Carbon-nitrogen interactions in forest ecosystems

final report

Gundersen, Per; Berg, Bjørn; Currie, W.S.; Dise, N.B.; Emmett, B.A.; Gauci, V.; Holmberg, M.; Kjønaas, O.J.; Mol-Dijkstra, J.; van der Salm, C.; Schmidt, Inger Kappel; Tietema, A.; Wessel, W.W.; Vestergarden, L.S.; Akselsson, C.; Vries, W. De; Forsius, M.; Kros, H.; Matzner, E.; Moldan, F.; Nadelhoffer, K.J.; Nilsson, L.-O.; Reinds, G.J.; Rosengren, U.; Stuanes, A.O.; Wright, R.F.

Carbon-Nitrogen Interactions in Forest Ecosystems

Final Report

P. Gundersen, B. Berg, W. S., Currie, N.B. Dise, B.A. Emmett,
V. Gauci, M. Holmberg, O.J. Kjønaas, J. Mol-Dijkstra,
C. van der Salm, I.K. Schmidt, A. Tietema, W.W. Wessel,
L.S.Vestgarden, C. Akselsson, W. De Vries, M. Forsius, H. Kros,
E. Matzner, F. Moldan, K. J. Nadelhoffer, L.-O. Nilsson,
G.J. Reinds, U. Rosengren, A.O.Stuanes and R.F Wright

KVL | **A UNIVERSITY OF LIFE SCIENC**

Preface

This report is a summary of the main results from the EU project "Carbon – Nitrogen Interactions in Forest Ecosystems" (CNTER) which was undertaken in 2001-2005 by eleven partners from seven countries.

The partners and the involved principal investigators were: Open University in Milton Keynes, UK (Nancy B. Dise, Vincent Gauci), Centre for Ecology and Hydrology in Bangor, UK (Bridgett Emmett), University of Bayreuth, Germany (Egbert Matzner, Bjørn Berg), Norwegian Forest Research Institute in Ås, Norway (Janne Kjønaas), Agricultural University of Norway (Arne Stuanes, Live S. Vestgarden), Alterra Green World Research in Wageningen, the Netherlands (Janet Mol-Dijkstra, Caroline van der Salm), University of Amsterdam, the Netherlands (Albert Tietema, Wim W. Wessel), Finnish Environment Institute in Helsinki (Martin Forsius, Maria Holmberg) University of Michigan, USA (William S. Currie, Knute Nadelhoffer), Lund University, Sweden (Ulrika Rosengren) and the co-ordinating partner Danish Centre for Forest, Landscape and Planning (Inger K. Schmidt, Per Gundersen).

The starting point for the CNTER project was 'what can we learn about carbon cycling from our knowledge on the nitrogen cycle'. We used our long-term nitrogen manipulation experiments started in the NITREX-project, and we improved European databases on observed element pools and fluxes from several hundred forest sites with a new focus on carbon. Since carbon and nitrogen are bound together in organic matter we studied both the effect of nitrogen deposition on carbon cycling in forest ecosystems, and the effect of carbon accumulation on nitrogen storage and release.

June 2005

Per Gundersen
CNTER co-ordinator

Contents

Summary

Databases on carbon (C) and nitrogen (N) fluxes and pools in European forests were compiled for 400 sites and explored thoroughly to create empirical models that predict C accumulation and N retention/nitrate leaching from N input, climate, and ecosystem characteristics.

For nitrate leaching, analyses show that there is a threshold N deposition of 8-10 kg N/ha/yr below which almost no leaching occurs. The important parameters that determine N leaching (and thus N retention) are: N deposition, the organic layer carbon to nitrogen ratio (C/N ratio) and annual temperature. At low C/N ratios (below 23) N input determines N leaching. At higher C/N ratios both N input and temperature are important. Adding more sites throughout the project did not change these relationships and they were robust in validation tests.

Based on a 'N balance' approach, estimates of soil C sequestration rates were calculated by multiplying soil N retention with soil C/N ratio. The mean for European forest with data was 190 kg C/ha/yr, but these have a geographical bias towards central Europe where the estimated C sequestration rates are highest. An unbiased but more uncertain extrapolation to Europe had an overall mean of 70 kg C/ha/yr.

Estimates of C and N sequestration rates in the organic layer of forest soils have been calculated for specific sites based on the 'limit value' concept that uses data from studies of the decomposition of organic matter. The method was further validated in CNTER. This method can be upscaled to Europe, and for 150 sites for which data are sufficient, a mean of 400 kg C/ha/yr was obtained. The method has also been applied for the whole of Sweden, where the range in estimated soil C sequestration was 40-400 kg C/ha/yr. Estimates obtained by the N-balance approach throughout Sweden were below those for the limit value but followed the same spatial gradients. Estimates of C sequestration in the organic layer (using the limit value method) are usually higher than those using the N balance approach for two reasons: i) unlike the N balance method, the limit value approach cannot account for negative C sequestration (i.e. a C loss), and ii) the limit value approach estimates C accumulation in the forest floor (which has the highest C accumulation rate of the soil) whereas the N balance approach accounts for the whole soil profile. This is most noticeable after land use change (i.e. afforestation) where an organic layer accumulates, but where C may be lost from the mineral soil. Efforts on modelling C sequestration have shown that traditional concepts for decomposition assuming a steady state at some point yield too low C accumulation.

We have further made regional and European estimates of the present C sequestration rates in forest soils using several methods. The estimates have consistently shown that C sequestration rates in soil are low: 0-400 kg C/ha/yr. A probable mean is c. 100 kg C/ha/yr. Converted to a European scale this is 13 Mt C/yr, where trees additionally have a net accumulation in the biomass of c. 70 Mt C/yr. Our estimates are much lower than estimates published earlier based on other approaches. For the CNTER estimates it is assumed that C accumulates with N i.e. that soil C/N ratios do not increase. With the elevated N deposition in Europe, the soil C/N ratios are more likely decreasing, thus C-sequestration rates are probably even lower than we estimated. We find it very important to arrive at well-established soil C sequestration numbers, since the size of the possible biological C sink in Europe is uncertain and under debate. We will seek to continue to improve on our estimates of soil C sequestration rates.

To gain insight into C and N interactions, soil and vegetation from long-term field experiments using additions of stable N isotopes have been re-sampled. After ten years, most of the N applied over one year is still present in the soil in amounts not very different from those measured after 1-3 yrs. These experiments allow for a thorough testing and validation of a process model that predicts the fate of N in the ecosystem. When applied to multiple sites we gain an insight in the fate of N that will allow calculation of C accumulation and fluxes in the systems, which can not be obtained in other ways.

Tree species trials, plantation mosaics and felling experiments were re-sampled to gain insight into forest management options (tree species, age and felling regime) for increasing C sequestration and protecting downstream water against eutrophication. The indicators sampled were concentrations of nitrate, DOC and pH in seepage water below rooting zone, forest floor and upper mineral soil C and N pools and CN ratio. The impact of felling was increased with decreasing depth of the organic layer at humid temperate climate. There was no consistent effect of tree species on N leaching between regions in these trials. On a cross-European basis, conifer forests receiving inorganic N in throughfall from 10-25 kg N ha^{-1} y^{-1} appear to have enhanced N leaching over hardwood forests receiving the same amount of N deposition.

1 Introduction

1.1 Scientific background

The carbon (C) and nitrogen (N) cycles are interdependent. In forests, C and N accumulate together in soil organic matter, and the process dynamics of C and N are closely linked. In this project we thus studied *both* the effect of nitrogen deposition on carbon cycling in forest ecosystems, *and* the effect of carbon accumulation on nitrogen storage and release.

Nitrogen interactions in the carbon cycle: By far the largest amount of C stored in forest ecosystems in the Northern Hemisphere is stored in soil. Carbon fixed by photosynthesis ultimately moves via litterfall to the soil, where it is only partially decomposed. Thus, over the long term the soil is the ultimate sink or source of CO_2 in forests. Nitrogen is usually the limiting nutrient in forest ecosystems, and thus sequestration of C is closely linked to the N cycle. Increased N deposition is characteristic of large regions of Europe (and of parts of North America) and may thus have led to increased net primary production in many forest ecosystems, thereby increasing sequestration of CO_2 from the atmosphere. Understanding the N cycle in forests is therefore the key to understanding sequestration of C and the strength of the long-term source or sink for carbon in soils. The effect of increased N deposition on C sequestration in forests may be a significant sink in the global C budget (Holland et al. 1997) but the magnitude of this sink is unknown and under debate. Members of CNTER have in the past suggested that currently this sink is minor and that the sink capacities of soil and vegetation are about equal (Nadelhoffer et al.1999). In the Kyoto Protocol, terrestrial sinks for CO_2 are recognised and may in the future be accounted for in the agreed international CO_2 emission reductions. This will require methods for reliable quantification of these C sinks.

Carbon interactions in the nitrogen cycle: In many forest ecosystems, especially in northwestern and central Europe, N deposition currently exceeds the capacity of the vegetation to accumulate and remove nitrogen through net primary production. Most of the excess N is retained in the soil or leached as nitrate, which can cause acidification and eutrophication of soils and downstream water bodies. The capacity of soils to retain N seems to be dependent on the availability of C in active soil organic matter pools, which is related to the C/N ratio of these pools (e.g. Gundersen et al 1998). The C/N ratio may decrease over time if more N-rich organic matter is accumulating. Ultimately, if the C/N ratio reaches a critical threshold, the ecosystem may become nitrogen saturated (Aber et al. 1989) and nitrate will eventually 'break through' and leach.

Quantifying the rate of such change in C/N ratio over time is the key to predicting future N retention and losses of nitrate from forest ecosystems. In the emissions reductions (UN-ECE Multi-pollutants-Multi-effects protocol) and in the EU directive on emission ceilings, the concept of critical loads is used to describe present and future impacts of N deposition in forest ecosystems. In calculation and mapping of critical loads, estimates of soil N retention are the most uncertain parameter. Current models assume a steady state and do not take into account changes in C/N ratio, ecosystem N status, N deposition, or change of tree species.

Since soil processes account for the most significant unknowns in the C and N cycles (Bird et al., 2001) our focus in this project has been on the dynamics and sequestration rates of C and N in forest soils. In this work our main questions were:

1. Does N deposition affect (enhance) C sequestration rates? If so, when and where will this occur?

2. How will current European environmental policy affect C sequestration?

3. How important is the soil C sequestration rate to the European C-sink?

4. How can we manage soil C storage in the short and in the long term? What reduces or increases soil C pools?

5. What are the key indicators for risk of N leaching? When and where will N deposition cause N leaching?

6. Are C and N storage processes in soil always coupled?

7. What are the uncertainties for future C sequestration rates and N retention?

8. Can forest be managed to reduce leaching of reactive N and DOC?

1.2 CNTER approach

The complexity of the C and N cycles as well as of their interactions in soil processes may limit the potential for extrapolation in time and space even when based on very detailed process understanding. Further, experimentation is complicated by the large stores of C and N, where detectable changes including changes in C/N ratios may only appear over decades. To overcome these problems our approaches in CNTER were (1) to combine and analyse data on C and N fluxes and pools from several hundred case studies within Europe during the last two decades; (2) to integrate empirical information on litter decomposition and thereby estimate the possible effect of N deposition on decomposition rates; (3) to resample existing long-term N input manipulation and ^{15}N labelling experiments in forests in Europe and North America to determine the fate of added nitrogen after 8-10 years, (4) to improve and validate existing dynamic models of C and N cycling, and (5) to use well-established and well-described forest management trials to evaluate the impact of forest management on C and N sequestration.

The general objectives were to:
1. Identify and validate indicators of the state of forest ecosystems, and environmental risks and impacts related to C and N (i.e. C sequestration, acidification and nitrate pollution of forest waters).
2. Gain new insights into C and N interactions in forest soils, especially the effect of N deposition on the soil C sink.
3. Provide estimates of C sequestration rates and N retention in forest ecosystems under current and future conditions given scenarios for major environmental change drivers: land use, air pollution and climate.
4. Develop empirical and dynamic models that allow extrapolation of C and N risks and impacts in forests to regional and European scales.
5. Provide guidelines for forest management to optimise environmental benefits (C sequestration and N retention).

2 Analysis of empirical data

As part of the research effort related to acid deposition, input-output budgets of major elements had been compiled for approx. 200 forest sites in the IFEF (Indicators of Forest Ecosystem Functioning) database developed from the papers Dise and Wright (1995), Gundersen (1995) and Dise et al. (1998a,b). Further, the European monitoring programmes under UN-ECE have detailed data from >100 forest plots (ICP Forest, Level II) where element budgets were made available from a parallel project (van der Salm et al., 2004). Smaller subsets of these data had been exploited with respect to acidification and nitrogen issues and promising empirical relationships had been identified although with large confidence intervals due to a limitation in the number of sites with full datasets. In CNTER we filled data gaps, added new data relevant to carbon pools and fluxes, compiled data from new sites and combined the databases to one European database for forests several times larger than previously analysed (section 2.1).

The aims were i) to achieve a more definite identification of indicators (and of empirical models for prediction) of N retention and nitrate leaching (section 2.2.1), ii) to explore the dataset for empirical relationships with C parameters, and iii) to calculate (and extrapolate) estimates of soil C accumulation (C sequestration) in Europe (section 2.5). To further validate relationships, less intensive regional surveys were (re)sampled for soil C and N parameters (section 2.2.2). The analysis of the datasets was based on traditional statistical methods (multiple regressions, principal component analysis, ANOVA, and non-parametric equivalents). However, we also used neural networks as a new method of data exploration. Neural networks analysis was used on other datasets due to technical requirements (section 2.3).

Decomposition of plant litter is one of the most important processes determining C turnover and soil C storage. In previous work Berg et al. (2001) found empirical relationships between litter chemical composition and the recalcitrant fraction remaining. This finding was based on a data collection that was expanded to a larger database in CNTER. This new database DELILA (Decomposition, LItter, Limit vAlues) was further explored and used for calculation and extrapolation of C sequestration in organic layers (section 2.4).

2.1   European databases

Database for C and N pools and fluxes in forests
Over the course of the CNTER project, the number of sites in the IFEF database with N leaching fluxes calculated over at least one full year was increased from 202 to 280. The sites are forested plots and catchments published in scientific papers over the last decade (Fig 1A). The database now contains input-output budgets, stand/site characteristics and information on ecosystem properties that are important to C storage and C fluxes, e.g. C and N pools in vegetation and soil, driving variables such as climate and tree species, response variables such as DOC and nitrate leaching, and process variables such as C and N accumulation. Not all data are available for all sites.
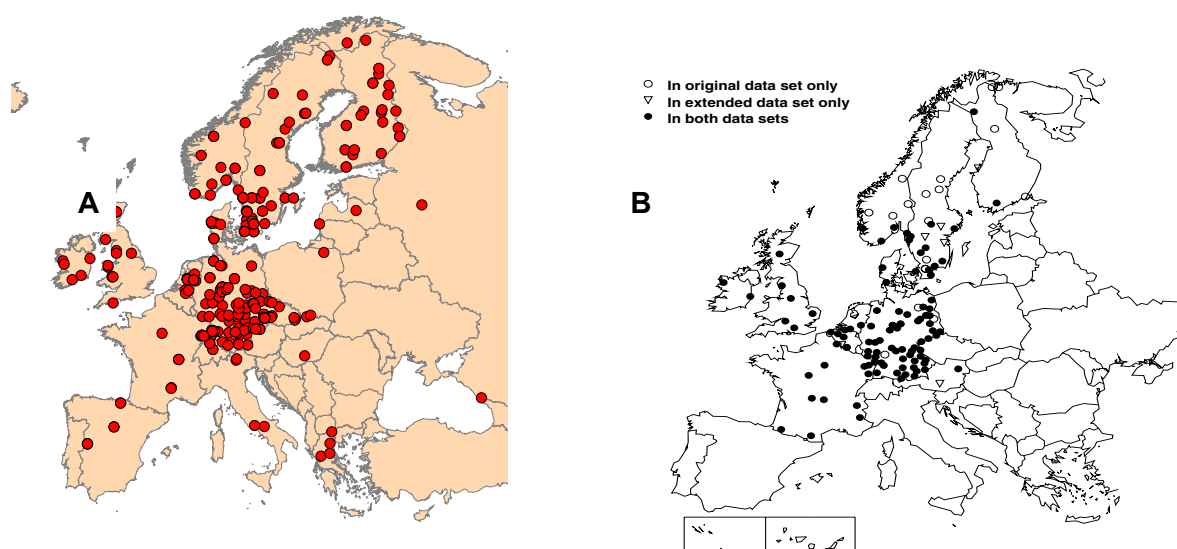
Figure 2.1: Map showing distribution of sites included in the (A) IFEF database, and (B) Level II database. For Level II: 'original' dataset refers to sites with data up to 1998; 'extended' includes sites with data from 1999 and 2000.

The original Level II database (De Vries et al., 2001), containing input-output budgets derived for 121 UN-ECE/EC Intensive monitoring plots for the period 1995-1998 (Fig 2.1B), was amended with two more years' of calculated leaching fluxes, giving up to 6 full years of data for these forests. Leaching fluxes are calculated by multiplying measured soil solution concentrations with simulated water fluxes. Water fluxes are simulated based on daily meteorological data and generic soil physical characteristics using a Richards' model approach.

The Level II database is an improvement over IFEF in that it covers similar years and employs standard methods with a set of quality control checks, but it contains fewer sites, a narrower range of variables (e.g. range of input N, climate), and a more generic water flux calculation than the IFEF forests. Quality control checks on both databases consisted of tests on the Cl⁻ balance (where available), with the ratio between Cl⁻ input and output required to be between 0.5 and 1.5. An additional check was that N leaching should be no more than 10% higher than N in throughfall, with sites exceeding this value indicating a disturbance in the N cycle.

In order to provide background data for the statistical analysis of N retention indicators, cumulative N deposition estimates were derived at SYKE in co-operation with the Coordination Centre for Effects at RIVM, The Netherlands. Estimates of European N emissions for 1880-2000, reported in Schöpp et al. (2003) were used. Transfer matrices of the EMEP MSC-W Centre (150x150 km^2 grid) were then used to derive European-scale estimates of cumulative N deposition for the wet and dry deposition fractions for each grid. These estimates can then be scaled using the measured site-specific deposition values (measured throughfall and open field deposition) for each site. The methodology was successfully tested on a subset of 21 sites belonging to the ICP Integrated Monitoring network before addition to the full database. Vegetation pools of C and N as well as growth (C accumulation) and vegetation N uptake were calculated from standing wood volume and changes therein where known, or else derived from stand age and available site quality characteristics, using forest yield tables to estimate the actual forest growth (Klap et al., 1997) as described in de Vries et al. (in press).

Both databases (IFEF and Level II) were greatly improved, thus allowing for new analyses. Statistical analyses consisting of stepwise regression and principal components analyses

were run on the separate and joint databases, and cross-validated. The results of the analyses are summarised in sections 2.2.1 (nitrogen), and 2.4 (carbon). The databases are a deliverable product, which will be a highly valuable resource for future scientific study on forest ecosystem properties across Europe.

Litterfall and decomposition
The database DELILA contains information about quantitative litterfall, litter chemical composition and limit values for decomposition for mainly Scots pine and Norway spruce but including deciduous tree species, with focus on northern Europe. DELILA has been divided into two subunits. DELILA I has about 75 values quantitative litterfall values from each of pine and spruce forests, plus about 40 values for different deciduous species; in all ca 190 values. DELILA II contains ca 170 sets of values for initial chemical composition for foliar litter and ca 150 limit values for decomposition and initial litter chemical composition. The litterfall data encompass mainly foliar litterfall; other litter fractions are included only to a limited extent. The data set for litterfall and that for limit values cover each other geographically to a certain extent, and both are representative of Nordic forests.

2.2 Nitrogen retention indicators

2.2.1 Analysis of European datasets

The relationship between the throughfall input of N ('Nin') and the leaching of N ('Nout') showed a general positive trend, with a large number of sites leaching undetectable levels of nitrogen at N input fluxes below around 8 kg N ha^{-1} y^{-1} (Fig. 2.2). Joining the two databases also showed that IFEF spanned almost twice the range in N deposition as the Level II sites, with more sites at both the low and high ends of the N deposition range. There are two reasons for this: i) The IFEF sites cover a geographically larger area than Level II (Figures 2.1a,b), and ii) flux measurements from the Level II sites are all more recent than 1996, when substantial reductions in N emission and deposition had begun in high-deposition areas such as the Netherlands and Denmark.

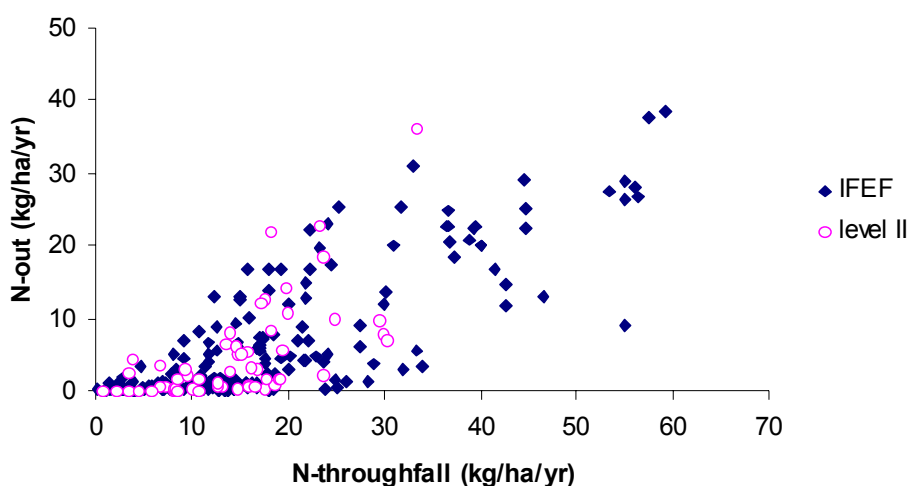*Figure 2.2: N leaching fluxes (kg N ha^{-1} y^{-1}) against N input in throughfall, IFEF and Level II sites.*

Both the IFEF and Level II datasets show that the primary correlation to N leaching is the input flux of nitrogen (Fig. 2.2), explaining about 50% of the variability in N-out. On average approximately 55% of N input in throughfall is leached, and in nearly all sites at least 3-4 kg

N ha^{-1} yr^{-1} are accumulated. In general, the amount of N accumulated increases as N deposition increases.

Estimates of N leaching can be significantly improved by considering some simple characteristics of the sites, resulting in a classification of forests across Europe into three categories:

      I.   'Low N-in' (N in throughfall < 8-10 kg N ha^{-1} yr^{-1}),

      II.   'N-enriched' (forest floor C/N $\leq$ 23), and

      III.   'C-enriched' (N-in $\geq$ 8-10 N kg ha^{-1} y^{-1}; C/N > 23)

All sites fall uniquely within one of the three categories.

The first category encompasses low N deposition forests, in which significantly more N is retained than would be predicted based on all sites together. These sites show strong N limitation of vegetation, soil microbes, or both. In IFEF, N leaching is <2 kg N ha^{-1}yr^{-1} for all but 2 forests (these leached 2-4 kg N ha^{-1}yr^{-1}), which are the only ones located at altitudes above 1000 m. After accounting for altitude, there is a weak positive relationship between N input and N leaching. This is probably due to N lost through snowmelt or otherwise hydrologically bypassing soil adsorption or biological uptake (e.g. through macropore flow).
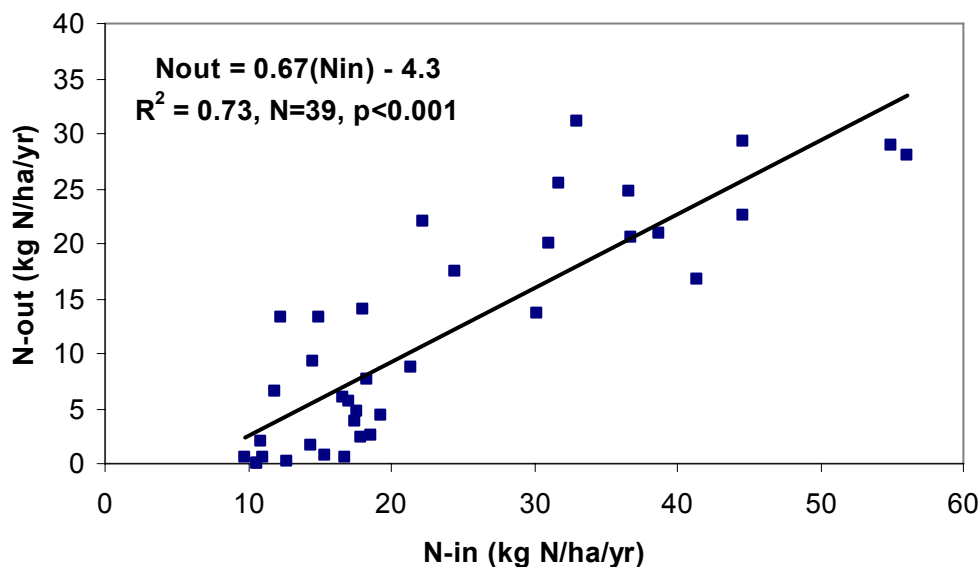
Figure 2.3: N leaching fluxes (kg N ha^{-1} y^{-1}) against N input in throughfall for forests with C/N $\leq$ 23.

After N deposition, the next most important correlate to N leaching is the C:N ratio of the forest floor (in the database this is primarily the OH or Oa horizon), which we view as a proxy for overall ecosystem N enrichment. Ecosystem N enrichment can be thought of as how much reactive N is stored in the system, and is primarily a function of the climate, site history (recent and geological, including land use), vegetation type and N deposition. Sites that are highly N-enriched (C:N $\leq$ 23) can be well described as leaching N in an approximately linear relationship to N-input (Fig 2.3), suggesting overall N saturation of vegetation and soil. For these, the best estimate of N leaching is 67% of N input minus 4 kg N ha^{-1} yr^{-1}, with an r^2 of 0.73 (linear) to 0.75 (logarithmic). Within our databases there are no N-enriched forests receiving less than 9 kg N ha^{-1} yr^{-1}.

Because the nitrogen enrichment of a site is only partly due to the nitrogen deposition, there are many sites receiving high N deposition but with a high forest floor C/N. N leaching from these 'C-enriched' forests is the most difficult to model empirically. Regression analysis shows N

leaching to be on average 48% of N input minus 3.5 kg ha^{-1} yr^{-1}, although significant variability remains. Of the three categories, this is the only one in which forest characteristics play a significant role in regional-scale prediction of N leaching. Tree type (conifers leach more N than hardwoods), and tree age (older conifer stands leach more N than younger conifer stands) account for some of the variability in N leaching, as well as temperature (cooler sites leach more N than warmer sites). It is apparent that the response to N deposition in these forests is much more dependent upon factors such as climate, site history and management than for either the low Nin or the N-enriched forests.

The most significant of these variables in a model already containing N deposition is mean annual temperature (MAT). Fitting a spline function to N-in, N-out, and MAT shows a curvilinear relationship between Nout and temperature for C-enriched forests, with peak N leaching at MAT 7.5°C (Fig 2.4). One hypothesis is that vegetation uptake of nitrogen in relatively cold sites is temperature-limited for significant parts of the year, with nitrogen turnover microbially-dominated. Higher leaching of N as MAT increases then reflects enhanced rates of N mineralisation and nitrification uncoupled from uptake during the non-growing season. As temperature further increases, vegetation uptake can potentially occur throughout the year. Nitrogen that is released through mineralisation or nitrification is taken up by plants, with higher temperatures increasing productivity and rates of uptake.

To gain insight into the strength and weakness of the developed relationships, and to identify sites and regions at risk from enhanced N leaching, linear regression models developed on the Level II data were validated using the IFEF database. Overall, the best predictions of N leaching are obtained for the low N-in forests and the N-enriched sites. N leaching from the C-enriched sites, about 40% of the forests in the databases, continues to be poorly predicted. Adding mean annual temperature leads to higher correlation coefficients on the calibration sets, but so far does not lead to better results when applying the model to the validation set. Other temperature relationships will be investigated in the future. This problem is most dramatic for mid-latitude sites and Mediterranean sites, especially those receiving a moderate N input, with an above-average temperature, and relatively high C/N ratio. At these forests nitrogen leaching fluxes are often strongly overestimated.

Thus, on a regional scale, CNTER has resulted in good prediction of N leaching levels from the 'extremes', that is, sites that currently and historically receive low levels of N deposition, and sites that are highly N-enriched. Attention should be focused in the future on understanding the nitrogen dynamics of forests that are not nitrogen enriched and receive moderate to high levels of N deposition. Further improvements in the databases, potentially leading to better predictions and more insights into processes, can be made by collecting more information on local climate, vegetation type, forest floor and soil carbon and nitrogen stocks (which may be better predictors of N enrichment than C:N), and DOC and DON leaching (which may be useful proxies for C and N stocks, respectively).

As mentioned above N deposition is not the only determinate of forest floor C/N ratios. Although a significant decrease in C/N is observed with increasing N deposition this is confounded with MAT (N deposition correlates with MAT). In our calculation of soil C sequestration rates (section 2.5) it is important to know if N deposition influences forest floor C/N, but no definite conclusions can be made from these datasets.
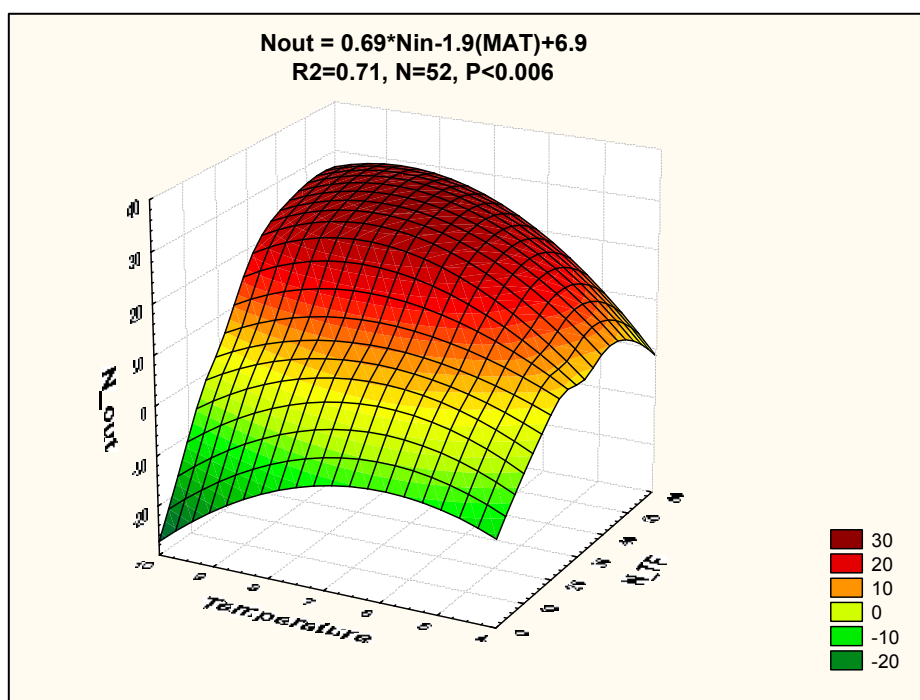
Figure 2.4: N leaching fluxes (kg N ha^{-1} y^{-1}) against N input in throughfall and mean annual temperature for forests with C/N $\geq$ 23.

Corresponding to the European effort in CNTER compiling and analysing observational data, Aber et al. (2003) have examined the response of natural ecosystems – primarily forests – to atmospheric N deposition in NE USA in a synthesis supported by NERC (Northeastern Ecosystem Research Cooperative). In a new project, some members of the CNTER consortium will compare and contrast ecosystem response to N deposition in the NE USA and Europe, beginning with portions of the IFEF and NERC databases. All sites considered to date have information on inorganic N deposition and N output. Nitrogen deposition is typically recorded as throughfall for Europe, and modelled wet + dry deposition for the NE USA. N output is runoff (from catchments) or leachate from deep lysimeters (on plots). A preliminary assessment indicates that inorganic N output increases with N deposition in both Europe and the NE USA and suggests a common threshold of ~8 kg ha^{-1} yr^{-1}, above which some – but not all – systems begin to export significant quantities of nitrate. The present data compilation only weakly suggests relationships between forest floor C/N and nitrate export, although only 134 of 328 sites had C/N data. Multiple regression suggests that N deposition explains most of the variation in N export, but that inclusion of estimated C/N ratio and mean precipitation can provide modest improvements (Goodale et al. 2005). Anticipated future efforts include filling in missing data values for key parameters at existing sites and new statistical analyses to better determine and compare threshold values of response across regions.

2.2.2   Regional studies

The regional datasets are collected in a comparable way from three countries (SE, DK and NL) based on previous sampling networks. In addition, two datasets (Sitka spruce and common oak) based on more intensive monitoring was available from Wales, UK. The forest Edese bos is situated in the central part of The Netherlands. The area consists of nutrient-poor cover sands, which have been ice-pushed in the Saalien ice age. The Scanian data set is collected from long-term forest monitoring plots in the southernmost part of Sweden. The distribution of plots mirrors the natural variation in Scanian bedrock and moraine. Soil types range from podzols, through transition types, to cambisols, and soils influenced by agricul-

ture. Soil texture is loamy with varying amounts of sand, silt and clay, the dominating texture being sandy, silty loam. The Danish data set consists of a national survey covering most of the country. Norway spruce and Scots pine forests were sampled in all three countries whereas common beech sites were sampled in The Netherlands and Denmark; larch and Douglas fir were sampled in The Netherlands only. Soils were sampled from the forest floor for the C/N analyses and below the rooting zone (50-100 cm depth) for the extraction of nitrogen. Ambient deposition of nitrogen at the different sites was estimated by computer modelling. The Wales data were collected from a chronosequence of Sitka spruce on afforestation sites and from 19 natural oak stands where input-output budgets for N were established earlier.
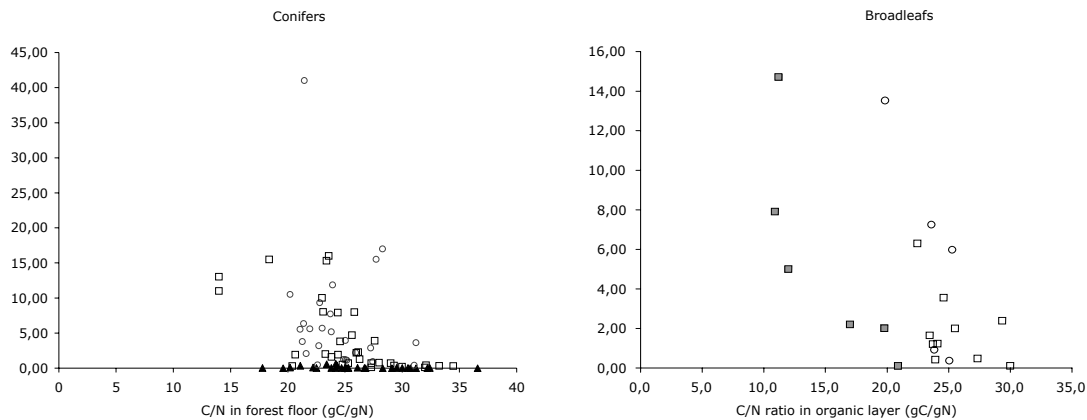
Figure 2.5: Relationship between the C/N ratio in the forest floor and NO3-N concentrations (mg/L, y-axis's) from below the rooting horizon a) (left panel) in 86 conifer stands from Denmark (national survey, squares), The Netherlands (Edese bos, circles) and Sweden (Scania, triangles) and b) (right panel) 20 broadleaf stands from Denmark with (open square) or without (filled square) organic layer and The Netherlands (Edese bos, circles)

The dataset encompassing coniferous stands from three countries collected with comparable methods show a significant correlation between C/N in forest floor and nitrate in soil water below the rooting zone (Fig. 2.5a; r= –0,41; p<0.0001) but the relationship is to a large part governed by the Danish survey. Data from the broadleaf forest sites (Fig. 2.5b) show a similar trend but also points out the importance of soil types – at sites with very nutrient rich loamy soils with no real accumulation of an organic layer, 0-5 cm mineral soil C/N ratio was used – these have clearly lower C/N ratios and increasing nitrate concentrations as C/N ratio decrease, but can not easily be compared with broadleaf forest sites with organic layer.

In the Welsh datasets controls on nitrate leaching from common oak stand could be compared to the Sitka spruce stands. A significantly lower threshold has been identified for common oak both in terms of the throughfall flux (Figure 2.6a) and C/N ratio (Figure 2.6b). This illustrates that there may be differences in the behaviour of broadleaf and coniferous forests. However in the Welsh case differences in management (oak natural and unmanaged, Sitka spruce planted and managed) may be of importance.

Analysing data from the Scanian data set reveals a significant correlation between total nitrogen deposition and C/N in the forest floor (Fig. 2.7a; r=–0,47; p=0.028) and an even stronger relationship to the deposition of NO_x only (r= –0,65; p<0.002). This seems to confirm that N deposition do enrich the soil, a hypothesis that could not be confirmed in the Europe wide data above due to a confounding effect of MAT. In Scania MAT is constant, thus the negative effect of N deposition is more likely to be real. The N deposition was also significantly correlated to the N concentration in the needles (r=0,56; p<0.01) and to the C/N ratio

in the forest floor. The C/N clearly affects N concentration in soil water (Fig. 2.7b) but it is likely that factors like age and soil type are important for determining actual N-concentrations in the soil water.

In the Dutch data set, where the highest levels of N deposition were found, no significant correlations were found between either N deposition and forest floor C/N ratio (Fig. 2.8) or N in soil water and forest floor C/N ratio.
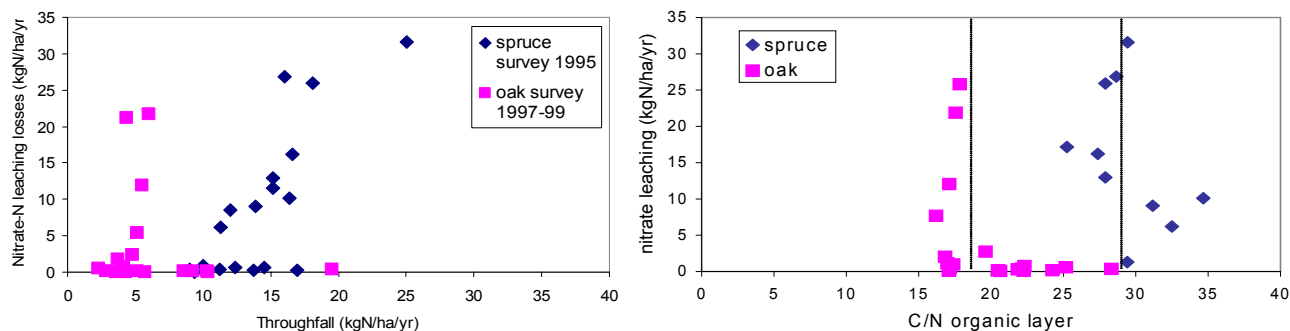
Figure 2.6: The relationship between a) throughfall N flux and nitrate leaching and b) organic layer C/N and nitrate leaching for Sitka spruce and common oak in Wales on similar soils.
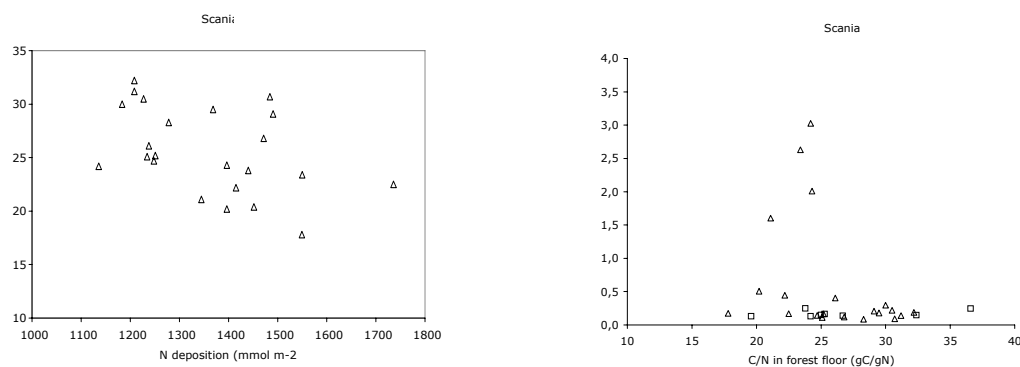
Figure 2.7: a) Relation between modelled total N deposition (mmol$_c$ m^{-2} y^{-1}) and forest floor C/N ratio (gC/gN on y-axis) and b) relation between the C/N ratio (gC/gN) in the forest floor and total N concentration in soil solution below the rooting zone (mg N/L, y-axis) in 22 Norway spruce (triangle) and 8 Scots pine (square) stands in Scania, Southern Sweden.
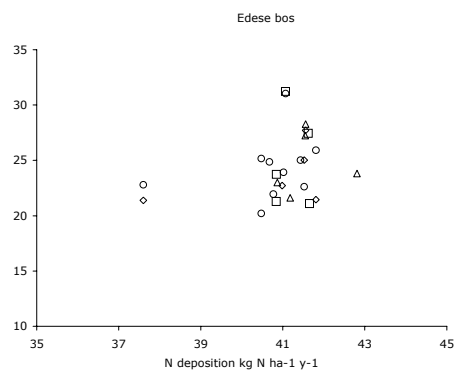
Figure 2.8: a) Relationship between modelled total N deposition (kg N ha^{-1} y^{-1}) and C/N ratio (y-axis) in the forest floor in 20 coniferous stands in Edese bos, The Netherlands.

2.3 Empirical estimates with artificial neural networks

The neural networks techniques were first tested in a temporal application with the aim of analysing the effect of climate change on C and N leaching. The second application was on using spatial data to see if soil C and N contents could be predicted by other site parameters.

a) Temporal application
An empirical model of streamwater quality was designed on the basis of artificial neural networks to reproduce daily total organic carbon (TOC), total nitrogen (N_{tot}) and total phosphorus (P_{tot}) concentrations (Holmberg 2003; Holmberg et al. 2006). The streams that we modelled drain two forested catchments located in southern and eastern Finland. The simulated concentrations were used to predict future fluxes under scenarios of climate change. In the period 1990 to 2000, observed TOC, N_{tot} and P_{tot} concentrations were in the range 2 – 60 mg C L^{-1}, 0.1 – 1.4 mg N L^{-1} and 1 – 60 µg P L^{-1} with a mean discharge per unit area in the range of 10 to 15 L s^{-1} km^{-2} for the three streams.

Our artificial neural networks consisting of 13 input variables, one hidden layer with 7 nodes and one output variable each were trained with the back-propagation algorithm to estimate the concentration of TOC, N_{tot} and P_{tot} in streamwater. Daily air temperature, precipitation and runoff observations were included in the input variables as well as catchment characteristics such as catchment area and the area of lakes and peatland within the catchment. The networks performed well in comparison with the alternative method, i.e. flow-weighted average concentrations. Artificial neural networks are a useful method for creating black-box models of streamwater quality in cases where the involved processes are too complex to simulate directly.

Projected changes in monthly temperature and precipitation under a changing climate were used to generate daily temperature and precipitation series for the 2050's (Fig 2.9). Daily runoff values for this hypothetical year were produced with an operational runoff model. Carbon and nitrogen loads were calculated for the 2050's using the neural network models of TOC, N_{tot} and P_{tot} concentrations in stream water. The low change scenario resulted in annual fluxes close to present fluxes, while the high change scenario gave an increase of approximately 26 per cent in annual TOC, N_{tot} and P_{tot} fluxes.
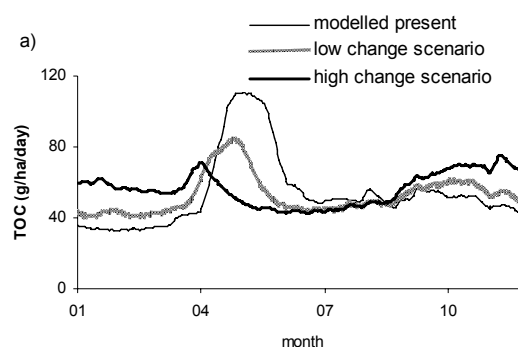
Figure 2.9: Predicted daily values of fluxes of TOC at the Hietapuro catchment for present conditions (1990's) and for a low climate change scenario (2050's) and a high scenario (2050's). Predicted fluxes are obtained from concentration values obtained as output of a network driven by 13 input forcing variables (Holmberg et al. 2006).

To evaluate our results, we made a comparison with results from a dynamic soil model (SMART2, Chapter 4) that operates with process-based descriptions of the mechanisms that determine the concentrations of NO_3 and NH_4 in soil leachate. As described in Chapter 4,

SMART2 was applied to predict changes in soil solution concentrations for a set of plots, including two located close to our catchments. Because the process-based SMART2 model accounts for changes in soil leachate at the plot-level and the empirical ANN-model reproduces stream chemistry in the runoff from the whole catchment, the results of the two models are not directly comparable. Furthermore, the scenarios used by SMART2 to predict future soil conditions include assumptions concerning both climate and deposition levels, while no information about future deposition was used to drive the empirical stream model. Therefore a quantitative juxtaposition of the predictions obtained by the two models is not meaningful.

On a methodological level the approaches may, however, be compared. The empirical model, based on artificial neural networks, was able to successfully deal with short-term dynamics of N_{tot}, P_{tot} and TOC in runoff from the catchments, driven by the variability in the climatic drivers. The process-based soil model is more difficult to calibrate to reproduce all aspects of the short term dynamics, although it does perform plausibly over a longer time scale. As the empirical model is driven by present day climatic variability, it is possible, in spite of the careful selection of driving variables used in the predictions, that the credibility of the future assessments approaches its limits.

b) Spatial application
SYKE has been cooperating with the Finnish Forest Research Institute (FFRI) in order to predict soil C and N amounts using national forest inventory site and stand data (Tamminen and Starr 1990). FFRI has investigated 488 sample sites (Tamminen and Starr 1990) from a systematic network of 3000 permanent plots established as part of the eight national forest inventory with stand data on tree species, basal area and age, and site information on latitude, elevation, slope, soil type and depth, and taxation class. Soil carbon and nitrogen amount in samples from the humus layer and the 0-5, 5-20 and 20-40 mineral soil layers have been included in the prediction task, using both stepwise regression (FFRI) and artificial neural networks (SYKE).

In the results of the stepwise regression analysis the amount of carbon in the organic layer was a function of organic layer thickness, mineral soil texture, site fertility, site wetness, tree species and stand age. The organic layer C/N ratio increased to the north, with decreasing effective temperature sum. It increased also with increasing stand age and was higher in pine stands and on paludified sites and sites with low fertility. Nitrogen amount in the organic layer was correlated positively with organic layer thickness and variables indicating wet conditions and peaty organic layers and negatively with nutrient-poor site conditions, here the poorest site type and sorted fine sand soils. The amount of nitrogen in the mineral soil correlated positively with soil fine fraction proportions and site's production capacity, represented here by the best wood production classes.

In the neural network analysis, networks of feed-forward-type were used with sigmoid activation functions and back-propagation of errors. The amounts of carbon and nitrogen in the organic layer and in the mineral layer 0-20 cm were modelled with the networks. Through these networks, the amount of carbon and nitrogen were expressed as non-linear functions of several variables observed or measured at the site - the effective temperature sum and the elevation above sea level, the tree species and age of the stand, the thickness of the organic layer, the fertility and stoniness of the site. The correlation between the modelled results and the validation data was between 0.7 and 0.8. for these networks. Reporting the comparisons between the two approaches and the conclusions concerning site forming processes is in progress (Holmberg et al. in prep.).

2.4 Carbon accumulation in organic layers

2.4.1 The limit-value concept

Litter decomposition rates normally decrease the further the decomposition process goes. This may be a consequence of the relative enrichment of resistant compounds and the formation of new compounds that are resistant/recalcitrant. In some cases decomposition rates may even decrease very quickly. As the decomposition rates approach zero it is possible to calculate limit values for decomposition, viz. the accumulated mass loss approaches asymptotically a highest value, as can be expressed with the asymptotic function:

Eq 2.1
$$m.l. = m(1-e^{-kt/m}),$$

in which $\underline{m.l.}$ is accumulated litter mass loss, $\underline{m}$ is the limit value, $\underline{t}$ is time and $\underline{k}$ is the decomposition rate.

Limit values calculated for different foliar litter species range between ca 50 and 100 % decomposition (e.g. Berg & McClaugherty 2003). Based on information about factors regulating the degradation of lignin, the following hypotheses were tested; (i) that the limit value was negatively related to litter N concentration (the higher the initial litter N concentration the more recalcitrant material remained), (ii) that the limit value was positively related to litter Mn concentration (the higher the initial litter Mn concentration the less recalcitrant matter remains). The physiological background is that high N concentrations hamper the degradation of lignin and ligninlike compounds (Eriksson et al. 1990) and raised concentrations of Mn (Hatakka 2001) enhance it, observations that now are ca 25 and 15 years old, respectively and relate to formation of lignin-degrading enzymes. That Mn is important as a rate regulator has been observed for Norway spruce needle litter (Berg et al. 2001) and now also for oak leaf litter (Box 1). Further, an effect of litter Ca concentration has been observed. Although Ca has no direct causal involvement in lignin degradation, such an effect may be ascribed to its general stimulating effect on the microbial population.
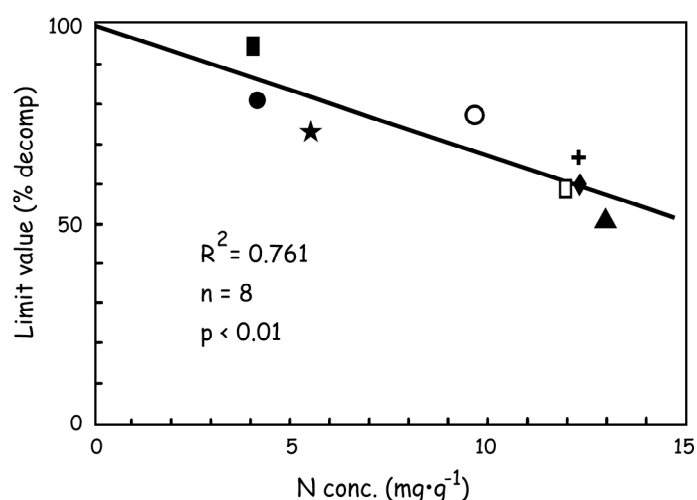
Figure 2.10: The average initial litter N concentration and limit values for needle litter of lodgepole pine, green and brown Scots pine needles, Norway spruce, silver birch, silver fir, Pyrenean oak, and common beech (from Berg (2000)).

The available limit values from unpolluted ecosystems (ca 120) were related to initial litter chemical concentrations of N and Mn and highly significant relationships were found both for the combined data set and for average values for species (e.g. Fig 2.10). The inclusion of litter lignin concentration improved the relationship, although the range in litter lignin concentrations was considerably lower than for the two nutrients. This was a first step to allow a

more advanced theory that could be a base for a further validation, based on quantifications of humus accumulation.

Based on the limit value, the fraction of litter that remains as stable humus can be estimated, and if the litterfall flux is also known, it is possible to calculate the accumulation rate of humus and thereby the sequestration of C (and N) in the organic layer.

2.4.2 Calculation of carbon sequestration rates, and their validation.

Rates for C sequestration that we have been calculating on a theoretical basis need to be validated and such validations have been made. A limitation to this approach is the availability of reliable validation data. We have made validations in two dimensions, first using litter chemical composition and stored amounts of SOM (C), and in a second step by direct quantifications using calculated and modelled limit values and litter fall that are compared to stored amounts of C.

In two cases validation data have indicated that for forest stands with more N-rich litter a higher C accumulation has taken place in spite of the fact that litter fall in the N-rich stands was lower; (i) thus in a set of 7 paired equal-aged stands of Scots pine and Norway spruce the accumulated amounts of C in the organic layer were in all cases higher in the stands with higher N concentrations in the litter fall (mainly spruce) in spite of a higher litter fall in the Scots pine stands (Berg et al. 2001). (ii) For paired stands of Douglas fir and red alder the difference in limit-value-predicted amounts of SOM and those actually measured was low enough to support that N concentration had an influence on the limit value and the stored amounts (Berg et al. 2001).

A set of mixed conifer-dominated unpolluted boreal stands of Scots pine and Norway spruce (Berg et al. 2001; Berg & Dise 2004a,b) and two paired temperate N-polluted stands (Norway spruce and common beech)(Berg 2004) were used in two different approaches, both intended to quantify the humus accumulation (C sequestration).

In the former case a 120-yr-old stand was used as well as three groups of stands with the average ages of 1106, 2081, and 2984 years, stand age measured as the time since the latest forest fire, thus giving the time when humus accumulation started (Table 2.1). For the 120-yr-old stand the average limit value was based on 9 separate measurements of limit values. Litter fall was measured in a chronosquence and the accumulated litter fall could be calculated over 120 years. For the older stands average litter fall was modelled, using ca 39 average litter fall values from the same region. Limit values were averages calculated based on limit values from 20 measured Norway spruce, Scots pine and silver birch stands from the region (Berg & Dise 2004a,b). The actually stored amounts of humus (and C) were in all

Box 1: Decomposition of common oak litter

To fill one of the significant gaps in the DELILA database on decomposition and limit values litter was collected from 19 common oak stands in Wales (see section 2.2.2). Litter decomposition bags were installed in a typical oak stand (Aber, Wales) in February 2002 to determine the effect of litter quality on the controls on decomposition rates for broad leaf species and the 'critical limit value' for a hardwood species. Results indicate that the most important factor determining initial decomposition rates is the initial manganese content of the litter ($P = 0.007$, $R^2 = 0.94$) although relationships were also significant for calcium, nitrogen and lignin (Emmett et al. in prep.)
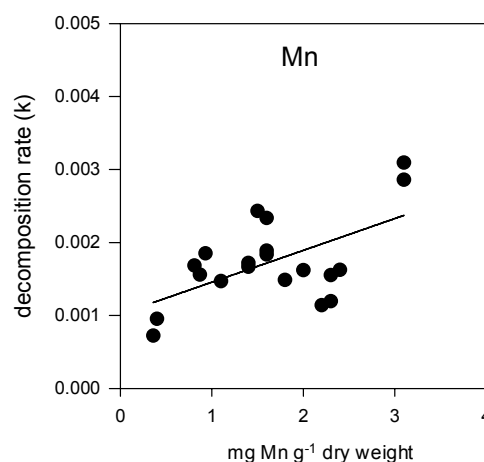
Figure 2.11 Decomposition rate as a function of initial litter manganese content.

cases measured (Staaf & Berg 1977, Wardle et al. 1997). The resulting comparison of measured and calculated amounts of C (Berg et al. 2001) was improved (Berg & Dise 2004a,b) and over 3000 years this resulted in deviations ranging between 2 and 11 %.

Table 2.1. Estimated and observed accumulation of organic matter and nitrogen in the forest floor of known age in Swedish boreal forests. The table from Berg & Dise (2004b)

| | North Sweden - islands | | | Jädraås |
	n=14	n=24	n=12	n=1
Mean stand age (yrs)	2984	2081	1106	120
a. Estimated total litterfall (kg m^{-2})	242	169	89.6	15.2
b. Weighted limit value[1]	80.5	81.7	83.8	89.0
c. Remaining fraction	0.195	0.183	0.162	0.11
d. Estimated forest floor mass (a X c, kg m^{-2})[2]	47.2	31.0	14.5	1.67
e. Measured forest floor mass (kg m^{-2})	49.1	34.6	14.3	1.5
f. (% difference, measured vs. estimated)	3	10.5	-1.6	-8.4

[1]Calculated by weighting the limit values of the individual species by their proportional biomass in the stand.
[2] Slight differences between calculated and displayed (table) values due to rounding of weighted limit values in table.

It may be pointed out that when the measured, accumulated amounts of C are plotted *vs* time a straight line results indicating that the concept "steady state" is by no means reached, not even after 3000 years and with more than 24 kg C m^{-2} accumulated in the organic layers.

The validation in paired temperate N-polluted stands (Norway spruce and common beech) was made over a period of 36 years in stands where litter fall was measured as well as the growth of the humus layers every 3-4 years. The limit values were modelled using 6-yr average values for litter chemical composition. The difference between calculated and measured amounts was less than 20% (Berg, 2004).

Since C and N are bound together in SOM a similar method based on the limit value can be developed for N sequestration in the organic layer (Berg & Dise 2004b). This method was validated as described above for C (Berg & Dise 2004b), but it has not been further exploited in CNTER.

2.4.3   Extrapolation

Scaling up. After the successful validations of the limit-value concept and the C sequestration rates, an upscaling effort was made to a case study country. In addition, we also attempted to validate the method at the same scale. An estimate was thus made for C sequestration rates in the organic matter layer in Swedish forests, encompassing the forested area (23 * 10^6 ha) of Sweden ranging from about 55°N to 69°N (Akselsson et al. 2005).

Also in the scaling up approach we combined amounts of litter fall with the concept of limit values to estimate recalcitrant litter remains. Four groups of tree species were identified (pine, spruce, birch and "other deciduous species". Annual actual evapotranspiration (AET) was estimated for each of ca 15 000 grids of 5 * 5 km covering Sweden. For each grid, data of forested area and main species composition were available through a data base. The annual input of foliar litter into each grid was calculated using empirical relationships between AET and foliar litter fall in the four groups. Litter input was combined with average limit values for decomposition for the four groups of litter, based on empirical data. Finally C sequestration rate was calculated using a constant factor of the C concentration in the litter decomposed to the limit value, thus forming soil organic matter (SOM).

We obtained a value of $4.8*10^6$ tons of C annually sequestered in SOM in forest soils calculated as that of mature forests in Sweden, with an average of 190 kg ha^{-1} and a range from 40 in the north to 410 kg ha^{-1} in the south/southwest. Norway spruce forests accumulated annually on the average 200 kg ha^{-1}. The pine and birch groups had an average of 150 kg ha^{-1} and for the group of other deciduous trees, which is limited to south Sweden, the C sequestration rate was around 400 kg ha^{-1}.

The rates given by the limit value approach are potential rates, without considering the effect of a.o. fire. The calculated C sequestration thus implies a large potential for SOM buildup in undisturbed systems, especially with wild fires today kept to a minimum. Further, these potential rates allow a differentiation in C sequestration rates, considering the effect of tree species.

Upscaling approaches as compared to countrywide, directly measured values for Sweden. A new comparison is being made, in which we compare the results from the limit value approach and the N-balance method to directly measured values from numerous humus samplings over 40 years covering Sweden, based on ca 200 000 humus measurements (pers. comm. from Prof. Maj-Britt Johansson, Dept of Forest Soils SLU, Uppsala). The measured values gave an average carbon sequestration rate of ca 177 kg C ha^{-1} yr^{-1} to be compared to the limit value approach with ca 190 kg C ha^{-1} yr^{-1} and the N-balance method with 96 kg C ha^{-1} yr^{-1}. This indicates that the two calculation methods give realistic and rather correct results. The patterns over the case-study country that are obtained with the different ap-

Box 2: Regional upscaling of soil C sequestration, Sweden

The upscaling of the limit value estimates performed by Akselsson et al. 2005 was compared to upcaled estimates based on the N balance approach (section 2.5.1) by Akselsson et al. (in review). The mean C sequestration in the organic layer was estimated to 190 kg C/ha/yr (a) whereas estimate for the whole soil was 96 kg C/ha/yr (b). The north - south gradient were similar for both methods.

(a)   'Limit value method'   (b)   'N balance method'

C sequestration (kg ha^{-1} yr^{-1})
< 0
0 - 100
100 - 200
200 - 300
> 300
Lake
Protected area
Non forested area
Not included in mapping

Figure 2.12: Estimates of C sequestration in 5x5 km grids covering Sweden based on two fundamentally different methods (Akselsson et al., in review).

proaches still need to be analysed.

Upscaling approaches as compared to that of the Swedish MISTRA project "Lustra". The CNTER project used two approaches (Box 2) in which forest soil over Sweden gave the same gradient for C sequestration rates (Akselsson et al., in review). A comparison to stored C amounts (Anon. 1999) was done for a study covering the same case-study country. We have made this comparison since the "Lustra" program is the largest research program reporting on amounts of C in Swedish soils.

The average C content in Swedish forest soil (incl. the mineral soil to the depth of one meter), according to the Swedish Forest Inventory, is 80 ton ha^{-1}, developed since the end of the last glaciation (Anon., 1999). This implies a low average net accumulation of less than 10 kg C ha^{-1} yr^{-1}. This average accumulation is a net rate including the effect of several factors such as numerous wild fires, climate change, etc. Also forest management such as site preparation has affected these values. With regular intervals forest fires have removed at least part of the SOM layer but since the 1950s more efficient prevention of forest fires has mainly removed fire as an agent that decrease SOM layers. Considering that about 80% of the presently stored soil C is found in the mineral soil (Anon. 2003) this leaves ca 16 tons per hectar in the organic layers thus giving a C sequestration rate of ca 1.6 kg C ha^{-1} yr^{-1} to be compared to the average 100-190 kg C ha^{-1} yr^{-1} as estimated for the same case study area.

2.5   Carbon sequestration in European forest soils

Due to the heterogeneity of soils it is difficult to measure C sequestration rates in forest soils by direct sampling within a reasonable time frame (Yanai et al., 2003). Thus there are almost no estimates of this process, except for some case studies related to land use change (see de Vries et al. (in review). The calculations above of C sequestration in the organic layer based on decomposition limit values is one approach to collect some information for specific sites as well as to upscale to regions or continents. The organic layer is however only a component of soil organic matter, and the mineral soil may be both a sink and a 'source of' C. We have thus suggested another relative simple approach to estimate at least an upper limit for soil C sequestration rates, which will be explored in the following sections.

2.5.1   Nitrogen balance approach

Carbon and nitrogen in soils are bound together in organic matter and the accumulation of C and N in soil organic matter occurs through the same mechanisms, production of dead organic matter and microbial turnover. We thus assume that soil C sequestration is related to N sequestration (immobilisation) in the soil and calculate an upper estimate of net C sequestration as immobilisation of nitrogen in the soils, multiplied by the C/N ratio of the forest soils. In the most simplified calculation all N (and C) is assumed to accumulate in the forest floor:

Eq. 2.2 $C_{seq} < N_{imm} * C/N_{forest floor}.$

This is an upper estimate for soil C sequestration rate since especially at high N input 'new' organic matter may accumulate at a lower C/N than the current bulk forest floor C/N (as observed in some regions, section 2.2.2). Further, N may also be incorporated in soil organic matter in the mineral soil where C/N ratio usually is lower than in the forest floor.

These calculations require an accurate estimate of N immobilisation from the N mass balance of the root zone:

Eq. 2.3 $N_{imm} = N_{dep} - N_{uptake} - N_{leach}$

It is assumed that denitrification can be neglected (or is balanced by a small N_2 fixation). Denitrification fluxes are known to be small in well-drained soils (Persson et al. 2000).

The databases IFEF and Level II (section 2.1) and the Welsh dataset (section 2.2.2) allow estimation of soil C sequestration rates for specific sites and the empirical models derived for N leaching give possibilities to predict and upscale the estimates.

2.5.2   Carbon sequestration estimates

Carbon sequestration rates were estimated from the IFEF-Level II/I sites by using the mass balance approach and site-specific vegetation N uptake estimates. For IFEF, C sequestration was calculated using both measured and modelled (from regression equations) N leaching.

Using the N balance approach, C sequestration rates were highest between 51 and 55 degrees latitude for both the IFEF and Level II/I sites (Fig 2.13). The potential maximum C sequestration estimated via the limit values approach is in the 46-50 degrees latitudinal zone. For IFEF, mean C sequestration using *measured* N leaching was 187 kg C ha^{-1}yr^{-1}; the mean for *modelled* values of N leaching (based on empirical regressions using N input, C/N and mean annual temperature) was 192 kg C ha^{-1}yr^{-1} (Fig. 2.13). The good agreement between these two approaches indicates that carbon sequestration can be estimated via the N mass balance approach from any European forest where N input in throughfall, forest floor C/N and mean annual temperature can be confidently estimated.

Mean C sequestration rates for Level II+I were also calculated using the mass balance approach amounting to 63 kg C ha^{-1}yr^{-1} 67% less than the IFEF derived estimates. For the limit value method of calculating C sequestration we estimate a rate of 412 kg C ha^{-1}yr^{-1} which is 117% higher than the IFEF derived estimates.
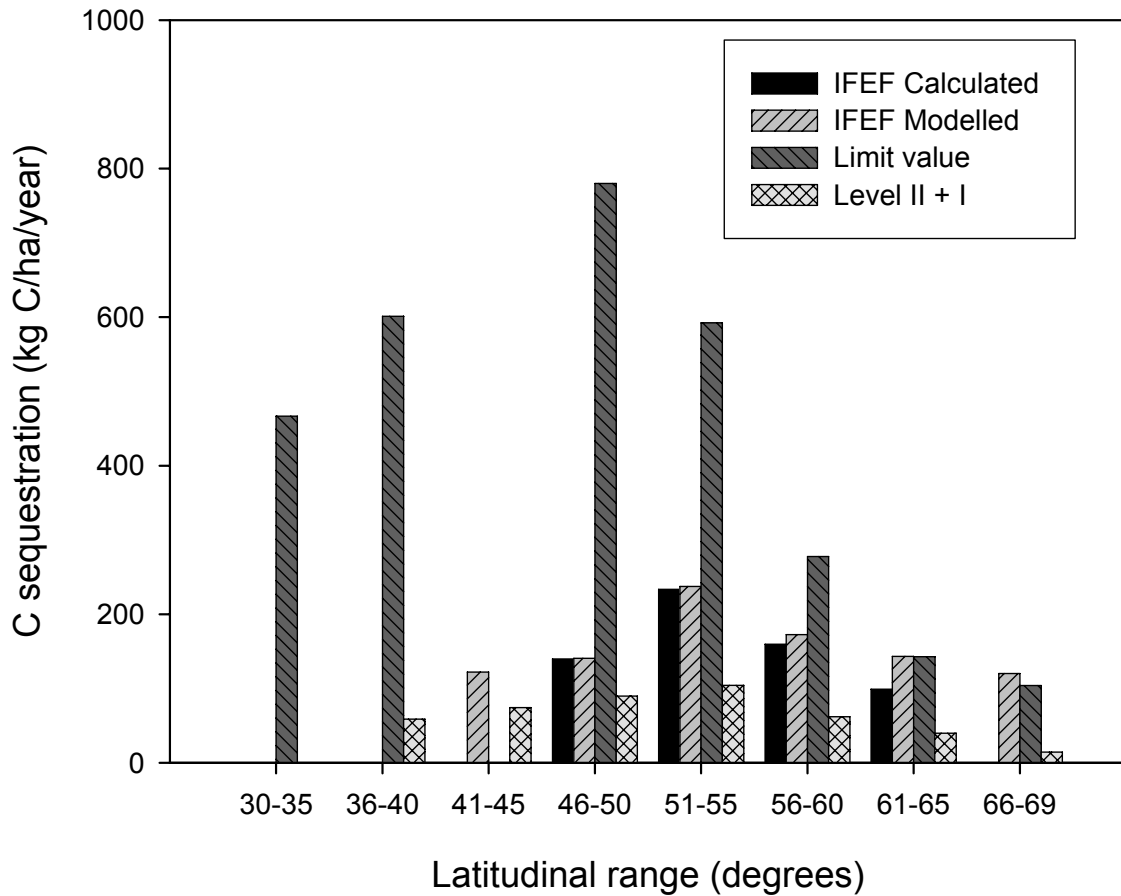
Figure 2.13: C sequestration rates across Europe as estimated by the N balance (IFEF and Level II + I) and the limit value methods.

The reasons behind these differences are not immediately apparent. 10 sites (where C sequestration rates could be calculated or modelled) were common to both the Limit values database (DELILA) and IFEF allowing direct comparison of the two approaches (Fig. 2.14). As already demonstrated in Fig. 2.13. C sequestration estimates from IFEF, both calculated from measured data (Fig. 2.14a) and modelled estimates (fig. 2.14b) consistently are lower than those derived via the limit value method of calculation. There is, however, good agreement between estimates from the two approaches in high latitude zones i.e. >60°N (Fig. 2.13). One reason behind the discrepancy between the IFEF mass balance and the limit value approaches is that C sequestration can be negative when estimated for individual sites via mass balance methods where as only positive values of C sequestration can be achieved via the limit values method. The differences between mass balance C sequestration estimates from the IFEF and Level II + I databases are less clear.
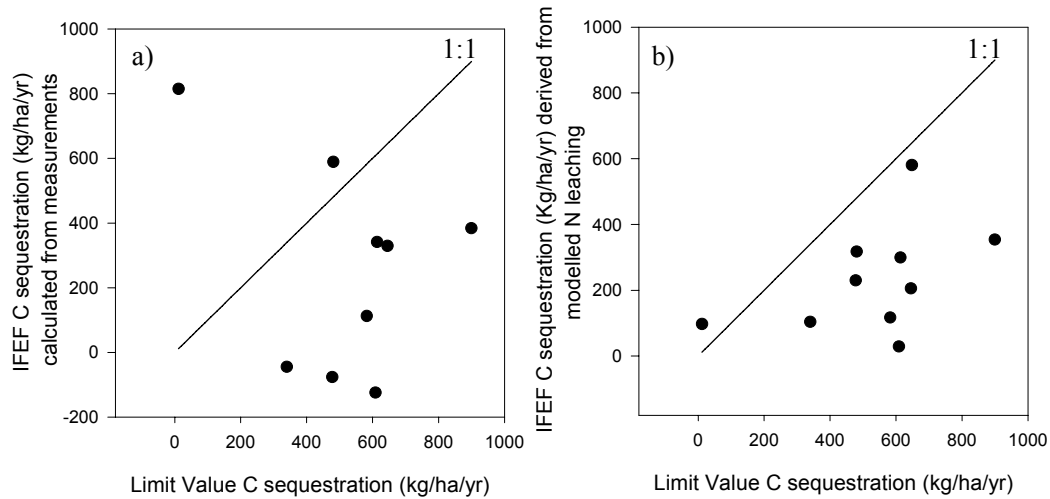
Figure 2.14: Comparison of C sequestration estimates via limit value and IFEF N balance methods.

2.5.3 CN ratio and C sequestration.

Mean C sequestration rates from 'N-enriched' (forest floor C/N < 23) and 'C-rich' (forest floor C/N ≥ 23) sites was 165 and 213 kg C ha^{-1}yr^{-1}, respectively. For Level I + II, N-enriched and C-rich sites (see section 2.2.1) are accumulating 18 and 103 kg C ha^{-1}yr^{-1}, respectively (Fig. 2.15 a and b). For C rich sites (Fig. 2.3) there is closer agreement between estimates from the two databases for the 45-50 degrees and 60-65 degrees latitudinal classes than exists between 50 and 60 degrees where IFEF estimates are 2-3 times larger than those of Level I + II. For N-enriched sites (Fig 2.15b) the difference is far larger with IFEF estimates being 5-10 times larger. This could be explained by a greater proportion of low C/N sites within Level II+I than is currently featured in IFEF. For example 11% of N-enriched Level II+I sites have C/N of <15 as opposed to only 6% of IFEF N-enriched sites and there are no IFEF sites with C/N<12 in IFEF although these comprise ~5% of Level II+I.
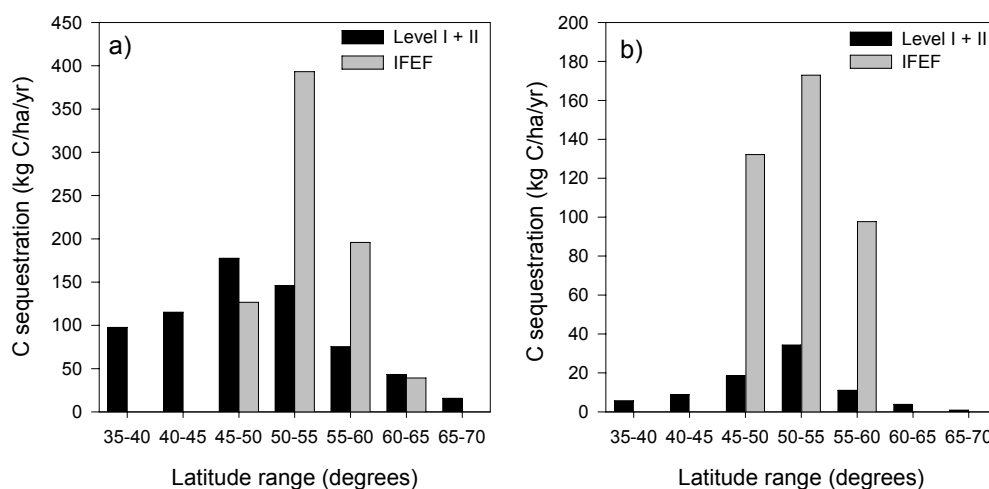
Figure 2.15: Comparison of C sequestration estimates via IFEF (calculated using measured N leaching) and Level I+II mass balance methods a) CN ≥ 23 i.e. C rich sites b) CN ≥ 23 i.e. N enriched sites.

2.5.4 Regional case studies

Biogeochemical databases for 20 spruce and 19 oak stands were collected (see also section 2.2.2) and used to provide estimates of N accumulation rates and carbon sequestration rates using the N balance approach. Results indicated that our estimates were highly dependent on the assumption of throughfall fluxes as a good indicator of N deposition. If this assumption is accepted, there is evidence that C sequestration rates are negative in soils beneath spruce stands in Wales (-486 to +7kgC/ha/yr) (Figure 2.16). This is not the case for most oak stands where positive C sequestration rates we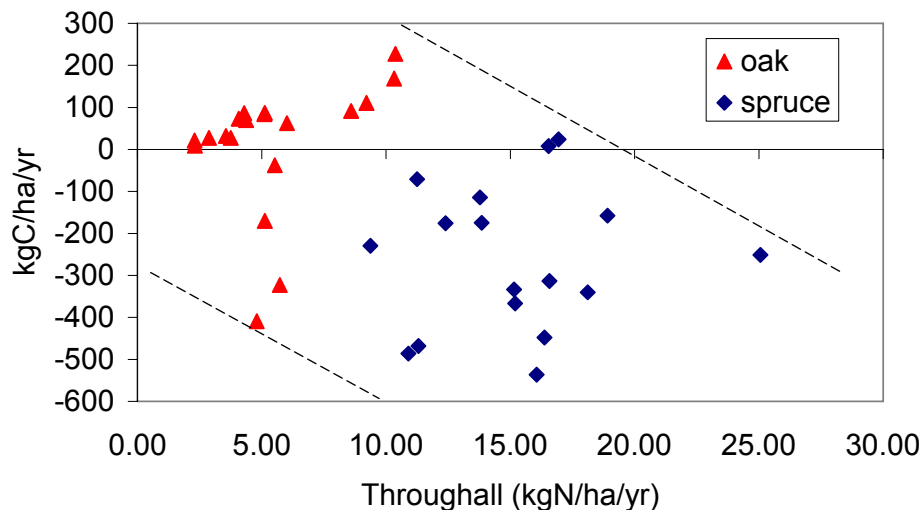re observed (-409 to +228 kgC/ha/yr), negative values from stands loosing nitrate. Likely explanations for the C loss from the spruce sites are that the disturbance and increased evapotranspiration rates beneath spruce after plantation on wet grass- or moorland accelerates decomposition and thus carbon loss from the soil. The negative rates calculated for spruce is thus an effect of the land use change. Negative N accumulation rates are calculated throughout the succession (Figure 2.17) However, we do not know if soil C/N ratios have increase somewhat from this change and thus in part may compensate the calculated C loss.

Figure 2.16: Carbon sequestration rates for Sitka spruce and common oak stands in Wales calculated using the N balance approach.

Forest floor accumulation in a chronosequence of Sitka spruce stands was also considered as a method of providing a second estimate of C accumulation analogous to the estimates based on the limit value concept. As seen from Fig. 2.18 the organic layer mass increased c. 1000 kg/ha/yr or c. 500 kgC/ha/yr, which is an amount comparable to the accumulation in other systems (i.e. Fig. 2.13). This approach was considered fundamentally flawed in this type of intensive forestry, as there is no consideration of the impact of afforestation on lower mineral soils. The N balance (Fig. 2.17) and modelling work indicated that net loss of N from the mineral soil occurs in both young and old stands in this type of intensive plantation forestry probably causing a loss of C as indicated in Fig. 2.16. Thus, C accumulation in a forest floor (Fig. 2.18) provides only one component of the carbon fluxes.

Afforestation chronosequences of both Norway spruce and common oak from Vestskoven, Denmark showed similar results as the Welsh Sitka spruce sequence. An accumulation of organic layers occurred with stand age but a greater loss of C from the mineral soils was indicated from soil sampling (Vesterdal et al. 2002)
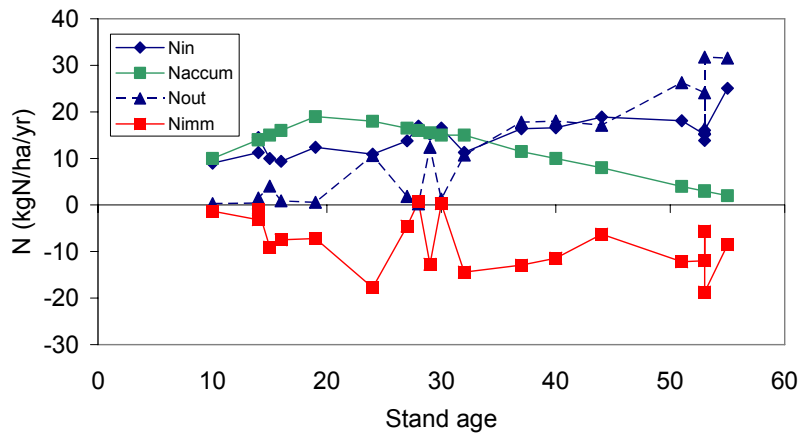
Figure 2.17: Nitrogen fluxes over the Sitka spruce chronosequence from Wales showing negative nitrogen accumulation (i.e. net loss of nitrogen from the soil).
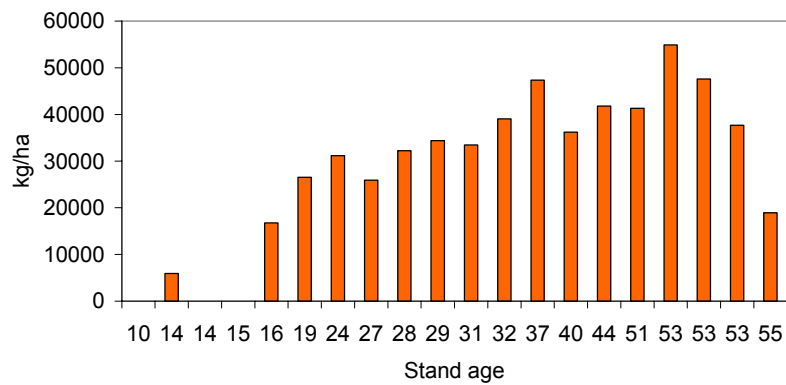
Figure 2.18: Forest floor development across a chronosequence of Sitka spruce stands in Wales, UK.

2.5.5 Conclusion

As the two different methods (limit value method and N-balance method) calculate two different carbon sequestration rates, namely the potential sequestration and a sequestration that considers forest management, respectively, they will serve different purposes. Although the latter, as expected, gives lower values than the former, they both give carbon sequestration rates that agree with well controlled measured values both on single spots and over a larger region.

Both methods should be used in parallel to calculate carbon sequestration rates. For coniferous forests in the boreal and the northern part of the temperate zone both methods appear to function and appear ready to be used and the now obtained values can be considered to be realistic and reliable. We would also conclude that for deciduous tree species and forests (mainly temperate) more work is needed to determine mechanisms for carbon sequestration. Such work should ideally be based on both methods.

3 Long-term experiments, impact of N deposition on C-N interaction

3.1 Sites and experiments

In the late 1980s and early 1990s, field manipulation experiments in forests were started with nitrogen deposition. The objective of these studies was to investigate the effect of decreased or increased N deposition on ecosystem functioning, with the focus on N saturation. Also, in most of these experiments ^{15}N was applied for a year as deposition. In CNTER the focus was on the effect on all relevant carbon (C) pools and processes. The use of ^{15}N made it possible to follow the long-term fate of this deposited N through the ecosystem. Moreover, it allowed to answer the question how much carbon (C) can be stored as a result of this extra deposited N. This method was applied for many of these sites by Nadelhoffer et al. (1999) for the short-term effect of N, and a new sampling in some of the sites was conducted in CNTER to study the long-term effect of N.

A nitrogen saturated ecosystem was defined by Aber et al. (1989) as an ecosystem exhibiting an increased loss of inorganic N in solution below the rooting zone. These experiments were designed to study the consequences of increased N deposition on N saturation and to investigate if N saturation was reversible. In all of our studies where N deposition was increased, inorganic nitrogen concentrations in solution increased. However, still after 10-15 years of increased N input a high percentage was retained in the trees or in the soil. This retention is either biotic by microbes or abiotic, if the increased N immobilisation doesn't match an increased production of CO_2. Many other measurements show that increased N deposition has influenced biota. The C/N ratio of the organic layer is another symptom of N enrichment, although the time-scale of these effects may be completely different.

Table 3.1. Characteristics of the sites and the experiments. The long-term fate of ^{15}N was determined in all sites; the shaded sites have long-term measurements on the C cycle.

	Gårdsjön	Klosterhede	Speuld	Ysselsteyn	Aber	Harvard Forest	Bear Brook
Country	Sweden	Denmark	Netherlands	Netherlands	Wales, UK	USA	USA
Tree specie	Norway spruce	Norway spruce	Douglas fir	Scots pine	Sitka spruce	Red pine / Oak	Beech-maple spruce
Age (y)	91-114	85	43	55	45	60	55
Ambient N deposition (kg N ha^{-1} y^{-1})	13	20	50	58	15	8	4
Manipulated N deposition (kg N ha^{-1} y^{-1})	49	55	4	6	35	58	29
Since	1991	1992	1990	1989	1991	1988	1990
^{15}N applied	Apr 92 - 93	Apr 92 - 93	May 92 - 93	May 92 - 93	Apr 92 - 93		1991 & 1992
N form applied	^{15}NH$_4$^{15}NO$_3$	^{15}NH$_4$^{15}NO$_3$	^{15}(NH4)$_2$SO$_4$	^{15}(NH4)$_2$SO$_4$	^{15}NH$_4$^{15}NO$_3$	^{15}NH$_4$^{15}NO$_3$	^{15}NH$_4$

Another question is addressed in the two Dutch forests. Is nitrogen saturation reversible, or is it possible to decrease nitrate leaching as a result of a sudden decrease in input. The first-year results showed an immediate reaction to decreased input, indicating that a large part of the leached nitrate originated from the deposited ammonium. This short-term reaction was confirmed by the application of ^{15}N. After 15 years of decreased N deposition, nitrate leach-

ing is still very low in these sites. Also, other measurements show that the decreased N input has lead to significant changes in internal N cycling. Trees grow faster, the N concentration in current-year needles has decreased dramatically and some endangered species of mushrooms have reappeared.

The characteristics of the sites and the experiments are summarized in Table 3.1. Measurements of relevant C processes were tree growth, litter production, litter decomposition, soil respiration (field and laboratory), and DOC leaching (Table 3.2). The long-term fate of ^{15}N was followed in the trees (needles, branches, wood) and in soil (organic and mineral soil).

3.2   Impact of N deposition on C sequestration

Tree growth: As long as trees can successfully compete with the microbes for the deposited N, growth can theoretically be stimulated by N additions. In the oak trees at Harvard Forest and in Bear Brook an increase was indeed measured, whereas in Klosterhede no change in tree growth could be detected. Obviously in the latter site most of the added N was retained in the organic and mineral soil. Alternatively, the measurements of the tree growth were not sensitive enough to detect changes. Surprisingly, a decrease in tree growth was measured in Gårdsjön after increased deposition. In Ysselsteyn, the high N deposition had been hampering tree growth. A decrease in N deposition improved tree growth significantly.

Table 3.2. Overall ecosystem response to 10-15 years of changed N deposition.								
	Gårdsjön	Klosterhede	Speuld[1]	Ysselsteyn	Aber[1]	Harvard Forest: Red pine	Harvard Forest: Oak	Bear Brook
Treatment	Increase	Increase	Decrease	Decrease	Increase	Increase	Increase	Increase
Tree growth	Decrease	Unchanged		Increase*	Un-changed	Unchanged	Increase*	Increase
Foliar N concentration		Increased*		Decrease*		Increased*	Increase*	Increase*
Litterfall N		Increased*		Unchanged		Unchanged	Un-changed	
Decomposition rate of fresh litter	Unchanged	Decrease* of N (external and internal N)	Un-changed	Decrease* of N (external N)		Unchanged*	Decrease*	
Respiration (field)	Unchanged	Unchanged		Decrease*		Decrease*	Decrease*	
Respiration (lab)	Temporal increase*	Increase* in new SOM decrease* in old SOM		Increase* in new SOM decrease* in old SOM		Decrease*	Decrease*	
DON leaching	Unchanged			Unchanged	Un-changed	Increase*	Increase*	
DOC leaching		Unchanged		Decrease		Unchanged	Un-changed	
NO$_3$ leaching	Increase*		Decrease*	Decrease*	Increase*	Increase*	Increase*	Increase*
* Means significant, if not significant it is a trend; 1. after 5-10 years, the rest after 10-15 years								

Box 3: Test of N effect on respiration

At Klosterhede, DK ,"new" SOM (LF layer) had higher respiration under N addition than control. For "old" SOM it is the opposite, reduced respiration after N addition (Fig. 3.1). This is in correspondence with the hypothesis by Berg & Matzner (1997) - that N stimulate in the first stage of decomposition and hampers decomposition in later stages. The net effect of this in the field may be hard to detect, but in the long run we would expect a slight increase in SOM.
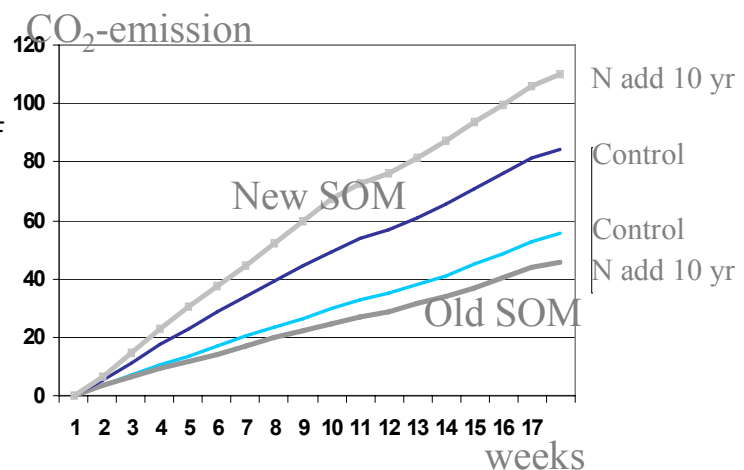

Figure 3.1: Respiration in a laboratory incubation of new and old SOM derived at ambient and elevated N deposition at Klosterhede, Denmark.

Soil respiration: An important carbon flux out of the ecosystem is soil respiration. It is difficult to separate the CO_2 efflux out of the soil between respiration by microbes decomposing organic material and respiration by plant roots. However, the decomposition rate of fresh litter is measured as well by a litterbag experiment. The storage of C in older litter will be addressed by the limit value concept (section 2.4).

The effect of changed N deposition on litter decomposition rate is a long term effect. In all European sites there was no or only a temporarily effect after 5-10 years of changed deposition. On the sites where the effect was measured after 10-15 years, the decomposition rate decreased. This was a result of simulated increased N deposition in Klosterhede and Harvard forest (oak litter), but due to decreased N deposition in Ysselsteyn. No effect was measured in Gårdsjön after 10-15 years. The effect on CO_2 loss by soil respiration showed a different picture. Field measurements of soil respiration revealed a decrease in CO_2 loss in Harvard forest as a result of N addition, and in Ysselsteyn as a result of a decrease in N deposition. For both sites, both field and laboratory measurements showed an effect, meaning that at least microbial activity was influenced and probably also root respiration. No changes in field rates of CO_2 loss could be measured in Gårdsjön and Klosterhede. The laboratory respiration measurements in these latter sites showed that there was an effect on microbial activity, but it was more complex especially for Klosterhede (Box 3). In the laboratory decomposition (respiration) of "new" SOM increased by N addition whereas it decreased for "old" SOM, thus these opposite directions of change may outweigh each other which explain that there were no 'net' effect on soil respiration in the field. However, the litterbag experiment in the field showed a significant decrease in decomposition (weight loss) as a result of the increased N deposition and is thus not consistent with the results from the laboratory. The decrease in decomposition was caused by external N (the fertilisation itself) as well as by internal N (the changed chemistry as a result of 10-15 years of increased N deposition).

The results after 10-15 years of changed N deposition give insight in the effect of N deposition in C sequestration. Increased N deposition did in some of our sites lead to increased amount of C in trees as long as the trees can compete with the soil microbes and the abiotic immobilisation for the added N. On the short term there is no obvious effect of increased N on the C loss by respiration of fresh litter. However, as a result of high N deposition, the chemistry of the needles will change on the long term which might decrease the C loss from

fresh litter. As a result, an overall positive effect of N deposition on C sequestration in the tree as well as in the (organic) soil is expected.

This effect can turn negative if N deposition remains high for a very long time, inducing for instance soil acidification. The experiment in Ysselsteyn showed that under that circumstance a decreased N deposition increased the tree growth and decreased the decomposition rate of fresh litter (only by external N), respiration rate measured in the laboratory and in the field. As a result, C sequestration in the ecosystem will increase. So, in a N limited forest receiving increased N deposition, as well as in an N saturated forest where input is drastically decreased, more C is sequestered (Figure 3.2).
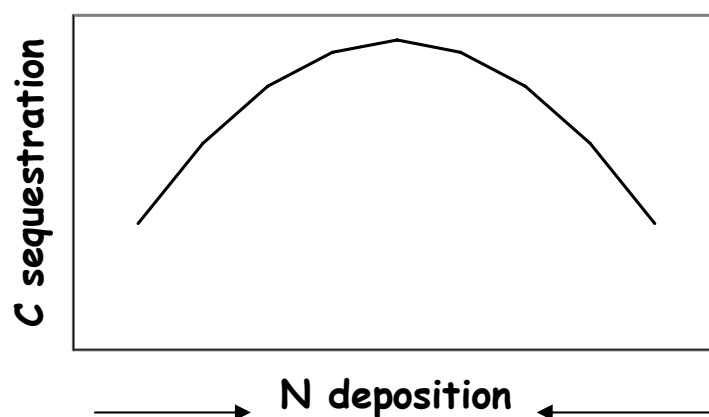

Figure 3.2: Conceptual and generalised model of the effect of N deposition on C sequestration in forests.

3.3 Long vs short term fate of 15-N

The long term, N manipulation experiments shown in Table 3.1 were intensively sampled (European sites in 2002/03) to identify the long term fate of ^{15}N additions applied in early 1990s. In the publication in Nature (Nadelhoffer et al.1999) we used the short-term fate of ^{15}N additions to roughly estimate soil C sequestration. With the long-term N fate we can now recalculate a more reliable estimate of the C sequestered with one year cohort of N deposition. In Fig. 3.3 an example of the type of data is given from the UK site, where some nitrate leaching occur, thus labelled N is leaving the system. However, the soil still contains the largest part of the added label. At the DK site in Klosterhede where almost all N is still retained in the system (no nitrate leaching) the soil is slightly increasing its share of the added label. This means that the conclusion from the Nadelhoffer paper that most of the deposition N is stored in the soil is also valid for the longer-term (decadal) fate.

In order to be able to fully analyse ^{15}N experiments, mathematical simulation models are necessary. These models can be of varying level of complexity but at least must contain the pools and fluxes or processes the researcher wants to quantify. Predictions of the movement of the stable isotope ^{15}N into ecosystem pools can be tested because changes in ^{15}N/^{14}N ratios in vegetation and soil pools can be easily detected. Using a biogeochemical process model to interpret the redistributions of field-applied ^{15}N tracers provides an opportunity to test model formulations of C/N interactions with more sensitivity than can be achieved with non-isotope models.
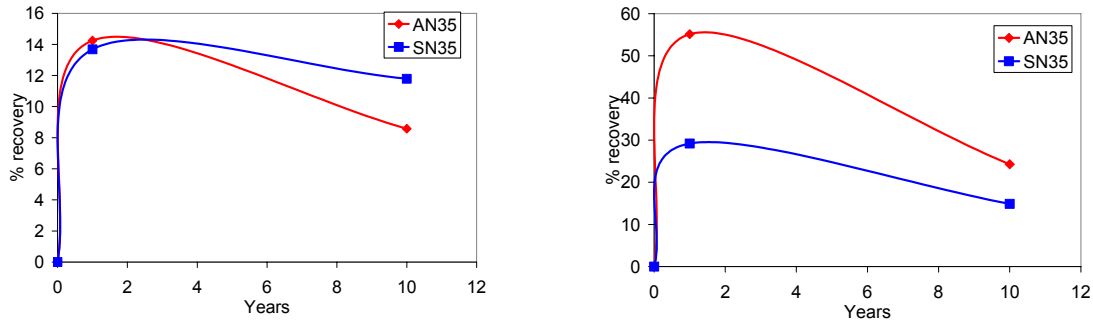
Figure 3.3. The change in recovery of ^{15}N in foliage (left) and soil (right) in a Sitka spruce plantation in Aber, Wales, UK (AN35 = ammonium nitrate 35 kg N/ha/yr added; SN35 = sodium nitrate 35 kg N/ha/yr added).

For recoveries of ^{15}N tracers in order to compare field data against model simulations, we express results as the mass of tracer ^{15}N recovered above background in each ecosystem compartment. We refer to this quantity as PR^{15}N (percent recovery of ^{15}N tracers), calculated as follows:

$$PR^{15}\text{N}(C_i,t) = \frac{\text{N}_{Ci}(t)\left(\text{atom\%}^{15}\text{N}_{Ci}(t) - \text{atom\%}^{15}\text{N}_b\right)}{A(t\text{-}t_0)\left(\text{atom\%}^{15}\text{N}_a - \text{atom\%}^{15}\text{N}_b\right)} \qquad (1)$$

where $\text{N}_{Ci}(t)$ is the amount of N (g / m^2) in C_i at time t, $A(t\text{-}t_0)$ is the sum of N amendments (g/m^2) to time t, C_i is an ecosystem compartment, and the 'a' subscript denotes amendment, 'b' denotes background (Currie & Nadelhoffer, 1999).
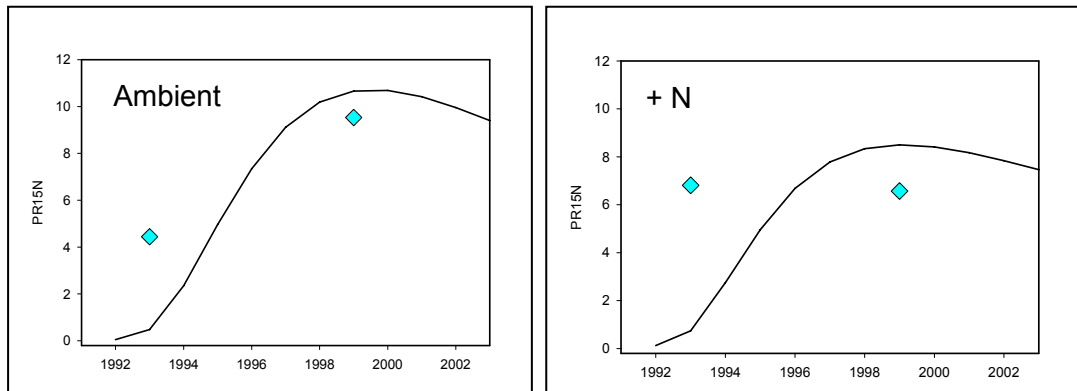

Fig. 3.4. Comparisons of TRACE model simulations (lines) and field data (symbols) for ambient (no N addition) and +N treatments at Klosterhede, Denmark. Currie et al. in prep.

For the CNTER project, an ecosystem process model also provides the ability to simulate the effects of elevated N deposition on rates of C storage in forests considered in this project. Progress on the topic requires analyses that go beyond simple stoichiometric calculations to those that can consider realistically integrated ecosystem responses to elevated N deposition. Forest ecosystems have the ability to shift C/N stoichiometries, change patterns of N allocation in tissues, and alter the fluxes of N among vegetation and soil pools. The retention of elevated N inputs in a particular forest ecosystem results from both short-term partitioning and longer-term redistributions that are likely to include complex interactions with C, light, water, and ecology of the existing vegetation.

Dynamic biogeochemical models provide a means of organising this understanding. Models allow us to quantitatively link fine-scale processes in an effort to understand the manner in which their interactions may control system-level dynamics. This is a key step in gaining a predictive understanding of ecosystem responses to global change. The TRACE model is an ecosystem process model that includes complex plant phenology and physiology, together with complex soil processes including gross N transformations, litter decay and humification, together with C-N interactions in soil.
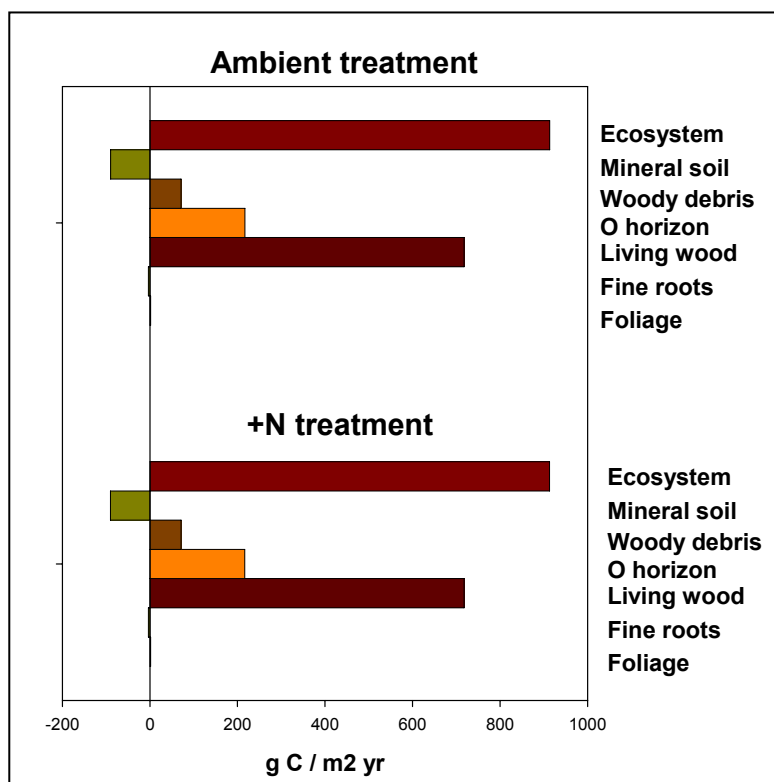

Figure 3.5: Present-day rates of C storage at Klosterhede, DK, modeled by TRACE. The model indicates little difference in the rate of C storage under +N treatment.

After parameterizing and calibrating the TRACE model to simulate C and N pools and fluxes at Klosterhede, model simulations of the redistribution of ^{15}N tracers were tested against data (Fig. 3.4). The model captured the general range of ^{15}N tracer recovery in foliage, lending credence to the modelled soil and plant processes for this site. The model also captured key aspects of an apparent difference in the 7-year recoveries of ^{15}N in foliage (Fig. 3.4): both the fact that the difference between the ambient and +N treatments were slight, and the direction of this slight difference. Model results were then analysed to assess simulated differences in rates of C storage between the ambient and +N treatments at Klosterhede. The model indicated no significant differences in C storage between the ambient and +N treatments at this forest site (Fig. 3.5). Similar results are obtained in the first applications to the sites in Ysselsteyn, NL and Aber, UK, but better calibration to simulate the actual C and N pools are needed for the final interpretation of the tracer data.

4 Modelling C and N dynamics

4.1 Introduction

Estimates of the carbon sink in European forests vary between 0.04 and 0.7 Gg C yr^{-1} (Janssens et al., 2003). This wide range over two orders of magnitude results from different assumptions in the methods available (IPCC, inventories, extrapolation from CO_2 eddy flux measurements, extrapolation from carbon pools and carbon turnover studies; inverse models as constraints for the European continental fluxes), from whether or not all the compartments are considered (wood, above ground biomass, below ground biomass, harvest, soil, distur-bance, etc.) and from huge errors in the extrapolation of tree or stand level data to much lar-ger scales. In the long run the soil is the ultimate sink or source of CO_2 for terrestrial ecosys-tems. However, soil processes account for the major uncertainties in the C cycle. The amount of carbon that will be fixed in the soil is the net result of CO_2 fixed by the vegetation followed by litter fall to the soil where it is ultimately (partly) decomposed. Since these sys-tems are generally limited by nitrogen and the C cycle is closely linked to the N-cycle, under-standing the N cycle is therefore crucial for the quantification of the long term behavior of soil carbon.

Important questions with respect to carbon sequestration are related to the cause of the large uptake of the mid-latitude forests and the time period in which the terrestrial sink will be satu-rated (Houghton et al., 1998). European forests have a role in net carbon sequestration of the biosphere due to growing stock (influenced by forest management) and changes in net primary productivity hypothesised to be due to increases in atmospheric CO_2 concentrations, nitrogen deposition (Holland et al., 1997; Nadelhoffer et al., 1999) and temperature and a prolonged growing season (e.g. Myneni et al., 1997). Using a modelling approach, tempera-ture has been claimed to be relatively unimportant, whereas the combination of CO_2 rise and elevated N deposition may account for a 15-20% increase in forest net primary productivity (Rehfuess et al., 1999). In this context, N deposition is claimed to be most important (Rehfuess et al., 1999, last ref recognition). The remaining explanation would then be the impact of forest management.

By far the largest amount of C stored in forests in the northern hemisphere is stored in the soil. Carbon fixed by photosynthesis ultimately moves via litter fall to the soil, where it is only partially decomposed. Thus, in the long term the soil is the ultimate sink or source of CO_2 for these ecosystems. Soil processes account for the most significant unknowns in the C and N cycle. Current hypotheses suggest that increased N deposition causes an increased rate of soil organic matter accumulation at least in two ways due to an increased leaf/needle bio-mass and litter production (e.g. Schulze et al., 2000) and a reduced decomposition of organic matter (Berg & Matzner, 1997). The N-content of forest litter and humus might thus be an important indicator of the C-sequestration.

In this chapter we present a quantification of carbon sequestration in the soil using the dy-namic model SMART2 (Kros, et al. in press), with a variant of the limit value concept (section 2.4) and with the N balance method (as in section 2.5.1) The use of a dynamic model makes it possible to quantify effects of scenario's, such as nitrogen deposition scenario's and tem-perature change, on C sequestration.

The aim of this study was: i) to model current C (and N) sequestration in soil with the dy-namic model SMART2 to compare with estimates based on the limit value and the N balance methods; ii) to predict the effect of changes in N deposition on C (and N) sequestration; iii) to gain insight in the long term change in C sequestration due to changes in N deposition on a European scale.

4.2 Sites and methods

From the European Intensive Monitoring Programme (Fig 2.1b) we have choosen 171 plots for the dynamic modeling, where a minimum set of surveys has been conducted so that validated information on bulk deposition, throughfall and soil solution chemistry is available and the water balance quantified. The distribution on tree species is shown in Table 4.1.

Table 4.1 Distribution of plots over combinations of tree species groups in ICP Level II

Species group	Species	Total
Spruce	Picea abies, Picea sitchensis	73
Pine	Pinus sylvestris	40
Other conifers	Abies alba, Pseudotsuga menzie-sii, Larix decidua, Pinus cembra, Pinus nigra	10
Standard oak	Quercus petraea, Quercus robur, Quercus cerris	17
Beech	Fagus sylvatica	29
Other broadleaves	Fraxinus excelsior	2

The model used for the *dynamic calculations SMART2* (Kros et al., 1995) is a simple, single-layer soil acidification and nutrient cycling model. It includes the major hydrological and biogeochemical processes in the vegetation, litter and mineral soil. The details on the C and N processing in the model are explained in Mol-Dijkstra et al. (submitted). Here also the calibration procedures are described in detail. The model results were compared to estimates based on the *Limit value concept* (section 2.4) modified to the data available and produced by upscaling procedures as describe in Mol-Dijkstra et al. (submitted) and based on *the N balance method* (section 2.5.1). Fundamental differences in the calculation of C sequestration by the three concepts are summarized in Table 4.2.

In SMART2, the calculated C pool changes in the soil are driven by C pool changes in the vegetation. The same is true for the limit value concept, but in the limit value concept, the C pool changes are related to the C/N ratio in litterfall. The N balance method is in principle different, since C pool changes in the soil are driven by N pool changes in soil. In fact, N immobilisation is calculated and C immobilisation is directly calculated from N immobilisation with a constant C/N ratio in the soil. A basic difference between the SMART 2 model on one hand and the limit value approach and N-balance approaches on the other is that in the two latter approaches recalcitrant remains are formed which simply are not degraded whereas in SMART 2 a slow degradation takes place also of the resistant components and a steady state is allowed to develop.

Table 4.2 Crucial processes for the calculation of C sequestration by SMART2, and the limit value and N balance concepts.

Processes	SMART2	Limit value	N balance
N deposition	X	(x)[1]	x
C and N litter fall	X	x	-
N uptake by growth	(x)[2]	-	x
C decomposition	X	x	-
N immobilization	X	-	x

1 Litter fall rates and their N contents are dependent on N deposition, but N deposition is

not included as a process as such.

2 Litter fall growth rates are linked to growth rates, but N growth itself does not influence C sequestration in SMART2

N deposition is the only parameter used by all three approaches. The total N deposition per plot was derived from both bulk deposition and throughfall of N, based on methods described in de Vries et al. (2001). Past and present scenarios for N deposition are based on transfer matrices of historic and future deposition in grids over Europe scaled to each site as explained in section 2.1 for estimation of cumulative deposition that were added to the databases.

Tree growth is used in both SMART2 and the N balance method. The actual N uptake was derived by multiplying changes in standing biomass in terms of stem wood with deposition dependent N contents in biomass as also explained for the databases (section 2.1).

For the modeling of C sequestration it is important to have reliable estimates of litterfall, which is strongly related not only to climate but also to tree species. These data are not currently available but were estimated for each tree species or group of species (in Table 4.1) by upscaling models suggested by Berg & Meentemeyer (2001) which was also used in the upscaling procedures used in the studies presented in section 2.4.3.

Calibration of SMART2 is done in several steps as detailed by Mol-Dijkstra et al. (submitted). As one of the lasts steps N leaching from SMART2 was compared with data and estimates based on empirical relationship as developed in section 2.2.2. In Fig 4.1 the estimates are compared in relation to C/N of the forest floor. The pattern is captured reasonably well by the model.
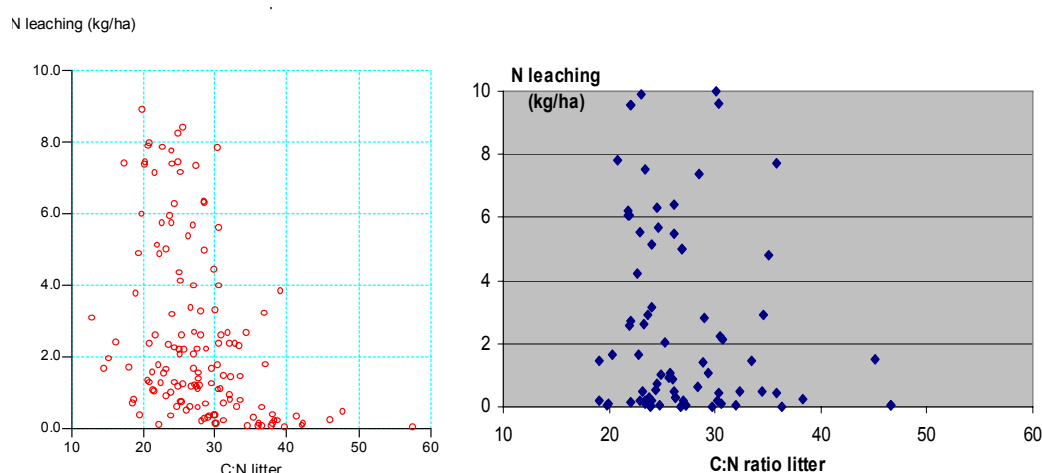

Figure 4.1: Simulated N leaching against simulated forest floor C/N with SMART2 (left) and from data and empirical relationships (right).

4.3   C-sequestration rate estimates modelled using three methods

Current C sequestration
The modelled C-sequestration rates in the soil varied largely between the three approaches. With SMART2, we calculated values from zero to 200 kg ha^{-1} yr^{-1} and with the limit value concept from 80 tot 600 kg ha^{-1} yr^{-1} (Fig. 4.2). The empirical method resulted in values between the results of SMART2 and the limit value concept. The large difference between the SMART2 and the limit value results is caused by difference in the decomposition concept. In

the SMART2 model, in principle all incoming litter fall is decomposed at the end. This means that it is possible to reach a steady state where decomposition equals the litter fall, resulting in a C sequestration of zero. In the limit value concept, there is an amount of fresh litter fall that is not decomposed, which directly contributes to the C sequestration. The limit value was allowed to vary between 43% and 87%, which means that at least 13% of the fresh litter is not decomposed. The empirical method is fundamentally different from the other two methods.
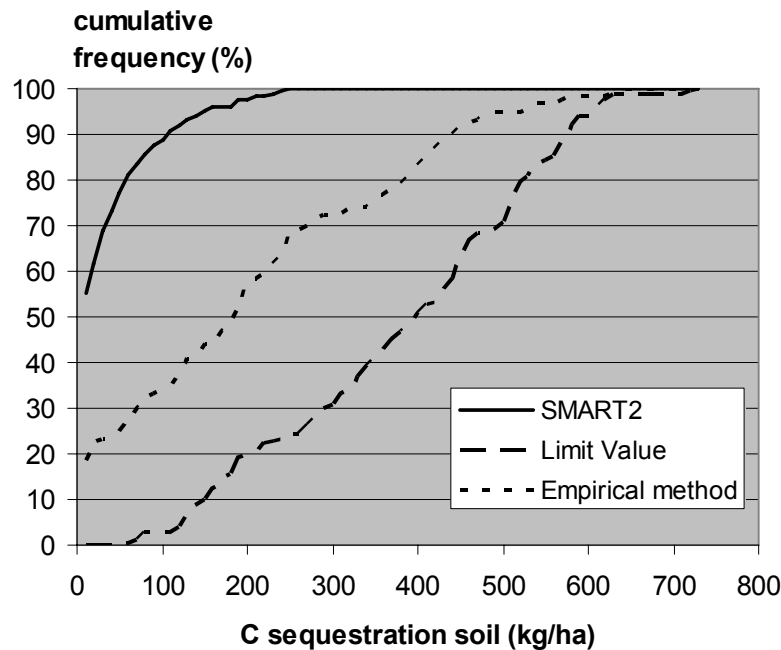

Figure 4.2 Cumulative frequency distribution of C sequestration in the year 2000, calculated with SMART2, the N balance (empirical) method and the limit value concept (kg m^{-2} yr^{-1}).

Besides the amount of C sequestered, there was also a difference in the distribution over Europe. The SMART2 model calculated higher values in Sweden and lower values in Germany, in contrast to the limit value concept and the N balance method (Fig. 4.3) The steady state will occur early when litter fall is low and the mineralisation rate is high. In Sweden for instance, SMART2 calculated higher C sequestration than in Germany. This has possibly to do with the effect of temperature on the mineralisation rate constant. Lower temperatures imply lower rate constants. On the other hand higher pH values were calculated in Sweden than in Germany, which would have an opposite effect.
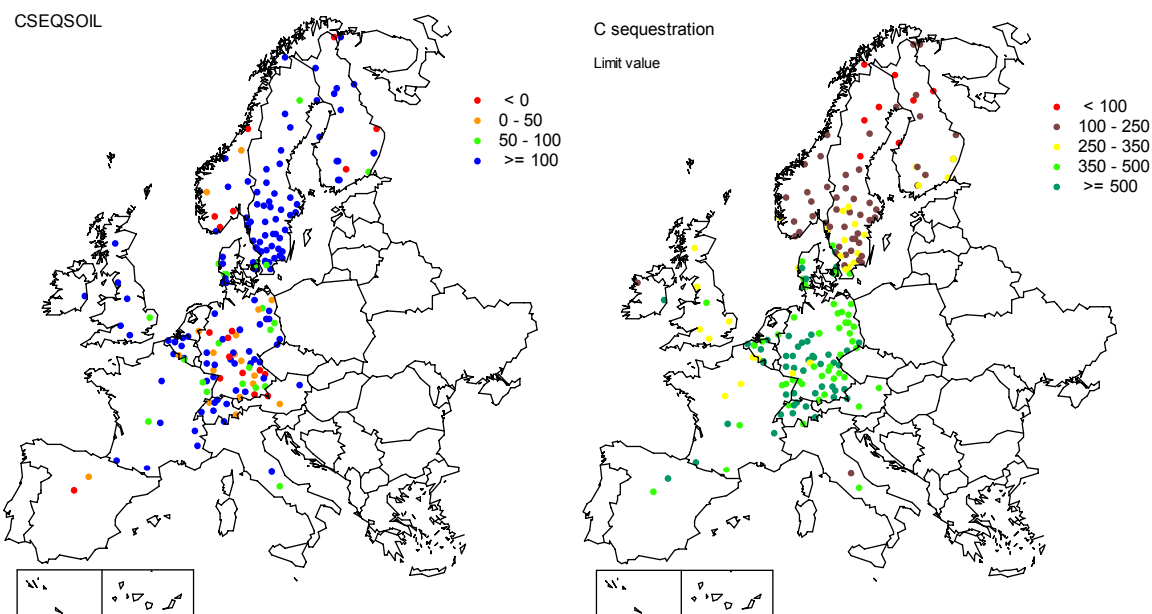
Figure 4.3 C sequestration calculated with SMART2 (left) and the limit value concept (right) at level II plots (kg ha^{-1} yr^{-1}).

C sequestration rates in the long run

Compared to 2000, both SMART2 and the limit value concept calculated lower C sequestration values in 2050 (Table 4.3) which was partly an effect of reduction in N deposition as the NEC scenario has just a slight reduction of N deposition in 2050 compared to 2000 (see Table 4.3) The lower C sequestration in the SMART2 modelling was the result of having more plots in a steady state in 2050, besides a somewhat lower input of litter fall. The lower litter fall and lower N concentrations in litter fall resulted in lower calculated C sequestration rates by the limit value (minus 5-6%) concept too. With the N balance method is was not possible to calculate C sequestration in 2050 as it uses measured N concentrations and measured C/N ratio's.

Effect of N deposition on C sequestration rates in the long run

With the reduced N deposition scenario (MFR) the calculated C sequestration rate decreased compared to the results with the no change scenario (NEC). With SMART2 the calculated reduction in C sequestration was 79% and with the limit value concept was 6-12%. For both methods, the decreasing litter fall was an important cause for the decreasing calculated C sequestration. An additional effect is that with lower litter fall input, the steady state in SMART2 is reached earlier. For the limit value concept, there was another additional effect of decreasing N deposition. Due to lower N contents in litter fall, the decomposition (limit value) increased and thus C sequestration decreased.

Table 4.3: Carbon sequestration (kg ha^{-1} yr^{-1}) calculated with three methods in the year 2000 and 2050 for the NEC (no change) and MFR (decrease) N deposition scenarios.

Year (scenario)	Percen-tile	Method		
		SMART2	Limit value	N balance
2000	5%	0	122	0
	50%	3	396	184
	95%	141	607	532
2050 (NEC)	5%	0	122	-
	50%	0	372	-
	95%	68	578	-
2050 (MFR)	5%	0	115	-
	50%	0	332	-
	95%	41	506	-

4.4 Discussion and conclusions

Litterfall: Good estimates of litterfall are crucial for the calculation of C sequestration with the SMART2 model and the limit value concept. Since measured litter fall data were not available we used regression equations to estimate litter fall. Using these equations with an R^2 of 43% for pine and 48% for spruce, means a quite large uncertainty of litter fall.

Tree growth: From 1980 onwards, a constant growth was considered. This was done in order to avoid interference of age and/or rotation period of the individual sites when performing long-term scenarios. Using a constant growth rate in SMART2 may ultimately lead to a steady state in soil carbon (see discussion about decomposition). We assumed no management when we used constant growth from 1980 and onwards. Management like thinning and clear-cut may have serious impacts on carbon pool changes in the soil.

Decomposition: The SMART2 model and the limit value concept differ fundamentally from the N balance method. Both for SMART2 and for the limit value concept litter fall is crucial input. Uncertainty in litter fall directly leads to uncertainty in C sequestration rates calculated with these methods. The N balance method just needs N fluxes, which are partly measured and may have less uncertainty. Remarkably, the N balance method does not need any C input to calculate C sequestration. The N balance model assumes a constant C/N ratio in the soil and links the C sequestration via this constant C/N ratio to the calculated N immobilisation. With high N-deposition rates, it is more likely that the C/N ratio decreases than increases and thus the calculated C-sequestration rate is an upper limit.

In another way, SMART2 differs fundamentally from the other two models in the results of C sequestration. With SMART2 it is possible to reach a steady state where decomposition equals incoming litterfall, resulting in a C sequestration of zero. Finally, in the long term all plots will reach a steady state when model inputs such as litter fall and growth are constant. With the limit value concept and the N balance method, the calculated C accumulation is always above zero. This implies that the litter layer and/or C stock in the soil would permanently be increasing in an unmanaged forest soil. Both the permanently increasing C stock in soil and the steady state can be queried.

C sequestration: The calculated C sequestration rates in the soil show large variations between the three methods. With SMART2, we calculated very low values (median of 5 kg ha^{-1} yr^{-1}), with the limit value concept high values around 400 kg ha^{-1} yr^{-1} and with the N balance method, the calculated C sequestration was in between the two other methods (around 180 kg ha^{-1} yr^{-1}). Liski et al. (2002), found an average value in 1990 of 190 kg ha^{-1} yr^{-1}, also with a large variability over Europe using a modified version of the ForClim-D model (Perruchoud et al., 1999). For Northern Europe they found increasing C sequestration rates in the exam-

ined period. Looking at the shape of the curves, the model used by Liski et al. (2002) seems to reach a steady state in the very long term. Dynamic models reach a steady state in undisturbed (unmanaged) forest soils.

Based on the comparison between the three methods, it would be appropriate to apply a slower decomposable pool to SMART2, with a possibly recalcitrant part. Besides, effects of management and land-use history should be taken into account. The presented simulations suggest that a decrease in nitrogen deposition may substantially decreases carbon sequestration in both coniferous and deciduous forest, which would jeopardize the fulfillment of the Kyoto protocol.

N leaching/retention: The simulated N concentrations with SMART2 did not fit the measurements very well, although both measured and simulated N concentrations were low. To improve the simulations, more site-specific model-input data are needed. Litterfall, its N contents and N mineralisation have a strong effect on N leaching in the SMART2 model, both are quite uncertain in this application. Furthermore N concentrations show large variability in time. As SMART2 calculates yearly values it is difficult to compare the model results with measurements (with just a few measurements in a year). On the other hand, improvements of SMART2 may be found in the description of the decomposition process and the effect of C/N ratio on decomposition. Decomposition is limited to the litter layer, but it may be extended to the mineral soil, by introducing a slow decomposable pool. This suggestion has also been made at the discussion about C sequestration.

Both modelled and measured N leaching varied between 0 and 30 kg ha^{-1} in 2000. In 2050, the ranges were comparable, just slightly lower, which is the consequence of different processes in the SMART2 model. Lower N deposition leads to lower N uptake, which means that the N surplus is not reduced equally to the N deposition reduction. On the other hand less growth implies less litter fall with possibly lower N contents, which leads to lower N leaching. Another effect of the lower N and S inputs is that pH increases, resulting in higher mineralisation rates. The net effect of all these processes in SMART2 is small.

5 Management impact on C and N pools and fluxes

5.1 Introduction

The focus on forest management practices has increase due to an interest in C sequestration in forest ecosystems acknowledged in the Kyoto protocol as a tool to reduce atmospheric CO_2. Maximising carbon sequestration rates is one of the compensatory measures possible to decrease the rising concentration of the greenhouse gas CO_2 in the atmosphere. Forest ecosystems sequester carbon in pools with strongly varying life spans. In case of soil organic matter with a residence time of more than 100 years, it would require a long accumulation period to reach to an equilibrium level (if it exists?, see section 2.4). It has been estimated that efficient management of the world's forests may increase their storage capacity by 1.2 Gt C per year (Winjum et al. 1992) by development of management strategies that increase C storage and prevent C loss by CO_2 emission.

There are several ways that forest management can influence C storage above ground as well as in the whole wood producing chain. The carbon in the soil compartment of forests may have a relatively long life span in itself, but can be managed. It may remain intact after harvesting the trees, but this is, however, subject of debate and conflicting observations exists (Wessel and Tietema, 2003)

The influence of forest management on soil N retention and nitrate leaching was the focus of an extensive review (Gundersen et al. 2006). Protection and improvement of forest water quality with respect to nitrate require both reductions in N emissions and use of management options that may alleviate N saturation. Management options reducing input or removing N from the forest ecosystem include replacement of conifers by deciduous trees, shorter rotations, whole-tree-harvesting, fertilising with limiting nutrients, prescribed fire, or protecting/improving the riparian zone. Most work has been done on coniferous forest and deciduous forests have not revealed the same clear correlation between N status and N leaching as coniferous species (see chapter 2).

Within CNTER we have evaluated the potential for environmental benefits of forests (C-sequestration and N-retention) through forest management and silvicultural practices. The major ecosystem impacts we have considered have been (1) changes in forest soil carbon storage, and (2) changes in nitrate concentrations in forest waters. Based on the review of the influence of forest management on soil C storage (Wessel and Tietema, 2003) and N leaching (Gundersen et al. 2006) we decided to focus on 1) the effect of tree species and 2) the impact of felling regime and age structure.

Although there is some information on differences between conifers and broadleafs, differences between species are poorly known. Since N-deposition in coniferous species is approximately two-fold higher than in broadleaves this dimension became increasingly important as deposition increase. The differences between species imply that management practices favouring certain species may delay or accelerate N-leaching and the potentially negative changes associated with this process. In C. and NW. Europe, where broadleaved species were replaced by conifers, trends to return to broadleaved species may alleviate the relative widely spread elevated nitrate level in seepage water from European forests.

In studying the effect of species it has been a problem that deciduous and coniferous forest types in Europe tend to grow on different soil types with deciduous forests on richer soils. Within CNTER, we have been able to study the effects in tree species trials established on similar soil types.

The major single man-made disturbance applied to forest ecosystems is felling. Although there is much information about the effect of felling it is conflicting with respect to C (Wessel and Tietema, 2003) and for N retention the effect of soil type and C/N ratio is not clear (Gundersen et al. 2006). The combined effect of biomass utilisation regime and site specific conditions may influence both carbon storage and N-retention as a number of processes are affected, which exert important controls on C-sequestration, N-retention and nitrate leaching after harvest.

5.2   National surveys – identification of relevant management options

To evaluate the potentials for environmental benefits from forest management we have revisited former management trials with a good historical record of the treatments/manipulation in DK, NL and UK. The medium- to long-term effects of the applied management options on C and N dynamics have been evaluated by sampling indicators (such as C/N ratio, soil nitrate concentration or soil C-pool) in these experiments. In DK 72 stands selected in large tree species trials planted in 1964/5 and 1973 were sampled for C and N pools, and soil nitrate. The contribution from the UK focused on a harvest/regeneration survey where nitrate leaching was already determined at >50 sites with known harvest history and cultivation intensity. Additional sampling of soil C and N pool and vegetation data has been performed. In NL 75 stands within the Edese bos area were sampled for soil C and N pools and soil nitrate. Results were analysed for effect of stand age and tree species.

5.2.1   The tree species trials, DK

The impact of tree species on carbon (C) sequestration and nitrogen (N) retention was studied in a Danish tree species trial established in 1965. At 13 different locations around the country 12 common species were planted in 1965. From 7 of these trials we sampled 4 species, beech, oak, larch and Norway spruce. Further, we included two other trials established in 1973 with beech, oak, larch, ash and Norway spruce. As all the tree species except ash were present on four different soil types the design enables us to discriminate between differences in tree species and site conditions including N-deposition.

In each stand, we sampled forest floor and the upper 30 cm of the mineral soil along 3 transects. In each transect 5 subsamples were taken and pooled to give 3 composite samples. We sampled the forest floor by area sampling and the upper 30 cm of the mineral soil by a 5 cm diameter auger in order to determine soil density. The forest floor was divided into fine litter remains and coarse woody debris and C and N concentrations and pools were determined on the forest floor fractions and the mineral soil in 0 – 30 cm depth.

In all the stands, soil NO_3-N concentrations were determined at 90 cm depth. Along the same three transects, five soil samples were taken by a soil corer and pooled to give three composite samples. All samples were taken in February and March 2003. Nitrate concentrations were measured after KCl extractions and converted to soil water concentrations taking the soil moisture into account.

Carbon content in the organic layer was highest in the larch stands but also Norway spruce accumulated significant amounts of organic matter whereas the deciduous species accumulated less with larch > Norway spruce >> beech > oak > ash. In the mineral soil, there was slightly more carbon in the spruce stands so that the larch and the spruce stand had similar amount of C in organic layer plus 0 – 30 cm mineral soil significantly larger than beech > oak > ash.

Concentrations of NO_3-N in the soil water varied among species and across sites for the same species. Across sites beech had the highest amount of nitrate in the soil solution with beech > spruce > larch > oak > ash (Fig. 5.1).
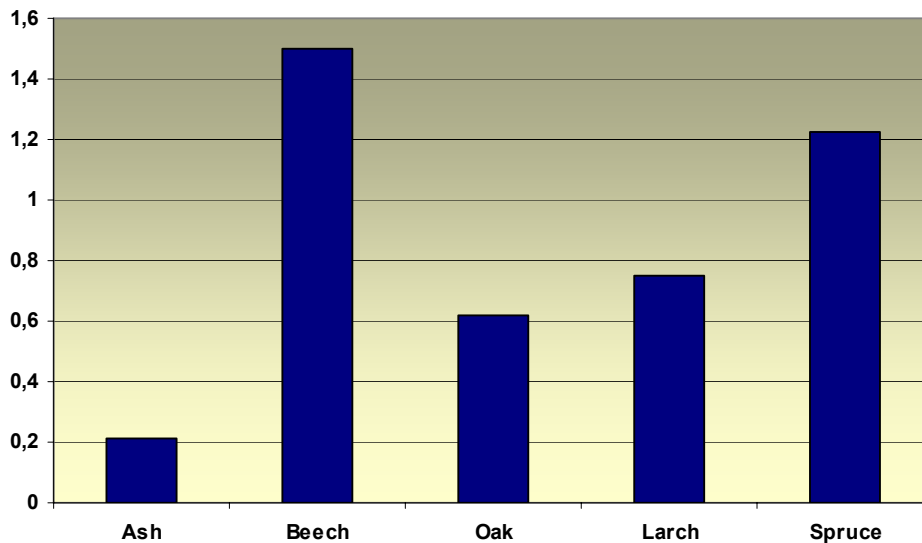

Figure 5.1: Average seepage water nitrate-N concentration in relation to tree species in even-aged stands at 9 sites in Denmark.

The highest nitrate concentrations despite tree species were found in the soils with the highest pH and clay content whereas very low concentrations were observed at the sandy podzolised soils (Fig. 5.2). This is negatively correlated with the organic layer mass and carbon content but is not related to the total carbon storage in the upper soil layers. In general though, the concentrations are low and among the 112 samples only two had concentrations above 5 $mgNO_3$-N/L. Ammonium-N was below 1 mg NH_4-N/L in all samples and in most cases below 0.5 mg NH_4-N/L.

The observed relationship between nitrate leaching and organic matter C/N ratio as reported in the literature for coniferous forests is difficult to evaluate in this material as the C/N ratio varies to a minor extent and mostly between 25 and 27, which is the threshold below which nitrate leaching increases. Beech and oak have the highest span in C/N ratio from 22 to 32 (beech) and 19 to 33 (oak). Beech shows both high and low leaching at each end of the range whereas nitrate concentrations in oak stands are low in the upper end of the range but both low and high when the C/N ratio drops below 25.
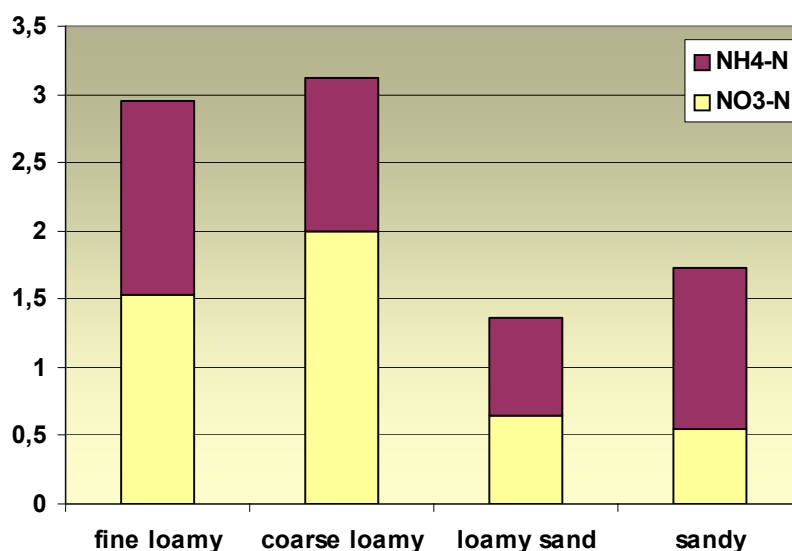
Figure 5.2. Average seepage water nitrate and ammonium concentrations (mgN/l) in relation to soil type in 36 even-aged stands at 9 sites in Denmark.

5.2.2   The effect of harvest regime, UK

Felling involves a significant disruption to the physical, chemical and biological processes within a forest. Initial work at Hubbard Brook identified the potentially large losses of nitrate during this period of disturbance (Likens et al. 1970). The magnitude of losses has been associated with a range of factors including intensity of harvesting, nitrogen status of the forest prior to felling, age of the forest and its fertility (Gundersen et al. accepted). Losses can continue for as long as 10 - 15 years after felling (e.g. Neal et al. 1998) until the biological sinks (ground vegetation and regrowing stand) have been re-established. Controls on dissolved organic carbon (DOC) loss have been less well studied but are an area of interest due to both concerns about colour in waters for the water industry and the potential importance of DOC to the carbon balance of a site particularly in areas dominated by soils rich in organic material.

A unique study of 60 catchments (2 – 5ha) in Wales (Neal et al. 1998) provided an opportunity to examine controls on nitrate, ammonium and dissolved organic carbon concentrations in a wide range of catchments. Streamwater in these catchments had been sampled in 1990 on up to 8 occasions during both baseflow and storm flow conditions. 53 of these sites were revisited in 2002/03 and soil sampling carried out. Soils were sampled on both the slope and valley bottom to identify the potential effect of proximity of soils to stream water quality. Twenty five environmental parameters were collected in total including; tree species, climate, deposition, altitude, slope, aspect, soil texture, C/N and pH of soils on slopes and in valley bottoms.
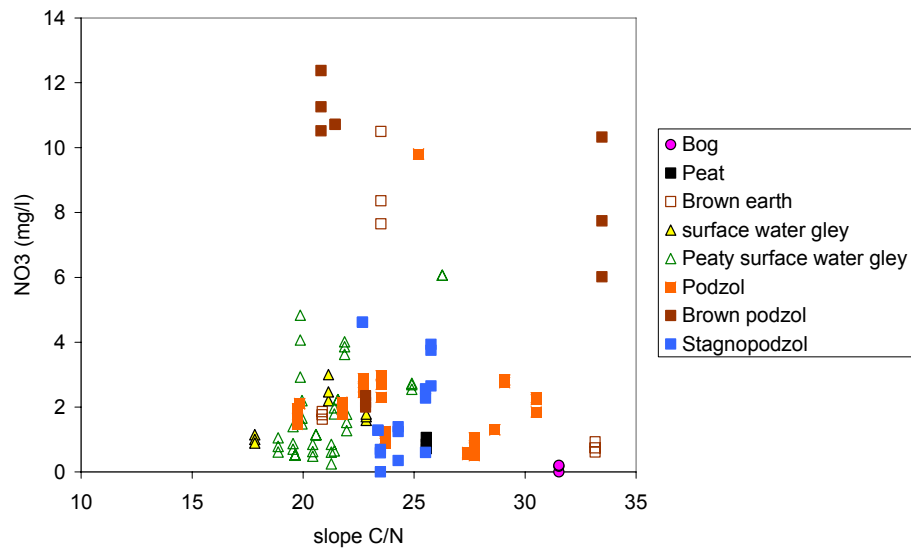
Figure 5.3 Nitrate concentrations and C/N of the dominant soils on catchment slopes at 53 Welsh sites

Contrary to expectations, results indicate no relationship between soil C/N and nitrate concentrations in streamwater either in the valley bottom or on the slope (Fig. 5.3) or for any soil type for either base flow or storm flow. Again no relationship was observed for DOC (Fig. 5.4). Ammonium concentrations are most strongly associated with DOC but not with nitrate (not shown).
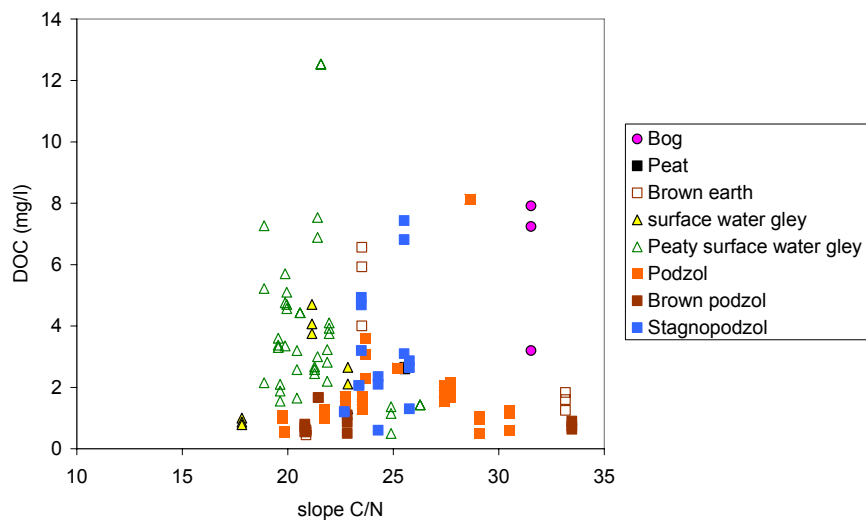

Figure 5.4: DOC concentrations and C/N of the dominant soils on catchment slopes at 53 Welsh sites

The major factor which appears to be related to nitrate concentrations in streamwater at both storm flow and baseflow (Fig. 5.5) is the depth of the organic layer in either the slope or valley bottoms. Similar relationships were observed for soils in both locations. Surprisingly, no relationships with DOC were observed.
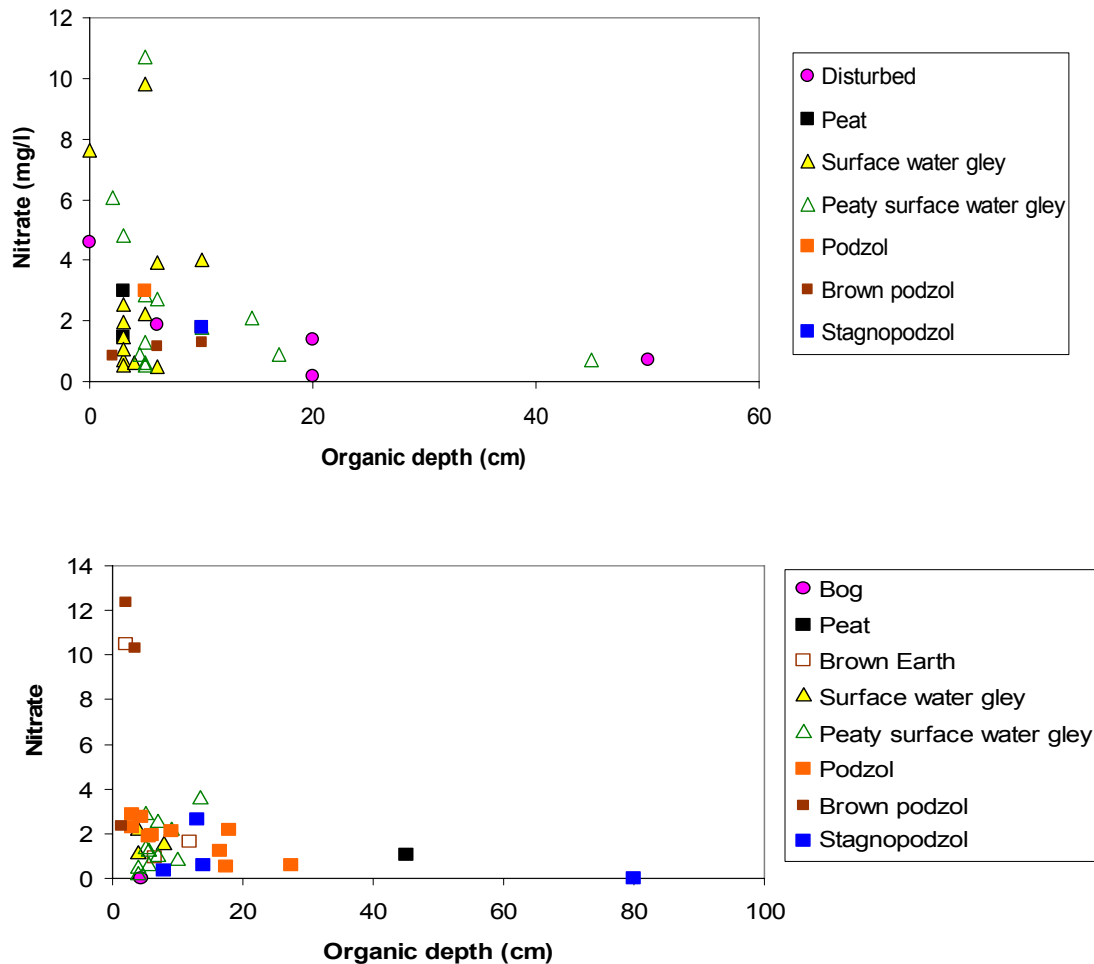
Figure 5.5: Depth of organic layer a) in the valley bottom slope and stormflow nitrate concentrations in streamwater (upper panel), and b) in the dominant soil on catchment slope and baseflow nitrate concentrations (lower panel).

This suggests that it is the size of the soil sink for nitrogen (ie. the amount of organic soil) which determines the retention capacity for N at a site. Shallow or mineral soils with < 10cm of organic layer have a lower retention capacity for N and therefore they have greater potential to cause eutrophication of streamwater. Surprisingly, this relationship was not observed for DOC due to confounding effects of rainfall. Analysis is continuing to separate out these confounding effects. In conclusion, this study has identified an easily measured indicator of N retention following felling. Work continues to identify a similar indicator for DOC.

5.2.3   Edese Bos, NL

The infiltration area "de Driesprong" (10 km^2) is situated north of the town of Ede in the centre of The Netherlands. The groundwater retention time, within the infiltration area, is 25 years. Land use of the infiltration area is mainly forest (55%) and heathland (25%), while the rest is urban and arable land. The forested area consists of three forests: "Edese bos", "Roekelse bos" and "de Valouwe". Scots pine is the dominant tree species (47% of the total forested area), followed by larch (15%), Douglas fir (12%), beech (12%) and Norway spruce (5%). In 75 out of 350 forest stands, the C/N ratio of the organic layer was determined.

These 75 stands consisted of Scots pine (28 stands), Douglas fir (15), larch (9), Norway spruce (9), common beech (10) and oak (4). In each stand, 4 samples of the organic layer were taken by means of a PVC tube (length = 20 cm, internal diameter = 11.8 cm). Woody particles and roots (> 2 mm) were removed, and the samples were dried at 70 C and weighed. After grinding, the 4 samples of each stand were bulked and C and N concentrations were determined in triplicate subsamples with a CN analyser. The C/N ratio (g C g^{-1} N) was calculated as the ratio between the C and N storage (g m^{-2}).

Statistical differences between forests and tree species were tested with a one-way analysis of variance (ANOVA). A Tukey t-test was used to test the significance between means. Significant differences between means are given for $p < 0.05$, unless otherwise noted.

In 30 of the 75 forest stands, NO3- leaching was determined. Stands of Scots pine (10), Douglas fir (5), larch (5), Norway spruce (5) and beech (5) were chosen to cover the whole range of C/N ratios and tree ages. Soil samples at 3 depths (1.20 m, 1.40 m and 1.60 m) were taken in January 1999. In each stand two soil cores were taken, which were mixed per depth. After determining the moisture content, the samples were extracted with water in a 1:1 extraction ratio and filtered (0.2 µm). Nitrate concentrations were measured by automated colorimetry using an autoanalyser.

To translate nitrate concentrations into nitrate leaching, water fluxes were calculated with the hydrological model SWIF. Daily precipitation of the previous 4 years at the neighbouring monitoring site Speuld were used as input. Tree-specific input parameters for describing throughfall and transpiration were used to differentiate between tree species. Soil physical parameters characteristic for loamy, sandy podsols were used. Nitrate leaching was calculated as the product of the mean nitrate concentration of all three depths, and the calculated water flux at a depth of 1.50 m for the year 1998. WANDA was applied on all 30 stands to compare the simulated results with measured nitrate concentrations and modelled water fluxes.

The measured C/N ratios in the organic layer of the 75 forest stands varied from 15.7 to 31.3. The C/N ratio varied significantly per tree species (Fig. 5.6a). The C/N ratios in the Scots pine stands were significantly higher than measured in the Douglas fir, oak and beech stands. No other significant differences in C/N ratio between the tree species were found (Fig. 5.6a).
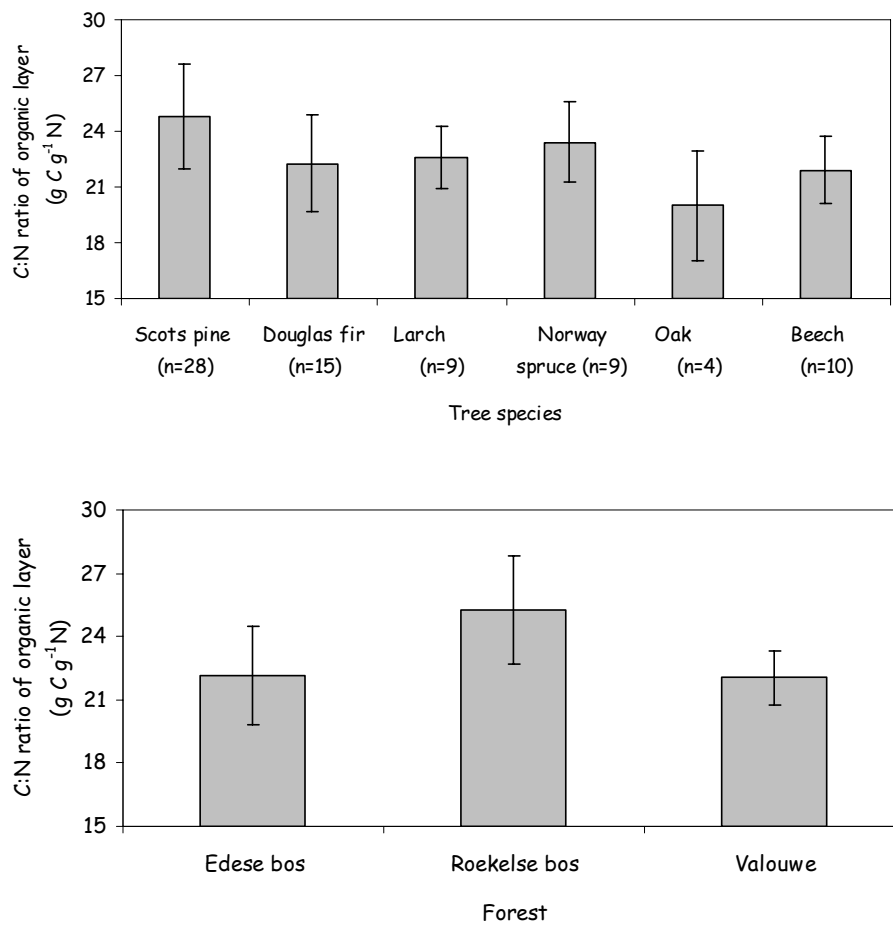
Figure 5.6: C/N ratio of the organic layer in 75 forest stands as a function of tree species (above panel) and the location in the three forests occurring in the infiltration area (lower panel). Bars show means with standard deviation.

Heterogeneity in the C/N ratio in the forested area is evident, even within only a limited area of 10 km². The C/N ratio in the organic layer of the Scots pine stands varied from 18.9 to 31.3. The "Roekelse bos" showed significantly higher C/N ratios (Fig. 5.6b). This is probably caused by significant difference in the location and age of the three forests. The "Roekelse bos" is located on the top of a ice-pushed moraine, compared to the other two forests which are located on a lower part of the moraine, leading to differences in chemical composition of the minerals. However, also differences in land-use in the period before it became forested might have caused this discrepancy. Variation in N deposition between the stands is expected to be relatively small given the small area. The data of the Scots pine stands showed a clear effect of age on the build-up of the organic layer. Total storage of C as well as N (not shown) in the forest floor was significantly correlated with tree age (Fig. 5.7), indicating a linear build-up of the organic layer over time. The C sequestration in this layer is approximately 600 kg C/ha/yr which compare well with estimates calculated for organic layers (limit value concept) in this region (see sections 2.4-5).
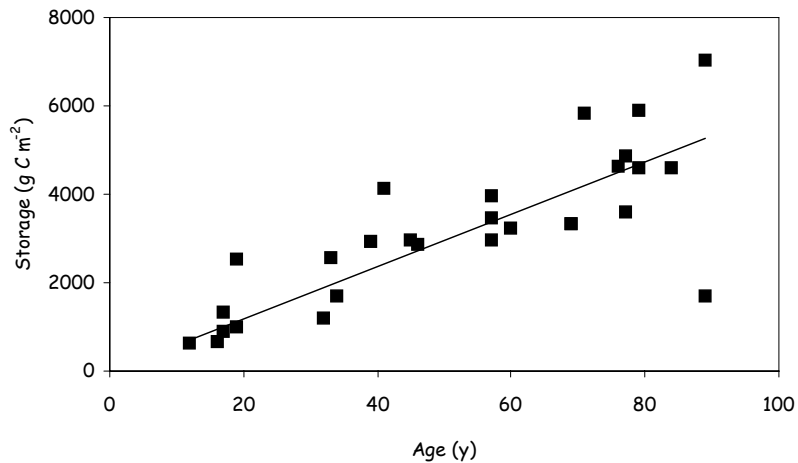
Figure 5.7: Total storage of C in the organic layer of Scots pine stands as a function of tree age in Edese Bos, NL.

The measured NO_3^- concentrations varied from 1.7 mg NO_3 l^{-1} to 75.3 NO_3 mg l^{-1} (Fig. 5.8). The highest NO_3^- concentrations were measured in the Douglas fir locations, whereas the lowest NO_3^- concentrations were measured in the beech stands. Ammonium concentrations were less than 8 mg l^{-1} at all depths (results not shown).
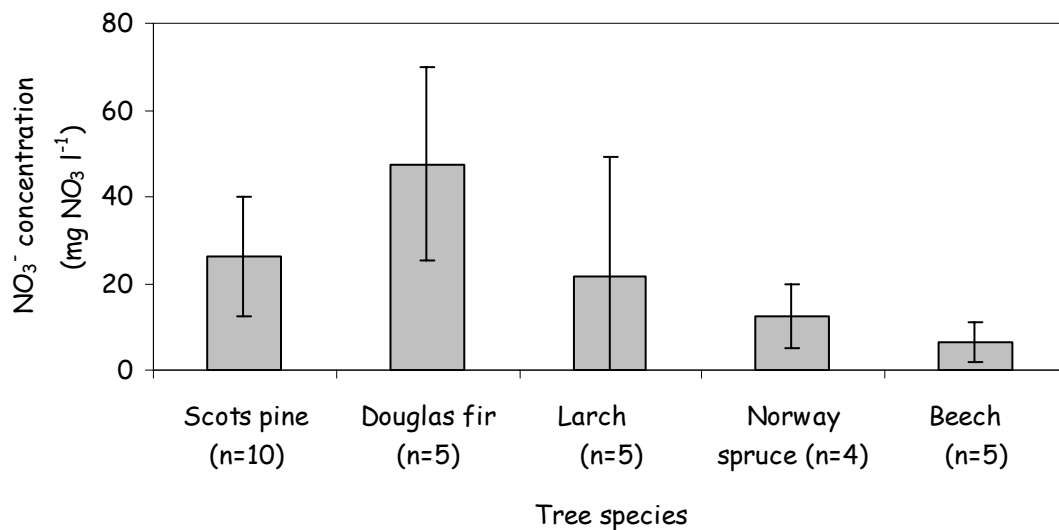

Figure 5.8: Nitrate concentrations in 29 forest stands as a function of tree species. Bars show means with standard deviation.

As with the C/N ratio, the spatial variability of the NO_3^- concentrations was high, which indicates high variability in N status across this relatively small area. The low NO_3^- concentrations in beech stands is a well-known phenomenon. This is often explained by the low decomposition rate of the beech litter leading to thick organic layers and to the production of phenolic substances preventing nitrification (Handley, 1961).

51

5.3 Analysis of management relevant parameters in European data

Evaluating the effects of different management strategies on nitrogen leaching and carbon storage on a European scale is limited by wide variation in factors such as climate, atmospheric deposition, geology, soils, and site history, and thus only broad-scale factors can be considered. We used the IFEF (section 2.1) database to evaluate the effects of tree 'type' (conifer versus hardwood) and tree age on general trends in nitrogen leaching across European forests, from southern Germany to Finland. Norway spruce generally dominates 'conifer' forests, and 'hardwood' forests are nearly always beech, oak, or mixtures of the two species. Tree age is expressed as age of the oldest trees; nearly all of the forests are even-aged or dominated by the oldest age class. We also considered soil (B-horizon) pH and the C/N ratio of the forest floor or OH horizon to shed light on some of the potential processes controlling effects of tree type or age on nitrogen and carbon retention.

Tree type: conifers vs hardwoods
Within the N deposition range in which both forest types occur in significant numbers (10-25 kg N ha^{-1} y^{-1}), conifers leach significantly more nitrogen than hardwoods. Mean carbon storage within this range also reflects these differences. A possible reason for this is that conifers filter more N from dry deposition than hardwoods. This is evident in the database: whereas at the lowest levels of bulk N deposition (0-12 kg N ha^{-1} y^{-1}), throughfall N fluxes are either lower than or equal to bulk N deposition fluxes for both tree types, above this level, conifers show significantly more N in throughfall than in bulk deposition data (roughly a factor 2). Hardwoods show no such throughfall enhancement of N deposition. This is likely due primarily to the different shape of conifers – many thin needles create a highly effective filter for dry deposition – as well as life history differences – unlike hardwoods, most conifers retain their needles throughout the year. It is likely that this throughfall enhancement of nitrogen deposition enriches the nitrogen status of the entire ecosystem.

Tree age
Across Europe, the only relationship between tree age and N leaching is that the very oldest forests (≥ 150 years) all leach low levels of nitrate (<ca 5 kg N ha^{-1} y^{-1}). The reason for this is that the oldest trees are found at the highest and (to a lesser extent) the lowest latitudes in IFEF, i.e. Fenno-Scandia, and high-altitude forests of e.g. southern Germany, where nitrogen pollution is lowest. To remove this confounding effect, sites were divided into low, intermediate and high N deposition ranges (<8, 8-33, and >33 kg N ha^{-1} y^{-1}). There is no relationship between tree age and N leaching for the lowest and highest ranges. At low N deposition, leaching of N is very low for nearly all sites; at high N deposition, all sites leach significant amounts of N regardless of age. However, at intermediate levels of N deposition there is some indication of enhanced N leaching for the oldest conifer stands (>100 years old).

The observation of essentially complete N retention regardless of age class at low N deposition is probably due to overall N limitation of the ecosystem. At high N deposition, overall elevated N leaching, again unrelated to age class, is likely due to N saturation of the vegetation and soil microbes. At intermediate N deposition, age-related N leaching of conifers suggests that the soils may be approaching N saturation, but that younger, aggrading vegetation can take up excess N. The evidence that nitrogen leaching increases with age for conifers older than 100 years receiving intermediate levels of N deposition supports the results of a chronosequence in Wales (see Fig 2.17), but limits its extrapolation to a restricted range of N deposition and where other important factors (e.g. climate, species, long-term site history) are relatively constant.

Thus, the cross-European dataset shows that, unlike hardwoods, conifers receiving intermediate levels of N deposition strongly enrich the nitrogen content of deposition through filtering of dry deposition in needles, and that this may in turn lead to an enhancement of the N status of conifer forests. It also suggests that, again within intermediate levels of N deposition, older conifers, but not hardwoods, show enhanced N leaching with age. Analysis of the relationship between site factors and N leaching from datasets such as these covering a wide area

must be carefully made, since tree species and age tend to be correlated with numerous factors such as region, climate and N deposition. To be more conclusive, such analyses need to be complemented with regional studies such as presented in section 5.2. These regional studies supply data obtained in forest management trials where differences in soil type, climate and N-deposition have minor importance.

Despite the advantage of the regional studies they do not yield consistent results between regions. In the Danish dataset, common beech had the highest soil nitrate concentration whereas it had the lowest one in the Dutch forests. Further investigations of these sites are needed to uncover the reasons for these marked differences. Site specific conditions including former land use strongly influence C sequestration and N-leaching.

6 Integrated discussion

The databases on C and N fluxes and pools in European forests were expanded to cover more than 400 sites and have been explored more thoroughly to create empirical models that predict C accumulation and N retention/nitrate leaching from N input, climate, and ecosystem characteristics. Different ways of validating the empirical models have been applied and improved in comparison with earlier analyses of such databases (e.g. McDonald et al. 2002). For nitrate leaching, analyses showed that there is a threshold N deposition of 8-10 kg N/ha/yr below which almost no leaching occurs. A similar threshold is observed in a compilation of stream and lake data from north-eastern USA (Aber et al. 2003). In the new analysis with more sites the important parameters that determine N leaching and thus N retention are N deposition, the organic layer C/N ratio and annual average temperature. At low C/N ratios (below 23) N input determines N leaching. At higher C/N ratios both N input and temperature are important. These relationships have been consistent throughout the project and have been robust in validation tests.

Based on measured as well as modelled data in our databases for the N balance in European forest stands, soil C sequestration rates were calculated from a simple equation, in which the amount of stored soil N is multiplied by the soil C/N ratio (N-balance approach). The overall mean C sequestration rate was 190 kg C/ha/yr with somewhat higher values for N-poor sites and lower for N rich ones. The estimates are highest for Central and Western Europe (mid latitude range) and decrease toward the north and the south, which follow the general trends in net primary production. An extrapolation using the N balance calculation approach for the ICP Forest Level I plots yielded an overall mean of 70 kg C/ha/yr (de Vries et al. in press). The reason for the lower overall mean in the extrapolation compared to the mean calculated form the database sites is that the database has a geographical bias towards the mid latitude range.

We have also estimated C and N sequestration rates in the organic layer of forests using 'the limit value concept' that is based on data from studies of the decomposition of organic matter. The approach has been successfully validated for both C and N on a few sites where humus accumulation and age are known (Berg 2004; Berg and Dise 2004a, b). Using this method we have calculated C sequestration rates in the organic layer over Europe for c. 30 sites. The pattern agrees well with the pattern obtained by the 'N-balance approach', but the estimates were somewhat higher. This may be expected as the limit-value approach gives the potential sequestration rate.

The limit value method has been applied onto the whole of Sweden in an upscaling study (Akselsson et al., 2005). The range in estimated soil C sequestration rates was 40 to 410 kg C/ha/yr. For Sweden, a calculation based on the 'N-balance approach' was made at the same resolution and this gave a similar range of results as well as a similar pattern in the estimates throughout the country (Akselsson et al. in review). Further, for Sweden both methods agreed very well with country-wide directly measured values (MB Johansson, pers comm.). It appears that for this mainly boreal case study country with mainly little disturbance of the forest soil, our calculations are realistic and correct.

We have calculated present day C sequestration rates in forest soils for Europe and made regional estimates in several ways. The estimates have consistently shown that C sequestration rates in the soil are low: 0-400 kg C/ha/yr. The mean value for sites with data is ca 200 kg C/ha/yr, but when extrapolated without geographical bias the mean value is only 70 kg C/ha/yr. The extrapolation involves assumptions and simplifications that may increase the uncertainty. We think a probable mean for Europe is 100 kg C/ha/yr, when converted to a

European scale this is 13 Mt C/yr. Additionally tree biomass has a net accumulation of ca 70 Mt C/yr. Our estimates are much lower than the estimate of 1400 kg C/ha/yr published earlier by Schulze et al. (2000) based on the difference between litterfall and mineralisation rates in a European transect. When adjusted for losses from harvest operations the estimate was 1100 kg C/ha/yr (Janssens et al. 2003).

For the CNTER estimates of soil C sequestration calculated from N budgets it is assumed that C will not accumulate without N i.e. that soil C/N ratios do not increase. However, with the elevated N deposition in Europe, the soil C/N ratios are more likely to be decreasing, thus C-sequestration rates for Europe probably are even lower than our estimate. Based on a modelling approach, Liski et al. (2002) reached estimates comparable to ours. It is very important to arrive at well-established soil C sequestration numbers, since the size of the possible biological C sink in Europe is uncertain and under debate (Janssens et al. 2003). An additional problem is the leaching of carbon from the mineral soil. This leaching, which is considerably lower than the sequestration of C in the humus layer is still of significance and its causes are still not clear.

The estimated C sequestration rates from the N balance method rely heavily on the how accurate the parameters in the N balance equation (eq. 2.3) are as well as on the assumption that denitrification and N_2 fixation can be neglected. It is difficult to estimate the uncertainty on each of the parameters (deposition, net uptake and leaching) and it is thus even more difficult to estimate the uncertainty on the data for N immobilisation in the soil. This issue requires further attention in the future. If the calculated N immobilisation is 1 kg N/ha/yr off the actual value, the estimate for C sequestration will be 20-30 kg C/ha/yr off the actual rate.

A method to calculate N sequestration rates (based on the limit value method) was developed and validated (Berg & Dise, 2004b). The method is ready to be scaled up to a region and may be used as a tool for boreal coniferous and some temperate forests (common beech, common oak, spruce spp and pine spp). The method needs to be more investigated for further temperate forests and for forest with heavy disturbances. This method may prove to be useful in calculation of critical loads for N, since one of the significant unknowns in these calculations is the long-term accumulation of N in SOM (e.g. Gundersen, 1992).

To gain an insight in C and N interactions long-term field experiments simulating increased or decreased N deposition with stable N isotopes have been re-sampled. This allowed revaluation of the results published by Nadelhoffer et al. (1999) showing that in the short-term (1-3 yrs) soils were the strongest sink for deposition N. Our new results show that after 10 yrs this is still the case.

These experiments allow for thorough testing and validation of a process model that predicts the fate of N in the ecosystem. The model has been applied to one US site (Currie and Nadelhoffer, 2002; Currie et al. 2004). With the model applied to multiple sites we expect to gain new insight and more confidence in its predictions. From detailed knowledge on the fate of N, the model can give estimates of C accumulation rates and fluxes in the systems and can calculate ecosystem C-N stoichiometry that may be useful for upscaling.

A dynamic biogeochemistry model SMART2 was amended and improve to model C and N fluxes and applied at 171 European sites (Mol-Dijkstra et al., submitted). The model could reasonably well simulate the cycling and leaching loss of N. However, the accumulation of C in SOM seemed strongly underestimated in comparison with the estimates based on the two other CNTER methods which were also calculated for each site. The reason for this was that the model (as most other ecosystem models) use an assumption of steady state where decomposition equals incoming litterfall, thus after a certain (site-specific) period soil C sequestration rates become zero. To capture the accumulation of C and N that we observe in forest soils (and predict it by the empirical methods) ecosystem models for C and N cycling may need to be reformulated.

Soils in tree species trials, plantation mosaics and felling experiments were sampled to gain insight in forest management options for improving C sequestration and water protection

(nitrate). These results were compared with effects of management (tree species, age) obtained from the IFEF database. On a cross-European basis, coniferous forests receiving inorganic N in throughfall from 10-25 kg N/ha/yr appear to have a higher N leaching as compared to hardwood forests receiving the same amount of N in deposition. The most likely reason for this is that full-year filtering of atmospheric dry deposition by needles enriches the N status of the site. In the regional dataset (DK and NL) we see the same pattern for conifers versus broadleafs but comparing single tree species showed no consistent effect of N leaching. There is also some evidence that nitrogen leaching increases with age for conifers older than 100 years receiving intermediate levels of N deposition (8-33 kg N/ha/yr). Such age-related enhanced N leaching does not appear to occur in the broadleaf forests studied. The enhanced leaching of N with age observed in this regional analysis supports the results from a chronosequence in Wales, but limits its extrapolation to a restricted area of intermediate N deposition, where other important factors such as climate, species and long-term site history are relatively constant. Differences in N retention between tree types and age classes are reflected in differences in calculated C storage.

7 Conclusions

7.1 Carbon sequestration in European forest soils

In CNTER we studied both carbon (C) and nitrogen (N) cycling in forest ecosystems through several different approaches because C and N processes are connected and because C and N are accumulate bound together in soil organic matter compounds. Our approach has been successful in numerous aspects as listed below in section 7.2.

Most importantly we developed and tested two different methods, namely the limit value method and the N-balance method, to calculate two different carbon sequestration rates, in the first case the potential sequestration in the forest floor only and in the latter a sequestration that considers the C accumulation in the whole system, incl. such influencing factors as direct forest management they will serve different purposes. Although the latter, as expected, gives lower values than the former, they both give carbon sequestration rates that agree very well with well controlled measured values both on single spots and over a larger region.

These methods are relative simple and can be extrapolated to the European scale. Our calculations show that the overall mean C sequestration rate for European forest soils is 100 kg C/ha/yr or less, which on the European scale is 13 Mt C/yr (for comparison the net accumulation in tree biomass is estimated to 70 Mt C/yr). Our esimate is much lower than those published in the literature earlier.

Although our methods have shown promising results, they need to be further developed and validated, especially with respect to the upscaling procedures. Both methods should be used in parallel to calculate carbon sequestration rates. For coniferous forests in the boreal and the northern part of the temperate zone the now obtained values can be considered to be realistic and reliable and both methods appear to function and appear ready to be used. We would also conclude that for deciduous tree species and forests (mainly temperate) more work is needed to determine mechanisms for carbon sequestration. Such work should ideally be based on both methods. Analyses of C sequestration patterns on a regional scale should be compared with data from directly measured values.

7.2 Major achievements in brief

1. Database for C and N pools and fluxes in forests
The database is the most prominent set of data on biogeochemical cycling existing for forest. It will be a highly valuable resource for future scientific study on forest ecosystem properties across Europe.

2. Database on litter fluxes and litter chemistry
This type of data has not been presented before at the European level. The Forest Focus programme is currently considering initiating such monitoring. In this the database is a valuable (historical) record.

3. Indicators and empirical models (mostly N retention)
An improvement over previous attempt to establish empirical models is that the presented models build on (several times) more sites, thus prediction have lower variance. Indicators are of importance in upscaling and in decisions on measuring the right and important parameters in observational studies.

4. Method to estimate soil C sequestration (N balance)
To our knowledge this is the only method suggested that can yield estimates of soil C sequestration that can be upscaled to Europe; an important component of the European C balance. This is of importance in evaluating biological sinks in relation to the Kyoto-protocol.

5. Method to estimate C and N sequestration in organic layers (forest floor)
Another method to esimate (part of) the soil C sink. The rates obtained pose new questions to our understanding of the decomposition process.

6. Experimental data from long-term N experiments
The N manipulation experiments are unique and produce results that receive international attention, especially the tracer studies, where the fate of N have been studied over a decade. Two sites have become a resource for an upcoming EU project NitroEurope (NEU).

7. Improvement of process modelling continue in NEU
First attempts on a model system that can produce predictions of C and N fluxes at a European scale.

8. Experimental data on the impact of management on C and N sequestration
Although not conclusive in all aspects these data and analysis give valuable information on how to manage forest to enhance C and N sequestration.

8 References

Aber, J.D., Nadelhoffer, K.J., Steudler, P. and Melillo, J.M. (1989). Nitrogen saturation in northern forest ecosystems. BioScience, 39, 378-386.

Aber, JD, CL Goodale, SV Ollinger, M-L Smith, AH Magill, ME Martin, RA Hallett, JL Stoddard, & NERC Participants. 2003. Is nitrogen deposition altering the nitrogen status of northeastern forests? BioScience 53(4): 375-389.

Akselsson, C., Berg, B., Meentemeyer, V., & Westling, O. 2005. Carbon sequestration rates in organic layers of boreal and temperate forest soils - Sweden as a case study. Global Ecology and Biogeography 14; 77-84.

Akselsson, C., Berg, B., Gundersen, P., and Westling, O. (in review). Comparing two methods to calculate terrestrial carbon sequestration in boreal forests soils. Global Ecology and Biogeography.

Anonymous 2003. LUSTRA-programmet 2001 och 2002. Årsrapport. Department of Forest Soils. Swedish University of Agricultural Sciences, Uppsala. (In Swedish).

Berg, B. & Dise, N. 2004a. Calculating the long-term stable nitrogen sink in northern European forests? Acta Oecologica 26; 15 - 21.

Berg, B. & Dise, N. 2004b. Validating a new model for N sequestration in forest soil organic matter. Water, Air and Soil Pollution. Focus 4: 343-358.

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

Berg, B. & H. Staaf, 1981. Leaching, accumulation and release of nitrogen in decomposing forest litter. In: Clark, F.E. & T. Rosswall (Eds). Terrestrial Nitrogen Cycles. Ecol. Bull. 33. Stockholm, pp. 163-178.

Berg, B. & McClaugherty, C. 2003. Plant litter. Decomposition, Humus Formation, Carbon Sequestration. Springer Verlag Heidelberg, Berlin. 296 pp

Berg, B. & V. Meentemeyer, 2001. Litter fall in some European coniferous forests as dependent on climate: a synthesis. Can. J. For. Res. 31, 292–301.

Berg, B. 2000. Litter decomposition and organic matter turnover in northern forest soils. Forest Ecology and Management 133; 13-22.

Berg, B. 2004. Sequestration rates for C and N in humus at four N-polluted temperate forest stands. In; Matzner, E, (Ed). Biogeochemistry of Forested Catchments in a Changing Environment. A German Case Study. Ecological Studies Vol 172, pp 361-376. Springer Verlag, Berlin - Heidelberg.

Berg, B. McClaugherty, C. Virzo De Santo, A. & Johnson, D. 2001. Humus build-up in boreal forests - effects of litter fall and its N concentration. Canadian Journal of Forest Research 31, 988-998.

Bird MI; Santrùcková H; Lloyd J; Veenendaal EM (2001): Global soil organic carbon pool. In: Schulze E-D; Harrison Sandy P; Heimann M; Holland EA; Lloyd J; Prentice I Colin; Schimel D (eds): Global biogeochemical cycles in the climate system. Academic Press, San Diego. pp. 185-197

Currie, W.S. and K. Nadelhoffer, 1999, Dynamic redistribution of isotopically labelled cohorts of nitrogen inputs in tow temperate forests. Ecosystems 2, 4-18.

Currie, W. S. and K. J. Nadelhoffer. 2002. The imprint of land use history: Patterns of carbon and nitrogen in downed woody debris at the Harvard Forest. Ecosystems 5(5): 446-460.

Currie, W. S., K. J. Nadelhoffer, and J. D. Aber. 2004. Redistributions of 15N highlight turnover and replenishment of mineral soil organic N as a long-term control on forest C balance. Forest Ecology and Management, 196:109-127.

De Vries, W., Gert Jan Reinds, P. Gundersen, Sterba, H., (in press) Carbon pool changes in European forests and forest soils in view of the impacts of nitrogen deposition Global Change Biology.

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

Dise, N.B. and Wright, R.F. 1995. Nitrogen leaching from european forests in relation to nitrogen deposition. Forest Ecology and Management 71: 153-161.

Dise, N.B., E. Matzner and M. Forsius, 1998a. Evaluation of organic horizon C:N ratio as an indicator of nitrate leaching in conifer forests across Europe. Env. Poll. 102: 453-456

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

B.A. Emmett, D.L. Williams , S.A. Brittain and B. Williams (submitted to Acid Rain 2005) A comparison of N and C cycling in spruce and oak stands.

Eriksson K-E. Blanchette R.A. & Ander P. 1990. Microbial and enzymatic degradation of wood and wood components. Springer, Berlin Heidelberg New York, 407 pp.

Goodale, C.L., Dise, N.B., and Lovett, G.M. Initiation of a trans-Atlantic synthesis of ecosystem responses to nitrogen deposition. Poster accepted at the 7th International Conference on Acid Deposition, Prague, Czech Republic, June 2005.

Gundersen, P., 1992. Mass balance approaches for establishing critical loads for nitrogen in terrestrial ecosystems. In: Grennfelt, P. & Törnelöf, E. (Eds.) "Critical loads for nitrogen". NORD 1992:41, Nordic Council of Ministers, Copenhagen, pp. 55-110.

Gundersen, P., 1995. Nitrogen deposition and leaching in European forests - preliminary results from a data compilation. Water, Air and Soil Pollution, 85:1179-1184.

Gundersen, P., Callesen, I. and de Vries, W., 1998. Nitrate leaching in forest ecosystems is controlled by forest floor C/N ratio. Environmental Pollution 102: 403-407.

Gundersen, P.; K. Raulund-Rasmussen and I. K. Schmidt, 2006. Leaching of nitate from temperate forests – effects of air pollution and forest management. Environmental Reviews, 14: 1-57.

Hatakka, A. 2001. Biodegradation of lignin. In:Biopolymers, Vol. 1. Lignin, Humic substances and Coal (Edited by M. Hofman, and A. Stein). Wiley, Weinheim. pp 129-180.

Hatakka, A., 2001. Biodegradation of lignin. In: Hofrichter, M. & A. Steinbüchel (Eds). Lignin, humic substances and coal, vol. 1. Wiley-VCH, Weinheim, Germany, pp. 129-180.

Holland, E.A., B.H. Braswell, J.F. Lamarque, A.R. Townsend, J. Sulzman, J.F. Müller, F. Dentener, G. Brasseur, H. Levy, II, J.E. Penner & G.J. Roelofs, 1997. Variations in the predicted spatial distribution of atmospheric nitro-gen deposition and their impact on carbon uptake by terrestial ecosystems. J. Geophys. Res. 102 (13), 15849-15866.

Holmberg, M. (2003). Modelling studies on soil-mediated response to acid deposition and climate variability. PhD thesis. Helsinki University of Tehcnology, Systems Analysis Laboratory Research Reports A87. Espoo 2003. 33 p + vii app.

Holmberg, M., Forsius, M., Starr, M. and Huttunen, M. 2006. An application of artificial neural networks to carbon, nitro-gen and phosphorus concentrations in three boreal streams and impacts of climate change. Ecological Modelling (published in online-early).

Holmberg, M., Starr, M., Tamminen, P., Forsius, M., Rita, H., Posch, M. Soil carbon and nitrogen reserves in Finnish upland forest soils: Extrapolation using national forest inventory data and cumulative nitrogen deposition estimates (in prep).

Houghton, R.A., E.A. Davidson & G.M. Woodwell, 1998. Missing sinks, feedbacks, and understanding the role of terrestrial ecosystems in the global carbon balance. Global Biogeochemical Cycles 12 (1), 25-34.

Janssens, I.A., A. Freibauer, P. Ciais, P. Smith, G.-J. Nabuurs, G. Folberth, B. Schlamadinger, R.W.A. Hutjes, R. Ceulemans, E.-D. Schulze, R. Valentini & A.J. Dolman, 2003. Europe's Terrestrial Biosphere Absorbs 7 to 12% of European Anthropogenic CO2 Emissions. Science 300 (5625), 1538-1542.

Kirschbaum, M.U.F. & K.I. Paul, 2002. Modelling C and N dynamics in forest soils with a modified version of the CENTURY model. Soil Biology and Biochemistry 34 (3), 341-354.

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

Kros, H., Mol, J., Bert Jan Groenenberg, Gert Jan Reinds, Gerrit Heil, Willem van Deuren and Wim de Vries (in press). Modelling the afforested system: the soil and water compartment. in: Heil, Muys and Hansen (eds).Environmental effects of afforestation. Kluwer, Dordrecht, Netherlands.

Kros, J., Reinds, G.J., De Vries, W., Latour, J.B. and Bollen, M. (1995) Modelling the response of terrestrial ecosystems to acidification and desiccation scenarios. Water, Air and Soil Pollution 85:1101-1106.

Lek, S., Delacoste, M., Baran, P., Dimopoulos, I., Lauga, J., Aulangier, S. 1996. Application of neural networks to modelling nonlinear relationships in ecology. Ecological Modelling 90:39-52.

Likens GE, Bormann FH, Johnson, NM, Fisher DW and Pierce RS 1970 Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecological Monographs 40:23-47.

Liski, J., D. Perruchoud & T. Karjalainen, 2002. Increasing carbon stocks in the forest soils of western Europe. Forest Ecol. Manage. 169 (1-2), 159-175.

MacDonald, J.A., N.B. Dise, E. Matzner, M. Armbruster, P. Gundersen and M. Forsius, 2002. Nitrogen input together with ecosystem enrichment predict nitrate leaching from European forests. Glob. Change Biol. 8, 1028-1033.

Mol-Dijkstra, J. Gert Jan Reinds, Hans Kros, Bjørn Berg and Wim de Vries (submitted). Prediction of carbon sequestration on intensively monitored plots in relation to N deposition.

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

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

Neal, C., Reynolds, B., Wilkinson, J., Jill, T., Neal, M., Hill, S. and Harrow, M. 1998. The impacts of conifer harvesting on runoff water quality: a regional survey for Wales.

Perruchoud, D., L. Walthert, S. Zimmermann & P. Lüscher, 1999. Contemporary carbon stocks of mineral forest soils in the Swiss Alps. Biogeochem. 50 (2), 111–136.

Persson, T., Rudebeck, A., Jussy, J.H., Colin-Belgrand, M., Priemé, A., Dambrine, E., Karlsson, P.S., & Sjöberg, R.M. (2000) Soil nitrogen turnover - mineralisation, nitrification and denitrification in European forest soils. In: Carbon and nitrogen cycling in European forest ecosystems. Ecological Studies 142 (ed. by E.-D. Schulze), 297-231, Springer Verlag Berlin Heidelberg.

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

Schulze, E.D., L. Högberg, H. van Oene, T. Persson, A.F. Harrison, D. Read, A. Kjøller & G. Matteucci, 2000. Interactions between the carbon and nitrogen cycle and the role of biodiversity: A synopsis of a study along a North-south transect through Europe. In: Schulze, E.D. (Ed). Carbon and Nitrogen Cycling in European Forest Ecosystems. Ecological Studies 142. Springer, Berlin Heidelberg New York, pp. 468-492.

Schöpp, W., Posch, M., Mylona, S. and Johansson, M. 2003. Long-term development of acid deposition (1880-2030) in sensitive freshwater regions in Europe. Hydrol. Earth Syst. Sci., 7,436-446.

Staaf, H. and Berg, B. 1977. A structural and chemical description of litter and humus in the mature Scots pine stand at Ivantjärnsheden. Swed. Con. For. Proj. Int. Rep. No. 65:31 pp.

Tamminen, P. & Starr, M.R. 1990. A survey of forest soil properties related to soil acidification in southern Finland. In: Acidification in Finland (P. Kauppi, K. Kenttämies & P. Anttila. Eds.). Springer-Verlag, Berlin - Heidelberg. pp. 231-247.

Tietema, A., B.A. Emmett, P. Gundersen, O.J. Kjønaas & C. Koopmans, 1998. The fate of ^{15}N-labelled nitrogen deposition in coniferous forest ecosystems. Forest Ecol. Manage. 101 (1-3), 19-27.

Van der Salm, C.; G.J. Reinds, W. de Vries, 2004. Assessment of the Water Balance of European Forests: A Model Study, Water, Air, & Soil Pollution: Focus, 4, 175 – 190.

Vesterdal, L.; E. Ritter, and P. Gundersen, 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecology and Management 169, 137-147

Winjum JK, Dixon RK, Schroeder PE 1992. Estimating global potential of forest and agroforest management-practices to sequester carbon Water Air Soil Pollut. 64 (1-2): 213-227

Wardle, D.A. Zachrisson, O. Hörnberg, G. & Gallet, C. 1997. The influence of island area on ecosystem properties. Science 277, 1296-1299.

Wessel, W.W. and A. Tietema. 2003. The effect of forest management on carbon sequestration in temperate forests: a literature review. ICG Report 03-6, Amsterdam University

Yanai, R.D., Stehman, S.V., Arthur, M.A., Prescott, C.E., Friedland, A.J., Siccama, T.G., Binkley, D. (2003) Detecting Change in Forest Floor Carbon. Soil Sci. Soc. Am. J. 67, 1383-1593.

Forest & Landscape Working Papers