



Modelling soil carbon sequestration of intensively monitored forest plots in Europe by three different approaches

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ABSTRACT

Information on soil carbon sequestration and its interaction with nitrogen availability is rather limited, since soil processes account for the most significant unknowns in the C and N cycles. In this paper we compare three completely different approaches to calculate carbon sequestration in forest soils. The first approach is the limit-value concept, in which the soil carbon accumulation is estimated by multiplying the annual litter fall with the recalcitrant fraction of the decomposing plant litter, which depends on the nitrogen and calcium content in the litter. The second approach is the N-balance method, where carbon sequestration is calculated from the nitrogen retention in the soil multiplied with the present soil C/N ratio in organic layer and mineral topsoil. The third approach is the dynamic SMART2 model in combination with an empirical approach to assess litter fall inputs. The comparison is done by first validating the methods at three chronosequences with measured C pools, two in Denmark and one in Sweden, and then application on 192 intensive monitoring plots located in the Northern and Western part of Europe. Considering all three chronosequences, the N-balance method was generally most in accordance with the C pool measurements, although the SMART2 model was also quite consistent with the measurements at two chronosequences. The limit-value approach generally overestimated the soil carbon sequestration. At the intensive monitoring plots, the limit-value concept calculated the highest carbon sequestration, ranging from 160 to 978 kg ha⁻¹ year⁻¹, followed by the N-balance method which ranged from 0 to 535 kg ha⁻¹ year⁻¹. With SMART2 we calculated the lowest carbon sequestration from -30 to 254 kg ha⁻¹ year⁻¹. All the three approaches found lower carbon sequestration at a latitude from 60 to 70° compared to latitudes from 40 to 50 and from 50 to 60. Considering the validation of the three approaches, the range in results from both the N-balance method and SMART2 model seems most appropriate.

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1. Introduction

Information on soil carbon sequestration and its interaction with nitrogen availability is rather limited, since soil processes account for the most significant unknowns in the C and N cycles. Since forest systems are generally nitrogen limited, nitrogen deposition may increase forest growth and litter fall, which when shed is only partially decomposed on the soil, thus increasing soil carbon sequestration. Recent experimental N fertilization results in Sweden and Finland (Hyvönen et al., 2008), investigating the impact of long-term nitrogen addition on carbon stocks in trees

and soils in Northern Europe, showed an average response near 25 kg C/kg N for trees and 11 kg C/kg N per year for soil.

By far the largest amount of C stored in forests in the northern hemisphere is stored in the soil. While carbon sequestration in trees due to elevated growth is largely a transitory phenomenon, elevated soil carbon sinks could last for a much longer period, since below-ground carbon has much lower turnover time than above-ground carbon or alternatively it is stable. Thus, in the long term the soil is the ultimate sink or source of CO₂ for these ecosystems. Current hypotheses suggest that increased N deposition causes an increased rate of soil organic matter accumulation at least in two ways due to an increased leaf/needle biomass and litter production (Schulze et al., 2000) and a reduced decomposition of organic matter (Berg and Matzner, 1997; Harrison et al., 2000). Bragazza et al. (2006), however, found higher decomposition rates in recently formed peat samples under higher atmospheric N supplies. Soil processes account for the most significant unknowns

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in the C and N cycles. Information on soil carbon sequestration and its interaction with nitrogen availability is, however, rather limited.

The retention or sequestration in forest soils can be derived from repeated soil inventories. In the Swedish Forest Inventory, for example, monitoring of humus C increase is based on measurements of humus depth at an annual interval and the samples are analyzed for C. Long-term trends can thus be followed up. In the Netherlands humus C and soil C in the top 30 cm was measured in the period 1990–2000 with a 5-year interval (Leeters and de Vries, 2001). Results show that the change is hard to detect within a short period of time, considering the large present pools with the exception of the organic layer (see also De Vries et al., 2000). An alternative approach is the measurement of carbon stocks in chronosequences, being a series of forest stands planted in different years on similar soils in the same area (e.g. Vesterdal et al., 2007). One can also estimate the net C sequestration in the soil from direct measurements of the carbon input to the soil by litter fall and root decay and carbon release by net mineralization. Such an approach, which was, e.g. used by Schulze et al. (2000) is also hampered by the fact that the result is based on subtracting large numbers with relative high uncertainties. An empirical approach related to this principle is the limit-value concept, in which a maximum percentage of the litter fall (limit value) is able to decompose. Soil carbon accumulation is thus estimated by multiplying the annual litter fall with the recalcitrant fraction of the decomposing plant litter (Berg et al., 2001; Berg and McLaugherty, 2007).

A completely different approach to assess carbon sequestration is based on the principle that the potential C fixation response to elevated N deposition is restricted by the stoichiometry of C and N of the forest ecosystem compartments (Gundersen et al., 2006). Because of the different C/N ratios, much more N is required to bind up C in soils than in woody biomass. This aspect is the rationale behind using information on the N immobilization in soils, in response to N deposition, to assess the related soil carbon sequestration by multiplying it with C/N ratio by which carbon is sequestered as compared to nitrogen. As a first approach, the present soil C/N ratio can be used. Examples of the use of this so-called N-balance method are given by Nadelhoffer et al. (1999) and De Vries et al. (2006). By combining various tracer experiments, however, Nadelhoffer et al. (1999) showed that only a very small part of the added N (~5%) is stored in stem wood with a high C/N ratio (250–500) whereas most of the deposited N (~70%) is actually stored in soils with a much lower C/N ratio (10–30). De Vries et al. (2006) used this approach to estimate net carbon (C) pool changes and long-term C sequestration in trees and soils at more than 100 Intensively Monitored forest plots (Level II plots) and to scale up results to Europe based on data for more than 6000 forested plots in a systematic 16 km × 16 km grid (level I plots).

Finally, carbon sequestration calculations for soil can be based on dynamic models including the C cycle. Dynamic soil carbon models vary in complexity and in what input information they require (Powlson et al., 1996). Examples of process-oriented soil carbon models are CENTURY (Parton et al., 1987) and RothC (Coleman and Jenkinson, 1996), ANIMO (Wolf et al., 2005) and CESAR (Vleeshouwers and Verhagen, 2002), most of them being developed for agricultural soil, although CENTURY and RothC for arable soils have also been applied to forest soils (e.g. Peng et al., 1998; Falloon et al., 2002). Examples of such models for forest ecosystems are the models SUMO2 in combination with SMART2 (Wamelink et al., 2009), EFIMOD, EFM, FinnFor and Q (Van Oijen et al., 2008) and the Yasso soil model by Liski et al. (2002) in combination with forest stand growth simulation models, such as CO2FIX (Masera et al., 2003) or the region-scale forestry model

EFISCEN (Karjalainen et al., 2002). Such models need to include both carbon inputs to the soil by litter fall and root decay and carbon release by net mineralization to allow the calculation of soil carbon sequestration.

This paper presents a quantification of carbon sequestration in the forest soils, using (i) the limit-value concept (Berg et al., 2001; Berg and McLaugherty, 2007) (ii) the N-balance method (Gundersen et al., 2006) and (iii) the dynamic soil model SMART2 (Kros et al., 1995). The applicability of the various methods was first evaluated by comparing results obtained for soil carbon sequestration rates by these methods with measurements derived from three chronosequences. To gain more insight in soil carbon sequestration on a European wide scale, we then present an estimate of carbon sequestration rates in nearly 200 European forest soils that are part of the Pan-European “Programme for Intensive and Continuous Monitoring of Forest Ecosystems”, carried out since 1994. The aim of this study is to evaluate the consistency of estimates of soil carbon sequestration rates by these three completely different approaches on a European wide scale as a means to gain more insight in the reliability of such estimates.

2. Locations

2.1. Chronosequences in Sweden and Denmark

We made a comparison between the three calculation methods for carbon sequestration (i) the limit-value concept, (ii) the N-balance method and (iii) the SMART2 model using three chronosequences in Sweden and Denmark where carbon stocks in the soil were measured. All stands in a chronosequence are assumed to having been exposed to the same environmental and climatic conditions and have experienced the same pollution regime. Apart from measurements of the C pool in the organic layer and mineral soil (up to 25 cm or the bottom of the AP horizon), the carbon pools in above ground biomass and the carbon input by litter fall were measured (Vesterdal et al., 2007). Furthermore, data were gathered on the water fluxes (Van der Salm et al., 2007b) and on the N deposition and N concentrations in soil solution, thus allowing the calculation of N budgets (Rosenqvist et al., 2007).

The field site of Tönnersjöheden in Sweden (56°40'N, 13°04'E) is an experimental forest area located 20 km from the city of Halmstad (Heil et al., 2007). Afforestation of land taken out of agriculture has been an ongoing large-scale change in land use during the 20th century, particularly over the last 50 years. Soils in the area are represented by till, glaciofluvial material, out-wash sand and peat. The soil type is Arenosol (FAO) developed on sandy glaciofluvial material. The soil contains a large amount of stones. Furthermore, the soils are well drained and have deep groundwater tables. The area has flat topography (altitude 50–65 m a.s.l.), a mean annual temperature between 6.1 and 7.3 °C and an annual mean precipitation of 800–1050 mm. The former arable land has mainly been afforested with Norway spruce. Five stands were selected to represent a chronosequence of 90 years. Three of these stands were located within an area of 1 km × 1 km, and the distance to the other two stands was 4–6 km. It was verified for each stand that the former land use had been agriculture or pasture. No preparatory deep ploughing was performed prior to afforestation.

Vestskoven in Denmark (55°41'N, 12°21'E) is an afforestation area located 15 km west of Copenhagen (Heil et al., 2007). In 1967 it was decided to establish an urban forest, and arable land has successively been bought up for afforestation over the years. The area presents a unique opportunity to study the influence of afforestation within the same soil type. The soils are nutrient-rich and moist Mollic Hapludalfs with a texture of sandy loam

developed from calcareous till deposits. The topography in the area is flat (altitude 20–28 m a.s.l.). The climate is temperate with a mean annual temperature of 7.7 °C and a mean annual precipitation of around 625 mm for the period 1960–1990. Seven stands of Norway spruce (*Picea abies*) and seven stands of common oak (*Quercus robur*) were selected to represent chronosequences of almost 30 years. The stands were located within an area of 1 km × 3 km and it was verified for each stand that the land use had been agriculture (cropland) or horticulture for centuries until afforestation. The trees in all stands were planted following conventional tillage, i.e. soils were not disturbed by preparatory deep plowing. Weeds were controlled mechanically for about 3 years after planting by harrowing a couple of times during summer.

2.2. Intensive monitoring plots in Northwestern and Central Europe

We used selected plots of the “Programme for Intensive and Continuous Monitoring of Forest Ecosystems”, carried out since 1994. This so-called Level II Monitoring Programme includes approximately 860 permanent observation plots in 30 participating countries with data from: (i) mandatory surveys on a 1–10 yearly basis, carried out at all plots (crown condition, at least once a year; chemical composition of live needles and leaves, at least every 2 years; soil chemistry, every 10 years; stem increment, every 5 years) and (ii) optional surveys on a daily to biweekly basis, carried out on a subset of plots (atmospheric deposition, soil solution chemistry and meteorology). In this study, the data were limited to plots which allowed the assessment of N budgets by including atmospheric deposition, in terms of bulk deposition and throughfall, and soil solution chemistry data, mainly in the period 1995–2000. Furthermore, plots were excluded in which (i) soil solution was not sampled with tension lysimeters; (ii) reliable throughfall fluxes could not be calculated and (iii) the soil type did indicate the presence of ground water in the soil profile, since the hydrological simulations were made assuming free drainage (see also De Vries et al., 2007; Van der Salm et al., 2007a). This resulted in a total number of 192 plots as presented in Fig. 1. The map shows that most plots are located in Northern Europe (Scandinavia), Western Europe (the British Isles, the Benelux, and France) and Central Europe (Germany, Austria and Switzerland). Only a few plots are found in the Mediterranean countries (two in Spain and two in Italy). Most common tree species in the plots are Norway spruce (*P. abies*), Scots pine (*Pinus sylvestris*), common beech (*Fagus sylvatica*) and to a less extent common oak (*Q. robur* and *Q. petraea*; Table 1).

3. Methodological aspects

3.1. Model approaches

3.1.1. Limit-value concept

Tree foliar litter and other litter types as well almost never decompose completely but leave recalcitrant remains that vary in size with litter type and a range from ca. 50% to 0% of stable remains have been found. The principle of the limit-value concept (Berg and McClaugherty, 2007) is that soil carbon sequestration can be calculated by multiplying the annual litter fall by a recalcitrant fraction of the decomposing plant litter (e.g. Berg et al., 2001). This so-called recalcitrant fraction, that decomposes very slowly, is based on that the accumulated mass loss of litter in time ultimately reaches a limit value (Berg and Eckbohm, 1991; Coûteaux et al., 1998) that can be calculated. The limit value (LimVal) is defined as the maximum percentage of litter that will be decomposed. The recalcitrant fraction (rf) is thus related to the limit value

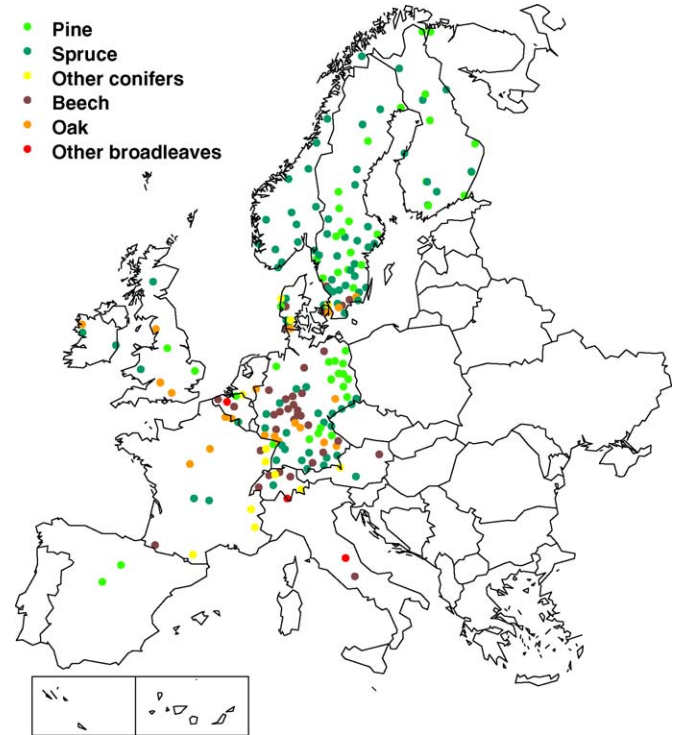


Fig. 1. Locations of the intensive monitoring plots used for calibration and application the dynamic SMART2 model.

according to $rf = 100 - \text{LimVal}/100$. The C sequestration, according to the limit-value concept ($C_{\text{seqLimVal}}$) ($\text{kg ha}^{-1} \text{ year}^{-1}$) is thus calculated as

$$C_{\text{seqLimVal}} = \frac{100 - \text{LimVal}}{100} \times C_{\text{if}} \quad (1)$$

where C_{if} is the amount of carbon in litter fall ($\text{kg ha}^{-1} \text{ year}^{-1}$). The model has been validated against measured values of C accumulation with up to 25 kg C/m^2 during a period of 3000 years (Berg et al., 2001; Berg and Dise, 2004). The stability of the remains are thus valid for periods of millennia. The capacity to store C varies with the litter chemical composition with more N-rich litter types having a larger recalcitrant fraction. The effect of N on the limit value has been observed in the interval from 0.3% to 1.4% N in the newly shed litter (Berg et al., 1999). That range includes most boreal and temperate foliar litter species.

3.1.2. N-balance method

The basic concept of the N-balance method (e.g. De Vries et al., 2006) is that carbon sequestration can be calculated from nitrogen retention in the soils, since carbon and nitrogen accumulation in organic matter occur through the same mechanisms. Calculation of

Table 1
Distribution of plots over combinations of tree species group

Species group	Species	Total
Spruce	<i>Picea abies</i> , <i>Picea sitchensis</i>	83
Scots pine	<i>Pinus sylvestris</i>	45
Other conifers	<i>Abies alba</i> , <i>Pseudotsuga menziesii</i> , <i>Larix decidua</i> , <i>Pinus cembra</i> , <i>Pinus nigra</i>	11
Standard Oak	<i>Quercus petraea</i> , <i>Q. robur</i>	19
Beech	<i>Fagus sylvatica</i>	31
Other broadleaves	<i>Q. cerris</i> , <i>Fraxinus excelsior</i>	3
Total		192

soil carbon sequestration is based on the calculated nitrogen immobilization (sequestration) in the soils, multiplied by the C/N ratio of the forest soils (Gundersen et al., 2006). N immobilization (sequestration) is calculated as the difference of total N deposition minus N uptake and N leaching:

$$N \text{ immobilisation} = N \text{ deposition} - N \text{ leaching} - N \text{ uptake} \quad (2)$$

This approach is based on the assumption that denitrification can be neglected in the organic layer and the mineral topsoil, where both N and C sequestration is assumed to occur. When multiplying the net N immobilization rate with the C/N ratio, the variation of the C/N ratio with the depth of the soil profile was accounted for since there is a large difference between C/N ratio in the organic layer (forest floor) and that in the mineral soil. Based on ^{15}N experiments (Tietema et al., 1998; Nadelhoffer et al., 1999) we modelled the partitioning of N retention between forest floor and mineral soil as a function of the N input and the C/N ratio of the forest floor according to:

$$C\text{seq}_{N \text{ balance}} = N \text{ immobilisation} \left(\text{fret}_{\text{ff}} \frac{C}{N_{\text{ff}}} + (1 - \text{fret}_{\text{ff}}) \frac{C}{N_{\text{ms}}} \right) \quad (3)$$

where C/N_{ff} and C/N_{ms} are the C/N ratios of the forest floor and the mineral soil (up to a depth of 20 cm), and fret_{ff} is the N retention fraction in the forest floor, being the ratio of the N retention in the forest floor and the N retention in the complete soil profile (forest floor and mineral soil to 20 cm). The N retention fraction in the forest floor was calculated as a function of the NH_4 -fraction in the N input and the C/N ratio of the forest floor, as given by De Vries et al. (2006).

3.1.3. SMART2 model

SMART2 (Kros et al., 1995) is a simple dynamic, single-layer soil acidification and nutrient cycling model. It includes the major hydrological and biogeochemical processes in the vegetation, litter and mineral soil. Crucial for the calculation of carbon sequestration in the soil by SMART2 is the litter fall rate and the decomposition rate, since the C sequestration rate is calculated as the difference between these two C fluxes, according to:

$$C\text{seq}_{\text{SMART2}} = C_{\text{if}}(t)(1 - f_{\text{mi},n}) - C_{\text{mi},lt}(t) \quad (4)$$

where C_{if} is the carbon input by litter fall ($\text{kg ha}^{-1} \text{ year}^{-1}$), $f_{\text{mi},n}$ is the mineralization fraction of fresh litter (–), and $C_{\text{mi},lt}$ is the mineralization rate of carbon from the organic layer ($\text{kg ha}^{-1} \text{ year}^{-1}$), t is the actual year. The litter fall rate is not simulated by SMART2 but needs to be assessed on the basis of measured data, empirical relationships or an external forest growth model, such as SUMO2 to which SMART2 has been coupled (Wamelink et al., 2009). In our approach, we used the first two options namely measured data at the chronosequences and empirical relationships at the intensive monitoring plots. Decomposition was modelled as a function of temperature and pH, which is dynamically simulated by SMART2. To assess pH, the model dynamically simulates changes in concentrations of major cations and anions (Al, Ca, Mg, K, Na, NH_4 , NO_3 , SO_4 , HCO_3 and Cl) concentrations in the soil solution. In addition, it simulates changes in solid-phase characteristics connected to the acidification status, i.e. carbonate content, base saturation and amorphous Al precipitates (Kros et al., 1995).

3.1.4. Comparison of principles of calculation of carbon sequestration by the three models

The limit-value concept and the N-balance method are both approaches that are typically applicable for a long-term perspective. They refer to the average carbon sequestration during a period

of decades and results can deviate during shorter time periods due to impacts of, e.g. climate on the carbon input by litter fall and the release by decomposition and thereby on the carbon sequestration. The dynamic modelling approach typically includes those variations and a comparison of results thus requires the calculation of a long-term model average. Another aspect in which SMART2 differs fundamentally from the other two methods is that all plots will in the long term reach a steady state, at which decomposition equals incoming litter fall, resulting in a carbon sequestration of zero when model inputs such as litter fall and growth are constant. In the limit-value concept, the calculated C accumulation never approaches zero as it equals the recalcitrant fraction of the litter fall input. This is also the case when applying the N-balance method as done in this study, where the N immobilization is derived from measured or estimated N deposition, N uptake and N leaching (Eq. (2)). In time this could go to zero when the impact of N deposition on the N saturation in terms of a decline in soil C/N ratio and a related decrease in N immobilization (increase in N leaching) would change. This is likely hardly ever to be reached.

In the limit-value concept and the SMART2 model, the calculated changes in the soil C pool are mainly driven by C pool changes in the vegetation and related changes in litter fall. In the N-balance method, calculated C pool changes are fully driven by the N deposition, together with the soil C/N ratio, which in turn affects N immobilization. The crucial processes for the calculation of C sequestration by the three concepts are summarized in Table 2 and described in the following sections. In the model calculations, the various model inputs used for the limit-value concept and the N-balance method refer to the measuring period. For SMART2, however, it includes a calculation period, starting at the date of afforestation in case of the chronosequences and starting in 1880 at the intensive monitoring plots. Results were evaluated on the present carbon pool in the chronosequences or in the organic layer of the intensive monitoring plots (see Section 4 on model calibration of SMART2). Since the chronosequences are located on former agricultural area, we knew that the initial litter pool was zero at the time of planting. For the intensive monitoring plots, a period of 120 years (1880–2000) was used to give robust model results for the calibrated mineralization rates.

3.2. Litter fall

Litter fall is the driving force of the calculation of carbon sequestration by SMART2 and by the limit-value concept. Litter fall is the largest natural inflow of organic material and nutrients to the forest floor and in most European forests it is dominated by that from the trees. Some large compilations have been published on foliar litter fall. Liu et al. (2004) related litter fall for Eurasia to

Table 2

Crucial processes and concepts for the calculation of carbon sequestration rates by SMART2, the limit value approach and the N-balance method

Processes	Limit value	N-balance	SMART2
C and N in litter fall	x^a	–	x^b
C decomposition	$(x)^a$	–	x
N deposition	–	x	$(x)^c$
N uptake	–	x	$(x)^d$
N immobilization/leaching	–	x	$(x)^d$

^a N contents in litter fall affect the decomposition pattern and the limit value approach, but C decomposition is not included as a process as such.

^b Both C and N fluxes by litter fall are included in SMART2. N fluxes by litter fall influence C sequestration indirectly by affecting the pH, which influences decomposition.

^c N deposition affects N uptake and thereby soil pH and thus C sequestration.

^d N uptake and N immobilization included in SMART2 influence C sequestration through soil pH.

climate factors and found separate patterns among the climatic zones, e.g. in the boreal zone the litter-fall rate was higher in coniferous as compared to deciduous forests. Meentemeyer et al. (1982) used data sets with nearly global coverage and related litter fall with actual evapotranspiration (AET), and other climatic variables. In a regional study for Fennoscandia, Berg and Meentemeyer (2001) related foliar litter fall to AET.

3.2.1. Chronosequences

At the investigated chronosequences, the amount of litter fall was based on measurements during a 2-year period in Denmark and a 1-year period in Sweden (Vesterdal et al., 2007). For the limit-value concept, we directly used these values. For the SMART2 model, we needed litter fall for the period since afforestation. We fitted a logistic growth function, assuming that litter fall has a similar curve as growth (see Eq. (10)) but reaching the maximum amount of leaves three times faster than the maximum stem volume.

3.2.2. Intensive monitoring plots

For the intensive monitoring plots, the maximum amount of litter fall, related to a mature tree species was calculated by regression functions derived from an independent database (Gundersen et al., 2006; Berg and McClaugherty, 2007). This database consisted of 153 observations: 13 for deciduous forest, 67 for pine and 73 for spruce. The latitude in this data set ranges from 37°12' to 69°45'N with an average of 56°35'. The longitude ranges from -9°25' until 30°97'E with an average of 11°80'.

We used the following explanatory variables to calculate the amount of litter fall for pine and spruce: mean temperature for the period May–October (XT), effective temperature sum (ETS), which is the temperature sum of temperatures above 0 °C in the period May–October, precipitation minus potential evapotranspiration (Thorntwaite and Mather, 1957) for the period May–October (PES) and latitude (LAT). For pine and spruce different combinations of the explanatory variables resulted in the best relationships, which had an R^2_{adj} of 0.44 and 0.49, respectively. For reasons of consistency, we decided to use the same model for both forest types and we selected the model with PES, ETS and LAT as explanatory variables, although the R^2_{adj} per tree type was a bit lower with 0.43 for pine and 0.46 for spruce.

$$\text{litter fall} = \text{Constant} + a \times \text{PES} + b \times \text{ETS} + c \times \text{LAT} \quad (5)$$

The coefficients are given in Table 3. Since the amount of data for deciduous trees was too small to find significant relationships, we used literature values for leaf biomass and litter-fall rate as presented in Table 4 (De Vries et al., 1990).

Litter fall was calculated by multiplying leaf biomass and litter fall rate, which is the fraction of leaves that is lost each year. Except for evergreen oak, each deciduous tree drops all leaves each year. The deciduous forest plots mainly occurred in the low-altitude parts of Germany, Belgium and France, with comparable climatic circumstances and thus no strong climate influence on litter fall rates is likely in this region.

Table 3

Values for coefficients in regression functions for Scots pine and other conifers to calculate litter fall on level II plots

Coefficient	Scots pine	Spruce plus other conifers
Constant	2893	16,619
<i>a</i>	-1.26	0.58
<i>b</i>	47.7	-5.4
<i>s</i>	-55.1	-225.0

The letters *a*, *b* and *c* refer to the constants in Eq. (6).

Table 4

Leaf biomass and relative litter fall rates for different broadleaves

Tree type	Leaf biomass (kg ha ⁻¹)	Litter fall rate
Remaining broadleaves (<i>Fraxinus excelsior</i>)	3255	1
Beech (<i>F. sylvatica</i>)	3020	1
Oak (<i>Q. petraea</i> , <i>Q. robur</i>)	3490	1
Oak evergreen (<i>Q. cerris</i>)	2792	0.3

For the development of litter fall on the Level II plots, we used a logistic growth function, in a similar way as the stem growth, according to Eq. (10). The estimation of maximum amount of litter fall for each plot was derived from the regression relation described in Eq. (5). In the limit-value concept we used the 5 years average over the period 1995–2000. In the SMART2 model we used the yearly calculated litter fall during the period 1950–2000.

3.3. Carbon decomposition

The decomposition of organic matter, which is a crucial process in the calculation of carbon sequestration in the limit-value concept and the SMART2 model, is estimated as described below.

3.3.1. Limit-value concept

In this approach, the limit value (LimVal), which is defined as the maximum percentage of litter that will be decomposed, is controlled by factors that determine the degradation and modification of lignin, e.g. the initial litter N and Ca concentrations with a suppressing and stimulating effect on lignin degradation, respectively (e.g. Eriksson et al., 1990; Hatakka, 2001), according to:

$$\text{LimVal} = \text{Constant} - a \times \text{ctN}_{lf} + b \times \text{ctCa}_{lf} \quad (6)$$

where ctN_{lf} is the N content in litter fall (g kg⁻¹) and ctCa_{lf} is the Ca content in litter fall (g kg⁻¹). The coefficients for three tree types are given in Table 5. For plots with 'other conifers' we used the limit value for spruce, whereas for plots with standard oak, beech and other broadleaves we used the limit value for deciduous forests.

3.3.2. SMART2 model

In SMART2, part of the incoming fresh litter is decomposed within the first year after falling, while the remaining is transferred to the organic layer, as given in Eq. (4). The decomposition of this organic layer pool is described by first-order kinetics (Van Veen, 1977):

$$C_{mi,lt} = k_{mi,lt} \times \text{redT}(t) \times \text{redpH}(t) \times \text{AmC}_{lt}(t) \quad (7)$$

where $k_{mi,lt}$ is the mineralization rate constant for litter (year⁻¹) at optimal temperature and pH, AmC_{lt} is the amount of carbon in the organic layer (kg ha⁻¹) and redT and redpH are reduction functions for temperature and pH, respectively. Mineralization of organic matter in the mineral soil layers is not considered in SMART2, except for the mineralization from root necromass, which is fed by root decay. The mineralization rate constant $k_{mi,lt}$ was calibrated to the measured C pool in the organic layer. The initial value was 0.05, but the calibrated values (see Section 4) had a large variation with a median value of 0.044 and with a coefficient of variation of 2.33.

Table 5

Coefficients for calculation of the limit value for spruce, pine and deciduous forest

Vegetation type	Constant	<i>a</i>	<i>B</i>	<i>R</i> ²
Spruce	96.2	3.50	0.42	77.5
Scots pine	65.3	0.22	2.65	75.7
Deciduous forests	94.6	2.10	0.05	26.2

Decomposition is reduced in SMART at low pH in a range between pH 2.5 and 6, according to Kros et al. (1995). Therefore, it is important to have good estimates of pH which is strongly affected by S and N deposition and by weathering of base cations, which is all included in SMART2.

Furthermore a temperature effect was included on the decomposition rates according to Kirschbaum and Paul (2002), which we scaled to 10 °C as reference temperature:

$$rf_T = e^{3.36((T-40)/(T+31.79)-(10-40)/(10+31.79))} \quad (8)$$

where rf_T is the reduction factor for temperature (–) and T is temperature (°C). The scaling to 10 °C as a reference temperature results in a reduction factor equal 1.0 at a temperature of 10 °C.

The N-mineralization rate, which strongly affects N concentrations in the soil solution, is reduced in SMART at low N contents (high C/N ratios) to account for immobilization by microbes according to Janssen (1984). The dissimilation to assimilation ratio DA_{mo} was calibrated to the measured C:N ratio of litter. The default value was 5, the calibrated values had a median of 3.47 and a variation coefficient of 0.68. Even though this aspect is not so relevant for the calculation of the carbon sequestration, it has an influence through its effect on soil pH.

3.4. Tree growth

Both in SMART2 and the N-balance method, tree growth is used for the calculation of soil C sequestration. In the N-balance method, it refers to the measuring period, whereas for SMART2 it includes the whole calculation period starting at the date of afforestation in case of the chronosequences and in 1880 at the intensive monitoring plots.

3.4.1. N-balance method

Growth data at the intensive monitoring plots were calculated from biometric observations, such as diameter at breast height (DBH), height and stem numbers to assess the amounts of above ground biomass in 1995 and 2000, as described in De Vries et al. (2003) and summarized in De Vries et al. (2006).

The N content in stems was assumed to vary with the N deposition, being higher in high deposition areas and lower in low deposition areas. This was based on the hypothesis that at a high N availability, a higher uptake takes place and that the additional N uptake is only partly leading to additional growth (C-pool change) and part is just leading to higher N contents (lower C/N ratios) in stem wood. Actually, the N content was assumed to range from 0.1% (C/N ratio of 500), the constant value used by Nadelhoffer et al. (1999) in areas with a deposition below 1400 mol ha⁻¹ year⁻¹ (approximately 20 kg ha⁻¹ year⁻¹) to 0.2% (C/N ratio of 250) in areas with a deposition above 4000 mol ha⁻¹ year⁻¹ (approximately 60 kg ha⁻¹ year⁻¹). This relation is based on a variation of (stem wood) N contents between 0.1% and 0.2% (at a constant C content of 50%) in comparatively low deposition areas (Scandinavia) to high deposition areas (The Netherlands). In the calculation the following relation was used:

$$ctN_{st} = b + 0.42 \times N_{dep} \times 0.0001 \quad (9)$$

where ctN_{st} is N content in stems (%), b is a constant which is 0.05 for conifers and 0.1 for deciduous and N_{dep} is N deposition (mol ha⁻¹ year⁻¹). A similar relationship was used for the relation between N deposition and the N contents in branches.

3.4.2. SMART2 model

For the application of SMART2 on the intensive monitoring plots between 1880 and 2000 we used logistic growth functions to calculate growth rate with a rotation period of 100 years for all

species except for the 'other broadleaves' for which we used 70 years. A rotation period of 100 years appears to be common practice in Europe for most tree species, except for particular broad-leaf species such as poplar and aspen (EFI, 2002). The calculated amounts of biomass in 1995 and 2000 for the intensive monitoring plots, as described in De Vries et al. (2003), were used for fitting logistic growth curves according to:

$$Am_{st}(t) = \frac{Am_{st,mx}}{1 + \exp(-k_{gl} \times (t - t_{1/2}))} \quad (10)$$

where $Am_{st}(t)$ is the biomass of stems and branches for the simulation year t (kg ha⁻¹) as measured for the years 1995 and 2000, $Am_{st,mx}$ the (maximum) amount of stem biomass for a mature tree (kg ha⁻¹), $t_{1/2}$ the half life-time (year), and k_{gl} is the logistic growth-rate constant (year⁻¹). Values for $Am_{st,mx,act}$, $t_{1/2}$ and k_{gl} were estimated for each site by a non-linear optimization software, described in an unpublished documentation by M. Posch, A. Grübler and N. Nakicenovic (1987). The actual growth rate as function of time, $G_{st}(t)$ (kg ha⁻¹ year⁻¹) was derived through the first derivate of the logistic growth function, according to:

$$G_{st,act}(t) = \frac{k_{gl} \times Am_{st,mx} \exp(-k_{gl}(t - t_{1/2}))}{[1 + \exp(-k_{gl}(t - t_{1/2}))]^2} \quad (11)$$

where $G_{st,act}(t)$ is the actual growth for simulation year t (kg ha⁻¹ year⁻¹). By multiplying this growth with element contents, which are N deposition dependent in the case of N, the relevant annual net uptake is calculated.

3.5. N deposition

3.5.1. N-balance method

For the N-balance method total N deposition refers to the measurement period of the chronosequences (2000–2003) and the intensive monitoring plots (1995–2000). The total N deposition (N_{td}) in the chronosequences was calculated as the sum of N in throughfall (N_{tf}), measured at each plot (Rosenqvist et al., 2007), an estimated N input by stemflow (N_{sf}) and an estimated canopy exchange (uptake) of inorganic nitrogen (N_{ce}). Stemflow flux was estimated from the annual through fall according to Ivens (1990):

$$N_{sf} = N_{tf} \times \frac{\alpha}{(1 - \alpha)} \quad (12)$$

where α is an empirical value. For deciduous forest, α was set to 0.12 independently of age. For coniferous forests, the value of α was calculated as a function of stand age according to Ivens (1990):

$$\alpha = \begin{cases} 0.24 & \text{age} < 20 \\ 0.31 - 0.0034 \times \text{age} & 20 < \text{age} < 90 \\ 0.0 & \text{age} > 90 \end{cases} \quad (13)$$

The canopy exchange of N (N_{ce}) was obtained by a relationship described by De Vries et al. (2001), using empirical relationships between throughfall, stemflow and independently measured total deposition, as reported by Johnson and Lindberg (1992) for 12 sites in the USA:

$$N_{ce} = 0.69 \times (N_{tf} + N_{sf}) + 91.9 \quad \text{for } N_{tf} + N_{sf} < 1000 \text{ mol}_c \text{ ha}^{-1} \text{ year}^{-1} \quad (14)$$

Johnson and Lindberg (1992) made their measurements at sites situated in areas with relatively low N deposition with throughfall and stemflow fluxes of inorganic nitrogen ranging between 100 and 1000 mol_c ha⁻¹ year⁻¹. Eq. (14) therefore can only be applied for this range of inorganic nitrogen fluxes, which appeared to be the case for the investigated chronosequences. In those rare cases where the N deposition exceeded this range an N exchange value of

Table 6

Used data for the calculation of the C sequestration using the limit value concept and the N-balance method

Location	Age (year)	N total deposition (kg ha ⁻¹)	N uptake (kg ha ⁻¹)	N leaching (kg ha ⁻¹)	C/N (–)		Litter fall (kg ha ⁻¹)	Content in litter fall (g kg ⁻¹)	
					ff	ms		N	Ca
Tönners-jöheden	19	19.2	7.6	0.2	27	15	905	11.5	4.15
	30	16.6	9.9	5.5	23	13	1390	11.5	4.15
	63	23.7	4.2	0.1	28	13	1170	11.5	4.15
	74	21.3	4.0	0.2	29	20	1055	11.5	4.15
	92	26.4	4.9	0.0	33	21	1310	11.5	4.15
Vestkoven, Norway spruce	11	19.7	7.6	0.	42.0	12.2.	202	11.9	5.4
	13	22.1	7.6	0.	42.0	12.2.	1219	8.9	4.0
	28	28.0	9.6	0.2	30.8	12.3	1745	10.7	4.8
	32	40.1	9.6	11.8	30.8	12.3	1657	9.8	4.4
Common oak	8	17.6	8.8	3.2	25.8	11.2	1469	17.3	7.6
	13	15.7	8.8	0.1	25.8	11.2	2100	16.1	7.1
	22	18.6	9.7	7.2	35.2	11.7	2392	20.1	8.8
	24	19.8	9.7	11.6	35.2	11.7	2291	17.2	7.6
	31	19.5	10.2	10.7	34.1	10.9	2797	18.4	8.1

780 mol_c ha⁻¹ year⁻¹ was used, being the calculated value for N exchange when the sum of N_{lf} and N_{sf} is 1000 mol_c ha⁻¹ year⁻¹. This occurred at the highest stand age at Tönnersjöheden and at the two oldest stands of the spruce chronosequence in Vestkoven. For the rest of the cases the sum of N_{lf} and N_{sf} ranged between 646 and 947 mol_c ha⁻¹ year⁻¹.

At the intensive monitoring plots, total N deposition, was calculated by a comparable approach but here the canopy exchange was derived by using data on both bulk deposition and throughfall of N (NH₄ and NO₃) and other major elements, such as Cl, Ca, Mg, K and Na, based on a canopy exchange model described by De Vries et al. (2001).

3.5.2. SMART2 model

For the application of the SMART2 model, we used N depositions since the date of afforestation in case of the chronosequences (see age information in Table 6) and since 1880 on the intensive monitoring plots (see before). For these simulations, the trends in SO₂, NO_x and NH₃ deposition were derived using RAINS country emissions (Cofala and Syri, 1998a, b) and transfer matrices derived from the EMEP long-range transport model (Bartnicki et al., 2002) for 50 km × 50 km EMEP grid cells for the period 1960–2000. These trend curves were scaled by the average computed total deposition based on bulk and throughfall measurements as described above for the period 1996–2000 so that the EMEP time series coincide with the plot-specific deposition for that period. Deposition trends between 1880 and 1960 were based on Schöpp et al. (2003), by using one trend line for SO₂, NO_x and NH₃ independent of their location.

3.6. N immobilization and N leaching

Nitrogen immobilization is used in the N-balance method to calculate C sequestration. In this approach, N immobilization is derived from the difference between calculated total N deposition (Section 3.5) and the sum of N uptake (Section 3.4) and N leaching (see also Eq. (2)), as described in the next section. In the SMART2 model N immobilization is also included, but this process does not affect C decomposition and thereby C sequestration.

For the chronosequences, the N leaching was derived by multiplying measured N concentration in the soil solution by the water leaching modelled with SWAP (see Van der Salm et al., 2007a). The N-uptake was calculated by multiplying the measured stem increase by the N content in stems, which was taken from Jacobsen et al. (2002).

For the intensive monitoring plots, N leaching is calculated in the N-balance by multiplying the measured N concentrations in the soil solution with the calculated water leaching by the hydrological model WATBAL (Starr, 1999), which is a monthly water balance model for forest soils based on the following water balance equation:

$$P = ET + R + \Delta SM \quad (15)$$

where P is precipitation (mm day⁻¹), ET is evapotranspiration (mm day⁻¹), R is soil water flux (mm day⁻¹) and ΔSM is change in soil moisture storage (mm day⁻¹). Evapotranspiration is calculated using an estimation of global radiation using a reference crop equation adjusted by a crop (forest stand) coefficient to take into account the greater evapotranspiration from forests. Data on monthly precipitation are measured at all sites, while data on air temperature and radiation are partly measured and partly derived from nearby meteorological station. To calculate R and ΔSM , the available water capacity (AWC) of the soil is needed, which was derived with transfer functions based on soil texture, bulk density and organic matter content. WATBAL has been validated by a comparison with measured soil water fluxes from in situ (zero-tension) gravity lysimeters and soil moisture content measured with TDR probes are made.

4. Calibration of the SMART2 model

Predicting C pools adequately is important to calculate C sequestration. Model inputs such as tree growth and litter fall are of great importance. We have assumed observed growth and litter fall as given values, because the observed values are reliable and therefore we did not calibrate them. Another important model input is the initial C pool of the litter layer. We decided to calibrate this initial pool, since measurements of the initial pools were not available. The main process in the SMART2 model that determines the C sequestration in the soil is mineralization of the litter layer, with litter older than 1 year. Since the mineralization is affected by pH, it is important to model the chemical composition of the soil solution adequately. Therefore we decided to calibrate C/N ratio of the litter layer and Gapon exchange constants, which regulate the exchange of Al, H and base cations between the solid phase (exchange complex) and soil solution.

We did the calibration of the SMART2 model in four successive steps. First the initial litter pool at the beginning of the simulation run (1880) was calibrated to the measured present C pool of litter,

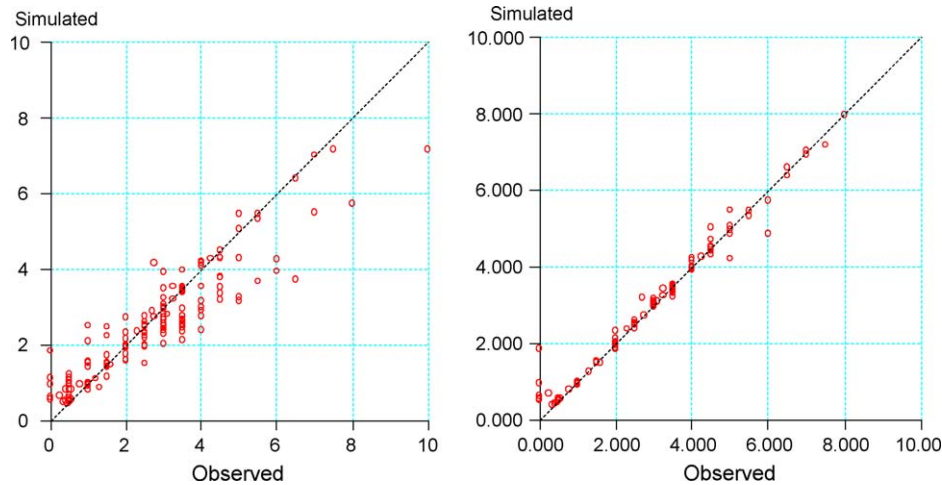


Fig. 2. Simulated against observed C pool of the litter layer (kg m^{-2}), before (left) and after (right) calibration of initial C pool and mineralization rate constant.

considering that the initial litter pool was between zero and twice the measured C pool of litter. In the first step the initial value of the mineralization constant, being 0.05 year^{-1} , was used. The second step was the calibration of the mineralization constant, starting with the last value of initial litter pool. This was only done when the first step did not result in adequate values of the simulated C pools. Since there is a large uncertainty about the mineralization rate constant, we varied it between 0.01 and 10 times the initial value of the mineralization rate constant. An example of the result of this calibration is given in Fig. 2 for the intensive monitoring plots.

The C pool of old litter was simulated well, which means that the calibration of initial litter pool and or mineralization rate constant was successful. In Norway, the C pool of litter was not measured and we assumed a C pool of 3 kg m^{-2} for Scots pine and 3.5 kg m^{-2} for Norway spruce, the median values for the Swedish plots. It was necessary to calibrate the mineralization constant for most plots.

The third step was the calibration of the assimilation/dissimilation ratio of the decomposing microbes to the measured C:N ratio in litter. We varied the assimilation/dissimilation ratio between 1.5 and 10 using an initial value of 5. The result of this calibration for the intensive monitoring plots is given in Fig. 3. The C:N ratio in old litter was simulated well too, although larger deviations were found. The range of dissimilation/assimilation

ratio (D/A) was not always broad enough to meet the criterion. When the C:N ratio of litter was not reached the latest value of D/A was taken.

Finally, the Gapon exchange constants were calibrated to the measured base saturation, which is strongly related to the pH. The simulated pH before and after calibration did not differ much for the intensive monitoring plots (Fig. 4).

In general, the results agreed quite well with the observed values, especially between the values 4 and 5. Divergences had different causes. First, measurements were not always from the same depth as the model output. Second, some plots seem to have SO_4 weathering in the soil, which is not in the SMART2 model. At these plots the simulated pH was too high, whereas the SO_4 concentrations were too low. The plots with assumed SO_4 weathering were skipped.

We tested this procedure at the described chronosequences where, in contrast to the intensive monitoring plots, carbon sequestration, in a certain sense, is measured several times over a longer period. We did the calibration at the chronosequence as if we had just one measurement in time and chose therefore the measurements of the middle stand age of each chronosequence, which were the 1967 measurements for Sweden, 1991 for the spruce site in Denmark and 1992 for the oak site in Denmark. Afterwards we compared the simulated carbon sequestration for

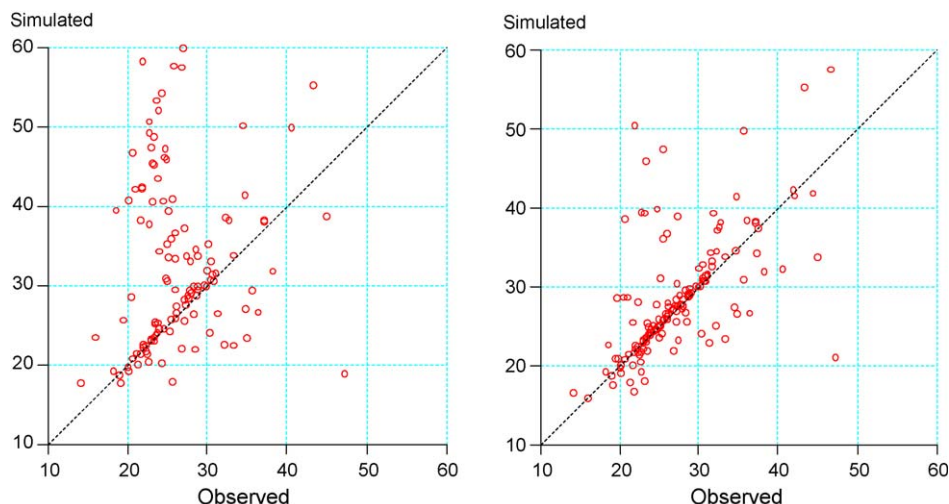


Fig. 3. Simulated against observed C:N ratio of the litter layer, before (left) and after (right) calibration of the dissimilation-to-assimilation ratio.

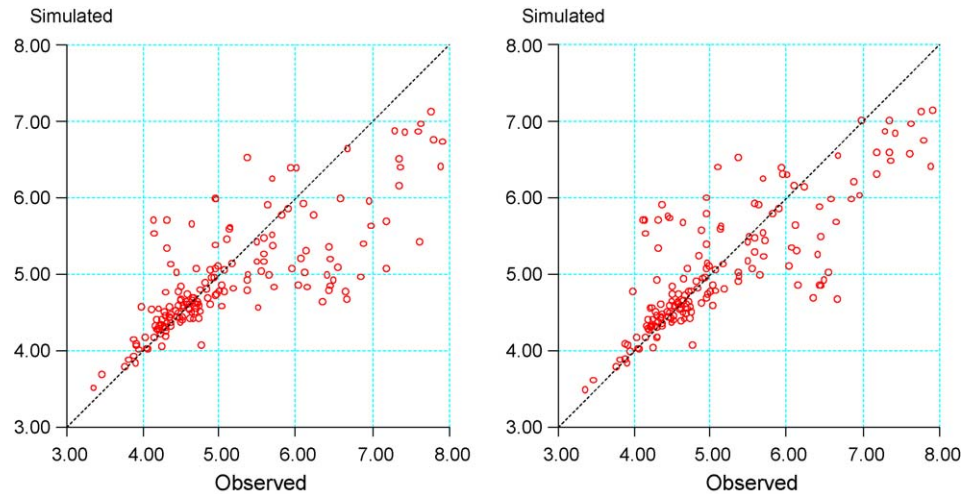


Fig. 4. Simulated against observed pH, before (left) and after (right) calibration.

different times with the measured carbon sequestration at the corresponding age.

Since the chronosequences are located on former arable land, we knew that the initial litter pool was zero so that we did not need to calibrate the initial litter pool. We started the procedure with the calibration of the mineralization rate constant and then continued the calibration with the assimilation/dissimilation ratio and finally the Gapon exchange constants. The calibrated mineralization constants were 0.0005 year^{-1} for both chronosequences planted with Norway spruce and 0.5 year^{-1} for the one planted with common oak.

5. Carbon sequestration estimates using SMART 2, the limit-value approach and the N-balance method

5.1. Results obtained for the chronosequences

We compared the three described methods on the chronosequences using as much as possible measured data from these

locations (Table 6). For the limit-value concept, we used the measured N and Ca contents in the litter fall and the measured litter fall in each stadium. For the N-balance method, we used the measured throughfall, stem wood increase and N concentration in the soil solution, to assess total N deposition N uptake and N leaching as described before. For the SMART2 model we used these data to derive those values since the time of afforestation (see before). The measured C sequestration per year (Table 7) was determined for each stadium by dividing the total C pool of litter by the age of the site (Vesterdal et al., 2007). The results of the three methods were most consistent at Tönnersjöheden, the oldest chronosequence. The results of the three calculation methods agreed quite well with the measured C sequestration. At the older stadiums, all methods underestimated the carbon sequestration. At the two Danish chronosequences, however, the results diverged considerably. At the Danish location planted with Norway spruce, both the limit value method and the N-balance method overestimated the C sequestration, whereas SMART2 underestimated it. At the oak chronosequence, the limit value method calculated a

Table 7
Measured and calculated carbon sequestration rates ($\text{kg ha}^{-1} \text{ year}^{-1}$) using the limit-value concept, the N-balance method and SMART2 at three chronosequences in Sweden and Denmark

Location	Species	Age (years)	Carbon sequestration rate ($\text{kg ha}^{-1} \text{ year}^{-1}$)			
			Measured	Limit value	N-balance	SMART2
Tönnersjöheden	Norway spruce	19	258	381	334	247
		30	327	586	207	290
		63	352	493	319	412
		74	563	445	301	444
		92	609	552	285	480
Average			422	491	289	375
Vestkoven	Norway spruce	11	347	87	397	81
		13	197	406	463	87
		28	320	685	450	131
		32	306	601	460	140
Average			293	445	443	110
	Common oak	8	55	607	115	97
		13	112	816	136	134
		22	116	1127	47	92
		24	63	942	-43	82
		31	71	1220	-37	56
Average			83	943	44	92

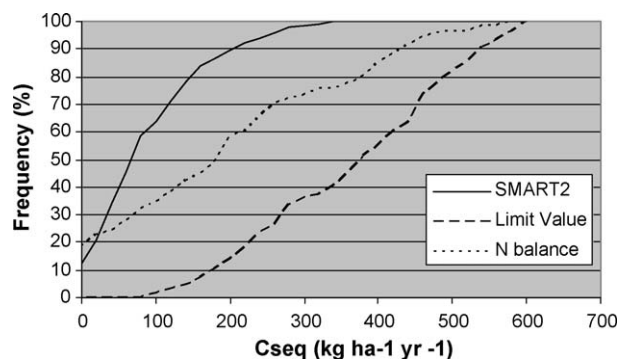


Fig. 5. Cumulative frequency distribution of the average C sequestration rate from 1950 to 2000, calculated with SMART2, the limit value concept and the N-balance method ($\text{kg m}^{-2} \text{ year}^{-1}$).

large overestimation of the sequestration. With the N-balance method, the overestimation was smaller. With the N-balance method we calculated two negative values at the two oldest stadiums due to the relative high N leaching at those locations (Table 6). The SMART2 model results agreed very well with the measurements, which was also true for the trend in time of the sequestration, with a decrease of C sequestration at higher age.

The results over all these chronosequences showed that the limit value approach leads to the largest calculated C sequestration, whereas the SMART2 model calculated the smallest C sequestration. Considering all three chronosequences, the N-balance method was generally most in accordance with the C pool measurements, although the SMART model was also quite consistent with the measurements at two chronosequences. The limit value approach generally overestimated the soil carbon sequestration.

5.2. Results obtained for the intensive monitoring plots

The modelled C-sequestration rates in the soil has a large variation between plots but also between the various approaches to assess the soil carbon sequestration rate, as shown in Fig. 5.

With the limit-value concept, the calculated carbon sequestration rates ranged from 160 to 978 $\text{kg ha}^{-1} \text{ year}^{-1}$ with a median value of 446 $\text{kg C ha}^{-1} \text{ year}^{-1}$ and with the N-balance method, the results ranged from 0 to 535 $\text{kg ha}^{-1} \text{ year}^{-1}$ with a median of 184 $\text{kg ha}^{-1} \text{ year}^{-1}$. With SMART2, we calculated a carbon sequestration between -30 and 254 $\text{kg ha}^{-1} \text{ year}^{-1}$ with a median of 64 $\text{kg ha}^{-1} \text{ year}^{-1}$. Low predicted values by SMART2 are caused by the dynamic of the system. After a clear cut, which happens each 100 year in the SMART2 simulation, a lot of organic matter is decomposed, resulting in negative carbon sequestration. Taking the average over 50 years with a clear cut in that period leads to low average carbon sequestration values.

Table 8

Calculated carbon sequestration ($\text{kg ha}^{-1} \text{ year}^{-1}$) with the limit value concept, N-balance method and SMART2 as a function of latitude

Latitude	Carbon sequestration ($\text{kg ha}^{-1} \text{ year}^{-1}$)		
	Limit-value concept	N-balance method	SMART2 model
40–50	512 (153–996)	233 (0–532)	64 (–9–255)
50–60	455 (182–974)	193 (0–634)	70 (–37–209)
60–70	291 (121–871)	54 (0–150)	39 (–11–258)
All	446 (160–978)	184 (0–535)	64 (–30–254)

Median values are given with 5 and 95 percentile between brackets.

For all three methods, the soil C sequestration rate was generally calculated to be lower in Northern Europe (latitude >60) than in Central and Southern in Europe, as seen in Table 8. In the N-balance method, which is directly related to N deposition, there was on average a fourfold difference between Northern Europe and the rest of Europe, whereas the difference was approximately twofold for the limit-value concept and the SMART2 model. An overview of the geographic differences is given in Fig. 6.

The large difference between the results of SMART2 and the limit-value approach is caused by differences in the decomposition concept. In the limit-value concept, there is a recalcitrant fraction of the fresh litter fall that is not decomposed, which varied between 12% and 57%. In the SMART2 model, in principle all incoming litter fall is ultimately decomposed. This means that it is possible to reach a steady state where decomposition equals the litter fall, resulting in a C sequestration of zero. The N-balance method is fundamentally different from the other two methods in that it focuses on N deposition impacts.

6. Discussion and conclusions

6.1. Plausibility of the derived carbon sequestration rates

The calculated C sequestration rates in the intensively monitored plots show large variations between the three described methods. The plausibility of the results can be evaluated against other literature results. Liski et al. (2002), using a modified version of the ForClim-D model (Perruchoud et al., 2000), later called the Yasso model (Liski et al., 2005), found an average rate in 1990 of 190 $\text{kg ha}^{-1} \text{ year}^{-1}$ and of 305 $\text{kg ha}^{-1} \text{ year}^{-1}$ in 2040, with a large variability, for the EU countries including Norway and Switzerland. Nabuurs and Schelhaas (2002) calculated a net carbon sequestration in soil for 16 typical forest types across Europe. The advancing mean of the net C sink of all forests was calculated to equal 110 $\text{kg ha}^{-1} \text{ year}^{-1}$. Results of the N-balance method as applied by Nadelhoffer et al. (1999) and De Vries et al. (2006) lead to an average increase of carbon sequestration of 21 and 15 $\text{kg ha}^{-1} \text{ year}^{-1}$, respectively per kg N deposition. Hyvönen et al. (2008) investigated the impact of long-term nitrogen addition on carbon stocks in trees and soils in northern Europe (Sweden and Finland). They quantified the effects of fertilizer N on C stocks in trees and soils (organic layer +0–10 cm mineral soil) by analyzing data from 15 long-term (14–30 years) experiments in *P. abies* and *P. sylvestris* stands in Sweden and Finland. Addition of a cumulative amount of N of 600–1800 kg N ha^{-1} resulted in a mean increase of 11 kg C/kg N in soil, respectively. These results of 11–25 kg C/kg N can be used to derive a range in carbon sequestration rates by multiplying them with the additional N input on the intensive monitoring plots, compared to a background N deposition of 3 $\text{kg N ha}^{-1} \text{ year}^{-1}$. The additional N input above 3 $\text{kg N ha}^{-1} \text{ year}^{-1}$ at the investigated plots ranged from 0.2 to 36 $\text{kg N ha}^{-1} \text{ year}^{-1}$, with a median of 13 $\text{kg N ha}^{-1} \text{ year}^{-1}$. On average, this implies a range in carbon sequestration from 142 to 322 $\text{kg ha}^{-1} \text{ year}^{-1}$.

It seems in general that the results obtained with the N-balance method at the intensive monitoring plots are most in line with literature data. The limit-value concept generally calculates too high values, whereas the calculated soil carbon sequestration rates by SMART2 seem generally too low. The results of the application of the three methods to the chronosequences also showed that, in general, the N-balance method corresponded best with the measured C sequestration. The limit value overestimated the C sequestration for some cases, whereas the SMART2 model sometimes underestimated

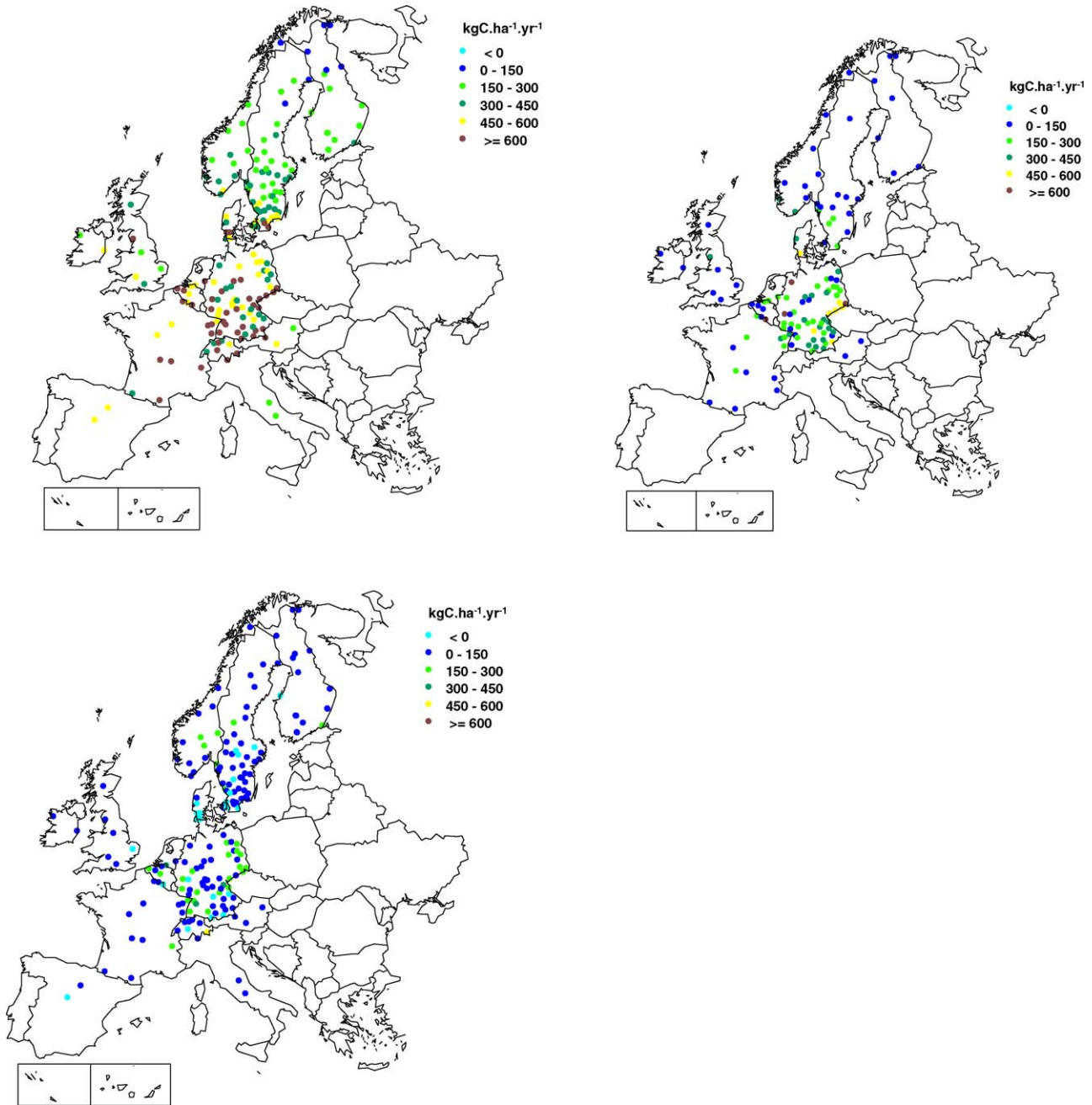


Fig. 6. Carbon sequestration rates ($\text{kg ha}^{-1} \text{ year}^{-1}$) calculated with the limit value concept (top left), the N-balance method (top right) and SMART2 (below) at the investigated 192 intensive monitoring plots.

the C sequestration. There is thus a need for improvement, as summarized below.

6.2. Uncertainties in litter fall in the limit value and SMART 2 model approach

Both for SMART2 and the limit-value concept, litter fall is a crucial input and an uncertainty in litter fall directly leads to uncertainty in C sequestration rates calculated with these methods. 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. An alternative is the use of SUMO2, which

calculates biomass production dependent on N availability (Wamelink et al., 2009).

6.3. Uncertainties in the carbon to nitrogen sequestration ratio in the N-balance method

The reliability of the soil carbon sequestration rates by the N-balance method relies on the accuracy of: (i) N deposition and N leaching, which are derived from measurements and water flux and (ii) present soil C/N ratios as a proxy for the ratio of C to N accumulation or sequestration, C/N_{seq} . The latter assumption is mainly determining the uncertainty in estimated carbon sequestration. Evidence that carbon accumulation in response to N

Table 9

Estimated soil carbon sequestration per kg nitrogen addition and soil C/N ratios at the Ruabon heathland N manipulation site (after Evans et al., 2006)

N input in 11 year (kg/ha)	Soil C pool (kg/ha)	Change in soil C pool compared to ambient (kg/ha)	C/N _{seq} (kg C/kg N)	C/N _{soil} (kg C/kg N)
0	105,360	–	–	33.4
440	120,360	15,000	34.1	31.9
880	125,640	20,280	23.0	31.2
1320	131,880	26,520	20.1	30.7

addition approximates the C/N ratio of the soil has been presented by Evans et al. (2006) for two heathland N manipulation sites. The manipulation experiment, established in 1989, includes a control treatment plus three N addition treatments of 40, 80 and 120 kg N ha⁻¹ year⁻¹, added monthly as finely sprinkled NH₄NO₃ solution. Measurements of soil C pools allowed to calculate changes in the C pools at given N inputs during the 11-year experiment, thus allowing to calculate the C/N sequestration ratio, being close to the C/N ratios of the soil, as presented in Table 9.

In applying the N-balance method, one should be aware that the N immobilization rate may decrease with time due to N saturation, which is reflected in a lowering of the C/N constant, which is an important indicator for N leaching and N accumulation. There is, however, no clear evidence for an impact of N deposition on the soil C/N ratio. Dise et al. (1998a, b) found significant relationships between the input flux of inorganic N and the N concentration of the forest floor, but no relationship between N input and the C/N ratio, supporting the hypothesis of carbon accumulation due to N fertilization on a regional scale. Furthermore, there are no clear long-term rising trends in inorganic N leaching (e.g. Monteith et al., 2000; Aber et al., 2002) which may also partly be caused by the limited effect of N deposition on the C/N ratio. When N deposition not only affects the N accumulation in litter but also the C accumulation from increased litter production and/or suppressed litter decomposition, such a limited impact is to be expected (Evans et al., 2006).

6.4. Uncertainties in the SMART2 model simulation

The reason why SMART2 seems to underestimate carbon sequestration could be that, the model underestimates the period before a steady state is reached. For example, Masera et al. (2003) using the YASSO model by Liski et al. (2002) linked to the forest stand growth simulation model CO2FIX, calculated a steady state after 200 years for a fir-beech stand and after 300 years for a spruce stand. SMART2 however, always reached steady state within 100 years. This aspect can be improved by including the effect of a slower decomposable pool in SMART2. In the current version of SMART2, the decomposition is furthermore only affected by pH and temperature. The effects of soil moisture content and nitrogen content in litter on decomposition are not included in the model, partly because the impacts are not trivial. For example, in many experiments, a positive relation was found between N-content in fresh litter and decomposition (Vesterdal, 1999; Hobbie, 2000; Silver and Miya, 2001; Sariyildiz, 2003; Xu and Hirata, 2005), but there are also examples of experiments where the effect of N content on decomposition was not found (Melillo et al., 1982; Murphy et al., 1998; Fourqurean and Schrlau, 2003).

In summary, SMART2 can be improved by including the effect of a slower decomposable pool, with a recalcitrant part, and possibly by accounting for the impact of moisture and N availability. Besides, effects of management during land-use history should be taken into account, since management aspects like thinning and clear-cutting, affect the carbon-pool build-up rates, and these were not included in the simulations.

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References

- Aber, J.D., Ollinger, S.V., Driscoll, C.T., Likens, G.E., Holmes, R.T., Freuder, R.J., Goodale, C.L., 2002. Inorganic nitrogen losses from a forested ecosystem in response to physical, chemical, biotic, and climatic perturbations. *Ecosystems* 5, 648–658.
- Bartnicki, J., Olendrzynski, K., Jonson, J.E., Berge, E., Unger, S., 2002. Description of the Eulerian acid deposition model. EMEP/MS-CW Report 1/98. EMEP/MS-CW.
- Berg, B., Eckbohm, G., 1991. Litter mass-loss rates and decomposition patterns in some needle and leaf litter types. Longterm decomposition in a Scots pine forest VII. *Canadian Journal of Botany* 69, 1449–1456.
- Berg, B., Matzner, E., 1997. Effect of N deposition on decomposition of plant litter and soil organic matter in forest systems. *Environmental Reviews* 5, 1–25.
- Berg, B., Meentemeyer, V., 2001. Litter fall in some European coniferous forests as dependent on climate: a synthesis. *Canadian Journal of Forest Research* 31, 292–301.
- Berg, B., Dise, N., 2004. Validating a new model for N sequestration in forest soil organic matter. *Water, Air, and Soil Pollution: Focus* 4, 343–358.
- Berg, B., McClaugherty, C., 2007. *Plant Litter: Decomposition, Humus Formation, Carbon Sequestration*, Second edition. Springer-Verlag, Heidelberg, Berlin.
- Berg, B., Laskowski, R., Virzo De Santo, A., 1999. Estimated nitrogen concentrations in humus based on initial nitrogen concentrations in foliar litter: a synthesis. XII. Long-term decomposition in a Scots pine forest. *Canadian Journal of Botany* 77, 1712–1722.
- Berg, B., McClaugherty, C., Virzo De Santo, A., Johnson, D., 2001. Humus buildup in boreal forests: effects of litter fall and its N concentration. *Canadian Journal of Forest Research* 31, 988–998.
- Bragazza, L., Freeman, C., Jones, T., Rydin, H., Limpens, J., Fenner, N., Ellis, T., Gerdol, R., Hajek, M., Hajek, T., Iacumin, P., Kutnar, L., Tahvanainen, T., Toberman, H., 2006. Atmospheric nitrogen deposition promotes carbon loss from peat bogs. *Proceedings of the National Academy of Sciences* 103, 19386–19389.
- Cofala, J., Syri, S., 1998a. Nitrogen oxides emissions, abatement technologies and related costs for Europe in the RAINS model database. International Institute for Applied Systems Analysis. IIASA Interim Report IR-98-88, Laxenburg, Austria.
- Cofala, J., Syri, S., 1998b. Sulfur emissions, abatement technologies and related costs for Europe in the RAINS model database. International Institute for Applied Systems Analysis. IIASA Interim Report IR-98-035, Laxenburg, Austria (update October 19, 1998).
- Coleman, K., Jenkinson, D.S., 1996. RothC-26.3—a model for the turnover of carbon in soil. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), *Evaluation of Soil Organic Matter Models, Using Existing Long-Term Datasets*. Springer-Verlag, Heidelberg, pp. 237–246.
- Coûteaux, M.M., McTiernan, K.B., Berg, B., Szuberla, D., Dardenne, P., Bottner, P., 1998. Chemical composition and carbon mineralisation potential of scots pine needles at different stages of decomposition. *Soil Biology and Biochemistry* 30, 583–595.
- De Vries, W., Hol, A., Tjalma, S., Voogd, J.C., 1990. Literatuurstudie naar voorraden en verblijftijden van elementen in een boscysteem (in Dutch). Staring Centrum, Wageningen. Rapport 94.
- De Vries, W., Reinds, G.J., van Kerkvoorde, M.A., Hendriks, C.M.A., Leeters, E.E.J.M., Gross, C.P., Voogd, J.C.H., Vel, E.M., 2000. *Intensive Monitoring of Forest Ecosystems in Europe*. Technical Report 2000. UN/ECE, EC, Forest Intensive Monitoring Coordinating Institute.
- De Vries, W., Reinds, G.J., van der Salm, C., Draaijers, G.P.J., Bleeker, A., Erisman, J.W., Auee, J., Gundersen, P., Kristensen, H.L., van Dobben, H., de Zwart, D., Derome, J., Voogd, J.C.H., Vel, E.M., 2001. *Intensive Monitoring of Forest Ecosystems in*

- Europe. Technical Report 2001. UN/ECE, EC, Forest Intensive Monitoring Coordinating Institute, Geneva, Brussels.
- De Vries, W., Reinds, G.J., Posch, M., Sanz, M.J., Krause, G.H.M., Calatayud, V., Renaud, J.P., Dupouey, J.L., Sterba, H., Gundersen, P., Voogd, J.C.H., Vel, E.M., 2003. Intensive Monitoring of Forest Ecosystems in Europe. Technical Report 2003. UN/ECE and EC, Forest Intensive Monitoring Coordinating Institute, Geneva, Brussels.
- De Vries, W., Reinds, G.J., Gundersen, P., Sterba, H., 2006. The impact of nitrogen deposition on carbon sequestration in European forests and forest soils. *Global Change Biology* 12, 1151–1173.
- De Vries, W., van der Salm, C., Reinds, G.J., Erismann, J.W., 2007. Element fluxes through European forest ecosystems and their relationships with stand and site characteristics. *Environmental Pollution* 148, 501–513.
- Dise, N.B., Matzner, E., Gundersen, P., 1998a. Synthesis of nitrogen pools and fluxes from European forest ecosystems. *Water, Air, and Soil Pollution* 105, 143–154.
- Dise, N.B., Matzner, E., Forsius, M., 1998b. Evaluation of organic horizon C:N ratio as an indicator of nitrate leaching in conifer forests across Europe. *Environmental Pollution* 102, 453–456.
- EFL, 2002. Forest management guidelines and practices in Finland, Sweden and Norway. EFL internal report 11.
- Eriksson, K.-E.L., Blanchette, R.A., Ander, P., 1990. *Microbial and Enzymatic Degradation of Wood and Wood Components*. Springer-Verlag, Berlin, Germany.
- Evans, C.D., Caporn, S.J.M., Carroll, J.A., Pilkington, M.G., Wilson, D.B., Ray, N., Cresswell, N., 2006. Modelling nitrogen saturation and carbon accumulation in heathland soils under elevated nitrogen deposition. *Environmental Pollution* 143, 468–478.
- Falloon, P., Smith, P., Szabó, J., Pásztor, L., 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. *Soil Use and Management* 18, 164–174.
- Fourqurean, J.W., Schrlau, J.E., 2003. Changes in nutrient content and stable isotope ratios of C and N during decomposition of seagrasses and mangrove leaves along a nutrient availability gradient in Florida Bay, USA. *Chemistry and Ecology* 19, 373–390.
- Gundersen, P., Berg, B., Currie, W.S., Dise, N.B., Emmett, B.A., Gauci, V., Holmberg, M., Kjonaas, O.J., Mol-Dijkstra, J., van der Salm, C., Schmidt, I.K., Tietema, A., Wessel, W.W., Vestergaard, L.S., Akselsson, C., de Vries, W., Forsius, M., Kros, H., Matzner, E., Moldan, F., Nadelhoffer, K.J., Nilsson, L.-O., Reinds, G.J., Rosengren, U., Stuaes, A.O., Wright, R.F., 2006. Carbon–nitrogen interactions in forest ecosystems—Final report. Danish Centre for Forest, Landscape and Planning, KVL Forest & Landscape Working Papers 17.
- Harrison, A.F., Harkness, D.D., Rowland, A.P., Garnett, J.S., Bacon, P.J., 2000. Annual carbon and nitrogen fluxes in soils along the European forest transect determined using ¹⁴C-bomb. In: Schulze, E.D. (Ed.), *Carbon and Nitrogen Cycling in European Forest Ecosystems*. Ecological Studies, vol. 142. Springer, Berlin, Heidelberg, New York, pp. 237–256.
- Hatakka, A., 2001. Biodegradation of lignin. In: Hofrichter, M., Steinbüchel, A. (Eds.), *Lignin, Humic Substances and Coal*, vol. 1. Wiley-VCH, Weinheim, Germany, pp. 129–180.
- Heil, G.W., Hansen, K., Muys, B., van Orshoven, J., 2007. Introduction: demand for afforestation management in North-Western Europe. In: Heil, G.W., Muys, B., Hansen, K. (Eds.), *Environmental Effects of Afforestation in North-Western Europe*. From Field Observations to Decision Support. Plant and Vegetation, vol. 1. Springer Series, pp. 79–108.
- Hobbie, S.E., 2000. Interactions between litter lignin and soil nitrogen availability during leaf litter decomposition in a Hawaiian montane forest. *Ecosystems* 3, 484–494.
- Hyvönen, R., Persson, T., Andersson, S., Olsson, B., Ågren, G.I., Linder, S., 2008. Impact of long-term nitrogen addition on carbon stocks in trees and soils in northern Europe. *Biogeochemistry* 89, 121–137.
- Ivens, W.P.M.F., 1990. Atmospheric deposition onto forests: an analysis of the deposition variability by means of throughfall measurements. Ph.D. Thesis. University of Utrecht, Utrecht, The Netherlands.
- Jacobsen, C., Rademacher, P., Meesenburg, H., Meiwes, K.J., 2002. *Gehalte chemischer Elemente in Baumkompartimenten (in German)*. Niedersächsische Forstliche Versuchsanstalt, Göttingen.
- Janssen, B.H., 1984. A simple method for calculating decomposition and accumulation of 'young' soil organic matter. *Plant and Soil* 76, 297–304.
- Johnson, D.W., Lindberg, S.E., 1992. *Atmospheric Deposition and Forest Nutrient Cycling*. Ecological Studies, vol. 91. Springer, New York.
- Karjalainen, T., Pussinen, A., Liski, J., Nabuurs, G.-J., Erhard, M., Eggers, T., Sonntag, M., Mohren, G.M.J., 2002. An approach towards an estimate of the impact of forest management and climate change on the European forest sector carbon budget: Germany as a case study. *Forest Ecology and Management* 162, 87–103.
- Kirschbaum, M.U.F., Paul, K.I., 2002. Modelling C and N dynamics in forest soils with a modified version of the CENTURY model. *Soil Biology and Biochemistry* 34, 341–354.
- Kros, J., Reinds, G.J., de Vries, W., Latour, J.B., Bollen, M.J.S., 1995. Modelling of soil acidity and nitrogen availability in natural ecosystems in response to changes in acid deposition and hydrology. SC-DLO Report 95, Wageningen, The Netherlands.
- Leeters, E.E.J.M., de Vries, W., 2001. Chemical composition of the humus layer, mineral soil and soil solution of 200 forest stands in the Netherlands in 1995. Alterra, Green World Research, Wageningen, The Netherlands. Alterra rapport 424.2.
- Liski, J., Perruchoud, D., Karjalainen, T., 2002. Increasing carbon stocks in the forest soils of western Europe. *Forest Ecology and Management* 169, 159–175.
- Liski, J., Palosuo, T., Peltoniemi, M., Sievänen, R., 2005. Carbon and decomposition model Yasso for forest soils. *Ecological Modelling* 189, 168–182.
- Liu, C., Westman, C.J., Berg, B., Kutsch, W., Wang, G.Z., Man, R., Ilvesniemi, H., 2004. Variation in litterfall-climate relationships between coniferous and broadleaf forests in Eurasia. *Global Ecology and Biogeography* 13, 105–114.
- Masera, O.R., Garza-Caligaris, J.F., Kanninen, M., Karjalainen, T., Liski, J., Nabuurs, G.J., Pussinen, A., de Jong, B.H.J., Mohren, G.M.J., 2003. Modeling carbon sequestration in afforestation, agroforestry and forest management projects: the CO2FIX V.2 approach. *Ecological Modelling* 164, 177–199.
- Meentemeyer, V., Box, E.O., Thompson, R., 1982. World patterns and amounts of terrestrial plant litter production. *Bioscience* 32, 125–128.
- Melillo, J.M., Aber, J.D., Muratore, J.F., 1982. Nitrogen and lignin control of hardwood leaf litter decomposition. *Ecology* 63, 621–626.
- Monteith, D.T., Evans, C.D., Reynolds, B., 2000. Are temporal variations in the nitrate content of UK upland freshwaters linked to the North Atlantic oscillation? *Hydrological Processes* 14, 1745–1749.
- Murphy, K.L., Klopatek, J.M., Coe Klapotek, C., 1998. The effects of litter quality and climate on decomposition along an elevational gradient. *Ecological Applications* 8, 1061–1071.
- Nabuurs, G.J., Schelhaas, M.J., 2002. Carbon profiles of typical forest types across Europe assessed with CO2FIX. *Ecological Indicators* 1, 213–223.
- Nadelhoffer, K.J., Emmett, B.A., Gundersen, P., Kjonaas, O.J., Koopmans, C.J., Schleppe, P., Tietema, A., Wright, R.F., 1999. Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests. *Nature* 398, 145–148.
- Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of factors controlling soil organic matter levels in Great Plains Grasslands. *Soil Science Society of America Journal* 51, 1173–1179.
- Peng, C., Apps, J.M., Price, D.T., Nalder, I.A., Halliwell, D.H., 1998. Simulating carbon dynamics along the Boreal Forest Transect Case Study (BFTCS) in central Canada, 1, model testing. *Global Biogeochemical Cycles* 12, 381–392.
- Perruchoud, D., Walthert, L., Zimmermann, S., Lüscher, P., 2000. Contemporary carbon stocks of mineral forest soils in the Swiss Alps. *Biogeochemistry* 50, 111–136.
- Powlson, D.S., Smith, P., Smith, J.U., 1996. *Evaluation of Soil Organic Matter Models*. Springer, Berlin.
- Rosenqvist, L., Hansen, K., Vesterdal, L., Denier van der Gon, H., van der Salm, C., Bleeker, A., Johansson, M.-B., 2007. Nitrogen deposition and nitrate leaching following afforestation: experiences from oak and Norway spruce chronosequences in Denmark, Sweden and the Netherlands. In: Heil, G.W., Muys, B., Hansen, K. (Eds.), *Environmental Effects of Afforestation in North-Western Europe*. From Field Observations to Decision Support. Plant and Vegetation, vol. 1. Springer Series, pp. 79–108.
- Sariyildiz, T., 2003. Litter decomposition of *Picea orientalis*, *Pinus sylvestris* and *Castanea sativa* trees grown in Artvin in relation to their initial litter quality variables. *Turkish Journal of Agriculture and Forestry* 27, 237–243.
- Schöpp, W., Posch, M., Mylona, S., Johansson, M., 2003. Long-term development of acid deposition (1880–2030) in sensitive freshwater regions in Europe. *Hydrology and Earth System Sciences* 7, 436–446.
- Schulze, E.D., Högberg, L., van Oene, H., Persson, T., Harrison, A.F., Read, D., Kjoller, A., Matteucci, G., 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, vol. 142. Springer, Berlin, Heidelberg, New York, pp. 468–492.
- Silver, W.L., Miya, R.K., 2001. Global patterns in root decomposition: comparisons of climate and litter quality effects. *Oecologia* 129, 407–419.
- Starr, M., 1999. WATBAL: a model for estimating monthly water balance components, including soil water fluxes. In: Kleemola, S., Forsius, M. (Eds.), 8th Annual report 1999. UN ECE ICP Integrated Monitoring. The Finnish Environment 325. Finnish Environment Institute, Helsinki, Finland, pp. 31–35.
- Thornthwaite, C.W., Mather, J.R., 1957. Instructions and tables for computing potential evapotranspiration and the water balance. *Publications in Climatology* 10, 185–311.
- Tietema, A., Emmett, B.A., Gundersen, P., Kjonaas, O.J., Koopmans, C., 1998. The fate of ¹⁵N-labelled nitrogen deposition in coniferous forest ecosystems. *Forest Ecology and Management* 101, 19–27.
- Van der Salm, C., Reinds, G.J., de Vries, W., 2007a. Water balances in intensively monitored forest ecosystems in Europe. *Environmental Pollution* 148, 201–212.
- Van der Salm, C., Rosenqvist, L., Vesterdal, L., Hansen, K., Denier van der Gon, H., Bleeker, A., Wieggers, R., Van den Toorn, A., 2007b. Interception and water recharge following afforestation: experiences from oak and Norway spruce chronosequences in Denmark, Sweden and the Netherlands. In: Heil, G.W., Muys, B., Hansen, K. (Eds.), *Environmental Effects of Afforestation in North-Western Europe*. From Field Observations to Decision Support. Plant and Vegetation, vol. 1. Springer Series, pp. 53–77.
- Van Oijen, M., Ågren, G.I., Chertov, O., Kellomäki, S., Komarov, A., Mobbs, D.C., Murray, M.B., 2008. Evaluation of past and future changes in European forest growth by means of four process-based models. Chapter 4.4. In: Kahle, H.P., Karjalainen, T., Schuck, A., Ågren, G.I., Kellomäki, S., Mellert, K., Priezel, J., Rehfuess, K.E., Spiecker, H. (Eds.), *Causes and Consequences of Forest Growth Trends in Europe—Results of the RECOGNITION Project*. European Forest Institute Research Report 21. Brill., Leiden, pp. 183–199.
- Van Veen, J.A., 1977. The behaviour of nitrogen in soil: a computer simulation model. Ph.D. Thesis. Free University Amsterdam, Amsterdam.

- Vesterdal, L., 1999. Influence of soil type on mass loss and nutrient release from decomposing foliage litter of beech and Norway spruce. *Canadian Journal of Forest Research* 29, 95–105.
- Vesterdal, L., Rosenqvist, L., van der Salm, C., Hansen, K., Groenenberg, B.J., Johansson, M.-B., 2007. Carbon sequestration in soil and biomass following afforestation: Experiences from oak and Norway spruce chronosequences in Denmark, Sweden and the Netherlands. In: Heil, G.W., Muys, B., Hansen, K. (Eds.), *Environmental Effects of Afforestation in North-Western Europe. From Field Observations to Decision Support*. Plant and Vegetation, vol. 1. Springer Series, pp. 19–51.
- Vleeshouwers, L.M., Verhagen, A., 2002. Carbon emission and sequestration by agricultural land use: a model study for Europe. *Global Change Biology* 8, 519–530.
- Wamelink, G.W.W., Wieggers, H.J.J., Reinds, G.J., Kros, J., Mol-Dijkstra, J.P., van Oijen, M., de Vries, W., 2009. Modelling impacts of changes in carbon dioxide concentration, climate and nitrogen deposition on NPP and carbon sequestration of Intensive Forest Monitoring plots in Europe. *Forest Ecology and Management* 258, 1794–1805.
- Wolf, J., Hack ten Broeke, M.J.D., Rotter, R., 2005. Simulation of nitrogen leaching in sandy soils in The Netherlands with the ANIMO model and the integrated modelling system STONE. *Agriculture, Ecosystems & Environment* 105, 523–540.
- Xu, X., Hirata, E., 2005. Decomposition patterns of leaf litter of seven common canopy species in a subtropical forest: N and P dynamics. *Plant and Soil* 273, 279–289.