S between 1992 and 2005. We analyse long-term observational data on soil chemical characteristics and a wide array of bioindicators: forest floor vegetation, bryophytes, lichens, and birds. In addition, biodiversity and temporal biodiversity changes are assessed for all of these organisms.

Overview of the report

The present report is a synthesis of individual publications focused on each bioindicator which have been published already or are in preparation. The report starts with an executive summary (page 5 and 7) and a separate summary for policy makers (page 9). An introduction is then given in the UNECE ICP Integrated Monitoring programme, an area description and an overview of the data in chapter 2 and 3. This is followed by the results of the long-term measurement of airborne N and S deposition at the ICP-IM site „Zöbelboden". Other environmental and anthropogenic determinants of the forest ecosystems are shortly described in chapter 4. Chapter 5 is the core of the report with results of the changes found in the soils' chemical characteristics, bioindicators and biodiversity. Each section provides a short overview of the method, the main results and conclusions. Chapter 7 is integrating the individual results and states on the possible future scenario. Details of the applied methods are given in chapter 9 in the Appendix.

2 UNECE ICP-INTEGRATED MONITORING

The aims of integrated ecosystem monitoring

The integrated monitoring of ecosystems refers to the simultaneous measurement of physical, chemical and biological properties of an ecosystem over time and across compartments at the same location. The Integrated Monitoring programme (ICP IM; www.environment.fi/default.asp?node=6318&lan=EN) forms part of the effects monitoring strategy under the Long-range Transboundary Air Pollution Convention (LRTAP) of the United Nation's Economic Commission for Europe (UNECE; www.unece.org/env/lrtap).

The goal was originally to determine and predict the state and change of terrestrial ecosystems – ideally small, 10–1000 ha catchments – in a long-term perspective with respect to the impact of air pollutants, especially N and S. This was to provide a basis for decisions on emission control and an assessment of the ecological impact of such control within the LRTAP convention. Owing to its multi-disciplinary setting, the full implementation of ICP IM will allow for determination of the ecological effects of tropospheric ozone, heavy metals and persistent organic substances. In addition, it contributes to meeting international data requirements for examining the ecosystem impacts of climatic change, changes in biodiversity and depletion of stratospheric ozone.

Long-term data from Integrated Monitoring

ICP IM data are currently being submitted from about 50 sites in 17 European countries, the Russian Federation, and Canada. Additional data reported earlier are available from sites which have either been suspended or taken out of the ICP IM network, and are used for regional monitoring. Taken together, the ICP IM sites provide a source of information allowing for comparisons of complex and multiple effects of environmental changes in ecosystems and across biogeographic regions.

Figure 1: UNECE ICP Integrated Monitoring sites in Europe, the Russian Federation, and Canada by the year 2006.

3 THE AUSTRIAN IM SITE

3.1 Study area description

The Austrian UNECE ICP Integrated Monitoring site (www.umweltbundesamt.at/im) has a size of 90 ha and is situated in the northern part of the national park "Northern Limestone Alps" (N 47°50'30", E 14°26'30") (Figure 2). The altitude ranges from 550 m to 956 m a.s.l. The main type of rock is Norian dolomite (Hauptdolomit) which is partly overlayed by limestone (Plattenkalk). The catchment area is divided into a very steep (30–70°) slope from 550–850 m a.s.l. and an almost flat plateau (850–956 m a.s.l.) on the top of the mountain.

Soil types

On the slope, Lithic and Rendzic Leptosol (= rendsina) predominates with an average mineral soil depth of 12 cm. The plateau is characterised by relictic loams, particularly Dystric Planosol (= pseudogley) with an average soil depth of 32 cm. In some locations at the plateau Gleyic Planosol can be found. At intermediate positions Chromic Cambisol (= brown earth) occurs with an average soil depth of 22 cm. Mull and moder humus predominates but mor humus can also be found.

Climate

The long-term average annual temperature is 6.7° C. The coldest monthly temperature at 900 m a.s.l. is –0.9° C (January), the highest is 15.5° C (August). Annual rainfall ranges from 1500 to 1800 mm. Monthly precipitation ranges from 100 mm (October) to 230 mm (July). Snowfall occurs between October and May with an average duration of snow cover of about 4 months, although this is very variable.

Major forest types and forest management

The slope is mainly covered by mixed mountain forest with beech (*Fagus sylvatica*) as the dominant species, Norway spruce (*Picea abies*), maple (*Acer pseudoplatanus*) and ash (*Fraxinus excelsior*) (Adenostylo glabrae-Fagetum sensu WILLNER 2002), whereas *Picea abies* predominates on the plateau following plantation after a clear cut around the year 1910. The potential natural vegetation of the plateau is a mixed Fagus-Abies-Picea-forest (Cardamino trifoliae-Fagetum sensu WILLNER 2002). From the start of the project in 1992 onwards forest management has been restricted to single tree harvesting in case of bark beetle infestation. Wind throw is frequent in the area with single tree events and events affecting larger areas.

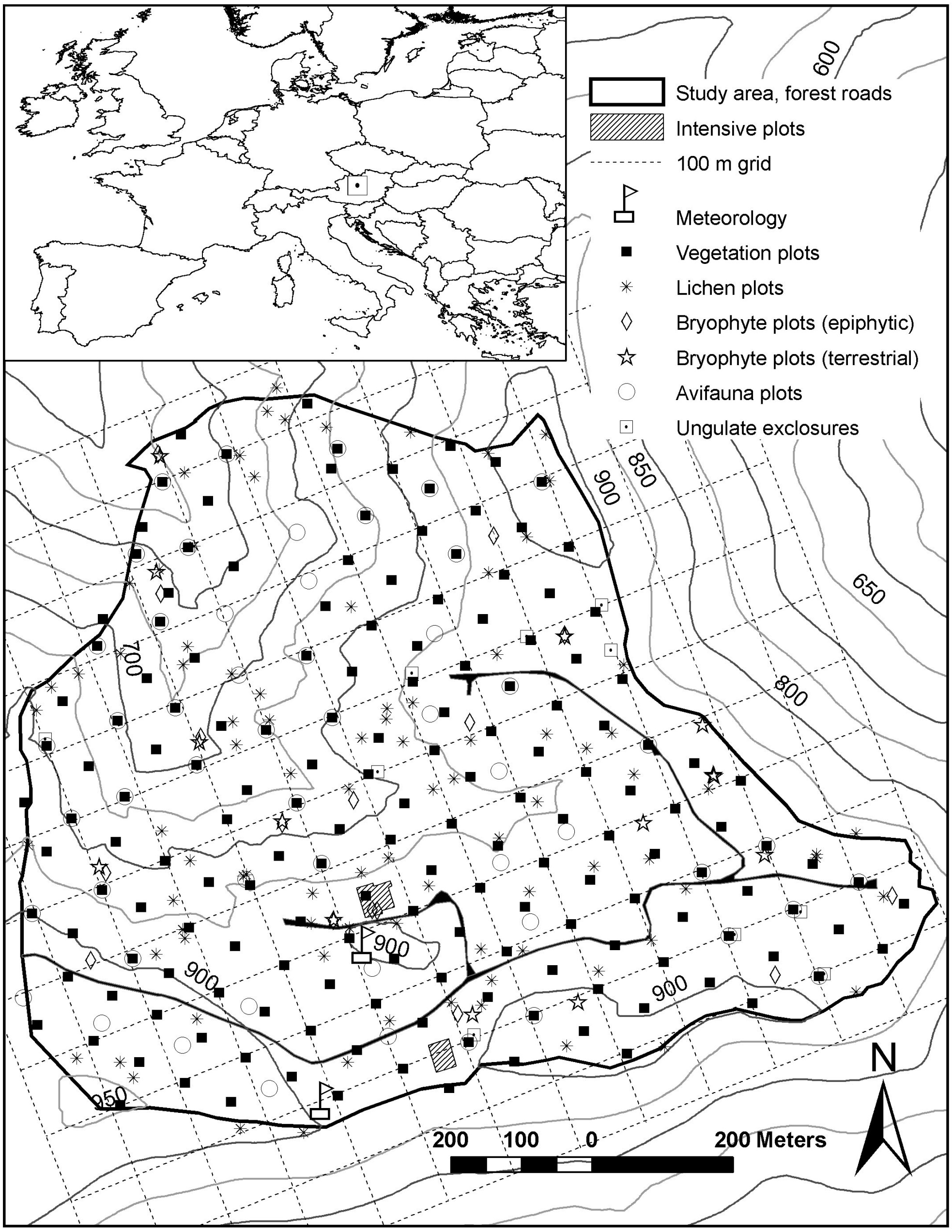

Figure 2: The Austrian UNECE ICP Integrated Monitoring site Zöbelboden. The location of the main meteorological measurements, the two intensive plots, and the distribution of all permanent plots for the monitoring of bioindicators are shown.

3.2 Spatio-temporal data

Meteorology, air chemistry, airborne deposition, bioindicator net

Meteorological and air chemistry measurement is located at a clearing area on the plateau (895 m a.s.l.). Additional meteorological data is acquired at a 46 m high tower reaching above the canopy for the measurement of mesoclimatic conditions (902 m a.s.l.). Two intensive plots for forest stand deposition measurement, soil, and soil water chemistry were set up, one being representative of the slope and the other one of the plateau (Figure 2). Permanent plots of the different bioindicators are distributed across the entire study area. In total 64 main permanent plots are located in a 100 m grid (Figure 2). It is designed in such a way that disturbance by sampling is minimal. Soil cores were sampled, a forest stand inventory was carried out, and forest floor vegetation was recorded according to Figure 3. Fenced and control plots for measuring large herbivore impact (by roe deer, red deer, chamois) on tree establishment are set up at 10 main plots. Additional plots for forest floor vegetation are located in the centre of each 100 m grid. Terrestrial and epiphytic bryophytes, epiphytic lichens, and birds are monitored through plots distributed irregularly according to specific needs like e.g. host trees for lichen monitoring (chapter 8).

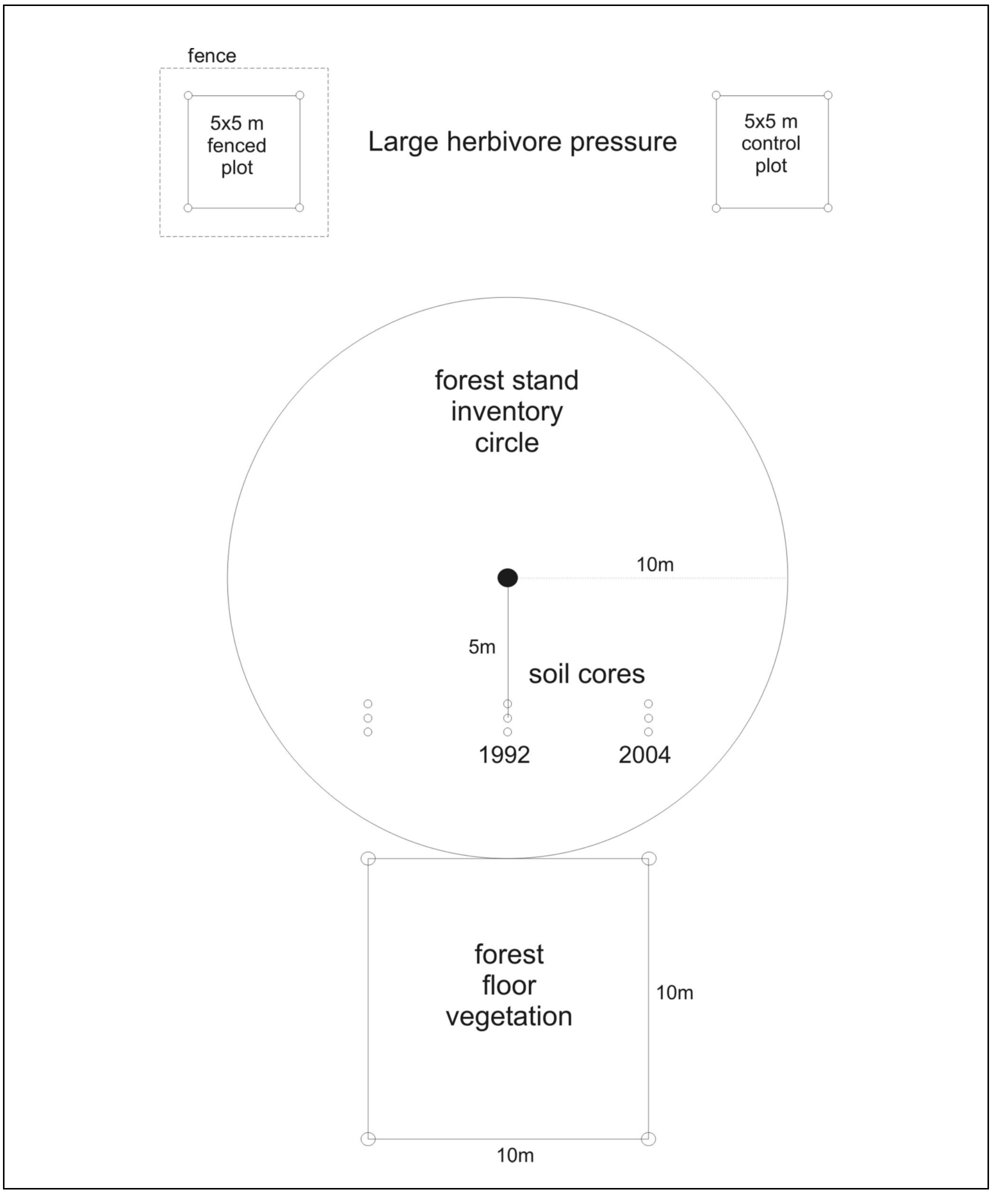

Figure 3: Design of the main plots for monitoring soils, forest stands, and forest floor vegetation. Small dark and open circles depict points that are permanently marked (metal poles) and geodetically surveyed (20 cm coordinate accuracy). Fenced and control plots for measuring herbivore impact are set up only on 10 main plots.

Long-term bioindicator observations

The first records of permanent plots were acquired in the years 1992/1993. A full repetition took place in the years 2004/2005. Herbivore impact and the bird inventory started in the 1995 and 1997 respectively. The observation period ranges between 8 and 13 years for the soil chemical characteristics and the different bioindicators. Herbivore impact, bryophyte, and lichen plots were recorded once again in-between (Table 1).

Table 1: Overview of long-term data of the Austrian ICP IM site which was used for the present study. Years of survey are given as dark shaded boxes.

year of survey	**1991–1999**									**2000–2005**					
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**0**	**1**	**2**	**3**	**4**	**5**
forest inventory		■													■
large herbivore pressure					■			■						■	
soil		■												■	
forest floor vegetation			■	■											■
terrestrial and epiphytic bryophytes		■	■					■						■	■
epiphytic lichens			■						■						■
birds							■								■

4 NITROGEN AND SULPHUR DEPOSITION

4.1 Measurements and models

Measurements of airborne Nitrogen and Sulphur deposition

Direct measurement of airborne deposition is carried out according to the methodological standards of the ICP IM programme. We used data on bulk deposition from a clearance area (895 m a.s.l.), throughfall deposition (= below canopy) in two intensively surveyed forest stands and on modelled total deposition (including dry, fog and cloud deposition) in order to describe the situation with reference to airborne deposition of N and S in the study area. Non-forest bulk deposition in the clearance area and throughfall bulk deposition in the two intensive plots was measured using funnel-type bulk precipitation samplers. The two intensively surveyed forest stands are located on the plateau (intensive plot I, 895 m a.s.l.) and the slope (intensive plot II, 880 m a.s.l., Figure 2). They represent two contrasting forest stands which are typical of the study area: the mixed beech-maple-spruce forests are characteristic of the slope and the spruce-dominated forests are characteristic of the plateau.

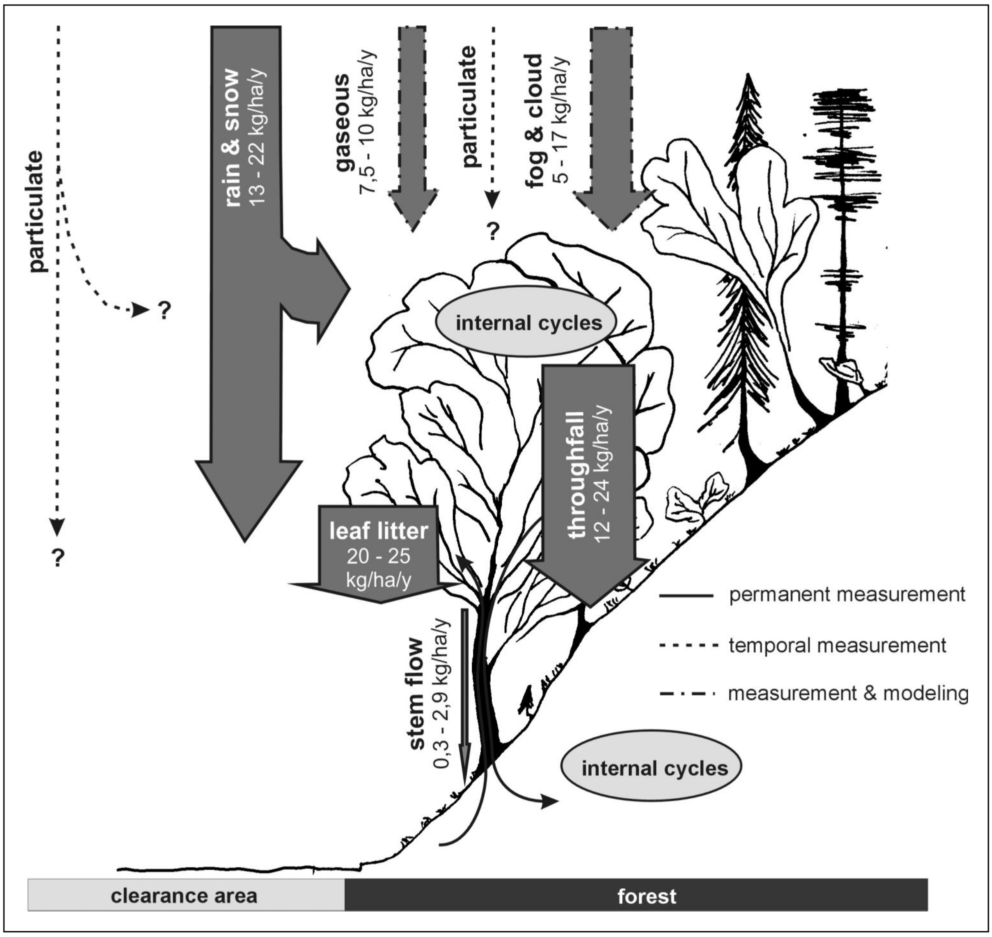

Figure 4: Paths of deposition of airborne pollution in the clearance area and forests of the ICP IM site Zöbelboden. As an example, the annual range of N deposition (in kg/ha/y) in different forest types is given. The estimates are derived from in situ measurements from the year 1995 to 2004 and modelling.

Modelling total deposition

The total deposition of S and N was estimated for the time period 1999-2002 by using measured wet only deposition of the clearance area, modelled occult deposition (CDM-Cloud Deposition Model after (LOVETT 1984, 1994)) and dry deposition (DDM-Dry Deposition Model after (BALDOCCHI et al. 1987) and (MEYERS et al. 1991)) with an active cloud water sampling technique (string collector NESA1) (KALINA & ZAMBO 2003). The estimation of total deposition was performed for 21 forest stands with forest inventory data on tree species, stem diameter, tree height, and canopy height. For an extrapolation of the results of the 21 forest stands to the entire study area, a forest biotope map from the year 2000 with data on tree species abundance for each polygon was used. The polygons were overlaid with the 21 forest stands and classified into four gross forest structural types for which the estimated total deposition had been established (Figure 8). These were henceforth referred to as "deposition clusters" (see also ZECHMEISTER et al. 2006).

4.2 Spatial distribution and temporal trends

Nitrogen deposition

Deposition measurements of nitrogen (N) in the clearance area *(895 m a.s.l.)* show annual average values of 16.5 ± 2.7 kg N-total*ha^{-1}*yr^{-1} (7.5 ± 2.6 kg NH_4-N*ha^{-1}*yr^{-1} and 6.9 ± 1.0 kg NO_3-N*ha^{-1}*yr^{-1}) in the period from 1995 to 2004 (Figure 5). Throughfall N deposition for intensive plot I was 20.9 ± 2.6 kg N*ha^{-1}*yr^{-1} and for intensive plot II 14.8 ± 2.4 kg N*ha^{-1}*yr^{-1} (Figure 6). Throughfall deposition of N-total slightly increased from 1995 to 2004. When modelled occult and dry deposition is taken into account, the different forest stands of the study area are exposed to N deposition ranging between 28 and 43 kg N-total*ha^{-1}*yr^{-1} (KALINA & ZAMBO 2003, see also HERMAN & SMIDT 1995). The substantial share of occult and dry deposition (45 to 59% of total N) is characteristic of mountain forests with frequent fog and cloud events (LOVETT 1984, 1994, WESELY & HICKS 2002). The empirical critical loads for deteriorating effects of excess N are between 10 to 20 kg N*ha^{-1}*yr^{-1} for the observed forest types (BOBBINK et al. 2003). Depending on whether the measured or the modelled deposition is considered, N deposition in the study area is within the range of critical loads or critical loads are exceeded by up to an order of magnitude.

Sulphur deposition

Sulphur (S) deposition shows annual average values of 6.2 ± 1.5 kg SO_4-S*ha^{-1}*yr^{-1} in the clearance area in the period from 1995 to 2004 (Figure 5). Average SO_4-S deposition was 7.1 ± 2.0 kg*ha^{-1}*yr^{-1} for intensive-plot I and 4.0 ± 1.5 kg*ha^{-1}*yr^{-1} for intensive-plot II (Figure 7). Taking occult and dry deposition into account the forests of the study area are exposed to SO_4-S deposition ranging between 10 to 18 kg*ha^{-1}*yr^{-1}. The share of occult and dry deposition is 40 to 67% of the total S deposition (KALINA & ZAMBO 2003). S deposition decreased like in many other parts of Europe so that acidification is becoming less of a problem and recovery from acidification can be expected (BOUWMAN et al. 2002) (Figure 5, Figure 7).

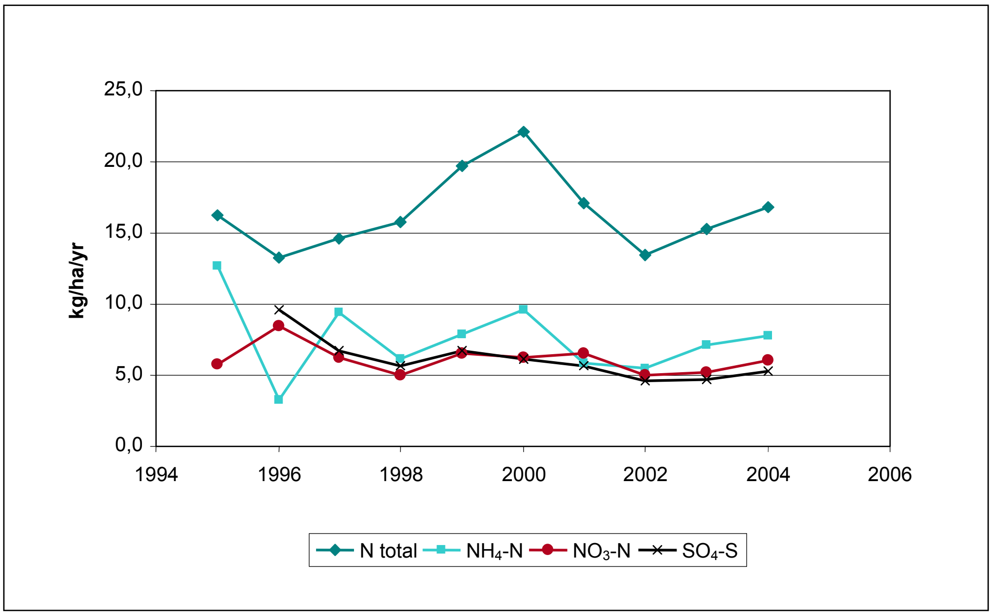

Figure 5: Non-forest bulk deposition of N-total, NH_4-N, NO_3-N, and SO_4-S measured in the clearance area (895 m a.s.l.) between 1995 and 2004.

Amounts of deposition in different forest stands

There is a significant difference between intensive-plot I and II with respect to the deposition amount of N-total and SO_4-S ($P < 0.001$ and $P = 0.005$; two-sided paired t-test). This difference can be explained by the stand structure and tree species composition of the two plots (KALINA & ZAMBO 2003). Intensive plot I, which is characteristic of spruce-dominated forests of the plateau, is exposed to significantly higher deposition than the mixed deciduous forests represented by Intensive plot II. Conifers like spruce are well known receptors of high amounts of particularly fog, cloud and dry deposition (LOVETT 1984, 1994). The plateau is thus exposed to higher amounts of deposition than the slopes (see Figure 8 with the plateau in the southern and eastern part of the study area). The snow regimes in the slope area (less snow due to wind in the upslope areas) and at the plateau (snow accumulation in flat areas) may additionally influence the differences in deposition.

Total deposition of S and N was estimated for different forest types which receive increasing deposition amounts (KALINA & ZAMBO 2003, Figure 8):

- low deposition (cluster 1): deciduous tree species dominate, a mixture of mostly beech, maple, ash, and spruce to a minor extent
- intermediate deposition (cluster 2): deciduous tree species, a mixture of mostly beech, maple, and ash
- high deposition (cluster 3): deciduous tree species and spruce (equal share of spruce with beech, maple, and ash)
- very high deposition (cluster 4): mono-dominant spruce forests.

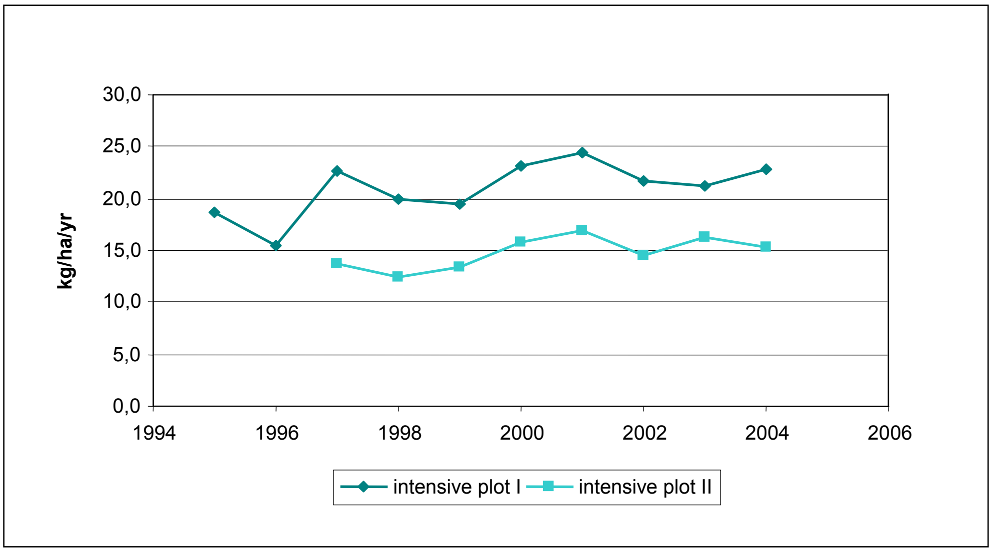

Figure 6: Throughfall bulk deposition of N-total in intensive plots I and II between 1996 and 2004.

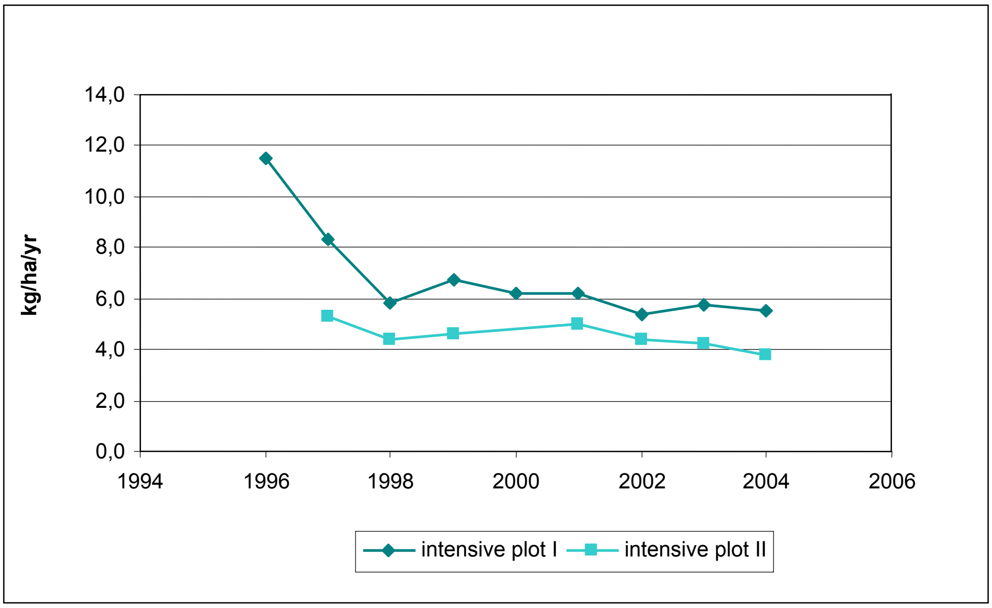

Figure 7: Throughfall bulk deposition of SO_4-S in intensive plots I and II between 1996 and 2004.

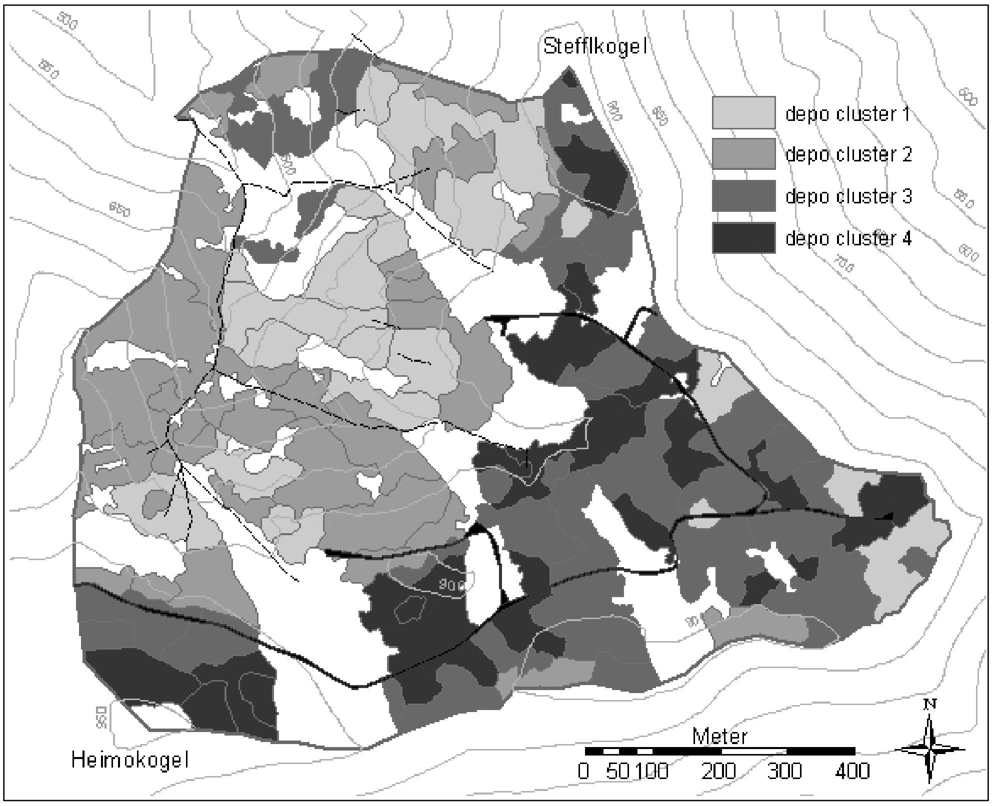

Figure 8: Spatial distribution of the total deposition of airborne N and S at the ICP IM site Zöbelboden as indicated by deposition clusters. Total deposition takes modelled occult and dry deposition into account. Low deposition (depo cluster 1), intermediate deposition (depo cluster 2), high deposition (depo cluster 3), very high deposition (cluster 4). The bold lines depict forest roads and the dashed lines permanent and temporary creeks.

4.3 Confounding environmental and human factors

Below we give an overview of a series of factors potentially confounded with the effects of N and S deposition. Their temporal development is described and they are discussed with reference to impacts on soils and bioindicators.

4.3.1 Climate

The long term average annual temperature is 6.7° C. The coldest monthly temperature (895 m a.s.l.) is –0.9° C (January), the highest is 15.5° C (August). Annual rainfall ranges from 1500 to 1800 mm. Monthly precipitation ranges from 100 mm (October) to 230 mm (July) (Table 2). At altitudes of 900 m a.s.l. snowfall occurs between October and May with an average duration of snow cover of about 4 months.

Table 2: Climatic characteristic of the IM site Zöbelboden derived from long-term data

precipitation	1650 mm/year (long-term average)
temperature	+ 6.7° C (long-term average)
vegetation period	approx. 190 days/year (>5° C)

No climatic trends in the observation period

Neither the mean nor the minimal and maximal daily day and night temperatures show an increasing or decreasing trend between 1992 and 2005. Year to year variation is however considerable (Figure 9). In particular precipitation varied substantially. Drought years like 2003 are beyond the average variation (Figure 10). The annual vegetation period (days/year with > 5° C) ranged between 156 and 246 days between 1992 and 2005 showing substantial year to year variations but no obvious increase or decrease.

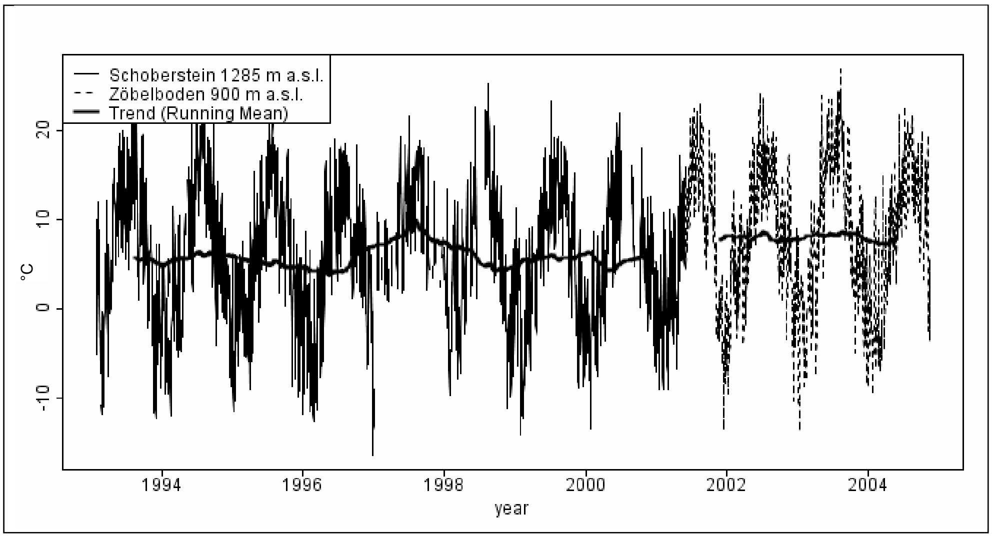

Figure 9: Mean daily temperature of the IM site Zöbelboden (895 m a.s.l.) and from Schoberstein (1285 m a.s.l.), a nearby meteorological station at comparable elevation with data prior to 2001. A trend (running mean with 365 days) is shown by the bold line (dashed lines indicate data from IM site Zöbelboden).

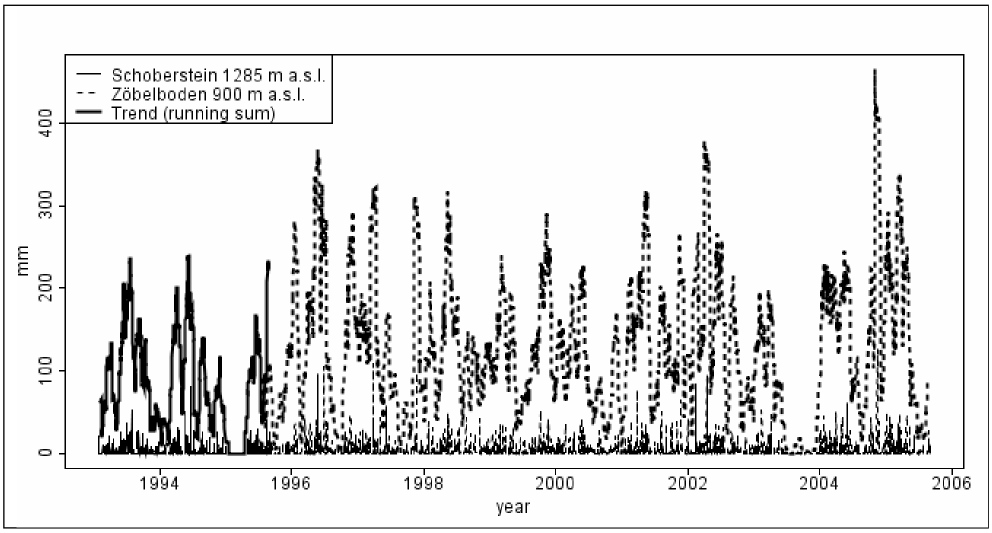

Figure 10: Daily precipitation of the IM site Zöbelboden (895 m a.s.l.) and from Schoberstein (1285 m a.s.l.), a nearby meteorological station at comparable elevation with data prior to 1995. A trend (running sum with 30 days) is shown by the bold line (dashed lines indicate data from IM site Zöbelboden).

In summary, the climatic parameters have not change significantly during the period of observation. A significant effect of climate warming on temporal trends of the bio-indicators observed might thus not have occurred. However, trends from nearby meteorological stations (e.g. Großraming, 379 m a.s.l.) indicate a continuous decade-long temperature increase. The forest ecosystems of the study area might thus respond to this long-term climate trend which is not captured by the meteorological measurements of the study area. In addition, extreme events occurred like the drought year 2003, which might have had an effect on the observation of bioindicators in the years 2004 and 2005. Nevertheless, the findings of (LEXER et al. 2001) indicate that particularly the montane forests of Austria – like those of the study area – are well buffered against contemporary climate changes. Taking all arguments into account, we do not expect a major influence of climate changes on the temporal development of the bioindicators at the IM site „Zöbelboden" during the observation period, with the exception of triggering effects of extreme climatic events.

4.3.2 Large herbivores

Trends in the abundance and impact of ungulate herbivores

Ungulate herbivores can have profound effects on tree recruitment by twig browsing and bark peeling (WEISBERG & BUGMANN 2003). The following ungulates can be found in the area: roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), and chamois (*Rupicapra rupicapra*). Since 2001 roe deer has been increasing, red deer decreasing and the numbers of chamois have varied substantially from year to year with no clear trend. In total, 10 fenced and their control plots were used to analyse changes in forest damage according to (REIMOSER et al. 1999) (see Figure 2 and Figure 3 for sampling details). Twig browsing predominantly occurs in winter and is most intensive on *Sorbus aria* and least on *Picea abies*. The plateau areas are more prone to twig browsing than the slope. Bark peeling decreased in the slope area and increased at the plateau. Overall herbivore damage increased from the mid-1990s until 2004 (REIMOSER & REIMOSER 2005).

Though there was an increasing impact of herbivores on tree establishment, no major indirect effect is to be expected during the relatively short observation period. A direct impact of ungulates on the composition of forest herbs and grasses was not investigated.

4.3.3 Forest and management

Intensive historical tree harvest

In the 19th century the demand for wood for the prosperous iron industry of the region was exceptional. The area of the „Reichraminger Hintergebirge", where the ICP IM site „Zöbelboden" is located, was extensively logged. Natural forests remained only in less accessible areas. The forests at the plateau of „Zöbelboden" were clear cut around the year 1910. Currently a plantation forest with spruce predominates on the plateau. Spruce was also planted on smaller clear cuts in the following decades. By contrast, the rather steep slopes largely remained untouched so that we find close to natural mixed deciduous forests with beech, maple, ash, spruce, larch and fir.

With the establishment of the national park, the „Nationalpark Kalkalpen" in the year 1997, forest management was restricted to single tree harvest in case of bark beetle attacks. The old forest road currently serves as the main access to the area.

The main impact of forest management on the area is thus historical. Young spruce plantations still exist but are excluded from permanent plot sampling. Older plantations, where permanent plots exist, are already mixed with deciduous trees (see chapter 4.3.4 for more details).

Intensive harvest in the past led to nutritional deficiencies in many central European forests and recovery processes might still be ongoing (e.g. HERMAN et al. 2001). In fact, the nutritional status of forest trees, as indicated by the needle chemistry of spruce in Austria, is low, particularly with regard to N (STEFAN & FÜRST 1998).

4.3.4 Forest tree layer changes

4.3.4.1 Forest floor light condition

Measurement of below canopy radiation

Changes of the tree canopy through growth, senescence, windthrow or bark beetle infestation, but also tree harvest directly influence the light conditions in the forest and the diversity of the entire forest stand. In order to directly measure forest floor light conditions, hemispherical photographs were taken at a height of 1 m in the year 2005 at all main 10*10 m permanent plot (Figure 2, Figure 3). Total below canopy radiation (in W/m^2) was calculated for the vegetation period (March to October) for each photograph using the software "Gap Light Analyzer" Version 2.0 (GLA 2.0, FRAZER et al. 1999). Since no hemispherical photographs were available for the year 1992 a multiple least square regression model was calibrated linking below canopy radiation with above canopy radiation (taken from the GLA calculations) and forest inventory parameters from the year 2005 ($R^2 = 0.69$, F-Test p-value < 0.001).

No temporal change of the below canopy light conditions

No significant temporal changes of the forest floor light conditions were found, neither on the plateau nor the slope ($p > 0.272$, two-sided t-test on below canopy radiation measures) (Figure 11). This was surprising as the slope was affected by a net decrease of tree individuals due to wind throw and senescence (Figure 12) and bark beetle attacks were frequent in the last years in the area. Apparently, closure of the canopy by the remaining tree individuals counterbalanced the net loss of tree individuals so that the light conditions, on average, were stable.

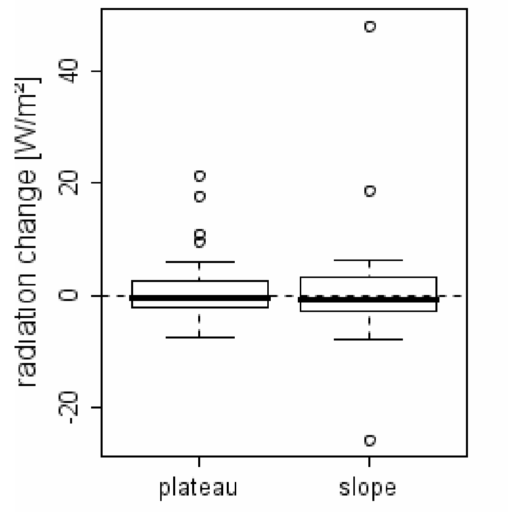

Figure 11: Box-and-whisker plot of the change of below canopy radiation (in W/m²) from 1992 to 2004 at the plateau and the slope (n = 62). Positive values indicate increasing and negative values indicate a decreasing radiation.

4.3.4.2 Tree species effect on soils

Tree layer changes were investigated through forest inventories in the year 1992 and 2005 on 64 10 m diameter circles in a 100 m regular grid (Figure 2, Figure 3). The method includes parameters about forest stands and single trees and followed the Austrian standards for forest inventories (SCHIELER & HAUK 2001).

Tree species composition indirectly affects forest floor vegetation via its influence on humus characteristics. Litter quality and the rates of airborne N and S deposition exert a combined effect on humus decomposition (KNORR et al. 2005). In particular the contribution of spruce is important in the area due to its known acidifying effect retarding humus decomposition and N mineralization (REHFUSS 1990).

Spruce decreased and beech increased

The share of spruce, which was planted on clear cuts, decreased in the entire area during the last 13 years. The number of beech trees increased at the plateau but decreased on the slope (Figure 12). Humus decomposition and thus N mineralization should thus have been ameliorated on the plateau. On the slope, no major changes are to be expected since the relative shares of beech and spruce remained more or less the same.

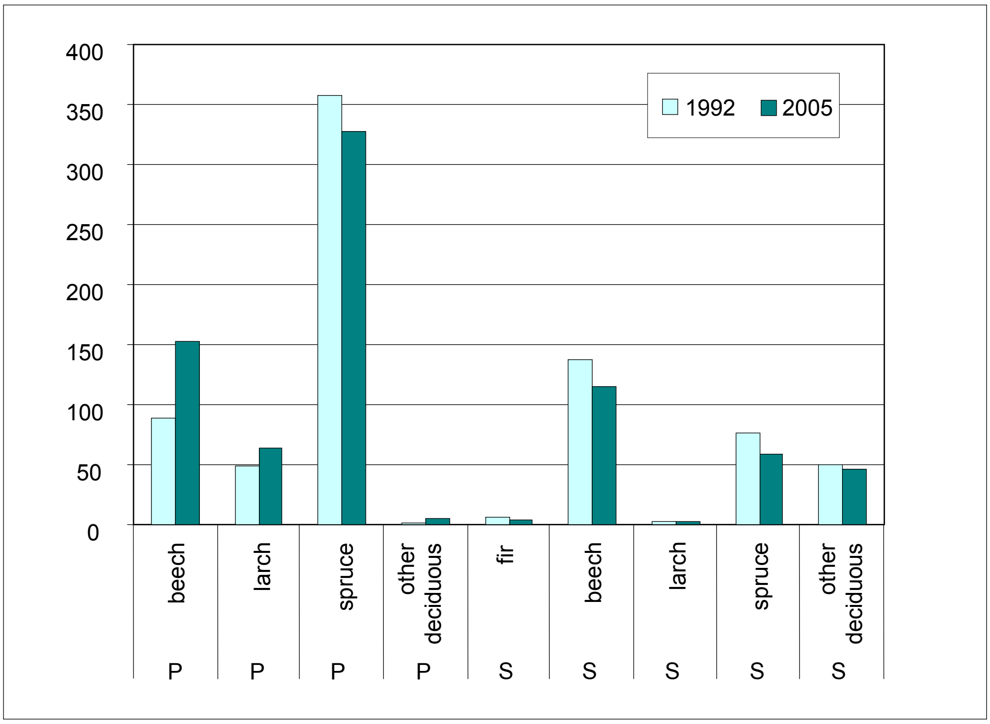

Figure 12: Comparison of the proportions of tree species between 1992 and 2005 at the plateau (P) and the slope (S). Units are total counts in 36 (plateau) and 27 (slope) forest inventory circles with 10 m diameter. Only living trees with a breast height diameter ≥ 10 cm were taken into account.

4.3.4.3 Tree layer diversity

Tree layer diversity indices derived from forest inventory data

Tree layer diversity, which is the number of tree species but also the horizontal and vertical canopy structure, determines much of total forest diversity. Heterogeneous tree layers create more diverse niches for plants and animals through their influence on light conditions, soil formation, the provision of forage, nesting opportunities and so on (NEUMANN & STARLINGER 2001). Trees as habitats for epiphytic organisms are the source of vivid population dynamics, thereby controlling diversity (ZECHMEISTER et al. 2006). We used three different diversity indices in order to reveal changes in the tree layer. The indices were calculated using the forest inventory data of 1992 and 2005. The integrated stand diversity index (JAEHNE & DOHRENBUSCH 1997) combines measures for the variation of species composition, tree stem diameter, tree distance and crown dimension. Forests stand are significantly more diverse in 2005 they were in 1992 (Table 3). The diversity of the vertical distribution of coverage within a forest stand – vertical evenness (NEUMANN & STARLINGER 2001) – is significantly increasing. Horizontal patchiness of trees, using the aggregation index according to (CLARK & EVANS 1954) did not change. The change of tree layer diversity is thus due to vertical structuring.

Table 3: Temporal trends of forest tree layer diversity indices derived from 64 forest inventory plots recorded in the years 1992 and 2005. Indices are given as mean and standard deviation; p-values were derived using a paired t-test.

diversity index	1992	2005	time trend	p-value
vertical evenness (NEUMANN & STARLINGER 2001)	0.49 ± 0.20	0.45 ± 0.21	+ 0.04	**0.034**
aggregation index (CLARK & EVANS 1954)	1.15 ± 0.25	1.16 ± 0.33	– 0.02	0.489
stand diversity index (JAEHNE & DOHRENBUSCH 1997)	2.95 ± 4.81	2.65 ± 1.52	+ 0.30	**0.001**

5 EFFECTS ON SOILS AND BIOINDICATORS

5.1 Soil changes

Soil sampling and chemical soil analyses

In 1992 and 2004 soil samples were collected at the 64 main permanent plot positions (Figure 2). At each location 3 soil samples were taken from 5 horizons and pooled (Figure 3). The humus layer and the mineral soil horizon from 0–10 cm was analysed for this study. At each sampling position the soil type was determined according to (NESTROY et al. 2000) and grouped into 3 gross soil types referred to as Leptosol (Rendsina), Cambisol (calcareous brown earth), and Planosol (Pseudogley) thereafter. The pooled soil samples were analysed for pH value, total organic and inorganic carbon (TOC, TIC), total nitrogen (N_{tot}), and total sulphur (S) content. C/N ratios were built with TOC and N_{tot} and C/S ratios with TOC and S respectively. For further details on sampling and chemical analysis see details in chapter 9.1.1.

5.1.1 Results

Soil chemical characteristics are mainly determined by soil type

The chemical characteristics of the soils in the area are mainly determined by the occurrence of relict loams, which are responsible for the development of Leptosol in the absence of loams and Cambisol or Planosol in the presence of loams. The humus layer of Leptosol shows highest N_{tot}, TOC and pH values, whereas Planosol shows the lowest. In this regard Cambisol takes an intermediate position. Base saturation is very high for all soil types (99%, unpublished data) and stems mostly from Ca and Mg ions. The mean C/N ratio of the humus layer is 34 in the year 1992 and 29 in the year 2004. It is lowest for Leptosol and Cambisol and highest for Planosol (Table 4).

Table 4: Chemical soil characteristics of the three main soil types and changes from 1992 to 2004.

			pH		**C/N**		**C/S**	
			1992	**2004**	**1992**	**2004**	**1992**	**2004**
Humus layer (n = 59)		Leptosol	5.3 ± 0.5	6.0 ± 0.4	32.8 ± 5.5	28.7 ± 6.1	221.9 ± 167.5	290.5 ± 84.3
		Cambisol	5.1 ± 0.5	5.1 ± 0.7	32.7 ± 4.7	29.7 ± 4.3	388.4 ± 281.4	308.4 ± 56.5
		Planosol	4.8 ± 0.6	4.5 ± 0.9	37.0 ± 6.8	30.3 ± 5.1	237.5 ± 89.7	310.8 ± 60.3
	Change	overall[1)]	**+ 0.2, p = 0.015**		**– 4.3, p < 0.001**		**+ 16.6, p = 0.009**	
		soil types[2)]	**p < 0.001**		p = 0.285		p = 0.055	
Mineral soil 0-10 cm depth (n = 56)		Leptosol	6.6 ± 0.3	6.9 ± 0.2	13.9 ± 2.4	15.3 ± 1.6	57.2 ± 14	125.7 ± 25.1
		Cambisol	6.2 ± 0.8	6.4 ± 0.7	15.6 ± 3.9	16.1 ± 1.7	57.3 ± 18.3	135.3 ± 25
		Planosol	5.3 ± 1.2	4.9 ± 1.0	16.5 ± 4.5	16.1 ± 1.9	48.2 ± 15	150.8 ± 15.5
	Change	overall[1)]	+ 0.1, p = 0.092		+ 2.2, p = 0.156		**+ 79.4, p < 0.001**	
		soil type[2)]	**p = 0.008**		p = 0.349		**p = 0.004**	

[1)] *Overall temporal changes were tested using a two-sided paired t-test.*

[2)] *Differences of changes between soil types were tested with ANOVA using the F-statistic. Values represent mean ± std.dev.*

Significant changes with p-value < 0.05 are given in bold letters.

Soil pH value increased

Between 1992 and 2004 the pH value increased overall by 0.2 in the humus layer and by 0.1 in the mineral soil from 0–10 cm. These changes were significantly different between soil types with highest increases for Leptosol (+ 0.7 for the humus layer) and a slight decrease for Planosol (Table 1). C/S ratio increased significantly in the humus layer (on average + 16.6) and in the organic soil (on average + 79.4). These changes were significantly different between soil types. The degree of change of the C/S ratio increases from Leptosol to Cambisol to Planosol.

Soil C/N ratio narrowed

The C/N ratio narrowed significantly (on average – 4.2) in the humus layer but remained stable in the mineral soil with no difference between soil types (Table 4, Figure 13, and Figure 14).

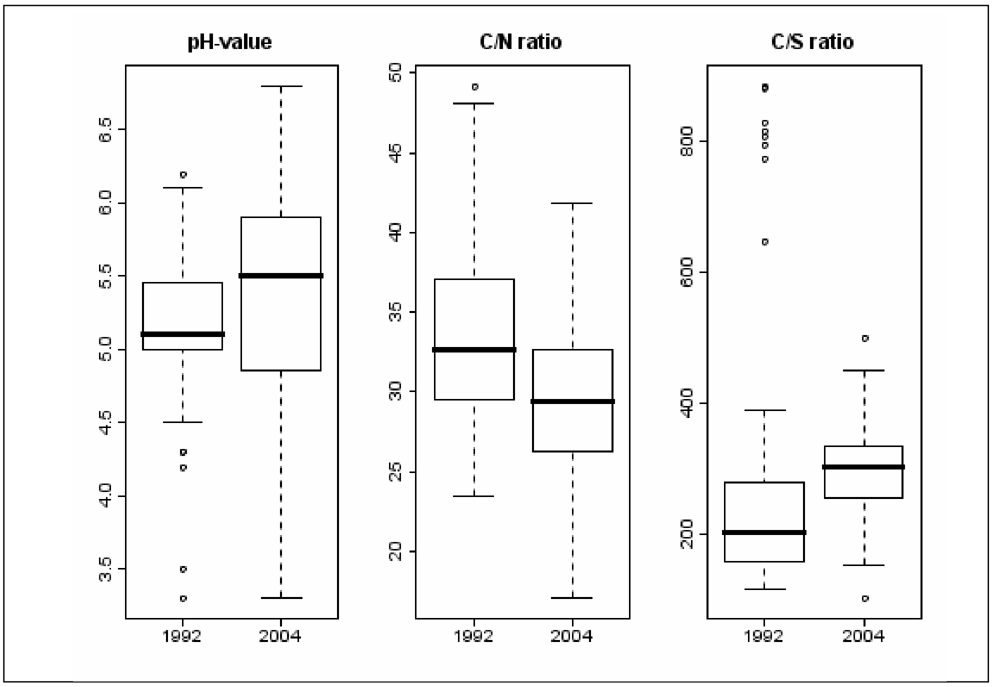

Figure 13: Box-and-whisker plots of changes in pH-value, C/N, and C/S ratio in the humus layer from the years 1992 and 2004 (n = 59).

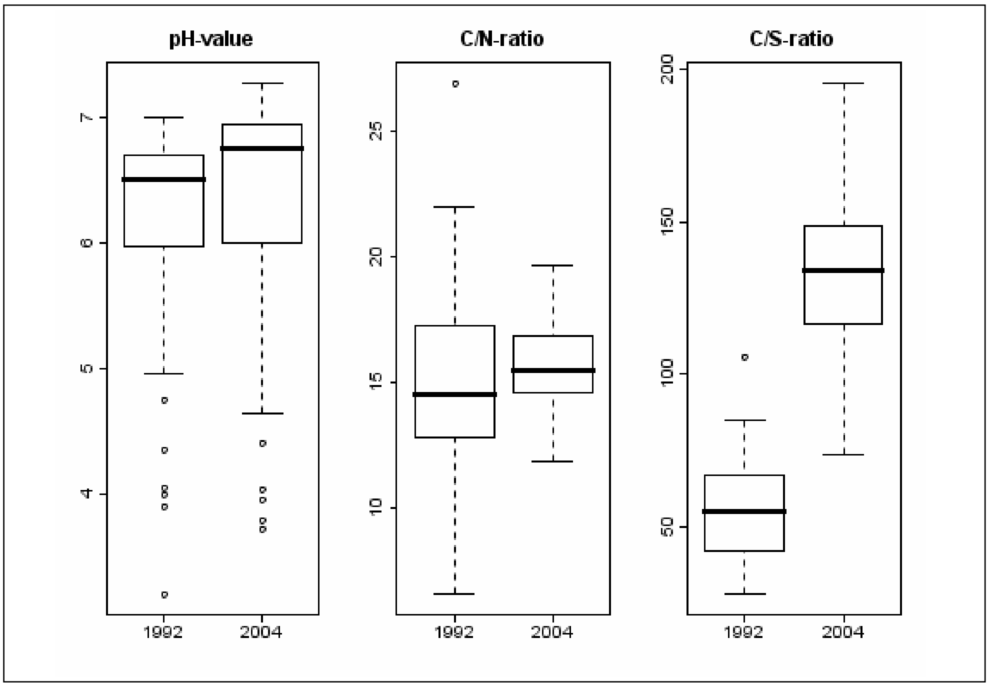

Figure 14: Box-and-whisker plots of changes in pH-value, C/N, and C/S ratio in the mineral soil from 0 to 10 cm from the years 1992 and 2004 (n = 59).

Less Sulphur deposition caused recovery from soil acidification

Taking the overall trend, soil pH values increased in the study area, which is in line with broad scale trend data (FORSIUS et al. 2001). This can be related to the decrease in S deposition which led to significantly increasing C/S ratios in the humus layer as well as the mineral soil (Figure 13, Figure 14). The magnitude and the direction of this trend differ between the soil types (Table 4). Leptosol and Cambisol exhibited a clear increase with Leptosol showing highest pH-increase. This is in line with the pronounced decrease of S deposition (Figure 7). Planosol on the other hand showed a decreasing pH value although the C/S ratios increased. In this case excess deposition of nitrous acids might be responsible for continuing acidification. Planosol occurs predominantly on the plateau where relatively high deposition occurs due to a high proportion of conifers (chapter 4). There, deposition of acids has apparently overruled the amelioration of acidification through increased litter quality, which can be expected given the relative decrease of the proportion of spruce (chapter 4.3.4.2).

Excess Nitrogen deposition caused soil eutrophication

The C/N ratio in the humus layer narrowed significantly (Table 4, Figure 13). Narrowing C/N ratios indicate enhanced N mineralisation and are used as robust indicators of excess deposition of airborne N (MACDONALD et al. 2002, FALKENGREN-GRERUP & DIEKMANN 2003, AUGUSTIN et al. 2005). Notably, both the soil C (TOC) and N pool (N_{tot}) decreased. Chronic N deposition might thus have stimulated the decomposer activity, thereby lowering the C and N storage of the soil, but less for the N than for the C pool. Additional factors exist which might have influenced N dynamics. First, canopy opening and enhanced light penetration may additionally stimulate N mineralisation. Though this might be important for single plots, the light conditions were on average stable during the observation period (chapter 4.3.4.1). Second, the nutritional status of the soils may still be undergoing a continuous recovery from the clear cut which occurred on the plateau in the beginning of the 20^{th} century.

5.1.2 Conclusion

Overall soil chemical changes

The soils show a significant recovery from acidification which is due to a decrease of S deposition to the forest soils. The recovery process is limited to Leptosol and Cambisol, both of which are predominant on the slopes of the study area. Planosol, the dominant soil type at the plateau, exhibited further acidification. The long-term excess N deposition caused soil eutrophication. C/N ratios narrowed significantly in all soil types.

5.2 Forest floor vegetation

Analysis of long-term trends of the forest floor vegetation

Rectangular 10*10 m permanent plots are positioned in a regular 100 m grid (with plots at each corner and in the centre) (Figure 2, Figure 3). For the present study we used only plots with at least 10% coverage of trees higher than 1.3 m in 1993 (in total 124). All vascular plant species present on each plot were recorded and a cover-abundance value estimated. The first recording took place in 1993, a repetition in the year 2005. In order to detect trends in species abundance and species composition we used Marginal Models for square contingency tables and Multidimensional Scaling. A potential causal relationship with air pollution of N and S was sought by using Ellenberg indicator scores (see 9.1.2 for more details).

5.2.1 Results

Nitrophilic forest floor plants increased

Overall, the abundance of forest floor plant species slightly increased. Whereas 13% of the plant species significantly decreased in abundance, 15% increased. The correlation between these abundance changes with species-specific Ellenberg indicator scores (ELLENBERG et al. 1992) showed a significant positive correlation with the N value (nutrient availability), whereas no correlation was found for L (light), M (moisture) and R (soil reaction) value (Table 5). This result is in line with many other studies stating that airborne N deposition favours species which are characterised by an enhanced N demand (BOBBINK et al. 2003).

Table 5: Correlation between abundance changes of plant species and their Ellenberg indicator scores. The β of Marginal Models, which indicates the degree and direction of abundance change, of 109 plant species was used. We used Kendall's rank correlation with exact p-values for ordinal data (significant values with p-value < 0.05 are given in bold). Positive correlation coefficients indicate positive correlation with Ellenberg scores and vice versa.

Ellenberg indicator	Kendall's Tau	p-value
N	**+ 0.14**	**0.026**
R	– 0.01	0.845
L	– 0.04	0.512
M	– 0.004	0.953

Trends in forest floor species composition differ between soil types

The changes in species composition from 1993 to 2005 are illustrated in Figure 16. Since species composition is largely determined by soil characteristics, plots group according to their soil type. On the right side base-rich, dry sites (Leposol) and on the left side increasingly moist, acidic sites prevail (Planosol) (Figure 16). There is an obvious and also significant change in species composition between 1993 and 2005. This change, moreover, is different for different soil types (overall change and group change has $p < 0.001$ using MANOVA). An analysis of Ellenberg indicator values for species composition does not reveal a clear relationship with N and S deposition and the observed soil changes (see 5.1). However, the convergence of the plots from the margins to the centre indicates an erosion at the infertile ends of the floristic gradient: at the base-rich, dry and at the acid, moist sites. Overall, forest floor vegetation is thus homogenising which is indicated by the shrinkage of compositional variation between 1993 and 2005 (Figure 15) (HÜLBER et al. submitted). Such a differential effect seems to be typical of forest floor vegetation (DIEKMANN & DUPRÉ 1997). Recently (GILLIAM 2006) hypothesised that the resulting homogenisation, which is caused by a decrease in the spatial diversity of N availability, is a major driver of biodiversity losses in forests.

Erosion of the infertile forest floor communities

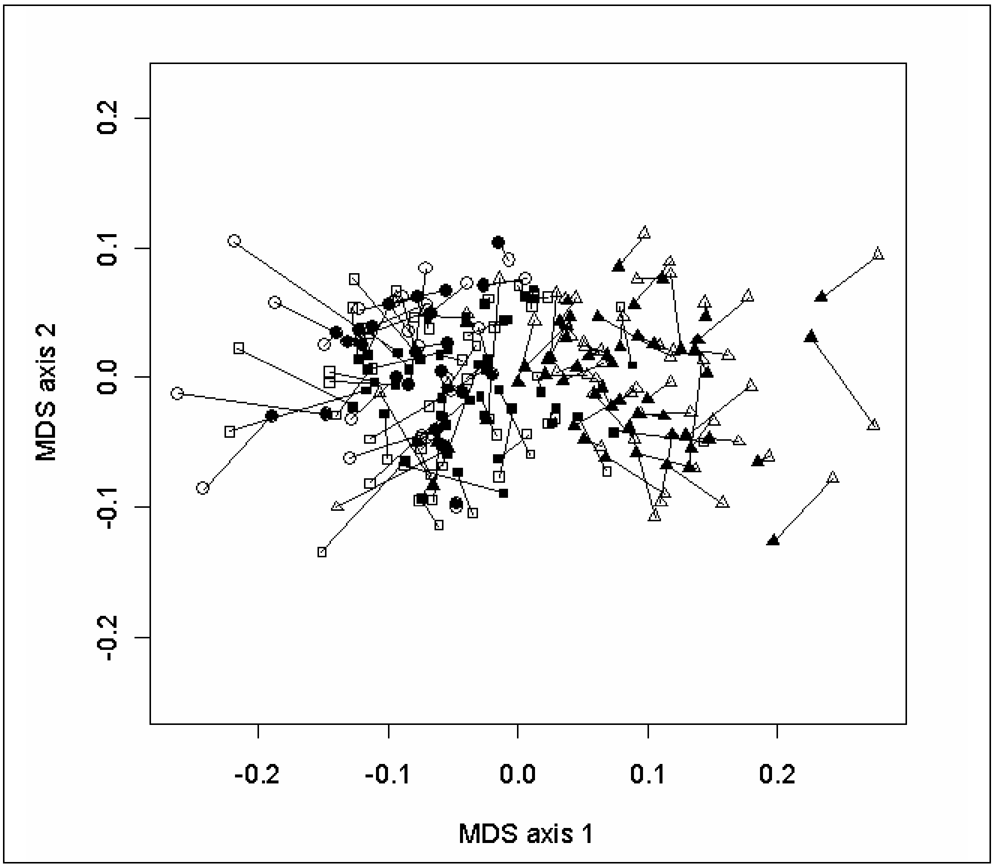

Figure 15: Temporal changes in species composition on 124 permanent plots of the Austrian Integrated Monitoring site from 1993 to 2005. The spacing of the plots indicates species compositional similarity. Filled symbols represent plot position in 2005, open symbols the position in 1993. Triangles represent Leptosol, circles Planosol and squares Cambisol. The graph shows the first and the second axis of the three-dimensional ordination space derived from non-metric Multidimensional Scaling (nMDS-space).

5.2.2 Conclusion

Overall changes of the forest floor vegetation

Chronic N and S depositions have differential effects on forest floor vegetation. Weak responses under intermediate site conditions may mask significant changes on oligotroph sites where species composition shifts towards the species composition of intermediate sites. As a likely long-term consequence, oligotrophic species will probably disappear from the regional species pool due to the erosion of oligotrophic sites. N deposition is thus a threat to biodiversity even in areas where N supply is favourable at the majority of sites. Compared to the trends found in other forests, the overall impact of airborne N and S deposition is weak so far. This is due to the predominance of soils characterised by rather favourable nutrient conditions and thus low sensitivity to eutrophication and by a high buffer capacity against acidification.

5.3 Bryophytes

Analysis of long-term trends of terrestrial and epiphytic bryophytes

In total, 20 epiphytic and 14 terrestrial monitoring plots were recorded in the study area (Figure 2). The epiphytic plots were established on different tree species according to the ICP IM Manual sub-programme EP 'trunk epiphytes' (ICP IM Manual, 2004). This method was originally developed for lichens, and adapted to be used for bryophytes in several countries. The first recording took place in 1993, with repetitions in the years 1998 and 2004/2005. The terrestrial plots with a size of 50*50 cm were established in the year 1992. The total cover of each bryophyte species was recorded. The repetitions took place in the years 1993, 1998 and 2004/2005. In order to detect trends in the abundance of single species we used Wilcoxon Matched Pairs test. Temporal trends in species composition of the permanent plots were analysed using Multidimensional Scaling (see chapter 9.1.2.2 for more details).

5.3.1 Results

In total, at least 55 species were found on all the permanent plots during one observation year. 33 of these were found on epiphytic and 33 on terrestrial plots. 11 species occurred on both types of plots. With the exception of a few dominants, coverage was low for most of the species and the turnover rate high over the observed period. When looking at all the permanent plots, only six terrestrial (18.2%) and eight epiphytic species (24.2%) covered more than 5% of the plot area. *Hypnum cupressiforme* was the only species covering more than 5% on both types of permanent plots.

The epiphytic plots exhibited an overall increase in bryophyte cover over the observation period, associated with a significant reduction of empty bark (Wilcoxon Matched Pairs test, $p = 0.035$) and a constant but not significant reduction of lichen coverage. For the terrestrial plots, the area covered by single bryophyte species differed extremely between plots during one observation, and also between the repeated observations of a plot.

Few bryophyte species increased or decreased directionally

Temporal changes of the observed species were inhomogeneous between the two periods (1992 to 1998 and 1998 to 2004/2005) and only very few species showed a constant directional trend over the entire observation period. On the epiphytic plots, significant increases of coverage were found for *Hypnum cupressiforme* (Wilcoxon Matched Pairs test, $p = 0.009$) for the entire observation period from 1992 to 2004.

Leucodon sciuroides did not increase from the year 1992 to 1998 but increased significantly from 1998 to 2004 (Wilcoxon Matched Pairs test, p = 0.027). On the terrestrial plots, only a single species (*Dicranodontium denudatum*) showed a significant decrease (Wilcoxon Matched Pairs test, p = 0.043) during the entire observation period (ZECHMEISTER et al. 2006).

No directional trend of the bryophyte species composition

In contrast to significant temporal trends in the abundance of single bryophyte species, no such trends were found for species composition of the permanent plots. In fact, there was no significant difference in species composition found between the years of observation, neither for the epiphytic nor the terrestrial plots. Some plots showed clear trends in a single direction, on others there were trends in the opposite direction. Several plots showed no discernible trends within the periods of investigation, which are illustrated by a zigzag line (Figure 16). Directional trends of some plots were thus counterbalanced by an overall pronounced variance.

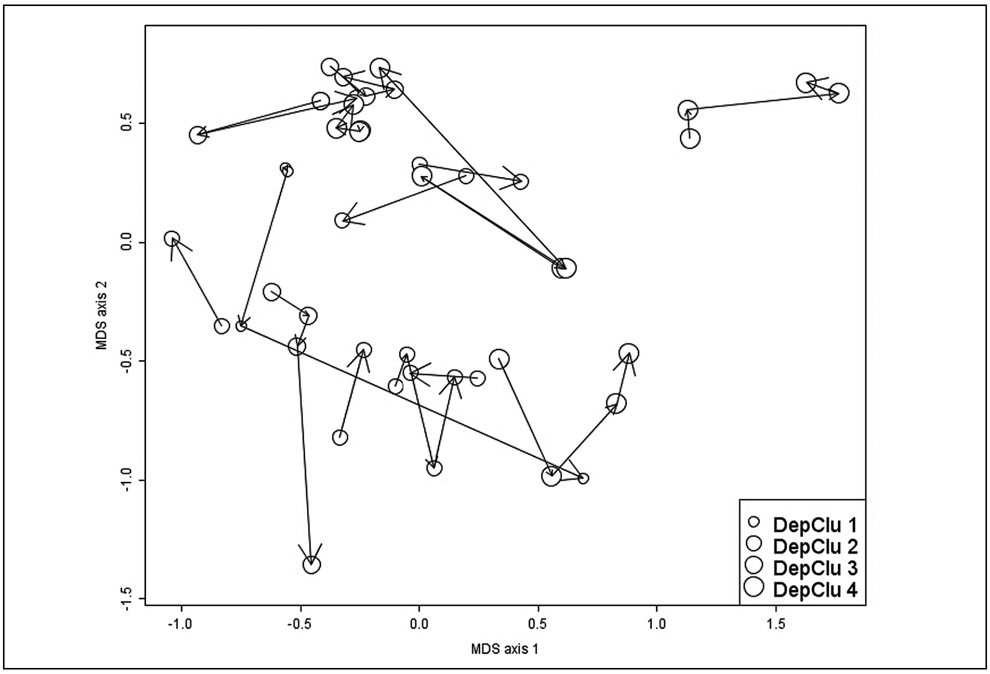

Figure 16: Temporal changes in species composition of 14 terrestrial bryophyte plots (Nonmetric Multidimensional Scaling (nMDS), axes 1 and 2). Distances between plots represent similarity of the species composition. The size of the circles represents the amount of N and S deposition (deposition clusters 1–4) on each terrestrial plot. The direction and size of the arrows represent the temporal development of each plot from the first (1993) to the last (2005) observation.

Bryophyte species composition correlates with the amount of airborne Nitrogen and Sulphur deposition

Notwithstanding the ambiguity of the temporal trends of bryophyte species composition at the permanent plots, some comparable patterns of airborne pollution and changes in overall species composition could be found. First, the peak in N deposition in the year 2000 (Figure 5) could be related to a relatively big change in species composition, which is reflected by elongated arrows in Figure 16. Eutrophication through N seems to be responsible for this change (ZECHMEISTER et al. 2006). Secondly, the species composition correlates with the deposition cluster representing different amounts of total N and S deposition (Table 6).

*Table 6: Correlation between the bryophyte species composition (multidimensional nMDS space with 3 axes) of the epiphytic and terrestrial plots and the deposition amount of N and S ("deposition cluster"). Values represent Kendall´s tau correlation coefficient between nMDS axis scores and the deposition cluster. Significant correlations are given in bold letters (*p-values < 0.05, ** < 0.01).*

nMDS axis	deposition cluster	
	epiphytic plots	**terrestrial plots**
1	0.012	**0.283***
2	**– 0.328****	**0.256***
3	0.002	0.218

5.3.2 Conclusion

Overall changes of terrestrial and epiphytic bryophytes

During the 14 year period of observation most bryophyte species remained stable in their overall abundance in the area. The observed changes can only to some extent be attributed to effects of airborne N and S pollution. A few bryophytes responded to N deposition levels in the observed period by an increment in their population coverage. Accordingly, the changes in only a few other species point to a recovery from acidification owing to a decrease in S deposition. The bryophyte communities as a whole did however not show directional changes attributable to the observed amounts of N deposition and the decrease of S deposition. Thus, the substantial exceedance of critical loads for eutrophication effects did not lead to acute injuries. If at all, such injuries tended to be chronic injuries on single bryophyte species and recovery processes from former acidification seem to be slow. Overall, this resulted in a spatial distribution of bryophyte communities which was at least to some extent related to different deposition regimes of airborne N and S.

5.4 Lichens

Analysis of long-term trends of epiphytic lichens

Epiphytic lichens were first recorded in the year 1993. Records are set up on maple (*Acer pseudoplatanus*), beech, (*Fagus sylvatica*), ash (*Fraxinus excelsior*), larch (*Larix deciduas*) und spruce (*Picea abies*). Two repetitions took place in the years 1999 and 2005. Tree selection was done so that an even distribution of tree species was achieved. From a total of 70 plots which were recorded in the year 1993, a subset of 49 remained in 2005. All plots were recorded according to four methods, three of which were used here. In order to detect trends in species abundance and species composition Marginal Models for square contingency tables and Multidimensional Scaling were used (see details in chapter 9.1.4).

5.4.1 Results

Lichen abundance decreased in response to airborne Nitrogen and Sulphur deposition

Epiphytic lichen cover per permanent plot decreased slightly from 1993 to 1999 (p-value = 0.094, Wilcoxon rank sum test), and significantly between 1999 and 2005 (p-value = 0.039, Wilcoxon rank sum test). This negative trend occurred particularly on conifers and in the lower and the upper parts of the study area. This decrease of abundance did not lead a simultaneous decrease of the epiphytic lichen diversity. On the contrary, diversity even increased (Table 8). A number of species, which show a significant decrease of abundance using Marginal Models, or even total extinction, are sensitive to eutrophication (e.g. *Hypogymnia physodes*) and acidification (e.g. *Bryoria sp*., *Usnea sp*.).

Lichen communities are continuously deteriorating

The temporal trends in the composition of the species show some differences between the host tree species. Though significant directional trends were found for the plots on conifers such trends were only weak on deciduous trees (Figure 17, Table 7). The epiphytic lichen communities were already impacted by airborne pollution during the first recording in 1993 and exhibited a continuing long-term deterioration without any recovery following the decrease in S deposition. This is in contrast to other areas in Europe where such a recovery was observed (e.g. GILBERT 1992). To date only rudimental communities with regard to species diversity and abundance can be found in the study area.

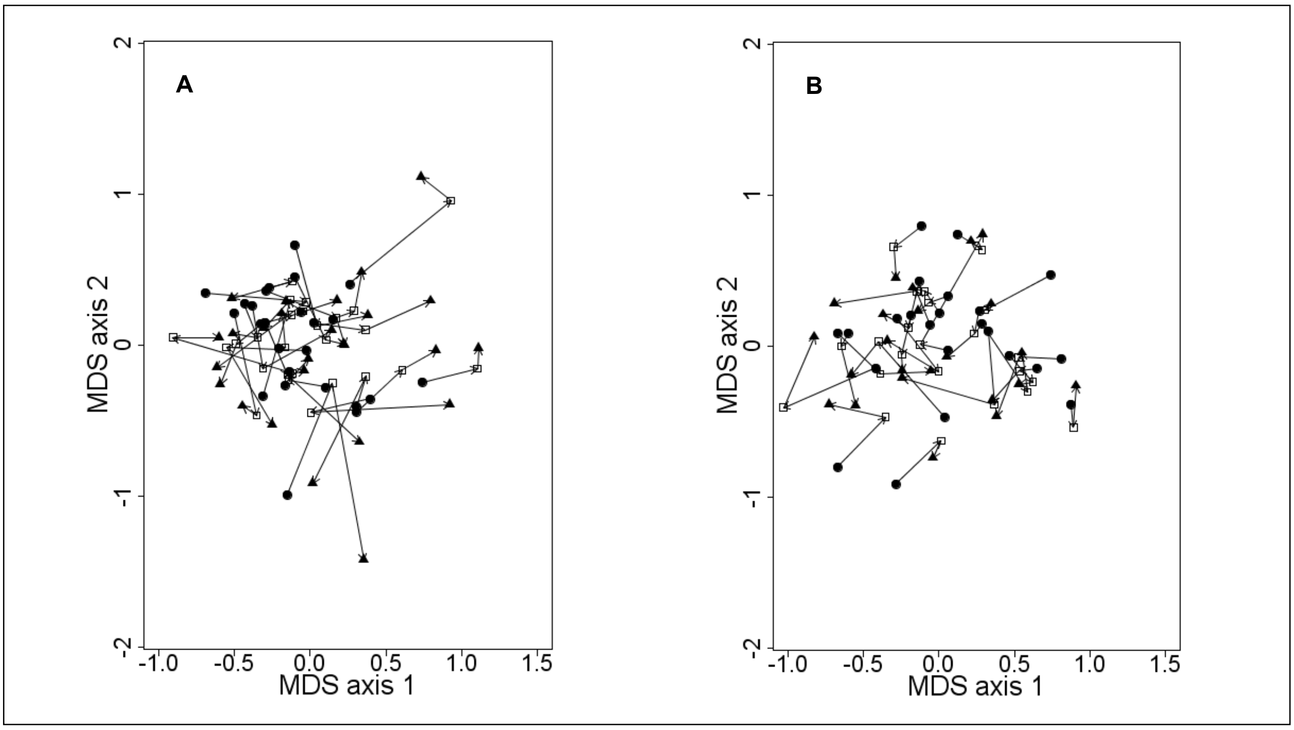

Figure 17: Temporal changes in lichen species composition on 49 permanent plots of the Austrian Integrated Monitoring site from 1993 to 2005 (recording method according to WITTMANN et al. 1989). A) plots on conifers; B) plots on deciduous trees. The spacing of the plots indicates species compositional similarity. Arrows start at the position of each plot in the year 1993 and end in 2005. The graph is spread between the first and the second axis of the three-dimensional ordination space derived from a non-metric Multidimensional Scaling (nMDS-space).

Table 7: Temporal trends of epiphytic lichen species composition on conifers and deciduous trees using the axis scores of 3-dimensional non-metric Multidimensional scaling with MANOVA. Calculations were done for three different recording methods (see chapter 9.1.4.1). Significant trends with p-value < 0.05 are given in bold.

method	observation period	p-value	
		conifers	deciduous
WITTMANN et al. (1989)	1993-1999	0.192	0.380
WITTMANN et al. (1989)	1993-2005	**0.031**	0.249
WITTMANN et al. (1989)	1999-2005	0.298	0.361
VDI-Richtlinie 3799 (1995)	1999-2005	0.153	0.637
TÜRK & ZIEGELBERGER (1982)	1999-2005	**0.033**	0.085

Low air pollution impact classes decreased and higher impact increased

The overall effect of airborne N and S deposition is reflected in the air pollution impact classes, which takes into account not only changes in species abundance but also injuries to individuals. From the beginning of the observation in the year 1993 low impact classes decreased whereas high impact classes increased (Figure 18). The trend between 1993 and 1999 was more pronounced than between 1999 and 2005.

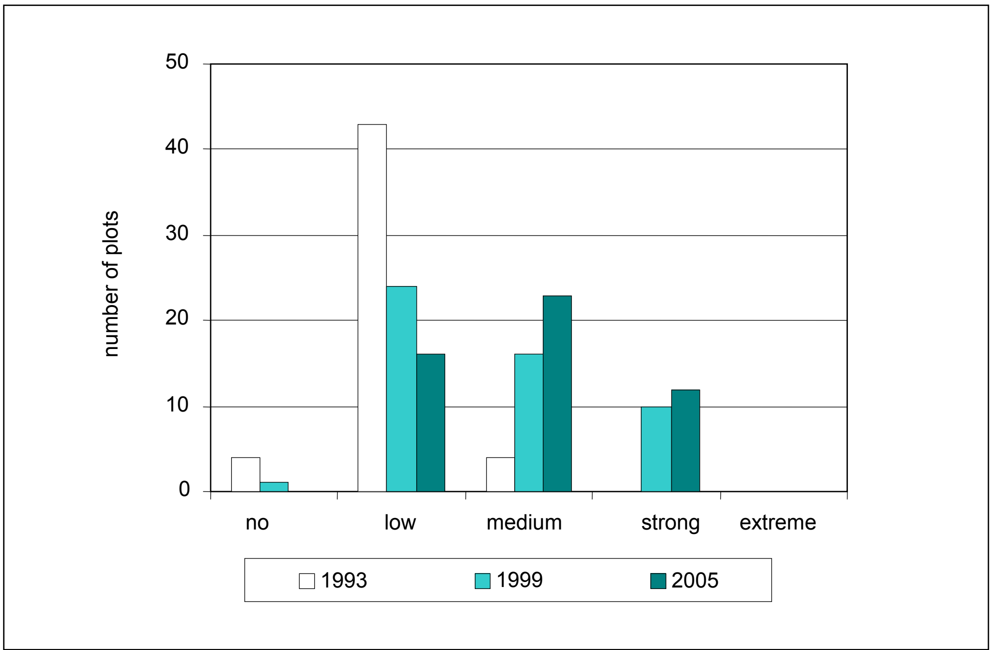

Figure 18: Distribution and temporal trend of air pollution impact according to the classification of (TÜRK & ZIELGELBERGER 1982) in the study area. Records according to (WITTMAN 1989) were used and only those plots which were recorded in all observation years.

The combined impact of airborne N and S deposition on lichens shows a diverse distribution within the study area. Most impacts were found in the lower and the upper parts. Frequent fogs in periods of temperature inversion and enhanced deposition of pollutants in the upper, exposed parts are probably responsible for this pattern.

5.4.2 Conclusion

Overall changes of epiphytic lichens

Long-term airborne N and S deposition had a significant impact on epiphytic lichens. The overall abundance of epiphytic lichens and that of sensitive species decreased. Some sensitive species even became extinct. Due to the colonisation on the permanent plots by other species this did not lead to an overall decrease of lichen diversity. Lichen communities show a deteriorating development in response to air pollution and are becoming increasingly dissimilar from communities typical of clean air conditions. The impact of airborne pollution increased continuously from 1993 to 2005. Currently all plots are affected. The plots indicating low impact decreased while those indicating medium and strong impact increased. Though acid deposition through S decreased, epiphytic lichens did not recover. Lichens show clear signs of eutrophication due to excess N deposition.

5.5 Birds

Analysis of bird data

A total of 64 circles of 100 m in diameter and distributed over the study area were recorded three times a year during the breeding period (Figure 2). At each visit, birds were recorded during a time span of 5 minutes mainly via acoustic detection (LANDMANN et al. 1990). The first record was in the year 1997 and a repetition took place in 2005. Analyses were done separately for 7 gross forest types (see chapter 9.1.5 for more details).

5.5.1 Results

The changes in the frequency of bird species were different between forest types. However, the abundance and composition of birds did not change substantially from 1997 to 2005. Observed changes were attributable to natural variations of bird populations and to structural changes of forest stands. Closure of forest canopies by tree growth, changing shares of dead wood, and gap formation affected the diversity and abundance of specific species groups like hole-breeder, trunk-climber, and carnivores. With respect to airborne N and S deposition, only hypothetical cause-effect relationships can be established (see chapter 6). No trends related to broad scale climate changes were detected (HOCHRATHNER 2006).

5.5.2 Conclusion

The short-term bird data does not allow conclusions on temporal trends

In general, birds are robust indicators for environmental changes. A particular advantage is that a substantial amount of long-term data exists for many countries in Europe and beyond (DVORAK & WICHMANN 2003). The relatively short-term data available for the study area does not allow for clear conclusions on effect related trends. Hypotheses on the likely future fate of bird diversity, which is tightly linked to other ecosystem properties, can be found in chapter 6. In future, with longer-term data available, broad scale and regional scale trends can be compared which will allow for a causal interpretation of trends.

6 EFFECTS ON BIODIVERSITY

Analysis of the distribution and trends of biodiversity

The temporal trends of diversity of each taxonomic group are assessed with regard to the total number of species in the study area (PSN) and at the plot-scale (SN), and the Shannon diversity of plots (SH). The latter index takes the relative abundance of species into account. Apart from evaluating the temporal trend of each taxonomic group, an analysis is performed to detect the factors controlling diversity. Variables which account for site (habitat) condition, site (habitat) heterogeneity, and tree layer diversity are taken into account. The potential effect of air pollution is analysed through the inclusion of the pH-value and the C/N ratio of the soils for forest floor vegetation and the air quality class for lichens (see chapter 5.1 and 5.4 for more details).

6.1 Results

Overall decline of the diversity of forest biota

Averaged over all organisms, diversity has declined from the beginning of the 1990s until 2004/2005. PSN decreased for all but epiphytic lichens. SH, which indicates the heterogeneity of species frequency, also shows predominantly decreasing trends. Epiphytic lichens in general are an exception to this trend. The lichen species number increased in the entire study area and at the plot scale. This is a trend which was also found in other areas (HAUCK & RUNGE 2002). Only a few diversity trends are statistically significant. This is due to the rather high inter-plot variability compared to predominantly weak intra-plot temporal changes (Table 8).

Table 8: Time trends of species number for the entire study area (PSN, sum of species of all plots), species number per plot (SN, mean and standard deviation), and Shannon diversity per plots (SH, mean and standard deviation) for forest floor vegetation, epiphytic bryophytes, terrestrial bryophytes, epiphytic lichens, and birds. The arrow indicates increasing or decreasing trends from the first until the last observation year.

Taxonomic group	Biodiversity index	1992/1993	1997/1999	2004/2005	trend	p-value[5)]
forest floor vegetation (n = 124)	PSN[1), 2)]	195	–	181	↓	–
	SN	31 ± 10.0	–	30.4 ± 8.0	↓	0.100
	SH	2.30 ± 9.93	–	2.39 ± 8.07	↑	0.014
epiphytic bryophytes (n = 11)	PSN[1)]	16	17	15	↓	–
	SN	4.4 ± 2.0	4.1 ± 2.5	3.8 ± 2.4	↓	0.373
	SH	1.09 ± 0.57	0.98 ± 0.62	0.87 ± 0.57	↓	0.302
terrestrial bryophytes (n = 9)	PSN[1)]	53	66	45	↓	–
	SN	2.9 ± 2.0	4.1 ± 3.1	2.6 ± 2.4	↓	0.373
	SH	0.43 ± 0.55	0.52 ± 0.65	0.43 ± 0.51	→	–
epiphytic lichens (n = 80)	PSN[1), 3)]	(40)	65	69	↑	–
	SN[4)]	(4.4 ± 2)	6.0 ± 3.1	6.7 ± 3.4	↑	0.014
	SH[4)]	(0.58 ± 0.43)	0.88 ± 0.51	0.86 ± 0.54	↓	0.484
birds (n = 55)	PSN[1)]	–	50	48	↓	–
	SN	–	14.4 ± 4.9	13.1 ± 3.4	↓	0.015
	SH	–	2.48 ± 4.91	2.23 ± 3.4	↓	< 0.001

[1)] *only plots which were recorded in all observation years were used;*

[2)] *only plots with tree cover > 10% and only herb layers with a height < 60 cm were taken into account;*

[3)] *represents pooled species numbers derived from different (up to 4) methods (see chapter 9.1.4);*

[4)] *in the years 1992/1993 only one method was applied so that these values were not used for further analyses;*

[5)] *significance of the temporal changes were tested using one-sided paired Wilcoxon rank sum test for SN and one-sided paired t-test for SH (only the difference between first and last observation year was tested)*

Habitat condition and heterogeneity controls forest biodiversity

Habitat condition, habitat heterogeneity, and air quality explains between 30.9% and 64.6% of the variation of SN for lichens, forest floor vegetation and birds. SH is explained by 30.8% up to 79.7%. Each taxon is controlled by both mean habitat condition, like altitude or radiation, and habitat heterogeneity, like the variation in soil depths of the permanent plots. Significant effects on forest biodiversity are exerted through the forest tree layer and described by tree layer diversity indices (Figure 19). In general, diverse tree layers hold more subordinate species and more heterogeneously distributed species frequencies than less diverse tree layers. Interestingly, diversity decreased over time although tree layer diversity increased (Table 3).

The diversity at one trophic level is linked to the diversity at other trophic levels. An example is epiphytic lichen diversity, which is not only controlled by tree species identity but also by tree layer diversity (Figure 19). Another example is the interaction of vegetation with birds. The diversity of birds increases with increasing tree layer and forest undergrowth diversity (Figure 19).

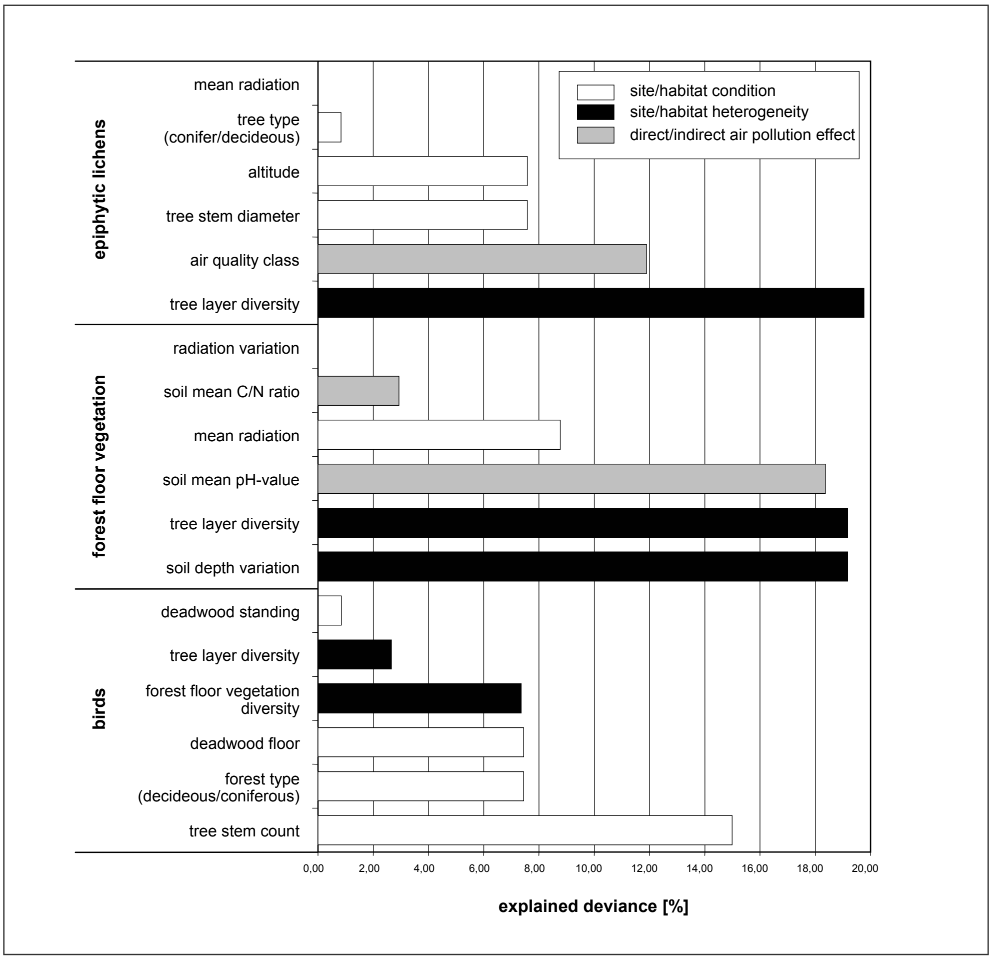

Figure 19: Relative importance of factors controlling the diversity (species number) of epiphytic lichens, forest floor vegetation, and birds. Importance is indicated by explained deviance deduced from generalised additive models of each organism group with the species number as response and the given factors (see chapter 9.1.6.1) as predictors. The percentage of explained deviance is calculated by dropping the respective variable from the full GAM. The shape of bars represents groups of variables indicating site/habitat condition effect, site/habitat heterogeneity effect, and air pollution effects through N and S deposition.

Airborne pollution through Nitrogen and Sulphur is codetermining the diversity of forest organism

In addition to habitat condition and diversity, air pollution with S and N directly or indirectly controls species diversity. Air quality is the second most important determinant for epiphytic lichen diversity. However, the temporal development of lichen diversity indicates that diversity can increase in the presence of serious air pollution, at least within the time frame of the present study. Forest undergrowth diversity is, among other factors, controlled by chemical soil characteristics (Figure 19). The number of forest floor species increases with increasing soil pH-value and C/N ratios (Figure 20) and these soil characteristics are influenced by air pollution (see chapter 5.1). Soil recovery from acidification due to a decrease in S deposition (leading to increasing pH-value) and eutrophication following excess N deposition (leading to decreasing C/N ratio) exert contrasting effects on forest floor diversity with a potentially increasing effect of the first and a decreasing effect of the latter. Air pollution effects on consumers, like birds, are highly oblique, depending on a range of other ecosystem properties which are in turn influenced by air pollution (e.g. forest undergrowth diversity).

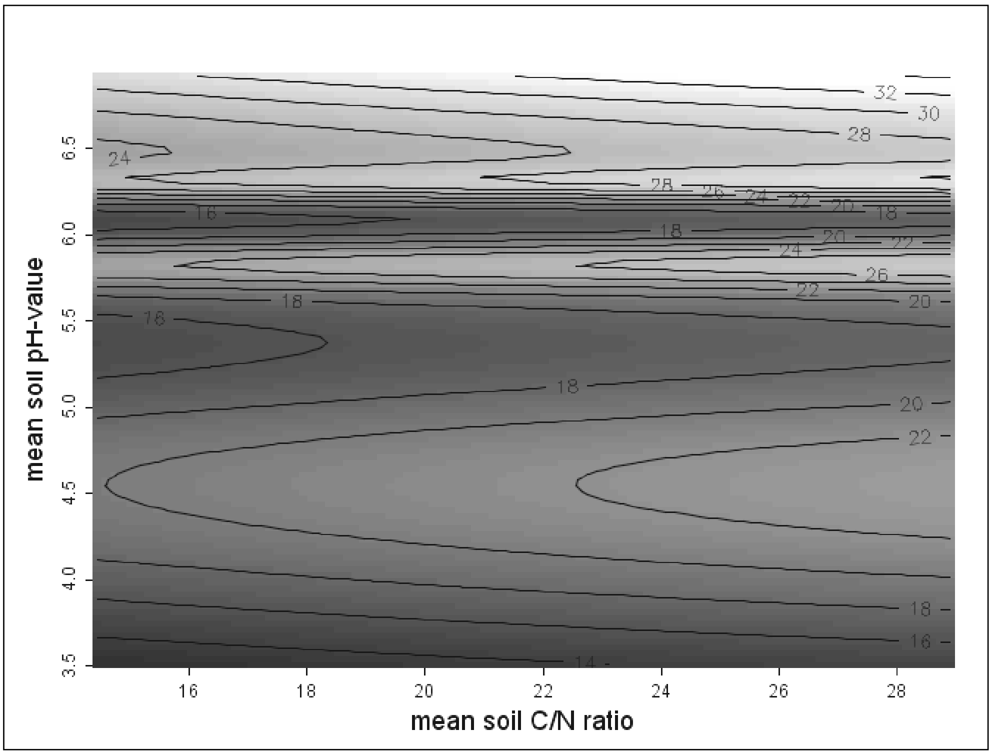

Figure 20: Contours of forest floor species number per plot (SN) in relation to soil C/N ratio and pH-value derived from a generalised additive model. Species number is increasing from the dark to the light shaded areas. All other predictor variables of the model (see Figure 19) are set to the mean value. Soil C/N ratio and pH-value are measured as the mean values of the humus layer and the mineral horizon of the first 10 cm of the soil core.

6.2 Conclusion

Interpreting the trends of forest diversity

In general, biodiversity declined from the beginning of the 1990s until the year 2005. With only one exception – epiphytic lichens – this trend is consistent for the species numbers of the entire study area, and for the mean species numbers at the scale of permanent plots. The decreasing diversity trend is less consistent at the plot scale – at least when an abundance based diversity index (Shannon) is taken into account – than for the entire study area.

Biodiversity is controlled by a range of interlinked factors with air pollution as a significant contributor. Tree layer diversity is one of the main factors controlling the diversity of the entire forest ecosystems. Linkages also exist between different trophic levels, such as tree and forest floor diversity, which favours the coexistence of a large variety of bird species.

Though long-term trends of cross-taxon biodiversity were observed, these trends should be interpreted with caution due to their low significance. Compared with the pronounced variability of diversity between permanent plots, the overall temporal trends are rather weak. It remains thus to be seen if the decline of biodiversity will continue and to which extent air pollution can be made responsible.

7 SYNTHESIS AND FUTURE SCENARIOS

Owing to the extensive canopy of forests, airborne pollutant levels are elevated compared to non-forest vegetation. Moreover, deposition substantially increases in mountain forests, particularly due to high levels of cloud and fog deposition (LOVETT 1984, 1994). Forests are thus particularly vulnerable to emissions of N and S. Detrimental effects on the vitality of forests and other natural and semi-natural ecosystems due to the excess deposition of both substances have been observed (FENN et al. 1998, FLÜCKIGER & BRAUN 1998, ERISMAN & DEVRIES 2000, ABER et al. 2001, VANDOBBEN et al. 1999, ASHENDEN 2002, LEGGE & KRUPA 2002, PAGE et al. 2004). With increasing evidence from experimental results and observations of monitoring data in the last two decades, the negative impact of N and S emissions on biodiversity has become an issue (BOUWMAN et al. 2002, MEA 2005, SUDING et al. 2005, GILLIAM 2006, PHOENIX et al. 2006).

Soils and bioindicators responded to airborne pollution at the ICP IM site Zöbelboden

Both S and N deposition influenced the forest ecosystems of the ICP IM site Zöbelboden in Austria. In accordance with the nitrogen saturation hypothesis, soils show increased N mineralization in response to excess N deposition which is indicated by lower C/N ratios. The abundance of nitrophilic forest floor species bryophytes and lichens increased. Sensitive epiphytic lichen species decreased in abundance. The composition of bryophyte communities is currently co-determined by the amount of airborne N and S deposition in the forest stands. Soils, with some exceptions, recovered from acidification due to a significant decrease of S deposition in the last decade. So far, forest floor vegetation did not respond unambiguously to the increase of soil pH. Some bryophyte species increased in abundance in response to lower acid deposition. Lichens, on the other hand, are still highly impacted by acid deposition.

The observed decline of biodiversity is to some extent due to excess Nitrogen deposition

Though long-term trends of cross-taxon biodiversity were observed, these trends should be interpreted with caution due to their low significance and because of the existence of organism groups whose diversity increased. In addition, it has to be taken into account that the biodiversity of forests is largely determined by below-ground organisms. Usually, soil biota exceeds plant diversity by some orders of magnitude and controls a series of important ecosystem processes (BARDGETS 2005, ZECHMEISTER-BOLTENSTERN et al. 2005). Nevertheless, airborne N and S deposition can clearly be made responsible for the decline of single species of some organism groups.

Forest ecosystems and forest diversity depends on a multitude of simultaneously changing factors

Forests exhibit characteristic spatial and temporal dynamics – among others gap formation and closure – which determine the degree and the patterns of biodiversity and ecosystem processes (LINDENMAYER & FRANKLIN 2002). In order to facilitate an interpretation of the effects of N and S, a range of environmental and human factors which control forest dynamics have to be taken into account. Site conditions have a prominent role with regard to N fluxes (ZECHMEISTER-BOLTENSTERN et al. 2005), thus predetermining the availability of nutrients for plants. Also, the amount of acidification depends strongly on the parent material, among many other factors (REHFUSS 1990). Also climate changes may enter into potential conflict with the effects of airborne N and S deposition and any action of forest management interferes with natural forest dynamics. An illustrative example of interlinked the manifold factors controlling forest ecosystems was given by (DE VRIES et al. 2006). They showed that the carbon sequestered by European forests in last decades was clearly controlled by a combined influence of forest management, climate change and N deposition. A substantial share of forest biodiversity is controlled by trees and tree canopy architecture (NEUMANN & STARLINGER 2001). Any change in the tree layer – be it by management or natural – thus indirectly affects forest biodiversity (EWALD 2002). Herbivore pressure, often controlled by hunting practices, also has an impact on forest dynamics through its effect on tree recruitment and growth (WEISBERG & BUGMANN 2003).

With continuous Nitrogen emission biodiversity loss in the Northern Limestone Alps of Austria is a likely scenario

The present study exemplifies the importance of the simultaneous measurement of all these influences and a wide array of bioindicators. Though many of the above factors have been dynamic and thus co-influenced the behaviour of bioindicators, we have been able to identify the single effect of airborne N and S deposition. It can be assumed from these results and from critical load mapping (UMWELTBUNDESAMT 2005) as well as from deposition measurements at ICP forest level II plots (SMIDT & OBERSTEINER submitted) that continuous eutrophication is taking place in large parts of the Northern Limestone Alps in Austria where airborne pollutant levels are elevated. If N emissions are not abated efficiently, biodiversity loss and effects on ecosystem functions will be a likely scenario in future. Even in the presence of an eventual reduction of N emission, ecosystem changes will still occur due to the time-lag inherent in recovery processes. The slow recovery from acidification shows that time-lags are to be expected. The net decrease of acid deposition through S during the last 30 years caused an increase in the soil's pH value and bryophytes responded slightly. However, forest floor vegetation remains rather inert while epiphytic lichens are continuously affected by acid deposition.

8 REFERENCES

ABER, J.; NEILSON, R.P.; MCNULTY, S.; LENIHAN, J.M.; BACHELET, D. & DRAPEK, R.J. (2001): Forest processes and global environmental change: predicting the effects of individual and multiple stressors. *Bioscience.* No. 51. pp. 735-751.

ADLER, W.; OSWALD, K.; & FISCHER, R. (1994): Exkursionsflora von Österreich. Verlag Eugen Ulmer, Stuttgart and Vienna.

ASHENDEN, T.W. (2002): Effects of wet deposition. In: BELL, J.N.B. & TRESHOW, M., eds. *Air Pollution and Plant Life*. John Wiley & Sons, Ltd., Chichester. pp. 238-249.

AUGUSTIN, S.; BOLTE, A.; HOLZHAUSEN, M. & WOLFF, B. (2005): Exceedance of critical loads of nitrogen and sulphur and its relation to forest conditions. *European Journal of Forest Research.* No. 124. pp. 289-300.

AUGUSTIN, S.; EVERS, J.; DIETRICH, H.-D.; EICHHORN, J.; HAUSSMANN, T.; ICKE, R.; ISENBERG, A.; LUX, W.; MUSIO, M.; PRETZSCH, H.; RIEK, W.; RÖTZER, T.; SCHULTZE, B.; SCHULZE, A.; SCHRÖDER, J.; SEIDLING, W.; WELLBROCK, N.; WILPERT, K. & WOLFF, B. (2005): Concept and feasibility study for the integrated evaluation of environmental monitoring data in forests. *European Journal of Forest Research.* No. 124. pp. 251-260.

BALDOCCHI, D.D.; HICKS, B.B. & CAMARA, P. (1987): A canopy stomatal resistance model for gaseous deposition to vegetated surfaces. *Atmospheric Environment.* No. 21. pp. 91-101.

BARDGETT, R. (2005): The biology of soil: a community and ecosystem approach. Oxford University Press, Oxford.

BEIER, C.; MOLDAN, F. & WRIGHT, R.F. (2003): Terrestrial ecosystem recovery – modelling the effects of reduced acidic inputs and increased inputs of sea-salts induced by global change. *Ambio.* No. 32. pp. 275–282.

BFH – Federal Research Centre for Forestry and Forest Products (2005): The condition of forests in Europe, 2005 Executive Report. United Nations Economic Commission for Europe, Geneva.

BFH – Federal Research Centre for Forestry and Forest Products (2006): The condition of forests in Europe, 2005 Executive Report. United Nations Economic Commission for Europe, Geneva.

BOBBINK, R.; ASHMORE, M.; BRAUN, S.; FLÜCKINGER, W. & VAN DEN WYNGAERT, I.J.J. (2003): Empirical nitrogen critical loads for natural and semi-natural ecosystems: 2002 update. In: ACHERMANN, B. & BOBBINK, R. (eds.) *Empirical critical loads for nitrogen*. Expert Workshop Proceedings, SAEFL, Berne. pp. 43-170.

BOUWMAN, A.F.; VAN VUUREN, D.P.; DERWENT, R.G. & POSCH, M. (2002): A global analysis of acidification and eutrophication of terrestrial ecosystems. *Water Air and Soil Pollution.* No. 141. pp. 349-382.

BRAUN-BLANQUET, J. (1964): Pflanzensoziologie, Grundzüge der Vegetationskunde. 3rd ed.: Springer Verlag, Vienna.

CLARK, F.J. & EVANS, F.C. (1954): Distance to the nearest neighbour as a measure of spatial relationships in populations. *Ecology.* No. 5. pp. 445-453.

IM PROGRAMME CENTRE (1994): Code List M2: M2 BRYOPHYTA – hepatic and musci, Europe and the Azores. Code list for bryophytes. IM Programme Centre. Finnish Environment Institute, Helsinki.

CORLEY, M.F.V.; CRUNDWELL, A.C.; DÜLL, R.; HILL, M.O. & SMITH, A.J.E. (1981): Mosses of Europe and the Azores: an annotated list of species, with synonyms from the recent literature. *Journal of Bryology.* No. 11. pp. 609-689.

DE VRIES, W.; REINDS, G. J.; GUNDERSEN, P. & STERBA, H. (2006): The impact of nitrogen deposition on carbon sequestration in European forests and forest soils. *Global Change Biology.* No. 12. pp. 1151-1173.

DIEKMANN, M. & DUPRÉ, C. (1997): Acidification and eutrophication of deciduous forests in northwestern Germany demonstrated by indicator species analysis. *Journal of Vegetation Science.* No. 8. pp. 855-864.

DVORAK, M. & WICHMANN, G. (2003): Die Vogelwelt Österreichs im dritten Jahrtausend – Monitoring-Programme für Vögel in Österreich. BirdLife Austria, Vienna.

ELLENBERG H.; WEBER H.E.; DÜLL R.; WIRTH V.; WERBER W. & PAULISSEN, D. (1992): Zeigerwerte von Pflanzen in Mitteleuropa. 3rd ed. *Scripta Geobotanica.* No. 18.

EMEP – Co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe (2006): EMEP Assessment Report (20. June 2006). www.emep.int/index_assessment.html.

ERISMAN, J.W. & DEVRIES, W. (2000): Nitrogen deposition and effects on European forests. *Environmental Review.* No. 8. pp. 65-93.

ERISMAN, J.W.; GRENNFELT, P. & SUTTON, M. (2002): The European perspective on nitrogen emission and deposition. *Environment International.* No. 9. pp. 311-325.

EWALD, J. (2002): Multiple controls of understorey plant richness in mountain forests of the Bavarian Alps. *Phytocoenologia.* No. 32. pp. 85-100.

FALKENGREN-GRERUP, U. & DIEKMANN, M. (2003): Use of a gradient of N-deposition to calculate effect-related soil and vegetation measures in deciduous forests. *Forest Ecology and Management.* No. 80. pp. 113-124.

FENN, M.E.; POTH, M.A.; ABER, J.D.; BARON, J.S.; BORMANN, B.T.; JOHNSON, D.W.; LEMLY, A.D.; MCNULTY, S.G.; RYAN, D.F. & STOTTLEMYER, R. (1998): Nitrogen excess in North American ecosystems: predisposing factors, ecosystem response, and management strategies. *Ecological Applications.* No. 8. pp. 706-733.

FERRETTI, M.; PETRICCIONE, B.; FABBIO, G. & BUSSOTTI, F. (eds) (2006): Aspects of biodiversity in selected forest ecosystems in Italy: status and changes over the period 1996-2003. Third report of the Task Force on Integrated and Combined (I&C) evaluation of the CONECOFOR programme 2006. *Annali Istituto Sperimentale per la Selvicoltura*. Special Issue 30/2. Arezzo.

FLÜCKIGER, W. & BRAUN, S. (1998): Nitrogen deposition in Swiss forests and its possible relevance for leaf nutrient status, parasite attacks and soil acidification. *Environmental Pollution.* No. 102. pp. 69-76.

FORSIUS, M.; KLEEMOLA, S.; VUORENMAA, J. & SYRI, S. (2001): Fluxes and trends of nitrogen and sulphur compounds at Integrated Monitoring sites in Europe. *Water Air and Soil Pollution.* No. 130. pp. 1641-1648.

GALLOWAY, J.N.; DENTENER, F.J.; CAPONE, D.O.; BOYER, E.W.; HOWARTH, R.W.; SEITZINGER, S.P.; ASNER, G.P.; CLEVELAND, C.C.; GREEN, P.A.; HOLLAND, E.A.; KARL, D.M.; MICHAELS, A.F.; PORTER, J.H.; TOWNSEND, A.R. & VÖRÖSMARTY, C.J. (2004): Nitrogen cycles: past, present, and future. *Biogeochemistry.* No. 70. pp. 153-226.

GILBERT, O.L. (1992): Lichen reinvasion with declining air pollution. In: BATES, J. & FARMER, A. *Bryophytes and lichens in a changing environment.* Clarendon Press, Oxford.

GILLIAM, F.S. (2006): Response of the herbaceous layer of forest ecosystems to excess nitrogen deposition. *Journal of Ecology.* No. 94. pp. 1176-1191.

GLUTZ V. & BLOTZHEIM U. (1962): Die Brutvögel der Schweiz. Verlag Aarauer Tagblatt, Aarau.

GROLLE, R. (1983): Hepatics of Europe including the Azores: an annotated list of species, with synonyms from the recent literature. *Journal of Bryology.* No. 12. pp. 403-459.

HACKL, E.; BACHMANN, G. & ZECHMEISTER-BOLTENSTERN, S. (2004): Microbial nitrogen turnover in soils under different types of natural forest. *Forest Ecology and Management.* No. 188. pp. 101-112.

HAFELLNER, J. & TÜRK, R. (2001): Die lichenisierten Pilze Österreichs – eine Checkliste der bisher nachgewiesenen Arten mit Verbreitungsangaben. *Stapfia.* No. 76. pp. 1-167.

HAUCK, M. & RUNGE, M. (2002): Stemflow chemistry and epiphytic lichen diversity in dieback-effected spruce forest of the Harz Mountains, Germany. *Flora.* No. 197. pp. 205-261.

HERMAN, F. & SMIDT, S. (eds) (1995): Ökosystemare Studien im Kalkalpin. FBVA Berichte No. 87. Forstliche Bundesversuchanstalt, Vienna.

HERMAN, F.; SMIDT, S. & ENGLISCH, M. (eds) (2001): Stickstoffflüsse am Mühleggerköpfl in den Nordtiroler Kalkalpen. FBVA Berichte No. 119. Forstliche Bundesversuchanstalt, Vienna.

HOCHRATHNER, P. (2006): Wiederholungsinventur Ornithologie 2005, Integrated Monitoring Zöbelboden. Umweltbundesamt, Wien. (unpublished report).

HÜLBER, K.; DIRNBÖCK, T.; KLEINBAUER, I.; WILLNER, W.; DULLINGER, S.; KARRER, G. & MIRTL, M. Long-term trends of forest floor vegetation exposed to airborne N and S deposition. (submitted to Basic and Applied Ecology).

JAEHNE, S. & DOHRENBUSCH, A. (1997): Ein Verfahren zur Beurteilung der Bestandesdiversität. *Forstwissenschaftliches Centralblatt.* No. 116. pp. 333-345.

KALINA, M.F. & ZAMBO, E. (2003): Abschätzung der okkulten und trockenen Deposition am Integrated Monitoring Standort Zöbelboden. Umweltbundesamt, Wien. (unpublished report).

KLEEMOLA, S. & FORSIUS, M. (eds) (2006): 15th Annual report 2006, ICP Integrated Monitoring. Working Group on Effects of the UNECE Convention on Long-range Transboundary Air Pollution. The Finnish Environment, Helsinki.

LANDMANN A.; A. GRÜLL; P. SACKL & A. RANNER (1990): Bedeutung und Einsatz von Bestands-erfassungen in der Feldornithologie: Ziele, Chancen, Probleme und Stand der Anwen-dung in Österreich. *Egretta.* No. 33. pp. 11-50.

LANG, J. B. (2002): Maximum Likelihood Fitting of Multinomial-Poisson Homogeneous (MPH) Models for Contingency Tables using MPH.FIT. www.stat.uiowa.edu/~jblang/mph.fitting/mph.fit.documentation.htm.

LEGGE, A.H. & KRUPA, S.V. (2002): Effects of sulphur dioxide. In: BELL, J.N.B. & TRESHOW, M. (eds). *Air Pollution and Plant Life*. pp. 135-162. John Wiley & Sons Ltd., Chichester.

LEXER, M.J.; HÖNNINGER, K.; SCHEIFINGER, H.; MATULLA, C.; GROLL, N.; KROMP-KOLB, H.; SCHADAUER, K.; STARLINGER, F. & ENGLISCH, M. (2001): The sensitivity of the Austrian forests to scenarios of climatic change. Monographien, Bd. M-132. Umweltbundesamt, Wien.

LINDENMAYER, D.B. & FRANKLIN, J.F. (2002): Conserving forest biodiversity, a comprehensive multiscaled approach. Island Press, Washington, US.

LOVETT, G.M. (1984): Fates and mechanisms of cloud water deposition to subalpine balsam fir forest. *Atmospheric Environment.* No. 18. pp. 361-371.

LOVETT, G. M. (1994): Atmospheric deposition of nutrients and pollutants in North America: an ecological perspective. *Ecological Applications*. No. 4. pp. 629-650.

MACDONALD, J.A.; DIESE, N.B.; MATZNER, E.; ARMBRUSTER, M.; GUNDERSEN, P. & FORSIUS, M. (2002): Nitrogen input together with ecosystem nitrogen enrichment predict nitrate leaching from European forests. *Global Change Biology.* No. 8. pp. 1028-1033.

MEA – Millennium Ecosystem Assessment (2005): Ecosystems and Human Well-being. Biodiversity Synthesis. World Resources Institute, Washington, DC.

MEYERS, T.P.; HICKS, B.B.; HOSKER, R.P.; WOMACK, J.D. & SATTERFIELD, L.C. (1991): Dry deposition interferential measurement techniques -II. Seasonal and annual deposition rates of sulphur and nitrate. *Atmospheric Environment.* No. 10. pp. 2361-2370.

NESTROY, O.; DANNEBERG, O.H.; ENGLISCH, M.; GEßL, A.; HAGER, H.; HERZBERGER, E.; KILIAN, W.; NELHIEBEL, P.; PECINA, E.; PEHAMBERGER, A.; SCHNEIDER, W. & WAGNER, J. (2000): Systematische Gliederung der Böden Österreichs (Österreichische Bodensystematik 2000). *Mitteilungen der Österreichischen Bodenkundlichen Gesellschaft.* No. 60. pp. 1-99.

NEUMANN, M. & STARLINGER, F. (2001): The significance of different indices for stand structure and diversity. *Forest Ecology and Management* No. 145. pp. 91-106.

OKSANEN, J.; KINDT, R. & O'HARA, B. (2005): The vegan package, Version 1.6-10. http://cc.oulu.fi/~jarioksa/softhelp/vegan.html.

PAGE, T.; BEVEN, K. J. & WHYATT, D. (2004): Predictive capability in estimating changes in water quality: Long-term responses to atmospheric deposition. *Water Air and Soil Pollution.* No. 151. pp. 215-244.

PHOENIX, G.K.; HICKS, W.K.; CINDERBY, S.; KUYLENSTIERNA, J.C.I.; STOCKS, W.D.; DENTENER, F.J.; GILLER, K.E.; AUSTIN, A.T.; LEFROY, R.D.B.; GIMENO, B.S.; ASHMORE, M.R. & INESON, P. (2006): Atmospheric nitrogen deposition in world biodiversity hotspots: the need for a greater global perspective in assessing N deposition impacts. *Global Change Biology.* No. 12. pp. 470-476.

R-DEVELOPMENT-CORE-TEAM (2004). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna. http://www.R-project.org.

REHFUSS, K. (1990): Waldböden. Entwicklung, Eigenschaften und Nutzung. 2nd ed. Pareys Studientexte 29. Verlag Paul Parey, Hamburg.

REIMOSER, F. & REIMOSER, S. (2005): Wildökologische Auswertung für den UNECE Integrated Monitoring (IM) Standort Zöbelboden. Umweltbundesamt, Wien. (unpublished report).

REIMOSER, F.; ARMSTRONG, H. & SUCHANT, R. (1999): Measuring forest damage of ungulates: what should be considered. *Forest Ecology and Management* No. 120. pp. 47-58.

ROGORA, M.; MOSELLO, R.; ARISCI, S.; BRIZZIO, M.C.; BARBIERI, A.; BALESTRINI, R.; WALDNER; SCHMITT, M.; STÄHLI, M.; THIMONIER, A.; KALINA, M.; PUXBAUM, H.; NICKUS, U.; ULRICH, E. & PROBST, A. (2006): An overview of atmospheric deposition chemistry over the Alps: present status and long-term trends. *Hydrobiologia.* No. 562. pp. 17-40.

SCHIELER, K. & HAUK, E. (2001): Instruktion für die Feldarbeit - Österreichische Waldinventur 2000/2002. Federal Research and Training Centre for Forests, Natural Hazards and Landscape (BFW).

SLIGGERS, J. & KAKEBEEKE, W. (eds) (2004): Clearing the air: 25 years of the Convention on Long-range Transboundary Air Pollution. United Nations Economic Commission for Europe, UN, Geneve.

SMIDT, S. & OBERSTEINER, E.: 10 Jahre Depositionsmessung im Rahmen des europäischen Waldschadensmonitorings. (submitted to Austrian Journal of Forest Research).

STEFAN, K. & FÜRST, A. (1998): Indication of S and N inputs by means of needle analyses based on the Austrian bio-indicator grid. *Environmental Science and Pollution Research.* Special 5/1. pp. 63-69.

SUDING, K.N.; COLLINS, S.L.; GOUGH, L.; CLARK, C.; CLELAND, E.E.; GROSS, K.L.; MILCHUNAS, D.G. & PENNINGS, S. (2005): Functional- and abundance-based mechanisms explain diversity loss die to N fertilization. *PNAS.* No. 102. pp. 4387-4392.

THOMPSON, L. (2004):. *S-Plus (and R) Manual to Accompany Agresti´s Categorial Data Analysis (2002)* (2nd ed.). https://home.comcast.net/~lthompson221/.

TÜRK, R. & G. ZIEGELBERGER (1982): Die Luftqualität von Salzburg – dargestellt anhand der Verbreitung epiphytischer Flechten. Luftgüteuntersuchungen mit Bioindikatoren im Land Salzburg. Amt der Salzburger Landesregierung (eds.). *Schriftenreihe Luftgüteuntersuchung.* No. 7. pp. 78-141. Salzburg.

UMWELTBUNDESAMT (2005): Critical Loads für Schwefel- und Stickstoffeinträge in Ökosysteme, Datenanfrage 2004. Umweltbundesamt, Wien. (unpublished report).

VAN DOBBEN, H.F.; TER BRAAK, C.J.F. & DIRKSE, G.M. (1999): Undergrowth as a biomonitor for deposition of nitrogen and acidity in pine forest. *Forest Ecology and Management.* No. 114. pp. 83-95.

VDI-Richtlinie 3799 (1995): Blatt 1. Flechten-Kartierungsverfahren. www.vdi.de.

WEISBERG, P. J. & BUGMANN, H. (2003): Forest dynamics and ungulate herbivory: from leaf to landscape. *Forest Ecology and Management.* No. 181. pp. 1-12.

WESELY, M.L. & HICKS, B.B. (2002): A review of the current status of knowledge on dry deposition. *Atmospheric Environment.* No. 34. pp. 2261-2282.

WILLNER, W. (2002): Syntaxonomische Revision der südmitteleuropäischen Buchenwälder. *Phytocoenologia.* No. 32. pp. 337-453.

WITTMANN, H., TÜRK, R., SCHERNTHANER-BLIEBERGER, E. & E. KUPFER-WESELY (1989): Immissionsökologische Studie über die epiphytische Flechtenvegetation in den geschädigten Wäldern Vorarlbergs (Österreich). *Lebensraum Vorarlberg.* No. 3. pp. 47-96. Waldforschung in Vorarlberg, Amt der Vorarlberger Landesregierung.

WOOD, S. (2006): The mgcv package. Version 1.3-22. http://cran.r-project.org/.

ZECHMEISTER, H. G., DIRNBÖCK, T., HÜLBER, K. & MIRTL, M.: Assessing airborne pollution effects on bryophytes – Lessons learned through long-term integrated monitoring in Austria. *Environmental Pollution.* (in press).

ZECHMEISTER-BOLTENSTERN, S.; HACKL, E.; BACHMANN, G.; PFEFFER, M. & ENGLISCH, M. (2005): Nutrient turnover, greenhause gas exchange and biodiversity in natural forests of central Europe. In: BINKLEY, D. & MENYAILO, O. (eds). *Tree species effects on soils: implications for Global Change*. NATO Science Series. Kluwer Academic Publisher, Dordrecht. pp. 1-19.

9 APPENDIX

9.1 Methodological details

9.1.1 Soil

9.1.1.1 Sampling

In 1992 and 2004 soil samples were collected at every main permanent plot position, making up 64 samples. At each location three soil cores were taken and pooled (Figure 3). The humus layer was sampled with a 30 by 30 cm frame. Mineral soil horizons taken at sequential depths of 0–5 cm, 5–10 cm, 10–20 cm, 20–30 cm and 30–50 cm were sampled with a 7 cm diameter soil core borer. Only the humus layer and the first two horizons of the mineral soil were used for the analyses. For each position the soil type was determined according to (NESTROY et al. 2000) and grouped into three soil classes (Leptosol, Cambisol, Planosol) referred to as soil type thereafter.

9.1.1.2 Chemical analyses

The three soil samples per plot were pooled and prepared following ÖNORM L 1060. Water content was measured as the mass loss at 105° C of the air dried fine earth. Soil pH was measured in $CaCl_2$ solution (ÖNORM L 1083). The pH values of the two mineral horizons (0–5 and 5–10 cm) were averaged. Total organic carbon (TOC) was calculated as the difference between total carbon (TC) and total inorganic carbon (TIC). TC was detected colourimetrically (ON EN 13137, ÖNORM L 1080) and TIC was quantified gasvolumetrically (ON EN 13137, ÖNORM L 1084). Total Nitrogen (N_{tot}) was quantified with potentiometric titration (ÖNORM L 1082). Relative amounts of TOC and N_{tot} were corrected for the water content of the fine earth. S was analysed differently. In 1992 S was determined colourimetrically; in 2004 S was pulped in $HClO_4$ and determined using ICP-OES Perkin-Elmer Optima 3000 DV (ÖNORM EN ISO 11885). TOC, N_{tot}, and S were pooled for the two mineral soil horizons. C/N ratios were established with TOC and N_{tot} and C/S ratios with TOC and S respectively.

A subset of stored soil samples from the year 1992 was re-analyzed in order to detect analytical errors due to the use of newer instruments. In total, 20 samples (humus and mineral soil) were selected. A re-analysis of pH and N_{tot} resulted in significantly higher values (mean difference in pH: + 0.36, $p < 0.001$; N_{tot}: + 0.09%, $p = 0.047$; two-sided paired t-test). For TOC no difference was found ($p = 0.225$). S showed significantly higher values (mean = + 0.065, $p = 0.003$). The error variability introduced by chemical analysis is negligible for N_{tot} and insignificant for TOC. The higher pH-values might be due to soil storage rather than analytical errors. The higher S values pointed in the opposite direction of the change observed over time.

9.1.2 Forest floor vegetation

9.1.2.1 Sampling

In 1993, observations were carried out by teams of one and two field botanists, in the year 2005 all plots were recorded by two persons. For recording, a tape measure was used delineating the plots. All plant species present on the plot were recorded with a cover-abundance value estimated on a 7-level ordinal scale (BRAUN-BLANQUET 1964). Woody plants were considered only up to a height of 0.6 m. Plant nomenclature follows (ADLER et al. 1994). Recording in the two observation years was carried out by different field botanists. In order to estimate the errors occurring when plant species abundance is recorded by different observers ("observer error"), a number of 14 permanent plots were selected randomly. In the year 2005 the observer plots were recorded by six independent observer teams consisting of one and two persons. The temporal shift in species composition proved to be more pronounced than the variation in estimates between different observers (see HÜLBER et al. submitted).

9.1.2.2 Statistics

To quantify the changes in cover-abundance estimates of plant species on all plots over time, Marginal Models for square contingency tables were applied (THOMPSON 2004). For each species occurring at least five times in one observation year, a contingency table arranged with defined cover-abundance estimation categories (estimation categories of 1993 as columns, estimations of 2005 as rows) was calculated. The absence of a species was considered as the lowest cover-abundance category. Parameter ß is a quantitative measure of the changes in species cover-abundance from 1993 to 2005. $\beta = 0$ refers to marginal homogeneity (= no change), $\beta < 0$ to decreasing, $\beta > 0$ to increasing cover-abundance. The model was fitted with the MPH.fit function for Multinomial-Poisson Homogeneous Models (LANG, 2002). We sought causes of single species change by using Ellenberg's indicator scores (ELLENBERG et al. 1992) for nitrogen (N), reaction (R), light (L), and moisture (M) (ELLENBERG et al. 1992). Correlation between β of the Marginal Models and Ellenberg species scores was tested using Kendall´s Tau with exact p-value. Only species models with the p-value of the Marginal Model > 0.05, indicating a reasonable fit of the defined logit-model with the data, were considered.

Changes in the vegetation composition of the permanent plots between observations in 1993 and 2005 were evaluated using non-metric Multidimensional Scaling (nMDS) and Multivariate Analysis of Variance (MANOVA). The dissimilarity matrix applied in the nMDS ordination was constructed using Bray-Curtis distances. Vegetation data from both years of observation were included in this matrix. A three dimensional nMDS-space was chosen. Axis values in nMDS-space derived from plot observations in 1993 were subtracted from the values of their corresponding plots in 2005 to obtain vectors (each with three elements) of the distances from paired plots in the nMDS-space. Overall changes in species composition and differences in the changes between different soil types were analysed by using these vectors as response in a MANOVA with soil type as grouping variable. Significant deviation of the intercept from zero would indicate an overall temporal shift in species composition from 1993 to 2005. Any differences in the vegetation shifts between the soil types were indicated by the coefficient of the soil type. The Pillai-Bartlett test statistic was applied. In order to determine causes for the changes of entire plant communities we calculated the unweighted mean values of Ellenberg's N, R, L, and M indicator score of all present species per plot. These figures for 1993 and 2004 were compared using a paired Wilcoxon rank sum test (see HÜLBER et al. submitted).

9.1.3 Bryophytes

9.1.3.1 Sampling

Epiphytic bryophytes

Tree selection: Ten beech trees (*Fagus sylvatica*), ten maple trees (*Acer pseudoplatanus*), and two ash trees (*Fraxinus excelsior*) were used for bryophyte recording. Half of the plots were located on the steep slope, and the other half on the plateau. The trees were distributed evenly over the ICP IM site. Epiphytic plots were established in 1993. Several plots had to be substituted following a loss of recording sites through windfall. Some plots were not established until 1998, and their introduction guaranteed an even distribution across the site, even on the steepest slopes (Figure 2). The bryophytes were recorded along a measuring tape fastened around the trunk of the sample tree at a level of around 120 cm above ground. The ring was fixed into place with a permanent nail, so that the recordings were in the same place for each period. Every 5 cm a recording was taken and the species hitting the tape at these points from the upper edge were recorded. Samples of species unidentifiable in the field were taken for identification in the lab. The frequency of each bryophyte species was recorded and the percentage of coverage was calculated in relation to the stem circumference.

Terrestrial bryophytes

The plots were selected in 1992 on the basis of an even distribution over the site (Figure 2). Terrestrial plots were marked by four steel posts and were recorded by hand drawing in 1992 and 1993. The contour lines of the bryophyte patches were traced onto plastic shields that were put over the terrestrial plot. Where several species were mixed up very closely, they were included in one recording patch and the percentages of the species were recorded in an extra file. The total coverage (cm^2) of each bryophyte species and additional structures was calculated from these maps. From 1998 onwards, the plots were recorded by taking photographs with high quality photographic cameras. Photographs were taken by standardising the height and angle of the camera using a self-made tripod fixed to the four posts of the terrestrial plot. To assure the congruency of both methods, a simultaneous comparison of the methods (hand drawing/photographs) was carried out in 1998 by using both methods on the same plots. No significant differences could be found.

The nomenclature of bryophytes follows the code list for Integrated Monitoring (Code List M2) (IM PROGRAMME CENTRE 1994), which is closely related to the code lists of (CORLEY et al. 1981) and (GROLLE 1983).

9.1.3.2 Statistics

Wilcoxon Matched-Pairs Signed Rank Test was used to test for significant differences in each bryophyte species.

An analysis of the temporal changes in the species composition of bryophyte communities was performed according to the methods applied for forest floor vegetation (chapter 9.1.2.2) with some exceptions: For the epiphytic plots, percentage coverage was used, and for the terrestrial plots the area of bryophyte species per plot. The dissimilarity matrix applied in the nMDS arrangement was constructed using Jaccard distances and thus accounted only for species composition. Results using abundance-based dissimilarity indices like Bray-Curtis did not differ significantly. In order to test the significance of different amounts of N and S deposition on the bryophyte species composition of the terrestrial and epiphytic plots, Kendall´s tau correlation coefficient between nMDS-axis scores and the ranked deposition clusters (see chapter 4.2) were used.

9.1.4 Lichens

9.1.4.1 Sampling

Epiphytic lichens were first recorded in the year 1993. Records are set up on maple (*Acer pseudoplatanus*), beech, (*Fagus sylvatica*), ash (*Fraxinus excelsior*), larch (*Larix deciduas*) und spruce (*Picea abies*). Two repetitions took place in the years 1999 and 2005. Tree selection was done so that an even distribution of tree species was achieved. During the observation period individual trees dropped out due to senescence and wind throw. These trees were substituted by trees of the same species nearby. From a total of 70 plots which were recorded in the year 1993 a subset of 49 remained in 2005. The plots were recorded according to four methods of which three were used here. All plots were marked with a coloured pin in order to guarantee the same position in each observation year. First, the standard methods according to *VDI Richtlinie* [VDI guideline] 3799 (1995) were applied: At a stem height between 1.2 and 1.7 m a 50 * 20 cm mesh with 10 grid cells is fixed. Species frequency is calculated as the number of cells occupied. The air pollution impact is calculated for groups of trees characterised by the same topographic position. The method according to (TÜRK & ZIEGELBERGER 1982) uses also plots between 1.2 and 1.7 m but records half of the stem perimeter and estimates the cover on an 8-level ordinal scale. Air pollution impact is, in addition to species abundance, defined by the vitality of lichens (percentage of injured thallus) and specific injuries (bleach, morphological anomalies, etc.). The only difference of the method according to (WITTMANN et al. 1989) is that plots of undefined size are set up in the lower part of a tree stem. The nomenclature follows (HAFELLNER & TÜRK 2001).

9.1.4.2 Statistics

Trends in the total cover of lichens were analysed using a paired Wilcoxon rank sum test. The analysis of temporal changes of the abundance of single species and of the species composition of epiphytic lichens communities was performed according to the methods applied for forest floor vegetation (chapter 9.1.2.2). The analyses were performed with the data derived from each of the four methods separately and compared thereafter.

9.1.5 Birds

9.1.5.1 Sampling

Sampling of birds was carried out at 64 evenly distributed permanent plots (circles with 50 m radius) using the point taxation method (LANDMANN et al. 1990) (Figure 2). The observational time span per point was five minutes. Each “contact” with a bird species was recorded, singing males were noted as breeding couples (GLUTZ 1962). All permanent plots were visited three times per year and the maximum count of contacts was used further on.

9.1.5.2 Statistics

The maximum count values were averaged for six forest habitat types (Table 9) to which the plots were assigned which led to the density of each bird species per habitat type. The dominance of each species was calculated by its maximum count divided by the sum for all counts of all species. The distribution of birds and the temporal changes were interpreted with regard to nesting and foraging guilds.

Table 9: Forest habitat types used to determine average bird abundance and dominance, together with their area (in hectares) at the ICP IM site.

forest habitat type	area (ha)
Pine *Pinus sylvestris* – Beech *Fagus sylvatica* – Fir *Abies alba* – Spruce *Picea abies* – Forest	11.6
Beech *Fagus sylvatica* – Fir *Abies alba* – Spruce *Picea abies* – Forest	26.8
Artificial Spruce Mixed Forest	35.5
Artificial Larch Larix decidua – Spruce Mixed Forest	2.6
Young Forest Plantations	3.5
Complete Deforestation	8.6

9.1.6 Effects on biodiversity

9.1.6.1 Sampling

A trend analysis was carried out with permanent plot data for all taxonomic groups with the restriction that only those plots were used which were recorded in each observation year. As diversity indices the total number of species in the study area (PSN), the number of species on a permanent plot (SN), and Shannon diversity of species on a plot (SH) were used. SN and SH of the permanent plots for forest floor vegetation, epiphytic lichens and birds were further analysed with respect to their environmental and anthropogenic determinants. In order to do so only the main plots with data on the soil chemical condition and its heterogeneity, the mean radiation and its heterogeneity, on tree layer and tree layer diversity as well as air pollution impact were used. Using these factors allows for an interpretation of the determinants of diversity in different taxa regarding the three main categories:

- site/habitat condition effect
  - forest floor vegetation & epiphytic lichens: mean below canopy radiation of four hemispherical photographs taken at the permanent plot (see chapter 4.3.4.1)
  - epiphytic lichens: tree species identity
  - epiphytic lichens: stem diameter at 1.3 m height as a surrogate for tree age
  - epiphytic lichens: altitude as a surrogate for gross climatic conditions
  - birds: forest type with the two categories deciduous and coniferous
  - birds: amount of standing and lying dead wood (see chapter 4.3.4)
  - birds: tree density (see chapter 4.3.4)

- site/habitat heterogeneity effect
  - forest floor vegetation: standard deviation of below canopy radiation of four hemispherical photographs taken at the permanent plot (see chapter 4.3.4.1)
  - forest floor vegetation: standard deviation of four soil depths from the corners of the 10*10 m permanent plots indicating the degree of variability of soil characteristics
  - forest floor vegetation, epiphytic lichens & birds: tree layer diversity, measured using the stand diversity index according to (JAEHNE & DOHRENBUSCH 1997) (see chapter 4.3.4.3)
  - birds: forest floor vegetation diversity indicated by the Shannon diversity index
- direct or indirect air pollution effect
  - forest floor vegetation: mean pH value of the humus layer and the mineral soil (0–10 cm depth) as influenced by S and N deposition (see chapter 9.1.1)
  - forest floor vegetation: mean C/N ratio of the humus layer and the mineral soil (0–10 cm depth) as influenced by N deposition (see chapter 9.1.1)
  - epiphytic lichens: direct air pollution effect indicated by injuries to lichen individuals.

Soil, radiation and tree layer data is only available for 64 main plots. For all epiphytic lichen plots and a number of bird plots, which are not congruent with the main plots, data from the closest main sampling point was used (< 65 m for lichens and < 100 m for birds). Bryophyte data was not amenable to the analysis since there are not enough plots close to the main permanent plots.

9.1.6.2 Statistics

PSN was calculated as the sum of species occurring on all plots used in each observation year. Temporal trends of SN and SH were tested using a one-sided paired Wilcoxon rank sum test for SN and one-sided paired t-test for SH. For each taxon only the difference between first and last observation year was tested. We used the VEGAN package for calculating diversity indices together with R 2.3.1 (OKSANEN 2004).

Environmental and anthropogenic determinants were analysed using generalised additive models (GAM) with SN as response and the above determinants as predictors. Smooth terms are represented using penalized regression splines with smoothing parameters selected by the Un-Biased Risk Estimator criterion (UBRE). Log-transformation was used for skewed predictors. We used the log-link function with poisson distributed numbers of species. Explained deviance of all partial effects was calculated by dropping terms from the full model. We used the MGCV package and R 2.3.1 for the analysis (WOOD 2006).