

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

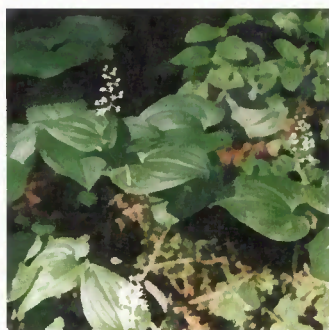
United Nations  
Economic Commission  
for Europe

European Commission

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# Intensive Monitoring of Forest Ecosystems in Europe

Technical Report 2001



Prepared by: **Forest Intensive Monitoring Coordinating Institute, 2001**



Forest Intensive Monitoring  
Coordinating Institute

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Technical Report 2001

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**Forest Intensive Monitoring Coordinating Institute, 2001**



Forest Intensive Monitoring  
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## Abstract

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

Apart from an overview of the implementation of the Pan-European Intensive Monitoring Programme of Forest Ecosystems up to 1998, this year's report focuses on water and element fluxes through the forest ecosystem. Furthermore, first results on the assessment of species composition of the ground vegetation are presented. This year's report forms part of a series of thematic reports. Major conclusions with respect to the ground vegetation assessment are:

- Species numbers show a slight North-South gradient with increasing species numbers in the Mediterranean areas compared to the boreal forests, except for some plots in Norway and Finland. This finding is in accordance with common knowledge.
- The species diversity based on abundance weighting according to the Simpson index varied strongly within countries and there were no clear gradients over countries. High Simpson indices (high diversity) are generally associated to plots with a high number of species and very low Simpson indices (low diversity) to plots with only a few species.

Major conclusions with respect to the water and element budgets are:

- Mean yearly interception ranged from approximately 160 mm for Pine and Oak to approximately 250 mm for Beech and 300 mm for Spruce, reflecting the increasing interception capacity of those tree species. Median transpiration fluxes were rather constant among the tree species and ranged from 325 mm.yr<sup>-1</sup> for Pine to 385 mm.yr<sup>-1</sup> for Spruce stands. This is consistent with literature data. Leaching fluxes mainly reflected the difference in precipitation on tree species. Median values increased from approximately 80 mm under Pine stands to 240 mm under Spruce stands.
- At most plots, the leaching flux of SO<sub>4</sub> is higher than that of NO<sub>3</sub>, despite the generally lower input of S than of N, indicating that SO<sub>4</sub> is still the dominant source of actual soil acidification. The median sulphur budget is close to zero, but at a considerable number of those sites, sulphur is released by the soil, indicating that these systems are recovering from previous episodes of high sulphate input.
- The leaching of N is generally negligible below throughfall inputs of 10 kg.ha<sup>-1</sup>.yr<sup>-1</sup>. There is a significant relationship between N leaching and N deposition but no significant relationship was found with the soil C/N ratio. Nitrogen budgets show that at most sites (90%) the N input is higher than the N leaching.
- Variations in BC leaching were significantly related to the S input and the pH and base saturation. The median base cation balance is close to zero, implying a net adsorption and a net release of base cation at approximately 50% of the plots.
- The Al leaching flux was significantly related to the SO<sub>4</sub> input, reflected by the fact that sites with a high Al leaching coincide with sites with a high input of SO<sub>4</sub>. The geographic patterns of both elements, however, did not coincide very well since soil base saturation was also significantly related to the Al leaching flux.

Keywords: Intensive monitoring, forest, ground vegetation, atmospheric deposition, soil solution chemistry, input-output balances, acidification, eutrophication.

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## Preface

The 'Pan-European Programme for Intensive and Continuous Monitoring of Forest Ecosystems' has been implemented to gain a better understanding of the effects of air pollution and other stress factors on forests. At present 862 permanent observation plots for Intensive Monitoring of forest ecosystems have been selected (512 in the European Union and 350 in several non-EU countries). 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 (formerly SC-DLO) and Oranjewoud International, has been set up to validate, store, distribute and evaluate the data at European level. Apart from the data management, FIMCI also acts as an information centre for National Focal Centres (NFC's), of both EU-Member States and the other participating countries of ICP-Forests.

In the year 1997, the first Technical Report was written including information on the data received up to 1994. This report did not yet contain results obtained from a data evaluation. In the period 1998-2000, three annual Technical reports were published, including information on the data received up to 1995-1997, respectively. Those reports contained results of nearly all the surveys carried out (crown, soil, foliage, increment, atmospheric deposition, meteorology and soil solution), including relationships between results from different surveys, while using statistical techniques for interpretation. From now on, another publication strategy will be followed, highlighting certain topics/themes by more in-depth studies.

The focus of this year's report is on water and element fluxes through the forest ecosystem. This topic was chosen since it is an important aspect in the chemical cycling in forests, which in turn is important in relation to effects. Furthermore, it is a prerequisite for the calculation of critical loads, being an important theme. The report also contains first data on the species diversity of the ground vegetation, focusing on data assessment methods, data comparability and a presentation of the results. Unlike previous years, many external experts have been involved in writing this report, who based their participation on ongoing projects, financed by the EC, related to the topics of this year's report.

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 monitoring programme***

The Pan-European Intensive Monitoring Programme of Forest Ecosystems was 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 factors on forest ecosystems. At present, the programme covers 862 plots in 30 participating countries (512 plots in the EU and 350 plots in non-EU countries). Some surveys are carried out on nearly all 862 plots (crown condition, soil chemistry, foliage and forest growth). At part of those plots, assessments of atmospheric deposition (496 plots), meteorology (201 plots), soil solution chemistry (250 plots), ground vegetation (634 plots) and remote sensing (approximately 160 plots) are carried out. In total 785 Intensive Monitoring plots have been installed. For most of the plots (around 85%) information on the methods applied has been received, validated and stored. The results up to 1998 include validated and stored data for 760 plots with respect to crown condition, 732 plots for foliar composition, 681 plots for soil, 557 plots for forest growth, 502 plots for atmospheric deposition, 416 plots for ground vegetation, 228 plots for soil solution and 151 plots for meteorology. Furthermore, data on ambient air quality and phenology are available at a limited number of plots.

### ***Objectives***

This years report forms part of a series of new thematic reports aiming at an adequate supply of policy relevant information for the coming period and an alternation of a focus on abiotic and biotic aspects. This approach increases the value of the monitoring results and stimulates a good co-operation between FIMCI and other users of the data. The focus of this year's report is on water and element fluxes through the forest ecosystem. This relates to the aim of evaluating the fate of atmospheric pollutants in the ecosystem in terms of accumulation, release and leaching. This topic was chosen in 2001, as it is a prerequisite for the calculation of critical loads. Furthermore, first data on the assessment of species composition of the ground vegetation, submitted at the end of 1999, are presented. The description focuses on data assessment methods, data comparability and straightforward presentation of the results.

### ***Species composition of the ground vegetation***

#### ***Approach***

After the UNCED conference in Rio de Janeiro in 1992, there was a growing concern over the world-wide loss of biodiversity due to anthropogenic influences including the deposition of nitrogenous compounds. The species composition of the ground vegetation, which is assessed at Intensive Monitoring plots, is an indication of the floristic biodiversity of forest ecosystems and can thus contribute to the ongoing discussion. Presently ground vegetation data are available for more than 400 plots. First evaluations focused on the number of species and the abundance weighted species diversity per plot, while checking for data comparability in relation to fenced and unfenced plots and differences in sample area.

#### ***Data comparability***

The comparability of species numbers inside and outside a fence at the same plot was high when the same data assessment methods are used. Effects of fencing, due to the exclusion of grazing, apparently did not yet take place in the relatively short time period between fencing and data

assessment. As sampling area and design varies between countries, a regression analysis was performed in order to check whether these differences would influence the resulting species numbers. Sample area, climatic zone, tree cover indexes were included as predictor variables. Results showed that the influence of sample area on species numbers was not significant, when accounting for differences in other environmental factors. An unbiased assessment of the impact of sample area on the species number requires an assessment of the species number for various sample areas at the same plot, which was not available.

### *Results*

Species numbers showed a North-South gradient with increasing species numbers in the Mediterranean areas compared to the boreal forests, except for some plots in Norway and Finland. In Poland and France, species numbers showed a gradient from West to East. The high number of species in the Mediterranean area (Spanish plots), however, may be influenced by the large sampling area compared to other countries. The species diversity based on abundance weighting according to the Simpson index varied strongly within countries and there were no clear gradients over countries. High Simpson indices (high diversity) were, however, often associated to plots with a high number of species and very low Simpson indices (low diversity) to plots with only a few species. Possibilities to do in-depth evaluations are considerable considering the availability of both species and their abundance in combination with environmental factors influencing those species, such as stand and site characteristics, climatic variables, soil properties and atmospheric deposition. Those possibilities will be investigated and published in next years report.

### ***Water fluxes through the forest ecosystem***

#### *Modelling approach*

Hydrological fluxes were calculated for 245 monitoring sites with the hydrological model SWATRE, using the Penman-Monteith equation for evapotranspiration, the Gash model to calculate interception and Richards' equation to calculate transport of water within the soil. Calculations were carried out with daily meteorological data on precipitation, temperature, net radiation, relative humidity and wind speed. Measurements were used for the plots where a meteorological survey is carried out, whereas interpolated data from nearby meteorological stations were used for plots where only a deposition survey is carried out. A comparison of measured and interpolated meteorological data showed good agreement for relative humidity, reasonable agreement for temperature and net radiation and poor agreement for wind speed. At 80% of the sites average transpiration fluxes were overestimated when interpolated data were used. The median difference in simulated transpiration fluxes was 45 mm. The observed differences in simulated leaching fluxes were comparable to the differences in transpiration fluxes. The use of interpolated meteorological data instead of local measured data leads to lower element leaching fluxes. The deviation in median leaching fluxes is quite limited and ranges from 15 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for Al up to 73 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations.

#### *Reliability of the model results*

Simulated water fluxes through the forest ecosystem were generally plausible in view of available measurements and literature data. The simulated throughfall could be calibrated on measured throughfall figures such that the simulated yearly values were within 5% of the measurements at 85% of the monitoring sites. Biweekly or monthly values were also quite well simulated. Simulated transpiration fluxes and leaching fluxes could not be validated on data for any of the individual sites. However, detailed studies on two sites in Germany and the Netherlands indicated that the model was able to simulate changes in soil water contents and thus in the transpiration

and leaching fluxes quite well. The simulated transpiration fluxes were also consistent with literature data, which indicate that transpiration fluxes for European forest are in a very narrow range around  $335 \text{ mm}\cdot\text{yr}^{-1}$  due to feedback mechanisms with soil and atmosphere.

#### *Model results*

Mean yearly interception evaporation ranged from approximately 160 mm for Pine and Oak to approximately 250 mm for Beech and 300 mm for Spruce. The interception fluxes for Pine and Oak were relatively low due to the relatively low rainfall on those tree species. Interception fractions increased going from Oak (0.22) < Pine (0.24) < Beech (0.27) < Spruce (0.30), reflecting the increasing interception capacity of those tree species. Median transpiration fluxes were rather constant among the tree species and ranged from  $325 \text{ mm}\cdot\text{yr}^{-1}$  for Pine to  $385 \text{ mm}\cdot\text{yr}^{-1}$  for Spruce stands. The range in the sum of soil evaporation and transpiration is also narrow and median values range from approximately  $400\text{-}450 \text{ mm}\cdot\text{yr}^{-1}$  for the various tree species. Leaching fluxes mainly reflected the difference in precipitation on tree species. Median values increased going from 81 mm under Pine stands to 236 mm under Spruce stands. The plots with the lowest leaching fluxes are found in area with relatively low precipitation such as north-eastern Germany parts of Sweden and Finland and locally in southern Europe.

The limited amount of available water in the examined Pine stands was reflected in the calculated mean transpiration reduction which was highest for Pine, with a median value of 15%. For the other tree species, the median value was 10%. In the future reports, more attention will be given to various drought stress parameters, including relative transpiration, that can be used in subsequent analyses relating drought stress to forest growth and forest vitality.

#### ***Element fluxes through the forest ecosystem***

##### *Modelling approach*

Input fluxes from the atmosphere were derived from fortnightly or monthly measurements of the chemical composition of bulk deposition and through fall water, multiplied by the water fluxes while correcting for canopy uptake. An available canopy exchange model was used as a basis and further improved. The resulting canopy exchange was related to available data on bulk deposition, meteorological parameters and foliar chemistry performing multiple regression analysis. Element outputs from the forest ecosystem were derived at intensively monitored plots by multiplying fortnightly or monthly measurements of the soil solution composition at a depth below the rootzone (mostly near 80 cm) with simulated soil water fluxes during those periods (see above). Element retention or release was assessed from the difference between the leaching from the bottom of the root zone and the element input from the atmosphere.

##### *Reliability of the model results*

There are considerable uncertainties in the calculated budgets, considering the uncertainties in calculated water fluxes and measured element concentrations in view of spatial variability within a plot. Furthermore, the budgets are based on measurements during a relatively limited number of years. For most sites (58%), budgets were limited to a three-year period (1996-1998) and for 28% of the sites it was even less, while data for a four year period (1995-1998) were available at 14% of the sites. This relatively short time span may lead to over- or underestimation of the budget compared to the long-term situation due to particular hydrological or biological circumstances in specific years. The remarkable high N retention in south-eastern Germany, for example, may be due to the fact that these budgets are mainly based on the year 1996 with a relatively low precipitation, which may lead to unrepresentative budgets.

### *Model results*

Element budgets for sulphur, nitrogen, base cations and aluminium clearly reflected the behaviour of those elements in response to atmospheric deposition. Median values for S leaching were nearly equal to the median S deposition, indicating the overall tracer behaviour of S. On a considerable number of sites S leaching was, however, higher than S deposition. Sites with the highest sulphur release are located in central Europe, where the strongest reduction in sulphur deposition has taken place over the last decade. This indicates that these systems are releasing sulphur stored in the soil in previous episodes of higher sulphate input.

In accordance with the available literature, N leaching was generally negligible below N inputs of  $10 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ . At higher inputs N leaching increased, but at most sites (90%) the N input was higher than the N leaching, reflecting N retention in the soil. Sites with a net release of nitrogen were found in areas with a high N deposition over a prolonged period of time, such as Belgium and north-western Germany. There was a significant relationship between N leaching and N deposition, but not with the soil C/N ratio, although the C/N ratio appeared to influence the average nitrate concentrations. Furthermore, N leaching was limited at high C/N ratios ( $>30$ ) in the organic layer. Due to the different behaviour of S and N, the leaching flux of  $\text{SO}_4$  was mostly higher than that of  $\text{NO}_3$ , indicating that  $\text{SO}_4$  is still the dominant source of actual soil acidification despite the generally lower input of S than N.

The median base cation balance was close to zero, implying a net adsorption and a net release of base cation at approximately 50% of the plots. The phenomenon of base cation removal due to man-induced soil acidification is thus limited, specifically since high leaching values were partly due to natural acidification in soils with a high pH and base saturation. The impact of air pollution on base cation removal is, however, clear since the leaching flux of base cations (Ca+Mg+K) increased significantly with an increase in the sulphur (acid) input. The Al leaching flux was also significantly related to the  $\text{SO}_4$  input (and leaching) reflected by the fact that sites with a high Al leaching coincide with sites with a high input (leaching) of  $\text{SO}_4$ . The geographic patterns of both elements did not coincide very well, however, since soil base saturation was also significantly related to the Al leaching flux.

# 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 (Council Regulation (EEC) No 3528/86) 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 under regulation No. 1091/94 with the aims to (ICP Forests, 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.

Based on the agreed selection criteria, laid down in Commission Regulation (EC) N° 1091/94, the EU Member States started to select and install their plots in 1994. After revision and acceptance of the ICP Forests Manual (Task Force meetings in Lillehammer and Prague, 1994 and 1995), also the non-EU countries started with the selection and installation of plots. In January 1995 and January 1996 progress reports on the selection and installation of the Intensive Monitoring plots were published (e.g. EC, 1996; 'orange brochure'). Since then, several countries have reviewed their selection and in some cases amendments have been made. Details on the plots and assessments can be found 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 a limited number (at least 10%) of the plots include atmospheric deposition, meteorological parameters, soil solution chemistry and ground vegetation. In the expert panels on deposition and meteorology, the possible inclusion of ambient air quality ( $O_3$ ,  $SO_x$ ,  $NO_x$  and  $NH_x$ ) and of phenology has been discussed and a manual is in preparation. 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 (continuous)

- ground vegetation (every 5 years)
- remote sensing/aerial photography (once)

### ***Aims of the Programme***

The 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 (specifically the elevated deposition levels of SO<sub>x</sub>, NO<sub>x</sub> and NH<sub>x</sub>) 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. More specific objectives in the context of air pollution are the assessment of:

- Responses of forest ecosystems to changes in air pollution by deriving trends in stress factors and ecosystem condition.
- The fate of atmospheric pollutants in the ecosystem in terms of accumulation, release and leaching.
- Critical loads and critical levels of atmospheric pollutants (SO<sub>2</sub>, NO<sub>x</sub>, NH<sub>3</sub>, metals) in view of ecosystem effects in relation to present loads.
- Impacts 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' (Min conference, III.). Objectives of the Pan-European Programme related to this topic can be formulated as:

- 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**

### ***Aims and contents of previous reports***

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. In the year 1997, the first Technical Report was written containing information on the data received until 1994. This report did not contain results obtained from a data evaluation. In the period 1998-2000, three Technical reports were published of a similar character. Those reports contained information on all the surveys carried out until 1995-1997, respectively (crown, soil, foliage, increment, atmospheric deposition, meteorology and soil solution), describing the results of the different surveys, partly in relation to each other, while using statistical techniques. The second Technical Report focused on a description of data assessment methods and on preliminary results of key parameters and correlative studies for the surveys on crown condition, soil, foliar

composition, forest growth and atmospheric deposition. More in-depth evaluations were included in the third and fourth Technical Reports of 1999 and 2000. Methods and results specifically aimed at insight in: (i) ranges in atmospheric deposition and soil solution chemistry in view of critical limits and their interrelationship (De Vries et al., 1999, 2000a) and (ii) relationships between crown-, soil- and foliar condition and environmental factors, such as meteorology and atmospheric deposition, accounting for differences in stand and site characteristics (De Vries et al., 2000a).

It now seems worthwhile to follow another publication strategy, since publication of new results from all the (bi) annual surveys would mean a repetition of comparable information already published before. Instead, it is more relevant to focus on certain topics/themes in the following years, thus highlighting a specific topic by more in-depth studies. In this context, it is worthwhile to make as much as possible use of the results of separate ongoing projects, of which many are co-financed by the EC, that make use of the Intensive Monitoring data. This years report therefore forms part of a series of new thematic reports. An overview of the possible central themes and additional topics in the Technical Reports for the period 2001-2005, is given in Annex 1. The publication strategy follows from the strategy for Intensive Monitoring for that period (De Vries, 2000). 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. This approach also increases the value of the monitoring results and stimulates a good co-operation between FIMCI and other users of the data.

### *Aim of this report*

Aspects that have been investigated in this year's report are illustrated in Figure 1.1.

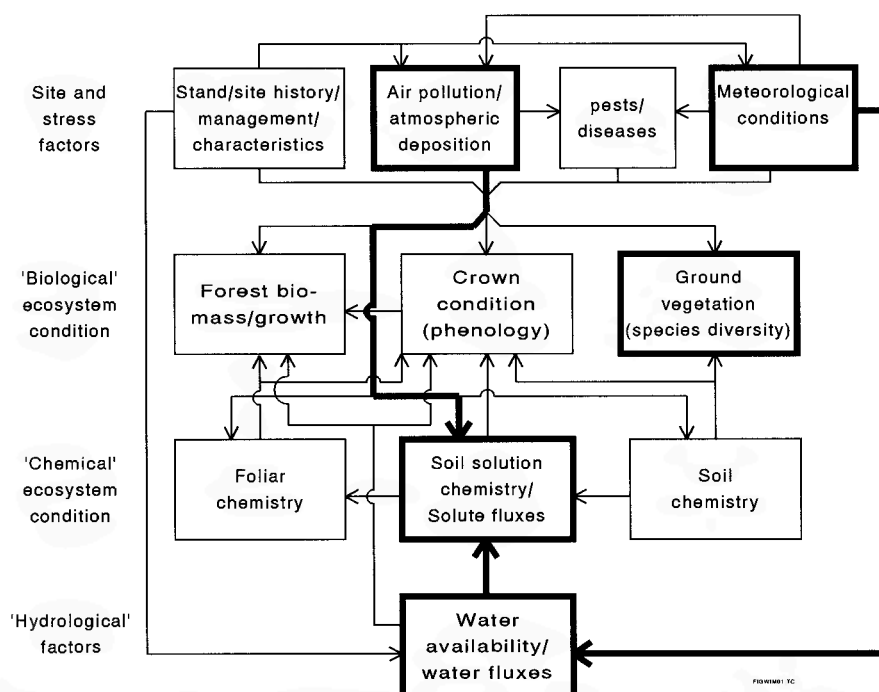


Figure 1.1 Flow diagram illustrating the relationships between site and stress factors and the forest ecosystem condition. Boxes and arrows in bold are specifically investigated in this year's report.

The focus of this year's report is on water and element fluxes through the forest ecosystem. A comparison of element inputs from the atmosphere and element outputs leaching from the bottom



of the root zone give insight in the fate (accumulation or release) of sulphur, nitrogen, base cations and aluminium in the ecosystem. As such, it is of crucial importance to assess the present and future impacts of atmospheric deposition on the element cycle and nutrient availability. This topic was also chosen, as it is a prerequisite for the calculation of critical loads, being an important theme in 2002 (see Annex 1). Furthermore, first data on the assessment of species composition of the ground vegetation, submitted at the end of 1999, are presented. The description focuses on data assessment methods, data comparability and on a simple presentation of the results, only describing possibilities for an in-depth evaluation to be carried out next year.

### **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 1998.

First results on the species diversity of ground vegetation at the plots are presented in chapter 3. This chapter focuses on data assessment methods, data comparability and presentation of the results, such as the number of species per plot and the abundance weighted species diversity. Possibilities to derive relationships between the species diversity of ground vegetation and environmental factors are described but actual results await for next year's report.

The major theme of this report is the assessment of water and element fluxes through the ecosystem, presented in Chapter 4 and 5. The methods and results of the assessment of water fluxes through the ecosystem in Chapter 4 focus on: (i) interception and throughfall and (ii) transpiration and leaching. Specific attention is given model parameterisation and calibration and to data comparability, including a comparison of measured and estimated site-specific meteorological data and the consequences for the simulated water fluxes (Chapter 4).

The methods and results related to element fluxes through the ecosystem in Chapter 5 focus on: (i) canopy exchange and element input and (ii) leaching and element retention. The methods include a description, parameterisation and calibration of the models and the statistical approaches used to assess relationships with stand and site characteristics. Attention is given to comparability of deposition data and of soil solution chemistry data, based on recent and ongoing literature, laboratory and field studies. The results include ranges and geographic variation in element input, element output and element retention and their relationship with environmental factors (Chapter 5).

Chapter 6 contains the discussion and conclusions related to the results in the previous chapters.

## 2 Programme

The Intensive Monitoring Programme is carried out on plots that were selected in such a way that this include the major tree species-soil type combinations 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 1998 (Section 2.2) are presented.

### 2.1 Selected plots in the various surveys

The Intensive Monitoring Programme now includes 862 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 785 of the 862 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 496 plots. Surveys with respect to meteorology and soil solution measurements are carried out at 201 and 250 plots respectively. Furthermore, it can be concluded that ground vegetation surveys will be carried out at 634 plots, whereas the application of aerial photography is foreseen at more than 160 plots (Table 2.1). Several countries also plan to or do carry out additional surveys on the plots, such as phytopathology, litterfall, phenology, mycorrhiza and/or fungi and other in-depth studies to soil water regimes, gas exchange and intensive 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 2001. The map makes a distinction between plots where:

- all surveys are carried out
- all surveys are carried out except for meteorological measurements
- the mandatory surveys (crown condition, soil, foliage and increment) and deposition are carried out and partly also ground vegetation.
- only the mandatory surveys are carried out and partly also ground vegetation.

The map shows that number of plots at which all surveys or all surveys excluding meteorological measurements are carried out occur mainly in a north-south transect going from Scandinavia over Germany to France and Spain. It also shows that atmospheric deposition (at least bulk deposition, but mostly also including throughfall) is measured at much more plots, including a west-east gradient going from the UK to Poland/Hungary.

Table 2.1 Overview of the number of selected plots for the main surveys.

Countries	Total	Crown	Soil	Foliar	Growth	Atm. Dep.	Meteo	Soil sol.	Gr. Veget.	Rem. Sens.
<b>EU countries</b>										
Austria	20	20	20	20	20	20	2	2	20	20
Belgium Flanders	12	12	12	12	12	6	2	6	12	-
Belgium Wallonia	8	8	8	8	8	2	1	2	8	-
Denmark	16	16	16	16	15	10	3	10	15	5
Germany	89	89	89	89	89	86	66	78	80	49
Greece	4	4	4	4	3	4	4	-	4	-
Spain	53	53	53	53	53	12	12	6	52	-
France	100	100	100	100	100	25	25	15	99	14
Ireland	15	15	15	15	15	3	8	3	9	15
Italy <sup>2)</sup>	25	25	25	25	25	17	15	2	25	20
Luxembourg	2	2	2	2	2	1	2	-	2	-
Netherlands	14	14	14	14	14	4	-	14	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	13	16	31	?
Sweden	100	100	100	100	100	46	-	46	-	12
United Kingdom	10	10	10	10	10	10	2	7	10	-
<b>Total EU</b>	<b>512</b>	<b>512</b>	<b>512</b>	<b>512</b>	<b>510</b>	<b>263</b>	<b>159</b>	<b>209</b>	<b>390</b>	<b>135</b>
<b>Non-EU countries</b>										
Bulgaria	3 <sup>1)</sup>	3	3	3	3	3	3	3	3	?
Belarus	81 <sup>1)</sup>	81	81	81	81	-	-	-	-	-
Switzerland	16	16	16	16	16	13	16	7	16	16
Czech Republic	10	10	10	10	10	2	1	1	10	-
Estonia	6	6	6	6	6	5	-	2	6	-
Croatia	7	7	7	7	7	2	3	3	4	?
Hungary	14	14	14	14	14	14	14	-	14	-
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	148	148	148	148	148	148	-	-	148	?
Romania	13	13	13	8	13	4	-	4	13	?
Russia	12	12	12	12	12	12	-	-	-	?
Slovenia	3 <sup>1)</sup>	3	3	3	3	2	3	-	-	?
Slovak Republic	7	7	7	7	7	7	-	-	-	2
<b>Total non-EU</b>	<b>350</b>	<b>350</b>	<b>350</b>	<b>343</b>	<b>348</b>	<b>233</b>	<b>42</b>	<b>41</b>	<b>244</b>	<b>17</b>
<b>Total</b>	<b>862</b>	<b>862</b>	<b>862</b>	<b>855</b>	<b>858</b>	<b>496</b>	<b>201</b>	<b>250</b>	<b>634</b>	<b>162</b>

<sup>1)</sup> In these countries plots have not yet been installed.

<sup>2)</sup> Selected plots: 27

## 2.2 Submitted data and information until 1998

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 this difference are:

- 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.

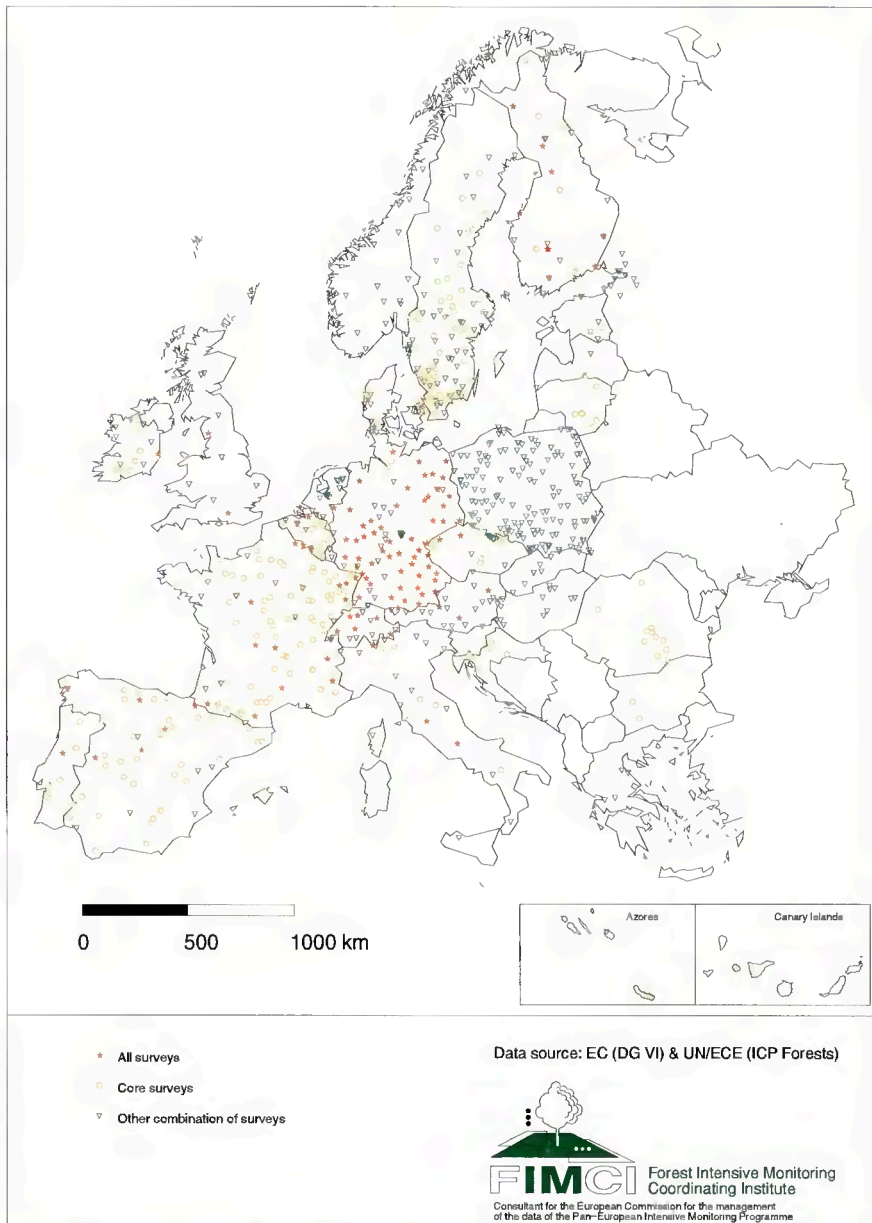


Figure 2.1 Geographical distribution of installed Intensive Monitoring plots based on information received until February 2001.

Compared to last years' report, the number of plots with data has only slightly increased for most surveys. The largest increase is found for ground vegetation as many countries have submitted both data and DAR-Q information last year.

Table 2.2 Overview of the number of plots for which data and/or information was submitted for the eight surveys until the year 1998<sup>1)</sup>.

Survey	Selected plots <sup>2)</sup>		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	512	350	509	251	498	244	497	230
Soil condition	512	350	491	190	446	216	440	190
Foliar condition	512	343	503	229	444	241	441	225
Growth	510	348	481	76	458	89	435	53
Deposition <sup>3)</sup>	263	233	294	208	247	195	243	173
Meteorology	159	42	144	7	159	5	141	5
Soil solution	209	41	208	20	125	17	125	17
Ground vegetation	390	244	236	180	261	172	155	170

<sup>1)</sup> For soil, foliage and increment, also data from earlier years have been used.

<sup>2)</sup> The number of plots for which plot characteristics were received was 785 (see Section 2.1)

<sup>3)</sup> The number of plots for which deposition data are stored (294) is higher than the number of plots where currently deposition is measured (263, see table 2.1) because at a number of plots deposition measurements were stopped in the last years: for these plots only short datasets of the period before 1997/1998 are available.

## 3 The species composition of ground vegetation

### 3.1 Introduction

The concern about forest decline in the 1980s led to the initiation of nation-wide research programs, which mainly focused on the relation between atmospheric deposition and tree health or tree growth (e.g. Heij and Schneider, 1991). The strong research effort in these programs yielded many new insights into the ways in which atmosphere, soil and vegetation interact. Later, forest dieback appeared to be a rather localised phenomenon, although large scale effects on forest due to multiple stress have been found (Klap et al., 1997, 2000). At the same time the atmospheric concentrations of sulphur dioxide strongly decreased all over Europe, and the fear for large-scale forest dieback decreased concomitantly. After the Rio convention (Agenda 21, 1992), however, there was a growing concern over the world-wide loss of biodiversity. Atmospheric deposition was considered as one of the factors that might be responsible for this, and the research focus has partly shifted from tree growth to biodiversity. Both acidification and nitrogen enrichment are now accepted as factors that negatively affect biodiversity (e.g. Bobbink et al., 1998).

The species composition of the ground vegetation, which is an indication of the floristic biodiversity of forest ecosystems, is strongly affected by human activities like thinning, cutting, re-planting and removal of dead wood. During the past century, the effects of atmospheric deposition have been added to these activities. Especially deposition of nitrogenous compounds has led to dramatic changes in a number of cases. In many countries a shift in species composition is reported. This shift usually entails a transition from a cryptogam-dominated to a grass-dominated undergrowth, and an increase in species indicative for nitrogen-rich circumstances (see Van Dobben et al., 1999). The combination of ground vegetation data and environmental data sampled over a large part of Europe that are now available offer a unique opportunity to achieve a better understanding of the relation between the species composition of the ground vegetation and environmental factors, including atmospheric deposition. In this way, it may be possible to identify those environmental factors that most strongly determine the species diversity of the ground vegetation. If such factors are known, it might be possible to assess more precisely threats to species diversity, to which local governments might anticipate.

The four major aims of ground vegetation monitoring can be summarised as to:

- Relate vegetation composition to environmental conditions at a given point in time;
- Detect temporal changes in vegetation, using vegetation as early warning signal for environmental impacts;
- Relate the changes in vegetation to environmental changes by a quantitative evaluation;
- Parameterise and validate models predicting the long-term impacts of environmental changes on species diversity.

This chapter focuses on data assessment methods and data comparability and on a simple presentation of the results, as it is not a straightforward issue how to characterise biodiversity. A description of possibilities is given. However, we limit ourselves to simple presentations (Section 3.2). Such presentations, including the number of species per plot and the abundance weighted species diversity, are given in Section 3.3. In this section, we also describe possibilities (models) to: (i) derive relationships between species diversity of ground vegetation and environmental

factors and (ii) predict long-term impacts of environmental changes on species diversity. In-depth evaluations focusing on both aspects, however, are intended for next years report. Conclusions of this first preliminary evaluation are given in Section 3.4.

### 3.2 Methods

#### 3.2.1 Locations

Figure 3.1 show the locations for which ground vegetation data are currently available.

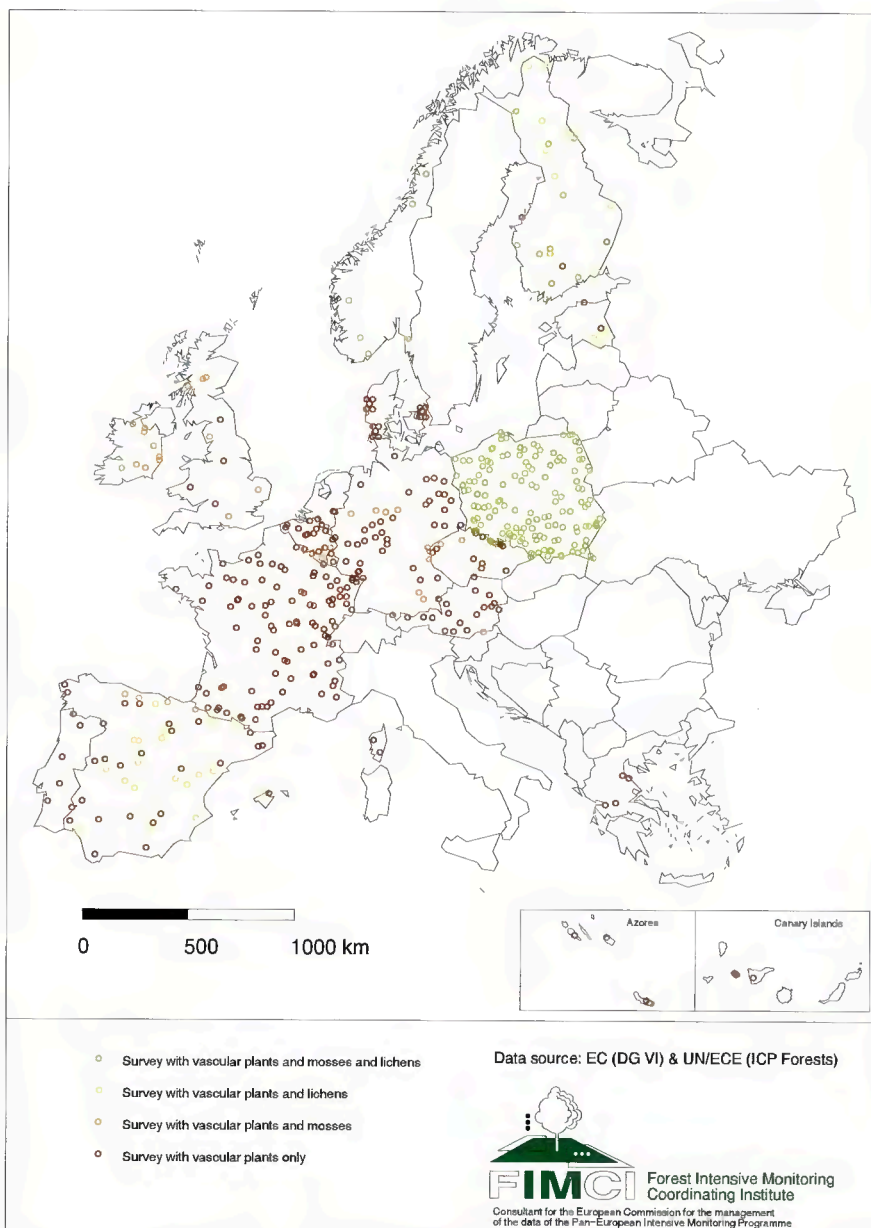


Figure 3.1 Geographical distribution of plots with available ground vegetation data up to 1998, distinguishing between the assessment of vascular plants, lichens and mosses.

For each of the Intensive Monitoring plots, the map shows for which type(s) of ground vegetation (vascular plants, mosses and lichens) data have been received. For Poland only the data for vascular plants were loaded in the database because the submitted species data on mosses and lichens still need to be transformed to the codes from the new mosses and lichens list. This means that currently no exact indication can be given on the occurrence of mosses and lichens on the Polish plots. Figure 3.1 shows that plots with data on both mosses and lichens are mainly found in Scandinavia, Ireland and Poland. Data on mosses were also submitted for a number of plots in western and central Europe, whereas data on lichens are also available for most Spanish plots.

### 3.2.2 Data assessment methods

The vegetation of each stand was described by means of sample plots within that stand. In the sample plots estimates of the abundance were made, i.e. the quantity of each species. The number and size of the sample plots and the methods of quantitative assessment per species (most often in abundance classes) differ regionally. Usually vegetation layers (i.e., herb, tree) are distinguished, but the definition of each layer is also regionally different. Therefore, the present evaluation focuses on species numbers for the total ground vegetation layer only. This circumvents the problems arising from the use of differently defined layers and abundance classes. However, even when concentrating on species numbers, some form of standardisation is necessary. In a given stand, the number of species observed in sample plots a priori depends on (i) total sample area per plot, (ii) position and size of the sampling units, and (iii) the measurement of species inside or outside a fence. These differences are described in more detail below.

#### *Sample area and positioning of the sampling units*

In determining the species composition of the ground vegetation, many countries used a number of sampling units within the plot. In the present evaluation, it is assumed that the number of observed species only depends on the total sampled area, irrespective of the number and size of the sampling units. Table 3.1 presents an overview of the ranges in numbers of sampling units, the area of each sampling unit and the total sampled area used in the 504 plots for which ground vegetation data were submitted.

Table 3.1 Numbers of plots (N) for different ranges in numbers of subplots, the area of each subplot and the total plot area.

Nr of subplots	N	Area of subplot (m <sup>2</sup> )	N	Total sampled area (m <sup>2</sup> )	N
1	32	1-10	230	10-50	58
2-15	234	11-25	20	51-99	173
16-25	187	26-100	208	100-399	42
26-100	34	101-500	22	400-999	195
101-200	2	501-2500	9	1000-2500	21
No information	15	No information	15	No information	15

The total sampled area varied between 10 and 2500 m<sup>2</sup>. In the meeting of the Expert Panel on ground vegetation in Lillehammer, a recommended sampled area of 400 m<sup>2</sup> was suggested. Results show that only 128-143 plots (31-34%) would meet such a requirement (Table 3.1). In order to allow comparison of the data, normalisation of the number of species to a standard area (e.g. 400 m<sup>2</sup>) might be needed. Therefore, the (non-linear) relationship between numbers of species and the sampled area has to be determined accounting for differences in e.g. climate zone and tree species. This aspect is further discussed in Section 3.2.3.



### ***Fencing***

Fencing leads to incomparable results within a relatively short time (half a year) due to the exclusion of grazing (Kuiters et al., 1997; Kuiters and Slim, 2000). Therefore, we only present results of unfenced plots unless fencing took place recently. Table 3.2 shows that nearly all plots have at least an unfenced subplot where the vegetation survey was carried out.

*Table 3.2 Number of fenced and unfenced plots and a combination of both.*

Type of plot	Number of plots
Unfenced	312
Both unfenced and fenced	169
Fenced	8
No information	15

### ***Considered vegetation layers***

Ground vegetation was assessed in various layers that have been divided on the basis of either taxonomy (i.e., mosses, vascular plants), morphology (e.g. shrubs, trees), height (e.g. <0.01m, 0.01-0.5 m, 0.5-10 m and >10m) or a combination (Table 3.3).

*Table 3.3 Number of plots in which ground vegetation is divided by taxonomy, by height and a combination of both.*

Division in layers	Number of plots
Taxonomy	32
Height	26
Taxonomy and height	431
No information	15

### ***Used species codes***

The species are entered in the database coded as nine-digit codes: three digits for the family name, three digits for the genus name and three digits for the species name. This standardised code was taken from the PANDORA database, which is considered as the most up-to-date checklist of the European Flora. This list can be viewed on <http://www.rbge.org.uk/>. When species are encountered that are not in this list, countries are allowed to make their own additions to this list. Such additions are recognisable by an additional code for the latter three digits followed by a code for the country. Agreements that have been made with respect to the species codes for mosses and lichens can be viewed in [http://www.nisk.no/forskning/skogpatologi/ops/icp-for-veg/Code\\_Lists\\_Cryptogams/Default.html](http://www.nisk.no/forskning/skogpatologi/ops/icp-for-veg/Code_Lists_Cryptogams/Default.html).

### ***Description of species abundance***

Estimate of species cover were always made by eye in the field, as perpendicular projection of all living parts on the ground surface. Some fieldworkers first made their estimates in class intervals, for which various coding scales exist, such as Londo or Braun Blanquet (Table 3.4). All codes were then back transformed to percentages before entering the database.

Table 3.4 Number of plots in which the abundance ground vegetation is assessed directly or indirectly by a coding scale and a combination of both.

Abundance scale	Number of plots
% scale	297
Londo	12
Braun Blanquet	169
Barkman et al. + %scale	3
Braun Blanquet + %scale	2
Braun Blanquet + Londo	6
No information	15

### 3.2.3 Data quality assurance and data comparability

Data quality assurance procedures included basic checks whether:

- Codes for e.g. fencing and vegetation species are valid.
- The total cover for each layer is between 0 and 100%.
- The heights of the various distinguished layers are within plausible ranges, e.g. the herb layer is not higher than 2 m and the shrub layer is not higher than 10m.

### 3.2.4 Data evaluation methods

Data evaluation methods focus on the presentation of numbers of species including its abundance. The number of species in a given stand may differ from the 'real' biodiversity of the stand due to influences of: (i) the sample area, (ii) size and position of the sampling units and (iii) the presence of a fence.

The effect of size and position of the sampling units was disregarded in the present analysis. The impact of a fence was investigated by comparing the results of fenced and unfenced sample area for ground vegetation plots for those plots where both assessments were carried out in a similar way. To present comparable data, results that were based on measurements outside a fence were used only. The impact of the sample area was investigated as described below. Finally, results of the species diversity in view of both the number of species and its abundance in the plot are presented, as described further.

#### *Species numbers in relation to the sample area*

There is an ongoing debate in ecological literature as to the shape and nature of the so-called 'species-area curve'. Its principle can be easily understood if one imagines an area A with S species; if a larger area is considered, the same species will be met over and over again. Therefore the number of species increases more slowly than the area considered. The form that is most suitable to express this principle for areas that are not too large is (Lindeijer et al., 1998):

$$S = \beta + \alpha \cdot \log(A) \quad (3.1)$$

in which S is the number of species or species density, A is the plot size (m<sup>2</sup>),  $\alpha$  is a constant (the increase in species number when increasing the plot size by a factor 10) and  $\beta$  is a constant (the number of species in a plot of unit size).

This form is often encountered in literature, and is supported by many data, at least for values of  $A$  in the range  $1 \text{ cm}^2$  to several  $1000 \text{ km}^2$  (Sugihara, 1980; Tokeshi, 1993; Lindeijer et al., 1998). Using this relation, a biodiversity indicator  $H$  for each plot can be derived as:

$$H_i = S_i - (\beta + \alpha \cdot \log(A_i)) \quad (3.2)$$

in which  $H_i$  is the biodiversity indicator of plot  $i$ , being the differences in ‘real’ species number and expected species number considering the sample area,  $S_i$  is the number of species in plot  $i$ ,  $A_i$  is the total sample area, summed over all sampling units of plot  $i$  ( $\text{m}^2$ ) and  $\alpha$  and  $\beta$  are constants, estimated by linear regression.

The equations (3.1) and (3.2) can be used to estimate the number of species at a fixed area, e.g.  $400 \text{ m}^2$ . First, both  $\alpha$  and  $\beta$  have to be estimated with Eq. (3.1) at plots where the number of species,  $S$ , are known for different sample sizes. Then the average number of species at a standardised area can be estimated by equation (3.1), using the estimated constants and a fixed area of  $400 \text{ m}^2$ . Finally, for each plot the deviation  $H_i$  estimated with Eq (3.2) has to be added to get a normalised species number, assuming that the deviation  $H_i$  is independent of the Area  $A_i$ .

The problem with normalising the data this way was, however, that it requires an assessment of the species number for various sample areas at the same plot, which was done for only a very limited number of plots. Instead the sample areas varied over the plots, including variation that also affected the species number, such as the climate zone and the number of trees per hectare or stem density index. To see whether sample area is important we thus performed a regression analyses, with species number as the response variable and with sample area, climate zone number and trees per hectare or stem density index as the predictor variables.

### ***Species weighting***

In view of the available data, the number of species (probably after correction for plot size) seems a simple and practically feasible measure for biodiversity. There are, however, other options to present the data, using the idea of species weighting. Using this approach, one argues that some species have a larger contribution to biodiversity than others, or that measures based on a selection of species are more practical than those based on all species. Biodiversity is thus not simply measured as the number of species, but as:

$$H_i = \sum_{i=1}^n (\text{spec}_{i,j} \cdot \text{weight}_j) \quad (3.3)$$

with  $n$  as the total number of species. The following major criteria for species weighting have been considered (Lindeijer et al., 1998):

1. ‘Importance’ from a nature conservancy point of view (i.e. considering rarity, decline); the IUCN concept of ‘Red Lists’ is such a form of weighting;
2. Functional groups; e.g. primary producers should be present anyway, an ecosystem is only ‘complete’ if large carnivores are present, or, in a forest, mycorrhiza-forming mushrooms should be present, etc.
3. Taxonomic groups; often only mammals, birds and vascular plants are considered. This criterion is related to criterion (2), but is also used in response to practical constraints (e.g. availability of data);

4. Abundance of individuals per species. Some of the classical biodiversity measures, e.g. the Simpson and the Shannon-Weaver index (see e.g. Huston, 1994) use this form of weighting.

*Rarity weighting:* One could argue that the number of rare species is most important species diversity indicator, specifically when those species are declining. This concept is used in the assessment of so-called Red List species. There are, however, practical constraints to weighting the species composition based on Red Lists. Species of the Red Lists are inherently rare and will be absent in most plots. Furthermore, Red Lists are not available for all countries. However, it might be possible to construct a database of species weights based on the concept of the Red Lists, i.e. rarity and decline (Schouwenberg et al., 1997). As this form of species weighting comes closest to the concept of biodiversity as used in political negotiations (e.g. Agenda 21), a further elaboration of these lines seems a promising topic for in-depth evaluations to be presented in later Technical Reports.

*Group weighting:* Weighting to functional group is inherent to the present project, since flora constitutes just a single functional group (i.e., primary producers). Weighting species to taxonomic groups is inherent to the present project, which concentrates on flora. It is important to note that in doing so, the taxonomic group studied is considered to be representative for the biodiversity as a whole, i.e. for all taxonomic groups. A debate is going on in literature whether or not biodiversity is correlated among different taxonomic groups. There are a number of indications that this question may be answered affirmative (Hansson, 1997; Monkkonen and Viro, 1997; Kerr, 1997; Crisp et al., 1998).

*Abundance weighting:* Abundance weighting is an essential ingredient in the ‘classical’ biodiversity measures. These measures were derived from information theory, i.e. they quantify the amount of ‘information’ present in a given set of species (Pielou, 1969). This information quantity depends on two factors, namely the number of species, and a measure called ‘evenness’, this is the extent to which an ecosystem is composed of a few very common species, or of many rare species. The most popular measures for abundance-weighted biodiversity are the Shannon-Weaver index and Simpson’s diversity index. More information on relevant indices is given in Neumann and Starlinger (2001). Some ecological theories (that now have largely become obsolete, however; e.g. Hurlbert, 1971) state that a system’s evenness is related to its stability. In spite of their popularity (especially in the past) the ecological relevance of the abundance-weighted measures is, however, not so clear (Huston, 1994). Despite the critique mentioned above, we used the Simpson’s diversity index to gain insight in the ‘species diversity’:

$$D = 1 - \sum_{i=1}^n p_i^2 \quad (3.4)$$

in which  $p_i$  is the cover percentage of species  $i$  (measured as biomass or number of individuals) in the plot. In applying Eq. (3.5), the percentage cover of all species in a plot was normalised to 100%. It can be easily seen that  $D$  is a weighted number of species, with  $p_i$  as the weighting factor.  $D$  varies between 0 and 1, a higher value indicating that rare species contribute more to the diversity than abundant species. This is in line with the common feeling among ecologists in which ecosystems with many rare species are higher valued than ecosystems with a few abundant species.

In summary, in this report we only apply abundance weighting, with the cover percentage of each species as the weight factor, using the Simpson Index. In the future, the concept of rarity weighting will be explored further.

### 3.3 Results and discussion

#### 3.3.1 Impacts of data assessment methods on species density

Results of the multiple regression analyses showed no influence of the sample area on the species number at unfenced plots. The species diversity decreased significantly at an increased tree number or stem density index and the impact of climate zone was also significant (increasing species diversity from north to south). Sample area impacts, however, could not be detected in this analyses, while accounting for the influence of the climate zone and canopy closure in terms of trees per hectare or stem density index. Consequently, we could not normalise the data on species numbers to a fixed sample area.

A comparison of the species density in fenced and unfenced plots is given in Fig. 3.2. The comparison is limited to 69 assessments at 36 Intensive Monitoring plots, where ground vegetation was assessed on the same day and with the same sample area both inside and outside a fence. Since the date that the fence was installed is not recorded in the DAR-Q, this information could only to some extent be derived from the submitted data. Only if data are available for more than one year, some conclusions about how long a fence is present can be drawn. Because data for more than one year were available for one country only, and most countries submitted results from their first surveys, we assumed that the fences at the plots in this data set were present for a relatively short time period. Results show that a high comparability between the species number at fenced and unfenced plots. On the longer run, fencing will lead to incomparable results with unfenced plots due to the exclusion of grazing (Kuiters et al., 1997; Kuiters and Slim, 2000).

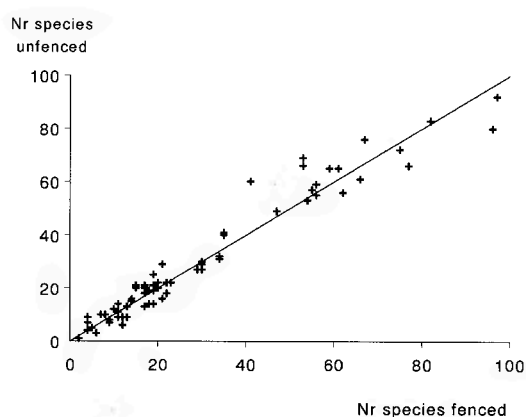


Figure 3.2 Comparison of the species numbers inside and outside a fence for Intensive Monitoring plots at which similar assessment methods were used

#### 3.3.2 Ranges in species density and species diversity

Ranges in number of species and in the abundance weighted species diversity, according to the Simpson index, are given in Table 3.5. The results are limited to vascular plants which were 30

assessed at all investigated plots. The mosses and lichens were excluded because these species have not been submitted by all countries. Tree species were also omitted because they are usually planted, and are therefore considered as a part of the abiotic environment rather than as a part of the natural (spontaneous) vegetation.

*Table 3.5 Ranges in species number and Simpson indices for vascular plants at the 409 unfenced plots for which ground vegetation data were available.*

Variable	Range				
	min	5%	50%	95%	max
Species number	1	7	24	80	134
Simpson index	0.0	0.2	0.7	0.9	1.0

At the 409 investigated plots, the total number of vascular plant species varied between 1 and 134 with a median value of 24 (Table 3.5). When including the tree layer, the species numbers hardly changed, the median being equal to 25 and the maximum to 136. Mosses and lichens were only assessed at 49 and 39 plots, respectively. The number of mosses and lichens at those plots varied between 1-7 and 1-6 respectively. The Simpson index varied from 0 (only 1 species) to 1.0 (a large number of species) with a median value of 0.7 (Table 3.5).

### **3.3.3 Geographic variation of species density and species diversity**

A map of the number of different species at the investigated unfenced plots using the original data is given in Figure 3.3. To make the results comparable, the map only includes the species of the vascular plants, since not all countries included mosses and lichens in their ground vegetation assessment (Fig. 3.1). Furthermore, the tree layer was also excluded. If a country has different sample areas per plot, the data from the largest sample area were taken. If a country performs more than 1 survey per year, all unique species from all surveys at the plot in the same year were taken.

The results show a slight North-South gradient with higher species numbers in the Mediterranean areas compared to the boreal forests, except for some plots in Norway, where a high number of species is recorded even though the plot size is small (50 m<sup>2</sup>). Also note the gradient in Poland and France where the number of species increases from West to East. The higher number of species for the Spanish plots might partly be influenced by the large sampling area (2500 m<sup>2</sup>) as compared to other countries (mostly between 50 – 200 m<sup>2</sup>) but our analyses did not show a clear effect of sampling area. Furthermore, the North-South gradient is in line with common knowledge, indicating that a possible bias by sampling area can not be large.

#### ***Abundance weighted species diversity***

A map of the abundance weighted species diversity according to the Simpson index is given in Fig. 3.4. The Simpson index was computed for all species in the shrub and herb layer, thus excluding mosses, lichens and the tree layer. Results show that the Simpson index can vary strongly within countries and that there are no clear gradients over countries. As expected, the highest Simpson indices (high diversity) are associated with plots containing a high number of species (compare figure 3.2) and very low Simpson indices (low diversity) with plots containing only a few species.

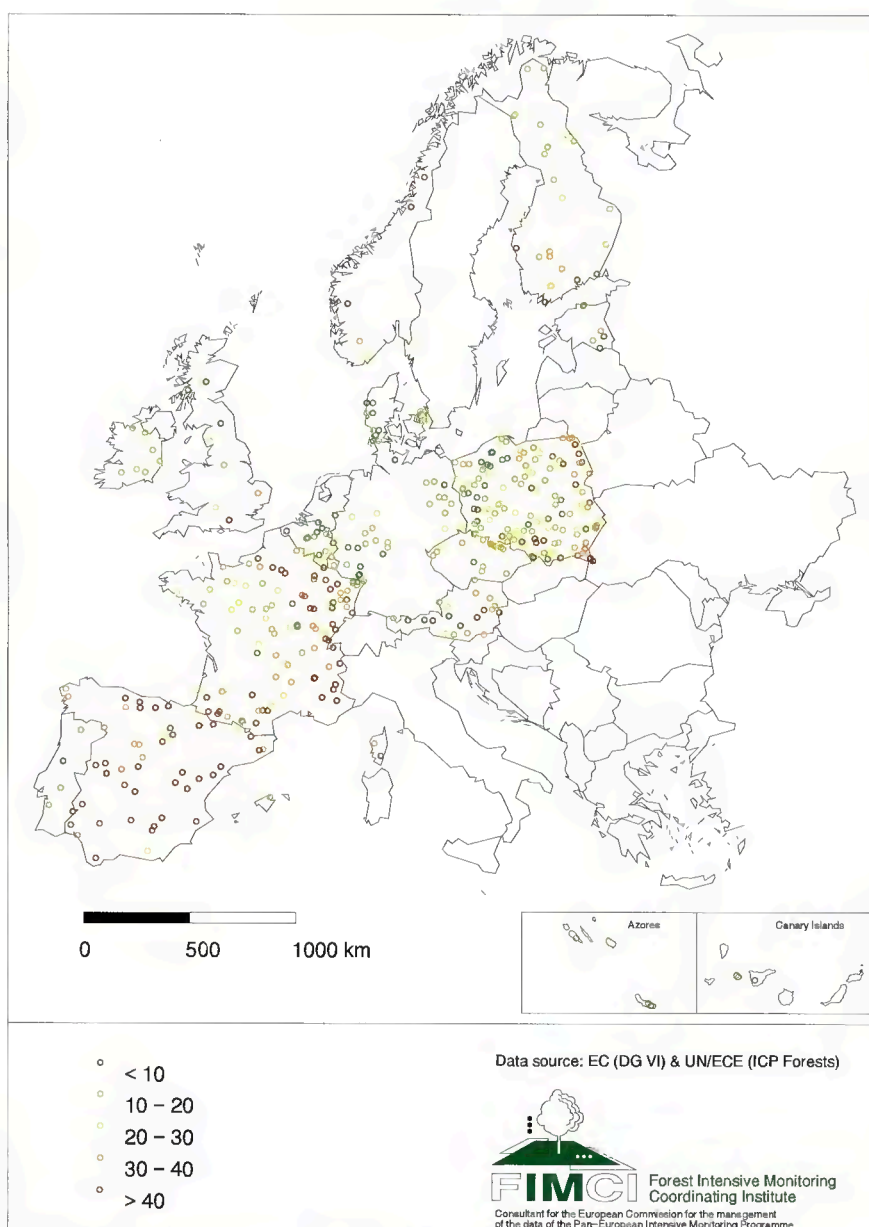


Figure 3.3 Species numbers for vascular plants at the 409 unfenced plots in 1998 for which ground vegetation data were available.

For some plots no Simpson index could be calculated since all species on the plot had a cover of 0.0% (plots with many different species but each with extreme low coverage). The value of the Simpson index is also influenced by the way the species abundance was determined in the original assessment (Table 3.4) and the conversion to cover percentages afterwards. This means that further analysis is needed of the submitted data in conjunction with the DAR-Q information, to see whether these kind of indices indicate real differences in species diversity or mostly reflect differences in assessment methods.

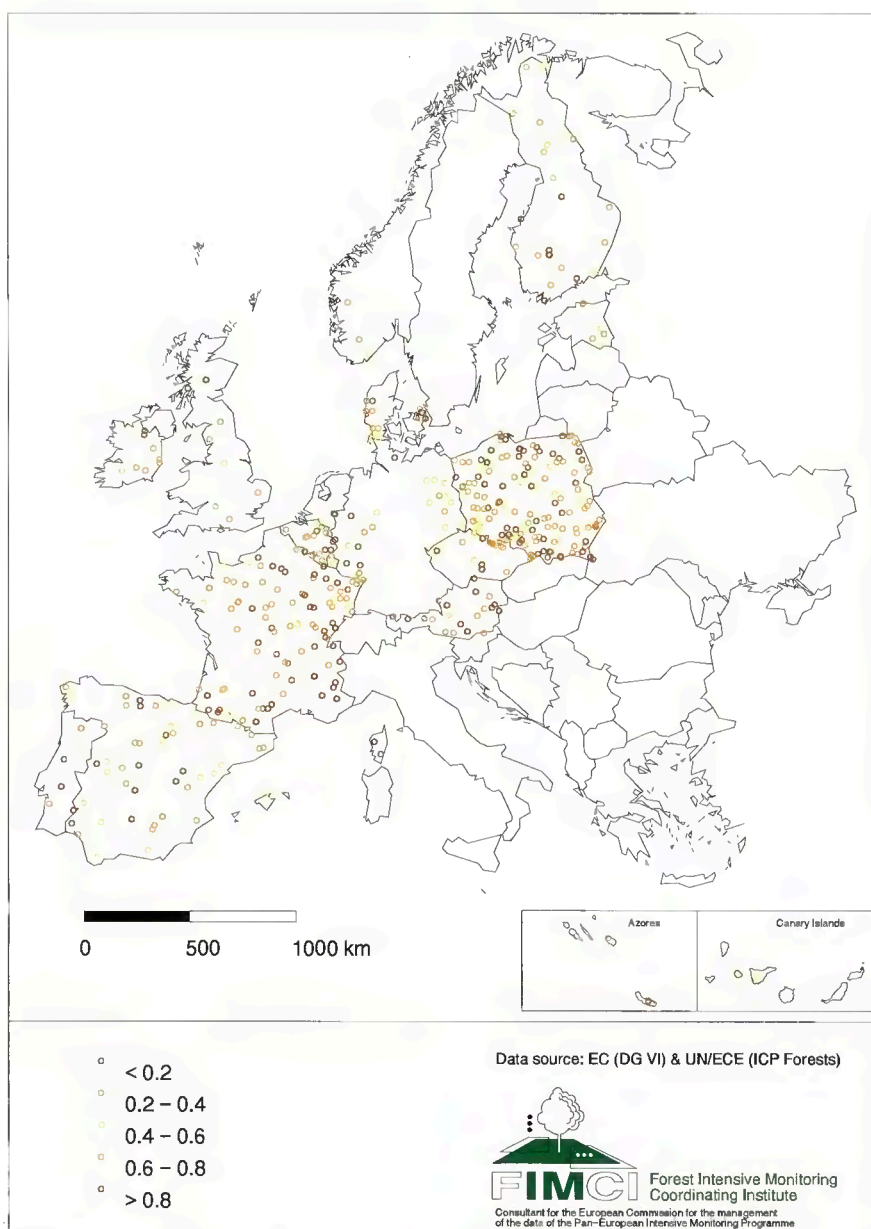


Figure 3.4 Abundance weighted species diversity according to the Simpson index at the 409 unfenced plots in 1998 for which ground vegetation data were available.

### 3.3.4 Possibilities to derive relationships between the species diversity of ground vegetation and environmental factors

Many models that are applied to relate ecological effects to variations in single environmental variables are functionally mechanistic of nature. It is considered questionable (Latour et al., 1993) if mechanistic modelling can predict the combined ecological effects of various environmental perturbations. As the only alternative, the effects of variations in a multitude of environmental variables may be estimated by applying a statistical approach. The statistical derivation of the relation between species diversity and environmental factors can be done by applying two different techniques:



1. Multivariate analysis, giving information on species assemblies as a whole: the ecosystem level approach.
2. Multiple regression, giving insight in the requirements of individual species: The species centred approach.

In both types of statistical models, the availability of a comprehensive data set is a prerequisite. For each plot, the data should contain a complete set of environmental observations (predictor variables) in conjunction with a list of occurring species and their abundance (response variables). On a European scale, the extensive set of physico-chemical predictor variables, of which an indication is given in table 3.6, should preferably be composed of continuous, independent and large gradients. In general, such models are specifically useful to test hypotheses on the impact of (relationships between) environmental factors on species diversity. When such relationships are strong, the models can also be used to make predictions of the ground vegetation composition in space and/or time.

*Table 3.6 Indication of relevant environmental predictors for ground vegetation species.*

Variable type
<b><i>Stand characteristics</i></b>
- Country/co-ordinates
- Soil type (categorical)
- Tree species (categorical)
- Stand age
- Altitude
- Stem density (index)
<b><i>Climatic variables</i></b>
- Annual precipitation
- Annual throughfall
- Annual average evapotranspiration
- Drought stress indices
- Temperature (Annual averages plus extremes)
- Temperature indices
<b><i>Soil properties affecting nutrition</i></b>
- Organic mass of humus layer
- C, N, P, Ca, Mg and K content in the humus layer
- C and N content and base saturation in the mineral topsoil
- The pH (CaCl <sub>2</sub> ) of the humus layer and mineral topsoil
<b><i>Air pollution influence</i></b>
- Bulk deposition and throughfall of NH <sub>4</sub> , NO <sub>3</sub> , SO <sub>4</sub> , Ca, Mg and K

In applying a probabilistic model, a distinction can further be made in models operating at the ecosystem level and at the species level as discussed below. More in-depth evaluations are intended for next years Technical Report.

### ***Statistical models at the “ecosystem level”***

Available ground vegetation data are purely observational, i.e. not resulting from an experiment that was set up to test a single hypothesis. This means that there is a large number of possible relationships to be examined. The floristic data comprise several thousands of species, and the abiotic data at least 100 variables (soil, climate, deposition). The method that is probably most appropriate to analyse this type of data is multivariate statistics (Ter Braak, 1995) or a family of methods to detect patterns in multidimensional data. In general, these methods attempt to reduce the dimensionality of a data set with a minimum loss of information. This is achieved by

extracting sources of variation in decreasing order of importance. In principle, there are many ways in which dimension reduction can be optimised (i.e., loss of information can be minimised), depending on aim of the project and structure of the data. Only two aspects of this optimisation are mentioned here, namely the incorporation of regression analysis, and the underlying response models.

Dimension reduction can be combined with multiple regression analysis, to detect relations between multidimensional response variables and explanatory variables. The simplest way to do this is to extract a few axes (as above) and use each of these as input variables in separate regression analyses (the so-called 'indirect gradient analysis'). However, more sophisticated methods integrate dimension reduction and regression analysis, so that not the mutual relationships between the dependant variables (i.e., the species) are optimally expressed, but their relationships with the explanatory variables ('direct' or 'canonical' gradient analysis). The differences between these two methods are only slight if the relations between the response and explanatory variables are strong.

In regression analysis the response of the dependant variable is usually assumed linear or at least (e.g. after log- or root-transformation) monotone. However, in ecological data the response is often unimodal. Each species occurs optimally at a certain position along an abiotic gradient (e.g. altitudinal or climatic), and decreases in both directions away from the optimum. The cubic transformation that would be appropriate in univariate statistics in that case cannot be applied in multivariate statistics, and more sophisticated transformation methods are needed (Jongman et al., 1995). Methods that assume a linear response are also termed 'PCA-related' (PCA = Principal Component Analysis), those that assume a unimodal response are termed 'CA-related' (CA = Correspondence Analysis).

### ***Predictive models at the species level***

Multiple regression can be used to formally express the occurrence probability of individual species as a function of the variability in predefined environmental factors and possibly their interactions. This type of regression modelling is actually based on covariance of the species and a variety of habitat factors. The required information is not necessarily quantitative; also class data (e.g., soil types or species presence/absence) can be analysed in this way.

Latour et al. (1993) developed a conceptual, species-centred, multiple-stress MOdel for VEgetation (MOVE), which relates the occurrence of individual species of plants to nutrient availability, pH and moisture content. The MOVE model has been extended by adding a soil module predicting those environmental variables as a consequence of environmental policy scenarios. The dynamic soil model SMART (De Vries et al., 1989) is used to generate the required abiotic input of MOVE. The combined model (SMART-MOVE) enables a prediction of the associated changes in the species composition of the vegetation (Figure 3.5).

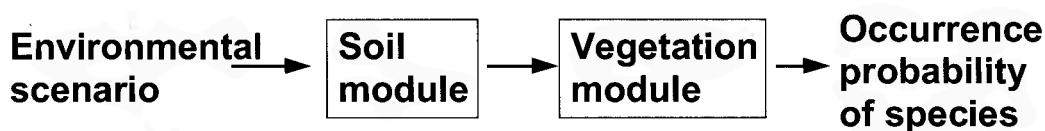


Figure 3.5 Schematic representation of the SMART-MOVE-model.

In order to calibrate the MOVE model, the response curves of 700 Dutch plant species have been constructed for the combination of soil moisture content, nutrient availability and soil acidity (Wiertz et al., 1992; Latour and Reiling, 1993) as illustrated in Fig. 3.6. The calibration process was executed by applying Gaussian logistic regression models on an extensive database developed in a revision of the Dutch classification system for plant communities.

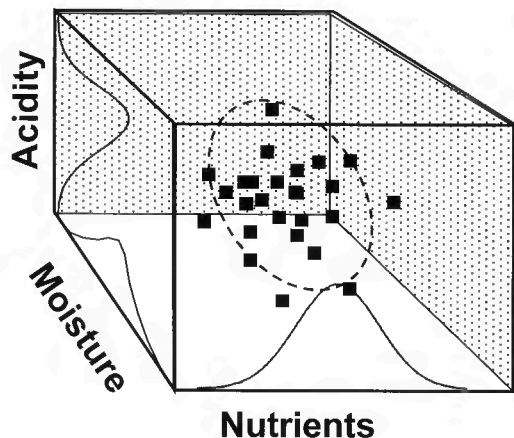


Figure 3.6 *A three-dimensional cube with dimensions defined by variables related to acidification, eutrophication and desiccation. Dots in the cube refer to the occurrence of a particular species. The bell-shaped solid lines are the probability densities projected on the abiotic axis. The dashed "95% probability response volume" describes the "normal operating range" for the species.*

It is possible that a species centred vegetation model reflecting variations in climate, acidification and eutrophication can be made to work on a European scale. Since specific abiotic factors are linked to the occurrence of long-range transboundary air pollution, cause-effect relationships can statistically be established. The possibilities for the application of predictive models are strongly depending on the scenario validity of the input models on a European scale. The SMART-model, that may act as one of the input sources to the vegetation model, has already specifically been developed and applied on a European scale in the context of dynamic scenario analyses (De Vries et al., 1994). Climatic change models, such as the IMAGE2-model (Rotmans, 1990) are also available on a continental scale.

### 3.4 Conclusions

The first results of the ground vegetation assessment lead to the following conclusions:

- The comparability of species numbers inside and outside a fence at the same plot is high when the same data assessment methods are used. Effects of fencing, due to the exclusion of grazing, apparently did not yet take place in the relatively short time period between fencing and data assessment.
- Results of a multiple regression analyses showed no influence of the sample area on the species number, when accounting for differences in climate zone and factors influencing the canopy closure, such as number of trees per hectare or stem density index. An unbiased assessment of the impact of sample area on the species number requires an assessment of the species number for various sample areas at the same plot, which was not available.
- In line with common knowledge, species numbers show a North-South gradient with increasing species numbers in the Mediterranean areas compared to the boreal forests, except for some plots in Norway and Finland. The high number of species in the Mediterranean area,

might partly be influenced by the large sampling area but the impact is likely to be small. In Poland and France, the number of species seems to increase in a West-East direction.

- The species diversity based on abundance weighting according to the Simpson index varied strongly within countries and there were no clear gradients over countries. High Simpson indices (high diversity) are generally associated to plots with a high number of species and very low Simpson indices (low diversity) to plots with only a few species.
- Possibilities to do in-depth evaluations are large, considering the availability of both species and their abundance in combination with environmental factors influencing those species. Examples are (i) stand and site characteristics (e.g. soil type, tree species, stand age, altitude, and soil cover index), (ii) climatic variables, such as precipitation, and soil input (throughfall plus stemflow), (iii) soil properties affecting nutrition such as the content of C and major nutrients (N, P, Ca, Mg and K) in the humus layer and (iv) the bulk deposition and throughfall of those nutrients (NH<sub>4</sub>, NO<sub>3</sub>, SO<sub>4</sub>, Ca, Mg and K). Those possibilities will be investigated and published in next years report. The number of plots that will be included in this evaluation will be larger than in this years report, due to the submission of ground vegetation data after the deadline.

## 4 Assessment of water fluxes through the forest ecosystem

### 4.1 Introduction

Water influences the availability of nutrients by affecting bio/geochemical processes and the loss of nutrients from the rooting zone by leaching. To understand the element cycles in a forest ecosystem (see also Chapter 5) and to predict its future development in response to atmospheric inputs, a quantification of the hydrological situation is therefore indispensable. The hydrological budget is mainly determined by the input of water by rainfall and the loss of water from the forest by interception evaporation, soil evaporation, transpiration and leaching. Apart from influencing nutrient availability, and thereby the vitality and growth of forest ecosystems, the availability of water directly influences forest growth by limiting the transpiration. Furthermore, drought stress may have a strong impact on forest condition in terms of defoliation. Water stress is considered very important with respect to forest condition. Innes (1993) mentioned that the most alarming and frequent observations of a decrease in forest condition in Central Europe coincided with the dry years 1982 and 1983. Landmann (1995) mentioned that defoliation appears to be highest in soils poorly supplied with water and/or in stands in which trees, at some stage of development, have suffered from competition for water. The effects of water stress may diverge from yellowing of the foliage, foliage necrosis, to complete defoliation following extreme drought events (Innes, 1993; Landmann, 1995).

Water fluxes due to interception evaporation can be derived at all Intensive Monitoring plots where bulk deposition and throughfall have been measured. Soil evaporation, transpiration and leaching, however, have not been measured at any of the Intensive Monitoring plots and have to be calculated with hydrological simulation models. Information on water stress, such as the ratio between actual and potential transpiration can also be derived from such models. Studies on water fluxes have been carried out for Intensive Monitoring plots in e.g. Ireland (Nunan, 1999), Germany (Hörmann and Meesenburg, 2000) and France (Granier et al., 2000) and also for forested plots in the Integrated Monitoring Network (Starr, 1999), using different model approaches. A comparison of results of various model approaches for Intensive Monitoring plots in Germany is given in Hörmann and Meesenburg (2000). Such a study was now also carried out for the Intensive Monitoring plots on a European scale, and is described in this chapter. In this chapter, the methods for the derivation of water fluxes due to interception evaporation, soil evaporation, transpiration and leaching are discussed (Section 4.2) and results are presented (Section 4.3). To derive these fluxes soil hydrological models have been used.

Section 4.2 includes a critical review of various hydrological models and gives a description of the particular models used in this study to calculate hydrological fluxes including their parameterisation. Specific emphasis is given to the derivation and use of daily meteorological data, i.e. precipitation, temperature, net radiation, relative humidity and wind speed, which are needed to calculate accurate hydrological fluxes. Those data are only available at the plots where a meteorological survey is carried out. For plots where only a deposition survey is carried out, data availability is, however, limited to monthly or (bi-)weekly rainfall and throughfall data. The missing data have been derived by interpolation using measurements from meteorological stations over Europe (Section 4.2). The results in Section 4.3 are focused on the hydrological budget as a prerequisite for the element budget (see also Chapter 5 on element fluxes). The effect of using interpolated data instead of on-site measured data has also been evaluated. In future reports, more

attention will be given to various drought stress parameters, including transpiration reduction, that can be used in subsequent analyses relating drought stress to forest growth and forest vitality.

## 4.2 Methods

### 4.2.1 Locations

The investigated plots were located in 13 different countries, mostly in Central and Western Europe (Fig. 4.1).

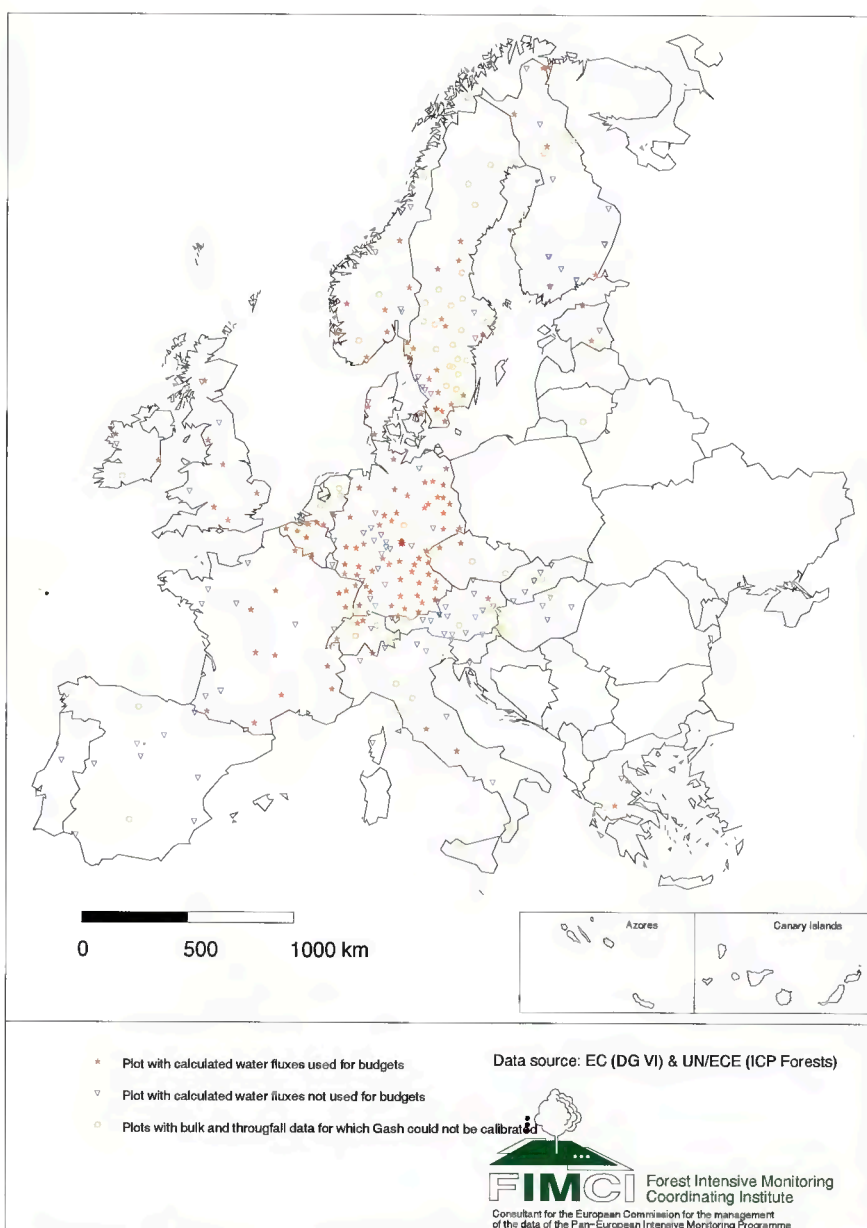


Figure 4.1 Geographical distribution of Intensive Monitoring plots for which water fluxes have been calculated, using data up to 1998.

Annual water fluxes could in principle be calculated for all Intensive Monitoring sites for which both rainfall and throughfall has been measured at a regular basis for a period of at least 300 days. This included 309 sites for the period up to 1998. The number of sites with both bulk and throughfall data increased from 96 in 1995 to 265 in 1996 and to nearly 300 sites in 1997 and 1998. The sites were located in 13 different countries, mostly in Central and Western Europe (Fig. 4.1). At 64 of the 309 plots, hydrological fluxes could not be successfully calculated due to inconsistencies in the provided data (c.f. Section 4.3.1). From the 245 remaining plots for which water fluxes could be calculated, the fluxes of only 121 plots could be used to assess element budgets due to the limited availability of soil solution chemistry data (Chapter 5). The geographic distribution of the different types of plots is given in Fig. 4.1.

#### 4.2.2 Data assessment methods

To calculate the various water fluxes and the transpiration reduction, a set of daily meteorological data on precipitation, net radiation, temperature, wind speed and relative humidity is required. Daily meteorological measurements were, however, only available for plots where a meteorological survey is carried out. For the remaining plots, data were derived by interpolation from existing meteorological databases. In this section, the measurement methods for meteorological data is first described followed by the derivation of scaled and interpolated meteorological data.

To obtain consistent budgets, we used the interpolated data for all plots throughout this study. The use of interpolated data instead of on-site measured data may lead to errors in the calculated budgets. We therefore also give information on the agreement between measured and interpolated data and on the impact of these discrepancies on the calculated hydrological fluxes at the plots where measurements were available (Section 4.3.2.1).

##### 4.2.2.1 Meteorological measurements

At approximately half of the 245 plots at which water fluxes could be calculated, meteorological measurements including methodological information was available. Most equipment, sensors and their placement was in accordance with the World Meteorological Organisation Standard. For the evaluations carried out in this report, it is assumed that no significant deviations in the measurements have occurred due to measuring errors in the equipment itself. Mandatory data submitted include precipitation (sum), air temperature (mean, min, max), relative humidity (mean, min, max), wind speed (mean, min, max), wind direction (mean) and solar radiation (sum). Available data for the years 1995-1998 used in this evaluation are given in table 4.1. The most relevant information on data assessment methods used is given below.

*Table 4.1 Number of plots with meteorological measurements up to 1998. Numbers in brackets include the available meteorological data at the 121 plots where element budgets could ultimately be carried out*

	1995	1996	1997	1998
Precipitation	29 (16)	83 (52)	114 (66)	125 (71)
Net. radiation	7 (3)	56 (37)	89 (53)	103 (61)
Temperature	26 (14)	87 (53)	120 (68)	133 (74)
Relative humidity	24 (12)	85 (51)	118 (66)	131 (72)
Wind speed	7 (3)	64 (39)	92 (51)	103 (57)

### ***Location of measurements***

Measurements for almost all mandatory meteorological parameters were mostly carried out on open field stations within the forest area (Table 4.2). Sometimes measurements were taken in close proximity (in general not more than 2 km distance) of the monitoring plot or above the forest stand canopy.

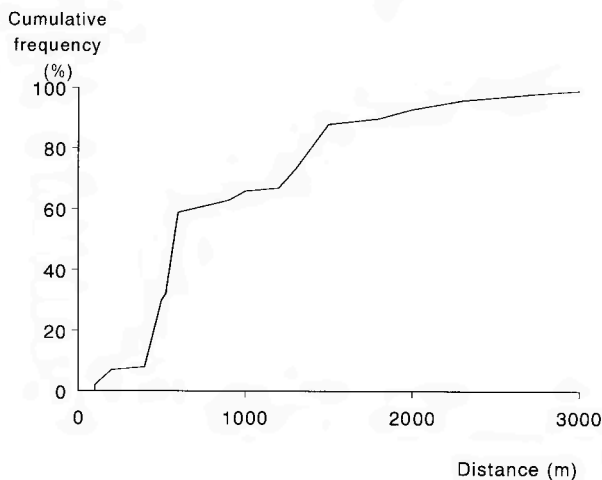
*Table 4.2 Location of meteorological measurements.*

Location	Number of plots				
	PR <sup>1)</sup>	AT <sup>1)</sup>	RH <sup>1)</sup>	WS <sup>1)</sup>	SR <sup>1)</sup>
Above the canopy	13	21	21	19	18
Open field in forest area	85	85	83	74	73
Open field outside forest area	9	6	6	6	6
Total	107	112	110	99	97

<sup>1)</sup> PR = precipitation, AT = air temperature, RH = relative humidity, WS = wind speed and SR = solar radiation.

### ***Distance of the meteorological stations to the Intensive Monitoring plots***

All meteorological stations for which DAR-Q Information has been received are located at or in the vicinity of the Intensive Monitoring plot. The distance between the meteorological plot and the Intensive Monitoring plot, for meteorological stations that are not on the Intensive Monitoring plot itself, lies within 3200 m (Fig. 4.2).



*Figure 4.2 Distance of meteorological stations to the Intensive Monitoring plots.*

#### **4.2.2.2 Rainfall and throughfall data and interpolated meteorological data**

##### ***Rainfall and throughfall***

At all investigated sites, precipitation data are available on a weekly, biweekly or monthly basis from the deposition survey. At 47% of the sites, data were available at a weekly basis, at 27% fortnightly measurements were carried out and at 26% of the sites data were collected monthly. Bulk precipitation is measured using funnels (bulk samplers) situated in the open field near the monitoring plot. Throughfall has been measured at the plots using either gutters or funnels. More detailed information on the number of samplers and the collecting area of the samplers can be found in Chapter 5.2.



Interpolated daily precipitation is based on data from the German Weather Service (Deutscher Wetter Dienst, DWD). Precipitation may vary considerably over short distances, leading potentially to large differences in daily precipitation between interpolated site data and the actual (weekly to monthly) measured precipitation at the site. Therefore, interpolated daily precipitation data has been corrected on the measured biweekly or monthly values according to:

$$P_{i,\text{site}} = P_{i,\text{int}} \cdot \frac{P_{\text{period,site}}}{P_{\text{period,int}}} \quad (4.1)$$

where  $P_{i,\text{site}}$  is the daily precipitation at the site,  $P_{i,\text{int}}$  is the daily interpolated precipitation for the site,  $P_{\text{period,site}}$  is the measured precipitation at the site during a given measurement period (two weekly or monthly) and  $P_{\text{period,int}}$  is the interpolated precipitation during this period.

### ***Other meteorological data***

To calculate the (potential) evapotranspiration flux, data on net radiation, temperature, wind speed and humidity have to be provided. Site specific meteorological data were derived by interpolation between data from the main European meteorological stations. Interpolation to each site was performed using an inverse distance weighting procedure for four meteorological stations located in the surroundings of the plot. More information on the interpolation procedure is given in Klap et al. (1997). Data for the meteorological stations were obtained from the NCAR (National Centre for Atmospheric Research), Boulder, USA and ECMWF.

## **4.2.3 Hydrological model approach**

### ***Possible model approaches***

To quantify evapotranspiration and leaching fluxes the various terms of the hydrological cycle have to be quantified. Water enters the forest by precipitation. Part of this precipitation is lost by interception evaporation. The remaining part enters the forest as throughfall or stem flow. The water that enters the forest may infiltrate into the forest floor or may be lost by surface runoff. The infiltrated water may be taken up by the roots, leached to a deeper layer or (in case of a saturated zone) be lost by lateral drainage (Fig. 4.3). The water balance of a forest stand can thus be described as:

$$P = E + R + D + L \quad (4.2)$$

in which  $P$  is the precipitation,  $E$  is the evapotranspiration,  $L$  is the leaching flux,  $R$  is runoff and  $D$  is the lateral drainage. Evapotranspiration is divided in interception evaporation ( $E_i$ ), transpiration ( $E_t$ ) and soil evaporation ( $E_s$ ) according to:

$$E = E_i + E_t + E_s \quad (4.3)$$

To obtain water fluxes for the above mentioned 309 European forest stands a model had to be selected that is able to calculate fluxes for a broad range of forests preferably on the basis of a relatively limited amount of data. Both requirements are difficult to fulfil because simple models

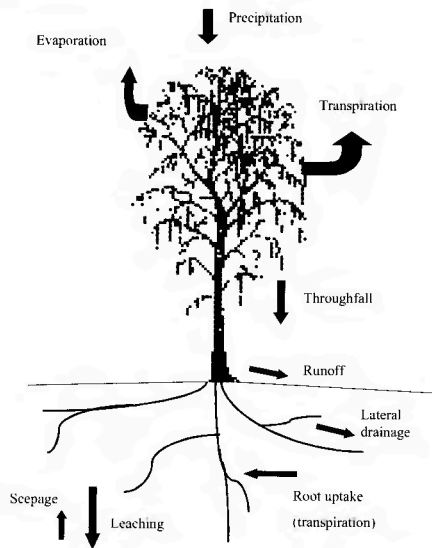


Figure 4.3 The hydrological cycle of a forest stand.

require generally less information but their range of application is often more limited, whereas more comprehensive models require more data which are not available. A range of models ranging from simple budget models to comprehensive process oriented soil hydrological models may be used to calculate output fluxes from forests. Although these models differ substantially, they all consist of three separate submodels to calculate:

- Potential evapotranspiration (upper boundary condition)
- Interception evaporation losses
- Water fluxes within the soil compartment

An overview of the most common (sub)models is given in Table 4.3.

Table 4.3 An overview of some common models used to calculate evapotranspiration, interception and transport of water in forests.

Process	Model complexity			
	Very Simple	Simple	Intermediate	Complex
Evapotranspiration	<b>Chloride balance</b>	<b>Thornwaite (1954)</b>	<b>Makkink (1957)</b>	<b>Penman-Monteith Monteith (1965)</b>
Data needs	$E=f(P,L,D)$ c.f. Eq. 1	$E=f(\text{Temperature})$	$E=f(\text{radiation, temperature, humidity, crop factor})$	$E=f(\text{radiation, temperature, humidity, wind speed, crop factor})$
Interception	<b>Measurements</b>	<b>Empirical relationship</b>	<b>Gash (1979)</b>	<b>Rutter et al. (1971)</b>
Data needs	Regular measurements of rainfall and throughfall	$E_i = a + b P$	$E_i = f(\text{storage capacity of the crown, soil coverage, rainfall intensity and evaporation rate})$	$E_i = f(\text{storage capacity of the crown, aerodynamic properties of the crown, soil coverage, rainfall intensity and evaporation rate on hourly basis})$
Transport	<b>Chloride balance</b>	<b>One layer capacity model</b>	<b>Multi Layer capacity model</b>	<b>Richards (Darcy) model</b>
Data needs	Long term records of throughfall quantities and Cl concentrations in throughfall and soil solution	Soil water retention curve (average for root zone)	Soil water retention curves for different horizons	Soil water retention curves and hydraulic conductivity for different horizons

### ***Potential evapotranspiration***

The evapotranspiration model calculates the potential loss of water to the atmosphere by evaporation of interception water, by evaporation from the soil surface and by transpiration. Inputs to the model are meteorological data and generally a parameter indicating the resistance of the vegetation to the loss of water from the stomata. The main difference between the various models is the detail of process descriptions and their amount of meteorological data needed to run this (sub)model. Complicated models like Penman-Monteith (Monteith, 1965) need daily data on temperature, radiation, humidity and wind speed. Simple models only require temperature data (e.g. Thornwaite, 1954) or temperature and radiation (e.g. Makkink, 1957). However, these simple models have been developed for specific regions and need to be calibrated when applied over a broader scale of meteorological conditions. This is a major disadvantage when selecting a model for calculating a water balance for all Intensive Monitoring plots. Meteorological data to feed the Penman-Monteith model are, however, measured at part of the Intensive Monitoring plots only. For other plots, data have to be derived from the European network of meteorological stations, as described before (Section 4.2.2.2).

### ***Interception evaporation***

Interception loss may be calculated using an empirical relation between rainfall and interception, or by simulation models such as the Gash model or the Rutter model. Empirical relationships may be successfully used for coniferous forests where interception losses are almost constant during the year. However, for deciduous forest different relationships have to be used for the winter and the summer period (Granier et al., 2000). The Gash model is a relatively simple model that calculates interception on basis of the interception capacity of a tree and the degree of soil cover. The model has been successfully used for a range of different situations. The Rutter model is much more complicated, requiring information amongst others on the albedo and aerodynamic properties of the crown and meteorological data on a hourly basis (Rutter et al., 1975).

### ***Soil water fluxes***

To calculate the soil hydrological fluxes, two types of models are often used: capacity models and models based on the Richards equation. The difference in data requirements for these two types of models is relatively small. Capacity model require only limited information (water content at field capacity and at the wilting point) on the soil physical characteristics of the soil, whereas Richards models require a full water retention curve and data on the (saturated) hydraulic conductivity. However, the use of capacity models is limited to well drained soils with a deep water table.

Soil water fluxes may also be calculated on basis of the chloride balance. This method, which is based on the assumption that chloride is inert in the soil, requires equilibrium between the chloride concentrations in the soil solution and the chloride input. Due to the assumption of equilibrium between the input and output of chloride from the soil, this method can only be successfully applied to long term measurements (5-10 years) where storage of chloride in the soil profile can be neglected. Therefore this method is not very suitable for most of the Intensive Monitoring sites, where the length of the data record is short (on average 3 years).

### ***Model selection***

On basis of the model limitations and the available data a selection has been made out of the above model components. Meteorological data can be obtained quite easily either from measured

data (limited number of plots) or by interpolation from existing databases. Therefore the relatively complicated Penman Monteith model was used to calculate evapotranspiration. Interception was calculated using the Gash model, because the Rutter model requires too much data whereas empirical rainfall interception relations are less suitable for deciduous forests. The selection of a soil hydrological model was more difficult. Theoretically, Richards models have clear advantages above the capacity models in particular when considering the one-layer models. However, the availability of data is very limited. Soil physical data are not readily available and have to be derived from texture data. In some cases, even texture data have to be estimated using the soil classification. Finally, it is difficult to characterise the different layers because both a profile description and information on root depth and root distribution are lacking for the considered sites. Due to this shortage of data, it is not yet fully clear whether the theoretical advantages of the Richards model will lead to significantly better results than a capacity model.

To select a model, both types of models were compared at a number of long-term Intensive Monitoring sites, using both site-specific soil physical data and estimated data using the same methods for data derivation as for the Intensive Monitoring sites. Results for two sites in Germany and the Netherlands indicate that chloride fluxes simulated with the Richards' model were close to the long-term chloride budget (Van der Salm, in prep). The capacity models both overestimated the long-term Cl output of the soil system by approximately 20%. The use of generic soil physical data instead of on-site measured data affected the hydrological fluxes by 10%, but the difference between the Richards' model and the capacity models remained almost the same. On basis of the theoretical considerations and the above results it was concluded that the Richards' model is most suitable to calculate output fluxes, in particular when focussing on relatively short periods of time.

The selected model components (Gash model, Penman-Monteith model and the Richards' model) form part of the hydrological model SWATRE (Belmans et al., 1983). This model has originally been developed to calculate hydrological fluxes in agricultural systems. Since then the model has been extended to make it more suitable for forests, by including the Gash interception model, a snow module and improving the feed-back mechanism for water uptake during drought conditions (Groenenberg et al., 1995; Tiktak et al., 1995). The model provides a finite difference solution to Richards equation and includes several different options to calculate interception evaporation and transpiration.

#### 4.2.4 Calculation of interception and throughfall

##### *Model description*

An important aspect in the calculation of the hydrological fluxes is the loss of water by interception evaporation. Daily interception losses at the 309 Intensive Monitoring sites were calculated using the Gash model (Gash, 1979; Gash et al., 1995). Using this model, first the amount of rainfall necessary to fill the canopy storage capacity ( $P_s$ ) is calculated, according to:

$$P_s = \frac{R_{avg}}{E_{avg}} \cdot S_{max} \cdot cc \cdot \ln \left( 1 - \frac{E_{avg}}{R_{avg}} \right) \quad (4.4)$$

where  $R_{avg}$  is the average rainfall rate ( $\text{mm}\cdot\text{hr}^{-1}$ ),  $E_{avg}$  the average daily evaporation rate during rainfall ( $\text{mm}\cdot\text{hr}^{-1}$ ),  $S_{max}$  the storage capacity of the crown (mm) and  $cc$  the canopy closure fraction (-) (mostly erroneously denoted as the soil cover fraction,  $sc$ ). The total amount of interception ( $E_i$ ) is then calculated as:

$$E_i = cc \cdot P \quad \text{when } P < P_s \quad (4.5)$$

or as

$$E_i = cc \cdot P_s + cc \cdot \frac{E}{R} \cdot (P - P_s) \quad \text{when } P > P_s \quad (4.6)$$

where  $P$  is the daily precipitation ( $\text{mm}\cdot\text{d}^{-1}$ ). The maximum interception evaporation is limited by the potential evapotranspiration rate of a wet canopy, e.g. the Penman-Monteith evapotranspiration at a crop resistance of zero (c.f. Eq. 4.8 and 4.11).

### ***Model parameterisation and calibration***

Apart of the meteorological data, the Gash model needs values for the parameters describing the storage and evaporation from the canopy. Normally parameter values for the storage capacity of the crown,  $S_{max}$ , and the canopy closure fraction,  $sc$ , are derived from analyses of data from single storms using the analysis method described by Leyton (Leyton et al., 1967). Such an analysis could not be made for the Intensive Monitoring plots because the Intensive Monitoring database provides only two-weekly or monthly data for throughfall. Instead the parameters  $S_{max}$ ,  $cc$  and  $R_{avg}/E_{avg}$  were calibrated on measured throughfall data for all Intensive Monitoring plots where throughfall has been measured at a regular (weekly to monthly) basis.

To calibrate these parameters an automatic calibration procedure was used based on the Levenberg-Marquardt algorithm (Marquardt, 1963). To guide the optimisation procedure, initial values, upper and lower boundaries for the above parameters had to be provided. Initial parameter estimates for  $S_{max}$  were based on Hendriks et al. (1997), as presented in Table 4.10 (initial values). Upper and lower boundaries were set by permitting a 50% deviation from the initial values for  $S_{max}$ . The initial value for the  $R/E$  ratio was set to 8.0 based on average values reported for a number of studies in Europe (Gash et al., 1980, 1995; Gash and Morton, 1978; Klaassen et al., 1998; Lankreijer et al., 1993; Llorens et al., 1997b; Lousteau et al., 1992b; Pearce et al., 1980; Teklehaimanot and Jarvis, 1991; Valente et al., 1997; Vroom, 1996). For this parameter, wide ranges were used for the upper and lower boundary to account for differences in average rainfall and evaporation rates over Europe.

A problem with the automatic calibration of the Gash model is that the canopy closure and  $S_{max}$  are correlated (comparable results may be obtained when lowering the canopy closure while increasing  $S_{max}$ ). To obtain a good calibration a reasonable initial estimate of the canopy closure is of major importance. The canopy closure of a site is related to stand characteristics such as the number of trees/ha, tree species, LAI and age of trees. To make the best estimate of canopy closure for the Intensive Monitoring sites two different approaches were used and the outcome of both approaches was averaged. The first approach is based on reported data on canopy closure and stand characteristics from interception studies (Carlyle-Moses and Price, 1999; Gash et al., 1980; Granier et al., 2000; Hendriks et al., 1990; Herbst et al., 1999; Hörmann et al., 1996; Lankreijer et al., 1993; Llorens et al., 1997a; Lousteau et al., 1992a; Musters, 1998; Robins, 1974;

Teklehaimanot and Jarvis, 1991; Valente et al., 1997). The collected data showed a considerable scatter in canopy closure but regression analyses on the collected data indicated a clear relationship between the (log of) number of trees/ha and the canopy closure fraction (Fig. 4.4; Table 4.4).

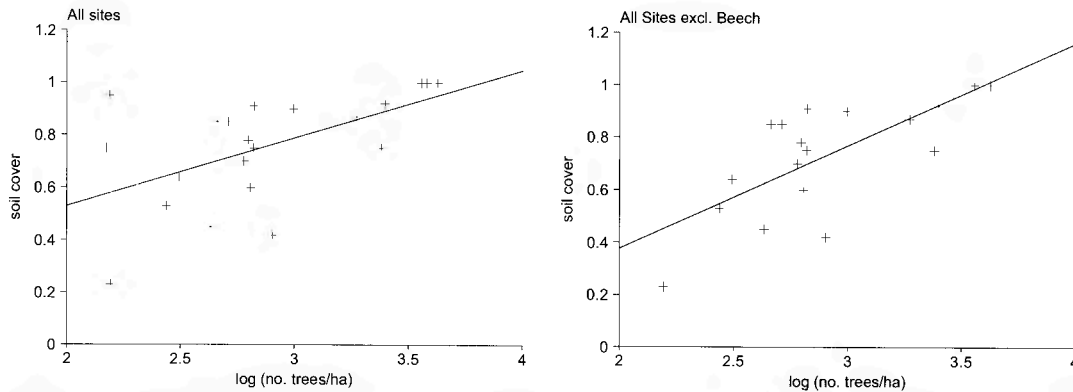


Figure 4.4 Relationship between tree density and observed canopy closure in literature data of 21 forest stands incl. Beech forest (left) and excluding Beech forest (right).

The relationship improved significantly when three sites with Beech forest were excluded from the data set. At two of these sites a high canopy closure (0.75 and 0.95) was reported (Hörmann et al., 1996; Herbst et al., 1999) despite the limited number (approx. 150) of trees.

Table 4.4 Relationship between tree density and canopy closure in literature data of 21 forest stands.

Data set	Relationship	No sites	R <sup>2</sup> <sub>adj.</sub>
All tree species	cc = 0.229 log (trees.ha <sup>-1</sup> ) + 0.09	21	0.21
Spruce, Pine, Douglas fir and Oak	cc = 0.396 log (trees.ha <sup>-1</sup> ) - 0.42	18	0.47

<sup>1)</sup> For Beech the relationship derived for all tree species was used

The second approach was based on relationships between the (average) crown dimensions of individual trees and the diameter of the tree (Nagel, 1999). For each plot canopy closure (the area occupied by tree crowns divided by the area of the plot) was calculated from the average crown diameter of the trees and multiplied by the number of trees, according to:

$$cc = \max\left(1, \frac{(a + b \cdot DBH)^2 \cdot 0.25 \cdot \pi \cdot ntrees \cdot 10^{-4}}{\text{plot size}}\right) \quad (4.7)$$

in which cc is the canopy closure, a (m) and b (m) are constants and DBH is the diameter at breast height (m), ntrees is the tree density (ha<sup>-1</sup>). Parameters for a and b are listed in table 4.5.

Table 4.5 Parameters used to calculate the canopy closure (based on Nagel, 1999).

Tree species	a	b
Oak	1.411	0.154
Beech	1.389	0.181
Spruce	0.842	0.110
Pine	0.714	0.133

This relationship is based on a large number of forest measurements in Northwestern Germany. This method leads to quite comparable estimates of the canopy closure for sparse forests. In dense forests, where crowns are overlapping, the method tends to overestimate the canopy closure. To include the uncertainty of both methods in the estimation of the initial value for the canopy

closure, initial values were set to the average of the canopy closure derived using both relationships. The lowest calculated value for each site was used as the lower limit and the highest obtained value was used as the upper limit.

#### 4.2.5 Calculation of transpiration and leaching fluxes

##### *Model description*

The potential loss of water by evapotranspiration was calculated using the Penman-Monteith equation:

$$E_{pm} = \frac{1}{\lambda} \cdot \frac{s \cdot R_n + \rho \cdot C_p \cdot \delta q / r_a}{s + \gamma \cdot (1 + r_s / r_a)} \cdot f_s \quad (4.8)$$

where  $E_{pm}$  is the potential evapotranspiration ( $\text{mm} \cdot \text{d}^{-1}$ ),  $\lambda$  is the specific heat of evaporation ( $\text{J} \cdot \text{kg}^{-1}$ ),  $s$  is the slope of the saturated water vapour curve ( $\text{hPa} \cdot ^\circ\text{K}^{-1}$ ),  $R_n$  is the net radiation ( $\text{W} \cdot \text{m}^{-2}$ ),  $\rho$  is the density of air ( $\text{kg} \cdot \text{m}^{-3}$ ),  $C_p$  is the specific heat capacity of the air ( $\text{J} \cdot \text{kg}^{-1} \cdot ^\circ\text{K}^{-1}$ ),  $\delta q$  is the water vapour deficit ( $\text{hPa}$ ),  $r_a$  and  $r_s$  are the aerodynamic and canopy resistance ( $\text{s} \cdot \text{m}^{-1}$ ),  $\gamma$  is a psychrometer coefficient ( $\text{mbar} \cdot ^\circ\text{C}^{-1}$ ) and  $f_s$  is the number of seconds per day.

For  $\gamma$ ,  $\rho$  and  $C_p$  constant values were applied. Values used equal  $0.67 \text{ mbar} \cdot ^\circ\text{C}^{-1}$  for  $\gamma$ ,  $1.2047 \text{ kg} \cdot \text{m}^{-3}$  for  $\rho$  and  $1004 \text{ J} \cdot \text{kg}^{-1} \cdot ^\circ\text{K}^{-1}$  for  $C_p$ . Net radiation ( $R_n$ ) was computed from measured global radiation (Beljaars and Holtslag, 1990), the slope of the saturation vapour pressure-temperature curve ( $s$ ) from the temperature and the water vapour deficit ( $\delta q$ ) from measured relative humidity (affecting actual vapour pressure) and temperature (affecting saturated and actual vapour pressure) as presented in Klap et al. (1997). This means that for the use of the Penman/Monteith equation, all mandatory parameters (except for precipitation) are needed.

The aerodynamic resistance was calculated as a function of the wind speed and the roughness of the canopy:

$$r_a(z) = \frac{1}{k^2 \cdot u} \cdot \left( \ln \left( \frac{z-d}{z_0} \right) \right)^2 \quad (4.9)$$

where  $k$  is the von Karman constant (-),  $u$  the wind speed at height  $z$  ( $\text{m} \cdot \text{s}^{-1}$ ),  $z$  is the height of the canopy (m),  $d$  is the zero plain displacement (m) and  $z_0$  is the roughness length (m). The wind speed above the canopy was calculated from the wind speed measured at a standard height of 2 m, assuming a logarithmic profile.

The canopy resistance ( $r_s$ ) is equal to the basic canopy resistance as long as the vapour pressure deficit ( $\delta q$ ) is less than 3.0 hPa. At a higher evaporative demand, the canopy resistance was assumed to increase according to:

$$r_s = r_{s,\text{basic}} + (25 \cdot (\delta q - 3.0)) \quad (4.10)$$

The potential evapotranspiration is divided over interception evaporation (c.f. 3.2.3), potential soil evaporation and potential transpiration. When the canopy is wet,  $r_s$  is zero and the Penman-Monteith equation reduces to:

$$E_{\text{wet}} = \frac{\frac{1}{\lambda} \cdot (sR_{\text{net}} + \rho C_p / r_a \cdot \delta q)}{(s + \lambda)} \quad (4.11)$$

Using this notation for the evaporation during rainfall, the Penman-Monteith total evapotranspiration can be rewritten to:

$$E_{\text{pm}} = \frac{(s + \gamma)}{(s + \gamma \cdot (1 + r_s / r_a))} \cdot E_{\text{wet}} \quad (4.12)$$

Interception evaporation equals the minimum of  $E_{\text{wet}}$  and the potential interception  $E_i$ .

Potential soil evaporation was calculated according to Ritchie (1972):

$$E_s^* = \frac{1}{\lambda} \cdot \left( \frac{s}{s + \gamma} \right) \cdot R_{\text{net}} \cdot e^{(-0.39 \cdot \text{LAI})} \quad (4.13)$$

where LAI is the leaf area index, which was calculated as a function of canopy closure,  $cc$ , according to:

$$\text{LAI} = a \cdot cc + b \cdot cc^2 + c \cdot cc^3 \quad (4.14)$$

where  $a$ ,  $b$  and  $c$  are constants depending on the tree species.

Actual soil evaporation was calculated from the potential soil evaporation and the time since the last rainfall event according to Black et al. (1969):

$$E_s = \varepsilon \cdot (\sqrt{t_d + 1} - \sqrt{t_d}) \cdot E_s^* \quad (4.15)$$

where  $E_s$  ( $\text{m} \cdot \text{d}^{-1}$ ) is the actual soil evaporation rate,  $t_d$  (d) is the number of days elapsed since the last rainfall and  $\varepsilon$  ( $\text{d}^{-0.5}$ ) is an empirical parameter.

Potential transpiration is calculated from the potential evapotranspiration, given by Eq. (4.8), by reducing the evapotranspiration during rainfall with the calculated interception evaporation according to:

$$E_t^* = \frac{(s + \gamma)}{(s + \gamma \cdot (1 + r_c / r_a))} \cdot (E_{\text{wet}} - E_i) \quad (4.16)$$

The actual transpiration depends on the availability of water in the root zone. Reduction of transpiration occurs when the soil water pressure head drops below a certain threshold value. Water uptake from a soil layer becomes zero when the pressure heads drops below the wilting point. The actual transpiration is obtained by summation of the uptake from the different layers.



To calculate the actual evapotranspiration, the potential transpiration was divided over the soil layers on basis of the effective root length. The effective root length ( $L_{ef}$ ) was calculated from the given root length distribution and the degree of water saturation in the different layers to allow for compensatory uptake from relatively wet layers (Tiktak and Bouten, 1994):

$$L_{ef} = \left( \frac{\theta}{\theta_s} \right) \cdot L \quad (4.17)$$

Surface runoff may occur when the rainfall intensity exceeds the infiltration flux of water into the soil. When this occurs water is stored on the surface until a given storage capacity is exceeded. The surplus of water is then lost by surface runoff. Options to calculate lateral drainage are not used in the present application of SWATRE.

The transport of water through the soil was obtained by a numerical solution of Richards' equation:

$$\frac{\delta\theta}{\delta t} = \frac{\delta}{\delta z} \cdot \left( K(h) \cdot \left( \frac{\delta h}{\delta z} + 1 \right) \right) - S(h) \quad (4.18)$$

where  $\theta$  ( $m^3 \cdot m^{-3}$ ) is the volumetric water content,  $t$  (d) is time,  $z$  (m) the vertical position in the soil,  $h$  (m) soil water pressure head,  $K$  ( $m \cdot d^{-1}$ ) hydraulic conductivity and  $S$  ( $d^{-1}$ ) the sink term accounting for root water uptake (actual transpiration).

### **Model parameterisation**

Apart from the meteorological data (c.f. 4.2.3), two main groups of input parameters are necessary for the model SWATRE: those related to the abiotic characteristics of the site (such as number and thickness of the soil layers, soil physical characteristics) and the vegetation dependant parameters (e.g. crop factor, evapotranspiration parameters). Part of this information was not available in the Intensive Monitoring database and had to be derived indirectly using literature data and transfer functions (Table 4.6).

*Table 4.6 Derivation of the main input parameters for SWATRE.*

Parameter	Obtained from
Depth of the soil profile	Soil type and soil phase
Lower boundary conditions	Soil type
Texture of the layers	Directly from the Intensive Monitoring database, from the parent material or based on the FAO soil code
Soil physical characteristics	Based on texture data using transfer function (Wösten et al., 1999)
Root distribution	Depth of the soil profile and literature data (De Visser and de Vries, 1989)
Basic canopy resistance	Literature values for different tree species (Hendriks et al., 1997)
Tree height	Intensive Monitoring database or estimated (Klap et al., 1997)
Interception parameters	Calibration on measured throughfall

The Intensive Monitoring database does not (yet) contain profile descriptions of the various sites and therefore the depth of the soil profile has been derived on basis of the soil type and soil phase. Soil phase for each plot was derived by overlaying the map with plot locations with the soil map of the Soil Geographical Database of Europe at a 1:1Million scale (Eurosoil, 1999). The depth of the profile was derived from a pedotransfer rule indicating the depth of soil profile (shallow (<40 cm), moderate (40-80cm), deep (80-120cm) and very deep (> 120 cm)) as a function of soil code

and phase (Pedotransfer rule 411, European Soil Bureau). A depth of respectively 30, 60, 100 and 230 cm has been allocated to the respective depth classes. The soil profile has been divided into a maximum of 7 horizons (Table 4.7). The thickness of the organic layer was calculated from the measured weight of the organic layer ( $\text{kg m}^{-2}$ ) and the bulk density of that layer ( $\text{kg m}^{-3}$ ) calculated with transfer function that relates bulk density to the measured C content of organic soils (Van Wallenburg, 1988).

With respect to the lower boundary conditions, for all soil profiles free drainage of soil water at the bottom of the soil profile was assumed, except when bedrock was found within 30 cm depth (lithic phase). No hydrology was calculated for soils with ground water influence. In the future this will be improved by assigning different lower boundary conditions to soils with ground water influence, which means that we will also define boundary conditions for soils with gleyic and stagnogleyic regimes.

Table 4.7 Schematisation of the soil profile.

Horizon	Depth (cm)		Number of soil layers and thickness (cm)	Type of compartment
	From	to		
1	>0		1	Litter layer
2	0	5	2 (2.5 cm)	Topsoil
3	5	10	2 (2.5 cm)	Topsoil
4	10	20	2 (5 cm)	Topsoil
5	20	40	4 (5 cm)	Subsoil
6	40	80	4 (10 cm)	Subsoil
7	80	230	6 (25 cm)	Subsoil

Physical characteristics for each horizon were derived using class transfer functions (Wösten et al., 1999), depending on the texture class and whether the horizon is assumed to be a topsoil (A or E horizon) or a subsoil compartment (B and C horizons). Texture for the different horizons were either obtained directly from the Intensive Monitoring database (voluntary submissions) or derived from the FAO soil code (FAO Composition Rules, 1981). Soil physical characteristics for the litter layer were taken from data for a Douglas fir stand, being the only data we could find in literature for these parameters (Tiktak and Bouten, 1994).

Root distribution was based on estimated soil depth and literature data for deciduous and coniferous forest (Table 4.8; c.f. De Visser and de Vries, 1989).

Table 4.8 Root distribution (%) as a function of the depth of the soil profile for coniferous and deciduous forest.

Compartment	Depth of the soil profile									
	20		30		60		70		>70	
	Con	Dec	Con	Dec	Con	Dec	Con	Dec	Con	Dec
Litter	5	5	5	5	5	5	5	5	5	5
0-10 cm	60	55	55	40	45	25	40	20	40	20
10-20 cm	35	40	30	35	20	20	20	20	15	20
20-30 cm			10	20	10	20	10	20	10	15
30-40 cm					10	15	10	15	10	15
40-50 cm					5	10	5	10	5	10
50-60 cm					5	5	5	5	5	5
60-70 cm							5	5	5	5
70-80 cm									5	5

The basic canopy resistance ( $r_s$ ) was derived from a literature overview (Hendriks et al., 1997). Values ranged from  $50 \text{ s.m}^{-1}$  for Oak species,  $85 \text{ s.m}^{-1}$  for Beech,  $90 \text{ s.m}^{-1}$  for Douglas fir to  $100 \text{ s.m}^{-1}$  for Pine. Tree height was based on the reported measurements in the Intensive Monitoring database. For plots where data for tree height were not reported (26 sites), it was estimated as a

function of tree species, tree age, climatic zone and the C/N ratio (Klap et al., 1997). Interception parameters were calibrated on measured throughfall at the plots (c.f. 4.2.5 and 4.3.1).

## 4.3 Results and discussion

### 4.3.1 Interception and throughfall

#### *Model calibration*

Throughfall fluxes were calibrated on measured throughfall at 309 sites for which a sufficiently long record of bulk deposition and throughfall measurements was available up to 1998 (c.f. Chapter 4.2.1). At 90% of the sites (277) throughfall parameters could be optimised using the above mentioned procedure (Table 4.9). At the remaining 32 sites, extreme (combinations of) parameter values had to be used to approach the measured throughfall data. For example on a quite dense spruce stand (704 trees.ha<sup>-1</sup>; height 22 m) throughfall amounted to 91% of the yearly precipitation. The measured throughfall at this site could only be simulated by assuming low values for both the storage capacity of the crown and the canopy closure and a very high value for the average rainfall intensity, which was considered unlikely. The problems on such sites may be caused by errors in the precipitation and/or throughfall measurements or due to incomplete or incorrect information on site characteristics such as tree height and number of trees/ha. An unsuccessful application of the Gash model may also be due to extreme temporal variation in meteorological conditions (e.g. rainfall intensity or evaporation rate) which are not taken into account in the Gash model.

Simulated throughfall data for the considered sites were close to measured values for 88% (245 sites) of the 277 calibrated sites (Table 4.9). At 32 sites the throughfall was not simulated very accurately resulting in a deviation (defined as the absolute value of (measured-modelled)/measured \* 100) between simulated and measured total throughfall of more than 10% or leading to a Normalised Root Mean Square Error of more than 10%. Those 32 sites were not used in the budget calculations.

*Table 4.9 Number of sites for which the Gash model could be successfully applied.*

Tree species	Total	Not calibrated	Calibrated		
			Total	Rejected <sup>1</sup>	Accepted <sup>2</sup>
Spruce	136	16	120 (88%)	13	107
Pine	71	6	65 (92%)	10	55
Beech	51	4	47 (92%)	2	45
Oak	38	5	33 (87%)	3	30
Others	13	1	12 (92%)	4	8
Total	309	32	277 (90%)	32	245

<sup>1</sup> Sites with calibrated parameters for which the NRMSE or the deviation between the total measured and modelled throughfall (expressed as (modelled-measured)/measured\*100) was more than 10%

<sup>2</sup> Sites with calibrated parameters passing the above mentioned quality criteria

The average calibrated parameter values for the storage capacity of the crown,  $S_{max}$ , and the canopy closure,  $sc$ , were close to the initial values (Table 4.10). Calibrated values for the ratio of the average rainfall rate over the average evaporation rate during rainfall,  $R/E$ , were generally lower than initial values, in particular for Beech. The low  $R/E$  values for Beech might be due to an underestimation of the canopy closure in Beech stand. The (limited) literature data indicated

extremely high values for canopy closure under Beech at relatively low tree densities (c.f. Chapter 4.2.4). Moreover, stem flow was neglected in the present application. This may lead to an overestimation of the interception parameters in tree species where stem flow is substantial, such as Beech.

Although the model simulated the measured throughfall quite well and average model parameters were feasible, the range in the optimised parameters was rather high as indicated by the standard deviation (Table 4.10).

Table 4.10 Initial and calibrated values for the Gash parameters.

Parameter	Tree species	Initial settings <sup>1</sup>			Calibrated values	
		Initial value	Lower boundary	Upper boundary	Average	Standard deviation
$S_{max}$ (mm)	Spruce	2.8	1.5	3.3	2.4	0.8
	Pine	1.0	0.5	1.5	0.9	0.3
	Oak	0.9	0.5	1.5	1.4	0.4
	Beech	1.0	0.7	2.0	1.7	0.5
cc (-)	Spruce	0.71	0.60	0.82	0.78	0.17
	Pine	0.71	0.61	0.81	0.77	0.16
	Oak	0.80	0.64	0.95	0.82	0.16
	Beech	0.76	0.56	0.95	0.79	0.16
R/E (-)	Spruce	8	2	13	5.5	2.7
	Pine	8	2	13	4.6	2.4
	Oak	8	2	13	5.4	2.6
	Beech	8	2	13	3.6	1.9

<sup>1</sup> initial settings for canopy closure (sc) refer to average values because values depend on number of trees/ha and/or DBH (see Table 4.5 and 4.6)

The ratio between average rainfall intensity and evaporation velocity (R/E) was significantly higher in the Boreal and Northern Boreal climate zones (approx. 6.5), whereas significantly lower than average values were found in the south Atlantic zone (R/E = 2.7). These results are partly confirmed when analysing the difference in average R/E between the various countries. For example Finland, where all sites are in the Boreal or northern Boreal zone has a significantly higher R/E ratio (8.0) than countries in central Europe. On the other hand, Sweden has an average R/E of only 4.0, although 30% of the sites have a boreal climate.

The observed variation in the calibrated parameter values for canopy closure and storage capacity of oak, spruce and pine trees were weakly related to the tree density, the altitude and/or the latitude. The observed parameters for Beech were not related to either the geographical characteristics of the site or the tree density (Table 4.11).

Table 4.11 Relationships between parameter values for canopy closure (cc) and storage capacity ( $S_{max}$ ) and site characteristics.

Parameter	Tree species	Relationship	$R^2_{adj.}$ (%)
cc (-)	Spruce	$0.67 + 1.5 \cdot 10^{-4}$ no trees	22.0
	Pine	$1.41 + 1.6 \cdot 10^{-4}$ no trees - $1.35 \cdot 10^{-2}$ latitude - $1.7 \cdot 10^{-4}$ altitude	45.0
	Beech	$0.77 + 4.9 \cdot 10^{-5}$ no trees	1.0
	Oak	$0.49 + 9.5 \cdot 10^{-5}$ no trees + $1.23 \cdot 10^{-2}$ height	15.4
$S_{max}$ (mm)	Spruce	$2.88 - 2.9 \cdot 10^{-4}$ no trees - $4.8 \cdot 10^{-4}$ altitude	5.7
	Pine	$0.79 + 2.9 \cdot 10^{-4}$ altitude	6.9
	Beech	$1.25 + 1.7 \cdot 10^{-2}$ tree height	4.0
	Oak	$2.30 + 3.7 \cdot 10^{-2}$ tree height - $3.61 \cdot 10^{-2}$ latitude	23.5

The limited physical relationships between the calibrated parameter values and site characteristics indicates that part of the observed variation in calibrated values is probably due to local minima

in the optimisation procedure. This means that a good correspondence between measured and simulated throughfall may have been reached by using a combination of incorrect parameters. In extreme situations this may lead to errors in the calculated yearly soil evaporation (strongly influenced by canopy closure) and transpiration fluxes or to deviations in the daily water fluxes due to errors in the daily throughfall fluxes. To avoid such errors, detailed information on either the canopy closure or the storage capacity of the crown is indispensable.

### Model results

Yearly but also monthly and two weekly measured data were quite well simulated for the 245 sites passing the calibration procedure and the quality checks (Fig. 4.5). However, the deviation for individual measurement periods was sometimes quite large, as indicated by the cumulative frequency distribution of the deviation between measured and simulated throughfall (Fig. 4.6).

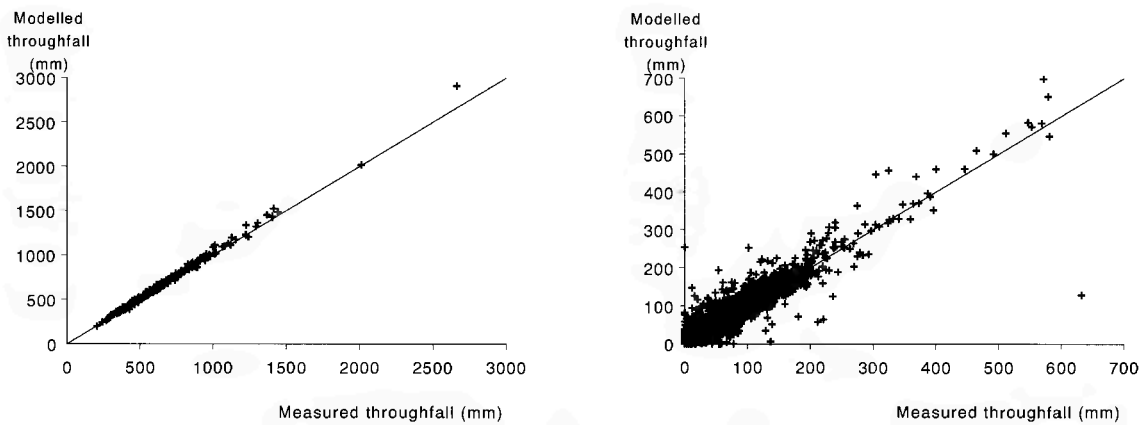


Figure 4.5 Measured and modelled yearly (left) and biweekly/monthly (right) throughfall at the Intensive Monitoring sites for the 245 sites that were successfully calibrated.

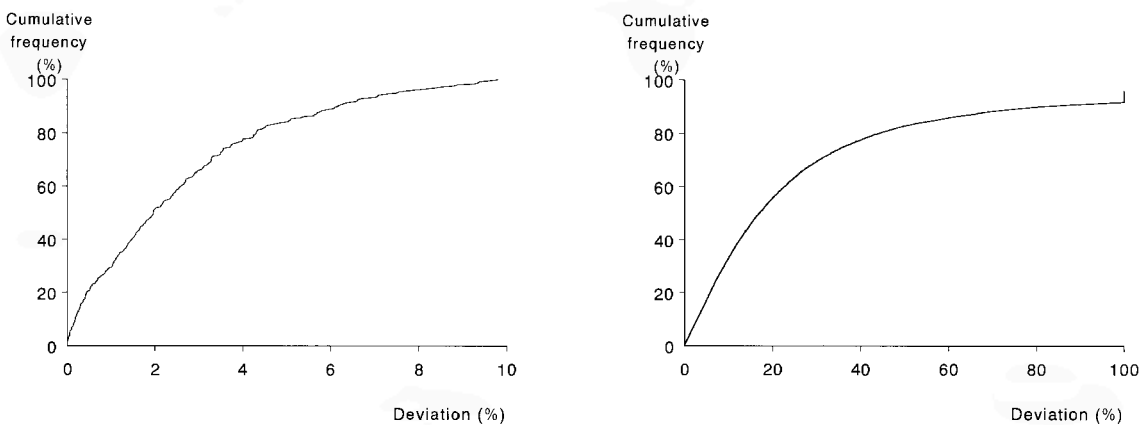


Figure 4.6 The cumulative frequency distribution of the deviation between measured and simulated yearly (left) and biweekly/monthly throughfall fluxes (right) for the 245 sites that were successfully calibrated.

Table 4.12 Measured precipitation and simulated interception and throughfall fluxes ( $\text{mm yr}^{-1}$ ) at 245 sites at which the Gash model was successfully calibrated.

Tree species	Precipitation			Interception			Throughfall		
	5%	50%	95%	5%	50%	95%	5%	50%	95%
Pine	42	64	112	4.5	15	32	35	50	76
Spruce	57	96	172	14.1	29	62	41	66	127
Oak	58	72	131	8.5	18	32	44	57	104
Beech	67	89	163	11.9	24	37	48	64	132
Other	42	112	213	7.7	30	61	34	75	167
All	52	86	162	9.3	24	52	38	60	130

Large differences in rainfall, interception and throughfall fluxes were found within the 245 plots for which fluxes were calculated (Table 4.12). Precipitation ranged between 520 and 1620 mm at 90% of the sites. Pine trees received a lower amount of precipitation compared to Beech and Spruce forest. Interception fluxes ranged from 93 to 520 mm (at 90% of the sites). Lowest values were found in Pine forest that received less rainfall compared to the other tree species. Deciduous forests (Oak and Beech) showed relatively lower interception fluxes (median values of 180 and 240 mm respectively) compared to the coniferous tree species. The observed differences in throughfall fluxes correspond to the trends in deposition and interception fluxes. The lowest fluxes are found under Pine trees (median value 500 mm) and higher values are found for Spruce trees and the deciduous tree species (median values 570-660 mm).

An overview of the average measured and modelled throughfall fluxes on the different sites is given in Fig. 4.7. Average yearly throughfall is high ( $> 1200$  mm) in mountainous areas in western Norway, central and southern Europe and in Ireland. Low values are found in the eastern part of Sweden and central Europe. The pattern of measured and modelled throughfall was highly comparable (Fig. 4.7).

### 4.3.2 Transpiration and leaching fluxes

In this section we report the modelled soil evaporation and transpiration fluxes and the resulting leaching fluxes from the root zone, that were used in the calculation of element budgets (Chapter 5). First the impacts of using interpolated meteorological data on water fluxes is discussed by a comparison of results with measured and interpolated meteorological data (Section 4.3.2.1). Then results obtained with interpolated data are presented, while paying attention to model validation on measurements and literature data (Section 4.3.2.2).

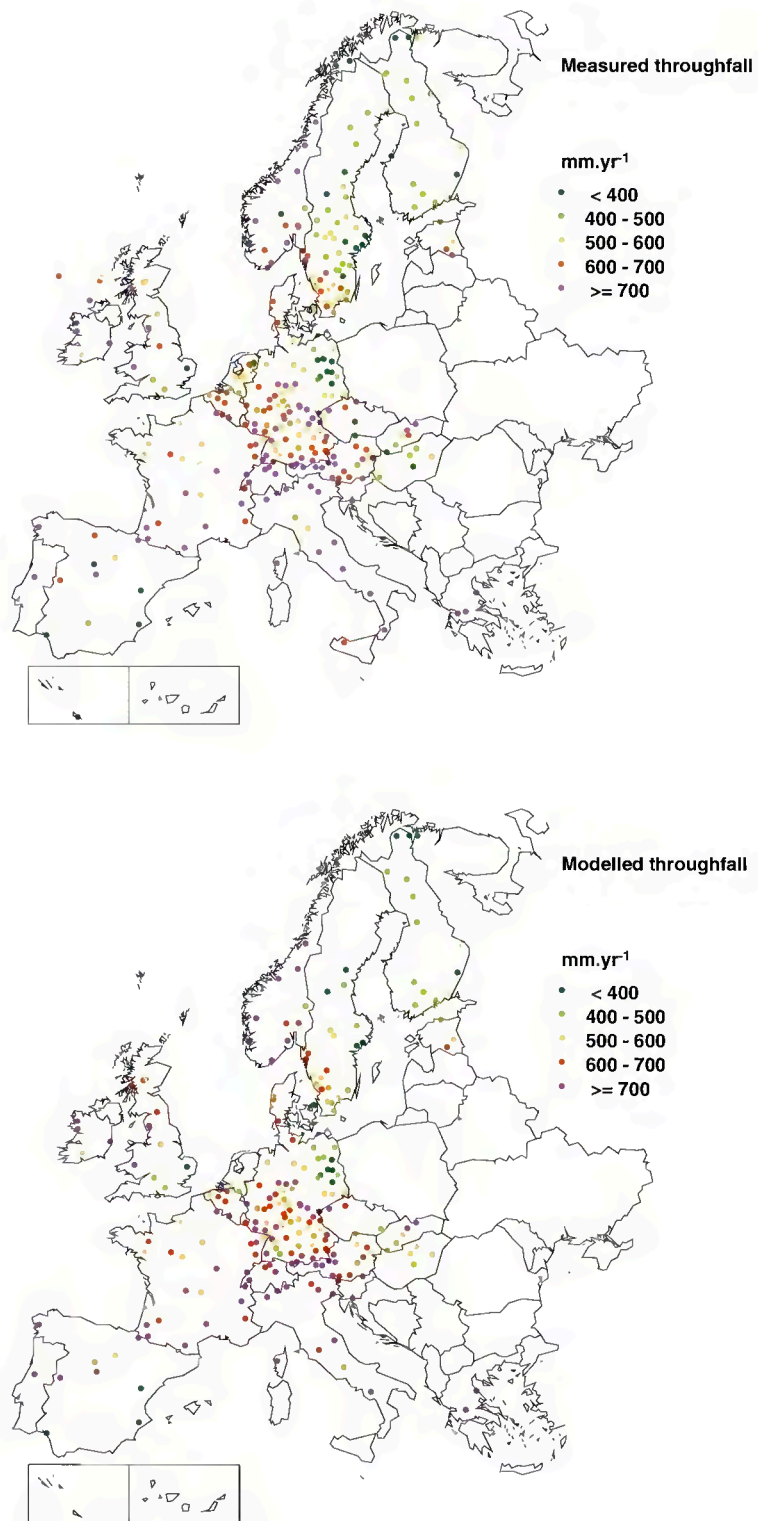


Figure 4.7 Average yearly measured (top) and modelled (bottom) throughfall fluxes ( $\text{mm.yr}^{-1}$ ) at the 245 sites for which both precipitation and throughfall data were available.

### 4.3.2.1 Comparison of results with measured and interpolated meteorological data

#### *Comparison of measured and interpolated meteorological data*

To obtain an indication of the adequacy of the interpolated daily meteorological data, we compared those data with measured data for the plots where a meteorological survey was carried out in the period 1995-1998. Comparisons were made on a daily, monthly and yearly basis. With respect to precipitation, use was made of the data that were scaled to the biweekly or monthly precipitation measured in the deposition survey. Results thus obtained for net radiation, temperature, wind speed and relative humidity are given in Fig. 4.8 for different time intervals. In Fig. 4.9, the daily averages over a period of a year are compared.

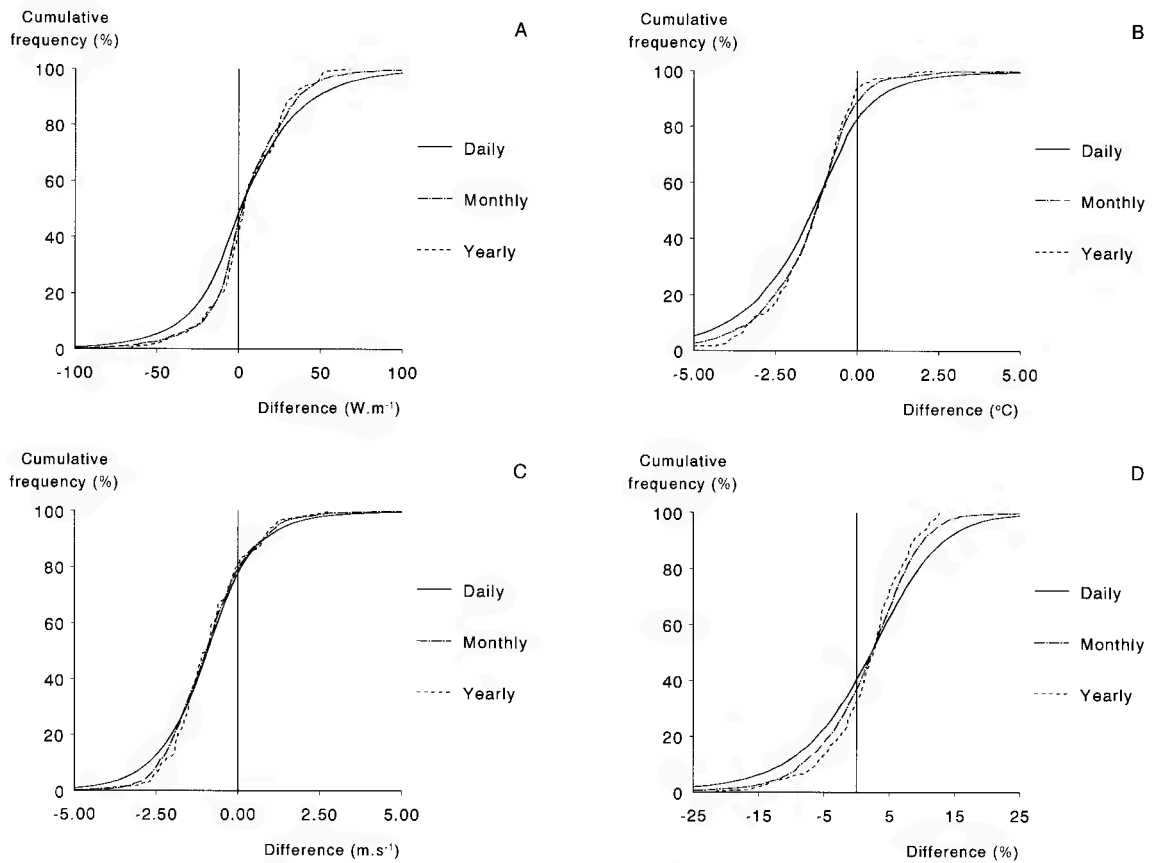


Figure 4.8 Difference, expressed as measured-interpolated, between net radiation (A), temperature (B), wind speed (C) and relative humidity (D) on a daily, monthly and yearly basis using interpolated and measured site-specific daily meteorological data at 77-118 plots in the period 1995-1998.

The results show reasonable agreement for relative humidity. Interpolated data were nearly always within 15% of the measured values and at most plots the relative deviations was less than 10%. Interpolated temperatures were also rather close to measured values. At 65% of the sites, daily temperatures were within 2.5 °C of the measured values. This percentage increased to respectively 72 and 85% when comparing average monthly or yearly values. Although the deviation was relatively limited it has to be concluded that measured temperatures are on average lower than the interpolated values. This phenomenon is clearly shown when comparing average measured and interpolated yearly temperatures for the different plots (Fig. 4.9). The interpolated



temperatures are closely related to the measured temperatures at the plots, but the interpolated temperatures are somewhat higher than the measured values.

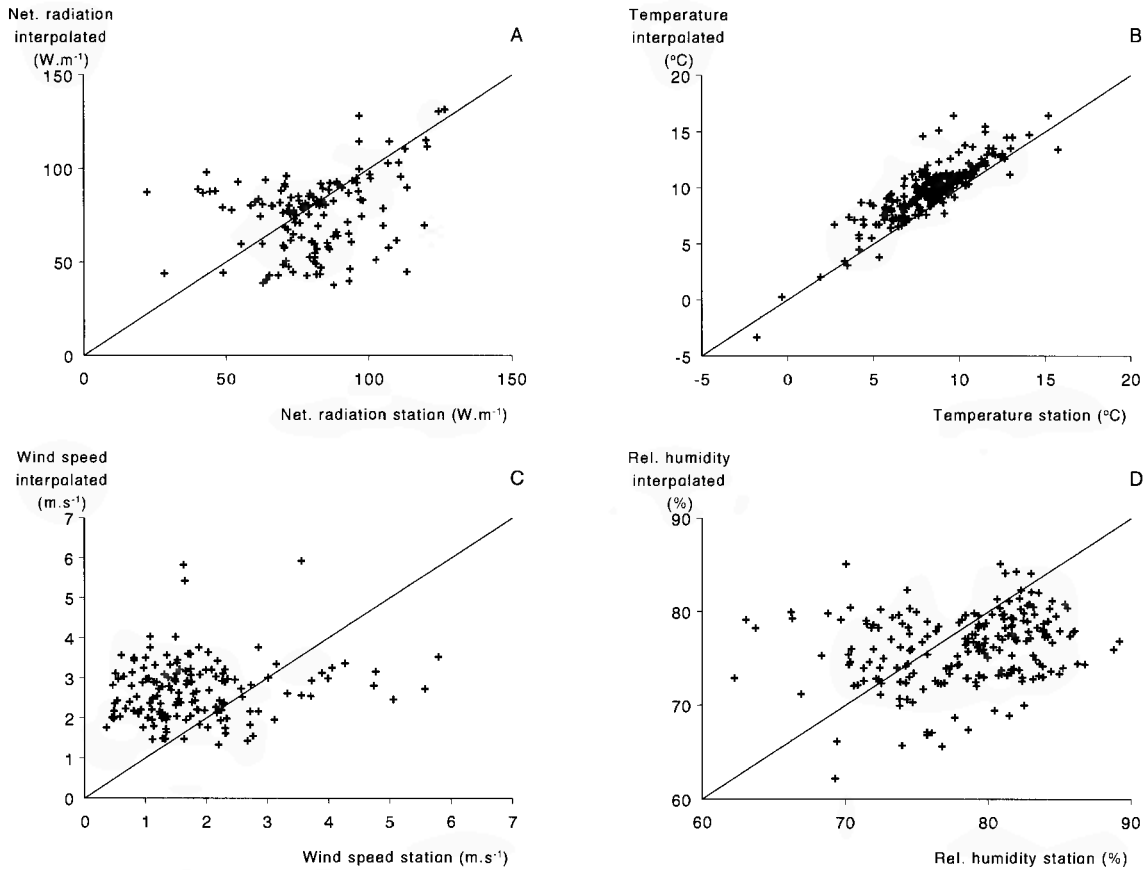


Figure 4.9 Comparison between average daily net radiation (A), temperature (B), wind speed (C) and relative humidity (D) on a yearly basis using interpolated and measured site-specific daily meteorological data at 77-118 plots in the period 1995-1998. The 1:1 line is given in the figure.

Net radiation was both underestimated and overestimated by the interpolation procedure. However, at 60 to 90% of the plots the deviation in interpolated and the measured net radiation was within 50%. The correspondence between interpolated wind speed and measured data was rather poor. At only 40% of the plots, the interpolated data was within 50% of the measured data. At 80% of the plots, the interpolated wind speed was higher than measured data, as indicated by the negative relative difference. This discrepancy between interpolated and measured data is prominent at low wind speeds. Measured yearly wind speed may be as low as 0.5 m.s<sup>-1</sup> whereas interpolated data rarely drop below 2 m.s<sup>-1</sup>. This difference is most probably due to the fact that meteorological stations whose data are used for the interpolation are generally located in large open areas. The wind speeds in or close to the (forested) plots may be expected to be smaller than the wind speed over vast open areas.

A comparison of the precipitation data on a daily, biweekly, monthly and yearly basis is given in Fig. 4.10. In those cases where biweekly data were not available in the deposition survey, they were estimated from the monthly values. Since scaled precipitation data were used, the comparison on a biweekly, monthly and yearly basis gives information on the comparability of measured (daily) precipitation in the meteorological survey and in the bulk deposition inventory. It thus provides information on the consistency of both data sets.

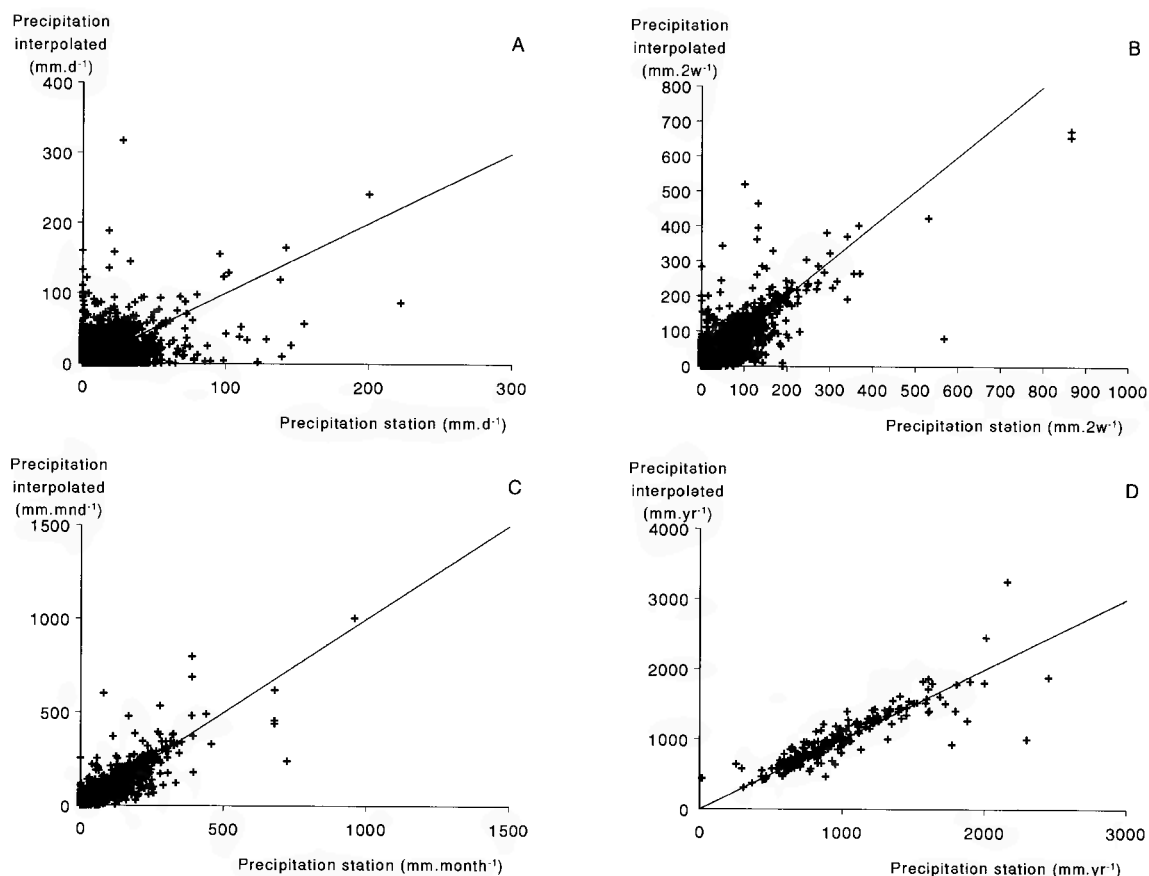


Figure 4.10 Comparison between precipitation on a daily basis (A), biweekly basis (B), monthly basis (C) and a yearly basis (D) using measured daily meteorological data and interpolated data scaled on measured biweekly/monthly precipitation data at 118 plots in the period 1995-1998. The 1:1 line is given in the figure.

Results show that the precipitation data have a high comparability on an annual basis. Results of a regression analysis, assuming a negligible intercept, gave a regression coefficient of 1.003. At high precipitation, the bulk measurements at the plot seem to underestimate the precipitation. There are indications that bulk deposition measurements do underestimate precipitation by approximately 15% due to evaporation from the measuring device in the field, but results given in Fig. 4.10 do not indicate a systematic difference (measurements at Speuld; Erisman, pers. comm.). On a monthly and biweekly basis, the comparison was reasonable, but on a daily basis the agreement was low. For 70% of the plots, the annual deviation was less than 5%; only occasionally, 10% deviation was found. This means that for plots where no meteorological measurements are carried out, biweekly, monthly and yearly precipitation sums can well be derived from the deposition data sets without introducing large errors. For computations based on daily values, the scaled interpolated data are less well suited, but their ultimate effects on e.g. water fluxes, drought stress indices and element fluxes need to be assessed, as presented below.

#### ***Comparison of calculated transpiration and leaching fluxes with measured and interpolated meteorological data***

Throughfall fluxes were calibrated on measured throughfall at the 125 sites where meteorological measurements were carried out at a regular basis, using the same calibration procedure as used for

the 309 sites for which interpolated meteorological data were used. At 78% of the sites (97) throughfall parameters could be optimised using the above mentioned procedure. At the remaining 28 sites, extreme (combinations of) parameter values had to be used to approach the measured throughfall data. Simulated throughfall data for the considered sites were close to measured values for 56 of the 97 calibrated sites. At 41 sites the throughfall was not simulated very accurately resulting in a deviation between simulated and measured total throughfall of more than 10% or leading to a Normalised Root Mean Square Error of more than 10%. Those 41 sites were not used in the comparison of fluxes derived using measured and interpolated data.

The number of sites for which the calibrated Gash model could not accurately simulate the measured biweekly or monthly throughfall was relatively large (42% of the sites) compared to the situation when interpolated data were used (12% of the sites). This is probably due to the fact that interpolated precipitation data were corrected on measured (biweekly/monthly) rainfall data at the plots. Although, the meteorological data were collected quite close to the plot (less than 1.5 km for 90% of the plots; c.f. Fig. 4.2), differences in measured precipitation between the meteorological station and the biweekly/monthly precipitation collected at the plot were sometimes considerable (Fig. 4.10). These discrepancies between measured precipitation at the meteorological station and the actual precipitation at the plot leads to (unrealistic) fluctuations in the throughfall fraction that can not be simulated by the Gash model.

The difference in the median calculated hydrological fluxes obtained with the two data sets was largest for the median transpiration flux and median leaching flux (near 50 mm, Table 4.13). The higher (potential) transpiration when interpolated data were used, leading to lower median leaching fluxes is most probably caused by the higher interpolated wind speed compared to the actual measurements.

*Table 4.13 Simulated median hydrological fluxes (mm yr<sup>-1</sup>) using local meteorological data and interpolated data.*

Flux (mm yr <sup>-1</sup> )	All plots (59)		Budget plots (27)	
	Local data	Interpolated data	Local data	Interpolated data
Precipitation	870	906	849	910
Interception	247	273	255	282
Throughfall	609	622	573	623
Transpiration	337	382	347	399
Soil evaporation	72	72	55	51
Leaching flux	195	148	173	130
Pot. transpiration	381	428	380	439
Pot. soil evaporation	104	98	98	53

An overview of the calculated yearly average hydrological fluxes using the local meteorological data and the interpolated data for the 59 plots is given in Fig. 4.11. For most sites differences in precipitation, throughfall and soil evaporation are quite small. At 80% of the sites transpiration fluxes are overestimated when interpolated data are used. This difference in transpiration flux amounts to more than 100 mm at some sites. However, at 50% of the sites the difference in simulated transpiration fluxes is less than 37 mm. The observed differences in simulated leaching fluxes are comparable to the differences in transpiration fluxes. At 85% of the sites simulated leaching fluxes are lower when interpolated data are used. The difference is often quite small (50% of the sites have a difference of less than 27 mm). At some sites the difference in simulated fluxes may increase to nearly 190 mm. However, these are generally sites for which the local meteorological measurements are limited to a 1 year period.

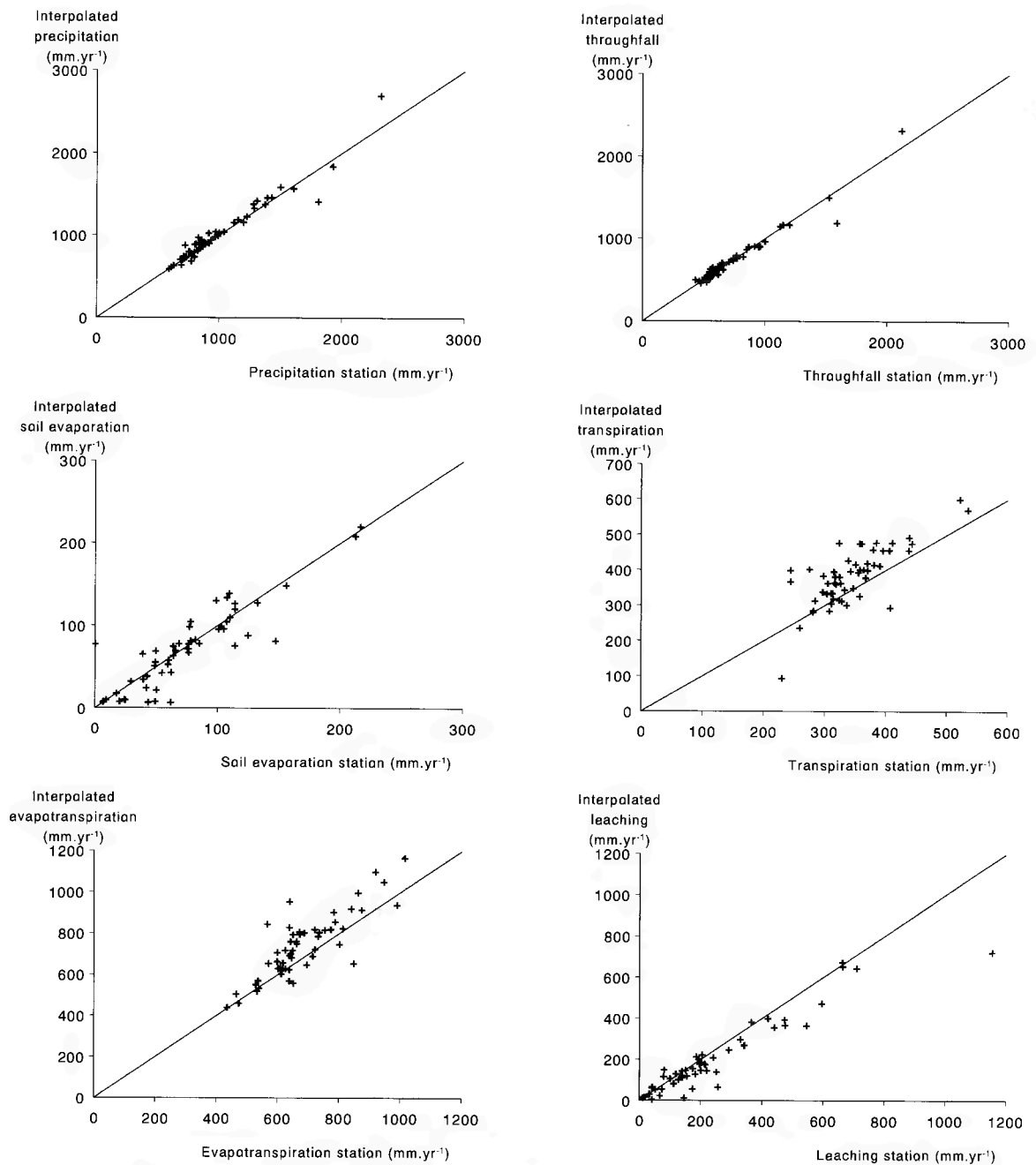


Figure 4.11 Comparison between average hydrological fluxes simulated when using local meteorological data and interpolated data.

This strong effect of uncertainty in meteorological data on the calculated transpiration and leaching fluxes for individual years is clearly reflected in the differences in calculated hydrological fluxes for individual plot-year combinations (Fig. 4.12). This graph shows that differences in rainfall and simulated throughfall, transpiration and leaching fluxes can be large for certain plot year combinations, although differences are quite limited (less than 50 mm) for more than 50% of the sites.

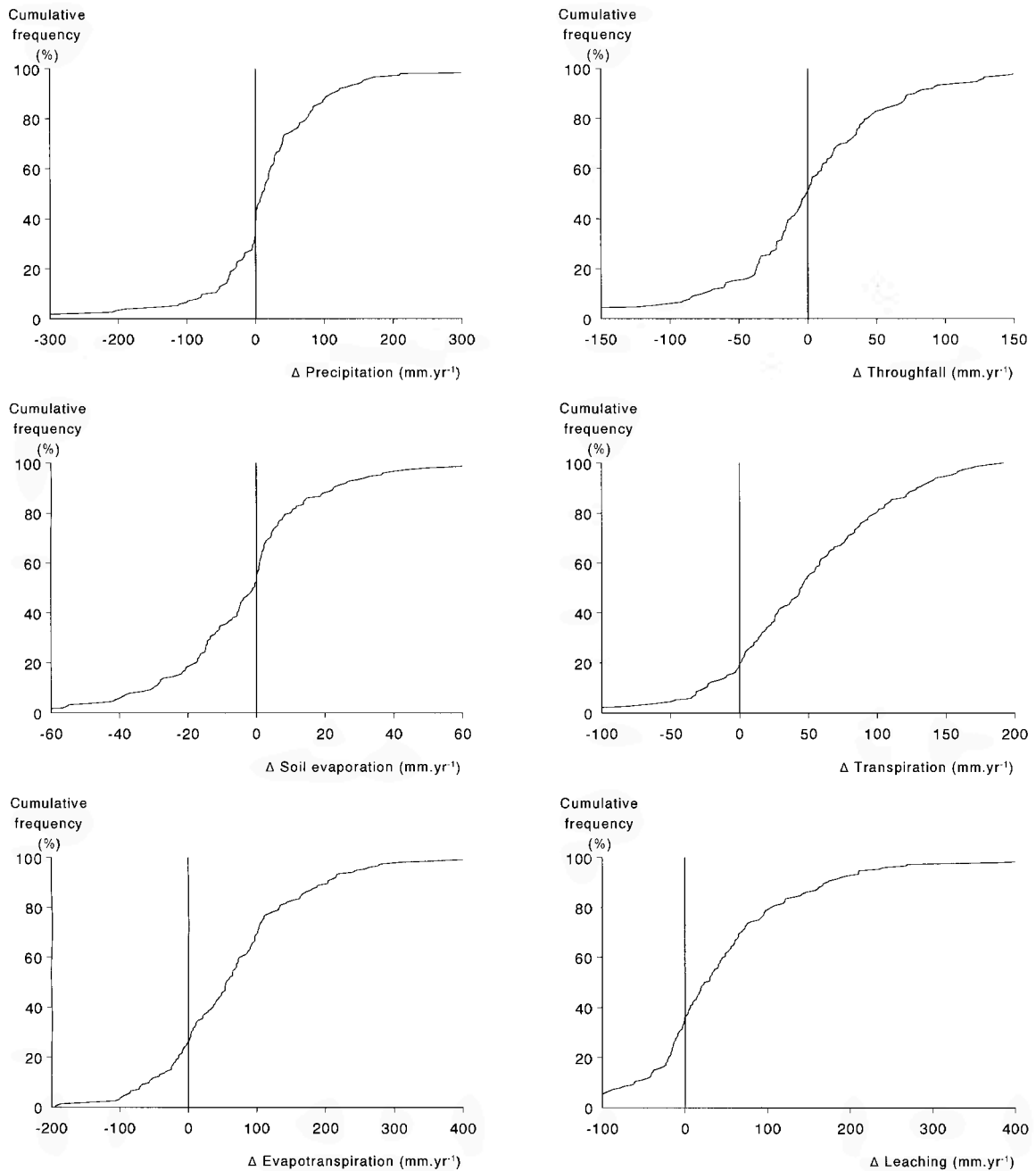


Figure 4.12 Cumulative frequency distribution of the difference in simulated water fluxes for individual plot-year combinations when using interpolated meteorological data and local measured data (interpolated - measured).

### 4.3.2.2 Results with interpolated meteorological data

#### *Model validation*

To gain confidence in the model and its parameterisation the model has been validated on two sites in the Netherlands and Germany for which extensive hydrological measurements were available (Van der Salm, in prep). An application to other sites (in Denmark and France) is planned. Preliminary results of the application of the hydrological model on a Douglas fir stand in Speuld and a Norway spruce stand in Solling indicate that the model was able to simulate

measured water contents and/or pressure heads quite well when site specific data are used (Fig. 4.13). The use of generic data, as used in the application for the Intensive Monitoring sites, led to a larger deviation between measured and simulated data (Fig. 4.13). The impact on simulated long-term leaching fluxes varied between 7% for Solling (13 year) and 25% for Speuld for which only a five year period was considered.

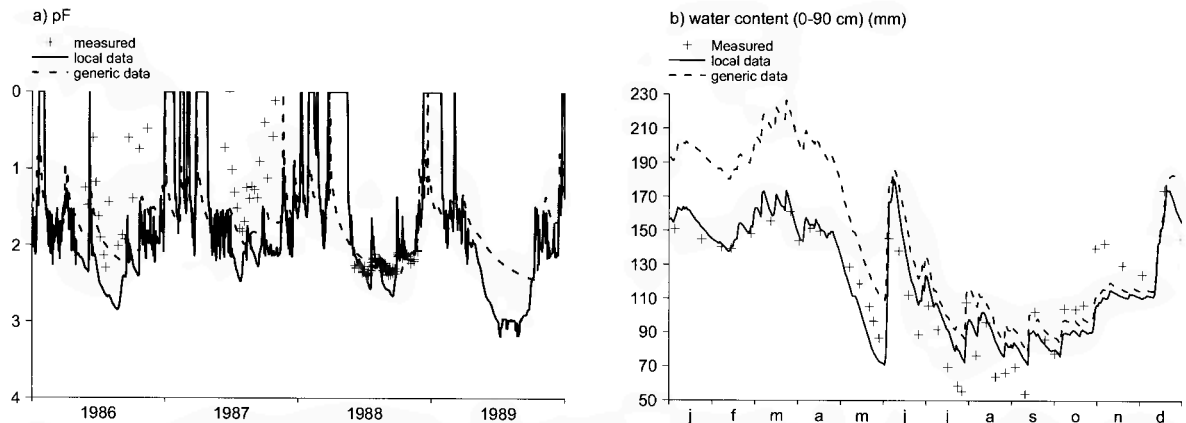


Figure 4.13 Measured and simulated pressure heads at 90 cm depth at Solling (left) and measured and simulated water contents in the upper 90 cm of the soil at Speuld (right).

### Model evaluation

A possible check of the modelled results can be based on the calculation of chloride budgets for the examined sites. Long-term chloride budgets may be expected to be close to zero. For the Intensive Monitoring sites the measurement period is too short (average 3 year) for such an evaluation. Nevertheless, the calculated Cl budgets do give some indication whether the order of magnitude of the calculated leaching fluxes is feasible (cf. Ch. 5.3.3).

Another possibility for a general evaluation of the plausibility of the model results is to compare simulated leaching fluxes with reported fluxes for Intensive Monitoring sites. Literature data were found for 34 sites mainly located in Northwestern Europe (Beier, 1998; Bouten and Jansson, 1995; Bouten et al., 1992; Boyle et al., 2000; Bredemeier et al., 1998; Dolman, 1988; De Visser and de Vries, 1989; Gärdenäs and Jansson, 1995; Granier et al., 2000; Grote and Suckow, 1998; Harding et al., 1992; Hendriks et al., 1990; Herbst et al., 1999; Jaeger and Kessler, 1997; Ladekarl, 1998; Musters, 1998; Tiktak and Bouten, 1990; Tiktak and Bouten, 1994; Van Grinsven et al., 1987). These data indicate that simulated leaching fluxes are within the range of literature data (Fig. 4.14), although a considerable part of the measurements is close to the upper site of the range of simulated fluxes. This bias can be partly explained by the fact that the literature data originate from sites in Northwestern Europe, where leaching fluxes are generally higher compared to sites in southern and central Europe. Another reason for this bias might be that leaching fluxes are slightly underestimated when interpolated meteorological data are used (c.f. section 4.3.2.1). To obtain more information on the accuracy of simulated fluxes, a closer comparison of the simulated leaching and transpiration fluxes at a site scale is necessary.

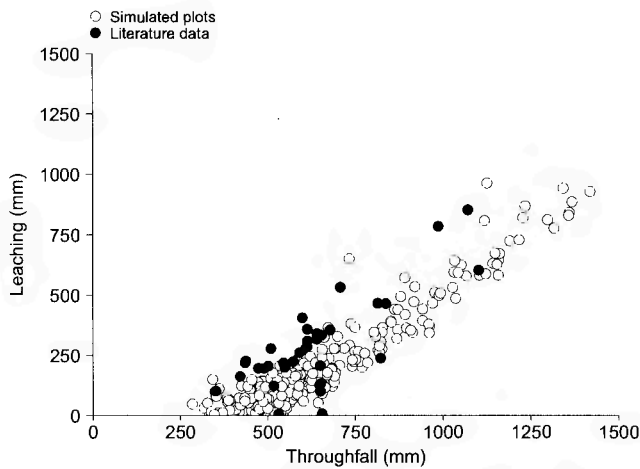


Figure 4.14 Simulated and measured yearly leaching fluxes as a function of the yearly throughfall.

### Model results

Transpiration and leaching fluxes were calculated for all 245 sites for which throughfall fluxes could be successfully simulated (see also Table 4.9). Chemical budgets could only be calculated for sites for which data on both soil solution and deposition were available. Results for this subset of 121 plots are also presented separately in this paragraph. This selection of plots is also used to calculate the element budgets described in Chapter 5.

Evapotranspiration and leaching fluxes varied considerably for the investigated sites (Fig. 4.15 and 4.16). The strongest variation in fluxes was found for the interception evaporation which ranged from less than 100 mm to more than 500 mm, depending on the amount of rainfall and the potential evapotranspiration. Ranges in transpiration and soil evaporation fluxes are much smaller. Transpiration varied between approximately 200 to 400 mm and soil evaporation fluxes ranged between 10 and 100 mm. Leaching fluxes ranged from 0 to more than 1500 mm yr<sup>-1</sup>. The budget plots showed a much smaller variation in leaching fluxes compared to the total set of plots for which hydrological fluxes were calculated. The median values for both datasets were comparable (approx. 150 mm) whereas the 90 percentile of the leaching fluxes was approximately 500 mm for the budget plots and 900 mm for all hydrological plots.

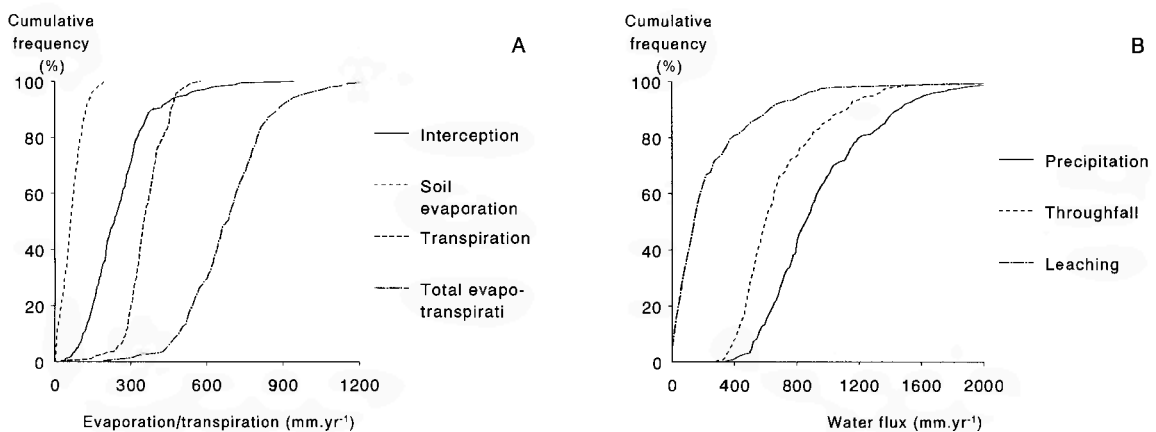


Figure 4.15 Cumulative frequency distributions of the evaporation and transpiration (A) and (B) at the 245 sites for which hydrological fluxes were calculated.

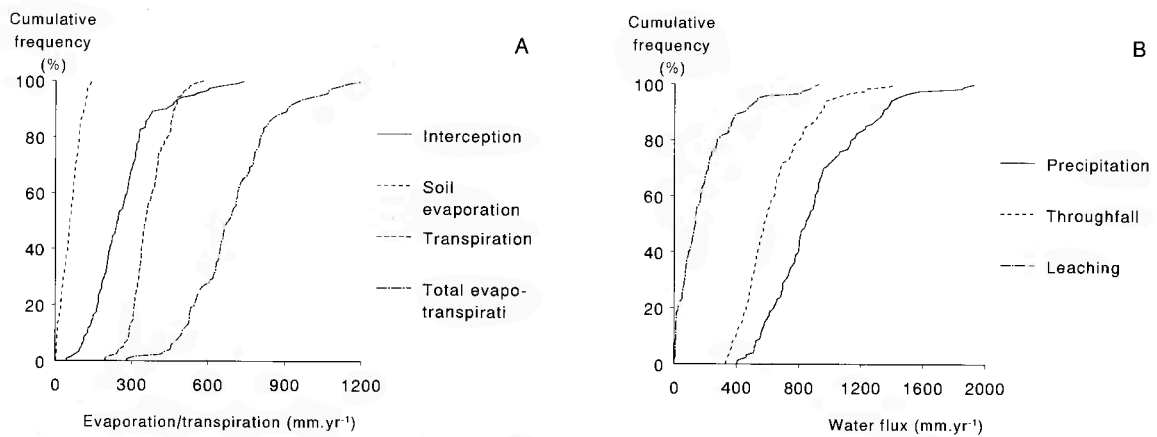


Figure 4.16 Cumulative frequency distributions of the evaporation and transpiration (A) and (B) at the 121 sites for which chemical budgets could be calculated.

An overview of the median simulated hydrological fluxes for the different tree species is given in Table 4.14. Differences between fluxes calculated for the 245 sites and the subset of 121 sites, used to calculate chemical budgets, are small. Median transpiration fluxes ranged from 314 mm.yr<sup>-1</sup> at the Pine stands to 385 mm.yr<sup>-1</sup> for Spruce. The calculated hydrological fluxes are generally in the range of values reported for various European forests. Median transpiration fluxes are close to the mean value of 333 mm.yr<sup>-1</sup> of Roberts (1983), who found that transpiration fluxes for European forest are in a very narrow range due to feedback mechanisms with soil and atmosphere. Soil evaporation fluxes were highest for the Oak stands, which are generally less dense. The median leaching fluxes ranged from 79 mm on Pine stands compared to 205 mm for Spruce. The leaching fluxes for Pine are quite low because the median precipitation is somewhat lower on Pine compared to Spruce and Beech.

Table 4.14 Simulated median hydrological fluxes (mm) for the 245 sites that were successfully calibrated and for the 121 sites for which chemical budgets were calculated (between brackets).

Tree species	Number of sites	Number of years	Calculated fluxes (mm)				
			P	E <sub>i</sub>	E <sub>t</sub>	E <sub>s</sub>	L
Pine	51 (29)	3 (3)	642 (602)	152 (152)	314 (328)	55 (55)	79 (17)
Spruce	98 (51)	3 (3)	963 (943)	290 (303)	385 (365)	32 (27)	205 (192)
Oak	24 (15)	3 (4)	725 (777)	177 (202)	338 (340)	105 (99)	123 (169)
Beech	45 (20)	3 (3)	891 (876)	241 (241)	356 (358)	94 (82)	138 (135)
Others	27 (6)	3 (6)	1122 (1233)	300 (437)	389 (483)	81 (9)	278 (218)
All	245 (121)	3 (3)	860 (846)	240 (247)	356 (356)	64 (57)	152 (136)

The differences in median precipitation fluxes between the tree species are reflected in the calculated average yearly transpiration reduction (Table 4.15). The highest transpiration reduction was found in Pine trees (0.84 at the plots for which budgets could be calculated), whereas transpiration reduction for spruce and deciduous trees was much smaller (0.89-0.93).

Table 4.15 Mean yearly transpiration reduction at Intensive Monitoring sites.

Tree species	Precipitation (mm)		Transpiration reduction (E <sub>t</sub> /E <sub>i</sub> )	
	All	budgets	all	budgets
Pine	642	602	0.86	0.84
Spruce	963	943	0.93	0.93
Oak	725	777	0.92	0.92
Beech	891	876	0.91	0.93
Others	1122	1233	0.83	0.89
All	860	846	0.91	0.91



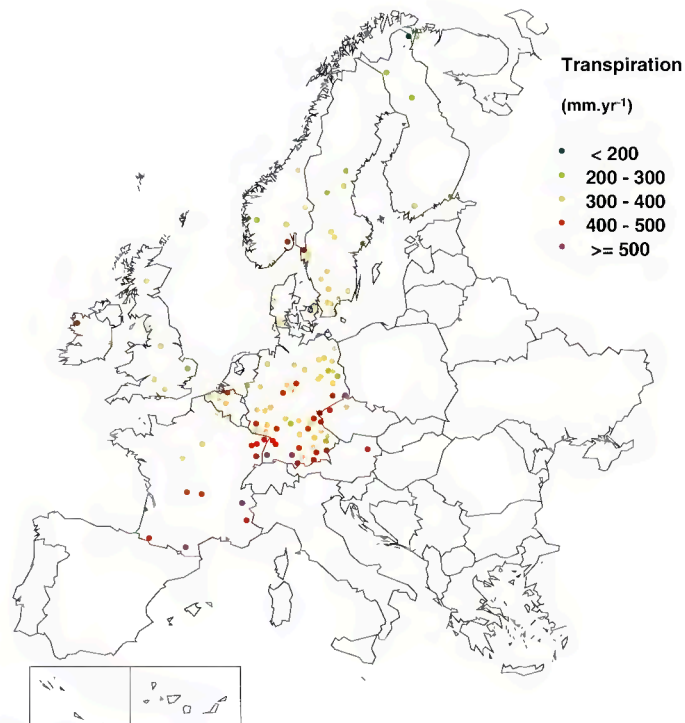
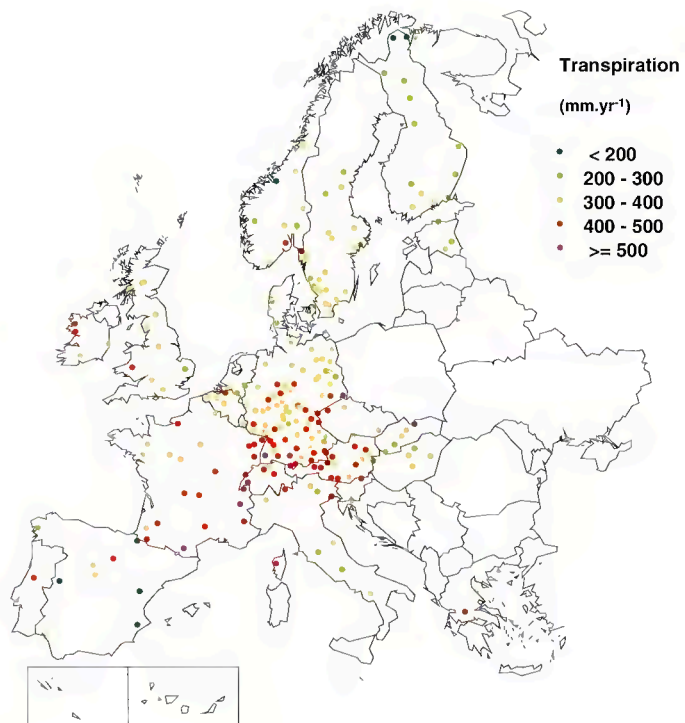


Figure 4.17 Average annual transpiration fluxes (mm.yr<sup>-1</sup>) for the 245 monitoring plots for which hydrological budgets have been calculated (top) and the 121 monitoring plots for which chemical budgets have been calculated (bottom).

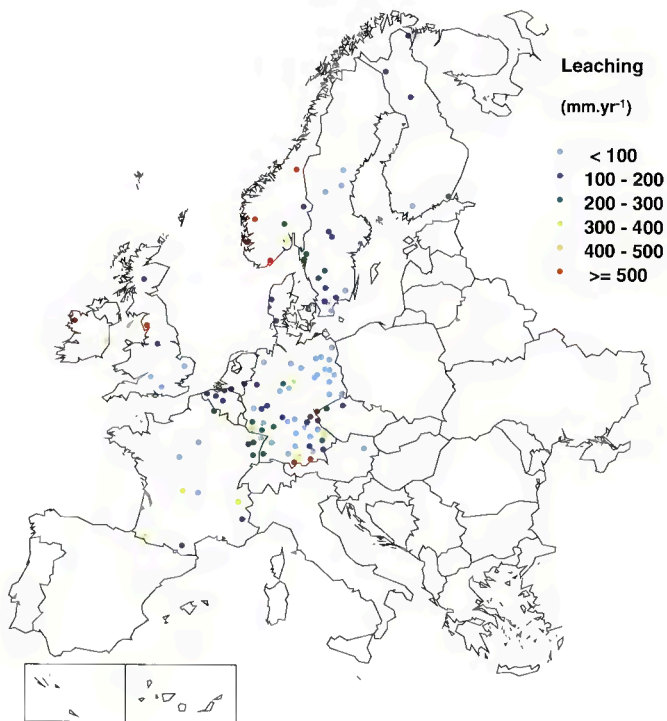
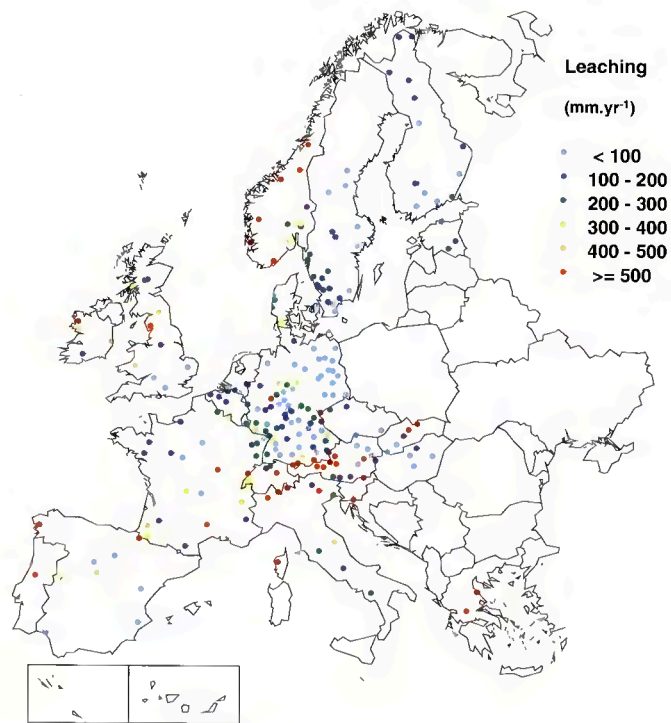


Figure 4.18 Average leaching fluxes ( $\text{mm.yr}^{-1}$ ) for the 245 monitoring plots for which hydrological budgets have been calculated (top) and the 121 monitoring plots for which chemical budgets have been calculated (bottom).

Transpiration fluxes are generally highest (> 450 mm) in central Europe where both rainfall and radiation are relatively high (Fig. 4.17). At most sites in southern Europe, transpiration is somewhat lower due to the limited rainfall. Exceptions are the sites in mountainous areas, which receive higher amounts of rainfall. In northern Europe transpiration fluxes are lower than in central Europe due to the decrease in radiation with increasing latitude and the relatively low precipitation in part of Sweden and Finland.

The map showing the leaching fluxes for all plots and for the plots for which budgets could be assessed (Fig. 4.18) closely resembles the patterns in throughfall maps (Fig. 4.7). High leaching fluxes are found in Northwestern Scandinavia, Britain and Ireland and on some plots in mountainous areas in central and southern Europe. Remarkable are the low leaching fluxes (< 100 mm) in parts of Sweden and Northeastern Germany that are caused by the combination of low rainfall and relatively high transpiration fluxes (coniferous forests).

#### 4.4 Conclusions

A comparison of measured and interpolated meteorological data showed good agreement for relative humidity, reasonable agreement for temperature and net radiation and poor agreement for wind speed. The effect of using interpolated data instead of on-site measured data led to an overestimation of the simulated transpiration fluxes and an underestimation of the leaching fluxes. However, on most sites the impact was relatively limited. More specifically, the following conclusions can be drawn with respect to the calculation of the hydrological fluxes:

- The simulated throughfall could be calibrated on measured throughfall data such that the simulated yearly values were within 5% of the measurements at 85% of the monitoring sites. Biweekly or monthly values were also quite well simulated as indicated by the relatively low normalised root mean square error with a mean value of 0.29.
- At 80% of the sites average transpiration fluxes were overestimated when interpolated data are used. The median difference in simulated transpiration fluxes was 45 mm. The observed differences in simulated leaching fluxes were comparable to the differences in transpiration fluxes. At 85% of the sites simulated leaching fluxes were lower when interpolated data are used. The median difference was 47 mm but at some sites the difference in average simulated fluxes increased to nearly 190 mm. However, this were generally sites for which the local meteorological measurements were limited to a 1 year period.
- Mean yearly interception evaporation ranged from approximately 160 mm for Pine and Oak to approximately 250 mm for Beech and 300 mm for Spruce. The interception fluxes for Pine and Oak were relatively low due to the relatively low rainfall on those tree species. Interception fractions increased going from Oak (0.22) < Pine (0.24) < Beech (0.27) < Spruce (0.30), reflecting the increasing interception capacity of those tree species.
- Simulated transpiration fluxes and leaching fluxes could not be validated on data for any of the individual sites. However, detailed studies on two sites in Germany and the Netherlands indicated that the model was able to simulate changes in soil water contents and thus in the transpiration and leaching fluxes quite well. The simulated (evapo)transpiration fluxes for the 245 sites were also in the range of data reported in the literature. However, leaching fluxes tend to be a bit low compared to literature data, which may be partly explained by the fact that measurements are biased to northwestern Europe.
- Median transpiration fluxes were rather constant among the tree species and ranged from 325 mm.yr<sup>-1</sup> for Pine to 385 mm.yr<sup>-1</sup> for Spruce stands. This is consistent with literature data, which indicate that transpiration fluxes for European forest are in a very narrow range around

- 335 mm.yr<sup>-1</sup> due to feedback mechanisms with soil and atmosphere. The range in the sum of soil evaporation and transpiration is also narrow and median values range from approximately 400-450 mm.yr<sup>-1</sup> for the various tree species. Leaching fluxes mainly reflected the difference in precipitation on tree species. Median values increased going from 81 mm under Pine stands to 236 mm under Spruce stands. The plots with the lowest leaching fluxes are found in area with relatively low precipitation such as northeastern Germany, parts of Sweden and Finland and locally in southern Europe.
- The limited amount of available water in the examined Pine stands is reflected in the calculated mean transpiration reduction which is highest for Pine, with a median value of 15%. For the other tree species the median value was 10%. In the future reports, more attention will be given to various drought stress parameters, including relative transpiration, that can be used in subsequent analyses relating drought stress to forest growth and forest vitality.

## 5 Assessment of element fluxes through the forest ecosystem

### 5.1 Introduction

A comparison of element inputs from the atmosphere and element outputs leaching from the bottom of the root zone give insight in the fate (accumulation or release) of sulphur, nitrogen, base cations and aluminium in the ecosystem. As such, it is of crucial importance to assess the present and future impacts of atmospheric deposition on the element cycle and nutrient availability. More specifically, budgets of  $\text{SO}_4$ ,  $\text{NO}_3$  and  $\text{NH}_4$  give insight in (i) the actual rate of acidification due to anthropogenic sources and (ii) the potential rate of acidification by immobilisation of S and N (e.g. Van Breemen et al., 1984; De Vries et al., 1995). Results about the input and output of Al and base cations (BC) give information about the mechanisms buffering the acid input (e.g. Mulder and Stein, 1994; Wesselink, 1995; De Vries et al., 1995). In general, the ratio of Al to BC release is a crucial aspect with respect to soil mediated effects of acid inputs (e.g. Cronan et al., 1989; Sverdrup and Warfvinge, 1993). These insights can therefore be used to derive critical deposition levels for forest soils (ecosystems). Comparison with available data on present loads, leads to insight in the stress of air pollution on the chemical ecosystem condition (e.g. De Vries et al., 2000b).

Element budgets have already been carried at Intensive Monitoring plots by several countries including Greece, (FIMCI, 1998), Ireland (FIMCI, 1998; Boyle, 2000; Farrel et al., 2001), Belgium (FIMCI, 1999, 2000), Germany (FIMCI, 1999; Sprangenberg, 1997; Wetzel, 1998; Block et al., 2000) and Slovakia (FIMCI, 2000). Furthermore, there are several literature compilations of element budgets, focusing on the behaviour of nitrogen (e.g. Dise et al., 1998a, b; Gundersen et al., 1998a, b). A European wide assessment of element budgets, using all available data on deposition, meteorology and soil solution chemistry at the Intensive Monitoring plots has, however, not yet been carried out. This chapter aims to fill this gap.

This chapter first describes the methods used to calculate element input by atmospheric deposition, element leaching and element retention (Section 5.2) and then gives results on the range and geographic variation of all these fluxes and their relation with environmental factors (section 5.3). In Section 5.2, we describe the methods to assess site-specific total atmospheric deposition and total element output for selected Intensive Monitoring plots based on measurements of both throughfall, bulk deposition and soil solution chemistry. Input fluxes were derived from fortnightly or monthly measurements of the chemical composition of bulk deposition and throughfall water, multiplied by the water fluxes while correcting for canopy uptake. A canopy exchange model, developed by Ulrich (1983) and extended by Draaijers and Erisman (1995), was used as a basis and further improved. The resulting canopy exchange was related to available data on bulk deposition, meteorological parameters and foliar chemistry performing multiple regression analysis. Element outputs from the forest ecosystem were derived at intensively monitored plots by multiplying fortnightly or monthly measurements of the soil solution composition at the bottom of the rootzone with simulated unsaturated soil water fluxes. Element retention or release was assessed from the difference between the leaching from the bottom of the root zone and the element input from the atmosphere.

Results on the derived input, output and retention or release of major elements (chloride, sodium, sulphate, nitrogen, base cations and aluminium) are described in Section 5.3. Apart from results

on ranges and geographic variation, this section focuses on relationships between canopy uptake, leaching or retention/ release of elements and readily available environmental variables using a statistical technique. Examples of environmental variables are stand and site characteristics, soil and foliar chemistry, precipitation and bulk deposition. Element leaching for example, is not only influenced by the atmospheric input, but also by the chemical interactions in the soil, which in turn are influenced by stand and site characteristics and soil chemical parameters. Nitrogen retention may e.g. be determined by the soil C/N ratio (e.g. Dise et al., 1998a, b; Gundersen et al., 1998a), whereas the pH and base saturation most likely influence the release of Al and base cations (BC). Reliable relationships can be used for upscaling the results to a larger scale. Conclusions are given in Section 5.4.

## 5.2 Methods

### 5.2.1 Locations

Element budgets were calculated for sites where precipitation and throughfall fluxes and element concentrations in soil solution have been measured up to 1998 for a period of more than 300 days. The criterion of 300 days was included because yearly average fluxes may differ substantially from those observed during a short measurement period. Furthermore sites were selected where:

- Soil solution is sampled with tension lysimeters (see section 5.2.2);
- Reliable water fluxes could be calculated (i.e. the Gash model could be successfully calibrated);
- The soil type does not indicate the presence of ground water in the soil profile (since the hydrological simulations were made assuming free drainage).

All these criteria were matched at 121 of the 228 sites with soil solution data (Table 5.1), located in Belgium, France, Denmark, Germany, Great Britain, Ireland, Norway, Sweden, Finland and Austria (Fig. 5.1). The number of selected sites for which budgets could be calculated based on annual input and output fluxes increased from 16 plots in 1995 to 85 plots in 1996, 113 in 1997 and 121 in 1998.

Table 5.1 Number of monitoring sites for which sufficiently long records of precipitation, throughfall and soil solution concentration measurements were available to calculate annual element budgets up to 1998.

Quality aspects	Number of sites	
	Bulk deposition <sup>1)</sup> and throughfall	Soil solution chemistry
Total number	309	228
- Input available	309	204
- Calibrated gash parameters	245	138
- Period of at least 300 days	-	164
- Acceptable techniques	-	128
- well drained soils	-	121
Available for budgets	121	121

<sup>1)</sup> Equal to precipitation

The element inputs were not only assessed at the 121 plots for which budgets could be calculated but also at all 309 plots with bulk and throughfall data. Fig. 5.1 shows the geographic variation of (i) the 121 plots for which budgets could be calculated, (ii) the 83 plots (204-121; see Table 5.1) for which data for bulk deposition, throughfall and soil solution chemistry were available that did not pass the various quality checks and (iii) the remaining 105 plots (309-204; see Table 5.1) with only data on bulk deposition and throughfall.

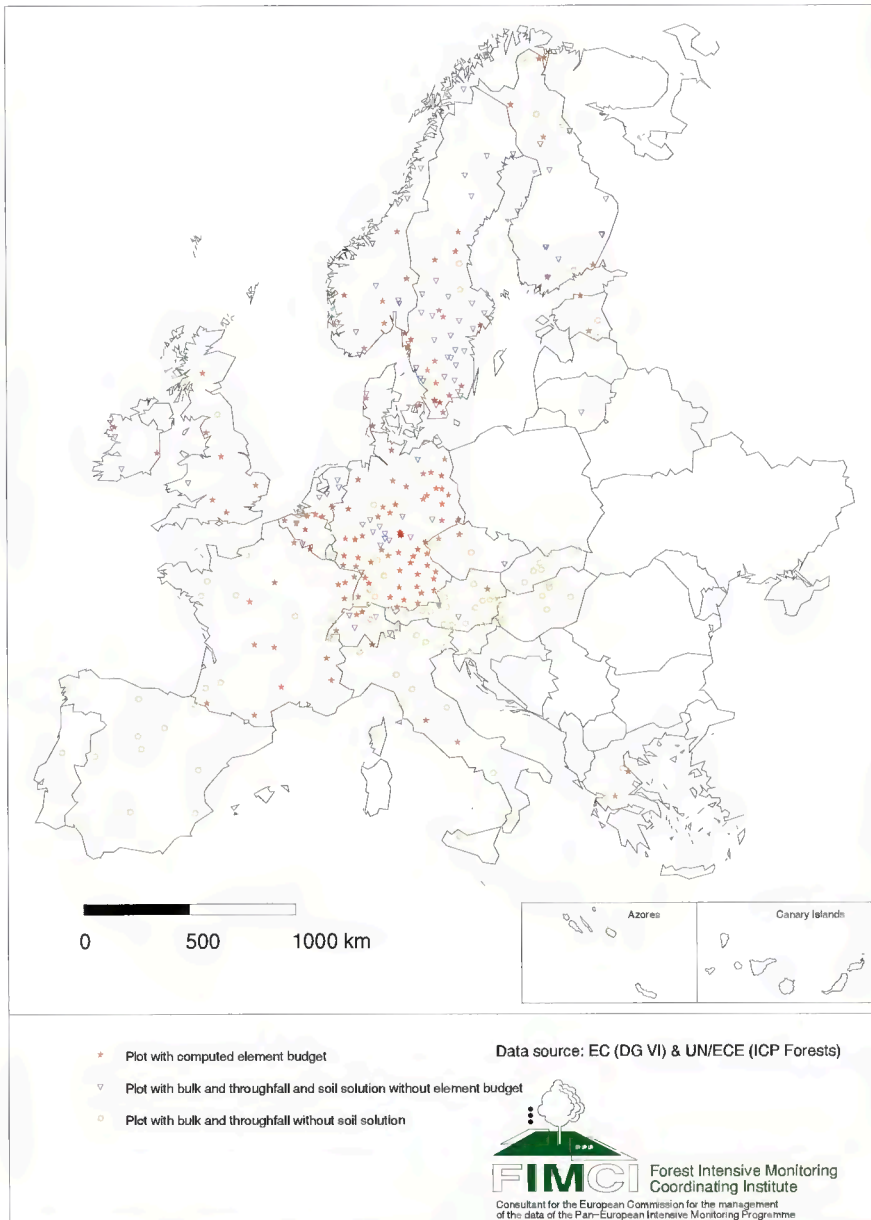


Figure 5.1 Geographical distribution of Intensive Monitoring plots for which element inputs and outputs have been calculated up to 1998.

### 5.2.2 Data assessment methods and data comparability

The relevant EC Regulations and manual of ICP Forests give standard methods for the sampling and analysis of bulk precipitation, throughfall, stemflow and soil solution. Nevertheless, the countries involved in the program sometimes use their own specific sampling equipment, sampling strategy, sample handling and analytical procedures and interpretation methods. This includes methods to interpolate missing values and to calculate annual fluxes. In this context field and laboratory intercomparisons are crucial and enable researchers to identify the most accurate analytical methods, sampling equipment, strategy and handling, thus leading to further harmonisation in methods. Results of such a joint field campaign carried out with respect to

deposition monitoring, comparing impacts of different sampler spacing and sampler numbers on throughfall in the Speulder forest and on bulk deposition at the Schagerbrug grassland in the Netherlands, are given in Annex 2.

Through the Data Accompanying Report Questionnaires (DAR-Q's) the participating countries submitted information on the applied methods for most of the plots. Here we report the most relevant aspects with respect to data assessment methods related to atmospheric deposition and soil solution. The presentation of applied methods concerns only those plots for which both atmospheric deposition and soil solution data were available for at least one year. This included 228 plots (Table 5.1). More detailed information on sampling techniques, the sampling material, sampling numbers, sampling frequencies and conservation and analyses is given in Annex 3.

### ***Deposition data***

Data assessment information for deposition monitoring indicates that throughfall and precipitation measurements were obtained mainly by funnels and sometimes gutters (in case of throughfall). The number of funnels was always above 10, being the recommendation in literature (Lövblad, 1994). At 18 of the 228 plots only 3 gutters were used, but those devices had a relatively large collecting area. Nevertheless, the required total collecting area of at least 3140 cm<sup>2</sup> according to the recommendation in the Manual of ICP Forests was not met at 108 sites (47%). From the deposition intercomparison project executed at the Speulder forest (see Annex 2) it was concluded that the accuracy of throughfall measurements will be significantly reduced when the total collecting area is smaller than 2000 cm<sup>2</sup> (Draaijers et al., 2001). Even in this case, 96 plots (42%) did not meet this requirement. We decided to still use the results from those plots but evaluated the results with care in view of plausibility.

The sampling frequencies for both bulk deposition and throughfall measurements were mostly weekly (47%) followed by either biweekly (27%) or 4-weekly periods (26%). Information on the conservation and analyses of samples is given in Annex 3.

### ***Soil solution data***

In the Intensive Forests Monitoring Programme, four different sampling techniques are used for soil solution sampling, i.e. by:

- Placing soil solution collectors in the field: suction cups and zero tension lysimeters.
- Taking soil samples and extraction of soil solution: centrifugation or extraction methods.

In order to ensure that results are as comparable as possible, assessment of the input-output budgets was limited to the results obtained with tension lysimetry, being used at 210 of the 228 plots. In selecting those plots we focused on the sampler within a depth of 40 –100cm, that was used to calculate the outflow from the forested ecosystem. We also skipped 7 plots in which perched water or ground water occurred within the soil-solution sampling layer as this causes problems in (i) assessing the effects of acidifying deposition on soil water quality (ii) deriving water fluxes. Since no information was available about the occurrence of ground water or perched water, we have only omitted results obtained from soils with gleyic features that indicate the sporadic presence of ground water within the root zone.

The equipment used in sampling soil solution (lysimeters, tubing, collection bottles etc.) should be made of materials that neither contaminate the samples nor decrease (e.g. through sorption) the concentrations of ions in the samples. A review of the materials used (Derome et al., 2001)



showed that ions whose concentrations may be biased due to the sampling equipment are not used in our budget studies (see also Annex 3). Consequently, we used the data from all the types of lysimeter in our calculations. As with deposition data, the recommended number of samplers for monitoring soil solution is at least 10 and this requirement was not met at 90 plots. At 21 plots, only three lysimeters were used. As with deposition, we have not removed those plots from the assessment of input-output budgets but used the information in evaluating the plausibility of the results.

## 5.2.3 Data quality assurance

### 5.2.3.1 Quality checks

Procedures for quality assurance and quality control (QA/QC) included a check on all individual measurements with respect to:

- the balance between cations and anions
- the difference between measured and calculated electric conductivity
- the ratio between Na and Cl concentrations

This is the standard procedure introduced by FIMCI in the Technical reports and will be summarised here. However in order to increase the number of data used in the evaluation, a correction procedure was developed which is described in Section 5.2.3.2.

#### *Ionic balance*

On an equivalent basis, the sum of all major cations should equal the sum of all major anions. The percentage difference was therefore calculated according to:

$$PD = 100 * \frac{(\sum \text{cat} - \sum \text{an})}{0.5 * (\sum \text{cat} + \sum \text{an})} \quad (5.1)$$

$$\sum \text{cat} = [\text{Ca}^{2+}] + [\text{Mg}^{2+}] + [\text{Na}^+] + [\text{K}^+] + [\text{H}^+] + [\text{NH}_4^+] \quad (5.2)$$

$$\sum \text{an} = \text{Alk} + [\text{SO}_4^{2-}] + [\text{NO}_3^-] + [\text{Cl}^-] + [\text{RCOO}^-] \quad (5.3)$$

where:

PD = percentage difference (%)

Alk = alkalinity ( $\text{mmol}_c \cdot \text{m}^{-3}$ )

In principle, H should be neglected when alkalinity is positive. The basic assumption is that the charge of the other cations and anions present in solution can be neglected. The concentration of organic acids in bulk deposition and throughfall was calculated from measured concentrations of DOC (when available), according to Oliver et al. (1983). More information is given in Annex 2.

When DOC was not measured, the charge balance check was still made assuming that the influence was small. This is specifically true in bulk deposition, where concentrations of low molecular organic acids, such as formic and acetic acid, only have a minor role in the ionic balance.

The difference between the sum of all major cations and anions in soil solution was calculated similarly. However, unlike atmospheric deposition, Al was included in the calculation of cations. Checks on the ionic balance were only made when all cations and anions were measured. The only allowances made were situations where (i) Al was missing at a pH > 5, (ii) alkalinity was missing at a pH < 5 and (iii) DOC was missing (separate calculation). In general, it is required that PD is less than 10% for bulk deposition and less than 20% for throughfall when the sum of cations and anions is larger than 500 mmol<sub>c</sub>.m<sup>-3</sup> (WMO, 1992; Ulrich and Mosello, 1998). Larger relative differences are acceptable at low concentrations of the sum of cations and anions (Table 5.2).

Table 5.2 The required criteria for the ionic balance (WMO, 1992).

Cations+anions (mmol <sub>c</sub> .m <sup>-3</sup> )	Acceptable difference (%)
≤ 50	≤ 60
50 – 100	≤ 30
100 – 500	≤ 15
> 500	≤ 10

### Electric conductivity

Another quality check is the difference between measured and calculated electric conductivity, when available. Electric conductivity (EC) is a measurement of the ability of an aqueous solution to carry an electric current. Apart from temperature, this ability depends on the type and concentration (activity) of ions in solution according to:

$$EC = \sum_i \lambda_i \cdot f_i \cdot c_i \quad (5.4)$$

where:

EC = electric conductivity (μS.cm<sup>-1</sup>)

λ<sub>i</sub> = equivalent ionic conductance, being the capacity of a single ion to carry an electric current in ideal conditions of infinite solution at 20 °C (kS.cm<sup>2</sup>.eq<sup>-1</sup>)

c<sub>i</sub> = concentration of ion i with i = H, Ca, Mg, K, Na, Al, NH<sub>4</sub>, NO<sub>3</sub>, SO<sub>4</sub>, Cl, Alk (mmol<sub>c</sub>.m<sup>-3</sup>)

f<sub>i</sub> = activity coefficient of ion i.

Values used for λ<sub>i</sub> for the various ions, are given in De Vries et al. (2000a). Activity coefficients were calculated as a function of the ionic strength (I), using the Davies equation (Stumm and Morgan, 1981). The percentage difference between calculated and measured conductivity was calculated as:

$$PD = 100 \cdot \frac{(EC_{\text{calc}} - EC_{\text{meas}})}{EC_{\text{calc}}} \quad (5.5)$$

According to WMO (1992), the discrepancy between measured and calculated conductivity should therefore be no more than 20% at a measured conductivity above 30 μS.cm<sup>-1</sup> (Ulrich and Mosello, 1998). At lower ionic strength, the acceptable difference was set at 30%.

### ***Sodium to chloride ratios***

The correlation between ions in solution and the covariance between ion concentration ratios is a third possibility to check the quality of the data. An important check is the ratio between Na and Cl. Assuming that seasalt is a dominant source of both ions, the Na to Cl ratio should resemble the ion ratio in seawater being equal to  $0.858 \text{ mol}_c \cdot \text{mol}_c^{-1}$ . Ivens (1990) found a Na to Cl ion ratio mostly varying between 0.7 and 1.0 in annual bulk deposition and throughfall fluxes with a median value resembling the ratio in seawater (0.84 in bulk deposition and 0.88 in throughfall). Draaijers (pers. comm.) stated that on an annual basis, the Na to Cl ratio should vary between 0.5 and 1.0.

### **5.2.3.2 Correcting outliers**

Throughfall, stemflow and bulk precipitation concentration data in the Intensive Monitoring database regularly did not pass the quality checks described in chapter 5.2.3.1. For this reason a 'correction' procedure was implemented resulting in more data available for further analysis. First, concentration data were log-transformed because they have log-normal distributions. Then for each plot the following sequential procedure was applied:

- The average log transformed concentration of each ion X (X = H, NO<sub>3</sub>, NH<sub>4</sub>, SO<sub>4</sub>, Mg, Ca, K, Na and Cl) and its standard deviation was determined as well as the average ratio of ion X to each other ion Y (8 ratio's) and the associated standard deviations, using all available measurements (mostly bi-weekly or monthly values).
- Approved log-transformed concentration data were then selected by removing concentration data outside 95% confidence interval (average + 2 \* standard deviation). With this approved data set, linear regression analysis (assuming that there is no trend in concentrations) was carried out to find the relationship for each combination of ions X and Y.
- For each measurement (sample) with an error in the charge balance exceeding the maximum allowable error, all ratio's of X to Y were determined. If more than 4 out of the possible 8 ratio's of ion X to the other ions were outside the 95% confidence interval, the concentration of ion X was considered to be an outlier. A 'corrected' concentration for ion X was then computed from the ion Y that has the highest correlation coefficient using the regression equation derived in step 2, with the restriction that the concentration-value of ion Y itself was not considered to be an outlier.

This procedure did correct a number of outliers in the data set that probably originate from analysis errors or errors in transferring the data, but was not always successful. To avoid the 'correction' of sound measurement data, finally a comparison was made for each period between the original and corrected data. Only if either the error in the charge balance or the error in the computed versus measured conductivity decreased when using the 'corrected' data, these corrected values were used, otherwise the original data were maintained.

### **5.2.4 Calculation of canopy exchange and element input**

#### ***Model description and parameterisation***

Total deposition was calculated according to a slightly adapted canopy budget model developed by Ulrich (1983) and extended by Bredemeier (1988), Van der Maas et al. (1991) and by Draaijers and Erisman (1995). In the canopy budget model, annual total deposition is derived by correcting the input by both throughfall and stemflow for exchange processes, occurring within

the forest canopy. At plots where stemflow data were missing, the annual stemflow was estimated from the annual throughfall according to Ivens (1990):

$$X_{sf} = X_{tf} \cdot \alpha / (1 - \alpha) \quad (5.6)$$

where:

X = a given ion (H, Ca, Mg, K, Na, NH<sub>4</sub>, NO<sub>3</sub>, SO<sub>4</sub>, Cl)  
 sf = stemflow (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>)  
 tf = throughfall (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>)  
 α = an empirical value

For coniferous forests, the value of α was calculated as a function of stand age according to (Ivens, 1990):

$$\begin{aligned} \alpha &= 0.24 && \text{age} < 20 \\ \alpha &= 0.31 - 0.0034 \cdot \text{age} && 20 < \text{age} < 90 \\ \alpha &= 0.0 && \text{age} > 90 \end{aligned} \quad (5.7)$$

For deciduous forests, α was set at 0.12 independent of age. The same values were used for coniferous forests with an unknown age (Ivens, 1990). More information is given in De Vries et al. (1999).

Total deposition fluxes of base cations were calculated according to (Ulrich, 1983):

$$BC_{td} = \frac{Na_{tf} + Na_{sf}}{Na_{bd}} \cdot BC_{bd} \quad (5.8)$$

where:

BC = Ca, Mg, K  
 td = total deposition (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>)  
 bd = bulk deposition (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>)

Eq. (5.8) is based on the assumption that (i) Na does not interact with the forest canopy (inert tracer) and (ii) the ratios of total deposition over bulk deposition are similar for Ca, Mg, K and Na. Specifically in coastal areas, this assumption is not always valid (Baloutes. Greece, pers. comm.) Canopy leaching induced by the internal cycle of these nutrients, was thus computed by the difference between the sum of BC in throughfall and stemflow minus total deposition according to:

$$BC_{ce} = BC_{tf} + BC_{sf} - BC_{td} \quad (5.9)$$

where:

ce = canopy exchange (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>)

Canopy exchange of SO<sub>4</sub><sup>2-</sup> is assumed negligible. The total deposition of this ion was thus calculated as:

$$SO_{4,td} = SO_{4,tf} + SO_{4,sf} \quad (5.10)$$

NH<sub>4</sub> and H interact with the forest canopy by exchange with base cations (Roelofs et al., 1985). We assumed that the total canopy uptake of H<sup>+</sup> and NH<sub>4</sub><sup>+</sup> is equal to the total canopy leaching of Ca<sup>2+</sup>, Mg<sup>2+</sup> and K<sup>+</sup> taking place through ion exchange, corrected for the leaching of weak acids. The NH<sub>4</sub> throughfall and stemflow flux was thus corrected for canopy uptake to calculate the total deposition of NH<sub>4</sub> according to (After Van der Maas et al., 1991; Draaijers and Erisman, 1995):

$$\text{NH}_{4,\text{td}} = \text{NH}_{4,\text{tf}} + \text{NH}_{4,\text{sf}} + \text{NH}_{4,\text{ce}} \quad (5.11)$$

with:

$$\text{NH}_{4,\text{ce}} = \text{BC}_{\text{ce}} - \text{WA}_{\text{ce}} - \text{H}_{\text{ce}} \quad (5.12)$$

and:

$$\text{H}_{\text{ce}} = \left( \frac{\text{H}_{\text{tf}} \cdot \text{xH}}{\text{NH}_{4,\text{tf}} + \text{H}_{\text{tf}} \cdot \text{xH}} \right) \cdot (\text{BC}_{\text{ce}} - \text{WA}_{\text{ce}}) \quad (5.13)$$

where:

xH = an efficiency factor of H in comparison to NH<sub>4</sub>  
 WA = weak acids

Based on experiments in the laboratory (Van der Maas et al., 1991), it was assumed that H<sup>+</sup> has per mol an exchange capacity six times larger than NH<sub>4</sub><sup>+</sup> (xH = 6). The estimation of the weak acid concentration was based on the sum of HCO<sub>3</sub><sup>-</sup>, derived from the pH and an assumed atmospheric CO<sub>2</sub> pressure, and RCOO<sup>-</sup> derived from DOC or the difference in concentration of cations minus strong acid anions (see Annex 4). The weak acid canopy exchange was calculated as:

$$\text{WA}_{\text{ce}} = \text{WA}_{\text{tf}} + \text{WA}_{\text{sf}} - 2\text{WA}_{\text{bd}} \quad (5.14)$$

The total deposition of protons was calculated as:

$$\text{H}_{\text{td}} = \text{H}_{\text{tf}} + \text{H}_{\text{sf}} + \text{H}_{\text{ce}} \quad (5.15)$$

Total deposition of NO<sub>3</sub><sup>-</sup> was calculated according to:

$$\text{NO}_{3,\text{td}} = \text{NO}_{3,\text{tf}} + \text{NO}_{3,\text{sf}} + \text{NO}_{3,\text{ce}} \quad (5.16)$$

where the canopy exchange of NO<sub>3</sub><sup>-</sup> equals the canopy exchange of nitrogen minus the canopy exchange of NH<sub>4</sub><sup>+</sup>. The canopy exchange of nitrogen was calculated according to:

$$\text{N}_{\text{ce}} = \text{NH}_{4,\text{ce}} \cdot \left( \frac{\text{NH}_{4,\text{tf}} \cdot \text{xNH}_4 + \text{NO}_{3,\text{tf}}}{\text{NH}_{4,\text{tf}} \cdot \text{xNH}_4} \right) \quad (5.17)$$

where  $NH_{4,ce}$  is calculated according to Eq. (5.12) and the  $xNH_4$  is an efficiency factor of  $NH_4^+$  in comparison to  $NO_3^-$  (we assumed that  $xNH_4 = 6$ ). Actually, Draaijers and Erisman (1995), assumed canopy uptake of  $NO_3^-$  to be negligible. There is, however, ample evidence that this is not true, specifically since  $NO_3^-$  in throughfall is often less than  $NO_3^-$  in bulk deposition in low deposition areas. Up to now, several basic assumptions in the model (e.g. the ratio in exchange capacity between  $H^+$  and  $NH_4^+$ ) are not properly evaluated for different environmental conditions (tree species, ecological setting, pollution climate) which limits its application (Draaijers et al., 1994; Draaijers and Erisman, 1995). Furthermore, the model has only been validated in relatively polluted areas, such as the Netherlands and Denmark. An independent estimate of the canopy exchange can be made by subtracting throughfall and stemflow data from estimated total deposition values obtained by an atmospheric deposition model such as EDACS (Erisman and Draaijers, 1995). Such an estimate is, however, completely dependant on the validity of EDACS model results. A comparison of EDACS model estimates and measurements at 223 Intensive Monitoring plots is given in Annex 5.

### *Assessment of relationships with stand and site characteristics*

To gain insight in the impact of various environmental factors on the calculated canopy exchange, multiple regression analysis was carried out. The canopy exchange of base cations ( $BC_{ce}$ ) for coniferous and deciduous tree species, was related to the deposition of base cations ( $BC_{dep}$ ), base cations content of the leaves/needles ( $BC_{fol}$ ), acid deposition ( $H_{dep}$ ), ammonium deposition ( $NH_{4,dep}$ ), total annual precipitation amount ( $P_{tot}$ ), surface wetness duration (SW) and vitality/defoliation (Def) according to:

$$BC_{ce} = f(BC_{dep}, H_{dep}, NH_{4,dep}, BC_{fol}, P_{tot}, SW, Def) \quad (5.18)$$

Surface wetness duration was approximated by taking the fraction of days in the year with (any amount of) rainfall. Defoliation was estimated according to the method presented by Klap et al. (1997). Biotic stress factors were not taken into account but used to explain possible outliers.

The canopy exchange of reduced nitrogen components ( $NH_{4,ce}$ ) was related to:

$$NH_{4,ce} = f(H_{dep}, NH_{4,dep}, N_{fol}, P_{tot}, SW, Def, R_{stom}) \quad (5.19)$$

where  $NH_{4,dep}$  is the deposition of reduced nitrogen,  $N_{fol}$  the nitrogen content in leaves/needles and  $R_{stom}$  the stomatal resistance. By using the model of Baldocchi et al. (1987), it was assumed that  $R_{stom}$  only depends on surface temperature and global radiation (Q):

$$R_{stom} = R_i \cdot \left( 1 + \left[ \frac{200}{Q + 0.1} \right]^2 \right) \cdot \left( \frac{400}{T_s(40 - T_s)} \right) \quad (5.20)$$

Values for the internal resistance ( $R_i$ ) were obtained from a look-up table for different land-use categories and seasons, as described by Wesely (1989).

The canopy exchange of oxidised nitrogen components ( $NO_{3,ce}$ ) was related to:

$$NO_{3,ce} = f(NO_{3,dep}, N_{fol}, Def, R_{stom}) \quad (5.21)$$

where  $\text{NO}_{3,\text{dep}}$  is the deposition of oxidised nitrogen. Finally, the canopy exchange of total nitrogen ( $N_{\text{ce}}$ ) was related to the following environmental factors:

$$N_{\text{ce}} = f(N_{\text{dep}}, H_{\text{dep}}, N_{\text{fol}}, P_{\text{tot}}, \text{SW}, \text{Def}, R_{\text{stom}}) \quad (5.22)$$

in which  $N_{\text{dep}}$  is equal to the sum of oxidised and reduced nitrogen deposition. As for base cations, biotic stress factors were not taken into account but used to explain outliers.

## 5.2.5 Calculation of leaching fluxes and element retention

### *Model description*

Leaching fluxes of the considered elements were calculated by multiplying the measured soil solution concentrations with the water leaching fluxes at the corresponding depth. For sites with a thick soil profile ( $> 1$  m), element leaching fluxes were calculated at a depth of approx. 80 cm. This is often the deepest lysimeter cup and may be assumed to represent the leaching fluxes at the bottom of the root zone. At sites with more shallow soils the deepest available lysimeter cup was used.

Concentrations in lysimeter cups were generally measured with weekly to monthly intervals, whereas water fluxes were calculated on a daily basis (Chapter 4). To calculate leaching fluxes, concentrations either have to be interpolated or water fluxes have to be accumulated over the measuring period. Both methods were applied in calculating leaching fluxes by multiplying:

1. The (average) measured concentration during the period between two subsequent measurements with the accumulated water flux during that period.
2. The daily interpolated concentrations with daily water fluxes.

Multiplication of yearly average concentrations with yearly average leaching fluxes is also possible, but the reliability of this approach is relatively small (De Vries and Janssen, 1994). Application of this method may only be preferred when large errors occur in the measured concentrations, when measurement are infrequent or when daily calculated leaching fluxes are assumed to be very uncertain. Since we only used data from sites where concentrations are measured at regular intervals, we only used the two methods described above.

The choice between the first and second method described above depends somewhat on the method and the measurement frequency used to sample the soil solution. In most cases samples are obtained by putting suction on lysimeter cups and collecting the water samples after a period of 1 to 4 weeks. During the sampling periods the suction of the cups gradually reduces and accordingly more soil water is collected at the start of the sampling period compared to the end of the period. When the sampling frequency is high, the collected samples will be representative for the average concentration during the sampling period and method 2 may be preferred. When longer sampling intervals are used, interpolation of the measured concentrations between two subsequent measurements may be preferred. In that case the measured concentrations are assumed to represent the concentration at the day of the start of the sampling period. Concentrations at the following days are calculated by interpolation between these concentrations and the concentrations measured during the next sampling period (method 2). In this study both method 1 and 2 have been used to indicate the uncertainty of the calculated leaching fluxes.

Element budgets were calculated by subtracting the leaching flux from the total deposition flux, in which total deposition fluxes were derived by both the canopy exchange model and by regression models (c.f. 5.2.4). For each site four estimates of the element budgets were thus obtained based on (i) the standard canopy uptake and average soil solution concentrations, (ii) the standard canopy uptake model and interpolated soil solution concentrations, (iii) the regression canopy uptake and standard soil solution concentrations and (iv) the regression canopy uptake model and interpolated soil solution concentrations. Positive budgets indicate that a certain element is retained in the soil, whereas negative budgets indicate a net release of this element from the soil.

### *Assessment of relationships with stand and site characteristics*

To investigate relationships between element leaching and element input, use was made of multiple regression models relating element leaching to the input and all predictor variables affecting element retention. Regression analysis relates a given response variable to (environmental) predictor variables. The term response variable stems from the idea that it responds to the environmental variables in a causal way, but causality cannot be inferred from regression analysis. Supposed relationships were of the form:

$$\log y = \alpha_0 + \alpha_1 x_1 + \alpha_2 x_2 + \dots + \alpha_n x_n \quad (5.23)$$

where  $\log y$  is the expectation value of the response variable (atmospheric deposition, soil solution chemistry),  $x_1$  to  $x_n$  are predictor variables (stand and site characteristics, meteorological parameters etc.) and  $\alpha_1$  to  $\alpha_n$  are the regression coefficients.

The regression analyses was applied by using a so-called Select procedure. This procedure combines predictor variables that were qualitative (indicator variables), such as tree species and/or soil type with quantitative variables. This approach combines forward selection, starting with a model including one predictor variable, and backward elimination, starting with a model including all predictor variables. The 'best' model was based on a combination of the percentage of variance accounted for ( $R^2_{adj.}$ ), that should be high and the number of predictor variables, that should be low. In order to meet the requirement of regression analyses that the response variable is normally distributed with a constant variance at fixed values of the predictor values, the considered responses (the leaching data) were log-transformed. This also causes interaction to be less significant. Normality was checked by a scatter plot of the residuals against the fitted values.

An overview of the predictor variables included in the various regression models is given in Table 5.3. Site characteristics included as predictor variables were soil type and humus type. Stand characteristics included tree species and stand age, being all variables that appear to influence element retention or uptake (e.g. Dise et al., 1998a, b; Gundersen et al., 1998a). All stand and site characteristics were included as qualitative variables. Regarding soil type, 3 main groups were distinguished i.e.: Podzols and Arenosols, being acidic sandy soils, Cambisols and Luvisols being slightly acidic sandy soils or clayey soils and remaining non-calcareous soils. The humus types were distinguished in mor, moder, moder/mull and mull. For tree species 5 major groups were distinguished: pine, spruce, other coniferous, oak, and beech (see also De Vries et al., 1998). Regarding stand age, a distinction was made in young (< 30 years) and mature (> 30 years) stands. The limitation of the various characteristics to the distinguished groups was based on the expected differences, considering the limited number of data (only 121 plots at maximum). As a



rule of thumb, the number of observations should exceed 4 times the degrees of freedom (Oude Voshaar, 1994).

Table 5.3 Overview of the predictor variables used to explain element leaching and element budgets at the 121 Intensive Monitoring plots where both deposition data and leaching fluxes were available.

Predictor variables	S	N	BC	Al
<i>Site/Stand characteristics</i>				
Tree species	X	X	X	X
Soil type group	X	X	X	X
Humus type group	X	X	X	X
Stand age	X	X	X	X
<i>Deposition</i>				
S	X		X	X
N		X	X	X
FrNH <sub>4</sub>		X	X	X
BC			X	X
<i>Foliage</i>				
N-content		X		
<i>Soil chemistry</i>				
Organic C pool		X		
C/N ratio litter		X		
C/N ratio mineral layer		X		
pH topsoil	X	X		
pH subsoil	X		X	X
Base saturation			X	X
<i>Interactions</i>				
N deposition.C/N ratio		X		

With respect to atmospheric inputs, use was made of the calculated total deposition using the original canopy exchange model. Results obtained from the regression equations were considered less adequate, specifically for base cations (Section 5.3.1). Soil chemical variables included were those that were assumed to influence either N retention processes (C/N ratio in the organic layer and mineral topsoil and soil pH) and BC or Al release (base saturation and pH).

## 5.3 Results and discussion

### 5.3.1 Element input by deposition

In Table 5.4 results are presented of the quality checks applied on annual throughfall and bulk precipitation fluxes for the period 1995-1998 stored in the Intensive Monitoring database.

Table 5.4 Overview of the results of the quality checks performed on available annual throughfall and bulk precipitation data for the period 1995-1998 as stored in the FIMCI database.

	Bulk precipitation	Throughfall
Total number before quality checks	820	820
Missing data	86	97
Data failing the ionic balance check	301	169
Data failing the sodium-to-chloride check	75	71
Total number passing quality checks	411	515

For 309 sites annual throughfall and bulk precipitation fluxes were available for all components for one or more years, leading to 820 plot-year combinations. These sites were distributed over 21 different countries in Europe. Quality checks included a check on the ionic balance and on

sodium-to-chloride ratios. Checks on measured and calculated conductivity and on phosphate concentrations could not be applied due to lack of information (data).

Applying the quality checks resulted in about 50% and 37% of the data being lost for bulk precipitation and throughfall, respectively. The quality checks were applied on corrected data as described in Section 5.2.3.2). The correction procedure was applied on monthly data available, after which annual fluxes were calculated. These annual fluxes were then used in the multiple regression analysis. Comparisons with uncorrected data showed that an additional 10% of these data could be used for performing the multiple regression analysis.

### 5.3.1.1 Relationships between canopy exchange and environmental factors

Annual throughfall and bulk deposition data for the years in the period 1995-1998 that successfully passed the applied quality checks were subsequently used for multiple regression analysis using the functions presented before. Sites passing the quality checks were located in northwestern Europe (France, Norway, Sweden, Finland, Germany and Ireland). By far most sites were coniferous forest stands. The relationships described below were derived for both coniferous and deciduous forests. For this study the tree species *Picea abies* and *Pinus silvestris* were selected to represent the coniferous forests, while for deciduous forest the selected tree species were: *Fagus sylvatica*, *Quercus petraea* and *Quercus robur*. For all deposition parameters in the regression functions up to now bulk precipitation fluxes are used. Instead of bulk precipitation fluxes, total deposition estimates will be used when modelled dry deposition fluxes are available from the EDACS model (Erisman and Draaijers, 1995). The relationships described below were derived according to the functions shown in Section 5.2.4. Parameters not significantly contributing to the regression were excluded.

For total base cations the following relationships were derived for coniferous forests (n=180) and deciduous forests (n=68), respectively:

$$BC_{cc} = -0.353 + 0.850 \cdot BC_{fol} + 0.828 \cdot NH_{4,dep} + 0.002 \cdot P_{tot} \quad R^2_{adj} = 0.53 \quad (5.24)$$

$$BC_{cc} = 0.085 + 0.911 \cdot BC_{fol} \quad R^2_{adj} = 0.10 \quad (5.25)$$

in which  $BC_{cc}$  and  $NH_{4,dep}$  are in  $kmol_c \cdot ha^{-1}$ ,  $BC_{fol}$  in  $mmol_c \cdot kg^{-1}$  and  $P_{tot}$  in  $mm \cdot yr^{-1}$ . Instead of calculating the total canopy exchange of base cations, also the canopy exchange of the separate components was calculated leading to the following relationships for coniferous forests (n=180) and deciduous forests (n=68), respectively:

$$Mg_{le} = -0.079 + 0.142 \cdot BC_{fol} + 0.098 \cdot NH_{4,dep} + 0.0005 \cdot P_{tot} \quad R^2_{adj} = 0.22 \quad (5.26)$$

$$Mg_{le} = -0.022 + 0.158 \cdot BC_{fol} \quad R^2_{adj} = 0.10 \quad (5.27)$$

$$Ca_{le} = -0.112 + 0.270 \cdot BC_{fol} + 0.298 \cdot NH_{4,dep} - 0.0005 \cdot P_{tot} + 2.787 \cdot H_{dep} + 0.010 \cdot Def \quad R^2_{adj} = 0.41 \quad (5.28)$$

$$Ca_{le} = -0.061 + 0.480 \cdot BC_{fol} - 0.569 \cdot Ca_{dep} \quad R^2_{adj} = 0.31 \quad (5.29)$$

$$K_{le} = 0.052 + 0.349 \cdot BC_{fol} + 0.408 \cdot NH_{4,dep} + 0.002 \cdot P_{tot} - 0.976 \cdot K_{dep} - 0.013 \cdot Def - 0.298 \cdot SW \quad R^2_{adj} = 0.48 \quad (5.30)$$

$$K_{le} = 0.228 + 0.347 \cdot BC_{fol} + 0.001 \cdot P_{tot} - 1.640 \cdot K_{dep} \quad R^2_{adj} = 0.21 \quad (5.31)$$

in which  $Mg_{le}$ ,  $Ca_{le}$ ,  $K_{le}$ ,  $NH_{4,dep}$ ,  $H_{dep}$ ,  $Ca_{dep}$  and  $K_{dep}$  are in  $kmol_c \cdot ha^{-1}$ ,  $BC_{fol}$  in  $mmol_c \cdot kg^{-1}$ ,  $P_{tot}$  in  $mm \cdot yr^{-1}$ ,  $SW$  in % and  $Def$  is the logit transformation of the defoliation (in %/100).

Surface wetness can be a difficult parameter to determine, due to lacking information on the precipitation data on a day by day basis. When leaving surface wetness out of the function for  $K_{le}$ , the relationship is very similar to the function mentioned above and becomes (coniferous forests;  $n=180$ ):

$$K_{le} = -0.169 + 0.440 \cdot BC_{fol} + 0.434 \cdot NH_{4,dep} + 0.002 \cdot P_{tot} - 0.843 \cdot K_{dep} - 0.015 \cdot Def \quad R^2_{adj} = 0.46 \quad (5.32)$$

Relationships were quite strong for coniferous forests but relatively weak for deciduous forests. Results clearly show that base cation leaching increases with increasing base cation foliar content and increasing  $NH_4$  deposition. Leaching of calcium positively relates to H deposition as well. Leaching of Ca and K is significantly smaller with increasing deposition of the mentioned cations. Leaching of base cations generally increases with increasing annual precipitation amount, Ca leaching being the exception showing significant larger leaching in less wet regions. K leaching increases with smaller vitality and smaller surface wetness duration. Ca leaching increases with less defoliation. For reduced nitrogen the following equations were derived:

$$NH_{4,up} = 0.106 - 0.278 \cdot N_{fol} + 1.171 \cdot NH_{4,dep} - 0.020 \cdot Def + 0.0004 \cdot R_{stom} \quad R^2_{adj} = 0.62 \quad (5.33)$$

$$NH_{4,up} = 0.449 + 0.404 \cdot NH_{4,dep} - 0.001 \cdot P_{tot} \quad R^2_{adj} = 0.22 \quad (5.34)$$

in which  $NH_{4,up}$  and  $NH_{4,dep}$  are in  $kmol_c \cdot ha^{-1}$ ,  $N_{fol}$  in  $mmol_c \cdot kg^{-1}$ ,  $P_{tot}$  in  $mm \cdot yr^{-1}$ ,  $R_{stom}$  in  $s \cdot m^{-1}$  and  $DEF$  is the logit transformation of the defoliation (in %/100).

As with surface wetness, stomatal resistance ( $R_{stom}$ ) can be a difficult parameter to determine. Leaving  $R_{stom}$  out of the function for  $NH_4$ , up for deciduous trees shows the following result, for which the relationship is less strong than the function mentioned above:

$$NH_{4,up} = 0.233 + 0.355 \cdot NH_{4,dep} - 0.001 \cdot P_{tot} \quad R^2_{adj} = 0.08 \quad (5.35)$$

For oxidised nitrogen the following equations were derived. Because for deciduous trees no significant relationship could be found, the equation derived for all tree species taken together can be used instead.

$$NO_{3,up} = -0.031 + 0.218 \cdot NO_{3,dep} - 0.008 \cdot Def \quad R^2_{adj} = 0.30 \quad (5.36)$$

$$NO_{3,up} = -0.017 + 0.179 \cdot NO_{3,dep} - 0.007 \cdot Def \quad R^2_{adj} = 0.23 \quad (5.37)$$

in which  $NO_{3,up}$  and  $NO_{3,dep}$  are in  $kmol_c \cdot ha^{-1}$  and  $Def$  is the log transformation of the defoliation (in %/100). As for base cations, relationships were (much) stronger for coniferous forests

compared to deciduous forests. Nitrogen uptake positively relates to nitrogen deposition. Ammonium uptake is found to be larger at lower nitrogen foliar content, small stomatal resistance (indicating  $\text{NH}_3$  uptake) and low vitality.

### 5.3.1.2 Ranges in total deposition

Results of a comparison of total element deposition for the 309 sites, using the canopy uptake model and regression equations derived from it, are shown in Fig. 5.2.

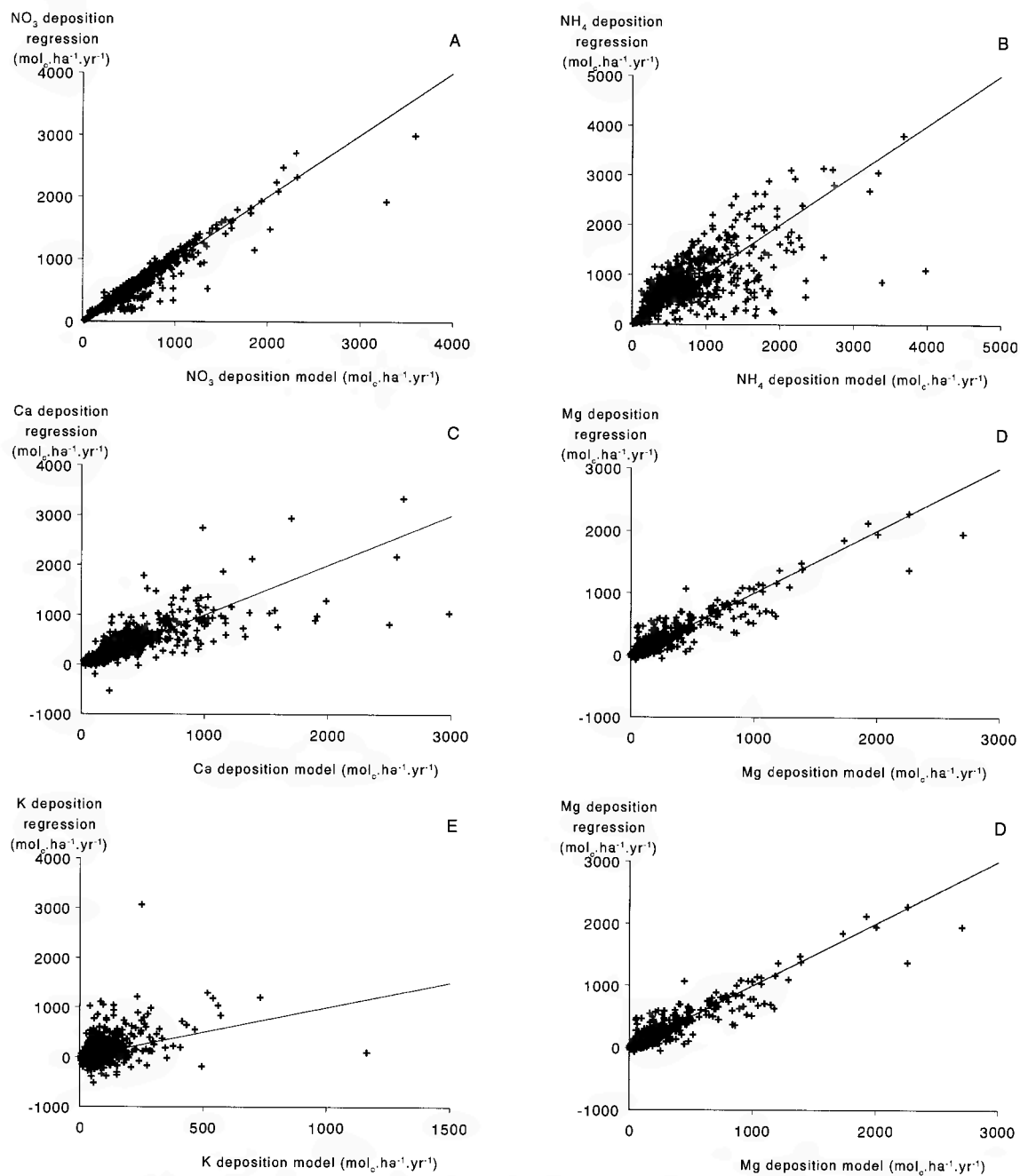


Figure 5.2 Comparison of total deposition fluxes of  $\text{NO}_3$  (A),  $\text{NH}_4$  (B), Ca (C), Mg (D), K (E) and BC (Ca+Mg+K) (F) for 309 sites using the canopy uptake model and regression equations derived from it.

For  $\text{NO}_3$  and Mg the comparison is reasonable, but for  $\text{NH}_4$ , Ca and specifically K it is weak. Note that the regression equations were only derived on a high quality subset of the data (Section 5.3.1). The results obtained by the regression model may cause negative deposition values in case of base cations, indicating that it is better to use the original canopy exchange model for those elements. Another reason for this choice is that the assumption of comparable dry to bulk deposition ratio for Na and the other base cations, applied in the canopy exchange model seems reasonable. The range in total deposition of the major elements is illustrated in Fig. 5.3.

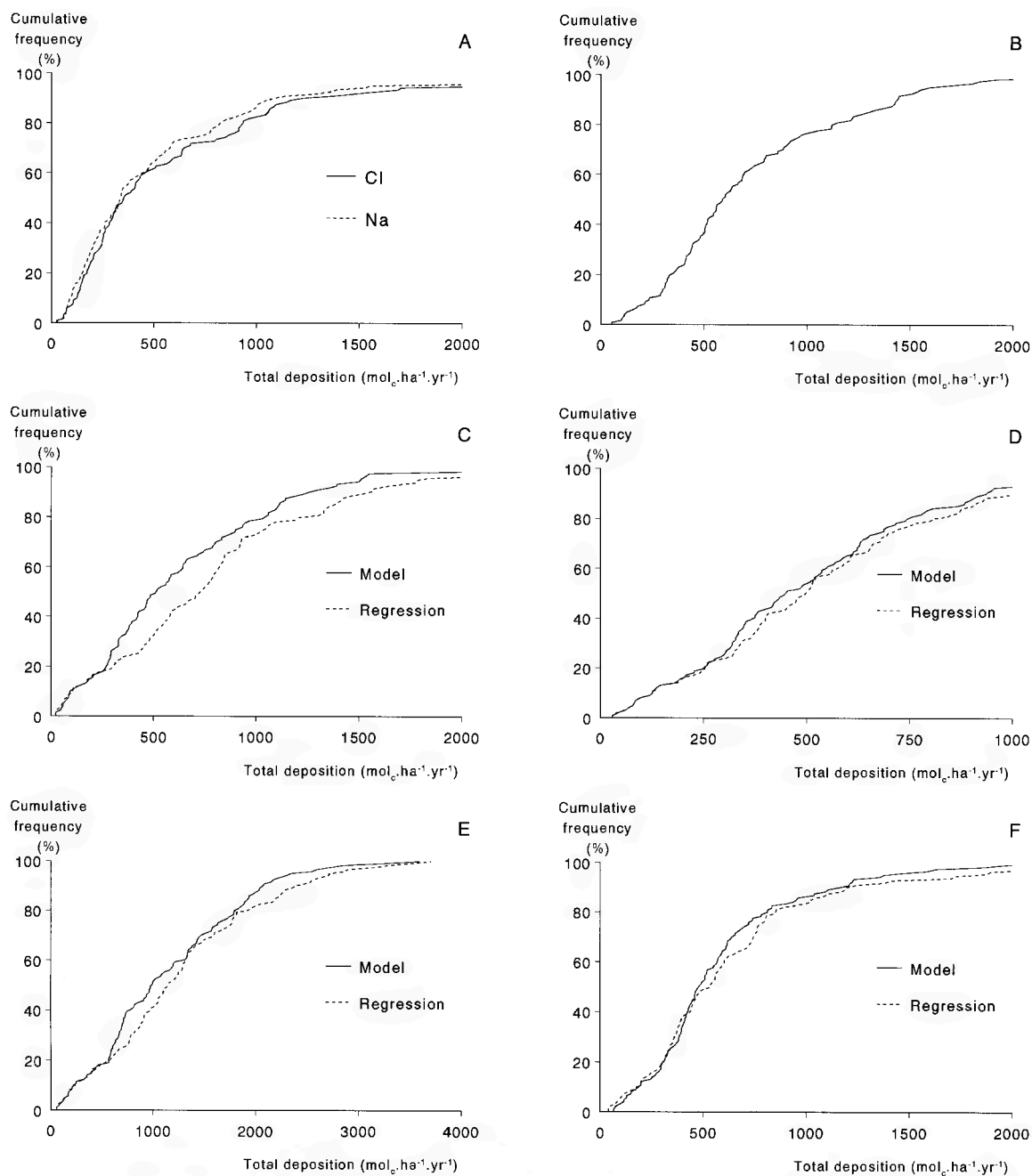


Figure 5.3 Cumulative frequency distributions of the total deposition of Cl and Na (A),  $\text{SO}_4$  (B),  $\text{NH}_4$  (C),  $\text{NO}_3$  (D), N (E) and BC (F).

Results show a comparable range in Cl and Na input (Fig. 5.3A). Total S deposition generally stays below  $2000 \text{ mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  (Fig. 5.3B) but the N deposition does exceed  $3000 \text{ mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  at

several plots (Fig. 5.3E). The figure illustrates that for both nitrogen and base cations, the regression equations generally lead to higher deposition estimates than the canopy exchange model (Fig. 5.3C- 5.3F).

The ratio in annual Na to Cl deposition ranged between 0.5 and 1.5 with most of the values near 0.86 being the Na/Cl ratio in seawater (Fig. 5.4A). The N/S ratio was mostly above 1.0, the percentage varying between approximately 65 and 85% depending on the canopy model used (Fig. 5.4B). Despite this uncertainty, the results show the dominating influence of nitrogen in the atmospheric input. The results for the  $\text{NH}_4/\text{NO}_3$  ratio (Fig. 5.4C) clearly illustrate the dominance of ammonium over nitrate in the deposition. This was already clear from the results presented in Figure 5.3, showing that the  $\text{NH}_4$  input can be as high as  $2000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$  (Fig. 5.3C), whereas the  $\text{NO}_3$  input stays nearly always below  $1000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$  (Fig. 5.3D). The results for the ratio of base cations to the sum of sulphur and nitrogen (Fig. 5.4D) illustrate that at approximately 30-50% of the plots (depending on the model used to calculate canopy exchange), the acid input by S and N is buffered by base cation input.

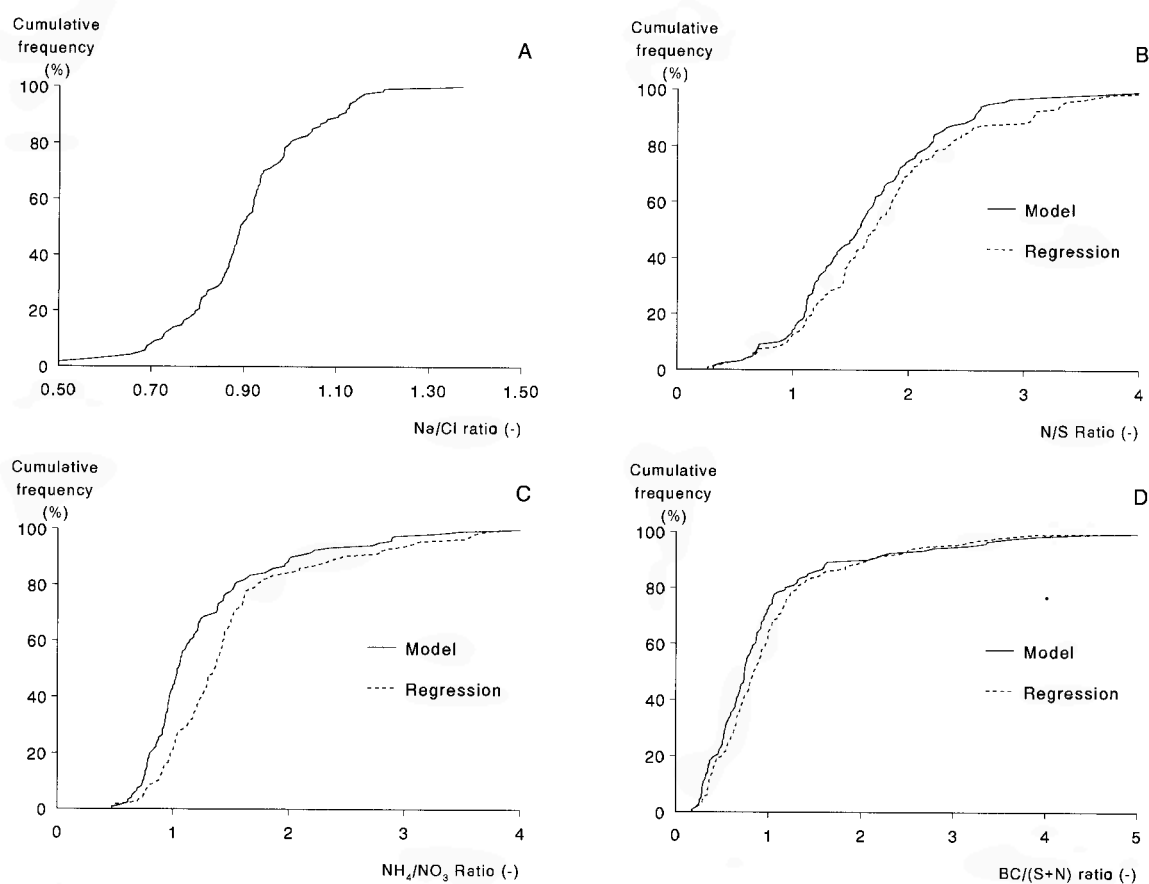


Figure 5.4 Cumulative frequency distributions of the ratio's in the total deposition of Na/Cl (A), N/S (B),  $\text{NH}_4/\text{NO}_3$  (C) and BC/(N+S) (D).

Results of the variation in element input fluxes for the 309 and 121 sites are shown in Table 5.5 and 5.6, respectively. As with Figure 5.3, both tables illustrate that for nitrogen compounds input fluxes are generally somewhat higher when calculated with the multiple regression models compared to the canopy budget model. Median nitrogen inputs in oak stands calculated from the multiple regression model were even twice as large when compared to the original canopy budget calculations. For base cations a mixed picture arises, with smaller median total base cation input

calculated using the multiple regression models for pine, beech and ‘other’ stands and higher median total base cation input calculated for spruce and especially oak stands.

Table 5.5 Median input fluxes (in mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) of sodium, chloride, sulphate, nitrate, ammonium and total base cations at 309 investigated sites using canopy exchange model to calculate canopy exchange. Values in brackets are the median input fluxes using the regression equations.

Tree species	Number of sites	Total deposition (mol <sub>c</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )						
		Cl	Na	SO <sub>4</sub>	N	BC		
Pine	67	236	210	408	621	(611)	394	(357)
Spruce	125	316	305	505	900	(844)	485	(544)
Oak	31	481	412	646	910	(1188)	562	(835)
Beech	51	346	334	663	1330	(1550)	629	(616)
Other	35	770	643	578	1047	(1078)	770	(723)
All	309	315	305	560	939	(1033)	534	(557)

Table 5.6 Median input fluxes (in mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) of sodium, chloride, sulphate, nitrate, ammonium and total base cations at 121 sites for which budgets were calculated using canopy exchange model to calculate canopy exchange. Values in brackets are the median input fluxes using the regression equations.

Tree species	Number of sites	Total deposition (mol <sub>c</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )						
		Cl	Na	SO <sub>4</sub>	N	BC		
Pine	29	280	241	517	714	(846)	491	(377)
Spruce	51	462	403	685	1198	(1312)	469	(559)
Oak	15	640	475	637	962	(1249)	519	(805)
Beech	20	398	389	634	1340	(1497)	489	(440)
Other	6	287	246	509	826	(796)	751	(653)
All	121	356	340	592	995	(1158)	482	(536)

Nitrogen (nitrate and ammonium) input fluxes are found highest in beech stands, that may possibly be explained by the location of these sites close to polluting sources. Relatively high inputs of nitrogen are also found for oak stands. Pine stands show much smaller input fluxes of acidifying compounds whereas spruce stands take up an intermediate position. The differences between tree species only partly reflect differences in surface roughness, influencing dry deposition. Part of the variation is explained by the location with e.g. pine stands being more dominant in Northern Europe where the N input is lower. For sulphate, the differences are much less distinct. Sodium and chloride input fluxes generally strongly depend on the input of sea salt and is generally found largest near sea areas. Largest inputs of sodium and chloride are found in oak stands and smallest inputs in pine stands. Base cation input is largest in beech and oak stands and smallest in pine stands.

### 5.3.1.3 Geographic variation in total deposition

The geographical distribution of the input fluxes is presented in the Figs 5.5-5.7. Chloride input is relatively large (>800 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) in coastal areas throughout Europe. Relatively high sulphate input (>800 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) can be found everywhere in Europe, except for central and northern part of Scandinavia. Many sites with high sulphate input are situated in central Europe (Fig. 5.5) High N inputs (>1800 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) occur in central Europe. Total nitrogen input is generally much smaller in northern and southern Europe (Fig. 5.6). Base cation input is relatively high (>800 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) in southern Europe and Lithuania, which is consistent with findings of Draaijers et al. (1997), whereas the input of base cations is low in Scandinavia (Fig. 5.7).

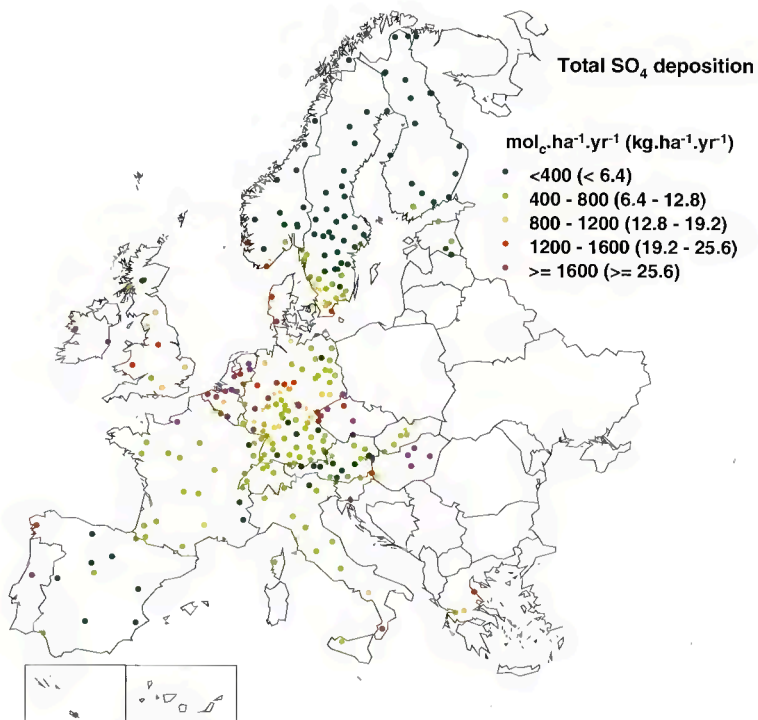
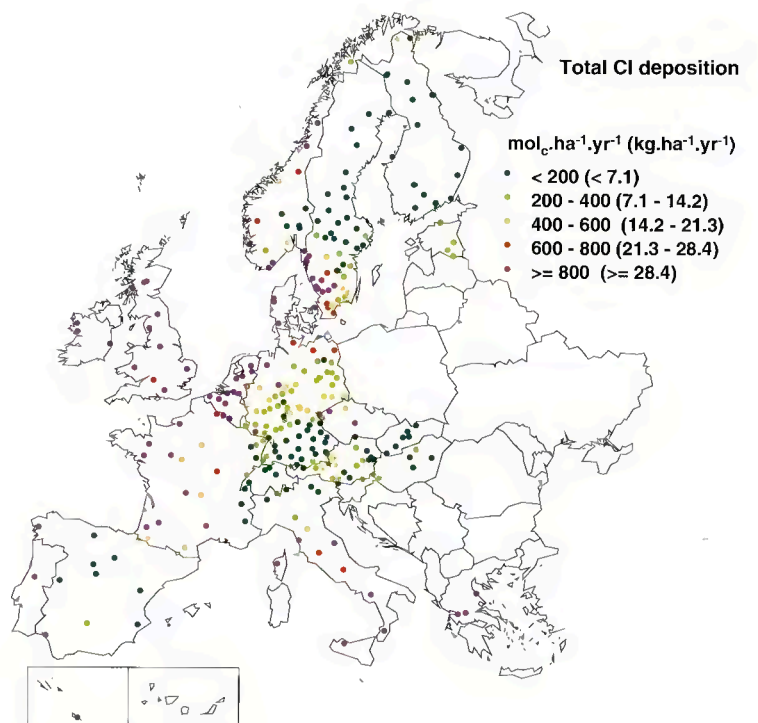


Figure 5.5 Geographical variation in input fluxes (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) for chloride (top) and sulphate (bottom) at the Intensive Monitoring plots throughout Europe.



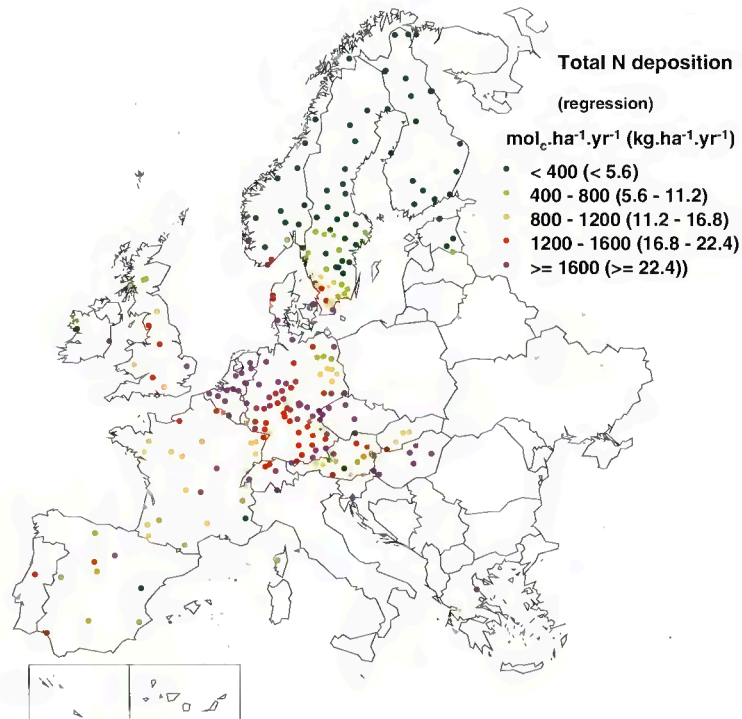
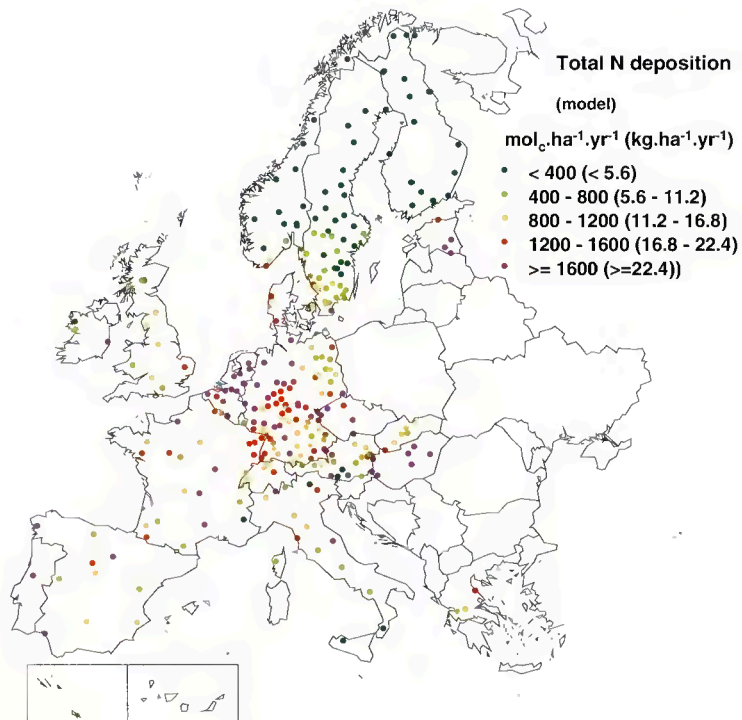


Figure 5.6 Geographical variation in total deposition ( $\text{mol}_e \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) at the Intensive Monitoring plots for nitrogen based on the canopy exchange model (top) and regression equations (bottom) to calculate canopy exchange (Fig. 5.5) throughout Europe.

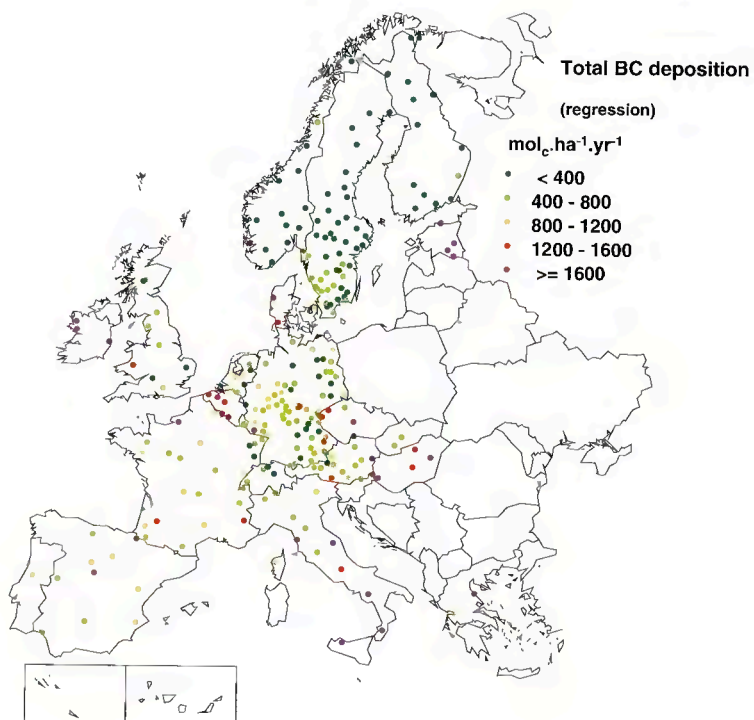
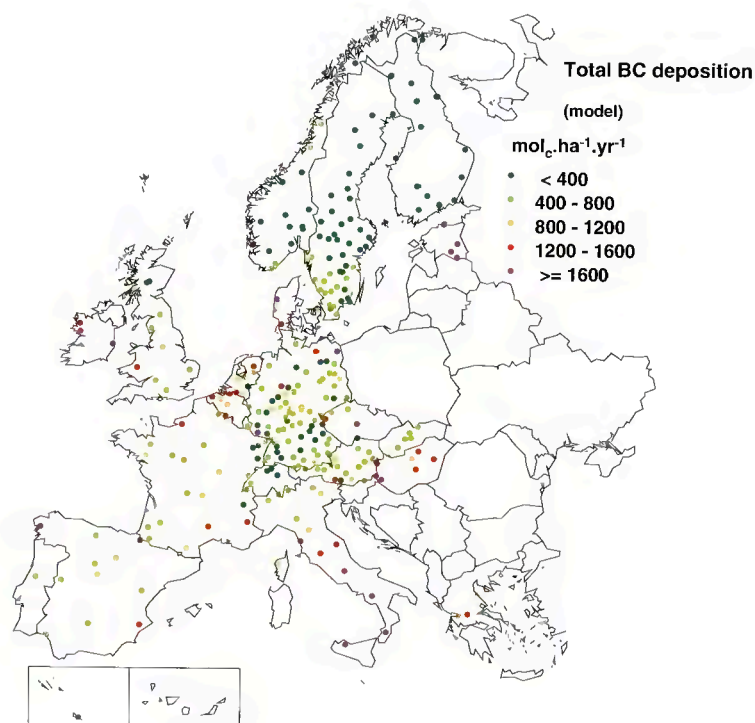


Figure 5.7 Geographical variation in total deposition ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) at the Intensive Monitoring plots for base cations (Ca+Mg+K) based on the canopy exchange model (top) and regression equations (bottom) to calculate canopy exchange (Fig. 5.5) throughout Europe.

### 5.3.2 Element output by leaching

#### 5.3.2.1 Comparison of results with measured and interpolated meteorological data

Fig. 5.8 shows the leaching fluxes based on interpolated data against the fluxes based on local data for 26 sites where both calculations could be made.

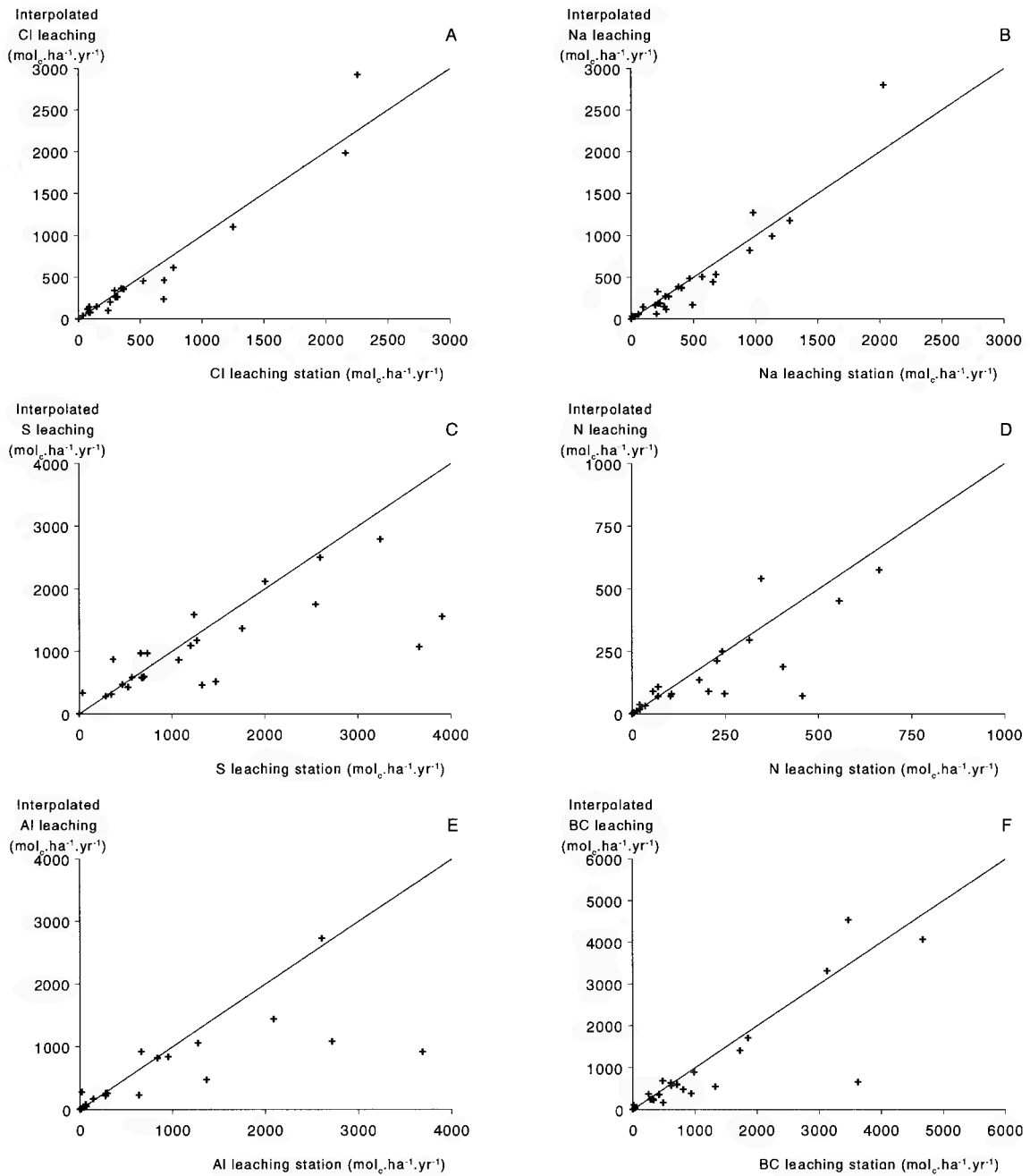


Figure 5.8 Leaching fluxes calculated using interpolated meteorological data compared to leaching fluxes calculated using locally measured meteorological data for Cl (A), Na (B), S (C), N (D), Al (E) and BC (F).

Due to differences in meteorological data (in particular with respect to wind speed), transpiration fluxes were overestimated when interpolated meteorological data were used, resulting in lower leaching fluxes (cf. section 4.3.2.1). These lower hydrological fluxes lead to slightly lower element leaching fluxes when interpolated data are used compared to the use of local meteorological data. The difference in median leaching fluxes for the 26 sites is quite limited and ranges from 15 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for Al up to 73 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations (Table 5.7). The difference in leaching fluxes are negligible (less < 4%) at 10% of the sites. At 50% of the sites the error is larger than 16 to 21%, and at 10% of the sites it is even larger than 60%.

Table 5.7 Leaching fluxes calculated using local and interpolated meteorological data and median, 10% and 90% differences in calculated leaching fluxes.

Element	Leaching fluxes (mol <sub>c</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )						Deviation		
	local meteorological data			interpolated meteorological data			(interpolated-local)/local (%)		
	10%	50%	90%	10%	50%	90%	10%	50%	90%
Cl	50	312	2217	61	260	2390	3	17	65
Na	46	301	1192	47	270	1215	2	20	67
S	317	907	2920	324	866	1935	2	18	68
N	5	106	517	5	80	497	0	21	64
BC	35	620	4008	80	547	4248	2	16	62
Al	5	210	2348	4	225	1069	4	21	66

### 5.3.2.2 Ranges in leaching fluxes

Differences in calculated element leaching fluxes using average or interpolated concentrations were small (Fig. 5.9). Leaching fluxes and budgets were calculated using the measured (average) concentrations, because this method is more straightforward. Results of element leaching fluxes obtained for the 121 investigated sites show a considerable variation as indicated in Figure 5.10.

The median water flux (not shown) was 144 mm.yr<sup>-1</sup>, but the 10 percentile was nearly 7 times as low, whereas the 90 percentile was approximately 4 times as high. For the major ions, the range is even higher. The 10 percentile leaching flux is lower than 100 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for most elements, except for sulphur which has a 10 percentile leaching flux of 130 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>. The 90 percentile leaching flux ranges from approximately 1000 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for Cl, Na and N up to almost 3000 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations. Remarkably are the strong differences for N which is almost negligible in 50% of the soils, whereas the 90 percentile leaching flux is 1000 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> (Fig. 5.10). This indicates that nitrogen is strongly retained or denitrified in the soil until a certain threshold in deposition levels is exceeded (cf. 5.3.2.2). Unlike the deposition, the leaching of ammonium is lower than that of nitrate.

Median leaching fluxes for Cl, Na, S, N, BC and Al concentrations for the various tree species are presented in Table 5.8. The highest leaching fluxes are found for sulphate corresponding to a relative high median input of sulphate. Median sulphate leaching fluxes for Oak are significantly higher than for pine which are relatively low compared to the median input of sulphur which is almost equal for the considered tree species. The high leaching fluxes for sulphate compared to nitrogen indicates that SO<sub>4</sub> is still the dominant source of actual soil acidification, although the total N deposition is generally larger than the total S deposition. This in turn indicates a clear difference between S and N retention in the forest ecosystem (see Section 5.3.3).

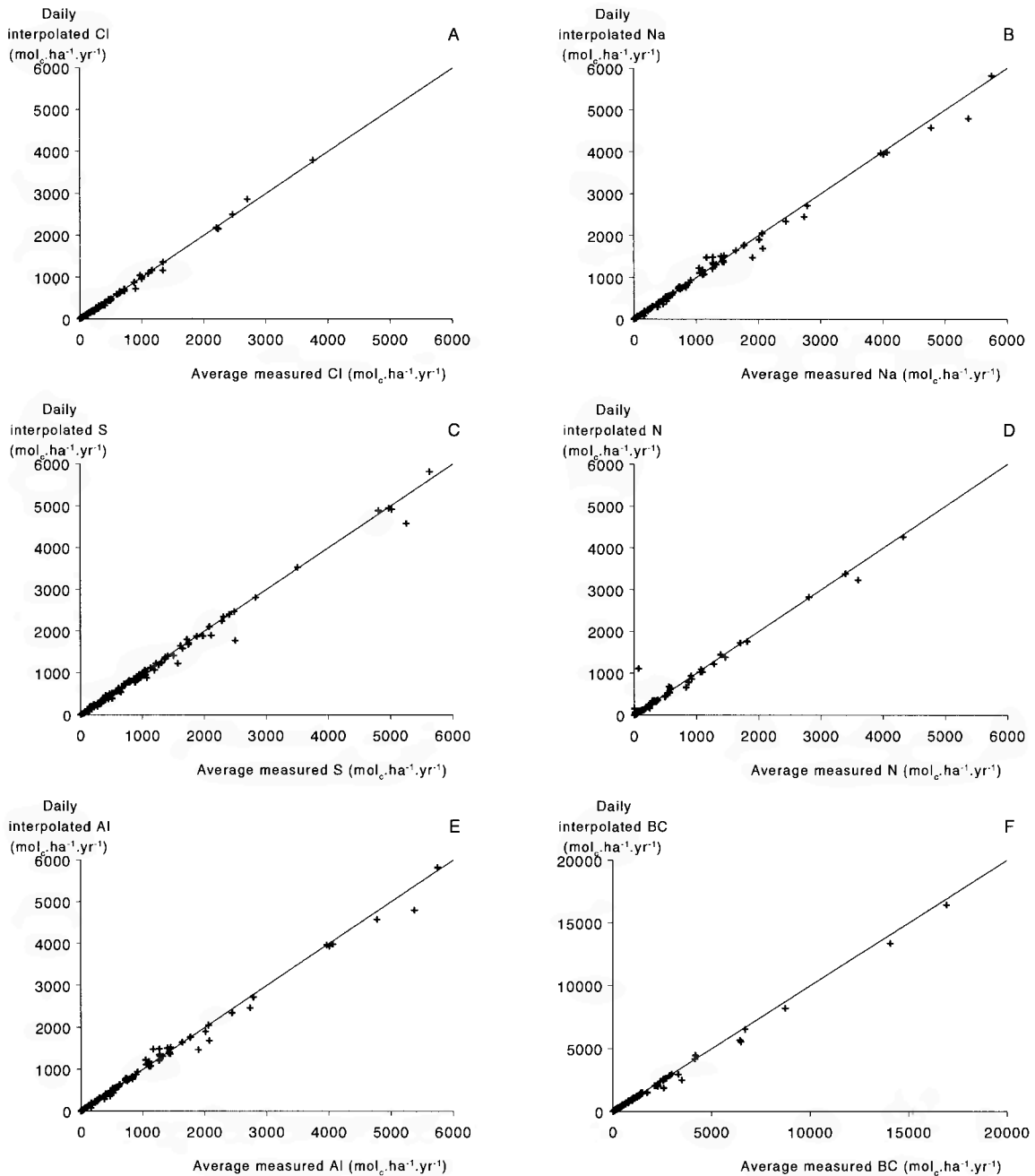


Figure 5.9 Comparison of calculated the Cl (A), Na (B), S (C), N (D), Al (E) and BC (Ca+Mg+K) (F) leaching fluxes using (average) measured concentrations and daily interpolated concentrations.

Table 5.8 Median element leaching fluxes (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) for chloride, sodium, sulphate, nitrogen, base cations (Ca+Mg+K) and aluminium at 121 investigated sites.

Tree species	Number of sites	Leaching fluxes (mol <sub>c</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )					
		Cl	Na	S	N	BC	Al
Pine	29	49	54	197	7	156	138
Spruce	51	203	233	590	112	331	774
Oak	15	427	548	1025	212	2184	30
Beech	20	272	269	604	135	717	326
Other	6	147	185	590	54	1149	31
All	121	200	211	509	60	377	294

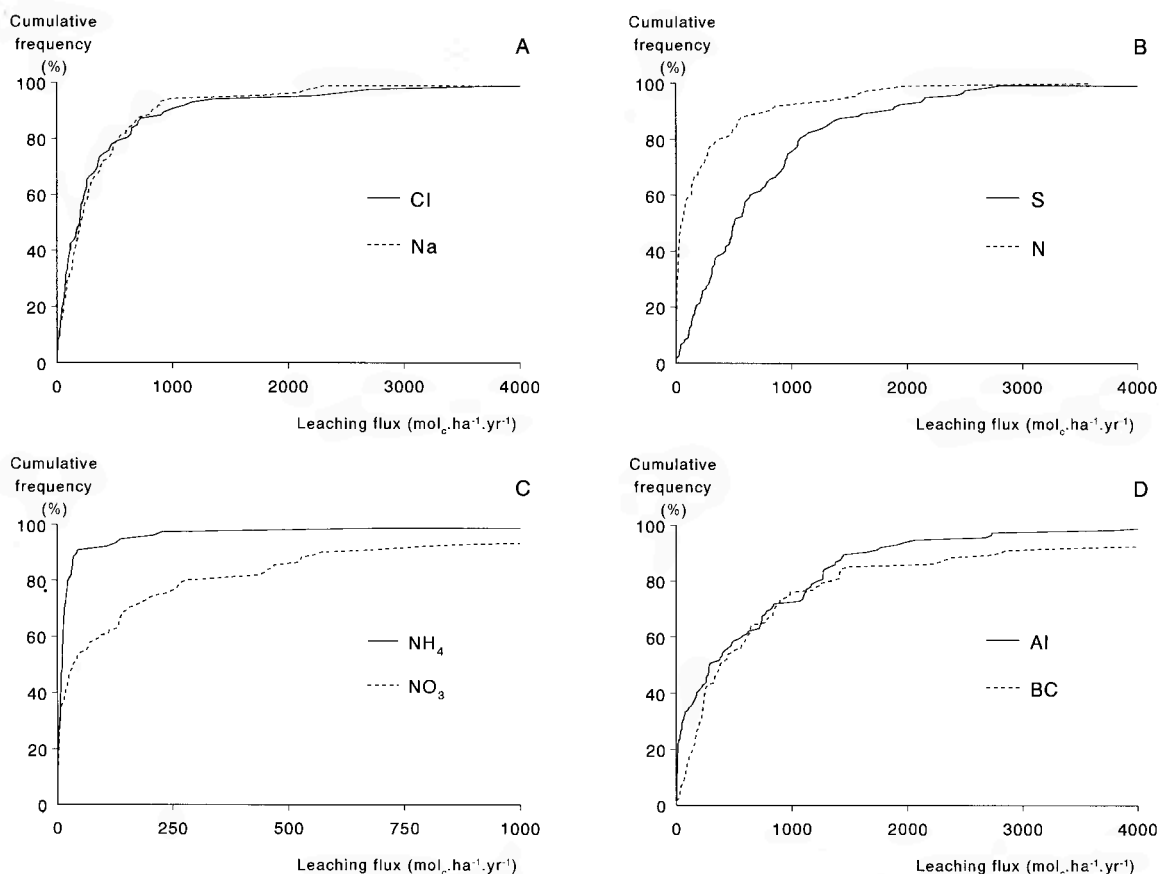


Figure 5.10 Cumulative frequency distribution (% plots) for the Cl and Na (A), S and N (B),  $\text{NH}_4$  and  $\text{NO}_3$  (C) and Al and BC (Ca+Mg+K) (D) leaching fluxes calculated using average concentrations.

Median leaching fluxes are lowest for nitrogen indicating that nitrogen is strongly retained in the soil or denitrified. A considerable difference in nitrogen fluxes is found between the different tree species. The lowest median N leaching flux is found under pine trees which is partly caused by the low hydrological leaching fluxes (Table 4.15) which also leads to low median leaching fluxes for Cl and Na under Pine. Nitrogen leaching fluxes under Spruce are also lower compared to the fluxes under the deciduous tree species.

The median leaching flux of base cations is generally higher than for aluminium, indicating that the annual average Al/BC ratio is generally less than 1.0, being considered as an average critical value with respect to impacts on roots (e.g. Sverdrup and Warfvinge, 1993). An exception is formed for Spruce where the median leaching flux of Al is almost twice as high as for base cations. The ratio between Al and BC in the leaching water does not directly give information on the ratio of Al and BC release by weathering and cation exchange buffering the soil system against acid atmospheric inputs. The reason is that Al input from the atmosphere is negligible, whereas this is not the case for base cations. More information on the latter aspect can be derived from the element budgets (Section 5.3.3).

### 5.3.2.3 Geographical variation in leaching fluxes

The geographic variation of the leaching fluxes is presented in Fig. 5.11 and 5.12.

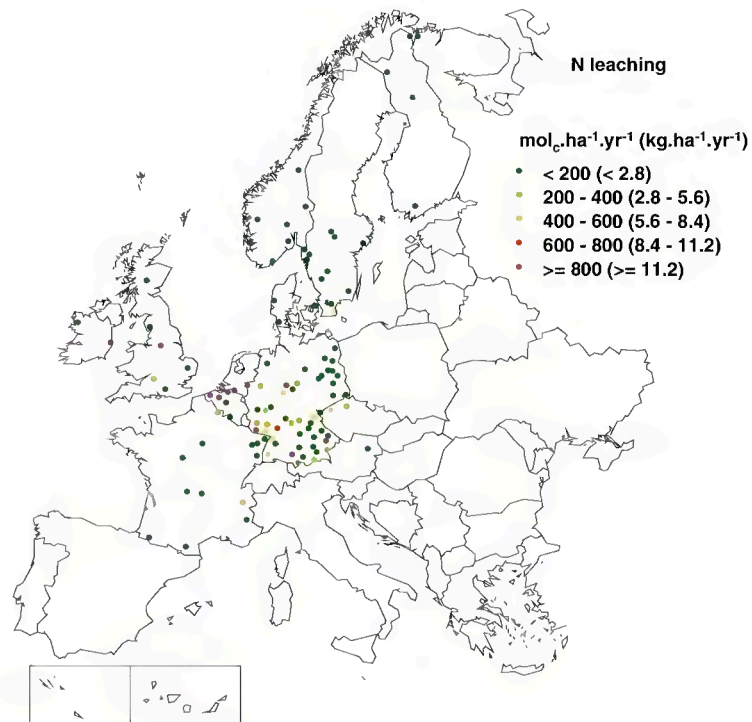
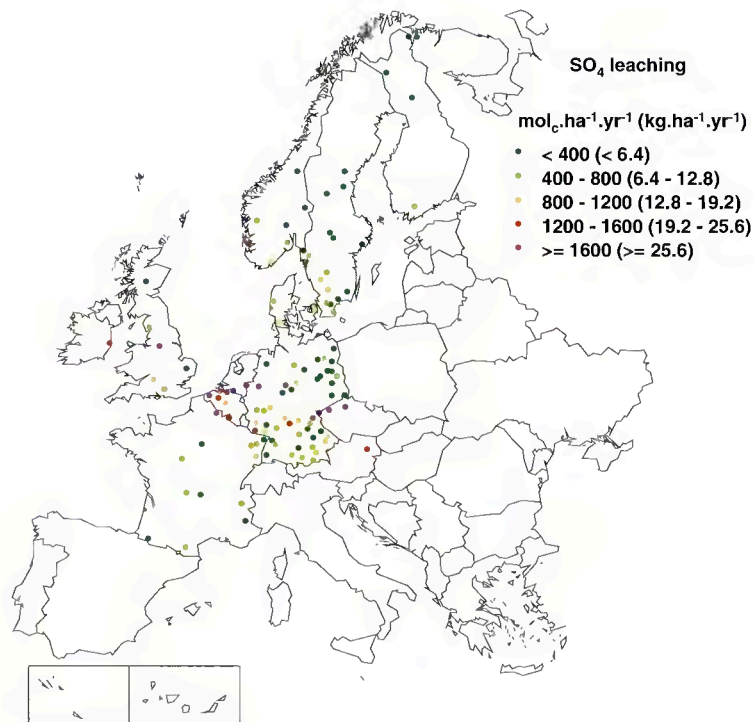


Figure 5.11 Geographical variation in leaching fluxes (mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) of sulphate (top) and nitrogen (bottom) at the Intensive Monitoring plots throughout Europe.

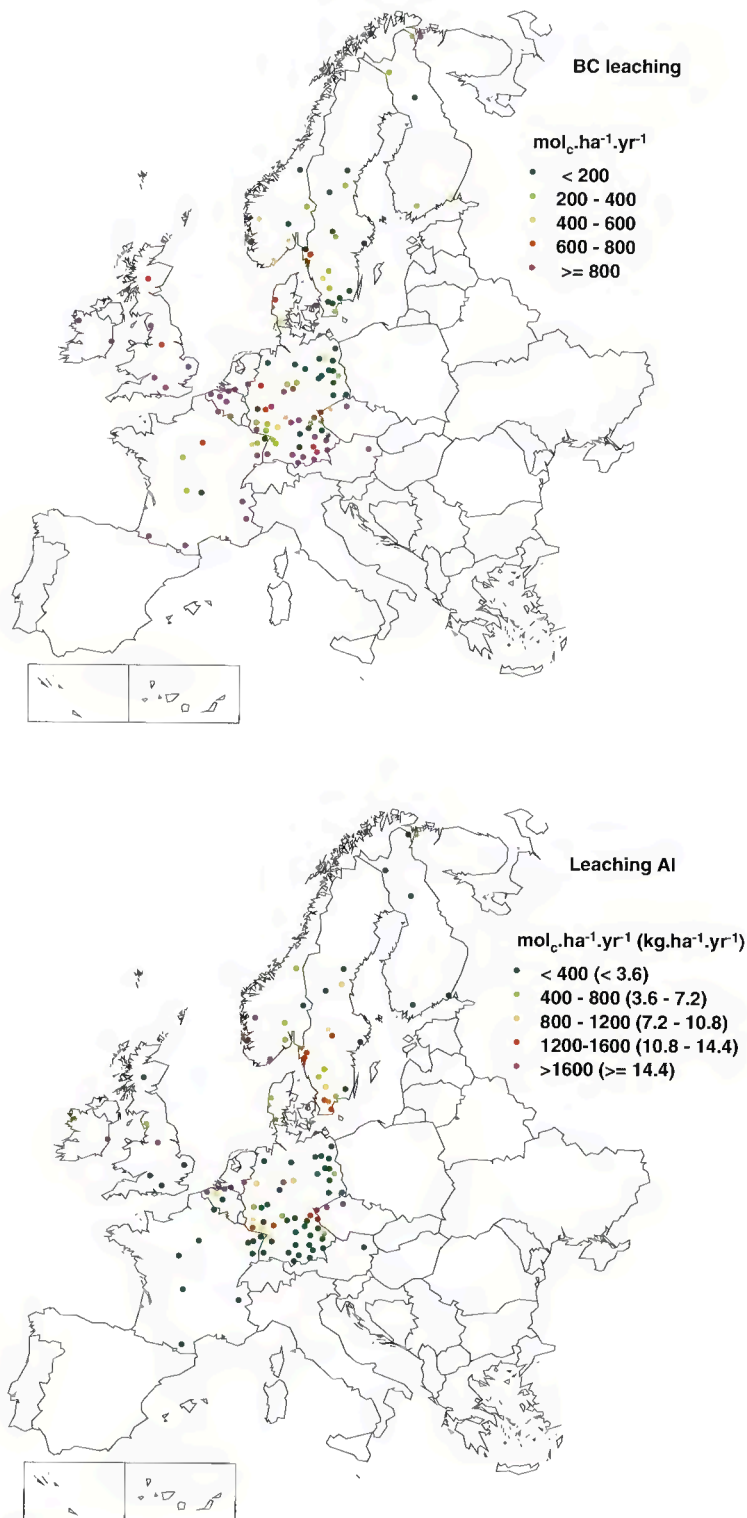


Figure 5.12 Geographical variation in leaching fluxes ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) of base cations Ca+Mg+K (top) and aluminium (bottom) at the Intensive Monitoring plots throughout Europe.

Extremely high Cl leaching fluxes (above 2000-3000  $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) mainly occur near the coast in Ireland and Denmark, where the input of Cl is very high. At more continental sites the Cl leaching



fluxes are generally less than  $500 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ . Very high leaching fluxes of  $\text{SO}_4$  mainly occur in Western and Central Europe (Belgium and parts of Germany and the Czech Republic), reflecting the high deposition at those sites. At part of those sites, the leaching flux of Al is also high indicating the occurrence of an acid soil releasing mainly Al in response to the high input (leaching) of  $\text{SO}_4$ . High N leaching fluxes ( $> 1000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ ) do occur in Belgium and central Germany where the input of N (specifically of  $\text{NH}_4$ ) is also high. Data for the Netherlands are not available due to the use of the centrifugation method and only one measurement per year, excluding the assessment of a reliable leaching flux. In northern Europe and in France N leaching fluxes are low ( $< 200 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ ). However, the geographic variation of N leaching is large (specifically in Germany), indicating that both N deposition and soil characteristics influence N leaching. The extremely high leaching fluxes for BC (above  $7000\text{-}8000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ ) all occur at near neutral or even calcareous sites in Central Europe, where the leaching of Ca is high due to natural decalcification. At these sites the leaching of bicarbonate (not shown) is also high.

### 5.3.2.4 Relationships between element leaching and environmental factors

The range in S and N leaching is generally comparable to the range in S and N deposition as illustrated in Figure 5.13.

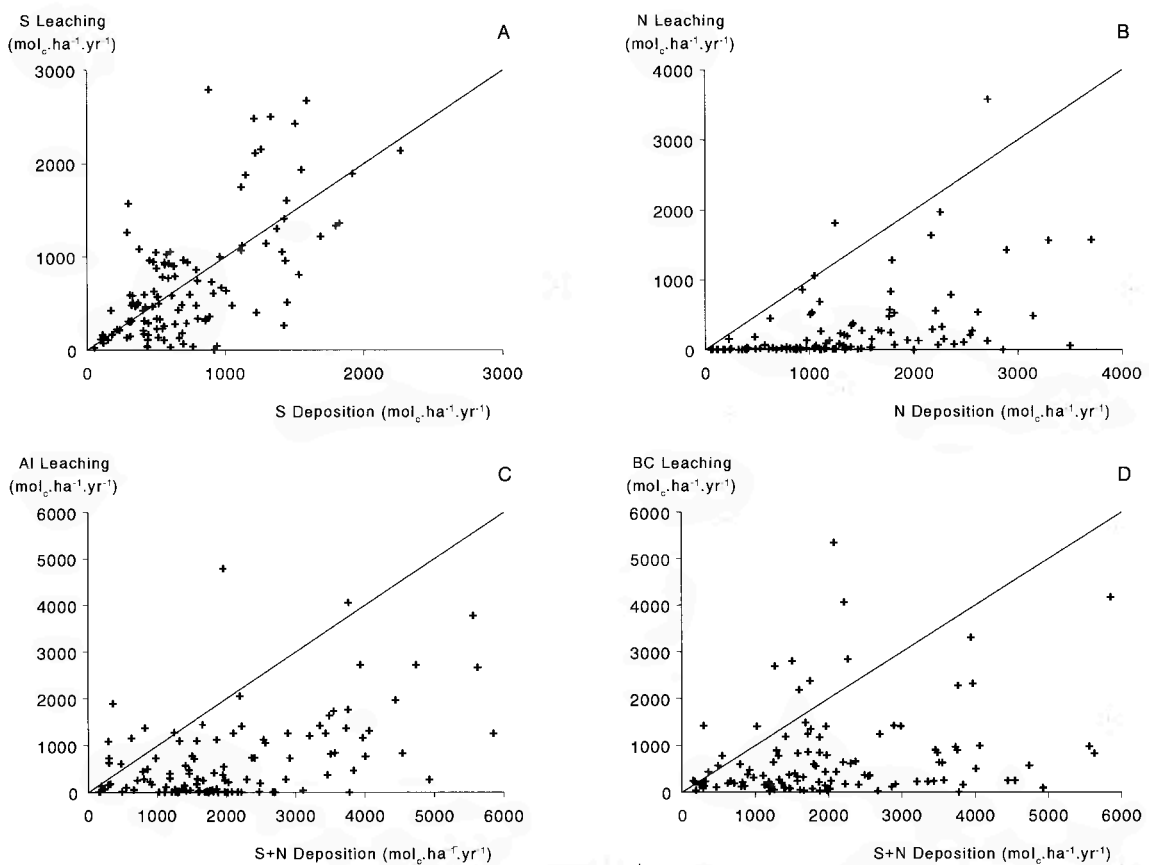


Figure 5.13 Relations between total deposition and leaching fluxes of S (A) and N (B) and between the Al leaching fluxes and the total deposition flux (using the deposition model including regression equations to estimate canopy uptake) of S+N (C) and BC leaching and S+N deposition (D) at the 121 monitoring sites.

On average S leaching is close to S deposition, but there is a large variation that can partly be attributed to errors in both the input and output assessment (Fig. 5.13A). At five sites the leaching of SO<sub>4</sub> is considerably higher (approximately 3000-5000 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) than the deposition (approximately 1500 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>), indicating a strong release of SO<sub>4</sub>. Most of these sites are located in areas that received a high sulphur deposition over the past decades (Czech-German border). The present high leaching fluxes at these sites are probably due to the release of sulphur, which has been adsorbed during previous decades.

In accordance with results found by e.g. Dise et al. (1998a, b) and Gundersen et al. (1998a), the leaching of N is generally negligible below a throughfall input of 10 kg.ha<sup>-1</sup>.yr<sup>-1</sup>. At N throughfall inputs above 10 kg.ha<sup>-1</sup>.yr<sup>-1</sup>, leaching of N is generally elevated, although lower than the input indicating N retention at most of the plots. At 2 sites, however, N leaching is larger than the N input (Fig. 5.13B), indicating the occurrence of a disturbance in the N cycle. The range in Al leaching fluxes is quite comparable to the range in throughfall S and N fluxes (Fig. 5.13C). At several sites, Al leaching fluxes are even higher than the acid deposition, indicating that the leaching of sulphate is higher than the input (S release from the soil causing acidification) and this acid input is almost completely buffered by the release of Al. At all other sites the Al leaching flux is lower, indicating that part of the potential acid input is buffered by N retention and/or base cation release. In soils with a pH above 5.0, the release of Al is generally negligible, independent of the S and N input, since BC release by weathering and cation exchange buffers the incoming net acidity in those soils. This is illustrated by a site with a high input of S and N by throughfall (near 8000 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>) and hardly Al leaching. The range in BC leaching fluxes is quite comparable to the range in Al leaching fluxes (Fig. 5.13D). As with Al, at several sites BC leaching fluxes are higher than the acid deposition. Apart from the possibility of S release from the soil, this is mostly due to natural acidification by bicarbonate leaching.

To gain insight in the various factors affecting element leaching, a multiple regression analyses was carried out (see Section 5.2.5). Results are presented in Table 5.9.

Table 5.9 Overview of the predictor variables explaining element leaching at 97-112 Intensive Monitoring plots with the number of plots and the percentage variance accounted for.

Predictor variables	S	N	BC	Al
<i>Site/Stand characteristics</i>				
Tree species	+			
Humus type group				+
Stand age	+			
<i>Deposition</i>				
S	++			++
N		++	+	
FrNH <sub>4</sub>				
<i>Foliage</i>				
N-content		+		
<i>Soil chemistry</i>				
pH subsoil			++	
N <sup>1)</sup>	112	97	109	106
R <sup>2</sup> <sub>adj</sub> (%) <sup>2)</sup>	46	24	30	35

++ significant at the p 0.01 level and t>3 and positive correlated with response variable

+ significant at the p 0.05 level and t>2 and positive correlated with response variable

-- significant at the p 0.01 level and t>3 and negative correlated with response variable

- significant at the p 0.05 level and t>2 and negative correlated with response variable

<sup>1)</sup> N = number of plots.

<sup>2)</sup> R<sup>2</sup><sub>adj</sub> = percentage variance accounted for.

The results show that variations in the element leaching fluxes are often related to the atmospheric deposition (the S input in case of Al being the major source of acidification).  $\text{SO}_4$  leaching was related to the deposition and was higher in mature stands (> 30 year old) compared to young stands. The relation with stand age may be explained by the fact that the cumulative sulphur accumulation in the soils of the young stands is less compared to the mature stands. Sulphur deposition was also (weakly) related to the tree species, being lowest under Pine trees, somewhat higher under Spruce and highest under deciduous forest. The reason for this relationship with tree species is unclear and may be an artefact of regional differences in S leaching.

N leaching is positively related to the N content in the foliage and the total N deposition. There was no significant correlation between the calculated  $\text{NO}_3$  and  $\text{NH}_4$  (and total N) leaching flux and the C/N ratio of the organic or mineral layer, which is contradictory to the results presented by Dise et al. (1998a, b) and Gundersen et al. (1998a).

A visual inspection of the relationship between N leaching and N deposition against C/N ratio of the organic layer and mineral soil, however, showed that soils with very high C/N ratios (> 30) in the organic layer tend to have lower leaching fluxes in particular when N deposition is low (Fig. 5.14). An exception are two plots with C/N ratios near 35 and an N leaching flux near 1500  $\text{mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$  (Fig. 5.14A). Below C/N ratios of 30, however, the scatter is quite large. For example, the N leaching ranges between 150 and 1500  $\text{mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$  at a C/N ratio of the organic layer of approximately 25 and a N deposition of more than 2000  $\text{mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ . We did find, however, a statistically significant impact of the C/N ratio on soil  $\text{NO}_3$  concentrations for approximately 240 Intensive Monitoring plots (see Annex 6). A more in-depth analysis, based on (much) more input-output budgets of N is thus necessary to further investigate the possible role of the C/N ratio of the soil on the N dynamics.

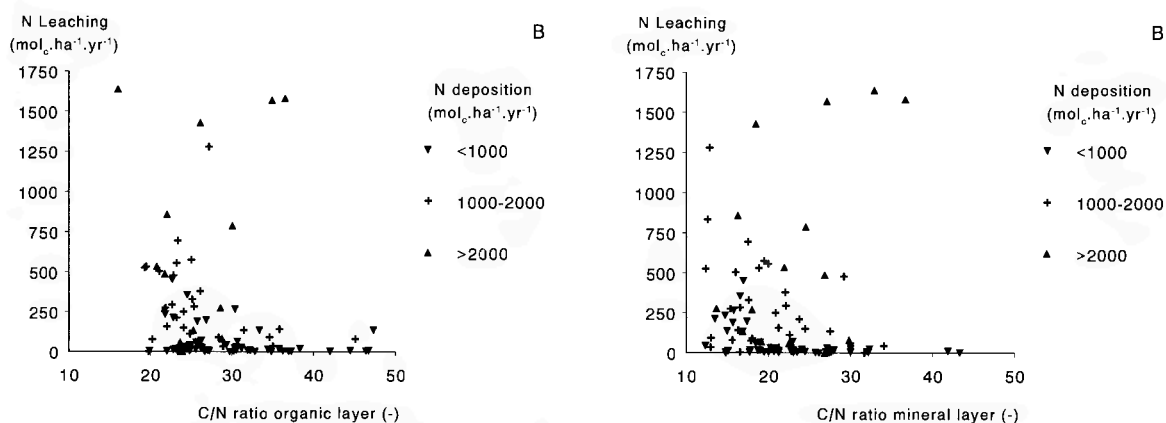


Figure 5.14 Relationships between N leaching and the C/N ratio of the organic layer (A) and mineral layer (B).

Variations in BC leaching were significantly related to the N input, but also the pH of the subsoil. Al leaching shows a positive relation with sulphur deposition and was higher in mull type humus compared to mor and moder type humus. This indicates that the solubility of Al is probably lower in soils with a mor type humus compared to more strongly decomposed humus types.

### 5.3.3 Element budgets

#### 5.3.3.1 The effect of the use of interpolated meteorological data on the calculated element budgets

The use of interpolated data instead of local measured meteorological data leads to an underestimation of the calculated leaching fluxes (cf. section 4.3.2.1 and section 5.3.2.1). This affects the calculated budgets in particular for elements where the output flux is comparable or higher than the input. This is shown in Fig. 5.15 where the budgets based on interpolated data are plotted against the budgets based on local data.

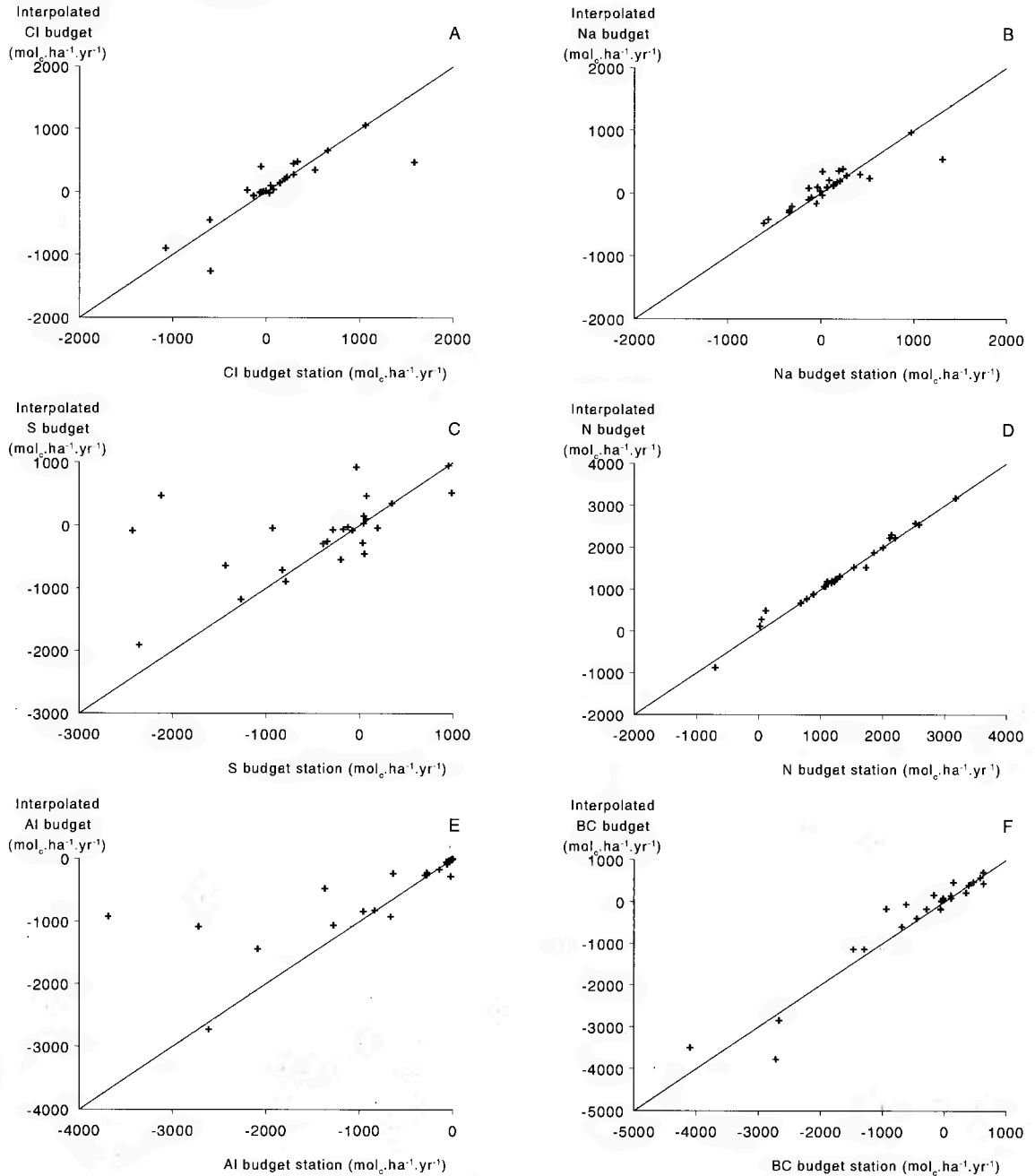


Figure 5.15 Element budgets based on interpolated meteorological data compared to local meteorological data for Cl, Na, S, N, Al and BC.

Differences in nitrogen budgets (an element that is strongly retained in the soil) are rather small, whereas differences in sulphur, aluminium or base cation budgets are much larger. The difference in median element budgets ranges from 12 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for N to 188 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations (Table 5.10). The difference in impact of the deviation in leaching fluxes on the calculated budgets is clearly reflected in the relative deviation of the budgets for the various elements. Nitrogen budgets are hardly affected at more than 50% of the sites, whereas S, Cl, Na and BC budgets are more strongly affected (deviation of more than 20% at 50% of the sites).

Table 5.10 Element budgets calculated using local and interpolated meteorological data and median, 10% and 90% differences in calculated budgets.

Element	Element budget (mol <sub>c</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )						Deviation		
	local meteorological data			interpolated meteorological data			(interpolated-local)/local (%)		
	10%	50%	90%	10%	50%	90%	10%	50%	90%
Cl	-600	36	603	-256	93	568	3	55	150
Na	-430	19	464	-285	117	450	2	33	330
S	-1776	-149	271	-800	-67	500	6	60	710
N (model)	115	973	2021	127	950	2067	0	2	120
N (regr.)	75	1222	2397	234	1200	2428	0	2	220
BC (model)	-3270	-166	516	-3605	22	517	0	38	213
BC (regr.)	-2991	-113	638	-3231	10	666	0	23	135
Al	-2347	-210	-5	-1068	-225	-4	4	21	66

### 5.3.3.2 Ranges in element budgets

Ranges in element budgets for chloride, sulphate, nitrate, base cations (Ca+Mg+K) and aluminium for the 121 monitoring sites are presented in Figure 5.16 and Table 5.11.

For most sites the Cl budget is close to zero: 80% of the sites have a Cl budget between -60 and 600 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>. This is quite close to values for the Cl budget at the Solling spruce stand for which long term measurements are available, At this site Cl budgets ranged from -560 to 830 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for a one year period (Van der Salm, in prep). For a four year period this range was somewhat smaller (-280 to 101 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>).

The median sulphur budget is also close to zero. However, the range in sulphur budgets is much broader compared to chloride. Budgets for 80% of the sites range from -700 to 550 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup>. At a considerable number of sites sulphur is released from the soil, indicating that these system are recovering from previous episodes of high sulphate input. This is affirmed by the geographical differences in sulphur budgets (Section 5.3.3.2).

Nitrogen budgets (NH<sub>4</sub> + NO<sub>3</sub>) range from 60 to 1800 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> when the canopy exchange model is used to calculate canopy exchange and total deposition. A slightly broader range is found when the regression model is used, due to the fact that the regression model yields a somewhat higher canopy uptake of N in areas with a relatively low N deposition, whereas differences are negligible in areas with a high N deposition. At most sites (85%) the N input is higher then the N leaching.

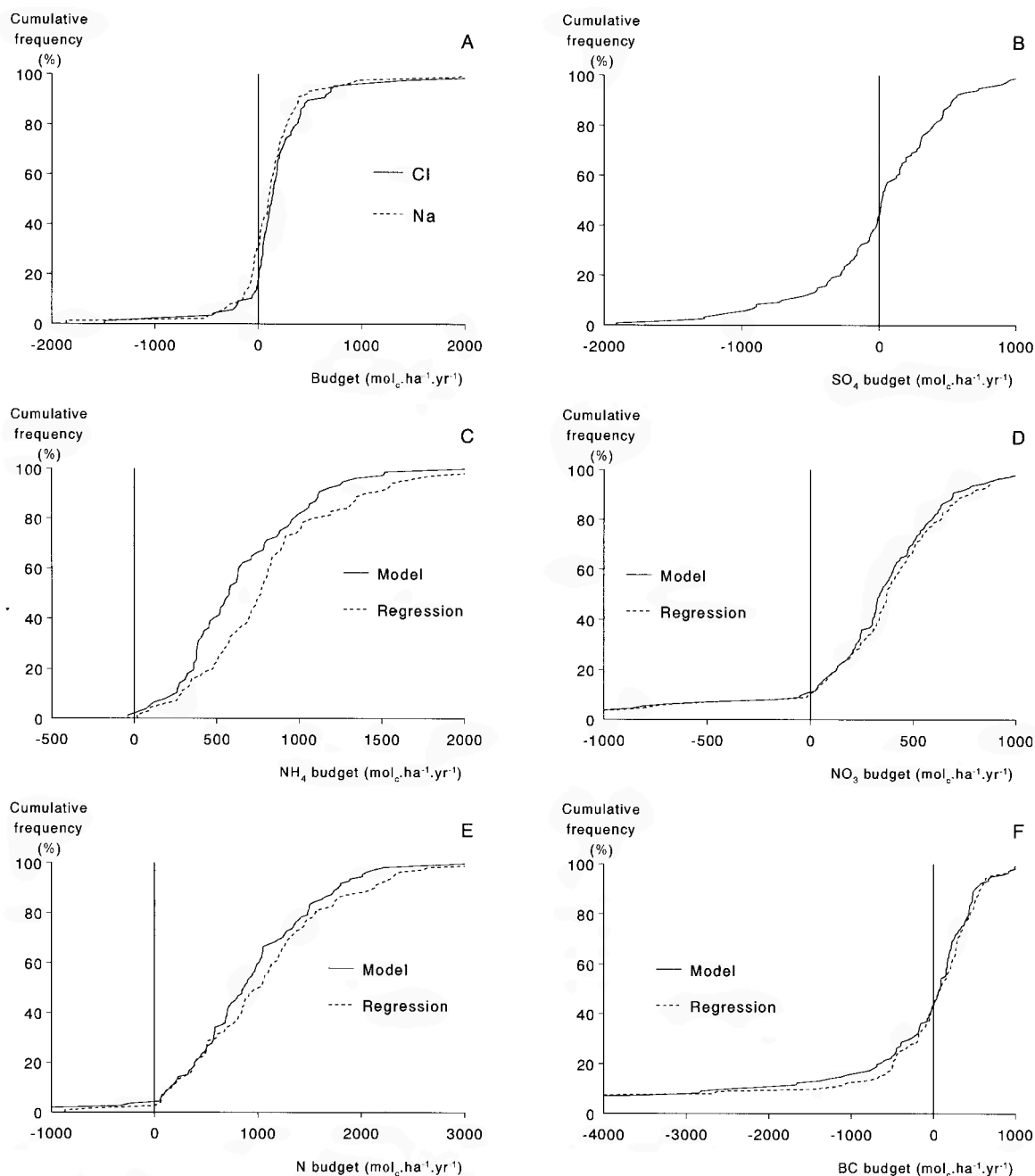


Figure 5.16 Cumulative frequency distribution (% plots) for the Cl and Na (A), S (B), NH<sub>4</sub> (C), NO<sub>3</sub> (D), N (E) and BC (Ca+Mg+K) (F) budgets calculated using (average) measured concentrations.

Table 5.11 Element budgets for chloride, sodium, sulphate, nitrogen, base cations (Ca+Mg+K) and Aluminium.

Element	Number of sites	Element budget (mol <sub>e</sub> .ha <sup>-1</sup> .yr <sup>-1</sup> )			Behaviour	
		10%	50%	90%	Retention	Release
Cl	87	-60	135	614	83	17
Na	89	-141	99	391	70	30
S	120	-672	21	540	57	43
N (regression)	111	169	1028	2129	97	3
N (model)	111	159	871	1787	96	4
BC (regression)	112	-1307	86	576	57	43
BC (model)	121	-2240	86	517	57	43
Al	114	-1648	-294	-7	1	99

At 45% of the sites the BC balance is positive (retention) indicating that the input of base cations by throughfall, weathering, mineralisation and ion-exchange is higher than the uptake by the plant. Some sites on highly weatherable parent material (chalk, gypsum etc.) have a strong negative BC balance (up to  $-15000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ ).

The Al balance is always negative (Al is leached) because the input of Al is negligible. The median Al balance is  $-300 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ . This corresponds to a median leaching concentration of  $0.25 \text{ mol}_c.\text{m}^{-3}$ , which is above a critical limit of  $0.2 \text{ mol}_c.\text{m}^{-3}$ , indicated in the literature with respect to impacts on roots for the most sensitive tree species (Cronan et al., 1989).

Median element budgets for chloride, sulphate, nitrate, base cations (Ca+Mg+K) and aluminium for the different tree species at the 121 monitoring sites are presented in Table 5.12. The median budget for Cl and Na is slightly positive, deposition exceeds leaching except for oak. The sulphur budget is close to zero in coniferous forests, in deciduous forest the median sulphur budget is slightly negative. Nitrogen is strongly retained in a large part of the sites and accordingly the median nitrogen budget is positive for all tree species. The median nitrogen budget is approximately  $300 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$  higher when canopy exchange is calculated by the regression model due to a lower canopy uptake estimated by the canopy exchange model (cf. 5.3.12). The median base cation budget is zero to slightly positive for the coniferous tree species and negative for the deciduous species. The high values for base cations budgets for soil under deciduous forest are partly due to the occurrence of calcareous soils.

Table 5.12 Median element budgets for chloride, sodium, sulphate, nitrogen, base cations (Ca+Mg+K) and Aluminium using the canopy exchange model. Values in brackets are the median input fluxes using the regression equations to calculate the input.

Tree species	Number of sites	Element budget ( $\text{mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ )							
		Cl	Na	S	N	BC	Al		
Pine	29	211	204	216	703	(861)	253	(210)	-138
Spruce	51	167	121	16	1040	(1094)	94	(216)	-774
Oak	15	-37	-86	-256	686	(846)	-911	(-445)	-30
Bccch	20	113	37	-22	984	(1225)	30	(53)	-326
Other	6	102	19	28	772	(782)	-862	(-864)	-31
All	121	135	99	21	871	(1028)	86	(86)	-294

In general, the range in element release is comparable to element leaching. This broad range in budgets is partly due to the wide geographic range in locations leading to very diverse circumstances with respect to deposition, hydrology and bio-geochemical processes. However, it has to be stated that the budgets are based on measurements during a relatively limited number of years. For 14% of the sites data for a four year period (1995-1998) where available. For most sites (58%) budgets were limited to the period 1996-1998. For 20% of the sites two years of measurements were available, whereas for 8% of the sites budgets are based only on data from 1999. This relatively short time span may lead to over- or underestimation of the budget compared to the long-term situation due to particular hydrological or biological circumstances in specific years. For example, the extraordinary high leaching of sulphur and nitrogen at two sites is caused by extreme leaching in 1998, whereas the other years have a much lower release of sulphur and nitrogen.

### 5.3.3.3 Geographic variation in element budgets

Extreme values for the Cl budget are mainly found in coastal areas (e.g. Ireland, western France) where the input of Cl is extremely high (Fig. 5.17). Rather high values for the Cl budget ( $> 600 \text{ mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) are also found for a number of sites at the Czech-German border. At these sites the input of chloride was up to three times higher compared to other sites in Germany. Sites with the highest sulphur release are located in central Europe, where the strongest reduction in sulphur deposition has taken place over the last decade.

Sites with a net release of nitrogen are found in Belgium and Northwestern Germany (Fig. 5.18). This corresponds to the area which have received a high N deposition over a prolonged period of time. Remarkable is the high N retention in Southeastern Germany. According to the present calculations these sites still retain a lot of nitrogen despite relatively high depositions of N. This discrepancy may be explained by a very long period (centuries) of intensive use of litter on these poor soils until the 1950th and a therefore still existing deficit in the N budget. High BC release values are generally found in areas with a high N or S deposition such as Belgium, Northwestern Germany and the area around the German-Czech border. The Al budget is equal to Al leaching (Fig. 5.12) and no map is therefore given here.

### 5.3.3.4 Relationships between element budgets and environmental factors

Results of the multiple regression analyses relating element budgets to environmental factors are presented in Table 5.13.

Table 5.13 Overview of the predictor variables explaining element budgets at 73-112 Intensive Monitoring plots with the number of plots and the percentage variance accounted for.

Predictor variables	S	N	BC
<i>Site/Stand characteristics</i>			
Tree species	-		
Stand age	-	+	
<i>Deposition</i>			
N		++	-
<i>Foliage</i>			
N-content		--	
<i>Soil chemistry</i>			
Organic C pool		-	
pH subsoil			--
N <sup>1)</sup>	112	73	109
R <sup>2</sup> <sub>adj</sub> (%) <sup>2)</sup>	9	61	29

++ significant at the p 0.01 level and  $t > 3$  and positive correlated with response variable

+ significant at the p 0.05 level and  $t > 2$  and positive correlated with response variable

-- significant at the p 0.01 level and  $t > 3$  and negative correlated with response variable

- significant at the p 0.05 level and  $t > 2$  and negative correlated with response variable

<sup>1)</sup> N = number of plots.

<sup>2)</sup> R<sup>2</sup><sub>adj</sub> = percentage variance accounted for.



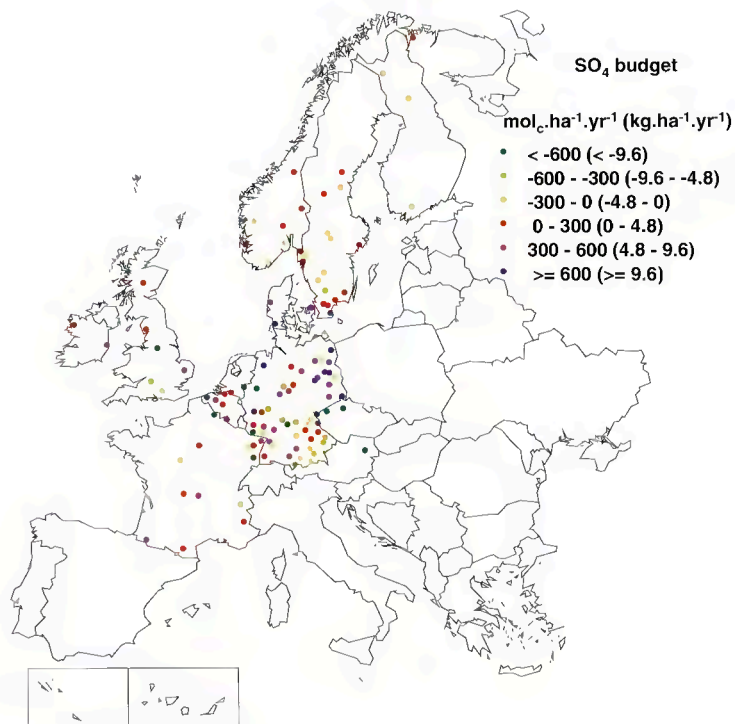
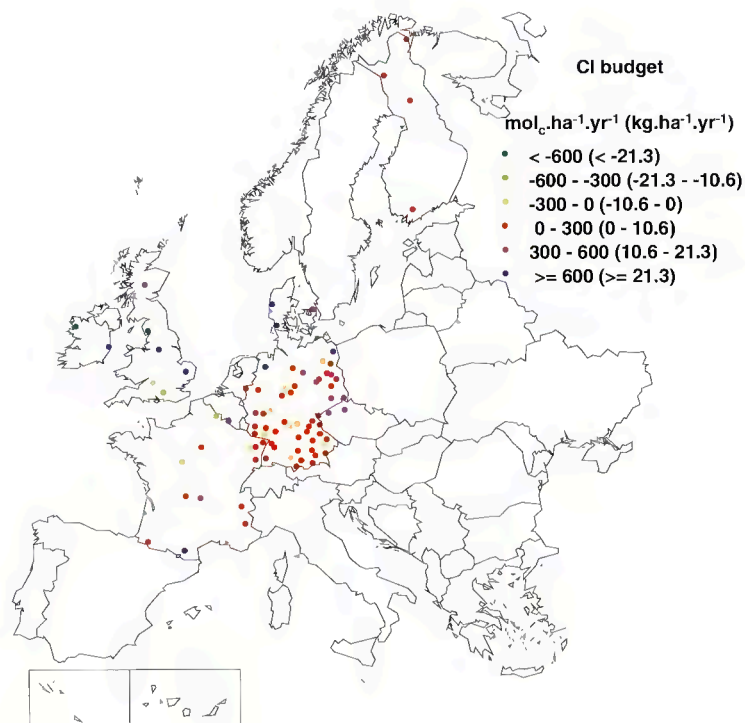


Figure 5.17 Geographical variation in budgets ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) for chloride (top) and sulphate (bottom) at the Intensive Monitoring plots throughout Europe.

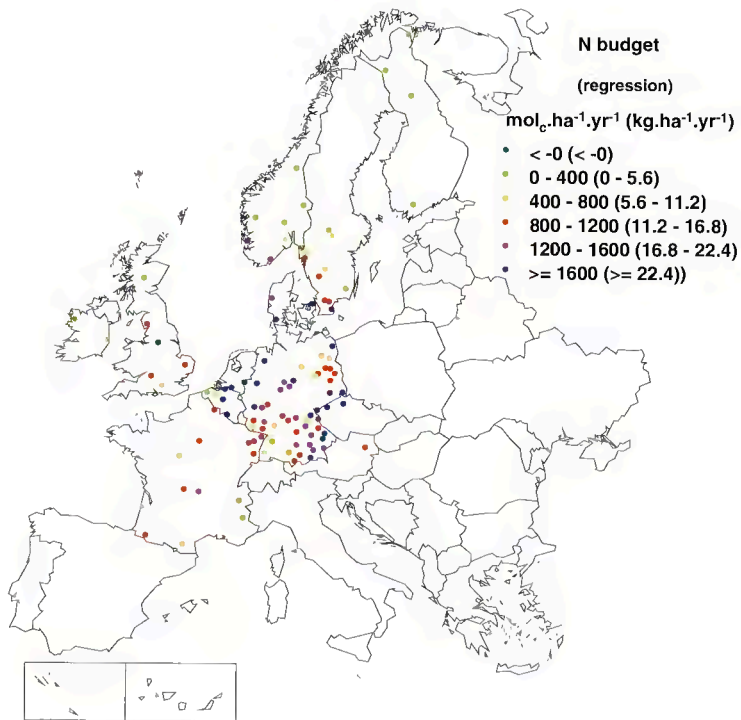
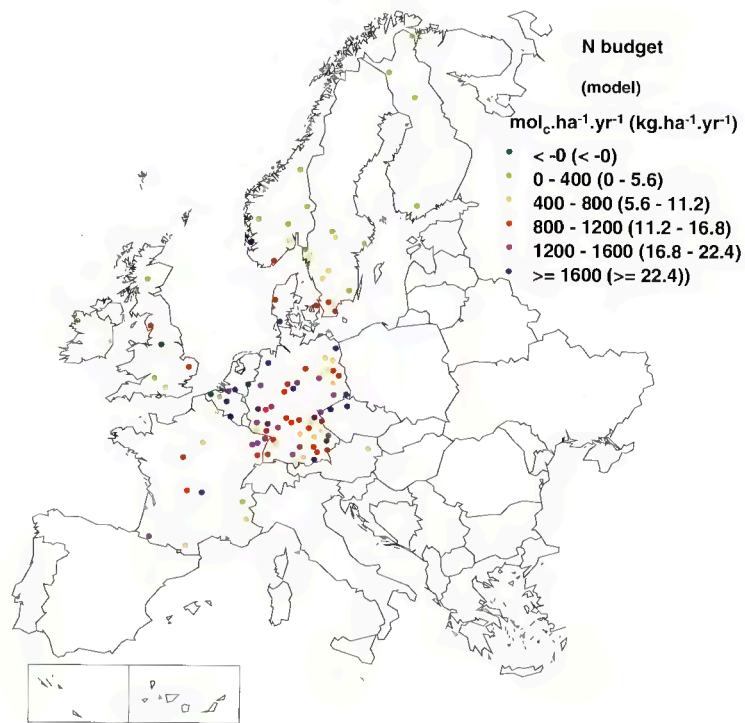


Figure 5.18 Geographical variation in budgets ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) for nitrogen using total deposition based on the canopy exchange model (top) and regression equations to calculate canopy exchange (bottom) at the Intensive Monitoring plots throughout Europe.

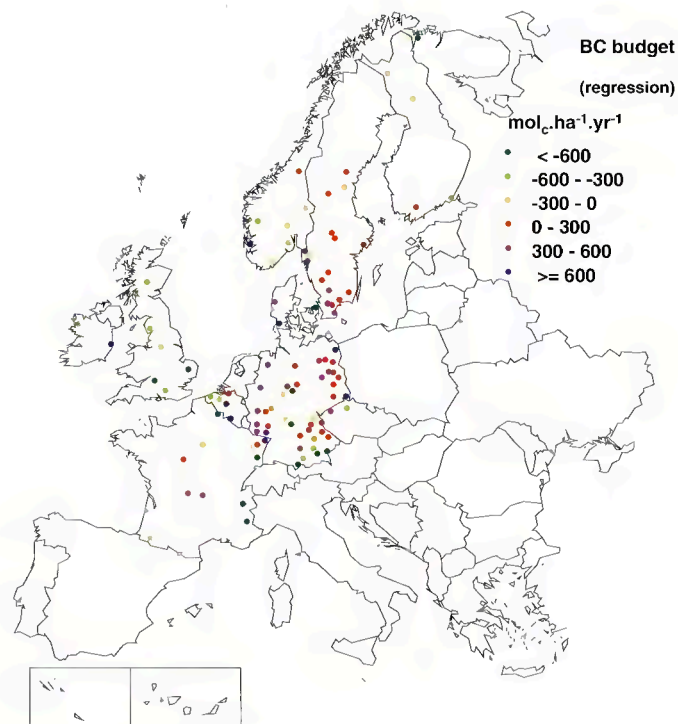
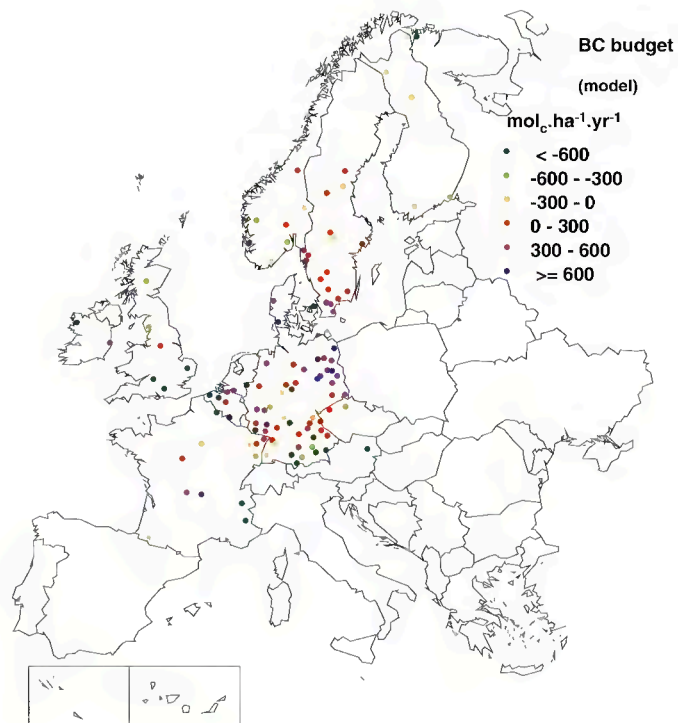


Figure 5.19 Geographical variation in budgets ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) for base cations (Ca+Mg+K) using total deposition based on the canopy exchange model (top) and regression equations to calculate canopy exchange (bottom) at the Intensive Monitoring plots throughout Europe.

The results show that S retention from the soil can hardly be explained from environmental variables. N retention increases with an increased stand age and N deposition and a decreased organic C pool and foliar N content. As with N leaching, the N retention fraction was not significantly related to the C/N ratio of the organic or mineral layer. A plot of the N retention fraction against the C/N ratio of the organic and mineral layer, however, shows a tendency of lower N retention fractions at low C/N ratios and high deposition levels but the scatter is large (Fig. 5.20). As expected, variations in BC release from the soil were positively to the pH (high weathering rates in soils with a high pH). Al release is equal to Al leaching and consequently, we refer to Table 5.10 for the effects of environmental variables on the Al budget.

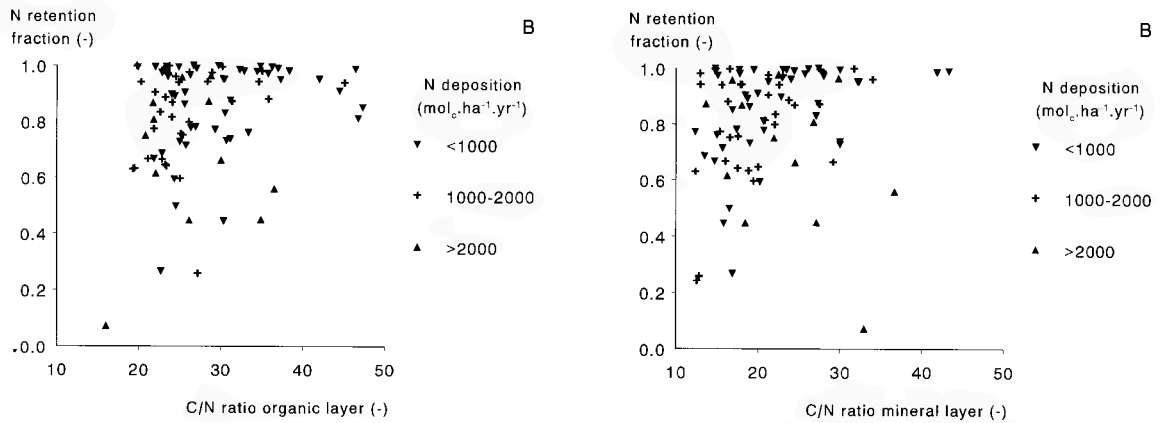


Figure 5.20 Relationships between N retention fraction and the C/N ratio of the organic layer (A) and mineral layer (B).

## 5.4 Conclusions

The following conclusions can be drawn with respect to the leaching and retention of the different elements:

- The use of interpolated meteorological data instead of local measured data leads to lower leaching fluxes. The deviation in median leaching fluxes is quite limited and ranges from 15 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for Al up to 73 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations. However, the (absolute) deviation in leaching fluxes at the individual sites is somewhat larger and ranges from 20 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for (a median) N leaching flux up to 180 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> at a medium S leaching flux. The lower leaching fluxes lead to a difference in median element budgets between 12 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for N and 188 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations.
- At nearly all plots, the leaching flux of SO<sub>4</sub> is much higher than that of NO<sub>3</sub> indicating that SO<sub>4</sub> is still the dominant source of actual soil acidification. The median sulphur budget is close to zero, but on average, S leaching is higher than S deposition. The median budget of zero does not imply tracer behaviour of S, since there is a large variation. Very high leaching fluxes of SO<sub>4</sub> mainly occur in Western and Central Europe (Belgium and parts of Germany and the Czech Republic), reflecting the high deposition at those sites. At a considerable number of those sites, sulphur is released by the soil, indicating that these systems are recovering from previous episodes of high sulphate input.
- The leaching of N is generally negligible below throughfall inputs of 10 kg.ha<sup>-1</sup>.yr<sup>-1</sup>. At sites with throughfall inputs above this level, leaching of N is generally elevated, although lower than the input indicating N retention at most of the plots. The significant relationship of N leaching with throughfall is reflected by the fact that highest N leaching fluxes do occur in

areas where the input of N (specifically of  $\text{NH}_4$ ) is also high. Nitrogen budgets show that at most sites (90%) the N input is higher than the N leaching. Despite the large variation in N leaching at comparable N inputs no significant relationships were found with either stand and site characteristics or with the soil C/N ratio. We did find, however, a statistically significant impact of the C/N ratio on soil  $\text{NO}_3$  concentrations. A more in-depth analysis, based on (much) more input-output budgets of N is thus necessary to further investigate the possible role of the C/N ratio of the soil on the N dynamics.

- At most of the plots, the leaching flux of base cations ( $\text{Ca}+\text{Mg}+\text{K}$ ) is higher than that of Al. Extremely high leaching fluxes for BC (above  $7000\text{-}8000 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ ) all occur at near neutral or even calcareous sites in Central Europe, where the leaching of Ca is high due to natural decalcification. Variations in BC leaching were significantly related to the S input and also to the pH and base saturation. The median base cation balance is close to zero, implying a net adsorption and a net release of base cation at approximately 50% of the plots.
- With the exception of some strongly acidified sites, the Al leaching flux was generally lower than the atmospheric input of S and N, indicating that part of the potential acid input is buffered by N retention and/or base cation release. The Al leaching flux was significantly related to the  $\text{SO}_4$  input (and leaching) reflected by the fact that sites with a high Al leaching coincide with sites with a high input (leaching) of  $\text{SO}_4$ . The geographic patterns of both elements, however, did not coincide very well since soil base saturation was also significantly related to the Al leaching flux. The median Al balance is  $370 \text{ mol}_c.\text{ha}^{-1}.\text{yr}^{-1}$ . High values are generally found in areas with a high N or S deposition.

## 6 Discussion and conclusions

Conclusions on the evaluations carried out in this year's Technical report were presented in the last sections of each chapter. Here we summarise those conclusions, focusing on the validity of the presented results, related to the species composition of the ground vegetation and water and element fluxes through the forest ecosystem.

### *Species composition of the ground vegetation*

The species composition of the ground vegetation, which is assessed at Intensive Monitoring plots, is an indication of the floristic biodiversity of forest ecosystems and can thus contribute to the ongoing discussion of the causes of biodiversity losses. Presently, information on plant species and their abundance in the ground vegetation are available for more than 400 plots in combination with environmental factors influencing those species, such as stand and site characteristics, climatic variables, soil properties and atmospheric deposition. Possibilities to do in-depth evaluations with those data are considerable and results of such investigations are aimed for next years report. In this years report the evaluations focused on a straightforward presentation of the number of species and the abundance weighted species diversity per plot, while checking for data comparability in relation fenced and unfenced plots and differences in sample area.

Species numbers of the vascular plants varied between 1 and 124 with a median value of 24. The numbers showed a North-South gradient with increasing species numbers in the Mediterranean areas compared to the boreal forests, except for some plots in Norway and Finland. In Poland, the species numbers increased from West to East. The species diversity based on abundance weighting according to the Simpson index varied strongly within countries and there were no clear gradients over countries. High Simpson indices (high diversity) were, however, often associated to plots with a high number of species and very low Simpson indices (low diversity) to plots with only a few species.

Results showed a high comparability of species numbers inside and outside a fence when the same data assessment methods are used. Apparently, effects of fencing due to the exclusion of grazing did not yet take place in the relatively short time period between fencing and data assessment. Results of a regression analysis also showed that the influence of sample area on species numbers was not significant, when accounting for differences in climatic zone and tree cover indexes on the species numbers. An unbiased assessment of the impact of sample area on the species number requires an assessment of the species number for various sample areas. Such an analysis can only be carried out at a limited number of plots for which such data are available.

### *Water fluxes through the forest ecosystem*

Leaching fluxes mainly reflected the difference in throughfall on forest stands. Median values increased going from approximately 80 mm under Pine stands to approximately 210 mm under Spruce stands. The strong relationship between annual throughfall and leaching is due to the calculated narrow range in the sum of soil evaporation and transpiration, with median values ranging from approximately 400-450 mm.yr<sup>-1</sup> for the various tree species. Plots with the lowest leaching fluxes are thus located in areas with relatively low precipitation such as north-eastern Germany, parts of Sweden and Finland and locally in southern Europe.

Simulated water fluxes through the forest ecosystem were generally plausible in view of available measurements and literature data. Simulated yearly throughfall values were within 5% of the measurements at 85% of the monitoring sites. Simulated transpiration fluxes and leaching fluxes could not be validated but detailed studies on two sites in Germany and the Netherlands indicated that the model was able to simulate changes in soil water contents quite well. Simulated yearly interception evaporation was well within the range observed in literature. The increase in average interception fractions going from Oak < Pine < Beech < Spruce from approximately 0.2 to 0.3 is well in line with the reported increasing interception capacity of those tree species (e.g. a literature compilation by Hiege, 1985). The simulated relatively constant transpiration fluxes among the tree species, with median values ranging from 314 mm.yr<sup>-1</sup> for Pine to 385 mm.yr<sup>-1</sup> for Spruce stands, is also consistent with literature data. A literature compilation by Roberts (1983) indicates that transpiration fluxes for European forest are in a very narrow range around 335 mm.yr<sup>-1</sup> due to feedback mechanisms with soil and atmosphere. The simulated evapotranspiration fluxes for the 245 sites were also in the range of data reported in the literature.

The above mentioned results were obtained using interpolated daily data for precipitation, relative humidity, temperature, net radiation and wind speed. A comparison of measured and interpolated meteorological data showed good agreement for relative humidity, reasonable agreement for temperature and net radiation and poor agreement for wind speed. The effect of using interpolated data instead of on-site measured data led to an overestimation of the simulated transpiration fluxes at 80% of the sites. However, at 50% of the sites, the effect is quite limited (deviation < 37mm). Due to the overestimation of the transpiration fluxes, leaching fluxes tend to be underestimated at 85% of the sites when interpolated meteorological data are used. At 80% of the sites average transpiration fluxes were overestimated when interpolated data were used. The median difference in simulated transpiration fluxes was 45 mm. The observed differences in simulated leaching fluxes were comparable to the differences in transpiration fluxes. The use of interpolated meteorological data instead of local measured data leads to lower element leaching fluxes. The deviation in median leaching fluxes is quite limited and ranges from 15 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for Al up to 73 mol<sub>c</sub>.ha<sup>-1</sup>.yr<sup>-1</sup> for base cations.

### ***Element fluxes through the forest ecosystem***

Element budgets for sulphur, nitrogen, base cations and aluminium clearly reflected the behaviour of those elements in response of atmospheric deposition. Median values for S leaching were close to the median S deposition. On a considerable number of sites S leaching was, however, higher than S deposition leading to a higher average S leaching than S deposition. Sites with the highest sulphur release are located in central Europe, where the strongest reduction in sulphur deposition has taken place over the last decade. This indicates that these systems are releasing sulphur stored in the soil in previous episodes of higher sulphate input.

In accordance with the available literature, N leaching was generally negligible below throughfall N inputs of 10 kg.ha<sup>-1</sup>.yr<sup>-1</sup>. At higher inputs N leaching increased, but at most sites (85%) the N input was higher than the N leaching, reflecting N retention in the soil. Sites with a net release of nitrogen were found in areas with a high N deposition over a prolonged period of time such as Belgium and north-western Germany. There was a significant relationship between N leaching and N deposition, but not with the soil C/N ratio, although the C/N ratio appeared to influence the average nitrate concentrations. Furthermore, N leaching was limited at high C/N ratios (>30) in the organic layer. Due to the different behaviour of S and N, the leaching flux of SO<sub>4</sub> was mostly

higher than that of  $\text{NO}_3$ . This indicates that  $\text{SO}_4$  is still the dominant source of actual soil acidification despite the generally lower input of S than N.

The median base cation balance was close to zero, implying a net adsorption and a net release of base cation at approximately 50% of the plots. The phenomenon of base cation removal due to man-induced soil acidification is thus limited, specifically since high leaching values were partly due to natural acidification in soils with a high pH and base saturation. The impact of air pollution on base cation removal is, however, clear since the leaching flux of base cations ( $\text{Ca}+\text{Mg}+\text{K}$ ) increased significantly with an increase in the sulphur (acid) input. The Al leaching flux was also significantly related to the  $\text{SO}_4$  input (and leaching) reflected by the fact that sites with a high Al leaching coincide with sites with a high input (leaching) of  $\text{SO}_4$ . The geographic patterns of both elements did not coincide very well, however, since soil base saturation was also significantly related to the Al leaching flux.

The large ranges in budgets for all the considered elements was to be expected considering the wide geographic range in locations with diverse circumstances with respect to deposition, hydrology and bio-geochemistry. However, there are also considerable uncertainties in the calculated budgets, considering the uncertainties in calculated water fluxes and measured element concentrations in view of spatial variability within a plot. Furthermore, the budgets are based on measurements during a relatively limited number of years. For most sites (58%), budgets were limited to a three-year period (1996-1998) and for 28% of the sites it was even less, while data for a four year period (1995-1998) were available at 14% of the sites. This relatively short time span may lead to over- or underestimation of the budget compared to the long-term situation due to particular hydrological or biological circumstances in specific years. Further improvements of the budgets can be expected when the time period increases. Furthermore, the regression relationships that were found between canopy exchange and environmental factors may be used to improve the existing canopy exchange model. Independent validation of the canopy model can be performed by using dry deposition estimates from the EDACS inferential deposition model (Annex 5). This validated model may subsequently be used for estimating atmospheric deposition on all Intensive Monitoring plots.



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## Annex 1 Suggested themes/topics in the coming Technical Reports

An overview of the possible central themes and additional topics in the Technical Reports for the period 2001-2005 is given in Table A1.1, including information on related projects.

*Table A1.1 Suggested publication strategy for the period 2001-2005.*

Year	Main topic	Additional topic	Relevant surveys	Related projects
2001	Water and element fluxes through the forest ecosystem	Species diversity of the ground vegetation	Deposition, meteorology, soil solution, ground vegetation	Bal-N-s (Input-output balance project funded by DGVI)
2002	Impacts of environmental factors on ground vegetation and possibly crown condition	Critical loads of atmospheric deposition for nitrogen, acidity and heavy metals	Crown condition, ground vegetation Deposition, meteorology, soil solution	Bal-N-s, Possibly an EU project related to impacts of environmental factors on ground vegetation
2003	Predictions of the long-term impact of emission-deposition scenarios on soil (solution) chemistry (and possibly the species diversity of ground vegetation)	Carbon sequestration of forest soils; Contribution of N deposition to C sequestration	Deposition, meteorology, soil, soil solution	Dymons (dynamic modelling project funded by Dutch Ministry VROM) Dynamic and Recognition (Carbon related projects funded by DGVI)
2004	Forest growth and carbon sequestration	The results (impacts) of ozone measurements	Increment, meteorology, deposition, soil, air quality	Center/Dynamic, Afforest and Carbo-Europe (Carbon related projects funded by DGXII and DG VI)
2005	Trends in atmospheric deposition, crown condition, foliar chemistry and soil solution chemistry	Comparison of measured and calculated trends in deposition and soil solution chemistry	Deposition, crown condition, foliar chemistry, soil solution chemistry	Not yet known

The publication strategy follows from the strategy for Intensive Monitoring (De Vries, 2000) focusing on the period 2001-2005. It also ensures an adequate supply of policy relevant information for the coming period and an alternation of a focus on abiotic and biotic aspects, as further elaborated in Table A1.2.

The focus on water and element fluxes in 2001 is logical as a precursor of the calculation of critical loads for 2002 and in line with ongoing activities within the Bal-N-s project. Furthermore, the presentation of data on the species diversity of the ground vegetation is included as an additional topic since those data, that have been submitted at the end of 1999, are new. It appears logical to focus the attention of the report of 2002 on the species diversity of the ground vegetation (main topic). A combination of this topic with crown condition in relation to environmental factors is attractive. This should be an improvement of a previous exercise in the Technical Report 2000 by including the calculated drought stress factors in the Technical Report 2001 (this report, Chapter 4) and information on stand history and stand management. A focus on critical loads in 2002 (additional topic) is a logical follow up of the element fluxes and inclusion of metals is in line with the attention it receives in the WGE and the ICP mapping and modelling.

The focus on model predictions in 2003 is in line with the expectations and focus of both the WGE and ICP mapping and modelling in this respect, as well as the project duration of Dymons (ending at the end of 2002). The focus on carbon sequestration in 2004 goes along very well with results from a repetition of the forest growth survey and the ending of various EU projects related to carbon sequestration. On the other hand it seems quite late and a publication of results seems

feasible in 2003 as well. For this reason, carbon sequestration in forest soils is suggested as an additional topic in 2003. Inclusion of all information on carbon sequestration will, however, be quite much. The focus on trends in 2005, finally is in line with the availability of data at that time allowing some first (very preliminary) trends.

*Table A1.2 Expected products described in the different Technical Reports in view of the aim of the Intensive Monitoring Programme and its policy relevance.*

Year	Focus	Expected products	Relation to aim of the Intensive Monitoring Programme	Policy relevance
2001	Abiotic	<b>Element retention</b> Relations with environmental factors	Provide insight in present impacts of meteorology and deposition forest soils	Information on the role of S and N in potential and actual soil acidification
2002	Biotic	<b>Relationships</b> Relations between abiotic stress factors (dose) and biotic effect indicators (response) <b>Critical loads</b> of nitrogen, acidity and heavy metals	Provide insight in present impacts of air pollution and other stress factors on forest ecosystems'  Contribute to indicators for sustainable forest management	Public and policy awareness of relative contribution of anthropogenic stress on forest ecosystems Guidance for implementation/evaluation of air pollution protocols
2003	Abiotic	<b>Model predictions</b> Predicted responses of forest ecosystems on (reduced) air pollution <b>Carbon sink potential</b> of forest soils	Provide insight in the future impacts of air pollution and other stress factors on forest ecosystems	Evaluation of multi-pollutant, multi effect protocol
2004	Biotic	<b>Carbon sink potential</b> of forests and forest soils Ozone stress	Provide insight in carbon sequestration	Guidance for implementation/evaluation of CO <sub>2</sub> and NO <sub>x</sub> emission protocols (Kyoto)
2005	Abiotic	<b>Monitoring data</b> Trends in stress factors and effect indicators	Contribute to the development and monitoring of 'criteria and indicators for sustainable forest management'.	Early warning system; insight in damage delay or recovery

## **Annex 2 Field intercomparison of bulk deposition and throughfall data at Speuld and Schagerbrug**

### **Aim**

Efforts to improve quality assurance and keep it at a high level are necessary to yield credibility in the data sets obtained. In this context, both field and laboratory intercomparisons are crucial (compare Lövblad, 1997; Mosello et al., 1999). Recently, in the Joint Research Centre of the EC (JRC) an important step forward was made, in view of the QA/QC of the laboratories involved. Those comparisons enable researchers to assess the relative importance of the different error sources (sampling methods versus chemical analysis) when measuring atmospheric inputs and soil outputs. It also enables them to select the most accurate analytical methods, sampling equipment, sampling strategy and sample handling, thus leading to further harmonisation in methods. At the same time knowledge transfer among participants is promoted and an assessment can be made on the quality and comparability of the results from the different monitoring sites. Here we report the first results of a field intercomparison related to the assessment of precipitation, throughfall, stemflow fluxes within the framework of the Intensive Monitoring Program (Draaijers et al., 2001).

A field intercomparison of sampling equipment, sampling strategy and sample handling has been set up aiming amongst others at:

- Assessing the quality and comparability of the results of the throughfall, stemflow and precipitation measurements performed within the framework of the Intensive Monitoring Program.
- Identifying throughfall, stemflow and precipitation measurements methods not fulfilling minimal requirements given the needs of and the accuracy needed in the Intensive Monitoring Program.
- Coming up with recommendations for optimal sampling equipment, sampling strategy and sample handling and on quality assurance and quality control procedures necessary concerning throughfall, stemflow and precipitation measurements performed within the framework of the Intensive Monitoring Program.

### **Methods**

The field intercomparison was conducted at the Speulder forest (throughfall and stemflow) and Schagerbrug (bulk precipitation) research sites, both situated in the Netherlands. The Speulder forest consists of a 2.5 ha homogeneous monoculture of Douglas fir (*Pseudotsuga menziesii*), 40 years old, with a stem density between 785 and 1250 trees.ha<sup>-1</sup> and a mean tree height of about 25 m. The experiment at Schagerbrug was hosted by the National Institute of Public Health and the Environment (RIVM, Bilthoven) and carried out by TNO Institute of Environmental Sciences, Energy Research and Process Innovation in cooperation with the ECN Netherlands Energy Research Foundation. Members of a Review Group act as external referees to control the quality of the set-up, interpretation and reporting of this study. The countries participating in the Intensive Monitoring Program were invited to install throughfall, stemflow and precipitation equipment at both research sites according to their own experimental protocol regarding sampling strategy (number and siting of the collectors) and sampling equipment. Sampling and chemical

analysis were conducted for half a year on basis of the countries own experimental protocol with respect to e.g. sample handling, sampling frequency and frequency of analysis. Experimental protocols were in part described in the so-called Data Accompanying Report Questionnaires (DAR-Q's), which the participating countries submitted to FIMCI. Additional information was obtained from the participating countries.

Data gathered during the field experiments are evaluated with respect to throughfall, stemflow and precipitation volumes, volume weighted average concentrations and fluxes. The evaluation concentrates on the mandatory parameters  $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Mg}^{2+}$ ,  $\text{Ca}^{2+}$ ,  $\text{NH}_4^+$ ,  $\text{Cl}^-$ ,  $\text{NO}_3^-$ ,  $\text{SO}_4^{2-}$ , alkalinity,  $\text{N}_{\text{total}}$  en pH. Deviations from the mean volume, volume weighted average concentration and flux are investigated in relation to the different technical aspects of the measurements, e.g. (i) sampling equipment (type, material, shape, size), (ii) sampling strategy (number and siting of samplers) and (iii) sample handling (e.g. sample frequencies, sample filtering, sample preservation, cleaning procedures and intervals, frequencies of analysis). To investigate the impact of the sampling frequency, a throughfall sample has been stored in the field and chemically analysed on a weekly basis for one month period. By digital processing of photographs taken of the canopy above each individual throughfall sampler, the crown coverage was assessed to investigate the impact of differences in canopy closure.

## Results

The number of countries that participated in the field intercomparison were 20 for throughfall, 6 for stemflow and 20 for bulk precipitation. The quality and comparability of the results of the different countries was assessed from statistical analysis. In Table A2.1, summary statistics for the deviation from the best estimate is presented for 6-month total fluxes of throughfall, stemflow and bulk precipitation, respectively. The best estimate was calculated as the mean flux of countries whose results were within the range of 2 times the standard deviation.

Table A2.1 Summary statistics for the deviation from the best estimate for 6-month total fluxes of throughfall, stemflow and bulk precipitation, respectively (in %). are presented as well (after Draaijers et al., 2001).

Statistic	Deviation (%) for throughfall										
	Na	Mg	K	Ca	$\text{NH}_4^+$	Alk.	$\text{NO}_3^-$	$\text{SO}_4^{2-}$	$\text{Cl}^-$	Kj N.	$\text{H}^+$
Mean	11.5	12.9	10.4	13.2	11.4	15.5	10.4	11.1	11.1	10.2	27.4
s.d.	9.1	9.0	6.2	8.9	9.8	13.8	9.2	10.2	8.3	8.4	17.8
Min	0.4	1.8	1.0	0.5	0.5	0.9	1.4	0.3	1.6	0.4	0.7
Max	33.3	35.5	21.2	29.9	36.0	45.8	35.0	36.7	30.8	31.5	73.2

Statistic	Deviation (%) for stemflow										
	Na	Mg	K	Ca	$\text{NH}_4^+$	Alk.	$\text{NO}_3^-$	$\text{SO}_4^{2-}$	$\text{Cl}^-$	Kj N.	$\text{H}^+$
Mean	24.8	31.5	25.5	25.7	21.2	41.5	19.3	25.6	22.3	19.4	30.8
s.d.	12.9	18.6	7.5	15.0	13.0	19.1	15.0	12.9	15.0	13.7	22.8
Min	8.0	1.1	15.9	1.6	7.7	12.8	0.4	4.8	8.2	5.7	2.5
Max	43.7	62.8	37.7	41.8	44.6	64.7	41.0	40.8	44.0	44.5	63.6

Statistic	Deviation (%) for bulk precipitation										
	Na	Mg	K	Ca	$\text{NH}_4^+$	Alk.	$\text{NO}_3^-$	$\text{SO}_4^{2-}$	$\text{Cl}^-$	Kj N.	$\text{H}^+$
mean	25.9	27.9	33.6	25.8	19.5	22.4	16.5	24.0	25.8	19.5	147.7
s.d.	16.2	18.9	26.1	16.4	19.3	17.0	10.0	16.7	14.0	19.2	305.3
min	0.3	0.5	1.0	3.9	0.7	1.9	3.4	2.5	1.5	2.6	42.3
max	66.3	74.6	112.2	80.8	90.6	64.9	36.4	78.7	56.7	80.9	1477.2



For throughfall fluxes, deviations from the best estimate up to 35% occurred, with the exceptions of  $H^+$  and alkalinity with deviations up to 75% and 45%, respectively. The average deviation from the best estimate for the 20 participating countries was between 10 and 15%, depending on component, with the exception of the throughfall flux of  $H^+$  with an average deviation of 25%.

For stemflow fluxes deviations from the best estimate up to 45% occurred,  $Mg^{2+}$ , alkalinity and  $H^+$  stemflow fluxes being the exceptions with deviations up to 65%. The average deviation from the best estimate for the 6 participating countries was between 20 and 30%, depending on component and alkalinity being the exception (average deviation of 40%).

For bulk precipitation fluxes deviations from the best estimate up to 90% occurred,  $K^+$  and  $H^+$  bulk precipitation fluxes being the exceptions with deviations up to 110% and 1500%, respectively. The extremely large deviations found for  $H^+$  are caused by very small fluxes. As a consequence, small absolute deviations result in large relative deviations. The average deviation from the best estimate for the 20 participating countries was between 15 and 35%, depending on component and  $H^+$  being the exception (average deviation of 150%).

More information on the results of the field inter-comparison project executed at the Speulder forest and Schagerbrug can be found in Draaijers et al. (2001).

## Evaluation

### *Throughfall*

#### *Representativity of measurements*

From the Field intercomparison project at the Speulder forest it became clear that the number of collectors (funnels) recommended in the Manual of ICP Forest (10-15) is generally too small to reach an accuracy of 10%. To reach the required accuracy forest for all components a minimum number of about 25 funnels would have been necessary at the Speulder forest. Performing a pre-study to determine the number of funnels (or gutters) necessary is recommended. The number of collectors necessary will depend on the homogeneity of the forest stand.

The Manual of ICP Forests (UN/ECE, 1998) recommends using at least 10-15 funnels with a diameter of 20 cm for collecting throughfall. This means that the total collecting area should exceed  $3140\text{cm}^2$ . For only 35% of countries participating in the field inter-comparison the total collecting area was found to exceed this value. For 20% of the countries the total collecting area not even exceeded half of the recommended value. From the field inter-comparison it was concluded that a total collecting area less than  $2000\text{cm}^2$  significantly reduces the accuracy of the throughfall measurements

Throughfall collectors should be placed in such a way that a large number of canopies are covered and results are representative for the forest plot. 75% of the countries fulfilled this requirement. When gutters are used instead of funnels, these gutters generally only cover a limited number of tree canopies.

#### *Determination of volumes*

It has been found relatively difficult to estimate throughfall volumes accurately. When averaging the deviations from all countries, the mean deviation from the best throughfall volume estimate equalled 8%, the largest deviation 23%. 35% of the countries were able to estimate throughfall

volumes within 5% of the best estimate. Apart from the representativity of the measurements, deviations could be explained by a number of other parameters. Important parameters included the collecting area of the funnels deviating from those reported and used in the flux calculations, and the collector height above the ground surface, influencing wind flow around the funnels. Sometimes there was a relatively loose connection between the funnel and storage container, allowing water to run down on the outside of the collector directly into the storage container. Reduced filter drainage and long tubes containing dips sometimes postponed and/or prevented through-flow and/or induced wetting loss.

#### *Determination of concentrations*

It also has been found quite difficult to estimate throughfall concentrations accurately. Deviations from the best estimate for volume-weighted mean throughfall concentrations were found dependent on component and country. When averaging the deviations from all countries, the mean deviation ranged between 8% ( $K^+$ ) and 21% ( $H^+$ ). Maximum deviations ranged between 21% ( $Cl^-$ ) and 61% ( $H^+$ ). When averaging the deviations from all components, the mean deviation ranged between 6% and 22%. Apart from non-representative sampling, deviations could be explained by the efficiency of the collectors to collect dry deposition (dependent on their height above the ground surface and the aerodynamic properties of the funnel), by algae present in tubes or litter present in the samples causing biochemical transformation, by restricted cleaning frequencies and/or by the chemical interactions with the collector material.

### *Stemflow*

#### *Representativity of measurements*

From the field inter-comparison project it was concluded that the number of stemflow collectors recommended in the Manual of ICP Forest (5-10) is too small to reach the required accuracy of 10%, even at a homogeneous forest stand like the Speulder forest. Most countries measure stemflow at less than 5 trees. Trees should be carefully selected to assure representative stemflow sampling within the forest plot. In a single species forest stand it is recommended to select trees on basis of information on diameter at breast height or tree height. A limited number of countries do not select their trees on these selection criteria.

#### *Determination of volumes*

It has been found relatively difficult to estimate stemflow volumes accurately. When averaging the deviations from all countries participating in the stemflow field inter-comparison, the mean deviation from the best stemflow volume estimate equalled 23%, with a maximum deviation of 50%. Apart from non-representative sampling, inaccurate stemflow volume determination results from leakage as a result of inadequate attachment of the spiral cord to the bark. Other explanations include the small capacity of the spiral cords and/or storage containers inducing overflow.

#### *Determination of concentrations*

It also has been found quite difficult to estimate stemflow concentrations accurately. Deviations from the best estimate for volume-weighted mean stemflow concentrations were found dependent on component and country. When averaging the deviations from all countries, the mean deviation ranged between 13% ( $NO_3^-$ ) and 52% ( $H^+$ ). Maximum deviations ranged between 19% (Kjehldall-N) and 136% ( $H^+$ ). When averaging the deviations from all components, the mean deviation ranged between 15% and 44%. Deviations could usually be explained by a number of

parameters. Apart from non-representative sampling, important parameters explaining deviations from the best estimate included planing of the bark surface (inducing bark leaching), the restricted cleaning frequency, the absence of a filter system, and chemical interactions with the sampler material.

## ***Bulk precipitation***

### *Representativity of measurements*

From the field inter-comparison project it was concluded that the minimum number of bulk precipitation collectors (funnels) recommended in the Manual of ICP Forests (2) is too small to reach the required accuracy of 10% at Schagerbrug for all components. The number of bulk collectors required will be site-specific and will mainly depend on the relative amount of dry deposition onto the collectors.

The Manual of ICP Forests recommends using at least 2 funnels with a diameter of 20cm. This means that the total collecting area should exceed 680cm<sup>2</sup>. For 70% of the countries participating in the field inter-comparison, the total collecting area was found not to exceed this minimum value. For 30% of the countries the total collecting area not even exceeded half of the recommended value. Several studies have indicated that the collecting area had actually only a minor influence on the precision of rainfall quantification (Thimonier et al., 1998). In the field-intercomparison, however, samplers with lower collection areas tended to collect higher precipitation amounts. This might be related to the specific weather situation during the field inter-comparison. Regularly showers with high winds frequently occurred. With such conditions a vacuum is formed in funnels. This vacuum is expected to be lower for small collectors, leading to better collection than larger samplers.

### *Determination of volumes*

It has been found relatively difficult to estimate precipitation volumes accurately. The deviation for precipitation volumes compared to the best estimate ranges from +103% to -27%. Deviations result from rainwater flowing directly into the storage container, aerodynamic blockage related to the height of the collector and its aerodynamic properties of the collector, and/or differences between actual and reported collecting areas.

### *Determination of concentrations*

It has been found difficult to estimate precipitation concentrations accurately because of the relatively large impact of dry deposition onto the collectors, strongly varying with funnel characteristics. Nitrogen components, alkalinity, K<sup>+</sup> and H<sup>+</sup> were found sensitive to biological conversion, which may happen when no conservatives are added and the sample stays in the field long than 2 weeks. In general deviations from the best estimate were found largest for alkalinity, Kjehldall-N, H<sup>+</sup> and K<sup>+</sup>, followed by Ca<sup>2+</sup>, Mg<sup>2+</sup>, Na<sup>+</sup> and Cl<sup>-</sup>. Deviations were found lowest for SO<sub>4</sub><sup>2-</sup>, NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup>.

A more extensive evaluation of the results of the field inter-comparison performed at the Speulder forest and Schagerbrug can be found in Draaijers et al. (2001).

## **Annex 3 Data assessment methods for deposition and soil solution monitoring**

### **Deposition data**

Data assessment information for deposition monitoring until 1998 has been stored for 226 of the 228 plots. For two plots, no DAR-Q information was available. Furthermore, the DAR-Q's were not always complete with respect to relevant aspects as shown below. Deposition estimates are derived from throughfall, stemflow and precipitation measurements using a canopy budget model.

#### ***Sampling equipment***

##### *Throughfall*

Throughfall is measured using either gutters or funnels. Gutters are sometimes preferred above funnels, because they integrate the input over a larger canopy area (Draaijers et al., 1998). However, generally only a small number of gutters placed under a limited number of trees is used, limiting the representativity of the throughfall measurements (Draaijers et al., 2001). For monitoring of throughfall, use was mostly made of funnels (86%), whereas three different types of gutters were used at 14% of the plots.

##### *Stemflow*

At 54 plots, with significant amounts of stemflow, stemflow was measured, mostly using a spiral cord around the tree trunk.

##### *Precipitation*

Bulk deposition is measured using funnels (bulk samplers) situated in the open field near the Intensive Monitoring plot. Sometimes wet-only samplers were used but bulk samplers were preferred because they are cheaper and do not require a power source. Sometimes precipitation volumes were also measured using a Hellman sampler and precipitation concentrations using a wet-only sampler. For 96% of all open field stations, the reported distance to the plot was not longer than 1 kilometre, 21.7% being within 100 m, 48.7% between 100 and 500 m and 25.6% between 500 and 1000 m. For the other 4% the distance increased up to 4300 m.

#### ***Number of samplers and collecting area***

##### *Throughfall*

To ensure that the determined atmospheric deposition data are representative for the monitoring plot, a good sampling set-up is essential. A sufficiently large number of samplers should be used to get a representative average value, considering the spatial variation in throughfall and stemflow. It is recommended to install at least 10 randomly placed funnels with a diameter of 20 cm (Lövblad, 1994). The comparison between these recommendations in literature and actually used number and type of throughfall samplers in the Intensive Monitoring, showed that in general the numbers of samplers were sufficiently large (Fig. 5.2). On the plots where funnels are used (85% of all deposition plots) the number of throughfall samplers used mostly ranged from 10-14 (78%). 15 or more samplers were used on 14% of the plots where funnels are applied. On 8% of these plots less than 10 samplers are placed. The minimum number of samplers per plot was reported to be 3. At these plots gutters are used with a relatively large collecting area.

*Table A3.1 Number of throughfall samplers used at the various Intensive Monitoring plots.*

Number of samplers	Number of plots		
	Gutters	Funnels	Total
< 5 (3)	18	-	18
5-9	-	-	-
10-14	14	104	118
15-19	-	61	61
20-24	-	27	27
>25	-	2	2
No information			2
Total	32	194	228

Information on the total collecting area was derived for the throughfall samplers by multiplying the number of samplers by the collecting area of each sampler, which is directly related to its diameter. Funnels had diameters ranging from 10 to 21 cm. Most of the plots had funnels with a diameter between 20 and 21 cm (38%), followed by a diameter < 12 cm (24%) and diameters between 14 and 18 cm (38%). Information on the sampler area is given in Table 5.3. A minimum total collecting area of 3140 cm<sup>2</sup> is required to keep in line with the recommendations made in the Manual of ICP Forests (UN/ECE, 1998). This requirement is met at only 108 sites (53%). From the deposition intercomparison project executed at the Speulder forest (see Annex 2) it was concluded that the accuracy of throughfall measurements will be significantly reduced when the total collecting area is smaller than 2000 cm<sup>2</sup> (Draaijers et al., 2001). At 96 Intensive Monitoring plots (42%) throughfall was sampled using samplers with a total collecting area less than 2000 cm<sup>2</sup>.

*Table A3.2 Total sampler area of throughfall samplers used at the various Intensive Monitoring plots.*

Sampler area (cm <sup>2</sup> )	Number of plots
<1500	33
1500-2000	63
2000-3000	12
3000-4000	7
4000-5000	64
>5000 (25500)	33
No information	16
Total	228

### *Stemflow*

Spatial variability of stemflow is generally larger than that of throughfall. The effective number of stemflow collectors required for representative sampling will depend on the tree species and stand characteristics. Fewer stemflow collectors will be needed for homogeneous, even-aged stands than for mixed stands with trees of different diameter and canopy size. Preferably trees should be selected on basis of information on diameter at breast height or tree height (Draaijers et al., 2001). A minimum number of 5-10 trees is mentioned in the literature (Lövblad, 1994) but this number is regarded to be a rather conservative estimate (Draaijers et al., 1998). The number of 5 trees was not reached on 81% of the plots. However, a sampling system in which multiple trees are connected to one collection container may have been reported as one sampler in the DAR-Q's. The number of sample trees may thus be underestimated to some extent.

### *Bulk deposition*

From the field intercomparison on bulk samplers performed at Schagerbrug it was concluded that the minimum number of bulk precipitation collectors (funnels) recommended in the Manual of

ICP Forests (two) is too small (Draaijers et al., 2001). The number of bulk collectors required will be site specific and will mainly depend on the relative amount of dry deposition onto the collectors. The number of bulk deposition samplers per site ranged between two and six on the majority of the 22 plots (97%) on which bulk is measured. For 20% of the plots no information on bulk deposition samplers has been received.

### ***Sampling frequencies***

The sampling frequencies for both bulk deposition and throughfall measurements were mostly weekly (47%) or fortnightly (27%). On 26% of the plots, for which sampling frequencies were reported, frequencies of 4-weekly periods were indicated. Stemflow is only measured on 56 plots. Sampling frequencies for stemflow at those 56 plots, were either weekly (57%) or biweekly (43%) (Table A3.3).

*Table A3.3 Sampling frequencies of bulk, throughfall and stemflow samplers used at the various Intensive Monitoring plots.*

Deposition	Nr of plots at a given sampling frequency		
	Weekly	Biweekly	Four weekly
Bulk deposition	106	61	59
Throughfall	106	61	59
Stemflow	31	23	-

### ***Conservation and analyses***

Most of the samples (81%) are stored less than a year, 18% of the samplers are not stored at all, and only 1% is stored for pooling. The storage temperature varies between a minimum of -22 °C and a maximum of 10 °C. Most of the samples (80%) are stored between 0 °C and 10 °C. At 46% of the plots a pre-treatment of the samples is done.

Insight in the impacts of analysis by various laboratories on the reliability and comparability of data can be obtained from ring tests in which water samples are distributed for analyses. Such a ringtest was carried out in 1996 and 1998. In 1996, 133 laboratories from different countries participated in this exercise, of which 18 laboratories are also participating in the Intensive Monitoring programme. The 18 laboratories in general had good results with only 5.9% outliers (Lövblad, 1997). In 1998, the number of participating laboratories was 188. In this year, the comparability between the different analyses was generally good, but for pH, alkalinity, calcium, sodium and potassium, some comparability problems do exist (Mosello et al., 1999).

### **Soil solution data**

In total methodological DAR-Q information has been submitted and stored for 168 of the 228 monitoring plots where soil solution chemistry is measured. As with deposition data, information is given on sampling techniques, the sampling material, sampling numbers, sampling frequencies and conservation and analyses. Attention is focused on the sampling techniques and the sampling material, that may strongly influence the comparability of the results obtained for the chemical composition of the soil solution. Information is also given on the procedures that have been adopted to make the results as comparable as possible.

In summary, at 18 plots the sampling techniques used were considered inadequate to use them in our budget studies, whereas differences in the sampler material were not considered of major

importance (see below). Ultimately, 210 plots were thus considered for use in input-output budgets. Information on the methodology, described below, focuses on the remaining 150 plots for which DAR-Q information was available. Finally 7 plots were skipped where perched water or ground water occurred within the soil-solution sampling layer. This section is based to a considerable extent on a recent review of the comparability of soil solution data obtained by varying techniques and approaches by Derome et al. (2001).

### **Sampling techniques**

In the Intensive Forests Monitoring Programme, several different sampling techniques are used for soil solution sampling, i.e. by:

- Placing soil solution collectors in the field, divided in tension lysimeters/suction cups and zero tension lysimeters for continuous monitoring.
- Taking soil samples and extraction of soil solution using either centrifugation or extraction methods.

The ion concentrations obtained do depend on the measuring devices, since different types of soil water are extracted. In general, concentrations increase going from zero tension lysimeters to suction cups and to centrifugation, whereas extraction may lead to completely different results (Derome et al., 2001). Information on differences resulting from various extraction methods has also been presented in e.g. Zabowski and Ugolini (1990), Giesler et al. (1996) and in a previous Technical Report (De Vries et al., 1999). The three procedures differ considerably with respect to the soil solution fraction sampled, the effects of sampling on the site and the extent to which they provide information about temporal and spatial variation in the properties of soil solution. The different soil solution fractions sampled by the three techniques are as follows:

- Zero-tension lysimetry: samples only water flowing through macropores, i.e. percolation water: Samples are obtained when field capacity is exceeded.
- Tension lysimetry and centrifuge drainage water samples water retained by capillary forces and percolation water if field capacity is exceeded during sampling.

In order to ensure that results are as comparable as possible, assessment of the input-output budgets was limited to the results obtained with tension lysimetry. This is the technique most widely used by the countries participating in the soil solution monitoring part of the Intensive Forests Monitoring Programme (Table A3.4). The collectors are related to the lowest sampler, that was used to calculate the outflow from the forested ecosystem.

*Table A3.4 Soil solution collection methods applied in the Intensive Monitoring Programme. Numbers in bold are the collectors used in further evaluations.*

Collectors	Number of plots
Tension lysimeters / suction cups	<b>175</b>
Tension lysimeters and Zero tension lysimeters	<b>21</b>
Zero tension lysimeters	2
Centrifugation	14
Extraction	2
No information	<b>14</b>
Total	228

We omitted the results obtained by zero-tension lysimetry (only two plots) since it includes a different fraction of the soil solution, with lower concentrations of ions. The results obtained by

centrifuge drainage were also omitted (14 plots in the Netherlands) as they include water held by stronger capillary forces than tension lysimeters, resulting in higher concentrations of ions. An additional reason for not using the results obtained from centrifugation was the annual frequency at which they were obtained. Finally, results obtained by extraction were deleted due to the completely different results obtained.

In summary soil solution chemistry data were used for 210 plots, including 196 plots for which we do have definite information on the use of lysimeters and 14 plots for which no information is available. For the latter plots we do know that neither centrifugation nor extraction was carried out and the chance of using zero tension lysimeters only was considered negligible (see Table A3.3).

### ***Sampler material***

*Material used:* The equipment used in sampling soil solution (lysimeters, tubing, collection bottles etc.) should be made of materials that neither contaminate the samples nor decrease (e.g. through sorption) the concentrations of ions in the samples. The description of sampler material is limited to suction cups, that were ultimately used in the assessment of input-output budgets. Suction cups or tension lysimeters used should be made of materials that are considered sufficiently free of contaminants, such that the sample solution is not influenced by the sampler itself. In the submanual for soil sampling (UN/ECE, 1998) a list of appropriate materials is presented, including ceramic (P80), Prenart teflon and plastic. The lysimeters used are made of the first two materials only, i.e. 153 ceramic cups and 43 teflon cups. The ceramic cups also include 4 cups made of the material aluoxide, which is comparable to ceramic. For the remaining 14 plots no information was available. Information on tubing and collection bottles was also not given in the DAR-Q's.

*Adsorption and retention by the lysimeter material:* Problems that may occur through adsorption or release specifically are primarily associated with heavy metals, phosphate, and DOC/TOC, as discussed below. Adsorption of heavy metals may be a problem, and at low concentrations this can have a considerable impact on heavy metal concentrations in the soil solution. In porcelain suction cups, aluminium contamination may also occur (Raulund Rasmussen, 1989) but those cups are not used in any of the Level II plots. Recent studies carried out in Denmark indicate that many heavy metals are adsorbed on the surface of the Prenart suction-cup lysimeters, and that it can also be a problem with ceramic and aluoxide lysimeters at pH values above 4.0 (Grossman et al. 1990; Derome et al. (2001). Reactions between the material and specific cations and anions can also occur, e.g. phosphate is partially retained by the ceramic P80 cup lysimeters as a result of the conversion of aluminium silicate into aluminium phosphate. The pore size of the lysimeters is critical with respect to DOC/TOC concentrations. Derome et al. (2001), for example, carried out laboratory experiment using soil water and a humus/water extract to test the retention of TOC and DOC by ceramic (P80) and teflon/ Prenart lysimeters. Samples taken before and after passing through the lysimeters were analysed before (TOC) or after (DOC) filtering through a 0.45 µm membrane filter. Significant differences were found between the TOC and DOC concentrations in the samples obtained with both type of lysimeter, and the TOC concentrations for the Prenart lysimeter were higher than those for the ceramic lysimeter. The explanation for this is that the Prenart lysimeters have a larger pore size.

*Interaction with sampling tubes and collection bottles:* A number of different plastic materials are available on the market. The use of nylon (PA, polyamide) tubing should be avoided because the



softener (benzene sulphonamide) added to certain grades of nylon tubing is water soluble and will result in elevated DOC, N<sub>tot</sub> and Stot concentrations in the soil solution samples. The bottles used in collecting water samples from tension lysimeters are invariably made of glass owing to the relatively high negative pressures applied. There is a considerable range of bottles on the market, and contamination from or adsorption by the glass material can pose a considerable problem because the samples remain standing in the bottles for periods of two to four weeks before they are taken to the laboratory or emptied. Derome et al. (2001) tested contamination from and adsorption by three types of glass bottles: a clear borosilicate bottle (Schott Duran) which is used in Denmark, a brown, reagent-quality borosilicate bottle used in Finland, and a plastic bottle claimed to be of extra high quality. The results showed that sodium is clearly released from the brown, reagent-quality borosilicate bottle, causing contamination problems, especially in non-maritime regions where the Na concentration in the soil solution is low. The brown bottle also had slightly higher boron and silicon concentrations. Since we have no information on the type of tubes and bottles currently in use, these could not be used as criteria for omitting certain measurements. Furthermore, Na is not used in the element budgets.

In general, the ions that may be problematic with respect to sampling equipment are not used in our budget studies and, consequently, we used the data from all the types of lysimeter in our calculations.

### ***Number of samplers***

Due to high spatial variation in both soil solution chemistry and percolation water fluxes, caused by e.g. variation in tree cover, ground vegetation and soil properties, a sufficient number of samples is needed to obtain a reliable estimate of the average concentration on a plot scale. This number increases when the spatial variability is larger. Table A3.5 gives the number of samplers used per plot for the lowest sampler in the mineral subsoil. Again, the information only applies to 150 plots for which data were available.

*Table A3.5 The number of samplers (tension lysimeters) used at the Intensive Monitoring plots for which data assessment information was available.*

Number of Tension lysimeters	Number of plots
< 5 (3)	21
5-9	69
10-14	73
15-19	7
20-24	13
>25	13
No information	14
<b>Total plots</b>	<b>210</b>

Numbers of samplers ranged from 3-48 in the mineral subsoil. An indication of the adequacy of the number of samplers or samples can be derived from the results of a comparative study of three methods to extract soil solution at two forest stands in the Netherlands at fifteen spots. The relative standard deviation of the estimated mean concentration at each depth varied nearly always between 20 and 60% (see also De Vries et al., 2000a). Requiring that the number of samples should be such that the plot mean is within  $\pm 20\%$  of the population mean with a confidence level of 95%, at least 10 samples are needed, assuming a relative standard deviation of 30% (De Vries et al., 2000a). Considering the variation in the two Dutch forest plots to be

representative, the number of suction cups used at 50 plots seems quite low. Although the number of three lysimeters seems very low, we have not removed those plots from the assessment of input-output budgets but used the information in evaluating the plausibility of the results.

### ***Sampler depths and sampling frequency***

The depths of the sampled layers in the subsoil varied from 40-80 cm as shown in Table A3.6. Sampling depths varied considerably and we always used the lowest lysimeter near 80 cm to calculate the element out put from the root zone. Sampling frequencies were almost equally divided over weekly, biweekly, four weekly or monthly and four monthly (three times a year). We still used the latter plots but took a close look at the results in view of plausibility.

*Table A3.6 The sample depths (tension lysimeters) and sampling frequencies used at the Intensive Monitoring plots.*

Sampling depth (cm)	Number of plots <sup>1)</sup>	Sampling frequencies	Number of plots
<50	63	Week	42
50-60	70	2-weeks	54
60-70	21	4-weeks	16
70-80	25	Monthly	38
>80	31	3 times a year	46
No information			14
Total	210		210

<sup>1)</sup> Information derived from the data base, not from the DAR-Q's. Note that the sum of the number of plots over all layers exceeds 210 since at a number of plots measurements are made in more than one of the indicated layers

### ***Conservation and analysis***

Good quality of the soil solution data requires a careful treatment of the sample water. Contamination of the samples (by e.g. algae) is prevented by keeping the samples cool and dark in the field and cleaning or replacing collection bottles periodically. Furthermore, the time between sampling and analyses (storage time) should not be too long. DAR-Q's report these measures for all plots as shown in Table A3.7. In addition, preservatives are used to diminish biological activity in the sample. All countries conserve their sample water in cool places with temperatures ranging from -20 up to 10 °C, mostly varying between 2-6 °C.

*Table A3.7 Conservation aspects applied at the Intensive Monitoring plots.*

Conservation aspect	Number of plots			Total nr of plots
	Favourable	Less favourable	No information	
Temperature	- 20 °C – 0 °C : 12	0 °C - 12 °C : 174	24	210
Storing	Darkness : 195	Light : 1	14	210
Preservatives	Added : 31	Not added : 165	14	210
Cleaning frequency	< 2 weeks :	> 2 weeks :		210
Storage time	≤ 1 week : 162	> 1 week : 34	14	210

The submanual on soil solution sampling (UN/ECE, 1998) states that samples should be analysed as soon as possible on the untreated samples for pH and conductivity. Furthermore, filtration (0.45 µm. membrane filter) should be applied. Hereafter at least the other mandatory parameters, dissolved organic carbon (DOC), K, Ca, Mg, Al total (if pH<5), NO<sub>3</sub>-N and SO<sub>4</sub>-S and optional parameters, should be measured. Based on a first inventory of the DAR-Q information no specific problematic pre-treatment and analysis methods were noticed. However, the use of ring tests on a European scale like the ones applied for deposition data is needed here as well to improve the insight into the data quality of the various analyses. Of the countries that have submitted DAR-Q information about 70% reported to be involved in such ring tests already.

### ***Perched water and ground water***

At many forested sites in northern and central Europe, especially under spruce, the surface soil is periodically subjected to a rise in the ground water table or perched water level. No detailed information is currently available about the hydrological conditions on the level II monitoring sites, apart from the information (insufficient, sufficient, excessive) provided under the title “water availability” in connection with the forest soil inventory. In addition, short-term waterlogging of the surface soil can also occur in Fennoscandia (as well as in mountainous regions in other parts of Europe) following snowmelt in the spring when the ground is still partly frozen.

From the point of view of soil solution monitoring, it is extremely important to know whether soil solution collected by lysimetry represents soil water that is derived directly from rainwater (or snowmelt) or indirectly from rainwater via perched water/ground water. To follow the effects of acidifying deposition on soil water quality and assess input-output budgets, it is clear that only water that is present in the unsaturated soil layers should be sampled. However, determining the origin of the soil water is not always easy in practice. The position of the ground water table or perched water level can be followed by means of observation pipes installed at suitable points on the plots and the soil moisture measurements carried out as part of the meteorological monitoring work. The main problem as far as tension lysimeters are concerned is that, even if the ground water/perched water level is below the location of the lysimeters, water may be carried from these sources up into the lysimeters via capillary flow.

At present no empirical information is available about the possible effects of ground water or perched water on the results obtained with lysimeters at greater depths. We have therefore only omitted results obtained from gley soils or soils with gleyic features that indicate the sporadic presence of ground water within the root zone.

## Annex 4 The use of canopy exchange of weak acids in the calculation of total deposition of ammonium and acidity

### Introduction

One of the aims of the Bal-N-s project is to assess site-specific atmospheric deposition for selected Intensive Monitoring plots (level II) based on measurements of throughfall and bulk deposition. To estimate atmospheric deposition, the canopy exchange model developed by Ulrich (1983), improved and extended by Bredemeier (1988), Van der Maas et al. (1991) and by Draaijers and Erisman (1995). In the canopy budget model, annual total deposition is estimated by correcting the input by both throughfall and stemflow for exchange processes, occurring at the forest canopy. At plots where stemflow data are missing, the annual stemflow is estimated from the annual throughfall according to Ivens (1990).

The major assumptions of the extended canopy budget model are as follows:

- Canopy exchange of  $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$  is assumed negligible.
- Total deposition of the base cations Ca, Mg and K is calculated by multiplying the bulk deposition of those cations with the ratio of the sodium input by both throughfall and stemflow to the sodium input in bulk deposition. Canopy exchange (leaching) is then computed by the difference between the sum of BC in throughfall and stemflow minus total deposition. This approach is based on the assumption that (i) Na does not interact with the forest canopy (tracer) and (ii) the ratios of total deposition over bulk deposition are similar for Ca, Mg, K and Na.
- Total canopy uptake of  $\text{NH}_4^+$  and  $\text{H}^+$  is assumed to be equal to the total canopy leaching of  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$  and  $\text{K}^+$  taking place through ion exchange, while subtracting the leaching of weak acids. In allocating the uptake of  $\text{H}^+$  and  $\text{NH}_4^+$ , it is assumed that  $\text{H}^+$  has per mol an exchange capacity six times larger than  $\text{NH}_4^+$  (Van der Maas et al., 1991) ( $x\text{H} = 6$ ).
- The dry total deposition of weak acids is assumed to equal the bulk deposition (total deposition equals twice the bulk deposition). The leaching of weak acids is calculated by subtracting the total deposition from the measured throughfall and stemflow.

The inclusion of weak acid leaching also requires an estimate of the concentration of weak acids (WA) in both bulk deposition and throughfall, since the concentration of bicarbonate and organic acids is not measured directly. The estimation of the weak acid concentration can be based on either (i) alkalinity, (ii) the sum of  $\text{HCO}_3^-$ , derived from the pH and an assumed atmospheric  $\text{CO}_2$  pressure, and  $\text{RCOO}^-$  derived from DOC or (iii) the difference in concentration of cations minus strong acid anions. A comparison of those estimates is needed to get an impression of the accuracy of the estimated WA concentration. If such estimates differ widely, it is not likely that the inclusion of weak acid leaching improves the assessment of the uptake of  $\text{NH}_4^+$  and  $\text{H}^+$ , also considering the uncertain assumption that bulk and dry deposition of weak acids are equal. The aim of this paper is to critically review the assumption that weak acid leaching needs to be included to adequately assess the uptake of  $\text{NH}_4^+$  and  $\text{H}^+$ .

## Mathematical formulation of the canopy budget model including and excluding weak acid leaching

### *Model including weak acid leaching*

In mathematical terms the exchange (leaching) of base cations and weak acids and the related uptake of  $\text{NH}_4^+$  and  $\text{H}^+$  is formulated as:

$$\text{BC}_{\text{td}} = \frac{\text{Na}_{\text{tf}} + \text{Na}_{\text{sf}}}{\text{Na}_{\text{bd}}} \cdot \text{BC}_{\text{bd}} \quad (\text{A4.1})$$

$$\text{NH}_{4,\text{td}} = \text{NH}_{4,\text{tf}} + \text{NH}_{4,\text{sf}} + \text{NH}_{4,\text{ce}} \quad (\text{A4.2})$$

with:

$$\text{NH}_{4,\text{ce}} = \left( \frac{\text{NH}_{4,\text{bd}}}{\text{NH}_{4,\text{bd}} + \text{H}_{\text{bd}} \cdot \text{xH}} \right) \cdot (\text{BC}_{\text{ce}} - \text{WA}_{\text{ce}}) \quad (\text{A4.3})$$

$$\text{BC}_{\text{ce}} = \text{BC}_{\text{tf}} + \text{BC}_{\text{sf}} - \text{BC}_{\text{td}} \quad (\text{A4.4})$$

$$\text{WA}_{\text{ce}} = \text{WA}_{\text{sf}} + \text{WA}_{\text{sf}} - 2\text{WA}_{\text{bd}} \quad (\text{A4.5})$$

$$\text{H}_{\text{td}} = \text{H}_{\text{tf}} + \text{H}_{\text{sf}} + \text{H}_{\text{ce}} \quad (\text{A4.6})$$

with:

$$\text{H}_{\text{ce}} = \text{BC}_{\text{ce}} - \text{WA}_{\text{ce}} - \text{NH}_{4,\text{ce}} \quad (\text{A4.7})$$

where:

- WA = weak acids
- BC = Ca, Mg, K
- td = total deposition ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ )
- bd = bulk deposition ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ )
- ce = canopy exchange ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ )
- xH = an efficiency factor of H in comparison to  $\text{NH}_4$  (xH = 6)

### *Model excluding weak acid leaching*

In the model excluding weak acid leaching,  $\text{WA}_{\text{ce}}$  is set to 0 (Eq A4.3 is not included) and the Eqs. A4.3 and A4.7 related to the uptake of  $\text{NH}_4^+$  and  $\text{H}^+$  thus change to

$$\text{NH}_{4,\text{ce}} = \left( \frac{\text{NH}_{4,\text{tf}}}{\text{NH}_{4,\text{tf}} + \text{H}_{\text{tf}} \cdot \text{xH}} \right) \cdot \text{BC}_{\text{ce}} \quad (\text{A4.8})$$

$$H_{ce} = BC_{ce} - NH_{4,ce} \quad (A4.9)$$

In the situation where weak acid exchange is negative (leaching is positive, so the throughfall and stemflow is larger than the estimated total deposition), the estimated uptake of  $NH_4^+$  and  $H^+$  decreases when neglecting this process (compare Eq. A4.3 and A4.8). The reverse is true when weak acid exchange is positive, implying that weak acids are taken up by the forest canopy, together with e.g. protons (a situation that is unlikely to occur)

### Estimation of the weak acid concentration

The estimation of the weak acid concentration was derived with three independent estimates. In the first approach, the WA concentration is estimated from the sum of  $HCO_3^-$ , derived from the pH and an assumed atmospheric  $CO_2$  pressure, and  $RCOO^-$  derived from DOC according to (Oliver et al., 1983):

$$WA = HCO_3^- + RCOO^- \quad (A4.10)$$

with:

$$HCO_3^- = \frac{K_{CO_2} \cdot pCO_2}{H} \quad (A4.11)$$

$$RCOO^- = m \cdot DOC \cdot \frac{K_a}{K_a + H} \quad (A4.12)$$

The constant  $K_a$  was estimated according to (Oliver et al., 1983):

$$pK_a = a + b \cdot pH - c \cdot pH^2 \quad (A4.13)$$

where:

$K_{CO_2}$  = Dissociation constant of  $CO_2$ , being the product of Henry's law constant for the equilibrium between  $CO_2$  in soil and air and the first dissociation constant of  $H_2CO_3$  ( $mol^2 \cdot l^{-2} \cdot bar^{-1}$ )

$pCO_2$  = Partial  $CO_2$  pressure in the soil (bar)

DOC = Dissolved organic carbon concentration ( $mg \cdot l^{-1}$ )

m = Concentration of acidic functional groups on dissolved organic carbon ( $mol_c \cdot kg^{-1}$ )

$K_a$  = Dissociation constant for organic acid ( $mol \cdot l^{-1}$ )

H = Proton concentration ( $mol \cdot l^{-1}$ )

For  $pK_{CO_2}$ , a value of 7.8 was used, the partial  $CO_2$  pressure was set at 0.3 mbar (0.0003 bar) and the value of m was set at  $5.5 \text{ mol}_c \cdot kg^{-1}$  or  $0.0055 \text{ mmol}_c \cdot mg^{-1}$ . Values of a, b and c have been derived by calibrating the  $pK_a$  value on soil chemistry data (e.g. Van Wesemael and Verstaten, 1993) and surface water chemistry data (e.g. Oliver et al., 1983; Driscoll et al., 1989). In this calculation, we used the values by Oliver et al. (1983) for both deposition and soil solution data (a = 0.96, b = 0.90 and c = 0.039). A comparison of results using values derived by Van Wesemael

and Verstaten (1993) or Driscoll et al. (1989) shows a deviation of 5-25%, depending upon the pH (De Vries and Bakker, 1998).

In the second approach, the WA concentration was calculated from the measured alkalinity, while correcting for the pH according to:

$$WA = Alk + H - OH \quad (A4.14)$$

since alkalinity can be defined as:

$$Alk = HCO_3 + RCOO + OH - H \quad (A4.15)$$

In the third approach, the WA concentration was calculated from the difference in concentration of cations minus strong acid anions according to:

$$WA = \sum cat - \sum an \quad (A4.16)$$

with:

$$\sum cat = [Ca^{++}] + [Mg^{++}] + [Na^+] + [K^+] + [H^+] + [NH_4^+] \quad (A4.17)$$

$$\sum an = [SO_4^{--}] + [NO_3^-] + [Cl^-] \quad (A4.18)$$

The comparison between the three estimates was made for the plots where data on major cations and anions (needed in the third approach), alkalinity (second approach) and DOC (first approach) were all available. Those plots were located in Finland, Germany and the Czech Republic only. The problem with the last approach (which is generally used to estimate weak acids) is that it requires a good quality of the assessment of all major cations and anions. This approach may even lead to negative WA concentrations in case of either an overestimate of the major anions and/or an underestimate of the major cations. The estimates rely on the accuracy of the concentrations of all major and cations and anions, thus giving easily rise to large uncertainties. We thus also made a comparison in which we required that the difference between the sum of all the major cations and anions was less than 20% (limited to plots in Finland and Germany). Furthermore, in both cases we put negative concentrations to zero.

## Results

### *Estimated concentrations of weak acids according to the three methods*

Results of the estimated WA concentration in bulk deposition and throughfall according to the three methods show that the estimate based on the alkalinity measurement generally strongly deviates from the two other assessments (Figure A4.1).

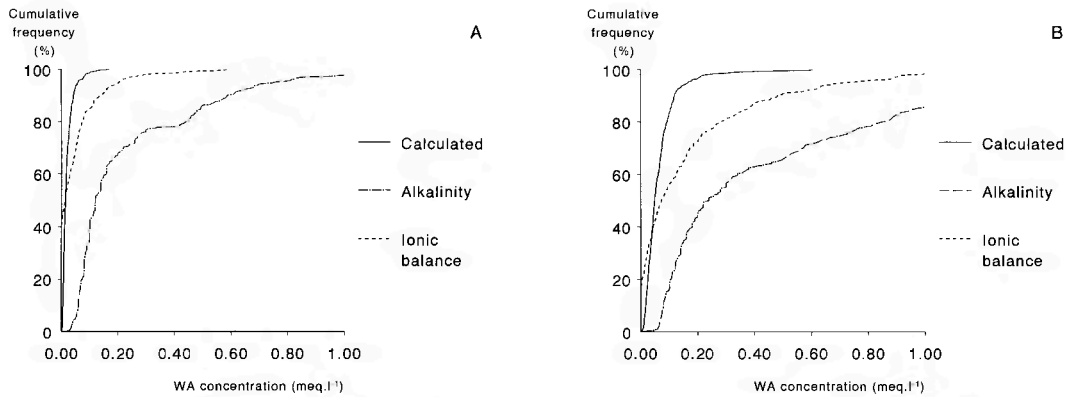


Figure A4.1 Estimated weak acid concentrations in bulk deposition (A) and throughfall (B) with three independent methods using all available data.

Without a quality check on the ionic balance data, the calculated alkalinity from pH and DOC is usually lower than the estimate based on the charge balance, both in bulk deposition and in throughfall. The difference is, however, small when allowing a relative difference of 20% in the charge balance only (Fig. A4.2). This requirement, however, hardly affected the percentage of plots where the estimated WA concentration is negative, being approximately 45% in case of bulk deposition and 20% in case of throughfall (Compare Fig. A4.1 and Fig. A4.2).

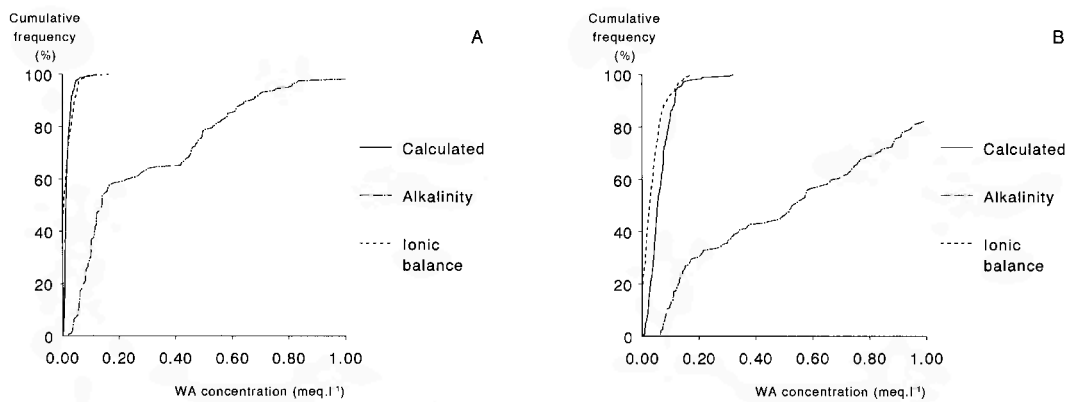


Figure A4.2 Estimated weak acid concentrations in bulk deposition (A) and throughfall (B) with three independent methods using only those data where the relative difference between the sum of cations and the sum of anions is less than 20%.

Considering the large deviation between the alkalinity assessment and the other two estimates, we considered the alkalinity measurement unreliable for the calculation of the weak acid exchange fluxes. A direct application of the ionic charge balance approach was also considered inadequate, due to the large percentage of plots with calculated negative (and thus zero) weak acid concentrations. Therefore, we further used a mixed approach in which the weak acid concentration was the maximum of a bicarbonate estimate based on the pH (according to Eq. A4.11) and the ionic balance (according to Eq. A4.16- A4.18). Using this mixed approach, weak acid concentrations are never zero, although it can be low at sites where the pH is high (low  $\text{HCO}_3^-$  concentration) and the charge balance is negative, since the  $\text{RCOO}^-$  concentration is assumed negligible in those situations. In the further comparison we thus focus on the calculation approach, which seems most adequate considering data quality and the mixed approach, which can also be applied when no DOC data are available, being the case at many plots.



**Concentrations and fluxes of weak acids in bulk deposition and throughfall using the calculation method and the charge balance method**

The parameters determining the estimated weak acid concentration according to the first assessment are the pH and the DOC concentration. Insight in the values of those parameters and the  $\text{HCO}_3^-$  and  $\text{RCOO}^-$  concentration thus derived is given in Figure A4.3. Results show that the pH and DOC concentration generally increases in throughfall compared to bulk deposition (Fig. A4.3A,B), thus leading to higher concentrations of bicarbonate and  $\text{HCO}_3^-$  and  $\text{RCOO}^-$  (Fig. A4.3C,D). The higher pH indicates the occurrence of acid buffering in the forest canopy, that might be due to uptake of H in exchange to base cations (or by release of  $\text{CO}_2$  causing a decrease of the bicarbonate flux). The higher DOC and  $\text{RCOO}^-$  concentrations might indicate the release of organic anions from the canopy, but this may also be due to dry deposition of those compounds.

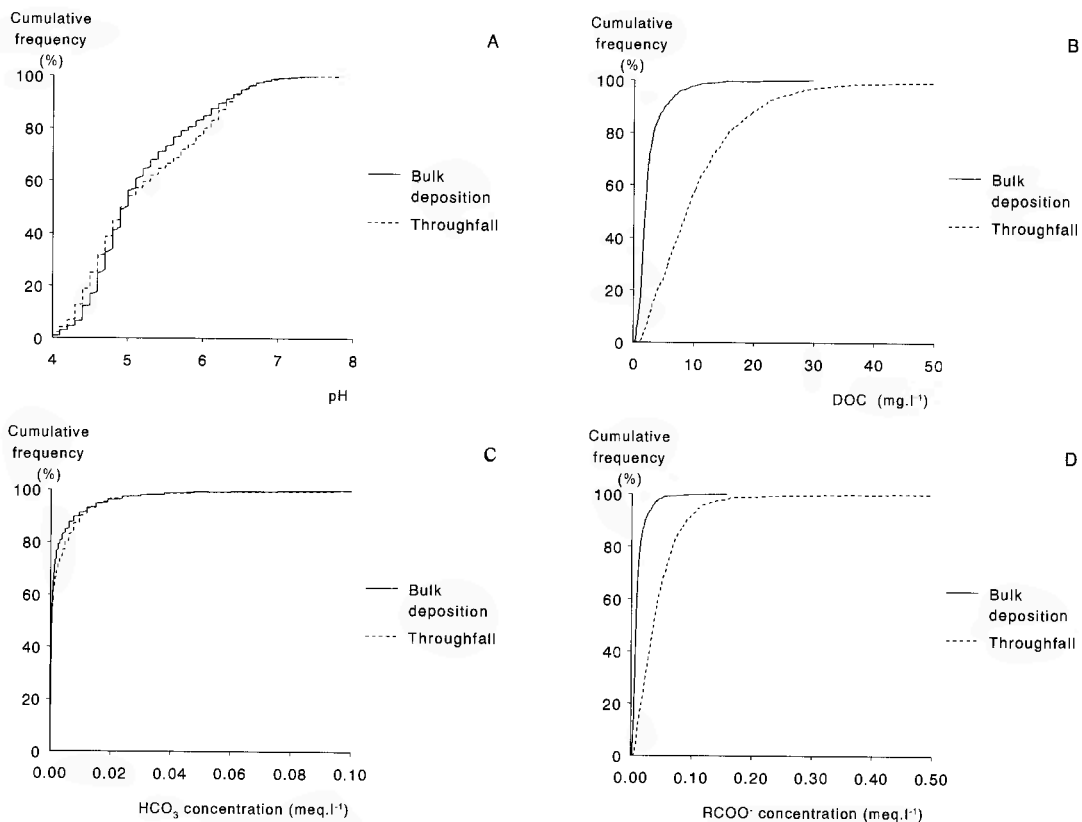


Figure A4.3 Values for the pH (A) and concentrations of DOC (B),  $\text{HCO}_3^-$  (C) and  $\text{RCOO}^-$  (D) in bulk deposition and throughfall.

The estimated WA concentration is also higher in throughfall than in bulk deposition, when using the calculation approach, as it reflects the concentration behaviour of bicarbonate and organic anions (Fig. A4.4A). Concentrations above  $0.05 \text{ mmol}_c \cdot \text{l}^{-1}$  are mainly due to high DOC values, specifically in throughfall, since  $\text{HCO}_3^-$  concentrations hardly exceed this value considering the low  $\text{CO}_2$  pressure of the atmosphere. Using the mixed (calculation and charge balance) approach, the concentrations in throughfall are also consistently higher in throughfall than in bulk deposition (Fig. A4.4B).

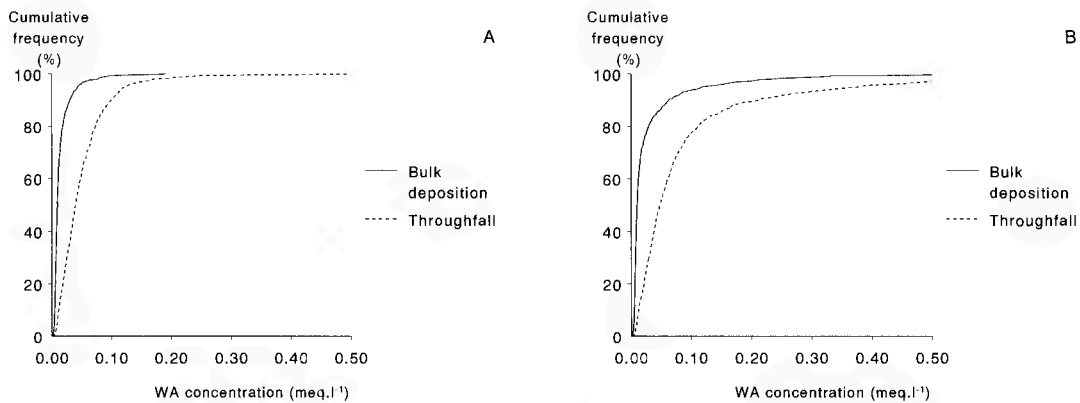


Figure A4.4 Estimated WA concentrations using the calculation approach (A) and the mixed approach (B) in bulk deposition and throughfall.

The difference in the two estimates is further illustrated in Fig. A4.5, where the annual weak acid flux in throughfall is plotted against the annual weak acid flux in bulk deposition.

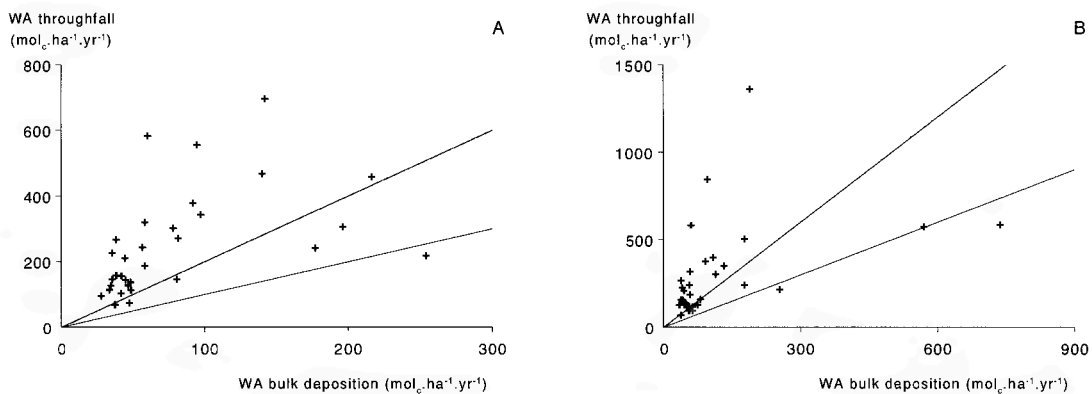


Figure A4.5 Relationships between estimated annual WA fluxes in bulk deposition and throughfall using the calculation approach (A) and the mixed approach (B).

Using the calculation approach, in nearly all cases (one exception) the throughfall flux is larger than the bulk deposition flux. At most plots, it is even larger than twice the bulk deposition flux, indicating weak acid leaching, since we assume that bulk and dry deposition of weak acids are comparable (Fig. A4.5A). Using the mixed approach, a similar conclusion holds (Fig. A4.5B). The plots do show directly the sensitivity of the assumption that bulk and dry deposition are equal with respect to the estimation of weak acid leaching or weak acid “uptake”.

#### **Calculated canopy exchange fluxes using the calculation method and the charge balance method**

The ranges in weak acid exchange fluxes based on the calculation approach and the mixed method are shown in Fig. A4.6. The figure also includes information about the range in calculated fluxes in throughfall plus stemflow below the forest canopy and the total deposition above the forest canopy (estimated as twice the bulk deposition).

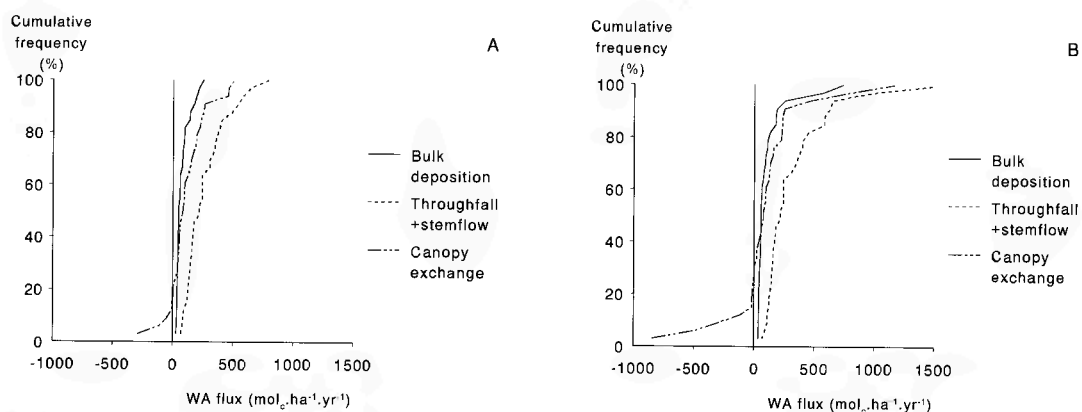


Figure A4.6 Estimated annual WA fluxes in total deposition, throughfall and stemflow and canopy exchange, using the calculation approach (A) and the mixed approach (B).

Results show a large range in exchange fluxes, ranging from highly positive values (indicating canopy leaching of weak acids, what is expected) to negative values indicating weak acid adsorption/uptake, which is unlikely to occur. In the latter case the calculated uptake of  $\text{NH}_4^+$  and  $\text{H}^+$  will increase compared to a calculation excluding the weak acid exchange. Negative weak acid leaching (uptake) fluxes occur in less than 20% of the plots in both procedures, but the negative values become higher in the mixed approach compared to the calculation approach.

Results illustrating the difference in calculated uptake of  $\text{NH}_4^+$  and  $\text{H}^+$  for the different methods are illustrated in Table A4.1. It also includes results of the charge balance approach only putting negative values to zero. The results show that on average both the  $\text{NH}_4^+$  and  $\text{H}^+$  uptake will decrease when including canopy exchange (in line with the average positive value for canopy leaching), but effects are largest for protons. Differences are small between the calculation approach and the mixed method indicating that this approach is a reasonable alternative when no DOC data are available. Even when neglecting the weak acid exchange, negative uptake fluxes for ammonium and protons are calculated, being due to estimated negative base cation leaching fluxes (Table A4.1). On average the proton adsorption is higher than the ammonium uptake, due to the preference factor of H compared to  $\text{NH}_4$  of 6.

Table A4.1 Ranges in calculated exchange fluxes of base cations, weak acids, ammonium and protons using various approaches.

Type of exchange Flux	Method	Exchange flux ( $\text{mol}_c \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ )					
		mean	Min	5%	50%	95%	max
Base cation leaching	Sodium	256	-524	-87	131	996	1164
Weak acid leaching	Calculated	114	-290	-103	86	461	506
	Charge balance	6	-1071	-423	-2	519	1168
	Mixed approach	95	-844	-458	78	738	1168
Ammonium uptake	Calculated	51	-526	-107	2	363	576
	Charge balance	89	-14	-3	20	453	491
	Mixed approach	65	-90	-35	7	366	487
Proton uptake	Without WA	90	-471	-14	20	465	1019
	Calculated	91	-89	-83	21	445	590
	Charge balance	161	-72	1	94	459	760
	Mixed approach	95	-85	-68	40	423	590
	Without WA	166	-79	-51	76	585	760

## Conclusions

This analyses leads to the following conclusions:

- Considering the large deviation between the alkalinity assessment and independent estimates based on pH and DOC measurements and a charge balance, the alkalinity measurement seems unreliable for the calculation of the weak acid exchange fluxes.
- Considering the calculation of negative weak acid concentrations in both bulk deposition and throughfall, the charge balance approach seems also unreliable for the calculation of the weak acid exchange fluxes, unless a severe quality control is carried out.
- The calculation approach based on pH and DOC measurements seems most reliable. When no DOC data are available, a mixed approach including the calculation of bicarbonate based on pH and of DOC based on the charge balance seems quite comparable to the calculation approach.
- The assumption that total deposition equals twice the bulk deposition may be questioned considering the occurrence of positive weak acid exchange fluxes at approximately 20% of the plots. This fact can only be explained by CO<sub>2</sub> release from the forest canopy, but the impact of this process is likely to be low.

Considering the results given above, it is suggested to:

- Further evaluate the assumption that total deposition equals twice the bulk deposition and that the preference factor of H compared to NH<sub>4</sub> of 6.
- Compare ammonium and proton adsorption while including and neglecting weak acid exchange and put attention to sites where weak acid adsorption is calculated.

## **Annex 5 Comparison of modelled deposition estimates with throughfall data at 223 Intensive Monitoring plots**

### ***Introduction***

Currently throughfall measurements are made at about 300 Intensive Monitoring plots to estimate the site specific inputs of nutrients. Deposition is site specific, determined by factors such as tree species, crown density, homogeneity of the stand, stand density, distance to the nearest edge, climate, environmental factors and vitality of the stand. Measurements are expensive and do not provide a direct link to emissions, which is necessary to develop and evaluate policies to abate pollution and forest damage. Therefore, there is a need for a tool to upscale results to sites where no measurements are available. The first tool to calculate site specific deposition fluxes, EDACS (European Deposition of Acidifying Components on a small Scale) was developed at RIVM (Erisman and Draaijers, 1995). The results of this model were used in the 10 years overview report to estimate deposition for Level I plots as described in Van Leeuwen et al. (2000). This model is now re-programmed and improved by ECN and meteorological input is updated, based on data from the European Centre for Meteorology and Weather Forecast (ECMWF). Concentrations of gases and aerosols are derived from the EMEP model. In this Appendix a short model description will be given and model results are compared with throughfall measurements at the Intensive Monitoring plots. Model results can also be used to estimate canopy exchange, assuming that the total deposition values are correct. The plausibility of the results are thus reviewed in the last section.

### ***Description of the EDACS model***

The basics of the EDACS model can be found in Erisman and Draaijers (1995), Van Pul et al. (1994) and Van Leeuwen et al. (2000). The basis for the total deposition estimates is formed by results of the EMEP long-range transport model. With this model dry, wet and total deposition is estimated on a 150x150 km grid over Europe using emission maps for SO<sub>2</sub>, NO<sub>x</sub> and NH<sub>3</sub>. The model results are used for estimating country-to-country budgets, as a basis of sulphur and nitrogen protocols, and for assessments. Calculated ambient concentrations of the acidifying components in 150x150 km grids by EMEP are multiplied with dry deposition velocity fields over Europe that are constructed with EDACS using a detailed land use map and meteorological observations to estimate small-scale dry deposition fluxes (Figure A5.1). The model input and output is flexible and depends on the land use information that is used. Currently land use maps for Europe (1/6 x 1/6°), for Germany on a 1 x 1 km<sup>2</sup> scale are available. This allows total deposition estimates on the European scale for 10 x 20km cells. In this approach we made a site specific calculation using site specific information available for the Intensive Monitoring plots. Every 6 hours the ECMWF meteorological data is used to calculate a deposition velocity for each grid or each plot. The deposition velocity is combined with a concentration obtained from calculations with the EMEP model to estimate the flux. An annual flux is the summation of all 6 hour values. The dry deposition flux is calculated as the product of the dry deposition velocity and air concentration at a reference height above the surface. In the inferential technique, the choice for a reference height (50 m) is a compromise between the height where the concentration is not severely affected by local deposition or emission and is still within the constant flux layer (Erisman, 1993). The parameterisation of the dry deposition velocity for particles was based on Erisman et al. (1994).

By using calculated concentration maps, the relationship between emissions and deposition is maintained and scenario studies, budget studies and assessments can be carried out on different scales. Wet deposition is added to the dry deposition to estimate total local scale deposition in Europe. Wet deposition can either be obtained directly from the EMEP model, or from measurements made in Europe.

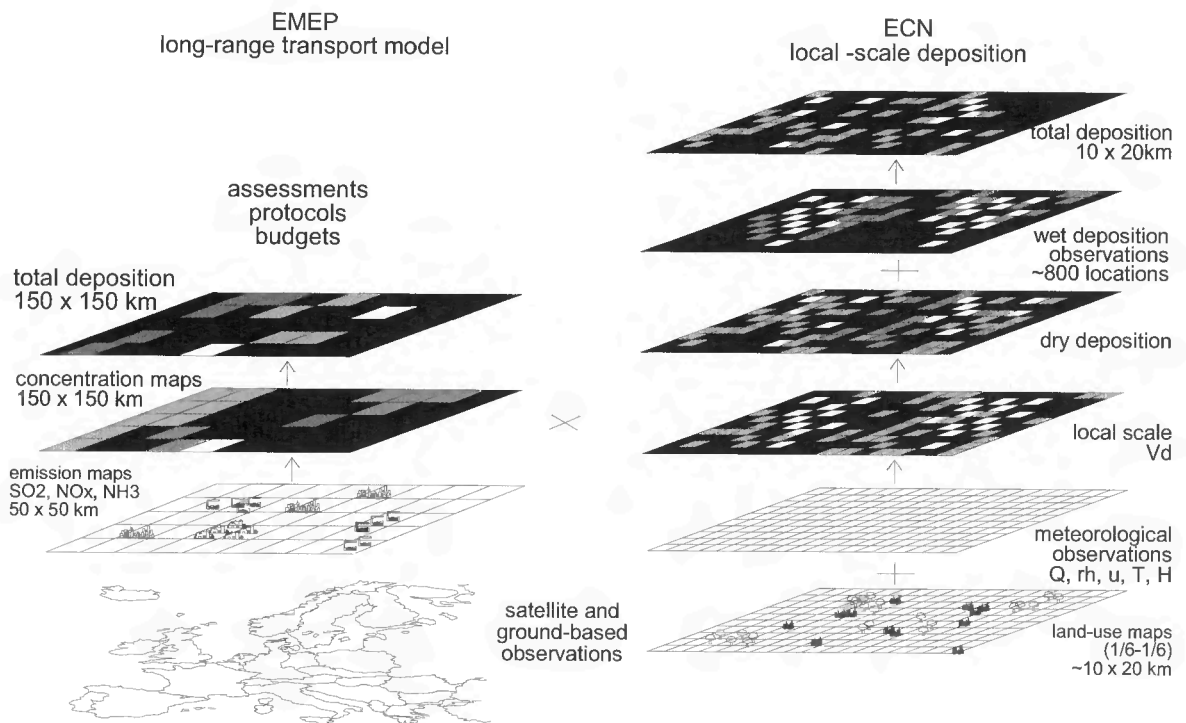


Figure A5.1. Outline of method to estimate local scale deposition fluxes.

### Comparison of model results with throughfall measurements

The EMEP model results are only available until 1996. Since then, a new model was developed and is still not operational. The comparison of model results with throughfall data has therefore been made for 1996 (223 plots). Figure A5.2 shows the model comparison with throughfall measurements at the plots. The correlation for sulphur is small (Fig. A5.2A). This is especially due to an overestimation of sulphur deposition at a few sites. These sites are all located close to or in the Black triangle. It appears that the EMEP model overestimates  $\text{SO}_2$  concentrations in these areas. On average the modelled S deposition and measured throughfall are comparable, however, indicating the nearly negligible influence of canopy uptake of sulphur. A best regression estimate was

$$\text{SO}_{4,\text{model}} = 530 + 0.63 \cdot \text{SO}_{4,\text{throughfall}} \quad R^2_{\text{adj}} = 0.32 \quad (\text{A5.1})$$

The N deposition, both of  $\text{NO}_3$  and  $\text{NH}_4$  are considerably larger than the measured throughfall, although the correlation is larger than for  $\text{SO}_4$  (Fig. A5.2B, C, D). Specifically the reduced N deposition is higher up to a factor of two, despite the high correlation. Best regression estimate were:

$$\text{NO}_{3,\text{model}} = 540 + 0.75 \cdot \text{NO}_{3,\text{throughfall}} \quad R^2_{\text{adj}} = 0.37 \quad (\text{A5.2})$$

$$\text{NH}_{4,\text{model}} = 610 + 1.54 \cdot \text{NH}_{4,\text{throughfall}} \quad R^2_{\text{adj}} = 0.64 \quad (\text{A5.3})$$

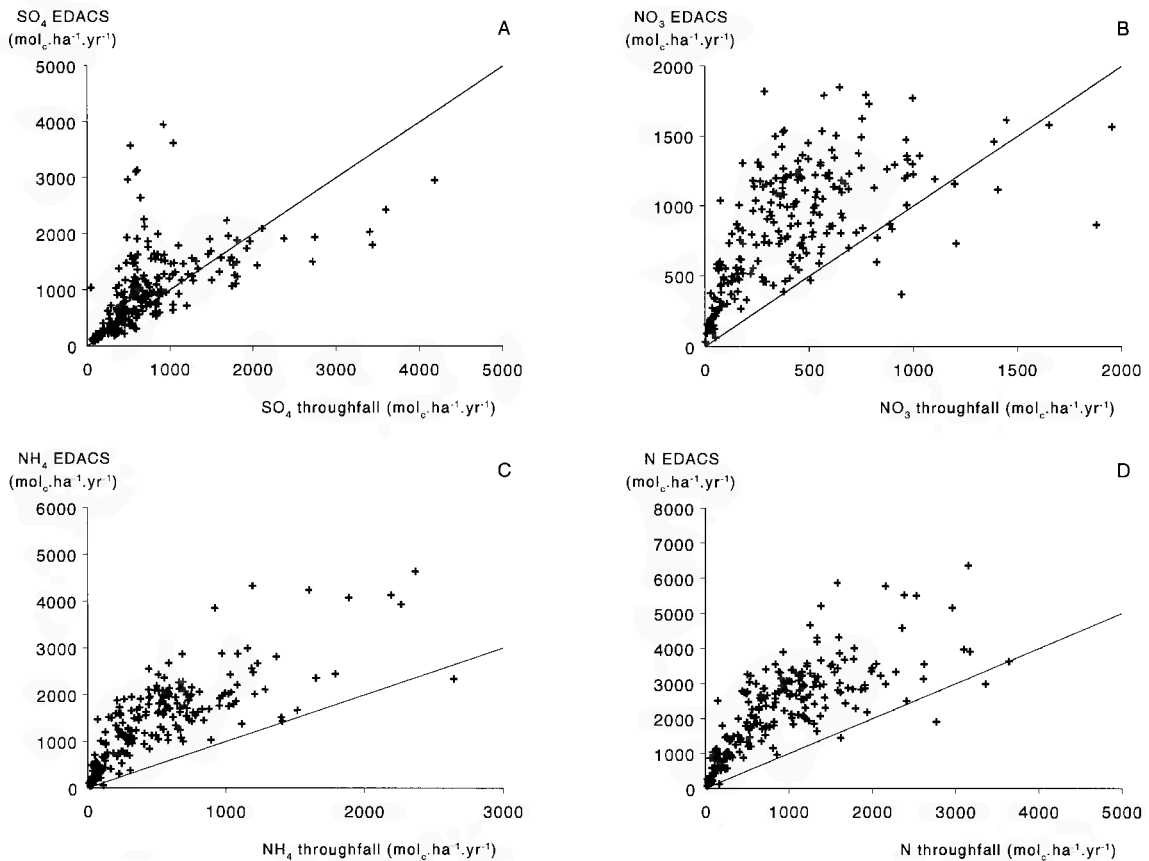


Figure A5.2 Comparison of total deposition calculated with EDACS with throughfall measurements of SO<sub>4</sub> (A), NO<sub>3</sub> (B), NH<sub>4</sub> (C) and total N (D).

The throughfall data are not corrected for canopy exchange. Assuming that total deposition model estimates are correct, canopy exchange can be estimated by subtracting throughfall measurements from the model estimates. This estimate also includes input by stemflow. Results for canopy exchange thus obtained for NH<sub>4</sub> and NO<sub>3</sub> as a function of the input by throughfall are given in Figure A5.3. Results show a decrease in calculated canopy uptake for NO<sub>3</sub> with increasing NO<sub>3</sub> input by throughfall (Fig. A5.3A). Considering the fact that NO<sub>3</sub> exchange in the canopy is low, this result indicates a tendency to overestimate NO<sub>3</sub> deposition at low throughfall inputs and underestimate NO<sub>3</sub> deposition at high throughfall inputs. As expected, NH<sub>4</sub> exchange is nearly always positive, but at a large number of plots the exchange exceeds 800 mol<sub>c</sub>·ha<sup>-1</sup>·yr<sup>-1</sup> which is generally considered a maximum. This implies that NH<sub>4</sub> deposition seems to be overestimated at a large number of plots.

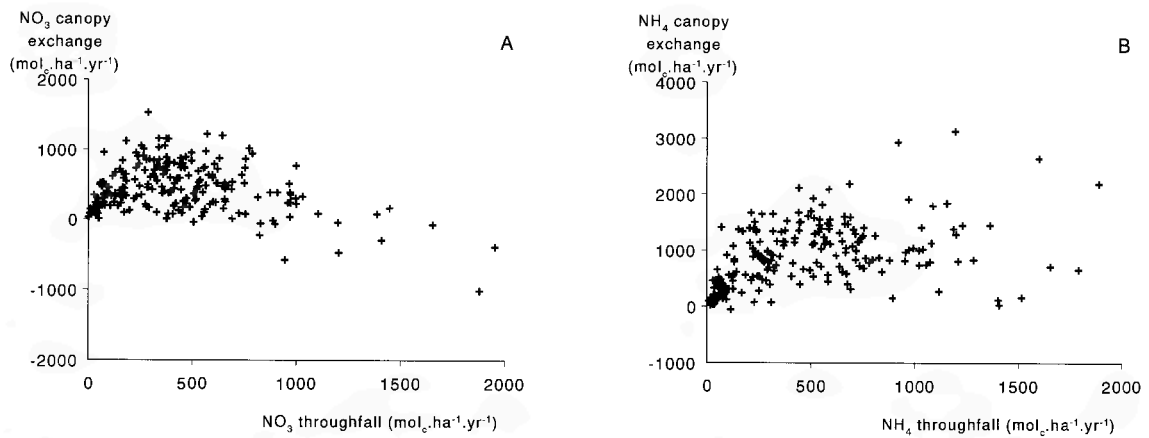


Figure A5.3 Comparison of canopy exchange calculated with from EDACS results and throughfall measurements as a function of throughfall for  $\text{NO}_3$  (A) and  $\text{NH}_4$  (B).

These preliminary results show the potential for the EDACS model to be used for upscaling. There is, however, a need to improve the results. The most important improvement is the use of concentration data at a smaller grid. These will be available from the new EMEP model. Furthermore, tests with the Dutch LOTOS model and with the DEM model from Denmark will be done. Both models calculate concentrations at a much smaller scale. Finally, the deposition velocity parameterisation used in the model can be improved using micrometeorological measurements, which are made in several countries.



## **Annex 6 Relationships between soil nitrate concentrations and environmental factors**

### **Introduction**

Increased nitrogen (N) deposition to forests may gradually change a range of N cycling processes and lead to nitrogen saturation of the ecosystem, i.e. elevated nitrate leaching will occur (Aber et al., 1995). Compilations of input-output budgets from European forests have shown that a major part of the investigated sites that received more than  $10 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  in throughfall are leaching  $>5 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  (Dise and Wright, 1995; Gundersen, 1995). However, the relationships between input and output of nitrogen were rather weak. This variability in forest ecosystem response to N inputs was further studied in data compilations on element cycling and ecosystem characteristics from published European studies in plots and catchments ( $n=139$ ) from 1970 to 1995 (Dise et al., 1998a, b) or in plots alone ( $n=80$ ) in a more narrow time span 1985-95 (Gundersen et al., 1998a). Incomplete data sets hampered a thorough statistical analysis of all the parameters considered. Nevertheless, these data sets revealed relationships between N deposition, N concentrations in foliage, some ecosystem characteristics and leaching of nitrate. The regression model that explained most of the variability in nitrate leaching included both N input and C/N ratio of the forest floor, but this was based on only 30-40 sites with a strong bias towards the coniferous forests. Differences in methodology, as well as in temporal and spatial scales may contribute an unknown error in detecting relationships in these databases.

With the growing database from the Intensive Monitoring programme some of the drawbacks of the literature compilations can be avoided. However, in plot studies leaching fluxes can only be obtained by construction of a detailed water balance for each forest site (Chapter 4). This is resource demanding for data collection and the calculations may add a considerable amount of error to the estimated fluxes. This may be a reason for the relative poor correlations found between element fluxes in chapter 5.

It may therefore be advantageous to use the measured soil solution nitrate concentrations directly in this kind of regional scale analysis. In a national survey in Denmark, Callesen et al. (1999) found significant relationships between nitrate concentrations in soil solution and forest characteristics such as management type and soil texture class. The approach, further made use of the concentration time series to extract information about seasonal and annual variations in nitrate concentrations.

The results of the Intensive Monitoring programme offer an opportunity for analyses of relationships between nitrate concentrations measured in soil solutions collected over time and a large number of site-specific factors. The purpose of the present study was to search for relationships between atmospheric N deposition, forest biogeochemistry and nitrate leaching based on the time series of nitrate concentrations in soil solution. The data was exploited by a statistical approach based on mixed linear model theory (Littell et al., 1996) as illustrated by Callesen et al. (1999).

## Materials and Methods

### *Site and data description*

The Intensive Monitoring programme is carried out on forest plots selected by each participating country in Europe. The forest plots included in the data base used in the present study were dominated by either broadleaf species (*Quercus robur*, *Quercus petraea*, *Fagus sylvatica*) or conifer species (*Picea abies*, *Picea sitchensis*, *Picea alba*, *Pinus sylvestris*, *Pinus nigra*, *Pseudotsuga menziesii*) with the average stand age ranging from 30 to 130 years. The plots are situated within the latitudes 43°N-69°N, at altitudes from 25 to 1375 m above sea level, and the pH<sub>(0.01 CaCl<sub>2</sub>)</sub> was in the range of 2.8-7.3 in the top 0-10 cm layer of mineral soil. The N deposition in throughfall covered a range of 0-42 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>. The Intensive Monitoring programme includes surveys of atmospheric deposition, meteorology, soil characteristics, tree species, foliar and soil solution chemistry among others. Details on sampling and chemical analyses are found in De Vries et al. (1998, 1999, 2000a) and in Annex 3. In the survey, the organic topsoil (forest floor) was defined as the O-horizon on top of the mineral soil (ICP-Forest Manual).

All sites where soil solution nitrate concentration had been measured in the period Jan. 1996 to Jan. 1998 and where data on main biogeochemical properties were available were included in the analyses (n=111). The soil nitrate concentration was at most sites measured at several depths. The concentration measured in the depth closest to 100 cm was chosen to represent nitrate concentration in soil water leaching from the rhizosphere. The chosen depths ranged from 40-250 cm. The results were obtained from analysis of solution collected continuously in suction cups or zero tension lysimeters. The results from some sites/countries were excluded from the analyses due to the use of centrifugation for collection of soil solution or due to very limited time resolution of the results (e.g. the Netherlands). Soil centrifugation has been found to give higher results for nitrate concentration compared to suction cup lysimeters (De Vries et al., 1999). The countries included in the analysis were France (n=15), Germany (n=55), Belgium (n=6), Sweden (n=2), Norway (n=15), Finland (n=4), UK (n=4), Ireland (n=3), Austria (n=1) and Denmark (n=6). Throughfall and bulk deposition of N was calculated as an average of the estimated annual deposition rates per plot varying between 1-5 estimates in the period 1993-1997. We used this longer-term average as a measure of 'pollution load' instead of the throughfall N deposition from the actual soil solution time series alone, since the lifetime of N in the ecosystem may be several years. In the statistical analyses we preferred the use of throughfall N deposition as a measure for N input to the sites instead of bulk N deposition. Throughfall N deposition includes the major part of the dry deposition and relates therefore more closely to the actual N input to the system than does bulk N deposition. Estimates of total deposition derived in Chapter 6.2.5 were not considered, since we wanted to concentrate on directly measured parameters.

### *Statistical models*

A preliminary statistical analysis was performed to indicate significant relationships between biogeochemical characteristics and soil solution nitrate concentration. Simple linear regression analyses and test of homogeneity of slopes (SAS proc glm) were performed on average soil nitrate concentration calculated for each site from measurements in the period Jan. 1996 to Jan. 1998. The factors found to be significantly related to average soil solution nitrate concentrations were climate, throughfall N, foliage N content, tree-type (conifers, broadleaves), stand age, C/N ratio of the organic layer, humus type (mull, moder, mor, raw humus), soil pH, phosphorous

content of mineral soil (0-10 cm) and soil classification. These factors were subsequently examined in the construction of a mixed linear model (SAS proc mixed).

The data on soil nitrate concentrations (response variable) used in the mixed model were the same time series as averaged in the preliminary analyses. However, to overcome the problem that different sampling frequencies were used at the sites, an average nitrate concentration was calculated for each season (spring (March-May), summer (June-Aug.), autumn (Sept.-Nov.) and winter (Dec.-Feb.)) in the period winter 1996 to winter 1998; that is, 3-9 repeated measurements per site.

The longitudinal character of the data was taken into account by using a split-plot type design with sites as 'main plots' and season results as 'subplots' (Christensen, 1996). Equivalently this means that the factor site has a random effect in the model, or that a compound symmetry dependence structure is assumed for the series of observations at each site (Littell et al., 1996). The soil nitrate concentrations were transformed prior to analysis by the function  $y = \log(x + 0.05)$ . The value 0.05 (mg NO<sub>3</sub>-N.l<sup>-1</sup>) was added to avoid observations equal to zero, which are not allowed for the transformation class. The log transformation was recommended by maximum likelihood estimation. Due to the transformation, the resulting estimates and confidence intervals were calculated by retransformation to the original scale and therefore correspond to median (midpoint) nitrate concentrations (Parkin and Robinson, 1994).

Throughfall N was found to have a strong relationship to soil nitrate when tested alone in the mixed model analysis. Each other factor was subsequently tested for significant effects on soil nitrate concentration in a preliminary model including throughfall N. The factors found to be significant were: tree-type, C/N ratio of the organic layer, humus type, phosphorous content of mineral soil (0-10 cm), and foliage N content. The factors that were found not to be significant were age, soil classification, climate and soil pH. The model was furthermore used to test for interactions (covariance). The significant factors and interactions were used to construct a common mixed model, which then was reduced for insignificant factors. The formal writing of the resulting model for soil solution nitrate observations at site *s* and time *t* is

$$\text{Soil NO}_3^-(s, t) = \text{season}(t) + \text{throughfall N}(s) + \text{tree-type}(s) + \text{throughfall N} \cdot \text{tree-type}(s) + \text{C/N organic layer}(s) + \text{site}(s) + \text{error}(s, t) \quad (\text{A6.1})$$

Here the terms 'site' and 'error' are random Gaussian variables. An  $r^2$  of 0.92 was calculated for the model by use of sum of squares of observed response variables and residuals from predicted response variables (SAS proc univariate). The  $r^2$  is not a true coefficient of determination of the complex model, but it illustrates the impact of variation in the model factors on soil nitrate concentrations. The  $r^2$  can not be interpreted like in a traditional regression model, since some of the variation is captured in the 'site' term.

The model was used to estimate medians for each level of main factors. The C/N of the organic layer was for this reason included in the model after classification into the levels  $C/N < 25$ ,  $25 < C/N < 30$ , and  $C/N > 30$  based on the hypothesis of Gundersen et al. (1998a). Medians were calculated as weighted values based on the distribution of sites at different levels of main factors. Differences between weighted medians were tested by Student *t*-tests with Bonferroni correction for multiple comparisons (Christensen, 1996). Due to significant interaction in the model between throughfall N and tree-type, it is called a covariate by treatment interaction model (Littell et al.,

1996). This means that differences between factors with covariates should be tested at different levels of the covariate. Therefore, differences between conifers and broadleaves were tested at the average throughfall N deposition calculated separately for conifers and broadleaves. Both averages happened to be at  $14.2 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ . Furthermore a low ( $4.0 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ ) and a high ( $22.8 \text{ kg N.ha}^{-1}.\text{yr}^{-1}$ ) value of throughfall N was tested. These values represented the minimum and maximum throughfall N measured at broadleaf sites. Foliage N content was not included in the mixed model analyses due to strong covariance with throughfall N and due to species specific differences.

The same procedure as described above was used to construct separate mixed models for conifers and broadleaves. The significance of all factors and interactions were tested followed by reduction of the resulting model by removal of insignificant factors and interactions. The formal writing of the resulting models is:

Conifers ( $r^2=0.93$ ):

$$\text{Soil NO}_3^-(s, t) = \text{season (t)} + \text{throughfall N (s)} + \text{C/N organic layer (s)} + \text{site(s)} + \text{error(s, t)} \quad (\text{A6.2})$$

Broadleaves ( $r^2=0.87$ ):

$$\text{Soil NO}_3^-(s, t) = \text{season (t)} + \text{throughfall N (s)} + \text{soil pH(0-10 cm) (s)} + \text{site(s)} + \text{error(s, t)} \quad (\text{A6.3})$$

## Results and Discussion

### *Seasonal and annual effects*

Differences in soil nitrate concentrations between seasons were significant ( $p=0.011$ ) in the overall model (Eq. A6.1) and a significant difference was found between winter and summer concentrations in the subsequent test between weighted medians (Table A6.1). No difference between years was detected. Winter concentrations were lower than summer concentrations, which may be explained by the diluting effect of increased water infiltration due to decrease of evapotranspiration, as well as by decreased nitrifying activity during the cold season. Although significant, the differences between seasons were relative small. This may mean that the uncertainty due to modelled timing of water flow (Chapter 4) may not add large errors to the estimates of nitrate leaching fluxes.

Table A6.1 Estimated medians of nitrate concentration in soil solution and test of significant differences between estimates at the average throughfall of 14.2 kg N.ha<sup>-1</sup>.yr<sup>-1</sup> in the common mixed model for broadleaves and conifers. Different letters indicate significant differences between estimated medians for each main factor.

Variable	Sites (n)	Observations (n)	Estimate mg.l <sup>-1</sup>	Confidence interval (95%)
Overall median	104	777	0.53	0.38 – 0.74
Season (p=0.011)				
Winter	103	225	0.47 a	0.33 – 0.67
Spring	104	183	0.55 ab	0.39 – 0.77
Summer	104	185	0.59 b	0.41 – 0.82
Autumn	103	184	0.52 ab	0.37 – 0.74
Throughfall N (p<0.0001)	104	777		
Tree-type (p=0.276)				
Tree-type:throughfall N (p=0.045)	104	777		
Conifers	73	545	0.32 a	0.24 – 0.43
Broadleaves	31	232	0.64 b	0.42 – 0.95
C/N organic layer (p=0.025)				
>30	30	228	0.13 a	0.07 – 0.22
25-30	25	188	0.47 b	0.24 – 0.88
<25	49	361	1.23 c	0.84 – 1.78

### Throughfall and bulk N deposition

The strongest relationship to nitrate concentrations was found for throughfall N deposition (p<0.0001). At the average throughfall of 14.2 kg N.ha<sup>-1</sup>.yr<sup>-1</sup> this resulted in significantly higher nitrate concentrations below broadleaves compared to conifers (Table A6.1). There was, however, also a significant interaction between throughfall N and tree-type (p=0.045). This interaction also appears from linear regression between throughfall N and the average nitrate concentration in soil solution (Fig. A6.1).

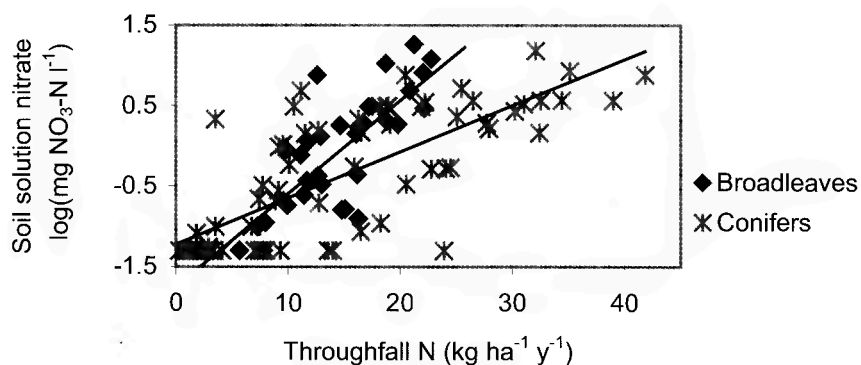


Figure A6.1 Average nitrate concentration in soil solution Jan. 1996 to Jan. 1998 (log transformed) vs throughfall N flux (1993-1997) at Intensive Monitoring plots. Linear regressions yield for broadleaves:  $\log(y) = 0.12x - 1.8$ ;  $r^2=0.65$ ; for conifers:  $\log(y) = 0.06x - 1.2$ ;  $r^2=0.59$ .

Thus the difference between nitrate concentrations below broadleaves and conifers depended on the level of throughfall N deposition and was therefore tested further for significant differences at a low and high level of throughfall N (see explanation in the Materials and Methods section). In accordance with the tree-type differences illustrated in Fig. A6.1, the test found the nitrate concentration to be higher below broadleaves compared to conifers (Table A6.2) both at average and high throughfall N deposition (14.2 and 22.8 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>), while no difference was found at

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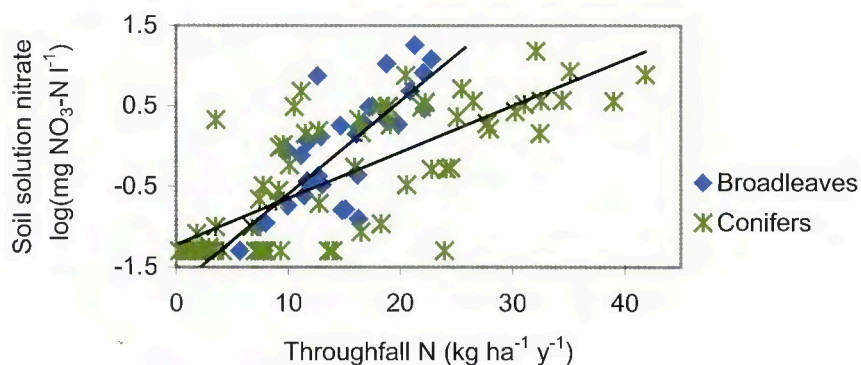


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Thus the difference between nitrate concentrations below broadleaves and conifers depended on the level of throughfall N deposition and was therefore tested further for significant differences at a low and high level of throughfall N (see explanation in the Materials and Methods section). In accordance with the tree-type differences illustrated in Fig. A6.1, the test found the nitrate concentration to be higher below broadleaves compared to conifers (Table A6.2) both at average and high throughfall N deposition (14.2 and 22.8 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>), while no difference was found at

low deposition level ( $4.0 \text{ kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ ). The regression lines in Fig. A6.1 indicate that nitrate concentration in soil solution under broadleaves will respond much more to deposition changes (at the double exponential rate) than under conifers.

Table A6.2 Estimated medians of nitrate concentration in soil solution ( $\text{mg N}\cdot\text{l}^{-1}$ ) and test of significant differences between estimates at three different levels of throughfall N in the common mixed model for broadleaves and conifers. Different letters indicate significant differences between estimated medians.

Tree type	Low throughfall N ( $4.0 \text{ kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ )	Average throughfall N ( $14.2 \text{ kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ )	High throughfall N ( $22.8 \text{ kg N}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ )
Conifers	0.04	0.33 a	0.97 a
Broadleaves	0.01	0.64 b	4.03 b

Bulk N deposition correlates with throughfall N deposition (Fig. A6.2) and can usually replace throughfall N deposition in the relationships found in this type of regional analysis (Dise and Wright, 1995; Gundersen, 1995; Tietema and Beier, 1995).

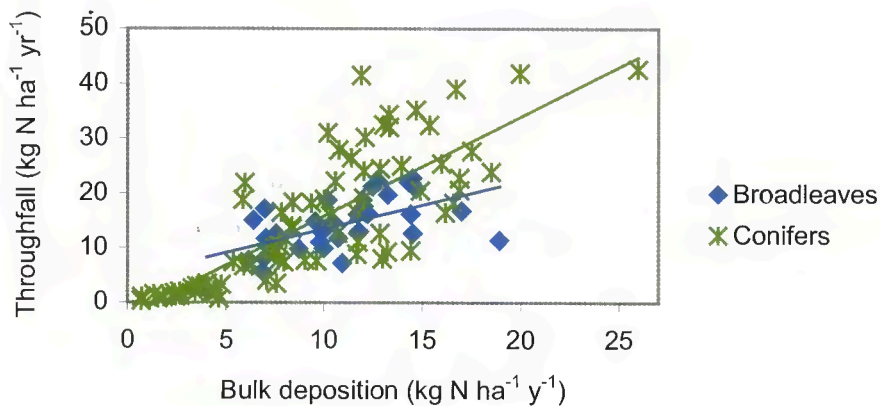


Figure A6.2 Throughfall vs bulk precipitation average N fluxes (1993-1997) at Intensive Monitoring plots ( $r^2=0.64$  for conifers;  $r^2=0.33$  for broadleaves).

When throughfall N deposition was replaced by bulk N deposition in the statistical model (Eq. A6.1) there was no interaction between tree-type and bulk N deposition ( $p=0.93$ ) and neither was there an effect of tree-type alone ( $p=0.73$ ). The other factors stayed significant in the model (results not shown). The differences found between nitrate concentrations below broadleaves and conifers may be interpreted as a result of the complex interactions between N deposition, forest biogeochemistry and nitrate leaching. Conifers are found to have higher throughfall N deposition than broadleaves at high levels of bulk N deposition (Fig. A6.2). This is due to the larger filtering effect of the conifer canopy compared to that of broadleaves caused by the larger canopy surface and roughness as well as the evergreen nature of conifers (e.g. Rothe et al., 2001). This difference is not observed at low N deposition levels where direct assimilation of ammonium in the canopy decrease throughfall N relative to bulk N deposition (Fig. A6.2). Ammonium is known to constitute a relatively larger part of throughfall N deposition compared to nitrate at higher deposition levels (Dise et al., 1998a), which is also apparent in the present dataset (Fig. A6.3).

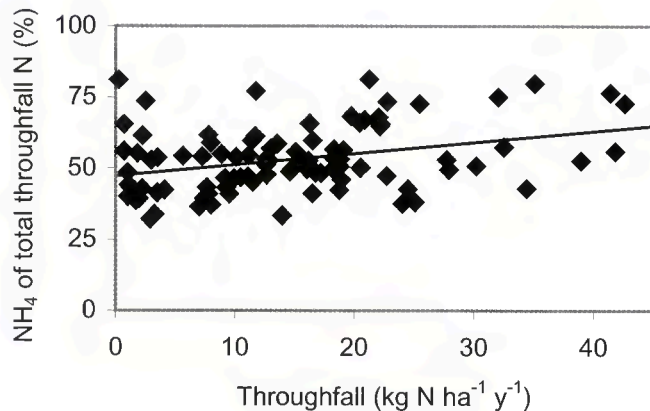


Figure A6.3 Ammonium contribution in throughfall in relation to the total throughfall N flux (average 1993-1997) at Intensive Monitoring plots.

Thus conifer stands experience higher throughfall N deposition levels and relatively higher contributions of ammonium N compared to stands dominated by broadleaves. Input-output budgets indicate that ammonium from atmospheric deposition is better retained in forest ecosystems compared to nitrate probably due to differences in mobility of the two ions (Gundersen, 1995; Dise et al., 1998a). This may explain why conifers (at throughfall  $>8 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ ) have lower soil nitrate concentrations than broadleaves at the *same throughfall N deposition* level (Fig. A6.1) and further that this differences is absent when compared at the *same bulk N deposition*.

### **The forest floor C/N ratio**

The C/N ratio of the organic topsoil was the only biogeochemical property that together with throughfall N deposition contributed significantly to explain nitrate concentrations in the common model (Eq. A6.1) for broadleaves and conifers (Table A6.1). In the preceding regression analysis on average nitrate concentrations the C/N ratio was not a significant factor ( $p=0.19$ ) in a linear model with throughfall N. This indicates that use of the time series information (seasonal fluctuations in nitrate concentrations) increase the analytical power of the statistical models.

Weighted medians of nitrate concentration in the three C/N classes were significantly different increasing from 0.13 over 0.49 to 1.24  $\text{mg} \cdot \text{l}^{-1}$  in the  $>30$ , 25-30 and  $<25$  class, respectively (Table A6.1). In the separate model analysis of the two tree-types (Eq. A6.2 and A6.3) C/N ratio of the organic topsoil entered the model for conifers ( $p=0.029$ ), but not for broadleaves. The results from conifers support the hypothesis of Gundersen et al. (1998a) based on analysis of published European data mainly from coniferous forests. The hypothesis says that the C/N ratio in the organic topsoil may be an indicator of the N status of the ecosystem and therefore also a predictor of the risk for nitrate leaching from the system. Gundersen et al. (1998a) suggested that a C/N ratio above 30 characterises forests with a low risk for nitrate leaching, a ratio between 25-30 characterises an intermediate risk, while a ratio below 25 characterises forests with a high risk for nitrate leaching. In an extended European dataset including also catchment studies Dise et al. (1998b) found that sites with throughfall N deposition below  $10 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  had low nitrate leaching regardless of C/N. At sites with deposition levels of  $10\text{-}30 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  nitrate leaching increased with decreasing C/N ratio. With deposition levels above  $30 \text{ kg N} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$  the results were more variable and nitrate leaching was observed at all sites including those with C/N ratio



above 30. Using the same deposition classes on the Intensive Monitoring data for conifers similar relations between average soil nitrate concentrations and C/N ratio of the organic topsoil was observed (Fig. A6.4):

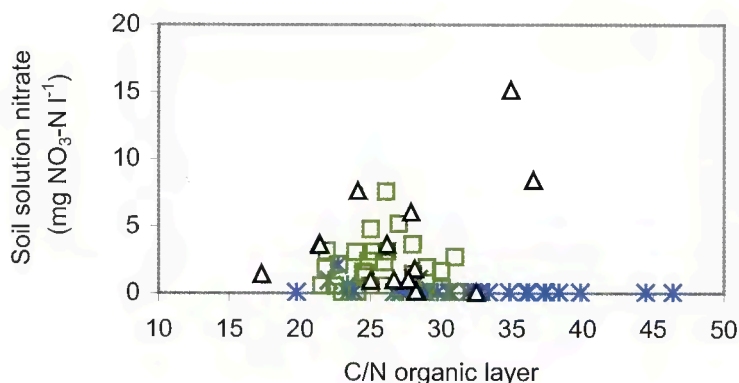


Figure A6.4 Average nitrate concentrations in soil solution Jan. 1996 to Jan. 1998 vs forest floor C/N ratio at low (\* < 10 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>), intermediate (□ 10-30 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>) and high (Δ >30 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>) throughfall N deposition at Intensive Monitoring plots.

Sites with deposition levels below 10 kg N.ha<sup>-1</sup>.yr<sup>-1</sup> in general had soil solution nitrate concentrations close to zero irrespective of C/N ratio. Sites with high nitrate concentrations at C/N ratios above 30 all had deposition levels well above 30 kg N.ha<sup>-1</sup>.yr<sup>-1</sup>. One of these sites (C/N ratio 36.5; nitrate concentration 8.4 mg.l<sup>-1</sup>) has been studied in detail by De Schrijver et al. (2000). They suggest that the mechanism behind leaching of nitrate at this high C/N ratio is due to transport of throughfall ammonium down through the upper soil layers to deeper layers with lower C/N ratio and higher nitrifying activity.

***Intercorrelations between C/N, throughfall N, foliage N and soil nitrate concentrations***

The C/N ratio of the organic layer is related to other ecosystem N fluxes and concentrations and may as such be an indicator of ecosystem N status (Gundersen et al., 1998b) as well as of the risk of nitrate leaching (Gundersen et al., 1998a). With time the elevated N input may increase N concentrations and fluxes in the ecosystem and the C/N ratio of the forest floor may thus decrease. The decades of elevated deposition seem to have influenced the C/N ratio of this layer in European forests, since a significant negative linear relationships was found between the throughfall N deposition and the C/N ratio for both tree-types (Fig. A6.5).

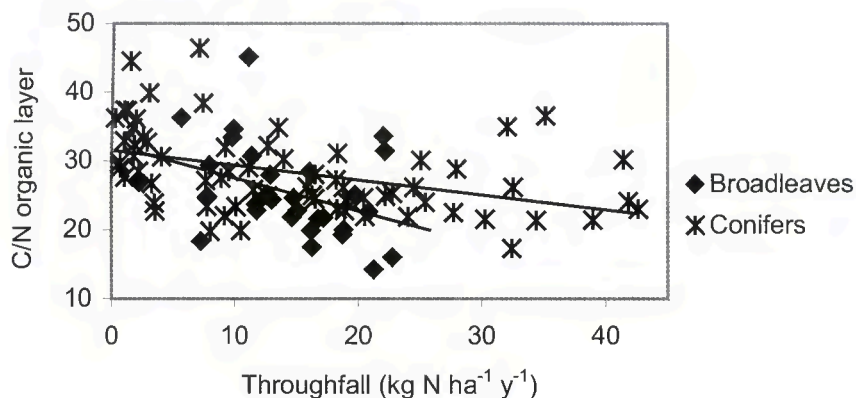


Figure A6.5 C/N ratio of the organic layer vs throughfall N flux (1993-1997) at Intensive Monitoring plots. The regressions are conifers:  $y = 31.5 - 0.21x$  ( $r^2 = 0.19$ ,  $p = 0.0002$ ); broadleaves:  $y = 32.3 - 0.48x$  ( $r^2 = 0.14$ ,  $p = 0.036$ ).

The relationships suggest a 2 unit decrease of forest floor C/N in conifers and a 5 unit decrease in broadleaves per 10 kg N.ha<sup>-1</sup>.yr<sup>-1</sup> increase in throughfall N (Fig. A6.5). However, since the deposition gradient decrease northward like temperature, the decreasing trend in C/N with deposition may have a climatic component as well. Also the relationship for broadleaves was rather weak.

Another factor of importance for the C/N ratio is the N status of the litter input. This was not measured at the sites, but so was foliage N concentration, that have been shown to correlate with litter N concentration (Tietema and Beier, 1995). A significant negative correlation between foliage N concentrations and the C/N of the organic topsoil was found for conifers (Fig. A6.6), however, not for broadleaves ( $p=0.52$ ).

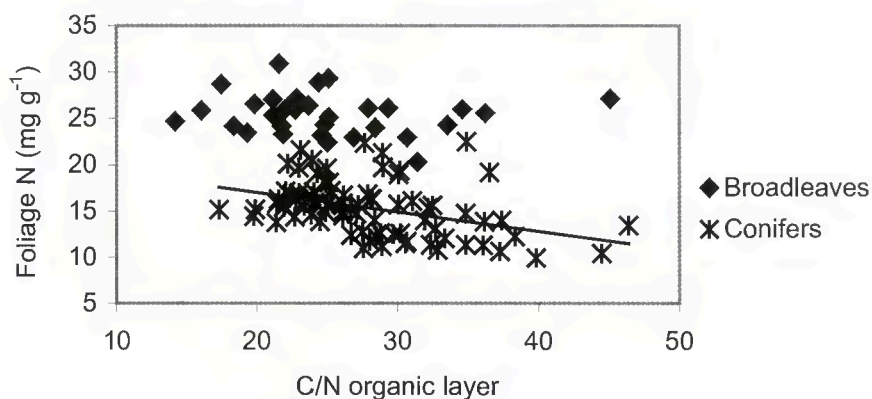


Figure A6.6 Foliage N concentration vs organic layer C/N ratio at Intensive Monitoring plots. The regression for conifers is  $y = 21.2 - 0.21x$  ( $r^2 = 0.16$ ,  $p = 0.0003$ ).

The figure further shows that the foliage N content differed between broadleaves and conifers with an average of 25.3 and 14.8 mg.g<sup>-1</sup>, respectively. The foliage N content was then again for conifers significantly related to the throughfall N deposition (Fig. A6.7), but not for broadleaves ( $p=0.71$ ).

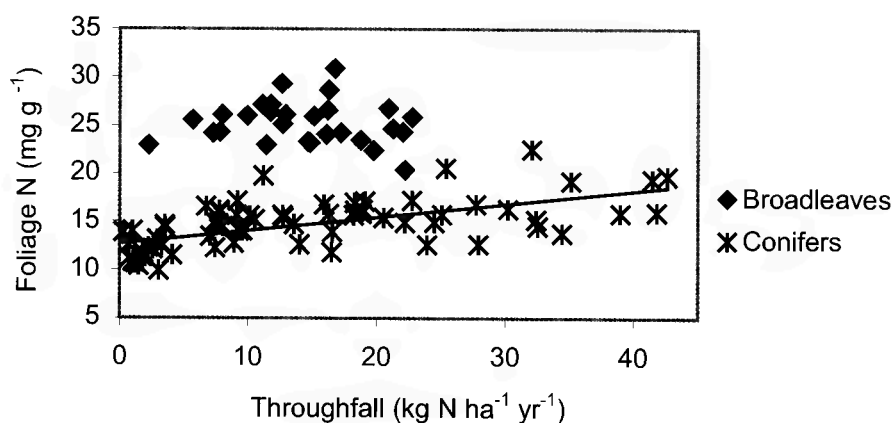


Figure A6.7 Foliage N concentration vs throughfall N flux (1993-1997) at Intensive Monitoring plots. The regression for conifers is:  $y = 12.7 + 0.14x$  ( $r^2=0.40$ ,  $p<0.0001$ ).

With the significant intercorrelation between throughfall N, C/N ratio and foliage N concentration for conifers a relationship between soil nitrate concentrations and foliage N content may be expected in conifers as shown in Fig. A6.8.

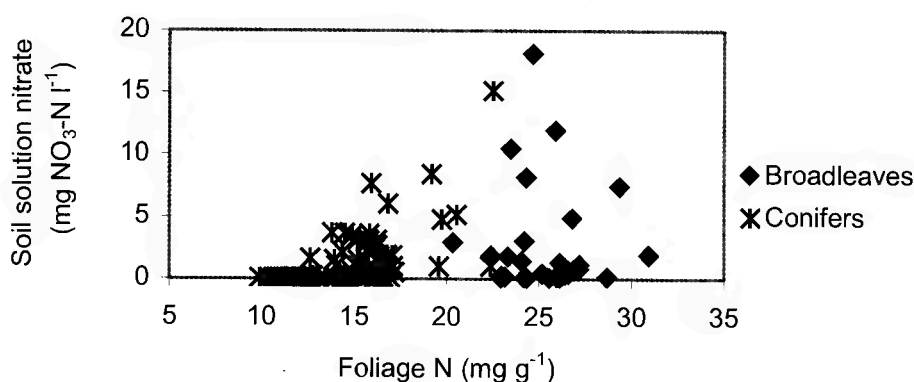


Figure A6.8 Average nitrate concentrations in soil solution Jan. 1996 to Jan. 1998 vs foliage N concentration at Intensive Monitoring plots. For conifers the correlation is significant ( $r^2=0.34$ ,  $p<0.0001$ ).

The results in Fig. A6.8 further show that a threshold can be set for conifer foliage N content of 12.6 mg.g<sup>-1</sup> below which no nitrate was found in soil solution, and of 17.0 above which the nitrate concentration was always above detection limit. These limits correspond well with those suggested by Gundersen (1999) from other European data. When replacing the C/N ratio in Eq. A6.2 with foliage N content in the mixed model analysis of conifers a significant relationship to nitrate concentrations ( $p<0.0001$ ) was also found.

In accordance with the lack of correlation for broadleaves in Fig. A6.6, A6.7 and A6.8, neither the C/N ratio ( $p=0.54$ ) nor the foliage N content ( $p=0.89$ ) showed significant relationships to nitrate concentration in soil solutions in the mixed model analysis for broadleaves.

### ***Relations to soil pH and humus type***

The pH of the mineral soil (0-10 cm depth) contributed to explain soil nitrate concentrations in the separate mixed model for broadleaves (Eq. A6.3). Previous studies have shown a relationship between pH of the B-horizon of European forests and nitrate leaching which may be explained by the acidifying effect on the soil of nitrification and leaching of nitrate from deposition (Dise et al., 1998a). Such relation was also found for conifers in the present study but only when throughfall N deposition was not included in the model (results not shown). Differently for broadleaves the pH was indicated to relate to nitrate concentrations more than what could be explained by the acidifying effect of N deposition, since both soil pH and throughfall N are included in the final model (Eq. A6.3). This may mean that soil type and characteristics are more important for the response to N deposition in broadleaf than in coniferous forests.

A biogeochemical property of humus type of the organic topsoil was found to be significantly related to nitrate concentrations when included in the common model for broadleaves and conifers ( $p=0.032$ ) as well as in the separate model for conifers ( $p=0.035$ ) (results not shown). The humus types of forest sites were classified as raw humus, mor, moder and mull. The estimated medians of nitrate concentration for the classes decreased in the order mor>moder>mull>raw humus. This result has no clear relation to the ecological function of the organic topsoil as based on present knowledge. The effect may further be questioned as the reliability of the humus classification is low (Vanmechelen et al. 1997).

### **Conclusion**

The mixed model analysis of the level II time series of soil nitrate concentrations increases the analytical power above that of correlation and regression statistics on concentrations means. Soil nitrate concentrations in winter are generally lower than in summer, but differences are small. The response of coniferous and broadleaf forest to N deposition is different and the tree-types need to be analysed separately.

In coniferous forests N input with throughfall, foliage N concentration, forest floor C/N ratio and nitrate leaching are interrelated variables. Soil nitrate concentrations are best explained by a model with throughfall N and forest floor C/N as main factors, though C/N ratio could be replaced by foliage N. The results confirm conclusions from other datasets, that the forest floor C/N ratio classes >30, 25-30 and <25 as well as the foliage N ( $\text{mg N.g}^{-1}$ ) classes <13, 13-17 and >17 indicate low, intermediate and high risk of nitrate leaching, respectively.

In broadleaved forests, correlations between N characteristics are less pronounced. A model including throughfall N and soil pH (0-10 cm) as main factors best explained soil nitrate concentrations. The responses of soil nitrate concentration to changes in N deposition will probably be more pronounced in broadleaf than in coniferous forests.

## Annex 7 Addresses

### 1. ICP Forests and European Commission

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