CONVENTION ON LONG-RANGE TRANSBOUNDARY AIR POLLUTION
INTERNATIONAL CO-OPERATIVE PROGRAMME ON ASSESSMENT AND MONITORING
OF AIR POLLUTION EFFECTS ON FORESTS
and
EUROPEAN UNION SCHEME
ON THE PROTECTION OF FORESTS AGAINST ATMOSPHERIC POLLUTION

United Nations
Economic Commission
for Europe

European Commission

Intensive Monitoring of Forest Ecosystems in Europe

Technical Report 2003

Prepared by: **Forest Intensive Monitoring Coordinating Institute, 2003**

Forest Intensive Monitoring
Coordinating Institute

Cover photo's by Gert Jan Reinds: Forests in national park Plitvice Lakes and its surroundings, Croatia.

ISSN 1020-6078

Printed in the Netherlands

CONVENTION ON LONG-RANGE TRANSBOUNDARY AIR POLLUTION
INTERNATIONAL CO-OPERATIVE PROGRAMME ON ASSESSMENT AND MONITORING
OF AIR POLLUTION EFFECTS ON FORESTS
and
EUROPEAN UNION SCHEME
ON THE PROTECTION OF FORESTS AGAINST ATMOSPHERIC POLLUTION

United Nations
Economic Commission
for Europe

European Commission

Intensive Monitoring of Forest Ecosystems in Europe

Technical Report 2003

W. de Vries
G.J. Reinds, M. Posch
M. J. Sanz, G.H.M. Krause, V. Calatayud
J.P. Renaud, J.L. Dupouey
H. Sterba, E.M. Vel, M. Dobbertin
P. Gundersen
J.C.H. Voogd

Forest Intensive Monitoring Coordinating Institute, 2003

FIMCI Forest Intensive Monitoring Coordinating Institute

Abstract

EC - UN/ECE, 2003; W. de Vries, G.J. Reinds, M. Posch, M. J. Sanz, G.H.M. Krause, V. Calatayud, J.P. Renaud, J.L. Dupouey, H. Sterba, E.M. Vel, M. Dobbertin, P. Gundersen and J.C.H. Voogd. *Intensive Monitoring of Forest Ecosystems in Europe, 2003 Technical Report.* EC, UN/ECE 2003, Brussels, Geneva, 163 pp.

Apart from an overview of the implementation of the Pan-European Intensive Monitoring Programme of Forest Ecosystems up to 2000, this year's report focuses on (i) ozone concentration data and ozone injury symptoms assessed in a test phase, (ii) ground vegetation species composition in view of environmental factors, (iii) carbon pool changes in trees and soil in relation to nitrogen deposition and (iv) long-term impacts of atmospheric deposition on soil and soil solution chemistry by means of dynamic modelling. Major conclusions are:

Ozone measurements and ozone injury impacts

- Measurements of ozone concentration derived by passive sampling, used at the Intensive Monitoring plots, compare very well to those derived by active monitoring.
- Validated ozone injury symptoms could be presented due to the further development of a sensitive species list, photo gallery, flow chart for injury discrimination and microscopic tools.

Ground vegetation species composition in view of environmental factors

- The various methods used in the first assessment, affect the possibility of integrated analyses, but most aspects have been taken up by the Expert Panel on ground vegetation.
- Mean Ellenberg indicator values reflect fairly well plot environmental conditions, thus being interesting indicators of long-term changes.
- The species composition at the European scale is mainly driven by climate, soils and forest types, but atmospheric (N and S) deposition has a significant impact on the variation.

Carbon pools and carbon pool changes in tree stem wood and soil

- On average, the estimated carbon pools in tree stem wood are approximately twice as low as in soil, but carbon pool changes in tree stem wood are generally 5-10 times as high as in soil.
- Net increases in the carbon pool by forests in Europe (both trees and soil) are in the range of 0.1-0.15 $Gton.yr^{-1}$, being about 50% of the estimated terrestrial carbon sink in Europe.
- The contribution of N deposition to the increase in carbon in standing biomass is approximately 10-20 $Mton.yr^{-1}$, being 3.5 to 7% of the annual estimated forest growth.

Long-term impacts of atmospheric deposition on soil and soil solution chemistry

- Application of a dynamic soil acidification model lead generally to a reasonable to good agreement between measured and simulated data for most of the Intensive Monitoring plots. Sometimes the intra-annual variation in especially nitrate and aluminium concentrations could not be reproduced.
- Evaluations of emission reduction scenarios during 1970-2030 show that strong reductions in dissolved sulphate concentrations have already taken place between 1980 and 2000, due to the high reductions in sulphur emissions in that period.
- Implementation of the Gothenburg protocol is predicted to lead to a significant reduction in nitrate and aluminium concentration by the year 2010, but concentrations of aluminium and their ratio to base cations do remain above critical values at several plots throughout the whole simulation period.
- Changes in the soil chemistry, such as the base saturation and soil C-N ratio, are much slower than in the soil solution chemistry and for a number of plots where acid inputs remains relatively high, base saturation will still decrease in the future.

Keywords: Intensive monitoring, ozone, forest growth, carbon sequestration, atmospheric deposition, dynamic modelling, acidification, eutrophication, soil solution chemistry.

Contents

Preface

The 'Pan-European Programme for Intensive and Continuous Monitoring of Forest Ecosystems' has been implemented in 1995 to gain a better understanding of the effects of air pollution and other stress factors on forests. At present 888 permanent observation plots have been selected. The Intensive Monitoring Programme includes the assessment of crown condition, increment and the chemical composition of foliage and soil on all plots, whereas atmospheric deposition, meteorological parameters, soil solution chemistry and ground vegetation composition are monitored at selected plots. Data are submitted to the Forest Intensive Monitoring Co-ordinating Institute (FIMCI), being a contractor of the European Commission (EC). FIMCI, which is a joint initiative of Alterra Green World Research and Oranjewoud International, has been set up to validate, store, distribute and evaluate the data at European level. FIMCI also acts as an information centre for National Focal Centres (NFC's) of the participating countries.

Between 1997 and 2000, four reports have been published with results from (nearly) all the surveys carried out. Results focused on relationships between crown condition, soil and soil solution chemistry and foliar chemistry on one hand and atmospheric deposition and meteorology on the other hand, using statistical techniques for interpretation. Since 2001, certain topics are highlighted. The focus of 2001 was on water and element fluxes through the forest ecosystem. It also contained first data on the species diversity of the ground vegetation. The report of 2002 focused on relations between plants species composition and environmental factors and on critical loads for nitrogen, acidity and heavy metals and their exceedances by present loads.

This last FIMCI report is the third in the series and includes the participation of various authors for the different chapters as presented below.

1. Introduction: Wim de Vries
2. The Intensive Monitoring Programme: Gert Jan Reinds and Wim de Vries
3. Ozone exposure and ozone injury symptoms at intensive monitoring plots: Maria Jose Sanz[1], George H.M. Krause[2], Vicent Calatayud[1] and Wim de Vries
4. Ground vegetation species composition: Jean Pierre Renaud[3] and Jean Luc Dupouey[3]
5. Carbon pools and carbon pool changes at intensive monitoring plots: Wim de Vries, Hubert Sterba[4], Gert Jan Reinds, Evert Vel, and Matthias Dobbertin[5]
6. Impacts of nitrogen deposition on carbon sequestration by forests in Europe: Wim de Vries, Gert Jan Reinds and Per Gundersen[6]
7. Modelling the long-term impact of deposition scenario's for nitrogen and acidity at intensively monitored forest plots: Gert Jan Reinds, Max Posch[7] and Wim de Vries

[1] CEAM, Parco Technologico, c/Charles H.Darwin 14, SP-46980 Paterna, Valencia, Spain.
[2] Landesumweltamt NRW, Wallneyersrtrasse 6, D-45133 Essen, Germany.
[3] INRA, Centre de Nancy, Unité ecophysiologie, F - 54280 Champenoux, France
[4] Universitat fur Bodenkultur Wien, Peter Jordanstrasse 82, A - 1190 Wien, Austria
[5] WSL, Zürcherstrasse 111, CH - 8903 Birmensdorf, Switserland.
[6] Danish Forest and Landscape Research Institute (DFLRI), Hoersholm Kongevej 11, DK-2970 Hoersholm, Denmark.
[7] RIVM- Coordination Center for Effects, P.O. Box 1, 3720 BA, Bilthoven, the Netherlands.

Overall editing was done by Wim de Vries, including the writing of the summary. Regarding chapter 4 on ground vegetation, we thank Dan Aamlid chairman of the ground vegetation expert panel (UN-ICP Forest / EU) and all participants and co-ordinators who kindly supplied information about their plots. The comments of Han van Dobben, Walter Seidling, and other reviewers were also greatly appreciated. For chapter 7 the hydrological model WATBAL developed by Mike Starr of the Finnish Forest Research Institute (METLA) was used; his support during the application of the model and the improvements he made to it during the last year are gratefully acknowledged.

The target groups of this report are the active participants of the Intensive Monitoring Programme (National Focal Centres, National Involved Research Institutes, Scientific Advisory Group, the Expert Panel Members, the Standing Forestry Committee of the European Union and ICP Forests) and the Scientific Community. The preparation of this report was possible thanks to the submission of data and information by the NFC's to FIMCI and the active participation and co-operation of the members and deputy members of the Scientific Advisory Group.

Extended Summary

The Intensive monitoring programme

The Pan-European Intensive Monitoring Programme of Forest Ecosystems started in 1994. The general aim of the Intensive Monitoring Programme is to contribute to a better understanding of the impact of air pollution and other stress factors on forest ecosystems. At present, the programme covers 888 selected plots in 30 participating countries. In total 791 Intensive Monitoring plots have been installed. Some surveys are mandatory and have to be carried out on all plots (crown condition, soil chemistry, foliage and forest growth). At part of the plots, assessments of atmospheric deposition (513 plots), meteorology (206 plots), soil solution chemistry (242 plots), ground vegetation (748 plots) and remote sensing (157 plots) are carried out. For nearly all the plots information on the methods applied is available. The results up to 2000 include data for 773 plots with respect to crown condition, 710 plots for soil, 753 plots for foliar composition, 766 plots for forest growth, 528 plots for atmospheric deposition, 694 plots for ground vegetation, 247 plots for soil solution and 189 plots for meteorology. Furthermore, data on ambient air quality and phenology are available at 170 and 64 plots, respectively.

Objectives

The aim of the thematic Technical Report is to inform active participants of the programme and scientist with relevant information from the monitoring programme and to promote co-operation between FIMCI and other (potential) users of the data. It includes the scientific background of the major results presented in the executive report, which is specifically aimed at both policy makers and the wider public. The focus of this year's report is on:

- Ozone measurements in view of possible impacts,
- Carbon sequestration in trees and soil and in relation to nitrogen deposition
- Long-term impacts of atmospheric deposition on soil and soil solution chemistry by means of dynamic modelling.

Furthermore, an update of the analyses of ground vegetation data was carried out, focusing on methodological aspects affected the comparability of ground vegetation data.

Ozone exposure and ozone injury symptoms: results of a test phase

Background

In nearly all regions of Europe ozone concentrations during the spring and summer are high enough to be of potential risk to sensitive plants. Threshold values for the protection of forest trees are frequently and repeatedly exceeded in many areas. Presently there are a few stations in Europe in rural areas compared to urban and suburban stations. Therefore the EU/ICP Forests programme launched a test-phase in order to explore the possibilities of monitoring ozone on its forest plots. The test-phase focussed on the use of passive samplers and the possibilities of visible ozone injury assessments.

Methods to derive ozone exposure

The total number of plots where passive samplers have been installed for ozone in 2001 equals 104 in nine countries. Despite various methodical difficulties with respect to concentration measurements by passive sampling, results show that the concentrations thus derived are very well comparable to those derived by active monitoring. For example, results in the case of Spain show that concentrations derived by active sampling are on average slightly (4%) lower than those obtained by passive sampling, but the relationship between both concentrations is strong (r^2 value of 0.91). This close relation shows that passive sampling can give reliable ozone concentration measurements over a given time period independently of the site characteristics. However, such relations have to be checked for other orographic and climatic types within Europe as well. Results of the test phase for 2001, although they are limited and do not represent all Europe (104 plots and 9 countries) reflect patterns already reported in the literature showing a gradient North to South with higher concentrations in the Southern Europe. Values themselves, due to the rather cold climate during summer months in 2001 in Central Europe, were comparatively low compared to former years (i.e. 1995 and 1997).

Methods to derive ozone injury symptoms
Nine countries reported results from 72 plots in the first ozone injury assessment in 2001. Light Exposed Sampling Sites (LESS) were established at 67 plots in eight countries. Three Validation Centres established for Southern, Central and Northern Europe functioned as centres of expertise/competence. Since 2000 three training courses were carried out in Spain, Switzerland, and France, where evaluators were trained on sampling procedures, symptom evaluation and other diagnostic tools. The test phase also lead to the further development of tools, such as a Sensitive Species List, a Photo Gallery, a Flow Chart for injury discrimination and Microscopy Protocols. At 18 plots visible ozone injury was reported. Among the 23 tree species screened, six showed validated ozone injury symptoms. Out of the total number of 67 LESS-sites, ozone symptoms were observed on one or more species in 37 sites (55%). Many of the species registered with ozone symptoms were not known to be ozone sensitive before.

Main result of the test phase
Main results form the test phase are that a monitoring system for ozone assessment at European scale, including concentrations and visible injury, is being defined and tested. By doing so, considerable knowledge was built up on the reliability of measuring ozone concentrations by using passive sampling and about ozone symptom expressions on different species in many countries. The assessment of ambient ozone concentration and of ozone injury on main tree species and ground vegetation has to be considered as a first attempt for a unique effects monitoring system on a European scale based on real field observations. However, it does not give an overall picture for Europe at the present stage.

Species composition of the ground vegetation

Approach
Ground vegetation represents a key component of the ICP-Forest programme. In last years report, ground vegetation data were related to stand characteristics, climatic conditions, soil conditions and deposition to analyse and understand the relation between the species composition of the ground vegetation and environmental factors. A further analysis of the data was carried out this year, focusing specifically on methodological heterogeneities within the ground vegetation network that could have affected survey comparability. We also examined species diversity indices and the influence of environmental factors on species composition at the European scale.

Methodological aspects
In this first round of ground vegetation assessment, several methods were used. These differences affect the possibility of integrated analyses at the European level in which heterogeneity in sampling area is the most important bias source. More species were generally recorded with larger sampling areas. In repeated surveys, an increase in species richness can also be due with the

familiarisation of the observers with the local flora. An important structural bias of the database is the lack of identification of the observers. With an increased number of observers per plot the number of species observed generally tends to increase. So this aspect of the database should be improved for further studies. Fencing tended to decrease species richness of many plots, most likely due to an increased biomass of competitive species (e.g. Rubus sp.) due to reduced herbivorous activities. If one wants to draw conclusions about spatial or temporal vegetation changes, comparison should be performed, using harmonised methods and should include several years per point compared, in order to cope with inter-annual variations and observer drifts. Most aspects have already been taken up by the Expert Panel on ground vegetation.

Diversity indices, Ellenberg indicators and influence of environmental factors
Diversity indices calculated for 671 Intensive Monitoring plots showed large geographical variation in species richness at the European scale. This variation was influenced by tree species, stand density, soil pH and plot nutritional aspects (foliar Ca concentrations). Mean Ellenberg indicator values reflect fairly well the plot environmental conditions. They could therefore constitute interesting indicators of long-term changes. A correspondence analysis, performed on 602 Intensive Monitoring plots showed that the species composition at the European scale is mainly driven by classical factors such as climate, soils and forest types. This conclusion is in line with results published the previous year. A small but significant part of the variation was related to atmospheric (N and S) deposition. Using logistic regression, several species showed their optimum located at low or at high N deposition. These species can be used as indicators of eutrophication or acidification of forest ecosystems.

Carbon pools and carbon pool changes in tree stem wood and soil

Approach
An estimate of net carbon pool changes in Intensive Monitoring plots was based on repeated forest growth surveys for trees. Carbon pools in trees in stem wood were calculated by multiplying stem wood volumes with stem wood density times the carbon content in stem wood. Stem volume was either submitted by the countries or calculated from the diameter of the tree at breast height and tree height. The pool of organic carbon in the soil was calculated by multiplying an estimated bulk density of the soil with the soil thickness and the carbon concentration in the soil. Carbon pool changes in the soil were based on calculated nitrogen retention (N deposition minus N leaching) rates in soils minus N uptake and multiplied by the C/N ratio of the forest soils. In order to scale up results to Europe, an estimate of soil carbon pool changes in was calculated for the more than 6000 level I plots using: (i) N deposition by model estimates, (ii) net N uptake by yield estimates as a function of site quality, (iii) N retention fractions in soil related to measured C/N ratios, based on results from level II plots and (iv) measured C/N ratios for forest soils.

Carbon pools and carbon pool changes at Intensive Monitoring plots
The geographic variation in the carbon pools in tree stem wood is related to the variation in standing biomass with lowest carbon pools (< 30 $ton.ha^{-1}$) in Northern and Southern Europe, due to temperature impacts (cold climate and water stress), and moderate (30-120 $ton.ha^{-1}$) to high carbon pools > 120 $ton.ha^{-1}$) in Central Europe. Results for the carbon pools in soils are deviating, with high pools occurring in Northern Europe because of the low mineralisation of carbon due to temperature extremes. In Southern Europe, there is a large variation in soil C pools. Results for the carbon pool changes show a comparable pattern to the carbon pools, with low changes in carbon pools in trees (< 1 ton $C.ha^{-1}.yr^{-1}$) in Northern and Southern Europe and moderate (1-4 ton to high changes (>4 ton $C.ha^{-1}.yr^{-1}$) in Central Europe.

Carbon sequestration on the European scale and the impact of N deposition
The carbon pool changes in the tree are generally 5-10 times as high as the estimated carbon pool changes in the soil. As expected the changes in the carbon pool in tree due to forest growth increase going from Northern to Central Europe. The calculated changes in the carbon pool in soil are high in Central Europe and low in Northern and Southern Europe, since it follows the N deposition pattern. Net increases in the carbon pool by forests in Europe (both trees and soil) are in the range of 0.1-0.15 $Gton.yr^{-1}$, being an important part (about 50%) of the terrestrial carbon sink in Europe, derived from atmospheric inversion models. The results furthermore show that the C sequestration by forest is mainly due to a net increase in forest growth, since the longer term C immobilisation in the soil is limited. The contribution of N deposition to the increase in carbon in standing biomass is approximately 10 and 20 $Mton.yr^{-1}$. If one relates the additional growth to the estimated forest growth of approximately 280 $Mton.yr^{-1}$, the contribution varies between approximately 3.5 and 7%.

Modelling the long-term impact of deposition scenario's for nitrogen and acidity

Approach
The dynamic soil chemistry model SMART was applied to about 200 Pan-European Intensive Monitoring plots for which both element input (deposition) and element concentrations in the soil solution were available. The plots occur in a transect from South -Western Europe to Scandinavia, the majority being located in Western and Northern Europe. Within the SMART model, parameters related to: (i) weathering of base cations, (ii) nitrogen transformations and budgets, (iii) cation exchange and (iv) aluminium to pH relationships were calibrated for each plot. Other processes such as uptake of N and base cations were (in) directly derived from measurements at the plot such as forest growth data. Hydrology was computed on a monthly basis with the WATBAL model. After calibration the model was applied for the plots and statistical measures were computed that indicate the goodness-of-fit between measured and simulated soil solution concentrations. Furthermore, two emission reduction scenario's were evaluated, one following the Gothenburg protocol and one using maximum feasible emission reductions, to determine (differences in) the recovery of forest ecosystems.

Model validation/calibration
Most of the processes in the SMART model could be successfully calibrated. The chloride and sodium budgets show that the hydrology at the plots is generally well simulated. The nitrogen budget could, however, only be closed assuming a time independent N immobilisation for a number of plots. Sulphate adsorption could not be modelled with the available data and has thus been ignored in the model applications. Calibration shows a much better agreement between pH and free (uncomplexed) aluminium than between pH and total aluminium. Base cation weathering rates computed from the base cation budget show the expected relationship with soil texture. Computed cation exchange constants show large variations over the plots, but this is not uncommon. There was generally a reasonable to good agreement between measured and simulated data for most of the plots, although some of the intra-annual variation in especially nitrate and aluminium concentrations could not be reproduced for a number of plots. Statistical measures for the goodness of fit indicate that pH is on average very well reproduced by the model, but not all variations within the year are accurately simulated. Relative deviations between measured and simulated nitrate and aluminium concentrations are sometimes considerable (mostly between -50 and +50 %), especially for plots with low average measured data, but absolute errors in the simulated concentrations are often very low.

Model application

Evaluation of the emission reduction scenarios in the period 1970-2030 shows a very strong reduction in sulphate concentrations between 1980 and 2000 in the soil due to the high reductions in sulphur emissions. By the year 2010, a significant reduction in nitrate concentration is predicted for most plots, but the effect is most striking for the plots with the highest present N concentrations. Aluminium concentrations above an assumed critical value of 0.2 $mol_c.m^{-3}$ occur at about 25 % of the plots in the beginning of the simulation period (1970). Simulations also show that implementation of the Gothenburg protocol causes a reduction of this percentage to about 5-10 % of the plots in 2030. Al/BC ratios above a critical value of 1 occur at about 5 % of the plots in 1970 and this percentage slightly decreases towards 2010. This slight decrease is because at a number of plots the decrease of base cation concentration, due to replenishment of the exchange complex, is stronger than the reduction in Al concentration. Base saturation improves over time for most plots but for a number of plots where acid inputs remains relatively high, base saturation will still decrease in the future. This shows the difference between the fast reactions in soil solution to emission reductions and the slower reactions of the soil solid phase. Future simulations show that the MFR scenario leads to lower sulphate and aluminium concentrations in 2030 than the Gothenburg scenario and that the MFR scenario is much more effective in reducing nitrate concentrations.

1 Introduction

In order to gain a better understanding of the effects of air pollution and other stress factors on forest ecosystems, the Pan-European Programme for Intensive and Continuous Monitoring of Forest Ecosystems was established. This chapter first presents information on the background and current status of the Intensive Monitoring Programme (Section 1.1). It then explains the focus of this year's Technical Report in view of the overall objectives of the programme (Section 1.2) and it ends with a description of the content of the Technical Report (Section 1.3).

1.1 Background and current status of the Intensive Monitoring Programme

Background of the programme

The Pan-European Programme is based on both the European Scheme on the Protection of Forests against Atmospheric Pollution and the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) under the Convention of Long-Range Transboundary Air Pollution (UN/ECE). In 1994, the Intensive Monitoring Programme was established by the EC with the aims to (ICP Forest, 2000):

- Monitor effects of anthropogenic (in particular air pollution) and natural stress factors on the condition and development of forest ecosystems in Europe.
- Contribute to a better understanding of cause-effect relationships in forest ecosystems functioning in various parts of Europe.

At present 862 permanent observation plots for Intensive Monitoring of forest ecosystems have been selected. Details on the plots and assessments are given in chapter 2.

The Intensive Monitoring Programme includes the assessment of crown condition, forest growth (increment) and the chemical composition of foliage and soil on all plots. Additional measurements on selected plots include atmospheric deposition, meteorological parameters, soil solution chemistry and ground vegetation. Within each of these surveys, a number of mandatory and optional parameters have been defined. The temporal resolution of the present surveys is scheduled as follows:

- Crown condition (at least once a year)
- Chemical composition of the concentrations of needles and leaves (at least every 2 years)
- Soil chemistry (every 10 years)
- Increment / forest growth (every 5 years)
- Atmospheric deposition (continuous)
- Soil solution chemistry (continuous)
- Meteorology and phenology (continuous)
- Ground vegetation (every 5 years)
- Remote sensing/aerial photography (once)
- Ambient air quality and ozone injury (continuous)

Aims of the Programme

A major objective of the 'Pan-European Programme for the Intensive Monitoring of Forest Ecosystems' is to gain a European wide overview of the impacts of air pollution and other stress factors on forest ecosystems. An overview of the most relevant relationships to be derived with the data in the Intensive Monitoring database is given in Fig. 1.1. The results should be useful for the evaluation of (protocols on) air pollution control strategies used within the UN/ECE Convention of Long-Range Transboundary Air Pollution and the EC. Specific objectives in the context of air pollution are the assessment of:

- The fate of atmospheric pollutants in the ecosystem in terms of accumulation, release and leaching.
- Critical loads and critical levels of atmospheric pollutants (SO_2, NO_x, NH_3, metals) in view of ecosystem effects in relation to present loads.
- Responses of forest ecosystems to (changes in) air pollution by deriving relationships between (trends in) stress factors and ecosystem condition.
- Influences of future scenarios of air pollution on the (chemical) ecosystem condition.

Recently, the aims of the Pan-European Programme have been widened towards the topics of biodiversity and climate change. In this context, the Programme aims to contribute to the development and monitoring of ‘criteria and indicators for sustainable forest management’. Objectives of the Pan-European Programme related to this topic are the:
- Assessment of net carbon sequestration in European forests, to improve the assessment of the global carbon balance and to evaluate the influence of changes in the climate due to atmospheric greenhouse gasses on the forest ecosystem.
- Further development and monitoring of indicators related to the various functions of forest ecosystems to assess its long-term sustainability, such as forest ecosystem health, forest production, species composition of ground vegetation and protective functions of soil and water resources.

1.2 Aim of the report

The contents of Technical Reports on the ‘Pan-European Programme for the Intensive Monitoring of Forest Ecosystems’ in Europe differ each year in view of the increased data availability in time. Since 2001, we focus on certain topics/themes by various in-depth studies, according to the publication strategy for the period 2001-2005 (See De Vries et al., 2001). It aims to ensure an adequate supply of policy relevant information for the coming period and an alternation of a focus on abiotic and biotic aspects.

The focus of 2001 was on water and element fluxes through the forest ecosystem, thus presenting information on the fate of atmospheric pollutants in the ecosystem in terms of accumulation, release and leaching. The report of 2002 focused on critical loads for nitrogen, acidity and heavy metals and their exceedances by present loads. It also contained relations between plant-species composition and environmental stress factors. Considering the aims of the programme mentioned above, the focus of this year’s report is on:
- Long-term impacts of scenarios of atmospheric deposition on soil and soil solution chemistry.
- Carbon sequestration in trees and soil and the role of nitrogen deposition

Furthermore, the report contains results of a test phase of ozone measurements in view of possible ozone injury impacts and an in-depth study on environmental impacts on ground vegetation. The latter study adds additional aspects to the study that has been presented in last years report (De Vries et al., 2002). The aspects that have been investigated in this year’s report are illustrated in Figure 1.1.

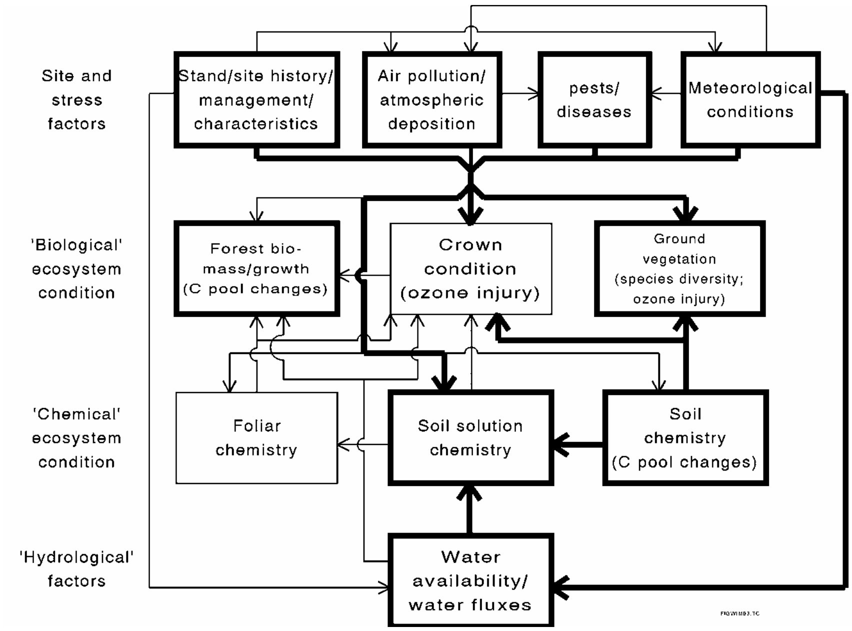


Figure 1.1 Flow diagram illustrating the relationships between site and stress factors and the forest ecosystem condition. Boxes and arrows in bold are specifically investigated in this year's report.

1.3 Contents of the report

Chapter 2 provides information on the current implementation of the Intensive Monitoring Programme, including information on the selected plots in the various surveys and the submitted data and information until 2000. Chapter 3 gives the results of a test phase on the assessment of ozone exposure by passive samplers and on visible ozone injury symptoms. Results on the species diversity of ground vegetation at approximately 670 plots are presented in chapter 4. This chapter focuses on methodological aspects of the ground vegetation survey that affected survey comparability. As in the previous year, relationships between the species composition of ground vegetation and environmental factors, such as tree species, soil factors related to acidity/ nutrient availability, climatic variables and atmospheric deposition.

An overview of carbon pools and carbon pool changes in tree stem wood, based on a repeated forest growth survey is presented in Chapter 5 for approximately 650 plots. For comparison, this chapter also gives information on carbon pools in the soil at those plots. A distinction has been made in the change in the living stock (trees that were alive in both surveys) and total stock (trees that were alive or dead in both surveys removed in the last survey). An assessment of carbon pool changes in trees and soil at approximately 120 Intensive monitoring plots and at the whole of Europe, based on approximately 6000 Level I plots is given in Chapter 6. Results for the tree are based on forest growth data and for the soil on the immobilisation of nitrogen and the C/N ratio of the soil. This chapter also contains information on the impact of nitrogen deposition on carbon sequestration by trees and soils. Finally Chapter 7 presents results of long-term impacts of the so-called Gothenburg scenarios of atmospheric deposition on the soil and soil solution chemistry of approximately 120 Intensive Monitoring plots. This chapter includes a comparison of measured and predicted soil solution concentrations to evaluate the dynamic soil model used for those plots.

2 The Intensive Monitoring Programme

The Intensive Monitoring Programme is carried out on plots that were selected in such a way that it includes the major combinations of tree species and soil type in a country. In this chapter an overview of plots in the various surveys (Section 2.1) and of the data that have been stored until 2000 (Section 2.2) are presented.

2.1 Selected plots in the various surveys

The Intensive Monitoring Programme now includes 888 plots from 30 participating countries. Some countries that participate in the ICP Forests programme, have indicated their participation in the Intensive Monitoring programme, but have not yet sent the general plot information. The number of plots that have presently been installed equals 791 of the 888 plots.

Table 2.1 shows the number of plots selected and installed and the number of plots on which the different surveys (crown condition, soil, foliage, forest growth, deposition, soil solution, meteorology and ground vegetation) are (planned to be) executed. Four surveys have to be conducted on all plots (crown condition, soil, foliage and forest growth). According to the information received, atmospheric deposition is carried out at 513 plots and ambient air quality at 160 plots. Surveys with respect to meteorology and soil solution measurements are carried out at 206 and 242 plots respectively. Ground vegetation surveys are carried out at 749 plots, whereas the application of aerial photography is foreseen at 157 plots (Table 2.1). Several countries also plan to or do carry out additional surveys on the plots, such as physiopathology, litter fall, phenology, mycorrhizae and/or fungi and other in-depth studies to soil water regimes, gas exchange and air quality measurements.

An overview of the surveys carried out at the different plots is given in Fig. 2.1. This map is based on information submitted until February 2003 and includes data up to the end of 2000. The map makes a distinction between plots where:

- Only mandatory core surveys (crown condition, soil, foliage and increment) are carried out.
- All surveys are carried out, including the core surveys and the optional surveys deposition, meteorology, soil solution chemistry and ground vegetation
- Core surveys are carried out in combination with one or more optional surveys (mostly deposition and ground vegetation).

The map shows that number of plots at which all surveys are carried out occur mainly in a north-south transect. It also shows that atmospheric deposition (at least bulk deposition, but mostly also including throughfall) is measured at much more plots than those where all surveys, including meteorological measurements and soil solution chemistry, are carried out.

2.2 Submitted data and information until 2000

Table 2.2 gives an overview of the number of installed plots, and the number of plots for which data, DAR-Q and both data and DAR-Q's are stored. Table 2.2 shows that for the vast majority of the plots with stored data, also the DAR-Q information is available. This table furthermore shows that the number of plots for which both data and DAR-Q information were stored is (slightly) lower than the number of installed plots. The main reasons for these differences are:

Table 2.1 Overview of the number of selected plots for the main surveys (Crown, Soil, Foliar and Growth are core surveys and the remaining surveys are optional).

Countries	Total	Crown	Soil	Foliar	Growth	Atm. Dep.	Meteo	Soil sol.	Ground Veget.	Rem. Sens.	Air Quality	Phen ology
EU countries												
Austria	20	20	20	20	20	20	2	2	20	20	-	-
Belgium Flanders	12	12	12	12	12	6	2	6	12	-	1	-
Belgium Wallonia	9	9	9	9	9	4	4	2	8	-	-	-
Denmark	17	17	15	17	17	8	3	8	17	-	3	-
Germany	91	91	91	91	91	88	68	80	82	49	60	15
Greece	4	4	4	4	3	4	4	-	4	-	4	4
Spain	53	53	53	53	53	12	12	2	52	-	12	12
France	100	94	100	94	94	24	25	14	94	14	24	75
Ireland	15	15	15	15	15	3	8	3	9	15	-	-
Italy	28	28	20	28	28	16	16	2	28	20	28	28
Luxembourg	2	2	2	2	2	1	2	-	2	-	-	-
Netherlands	14	14	14	14	14	4	-	4	14	-	-	-
Portugal	9	9	9	9	9	1	1	1	9	-	-	-
Portugal Azores	4	4	4	4	4	1	1	1	-	-	-	-
Finland	31	31	31	31	31	16	12	16	31	-	-	3
Sweden	100	100	100	100	100	46	-	46	98	12	17	-
United Kingdom	20	20	20	20	20	20	2	7	20	-	-	-
Total EU	529	523	519	523	522	274	162	199	501	130	149	137
Non-EU countries												
Bulgaria	3	3	3	3	3	3	3	3	3	-[2]	-	-
Belarus	81[1]	81	81	81	81	-	-	-	-	-	-	-
Switzerland	16	16	16	16	16	13	16	7	16	16	16	2
Czech Republic	14	14	14	14	14	5	2	3	11	-	5	-
Estonia	6	6	6	6	6	5	-	2	6	-	-	-
Croatia	7	7	7	7	7	2	3	3	4	-[2]	-	-
Hungary	15	15	15	15	15	15	15	-	15	-	-	-
Lithuania	9	9	9	9	9	-	-	-	9	9	-	-
Latvia	2	2	2	2	2	2	2	2	2	-	-	-
Norway	19	19	19	19	19	19	-	19	19	-	-	-
Poland	150	150	150	150	150	150	-	-	150	-[2]	-	-
Romania	13	13	13	8	13	4	-	4	13	-[2]	-	-
Russia	12	12	12	12	12	12	-	-	-	-[2]	-	-
Slovenia	3[1)]	3	3	3	3	2	3	-	-	-[2]	-	-
Slovak Republic	9	9	9	9	9	7	-	-	-	2	-	-
Total non-EU [3)]	359	359	359	354	359	239	44	43	248	27	21	2
Total	**888**	**882**	**878**	**877**	**881**	**513**	**206**	**242**	**749**	**157**	**170**	**139**

1) In these countries plots have not yet been installed.
2) At these plots, remote sensing measurements do take place but the number has not yet been confirmed.
3) Meanwhile also Cyprus has installed 4 plots; these were not yet included in this table

- Some countries have not submitted data for some of the surveys.
- Some countries submitted data that were not stored because the data were incomplete or problems exist with respect to their quality.
- At some of the installed plots, monitoring has started only very recently. Consequently, no data or DAR-Q information is available yet.

Inversely, the number of plots for which data are stored is generally larger than the number where assessments are presently carried out, since some plots have been abandoned (Compare Table 2.1 and 2.2). Compared to last years' report, the number of plots with data has only slightly increased for most surveys. The largest increase is found for forest growth as a number of countries have submitted both data and DAR-Q information last year. Ground vegetation data are now available for more than 85% of all plots, ground vegetation DAR-Q's for about 70% of the plots.

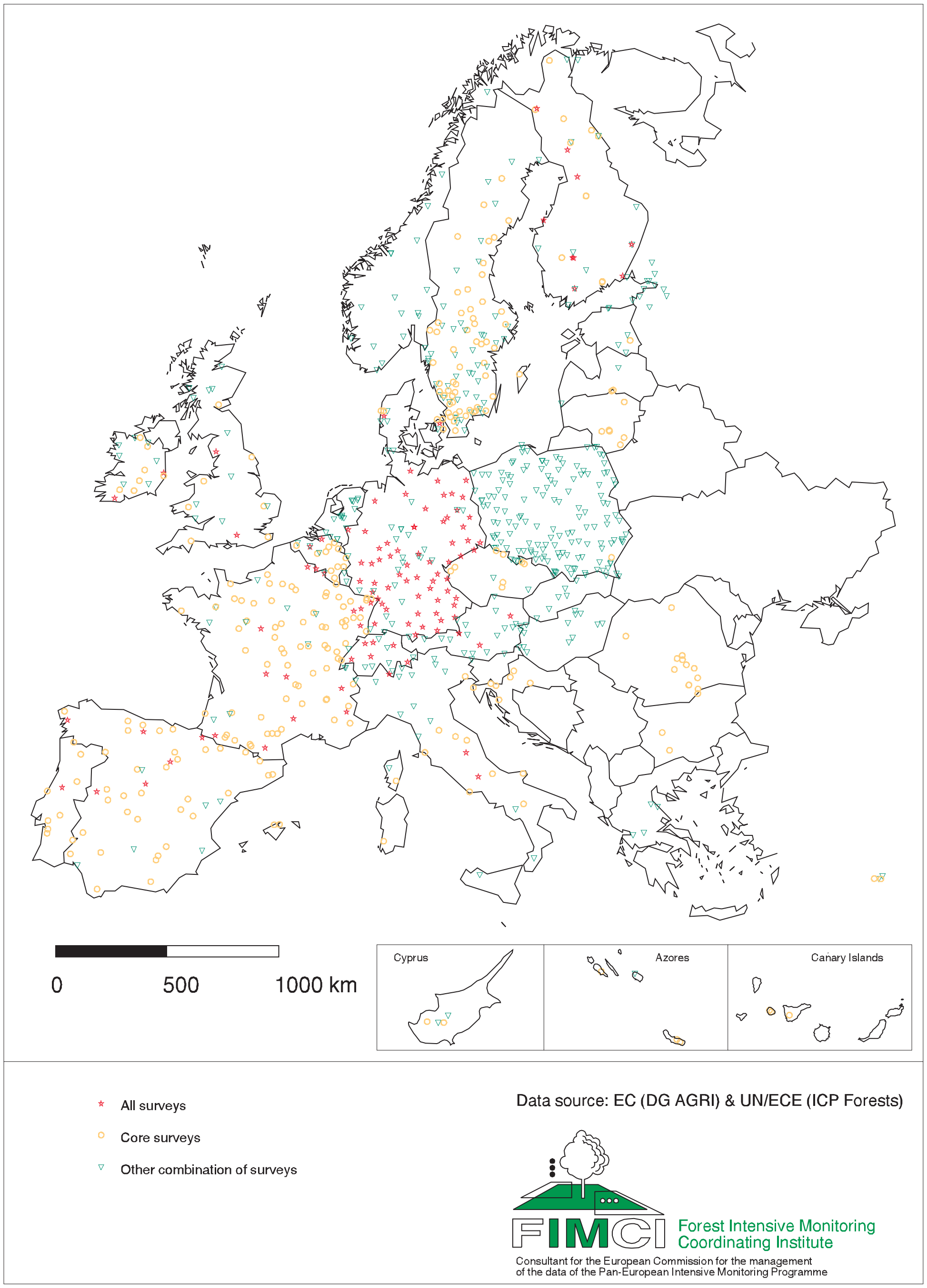


Figure 2.1 *Geographical distribution of installed Intensive Monitoring plots based on information received until February 2003. Core surveys include crown condition, soil, foliage and increment, whereas all surveys include the core surveys and deposition, meteorology, soil solution chemistry and ground vegetation*

Table 2.2 Overview of the number of plots for which data and/or information was submitted for the eight surveys until the year 2000[1)].

Survey	Selected plots[2)]		Data stored		DAR-Q information stored		Data and DAR-Q information stored	
	EU	non-EU	EU	non-EU	EU	non-EU	EU	non-EU
Crown condition	522	359	515	258	507	244	504	231
Soil condition	519	359	496	214	446	234	440	208
Foliar condition	522	353	515	238	453	244	449	229
Growth	521	359	511	255	472	94	470	80
Deposition	274	239	306	222	259	157	294	182
Meteorology	162	44	160	29	168	28	150	24
Soil solution	199	43	214	33	201	29	196	29
Ground vegetation	500	248	481	213	348	214	329	198

1) For soil, foliage and forest growth, also data from earlier years have been used.

2) The number of plots for which data are stored is sometimes higher than the number of plots selected, because at a number of plots measurements were stopped in the last years: for these plots only short datasets of the period before 1998/1999 are available.

3 Ozone exposure and ozone injury symptoms at intensive monitoring plots: results from the test phase 2001

3.1 Introduction

Background

Ozone measurements suggest that surface ozone concentrations at mid- to high latitudes have more than doubled during the past century (Sandroni et al., 1993; Marenco et al., 1994). Elevated levels are found in urban areas, but also in rural and remote mountainous regions due to the transport of ozone and its precursors. Elevated sites show little diurnal variation, and ozone concentrations increase with altitude, frequently resulting in a higher dose (concentration times time) as pollution burden. The concentrations of photochemical oxidants, ozone in particular, may exceed the thresholds for effects on sensitive tree species throughout Europe in the latitude range of 35^{o} to 70^{o} N (EC, 1999).

According to Fowler et al. (1999), ozone (O_3) is the most important regional air pollutant that may impact forest vegetation in Europe and elsewhere. A comparison of modelled ozone concentrations in the growing season on a world-wide scale compared to areas with occurrence of visible symptoms based on field and experimental evidences, seem to support the above concern (Collins et al., 2000). Recently, the UN/ECE, also concluded that tropospheric ozone is the main photochemical oxidant to be considered. In nearly all regions of Europe ozone concentrations during the spring and summer are high enough to be of potential risk to sensitive plants. The AOT40 value, that is recognised as an accepted standard for the protection of forest trees from adverse ozone effects (Kärenlampi and Skärby, 1996), is frequently and repeatedly exceeded in many areas (Sanz et al., 1999).

Ozone, unlike e.g. fluoride or sulphur dioxide pollution, leaves no elemental residue that can be detected by analytical techniques. Therefore, visible injury on needles and leaves is the only easily detectable evidence which can also be diagnosed in the field and considered as a distinct sign of potential impairment of forest ecosystems. Visible injury does not include all the possible forms of injury to trees and natural vegetation (i.e. pre-visible physiological changes, reduction in growth, etc.). Nevertheless, observation of typical symptoms on above ground plant parts in the field has turned out to be a valuable tool for the assessment of ozone injury in sensitive species in Europe. Furthermore, visible injury is regarded as a result of oxidative stress, leading to a cascade of adverse effects resulting in a reduced vitality of forest species and increasing predisposition to climatic, edaphic and biotic factors.

The evidence we have today strongly suggests that ozone occurs at concentrations causing visible foliar injury to sensitive plants (see also UNECE - CLRTAP, 1999).Surveys have recorded ozone-like symptoms on numerous native tree and shrub species in southern Switzerland (Innes et al., 1996; Skelly et al., 1998; Skelly et al., 2000; Guenthardt-Goerg, 2001; Van der Heyden et al., 2001), Greece (Velissariou et al., 1992), Italy (Cozzi et al., 2000), France (Dalstein et al., 2002) and Spain (Sanz et al., 2000; Skelly et al., 2000). However, little information is available on the effects of ozone on the multitude of native plant species throughout Europe (Ashmore and Davison, 1996).

Aims of the Working Group on Air Quality and the study

Up to now, the lack of ozone data was a serious limitation for the Intensive Monitoring (Level II) database. Besides the obvious connections with the potential effects on forests, ozone data are also relevant in relation to other themes which were subjected to important political agreement, like the tropospheric chemistry changes and the regional ozone formation. Examples are the CLRTAP Multi-Pollutant, Multi-effect Protocol ; the UN Convention of Biological Diversity; the EU Habitat Directive; the EU Acidification Strategy, the UN/ECE CLRTAP and the EU Air Quality directive (De Vries, 2000).

The Working group on Air Quality within the Expert Panel of Deposition aims to improve the knowledge of air concentration of various pollutants and effects associated with such an impact across forested areas in Europe. This is done by using the tools *passive sampling* and *visible injury assessment* on Main Tree Species (MTS) and Ground Vegetation (GV) on Intensive Monitoring Plots within ICP-Forests. The main pollutant considered is ozone for the reasons mentioned above, although other pollutants like SO_2, NO_2 and NH_3 are included to complete the deposition surveys that ICP-Forests is carrying out at the IMP. Since much of the ozone data at European level comes from monitoring devices located in urban/sub-urban areas (e.g. De Leeuw et al., 2001) often located at low altitude a comprehensive dataset about forest sites will provide a considerable input for a better understanding of ozone levels in remote areas including a broad range of altitudes.

ICP Forests Task Force mandated the Working Group on Ambient Air Quality, to have a one year test phase about the above mentioned work program. In order to come to a conclusive decision on how to continue the program, the Task Force asked the countries at its meeting 2001 in Lisbon, to report their data on passive sampling as well as ozone injury assessment as quick as possible. Data were send to FIMCI, and in parallel to CEAM Spain as the main Co-ordination Centre of the Working Group, for further evaluation. The following report corresponds to this request and highlights the first results and recommendations for further progress on the 2001 data base. Ozone is the main pollutant discussed further, focusing on the possibility to produce reliable information on both the air concentration at a given site and visible ozone injury on selected plant indicators (MTS and GV) using methods according to the manual. One has to be aware that the distribution of the plots with ozone concentration and ozone injury data is not representative in a statistical sense and the fact that only some countries volunteered in this pilot study make extrapolation or more general conclusions very difficult. It does, however, allow conclusions on methodological aspects, being the main focus of this test phase.

Contents of this chapter

In chapter 3.2, we describe the approaches that were used to assess ambient ozone concentration by Passive Sampling and the preliminary results obtained in this test phase. The approaches that were used to assess visible ozone injury and the first results that were obtained are described in Section 3.3, whereas the possibilities for combining information on ozone exposure and ozone injury are presented in Section 3.4. Finally, a discussion of the results and conclusions with recommendations for the future is presented in Section 3.5.

3.2 Assessment of ambient ozone concentration by Passive Sampling

The use of passive samplers (Passam) is meanwhile considered as a reliable and comparatively cheap method to gain information on ambient air quality, specifically in remote forest areas where no other technical facilities are available to operate continuous monitoring stations. Therefore, this method was chosen to obtain information on ambient air quality at IMP sites. However, it is necessary to elucidate how these data compare with data from continuous monitoring sites. This information is particularly important for QA/QC as well as for extending the data base for further modelling of ozone concentrations over Europe (e.g. EMEP). Passive sampling monitoring within the Working Group is done in accordance to the manual which it is strongly related to the CEN document 264 (CEN, 2001).

3.2.1 Materials and methods

The Working Group requested the NFC's in 2000 to supply information on methods, period of exposure, number of plots, sampling details and analytical procedure. A summary of the results is presented in table 3.1. Passive sampling is the main method on sites that did not currently monitor ozone using active samplers before 2001. Individual countries were free to select the type of passive sampling device they use. It was recommended that the samplers be run at selected sites during the vegetation period in parallel with active monitors according to the EU Daughter Directive (COM 1999, 125) reference method, UV-spectroscopy and/or with an instrument run at an EMEP site in accordance with the EMEP Manual (EMEP/CCC/ Report 1/15, NILU, Norway). However, one country selected as a method exclusively active monitors (Denmark, but only data from 2000 were submitted).

Table 3.1 shows the countries that participated in the passive sampling exercise, number of plots with passive samplers installed in 2000-2001 and compounds measured. The total number of plots where passive samplers have been reported to be installed for ozone since 2000 or 2001 equalled 104 in nine countries. Several countries keep the samplers all year, whereas others concentrate on the growing season. Analyses of the results are on their way by the countries.

Table 3.1. Number of plots per country where passive samplers, installed since 2000 or 2001, for ozone, sulphur dioxide were reported to the Working Group (Sanz and Krause, 2001).

Country	Nº plots*			
	Ozone	Sulphur dioxide	Nitrogen oxide	Ammonia
Austria	2	0	0	0
France	24	0	0	0
Germany	29	15	24	24
Greece	4	0	0	0
Italy	24	11	11	0
Luxembourg	2	0	0	0
Spain	12	12	12	12
Switzerland	5	0	11	11
UK	2	2	2	10
Total	104	40	60	57

* Numbers were updated after early data submission for 2001 ozone data. In the case of France, there is an apparent inconsistency, 26 plots with passive sampling data were submitted but only 24 belonged to Intensive Monitoring plots.

A general overview of the methods used by the various countries that carried out measurements of ambient air quality using passive samplers is given below:

- **Selection of plots**: Ambient air quality monitoring was in most cases carried out on plots where also meteorology and deposition data are available.
- **Location in the plot:** Monitoring was carried out in an open field, and devices are located close to the meteorological equipment at 2-4 m height.
- **Number of samplers and height:** Duplicate samplers at an appropriated height was the most frequent set up for all countries.
- **Period of sampling:** Sampling was carried out preferably on a 1 to 4-week basis depending on the countries. Measurements of ozone were mostly limited to the leafed period for deciduous species. Some countries, however, continued for the rest of the year for all pollutants (Table 3.2).
- **Analytical procedure:** Individual countries were free in their choice of methodology, as long as good quality was assured. Samplers were generally analysed at one laboratory per country.

Table 3.2 Ambient Air Quality information related to Passam activities reported to the Working Group in 2000 (Sanz and Krause, 2001).

	Exposure period (nr of months)	Exposure interval (weeks)	Passam type	Nº countries
Ozone	6,8,9,12	1,2,4	PDT, Indigo papers, IVL, Gradko, DPE/Passam, Ogawa	11
Sulphur dioxide	6,7,8,9,12	1,2,4	Potassium Carbonate + glicol, TEA, IVL, Gradko	8
Nitrogen oxides	6,7,8,9,12	1,2,4	IVL, TEA, Gradko	8
Ammonia	6,8,9,12	1,2,4	Sulphuric acid, Palmmes-Sammler, Gradko, IVL, Willems badges	7

PDT = Palmes diffusion tube home made IVL = Swedish Environmental Research Institute diffusion samplers
TEA = Triethanolamine, home made

Figure 3.2 shows the Intensive Monitoring locations where ozone data from passive sampling measurements are available. In several cases, the intensive monitoring locations with passive sampling were located close to active monitoring stations to control the quality of the passive sampling data. In several plots in the United Kingdom (2 plots), Austria (1 plot) and Italy (4 plots), the active monitors were very close or even in a level II plot (not more than a few km). Furthermore, in Spain (3 plots) and Switzerland (1 plot), continuous active monitors were also available but relatively far from the Intensive Monitoring locations (e.g. 50km). In the case of Denmark only 2 active monitors are reported, with data for 2000 (not included in the Report).

3.2.2 Preliminary results

Comparison of ozone concentrations by active and passive sampling

As pointed out before, it was considered necessary, to ensure the quality of the data gained by passive sampling by comparing these data with those from active monitoring sites. For this purpose, devices were exposed in close vicinity of the active monitoring station and at the Intensive Monitoring plot and concentration measurements compared on the basis of equal exposure periods.

Such a comparison was carried out in several countries (Austria, Germany, Italy, Spain and UK). As an example, the result of three Stations in Spain are shown in figure 3.1. One hour averaged means from real time measurements from each of the station (Valencian Community Air Quality Network) were aggregated to 14 day means and results compared with data of PASSAM´s exposed over 14 days at each station in accordance with CEN 264. Figure 3.1 A-C gives the individual linear regression curves for the 14 day intervals for the year 2001 (n= 48) for the different stations. The station characteristics and hence also the exposure conditions differ with respect to orography, climate and altitude. In figure 3.1 D data sets for all stations were pooled.

Results of the example show that concentrations derived by active sampling are on average slightly (2-11%) lower than those obtained by passive sampling, but the relationship between both concentrations is quite strong (R^2 value varying between 0.70 and 0.87). Pooling all samples increased the R^2 value to 0.91, while the average difference between active and passive sampling is only 4%. This close relation between ozone concentrations by PASSAM and active monitoring data shows that passive sampling can give reliable ozone concentration measurements over a given time period independently of the site characteristics.

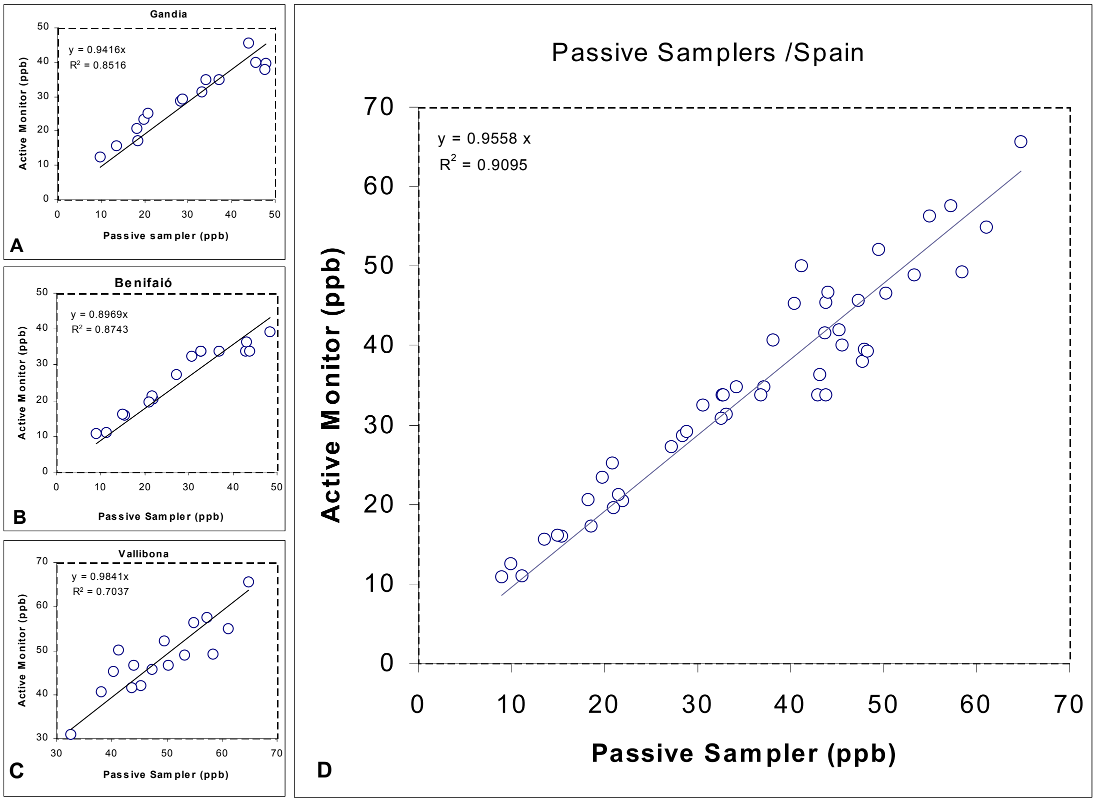


Figure 3.1 Comparison of ozone concentrations by active and passive monitoring at 3 stations within the Air Quality Monitoring Network of Valencia, Spain (A-C) and for a pooled data set (D).

However, such relations have to be established for other orographic and climatic types within Europe as well, before a more general statement can be given. Furthermore, the data have not yet been fully evaluated with respect to comparisons between Intensive Monitoring plots and nearest active monitoring stations in the vicinity.

Ozone concentration measurements by passive sampling at intensive monitoring plots

Although so far limited information on PASSAM data are available, a first evaluation on ambient ozone concentrations was carried out and results are presented in figure 3.2. The data represent a mean from April to September 2001. Data as shown are plausible as higher concentrations occur more or less in southern Europe, whereby 58 % of the Spanish Sites and 63 % of the Italian sites show a 6-month-time weighted-average concentration in the range of 45-60 ppb. In Germany, 65 % and 35 % of the sites showed an average concentration of 15-29 or 30-44 ppb, respectively. In France, nearly 50 % of the sites fell in either of the two latter categories. However, it has to be borne in mind that 2001 was generally considered as a rather low ozone year as compared to others (i.e. 1994, 1997, etc., see EEA reports).

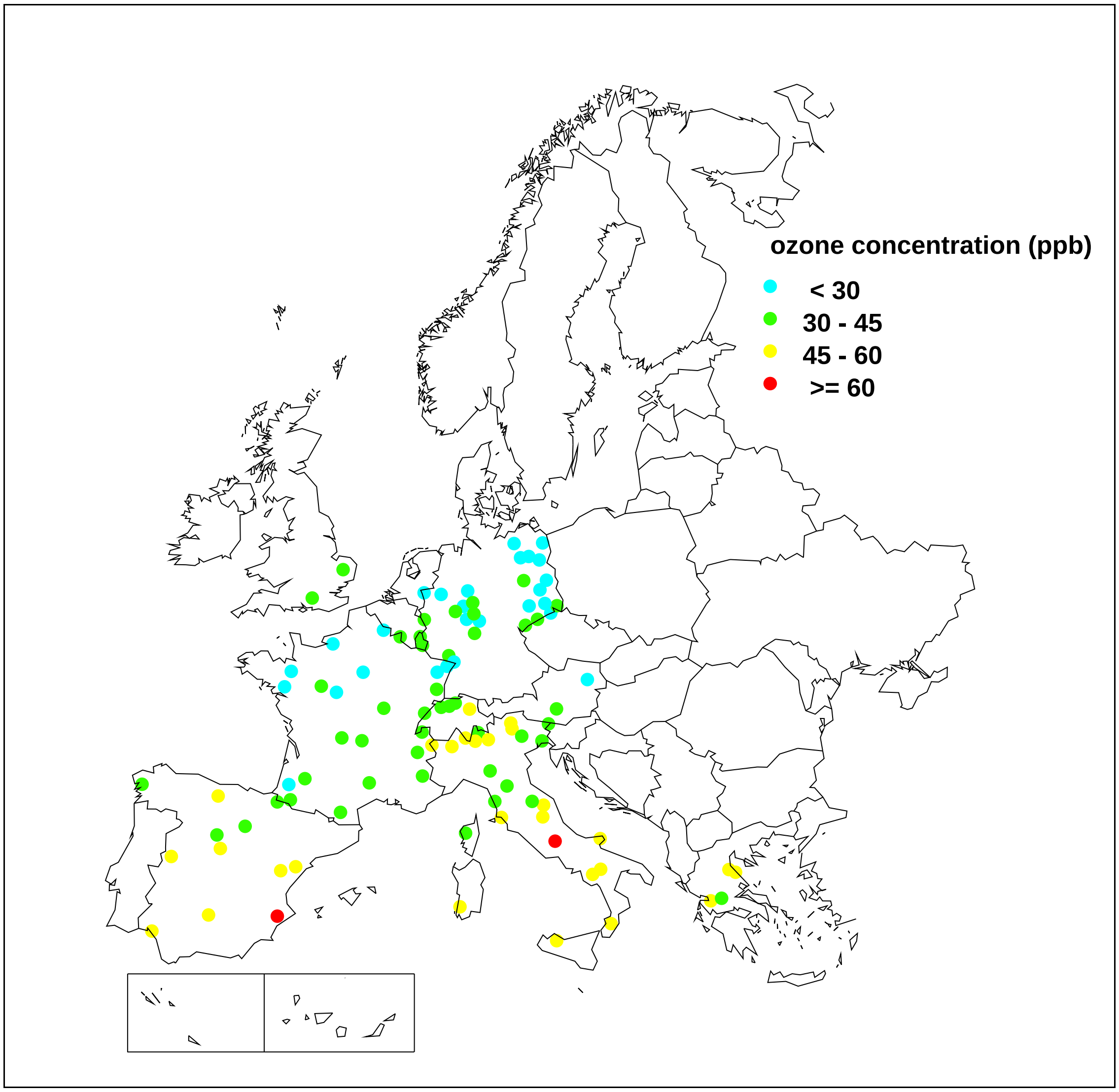


Figure 3.2 Time-weight average concentration of ozone (ppb) at selected plots included in the test phase for the season 2001 (April1-September 30 2001) measured by Passive Sampling

3.3 Assessment of visible ozone injury

3.3.1 Material and Methods

Assessment of visible ozone injury on Main Tree Species and Light Exposed Sampling Sites

Visible ozone injury assessment was carried out on Main Tree Species(MTS) in the plot and on existing native ground vegetation in a Light Exposed Sampling Sites (LESS) in a nearby forest edge. In these spots, foliage was screened for ozone injury with respect to the parameters *species*, *frequency* and *intensity* by optical observation. Leaves on which ozone injury surveys were carried out in the US as well as in Europe during the eighties, within the context of Novel Forest Decline, showed the difficulty of discrimination ozone injury from confounding symptoms, associated with biotic (fungi, insects, etc.) or abiotic (i.e. edaphic) stress factors. To facilitate this, tools such as a Sensitive Species List, a Photo Gallery, a Flow Chart for injury discrimination (Innes et al., 2001), as well as Microscopy Protocols were developed. The assessment on MTS was performed together with the sampling of leaf material for nutrient analysis in 2001 in order to limit resources to a minimum.

Training Courses, Validation Centers

The possibility of confounding factors with respect to diagnosis of ozone injury symptoms led to the necessity of building up expertise within the participating countries for differential diagnosis. Since 2000 the Working Group carried out three training courses in Spain, Switzerland, and France, where evaluators were trained on sampling procedures, symptom evaluation and other diagnostic tools. It is being shown that if injury needs to be quantify in the future, Training Courses should include field exercises organised in a manner to put under control the errors and the deviations among the field evaluators, for example using photographic standards for the main sensitive species. The three Validation Centers established for Southern, Central and Northern Europe functioned as centers of expertise/competence, which countries could address in cases of doubt concerning symptom expression. Methods on microscopical differentiation tools were developed in Germany and in Switzerland at WSL (http://www.wsl.ch), Zurich. The latter institute built up a *central differential diagnosis laboratory* for the Working Group and is developing an interactive web-based ozone-injury database recently (http:// www.ozone.wsl.ch/).

Photo Gallery, Sensitive species list

Several activities have been undertaken to help ozone specialist in the different countries to assess the symptoms in the field. The Central Coordination Center of the Working Group at CEAM, Valencia, Spain developed and continuously improves the WebPage of the Working Group (http://www.gva.es/ceam/icp-forests), among which is an extensive data bank of photographs on the expression of ozone symptoms, ozone like symptoms and typical confounds in several European species. Participating countries give continuous input to the picture gallery. Similarly a sensitive species list is provided to assess ozone injury at the LESS-sites. Again, this list is continuously improved by new information on sensitive species, not having been known to react sensitive to ozone. Additionally, WSL has developed a interactive, web-based ozone-injury database (http:// www.ozone.wsl.ch/).

3.3.2 Preliminary Results

The first survey on MTS was handicapped by pooling the sampling of leaves for nutrient analysis and ozone injury assessment for deciduous trees. Sampling of leaves for nutrient analysis had to be carried out in early summer months, where the expression of ozone symptoms is limited in comparison to the months of August and September. The sampling date for coniferous trees in January was acceptable, since symptoms developing during the year, will normally stay also during wintertime. Furthermore, sampling was done in 2001 and time was extremely short to build up the necessary expertise for the countries. This has been experienced during diagnostic exercises which were carried out at a forest sites in Moggio, Italy 2001 (Bussotti et al., 2003) and Nice, France in 2002 (Minutes of Training Course Nice 2002). Only ozone injury data that were validated by the Validation Centres are included. Ozone like injury was not included. The data presented here have therefore a rather limited value and the evaluation has to be considered as preliminary.

Main Tree Species

Nine countries reported results from 72 Intensive Monitoring Plots in the first ozone injury assessment in 2001. Data of only those species showing symptoms are compiled in Table 3.3. In Germany, Switzerland, and Italy *Fagus sylvatica* is the dominant deciduous trees species. Ozone symptoms were found at 3 of the 23 German Level II plots (13 %; but n=6 (26%) if also 3 other plots with similar but not validated symptoms are included) and 5 of the 7 Swiss plots (71 %). In Italy, one of the 8 Italian plots with *Fagus sylvatica* showed injury (but very scarce), and in Austria and Spain, the single plots of *Fagus sylvatica* examined also had symptoms. Symptoms were also detected in *Carpinus betulus* in Switzerland. No ozone injury was detected for *Quercus robur* or *Quercus petraea* (Germany) as well as other *Quercus spp* (Greece). From the coniferous species, verified observations on ozone injury were reported for *Pinus nigra*, *Pinus pinaster* (France), *Pinus halepensis* (Spain, Greece), and *Pinus strobus* (UK). With the exception of France, however, symptom expression was very weak and scattered. In summary, among the 23 tree species screened, 6 showed validated ozone injury symptoms. In total, the number of plots with ozone injury symptoms was 17 out of 72 (24%; Table 3.3).

However, as pointed out earlier due to the rather cold climate during summer months in 2001 in Central Europe, the ozone burden was comparatively low and results are in the range of expectations by the Working Group. For practical reasons, sampling generally took place during the early summer months, where symptom expression turned out to be weaker than in August/September.

Table 3.3 Main Tree Species (MTS) with ozone injury in Europe, 2001. Species that did not show injury in any plots are omitted.

MTS that showed Injury in 2001	Country	Nº of plots with ozone injury	Total Nº of plots
Carpinus betulus	Switzerland	1	1
Fagus sylvatica	Austria, Germany, Italy, Spain, Switzerland	11	44
Pinus halepensis	Greece*, Spain	2	2
Pinus nigra	France	1	2
Pinus pinaster	France	1	2
Pinus strobus	UK**	1	1

* Accesory plot, not yet integrated in level II plots
** Ozone symptoms mixed with aphid damage

Light Exposed Sampling Sites

Light Exposed Sampling Sites (LESS) were established in Austria (1), France (10), Germany (16), Greece (4), Italy (8), Spain (10), Switzerland (15), and the Slovak Republic (3) and vegetation screened according to the sensitive species list for ozone injury. Table 3.4 gives an overview of the species showing symptoms on the LESS observations for the year 2001 in Europe. Out of the total number of 67 plots in which LESS-sites were established, ozone symptoms were observed on one or more species in 32 LESS-sites (48%), whereas at 35 LESS sites no ozone injury was found.

Many of the species registered with ozone symptoms were not known to be ozone sensitive before (18 out of 61). Interestingly, several species showed symptoms in several countries, as shown e.g. in previous studies in Spain and Switzerland (Skelly et al., 2000). This confirms that the response of the herbaceous species is still largely unknown and the field survey can improve that. Especially in Switzerland, surveys were carried out very intensively, showing the highest density of plots as well as species with ozone injury. Although LESS sites are not likely to be representative of the ground vegetation within the plot itself in terms of ozone exposure (as concentrations are often significantly lower beneath the canopy), the survey provides biological monitoring data on the effects of ozone on easily accessible woody and not woody plant material.

3.4 Possibilities for combining information on ozone exposure and ozone injury

In the future, it is important to link the information of the information systems on *ambient air quality* and *ozone effects assessment.* Both systems are not completely overlapping, however. Data on the numbers of available plots with data on ozone injury symptoms at MTS and LESS plots and plots with data on both concentrations and injury are presented in Table 3.5. Data on ozone exposure are available at 104 plots, on injury symptoms at 87 plots and on both ozone concentrations and ozone injury symptoms at 48 plots. The latter plots allow the possibly in the future to assess relationships between concentrations and injury and possibly other environmental factors. A map of the plots with data on ozone exposure by passive sampling, ozone injury and both ozone exposure and ozone injury is given in Figure 3.3.

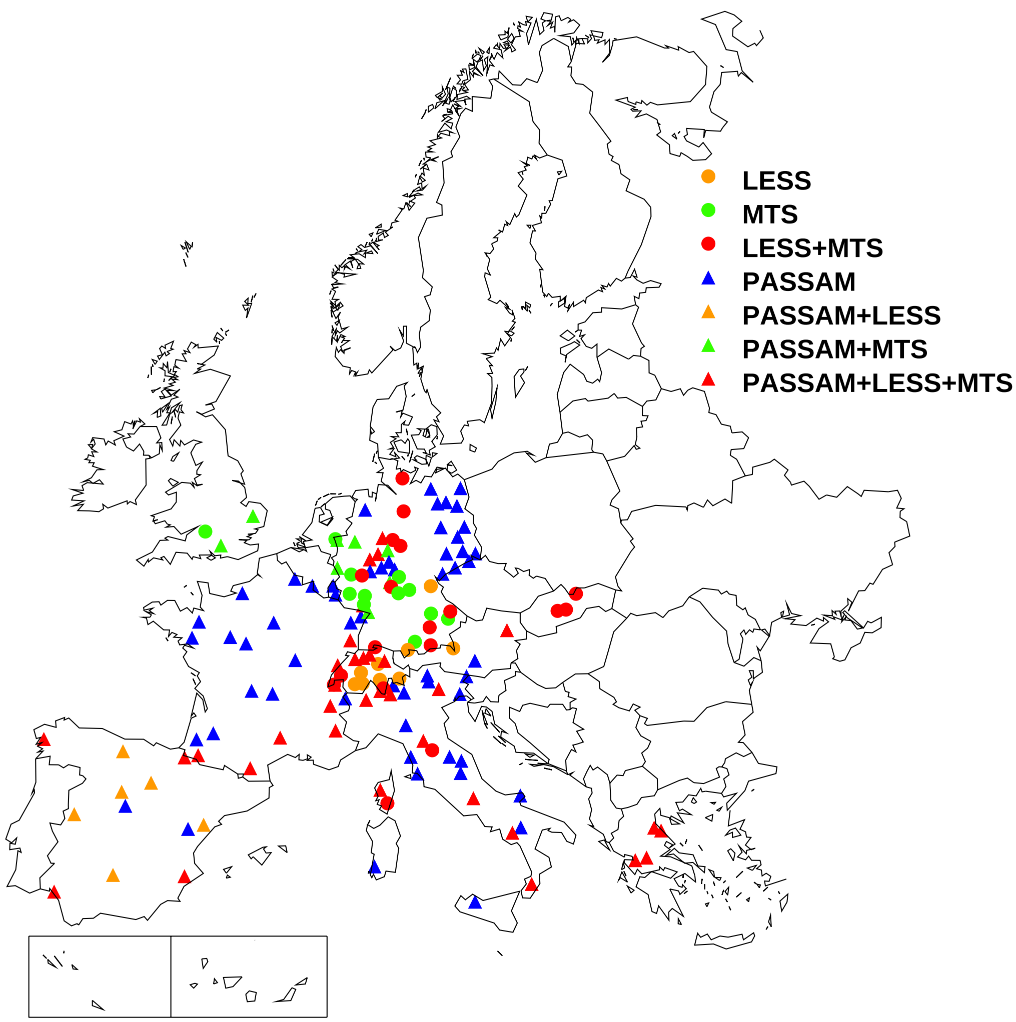


Figure 3.3 *Location of the plots with available ozone exposure and/or ozone injury data for main tree species MTS and light exposed sampling sites (LESS) in 2001. Circles represent plots with no ozone concentration data (PASSAM) available, and triangles correspond to plots where ozone concentration data are available.*

In this map, the different possibilities are represented separately (e.g. plots with only PASSAM data, or only with injury data either on the MTS or on the LESS or both, and other possible combinations of these three types of information).

Table 3.4 Species showing Ozone Injury on Light Exposed Sampling Sites in Europe, 2001

Type	Species	Country	Nº of plots per country			Total Nº of plots showing Injury
Trees	Acer campestre	Switzerland	1			1
	Acer pseudoplatanus	Switzerland	2			2
	Alnus glutinosa	France, Switzerland	1	2		3
	Alnus incana	Switzerland	2			2
	Alnus viridis	Switzerland	1			1
	Betula pendula	Germany	1			1
	Carpinus betulus	Switzerland	4			4
	Corylus avellana	France, Switzerland	1	3		4
	Fagus sylvatica	France, Germany, Switzerland	5	2	6	13
	Frangula alnus	Switzerland	1			1
	Fraxinus excelsior	Austria, France, Switzerland	1	1	6	8
	Picea abies	Germany, Switzerland	1	3		4
	Pinus halepensis	Spain	1			1
	Pinus sylvestris*	Switzerland	1			1
	Populus tremula	Switzerland	2			2
	Prunus avium	Switzerland	1			1
	Robinia pseudoacacia	Switzerland	1			1
	Salix alba	Switzerland	1			1
	Salix caprea	Germany, Switzerland	1	1		2
	Salix sp.*	Switzerland	1			1
	Sambucus nigra	Switzerland	1			1
	Sorbus aria	Switzerland	2			2
	Sorbus aucuparia	Switzerland	2			2
	Sorbus mougeotti	Switzerland	1			1
	Taxus baccata*	Switzerland	1			1
	Tilia cordata	Switzerland	1			1
	Ulmus glabra	Switzerland	1			1
Shrubs	Cornus sanguinea	Switzerland	4			4
	Crataegus laevigata	Switzerland	2			2
	Crataegus monogyna	France, Switzerland	1	3		4
	Crataegus oxyacantha*	France	1			1
	Evonymus europaeus	Switzerland	1			1
	Lonicera nigra*	Switzerland	1			1
	Lonicera xylosteum	Switzerland	4			4
	Prunus spinosa	Switzerland	4			4
	Rosa canina	Switzerland	3			3
	Rosa sp.	France	1			1
	Rubus fructicosus	France, Switzerland	4	2		6
	Rubus idaeus	France, Germany, Switzerland	3	1	1	5
	Rubus spp.	Italy	1			1
	Sambucus racemosa	Switzerland	2			2
	Sorbus chamaemespilus*	Germany	1			1
	Vaccinium myrtillus*	Switzerland	2			2
	Vaccinium uligunosum gaultherioides*	Switzerland	1			1
	Viburnum lantana	Switzerland	2			2
	Viburnum opulus	France, Switzerland	1	1		2
Herbs	Aquilegia vulgaris*	France, Switzerland	1	1		2
	Artemisia campestris*	France	1			1
	Astrantia major*	Italy	1			1
	Centaurea nigra*	Italy	1			1
	Cirsium helenidoides*	Switzerland	1			1
	Filipendula ulmaria*	France	1			1
	Helleborus niger*	Italy	1			1
	Heracleum sphondylium juranum*	Switzerland	1			1
	Impatiens parviflora*	Switzerland	1			1
	Lamium spp.	Italy	1			1
	Oenothera biennis*	Switzerland	1			1
	Plantago lanceolata	France	2			2
	Plantago major	France	1			1
	Senecio nemorensis*	Germany	1			1
	Senecio ovatus*	Switzerland	1			1

* Species that were listed in the Sensitive List as ozone sensitive from literature (See WebPage)

Table 3.5 Overview of the number of plots for which ozone injury data (MTS or LESS) and/or ozone concentration data were available in 2001. For more detailed information see Fig. 3.3.

Countries	Nr of plots with ozone concentration data	Nr of plots with ozone injury data			Nr of plots with ozone concentration and ozone injury data		
	PASSAM	MTS	LESS	Total	MTS	LESS	Total
Austria	2	1	1	1	1	1	1
France	24	10	10	10	9	9	9
Germany	29	30	16	33	10	4	10
Greece	4	4	4	4	4	4	4
Italy	24	8	8	8	8	8	8
Luxembourg	2	0	0	0	0	0	0
Slovak Republic	0	3	3	3	0	0	0
Spain	12	4	10	10	4	10	10
Switzerland	5	9	15	15	5	5	5
UK	2	3	0	3	2	0	2
Total	**104**	**72**	**67**	**87**[1]	**42**	**40**	**48**[1]

[1] Note that this total is not the sum of MTS and LESS of the previous columns since in many plots both MTS and LESS injury assessment was undertaken. It stands for MTS, LESS or MTS+LESS (87 plots) or Passam with MTS, LESS or MTS+LESS

3.5 Conclusions and recommendations

Conclusions on the test phase

The data presented here for ozone concentration measurements by passive sampling as well as the ozone injury assessment on main tree species and ground vegetation from the selected LESS-sites relate to a one-year test phase and should be regarded as preliminary. The locations are also by no means representative for Europe in neither space nor time. All methodical aspects of the program were discussed in depth at the 3rd International Training Course in Nice, 25.-27th September 2001 and participants of the workshop concluded that the test phase was successful and should be extended from 2003 to 2005 emphasising operational aspects. Problems to be addressed, if cause-effect relationships have to be developed, concern the representativity of the LESS and the MTS plots at a territorial scale, the knowledge of the behaviour of the herbaceous species and the validation of symptoms when we observe a large variability of the responses.

Despite various methodical difficulties with respect to concentration measurements by PASSAM's, results show that the concentrations thus derived are very well comparable to those derived by active monitoring. Considerable knowledge has been built up during the test period about ozone symptom expressions on different species in many countries and the sensitive species list has been considerably extended. The assessment of ozone injury on main tree species as well as ground vegetation at the LESS-sites has to be considered as a first phase towards a unique effects monitoring system on an European scale based on real field observations. Thus the programme serves as an independent information system together with the emission inventory data and the ambient air quality monitoring. While data derived from the latter can only give information an a potential risk, the effects monitoring gives information on effects truly manifested in form of visible plant reaction, hence allowing for the development of realistic risk scenarios. These data are further of use for model calibration purposes like in EMEP, or other air quality systems. In the recent Critical Level Workshop in Gothenburg, ICP-Forest was encouraged to continue ozone injury monitoring for developing a sound database on effects with ongoing time.

Outlook and recommendations

The Working Group follows the perspective to link the information of the two information systems *ambient air quality* and *ozone effects assessment* in a spatial analysis of ozone concentrations and relating the effects on vegetation to these data in a geographic information system (GIS). The data of the two systems are placed as individual layers over a given geographic area and analysis looks for systematic coincidences like hot spots, etc. Such an approach seems to be appropriate in a system where many of the variables follow stochastic processes. Similar approaches have been used successfully in epidemiology, however, methodology might need further development. In assessing relationships between ozone exposure indices and ozone injury, it is relevant for example to account for modifying factors such as geographical and meteorological factors. It is anticipated that a five-year measuring period will enable the necessary information basis.

Passive samplers are useful to get an idea about mean weekly/fortnightly ozone levels, but they do not provide indication about e.g. AOT40 values, i.e. the exposure indicator adopted to estimate the potential risk for forests as well as natural vegetation and crops (Führer et al., 1997). Such an estimate can, however been made from the data collected by passive samplers and their validation against continuous ozone monitors. In this context, some work has been done to estimate ozone concentrations under complex terrain condition as function of altitude and daytime (e.g. Loibl et al., 1994) and this provide the basis for calculating AOT40 values starting from e.g. weekly mean values obtained by passive samplers. Data presented by Krupa et al. (2003) show possibilities to reconstruct the frequency distribution for hourly ozone values from 14 day PASSAM data by special algorithms. Such a method would make it possible to calculate threshold values (AOT etc.). This kind of evaluation will be considered as a next step in monitoring ozone by diffusive sampling, besides further evaluations with respect to QA/QC. In Europe, a similar approach is currently being followed by Ferretti et al. (2001).

It is widely recognised, however, that plant response is actually more closely related to the internal ozone dose i.e. the ozone taken into the plant through the stomata, which in turn depends on a variety of ecological factors. Recently, considerable progresses has been done to estimate so-called AOT40 Level II values or other exposure indices that may provide more reliable estimates of the actual risk due to ozone (e.g. Grünhage et al., 2001). The AOT40 Level II value is an expression of exposure that incorporates the factors modifying the response of plants to ozone, thus providing an estimation of the actual ozone uptake by plants (Emberson et al., 1998; Emberson et al., 2000; Simpson et al., 2000). It is questionable whether such more detailed exposure values can be derived from PASSAM, but it is relevant to keep up with those developments when trying to derive relationships between ozone exposure and ozone injury symptoms. Other types of approaches were recently suggested for risk assessment oriented assessment, i.e. the MPOC approach (Grünhage et al., 2001).

4 Ground vegetation species composition

4.1 Introduction

Background

Monitoring of forest health was initiated in several countries in early 1980s due to increased interest in man-induced effects on forest ecosystems. In Europe, the international program of forest health monitoring, ICP-Forests, which started in 1994, has implemented monitoring of ground vegetation in a large number of plots and so far, more than 70% of the 862 plots (Level II) included in this program have been assessed for ground vegetation (De Vries et al., 2001). In a long-term monitoring program, ground vegetation is an essential element to be assessed since it constitutes the largest component of plant biodiversity in forest ecosystems, and often represents a driving force in forest dynamics. It influences not only water or mineral cycling, but also forest regeneration success through competition for light at early successional stages.

Ground vegetation is also a powerful bio-indicator of several environmental factors. It often gives integrated information about soil fertility, acidity, nitrogen status, water availability, or climatic conditions (Wittig et al., 1985; Ellenberg et al., 1992; Thimonier et al., 1992; Thimonier et al., 1994; Aarrestad and Aamlid, 1999; Van Dobben et al., 1999; Cluzeau et al., 2001). This aspect can be used to gain information about changes in site characteristics caused by stresses such as air pollution. However, the response of plant species to chronic air pollution, or climatic changes is usually small, gradual, and difficult to separate from natural succession. Therefore, it is important to examine closely the data collected so far by the ICP-Forest "ground vegetation" network in order to ascertain their potential to detect such changes.

This analysis, which uses the first round of ground vegetation assessment, offers the opportunity to examine "vegetation-environment" relationships at the European scale. It also brings new information about the main environmental factors controlling the distribution of forest plant species, and allows to quantify the impact of atmospheric deposition on forest ground vegetation, after taking into account bioclimatic, edaphic, and stand characteristics. This analysis was also performed to detect the main limitations in the current sampling strategies, in order to improve future interpretation of temporal trends that could possibly be observed in the next vegetation assessments (i.e. within 5 to 10 years).

Contents of this chapter

In chapter 4.2, we describe the approaches that were used to evaluate the ground vegetation data, focusing on both methodological biases and relationships between ground vegetation species composition and environmental factors, whereas results of these approaches are described in Section 4.3. Finally, a discussion of the results and conclusions is presented in Section 4.4.

4.2 Sites and methods

This analysis is based on 3870 ground vegetation surveys performed between 1994 and 2000, in 23 European countries (Table 4.1). Data were supplied by the Forest Intensive Monitoring Co-ordinating Institute (FIMCI), where the whole Intensive Monitoring database is being managed. During this time period, a total of 671 permanent plots were surveyed at least once for ground vegetation. For each country, Table 4.1 shows the surveys distribution in space and time, and presents also some of the methodological differences between participating countries.

Table 4.1 A total of 671 plots were analysed which could be subdivided into spatially and temporally distinct subplots. For the spatially distinct subplots, samples were collected from different areas that were either fenced or not. Some subplots were recorded more than once per year, or for several years (the number of subplots are shown for a given treatment only, fenced or unfenced). The main methods used to estimate cover are also presented (when the information was supplied) together with the number of teams that performed the surveys.

Country		Spatially distinct				Temporally distinct		Methods			
Name	abbrv.	Plots	Subplots per plot	Fenced	Area (m^2)	Range of sampled years	Intra-annual samples	Braun Blanquet	% Scale	Others	Number of Teams
Austria	AU	20	1 - 10	-	4 - 400	1995-96	1			20	1
Belgium	BL	20	1	11	200 - 400	1996-98	1 - 3	8		12	1
Czech Republic	CZ	10	1	-	2500	1995-98	1	10			1
Denmark	DK	14	1	-	9 - 11	1998-99	1		14		1
Estonia	EE	7	3 - 4	-	24 - 25	1997	1		7		1
Finland	SF	31	1	-	16 - 32	1998-99	1		31		2
France	FR	99	3 - 4	99	100	1994-97	1 - 6	99			12
Germany	DL	62	1 - 6	43	20 - 1600	1996-99	1 - 2	12	43	7	1
Greece	EL	4	1	4	100	1996	2	4			1
Ireland	IR	9	1	-	25	1997	1		9		1
Italy	IT	19	1	19	900 - 1200	1999	1	19			16
Latvia	LV	2	1	-	75	1999	1		2		1
Luxemburg	LX	2	4	2	100	1995	3	2			not given
Norway	NO	12	1	-	50	1998-99	1		12		1
Poland	PL	148	1	-	400	1998	1		148		7
Portugal	PO	9	1 - 2	-	50 - 2500	1998	1	9			1
Romenia	RO	7	1	-	5000	2000	1			7	not given
Slovak Republic	SR	7	1	2	2500 - 5500	1999	1 - 2			7	1
Spain	ES	52	1	-	2500	1999	1		52		3
Sweden	SW	98	1	-	900	2000	1		98		1
Switzerland	CH	16	1	-	516 - 1016	1995-99	1 - 2		16		1
The Netherlands	NL	14	1	-	300	1996-00	1		14		1
United Kingdom	UK	10	1 - 10	-	100	1998	1	9			1

4.2.1 Plot description and environmental parameters

For each plot, stand characteristics and basic environmental parameters were extracted from the FIMCI database (Table 4.2). 30-year averages of monthly temperature and precipitation were extracted from the Cramer and Leemans database (Leemans and Cramer, 1991; Cramer and Leemans, 2001). Total nitrogen and sulphur depositions were estimated for the 1990-2000 period using the EMEP Eulerian model (EMEP, 2001). Such averages, made over long time series, were considered more robust and representative than measurements in a single year, since ground vegetation generally responds in an integrative manner to environmental factors. Although allowing less accurate estimates, deposition and meteorological data on the basis of grid data were preferred over actual measurements at the sample sites (as was done by De Vries et al., 2002), since these allow the analysis of a larger number of plots. For example, throughfall, bulk and stem flow measurements were only available for 52 to 400 plots.

A large part of the studied plots were located in the atlantic (22%) or subatlantic (38%) climatic regions. Plots were mainly constituted of *Pinus sylvestris* (31%), *Picea abies* (26%), *Fagus sylvatica* (13%), *Quercus robur* (7%), or *Q. petraea* (5%) stands. The remaining 18% of the plots were constituted by more than 33 different dominant species. Globally, the surveyed plots were mostly coniferous (68%). This proportion is consistent with the one observed for the forested area of the countries involved in the program (65% coniferous) (FAO, 2000). The forest stands at the have a dominant age of 41 to 80 years (61% of the plots).

Table 4.2 Qualitative and quantitative environmental variables extracted from the FIMCI database with the number of plots (N) for which these variables were available. Note that for measured deposition (NH_4; NO_3; SO_4; K; Na; Ca; Mg; Cl; H in throughfall, bulk, and stem flow) the number of plots for which these data were available is low. In bold are the variables used in the present study.

Qualitative variables (N=671)	Quantitative variables	N	Quantitative variables	N
Dominant species	*Stand characteristics*		*Climate*	
Beech	Stand Density	671	Altitude	671
Oak	Yield	627	Precipitation	664
Pine	Basal area	303	Temperature	664
Spruce	Volume	29	Sum DD <0	668
Others	Dominant height	217	Sum DD >0	664
	Stand Density Index	303		
			Deposition (EMEP)	
Stand age	*Foliar variables*		Nitrogen_Mean	662
Irregular	Defoliation	670	Sulfur_Mean	662
Mature (>60 yrs old)	N	661	Nitrogen_StdDev	662
Young (<60 yrs old)	S	661	Sulfur_StdDev	662
	P	661	Quantity	420
Climatic zones	K	661		
Atlantic	Ca	661	*Measured deposition*	
Boreal	Mg	661	Throughfall	292
Continental			Bulk	400
Mediterranean	*Soil (0-10 cm)*		Stem flow	52
Mountainous	pH $CaCl_2$	596		
Subatlantic	Carbon	596		
	Nitrogen	595		
	C/N	595		
	pH H_2O	310		
	Ca	65		
	Mg	65		
	K	69		

4.2.2 Survey methods

The nomenclature used is based on the Pandora vascular plant checklist (see www.rbge.org.uk) which is considered as an up-to-date species list of the European flora. When species were not found in that list, country experts attributed relevant codes to these "new" species. Prior to further analyses, a careful examination of these "new names" was performed in order to avoid duplicates between countries, as well as synonymy and typing errors. For mosses and lichens the nomenclature used follows (Frey et al., 1995).

Species assessments were performed layer by layer, which were defined either on the basis of taxonomic groups (mosses, lichens, or vascular plants), morphology (e.g. herbs, ligneous shrubs), height, or a combination of these criteria. This induces substantial variation in herb and shrub heights from one country to another. Figure 4.1 shows the amplitude of this variation, which for herbs ranges from 0.15 to 1.8 m, and from 0.5 to 10 m for shrubs. In some plots the moss, lichen, or even tree layers have not been recorded (Table 4.3). This situation generated missing values that restricted analytical possibilities.

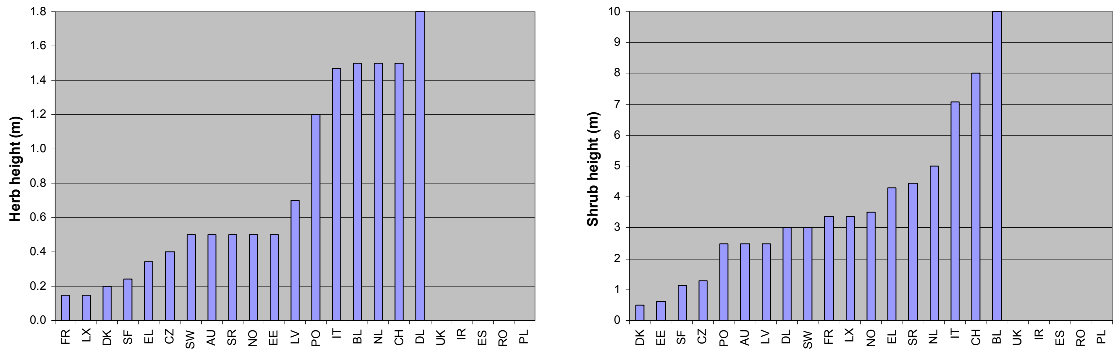


Figure 4.1 *Maximum height of herb (left) and shrub (right) layers grouped by country. Data were missing for 5 countries.*

Table 4.3 *Countries for which specific vegetation layers were not recorded in the vegetation survey.*

Trees	Mosses + Lichens	Lichens only
Denmark	Austria	Belgium
Estonia	Denmark	Switzerland
Finland	France	Czech Republic
Sweden	Luxemburg	Greece
United Kingdom	Poland	Latvia
	Romenia	Slovak Republic
	United Kingdom	

Species abundance was visually estimated in the field. The main method used, in two third of the plots, estimates species covers on a percent scale of the ground surface (Table 4.1). The second commonly used method is the one of Braun-Blanquet, where cover is estimated in class intervals. This method was used in 26% of the plots. Other modifications or combinations of the percent scale and class interval methods were also used (Table 4.1). Table 4.1 also shows the number of teams that performed ground vegetation surveys in each country.

4.2.3 Subplot identification

For most plots, surveys were performed within subplots, that varied in terms of number, fencing, or sampling area (Table 4.1). However, as data submitted to FIMCI did not include a formal identification of subplot, it was impossible, for any given subplot, to follow its ground vegetation dynamics whenever surveys were replicated in time. This evaluation was therefore restricted to the plot level.

Based on the survey list, the number of subplots per plot and their area were calculated. Figure 4.2 shows, for the unfenced subplots, their area distribution. It ranged from 9 to 5500 m^2, with particularly high frequencies for 100, 400, 900 and 2500 m^2. However, no information was available on the sampling extent (total area covered by the spatial subplots distribution). This could be much greater than the straight sum of subplot areas, when subplots are scattered.

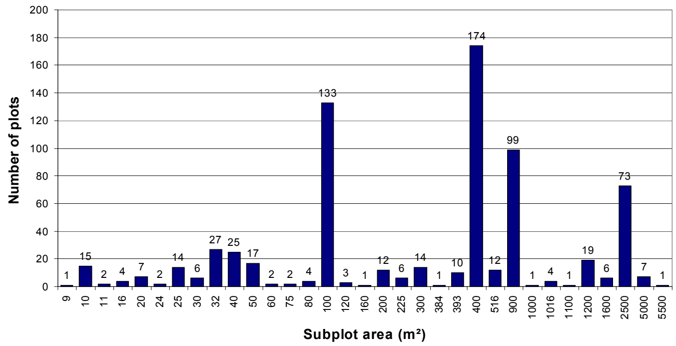


Figure 4.2. Distribution of subplot areas per plot for the unfenced treatment.

4.2.4 Species diversity indices

In ecological studies, 3 main types of diversity indices are generally used (Whittaker, 1972). The intra-habitat, or site diversity, which is called 'α diversity'. It represents the species richness at a local level. In this study, the number of species per plot was used as an α diversity index. The second type of diversity indices is called 'γ diversity'. It represents the species richness at a regional, or landscape level. In this study, the country level was used to calculate 'γ diversity' indices, which therefore represented the number of species found within a country. Finally, the last diversity index concerns the rate of species changes from one habitat to another. It is the 'β diversity'. It represents the length of the ecological gradient observed within a study. It could be calculated using detrended CCA (Jongman et al., 1995) or measured using indices such as the Sørensen similarity index (Legendre and Legendre, 1998). Only the first 2 types of indices (α and γ) are examined in more details in section 4.3.3.1.

4.2.5 Ellenberg indicator values

Based on their large floristic expertise, Ellenberg and co-workers (Ellenberg et al., 1992) developed a system of indicator values, classifying plant species (following a 9-point scale) according to their 'optimum' prevalence for several environmental factors. Using these indices, one can obtain an estimate of site environmental conditions based only on its floristic description. These indicator values were established for 7 environmental characteristics, namely temperature (T), continentality (K), light regime (L), soil acidity (R), soil nitrogen content (N), humidity (F), and salinity (S). These indices were developed for Central Europe, and their use outside this region may causes several problems (Schaffers and Sykora, 2000; Wamelink et al., 2002). For example, within a given region, the distribution of indicator values may be uneven and site scores may then be affected by this frequency distribution (Schaffers and Sykora, 2000). Also, Ellenberg's values for acidity and moisture were shown to vary widely around regression lines and to be biased toward experts expectations for given phytosociological classes (Ertsen et al., 1998; Schaffers and Sykora, 2000; Wamelink et al., 2002). To cope with these inaccuracies, several countries developed their own indices, or adapted the ones of Ellenberg (Donita et al., 1977; Landolt, 1977; Diekmann, 1995; Ertsen et al., 1998; Hill et al., 1999; Gégout et al., 2001; Wamelink et al., 2002). Nevertheless, until the development of a system based on vegetation surveys associated with actual measurements of abiotic factors, as suggested by Wamelink et al. (2002), the Ellenberg's indicator values still give sufficient insight into environmental conditions of a plot.

As the use of indicator values may represents an interesting tool to be used in several studies designed to evaluate ‘global changes’, we decided to use the Level II dataset to evaluate the strength of the relationship between Ellenberg’s indicator values and several environmental variables. This study, covering the European scale, offers therefore an opportunity to examine these relationships, for an application domain well exceeding the original one set by Ellenberg and co-workers.

4.2.6 Relationships with environmental parameters

Descriptive statistics were initially performed on the whole data set, and followed by multifactorial analyses. Ordinary least square and maximum likelihood methods were used. A particular effort have been made to keep the largest possible number of plots in the analyses. Therefore, some partly missing environmental factors were not included in order to maintain large sample sizes. The environmental factors available for the analyses are shown in Table 4.2. For the ordinary least square analyses, homoscedasticity of the residuals was visually checked and variables transformed when required. For counts data the variance-stabilising transformation used was a logarithmic (Ln) one. Impacts of environmental factors and methodological aspects (e.g. fencing, area) were evaluated on species richness variation (number of species per plot). Relationships between Ellenberg’s indicator values (see 4.2.5) and the assessed factors were also examined. For each indicator, averages per plot were made only when at least 5 species of known indicator values were present. Otherwise, plots were assigned missing values. Multivariate approaches were used to determined environmental effects on ground vegetation over Europe. Indirect and direct methods of gradient analysis were used. Analyses were carried out with SAS (SAS, 1990), except for canonical correspondence analyses (CCA) that were performed with CANOCO (Ter Braak and Smilauer, 1998).

Multifactorial Gaussian logit models

The relationship between a single species (presence/absence) and environmental variables was analysed using multifactorial Gaussian logit models which is a regression method based on maximum likelihood (Jongman et al., 1995). The model used had the following form :

$$P = \left(\frac{\exp\left(b_0 + \sum (b_{1,i} x_i + b_{2,i} x_i^{\,2}) + \sum b_{3,j} x_j \right)}{1 + \exp\left(b_0 + \sum (b_{1,i} x_i + b_{2,i} x_i^{\,2}) + \sum b_{3,j} x_j \right)} \right) \qquad (4.1)$$

where P is the expected probability of occurrence of a given species; x_i are quantitative explanatory variables; x_j are qualitative explanatory variables (dummies); and b_0, $b_{1,i}$, $b_{2,i}$, $b_{3,j}$ are parameter estimates.

Quantitative variables were standardised to zero mean and unit variance in order to ease comparison between different predictors.

Qualitative variables were restricted to 3 binary categories : stand type, age, and climatic zone. Stand type referred to: 1 for plots located within coniferous stands, and 0 for others. Age categories were: 1 for young stands (≤ 60 years olds), and 0 for irregular or older stands. Climatic zones were: 1 for plots located in the subatlantic zone (which is the main category in the data set) and 0 for others.

Model fitting was done using a logistic procedure associated with a stepwise selection at an α level of 0.05 (SAS, 1990). This α level is probably too stringent (Hosmer and Lemeshow, 2000), sometimes eliminating biologically meaningful variables, but it reflects the rather qualitative purpose of this study. Goodness-of-fit was estimated using the Hosmer-Lemeshow chi-square test (SAS, 1990; Pulkstenis and Robinson, 2002). The Nagelkerke generalised coefficients of determination (pseudo R^2) were also calculated (SAS, 1990). More details can be found in Renaud and Dupouey (2002). As a practical rule, Hosmer and Lemeshow (Hosmer and Lemeshow, 2000) suggested to restrict the number of parameters in the model to less than one tenth of the events (i.e. species presences, in our case). For that reason, only species that were present in at least 50 plots were retained for analysis. For each retained species (75), analyses were performed to estimate the impact of environmental predictors.

Ordinations

A preliminary correspondence analysis (CA) was made on a presence/absence basis for the shrub and herb layers merged together before analysis. This CA included all available plots (671). Tree, moss and lichen layers were not included, since they were not recorded in several plots. For each plot, only the last surveyed year was analysed, and all treatments such as fencing, surveyed periods, or areas were pooled together.

Following this preliminary analysis, many Spanish (including Canaries Islands) and all Portuguese plots were removed from further ordinations, since they showed drastic floristic differences compared to the other plots. CA and CCA based on species presence/absence were then performed on a selection of 602 plots. CCA included all predictors given in Table 4.2 (typed in bold). From these 26 variables, 12 were quantitative and standardized to zero mean and unit variance before CCA. The remaining qualitative variables were coded as 0 or 1 and these were also standardized as this is an automatic procedure in CANOCO.

In order to further reduce heterogeneity, rare species were either removed (when present in less than 5 plots) or down-weighted according to their frequencies. In such a case, species with frequencies lower than 20% of the most common ones received a weight inversely proportional to their frequencies (Ter Braak and Smilauer, 1998). Therefore, the resulting dataset included 482 species and 602 plots.

Spatial variability

In order to remove some spatial and methodological components of the ecological variability present in the CCA, covariables were used. As done in a previous study, countries were used as covariables to take into account methodological aspects (De Vries et al., 2002). In doing so, we expected to remove part of the variability associated with countries such as surveyed area, period, fencing, as well as the number of subplots, or sampling frequencies per plot. To take into account the spatial structure *per se*, a second-order linear equation was retained based on x-y geographical co-ordinates (Borcard et al., 1992). Using these two kinds of covariables, we expected to partial out the “spatial-methodological” variability from the remaining ecological variability.

In analyses of variance involving species richness, or Ellenberg indicator values as dependant variables, a similar approach was used. Country were not used as covariables, but spatial variability (x-y geographical co-ordinates) was taken into account using a cubic polynomial equation as covariable. Of course, this approach does not take into account differences in sylvicultural methods that could certainly occur from one country to another. But, since we were more interested in identifying impacts of survey methods (that even varied some times within countries) on species richness, we decided rather to include methodological aspects directly in the analysis (e.g. sampling area, number of visits per year, or number of subplot per plot). For Ellenberg indicator values, we assumed them to be less sensitive to methodological aspects and therefore restricted covariables to the geographical co-ordinates.

4.3 Results and discussion

4.3.1 Methodological bias

Many factors affect species richness. Part of this variability is associated with environmental factors, such as light, temperature, water regimes or even the degree of habitat patchiness at the landscape, or regional level (Palmer, 1994a, b; Bascompte and Rodriguez, 2001). Habitat productivity, and species competitiveness also contribute to species richness (Grime, 1973a; Waide et al., 1999; Mittelbach et al., 2001). However, methodological bias are also known to cause large variations in species diversity, often preventing direct comparisons between studies (Shmida, 1984). Therefore, methodological aspects deserves a special attention for the ground vegetation monitoring network.

Sampling area, frequency, or the season at which it occurs are of paramount importance when species diversity is estimated (Palmer, 1994a; Condit et al., 1996; Ney-Nifle and Mangel, 1999; Crawley and Harral, 2001; Madotz et al., 2002). For example, the shape of the sampling area, or the number of sampling units, through their influence on the sampling extent were shown to influence species richness (Palmer, 1994a; Legendre and Legendre, 1998). Studies with contiguous subplots are thus examining a narrower extent than the ones with sampling units distant from one another. In case of species presenting uneven distribution (e.g. spatial aggregation), or different levels of competition (Grime, 1973b), the sampling extent influences directly the number of species recorded (Palmer, 1994a; He and Legendre, 2002). Thus it is clear that sampling methodologies must be carefully analysed. In this context, impacts of sampling area, frequency and number of subplots per plot were evaluated in terms of species richness (α diversity).

Sampling area, frequency and number of subplots

As species richness has a skewed distribution, analyses of covariance were performed on transformed values (natural logarithm). Analyses were made for each layer separately. Sampling areas were divided in 5 classes to obtain an approximately even distribution of plots per class (0-50 m^2; 51-200 m^2; 201-600 m^2; 601-1000 m^2; and more than 1000 m^2). The number of subplots per plot was grouped into two classes (only 1, or more than 1 subplot per plot), as well as the number of times a plot was sampled per year (visited once, or more than once). As methodological variations had a spatial pattern, linked to countries, we partialled out spatial autocorrelation using a polynomial function of the plots co-ordinates (Borcard et al., 1992). Results are presented in Table 4.4.

The number of species is for the far larger part determined by the spatial variables, as apparent from a comparison of the sums of squares in Table 4.4. The methodological variation plays a minor, although significant, role in most cases. Only for the mosses the effect of the spatial variation is in the same order of magnitude as the effect of plot area, but for these organisms there is no influence of the sampling frequency (p=0.36).

*Table 4.4 Influences of total plot area, number of visits per year and number of subplots on layers species richness (Ln transformed). (DF: degree of freedom; Direction gives the way species richness changes with an increase in a given quantitative variable (it is similar for all 4 dependant variables, excepted for the mosses where an increase in area tends to reduce the number of species; R² is the determination coefficient for the whole model; R²spatial is for the spatial part only.) Values are Type III sums of squares (i.e. the decrease in regression sum of squares on dropping a given term from the model), followed by symbols showing the level of significance ((*): p< 0.10; *: p< 0.05; **: p< 0.01; ***: p< 0.001).*

Source	DF	Direction	Sum of Squares			
			Total	Herbs	Shrubs	Mosses
Model	15		101.08***	140.88***	151.86***	115.12***
Area classes	4	+	4.40*	7.74**	8.22**	66.04***
N. visits/yr	1	+	5.38***	7.50***	3.74*	0.30
N. subplots	1	+	0.20	0.00	0.16	6.39***
Spatial variables	9		83.49	107.58	131.00	42.75
		N. plots	671	669	499	296
		R^2	0.27	0.31	0.35	0.54

The relationship between species richness and sampling area is illustrated in Figure 4.3. For the herb layer, a clear increase in species richness is observed with an increase in sampling area. Results were similar for shrubs, although more variable. However, for the mosses and lichens, an opposite trend was observed. This discrepancy might be explained by the limited number of countries in which mosses and lichens were surveyed. As sampling area varied on a country basis, the relationship obtained could therefore be due to the fact that countries using large sampling areas had probably also a limited number of mosses, or lichens species.

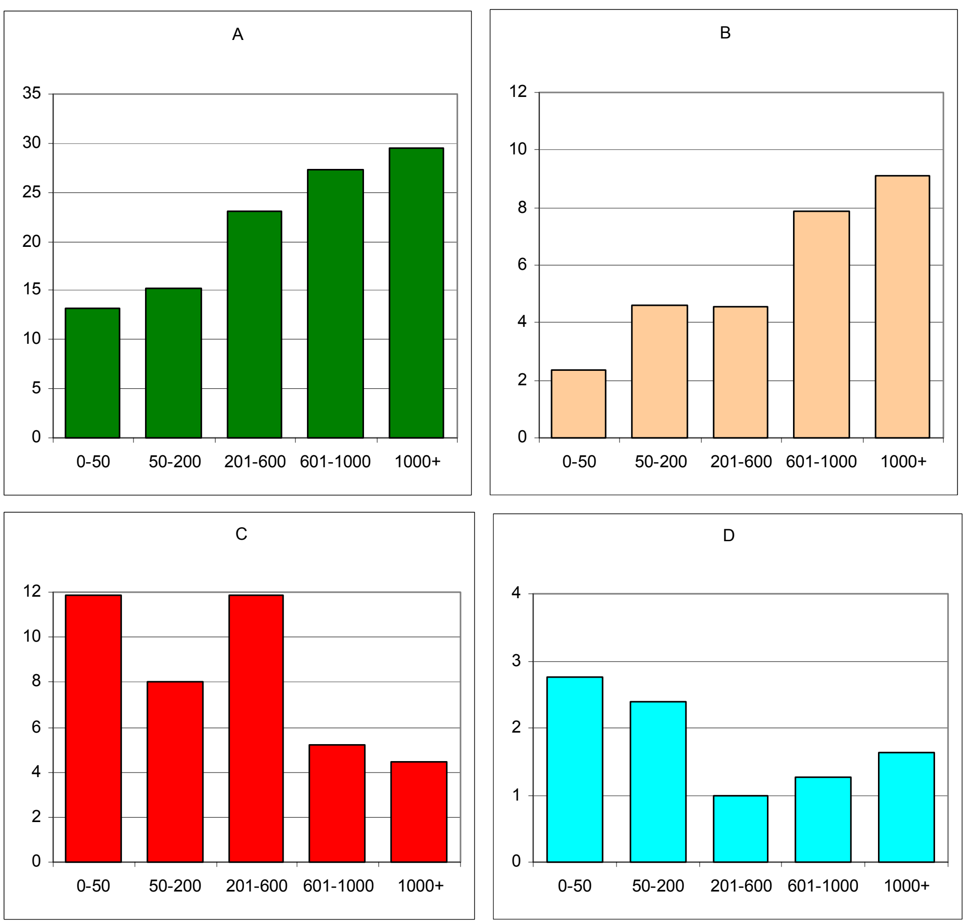


Figure 4.3 Impact of sampling area on species richness of herbs (A), shrub (B), moss (C) and lichen (D) layers.

Nevertheless, using the Arrhenius model (i.e. species richness (S) = cA^z, where Z is the slope of the relationship between the logarithm of S and the logarithm of the sampling area (A)), we obtained slope estimates based on the midpoint transformation of the area classes. We found slopes of 0.19 ± 0.03 for herbs, and 0.28 ± 0.05 for shrubs. For mosses and lichens, estimates were not found significantly different from zero ($p>0.05$). These values are comparable with the ones found in the literature, with Z ranging from 0.1 to 0.5 (Crawley and Harral, 2001). In Finland, Salemaa and co-workers (Salemaa et al., 1999) also obtained results that are in agreement with such values. Thus the increase in herbs and shrubs species that we observed is not surprising, since it supports the species-area relationship, one of the most robust generalisation in ecology, that is largely driven by species abundance and spatial distribution (He and Legendre, 2002).

Fencing

Fencing may have a strong effect on species richness within a relatively short time period (<5 years) (Grime, 1973b). For example Gough and Grace (Gough and Grace, 1998, 1999) showed that fencing led to a decrease in species diversity within a 2 years period after fence installation. This reduction of diversity was attributed to competitive exclusion. Similarly, Von Fischer (1993) also observed in the vicinity of Frankfurt, a rapid decrease in species richness, one year only after fence installation.

In the Level II network, 170 plots had fenced subplots. In sites where both fenced and unfenced subplots were present, there was an opportunity to evaluate the effect of grazing on species richness. Because the year of fence installation is not recorded in the database, we assumed that it was just before the first observations were made.

For Level II network, 5 countries only (Belgium, France, Germany, Italy, and Luxembourg) had paired treatments (fenced/unfenced subplots). This corresponds to 153 plots, among which some had unbalanced observations in terms of sampling area, periods, or frequencies. Therefore, the impact of fencing was evaluated on only 138 plots, for which paired t-test were performed. Results suggest that fencing tended to reduced species diversity ($p<0.05$) (Figure 4.4). This difference tended to be very large (>20 species) in 2 particularly rich plots (i.e. the French plot #34, and the Italian plot #9). Nevertheless, many plots remained within a ± 5 species interval between fenced and unfenced subplots. This suggests that while fencing tended to reduce the number of species per plot, it did not produced an even response over all plots. It seems that this reduction was more pronounced in plots with a high species diversity (Figure 4.4). Moreover, no clear time trends can be observed from 48 plots, that had been surveyed more than once. In Figure 4.4, red dots represent plots for which 3 years were spent between their first and last surveys. If fences were installed at, or just “before” the first surveys, we can then assume that fencing lasted since at least 3 years in these plots that showed no systematic decrease in species richness.

This absence of time trends suggests two things: 1) that the “fencing effect” might really occur, but in specific sites only, possibly the rich ones, or those that we may speculate under a strong grazing pressure; and 2) that a methodological bias could also be involved in such results. As many of the fenced subplots showing a decrease in richness are from France, we studied the design of these subplots (Figure 4.5). It shows that the unfenced subplots were systematically distributed outside the fenced area. Therefore unfenced subplots were constituting a sampling unit of a larger extent than the one of the fenced subplots. This larger extent, even though the total area surveyed was exactly the same, could have favoured a greater species richness, especially in heterogeneous stands such as the French plot number 34 (DOU65) (Dobremez et al., 1997) for example.

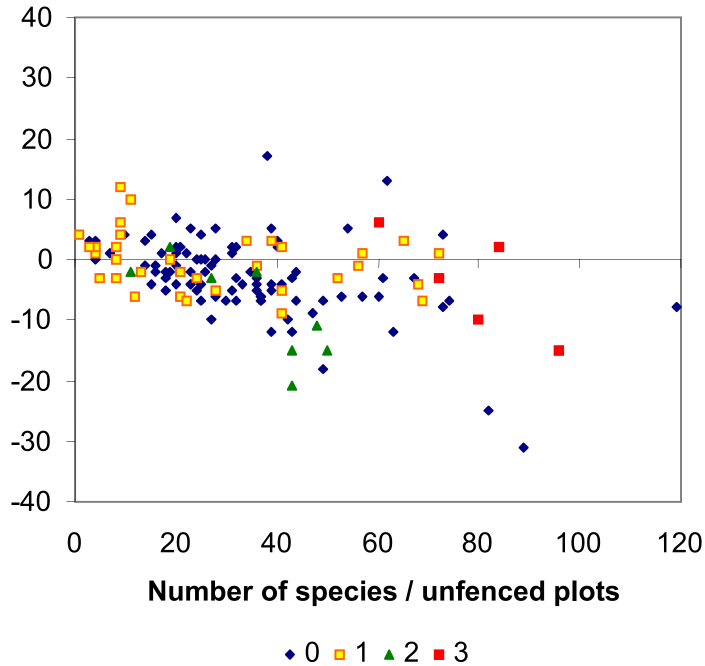


Figure 4.4 *Impact of fencing on species richness, showing the difference in number of species between the fenced and unfenced subplots presented against the number of species in the unfenced subplots. The colour of the symbols represents, for a given plot, the number of years between the last and the first survey (Blue = surveyed only once; yellow = 1yr; green =2 yrs; red = 3 yrs). The hazy zone covers an interval of ± 5 species.*

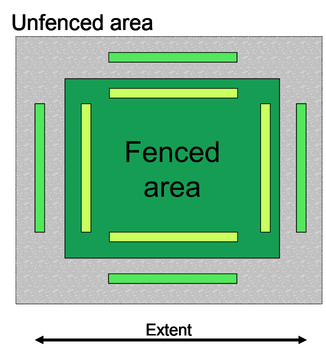


Figure 4.5 *Schematic representation of the French layout. The 4 rectangles located inside the fenced area and the 4 others located outside are the surveyed subplots.*

The fact that we do not have surveys prior to fencing makes the interpretation of the results presented in Figure 4.4 somewhat uncertain. The lower species richness we observed might be an artefact due to the larger extent of the unfenced plot (at least in France), but it might also be due to pre-existent spatial differences within plots, or to the impact of fencing *per se*.

Observer effects and temporal drift

Another important bias that has not been discussed yet concerns the observer errors (Kirby et al., 1986; Leps and Hadincova, 1992; Dupouey et al., 1998). Kirby and co-workers (Kirby et al., 1986) have shown that with only one pass, the number of species found by an observer alone, in a set of 6x200m² quadrats, was only 30 to 65% of the total number of species recorded by several observers. This result confirms that even for experienced botanists, it is nearly impossible to find all species within a given zone (Palmer et al., 2002). The number of species missed usually depends on the surveyed area, its heterogeneity, and the amount of time spent per survey (Kirby et al., 1986; Leps and Hadincova, 1992; Dupouey et al., 1998; Palmer et al., 2002).

With repeated surveys, observers also tend to be familiarised with the local flora. This phenomenon may lead to a spurious increase in species richness over time, due to the fact that if an observer has a check list of his previous visits, he may keep searching species that he may have overlooked otherwise. For example, Dupouey and co-workers (Dupouey et al., 1998) showed for 5 selected plots of the Level II network, that new species were regularly found during the first 3 years of observation. After this "adaptation" period, experienced botanists tend to find much less "new" species (Figure 4.6). Obviously, this potential bias is also confounded with "real" inter-annual variation that has probably occurred.

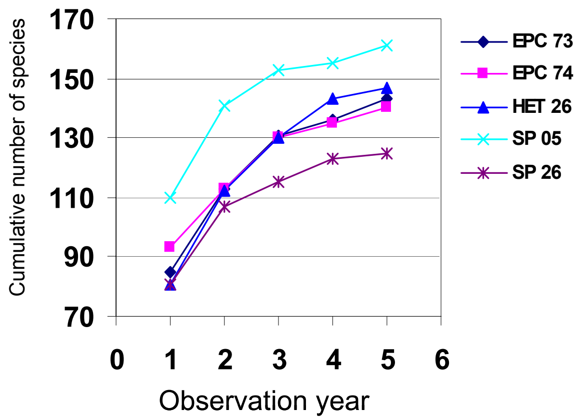


Figure 4.6 *Cumulative number of species per plot as influenced by the frequency of observation (in years) (from Dupouey et al., 1998).*

Figure 4.7 shows the increase in species richness for 78 plots that have been surveyed more than once. Large increases in number of species (>15 species) were observed for lag periods of only 1-4 years. As this is a rather short time interval, it is difficult to imagine that this large increase in species richness could have been caused by chronic environmental changes. We believe that this large difference probably includes observer bias. This idea is supported by the fact that in some countries (e.g. France), quality insurance programs were recently set, and data from these "control surveys" were also included in the ground vegetation database. In such situations, the comparison made between the "recent" richness and the one observed in earlier surveys is probably based on two different and also unequal sets of observers.

Ideally, assessments should therefore be conducted for several years (2-3) and pooled to a single sample in order to cancel out intra- and inter-annual sources of variations. Furthermore it is recommended to identify observers in the database since they constitute an extra source of variation.

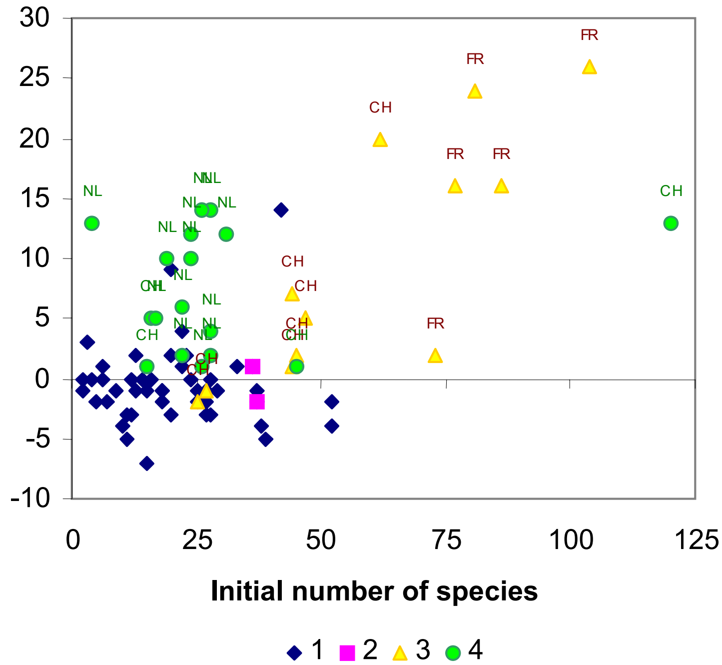


Figure 4.7 Inter-annual differences in number of species per plot for 78 plots that had the same area and number of visits for each year. The number of years spent between the first and the last survey are indicated in the legend below.

4.3.2 Correlation between environmental variables

As it is usually the case in ecology, many of the factors examined showed confounding effects. If we focus on nitrogen (N) deposition, a strong correlation with sulphur (S) deposition was observed (R=0.56). Positive relationships (R=0.41-0.46) were also observed between N deposition and foliar N and S concentrations, as well as with the sum of degree-days below 0°C. Conversely, a negative correlation (R=-0.31) was observed between N deposition and soil pH (0-10 cm).

In order to further examine the relation of qualitative variables with N deposition, an analysis of variance was performed (Table 4.5).

Table 4.5 Analysis of variance showing the relation of several predictors with nitrogen deposition (used as dependant variable). (R^2 = 0.62). Direction gives the way N deposition changes with an increase of a given quantitative variable. F values are followed by symbols showing the level of significance (: p< 0.05; **: p< 0.01; ***: p< 0.001).*

Source	DF	Direction	F value
S deposition	1	+	121.2***
Climatic zone	5		17.8***
Foliar N	1	+	73.7***
Dominant tree species	4		8.2***
Soil C/N	1	+	25.2***
Soil pH $CaCl_2$	1	-	17.0***
Stand density	1	-	8.7**
Precipitation	1	+	4.9*

Results suggest that N deposition differs between climatic zones and dominant tree species. Furthermore, N deposition also varies according to soil C/N, annual precipitation and tree density. Similar results have been obtained by De Vries et al. (2000) who showed that atmospheric deposition was influenced by the geographic region, rainfall, altitude and tree species. They also found that foliar N and S concentrations of coniferous species were related to N and S depositions.

Another confounding effect concerns the relation between stand age and climatic zones. For example, irregular stands were mostly located in the Mountainous and Mediterranean zones, whereas old stands (>100 years old) were rather found in the Mountainous and Subatlantic zones. On the opposite, very young stands (<40 years old) were found mainly in the Atlantic and Mediterranean zones. For example, Mediterranean plots were rather young or irregular, whereas Boreal ones were mostly between 40 and 80 years old. These confounding effects, or co-linearities, have to be kept in mind when interpreting results of such surveys.

4.3.3 Diversity indices and Ellenberg indicators

4.3.3.1 Overview of species diversity at the European scale

In this first round of ground vegetation surveys, a total of 2343 taxa were recorded, of which 2179 were identified at the species level. The richest layer was the herb one, with 1559 species, followed by the shrub (320 species), moss (178 species) and tree (100 species) layers. For 48% of the plots where lichens were found, only 22 species were recorded. Trees and shrubs were mostly identified at the species level (96%), whereas more difficulties were encountered for the identification of the mosses (87%) and lichens (85%), herbs displaying an intermediate value (93%).

Alpha diversity : species richness per plot

The α diversity is influenced by several methodological aspects as shown in section 4.3.1. However, apart from methodological aspects the α diversity has a strong between-country variation. On the basis of the maximum species richness per survey, Switzerland presented the largest α diversity (with 133 species observed in one plot). It is followed by Spain and Italy, with a maximum respectively of 93 and 89 species per plot. The mean α diversity per country is presented in Figure 4.8.

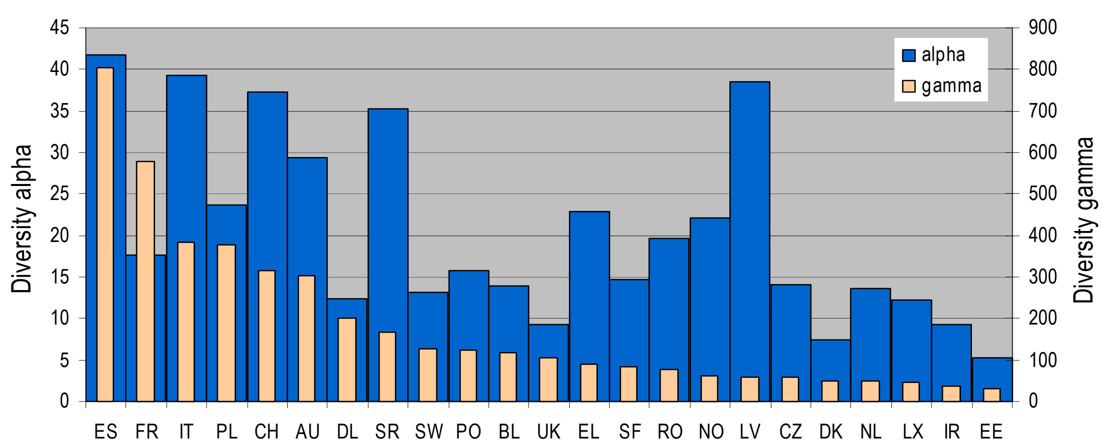


Figure 4.8 Comparison between the α (mean number of species per plot) and γ (overall number of species in all plots) diversities observed at the country level.

The largest values were found for Spain with an average of 42 species/plot, followed by Italy (39), Latvia (39), Switzerland (37), and Slovak Republic (35). These countries with high α diversities, have therefore rich habitats. On the contrary the United-Kingdom (9), Ireland (9), Denmark (7), and Estonia (5), had in average less then 10 species per survey. Detailed values per country are presented in Figure 4.8, together with the γ diversity indices.

A map of species richness per plot is presented in Figure 4.9. It gives an idea of the "species richness gradient" over the Level II network.

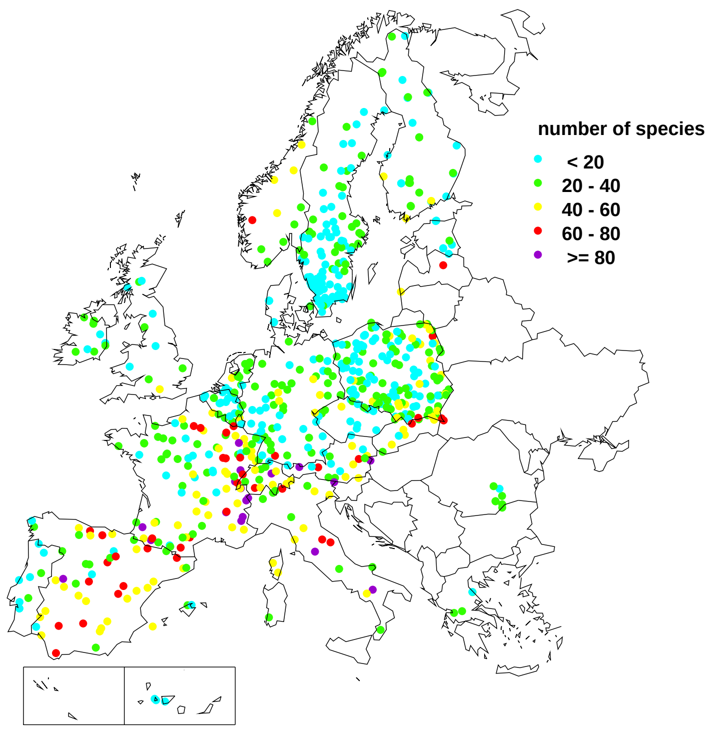


Figure 4.9 Map of the overall number of species per plot (includes fenced and unfenced subplots; not corrected for sampling area). Legend below represents the number of species per classes.

It clearly shows that species rich plots are mainly found in Spain and Italy, in the Alps (including a part of France, Switzerland, and Austria) and extending eastward to include Slovak Republic, and the South-Eastern part of Poland, along the Carpathian arch.

Gamma diversity : species richness per country

Gamma diversity is an estimate of the number of species that live in a region (Da Lage and Métailié, 2000). It depends both on the average α diversity and on the range of habitats found in this region. In Figure 4.8, we compare this measure with the plot richness (mean α diversity) at the national level. Of course, the γ diversity depends on the sampling intensity (number of plots and sampling design). Sampling is not systematic in the Level II network and therefore the ecological extent of the selected sites may vary between countries. Therefore the γ diversity estimates given in Figure 4.8 must be interpreted carefully.

From Figure 4.8, we can observe that Spain has both high α and γ diversities. It has particularly diversified habitats with a γ score of 805 species, for a total of 52 plots surveyed. France has also quite varied habitats (γ=577 species for a total of 99 plots), but with a rather median α diversity.

Environmental factors affecting species richness

At the plot level, we calculated the correlations among species richness observed for the different vegetation layers. Obviously, a strong correlation was observed between the herb and the total plot richness (R=0.95). This is not surprising since the herb layer generally constitutes the largest part (in average 81%) of the overall richness. Quite a good relationship was also observed between the shrub and the total plot richness (R=0.61). However, this relation was much weaker for the mosses (including lichens) (R=0.36). The correlation between the herb and moss layers was also very low (R=0.13).

Analyses of covariance were performed to examine the main factors affecting species richness. For each layer (lichens were pooled with mosses) the impact of the variables listed in Table 4.2 was evaluated. As Figure 4.9 showed a clear spatial structure, a polynomial equation based on plot co-ordinates was used to partial out spatial variance (Borcard et al., 1992). However, this procedure is also removing part of the environmental variability confounded with the geographical position. In the analyses, only the 4 most significant environmental factors were retained. They were: stand type, defined by the dominant tree species, stand density, soil pH (0-10 cm) and foliar calcium (Ca) concentration of the dominant tree species. All these variables showed significant effects on species richness (Table 4.6). Stand type and soil pH affected consistently all vegetation layers. In general, the species richness was higher in Oak stands (34 species in average) as compared to Beech (18 species) or Pine (19 species) stands. Species richness was also found to be positively related to soil pH. Stand density had a negative influence on the overall species richness, although for the shrub and moss layers it did not show a significant effect. Foliar Ca concentrations, was positively associated with the overall species richness, although no significant influence of this variable was detected for the moss layer.

These factors could be considered as the ‘traditional’ ones, affecting ground vegetation. A strong effect of soil pH and dominant tree species on floristic composition is certainly a ‘classical result’. Stand density also, has often shown major impacts on floristic diversity, through its influence on light availability. Interestingly mosses did not responded to this factor. However, the impact of foliar Ca concentrations is less frequently documented. We used this variable as a proxy of site fertility. The more fertile is a site, the larger is its potential to sustain a large number of species. However, even though Ca is a key element of site fertility, it is only one aspect of it and may not necessarily represent the main limiting factor in some sites.

*Table 4.6 Influences of plot characteristics on layers species richness (Ln transformed). (DF: degree of freedom; Direction gives the way species richness changes with an increase in a given quantitative variable; R^2 is the determination coefficient for the whole model; R^2spatial is for the spatial part only.) Values are Type III sums of squares (i.e. the decrease in regression sum of squares on dropping a given term from the model), followed by symbols showing the level of significance ((*): p<0.10; *: p<0.05; **: p<0.01; ***: p< 0.001).*

Source	DF	Direction	Sum of Squares			
			Total	Herbs	Shrubs	Mosses
Model	16		143.46***	159.12***	181.32***	69.03***
Main species	4		12.9***	6.7***	11.4***	6.3***
Stand density	1	-	37.6***	17.8***	2.9(*)	1.7
Soil pH	1	+	27.5***	20.6***	10.4**	16.8***
Foliar Ca	1	+	28.0***	34.0***	13.5***	3.1(*)
Spatial variables	9		76.82	97.86	117.37	37.14
		N. plots	595	594	436	263
		R^2	0.41	0.39	0.46	0.37

Furthermore, foliar Ca concentrations is not independent of soil acidity (R=0.59), and depends largely on the dominant tree species. For example, its optimum concentration is generally found to be higher in broadleaves than in conifers (Bonneau, 1995).

4.3.3.2 Ellenberg’s indicator values

In order to obtain stable plot estimates, mean Ellenberg’s indicator values (N, R, L, F, T and K) were calculated only when at least 5 species per plot had known Ellenberg’s values. Ellenberg indicators signification are as follows: N nitrogen; R acidity (from acid to basic), L light, F moisture, T temperature and K continentality. For a given plot, only herb and shrub layers were analysed and pooled together before calculation. Mean indicator values per plot were then calculated on a presence/absence basis. For the Intensive Monitoring plots, the amplitude of the distribution of the mean indicator values was relatively large for N and R, but more restricted for K and T values (which included only 5 out of the 9 Ellenberg’s classes). Interestingly, at the species level (i.e. for all species covered by this study), the distribution of K and T values varies also in a similar way, with only few species in the extreme classes. For K most species where found to be in classes 2 to 5, whereas for T, they were found mainly in classes 3 to 7. Considering that the Intensive Monitoring plots are covering the total area of validity of the Ellenberg’s values, this restricted distribution of K and T values may appears at first rather surprising. However, it is worth mentioning that among the list of species covered by Ellenberg, only a limited number belong to the extreme classes (less than 10% of the listed species).

Table 4.7 presents the results of a stepwise selection performed in order to relate environmental variables (including x-y co-ordinates) to indicator values.

Table 4.7 Influence of environmental variables on mean Ellenberg indicator values per plot. Values are Type III sums of squares (i.e. the decrease in regression sum of squares on dropping a given term from the model), followed by symbols showing the level of significance (: $p<0.05$; **: $p<0.01$; ***: $p<0.001$; DF: degree of freedom).*

Source	DF	Sum of Squares					
		F	K	L	N	R	T
Model		101.08***	153.87***	211.93***	310.54***	613.98***	117.72***
Climatic zone	5	3.17**	1.51*	4.40**	.	.	4.66***
Main species	4	.	13.37***	49.99***	22.69***	.	6.64***
Defoliation	1	.	0.76*	.	.	.	.
Stand density	1	.	.	7.69***	6.12***	.	.
Temperature	1	2.77***	1.23**	.	2.62*	9.60***	1.11***
Precipitation	1	7.21***	.	1.14*	.	.	1.91***
Altitude	1	.	.	.	4.31**	5.20**	.
N deposition	1	.	1.41***	3.25***	4.15**	.	.
S deposition	1	1.32**	.	.	.	.	.
Soil pH	1	4.52***	.	.	6.53***	94.44***	4.41***
C/N	1	.	.	1.24*	.	10.70***	.
Soil N	1	1.65**	.	2.50***	14.72***	.	.
Soil C	1	0.95*	.	.	14.79***	.	.
Foliar Ca	1	.	1.10**	.	4.34**	57.66***	.
Foliar K	1	.	1.20**	.	.	.	.
Foliar N	1	.	.	.	3.89**	.	.
Foliar P	1	.	.	.	3.87**	.	.
Spatial variables		66.32	123.84	63.90	147.21	301.06	75.67
	N. plots	445	585	546	501	497	391
	R^2	0.59	0.69	0.62	0.54	0.71	0.77

These results show a rather close relationship between indicator values and environment (R^2 = 0.54 to 0.77), which was also shown in previous studies (Thompson et al., 1993; Ertsen et al., 1998; Schaffers and Sykora, 2000; Dzwonko, 2001; Wamelink et al., 2002). In our study this relationship is closest for Ellenberg's T values (Figure 4.10), with a relatively good prediction (R^2 = 0.77) with a few parameters only. This close relationship of indicator values with environmental factors represents certainly an interesting aspect of the ground vegetation surveys, which could be favourably used in a long-term monitoring program, as bio-indicator of climatic changes.

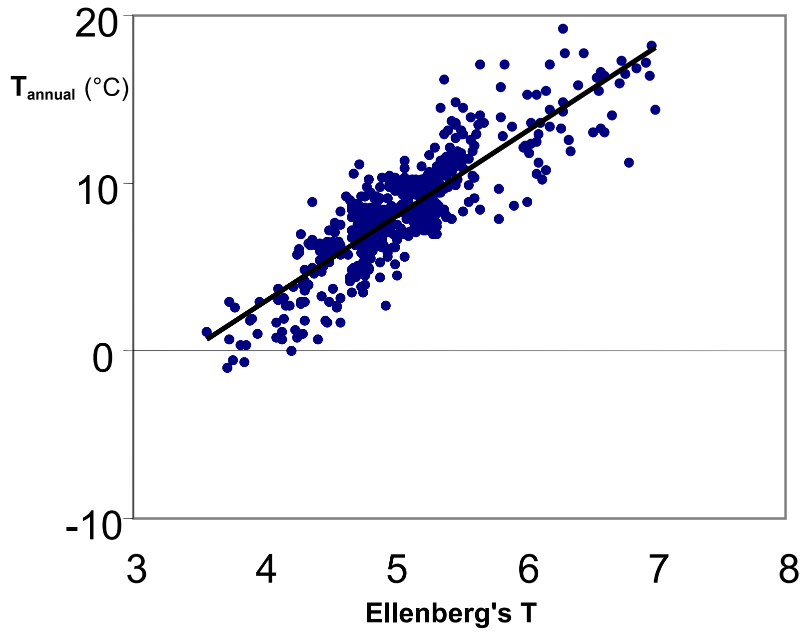


Figure 4.10 Relation between Ellenberg's T values and annual temperature. (R^2=0.54).

For R, Figure 4.11 shows the relationship between soil pH and plot mean R values. This relation appears to be linear for R larger than 4, but does not show any consistency for lower R values. Similar results were obtained by Wamelink et al. (2002).

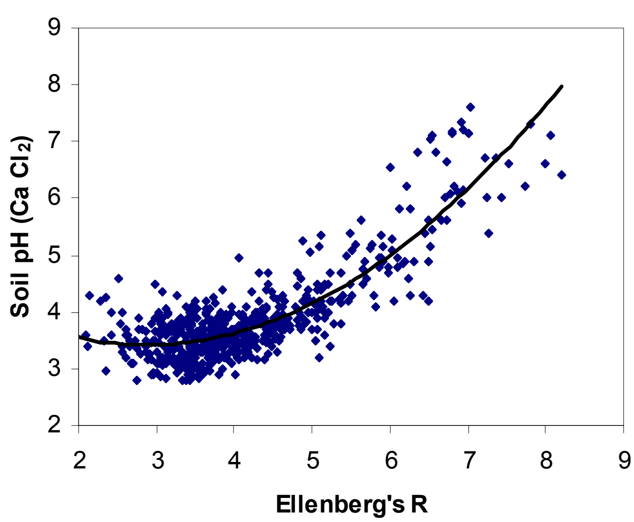


Figure 4.11 Relation between Ellenberg's R values and soil pH ($CaCl_2$) of the upper horizon (0-10 cm). (R^2=0.57)

For N values, several factors seem to be associated with this Ellenberg indicator. Figure 4.12 illustrates the relationship between N values, and N content in leaves, in soils, as well as N deposition. N values are only loosely linked to these factors taken individually, with R^2 ranging from 0.12 to 0.23. This result illustrates the complex nature of this Ellenberg indicator. The global model built in Table 4.7 was also the weakest (R^2 = 0.54) and least parsimonious (15 variables).

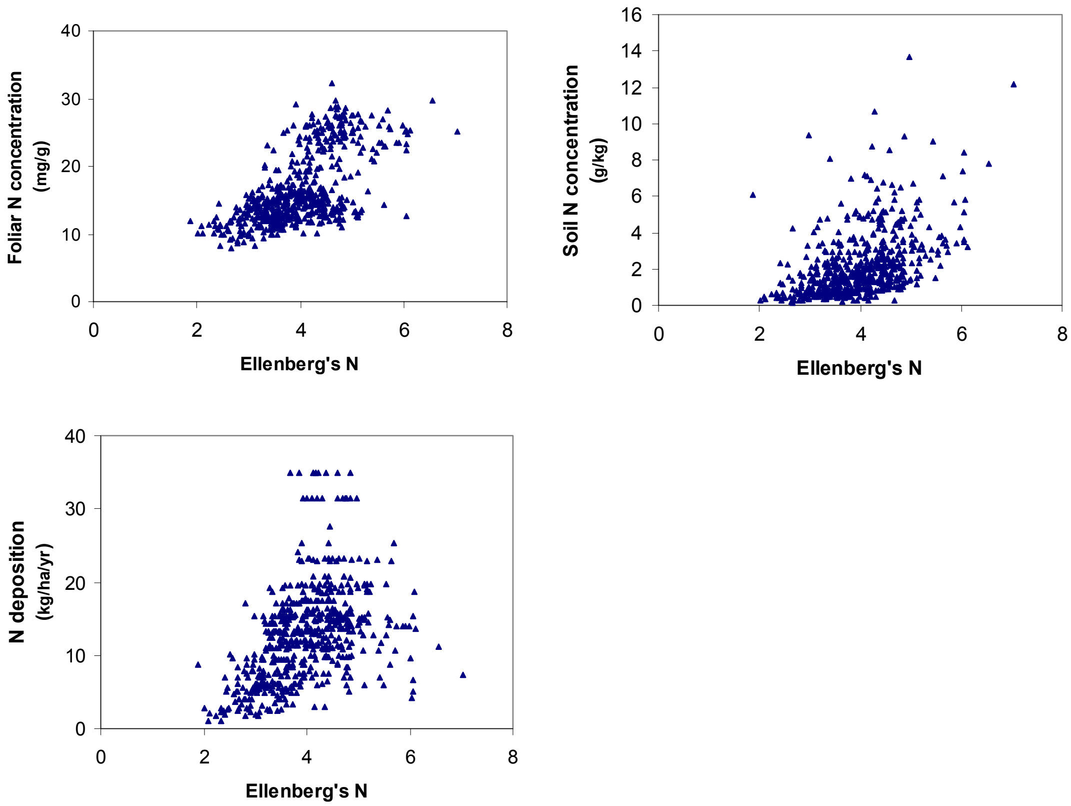


Figure 4.12 Relation between Ellenberg's N values and (a) leaves N concentrations in the dominant tree species (R^2=0.23), (b) soil N concentrations (R^2=0.13), and (c) N deposition (R^2=0.12).

An interesting aspect, shown in Table 4.7 is the fact that several environmental factors contribute significantly to indicator values. This result is in line with the original concept of Ellenberg, who proposed indicator values as global representations of environment conditions. Another interesting aspect concerns a variable used here as a proxy of soil fertility, i.e. foliar Ca concentrations, which showed significant effects on R, N and K values. Schaffers and Sykora (2000) had suggested that R values, normally considered as indicating "acidity", may be more representative of soil Ca availability. This idea is supported by the fact that foliar Ca concentration was the second factor, after soil pH, influencing R values (based on Type 3 F-values).

It is also worth mentioning that species ecological preferences may shift with their geographical locations. This problem (well recognised by Ellenberg) may therefore seriously affect the accuracy of the indicator values. In order to reduce the variance associated with such indices, the use of an index incorporating a notion of relative location compared to the species range could be imagined as a correction factor. The fact that the modelled indicator values had all a significant spatial component support the importance of the geographical location. In summary, the results obtained underline the integrative character of the Ellenberg's indicators, since several environmental and spatial factors contribute significantly to them.

4.3.4 Impact of environmental factors on species composition

The impact of environmental factors on ground vegetation was evaluated using 2 different approaches. An univariate one (individual species) which allowed to find the species optimum for several environmental factors. This approach modelled the probability to find a particular species using a multifactorial Gaussian equation. This approach is based on the maximum likelihood, using a binomial distribution and a logit link function (Renaud and Dupouey, 2002). We also used a multivariate approach based on ordinary least squares. In this case, species were analysed together on a presence/absence basis, using correspondence analyses.

4.3.4.1 Individual species approach

Based on the individual species approach, N or S deposition were found to affect the distribution of several species (Table 4.8). Some of them tended to react differently to N and S depositions. For example, *Cytisus scoparius, Galeopsis tetrahit, Lamiastrum galeobdolon* and *Solidago virgaurea* had an optimum at high N deposition, but tended to have optimum at low S deposition. This could represent an interesting area of investigation, finding out the mechanisms of vegetation changes due to changes in both N and S depositions. However, these results must also be considered with caution since the species distribution was not taken into account in the analysis.

Table 4.8 *Species influenced by nitrogen and sulphur depositions. (- : species having their optimum at low depositions; + : species having their optimum at high depositions) as well as there reported reactions to atmospheric deposition. (Sweden: reaction to a soil N index reported by Diekmann and Falkengren-Grerup 2002; Other experiments including fertilisations : from Van Dobben et al. 1999, and others reported by Diekmann and Falkengren-Grerup 2002; France : temporal changes observed in Alsace and Ardennes after a 10 years interval, from Cluzeau et al. 2001. Symbols : + and - = positively and negatively related to increases of N; -/+= both types of reactions were reported but mainly negatively related; +/-= both types of reactions were reported but mainly positively related; ns = not significant.)*

Species	Reaction to N deposition	Reaction to S deposition	Sweden	Other expt incl. Fert.	France
Anemone nemorosa		-			
Betula pubescens		-			
Convallaria majalis		-	-		-
Hedera helix	-				-/+
Lonicera periclymenum		-			ns
Maianthemum bifolium	-		-/+	-/+	ns
Melampyrum pratense	-		-		-/+
Viola reichenbachiana		-			
Viola riviniana	-		-/+	-	
Acer pseudoplatanus	+				+
Calamagrostis epigejos		+			+
Carex pilulifera	+		-	+	+
Carex sylvatica	+		-	+	+
Cytisus scoparius	+	-			+
Dryopteris filix mas	+		+/-	+	+
Epilobium angustifolium	+		+	+	+
Frangula alnus		+			-
Galeopsis tetrahit	+	-	+/-	+	+
Galium odoratum	+		+/-	+/-	ns
Lamiastrum galeobdolon	+	-	+	+/-	+
Luzula luzuloides	+				+
Milium effusum	+		+/-	+	ns
Rubus idaeus	+		+	+	ns
Sambucus nigra	+				+
Solidago virgaurea	+	-	-	+/-	+
Sorbus aucuparia		+			+
Urtica dioica	+		+	+	+
Vaccinium myrtillus		+	-	-	
Veronica officinalis	+		+/-	-	ns

This represents therefore a confusion source since the absence of a particular species could rather be associated with its distribution potential, instead of reflecting the direct effect of N deposition for example. Table 4.8 also reports results from previous studies showing species responses to N deposition and soil N availability, fertilisation, or temporal changes. For example, a strong negative relation between *Convallaria Majalis* and a soil N availability index was observed in

Sweden (Diekmann and Falkengren-Grerup, 2002). On the opposite, positive relationships were observed for e.g. *Melampyrum pratense Epilobium angustifolium, Rubus idaeus,* and *Urtica dioica*. A general agreement exists between the species that we found to react to N deposition and their changes over time, or under different N regimes (Table 4.8). These species could therefore be considered as good indicator candidates.

Maps of N deposition, as well as the distribution of *Galeopsis tetrahit* are presented in Figure 4.13. It shows that this species is mainly present in Central Europe where the N deposition is elevated, having values above 10 $kg.ha^{-1}.yr^{-1}$. A similar pattern was obtained for *Urtica dioica.* In the area of heavy N deposition (above 30 $kg.ha^{-1}.yr^{-1}$), mainly localised around Belgium and the Netherlands, however, *Urtica dioica* was not present. Therefore it is important to recognise that this uneven spatial distribution of atmospheric depositions is also confounded with bioclimatic factors, or species geographical distribution. In this perspective, species whose natural distribution area is restricted to this zone could be spuriously found to react positively to N deposition for example, even though they are indifferent to this factor. Even though this represents a potential bias source, the results obtained have shown species with a large geographical distribution, such as *Dryopteris filix-mas*, or *Urtica dioica*, to react positively to N deposition.

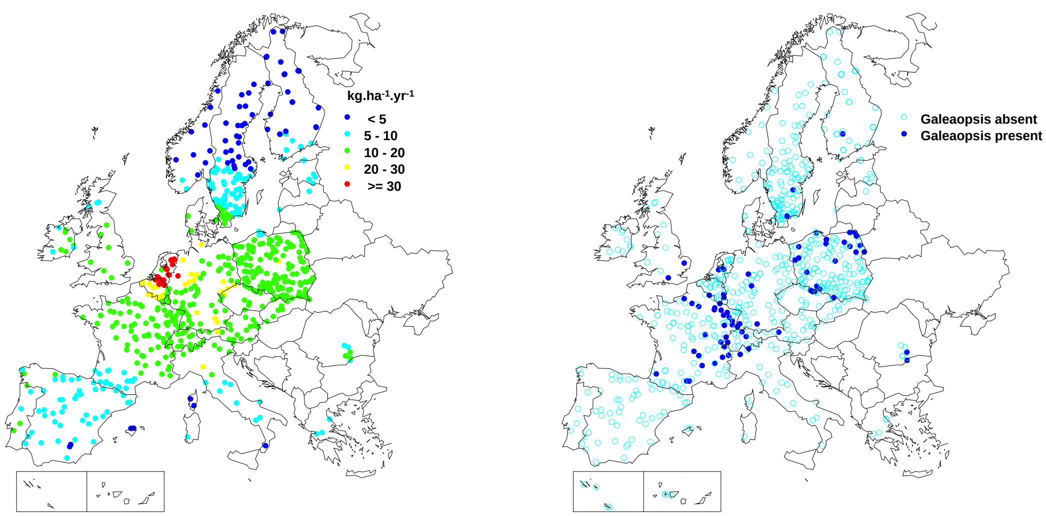


Figure 4.13 Maps of N deposition (in kg/ha/yr) (a) and of Galeopsis tetrahit (b) occurrences.

4.3.4.2 Multivariate approach

In order to identify the main factors influencing species communities at the European scale, CA and CCA were performed. These unimodal methods were preferred over linear ones, since the data set used covers a large ecological gradient. Whereas CA is an indirect method of gradient assessment, CCA is designed to detect the patterns of variation in species that can be explained by environmental variables, directly included in the analysis. It is therefore a direct method of gradient analysis.

Correspondence analyses

Following a preliminary CA performed on all available plots which has shown that Mediterranean plots, mainly the ones of Spain and Portugal, were floristically very different from the others, a second CA was performed after removal of these plots and keeping species that were present in at least 5 plots. This operation left 602 plots and 482 species for the analysis. The fist ordination plane obtained explains 6.4% of the total variance. Plot distribution on the first ordination plane is shown in Figure 4.14.

To have an insight about the factors associated to each axis, a stepwise regression was performed using environmental variables presented in Table 4.2, and mean plot values for the Ellenberg's indices. Results showed a strong association of the first axis (partial R^2=0.81) with the plot acidity (R) scores. The more "acid" plots are located on the left, and "alkaline" ones on the right of the first ordination plane. The second factor associated with axis 1 is the continentality (K) (partial R^2=0.11). Therefore, we can interpret this axis as an acidity one, linked to geographic, or latitudinal positions of the plots. Figure 4.14 shows plots of the Northern countries such as Finland and Sweden, on the left of the first axis, and Mediterranean, or more Southern countries on its right (e.g. Italy, Greece, Romania). For the second axis, results showed an association with the sum of degree days (R^2=0.40) and the Ellenberg's L values (partial R^2=0.30). On this axis, warmer and more "open" (i.e. mean L values ≥ 6) plots tend to be located at the bottom of the graph. Therefore it seems that the gradient observed using this CA are associated with 'classical' broad environmental factors including soil acidity, temperatures and light availability.

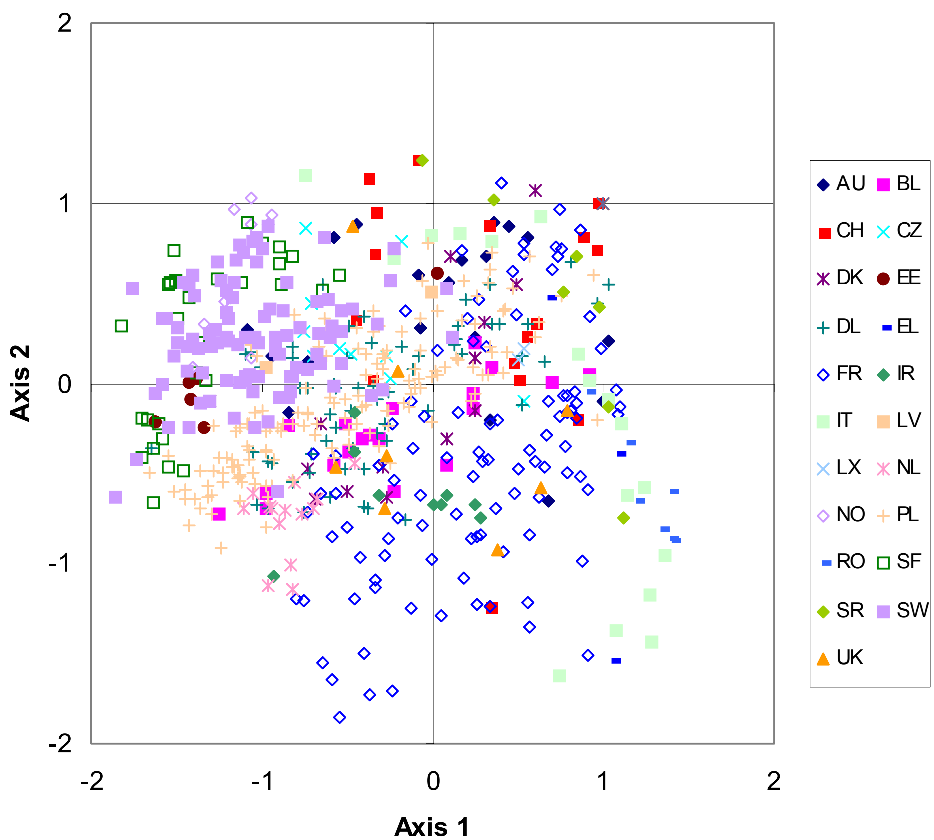


Figure 4.14 Plot distribution in the first ordination plane of a CA performed on 602 plots (plots are identified by country abbreviation. Axis1 explains 3.9% of the variance; Axis 2 2.5%).

Despite a large geographical amplitude, it seems that the first ordination plane is mainly of ecological nature (pH, and light) but also including some bioclimatical structure.

Canonical Correspondence analysis

The use of CCA allows to test directly the impact of environmental variables on species distribution, while taking into account the influence of covariables, such as spatial and methodological factors. In the present analysis, environmental variables were selected in order to keep the largest number of plots. Impact of these variables were tested using a forward selection and 199 Monte Carlo permutations (Ter Braak and Smilauer, 1998). Results showed that 22 variables out of 26 had a significant effect on species composition. However, 45% of the modelled variance could be attributed to only 3 variables: the presence of pine as dominant species, plot altitude, and soil pH. Nitrogen and sulphur depositions explained respectively only 2.7 and 1.4% of the modelled variance. Even though their explanatory power is low, their contribution remains significant with P-values lower than 0.02.

To summarise these results, the variables were grouped into 5 classes (Table 4.9): climate; forest types; soils; sylviculture (which includes stand density, age, and defoliation); and atmospheric deposition. The amount of variability explained by each of these classes were respectively of 37%, 32%, 18%, 10%, and 4%. This result is very similar to the one obtained by De Vries et al. (2002), who also found, using a more restricted number of Level II plots (360) for which direct measurements of atmospheric deposition were available, that atmospheric depositions were representing about 4% of the explained variance in species composition.

Table 4.9 Effects of the environmental variables summarised in broad environmental categories. Percentages explained variance are relative to $\Sigma\lambda_{can}$ / $\Sigma\lambda_{tot}$ (=10%).

Environmental variables	% explained variance
Climate	37.0
Forest type	31.5
Soil	17.8
Sylviculture	9.6
Deposition	4.1

It is quite common to relate floristic composition to climate and forest types. The role of soils and sylviculture as structuring factors contributing to ground vegetation dynamics is also well recognised. However, the demonstration of a significant impact of nitrogen and sulphur deposition on floristic composition at the European scale, as shown here and as found by De Vries et al. (2002), worth certainly some attention. In both studies, the part of variance explained is relatively low (c. 0.4%), but these results are emphasising the role of N deposition as contributing to ground vegetation composition. Other studies have also shown, at a more local or regional scale, the involvement of atmospheric deposition, or N enrichment, on floristic changes of the lichens and vascular plants communities (Thimonier et al., 1992; Van Dobben and ter Braak, 1998; Aarrestad and Aamlid, 1999; Van Dobben et al., 1999; Diekmann and Falkengren-Grerup, 2002). 7Our study is therefore supporting the idea that, at the European scale, a small but significant part of geographic variation in ground vegetation composition is related to atmospheric deposition.

A general picture of the species distribution and environmental influences can be obtained by combining Figure 4.15 (a) and (b). To ease reading, we presented them separately. Axis1 could be interpreted as opposing poor soils associated with Pine stands (with high C/N and low pH), to richer soils associated with Beech stands. Axis2 tends to oppose high precipitation and high altitude, associated with Spruce stands, to high temperatures associated with Oak stands on its positive side. S and N depositions, even though located toward the centre of the graph, are mainly associated with Axis1 (Figure 4.15a).

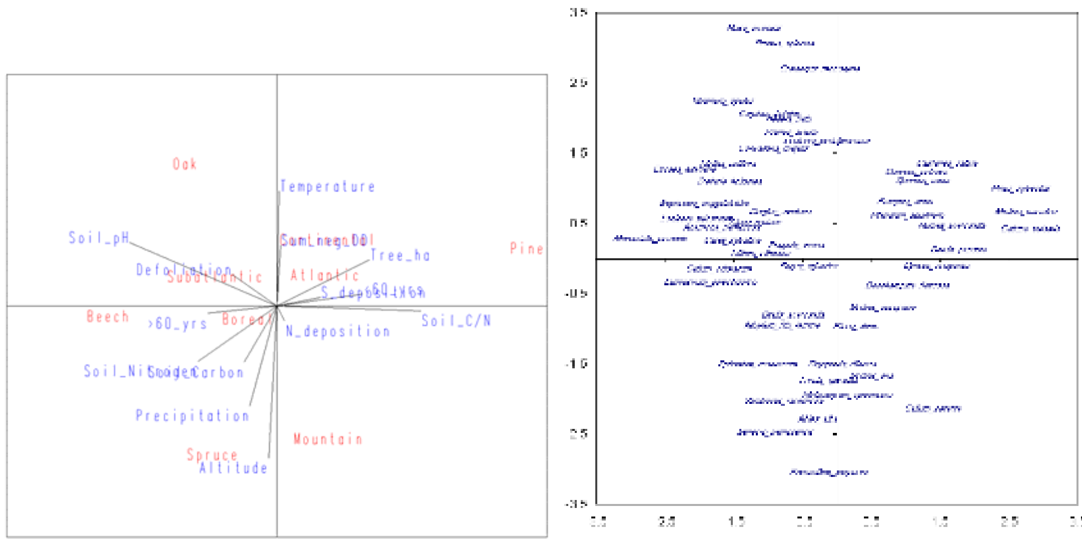


Figure 4.15 *Biplot of environmental variables (left) and species (right) for the first ordination plane of CCA ($\lambda tot=7.3$; $\sum\lambda can=0.73$; $\lambda 1 = 0.17$; $\lambda 2 = 0.14$). a). Environmental variables. b). Species (to ease reading, only species with the highest weight and percentage of explained variance in the model have been selected).*

Figure 4.15b presents species distribution. As expected from the previous graph, acidiphilous species, such as *Molinia caerulea, Calluna vulgaris,* or *Castanea sativa* are on the positive side of Axis1, as opposed to neutrophilous ones such as *Mercurialis perennis, Circaea lutetiana, Lamiastrum galeobdolon, Galium odoratum,* or *Stellaria holostea*, which are located on its negative side. Mountain species or hydrophilous ones such as *Senecio nemorensis, Prenanthes purpurea, Galium saxatile*, *Epilobium montanum, Gymnocarpium dryopteris, Dryopteris dilatata, Sambucus racemosa,* or *Melampyrum sylvaticum* are located on the negative side of Axis2, whereas more xeric ones such as *Rosa arvensis, Prunus spinosa*, or *Crataegus monogyna,* are on the positive side.

This CCA gives a similar picture of the main factors driving species composition throughout Europe as compared to CA results. Soils factors (pH an C/N) as well as temperature, precipitation and altitude are shown to be among the most important factors structuring plant communities at the European scale. The strength of the relationship between vegetation community composition and bioclimatic factors (such as temperature or elevation), as shown by CCA Axis 2, suggest that vegetation could be a potentially reliable indicator of ongoing climatic changes, as was already concluded from the study of Ellenberg index for temperature.

4.4 Conclusions

Species richness

Although a lack of methodological harmonisation introduces a certain amount of uncertainty in the results, the analysis of species numbers showed that the effect of geographical location far exceeds the effect of methodological differences. However, more harmonisation should be aimed at, and has for a large part been agreed upon after the collection of the present data.
Indices calculated from 671 Level II plots (α and γ diversities) are giving a picture of the geographical variation in species richness at the European scale. This variation is influenced by tree species, stand density, soil pH and plot nutritional aspects (foliar Ca concentrations). A significant spatial pattern was also observed, with richer plots being mainly located in Southern Europe. Some rich plots were also observed in the Alps and eastward.

Fencing

Using all paired fenced and unfenced treatments we showed that fencing tended to decrease species richness of several plots. This result could be due to fencing *per se*, as it support results of previous studies. However, we think that it could also be explained by a sampling bias, at least for the French plots, since in these plots the sampling extent of the unfenced subplots was larger than the one of the fenced ones. In further analysing the effects of fencing, the impact of game (and cattle) density on the relation of fenced-unfenced plots by browsing should be taken into account as well as the possible effect of favourising mice and rabbits by excluding foxes etc. by fencing.

Ellenberg's indicator values

Mean Ellenberg's indicator values were also shown to reflect fairly well the environmental conditions of a plot. However, these indices have integrative character and are only loosely linked to unique variables, such as soil pH for R values. Nevertheless, these indicators represent an interesting tool that could be used to monitor long-term changes in environmental conditions, especially climate warming. The indicator value of a plot is probably less influenced by methodological bias as compared to values of species richness.

Influence of environmental factors

The Canonical Correspondence Analysis performed on 602 Level II plots has shown that the species composition at the European scale is mainly driven by classical factors such as climate, soils and forest types, which explain more than 80% of the total modelled species variation. A small but significant influence of atmospheric deposition was also observed. Using logistic regression, several species were shown to have their optimum located at low, or at high N deposition. These species could therefore be used as potential indicators of N status changes along time.

From these results, it is clear that if one wants to draw conclusions about spatial or temporal vegetation changes, comparison should ideally include several successive years per point compared, in order to cope with inter-annual variations. Intercomparison exercises between countries could help improving the homogenisation of European observations. In this perspective, it appears important to continue the ground vegetation surveys on an annual, or a bi-annual basis, at least on a selected number of plots.

Finally, it appears important to mention that this study has underlined the difficulty to use species richness, and related measures of biodiversity, to detect small environmental changes. As these measures are sensitive to several bias sources, they could be seen as having a coarse detection limit. However, another way to investigate environmental changes would be to use the bio-indicator character of the ground vegetation, such as the Ellenberg's indicator values. This aspect appears to be more robust to methodological bias an should not be neglected in further studies.

5 Carbon pools and carbon pool changes at intensive monitoring plots

5.1 Introduction

Background

Information on periodic annual increment at the Intensive Monitoring plots has become available because of a first re-measurement of the trees, five years after installation. The repeated data on tree diameter (at breast height) and tree height can now be used to calculate standing wood volume and changes therein (Dobbertin, 2000). This chapter presents information on those changes in stem wood volume and the related carbon pools in stem wood, calculated from repeated data on both diameter at breast height (dbh) and tree height for all trees in the plot, distinguishing between the tree species occurring at the plot. In this chapter we only focus on the results of forest growth calculations for carbon pool estimations. In the future, an in-depth analysis on the deviation in expected growth (based on standard growth curves for the plot) and the natural and anthropogenic growing conditions, such as stand and site characteristics, soil chemical variables, meteorology and atmospheric deposition, is worthwhile to be carried out.

By multiplying single tree volume with wood densities and tree carbon contents, an estimate for the carbon pool stored in the stem was derived and extrapolated to carbon pools per hectare. Repeated surveys thus allowed the calculation of carbon pool changes. Similarly, carbon pools were calculated for the soil by multiplying soil thickness with soil bulk densities and soil carbon contents. In performing calculations the following data were used:

- Measurements: Diameter at breast height and tree height, soil thickness (volume) and soil carbon contents.
- Estimates: Form factors (to derive tree volume from diameter at breast height and tree height), stem wood density, soil bulk density and carbon contents in stem wood (close to 50%).

In performing the calculations, a differentiation was further made in stem wood and woody biomass, being stem wood and branch wood and in:

- Living standing stock: Stem wood volume of living trees.
- Total standing stock: Stem wood volume of living and dead trees.
- Total stock: Stem wood volume of living, dead and removed trees.

Contents of this chapter

In chapter 5.2, we describe the approach that was used to calculate changes in carbon pools in trees (standing biomass), based on repetitive measurements of diameter at breast height and tree height at Intensive Monitoring plots. This includes the methods (locations, data assessment methods and data evaluation methods) that are needed to calculate carbon pools in trees (standing biomass) and their changes over a five-year period due to forest growth (Section 5.2). Results are described in Section 5.3. This includes carbon pools in trees and soils at the beginning of the surveys (section 5.3.1) and changes in stem wood volume and carbon pools in stem wood due to forest growth in a five year period (Section 5.3.2). Finally, a discussion of the results and conclusions is presented in Section 5.4.

5.2 Methods

5.2.1 Locations

Considering the differences in the derivation of tree volume (see Section 5.2.3), countries were asked to perform volume calculations themselves. For the countries that did not submit information on stem wood volumes per hectare, calculations were made by FIMCI using data on diameter at breast height and tree height for either all trees or part of the trees on the plot. The Intensive Monitoring plots used for the calculation of carbon pools and carbon pool changes, distinguishing between calculations by the country and by FIMCI is given in Figure 5.1.

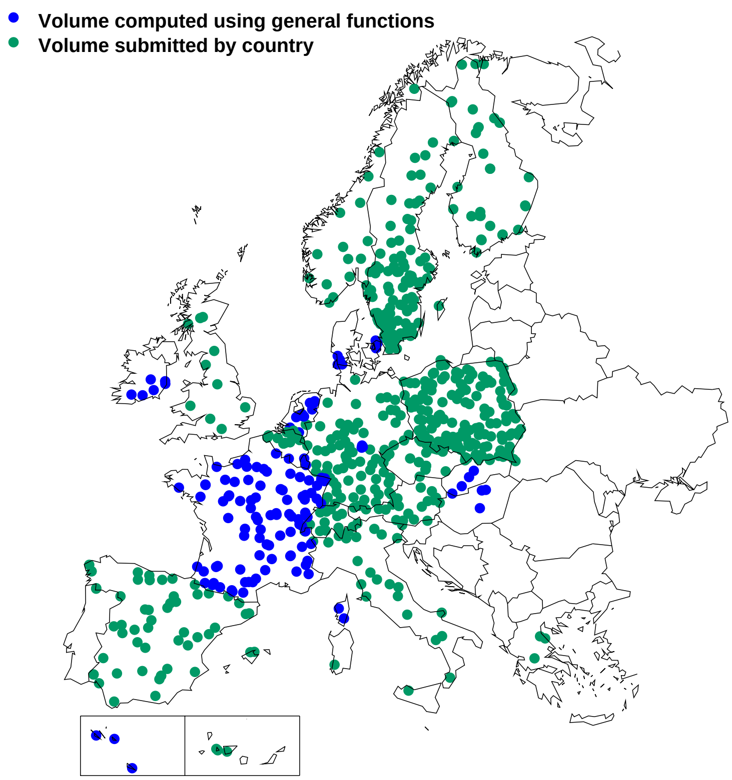


Figure 5.1 Locations of the 646 Intensive Monitoring plots used for the assessment of carbon pools and carbon pool changes in trees based on either submitted or calculated stem volumes.

Carbon pool changes could be calculated at 646 plots. For most countries, carbon pool calculations were based on country submitted volumes (Finland, Norway, Sweden, United Kingdom, Belgium Germany, Austria, Switzerland, Czech Republic, Poland, Spain, Italy and Greece). Countries for which volumes were calculated by FIMCI are Denmark, France, Hungary, Ireland, Netherlands and the Slovak Republic.

5.2.2 Data assessment methods

On all plots included in the forest growth survey, measurements of diameter at breast height (dbh) were carried out. Callipers or tape have been used to measure the diameter at breast height of the trees. On most plots, 'all' trees were assessed (either on the entire plot or on a subplot). When this was not the case, these plots were not been included in the evaluations. A similar strategy had to be applied for a limited number of plots for which only part of the trees (or no trees at all) were labelled with a unique tree number. Trees need such a fixed unique number in the database to facilitate the comparison between subsequent forest growth assessments. Tree height was only measured at part of the plots and mostly for part of the trees, using Blume-Leiss, Suunto hypsometers, Vertex instruments and Relaskop.

Data reliability is influenced by the number of assessment trees. Fig. 5.2 gives an overview of the numbers of assessment trees for diameter at breast height (dbh) and tree height combined with the number of plots. Data presented apply to the most recent forest growth measurements at each plot. On average 140 trees per plot were measured, ranging from 24 to 1135 trees (see also Fig. 5.2A). For a relatively large number of plots it could not be determined whether plots or subplots have been used.

At 83% of the plots where dbh was assessed also tree height was determined. It has to be noted that even measured tree height can be prone to large errors (Cluzeau et al., 1998). Approximately half of the countries determined tree height for all trees in the plot, whereas the other half reported measurements on a selection of trees (ranging between 8 and 50 trees). The average number of trees per plot at which tree height was measured was 55. A maximum of 472 trees per plot was assessed and minimal 3 trees were measured (see also Fig. 5.2B).

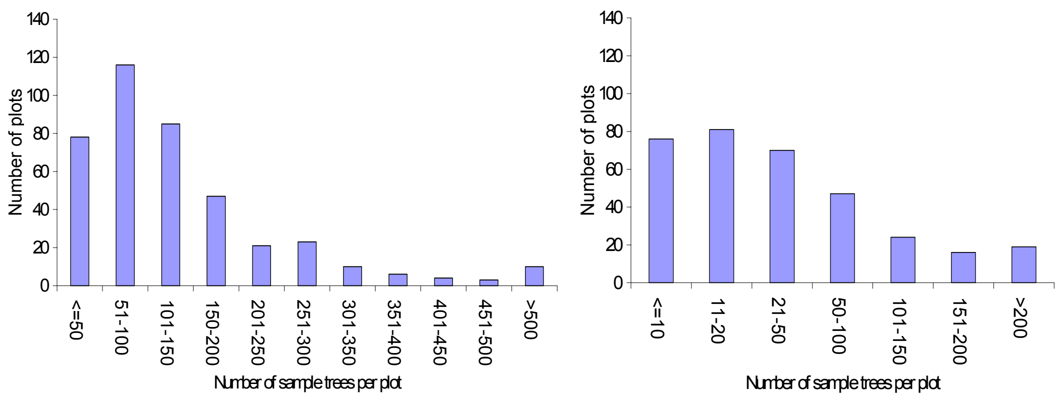


Figure 5.2 Number of sample trees used for the determination of diameter at breast height (A) and tree height (B).

5.2.3 Data evaluation methods

Considering impacts of tree species and geographic region on equations calculating tree volume from diameter at breast height and tree height, countries were asked to perform volume calculations themselves and send the result to FIMCI. For the countries that did not send this information, stem volume was calculated for different major tree species groups on the basis of selected volume equations. Below, we describe the information on stem volume submitted by different countries and the approach used to perform stem wood volume calculations for countries (plots) where this information was not received. Furthermore, the approach used to calculate carbon pool changes in tree stem wood is presented. In addition to the above ground biomass and the carbon sequestrated there an attempt was made to quantify the carbon stored in the soil. The approach used to calculate carbon pools in soil is therefore also shortly summarised.

Type of volume calculations

In presenting information on the volume and volume changes of wood in forests, it is important to define the type of wood included. In this context, a distinction can be made in:
- Stem wood: all stem wood from the bottom (forest floor or stump) to the tip of the tree

- Stem wood above a minimum diameter: stem wood and "thick branches" excluding part of the stem and branches below a minimum diameter (the tip of the tree).
- Total above-ground woody biomass: all stem wood and branch wood from the bottom to the tip of the tree
- Total above-ground woody biomass above a minimum diameter (usually only with broadleaves): stem wood and "thick branches", i.e. all stem wood and branches above a minimum diameter

In practice, a distinction is often made in stem wood, stem wood above a minimum diameter and tree wood above a minimum diameter, as visualised in the pictures below. The form factor, being the factor by which the product of the basal area and tree height has to be multiplied, varies depending on the definition of wood used. For stem wood, the form factor will decrease from a value of ∞ (infinite) at a dbh = 0, to some value of about 0.25 at large dbhs. For stem wood above a certain diameter, the form factor function starts with zero at dbh=0, then increases and finally approaches the stem wood function without minimum diameter. For tree wood above a minimum diameter the form factor function behaves at first like the stem wood function with a minimum diameter, and then ends with values somewhat larger than the stem wood function.

In this report, we reported stem wood. Some countries, however, reported total woody biomass till a certain top diameter. Furthermore, several volume equations refer to stem wood and thick branches. In these situations, correction functions were used to calculate stem wood from tree wood till a certain diameter, as described further.

Volume calculations by countries

Stem volume is related to the diameter (dbh or d) of the tree (normalised at breast height, 1.30m, and often denoted as dbh) and tree height (h). There are various equations that do relate tree volume to both parameters, depending on tree species and geographic region. Considering these differences, countries were asked to perform volume calculations themselves and send stem wood volumes per hectare ($m^3.ha^{-1}$) to FIMCI, while distinguishing between:
- Remaining alive stem wood: refers to trees that were alive in the first and second survey.
- Dead stem wood: refers to trees that were alive in the first survey and dead in the second survey (newly dead stem volume) and trees that were dead in both surveys.
- Removed stem wood: refers to trees that were either alive or dead in the first survey and removed in the second survey.

This information for the first and second growth survey allows the calculation of changes in volume ($m^3.ha^{-1}.yr^{-1}$) in terms of:
- Living standing stock: Changes in stem wood volume of trees that were alive in both the first and second survey.
- Total standing stock: Changes in stem wood volume of trees that were alive or dead in both the first and second survey. Removal of trees that were dead in the first survey is not reported.
- Total stock: Changes in stem wood volume of trees that were dead or alive in the first survey and dead or alive or removed in the second survey

Trees removed between the two observations were treated as having no increment. The countries that submitted information on volumes for the different categories, allowing those calculations are given in Table 5.1. Information was submitted by 13 countries, but in the case of Italy, the tree volume had to be recalculated to stem volume (see further). For six countries, the tree volume data were based on FIMCI calculations. A check was made on these calculations by comparing the results with the countries that did submit both data on diameter at breast height and tree height (allowing volume calculations) and tree volume as well.

Actually, all countries that submitted stem volumes differentiated between remaining alive stem wood, newly formed dead stem wood and removed stem wood, apart from Poland where only data for remaining alive stem wood were given. As asked for, the countries reported the volume categories for each forest stand but not per species in each stand. Since the conversion factors from volume to biomass are species specific, a correction procedure was applied. Volumes for all plots were calculated on a tree species basis by FIMCI, according to the approach described below. When the country also supplied total volume, we redistributed this total volume over the species, according to the volume share of the species in the FIMCI computations.

Table 5.1 Countries that submitted information on stem volume, tree volume and tree characteristics allowing the calculation of stem volume

Country	Submitted stem volume	Submitted tree volume recalculated to stem volume	Calculated stem volume
Austria	x		
Belgium	x		
Czech Republic	x		
Denmark			x
Finland	x		
France			x
Germany	x		
Greece	x		
Hungary			x
Ireland			x
Italy		x	
Netherlands			x
Norway	x		
Poland	x		
Slovak Republic			x
Spain	x		
Sweden	x		
Switzerland	x		
United Kingdom	x		

Volume calculations by FIMCI

For the countries that did not submit the results of their volume calculations, stem wood volume of each individual tree (V in m^3) was calculated as a function of the diameter at breast height (d, or dbh, in cm) and tree height (TH or h in m). The calculations were done for (clusters of) major tree species, while distinguishing between coppice forests and high forests. Differences between equations from different sources often originated from limited data, poor representation of sites, tree dimensions (no large dbh in the Scandinavian countries) and stand treatments. In the form factor equations for Austria, for example, half of the residual variance could be explained by an additional upper diameter and the dbh depends mostly on stand density (Pollanschütz, 1965). Consequently, the impact of regions was not included in the relationship used by FIMCI to calculate volume, as it might be strongly mixed with methodological aspects.

Two different type of volume equations were used to calculate the volume of each individual tree as a function of diameter and height:

1. A direct relationship between V and (d) and (h), according to some type of polynomial relationship, according to:

$$\begin{aligned} V = a + b_1 d + b_2 h + b_3 dh + b_4 d^2 + b_5 h^2 + b_6 d^2 h + b_7 dh^2 + b_8 d^2 h^2 + b_9 d^3 \\ + b_{10} d^3 h + b_{11} d^3 h^2 + b_{12}/h \end{aligned} \tag{5.1}$$

2. An indirect relationship between V and (d) and (h), according to the multiplication:

$$V = \sum_{i=1}^{N} d_i^2 \frac{\pi}{4} \cdot h_i \cdot f(d_i, h_i) \tag{5.2}$$

Where:
V = volume (m^3)
d = diameter at breast height (cm)
h = tree height (m)
f = an individual tree form factor equation

In situations where height data are missing, they were calculated from species and plot specific height curves, the parameters of which were calculated from the sample for which height and dbh were measured, according to (Prodan, 1965):

$$h - 1.30 = \frac{d^2}{\alpha_0 + \alpha_1 \cdot d + \alpha_2 \cdot d^2} \tag{5.3}$$

The form factor f was mostly calculated as (Pollanschütz, 1974; Schieler, 1988):

$$f = b_1 + b_2 \ln^2 d + \frac{b_3}{h} + \frac{b_4}{d} + \frac{b_5}{d^2} + \frac{b_6}{dh} + \frac{b_7}{d^2 h} \tag{5.4}$$

For some species, the coefficients were estimated separately for trees with d (dbh) < 10.4cm and d (dbh) > 10.4cm in a way that for dbh = 10.4cm both equations resulted in the same form factor (Pollanschütz, 1974; Schieler, 1988) Eq. (5.4) has also been used for Swiss pine, Black pine and the so called “subsidiary broadleaves” in the high forest system, while being valid for dbh > 5 cm (Schieler, 1988). Actually, 5 cm was the minimum diameter for tree to be included in the Intensive Monitoring plots.

An overview of the type of equations that were used as a function of tree species and geographic region is given in Table 5.2. The various coefficients that were used in the direct volume equation and in the form factor equation are presented in the Tables 5.3 and 5.4, respectively. The direct volume functions used for the calculation of volumes in high forests in the whole of Europe (G) were based on: (i) the literature (Austria, Germany (Bavaria), Switzerland) and (ii) an overview of form functions elaborated in an EU cost project for countries in the Northern countries (Finland, Sweden, Norway) and in Belgium (Flanders). These functions were all reviewed by Sterba to provide one most adequate equation per tree species.

The indirect volume functions were mostly related to tree species occurring in the Mediterranean area only. Data for the Mediterranean countries were provided by G. Fabio and originated from the analysis and processing of a set of existing tables on the occasion of the national Forest Inventory carried out in 1985. The analyses were performed by the Forest & Range Management Institute (ISAFA) of Trento-Lab of Forest Biometrics. The range of validity of the forest mensuration parameters (V in m^3; d or dbh in cm and h in m) for the Mediterranean equations (see Table 5.2) are from d = 15 cm onwards for high forests and d = 3 cm onwards for coppice forests. For those cases where the equations had to be extrapolated below the 15 cm limit we checked the form factors on plausibility (at least not less than 0 or higher than 1).

A list of all species occurring in the Level II plots and an allocation of these species to one of the main tree species (groups) is shown in Annex 1. This annex includes the allocation according to Table 5.2 plus an additional allocation of tree species not mentioned in this table. An overview of the parameters used in the direct and indirect volume equations for the distinguished tree species is given in Table 5.3 and 5.4, respectively.

Table 5.2 Type of equations that were used as a function of tree species and geographic region (G stands for general, being applicable for all countries except the Mediterranean, denoted as M).

Main Tree species	Species included	Direct volume equation	Indirect form factor equation
Norway Spruce	*Picea abies*	X (G)[1]	-
Scots Pine	*Pinus Sylvestris*	X (G)	-
Birch	*Betula pubescens, Betula pendula = B. verrucosa*	X (G)	-
Beech and subsidiary broadleaves high forest	*Fagus sylvatica, Prunus sp., Robinia sp., Sorbus sp.,Tilia sp*	-	X (G)
Beech coppice	*Fagus sylvatica*	X (M)	
Larch	*Larix europaea*	-	X (G)
Fir	*Abies alba*	-	X (G)
Swiss Pine	*Pinus cembra*	-	X (G)
Oak (Quercus)	*Q. robur, Q. cerris, Q. petraea, Castanea sativa*	-	X (G)
Oak (Quercus)	*Q. frainetto, Q. ilex, Q. pubescens, Q. suber*	X (M)[2]	-
Black Pine	*Pinus nigra*	-[1]	X (G)
Stone pine	*Pinus pinea*	X (M)	-
Maritime pine	*Pinus pinaster*	X (M)	-
Aleppo pine	*Pinus halepensis and Pinus brutia*	X (M)	-
Hornbeam, Hophornbeam[3]	*Carpinus (betulus), Ostrya sp.*	-	X (G)
Ash[3]	*Fraxinus sp.*	-	X (G)
Maple[3]	*Acer sp.*	-	X (G)
Elm[3]	*Ulmus sp.*	-	X (G)
Alder[3]	*Alnus sp.*	-	X (G)
White Poplar[3]	*Populus alba*	-	X (G)
Black Poplar[3]	*Populus nigra, Populus hybrides, Populus tremula,*	-	X (G)
Willow[3]	*Salix sp.*	-	X (G)
Subsidiary broadleaves coppice	*Acer sp.,Alnus sp., Betula sp., Carpinus sp., Fraxinus sp., Ostrya sp., Prunus sp., Robinia sp., Salix sp., Sorbus sp.,Tilia sp, Ulmus sp*	X (M)	

1 For these tree species, equations were also available for Mediterranean areas, but these functions were not used in the calculations.
2 For these tree species, different equations were used for stands managed under the high forest system and the coppice system.
3 For these tree species, the different equations all refer to stands managed under the high forest system.

In performing the calculations, we made the following assumptions:
- The Beech formula can be applied for all subsidiary broadleaves and the ratio is equal for stands managed under both the high forest and coppice system
- The Oak formula can be applied for stands managed under both the high forest system and the coppice system,
- The Scots pine formula was applied for the Stone pine and Aleppo pine

Corrections used to calculate stem wood from tree wood above a minimum diameter

In this report, we reported stem wood. Some countries (e.g. Italy), however, reported tree wood above a minimum diameter. Furthermore, several volume equations refer to stem wood + branch wood (thick branches or all branches), whereas it was decided that all volume equations will refer to stem wood only. The tree species for which this is the case are Beech (Fagus sylvatica) managed under the coppice system, Oak sp. (Quercus sp.) managed under the high forest system and under the coppice system, Stone pine, Aleppo pine and subsidiary broadleaves managed under both the high forest and coppice system.

Table 5.3 The values of the parameters (a and b_1-b_{12}) that were used in the direct volume equation:
$V= a + b_1d+b_2h+b_3dh+b_4d^2+ b_5h^2+ b_6d^2h + b_7dh^2+ b_8d^2h^2+b_9d^3+ b_{10}d^3h + b_{11}d^3h^2 + b_{12}/h$

Tree species	a	b_1 (d)	b_2 (h)	b_3 (dh)	b_4 (d^2)	b_5 (h^2)	b_6 (d^2h	b_7 (dh^2)	b_8 (d^2h^2)	b_9 (d^3)	b_{10} (d^3h)	b_{11} (d^3h^2)	b_{12} (1/h)
Norway Spruce					$0.115*10^{-3}$	$0.5618*10^{-4}$	$0.1746*10^{-4}$	$0.2022*10^{-4}$					
Scots Pine					$0.1072*10^{-3}$		$0.2427*10^{-4}$	$0.7315*10^{-5}$					
Birch[1]	-	-	-	-	-	-	-	-	-	-	-	-	-
Beech Mediterranean. Coppice						$0.187061*10^{-4}$	$0.524502*10^{-4}$		$-0.711644*10^{-6}$				
Oak Mediterranean. High forest		$0.172373*10^{-3}$	$-0.252758*10^{-3}$		$0.585386*10^{-4}$	$-0.120911*10^{-4}$	$0.307106*10^{-4}$	$0.113982*10^{-5}$	$0.152380*10^{-7}$	$0.488191*10^{-5}$		$0.397981*10^{-9}$	
Oak Coppice				$0.217520*10^{-4}$	$0.642760*10^{-4}$		$0.444912*10^{-4}$		$-0.728724*10^{-6}$		$-0.297849*10^{-6}$		
Black Pine Mediterranean.	$0.457023*10^{-3}$	$-0.423133*10^{-4}$	$0.160308*10^{-2}$	$-0.112508*10^{-3}$	$0.210093*10^{-4}$	$0.132827*10^{-4}$	$0.380346*10^{-4}$		$0.337571*10^{-8}$	$-0.177836*10^{-6}$		$-0.491192*10^{-9}$	
Stone pine		$-0.141070*10^{-2}$		$0.397095*10^{-3}$			$0.463730*10^{-4}$	$-0.146961*10^{-4}$		$-0.652415*10^{-6}$	$0.183120*10^{-6}$	$0.191249*10^{-8}$	
Maritime pine	$-0.319109*10^{-2}$	$-0.394309*10^{-3}$			$1.85128*10^{-4}$	$0.178057*10^{-4}$	$0.281052*10^{-4}$			$0.450308*10^{-6}$			
Aleppo pine	0.129174	$-1.41482*10^{-2}$	$-0.125895*10^{-1}$	$0.904472*10^{-3}$	$0.279059*10^{-3}$		$0.301592*10^{-4}$			$-0.645571*10^{-6}$		$0.258438*10^{-8}$	
Subsidiary broadleaves High forest	$0.140099*10^{-3}$	$0.370368*10^{-3}$	$0.151173*10^{-3}$	$-0.821778*10^{-4}$	$0.124442*10^{-3}$	$0.378640*10^{-6}$	$0.381535*10^{-4}$	$0.690078*10^{-6}$	$0.131890*10^{-7}$	$-0.277001*10^{-6}$	$0.959694*10^{-8}$	$0.103466*10^{-8}$	$0.489632*10^{-1}$
Subsidiary broadleaves Coppice	-0.001614	$0.959885*10^{-3}$	$-0.240608*10^{-3}$				$0.372428*10^{-4}$						

1 The equation for birch does not fit in the general scheme and was calculated as: $v[m^3] = 0.127764 \cdot 10^{-3} \cdot d^{2.27954} \cdot h^{-1.18672} \cdot (d+20)^{7.07362} \cdot (h-1,3)^{-5.45175}$ with d in cm and h in m

Table 5.4 The values of the parameters that were used in the form factor equation for a diameter at breast height less than 10.4 cm and more than 10.4 cm (b_1-b_7) in the equations $f = b_1 + b_2 \ln^2 d + \frac{b_3}{h} + \frac{b_4}{d} + \frac{b_5}{d^2} + \frac{b_6}{dh} + \frac{b_7}{d^2 h}$ *The form factor function was also used for dbh > 5 cm for the trees species Swiss pine, Black pine and the subsidiary broadleaves (Carpinus (betulus), Fraxinus sp., Acer sp. Ulmus sp., Alnus sp., Populus alba, Populus nigra, Salix sp.). In that case no function for a dbh < 5 cm is used*

Tree species	Dbh (cm)	b_1	b_2 (ln d/10)^2	b_3 (1/h)	b_4 (1/d)	b_5 (1/d^2)	b_6 (1/dh)	b_7 (1/d^{2}h)
Beech[1]	> 10.4	0.68625	-0.0371508	-3.10674	-3.8631	21.9462	49.6136	-223.719
	< 10.4	0.5173	-	-1.362144	-	-	9.9888	-
Larch	> 10.4	0.609443	-0.0455748	-1.86631	-2.48736	12.6594	36.9783	-142.04
	< 10.4	0.48727	-	-0.204291	-	-	5.9995	-
Fir	> 10.4	0.580223	-0.0307373	-1.71507	0.89869	-8.0557	19.661	-24.5844
	< 10.4	0.560673	0.15468	-0.065583	0.3321	-	-	-
Oak	> 10.4	0.115631	-	6.59961	12.0321	-93.0406	-215.758	1684.77
	< 10.4	0.417118	0.21941	1.332594	-	-	-	
Swiss Pine	>5	0.525744	-0.0334896	0.738943	-1.0646	-	-	33.4479
Black Pine	>5	0.5438	-0.00763	-		-	-	22.414
Carpinus (betulus)	>5	0.32473	0.02432	-	0.23972	-	-9.9388	-
Fraxinus sp.	>5	0.48122	-0.01489	-10.83056	-	-	9.3936	-
Acer sp.	>5	0.50101	-0.03521	-8.07176	-	0.03521	0.0	-
Ulmus sp.	>5	0.44215	-0.02446	0.0	-	0.0	0.0	2.87714
Alnus sp.	>5	0.42937	-	-4.10259	-	0.0	16.7578	-5.16631
Populus alba	>5	0.31525	-	0.0	0.51079	-0.34279	-26.08	28.6334
Populus nigra	>5	0.4115	-0.00989	-28.27478	0.35599	-0.21986	21.4913	-
Salix sp.	>5	0.54008	-0.002716	-25.11447	0.08327	-	9.3988	-

To relate stem wood to tree wood above a minimum diameter (stem wood and thick branches), correction functions were used as presented in Figure 5.3. The volume of broadleaf's above a certain diameter varies somewhere between 5 and 10 cm, thus being normalised to 7 cm. The ratios of stem wood (Vstem) and tree wood above 7 cm (V7) for oak, beech and Scots pine were based on an Austrian equation (Pollanschütz, 1974; Schieler, 1988) for stem wood only and a Bavarian equation (Kennel, 1973) for volume above 7 cm.

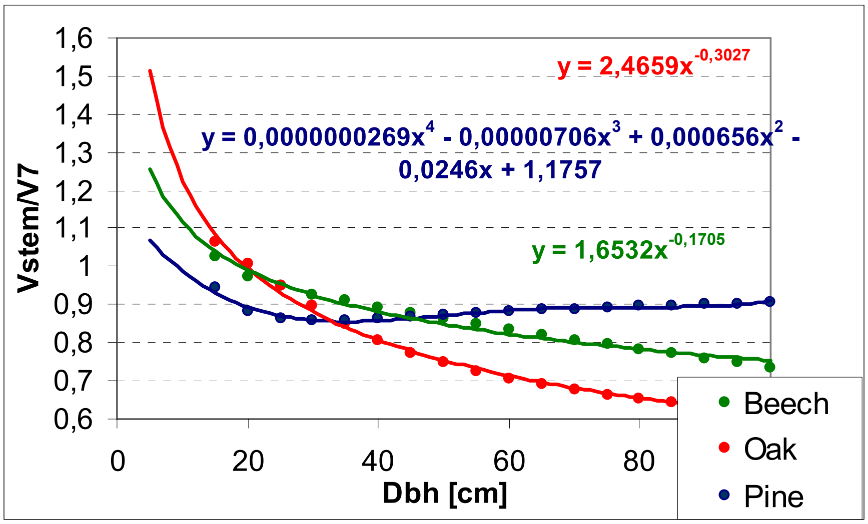


Figure 5.3 Equations used to relate stem wood to total woody biomass above a minimum diameter for oak, beech and Scots pine.

More precisely, first the form height was calculated for V7, which is the tree height times the form factor, according to (Kennel, 1973):

$$\ln fh = a + b \cdot \ln h + c \cdot \ln^2 h \quad (5.7)$$

with:

$$a = a_0 + a_1 \cdot \ln d + a_2 \cdot \ln^2 d \quad (5.8a)$$
$$b = b_0 + b_1 \cdot \ln d + b_2 \cdot \ln^2 d \quad (5.8b)$$
$$c = c_0 + c_1 \cdot \ln d + c_2 \cdot \ln^2 d \quad (5.8c)$$

The coefficients for the tree species in Figure are given in Table 5.5. With these equations and a height curve we calculated f = fh/h for V7. Then the form factor equations of Pollanschütz (1974) were used to calculate f for Vstem and Vstem/V7 thus calculated was depicted in figure 5.3.

Table 5.5 Coefficients for Beech, Oak and Pine used to calculate stem wood volume from total woody biomass.

Tree species	Coefficient a0	a1	a2	b0	b1	b2	c0	c1	c2
Beech	-2.7284	1.62283	-0.08797	0.83756	-0.21481	0.032567	-0.10534	0.028927	-0.00446
Oak	-3.06118	1.93898	-0.1651	1.45506	-0.68973	0.120127	-0.19992	0.112653	-0.02025
Pine	-5.80915	3.67116	-0.45928	3.387	-1.83211	0.29989	-0.49489	0.273999	-0.04449

Beware that the ratio in the figure depicted can be higher and lower than 1, as visualised in Figure 5.4. The first two pictures relate to a tree with branches and a top smaller than 7 cm. Here v7 is smaller than v_{stem}, because the volume of branches is excluded (less than 7 cm) and also the stem wood below 7 cm (red is stem wood, green is tree wood above 7 cm).The second two pictures relate to a tree with a diameter > 7 cm, which has already thick branches. Here v7 is larger than v_{stem}, because the volume of large branches is larger than the very small volume at the top of the tree below 7 cm, which is neglected. The ratio will approach a constant value (>1) assuming that there is a constant proportion of large branches per stem volume.

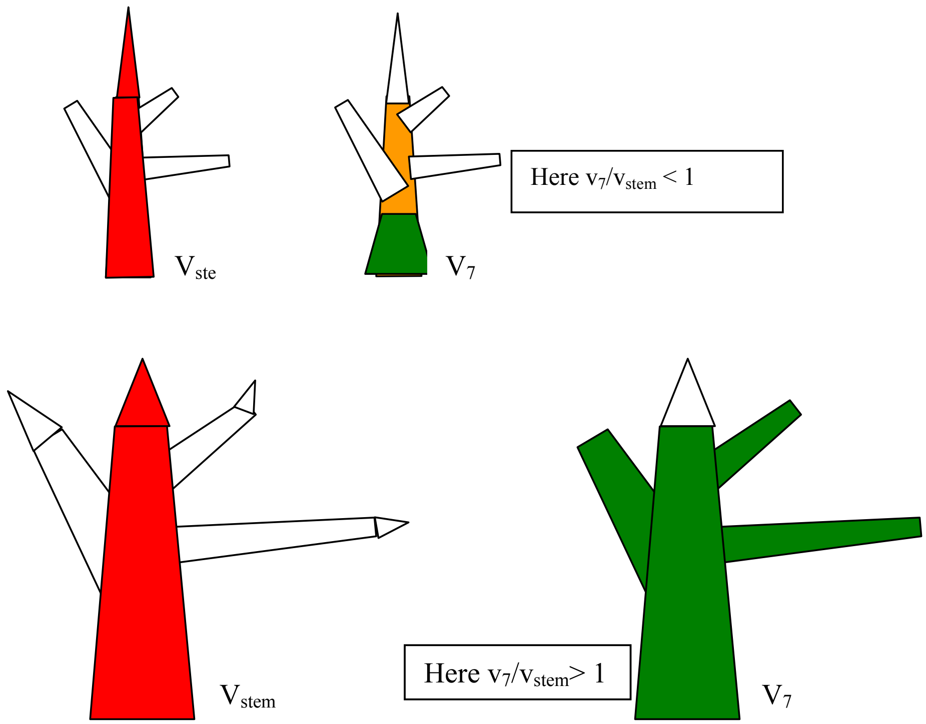


Figure 5.4 Pictures showing a tree with branches and a top smaller than 7 cm (v7/vstem <1) and a tree with a diameter > 7 cm, which has already thick branches (v7/vstem >1).

Calculation of carbon pools in trees and soil

Carbon pools in trees in stem wood were calculated by multiplying stem wood volumes ($m^3.ha^{-1}$) with stem wood density ($kg.m^{-3}$) and an assumed C content of 50% in stem wood. ($kg\ C.kg^{-1}$). Data on stem wood density per tree species that were used are presented in Table 5.6. Most data were derived from Wagenfuhr and Schreiber (1989), with data for a few species being based on Wiselius (1994). For the Eucalyptus, use was made of data in (Ilic et al., 2000).

In this chapter, we also present carbon pools in soil. Data assessment methods related to soil are described in detail in De Vries et al. (2000). Carbon pools were calculated for the organic layer and for the layer 0-80 cm. For 17% of the plots, C content was not measured for the entire soil compartment of 0-80 cm. For these plots (mainly located in Scandinavia) the C-pool was computed for the layer with measurements (mostly 0-40 cm) assuming that either the soil profile is not thicker than the lowest measurement depth (plots in e.g. Norway and Finland) or that the contribution of the deeper layer is not significantly contributing to the total C-Pool (other plots).

Carbon pools in the organic layer were calculated by multiplying the measured organic layer pool with the carbon content in that layer. Carbon pools in the mineral topsoil were calculated by multiplying an estimated bulk density of the soil with the soil thickness, the carbon content in the soil and an estimated coarse fraction (stones). More information on the calculation approach is given in De Vries et al. (2000).

Table 5.6 Stem wood densities per tree species that were used to calculate carbon pools in trees

Tree species Group	Included tree species	Wood density ($kg.m^{-3}$)
Salix	Salix alba, Salix caprea, Salix cinerea, Salix eleagnos, Salix fragilis, Salix sp.	330
Thuja	Thuja sp.	350
Cedrus	Cedrus atlantica, Cedrus deodara	400
Abies/Populus	Abies alba, Abies borisii-regis, Abies cephalonica, Abies grandis, Abies nordmanniana, Abies pinsapo, Abies procera, Pinus radiata, Pinus strobus, Populus alba, Populus canescens, Populus hybrides, Populus nigra, Populus tremula,	410
Picea	Picea abies, Picea omorika	400
Picea sitchensis	Picea sitchensis	350
Tsuga	Tsuga sp.	440
Other conifers	Cupressus lusitanica, Cupressus sempervirens, Juniperus communis, Juniperus oxycedrus, Juniperus phoenicea, Juniperus sabina, Juniperus thurifera, Taxus baccata, Chamaecyparis lawsoniana, Other conifers	450[1]
Pseudotsuga	Pseudotsuga menziesii	470
Pinus/Tilia	Pinus brutia, Pinus canariensis, Pinus cembra, Pinus contorta, Pinus halepensis, Pinus heldreichii, Pinus leucodermis, Pinus mugo, Pinus nigra, Pinus pinaster, Pinus pinea, Pinus sylvestris, Pinus uncinata, Tilia cordata, Tilia platyphyllos,	490
Alnus	Alnus cordata, Alnus glutinosa, Alnus incana, Alnus viridis	510
Prunus/Larix	Prunus avium, Prunus dulcis, Prunus padus, Prunus serotina, Larix decidua, Larix kaempferi	550
Juglans	Juglans nigra, Juglans regia	560
Olea/Platanus	Olea europaea, Platanus orientalis	580
Acer	Acer campestre, Acer monspessulanum, Acer opalus, Acer platanoides, Castanea sativa	590
Other broadleaves	Buxus sempervirens, Ilex aquifolium, Tamarix africana, Arbutus unedo, Arbutus andrachne, Ceratonia siliqua, Cercis siliquastrum, Erica arborea, Erica scoparia, Erica manipuliflora, Phillyrea latifolia, Phillyrea angustifolia, Pistacia lentiscus, Pistacia terebinthus, Rhamnus oleoides, Rhamnus alaternus, Betula tortuosa, Ceratonia siliqua (same as 75), Crataegus monogyna, Other broadleaves	595[1]
Betula	Betula pendula, Betula pubescens	610
Acer/Ulmus	Acer pseudoplatanus, Ulmus glabra, Ulmus laevis, Ulmus minor	640
Fraxinus/ Quercus	Fraxinus angustifolia, Fraxinus excelsior, Fraxinus ornus, Quercus cerris, Quercus coccifera, Quercus faginea, Quercus frainetto, Quercus fruticosa, Quercus ilex, Quercus macrolepsis, Quercus petraea, Quercus pubescens, Quercus pyrenaica, Quercus robur, Quercus rotundifolia, Quercus rubra, Quercus suber, Quercus trojana	600
Fagus	Fagus moesiaca, Fagus orientalis, Fagus sylvatica	680
Eucalyptus	Eucalyptus sp., Malus domestica, Pyrus communis, Laurus nobilis, Myrtus communis	700
Sorbus	Sorbus aria, Sorbus aucuparia, Sorbus domestica, Sorbus torminalis	730
Robinia pseudacacia	Robinia pseudacacia	740
Carpinus	Carpinus betulus, Carpinus orientalis, Corylus avellana, Ostrya carpinifolia	790

[1] Stem densities for the considered other conifers and other broadleaves have been set at the average of all species

5.3 Results

5.3.1 Carbon pools in soils and trees in the beginning of the monitoring period

Stem wood volumes

Calculations of stem wood volumes were made by countries or by FIMCI in cases that countries did not submit them. To gain insight in the adequacy of the volume calculations carried out by FIMCI, a comparison was made with the country submitted volumes. Results (Fig. 5.5) show that the comparison is generally good with some notable exceptions. Especially at some plots with stem wood volumes below 600 $m^3.ha^{-1}$, results were sometimes poor. This was the case in plots where hardly any height measurements were available. Furthermore, results were poor for exotic tree species in which the volume calculations were based on more general species (Section 5.2.3). For these tree species, however, we mostly had country submitted volumes thus allowing reasonable calculations of the carbon pools and their changes in time.

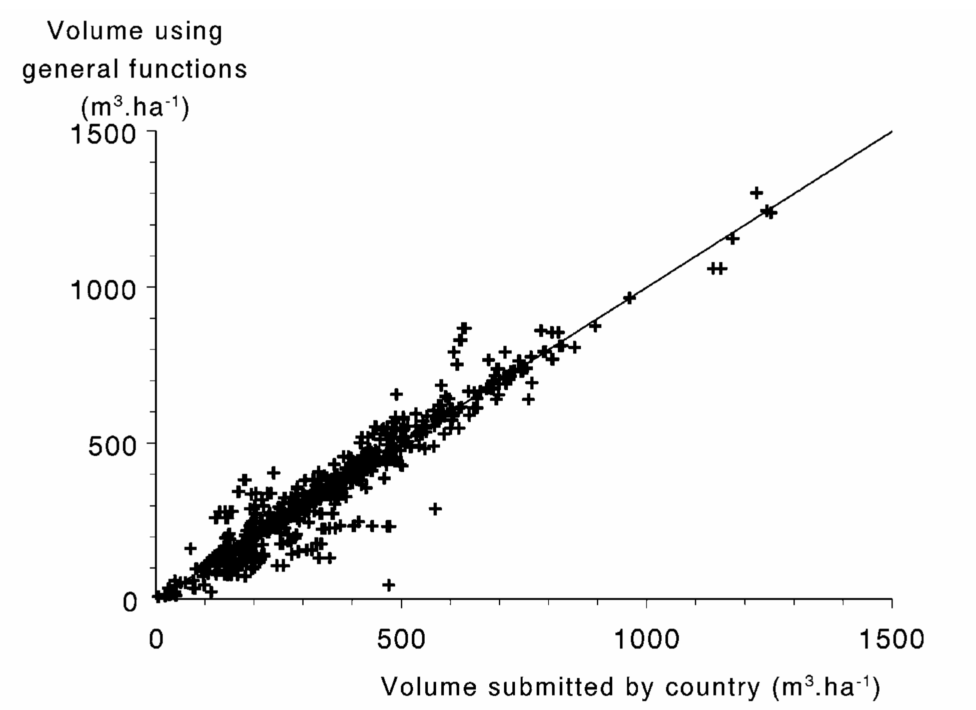


Figure 5.5 Comparison of stem wood volume calculations carried out by FIMCI with country submitted volumes

Ranges (median values and the 90% range as expressed by the 5 and 95 percentile) of calculated stem wood volume of living trees and all trees at the beginning of the monitoring period, are given in Table 5.7. The results show that the difference in stem wood volume of living trees and all (both living and dead) trees is negligible or marginal. Differences were only found for mixed broadleaves (median values of approximately 40-50 $m^3.ha^{-1}$).

Table 5.7 Ranges (medians, 5 and 95 percentile values) of stem wood volume of living trees and all trees.

Tree species group	Number of plots	Stem wood volume of living trees ($m^3.ha^{-1}$)		Stem wood volume of all trees ($m^3.ha^{-1}$)	
		Median	(5-95%)	Median	(5-95%)
Scots Pine	191	243	(91 - 363)	243	(91 - 363)
Spruce	154	342	(143 - 739)	344	(144 - 745)
Fir	16	472	(150 – 770)	472	(150 – 770)
High elevation conifers	5	277	(105 - 358)	277	(105 - 358)
Mediterranean pines	23	179	(51 – 700)	179	(51 – 700)
Remaining conifers	5	122	(8 - 644)	122	(8 – 644)
Mixed Conifers	28	398	(152 - 823)	398	(152 - 823)
Beech	71	346	(128 - 621)	346	(128 - 621)
Standard Oak	42	245	(93 - 417)	245	(93 - 417)
Oak evergreen	16	31	(11 - 205)	31	(11 - 205)
Oak other	12	126	(14 - 318)	127	(14 - 318)
Remaining broadleaves	3	227	(89 - 457)	227	(89 - 457)
Mixed Broadleaves	39	282	(141 - 474)	287	(151 - 474)
Mixed	39	339	(85 - 666)	339	(85 - 667)
All	644	285	(72 - 666)	287	(72 - 666)

An overview of the geographic variation of the stem wood volume of living trees, given in Figure 5.6, shows that low stem wood volumes (< 150 $m^3.ha^{-1}$) do occur in plots in Northern and Southern Europe, most likely due to temperature impacts (cold climate and water stress), and moderate (150-600 $m^3.ha^{-1}$) to high stem wood volumes (>600 $m^3.ha^{-1}$) in Central Europe, with highest volumes in Southern Germany, Switzerland and Austria.

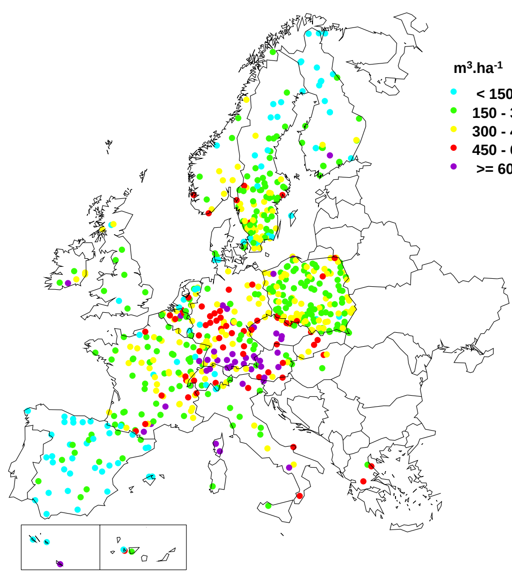


Figure 5.6 Stem wood volumes of living trees at 644 Intensive Monitoring plots

Carbon pools in stem wood and soil

Ranges (median values, 5 and 95 percentile) of calculated carbon pools in trees and soils (organic layer and mineral soil up to 80 cm) are given in Table 5.8. The results for trees are limited to the total standing stock in stem wood at the beginning of the monitoring period (living and dead trees). Results show that the median carbon pools in the soil are approximately 1.5 times as high as in the trees (overall median values of 105 and 70 ton.ha^{-1}, respectively) with some notable exceptions such as the evergreen oak with a median carbon pool in the soil that is more than 10 times as high than in the tree (Table 5.8).

Table 5.8 Ranges (medians, 5 and 95 percentile values) of calculated carbon pools in tree stem wood (living and dead) and soils.

Tree species group	Number		C pool in stem wood (ton.ha^{-1})		C pool soil (ton.ha^{-1})	
	Tree	Soil	Median	(5-95%)	Median	(5-95%)
Scots Pine	191	(188)	60	(22-89)	57	(21-228)
Spruce	154	(146)	65	(27-141)	130	(52-276)
Fir	16	(16)	98	(35-181)	114	(73-181)
High elevation conifers	5	(3)	75	(26-98)	109	(94-233)
Mediterranean pines	23	(23)	44	(13-172)	140	(20-232)
Remaining conifers	5	(2)	28	(1.8-146)	38	(38-106)
Mixed Conifers	28	(28)	85	(36-167)	111	(34-455)
Beech	71	(51)	118	(43-211)	144	(63-405)
Standard Oak	42	(26)	77	(28-125)	87	(47-389)
Oak evergreen	16	(16)	9.3	(3.6-67)	116	(36-266)
Oak other	12	(12)	41	(4.6-103)	145	(56-249)
Remaining broadleaves	3	(3)	68	(31-136)	473	(236-476)
Mixed Broadleaves	39	(38)	92	(44-154)	79	(50-223)
Mixed	39	(36)	82	(22-183)	126	(43-414)
All	644	(588)	70	(19-167)	105	(30-319)

An overview of the geographic variation in carbon pools in trees and soils is given in Fig. 5.7. The geographic variation in the carbon pools in trees is of course comparable to the variation in standing biomass, given in Figure 5.5, with lowest carbon pools (< 30 ton.ha^{-1}) in Northern and Southern Europe and moderate (30-120 ton.ha^{-1}) to high carbon pools > 120 ton.ha^{-1}) in Central Europe (Fig 5.7 left). The geographic variation of carbon pools in soils differs from trees, with high pools occurring in Northern Europe because of the low mineralization of carbon due to temperature extremes. In Southern Europe (e.g. Spain), there is a large variation in soil C pools (Fig. 5.7 right). A distinct feature is the low carbon pools in Poland. This is partly due to the fact that the organic layer has not been included in the Polish plots.

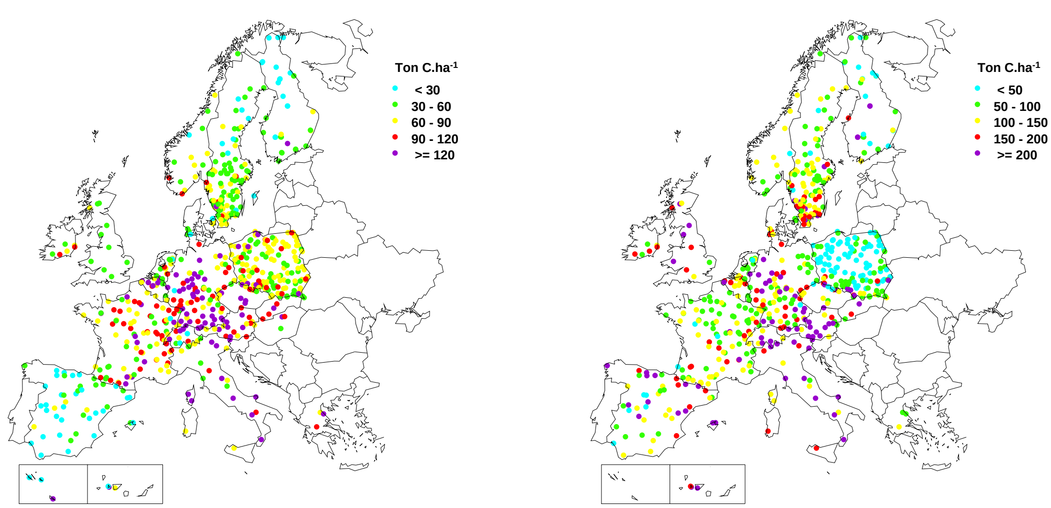


Figure 5.7 Carbon pools in tree stem wood (left) and soil (right) at 644 Intensive Monitoring plots

5.3.2 Carbon pool changes in trees

Changes stem wood volume during a five-year period

Ranges (median values and the 5 and 95 percentile) of calculated changes in stem wood volume of trees that were: (i) alive in both surveys (forest growth), and (ii) dead or alive in the first survey and dead or alive or removed in the second survey (total stock changes) are given in Table 5.9. Results show an overall median change in total stock of 8.6 $m^3.ha^{-1}.yr^{-1}$, with median values varying between 3 and 12 $m^3.ha^{-1}.yr^{-1}$ for most tree species with notable exceptions like the fir (21.5 $m^3.ha^{-1}.yr^{-1}$) and the evergreen oak and remaining broadleaves (1.3 $m^3.ha^{-1}.yr^{-1}$). The results also show that the difference between changes in living standing stock and in total standing stock are either negligible or very limited.

Table 5.9 Ranges (medians, 5 and 95 percentile values) of calculated changes in stem wood of living trees (living standing stock and living, dead and removed trees (total stock changes).

Tree species group	Number of plots	Changes in living standing stock ($m^3.ha^{-1}.yr^{-1}$)		Changes in total standing stock ($m^3.ha^{-1}.yr^{-1}$)	
		Median	(5-95%)	Median	(5-95%)
Scots Pine	192	7.4	(1.1 - 15.4)	7.4	(1.1 - 15.4)
Spruce	154	11.4	(2 - 51)	11.2	(2 - 51)
Fir	16	21.5	(6 - 28)	21.5	(6 - 28)
High elevation conifers	5	3.3	(1.2 - 7.5)	3.3	(2.7 - 7.5)
Mediterranean pines	23	3.4	(0.5 - 19)	3.4	(0.5 - 19)
Remaining conifers	5	6.8	(5 - 26.3)	6.8	(5 - 26.3)
Mixed Conifers	24	10.7	(1.6 - 19.9)	10.7	(1.6 - 19.9)
Beech	72	10.1	(2 - 27.3)	10.1	(2 - 27.3)
Standard Oak	41	6.3	(2.2 - 13.7)	6.3	(2.2 - 13.7)
Oak evergreen	16	1.3	(0.3 - 12.1)	1.3	(.3 - 12.1)
Oak other	12	3.5	(0.3 - 12.7)	3.5	(.3 - 12.7)
Remaining broadleaves	3	1.3	(0.5 - 9.2)	1.3	(.5 - 9.2)
Mixed Broadleaves	39	8.6	(2.6 - 14.9)	8.6	(2.6 - 14.9)
Mixed	42	7.9	(0.7 - 19.7)	7.9	(0.7 - 19.6)
All	644	8.6	(1 - 25.9)	8.6	(1 - 25.4)

These high and low changes do correlate with high and low volumes of standing biomass (Compare Table 5.7 and 5.9). In general, there is a positive relation between the increase in tree volume in the five-year period and the standing tree volume as shown in Figure 5.8, although the correlation is limited.

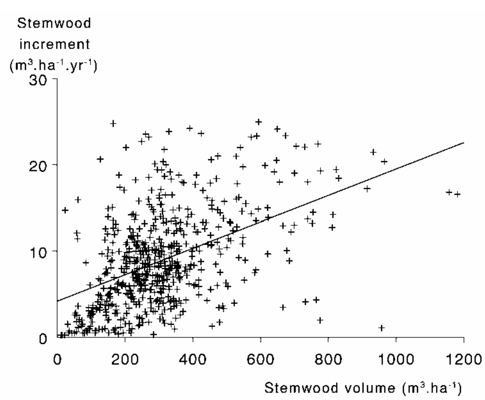


Figure 5.8 Relationship between stem wood increment in a five year period and the standing stem wood volume at the first survey

Carbon pool changes in trees during a five-year period

Ranges (median values and the 5 and 95 percentile) of calculated carbon pool changes in the living trees (forest growth), and total carbon pool changes (standing and removed trees) are given in Table 5.10. As with the results for the changes in stem wood volumes, the differences in carbon pool changes in living standing stock and in total standing stock are either negligible or very limited. The median carbon pool changes in both living and total standing stock vary mostly between 1000 and 2500 kg $C.ha^{-1}.yr^{-1}$ with the same notable exceptions, namely fir (4398 kg $C.ha^{-1}.yr^{-1}$) and the evergreen oak and remaining broadleaves (384 kg $C.ha^{-1}.yr^{-1}$). The overall median value is 2174 kg $C.ha^{-1}.yr^{-1}$ (Table 5.10)

Table 5.10 Ranges (medians, 5 and 95 percentile values) of calculated carbon pool changes in living trees and in living, dead and removed trees (total stock).

Tree species group	Number	Changes in living stock (kg $C.ha^{-1}.yr^{-1}$) Median (5-95%)		Changes in total stock (kg $C.ha^{-1}.yr^{-1}$) Median (5-95%)	
Scots Pine	192	1812	(280 – 3771)	1812	(280 – 3771)
Spruce	154	2199	(380 – 9625)	2199	(380 – 9625)
Fir	16	4398	(1236 – 6596)	4398	(1236 – 6596)
High elevation conifers	5	809	(288 – 2054)	809	(288 – 2054)
Mediterranean pines	23	838	(125 – 4663)	838	(125 – 4663)
Remaining conifers	5	1544	(1123 – 5963)	1544	(1123 – 5963)
Mixed Conifers	24	2379	(351 – 3980)	2379	(351 – 3980)
Beech	72	3428	(678 – 9205)	3428	(678 – 9205)
Standard Oak	41	1995	(646 – 4096)	1995	(646 – 4096)
Oak evergreen	16	385	(84 – 3937)	385	(84 – 3937)
Oak other	12	1136	(77 - 4117)	1136	(77 - 4117)
Remaining broadleaves	3	384	(186 - 2754)	384	(186 - 2754)
Mixed Broadleaves	39	2818	(830 – 5071)	2818	(830 – 5071)
Mixed	42	2273	(174 – 4325)	2263	(174 - 4253)
All	644	2174	(245 – 6073)	2160	(220 - 6073)

An overview of the geographic variation of changes in stem wood volumes and in carbon pools in stem wood is given in Fig. 5.9.

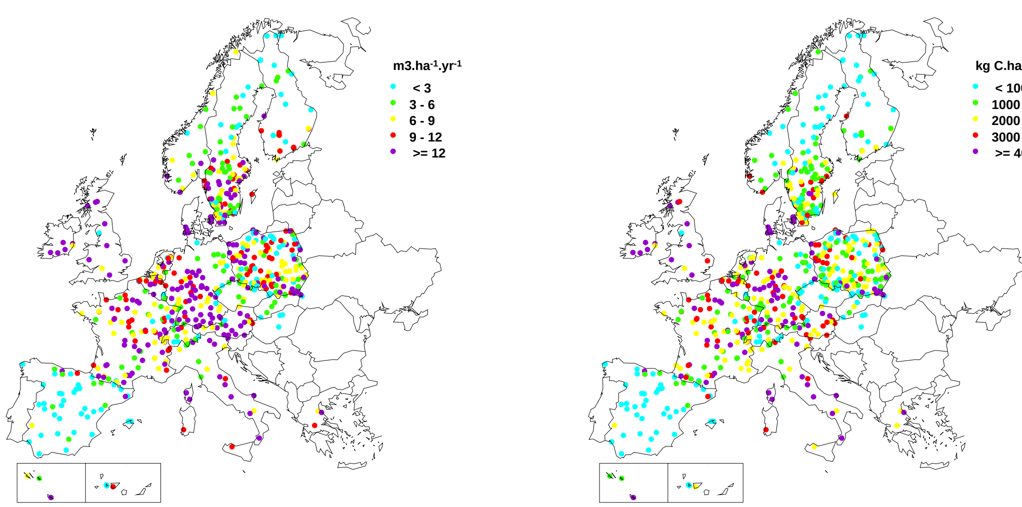


Figure 5.9 Map of changes in stem wood volumes (left) and carbon pools in stem wood (right) at 644 Intensive Monitoring plots

Results show a comparable pattern, with low changes in stem wood (< 3 $m^3.ha^{-1}.yr^{-1}$) and carbon pools in stem wood (< 1 ton $C.ha^{-1}.yr^{-1}$) in Northern and Southern Europe, due to temperature impacts (cold climate and water stress), and moderate (3-12 $m^3.ha^{-1}.yr^{-1}$ and 1-4 ton $C.ha^{-1}.yr^{-1}$) to high changes (>12 $m^3.ha^{-1}.yr^{-1}$ and >4 ton $C.ha^{-1}.yr^{-1}$) in Central Europe.

5.4 Discussion and conclusions

Comparison of country and FIMCI volume calculations

Using repeated data on tree diameter (at breast height) and tree height, changes in stem wood volume and the related carbon pools in stem wood, were calculated for 646 Intensive Monitoring plots. For most countries, carbon pool calculations were based on country submitted volumes, but for six countries volumes were calculated by FIMCI. Results showed that volume calculations carried out by FIMCI do compare well with country submitted volumes, except for plots where hardly any height measurements were available and plots with rare tree species. The volume calculations were carried out using elaborated polynomial equations with tree diameter (at breast height) and tree height as predictors (See Eq. 5.1, 5.2 and 5.4 and the Tables 5.3 and 5.4), based on a review of available equations for conditions in Central and Southern Europe. Simpler equations, calculating volume from direct relations with tree diameter only (e.g. Jenkins et al., 2003)**,** might have been used, but this choice was not made since tree height data were available for all plots, at least for part of the trees.

Comparison of calculated carbon pools with literature values

Apart from carbon pools in trees, carbon pools were also calculated for the soil by multiplying soil thickness with soil bulk densities and soil carbon contents. For the carbon pool calculations, measurements were used of the tree diameter, tree height, soil thickness and soil carbon contents, whereas estimates were used of form factors, stem wood density, soil bulk density and carbon contents in stem wood. Those estimates are rather robust; specifically the carbon contents in stem wood, but also the stem wood- and soil bulk densities and to a lesser extent the form factors. Regarding the stem wood densities used, the data compare generally well with those presented in overview papers focusing on carbon pool and carbon sequestration calculations for representative European forest ecosystems (Nabuurs and Schelhaas, 2002) and on a national scale (Jenkins et al., 2003). This implies that the estimated carbon pools are quite reliable.

Median soil C pools below the various tree species involved vary from approximately 50 - 200 $ton.ha^{-1}$ in the included 646 Intensive Monitoring plots. In general, this range is in line with literature results. For example, Nabuurs and Schelhaas (2002) gave exactly the same range in soil C pools for 16 typical forest types across Europe. Furthermore, average soil carbon pools in Intensive Monitoring plots in various countries appear to compare quite well with literature information on the country average carbon pools in soil. Examples, in which the first value is the average estimate derived for the monitoring plots and the second value is the country average value, are 63 versus 60 $ton.ha^{-1}$ for Finland (Liski and Westman, 1997), 121 versus 110 $ton.ha^{-1}$ for Germany (Baritz et al., 1999). In other countries, however, results deviate by a factor 1.5-2, such as 125 versus 74 $ton.ha^{-1}$ for Sweden (Lilliesköld and Nilsson, 1997), and 222 versus 103 $ton.ha^{-1}$ for Switzerland (Perruchoud et al., 1999). Nevertheless, these differences are still in line with deviations that one would expect when comparing country average values with average values of a limited number of plots. It only illustrates that the Intensive Monitoring plots are not always representative for the country as whole.

Regarding the C pools in tree stem wood, median values for the various tree species involved vary from approximately 10 - 100 ton.ha^{-1}. Results from dynamic modelling exercises, using data on stem wood volumes from forest resource information over Europe, leads to exactly the same range for the average tree carbon pools in the various countries over Europe in the year 1990 (Liski et al., 2002). The geographic variation of the tree carbon pools at the Intensive Monitoring plots also seems reasonably in line with the results obtained by Liski et al. (2002). In general, the country average carbon pools in stem wood based on results for Intensive Monitoring plots appear to slightly higher than the country average carbon pools presented by these authors. This is also true for the variation on a broad regional scale. Examples, in which the first value is the average estimate derived for the monitoring plots and the second value is the region average value obtained by Liski et al. (2002), are 51 versus 29 ton.ha^{-1} for Northern Europe (Finland, Sweden, Norway and Denmark), 60 versus 43 ton.ha^{-1} for North Western Europe (Ireland, UK, Belgium and the Netherlands) 93 versus 75 ton.ha^{-1} for Central Europe (France, Germany, Austria and Switzerland) and 41 versus 17 ton.ha^{-1} for Southern Europe (Spain, Portugal, Italy and Greece). These deviations in the values do indicate that the Intensive Monitoring plots are not representative for the various regions. Countries where the difference is small are e.g. Germany, Austria and Switzerland, in which the average values for both the monitoring plots and the country average values derived by Liski et al. (2002) are all near 100 ton.ha^{-1}. Large deviations, by a factor 2 or more do, however, occur for e.g. Ireland, Belgium, Spain, Italy and Greece. Possible differences in the type of trees (living or both living and dead) accounted for are not an explanation, since the difference in stem wood volume and related carbon pools of living trees and all (both living and dead) trees is negligible or marginal.

Comparison of calculated carbon pool changes with literature values
As with the carbon pools in standing biomass, the carbon pool changes are low in Northern and Southern Europe and moderate to high in Central Europe. The median carbon pool change in living trees at all plots equals 2175 kg.ha^{-1} with an overall variation of approximately 400 (evergreen oak) - 4500 kg C.ha^{-1}.yr^{-1} (fir). This is close to the maximum sequestration rates calculated by Nabuurs and Schelhaas (2002) after approximately 40 years for sixteen typical European forest types over Europe. These authors calculated an overall average value of 2980 kg.ha^{-1} varying between 1150 and 4100 kg.ha^{-1}, depending on the tree species. This result implies that the carbon pool changes at most plots are likely near their maximum. This is in line with the age class of the trees, which are mostly in the range of 40-60 years.

Possibilities for future research
Results presented in this chapter are just limited to a simple presentation of calculated changes in stem wood volume and related carbon pools, restricted to a five-year observation period. Interpretation in terms of expected changes in view of site quality and age class was not yet included. In the future, an in-depth analysis on the deviation in expected growth (based on standard growth curves for the plot) and the natural and anthropogenic growing conditions, such as stand and site characteristics, soil chemical variables, meteorology and atmospheric deposition, is worthwhile to be carried out. Such an analysis is presently carried out with results obtained for Austria, Bavaria, Czech Republic, France, and Switzerland The idea of this project, started in August 2002, is to: (i) use growth data from Intensive Monitoring plots in central European countries to parameterise a well defined basal area increment model and (ii) correlate the residuals, i.e. the ratio between observed and predicted basal area increment with data of environmental change, observed on a subset of these plots (Sterba, pers. Comm.). This approach might be well suited for all Intensive Monitoring plots.

Appendix 5.1 Available species at Intensive Monitoring plots for which forest growth data are available

Nr	Tree species	Nr	Tree species
1	Acer campestre	57	Salix alba
2	Acer monspessulanum	58	Salix caprea
3	Acer opalus	62	Salix sp.
4	Acer platanoides	63	Sorbus aria
5	Acer pseudoplatanus	64	Sorbus aucuparia
7	Alnus glutinosa	65	Sorbus domestica
8	Alnus incana	66	Sorbus torminalis
10	Betula pendula	68	Tilia cordata
11	Betula pubescens	69	Tilia platyphyllos
13	Carpinus betulus	70	Ulmus glabra
15	Castanea sativa	72	Ulmus minor
16	Corylus avellana	73	Arbutus unedo
17	Eucalyptus sp.	77	Erica arborea
18	Fagus moesiaca	81	Myrtus communis
20	Fagus sylvatica	82	Phillyrea latifolia
21	Fraxinus angustifolia	89	Ceratonia siliqua (same as 75)
22	Fraxinus excelsior	99	Other broadleaves
23	Fraxinus ornus	100	Abies alba
24	Ilex aquifolium	101	Abies borisii-regis
27	Malus domestica	103	Abies grandis
29	Ostrya carpinifolia	111	Juniperus communis
31	Populus alba	112	Juniperus oxycedrus
33	Populus hybrides	115	Juniperus thurifera
35	Populus tremula	116	Larix decidua
36	Prunus avium	117	Larix kaempferi
39	Prunus serotina	118	Picea abies
40	Pyrus communis	120	Picea sitchensis
41	Quercus cerris	122	Pinus canariensis
43	Quercus faginea	123	Pinus cembra
44	Quercus frainetto	124	Pinus contorta
46	Quercus ilex	125	Pinus halepensis
48	Quercus petraea	129	Pinus nigra
49	Quercus pubescens	130	Pinus pinaster
50	Quercus pyrenaica	131	Pinus pinea
51	Quercus robur	132	Pinus radiata
52	Quercus rotundifolia	133	Pinus strobus
53	Quercus rubra	134	Pinus sylvestris
54	Quercus suber	135	Pinus uncinata
56	Robinia pseudacacia	136	Pseudotsuga menziesii
		199	Other conifers

Appendix 5.2 Allocation of Available species at Intensive Monitoring plots for which forest growth data are available to main tree species (groups)

Main Tree Species Group	Species included[1]
Norway Spruce	Picea abies,
	Picea sitchensis Pseudotsuga menziesii
	Juniperus communis, Juniperus oxycedrus, Juniperus thurifera, Other conifers
Scots Pine	Pinus Sylvestris
Birch	Betula pubescens, Betula pendula = B. verrucosa
Beech and subsidiary broadleaves high forest	Fagus sylvatica, Prunus sp., Robinia sp., Sorbus sp.,Tilia sp
	Fagus moesiaca, Corylus avellana, Eucalyptus sp., Ilex aquifolium, Malus domestica, Pyrus communis, Sorbus aucuparia, Sorbus domestica, Sorbus torminalis, Arbutus unedo, Erica arborea, Myrtus communis, Phillyrea latifolia, Ceratonia siliqua, Other broadleaves
Beech coppice	Fagus sylvatica
Larch	Larix europaea
Fir	Abies alba
	Abies borisii-regis, Abies grandis
Swiss Pine	Pinus cembra
Oak (Quercus G)	Q. robur, Q. cerris, Q. petraea, Q. pubescens,
	Quercus rubra, Castanea sativa
Oak (Quercus M)	Q. frainetto, Q. ilex, Q. suber
	Quercus fagine, Quercus pyrenaica, Quercus rotundifolia
Black Pine	Pinus nigra
Stone pine	Pinus pinea
Maritime pine	Pinus pinaster
	Pinus canariensi, Pinus cembra, Pinus contorta, Pinus radiata, Pinus strobus, Pinus uncinata
Aleppo pine	Pinus halepensis and Pinus brutia
Hornbeam, Hophornbeam[2]	Carpinus (betulus), Ostrya sp.
Ash[2]	Fraxinus sp.
Maple[2]	Acer sp.
Elm[2]	Ulmus sp.
Alder[2]	Alnus sp.
White Poplar[2]	Populus alba
Black Poplar[2]	Populus nigra, Populus hybrides, Populus tremula,
Willow[2]	Salix sp.
Subsidiary broadleaves coppice	Acer sp.,Alnus sp., Betula sp., Carpinus sp., Fraxinus sp., Ostrya sp., Prunus sp., Robinia sp., Salix sp., Sorbus sp.,Tilia sp, Ulmus sp, Corylus avellana, Eucalyptus sp., Ilex aquifolium, Malus domestica, Pyrus communis, Sorbus aucuparia, Sorbus domestica, Sorbus torminalis, Arbutus unedo, Erica arborea, Myrtus communis, Phillyrea latifolia, Ceratonia siliqua, Other broadleaves

1 Tree species given in the first line were included in the allocation by Sterba (for subsidiary broadleaves coppice the first two lines), whereas the species given in the following lines were allocated to this species group by FIMCI.
2 For these tree species, the different equations all refer to stands managed under the high forest system.

6 Impacts of nitrogen deposition on carbon sequestration by forests in Europe

6.1 Introduction

The importance of assessing the terrestrial carbon sink

It is of importance to arrive at reliable estimates of CO_2 sequestration in forests since this may delay the rise in the atmospheric CO_2 concentration with implications for the speed of climate change. In the Kyoto Protocol, signed by 84 nations and ratified by 22 nations until January 2000, governments agreed to reduce emissions of CO_2 either by limiting fossil fuel consumption or by increasing net carbon sequestration in terrestrial sinks through afforestation and land use change or both. Even though increasing net carbon sequestration is still limited to strictly defined cases of afforestation and land use change, it has been advocated (IGBP Terrestrial Carbon Working Group, Steffen et al., 1998) to use a full carbon budget, including all potential terrestrial sinks over a sufficiently long time period, to be accounted for in international CO_2 emission reductions. This requires methods for reliable quantification of these C sinks.

The possible impact of nitrogen deposition on carbon sequestration in forests

Important questions with respect to carbon sequestration are related to the cause of the large uptake of the mid-latitude forests and the time period in which the terrestrial sink will be saturated (Houghton et al., 1998). European forests have a role in net carbon sequestration of the biosphere (i.e. Kauppi et al., 1992; Nabuurs et al., 1997). Apart from changes in standing growing stock (influenced by forest management), changes in net primary productivity may also play a role in this respect (Spiecker et al., 1996). Increased net primary productivity have been hypothesised to be due to increases in atmospheric CO_2 concentrations (e.g. Melillo et al., 1993; Friedlingstein et al., 1995), nitrogen deposition (Holland et al., 1997; Nadelhoffer et al., 1999) and temperature, increasing the growing season (e.g. Myneni et al., 1997). Increase in CO_2-concentrations on the other hand may favour growth as well as increase water use efficiency of trees. However, trees may adapt to changing CO_2-concentrations and the effect may diminish soon (Tognetti et al., 2000). Using a modelling approach, temperature has been claimed to be relatively unimportant, whereas the combination of CO_2 rise and elevated N deposition may account for a 15-20% increase in forest net primary productivity (Rehfuess et al., 1999). In this context, N deposition is claimed to be most important (Rehfuess et al., 1999). The remaining explanation would then be the impact of forest management.

Furthermore an elevated carbon sequestration in the soil, due to an increased accumulation of soil organic matter in response to elevated N inputs, may play a role. By far the largest amount of C stored in forests in the northern hemisphere is stored in the soil. Carbon fixed by photosynthesis ultimately moves via litter fall to the soil, where it is only partially decomposed. Thus, over the long term the soil is the ultimate sink or source of CO_2 for these ecosystems. Nitrogen is often the limiting nutrient in terrestrial ecosystems, and thus sequestration of C is closely linked to the N cycle. Soil processes account for the most significant unknowns in the C and N cycle. Current hypotheses suggest that increased N deposition causes an increased rate of soil organic matter accumulation at least in two ways due to an increased leaf/needle biomass and litter production (e.g. Schulze et al., 2000) and a reduced decomposition of organic matter (Berg and Matzner, 1997; Harrison et al., 2000) The N-content of forest litter and humus might thus be an important indicator of the C-sequestration. Understanding the N cycle in semi-natural ecosystems is therefore the key to understanding the long-term source or sink strength of soils for carbon.

Since nitrogen often is the limiting nutrient in forests, nitrogen deposition may increase wood production and accumulation of soil organic matter, thus increasing carbon sequestration into the forest. Earlier estimates suggested that this mechanism could take up one third of the global CO_2 emission from fossil fuel (or 2 x 10^{15} $g.yr^{-1}$) if most of the deposition nitrogen was taken up by trees and used to form new woody biomass (Holland et al., 1997). Recent data on the distribution of deposition nitrogen between trees and soil (Nadelhoffer et al., 1999), however, suggest that a large part of the nitrogen is accumulated in the soil at low carbon to nitrogen ratio (10-40) and not in the trees at carbon to nitrogen ratio (200-500). Thus the increase in nitrogen deposition may cause a much smaller additional CO_2 sequestration in forests (0.25 x 10^{15} $g.yr^{-1}$). This issue is a matter of ongoing scientific debate (e.g. Jenkinson et al., 1999; Schindler, 1999; Sievering, 1999) and continued research and deserves further attention as described in this background document. When the large uptake is mainly due to elevated growth, it is likely that this is a short transitory phenomenon, whereas it could be a carbon sink for a long period if soil accumulation is the main cause, since below ground carbon has much lower turnover times than above ground carbon.

Upscaling of carbon sequestration research in forests to the European scale

Information on C and N sequestration and their response to changes in N deposition on a European scale is presently still limited. The EUROFLUX project provided measurements of C fluxes above a range of forests across Europe (Thenhunen et al., 1998), but extrapolation of these results to a European scale is still prone to large uncertainties. The aboveground CO_2 sequestration in the trees can also be estimated from yield tables and models on tree growth. Furthermore, the net C sequestration can be based on repeated forest surveys (e.g. Kauppi et al., 1992; Nabuurs et al., 1997). But the CO_2 sequestration in forest soils is difficult to estimate from direct measurements at European or global scale. Recent EU projects (EXMAN, NITREX, NIPHYS, CANIF) have increased the understanding of controls in the N and C cycle in forests (e.g. Berg and Matzner, 1997; Dise et al., 1998a; Dise et al., 1998b; Gundersen et al., 1998a). However, the detailed process understanding has not yet led to major improvements on the ability to predict and extrapolate such impacts and to assess C sequestration on a European scale. Simulation models and local data are available for a number of sites, but extrapolation to a European scale has not yet taken place.

An example of upscaling forest ecosystem research to the continental and global scale (although based on rough generalisations) is the publication by Nadelhoffer et al. (1999). They calculated additional C sequestration on a global scale from additional N uptake by trees and N immobilisation in soils in response to N deposition. From their estimate, the authors conclude that C sequestration in forest trees and forest soils over the world is of equal magnitude. This estimate is, however, based on an estimated total world N deposition, averages values for the C/N ratio in stem wood and forest soils and constant N retention fractions in both compartments, based on the short-term fate (1-3 yr) of 15N labelled tracer experiments in nine temperate forests (Nadelhoffer et al., 1999). The upscaling of the results to a European scale (assumed constant N retention fractions and C/N ratios in stem wood and soil independent of the location) is extremely simple, thus hampering an adequate estimate on this large scale. The available data at level II and level 1 plots do allow for much better estimates as described below.

Contents of this chapter

This study presents an estimate of: (i) the current carbon sequestration in trees and soil in European forests for the year 2000, based on simple assumptions and empirical knowledge of the interaction of carbon (C) cycles and nitrogen (N) cycles in forests, and (ii) the likely impact of N deposition on the C sequestration rates in the last 40 years (the period 1960-2000). In making the calculations, use is made of data on N retention, N uptake and C/N ratios in soils, that were: (i) available at more than 100 Intensive Monitoring (level II) plots and (ii) estimated (N deposition, N uptake) and measured (C/N ratios) at more than 6000 Level 1 plots at a systematic rid, statistically representing approximately 2 million km^2 of forests in Europe (including part of Russia).

In Section 6.2, we describe the approaches that were used to calculate carbon sequestration both at Intensive Monitoring plots and at the European scale using Level I plot data. This includes a literature review of methods and results related to the assessment of the terrestrial carbon sink (Section 6.2.1) and the methods to calculate carbon sequestration at Intensive Monitoring plots (Section 6.2.2), extrapolate carbon sequestration to the European forested area (Section 6.2.3) and assessing nitrogen deposition impacts on the carbon sequestration for that area (Section 6.2.4). Results are described in Section 6.3. This again includes the carbon sequestration in soils at Intensive Monitoring plots (Section 6.3.1) and on the European scale (Section 6.3.2) and the impact of nitrogen deposition on carbon sequestration (Section 6.3.3). Finally, a discussion of the results and conclusions is presented in Section 6.4.

6.2 Methods

6.2.1 Methods and results related to the assessment of the terrestrial carbon sink

Measurements of atmospheric CO_2 indicate that from the estimated 7.1 Gton C released by man (5.5. fossil fuel and 1.6 from land use change and deforestation) only 3.4 Gton is found back in the atmosphere. From this, an estimated 1.5-2.0 Gton is being absorbed by the oceans (Bousquet et al., 1999). The remaining 1.5-2.0 Gton would be global terrestrial uptake (Ciais et al., 1995) but this estimate is also commonly referred to as the missing sink. Studies using global inversion models indicate that a significant portion of the net uptake of the terrestrial biosphere occurs at northern mid-latitude forest regions (Ciais et al., 1995; Fan et al., 1998; Bousquet et al., 1999).

Up to now, several studies have been carried out, to assess carbon sequestration in forests in Europe, but a direct comparison is hampered because of the measurement of different carbon sink terms. First of all, there is a difference in the assessment of the so-called net ecosystem production (NEP) or net ecosystem exchange (NEE), and the net biome production (NBP). The NEP or NEE stands for the total uptake of CO_2 by photosynthesis, corrected for plant and soil respiration, whereas the NBP equals the NEP corrected for CO_2 emissions due to harvest and forest fires. The latter term is critical with respect to long-term carbon storage, since an aggrading forest may temporarily sequester large carbon amounts, but most of it is re-emitted to the atmosphere after logging. Secondly, a distinction can be made in sequestration in the trees and in the soil. Over the long term, the soil is the ultimate sink or source of CO_2 for these ecosystems.

An overview of various estimates of the carbon sequestration in Europe, focusing on different ecosystem compartments and using different methods is given in Table 6.1. Apart from a distinction in the type of flux and the forest compartment, a differentiation has been made in the quality of the upscaling methods, going from individual sites to the European scale. A systematic discussion related to the various approaches and results is given below.

Table 6.1 Overview of different estimates of carbon sequestration on a European wide scale.

Type of C flux	Compartment	Method	Estimated sink Gton. yr^{-1}	Upscaling method	Reference
NBP landscape					
NBP	Landscape	Inversion modelling	0.30	Good	Bousquet et al. (1999)
NEE/NEP Whole forest/trees					
NEE	Whole forest	CO_2 net flux measurements	0.47	Neural networks	Papale and Valentini (2003)
			0.25	Forest maps	Martin et al. (1998)
NEP	Total above-ground biomass	Tree growth measurements	0.39-0.53[1]	Multiply with forested area	Schulze et al. (2000)
NBP whole forest/trees					
NBP	Trees (stem wood)	Repeated forest Inventories	0.10	Country inventory data	(Kauppi et al., 1992) (Nabuurs et al., 1997)
NBP	Trees (stem wood)	Modelling forest growth	0.08-0.12[2]	Country inventory data	Liski et al. (2002)
NEP contribution	Trees (above-ground biomass)	N retention	0.025[3]	World average values	After Nadelhoffer et al. (1999)
NBP forest soil					
NBP	Forest soil (below-ground biomass)	Carbon soil input minus carbon mineralisation	0.13	Multiply with forested area	Schulze et al. (2000)
NBP	Forest soil (below-ground biomass)	Modelling forest growth and decomposition	0.038-0.061[2]	Country inventory data	Liski et al. (2002)
NBP	Forest soil (below-ground biomass)	N retention	0.022[3]	World average values	After Nadelhoffer et al. (1999)

[1] The first estimates was derived by Schulze et al. (2000) based on a forested area in Europe of approximately 150 million ha, whereas the second estimate is based on an area of 200 million ha, used in this study
[2] These estimates were originally limited to the EU + Norway and Switzerland (approximately 138 million ha) but results were scaled to the European forested area, excluding most of Russia (approximately 200 million ha)
[3] These estimates were originally global but were scaled to the European N deposition and forest area. Actually , Nadelhoffer et al. (1999) also estimated a NEP of 0.025 for carbon sequestration in trees but this was presented as the contribution of N deposition to NEP in trees and not the total growth

Inversion modeling: Bousquet et al. (1999) estimated a carbon sink of 0.3 Gton C yr^{-1} for Europe used a global inversion model including data on regional CO_2 emissions and tropospheric CO_2 concentrations. This is the main approach used up to now to assess C sinks on a regional scale, since it includes regionally distributed data and models. Similarly, Bousquet et al. (1999) estimated a C sink in North Asia of 1.5 Gton C yr^{-1} and of 0.5 Gton C $year^{-1}$ in the Northern United States and Canada. Those models do not differentiate between forests and other land use types. In the Arctic and tropical Asia a net release of respectively 0.2 and 0.8 Gton C $year^{-1}$ was estimated.

NEE/NEP estimates of whole forests or trees from CO_2 net flux and tree growth measurements
An NEP estimate related to forests only is based on direct measurement of the net CO_2 exchange flux to the forest ecosystem at seventeen so-called EUROFLUX sites along a transect from North Sweden to Central Italy (Valentini et al., 2000). Tree species included were Norway spruce and beech. Results indicate that most forests act as sinks at present, and sequester CO_2 at an average rate of 3.03 $ton.ha^{-1}.yr^{-1}$. Scaling these results to the level of the continent remains, difficult. Recently, Papale and Valentini (2003), used the net CO_2 exchange flux collected in the EUROFLUX network at sixteen of these sites to train a neural network to provide spatial (1 x

1km) estimates of carbon fluxes of European forests. By using this approach, they estimated the total NEE to equal 0.47 Gton C yr^{-1}. This is almost equal to an estimate that can be derived by simply multiplying the average net CO_2 exchange flux of 3.03 $ton.ha^{-1}.yr^{-1}$ with the forested area 149 million ha of forests, which is generally used as an estimate for forests in Europe excluding Russia (Nabuurs et al., 1997), that would lead to an NEE of 0.45 Gton C yr^{-1}. Earlier, Martin et al. (1998), estimated that only between 0.17 and 0.31 Gton C yr^{-1} was sequestered by European forest in 1997, using an upscaling technique with forest maps, based on net CO_2 exchange fluxes ecosystem at eleven EUROFLUX sites. In both approaches, on an aerial basis the net sequestration was largest in Central Europe and lowest in Northern Europe, with Southern Europe in between.

At 11 forest sites, two of them overlapping with the Euroflux sites (so called Canif sites), the current carbon sequestration by tree growth or NEP (by trees), based on process studies and inventories, was estimated to equal 2.64 $ton.ha^{-1}.yr^{-1}$ (Schulze et al., 2000). Schulze et al. (2000) multiplied this value by 149 million ha of forests, to estimate an NEP of 0.39 Gton C yr^{-1} for Europe. Using a forested area of 200 million ha, applied in this study, it would lead to a sink of 0.53 Gton C yr^{-1}. Apart from the still relatively poor upscaling procedures, it should be noted that data on the present sequestration in the trees by uptake (and the same holds for the present CO_2 exchange) do overestimate the net carbon sink, as this approach does not account for C release after disturbances (NEE or NEP is larger than NBP).

NBP assessments for trees from repeated forest inventories and modelling forest growth
The net increase in carbon in forests (NBP) can be derived from repeated forest inventories on the standing biomass. Such data do indicate an increase in the period between 1970-1990 of 25% (Kauppi et al., 1992) leading to a net NBP in trees of approximately 0.1 Gton C yr^{-1}. A similar value was obtained by Nabuurs et al. (1997), using much more detailed information on forest inventories in most countries within Europe.

Liski et al. (2002) gave an estimate of the net carbon sequestration in trees based on a dynamic modelling exercises, using data on stem wood volumes from forest resource information over Europe. The growth of branches, foliage and roots is included by an additional allocation of dry matter increment, relative to the known stem wood increment data. The model was applied to the EU countries including Norway and Switzerland. The net carbon sequestration in trees was estimated at 390-600 $kg.ha^{-1}.yr^{-1}$ in 1990 and at 440-510 $kg.ha^{-1}.yr^{-1}$ in 2040. Considering the forested area of the included countries (138 million ha) this leads to a net carbon sequestration of 0.054-0.082 $Gton.yr^{-1}$ in 1990 and of 0.062-0.070 $Gton.yr^{-1}$ in 2040. Assuming that the average carbon sequestration is equal in the forests that are not considered, the net carbon sequestration equals 0.078-0.12 $Gton.yr^{-1}$ in 1990 and of 0.088-0.102 $Gton.yr^{-1}$ in 2040 for a forested area of 200 million ha.

A comparable model was used by Nabuurs and Schelhaas (2002) who calculated the net carbon sequestration in trees for 16 typical forest types across Europe. The advancing mean of the net sink of all forests was calculated to equal 800 kg $C.ha^{-1}.yr^{-1}$. Multiplication of this amount by the European forested area is not allowed, since the calculations are just mean to give indicative values for representative forest types. If one, however, simply multiplies this average value with a forested area of 200 million ha, it would lead to a net carbon sequestration of 0.16 $Gton.yr^{-1}$.

NEP assessments for soil from carbon cycling measurements and modelling soil C dynamics

As with CO_2 sequestration in tree, the retention in forest soils can be derived from repeated soil inventories, but those data are hardly available. An example of results thus obtained is presented in Leeters and de Vries (2001), but the results show that the change is hard to detect within a short period of time, considering the large present pools with the possible exception of the organic layer (see also De Vries et al., 2000). One can also estimate the net C sequestration in the soil from direct measurements of the carbon input to the soil by litterfall and root decay and carbon release by mineralization, but this approach is again hampered by the fact that the result is based on subtracting large numbers with relative high uncertainties. Such an approach was used by Schulze et al. (2000) at eleven "Canif" sites, mentioned above. These authors estimated an average C accumulation in soils of 0.86 ton C yr^{-1}. By simply multiplying this figure with 149 million ha of forests, they calculated a sink of 0.128 Gton C yr^{-1} at the European scale. Using a forested area of 200 million ha, as consistently applied in this study, it would lead to a sink of 0.172 Gton C yr^{-1}.

Apart from net carbon sequestration in trees, Liski et al. (2002) also gave an estimate of the net carbon sequestration in soil, based on the dynamic modelling exercise described before for the EU countries including Norway and Switzerland. The net carbon sequestration in soil was estimated at 190 $kg.ha^{-1}.yr^{-1}$ in 1990 and at 305 $kg.ha^{-1}.yr^{-1}$ in 2040. Considering the forested area of the included countries (138 million ha) this leads to a net carbon sequestration of 0.026 $Gton.yr^{-1}$ in 1990 and of 0.043 $Gton.yr^{-1}$ in 2040. Assuming that the average carbon sequestration is equal in the forests that are not considered, the net carbon sequestration equals 0.038 $Gton.yr^{-1}$ in 1990 and of 0.061 $Gton.yr^{-1}$ in 2040 for a forested area of 200 million ha. Nabuurs and Schelhaas (2002) also calculated the net carbon sequestration in soil for 16 typical forest types across Europe. The advancing mean of the net sink of all forests was calculated to equal 110 kg $C.ha^{-1}$.

Carbon sequestration derived from N retention

A completely different approach compared to all the former approaches is related to the possibility to assess C sequestration from N uptake by trees and N immobilisation in soils in response to N deposition. First estimates based on this approach suggested that this mechanism could take up one third of the global CO_2 emission from fossil fuel (or 2 Gton C yr^{-1}), being equal to the missing carbon sink (Holland et al., 1997). In this approach most of the deposition nitrogen was assumed to be taken up by trees to form new woody biomass. The assumption was that carbon and nitrogen accumulate in organic matter at the same relative rates through the same mechanisms. This means that nitrogen saturated forests with low nitrogen retention will have nearly no CO_2 sequestration in the soil.

Recent data on the distribution of deposition nitrogen between trees and soil, however, suggest that a large part of the nitrogen is accumulated in the soil at a low carbon to nitrogen ratio and not in the trees at a high carbon to nitrogen ratio. These results are based on the short-term fate (1-3 yr) of ^{15}N labelled tracer experiments in nine temperate forests (Nadelhoffer et al., 1999). Using a total world N deposition estimate of 5.1 $Mton.yr^{-1}$, average N retention fractions in stem wood (0.05) and in the soil compartment (0.7) and averages values for the C/N ratio in stem wood (500) and forest soils (30), these authors thus came to a ten times lower global estimate, than estimated by Holland et al. (1997). In Table 6.1 the estimates by Nadelhoffer et al. (1999) have been scaled to an estimated N deposition for Europe of 1.1 $Mton.yr^{-1}$. The results suggest that the sinks in forest trees and forest soils are of equal magnitude.

This estimate by Nadelhoffer et al. (1999) of the C sequestration gave rise to statements about the "mysterious" missing carbon sink (Schindler, 1999), since it would imply that forest are not responsible for the net uptake of the missing 1.5-2.0 Gton of CO_2 in the atmosphere. The estimated above-ground carbon sequestration is, however, likely to be underestimated since the authors neglected the effect of direct foliar uptake (Jenkinson et al., 1999; Sievering, 1999). The repeated forest inventory data by Kauppi et al. (1992) and Nabuurs et al. (1997) for Europe, leading to a net NBP in trees of approximately 0.1 Gton C yr^{-1} is already a strong indication for this. The problem in this discussion, however, is that Nadelhoffer et al. (1999) focused in principle on the additional C sequestration in response to N deposition and not on the total sequestration, as discussed further in Section 6.2.4.

6.2.2 Calculation of carbon sequestration in soils at Intensive Monitoring plots

An estimate of net C sequestration in Intensive Monitoring plots was based on the calculated nitrogen immobilisation (sequestration) in the soils, multiplied by the C/N ratio of the forest soils, distinguishing between the organic layer (forest floor) and mineral soil. As stated before, the basic assumption is that CO_2 sequestration can be calculated from nitrogen retention in the soils since carbon and nitrogen accumulation in organic matter occurs through the same mechanisms. N immobilisation (sequestration) was calculated as:

$$\text{N immobilisation} = \text{N deposition} - \text{N leaching} - \text{N uptake} \quad (6.1)$$

This approach is based on the assumption that denitrification can be neglected in the organic layer and the mineral topsoil, where both N and C sequestration is assumed to occur. Figure 6.1 shows the calculated N retention (N deposition minus N leaching) and N uptake for the Intensive Monitoring plots for which carbon pool changes in trees and soil were calculated (Fig 6.1). This included the plots with information on: (i) both bulk deposition and throughfall of N, thus allowing the calculation of total N deposition, and (ii) soil solution chemistry, thus allowing the calculation N leaching. Such budgets were only available for 124 plots, due to the limited availability of soil solution chemistry data. The budgets are an update of those described in De Vries et al. (2001) by including two additional years (the period 1995-2000). The plots were located in Belgium, France, Denmark, Germany, UK, Ireland, Norway, Sweden, Finland and Austria (Fig. 6.1). The actual N uptake was derived by multiplying changes in standing biomass in terms of stem wood, from repeated growth surveys in the period 1995-2000, as described in Chapter 5 with deposition dependent N contents in biomass. The uptake in branch wood was thus neglected. The results show that N uptake systematically increases going from Northern to Southern Europe while the N retention generally follows this pattern. In nearly all cases total retention (equal to uptake, denitrification and soil immobilisation) is larger than uptake implying that N is immobilised in the soil.

In multiplying the net N immobilisation with the C/N ratio, the variation of the C/N ratio with the depth of the soil profile must be accounted for. Especially there is often a large difference between C/N ratio in the organic layer (forest floor) and in the mineral soil. The retention of N in those layers is dependent on the transport of the mineral N input down the profile. From experiments simulating increased nitrogen deposition it is shown that nitrate is much more mobile than ammonium (Nadelhoffer et al., 1995; Emmet et al., 1998) and some transport of nitrate down the profile even occur at sites with high C/N in the forest floor (Gundersen and Rasmussen, 1995; Moldan et al., 1998).

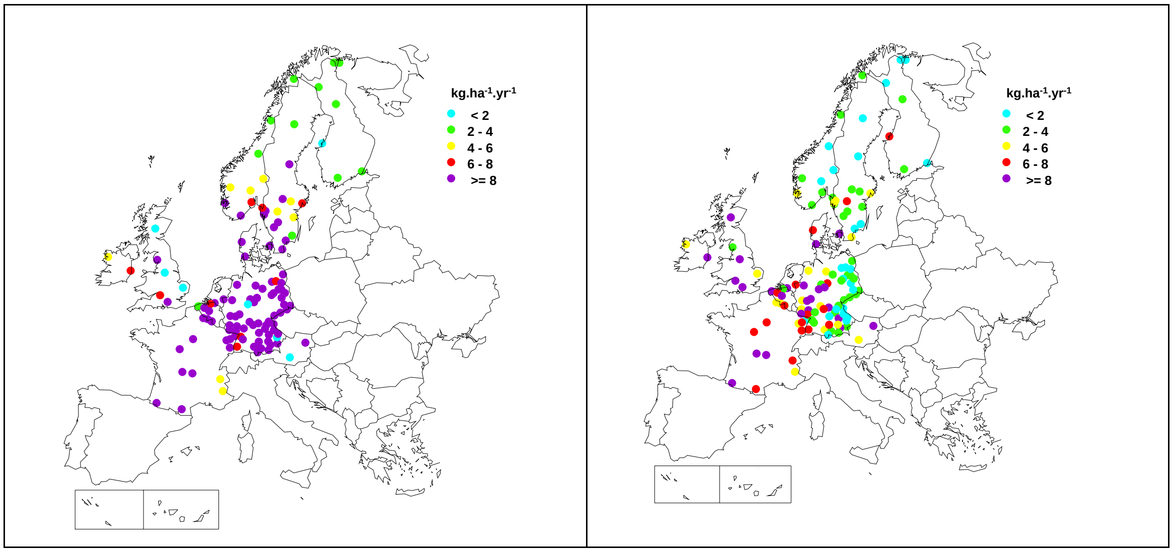


Figure 6.1 Nitrogen retention (N deposition minus N leaching; left) and N uptake (right) at the 121 Intensive Monitoring plots (kgN.ha^{-1}.yr^{-1}) that were used for the calculation of carbon sequestration in soils.

The fate of deposition N in forest soils has been studied by nitrogen tracer (15N) technique. In an ammonium-nitrate addition study by Nadelhoffer et al. (1999) in oak and pine stands nitrate and ammonium were labelled separately. Spraying of a NH_4NO_3 solution monthly in the growing season was performed to simulate deposition of 58 kg N/ha/yr. Results of the experiment are shown in Table 6.2. Most of the retained ammonium (80%) was found in the forest floor, whereas the retention of nitrate was about equal in the forest floor and the first 20 cm of the mineral soil. The data in Table 6.2 show a higher retention of both N compounds in the red pine stand, which had a higher C/N ratio than the oak stand. This indicates that at higher C/N ratio, which determines the sink strength of the forest floor, less N is transported down to the mineral soil.

Table 6.2 The fraction of added N (ammonium or nitrate) retained in the forest floor relative to the total soil N retention of the compound in both the forest floor and the mineral soil (0-20 cm) in two stands at Harvard Forest, USA. (Nadelhoffer et al., 1999).

Forest stand	N Labelling	Fraction N retained in the forest floor
Red Pine	Ammonium	0.82
(C/N = 26)	Nitrate	0.58
Oak	Ammonium	0.78
(C/N = 23)	Nitrate	0.52

Within the NITREX project, Tietema et al. (1998) performed 15N tracer studies with various combinations of input levels (current deposition and simulated increase or decrease of deposition) and dominating N compounds (NH_4-fractions from 0.1 to 0.8). Based on the fate of N added over one year, 40 to 75 % of soil N retention occurred in the forest floor. The lowest percentages were found at the highest nitrate depositions rates (lowest NH_4-fraction). In nitrifying soils (lower C/N ratio soils) labelled NH_4 may over time be transformed to nitrate and leached down the profile, which makes the interpretation of these numbers difficult. Based on these results we modelled the partitioning of N retention between forest floor and mineral soil as a function of the N input and the C/N ratio of the forest floor according to:

$$\text{C sequestration} = \text{N immobilisation} \cdot (fret_{ff} \cdot C/N_{ff} + (1 - fret_{ff}) \cdot C/N_{ms}) \qquad (6.2)$$

Where C/N_{ff} and C/N_{ms} are the C/N ratios of the forest floor and the mineral soil (up to a depth of 20 cm), and $fret_{ff}$ is the N retention fraction in the forest floor, being the ratio of the N retention in the forest floor and the N retention in the complete soil profile (forest floor and mineral soil). The N retention fraction in the forest floor was calculated as a function of the NH_4-fraction in the N input and the C/N ratio of the forest floor, by multiplication of two factors t and r according to:

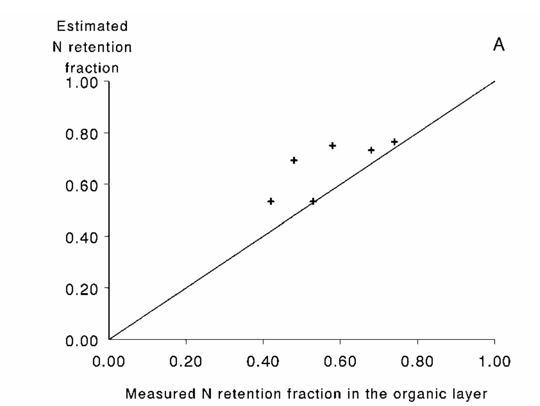
$$fret_{ff} = t \cdot r \tag{6.3}$$

With t and r are being values depending upon the NH_4-fraction in the N input and the C/N ratio of the forest floor, according to:

$$\begin{aligned} &t = 0.5 && \text{if } NH_4 \text{ fraction} < 0.5 \\ &t = NH_4 \text{ fraction} && \text{if } 0.5 < NH_4 \text{ fraction} < 0.75 \\ &t = 0.75 && \text{if } NH_4 \text{ fraction} > 0.75 \end{aligned} \tag{6.4}$$

$$\begin{aligned} &r = 1.0 && \text{if C/N ratio} < 20 \\ &r = 1.0 + 0.033 \cdot (\text{C/N ratio} - 20) && \text{if } 20 < \text{C/N ratio} < 30 \\ &r = 1.33 && \text{if C/N ratio} > 30 \end{aligned} \tag{6.5}$$

A comparison of calculated N retention fractions in the forest floor for sites and treatments included in Tietema et al. (1998) and the observed partitioning from the tracer experiments is presented in Fig. 6.2A. The figure shows a reasonable comparison, but the simple relationships may give a slight overestimation of the fraction of N retained in the forest floor. Values for the N retention fraction in the forest floor thus calculated for the intensive monitoring plots considered are presented in Fig 6.2B. In general the N retention fraction is higher than 50%. The C/N ratio of both organic layer and the mineral topsoil for the intensive monitoring plots is given in Figure 6.3. The figure shows that generally, the C/N ratios of the forest floor are much higher than in the mineral soil. Specifically in the Nordic countries the differnce can be large with C/N ratios in the organic layer often being higher than 35 and in the mineral layer varying between 20-30 (compare Fig. 6.3 left and right).



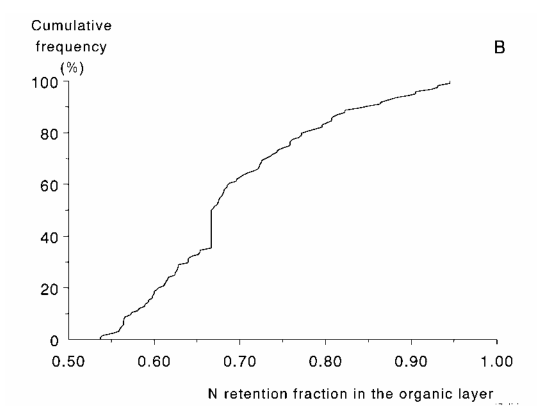


Figure 6.2 Comparison of predicted and measured N retention fractions in the organic layer of six forest plots (A) and predicted N retention fractions in the forest floor of the 121 Intensive Monitoring plots (B)

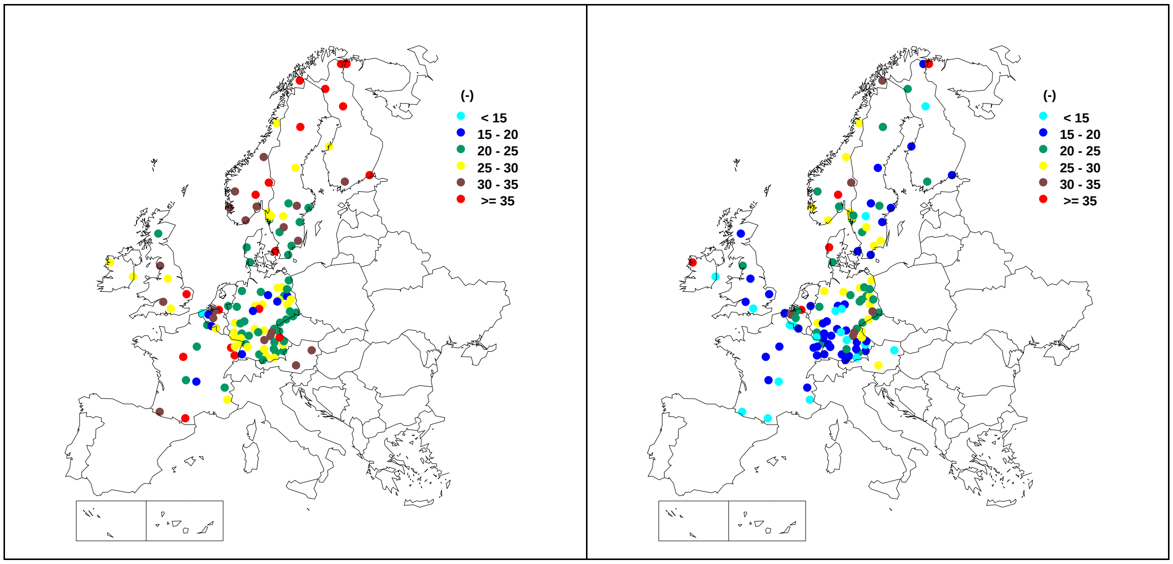


Figure 6.3 *C/N ratios in the organic layer (left) and mineral topsoil (right) at the 121plots that were used for the calculation of carbon sequestration in soils.*

6.2.3 Extrapolation of carbon sequestration to the European forested area

In order to scale up results to the European scale, an estimate of net C sequestration for more than 6000 forest soils located in a systematic grid of 16 km x 16 km (level I plots) was made, being representative for approximately 2,0 million km^2 for Forests in Europe, including part of Russia (each plot represents approximately 256 km^2). The assumed representative forest area of each grid cell in a country was scaled to the total forested area in each country given in the Annexes of the executive reports of ICP forests. As with the Intensive Monitoring plots, the calculation was based on calculated nitrogen retention in the soils, multiplied by the C/N ratio of the forest soil considered (see Eq. 1). N immobilisation (sequestration) was now calculated as a fraction of the N deposition corrected for N uptake, according to:

$$\text{N immobilisation} = \text{frN}_{\text{im}} \cdot (\text{N deposition - net N uptake}) \qquad (6.6)$$

The fraction frN_{im} was calculated as a function of the C/N ratio of the forest soil and the fraction NH_4 in deposition, using presently available results on this relationship given in e.g. Matzner and Grosholz (1997), Dise et al. (1998a; 1998b) and Gundersen et al. (1998a) and newly derived results from the Intensive monitoring plots. This relationship was derived by plotting the relation between N retention/N deposition and C/N ratio for several plots for which those data are available, including the Intensive Monitoring plots.

A relationship between the output (nitrate leaching)/input (throughfall N) ratio and C/N ratios in the forest floor in more than 30 forest conifer plots is presented in Figure 6.4 A (after Gundersen et al., 1998a). It should be noted that the estimates of output are relative uncertain and that throughfall N underestimate the total N input due to canopy uptake. N leaching appears to be negligble at all sites with an N input < 10 kgN/ha/yr. Based on those data, Gundersen et al. (1998a) presented a range in retention fractions as a function of the N status of the ecosystem, including C/N ratios, as given in Tabel 6.3

Table 6.3 An overview of ranges in N retention fractions as a function of the N status of the ecosystem based on results from Gundersen et al. (1998a; 2003) and De Vries et al. (2001).

Nitrogen status	Low (N limited)	Intermediate	High (N saturated)
Input (kg $N.ha^{-1}.yr^{-1}$)	0-15	15-40	40-100
Needle N%	< 1.4	1.4-1.7	1.7-2.5
Soil N flux density (litterfall + throughfall) (kg $N.ha^{-1}.yr^{-1}$)	< 60	60-80	>80
C/N ratio (g $C.g\ N^{-1}$)	> 30	25-30 or 20-30[1]	< 25 or <20[1]
Proportion of input retained (%)	>90	40-100	0-70

[1] the first criterion is based on Gundersen et al. (1998a) and the second on De Vries et al. (2001).

The reliability of the suggested N retention fractions as a function of C/N ratio is only partly substantiated by the results of the Intensive Monitoring plots as presented in Figure 6.4 B. The results show indeed that N retention is nearly complete (above 90%) at C/N ratios above 30-35 and very low at low C/N ratios (below 20) but in between it is highly variable. However, the Intensive Monitoring data include many sites with low input and low C/N (e.g. boreal forests in Scandinavia) that exhibit full retention simply because the input is low as illustrated in Dise et al. (1998b).

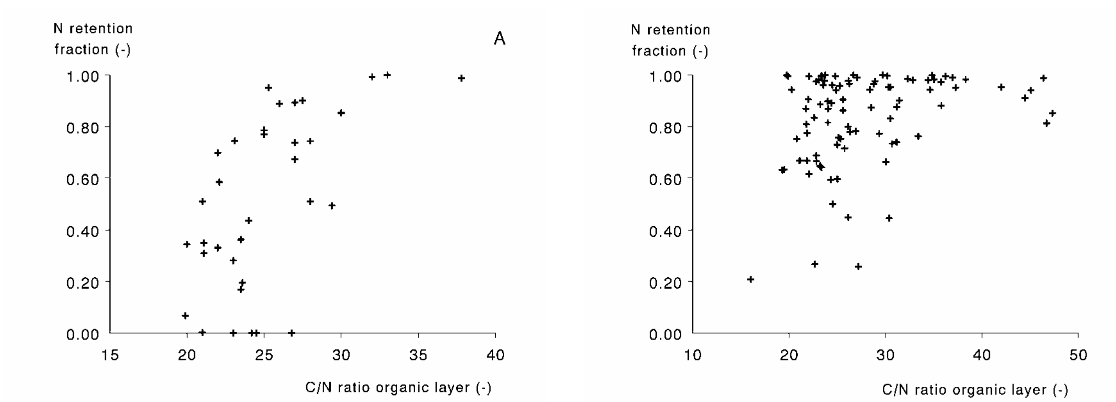


Figure 6.4 Relationship between N retention fraction and C/N ratios in the organic layer. The left graph refers to 34 forest plots of mainly conifers using ECOFEE-data from Gundersen et al. (1998a) with N input being (throughfall of N and N output referring to nitrate only (A). The right graph refers to 121 Intensive Monitoring plots with available data on total N deposition and N leaching (B).

Using the average fractions at each C/N ratio a logistic function of the N retention fraction as a function of C/N ratio was used in the calulations of N immobilisation as given in the following formula:

$$frN_{im} = \frac{1}{1+\alpha^{(\beta\cdot(CN-25))^{\gamma-1}}} \tag{6.7}$$

where $\alpha = 0.95$, $\beta = 0.4$ and $\gamma = -0.95$.

In the calculation use was made of site specific estimates for the more than 6000 forest soils in a systematic grid of 16 km x 16 km (level I) of

- N (NH_4, NO_3) deposition: EDACS model estimates
- Net N uptake: yield estimates as a function of stand age and site quality as described in Klap et al. (1997) multiplied by deposition dependent N contents in biomass.
- frNret: related to measured C/N ratios forest soil and modelled fraction NH_4 in deposition
- C/N ratios for forest soils: measurements, partly extrapolations

An overview of the calculated N deposition in 1960, 1990 and 2000 is given in Fig 6.5. Results show the large increase in N deposition in that period. The data for 2000 were used to calculate the carbon sequestration in the soil for that year. The data for the whole period 1960-2000 (data at 5 year intervals that were linearly interpolated) were used to assess the contribution of elevated N deposition in that period on the increase in carbon pools in standing biomass in that period (see the methodology described in Section 6.2.4 below).

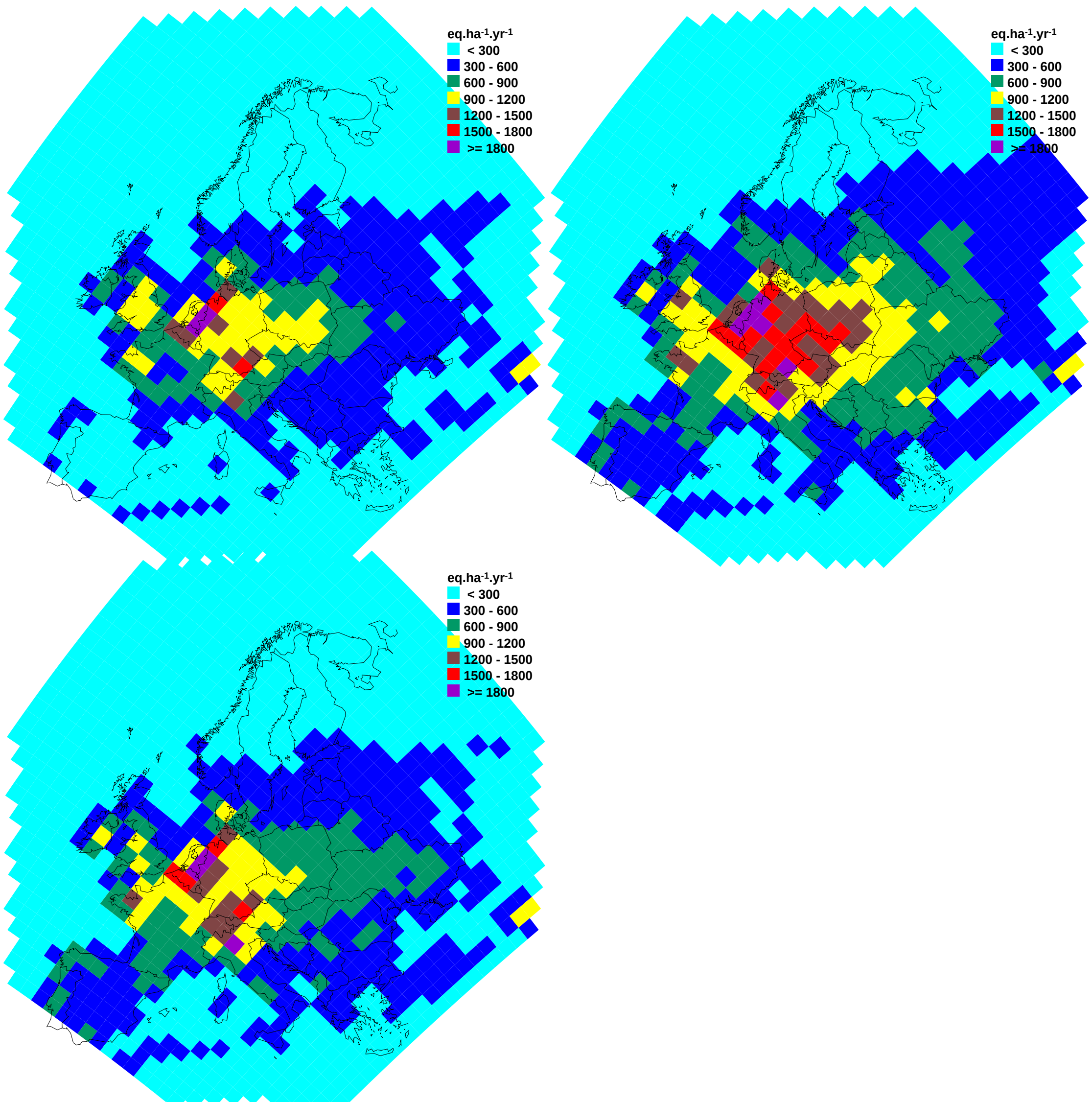


Figure 6.5 N deposition (mol N.ha^{-1}.yr^{-1}) for the year 1960, 1990 and 2000 as calculated by EMEP. Those data were allocated to the approximately 6000 level I plots based on a direct overlay

In this study, an estimate of C sequestration in stem wood of European forests (NEP) was derived from stand age and available site quality characteristics, using forest yield tables to estimate the actual forest growth (Klap et al., 1997), using a C content of 50%. This estimate was assumed to equal the baseline growth without impact of elevated N deposition. This estimate was increased with additional growth due to elevated N deposition as described in the following section. The net

C sink (NBP) in stem wood was calculated by assuming that NBP equals 33% of the NEP. This percentage is based on an estimated average NBP/NEP ratio for Europe, implying a net increase in standing forest biomass of 33% of the growth since 67 % is removed by harvesting or forest fires (Nabuurs and Schelhaas, 2003).

6.2.4 Assessing nitrogen deposition effects on carbon sequestration by European forests

The methodology used to calculate the impact of elevated nitrogen deposition on carbon sequestration by European forests is inspired by the approach of Nadelhoffer et al. (1999). These authors assessed additional C sequestration on a global scale from additional N uptake by trees and N immobilisation in soils in response to N deposition. Actually, the paper is sometimes rather unclear whether it derives the additional carbon sequestration due to N deposition, above the C sequestration due to 'normal' forest growth, or whether it calculates the total C sequestration using N retention as the indicator. This ambiguity is also partly reflected in the reactions on the paper debate (e.g. Jenkinson et al., 1999; Schindler, 1999; Sievering, 1999) and follows from the calculation. The estimate by Nadelhoffer et al. (1999), which suggest that C sequestration in forest trees and forest soils over the world is of equal magnitude is based on the following assumptions:

- Present total world N deposition estimate.
- Constant N retention fractions in trees (uptake; a fraction of 0.05) and soil (immobilisation; a fraction of 0.70), based on short-term fate (1-3 year) of ^{15}N labelled tracer experiments in nine temperate forests
- Averages values for the C/N ratio in stem wood (500) and forest soils (30).

Apart from the rough generalisation, the confusing aspect in this approach is that the present total world N deposition is used, whereas the paper discusses the possible impact of elevated N deposition. The 'unaffected' growth figures should be related to a certain N deposition as well. This implies that one should discuss the impact with reference to the increase in carbon pool in trees in the last decades, as presented by Kauppi et al. (1992) and Nabuurs et al. (1997). Those authors estimated a net increase in the C pool in trees in Europe of approximately 0.1 Gton C.yr^{-1} in the period 1970-1990. This implies that one has to estimate what the impact of increased N deposition in that period is on the C sequestration. In this study we used this approach but we extended the period to 1960-2000, assuming that the net C pool change in trees in that period is also 0.1 Gton C.yr^{-1}.

An overview of al differences used in this approach and those used by Nadelhoffer et al. (1999) is presented in Table 6.4. First of all, we used 1960 as the reference for N deposition (this leads to 'normal' growth) and calculated what the additional N deposition was in the period 1960-2000 compared to that reference year. Nadelhoffer et al. (1999) implicitly assumed that the reference N deposition is negligible. Unlike those authors we included the spatial differences in N deposition on the plots (EMEP estimates). Furthermore, we assumed that the additional N uptake due to N deposition is (uptake fraction) is a function of the N deposition, with values being higher in low deposition areas, because of N deficiencies, and lower in high deposition areas. Actually, the uptake fraction was assumed to vary from 10% in areas below 300 mol.ha^{-1}.yr^{-1} (approximately 5 kg.ha^{-1}.yr^{-1}) to 5% (the constant value used by Nadelhoffer et al. (1999) in areas above 1500 mol.ha^{-1}.yr^{-1} (approximately 20 kg.ha^{-1}.yr^{-1}). Similarly, the N immobilisation fraction was assumed to be a function of the C/N ratio of the organic layer and the NH_4/NO_3 in deposition, as described before, and not a constant of 70%.

Similar to the uptake fraction, the C/N ratios in trees were assumed to vary with the N deposition, values being higher in low deposition areas and lower in high deposition areas. This was based on the idea that luxury consumption takes place at a high N availability, meaning that the additional N uptake is only partly leading to additional growth (C pool change) since part is just leading to higher N contents (lower C/N ratios) in stem wood. Actually, the C/N ratio was assumed to vary from 500 (the constant value used by Nadelhoffer et al. (1999) in areas below 1500 $mol.ha^{-1}.yr^{-1}$ (approximately 20 $kg.ha^{-1}.yr^{-1}$) to 250 in areas above 5000 $mol.ha^{-1}.yr^{-1}$ (approximately 70 $kg.ha^{-1}.yr^{-1}$). This relation is based on a variation of N contents between 0.1 and 0.2% (at a constant C content of 50%) in comparatively low deposition areas (Scandinavia) to high deposition areas (The Netherlands). For the C/N ratio in the organic layer and mineral layer, we used the measured values at all Level I plots, instead of using a constant value of 30.

Table 6.4 Overview of differences between the approach used by Nadelhoffer et al. (1999) and in this study to calculate the impacts of N deposition on carbon sequestration.

Nadelhoffer et al. (1999)	Our approach
Reference N deposition is negligible	Reference N deposition is 1960
Constant average N deposition	Spatially distributed and time dependent N deposition[1)]
N uptake fraction is constant	N uptake fraction is f(N deposition)
N immobilisation is constant	N immobilisation fraction is f(C/N ratio humus layer/soil, NH_4/NO_3 in deposition)
C/N ratio tree is constant	C/N ratio tree varies in space and time as f(N $deposition_{x,t}$)[1)]
C/N ratio soil is constant in space and time	C/N ratio organic and mineral layer varies in space[2)]

[1)] Based on calculated EMEP N deposition
[2)] Based on the measured C/N ratio data at approximately 6000 forested plots

The above described methodological approach is presented below in mathematical terms. First the N sequestration in the tree is calculated from the additional N input in the period 1960-200 and the related C sequestration is calculated by multiplication with an N deposition dependent C/N ratio in the tree according to:

$$N_{seqtree}(extra) = \sum_{t=1960}^{t=2000}(Ndep_{(t)} - Ndep_{(1960)}).frup_{(t)} \tag{6.8}$$

$$C_{seqtree}(extra)_{(t)} = N_{seqtree}(extra)_{(t)} \cdot \frac{C}{N}tree_{(t)} \tag{6.9}$$

with the uptake fraction by stem wood being dependent on N deposition according to:

$$frup_{(t)} = 0.10 - 0.05 \cdot \frac{(Ndep_{(t)} - 300)}{1200} \tag{6.10}$$

for 300< Ndep < 1500, with $frup_{(t)}$ = 0.1 if = Ndep < 300 and $frup_{(t)}$ = 0.05 if = Ndep > 1500

and with the C/N ratio in the tree being dependent on N deposition according to:

$$\frac{C}{N}tree_{(t)} = 500 - 250 \cdot \frac{(Ndep_{(t)} - 1500)}{5000} \tag{6.11}$$

for 1500< Ndep< 5000, with $C/N_{(t)}$ = 500 if = Ndep < 1500 and $C/N_{(t)}$ = 250 if = Ndep > 5000

Then, the N immobilisation (sequestration) in the soil is calculated from the additional N input (N deposition in a given year minus the N deposition in 1960, corrected for the additional N uptake due to this increased N availability) for the period 1960-2000, multiplied with the N immobilisation fraction, according to (see Eq. 6.2):

$$N_{imsoil}(extra) = \sum_{t=1960}^{t=2000} (\delta Ndep_{(t)} - \delta Nup_{(t)}) \cdot frN_{im} \qquad (6.12)$$

with frN_{im} being calculated according to Eq. (6.7).Finally, the related C sequestration in the soil is calculated by multiplication of the calculated N immobilisation with the C/N ratio in the soil (organic and mineral layer), according to:

$$C_{seqsoil}(extra)_{(t)} = N_{imsoil}(extra)_{(t)} \cdot f(CN_{soil}) \qquad (6.13)$$

with f (CN_{soil}) being equal to the description of given in Eq. (6.2).

6.3 Results

6.3.1 Carbon pool changes in trees and soils at Intensive Monitoring plots

The result of the calculated annual carbon sequestration at the Intensive Monitoring plots obtained using Eq. (6.2) is given in Figure 6.6, together with the estimated sequestration due to forest growth in the last five years (see the previous chapter).

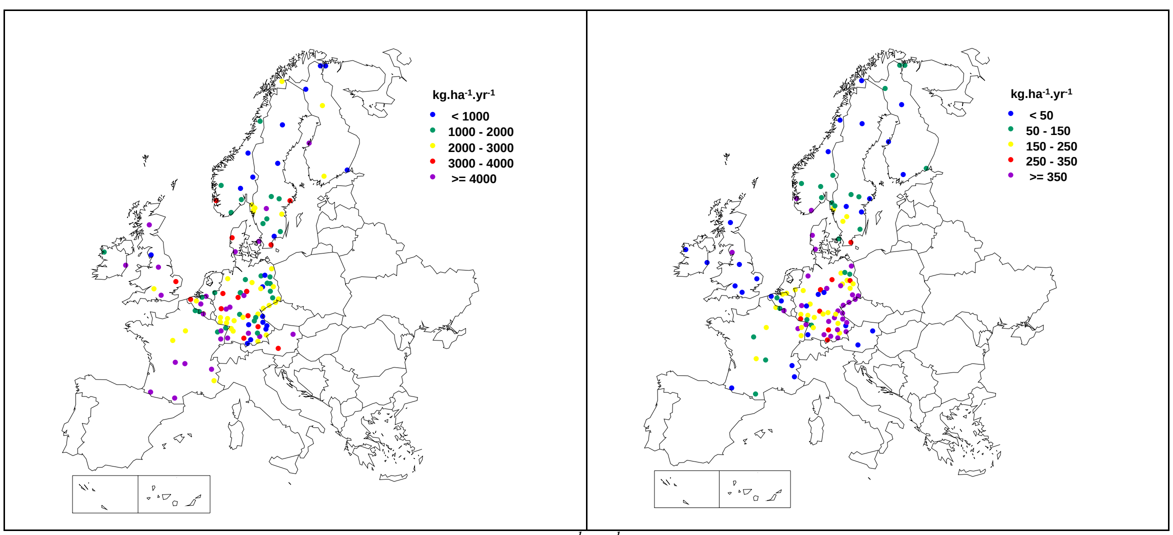


Figure 6.6 Calculated carbon pool changes (kgC.ha^{-1}.yr^{-1}) in trees (left) and soils (right) at the 121 Intensive Monitoring plots for the year 2000

The results show that the carbon pool changes in the tree are generally 5-10 times as high as the estimated carbon pool changes in the soil. As expected, the changes in the carbon pool in tree due to forest growth increase going from Northern to Central Europe and decrease again in Southern Europe. In line with the calculation procedure, the calculated changes in the carbon pool in soil do follow the N deposition pattern being high in Central Europe and low in Northern and Southern

Europe (Fig. 6.6). Interestingly, however, the same kind of pattern is found by Papale and Valentini (2003), presenting spatial (1 x 1km) estimates of carbon fluxes of European forests based on the net CO_2 exchange flux collected at sixteen of sites in the EUROFLUX network, using neural networks for the spatial extrapolation.

6.3.2 Carbon sequestration in soils and trees on the European scale

The estimated actual carbon sequestration in the tree wood (NEP) during the period 1960-2000 is given in Table 6.5. This estimate was based on the use of standard forest yield tables related to available site quality characteristics for each level I plot, while correcting for stand age increased with additional growth due to elevated N deposition as described before. Table 6.4 also includes an assessment of the net C sink (NBP) assuming that the latter value equals 33% of the NEP. The calculated carbon sequestration in stem wood due to forest growth equals approximately 0.28 Gton.yr^{-1}. Using a forested area of 200 million ha, the mean carbon sequestration rate in tree stem wood based on uptake is approximately 1400 kg.ha^{-1}.yr^{-1}. Assuming that NBP is 33% of the NEP, gives results close to 0.1Gton.yr^{-1}, being equal to a mean net sequestration rate of approximately 450 kg.ha^{-1}.yr^{-1}.

Table 6.5 Estimated total and net carbon sink for European forests due to tree growth (NEP) and increase in standing biomass (NBP, being 33% of the NEP) for the year 1960 and 2000.

Region	Carbon sequestration in wood (Gton.yr^{-1})			
	NEP 1960 -2000		NBP1960 -2000	
	No N impact	With N impact	No N impact	With N impact
EU	0.184	0.194	0.061	0.064
Candidate member states	0.036	0.038	0.012	0.013
Other European countries	0.059	0.063	0.020	0.021
Total	0.279	0.295	0.093	0.098

Note that total N uptake related to the above mentioned NEP growth figures is much higher than the additional N uptake mentioned by Nadelhoffer et al. (1999), which some authors related to total N uptake. The total estimated N uptake was 0.663 Mton.yr^{-1} at an estimated total deposition of 1.096 Mton.yr^{-1}. This implies a percentage uptake of 60% if one would relate N uptake to N deposition only, whereas Nadelhoffer et al. (1999) used a value of 5% for the additional N uptake related to N deposition. The impact of additional N input is 0.016 Gton.yr^{-1} (see also Section 6.3.3).

Estimated net carbon sequestration by accumulation in forest soils is given in Table 6.6. A distinction was made in the standard calculation with respect to N uptake and N accumulation (based on Eq. 6.2 and the extrapolation methods in section 6.2.3) and an alternative in which all the net incoming N was assumed to accumulate (total immobilisation, no leaching).

Table 6.6 Estimated net carbon sink by accumulation in European forest soils, for two different calculation scenarios for the year 2000.

Region	Net carbon sequestration in soil (Gton.yr^{-1})	
	Standard run	Total immobilisation
EU	0.0104	0.0183
Candidate member states	0.0013	0.0036
Other European countries	0.0020	0.0016
Total	0.0138	0.0235

The results using the standard run were lower than those derived by Nadelhoffer et al. (1999) (0.0138 vs. 0.022 Gton.yr^{-1}; compare Table 6.1 and 6.5). This is to be expected since these authors assumed a constant low net uptake (5%) and a constant high soil accumulation of 70% in the forest soil. Using the assumption that all the net incoming N is retained gives an estimate that is comparable to the estimate by Nadelhoffer et al. (1999), while the upper limit in this study appeared to be nearly twice as high compared than those derived by Nadelhoffer et al. (1999). Despite these possible uncertainties,

The geographic variation in carbon sequestration in trees and soils is illustrated in Fig. 6.7. The pattern in forest soil sequestration general follows the pattern of N deposition over Europe. It shows that C sequestration is small in Northern Europe, where the N input is low and nearly all incoming N is retained by the vegetation, and higher in Central and Eastern Europe where the N input is larger. This can, however, be a slight overestimate since part of the N accumulation may occur as a dilution of the C/N ratio at high deposition. The finding that C sequestration is negligible in northern boreal forest is in line with results from Martin et al. (1998) based on flux measurements for CO_2.

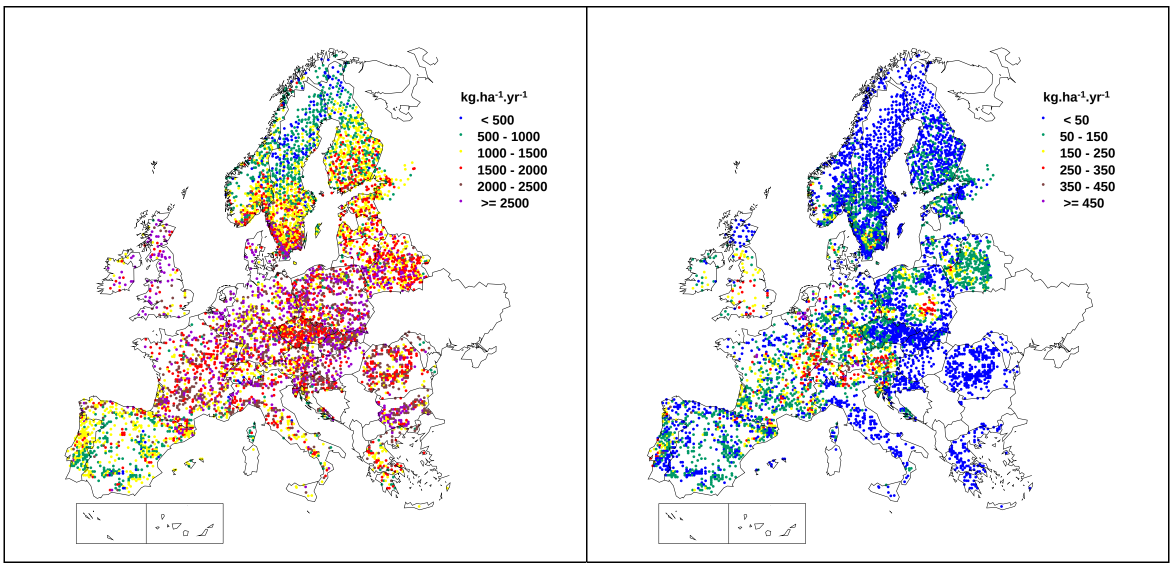


Figure 6.7 Geographic variation of the calculated carbon sequestration in trees and soil over Europe, using the standard run.

6.3.3 The impact of nitrogen deposition on carbon sequestration in European forests

Results obtained with respect to the total carbon sequestration in tree and soil using different assumptions regarding the parameters determining N retention are given in Table 6.7.

Table 6.7 Overview of the additional C sequestration in view of elevated N deposition in this period compared to the annual net carbon sequestration (Mton.yr^{-1})

Variation	Assumptions	Extra C sequestration 1960-2000 (Mton.yr^{-1})	
		Tree	Soil
1 This study: standard		16.1	6.2
- Uptake fraction	N dependent 5-10%[1]		
- Immobilisation fraction	N dependent 0-100%[1]		
- C/N ratio tree	N dependent 250-500[1]		
- C/N ratio soil	Site dependent[1]		
2 This study alternatives			
- Uptake fraction	Constant 5%[2]	9.9	5.6
- Uptake fraction	Constant 10%[2]	19.7	5.3
- Immobilisation fraction	Constant 70%[2]	14.6	6.3
- Immobilisation fraction	Constant 100%[2]	14.6	9.0
C/N ratio tree	Constant 500[2]	14.4	5.4
C/N ratio tree	Constant 400[2]	11.5	5.4
C/N ratio soil	Constant 30[2]	14.6	9.7
C/N ratio soil	Constant 25[2]	14.6	4.6
3 Nadelhoffer standard	Constant fractions and C/N ratios[3]	9.8	7.8
4 Nadelhoffer alternative	Adapted constant fractions and C/N ratios[4]	15.6	8.8

[1] The variation in fractions and C/N ratios depends amongst others on N deposition as described in the main text
[2] This variation implies that all other parameters are standard according to variation 1
[3] The standard Nadelhoffer calculation includes constant fractions for tree uptake (5%) and soil immobilisation (70%) and constant C/N ratios in tree (500) and soil (30)
[4] The alternative Nadelhoffer calculation includes adapted constant fractions for tree uptake (10%) and soil immobilisation (100%) and constant C/N ratios in tree (400) and soil (25)

Using the standard run, leads to an additional C sequestration in trees of approximately 16 Mton.yr^{-1} in the last 40 years. Comparing this estimate with the net increase in carbon pool in standing biomass of approximately 100 Mton.yr^{-1} (0.1 Gton.yr^{-1}) implies that 16% of this increase can be explained by an increase in N deposition. This contribution is however only the case if one relates the additional growth completely to the increase in standing biomass. If one relates it to the estimated growth without an impact of N deposition the contribution is only 16/279 is only 6% (see Table 6.4). The sensitivity analyses showed that the contribution varies between approximately 10 and 20 Mton.yr^{-1} implying a potential contribution of N deposition to the increase in standing biomass pool of 10-20 % (NBP) or 3.6-7% of the total growth (NEP).

The additional carbon sequestration in the soil is approximately 40% of the amount sequestered in the tree when using the standard with a variation between approximately 30-70% for the various alternatives. Using the standard Nadelhoffer approach, the additional sequestration in tree and soil is nearly equal (Table 6.6).

6.4 Discussion and conclusions

Comparison of calculated carbon sequestration estimates in stem wood with literature values
NEP: The average carbon pool change at all plots due to forest growth equals approximately 1400 kg.ha^{-1}.yr^{-1}, being approximately 1.5 times as low as the median change in carbon pool at all Intensive monitoring plots during a five year period (approximately 2175 kg.ha^{-1}.yr^{-1}; see Table 5.10). Apart from the fact that the Intensive Monitoring plots are not representative for the whole of Europe, this difference is most likely due to the fact that the calculated growth rates for the level I plots are average values over the total rotation period. For a total forested area in Europe of 200 million ha, the calculated carbon sequestration in stem wood due to forest growth (NEP) equals approximately 0.28 Gton.yr^{-1}.

Results of the NEP appear to be comparable to those based on CO_2 exchange fluxes (NEE) derived by Martin et al. (1998) based on the Euroflux sites (0.28 Gton.yr^{-1}), but the value is less (nearly twice as low) than the NEE value derived by Papale and Valentini (2003) from net CO_2 exchange fluxes collected at sixteen EUROFLUX sites. These fluxes do, however, include sequestration by trees and soil. The values are also twice as low as the NEP value derived from carbon flux data in forest at eleven "Canif" sites along a North –south transect through Europe (Schulze et al., 2000; compare Table 6.1). This illustrates that a simple extrapolation of results at a limited number of plots is highly questionable.

NBP: Assuming that NBP is 33% of the NEP, gives results close to 0.1Gton.yr^{-1}, being comparable to the estimates derived from repeated forest inventories (Kauppi et al., 1992; Nabuurs et al., 1997). Using a forested area of 200 million ha, the mean net carbon sequestration rate in tree stem wood is approximately 465 kg.ha^{-1}.yr^{-1}. This is close to net carbon sequestration rates in trees calculated by Liski et al. (2002) based on a dynamic modelling exercise, as described before. These authors calculated a net carbon sequestration in trees was at 390-600 kg.ha^{-1}.yr^{-1} in 1990 and at 440-510 kg.ha^{-1}.yr^{-1} in 2040. Upscaling their results to a forested area of 200 million ha, also gives results for the NBP near 0.1 Gton.yr^{-1} (see table 6.1). The average value is approximately twice as low as the average value of 800 kg C.ha^{-1}.yr^{-1} obtained by Nabuurs and Schelhaas (2002) for 16 typical forest types across Europe, but this in a comparable order of magnitude

Comparison of calculated carbon sequestration estimates in soil with literature values
The calculated net carbon sequestration in the soil of approximately 0.014 Gton.yr^{-1}, being equal to an average accumulation of 70 kg.ha^{-1}.yr^{-1}, is much higher than the value derived by Schulze et al. (2000) based on the C retention in eleven sites (0.13-0.17 Gton C.yr^{-1}). The latter estimate is likely to be an overestimate, as it would imply that the C/N ratio of European forest soils is strongly increasing. There are no indications that this is the case. To the reverse, it is more likely that C/N ratios are decreasing, especially in areas with an elevated N deposition. This result thus illustrates that it is dangerous to make estimates on a European scale based on a limited number of plots using a simple upscaling procedure.

The result is more in line with those derived by Liski et al. (2002), based on the dynamic modelling exercise described before, with net carbon sequestration in soil estimated at 190 -305 kg.ha^{-1}.yr^{-1} in 1990 and 2040, respectively, Even though this leads to clearly higher values on a European scale, the difference (see also Table 6.1) is by far not so large as with Schulze et al. (2000) . Furthermore, the results are in line with the net carbon sequestration in soil for 16 typical forest types across Europe derived by Nabuurs and Schelhaas (2002), being equal 110 kg C.ha^{-1}. If one would, again, simply multiply this average value with a forested area of 200 million ha, it would lead to a net carbon sequestration of 0.022 Gton.yr^{-1}, being comparable to the upper estimate in this study.

The conclusion that net sequestration potential of the below ground carbon in the soil, which has much lower turnover times than above ground carbon, is only small in forests is also in line with field data, showing that soil C sequestration is even small after afforestation on arable fields (Vesterdal et al., 2002). This implies that the terrestrial carbon sink can only be viewed as "buying variable time to address the most significant perturbation of the carbon cycle: fossil fuel emissions" (Steffen et al., 1998).

Impacts of nitrogen deposition on carbon sequestration

The basic assumption for carbon pool changes in both tree and soil was that the additional N uptake or immobilisation is reflected in carbon pool changes due to growth or organic matter accumulation according to the C/N ratio of the tree or the soil. The calculation focused on CO_2 sequestration in the soil from nitrogen retention assuming that nitrogen retained in the soil form organic matter with a constant carbon to nitrogen ratio. Most likely, the estimate constitutes an upper limit since nitrogen deposition tends to decrease the carbon to nitrogen ratio over time.

The conclusion that the increase in forest growth in trees is very small, about 5% only, seems contradictory with a generic more sophisticated modelling approach, in which the combination of CO_2 rise and elevated N deposition was estimated to account for a 15-20% increase in forest net primary productivity (Rehfuess et al., 1999). In this study, model predictions were made of carbon sequestration in view of changes in climatic variables, temperature and precipitation, CO_2 concentrations and nitrogen deposition. Results showed that the impact of temperature was much less important than that of CO_2 , whereas N deposition was claimed to be most important (Rehfuess et al., 1999). This contradiction is however due to the upscaling of the model to the European scale. The net impact of additional N deposition on forest growth was estimated at approximately 15/25 kg C per kg N by the various sophisticated forest growth models, being comparable to the result of the model applied in this study. It implies that in Central European areas with a large additional N input (e.g. of 10/20 kg.ha/1), the impact is large but not in Northern and Southern Europe, where the additional N input is generally low.

The predicted impact of N deposition in high deposition areas might even be overestimated. A positive effect can indeed be expected in areas where forest growth is limited by N availability, but a continuous high input of N may lead to a situation where other growth factors, such as other nutrients and water, become limiting for the growth of forest. The relation between water shortage and N surplus can be explained by the fact that a high N input favours growth of canopy biomass, whereas root growth may be relatively unaffected (shown only for seedlings). The increase in canopy biomass will lead to a higher demand for water and therefore to an increased risk of water shortage (drought). It also causes an increased demand of base cation nutrients (Ca, Mg, K) whereas the availability of these cations can be reduced by increased dissolved levels of NH_4 and/or Al (induced by NO_3 and SO_4). This effect may reduce the fertilising effect of high N deposition.

Conclusions and outlook

Based on soils data collected at the Level I and Level II monitoring plots and modelled nitrogen deposition data an estimate of CO_2 sequestration for European forests, divided in trees and soils could be made. Furthermore, the contribution of N deposition to forest growth and soil carbon sequestration could be assessed. Using the above mentioned approach, the following conclusions can be drawn from this study:

- Carbon pool changes in the tree are generally 5-10 times as high as the estimated carbon pool changes in the soil. As expected the changes in the carbon pool in tree due to forest growth increase going from Northern to Central Europe. The calculated changes in the carbon pool in soil do follow the N deposition pattern being high in Central Europe and low in Northern and Southern Europe. This follows from results at both level I and level II plots.
- Net increases in the carbon pool by forests in Europe (both trees and soil) are in the range of 0.1-0.15 $Gton.yr^{-1}$, being an important part (about 50%) of the terrestrial carbon sink in Europe, derived from atmospheric inversion models. The results furthermore show that the C sequestration by forest is mainly due to a net increase in forest growth, since the longer term C immobilisation in the soil is limited.

- The contribution of N deposition to the increase in carbon due to forest growth is approximately 10 and 20 $Mton.yr^{-1}$, being 3.5-7% of the carbon pool increase due to the average estimated forest growth in that period (approximately 280 $Mton.yr^{-1}$).

The result of this study implies that the impact of forest management is most important in explaining the carbon pool changes in forest in Europe. Combined with the conclusion that the increase in carbon pools in trees is mainly responsible for carbon sequestration in Europe, it implies that the current sequestration is only a transitory phenomenon. It is a gain due to the fact that forests in Europe are aggrading because the removal by harvesting and forest fires is less than the net growth. A further contribution to C sequestration on the forest area may come from earlier and recent afforestations on fields or grasslands. Effects of such land use change are not included in the calculations. On these areas the build up of C stock in trees may be substantial but still a transitory phenomenon lasting a forest generation.

For future predictions, models that are able to describe the effects of CO_2, water and nutrients on tree growth (Van Oijen et al., 2003a, b; Van Oijen et al., 2003c) are important tools. Simultaneously, many stress-factors tend to cause defoliation and decrease tree growth in European forests. Shortage of water, high pollutant concentrations in air, e.g. ozone, and high inorganic aluminium concentration in soil water are potential causes of defoliation. Process based models can be used to assess the importance of water shortage to photosynthesis and growth, as well as the importance of climate and water stress to defoliation and subsequently growth. Such models can also be used to describe tree growth and flows and accumulation of carbon, nitrogen, other nutrients and water in forests ecosystems. The lack of quality checked test data has considerably hindered model development. The large variation in climate and nitrogen deposition within Intensive Monitoring sites in Europe now enables the use of versatile and informative data sets to the testing of the models and testing of our understanding of the processes explaining tree growth and forest health. Such models combined information at level II and level I plots do form an important tool for future more elaborated studies on the prediction of carbon sequestration in Europe in response to changing environmental conditions.

7 Modelling the long-term impact of deposition scenario's for nitrogen and acidity at intensively monitored forest plots

7.1 Introduction

The relevance of dynamic soil models

Decisions on emission reductions policies require insight in the effectiveness of abatement strategies. In this respect, models are important tools to assist decision makers in their evaluation of strategies to control sulphur and nitrogen emissions. Up to now critical loads derived by steady-state models have been used in the negotiations of emission reductions in Europe; together with technical and economical aspects of emission reductions this had lead to cost-effective emission reductions based on effects on ecosystems. To gain insight into the time delay between the time-point of non-exceedance and actual chemical (and biological) recovery, however, dynamic models are needed.

In the causal chain from acid deposition to ecosystem-damage (damage to key indicator organisms) there are two major sources of response-delay. Biogeochemical processes can delay the chemical response in soil to acid deposition, and biological processes can further delay the response of indicator organisms, such as damage to trees in forest ecosystems. The static critical load model considers only the steady-state condition, in which the chemical and biological response to a change in deposition is complete. Dynamic models on the other hand, attempt to estimate the time evolution of soil (and biological) responses to changes in acid deposition and can be used to assess the time required for a new (steady) state to be achieved.

Relationships between critical load models and dynamic soil models

With critical loads, i.e. in the steady-state situation, only two cases can be distinguished when comparing them to deposition: (1) the deposition is below or equal to critical load(s), i.e. no exceedance, and (2) the deposition is greater than critical load(s), i.e. critical load exceedance. In the first case there is no (apparent) problem, so no reduction in deposition is necessary. In the second case there is, by definition, an increased risk of damage to the ecosystem, and therefore the deposition should be reduced to safeguard the ecosystem. Sometimes it is assumed that reducing deposition to (or below) critical loads immediately removes the risk of 'harmful effects', i.e. the chemical parameter (e.g. the Al/Bc ratio), which links the critical load to the effect(s), immediately attains a non-critical ('safe') value and that there is immediate biological recovery as well. But the reaction of soils, especially their solid phase, to changes in deposition is delayed by (finite) buffers, the most important being the cation exchange capacity (CEC). These buffer mechanisms can delay the recovery of e.g. chemical parameters, and it might take decades or even centuries before 'safe' values for a parameter or steady state is reached.

These finite buffers are not included in the critical load formulation, since they do not influence the steady state, but only the time to reach it. Therefore, dynamic models are needed to estimate the times to attain a certain chemical state in response to changes in acid deposition induced by agreements on emission reductions. In addition to the delay in chemical recovery, there is probably a further delay before the 'original' biological state is reached, i.e. even if the chemical criterion is met (e.g. $Al/Bc<1$), it will take time before full biological recovery is achieved. More information on the relationship between critical load models and dynamic soil models is given in Annex 1.

The role of ICP forests in dynamic modelling

Dynamic modelling is a relatively new topic for the effects-oriented work under the LRTAP Convention. Within the ICP on Integrated Monitoring, existing dynamic models have been applied in the mid-nineties at a few sites for which a sufficient amount of input data was available. By applying dynamic soil models at about two hundred Intensive Forest Monitoring sites, a picture appears of a transect through Europe going from southern France to northern Scandinavia. Especially the validation of dynamic soil models on those data is crucial since the new challenge within the ICP on Modelling and Mapping is to develop and apply dynamic model(s) on a European scale and to link them with the integrated assessment work under the LRTAP Convention in support of the review and potential revision of the Gothenburg Protocol.

Contents of this chapter

This chapter first presents the methods (locations, modelling approaches, model validation procedure and input data including deposition scenarios) that are needed to perform dynamic model calculations (Section 7.2). The dynamic modelling concepts and data requirements presented in the following are an extension of those employed in deriving the Simple Mass Balance (SMB) model, used in the previous Technical Report (De Vries et al., 2002) to derive critical loads for nitrogen and acidity. Results are described in Section 7.3. This includes results of a comparison of model results with measurements at more than 100 plots with both deposition and soil solution chemistry data (Section 7.3.1) and an application of the model to the same plots, using various relevant emission deposition scenarios (Section 7.3.2). Finally, a discussion of the results and conclusions is presented in Section 7.4.

7.2 Methods

7.2.1 Locations

For the dynamic modelling, validated information on bulk deposition, throughfall and soil solution chemistry is needed. In Fig 7.1 a map of the plots is presented for which these data are available and that were used for the dynamic model to predict the long-term impact of deposition scenario's for nitrogen and acidity Only for these plots a (partial) calibration of the model can be performed as both input (total deposition) and response (soil solution concentrations) are needed.

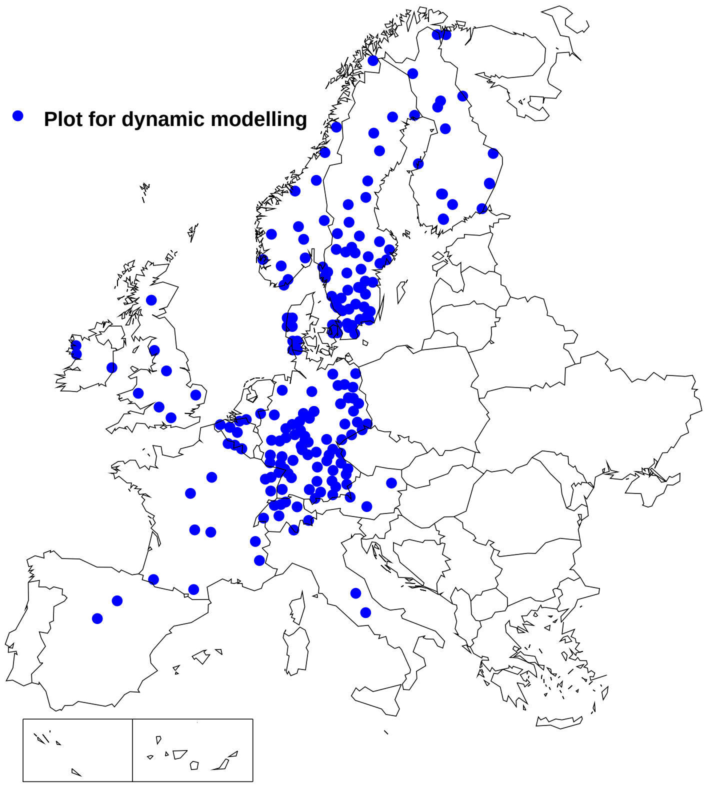


Figure 7.1 Locations of the Intensive Monitoring plots used for calibration and application the dynamic SMART model.

7.2.2 Modelling approach

Available dynamic soil models

For nearly 15 to 20 years, scientists have already been developing, testing and applying dynamic models to simulate the acidification of soils or surface waters. There is thus no shortage of soil (acidification) models, but most of them are not designed for regional applications. A comparison of 16 models can be found in a special issue of the journal "Ecological Modelling" (Tiktak and van Grinsven, 1995). These models emphasise either soil chemistry (such as SMART, SAFE and MAGIC) or the interaction with the forest (growth).

In addition to the large number of dynamic model applications to individual sites over the past 15 years, there are several examples of dynamic soil models that were developed and applied for application on a (large) regional scale. Earlier versions of the RAINS model (Alcamo et al., 1990) contained an effects module which simulated soil acidification on a European scale (Kauppi et al., 1986). De Vries et al. (1994b) employed the SMART model to simulate soil acidification in Europe, and Hettelingh and Posch (1994) used the same model to investigate recovery delay times on a European scale. Furthermore, De Vries et al. (1994a) used the RESAM model to simulate soil acidification in the Netherlands, while Alveteg et al. (1995) and Kurz et al. (1998) used the SAFE model to assess temporal trends in soil acidification in southern Sweden and Switzerland.

Constraints for dynamic models used in this study

Up to now critical loads, derived by steady-state models have been used to negotiate emission reductions in Europe. Thus the dynamic models to be used in the assessment of recovery under the LRTAP Convention have to be compatible with the steady-state models used for calculating critical loads. In other words, when critical loads are used as input to the dynamic model, the (chemical) parameter chosen as criterion in the critical load calculation has to attain the critical value once the dynamic simulation has reached steady state. But this also means that concepts used in the dynamic model have to be a continuation and extension of the concepts employed in deriving the steady-state model.

In order to meet this constraint the dynamic modelling concepts and data requirements presented in this chapter, are an extension of those employed in deriving the Simple Mass Balance (SMB) model. The SMB model is described in detail in the previous Technical report (De Vries et al., 2002). Earlier descriptions of the SMB model can be found in Sverdrup et al. (1990), De Vries (1991), Sverdrup and De Vries (1994) and Posch et al. (1995).

A model that does meet the constraints given above is the model SMART, described in De Vries et al. (1989) and Posch et al. (1993). This model contains basic extensions of the SMB model into a dynamic soil acidification model. An even simpler model, called Very Simple Dynamic (VSD) model, has recently been developed (Posch and Reinds, 2003).

The SMART model

The SMART model (**S**imulation **M**odel for **A**cidification's **R**egional **T**rends) is a relatively simple extension of the SMB model for critical loads. As with SMB, in the SMART model, the various ecosystem processes have been limited to a few key processes. Processes that are *not* taken into account are: (i) canopy interactions, (ii) nutrient cycling processes, (iii) N fixation and NH_4 adsorption, (iv) uptake, immobilisation and reduction of SO_4 and (v) complexation of Al with OH, SO_4.

The SMART model consists of a set of mass balance equations, describing the soil input-output relationships, and a set of equations describing the rate-limited and equilibrium soil processes. The soil solution chemistry in SMART depends solely on the net element input from the atmosphere (deposition minus net uptake minus net immobilisation) and the geochemical interaction in the soil (CO_2 equilibria, weathering of carbonates and silicates, and cation exchange). Soil interactions are described by simple rate-limited (zero-order) reactions (e.g. uptake and silicate weathering) or by equilibrium reactions (e.g. cation exchange and sulphate adsorption). It models the exchange of Al, H and Ca+Mg+K with Gaines-Thomas equations and sulphate adsorption with a Langmuir equation. Furthermore, it does include a balance for carbonate and Al, thus allowing the calculation from calcareous soils to completely acidified soils that do not have an Al buffer left. In this respect, SMART is based on the concept of buffer ranges expounded by Ulrich (1981). Recently a description of the complexation of aluminium with organic acids has been included. The interaction of Al with organic acids can be described as mono-, di- or tri-protic. A graphic representation of these processes is given in figure 7.2.

Solute transport is described by assuming complete mixing of the element input within one homogeneous soil compartment with a constant density and a fixed depth. Since SMART is a single layer soil model neglecting vertical heterogeneity, it predicts the concentration of the soil water leaving this layer (mostly the root zone). The annual water flux percolating from this layer is taken equal to the annual precipitation excess. The time step of the model is one year, i.e. seasonal variations are not considered. A detailed description of the SMART model can be found in De Vries et al. (1989) and Posch et al. (1993). The SMART model has been developed with regional applications in mind, and an early example of an application to Europe can be found in De Vries et al. (1994b).

The guiding principle of SMART is the compatibility with the critical load model SMB, since the steady-state solutions of the dynamic model employed should be the critical loads derived earlier. Dynamic models of acidification are based on the same principles as steady-state models: the charge balance of the ions in the soil solution, mass balances of the various ions, and equilibrium equations. However, whereas in steady-state models (SMB) only sources and sinks are considered which can be assumed infinite (such as base cation weathering), the dynamic model SMART includes finite sources and sinks of major ions, i.e. cation exchange, sulphate adsorption and nitrogen retention and a mass balance for cations, nitrogen and sulphate, in addition to the equations included in the SMB model. These are the three most important processes involving finite buffers and time-dependent sources/sinks. These finite buffers have not been included in the derivation of critical loads, since they do not influence the steady state. However, when investigating the chemistry of soils over time as a function of changing deposition patterns, these processes govern the long-term (slow) changes in soil (solution) chemistry. For example after an increase in acidifying input, cation exchange (initially) delays the decrease in the acid neutralisation capacity (ANC) by releasing base cations from the exchange complex, thus delaying the acidification of soil solution until a new equilibrium is reached (at a lower base saturation). On the other hand, cation exchange delays recovery since 'extra' base cations are needed to 'replenished' base saturation instead of increasing ANC of soil solution.

7.2.3 Model parameterisation and model calibration

Data needs

The input data needed to run dynamic models can be grouped into input- and removal fluxes and soil properties. Part of the input- and removal fluxes are also described in the previous technical Report (De Vries et al., 2002), since they were also needed in the SMB model, but for this study an update of those fluxes was made and the new results are shortly summarised. This includes the input of element by atmospheric deposition, the water fluxes through the forest ecosystems, the net uptake of nutrients by forests and the base cation weathering from the soil.

Furthermore this section describes how the (soil) data were derived that is needed to run dynamic models. Most important soil parameters are the cation exchange capacity (CEC) and base saturation and the exchange (or selectivity) constants describing cation exchange, as well as parameters describing sulphate adsorption or desorption, since these parameters determine the long-term behaviour (recovery) of soils. Also the parameters that determine nitrogen immobilisation, denitrification and nitrification were estimated.

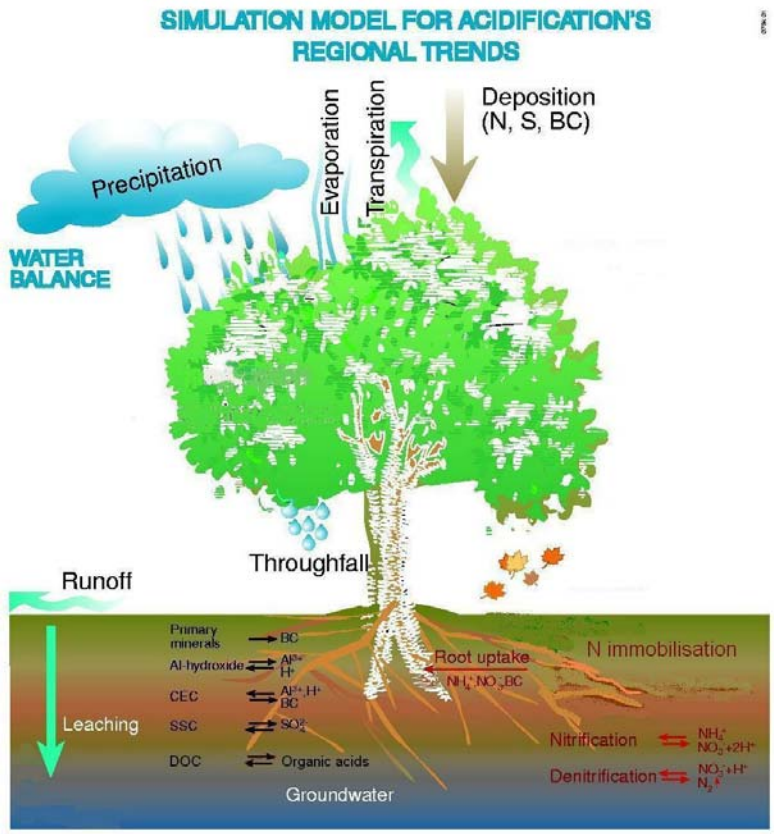


Figure 7.2 Graphic representation of the SMART model

Ideally, all input data are derived directly from measurements at the site for which the model is applied. This, however, is not always feasible for all input data. In this chapter we provide information on how the input data needed for SMART were derived, either derived directly from measurements at the sites or indirectly from model calibration.

Deposition of acidity and base cations

Total deposition of sulphur for the period 1995-2000 was computed by adding measured throughfall and measured or estimated stemflow values below the forest canopy, assuming that the effects of foliar sulphur uptake by the forest canopy is negligible. Base cation (Ca, Mg, K and Na) and nitrogen deposition data for each stand for the period 1995-2000 were derived on the basis of throughfall and bulk deposition data, accounting for canopy exchange, as described in De Vries et al. (2002).

Water fluxes

Water fluxes were calculated using the WATBAL model (Starr, 1999). This relatively simple water balance model uses a one-layer approach and monthly time-steps. Input data required are soil properties such as the available water content (AWC), rooting depth and texture dependent ratio's of the critical soil water content to the available water content, meteorological variables (rainfall, global radiation (or sunshine duration) and temperature), a number of generic site variables such as latitude, altitude, initial amount of snow on the ground and forest cover, and generic constants such as snow albedo. Measured throughfall was used as the water input at the top of the soil compartment, whereas monthly values of temperature and global radiation were derived from a data base with interpolated daily values for 50*50 km grid cells. AWC was estimated as function of soil type and texture class according to Batjes (1996) who provides texture class dependent AWC values for all FAO soil types based on an extensive literature review. Critical soil water: AWC ratios (the ratio between actual soil water content and AWC at which transpiration is reduced) where computed as a function of soil texture according to the standard WATBAL procedure.

The reliability of the water fluxes was checked by comparing the leaching of chloride (Cl) and sodium (Na) against the deposition. Both chloride and sodium can be considered as tracers (Cl) or nearly tracers (Na), i.e. the (long-term average) leaching computed from the modelled downward water flux and the measured concentration should match the deposition. In the case of Na, the leaching flux can be somewhat higher than the deposition flux due to weathering and cation exchange but generally these fluxes are negligible compared to the deposition of Na. The measurements of Na and Cl in deposition and soil solution thus allow checking whether the hydrology is modelled accurately.

Figure 7.3 shows the deposition-leaching relations for Cl and Na for plots with at least 2 years of measurements. As can be seen, the average slope for Cl is close to the perfect 1, indicating that there is no overall bias in the hydrological model. However, the graphs also show that there are several plots with rather unbalanced inputs and outputs. In the case of Na this could partly be explained by weathering. Part of the plots where chloride leaching is higher than chloride deposition are located close to the sea, which may cause imbalances in the budget due to sea-salt input on the soil that is not collected in the deposition samplers.

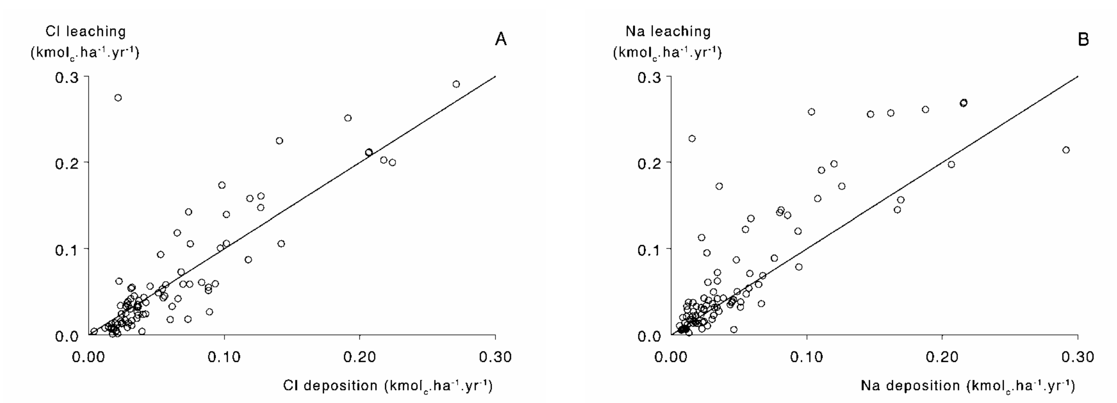


Figure 7.3 Cl and Na input-output (deposition-leaching) relationships.

Uptake of nitrogen and base cations

In SMART, nitrogen and base cation uptake is the net growth uptake, i.e. the net uptake by vegetation that is needed for long-term average growth. Input by litterfall and removal by maintenance uptake (needed to re-supply nitrogen and base cations to leaves) is thus not considered, assuming that both fluxes are equal in a steady-state situation. Thus the net uptake is calculated, being equal to the annual average removal in harvested biomass. In this calculation we assumed that this includes the removal of stems and branches.

Net growth uptake of N, N_{gu}, and base cations, Ca_{gu}, Mg_u and K_u, was computed by multiplying the current growth at the site with the densities of stem wood and standard element contents in stems and branches. For the densities of stem wood and the nitrogen and base cation contents in stems use was made of literature data (e.g. Kimmins et al., 1985; De Vries et al., 1994b), as presented in Table 7.1

Table 7.1 Average values used for the densities of stem wood and the N and BC (Ca+Mg+K) contents in stem wood of the considered main tree species

Tree species	Stem density ($kg.m^{-3}$)	N content in stem wood ($g.kg^{-1}$)	BC content in stem wood ($g.kg^{-1}$)
Scots Pine	510	1.2	1.6
Norway Spruce	460	1.2	1.2
Oak	700	1.7	2.5
Beech	700	1.4	2.4

Annual growth for the period 1995-2000 (for which the model was validated) was estimated from repeated surveys on stem diameter and tree height; as described in Chapter 5. Past and future growth, used in scenario analyses was derived by scaling the calculated yield for the period 1995-2000 with standard logistic growth curves available for combinations of species, climate and yield (Klap et al. (1997)). Stem to branch ratios were estimated as a function of tree species and stand age as described in Klap et al. (1997). Monthly net growth uptake was then derived by distributing the annual growth over the months within the growing season, weighted by the monthly fraction of temperature sum over the growing season. Begin and end of the growing season were computed as a function of climate zone, altitude and latitude according to Klap et al. (1997).

Figure 7.4 show the cumulative frequency distribution of computed N and BC uptake (in $kg.ha^{-1}.yr^{-1}$) for the considered plots. It clearly show that the uptake of especially N by broadleaved forest (median 5 $kg.ha^{-1}.yr^{-1}$) is much higher than for conifers forest (median value 2.7 $kg.ha^{-1}.yr^{-1}$), due to higher growth rates and higher N contents in stem- and branch wood.

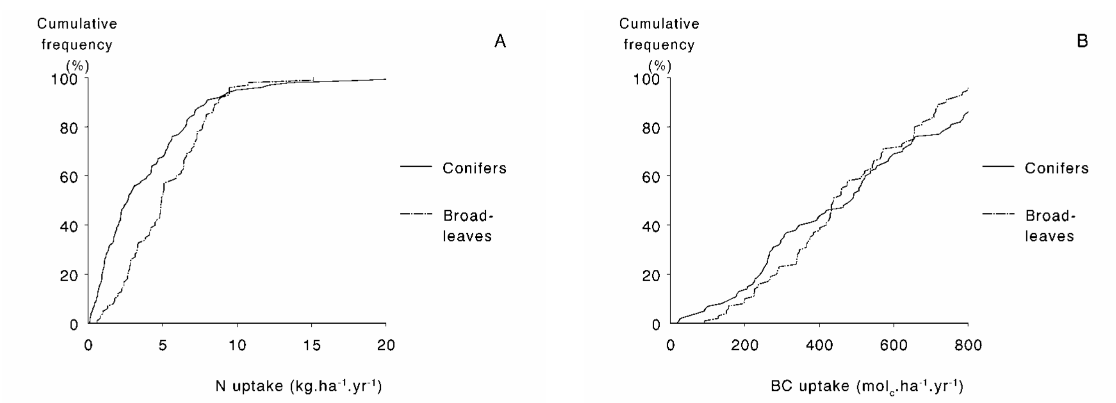


Figure 7.4 N uptake (left) and BC uptake (right) for conifers and broadleaved plots

Weathering of base cations

There are various possibilities to assess weathering rates including (UN/ECE, 1996):

- Estimation of the depletion of base cations in the soil profile by chemical analyses of different soil horizons including the parent material. This method, in which an extremely resistant

mineral, such as zirconium, is often used as an internal standard, gives the average weathering rate over the period of soil formation (De Vries and Breeuwsma, 1986; Starr et al., 1998).
- Correlation between the weathering rate and the total Ca and Mg content in the parent material multiplied by the present day effective temperature sum (Olsson and Melkerud, 1991). As with the previous approach, this gives the average weathering rate over the period of soil formation.
- The weathering rate model PROFILE, which calculates actual field weathering rates based on the soil mineralogy (Sverdrup, 1990; Sverdrup and Warfvinge, 1993).
- Assignment of an actual field weathering based on the parent material and texture class of a given soil (dominant) soil unit (De Vries et al., 1994b).

Apart from the last method, total cation concentrations in either the parent material (C horizon) or even in the complete soil profile, are needed to estimate the weathering rate. At present, PROFILE is most frequently used when detailed mineralogical data are available or when such data can be derived from a total cation analyses. Becker et al. (2000), for example, used PROFILE to calculate the weathering of Intensive Monitoring plots in Germany for the assessment of critical loads for those plots. Data on the total cation contents are, however, either not available for the plots or not submitted to FIMCI.

Since no measurements of weathering are available, the weathering of the base cations Ca, Mg, K and Na was estimated from the budget (the average of the differences between deposition and leaching corrected for base cation uptake) of the respective elements at the sites. If the budget yields a negative base cation weathering, the base cation uptake was adapted to close the budget and the weathering was set to zero. Simulations with Smart show that in many cases the contribution of cation exchange (release or adsorption) to the monthly budget can be considered negligible. Only for plots with a large base cation pool and rapidly changing acid input would it be of importance. However, the measurements do not allow an accurate assessment of the amount of base cations exchanged (only Ca +Mg in SMART) as only one observation of base saturation in time is available.

Thus the weathering of a base cation Y, Y_{we}, becomes:

$$Y_{we} = Y_{le} + Y_{gu} - Y_{dep} \tag{7.1}$$

where the subscripts *le*, *gu* and *dep* stand for leaching, net growth uptake and deposition of element Y. If Y_{we} calculated from Eq. 7.1 is negative, it is set to zero and Y_{gu} (for Y=Ca, Mg, K) is set to $Y_{dep}-Y_{le}$. Eq. 7.1 was applied to averages over the measurement period.

Figure 7.5 shows the cumulative frequency distributions of computed weathering rates for soil with sandy, clayey and calcareous parent materials. Its shows that the median value for sandy soils is about half that of clayey soils. As to be expected, weathering rates for calcareous soils are much higher and generally range between 1700 and 10000 eq.ha^{-1}.yr^{-1}.

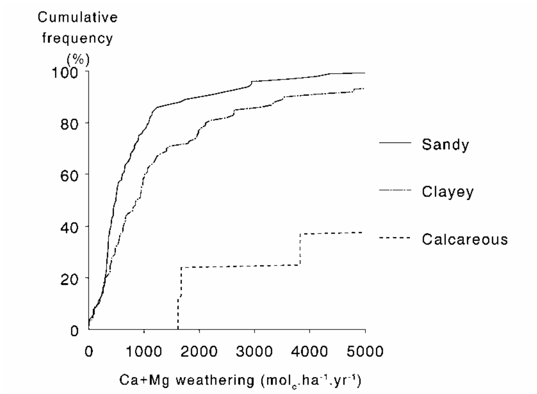


Figure 7.5 Base cation weathering for sandy, clayey and calcareous parent materials.

In the previous Technical Report, base cation weathering rates for the root zone were derived by a relationship with parent material class and texture class (either available or derived from soil type information). The estimates thus derived were updated on the basis of either the measured annual average temperature in the considered period 1995-1998 or interpolated annual average values for the 10-year period 1985-1995. More information on the reliability of this approach is given in the previous Technical Report (De Vries et al., 2002). A comparison of the results obtained from the present and the previous approach shows that there is only a very weak relationship between the base cation weathering rates based on classes and those based on calibration. This can partly be explained by the fact that the calibrated weathering is computed from a rest-term from the base cation input-output budget that includes uncertain terms as base cation uptake and leaching. Furthermore, base cation release from the exchange complex is now considered as weathering which, in some cases, might lead to unrealistic weathering rates. Most likely, the weathering rates of sandy soils above 1500 $mol_c.ha^{-1}.yr^{-1}$ are due to cation release from the exchange complex and the same is true with weathering rates of clayey soils above 3000 $mol_c.ha^{-1}.yr^{-1}$.

Parameters describing aluminium release

In SMART the concentration of free (uncomplexed) Al is modelled by a relationship with the H concentration according to:

$$[Al] = K_{Alox} \cdot [H]^{\alpha} \tag{7.2}$$

where α>0 is a site-dependent exponent and where KAl_{ox} = the Al dissolution constant. For α=3 this is the familiar gibbsite equilibrium ($KAl_{ox} = K_{gibb}$) used in the SMB model.

The equilibrium constant K_{Alox} and the exponent α are determined by linear regression after taking logarithms in eq. 7.2. For each plot for which observations of $[H]$, $[Al_{tot}]$ and DOC were available, the concentration of free Al was first calculated. This was done by using a triprotic model for the dissociation of DOC and a simple complexation model with Al, as implemented in the latest version of the SMART model, described in detail in Annex 3. For plots without DOC data constants were derived for the relation between pH and total Al.

Figure 7.6 show the relationships between pH and pAl (-log (Al)) for total Al (left) and free Al (right). This figure clearly shows that the relationship between free Al and pH is better than between total Al and pH especially for pH values greater than 5 where most Al is complexed. The horizontal line at pAl 5.82 in the leftmost graph represents an aluminium concentration of 0.04 mg.l^{-1}; for some countries this is the lowest value they submit (apart from 0) and might thus be the analytical detection limit for these countries. Figure 7.7 shows the correlation coefficient between pAl and pH for plots with an average pH lower than 5. This also illustrates the better correlation between pH and free aluminium than between pH and total aluminium as the median correlation coefficient between free Al and pH is about 0.7, whereas between total Al and pH it is about 0.4.

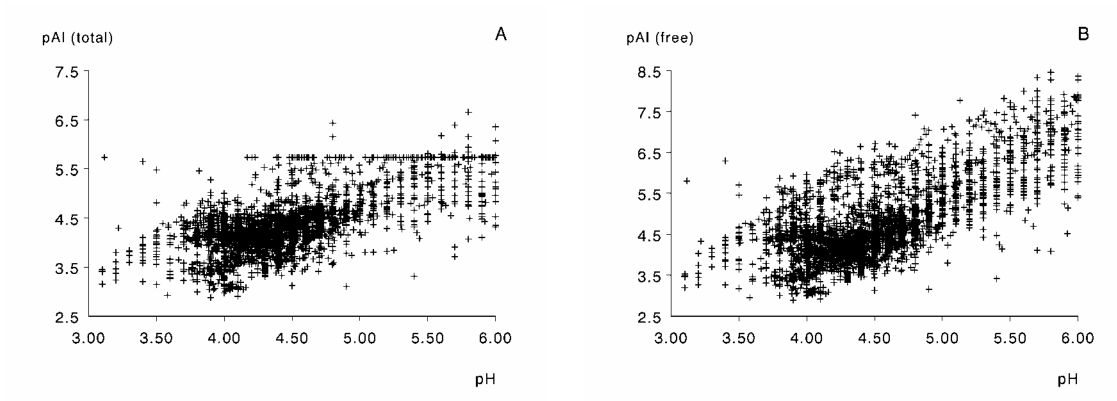


Figure 7.6 Relation between pH and pAl for total (A) and free Aluminium (B).

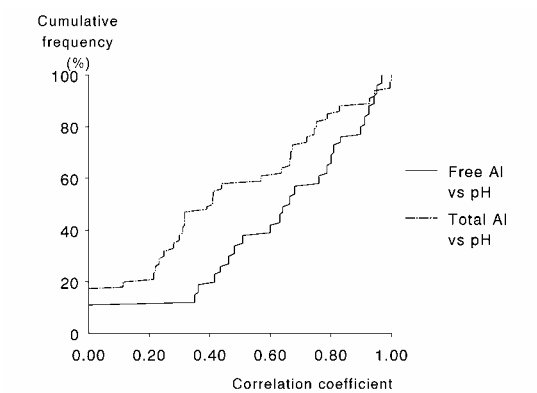


Figure 7.7 Cumulative frequency distribution of the correlation coefficient between pH and total and free Al

Figure 7.8 show the calibrated values of K_{Alox} and α in equation 7.2 for several selections from the data set. This figure shows α varies between 1 and 2.5 for plots with a good correlation between Al and pH (r > 0.5). Values for K_{Alox} are highly variable and mostly range between -5 and 3.

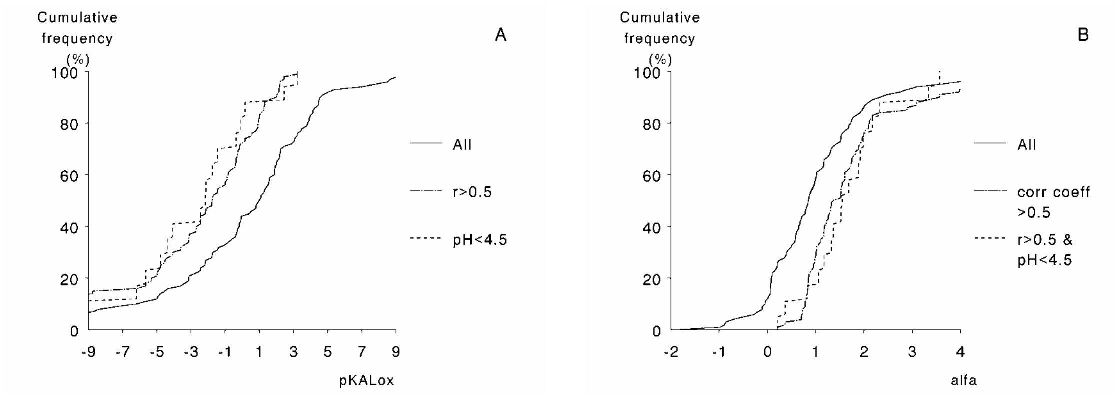


Figure 7.8 *Cumulative frequency distribution of the logarithmic Al dissolution constant (A) and the exponent α (B) in the free Al-pH relationship using the whole data set (all) and plots with a good correlation between Al and pH (r > 0.5) including all soil or acid soils only (pH<4.5).*

Plotting α against pK_{Alox} (figure 7.9) reveals a very strong relationship between these two parameters. This, at first sight remarkable relationship, is explained by the fact that for smaller α, K_{Alox} has to increase to get the same aluminium concentration in other words there is, of course, a strong relationship between the intercept from the regression of pKAlox against pH and the slope. It also shows that the simple gibbsite equilibrium with a -pK value of about 8 and α=3 holds only for a very few plots.

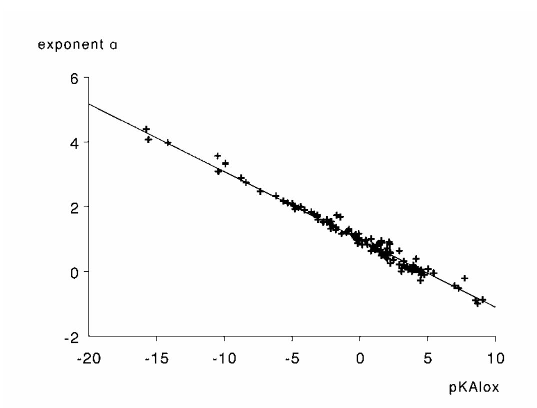


Figure 7.9 *Relationship between α and pK_{Alox}.*

Cation exchange constants

In SMART the exchange reactions in non-calcareous soils are described by Gaines-Thomas equations using concentrations instead of activities:

$$\frac{f_H^2}{f_{BC}} = K_{HBC} \cdot \frac{[H^+]^\beta}{[BC^{2+}]} \quad \text{and} \quad \frac{f_{Al}^2}{f_{BC}^3} = K_{AlBC} \cdot \frac{[Al^{3+}]^2}{[BC^{2+}]^3} \tag{7.3}$$

where K_{HBC} and K_{AlBC} are the Gaines-Thomas selectivity constants for H-BC exchange and Al-BC exchange, respectively. Since the exchange complex is assumed to comprise H^+, Al^{3+} and BC^{2+} only, charge balance requires that

$$f_{BC} + f_{Al} + f_H = 1 \tag{7.4}$$

Dividing the Eqs. 7.3, inserting the relationship between Al and H (Eq. 7.2) and taking the square root one can express f_{Al} as function of f_H and $[H]$ alone:

$$f_{Al} = K \cdot [H]^{-\gamma} \cdot f_H^3 \quad \text{with} \quad K = \sqrt{\frac{K_{AlBC} \cdot K_{Alox}}{K_{HBC}^3}} \quad \text{and} \quad \gamma = 3\beta/2 - \alpha \tag{7.5}$$

which allows to eliminate f_{Al} from eq. 7.4:

$$f_H + K \cdot [H]^{-\gamma} \cdot f_H^3 = 1 - f_{BC} \tag{7.6}$$

For a plot for which observations of $[H]$, $[Al]$ (as derived from $[H]$, $[Al_{tot}]$ and DOC), $[BC]$, f_{Al} and f_{BC} (and thus also f_H) are available, one could easily compute the site-specific values of K_{HBC} and K_{AlBC}. For the Intensive Monitoring plots, however, $[H]$, $[Al]$ and $[BC]$ are generally available but at the adsorption complex only f_{BC} (base saturation) is known. The values of K_{HBC} and K_{AlBC} can thus only be computed if one, e.g., assumes the pH-dependent relationship of eq. 7.5 between f_H and f_{Al} to be valid for all plots. Taking the logarithm of eq. 7.5,

$$\log_{10} \frac{fAl}{fH^3} = \gamma \cdot pH + \log_{10} K \tag{7.7}$$

and using the values on f_{Al}, f_H and $[H]$ from Dutch measurements for 531 forest soils layers (De Vries and Leeters, 2001; De Vries and Posch, 2003a), values of $\log_{10}K$=-3.53 and of γ=1.04 were obtained by linear regression.

Using those values of γ and K and measurements of $[H]$ and f_{BC} from the Intensive Monitoring plots, the cubic eq. 7.6 was solved to obtain f_H, and then f_{Al} from eq. 7.5. The exchange equations (eq. 7.3) were then used to estimate the exchange constants, using measurements of $[H]$ $[Al]$ and $[BC]$ in soil solution. The exponent β was obtained as β=2(γ+α)/3, with α derived earlier.

Figure 7.10 shows the cumulative frequency distributions of the calibrated values for the exchange constants and β. The figures shows that the exchange constants are highly variable, but similar ranges in K_{AlBC} have been found comparing data for about 200 plots in the Netherlands (De Vries and Leeters, 2001) for which the 5, 50 and 95 percentile are plotted in the graph as well. The variability in β is much smaller; it varies mostly between 1 and 2.5 with a median value of about 1.5. Figure 7.11 shows that the relationship between exchange constants and soil texture, which is sometimes assumed (especially for K_{AlBC}), is not really confirmed by the results from the calibration.

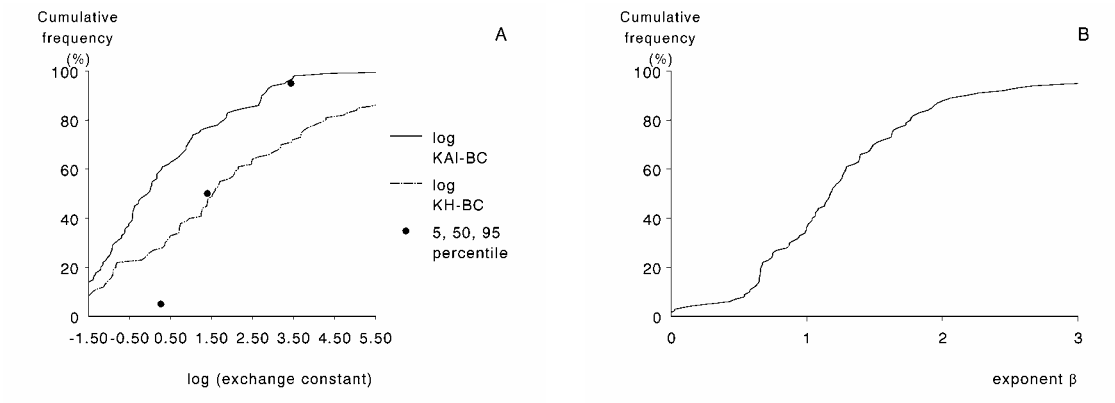


Figure 7.10 Cumulative frequency distributions of the calibrated exchange constants (A) and the exponent β (B)

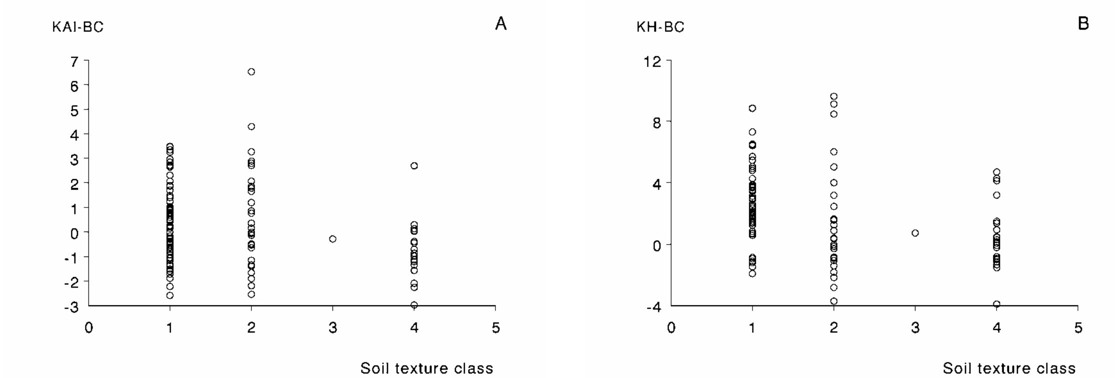


Figure 7.11 Exchange constants of Al against base cations (A) and protons against base cations (B) versus texture class

Nitrogen transformation parameters

In SMART the leaching of the nitrogen compounds NO_3 and NH_4 is calculated from the difference between input by deposition and the removal by growth uptake, immobilisation and denitrification, while accounting for the effect of nitrification. Denitrification and nitrification are modelled as fractions of the net nitrate and ammonium input, respectively. Growth uptake and immobilisation fluxes of both ions (NO_3^- or NH_4^+) are assumed proportional to their share in the deposition. The equations describing the nitrogen transformations in SMART are described in Annex 4.

Using the data from the Intensive Monitoring sites, the different N fluxes were estimated in the following way: Total deposition of NO_x and NH_3 was computed from bulk and throughfall measurements according to the procedure described in De Vries et al. (2001). The net growth uptake of N, N_{gu}, was derived as described earlier. The leaching fluxes of nitrate, ammonia, $NO_{3,le}$ and $NH_{4,le}$ were computed by multiplying (on a monthly basis) the measured concentrations of NO_3 and NH_4 with simulated downward water fluxes. Leaching of dissolved organic nitrogen, DON_{le}, was computed by multiplying the leaching of dissolved organic carbon (DOC) with the measured C/N ratio of the organic matter of the solid phase of the topsoil as this ratio gives a reasonable estimate of the C/N ratio of the dissolved organics (Michalzik and Matzner, 1999; Smolander et al., 2001).

In SMART the total N balance is given as:

$$N_{td} - N_{gu} - N_{im,t} - N_{im,acc} - N_{de} = N_{le} \tag{7.8}$$

The time (C/N ratio) dependent N immobilisation is computed from the immobilisation fraction fr_{im} through:

$$N_{im,t} = f_{im} \cdot \left(N_{td} - N_{gu} - N_{im,acc}\right) \tag{7.9}$$

Observational and experimental evidence (e.g. Gundersen et al., 1998b) shows a correlation between the C/N ratio and the amount of N retained in the soil organic layer. According to Dise et al. (1998b) and Gundersen et al. (1998a) the forest floor C/N ratios may thus be used to assess risk for nitrate leaching. Gundersen et al. (1998a) suggested threshold values of >30, 25 to 30, and <25 to separate low, moderate, and high nitrate leaching risk, respectively. This information has been used in SMART to calculate nitrogen immobilisation as a fraction of the net N input, linearly depending on the C:N ratio between a minimum and maximum value (see Annex 7.3). Below a C/N ratio of 15 there is no net immobilisation (fr_{im} = 0), whereas above a C/N ratio of 40 fr_{im} equals 1.

Denitrification is (in case of complete nitrification) computed by:

$$N_{de} = f_{de} \cdot \left(N_{td} - N_{gu} - N_{im,t} - N_{im,acc}\right) \tag{7.10}$$

The denitrification fraction f_{de} was assigned a default value depending on soil texture and gleyic features of the site according to Table 7.2

Table 7.2 Denitrification fraction fde as a function of texture and gley class.

Gley class	Denitrification fraction (-) per texture class				
	1	2	3	4	5
0	0.1	0.3	0.3	0.5	0.5
1	0.3	0.5	0.5	0.6	0.6
2	0.5	0.6	0.7	0.7	0.7
3	0.7	0.8	0.8	0.8	0.8

Using the measurements, N_{td}, N_{gu}, and N_{le} were computed as average values over the observation period, f_{im} was computed from the C:N ratio in the upper 20 cm of the soil and f_{de} was assigned according to Table 1. Then $N_{imm,acc}$ being the constant N immobilisation that is assumed to be associated to the build-up of organic matter is obtained by combining the equations 7.8, 7.9 and 7.10:

$$N_{im,acc} = N_{td} - N_{gu} - \frac{N_{le}}{(1 - f_{im}) \cdot (1 - f_{de})} \tag{7.11}$$

Although in the above derivations complete nitrification is assumed, one can compute the actual nitrification fraction (f_{ni}) from the measurements according to (see Annex 4):

$$f_{ni} = 1 - \frac{NH_{4,le}}{f_{td} \cdot NH_{3,td}} \quad \text{where} \quad f_{td} = \frac{N_{td} - N_{gu} - N_{im,t} - N_{im,acc}}{N_{td}} \tag{7.12}$$

Figure 7.12 shows the cumulative frequency distributions of the denitrification fraction (estimated from texture and gley class), time-dependent immobilisation fraction (estimated from C:N ration in the topsoil) and computed nitrification fraction (c.f. Eq. 7.12). This figure shows that the denitrification fraction is low at most plots (mostly below 0.3) indicating that most soils are reasonably well drained. Immobilisation fractions vary between 0 (for C:N ratio's below 15) to 0.9 (C:N ratio's close to 40) illustrating the range in N saturation over the various plots over Europe. The nitrification fraction exceeds 0.9 at 80 % of the plots which shows that assumption of complete nitrification (used in the derivation of Eq. 7.11) is valid for the vast majority of the plots.

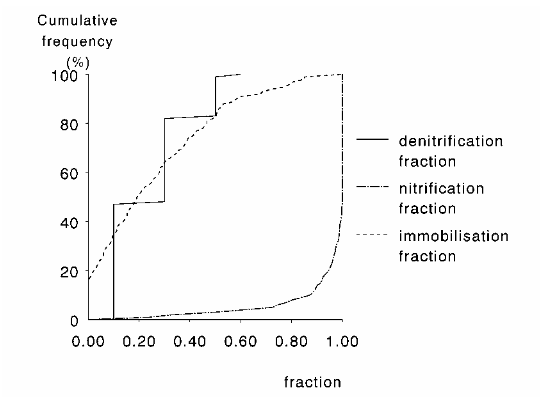


Figure 7.12 Cumulative frequency of estimated denitrification and time-dependent immobilisation fractions and of computed nitrification fraction at the plots

Time independent N immobilisation strongly varies between the plots. At about 20 % of the plots time independent N immobilisation equals 0 whereas the median value is about 5 $kg.ha^{-1}.yr^{-1}$. This shows that at many plots a substantial amount of N-loss cannot be explained by the assumed fractions of denitrification and C:N dependent N immobilisation. It must be kept in mind that this N immobilisation is a sort of rest-term in the N balance and therefore inherits all the uncertainties in the other terms such as leaching, uptake and total input.

Sulphate adsorption parameters

The amount of sulphate adsorbed, $SO_{4,ad}$ ($meq.kg^{-1}$), is assumed to be in equilibrium with the solution concentration and is described in SMART by a Langmuir isotherm (e.g. Cosby et al., 1986):

$$SO_{4,ad} = \frac{[SO_4]}{S_{1/2} + [SO_4]} \cdot S_{max} \tag{7.13}$$

where S_{max} is the maximum adsorption capacity of S in the soil ($meq.kg^{-1}$) and $S_{1/2}$ the half saturation concentration (eq/m^3). The parameters S_{max} and $S_{1/2}$ are not known, nor are there measurements of $SO_{4,ad}$. A *change* in $SO_{4,ad}$, i.e. the amount ad- or desorbed during one timestep, can in principle be estimated by looking at *differences* between S deposition and the amount of sulphur leached. However, the data available did not allow identifying meaningful values of S_{max} and $S_{1/2}$. For example, in many cases the estimated amount adsorbed increased despite a decrease in the concentration, in contradiction to the basic model assumption. Consequently, sulphate adsorption was neglected in the SMART simulations.

Soil properties

Data on soil properties that are needed are bulk density, cation exchange capacity (CEC) and sulphate sorption capacity (SSC) but the latter value was ultimately not used (see above). Values for the soil bulk density were taken from the voluntary soil physical data or, if not available, related to the organic carbon and clay content of the plot according to:

$$\rho = \begin{cases} 1/\left(0.625 + 0.05 \cdot C_{org} + 0.0015 \cdot clay\right) & \text{if} \quad C_{org} \le 5\% \\ 1.55 - 0.0814 \cdot C_{org} & \text{if} \quad 5\% < Corg < 15\% \\ 0.725 - 0.0337 \cdot \log_{10}\left(C_{org}\right) & \text{if} \quad C_{org} \ge 15\% \end{cases} \tag{7.14}$$

where ρ is the bulk density (g.cm^{-3}), C_{org} is the organic carbon content (%) and *clay* is the clay content (%). Equation 7.14 is based on data by Hoekstra and Poelman (1982) for mineral soils and from Van Wallenburg (1988) for peat soils. The middle part of equation 7.14 is a linear interpolation.

Values for cation exchange capacity (CEC) were taken from the soil survey data to arrive at a total value for the root zone. For plots where CEC was measured in an unbuffered solution, the CEC at pH 6.5 was calculated by assuming a relation between CEC, pH (in solution), clay and organic carbon (OC) according to (Helling et al., 1964):

$$CEC = (0.44 \cdot pH + 3.0) \cdot \%clay + (5.1 \cdot pH - 5.9) \cdot \%OC \tag{7.15}$$

Cation exchange capacity at pH 6.5 was then computed by scaling the measured CEC with the ratio between computed CEC at pH 6.5 and computed CEC at the soil solution pH according to:

$$CEC_{updated(pH6.5)} = CEC_{measured(pHactual)} \cdot (CEC_{calculated(pH6.5)} / CEC_{calculated(pHactual)}) \tag{7.16}$$

Evaluation of the modelling adequacy

In order to evaluate the quality of the model predictions, a comparison was made with the measurements by: (I) visual inspection for selected plots using monthly values, (ii) a scatter plot comparing the annual predictions and measurements for all plots and (iii) an evaluation of the deviation between predictions and measurements by various statistical measures. The measures used were the Normalised Mean Error (NME) and the Mean Absolute Error, as described below:

$$NME = \frac{\sum_{i=1}^{N}(P_i - O_i)}{N\overline{O}} \tag{7.17}$$

$$MAE = \frac{\sum_{i=1}^{N}| P_i - O_i |}{N} \tag{7.18}$$

where P_I *and* O_I are the predicted and observed value I, $\bar{O}$ is the average of the observations and N is the number of observations.

The NAE compares predictions and observations on an average level and expresses the bias in the average values of predictions compared to the observations (systematic underestimation or overestimation) but is rather sensitive to outliers. The closer the value to 0, the better. The mean absolute error is not so sensitive for outliers and does not allow for compensation of under- and overestimates, as the absolute value of the error is summed (Janssen and Heuberger, 1995)

7.2.4 Deposition scenarios used in model predictions

For the simulations, the trends in SO_2, NO_x and NH_3 deposition were derived using RAINS country emissions (Cofala and Syri, 1998a, b) and transfer matrices derived from the EMEP long-range transport model (Bartnicki et al., 2002) for 1960 to 2010. After 2010 deposition is assumed constant. These trend curves were scaled by the average computed total deposition (based on bulk and throughfall measurements) for the period 1996-2000 so that the EMEP time series coincide with the plot-specific deposition for that period. Base cation deposition was assumed constant over the entire simulation period and was set equal to the total deposition (based on bulk and throughfall measurements) for each plot. After 2000, two scenario's were used (1) the implementation of the Gothenburg Protocol by 2010 and (2) the implementation of maximum technically feasible reduction measures by 2010 (MFR). The Gothenburg protocol signed in 1999 is the latest of eight protocols that have been adopted under the Convention on Long-range Transboundary Air Pollution which set national emission ceilings. It aims to abate acidification, eutrophication and ground-level ozone. Once this Protocol is fully implemented, Europe's sulphur emissions should be cut by at least 63% and NOx emissions by 41% compared to 1990.

Figure 7.13 shows the historical and predicted future emissions of S and N for the two protocols. The figure clearly shows that for sulphur large reductions have been achieved and that future reductions are planned. For N some emission reduction occurred, but only with the MFR scenario a significant decrease in N emissions can be expected.

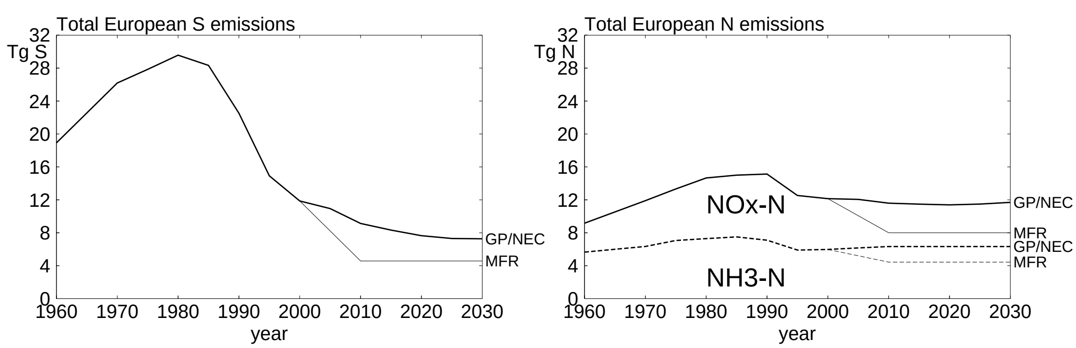


Figure 7.13 Emission reduction scenarios for S and N

7.3 Results

7.3.1 Model calibration

Comparison of model runs for selected plots

The model Smart was applied for the period 1960 to 2030. This means that initial estimates are needed (for the year 1960) for base saturation and C:N ratio. C:N ratio for 1960 was recomputed from the observed C:N ratio at the plots (mostly somewhere in the period 1990-1995) and the historical N input and uptake between 1960 and this time-point. For each plot, the base saturation in 1960 was calibrated by running the model between 1960 and the year with the observation of base saturation thereby adjusting the initial base saturation in 1960 until the observed base saturation is correctly reproduced. Figures 7.14 and 7.15 show examples of SMART model output compared with observations for various soil solution concentrations for two example plots in Germany (with a better than average fit) and Sweden (with a worse than average fit). Figure 7.14 shows a good agreement between (trends in) simulations and observations for all variables, especially for nitrate and aluminium.

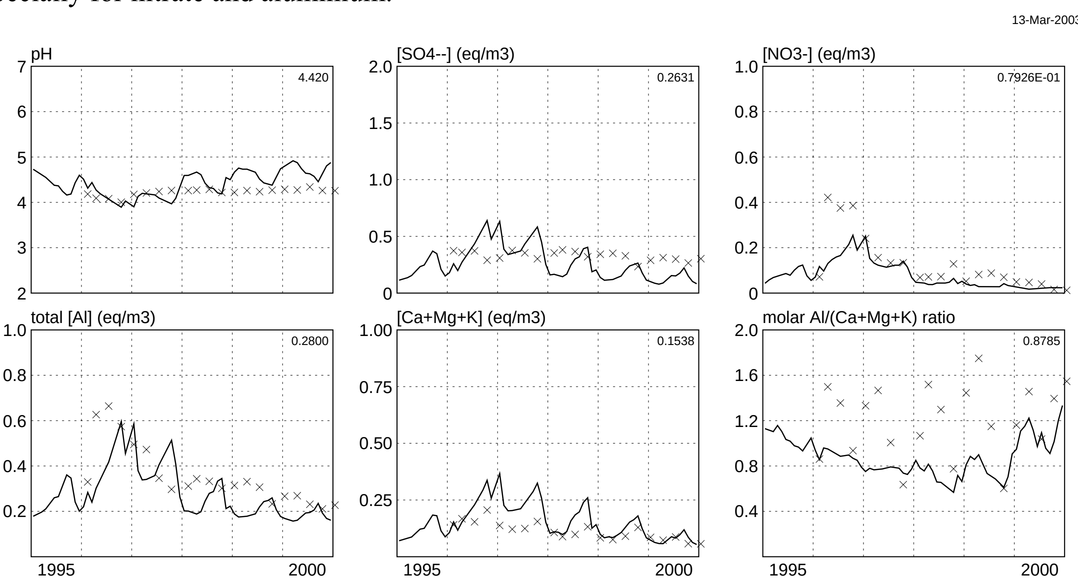


Figure 7.14 Observed (crosses) and simulated monthly soil solution concentrations for a plot in Belgium

Figure 7.15 shows the results of SMART for a plot in Sweden. For this plot pH and base cations are well simulated but not all peaks in Al concentration could not be reproduced and SO_4 is underestimated.

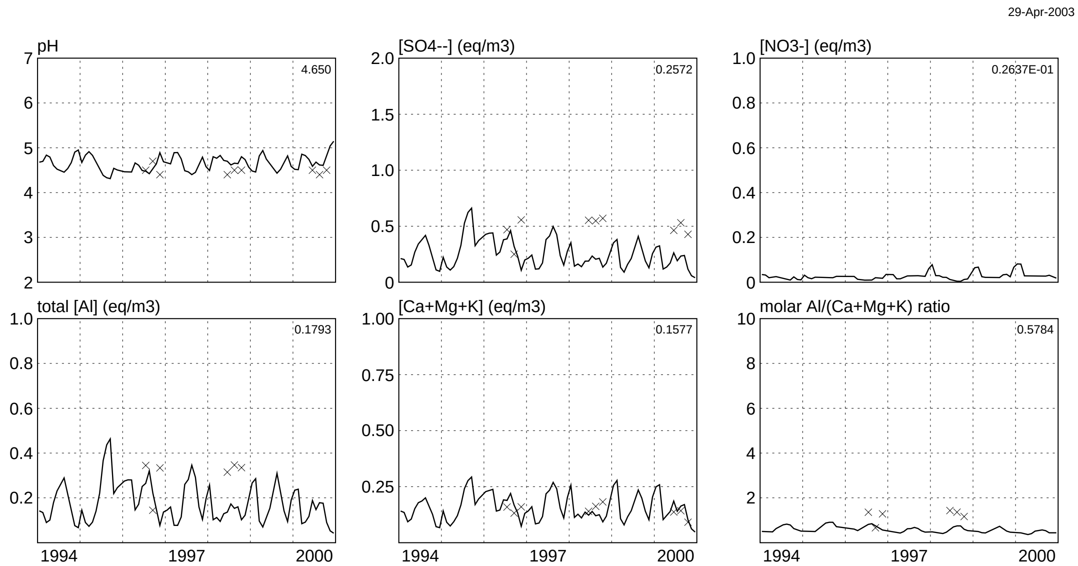


Figure 7.15 Observed (crosses) and simulated monthly soil solution concentrations for a plot in Sweden

Comparison of model runs for all plots:

Figure 7.16 shows the goodness-of-fit computed for all plots with at least 24 concentration measurements (two years of monthly data). Figure 7.16a shows the distribution of the Normalised Mean Error (NME) which gives a crude impression whether the simulations are an underestimate (negative NME) or an overestimate (positive NME). Ideally NME should be close to zero. This figure shows that pH is well simulated only for a low percentage of plots pH is underestimated by about 20-30 %. For sulphate and nitrate the median NME is close to 0 (no systematic under or overestimation) but high discrepancies between observations and model results occur for a number of plots. For sulphate and aluminium the deviation lies mostly between -50 and 50 %, for nitrate much higher overestimations occur. Very high errors in the simulated sulphate concentrations occur for some plots where sulphate is 'generated' in the soil itself due to weathering of parent material containing sulphate (Gypsum) or, for one plot, due to pyrite oxidation (!), processes not accounted for in the model.

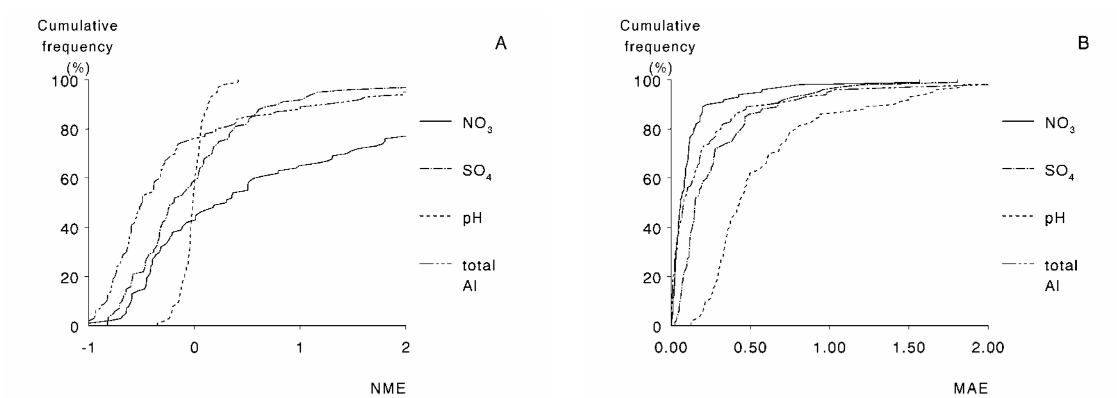


Figure 7.16 Goodness-of-fit expressed by the NME (A) and MAE (B) measures

Because the NME is the difference between modelled and observed values divided by the mean of the observations, low mean values can create high NME's. Furthermore this measure is rather sensitive to outliers which often occur for nitrate and aluminium, as for many plots the measurements show high variations over short time periods that cannot be reproduced by the model (see Figure 7.15). Therefore Figure 7.16b shows the Mean Absolute Error (MAE) that simply gives the average absolute difference between observed and simulated values (for concentrations in eq.m^{-3} and for pH in pH units). This figure shows that the absolute error in simulated nitrate concentrations is low, mostly below 0.1 eq.m^{-3}. For pH the median average absolute error over all plots is about 0.45 pH units: in combination with figure 7.16a this shows that on average the pH is well simulated (expressed by an NME close to zero) but that both under- and over estimations occur during the simulation (expressed by a median MAE of about 0.45 pH units).

Figure 7.17 shows the observed values averaged over the entire measurement period for each plot versus the simulated concentrations for this period for sulphate, nitrate, pH and total aluminium.

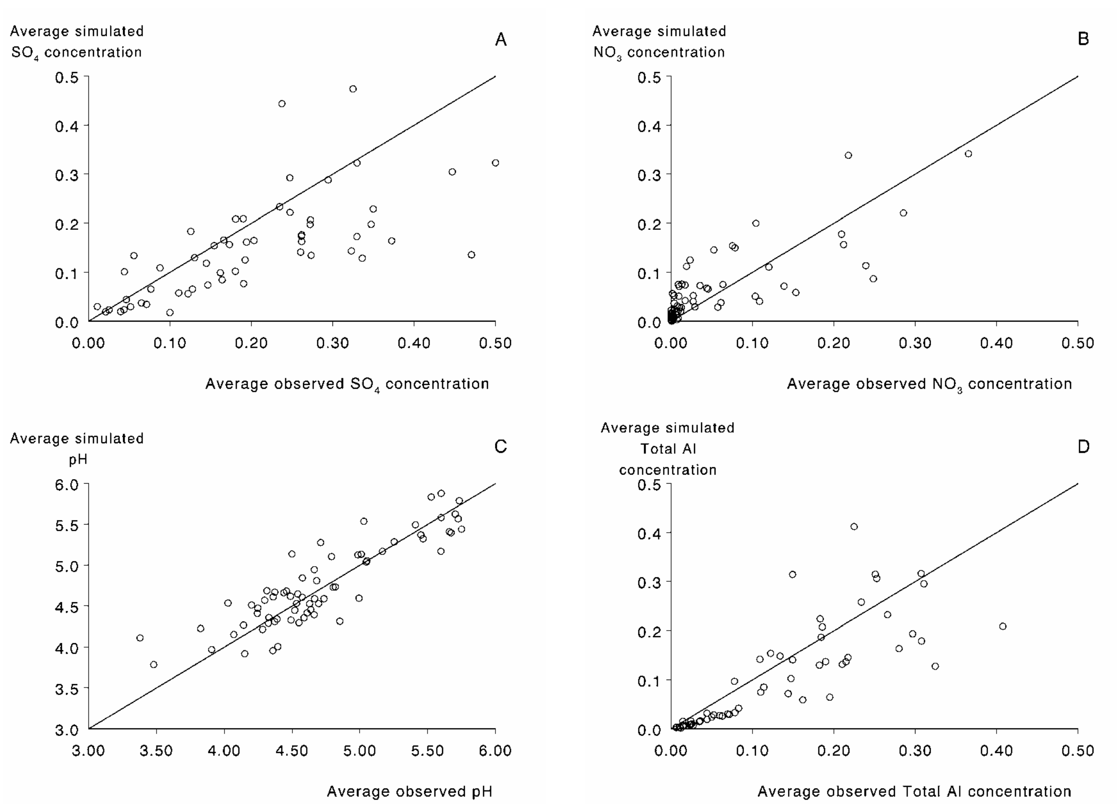


Figure 7.17 Observed versus simulated values for SO_4 (A), NO_3 (B), pH (C) and total Al (D) for 75 fully calibrated plots

Only those plots are shown for which nitrogen transformations, cation exchange and the free aluminium equilibrium could be calibrated. In practice this means that these are the 75 plots with measurements of all major cations and anions in both deposition and soil solution (including DOC in soil solution). The graphs show that there is a good correlation between measured and observed concentrations for most plots especially for pH and aluminium. Sulphate is for some plots clearly underestimated which might be due to weathering of sulphate containing parent material or release of sulphate by sulphate desorption, processes not accounted for in the simulations.

Underestimation of the aluminium concentration mainly occurs at plots with a high variation in measured concentrations that include peaks in Al concentration that the model cannot reproduce.

7.3.2 Model application for the period 1970-2030

Model runs for selected plots

Figures 7.18 shows an example of SMART model predictions for the period 1970-2030 for the same example plot in Germany for which we previously compared output with observations for various soil solution concentrations. The figure shows that recovery of the chemical soil status takes place under both the Gothenburg protocol and the Maximum Feasible Reduction (MFR) scenario as illustrated by the increase in pH and the decrease in Al/BC ratio. The MFR scenario is more effective than the Gothenburg scenario, but differences are small. The strongest effect of the MFR scenario is on the nitrate concentration: by 2030 the concentration simulated for the MFR scenario is much lower than simulated for the Gothenburg scenario.

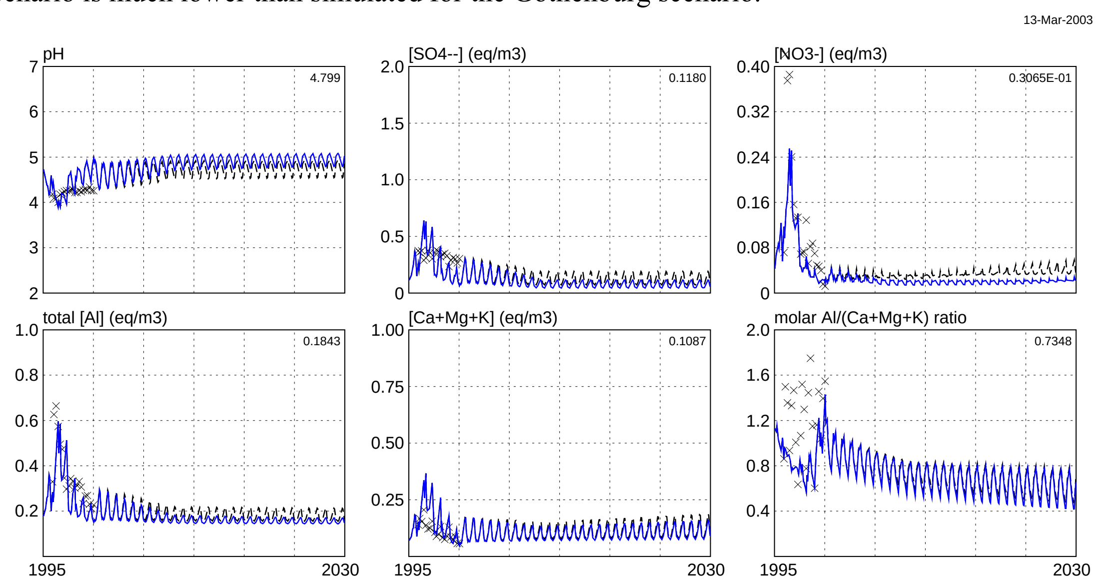


Figure 7.18 Scenario analysis for the plot of Figure 7.14 for the Gothenburg scenario (black lines) and the Maximum Feasible Reduction scenario (blue lines)

Model runs for all plots

Figure 7.19 and 7.20 shows the temporal evolution of 7 percentiles (5, 25, 50, 75 and 95) for 7 output variables of SMART for the period 1970 - 2030. The non-smooth behaviour of the lines between 1996 and 2000 reflects the use of year-specific data within this period (especially precipitation surplus), whereas for the other years average values were used. The figure shows a steep decline in SO_4 concentration caused by the strong reduction of sulphur deposition over Europe during the last two decades. This is accompanied by a decline in the loss of divalent base cations from the exchange complex. Patterns for pH are less distinct, only the lowest percentiles show a clear improvement over time. Aluminium follows the pH pattern; here the high percentiles show the most improvement (fewer high Al concentrations over time). Nevertheless about 25% of the plots have Al concentrations above the critical value of 0.2 $meq.l^{-1}$. About 5 % of the plots have Al/BC ratios above 1.0 in 1970 but this percentage decreases towards 2010. Nitrate concentrations also decline between 1990 and 2030 but at a number of plots high concentrations persist. At a few plots nitrate concentrations will even increase as a result of reduced forest growth, and thus reduced N uptake, due to ageing of the forest that exceeds the (limited) reduction in N input.

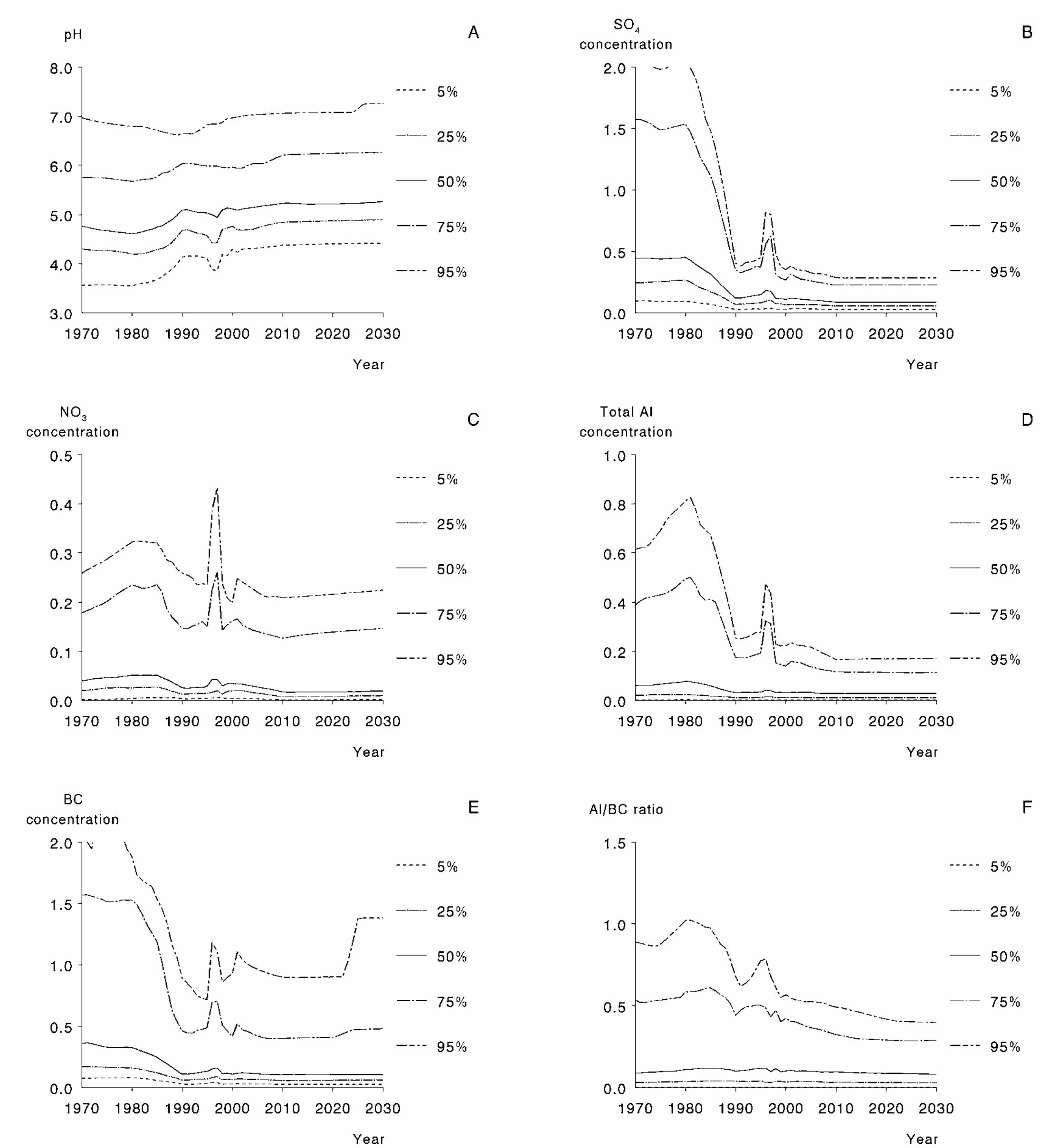


Figure 7.19 Graphs of the 5th, 25th, 50th, 75th and 95th percentiles of the distributions of 6 output variables of SMART between 1970 and 2030 for about 200 plots.

Base saturation (fBC, figure 7.20) improves over time for most plots but for a number of plots where acid inputs remains relatively high, base saturation will still decrease in the future as illustrated by the lines for the 25 and 5 percentile.

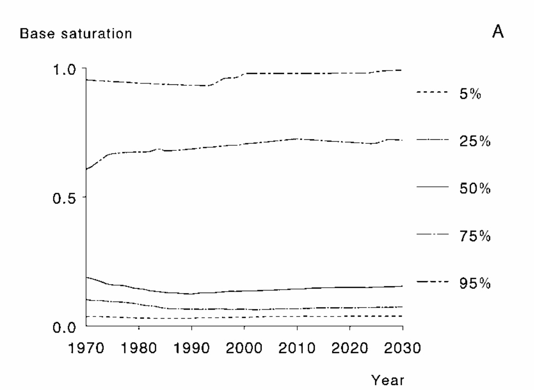


Figure 7.20 Graph of the 5th, 25th, 50th, 75th and 95th percentiles of the distributions of base saturation simulated by SMART between 1970 and 2030 for about 200 plots.

Figure 7.21 shows the time development of the median sulphate, nitrate and aluminium concentrations and pH for the two emission reduction scenarios.

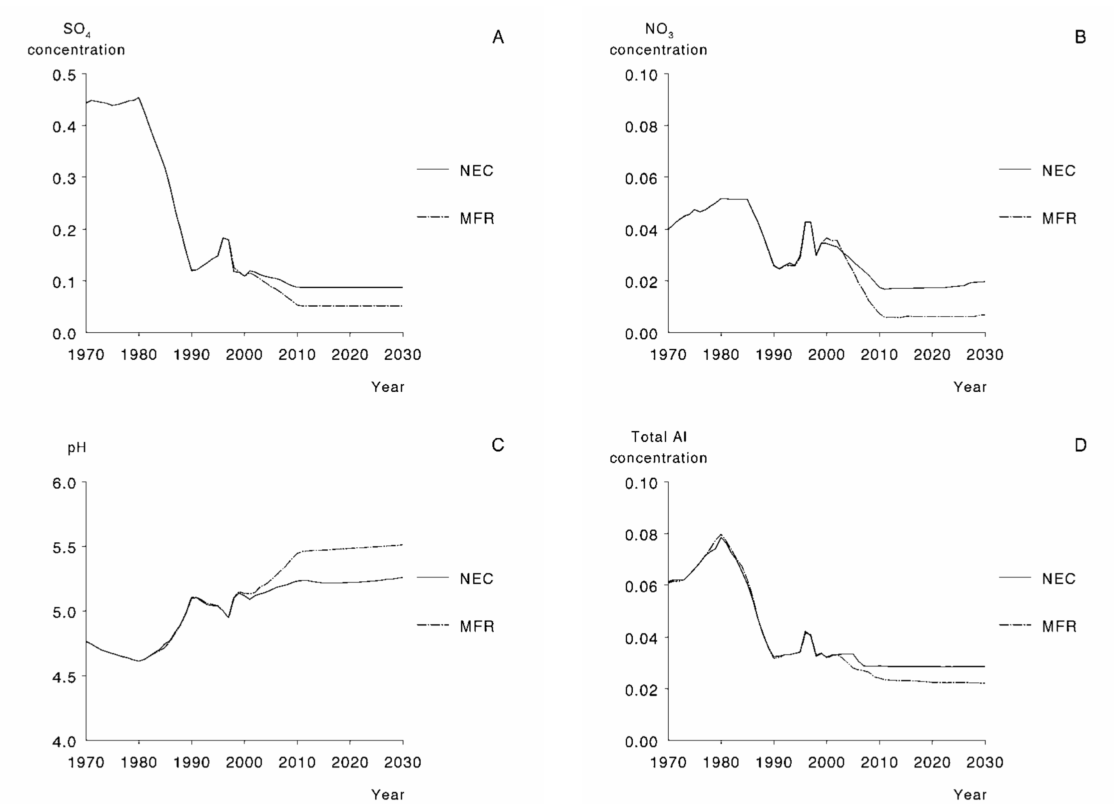


Figure 7.21 Graphs of the median values for SO_4, NO_3, Al and pH simulated by SMART between 1970 and 2030 for about 200 plots using two different emission reduction scenarios.

It shows that the MFR scenario leads to lower aluminium concentrations and higher pH values in 2010 than the Gothenburg scenario. The most pronounced difference occurs for nitrate as the median concentration over all plots in 2010 under the MFR scenario is less than half the median concentration under the Gothenburg scenario. This is due to the fact that under the Gothenburg scenario reductions in especially N emissions are much lower than when all available technology would be applied (MFR). The increase in median concentrations between 1995 and 2000 is an artefact caused by the use of year-specific hydrology for this period.

Geographic variation of predicted soil solution chemistry over time

Figure 7.22 shows the sulphate and nitrate concentrations at the modelled sites for the years 1970 and 2030 for the Gothenburg protocol. This figure illustrates the strong decline in sulphur concentrations also seen in figure 7.19. It also shows the high spatial variability in SO_4 soil solution concentration with the highest values in Central Europe. It also shows that the reduction in N emissions leads to lower nitrate concentrations at the plots, but that the decrease in concentration is much less than for sulphate (see also figure 7.19). Highest nitrate concentrations are found in Belgium and parts of Germany, the United Kingdom and Denmark.

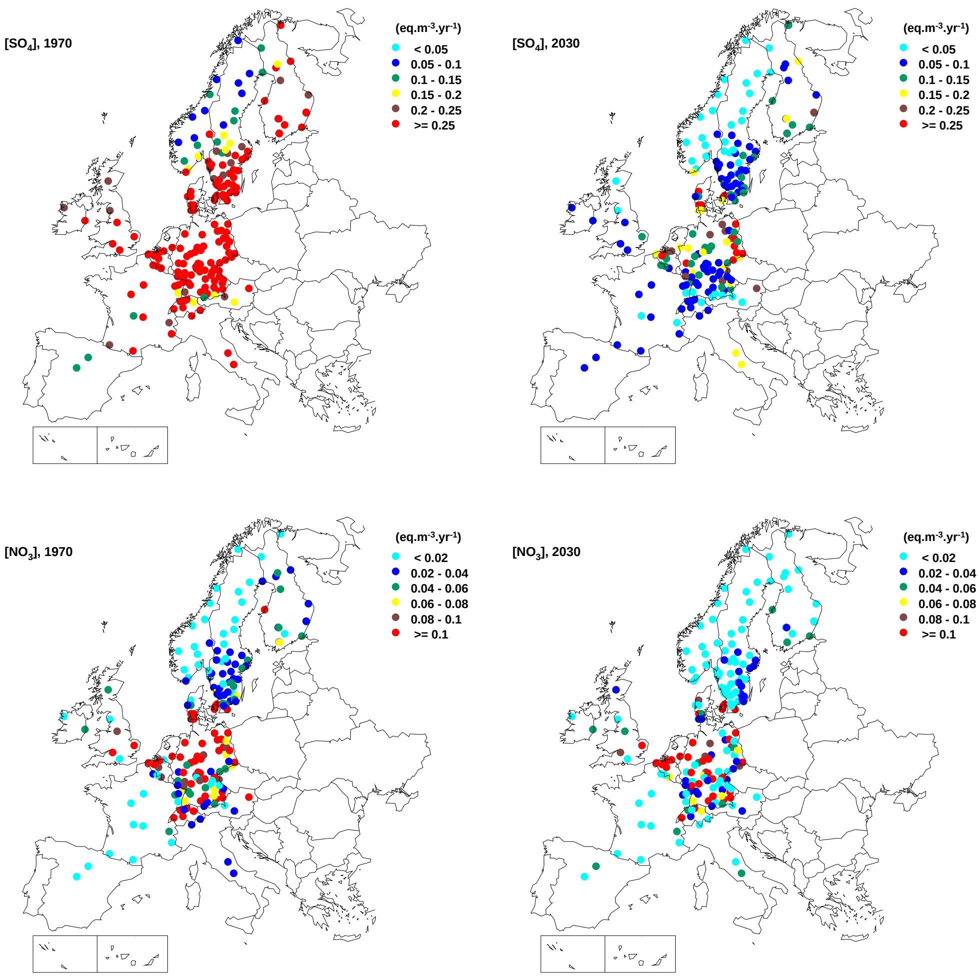


Figure 7.22 Annual mean soil solution concentrations of SO_4 and NO_3 at the modelled sites in 1970 and 2030 for the Gothenburg protocol

Figure 7.23 shows the time development in pH and the total Al concentration. It shows that for many plots simulated pH in 2030 is substantially higher than in 1970. For Al only the higher concentrations are strongly reduced over time (see also figure 7.19). It also shows that the number of plots where Al concentrations are above the critical value of 0.2 meq.l^{-1} hardly reduces over time.

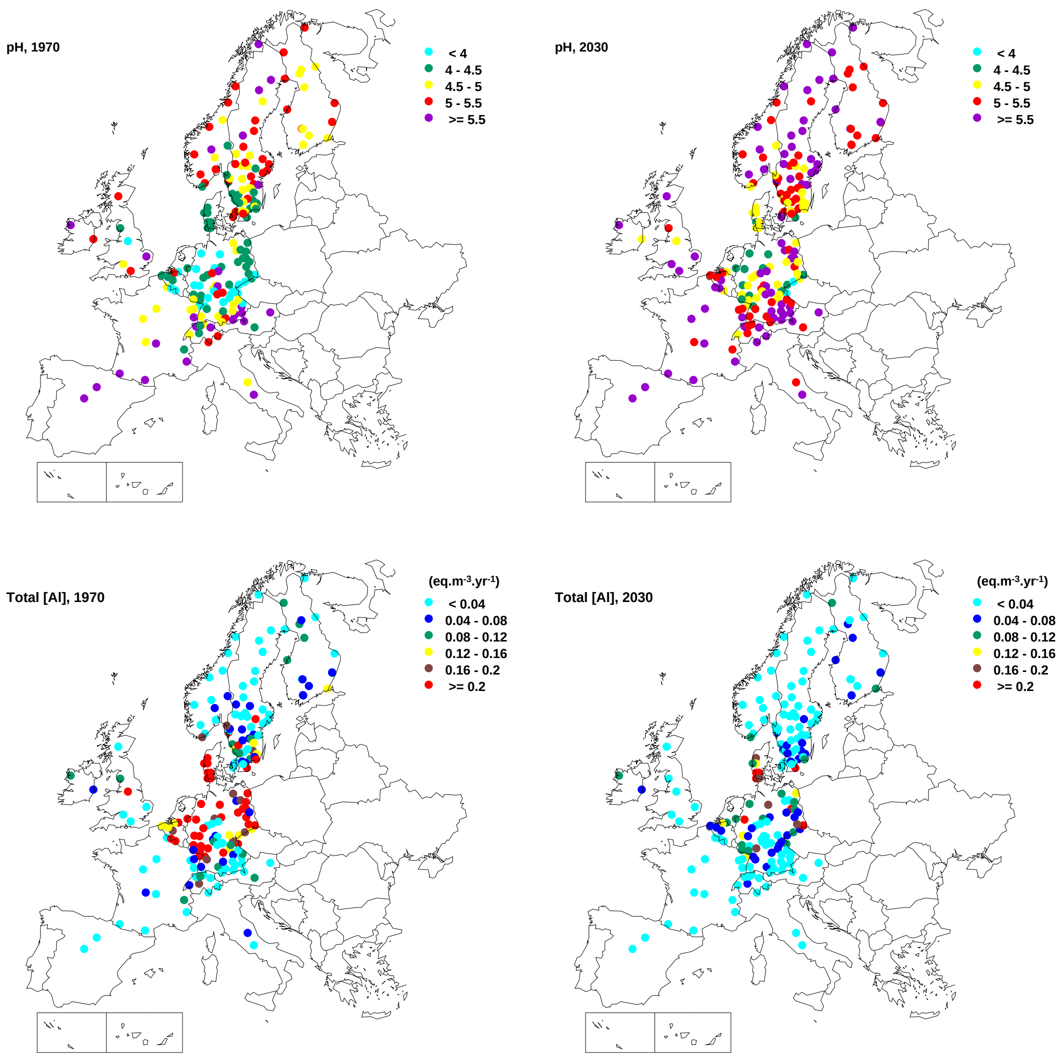


Figure 7.23 Annual mean pH and soil solution concentration of total Al at the modelled sites for 1970 and 2030 for the Gothenburg protocol

7.4 Discussion and conclusions

To evaluate effects of future deposition scenario's on Intensively Monitored plots, the model SMART was applied to about 200 sites where data on deposition, soil and soil solution were available. A number of parameters in the model could be computed from the measurements of element input (total deposition) and element concentrations in the soil solution. These plot-specific parameters where then used to apply the model at the sites. The computation of plot-specific model-parameters showed e.g.:

- The relationship between pH and free Al is better than between pH and total Al, especially for pH values greater than 5, and can for most plots be well described by an equilibrium reaction

using a variable exponent. This exponent is mostly lower that 3 (as it is assumed in a gibbsite equilibrium).
- Exchange constants for Al-BC and H-BC are highly variable, but the exponent β in the relationship is mostly between 1 and 2. This high variability in exchange constants was also observed within the Netherlands (De Vries and Posch, 2003b).
- The total N budgets reveals that for many plots the loss (removal) of N cannot be described by uptake, leaching, denitrification and C:N dependent N immobilisation alone. Only when a time-independent N immobilisation is assumed the budget can be closed. Uncertainties in the various terms within this budget are, however, high.
- Nitrification is almost complete at most of the plots, i.e. hardly any ammonia is leached.
- Parameterisation of the sulphur adsorption process in SMART was not possible with the available data on S input and -output.
- On average pH values are very well simulated although the model is not able to simulate all intra-year variability. There is no systematic deviation in simulated sulphate and nitrate concentrations but at some plots the model cannot accurately predict measured concentrations. For a number of plots peaks in the, on average low, aluminium concentration cannot be reproduced by the model leading to an underestimation of average Al concentrations. However, nitrate and aluminium concentrations on many plots are very low, leading to high relative errors in the simulations but absolute errors in the model predictions are often very low.

Evaluation of the Gothenburg protocol on emission reductions for the period 1970-2030 shows:
- A very strong reduction in sulphate concentrations between 1980 and 2000 in the soil due to the high reductions in sulphur emissions
- A reduction of the nitrate concentrations by the year 2010 for most plots but most striking for the plots with the highest present N concentrations
- A reduction in the aluminium concentrations over time, most clearly for those plots where aluminium concentrations are currently (very) high.
- Aluminium concentrations above 0.2 $mol_c.m^{-3}$ occur for about 25 % of the plots in the beginning of the simulation period; this percentage decreases to about 5-10 % of the plots in 2030.
- Al/BC ratios above a critical value of 1 occur at about 5 % of the plots in 1970 and this percentage even decreases towards 2010.

It should be kept in mind, however, that it not sure whether the planned reductions foreseen in the Gothenburg protocol will be reached in 2010 as a number of countries still needs to strongly decrease emissions to attain the targets set.

Comparison of the Gothenburg protocol with the maximum feasible reduction scenario shows:
- that the MFR scenario leads to lower sulphate and aluminium concentrations in 2030 than the Gothenburg scenario
- that the MFR scenario is much more effective in reducing nitrate concentrations than the Gothenburg scenario

Appendix 7.1 Relationships between critical load models and dynamic soil models

To illustrate the relationships between critical load models and dynamic soil models, Figure A7.1.1 summarises the possible development of a (soil) chemical and biological variable in response to a 'typical' temporal deposition pattern (after Posch et al., 2002).

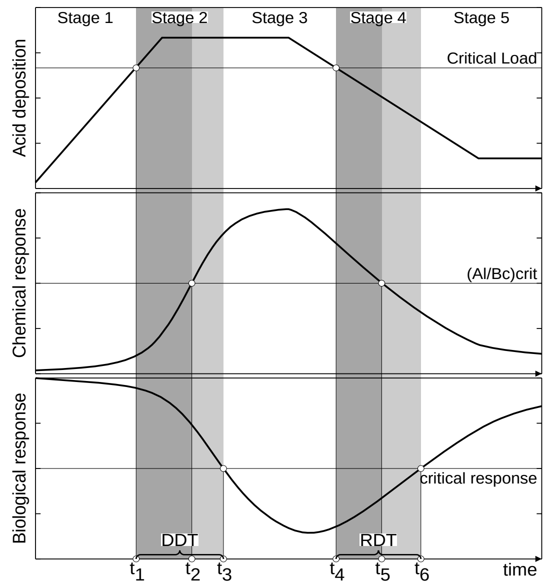


Figure A7.1.1 'Typical' temporal (past and future) development of the deposition (top), a soil chemical variable and the corresponding biological response. Also depicted are the critical values of those (chemical and biological) variables and the critical load derived from them. The delay between the (non-)exceedance of the critical load, the (non-)violation of the critical chemical criterion and the crossing of the critical biological response is indicated in grey shades, highlighting the Damage Delay Time (DDT) and the Recovery Delay Time (RDT) of the system.

Five stages can be distinguished:

1. *Stage 1*: Deposition was and is below the critical load (CL) and the chemical and biological variables do not violate their respective criteria. As long as deposition stays below the CL, this is the 'ideal' situation.
2. *Stage 2*: Deposition is above the CL, but (chemical and) biological variables still don't violate their respective criteria; there is a delay before this happens. Therefore, no damage is likely at this stage, despite the exceedance of the CL. We call the time between the first exceedance of the CL and first violation of the biological criterion the *Damage Delay Time* ($DDT=t_3–t_1$).
3. *Stage 3*: The deposition is above CL and both the chemical and biological criteria are violated. Measures have to be taken to avoid a (further) deterioration of the ecosystem.
4. *Stage 4*: Deposition is below the CL, but the chemical and biological criteria are still violated, and thus recovery has not yet occurred. We call the time between the first non-exceedance of the CL and the subsequent non-violation of both criteria the *Recovery Delay Time* ($RDT=t_6–t_4$).
5. *Stage 5*: This stage is similar to Stage 1. Deposition is below the CL and both criteria are no longer violated. Only at this stage can one speak of full ecosystem recovery.

Stages 2 and 4 can be further subdivided into two sub-stages each: Chemical delay times ($DDT_c=t_2–t_1$ and $RDT_c=t_5–t_4$; dark grey in Fig.A.1) and (additional) biological delay times ($DDT_b=t_3–t_2$ and $RDT_b=t_6–t_5$; light grey).

Appendix 7.2 The VSD, SMART and SAFE soil models

An overview of VSD, SMART and SAFE, being soil models of increasing complexity, is given in Table A7.2.1. Only a short description of the models can be given, but details can be found in the references cited. It should be emphasised that the term 'model' used here refers, in general, to a model system, i.e. a set of (linked) software (and databases) which consists of pre-processors for input data (preparation) and calibration, post-processors for the model output, and - in general the smallest part - the actual model itself.

Table A7.2.1 Overview of dynamic models that may be applied on a regional scale.

Model	Essential process descriptions	Layers	Essential model inputs
VSD	ANC charge balance Mass balances for BC and N (complete nitrification assumed)	One	CL input data + CEC, base saturation C/N ratio
SMART	VSD model + SO_4 sorption Mass balances for $CaCO_3$ and Al Separate mass balances for NH_4 and NO_3, nitrification Complexation of Al with DOC	One	VSD model + S_{max} and $S_{1/2}$ Ca-carbonate, Al_{ox} Nitrification fraction, f_{ni} pK values
SAFE	VSD model + Separate weathering calculation Element cycling by litterfall, root decay, mineralisation and root uptake	Several	VSD model + Input data for PROFILE Litterfall rate, parameters describing mineralisation and root uptake

The VSD model: The VSD model can be viewed as the simplest extension of the SMB model for critical loads. It only includes cation exchange and N immobilisation, and a mass balance for cations and nitrogen as described above, in addition to the equations included in the SMB model. In the VSD model, the various ecosystem processes have been limited to a few key processes. Processes that are *not* taken into account are: (i) canopy interactions, (ii) nutrient cycling processes, (iii) N fixation and NH_4 adsorption, (iv) interactions (adsorption, uptake, immobilisation and reduction) of SO_4, (v) formation and protonation of organic anions, (RCOO) and (vi) complexation of Al with OH, SO_4 and RCOO.

The VSD model consists of a set of mass balance equations, describing the soil input-output relationships, and a set of equations describing the rate-limited and equilibrium soil processes. The soil solution chemistry in VSD depends solely on the net element input from the atmosphere (deposition minus net uptake minus net immobilisation) and the geochemical interaction in the soil (CO_2 equilibria, weathering of carbonates and silicates, and cation exchange). Soil interactions are described by simple rate-limited (zero-order) reactions (e.g. uptake and silicate weathering) or by equilibrium reactions (e.g. cation exchange). It models the exchange of Al, H and Ca+Mg+K with Gaines-Thomas or Gapon equations. Solute transport is described by assuming complete mixing of the element input within one homogeneous soil compartment with a constant density and a fixed depth. Since VSD is a single layer soil model neglecting vertical heterogeneity, it predicts the concentration of the soil water leaving this layer (mostly the root zone). The annual water flux percolating from this layer is taken equal to the annual precipitation excess. The time step of the model is one year, i.e. seasonal variations are not considered. A detailed description of the VSD model can be found in Posch and Reinds (2003) and Posch et al., (2002).

The SMART model: The SMART model (**S**imulation **M**odel for **A**cidification's **R**egional **T**rends) is similar to the VSD model, but somewhat extended and is described in De Vries et al. (1989) and

Posch et al. (1993). As with the VSD model, the SMART model consists of a set of mass balance equations, describing the soil input-output relationships, and a set of equations describing the rate-limited and equilibrium soil processes. It includes most of the assumptions and simplifications given for the VSD model; and justifications for them can be found in De Vries et al. (1989). As with the VSD and SAFE model, it models the exchange of Al, H and divalent base cations, but describes them with Gaines-Thomas equations. Additionally, sulphate adsorption is modelled using a Langmuir equation (as in MAGIC) and organic acids can be described as mono-, di- or tri-protic. Furthermore, it does include a balance for carbonate and Al, thus allowing the calculation from calcareous soils to completely acidified soils that do not have an Al buffer left. In this respect, SMART is based on the concept of buffer ranges expounded by Ulrich (1981). Recently a description of the complexation of aluminium with organic acids has been included. The SMART model has been developed with regional applications in mind, and an early example of an application to Europe can be found in De Vries et al. (1994b).

The SAFE model: The SAFE (**S**oil **A**cidification in **F**orest **E**cosystems) model has been developed at the University of Lund (Warfvinge et al., 1993) and a recent description of the model can be found in Alveteg (1998) and Alveteg and Sverdrup (2002). The main differences to the SMART and MAGIC models are: (i) weathering of base cations is not a model input, but it is modelled with the PROFILE (sub-)model, using soil mineralogy as input (Warfvinge and Sverdrup, 1992); (ii) SAFE is oriented to soil profiles in which water is assumed to move vertically through several soil layers (usually 4), (iii) Cation exchange between Al, H and (divalent) base cations is modelled with Gapon exchange reactions, and the exchange between soil matrix and the soil solution is diffusion limited, (iv) SAFE assumes no retention of S in the soil (although a sulphate adsorption model depending on sulphate concentration and pH has been tested, Fumoto and Sverdrup, 2001). The SAFE model has been applied on many sites and more recently also regional applications have been carried out for Sweden (Alveteg and Sverdrup, 2002) and Switzerland (Kurz et al., 1998).

Appendix 7.3. Calculation of aluminium complexation, nitrogen transformations and sulphate adsorption in SMART

Calculation of the free aluminium concentration in SMART

In SMART the concentration of free (uncomplexed) Al can be calculated from observations of the concentrations of [*H*], [Al_{tot}] and DOC using relatively simple models for the dissociation of DOC and complexation of Al with organic anions. The dissociation of DOC is described by a triprotic model for soil organic acids, according to:

$$\begin{array}{lcl} H_3A & \Leftrightarrow & H^+ + H_2A^- \\ H_2A^- & \Leftrightarrow & H^+ + HA^{2-} \\ HA^{2-} & \Leftrightarrow & H^+ + A^{3-} \end{array} \qquad \text{(A7.3.1)}$$

or in mathematical form:

$$\begin{array}{l} [H^+]\cdot[H_2A^-] = K_1\cdot[H_3A] \\ [H^+]\cdot[HA^{2-}] = K_2\cdot[H_2A^-] \\ [H^+]\cdot[A^{3-}] = K_3\cdot[HA^{2-}] \end{array} \qquad \text{(A7.3.2)}$$

This system of equations for the concentrations of organic anions is easily solved, yielding (assuming that $m{\cdot}DOC$ is the sum of all organic species):

$$\begin{array}{l} [H_2A^-] = \dfrac{m\cdot DOC\cdot K_1\cdot[H]^2}{[H]^3 + K_1\cdot([H]^2 + K_2\cdot([H]+K_3))} \\ [HA^{2-}] = [H_2A^-]\cdot K_2/[H] \\ [A^{3-}] = [HA^{2-}]\cdot K_3/[H] \end{array} \qquad \text{(A7.3.3)}$$

The complexation of Al with organic anions is modelled according to (Santore et al., 1995):

$$\begin{array}{lcl} Al^{3+} + A^{3-} & \Leftrightarrow & AlA \\ Al^{3+} + A^{3-} + H^+ & \Leftrightarrow & AlHA^+ \end{array} \qquad \text{(A7.3.4)}$$

In their mathematical form these equations read:

$$\begin{array}{r} [AlA] = K_{31}[Al^{3+}][A^{3-}] \\ [AlHA^+] = K_{32}[Al^{3+}][A^{3-}][H^+] \end{array} \qquad \text{(A7.3.5)}$$

where K_{31} and K_{32} are the equilibrium constants. Defining pK=-$\log_{10}(K)$ (with K expressed in mol.l^{-1} or powers thereof), we use pK_{31}=-7.89, pK_{32}=-12.86 (Santore et al., 1995). Note that [Al^{3+}] refers to the concentration of *free* aluminium. From measurements the only quantity known is the concentration of *total* aluminium, [Al_{tot}]=[Al^{3+}]+[ΣAl_{org}], where [ΣAl_{org}] is the sum of the organically complexed Al-species, [ΣAl_{org}]=[AlA]+[$AlHA^+$]. Combining the various equations, the concentration of free aluminium is obtained as:

$$[Al^{3+}] = \frac{[Al_{tot}]}{1 + [A^{3-}](K_{31} + K_{32}[H^+])} \quad \text{(A7.3.6)}$$

Calculation of nitrogen transformations in SMART

In SMART the balance for the nitrogen compounds NO_3 and NH_4 is defined as:

$$NO_{3,le} = NO_{x,td} + NH_{4,ni} - NO_{3,de} - NO_{3,gu} - NO_{3,im} \quad \text{(A7.3.7)}$$

and

$$NH_{4,le} = NH_{3,td} - NH_{4,ni} - NH_{4,gu} - NH_{4,im} \quad \text{(A7.3.8)}$$

where the subscripts *le*, *td*, *ni*, *gu*, *de* and *im* stand for leaching, total deposition, nitrification, net growth uptake, denitrification and immobilisation, respectively. Adding equations A7.3.7 and A7.3.8 one obtains the mass balance for total nitrogen:

$$N_{le} = N_{td} - N_{gu} - N_{de} - N_{im,t} - N_{im,acc} \quad \text{(A7.3.9)}$$

where nitrogen immobilisation has been split into a time-dependent part $N_{im,t}$ and a constant part $N_{im,acc}$.

Denitrification and nitrification are modelled as fractions of the net nitrate and ammonium input, respectively:

$$NO_{3,de} = f_{de} \cdot (NO_{x,td} + NH_{4,ni} - NO_{3,gu} - NO_{3,im}) \quad \text{(A7.3.10)}$$

and

$$NH_{4,ni} = f_{ni} \cdot (NH_{3,td} - NH_{4,gu} - NH_{4,im}) \quad \text{(A7.3.11)}$$

where f_{de} and f_{ni} are the denitrification and nitrification fractions, respectively. For complete nitrification adding Eqs. A3.10 and A3.11 yields the equation for the denitrification of total N:

$$N_{de} = f_{de} \cdot \left(N_{td} - N_{gu} - N_{im}\right) \quad \text{(A7.3.12)}$$

Assuming no preference in the uptake and N immobilisation of NO_3^- and NH_4^+, growth uptake and immobilisation fluxes is calculated according to:

$$NO_{3,gu} = N_{gu} \cdot \frac{NO_{x,td}}{N_{td}} \quad , \quad NH_{4,gu} = N_{gu} \cdot \frac{NH_{3,td}}{N_{td}} \quad \text{(A7.3.13)}$$

and

$$NO_{3,im} = N_{im} \cdot \frac{NO_{x,td}}{N_{td}} \quad , \quad NH_{4,im} = N_{im} \cdot \frac{NH_{3,td}}{N_{td}} \quad \text{(A7.3.14)}$$

Combining Eqs. A7.3.7 to A7.3.14, one obtains for the balances:

$$NO_{3,le} = f_{td} \cdot (1 - f_{de}) \cdot (NO_{x,td} + f_{ni} \cdot NH_{3,td}) \qquad \text{(A7.3.15)}$$

and

$$NH_{4,le} = f_{td} \cdot (1 - f_{ni}) \cdot NH_{3,td} \qquad \text{(A7.3.16)}$$

where

$$f_{td} = \frac{N_{td} - N_{gu} - N_{im}}{N_{td}} \qquad \text{(A7.3.17)}$$

While f_{de} and f_{ni} are inputs to SMART, the fraction f_{im} of N immobilised is computed as a function of the C:N ratio in the topsoil according to (De Vries et al., 1994b):

$$f_{im} = \begin{cases} 0 & \text{if} \quad CN \leq CN_{min} \\ \dfrac{CN - CN_{min}}{CN_{max} - CN_{min}} & \text{if} \quad CN_{min} < CN < CN_{max} \\ 1 & \text{if} \quad CN \geq CN_{max} \end{cases} \qquad \text{(A7.3.18)}$$

where CN_{min} and CN_{max} are given limit values.

References

Aarrestad, P.A. and D. Aamlid, 1999. *Vegetation monitoring in South-Varanger, Norway-species composition of ground vegetation and its relation to environmental variables and pollution impact.* Environ. monit. assess. 58, 1-21.

Alcamo, J., R. Shaw and L. Hordijk, 1990. *The RAINS model of acidification*. Science and Strategies in Europe, Kluwer Academic Publishers, Dordrecht, 402 pp.

Alveteg, M., 1998. *Dynamics of forest soil chemistry.* PhD thesis, Reports in Ecology and Environmental Engineering 3:1998, Department of Chemical Engineering II, Lund University, Lund, Sweden, 81 pp.

Alveteg, M. and H. Sverdrup, 2002. *Manual for regional assessments using the SAFE model (draft version 8 April 2002)*. Department of Chemical Engineering II, Lund University, Lund, Sweden.

Alveteg, M., H. Sverdrup and P. Warfvinge, 1995. *Regional assessment of the temporal trends in soil acidification in southern Sweden, using the SAFE model.* Water Air Soil Poll. 85, 2509-2514.

Ashmore, M.R. and A.W. Davison, 1996. *Towards a Critical Level of Ozone for Natural Vegetation.* In: L. Kärelamppi and L. Shärby (Eds). Critical levels for ozone in Europe: testing and finalizing the concepts. UN-ECE Workshop Report. Univ. Kuopio.

Baritz, R., G.H. Adler, B. Wolff and B.-M. Wilke, 1999. *Regional distribution of carbon in German forest soils and its relation to climate change.* Zeitschrift fur Angewandte Geologie 45 (4), 218-227.

Bartnicki, J., K. Olendrzynski, E.J. Jonson, E. Berge and S. Junger, 2002. *Description of the Eularian acid deposition model*. EMEP/MSC-W.

Bascompte, J. and M.A. Rodriguez, 2001. *Habitat patchiness and plant species richness.* Ecology Letters 4, 417-420.

Batjes, N.H., 1996. *Development of a world data set of soil water retention properties using pedotransfer rules.* Geoderma 71, 31-52.

Becker, R., J. Block, C.-G. Schimming, T. Spranger and N. Wellbrock, 2000. *Critical loads für Waldökosysteme - Methoden unf ergebnisse für Standorte des Level II-Programms*. Bundesministerium für Ernährung, Landwirtschaft und Forsten, Bonn, Germany (in German).

Berg, B. and E. Matzner, 1997. *Effect of N deposition on decomposition of plant litter and soil organic matter in forest systems.* Environ. rev. 5, 1-25.

Bonneau, M., 1995. *Fertilisation des forêts dans les pays tempérés. Théorie, bases du diagnostique, conseils pratiques, réalisations expérimentales.* ENGREF-Nancy, 367 pp.

Borcard, D., P. Legendre and P. Drapeau, 1992. *Partialling out the spatial component of ecological variation.* Ecology 73 (3), 1045-1055.

Bousquet, P., P. Ciais, P. Peylin, M. Ramonet and P. Monfray, 1999. *Inverse modelling of annual atmospheric CO2 sources and sinks 1. Methods and control inversion.* Journal of Geophysical Research 104, 26161-26178.

Bussotti, F., M. Schaub, A. Cozzi, N. Kräuchi, M. Ferretti, N. Novak and J. Skelly, 2003. *Assessment of ozone visible symptoms in the field: perspectives of quality control.* Environ. pollut. ((in press)).

CEN, 2001. *Ambient Air Quality - Diffusive samplers for the determination of concentrations of gases and vapours - Part 3: Guide to selection, use and maintenance*. TC. No: 264; European Rwef. Nr. 13528-3.

Ciais, P., P.P. Tans, J.W.C. White, M. Trolier, R.J. Francey, J.A. Berry, D.R. Randall, P.J. Sellers, J.G. Collatz and D.S. Schimel, 1995. *Partitioning of ocean and land uptake of CO2 as inferred by 13C measurements from the NOAA Climate Monitoring and Diagnostics Laboratory global air sampling network.* Journal of Geophysical Research 100, 5051-5070.

Cluzeau, C., E. Ulrich, M. Lanier and F. Garnier, 1998. *RENECOFOR - Interprétation des mmesures dendrométriques de 1991 à 1995 des 102 peuplements du réseau*. Office National des Forêts Département des Recherches Techniques ISBN 2-84207-151-4, 309 pp.

Cluzeau, C., J. Drapier, R. Virion and J.L. Dupouey, 2001. *Changements à long terme de la végétation forestière : apport des données de l'IFN.* Projet GIP-Ecofor, Programme "Forêt et modifications de l'environnement", 36 + annexes pp.

Cofala, J. and S. Syri, 1998a. *Sulfur Emissions, Abatement Technologies and Related Costs for Europe in the RAINS Model Database.* IIASA Interim Report IR-98-35 (update 19.10.1998).

Cofala, J. and S. Syri, 1998b. *Nitrogen Oxides Emissions, Abatement Technologies and Related Costs for Europe in the RAINS Model Database.* IIASA Interim Report IR-98-88.

Collins, W.J., D.S. Stevenson, C.E. Johnson and R.G. Derwent, 2000. *The European regional ozone distribution and its links with the global scale for the years 1992 and 2015.* Atmospheric Environment 34, 255-267.

Condit, R., S.P. Hubell, J.V. LaFrankie, R. Sukumar, N. Manokaran, R.B. Foster and P.S. Ashton, 1996. *Species-area and species-individual relationships for tropical trees: a comparison of three 50-ha plots.* J. Ecol. 84, 549-562.

Cosby, B.J., G.M. Hornberger, R.F. Wright and J.N. Galloway, 1986. *Modeling the effects of acid deposition: Control of long-term sulfate dynamics by soil sulfate adsorption.* Water resour. res. 22 (8), 1283-1291.

Cozzi, A., M. Ferretti and J. Innes, 2000. *Sintomi fogliari attribuibili ad ozono sulla vegetazione spontanea in Valtellina.* Monti e Boschi ¾, 42-49.

Cramer, W.P. and R. Leemans, 2001. *Global 30-year mean monthly climatology, 1930-1960, Version. 2.1 (Cramer and Leemans).* In Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, Tennessee, U.S.A. [http://www.daac.ornl.gov].

Crawley, M.J. and J.E. Harral, 2001. *Scale dependence in plant biodiversity.* Science 291, 864-868.

Da Lage, A. and G. Métailié, 2000. *Dictionnaire de biogéographie végétale*. CNRS editions, Paris, 579 pp.

Dalstein, L., P. Vollenweider, N. Vas and M. Güenthardt-Goerg, 2002. *Lózone et les conifères du Sud-Est de la France.* Foret Méditerranéenne XXIII (2), 105-116.

De Leeuw, F., R. Sluyter, P. van Breugel and F. Bogman, 2001. *Air pollution by ozone in Europe in 1999 and the summer of 2000*. European Environment Agency, Topic Report 1/2001, 55 pp.

De Vries, W., 1991. *Methodologies for the assessment and mapping of critical loads and the impact of abatement strategies on forest soils.* Staring Centre Report 46. Wageningen, 109 pp.

De Vries, W., 2000. *Intensive Monitoring of Forest Ecosystems in Europe. Evaluation of the programme in view of its objectives studies to reach the objectives and priorities for the scientific evaluation of the data.* Forest Intensive Monitoring Coördinating Institute, A strategy document, Heerenveen, the Netherlands, 45 pp.

De Vries, W. and A. Breeuwsma, 1986. *Relative importance of natural and anthropogenic proton sources in soils in the Netherlands.* Water Air Soil Poll. 28 (1), 173-184.

De Vries, W. and E.E.J.M. Leeters, 2001. *Chemical composition of the humus layer, mineral soil and soil solution of 150 forest stands in the Netherlands in 1990.* Alterra rapport 424.1. Alterra, Green World Research, Wageningen (Netherlands), 113 pp.

De Vries, W. and M. Posch, 2003a. *Critical Levels and Critical Loads as A Tool For Air Quality Management.* In: C.N. Hewitt and A. Jackson (Eds). Handbook of Atmospheric Science. Principles and Applications. Blackwell Publishing, in press, London, UK, pp. 656.

De Vries, W. and M. Posch, 2003b. *Derivation of cation exchange constants for sand, loess, clay and peat soils on the basis of field measurements in the Netherlands.* Alterra Report (in press).

De Vries, W., M. Posch and J. Kämäri, 1989. *Simulation of the long-term soil response to acid deposition in various buffer ranges.* Water Air Soil Poll. 48 (3), 349-390.

De Vries, W., J. Kros and C. Van der Salm, 1994a. *Long-term impacts of various emission deposition scenarios on Dutch forest soils.* Water Air Soil Poll. 75 (1-2), 1-35.

De Vries, W., G.J. Reinds, M. Posch and J. Kämäri, 1994b. *Simulation of soil response to acidic deposition scenarios in Europe.* Water Air Soil Poll. 78, 215-246.

De Vries, W., G.J. Reinds, M.A. van Kerkvoorde, C.M.A. Hendriks, E.E.J.M. Leeters, C.P. Gross, J.C.H. Voogd and E.M. Vel, 2000. *Intensive Monitoring of Forest Ecosystems in Europe. Technical Report 2000*. UN/ECE, EC, Forest Intensive Monitoring Coordinating Institute, 193 pp.

De Vries, W., G.J. Reinds, H. van Dobben, D. de Zwart, M. Posch, J.C.H. Voogd, J. Auee and E.M. Vel, 2002. *Intensive Monitoring of Forest Ecosystems in Europe. Technical Report 2002*. UN/ECE and EC, Forest Intensive Monitoring Coordinating Institute, Geneva and Brussels, 175 pp.

De Vries, W., G.J. Reinds, C. Van der Salm, G.P.J. Draaijers, A. Bleeker, J.W. Erisman, J. Auee, P. Gundersen, H.L. Kristensen, H. Van Dobben, D. De Zwart, J. Derome, J.C.H. Voogd and E.M. Vel, 2001. *Intensive Monitoring of Forest Ecosystems in Europe. Technical Report 2001*. UN/ECE, EC, Forest Intensive Monitoring Coordinating Institute, Geneva and Brussels, 177 pp.

Diekmann, M., 1995. *Use and improvement of Ellenberg's indicator values in deciduous forests of the Boreo-nemoral zone in Sweden.* Ecography 18, 178-189.

Diekmann, M. and U. Falkengren-Grerup, 2002. *Prediction of species response to atmospheric nitrogen deposition by means of ecological measures and life history traits.* J. Ecol. 90, 108-120.

Dise, N.B., E. Matzner and M. Forsius, 1998a. *Evaluation of organic horizon C:N ratio as an indicator of nitrate leaching in conifer forests across Europe.* Environ. pollut. 102 (Supp 1), 453-456.

Dise, N.B., E. Matzner and P. Gundersen, 1998b. *Synthesis of nitrogen pools and fluxes from European forest ecosystems.* Water Air Soil Poll. 105, 143-154.

Dobbertin, M., 2000. *Forest Growth.* In: T. Haussmann, M. Lorenz and R. Fischer (Eds). International Review of ICP Forests. UN/ECE, Federal Research Centre for Forestry and Forest Products, pp. 84-102.

Dobremez, J.F., S. Camaret, L. Bourjot, E. Ulrich, A. Brêthes, P. Coquillard, G. Dumé, J.L. Dupouey, F. Forgeard, C. Gauberville, J. Gueugnot, J.F. Picard, J.M. Savoie, A. Schmitt, J. Timbal, J. Touffet and M. Trémolières, 1997. *RENECOFOR - Inventaire et interpretation de la composition floristique de 101 peuplements du réseau (Campagne 1994/95)*. Département des Recherches Techniques, Office National Forêts, Fontainebleau, Université de Savoie, 513 pp.

Donita, N., S. Purcelean, I. Ceianu and A. Beldie, 1977. *Ecologie Forestiera (cu elemente de ecologie generala)*. CERES, Bucuresti, Roumania.

Dupouey, J.L., L. Bourjot, S. Camaret, J.F. Dobremez, F. Forgeard, J.F. Picard and J. Touffet, 1998. *Study of sampling frequency and comparison of cover estimation methods for ground vegetation assessment*. Oxalis Projet - EU Project No. 96.60.FR.005.0, 65 pp.

Dzwonko, Z., 2001. *Assessment of light and soil conditions in ancient and recent woodlands by Ellenberg indicator values.* J. appl. ecol. 38, 942-951.

EC, 1999. *Ozone Position Paper*. Ad Hoc Working Group on Ozone Directive and Reduction Strategy development. Off. for Official Publications of the EC, Luxemburg.

Ellenberg, H., H.E. Weber, R. Düll, V. Wirth, W. Werner and D. Paulißen, 1992. *Zeigerwerte von Pflanzen in Mitteleuropa.* Scripta geobotanica 18, 1-248.

Emberson, L.D., M.R. Ashmore and H.L. Cambridge, 1998. *Development of Methodologies for Mapping Level II Critical Levels of Ozone*. DETR Report n. EPG. 1/3/82, Imperial College Centre for Environmental Technology, London, 113 pp.

Emberson, L.D., M.R. Ashmore, H.M. Cambridge, S. D. and T.J. P., 2000. *Modelling stomatal ozone flux across Europe.* Environ. pollut. 109 (3), 403-414.

EMEP, 2001. *Transboundary acidification and eutrophication and ground level ozone in Europe.* In EMEP Summary Report 2001 EMEP Report 1/01, July 2001. Joint CIAM & CCC & MSC-W report Norwegian Meteorological Institute.

Emmet, B.A., D. Boxman, M. Bredemeier, P. Gundersen, O.J. Kjønaas, F. Moldan, P. Schleppi, A. Tietema and R.F. Wright, 1998. *Predicting the effects of atmospheric nitrogen deposition on coniferous stand: evidence from the NITREX ecosystem-scale experiments.* Ecosystems 1, 352-360.

Ertsen, A.C.D., J.R.M. Alkemade and M.J. Wassen, 1998. *Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and slinity in the Netherlands.* Plant Ecology 135, 113-124.

Fan, S.M., M. Gloor, J. Mahlman, S. Pacale, J. L.Sarmiento, T. Takahashi and P.P. Tans, 1998. *A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models.* Science 282, 442-446.

FAO, 2000. *Global Forest Ressources Assessment 2000*. Main report. FAO forestry paper 140, 479 pp.

Ferretti, M., S. Allavena, N. Kräuchi, E. Pompei, B. Petriccione, G. Sanchéz-Peña, M.J. Sanz and E. Ulrich, 2001. *Ozone at the intensive monitoring plots in South-Western European forests: levels, risks, actual and potential effects*. Project proposal (unpublished).

Fowler, D., N. Cape, Coyle, C. Flechard, J. Kuylenstierna, K. Hicks, D. Derwent, C. Johnson and D. Stevenson, 1999. *The global exposure of forest ecosystems to air pollutants.* Water Air and Soil Pollution 116, 5-32.

Frey, W., J.P. Frahm, E. Fischer and W. Lobin, 1995. *Die Moos- und Farnpflanzen Europas.* In: Kleine Kryptogamenflora. Vol. Vol. IV Gustav Fischer Verlag, Stuttgart-Jena-New York.

Friedlingstein, P., I. Fung, E. Holland, J. John, G. Brasseur, D. Erickson and D. Schimel, 1995. *On the contribution of CO2 fertilization to the missing biospheric sink.* Global biogeochemical cycles 9 (4), 541-556.

Führer, J., L. Skärby and M.R. Ashmore, 1997. *Critical levels for ozone effects on vegetation in Europe.* Environ. pollut. 97, 91-106.

Fumoto, T. and H. Sverdrup, 2001. *Implementation of sulfate adsorption in the SAFE model.* J. Environ. Qual. 30, 45-57.

Gégout, J.C., H. Brisse and G. Grandjouan, 2001. *Comportement écologique des espèces végétales vis-à-vis du climat et du sol en France. Application à l'évaluation des charges critiques d'acidité et d'azote.* Rapport intermédiaire de la convention de recherche ADEME/ENGREF n° 99 62 003., ENGREF, Nancy, 104 pp.

Gough, L. and J.B. Grace, 1998. *Herbivore effects on plant species density at varying productivity levels.* Ecology 79 (5), 1586-1594.

Gough, L. and J.B. Grace, 1999. *Effects of environmental change on plant species density : comparing predictions with experiments.* Ecology 80 (3), 882-890.

Grime, J.P., 1973a. *Control of species density in herbaceous vegetation.* Journal of Environmental Management 1, 151-167.

Grime, J.P., 1973b. *Competitive exclusion in herbaceous vegetation.* Nature 242, 344-347.

Grünhage, L., G.H.M. Krause, B. Köllner, J. Bender, H.-J. Weigel, H.-J. Jäger and R. Guderian, 2001. *A new flux-oriented concept to derive critical levels for ozone to protect vegetation.* Environ. pollut. 111, 355-362.

Guenthardt-Goerg, M., 2001. *Ozonsymptome an Waldbaumarten.* Wald und Holz 10/01, 30-33.

Gundersen, P. and L. Rasmussen, 1995. *Nitrogen mobility in a nitrogen limited forest at Klosterhede, Denmark, examined by NH4NO3 addition.* For. ecol. manage. 71, 75-88.

Gundersen, P., I. Callesen and W. de Vries, 1998a. *Nitrate leaching in forest ecosystems is related to forest floor C/N ratios.* Environ. pollut. 102 (1), 403-407.

Gundersen, P., K. Raulund-Rasmussen and I.K. Schmidt, 2003. *Leaching of nitrogen from temperate forests - effects of air pollution and forest management.* In: R. Vertessy and H. Elsenbeer (Eds). Forests and Water. IUFRO (In press).

Gundersen, P., B.A. Emmet, O.J. Kjønaas, C. Koopmans and A. Tietema, 1998b. *Impact of nitrogen deposition on nitrogen cycling: a synthesis of NITREX- data.* For. ecol. manage. 101, 37-55.

Harrison, A.F., D.D. Harkness, A.P. Rowland, J.S. Garnett and P.J. Bacon, 2000. *Annual carbon and nitrogen fluxes in soils along the European forest transect determined using 14C-bomb.* In: E.D. Schulze (Ed). Carbon and nitrogen in forest soils. Ecological Studies 142. Springer Verlag, Heidelberg.

He, F. and P. Legendre, 2002. *Species diversity patterns derived from species-area models.* Ecology 83, 1185-1198.

Helling, C.S., G. Chesters and R.B. Corey, 1964. *Contribution of organic matter and clay to soil cation exchange capacity as affected by the pH of the saturating solution.* Soil Sci. Soc. Am. J. 28, 517-520.

Hettelingh, J.-P. and M. Posch, 1994. *Critical loads and a dynamic assessment of ecosystem recovery.* In: J. Grasman and G. Van Straten (Eds). Predictability and Nonlinear Modelling in Natural Sciences and Economics. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 439-446.

Hill, M.O., J.O. Mountford, D.B. Roy and R.G.H. Bunce, 1999. *Ellenberg's indicator values for British plants. ECOFACT, volume 2. Technical Annex.* Institute of Terrestrial Ecology, Huntingdon, UK.

Hoekstra, C. and J.N.B. Poelman, 1982. *Dichtheid van gronden gemeten aan de meest voorkomende bodemeenheden in Nederland.* Rapport nr. 1582. Stichting voor Bodemkartering, Wageningen, 47 pp.

Holland, E.A., B.H. Braswell, J.F. Lamarque, A.R. Townsend, J. Sulzman, J.F. Muller, F. Dentener, G. Brasseur, H. Levy, II, J.E. Penner and G.J. Roelofs, 1997. *Variations in the predicted spatial distribution of atmospheric nitrogen deposition and their impact on carbon uptake by terrestrial ecosystems.* Journal of Geophysical Research 102, 15849-15866.

Hosmer, D.W. and S. Lemeshow, 2000. *Applied logistic regression*. Second, John Wiley and Sons, inc., New-York, 375 pp.

Houghton, R.A., E.A. Davidson and G.M. Woodwell, 1998. *Missing sinks, feedbacks and understanding the role of terrestrial ecosystems in the global carbon balance.* Cycles 12, 25-34.

ICP Forest, 2000. *Strategy of ICP forest for the period of 2001-2006*. Convention of Long-Range Transboundary Air Pollution International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests). UN/ECE, Intenal report, Geneva, 19 pp.

Ilic, J., D. Boland, M. Mc Donald, G. Downess and P. Blakemore, 2000. *National carbon accounting system. Wood density Phase 1- State of knowledge.* Technical Report 18. Australian Greenhouse Office.

Innes, J., J. Skelly, W. Landolt, C. Hug, K.R. Snyder and K.R. Savage, 1996. *Development of visibleinjury on the leaves of Prunus serotina in Ticino, southern Switzerland, as a result of ozone exposure: preliminary results.* In: M. Knoflacher, J. Sneider and G. Soja (Eds). Exceedance of Critical Loads and levels. Convention on Long Range Transboundary Air Pollution, Vienna, Austria, pp. 146-154.

Innes, J.L., J.M. Skelly and M. Schaub, 2001. *Ozone and broadleaved species*. Eidgenoessische Forschungsanstalt WSL, Birmensdorf, Verlag Paul Haupt, Bern, Stuttgart, Wien, 136 pp.

Janssen, P.H.M. and P.S.C. Heuberger, 1995. *Calibration of process-oriented models.* Ecological Modelling 83, 55-66.

Jenkins, J.C., D.C. Chojnacky, L.S. Heath and R.A. Birdsey, 2003. *National-Scale Biomass Estimators for United States Tree Species.* For. sci. 49 (1), 12-35.

Jenkinson, D.S., K. Goulding and D.S. Polson, 1999. *Reply to Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests.* Nature 400, 629.

Jongman, R.H.G., C.J.F. ter Braak and O.F.R. van Tongeren, 1995. *Data analysis in community and landscape ecology*. Cambridge university press, Cambridge, 299 pp.

Kärenlampi, L. and L. Skärby, 1996. *Critical levels for ozone in Europe: testing and finalizing the concepts*. Univ. of Kuopio, Dept. of Ecology and Environmental Science, Kuopio, Finland, 363 pp.

Kauppi, P., J. Kämäri, M. Posch, L. Kauppi and E. Matzner, 1986. *Acidification of forest soils: Model developement and application for analyzing impacts of acidic deposition in Europe.* Ecological Modelling 33, 231-253.

Kauppi, P.E., K. Mielikäinen and K. Kuusela, 1992. *Biomass and carbon budget of European forests, 1971 to 1990.* Science 256, 70-74.

Kennel, E., 1973. *Bayerische Waldinventur 1970/71*. Forschungsberichte der Forstlichen Forschungsanstalt, München, 11 pp.

Kimmins, J.P., D. Binkley, L. Chatarpaul and J. de Catanzaro, 1985. *Biogeochemistry of temperate forest ecosystems Literature on inventories and dynamics of biomass and nutrients.* Information Report PI-X-47E/F. Petawawa National Forestry Institute, Canada, 227 pp.

Kirby, K.J., T. Bines, A. Burn, J. MAckintosh, P. Pitkin and I. Smith, 1986. *Seasonal and observer differences in vascular plant recordes from British woodlands.* J. Ecol. 74, 123-131.

Klap, J.M., W. de Vries, J.W. Erisman and E.P. van Leeuwen, 1997. *Relationships between forest condition and natural and anthropogenic stress factors on the European scale; pilot study.* Report 150. DLO Winand Staring Centre for Integrated Land, Soil and Water Research, Wageningen, the Netherlands.

Krupa, S., M. Nosal, J.A. Ferdinand, R.E. Stevenson and J.M. Skelly, 2003. *A multi-variate statistical model integrating passive sampler and meteorology data to predict the frequency distributions of hourly ambient ozone (O3) concentrations.* Environ. pollut. in press.

Kurz, D., M. Alveteg and H. Sverdrup, 1998. *Acidification of Swiss forest soils - Development of a regional dynamic assessment*. Environmental Documentation No.89, SAEFL, Berne, Switzerland, 115 pp.

Landolt, E., 1977. *Ökologische Zeigerwerte zur Schweizer Flora*. Zurich, ETH, 208 pp.

Leemans, R. and W. Cramer, 1991. *The IIASA database for mean monthly values of temperature, precipitation and sunshine on a global terrestrial grid.*, IIASA Reseach Report RR-91-18, Laxenburg, Austria, 61 pp. pp.

Leeters, E.E.J.M. and W. de Vries, 2001. *Chemical composition of the humus layer, mineral soil and soil solution of 200 forest stands in the Netherlands in 1995.* Alterra rapport 424.2. Alterra, Green World Research, Wageningen (Netherlands), 108 pp.

Legendre, P. and L. Legendre, 1998. *Numerical ecology*. 2nd English, Elsevier Science BV, Amsterdam, . xv + 853 pp.

Leps, J. and V. Hadincova, 1992. *How reliable are our vegetation analyses?* J. Veg. Sci. 3 (1), 119-124.

Lilliesköld, M. and J. Nilsson, 1997. *Kol i marken, Konsekvenser av markanvändning i skogs- och jordbruk.* Naturvårdsverket Rapport 4782. Naturvårdsverket, Stockholm.

Liski, J. and C.J. Westman, 1997. *Carbon storage in forest soil of Finland 2. Size and regional patterns.* Biogeochem. 36 (3), 261-274.

Liski, J., D. Perruchoud and T. Karjalainen, 2002. *Increasing carbon stocks in the forest soils of western Europe.* For. ecol. manage. 169 (1-2), 159-175.

Loibl, W., W. Winiwarter, A. Kopcsa, J. Züger and R. Baumann, 1994. *Estimating the spatial distribution of ozone concentrations in complex terrain using a function of elevation and day time and kriging techniques.* Atmospheric Environment 28, 2557-66.

Madotz, N., R.Y. Cavero and A. Ederra, 2002. *Estudio de la vegetacion del hayedo donde se situa la parcela 15Fs de la Red Europea de Seguimiento Intensivo y Continuo de los Ecosistemas Forestales (Auritz-Navarra).* Publ. Bio. Univ. Navarra, Ser. Bot. 14, 155-215.

Marenco, A., H. Gouget, P. Nédeélec and F. Karcher, 1994. *Evidence of long-term increase in tropospheric ozone from Pic du Midi data series. Consequences : Positive radiative forcing.* J. Geophys. Res. 99(D8) (16), 16,616-16,632.

Martin, P.H., R. Valentini, M. Jacques, K. Fabbri, D. Galati, R. Quaratino, J.B. Moncrieff, P. Jarvis, N.O. Jensen, A. Lindroth, A. Grelle, M. Aubinet, R. Ceulemans, A.S. Kowalski, T. Vesala, P. Keronen, G. Matteucci, A. Grainer, P. Berbingier, D. Loustau, E.D. Schulze, J. Tenhunen, C. Rebman, A.J. Dolman, J.E. Elbers, C. Bernhofer, T. Grunwald, H. Thorgeirsson, P. Kennedy and S. Folving, 1998. *New estimate of the carbon sink strength of EU forests integrating flux measurements, field surveys, and space observations : 0.17-0.35 Gt (C).* Ambio 27 (7), 582-584.

Matzner, E. and C. Grosholz, 1997. *Beziehung zwischen NO3-Austrägen, C/N-Verhältnissen der Auflage und N-Einträgen in Fichtenwald (Picea abies Karst.) - Ökosystemen Mitteleuropas.* Forstw. Cbl. 116, 39-44.

Melillo, J.M., A.D. McGuire, D.W. Kicklighter, B. Moore, III, C.J. Vorosmarty and A.L. Schloss, 1993. *Global climate change and terrestrial net primary production.* Nature 363, 234-240.

Michalzik, B. and E. Matzner, 1999. *Dynamics of dissolved organic nitrogen and carbon in a Central European Norway spruce ecosystem.* Eur. J. Soil Sci. 50, 579 - 590.

Mittelbach, G.G., C.F. Steiner, S.M. Scheiner, K.L. Gross, H.L. Reynolds, R.B. Waide, M.R. Willig, S.I. Dodson and L. Gough, 2001. *What is the observed relationship between species richness and productivity?* Ecology 82 (9), 2381-2396.

Moldan, F., R.F. Wright, K.W. van der Hoek, J.W. Erisman, S. Smeulders and J.R. Wisniewski, 1998. *Episodic behaviour of nitrate in runoff during six years of nitrogen addition to the NITREX catchment at Gardsjon, Sweden.* Environ. pollut. 102 (Supp 1), 439-444.

Myneni, R.B., C.D. Keeling, C.J. Tucker, G. Asrar and R.R. Nemani, 1997. *Increased plant growth in the northern high latitudes from 1981 to 1991.* Nature 386, 698-702.

Nabuurs, G.J. and M.J. Schelhaas, 2002. *Carbon profiles of typical forest types across Europe assessed with CO2FIX.* Ecological Indicators 1 (3), 213-223.

Nabuurs, G.J. and M.J. Schelhaas, 2003. *Spatial distribution of whole tree carbon stocks and fluxes of forests in Europe; where are the options for bio-energy?* Biomass & Bioenergy 24, 311-320.

Nabuurs, G.J., R. Päivinen, R. Sikkema and G.M.J. Mohren, 1997. *The role of European forests in the global carbon cycle-A review.* Biomass and bioenergy 13 (6), 345-358.

Nadelhoffer, K.J., M.R. Downs, B. Fry, J.D. Aber, A.H. Magill and J.M. Melillo, 1995. *The fate of 15N labelled nitrate additions to a northern hardwood forest in eastern Maine, USA.* Oecologia 103, 292-301.

Nadelhoffer, K.J., B.A. Emmett, P. Gundersen, O.J. Kjønaas, C.J. Koopmans, P. Schleppi, A. Tietema and R.F. Wright, 1999. *Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests.* Nature 398, 145 -148.

Ney-Nifle, M. and M. Mangel, 1999. *Species-area curves based on geographic range and occupancy.* J. Theor. Biol. 196, 327-342.

Olsson, M. and P.A. Melkerud, 1991. *Determination of weathering rates based on geochemical properties of the soil.* In: E. Pulkinen (Ed). Environmental geochemistry in Northern Europe. Proceedings of a Symposium held in Rovaniemi, Finland, 17-19 October, 1989. Geological Survey of Finland, Special paper 9. pp. 69-78.

Palmer, M.W., 1994a. *Scale dependence and the species-area relationship.* The American Naturalist 144 (5), 717-740.

Palmer, M.W., 1994b. *Variation in species richness: towards a unification of hypotheses.* Folia Geobotanica Phytotaxonomica 29, 511-530.

Palmer, M.W., P.G. Earls, B.W. Hoagland, P.S. White and T. Wohlgemuth, 2002. *Quantitative tools for perfecting species lists.* Envorinmetrics 13, 121-137.

Papale, D. and R. Valentini, 2003. *A new assessment of European forests carbon exchanges by eddy fluxes and artificial neural network spatialization.* Glob. Change Biol. 9 (4), 525-535.

Perruchoud, D., L. Walthert, S. Zimmermann and P. Lüscher, 1999. *Contemporary carbon stocks of mineral forest soils in the Swiss Alps.* Biogeochem. 50, 111–136.

Pollanschütz, J., 1965. *Eine neue Methode der Formzahl- und Massen-bestimmung stehender Bäume.* Mitteilungen der Forstlichen Bundesversuchsanstalt Mariabrunn 68, 186 pp.

Pollanschütz, J., 1974. *Formzahlfunktionen der Hauptbaumarten Österreichs.* Allg. Forstzeitung 85, 341-343.

Posch, M. and G.J. Reinds, 2003. *VSD - User Manual of the Very Simple Dynamic soil acidification model.* Coordination Center for Effects, National Institute for Public Health and the Environment (RIVM), Bilthoven, The Netherlands (in preparation).

Posch, M., G.J. Reinds and W. de Vries, 1993. *SMART: A Simulation Model for Acidification's Regional Trends. Model Description and User Manual*. Mimeograph Series of the National Board of Waters and the Environment 477, Helsinki, Finland, 43 pp.

Posch, M., P.A.M. de Smedt, J.P. Hettelingh and R.J. Downing, 1995. *Calculation and mapping of critical thresholds in Europe: CCE status report 1995.* Report 259101004. Coordination Center for Effects, National Institute of Public Health and the Environment, Bilthoven, the Netherlands, 198 pp.

Posch, M., J.P. Hettelingh, W. de Vries, H. Sverdrup and R.F. Wright, 2002. *Manual for dynamic modelling of soil response to atmospheric deposition.* RIVM-CCE report (in press).

Prodan, M., 1965. *Holzmeßlehre*. J. D. Sauerländer, Frankfurt/M., 644 pp.

Pulkstenis, E. and T.J. Robinson, 2002. *Two goodness-of-fit tests for logistic regression models with continuous covariates.* Statistics in Medicine 21, 79-93.

Rehfuess, K.-E., G.I. Ågren, F. Andersson, M.G.R. Cannell, A. Friend, I. Hunter, H.-P. Kahle, J. Prietzel and H. Spiecker, 1999. *Relationships between recent changes of growth and nutrition of Norway spruce, Scots pine and European beech forests in Europe-RECOGNITION*. Working Paper 19, European Forest Institute, 94 pp.

Renaud, J.-P. and J.-L. Dupouey, 2002. *Data analysis of the first European vegetation survey in Level II plots ground vegetation species composition*. Report, Program 2001.60.FR, Regulation 3528/86, Protection of forests against atmospheric pollution. UN-ICP-Forest / EU, 34 pp.

Salemaa, M., S. Monni, F. Royo Peris and C. Uhlig, 1999. *Sampling strategy for the assessment of temporal changes in ground vegetation in boreal forests.* Forest condition monitoring in Finland. National report 1998, Research papers 743, 117 - 127.

Sandroni, S., P. Bacci, G. Boffa, U. Pellegrini and A. Ventura, 1993. *Tropospheric ozone in the Pre-Alpine and Alpine regions.* In: Physico-chemical behaviour of atmospheric pollutants. Proceedings of the sixth European symposium, Varese 18-22 October 1993, Report EUR 15609/1 EN. pp. 513.

Santore, R.C., C.T. Driscoll and M. Aloi, 1995. *A model of soil organic matter and its function in temperate forest soil development.* In: W.W. McFee and J.M. Kelly (Eds). Carbon forms and functions in forest soils. Soil Sci. Soc. of America, Madison, Wisconsin, pp. 275-298.

Sanz, M.J. and G. Krause, 2001. *State of the art of passive sampling in the Pan European Program for Forest.* In: Proceedings of the International Conference Measuring Air Pollutants by Difusive Sampling. (2001), Montpellier, France,

Sanz, M.J., V. Calatayud and E. Calvo, 2000. *Sapatial pattern of ozone injury in Aleppo pine related to air polution dynamics in a coastal-mountain region of eastern Spain.* Environ. pollut. 108, 239-247.

Sanz, M.J., A. Carratalá, E. Mantilla, J.J. Dieguez and M. Millán, 1999. *Daily ozone patterns and AOT40 index on the east coast of the Iberian Peninsula.* Phys. Chem. Earth 24(5), 491- 494.

SAS, 1990. *SAS/Stat user's guide version 6, fourth edition, vol. 1 and 2*. SAS Institute inc., Cary, N.C., USA.

Schaffers, A.P. and K.V. Sykora, 2000. *Reliability of Ellenberg indicator values for moisture, nitrogen and soil reaction : comparison with field measurements.* J. Veg. Sci. 11, 225-244.

Schieler, K., 1988. *Methodische Fragen in Zusammenhang mit der Österreichischen Forstinventur*. Dipl. Arb. Univ. f. Bodenkultur, Wien, 99 pp.

Schindler, D.W., 1999. *The mysterious missing sink.* Nature 398, 105-106.

Schulze, E.D., L. Högberg, H. van Oene, T. Persson, A.F. Harrison, D. Read, A. Kjöller and G. Matteucci, 2000. *Interactions between the carbon and nitrogen cycle and the role of biodiversity: A synopsis of a study along a North -south transect through Europe.* In: E.D. Schulze (Ed). Carbon and nitrogen in forest soils. Ecological Studies 142. Springer Verlag, Heidelberg.

Shmida, A., 1984. *Whittaker's plant diversity sampling method.* Israel Journal of Botany 33, 41-46.

Sievering, H., 1999. *Reply to Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests.* Nature 400, 629-630.

Simpson, D., L. Emberson, J.P. Tuovinen and M. Ashmore, 2000. *Ozone deposition and stomatal flux modelling.* In: EMEP report 2/2000, Transboundary Photo-oxidants in Europe. EMEP summary report 2000. pp. 5-9.

Skelly, J., J. Innes, K.R. Snayder, J.E. Savage, C. Hug, W. Landolt and M.J. Sanz, 1998. *Investigations of ozone injury in forest of southern Switzerland : field surveys and open-top chamber experiemnts.* Chemosphere 36 (4-5), 995-1000.

Skelly, J.M., J.L. Innes, J.E. Savage, K.R. Snyder, D. van der Heyden, J. Zhang and M.J. Sanz, 2000. *Observation and confirmation of foliar ozone symptoms of native plant species of Switzerland and Southern Spain.* Water Air Soil Poll. 116, 227-234.

Smolander, A., V. Kitunen and E. Mälkönen, 2001. *Dissolved organic nitrogen and carbon in a Norway spruce stand and an adjacent clear-cut.* Biol. fertil. soils 33, 190-196.

Spiecker, H., K. Mielikäinen, M. Kölh and J.P. Skovsgaard (Eds.), 1996. *Growth trends in European forests.* Springer-Verlag, Berlin, Heidelberg, 372 pp.

Starr, M., 1999. *WATBAL: a model for estimating monthly water balance components, including soil water fluxes.* In: S. Kleemola and M. Forsius (Eds). 8th Annual report 1999. UN ECE ICP Intergrated Monitoring. The Finnish Environment 325. Finnish Environment Institute, Helsinki, Finland, pp. 31-35.

Starr, M., A.J. Lindroos, T. Tarvainen and H. Tanskanen, 1998. *Weathering rates in the Hieta jarvi integrated monitoring catchment.* Boreal Environment Research 3, 275-285.

Steffen, W., I. Noble, J. Canadel, M. Apps, E.-D. Schultze, P. Jarbis, D. Baldocchi, P. Ciais, W. Cramer, J. Ehleringer, G. Farquhar, C.B. Field, R. Gifford and M. Heimann, 1998. *The terrestrial carbon cycle: implications for the Kyoto protocol.* Science 280, 1393-1394.

Sverdrup, H. and P. Warfvinge, 1993. *The effect of soil acidification on the growth of trees, grass and herbs as expressed by the (Ca+Mg+K)/Al ratio. Reports in ecology and environmental engeneering.* Reports in Ecology and Environmental Engineering 1993: 2. Lund University, Department of Chemical Engineering II, 108 pp.

Sverdrup, H. and W. de Vries, 1994. *Calculating critical loads for acidity with the simple mass balance method.* Water Air Soil Poll. 72 (1/4), 143-162.

Sverdrup, H., W. de Vries and A. Henriksen, 1990. *Mapping critical loads. A guidance manual to criteria calculation methods data collection and mapping.* Miljø rapport 1990: 14. Nordic Council of Ministers Copenhagen 1990, 124 pp.

Sverdrup, H.U., 1990. *The kinetics of base cation release due to chemical weathering*. Lund University Press, Sweden, 246 pp.

Ter Braak, C.J.F. and P. Smilauer, 1998. *CANOCO reference manual and user's guide to CANOCO for Windows : Software for Canonical Community Ordination (version 4)*. Microcomputer Power, Ithaca, NY, USA, 352 pp.

Thenhunen, J.D., R. Valentini, B. Kostner, R. Zimmermann and A. Granier, 1998. *Variation in forest gas exchange at landscape to contintental scales.* Annales des Sciences Forestrieres 55 (1-2), 1-11.

Thimonier, A., J.L. Dupouey and J. Timbal, 1992. *Floristic changes in the herb-layer vegetation of a deciduous forest in the Lorraine plain under the influence of atmospheric deposition.* For. ecol. manage. 55, 149-167.

Thimonier, A., J.L. Dupouey, F. Bost and M. Becker, 1994. *Simultaneous euthrophication and acidification of a forest ecosystem in North-East France.* New Phytologist 126, 533-539.

Thompson, K., J.G. Hodgson, J.P. Grime, I.H. Rorison, S.R. Band and R.E. Spencer, 1993. *Ellenberg numbers revisited.* Phytocoenologia 23, 277-289.

Tietema, A., B.A. Emmett, P. Gundersen, O.J. Kjønaas and C. Koopmans, 1998. *The fate of 15N labeled nitrogen deposition in coniferous ecosystems.* For. ecol. manage. 101, 19-27.

Tiktak, A. and J.J.M. van Grinsven, 1995. *Review of sixteen forest-soil-atmosphere models.* Ecological Modelling 83, 35-54.

Tognetti, R., P. Cherubini and J.L. Innes, 2000. *Comparative stem-growth rates of Mediterranean trees under background and naturally enhanced ambient CO2 concentrations.* New Phytologist 146, 59-74.

Ulrich, B., 1981. *Okologische Gruppierung von Boden nach ihrem chemischen Bodenzustand.* Z. Pflanzenernähr. Bodenk. 144, 289-305.

UN/ECE, 1996. *Manual on methodologies and criteria for mapping critical levels/loads and geographical areas where they are exceeded*. Texte 71/96 Umweltbundesamt, Berlin, Germany, 144+lxxiv pp.

UNECE - CLRTAP, 1999. *The multi-effects, multi-pollutant Protocol, Gothenburg 1999 (*http://www.unece.org/env/lrtap/multipro.htm).

Valentini, R., G. Matteucci, A.J. Dolman, E.D. Schulze, C. Rebmann, E.J. Moors, A. Granier, P. Gross, N.O. Jensen, K. Pilgaard, A. Lindroth, A. Grelle, C. Bernhofer, T. Grünwald, M. Aubinet, R. Ceulemans, A.S. Kowalski, T. Vesala, Ü. Rannik, P. Berbigier, D. Loustau, J. Gudmundsson, H. Thorgeirsson, A. Ibrom, K. Morgenstern, R. Clement, J. Moncrieff, L. Montagnani, S. Minerbi and P.G. Jarvis, 2000. *Respiration as the main determinant of carbon balance in European forests.* Nature 404 (20 April), 861-865.

Van der Heyden, D., J. Skelly, J. Innes, C. Hug, J. Zhang, W. Landolt and P. Bleuler, 2001. *Ozone exposure thresholds and foliar injury on Forest plats in Switzerland.* Environ. pollut. 111, 321-331.

Van Dobben, H.F. and C.J.F. ter Braak, 1998. *Effects of atmospheric NH3 on epiphytic lichens in the Netherland : The pitfalls of biological monitoring.* Atmospheric Environment 32 (3), 551-557.

Van Dobben, H.F., C.J.F. ter Braak and G.M. Dirkse, 1999. *Undergrowth as a biomonitor for deposition of nitrogen and acidity in pine forest.* For. ecol. manage. 114, 83-95.

Van Oijen, M., G.I. Ågren, O. Chertov, S. Kellomäki, A. Komarov, D.C. Mobbs and M.B. Murray, 2003a. *Evaluation of past and future changes in European forest growth by means of four models (including underlying processes and uncertainties).* In: T. Karjalainen and A. Schuck (Eds). Causes and Consequences of Forest Growth Trends in Europe - Results of the RECOGNITION Project, Chapter 4.4.

Van Oijen, M., G.I. Ågren, O. Chertov, S. Kellomäki, A. Komarov, D.C. Mobbs and M.B. Murray, 2003b. *Identification of the major factors which determine forest response to environmental change: A methodology using multiple simulation models*. In preparation.

Van Oijen, M., J. Prietzel, G.I. Ågren, O. Chertov, H.P. Kahle, S. Kellomäki, A. Komarov, K.H. Mellert, H. Spiecker and R. Straussberger, 2003c. *Comparison of the results of the different approaches.* In: T. Karjalainen and A. Schuck (Eds). Causes and Consequences of Forest Growth Trends in Europe - Results of the RECOGNITION Project, Chapter 5.1.

Van Wallenburg, C., 1988. *De dichtheid van moerige gronden.* Internal Report Stichting voor Bodemkartering, Wageningen, the Netherlands, 5 pp.

Velissariou, D., A.W. Davison, J.D. Banes, T. Pfirrmann, D.C. Maclean and C.D. Holevas, 1992. *Effects of air pollution on Pinus halepensis Mill.: pollution levels in Attica, Greece.* Atmospheric Environment 26A(3), 373-380.

Vesterdal, L., E. Ritter and P. Gundersen, 2002. *Change in soil organic carbon following afforestation of former arable land.* For. ecol. manage. 169, 137-147.

Von Fischer, A., 1993. *Zehnjährige vegetationskkkundliche Dauerbeobachtungen stadnaher Waldbestände Reaktionen der Waldvegetation auf anthropogene Beeinflussungen.* Forstw. Cbl. 112, 141-158.

Wagenfuhr, R. and C. Schreiber, 1989. *Holzatlas*. 3 Auflage, Fachbuchverlag, Leipzig, Germany, 720 pp.

Waide, R.B., M.R. Willig, C.F. Steiner, G. Mittelbach, L. Gough, S.I. Dodson, G.P. Juday and R. Parmenter, 1999. *The relationship between productivity and species richness.* Annu. Rev. Ecol. Syst. 30, 257-300.

Wamelink, G.W.W., V. Joosten, H.F. van Dobben and F. Berendse, 2002. *Validity of Ellenberg indicator values juged from physico-chemical field measurements.* Journal of vegetation science 13, 269-278.

Warfvinge, P. and H. Sverdrup, 1992. *Calculating critical loads of acid deposition with PROFILE ? A steady-state soil chemistry model.* Water Air and Soil Pollution 63, 119-143.

Warfvinge, P., U. Falkengren-Grerup, H. Sverdrup and B. Andersen, 1993. *Modelling long-term cation supply in acidified forest stands.* Environ. pollut. 80, 209-221.

Whittaker, R.H., 1972. *Communities and ecosystems*. 2nd, (1975), Macmillan.

Wiselius, S.I., 1994. *Houtvademecum*. Stichting Centrum Hout, Almere, The Netherlands, 380 pp.

Wittig, R., H.J. Ballach and C.J. Brandt, 1985. *Increase of number of acid indicators in the herb layer of the millet grass-beech forest of the westphalian bight.* Angewandte Botanik 59, 219-232.

Annex 1 Addresses

1. ICP Forests and European Commission

ICP Forests	Mr Th. Haußmann ICP Forests/BMVEL P.O. Box 14 02 70 D-53107 Bonn Germany	Tel: Fax: e-mail:	+49.228.529.4321 +49.228.529.4318 thomas.haussmann@bmvel.bund.de
European Commission	Mr. L. Maier/I. Seoane DG AGRI F.1 B - 1049 Brussels Belgium	Tel: Fax: email:	+32.229.62022 +32.229.66255 Ignacio.seoane@cec.eu.int
	Mr. M. Hamell/B. Winkler DG ENV B.3 B - 1049 Brussels Belgium	Tel: Fax: email:	+32.229.53176 +32.229.90895. bernd.winkler@cec.eu.int
FIMCI	FIMCI Secretariat Thialf Business park 31 8443 DA Heerenveen The Netherlands	Tel: Fax: e-mail:	+31.513.855.016 +31.513.855.010 fimci@cybercomm.nl

2. Chairpersons of Expert Panels and Working Groups

Expert Panel on Crown Condition	Mr Joh. Eichhorn Hess. Landesanstalt für F.W.W. Prof. Oelkersstr. 6 D - 34346 Hann. Muenden Germany	Tel: Fax: e-mail:	+49.5541.7004.16 +49.5541.7004.73 eichhornj@forst.hessen.de
Soil Expert Panel	Mr E. Van Ranst University of Gent Lab for Soil Science Krijgslaan 281-S8 B - 9000 Gent Belgium	Tel: Fax: e-mail:	+32.9264.46.26 +32.9264.49.97 eric.vanranst@rug.ac.be
Foliar Expert Panel	Mr H. Raitio Finnish For. Research Inst. Kaironimentie FIN-39700 Parkano Finland	Tel: Fax: e-mail:	+358.34.4352.41 +358.34.4352.00 Hannu.Raitio@metla.fi
Forest Growth Expert Panel	Mr M. Dobbertin WSL/FNP Birmensdorf Zürcherstrasse 111 CH – 8903 Birmersdorf Switzerland	Tel: Fax: e-mail:	+41.1.739.2594 +41.1.739.2215 dobbertin@wsl.ch

Deposition Exert Panel	Mr E. Ulrich Off. Nat. des Forêts Bd. de Constance F - 77300 Fontainebleau	Ph: Fax: E-m:	+33.1.60.749.221 +33.1.64.224.973 erwin.ulrich@onf.fr
Remote Sensing Working Group	Mr C.P. Gross Uni. Freiburg, Fernerkund.&LIS Werderring 6 D - 79098 Freiburg Germany	Tel: Fax: e-mail:	+49.761.400.2881 +49.761.400.2883 grosscp@combo.felis.uni-freiburg.de
Meteorology Expert Panel	Mr T. Preuhsler Bayerische Landesanstalt für Wald und Forstwirtschaft Am Hochanger 11 D - 85354 Freising Germany	Tel: Fax: e-mail:	+49.8161.71.4910 +49.8161.71.4971 pre@lwf.uni-muenchen.de
Vegetation Expert Panel	Mr D. Aamlid Norw. Forest Research Inst. Hogskoleveien 12 N-1432 Ås Norway	Tel: Fax: e-mail:	+47.6494.89.92 +47.6494.29.80 dan.aamlid@skogforsk.no

3. Members and their deputies of the Scientific Advisory Group

Member	*Deputy*
Mr G. Landmann (chairman) Groupement d'Intérêt Public ECOFOR 19, av du Maine F - 75732 Paris, Cedex 15 Tel: +33.1.45.49.88.37 Fax: +33.1.45.49.88.39 e-mail: guy.landmann@engref.fr	N.N.
Mr H. Sterba Univ. f. Bodenkultur Wien Peter Jordanstrasse 82 A - 1190 Wien Tel: +43.1.47654.4200 Fax: +43.1.47654.4242 e-mail: sterba@edv1.boku.ac.at	Mr Joh. Eichhorn Hess. Landsanst. für Forsteinrichtung, Waldforschung und Waldökologie Prof. Oelkersstr. 6 D - 34346 Hann. Muenden Tel: +49.5541.7004.16 Fax: +49.5541.7004.73 e-mail: eichhornj@forst.hessen.de
Mr T. Preuhsler Bay. Landesanst. f. Wald&Forst Am Hochanger 11 D - 85354 Freising Tel: +49.8161.71.4910 Fax: +49.8161.71.4971 e-mail: pre@lwf.uni-muenchen.de	Mr N.O. Jensen Forskningcenter Riso P.O. box 49 DK - 4000 Roskilde Tel: +45.4677.4677 Fax: +45.4675.5970 e-mail: n.o.jensen@risoe.dk

Mr M. López-Arias
CIT – INIA
Carretera de la Coruna km 7,5
E - 28.040 Madrid
Tel: +34.91.34.76.856
Fax: +34.91.35.72.293
e-mail: larias@inia.es

Mr J.M. Rodrigues
Direccao Geral das Florestas
Av. de Joao Cristostomo 26
P - 1050 Lisboa
Tel: +351.1.312.4899
Fax: +351.1.312.4987
e-mail: jrodrigues@dgf.min-agricultura.pt

Mr P. Roskams
Inst. f For.& Game Management
Gaverstraat 4
B - 9500 Geraardsbergen
Tel: +32.54.43.71.15
Fax: +32.54.43.61.60
e-mail: peter.roskams@lin.vlaanderen.be

Mr J.W. Erisman
ECN
Postbus 1
NL - 1755 ZG Petten
Tel: +31.224.5641.55
Fax: +31.224.5634.88
e-mail: erisman@ecn.nl

Mr M. Starr
Finnish Forest Res. Inst.
P.O. Box 18
FIN- 01301 Vantaa
Tel: +358.9.857.65472
Fax: +358.9.857.2575
e-mail: michael.starr@metla.fi

Mr R. Ozolincius
Lith. Forest Research Inst.
Liepu 1, Girionys
LT - 4312 Kaunas District
Tel: +370.7.547.310
Fax: +370.7.547.446
e-mail: miskinst@mi.lt

Mr. O. Westling
IVL – Aneboda
S - 36030 Lammhult
Tel: +46.472.62075
Fax: +46.472.62004
e-mail: olle.westling@ivl.se

Mr. S. Solberg
Norw. Forest Research Inst.
Høgskoleveien 12
N - 1432 Ås
Tel: +47.64.94.89.96
Fax: +47.64.94.89.71
e-mail: svein.solberg@nisk.no

Mr S. Evans
Forest Research Station
Alice Holt Lodge, Wrecclesham
UK - GU 104LH Farnham Surrey
Tel: +44.1.420.222.55
Fax: +44.1.420.236.50
e-mail: sam.evans@forestry.gsi.gov.uk

Mr E.P. Farrell
Dept. of Environm. Res. Man.
University College of Dublin
IRL - Dublin 4
Tel: +353.1.706.7716
Fax: +353.1.706.1102
e-mail: ted.farrell@ucd.ie

Mr K. Raulund Rasmussen
Forest & Landsc. res inst.
Hörsholm Kongevej 11
DK - 2970 Hörsholm
Tel: +45.45.76.32.00
Fax: +45.45.76.32.33
e-mail: krr@fsl.dk

Mr A. Moffat
Forest research station
Alice Holt Lodge, Wrecclesham
UK - GU 104LH Farnham Surrey
Tel: +44.1.420.222.55
Fax: +44.1.420.236.53
e-mail: andy.moffat@forestry.gsi.gov.uk

Mr G. Baloutsos
Forest Research Institute
Terma Alkmanos
EL - 115 28 Athens
Ph: + 30.210.77.93142
Fax: + 30.210.77.84602
e-mail: noaimiar@compulink.gr

Mr M. Ferretti
LINNAEA Ambiente
Via G. Sirtori 37
I 50137 Firenze
Ph: +39.055.608.073
Fax: +39.055.608.311
e-mail: m.ferretti@linnaea.org

Mr B. Petriccione
Nat. Forest Service (V Unit)
Via Giosue Carducci 5
I - 00187 Roma
Ph: +39.06.4665.6523/4
Fax:+39.06.483498
e-mail: conecofor@corpoforestale.it

Mr A. Economou
Forest Research Institute
Terma Alkmanos
EL - 11528 Athens Ilissia
Tel: +30.210.77.84240
Fax: +30.210.77.84602
e-mail: naoimiar@compulink.gr

Mr. A. Szepesi
State Forest Service
P.O. Box 10
H - 1355 Budapest
Tel: +36.1.3743.216
Fax: +36.1.3126.112
e-mail: aesz@aesz.hu

Mr M. Bíba
For. & Game Man. Res. Institute
Strnady 136
CZ - 15604 Praha 516
Tel: +420.2.579.21.276
Fax: +420.2.579.21.444
e-mail: biba@vulhm.cz

Mr V. Henzlik
Lesprojekt
Nabrezn, 1326
CZ - 25044 Brandys N.L.
Tel: +420.2.02.800.137
Fax: +420.2.02.802.434
e-mail: henzlik@uhul.cz

Mr J. Mindás
Forest Research Institute
T.G. Masaryka str. 22
SL - 96092 Zvolen
Tel: +421.8555.314.206
Fax: +421.8555.321.883
e-mail: jozef.mindas@fris.sk

Mrs D. Bezlova
University of Forestry
Kl. Ohridski 10
BU - 1756 Sofia
Tel: +359.2.6301.357
Fax: +359.2.622.830
e-mail: bec@astratec.net

Ms N. Goltsova
SRCES of RAS, SP&SU
Oranienbaum skoje sch. 2
RUS-198904 Sankt Petersburg
Tel: +7.812.427.3258
Fax: +7.812.427.7310
e-mail: corina@mail.dux.ru

4. National Focal Centre Coordinators

Austria	Mr M. Neumann BF für Wald Seckendorff-Gudentweg 8 A - 1131 Wien	Ph: Fax: E-m:	+43.1.878.38.1327 +43.1.878.38.1250 markus.neumann@fbva.bmlf.gv.at
Belarus	Mr A.S. Poukovski P.O. Belgosles Zeleznodoroznaja 27 - 220089 Minsk	Ph: Fax: E-m:	+375.172.26.3107 +375.172.26.3115
Belgium (Wallonia)	Mr C. Laurent Min. Region Wallone, Div. N+F Avenue Prince de Liege 15 B - 5100 Jambes	Ph: Fax: E-m:	+32.8133.5842 +32.8133.5833 c.laurent@mrw.wallonie.be
Belgium (Flanders)	Mr P. Roskams Inst. f For.& Game Management Gaverstraat 4 B - 9500 Geraardsbergen	Ph: Fax: E-m:	+32.5443.71.15 +32.5443.61.60 peter.roskams@lin.vlaanderen.be
Bulgaria	Mrs D. Bezlova University of Forestry Kl. Ohridski 10 BU - 1756 Sofia	Ph: Fax: E-m:	+359.26.301.357 +359.26.22.830 bec@astratec.net
Croatia	Mr J. Gracan Forest Research Institute Cvjetno Naselje 41 CROA - 10450 Jastrebarsko	Ph: Fax: E-m:	+385.1.6273.000 +385.1.6273.035 josog@sumins.hr
Cyprus	Mr Christou Andreas Ministry of Agriculture Cyprus Forestry Department CY – 1414 Lefcosia	Ph: Fax: E-m:	+357.22.303.836 +357.22.303.935 publicity@cytanet.com.cy
Czech Republic	Mrs H. Uhlírová For. & Game Man. Res. Inst. VULHM Strnady 136 CZ - 15604 Praha 516	Ph: Fax: E-m:	+420.2.579.21.276 +420.2.579.21.444 uhlirova@vulhm.cz
Denmark	Mrs A. Bastrup-Birk Danish For. And Landsc. Res. Inst. Hoersholm Kongevej 11 DK- 2970 Hoersholm	Ph: Fax: E-m:	+ 45.4576.3200 + 45.4576.3233 abb@fsl.dk
Estonia	Mr K. Karoles Estonian Centre of Forest Prot Roomu tee 2 EE - 51013 Tartu	Ph: Fax: E-m:	+372.7.339.713 +372.7.339.464 mmk@uninet.ee
Finland	Mr H. Raitio Finnish For. Research Inst. Kaironiementie 54 FIN - 39700 Parkano	Ph: Fax: E-m:	+358.344.35.241 +358.344.35.200 Hannu.Raitio@metla.fi

France	Mr E. Ulrich Off. Nat. des Forêts Bd. de Constance F - 77300 Fontainebleau	Ph: Fax: E-m:	+33.1.60.749.221 +33.1.64.224.973 erwin.ulrich@onf.fr
Germany	Mr Th. Haußmann ICP Forests/BMVEL P.O. Box 14 02 70 D-53107 Bonn	Ph: Fax: E-m:	+49.228.529.4321 +49.228.529.4318 thomas.haussmann@bmvel.bund.de
Germany	Mr. W. Lux BFH-inst. f Forstoecol. u wald P.O.Box 100147 D-16201 Eberswalde	Ph: Fax: E-m:	+49.333.465.346 +49.333.465.354 lux@holz.uni-hamburg.de
Greece	Mr G. Baloutsos Forest Research Institute Terma Alkmanos EL - 11528 Athens	Ph: Fax: E-m:	+30.210.77.931.42 +30.210.77.846.02 mpag@fria.gr
Hungary	Mr P. Csoka State Forest Service P.O. Box 10 H - 1355 Budapest	Ph: Fax: E-m:	+36.1.3743.201 +36.1.3126.112 aesz@aesz.hu
Ireland	Mr P. Neville Coillte, The Irish Forestry Board Newtownmountkennedy IRE - County Wicklow	Ph: Fax: E-m:	+353.1.20.111.62 +353.1.20.111.99 Pat.Neville@coillte.ie
Italia	Mr D. De Laurentis Nat. Forest Service (V Unit) Via Giosue Carducci 5 I - 00187 Roma	Ph: Fax: E-m:	+39.06.4665.6523/4 +39.06.483498 conecofor@corpoforestale.it
Latvia	Ms L. Ziedina State Forest Service of Latvia 13. Janvara iela 15 LV - 1932 Riga	Ph: Fax: E-m:	+371.722.2820 +371.721.1176 liene@vmd.gov.lv
Lithuania	Mr R. Ozolincius Lith. forest research inst. Liepu 1, Girionys LI - 4312 Kaunas District	Ph: Fax: E-m:	+370.7.547.310 +370.7.547.446 miskinst@mi.lt
Luxembourg	Mr J.P. Arend Adm. des Eaux et Forêts B.P. 2513 L - 1025 Luxembourg	Ph: Fax: E-m:	+352.402.201-1 +352.402.201-250 jean-pierre.arend@ef.etat.lu
Moldavia	Mr D.F. Galuppa Min. Agriculturii si Alimentat MO - 277001 Chisinau	Ph: Fax: E-m:	+373.226.22.56 +373.222.32.51
Netherlands	Mr G. Tol, van Ministerie LNV, IKC-N P.O. box 30 NL - 6700 AA Wageningen	Ph: Fax: E-m:	+31.317.474.875 +31.317.474.930 g.van.tol@eclnv.agro.nl

Norway	Mr D. Aamlid	Ph:	+47.64.94.89.92
	Norw. For. Res. Inst.	Fax:	+47.64.94.29.80
	Hogskoleveien 12	E-m:	dan.aamlid@skogforsk.no
	N - 1432 Ås		
Poland	Mr J. Wawrzoniak	Ph:	+48.22.7150.463
	Forest Research Institute	Fax:	+48.22.8224.935
	Bitwy Warszawskiej 1920 nr 3	E-m:	j.wawrzoniak@ibles.waw.pl
	PL - 00973 Warsaw		
Portugal	Ms M. Barros	Ph:	+351.21.312.4896
	Direccao Geral das Florestas	Fax:	+351.21.312.4987
	Av. de Joao Cristostomo 28-6	E-m:	mbarros@dgf.min-agricultura.pt
	P–1069-040 Lisboa		
Romania	Mr N. Geambasu	Ph:	+40.1.240.6095
	ICAS Forest Research Institute	Fax:	+40.1.240.6845
	Soseaua Stefanesti 128, sect.2	E-m:	icasmb@bx.logicnet.ro
	RO - 7000 Bucharest		
Russia	Ms N. Goltsova	Ph:	+7.812.427.3258
	SRCES of RAS, SP&SU	Fax:	+7.812.427.7310
	Oranienbaum skoje sch. 2	E-m:	corina@mail.dux.ru
	RUS-198904 Sankt Petersburg		
Slovak Republic	Mr T. Bucha	Ph:	+421.8555.314.149
	Forest Research Institute	Fax:	+421.8555.321.883
	T.G. Masaryka str. 22	E-m:	tomas.bucha@fris.sk
	SL - 96092 Zvolen		
Slovenia	Mr P. Simoncic	Ph:	+386.1.200.78.07
	Slovenian Forestry Inst. GIS	Fax:	+386.1.257.35.89
	Vecna pot 2	E-m:	primoz.simoncic@gozdis.si
	SLO - 1000 Ljubljana		
Spain	Mr G. Sánchez Peña	Ph:	+34.91.596.4812
	Min.del Medio Ambiente(DGCONA)	Fax:	+34.91.597.5565
	Gran Via de San Francisco 4	E-m:	gsanchez@mma.es
	E - 28.005 Madrid		
Sweden	Mr S. Wijk	Ph:	+46.36.155.759/600
	National Board of Forestry	Fax:	+46.36.166.170
	Skogsstyrelsen, Vallgatan 6	E-m:	sture.wijk@svo.se
	S - 55183 Jönköping		
Switzerland	Mr N. Kräuchi	Ph:	+41.1.793 2595
	WSL/FNP Birmensdorf	Fax:	+41.1.739.2215
	Zürcherstrasse 111	E-m:	kraeuchi@wsl.ch
	CH - 8903 Birmensdorf		
Ukraine	Mr I.F. Buksha	Ph:	+38.0572.43.15.49
	Forest Research Institute	Fax:	+38.0572.43.25.20
	Pushkinska 86	E-m:	buksha@uriffm.com.ua
	61024 Kharkiv		
United Kingdom	Mr A. Moffat	Ph:	+44.1.420.22255
	Forest Research Station	Fax:	+44.1.420.23653
	Alice Holt Lodge, Wrecclesham	E-m:	andy.moffat@forestry.gsi.gov.uk
	UK - Farnham Surrey, GU10 4LH		

Last page on the inside of the cover:

For further information please contact:

European Commission
DG AGRI F.1 / DG ENV B.3
Rue de la Loi 200
B – 1-49 BRUSSELS

ICP Forests
P.O. Box 14 02 70
D – 73107 BONN

Forest Intensive Monitoring Coordinating Institute (FIMCI)
Thialf Business Park 31
NL - 8443 DA Heerenveen