CHAPTER 2

CARBON SEQUESTRATION IN SOIL AND BIOMASS FOLLOWING AFFORESTATION: EXPERIENCES FROM OAK AND NORWAY SPRUCE CHRONOSEQUENCES IN DENMARK, SWEDEN AND THE NETHERLANDS

L. VESTERDAL[1], L. ROSENQVIST[2], C. VAN DER SALM[3],
K. HANSEN[1], B.-J. GROENENBERG[3] and M.-B. JOHANSSON[2]

[1]Forest and Landscape Denmark, KVL, Hørsholm Kongevej 11, 2970 Hørsholm, Denmark, E-mail: lv@kvl.dk, [2]Swedish Agricultural University, Box 7001, 750 07 Uppsala, Sweden, [3]Alterra Green World Research, P.O. Box 47, 6700 AA Wageningen, the Netherlands

Abstract. There is limited knowledge of the contribution of afforested arable land to mitigation of greenhouse effects. In the AFFOREST project we evaluated the rate and magnitude of carbon (C) sequestration in biomass and soils following afforestation of cropland. Two oak (*Quercus robur*) and four Norway spruce (*Picea abies*) afforestation chronosequences (age range 1 to 90 years) were studied with respect to C sequestration in Denmark, Sweden and the Netherlands.

Biomass C sequestration ranged between 2.7 and 4.6 Mg C ha^{-1} yr^{-1} for stands younger than 45 years with no clear influence of different site characteristics. Such effects were probably masked by the soil enrichment, which is a legacy of former agriculture. Biomass C sequestration differed more between sites after 40-50 years owing to different management, tree species-specific growth patterns and less influence of former fertilization.

For the total soil compartment studied, i.e. forest floor and mineral soil 0-25 cm, afforestation of cropland as a minimum resulted in unchanged soil C contents and in most cases led to net C sequestration. Rates of soil C sequestration ranged from being negligible in two of the Danish chronosequences to 1.3 Mg C ha^{-1} yr^{-1} for the Dutch chronosequence. The allocation of sequestered soil C was also quite different among chronosequences. While forest floor development consistently led to C sequestration, there was no general pattern in mineral soil C sequestration. In the short term (30 years), tree species had little influence on total soil C sequestration. Afforestation of nutrient-poor sandy soils seemed to result in larger C sequestration in forest floors and the whole soil than afforestation of nutrient-rich, clayey soils.

G.W. Heil et al. (eds.), Environmental Effects of Afforestation in North-Western Europe, 19–51.

For the afforested ecosystem as a whole, the general contribution of soils to C sequestration (i.e. to a net gain in C stock) was about one third of the total C sequestration. The contribution of soil varied among the chronosequences from none to 31%, which is not far from reported contributions of soil in similar studies. In the short term (30-40 years), total C sequestration was higher in Norway spruce than in oak whereas soil type did not clearly influence the rate of C sequestration.

The work in AFFOREST has improved the knowledge of C sequestration in afforested cropland. The new results may help to bridge the gap between existing knowledge and policy demands.

1. INTRODUCTION

Sequestration of carbon (C) has recently been recognized as an important environmental effect of afforestation, and in a European perspective afforestation may provide the greatest potential for C sequestration in agricultural soils (Powlson et al. 1998). A new challenge in the context of climate change mitigation is the management of terrestrial ecosystems to conserve existing carbon stocks and to remove carbon from the atmosphere by increasing the existing stocks (Malhi et al. 1999). Afforestation of former arable land has been acknowledged under the Kyoto Protocol (article 3.3) as an eligible activity that contributes to the mitigation of increasing atmospheric CO_2 concentrations. As a specific change in land use, increases in terrestrial C stocks due to afforestation may serve as one of the measures to meet national reduction commitments.

Forests are generally characterized by a higher density of carbon than arable land, mainly due to the presence of perennial vegetation with a high biomass. Soils under forest are often considered to contain larger stocks of C than cropland soils, but the literature is not consistent (Rodriguez-Murillo 2001; Krogh et al. 2003; Lettens et al. 2004). What happens when cropland is converted to forest? Based on comparisons of C stocks in forests and arable land afforestation is expected to result in significant sequestration of C due to accumulation of woody biomass and further accumulation of organic matter in the soil. Already Billings (1938) reported quantitative data on soil C sequestration in former arable soils based on a study of shortleaf pine (*Pinus echinata* Mill.) afforestation in North Carolina. It is nevertheless still debated whether soil C stocks generally will increase following afforestation. For instance, the previous arable land use may be important for predicted increases in soil carbon, as soil C differ between permanent pasture, annually tilled cropland and no-till management (Römkens et al. 1999; Dick 1983; Denef et al. 2004).

There are, in addition, large uncertainties in terms of the rate of C sequestration and also the allocation of carbon to soil and biomass following afforestation. The contribution of soils to C sequestration is especially uncertain and it is a significant future challenge to quantify the potential soil sinks for CO_2 (Smith 1999; Garcia-Oliva & Masera 2004). In a review Johnson (1992) concluded that the reversion of former agricultural land to forest usually results in substantial increases in soil C, and Bouwman & Leemans (1995) suggested that 50 Mg C ha^{-1} would be sequestered in afforested tropical soils in 30 years. On the other hand some studies indicate that these expectations for soils are far too high (Hamburg 1984; Jug et al. 1999; Richter et al. 1999; Paul et al. 2002). The aggrading biomass of forest trees accounts for much of the C sequestered after afforestation. However, the relative contributions of biomass and soil are uncertain and must be expected to vary between forest, soil and climate types. Few studies included concurrent measurements of soil and biomass C sequestration, but these studies suggested that about 25% of the total C can be sequestered in the soil (Richter et al. 1999; Hooker & Compton 2003).

While C stored in forest biomass is strongly influenced by changing forest management or by disturbances such as clearcutting and replanting, C stored in mineral soils is less susceptible to such changes (Dewar & Cannell 1992). Soils may be a more permanent sink for C and the question remains to what extent they contribute to the total C sequestration of an afforested ecosystem. The quality of C stores with respect to permanency should be considered together with the quantity of C. Better quantitative information on C sequestration is needed in several European countries in order to evaluate the potential contribution of terrestrial systems and to meet the obligations following ratification of the Kyoto Protocol.

Direct measurements of C sequestration would require flux towers or the reassessment of C pools over a considerably long time period to assess changes. In order to save time, sampling of paired plots of afforested and arable land has often been undertaken. Many of the previous studies on afforestation were done on marginal land, on former pastures or on land abandoned before frequent fertilization and liming were introduced in agriculture. Such studies may not be representative for the intensively managed arable soils afforested today. Also, there is a need for studies that provide a basis for reporting and modeling national C sinks in the short-term, e.g., during the Kyoto commitment period 2008-2012.

This chapter evaluates the general effect of afforestation of former cropland on C sequestration in a synthesis of AFFOREST chronosequence experiments in three European countries. The results from the AFFOREST project are compared to previous studies of afforested cropland. The specific objectives were i) to estimate the rates of total C sequestration, ii) to determine the relative contributions to C sequestration of the soil and biomass components of the new forest ecosystems, iii) to study the possible differences in C sequestration between deciduous (oak) and coniferous (Norway spruce) tree species, and iv) to explore C sequestration at contrasting soil types.

2. MATERIALS AND METHODS

2.1. General approach

To study the effect on carbon sequestration following afforestation with oak and spruce, chronosequences of afforestation stands were selected in Denmark, southern Sweden and the Netherlands (Chapter 1). Measurements were conducted in each chronosequence stand to assess the changes in C storage. Biomass was measured for estimation of above- and belowground biomass C storage, and forest floors and mineral soils were sampled to assess soil C storage at different points in time following afforestation of former arable land.

2.2. Study sites

The study included six chronosequences of which two were differently aged oak stands and four were differently aged Norway spruce stands. In Denmark, one oak chronosequence and one Norway spruce chronosequence were assessed within the same forest with clay-rich and nutrient-rich soil in Vestskoven close to Copenhagen. Just outside Vestskoven, a 200-year-old afforested oak stand, Ledøje Plantage, was included for comparison. In contrast to all other stands Ledøje Plantage had a multi-layered stand structure, i.e. with beech and sycamore maple forming a subcanopy. Another spruce chronosequence in a contrasting environment was studied on sandy, poor soil at Gejlvang west of Vejle in southern Jutland. In the Netherlands, one chronosequence of oak and one chronosequence of spruce were studied on similar sandy soil close to Sellingen. The last chronosequence of spruce was situated in south-western Sweden, at Tönnersjöheden east of Halmstad. A map of the locations is found in Chapter 1 (Figure 1.8), and detailed site information is given on the AFFOREST web site (www.sl.kvl.dk/afforest). Briefly, annual temperatures ranged from 6-7°C in Sweden over 7.7°C in Denmark to 9°C in the Netherlands. Annual precipitation ranged from about 640 mm at Vestskoven, Denmark over ca. 800 mm in the Netherlands and to ca. 1000 mm at Gejlvang, Denmark and in Sweden. Soil types ranged from loamy Hapludalfs at Vestskoven, Denmark, to sandy Spodosols in the other chronosequences (Soil Survey Staff 1992). All sites were former cropland with annual tillage. The soils are mainly well drained except in the Netherlands where groundwater levels may be within 50 cm depth during wet periods of the winter. Nitrogen deposition was lowest to the Swedish spruce chronosequence (ca. 20 kg ha^{-1} yr^{-1}) and highest to the Danish spruce chronosequence at Gejlvang and the Dutch oak chronosequence (16-33 kg ha^{-1} yr^{-1}) (Chapter 4).

In all three countries, soil and biomass C stocks were measured once whereas rates of litterfall C were measured for 1-2 years. Table 2.1 gives an overview of the number of stands in each of the chronosequence for litterfall C and biomass and soil C stocks. In Denmark, all chronosequence stands were characterized in terms of biomass and soil C stocks, whereas biomass C was only estimated for a subset of the Swedish and Dutch stands. In the Swedish and Dutch chronosequences, baseline data on soil C data were included from fields still in arable use. Rates of litterfall C were only measured in selected chronosequence stands in all three countries.

Table 2.1. *The number of stands and their range in age with respect to litterfall, biomass and soil C measurements in each chronosequence. Age ranges deviate slightly between the C pools as they were not assessed in the same year.*

Chronosequence	Litterfall C		Biomass C		Soil C	
	Stands	Age range, (years)	Stands	Age range, (years)	Stands	Age range, (years)
Oak, Vestskoven, DK	5	8-31	8	5-28, 200	8	5-28, 200
Norway spruce, Vestskoven, DK	4	11-32	7	1-29	7	1-29
Norway spruce, Gejlvang, DK	3	20-41	5	4-41	5	4-41
Norway spruce, Tönnersjöheden, SE	4	20-93	5	19-92	24 (15)*	0-87
Oak, Sellingen, NL	4	4-19		4-19	12 (10)*	0-29
Norway spruce, Sellingen, NL	-	-	3	9-15	3	9-15

**In these two chronosequences, data on arable land were obtained from 9 fields (SE) and 2 fields (NL) within the afforestation area. The number of afforested stands is given in brackets.*

2.3. Biometrics and calculation of carbon stocks in biomass

In Denmark, stand characteristics at Vestskoven and Gejlvang were assessed in June 2001 according to the methods used in the Danish National Forest Inventory (Johannsen et al. 2004). Briefly, the three circular plots per stand were used as inventory plots for stem number, diameter at breast height and tree height. Diameter at breast height was measured for all trees within plots of different radii according to the diameter of trees, whereas height was measured for a subgroup of these trees randomly selected with probability of selection proportional to diameter. Heights of single trees were subsequently estimated by diameter-height regressions (Näslund 1936). Standing volume of merchantable wood (stem and crown wood for oak and stem wood for spruce) was estimated using equations developed by Madsen (1987) and Madsen & Heusérr (1993).

In Sweden, stand mensuration was done in the five intensively studied stands of the Tönnersjöheden chronosequence in September 2002. Stand data (stem number, diameter at breast height and tree height) were gathered on plots of 25 m × 25 m. Mean diameter, basal area and stand density were calculated. Diameter at breast height was recorded for all trees whereas height was measured for a subgroup of trees selected according to a procedure described by Karlsson (1998). Heights of remaining trees were subsequently estimated by diameter-height regressions. Standing volume of merchantable wood was estimated using equations developed by Brandel (1990).

In the Netherlands, aboveground biomass of trees was estimated on the 15 m × 15 m measurement plots in the 4 oak stands and the 3 spruce stands in October 2002. In order to save time per monitoring area and to be able to estimate the variability in the estimated biomass 3 sections of 7 × 7 m^2 were selected within the measurement plots. Within each section diameter at breast height (DBH), tree height (H) and stand density (N) were measured. DBH and N were measured for each individual tree, whereas H was based on an average estimate per 7 m × 7 m plot. The volume per tree (V) was calculated by using the equations for oak and Norway spruce in Jansen et al. (1996).

Standing volume of merchantable wood was subsequently expanded to total above- and belowground biomass and carbon stock of the total biomass. In Denmark this was done using the expansion factors, basic wood densities and carbon content used for oak and Norway spruce in annual Danish reports on LUCF to UNFCCC (Illerup et al. 2005). Briefly, total biomass was estimated by use of expansion factors 1.2 and 1.8 for oak and Norway spruce, respectively. The lower expansion factor for oak is partly due to inclusion of crown biomass in the Danish volume equations. The volume of wood was converted to mass by use of basic densities of 0.56 and 0.38 Mg m^{-3} for oak and Norway spruce, respectively. In Sweden, specific biomass functions were available for estimating total dry mass (Marklund 1988). In the Netherlands, tree volumes were transformed to biomass by using an expansion factor of 1.8 and wood densities similar to those used in Denmark. Carbon content for stands in all three countries was estimated using a common C concentration of 500 mg g^{-1} (IPCC 2003).

2.4. Litterfall

Total litterfall was measured for two years in Denmark and for one year only in Sweden and the Netherlands. In Denmark, five oak stands and four Norway spruce stands were sampled at Vestskoven and three Norway spruce stands were sampled at Gejlvang. Five litter traps (diameter 31 cm) per stand were placed 1 m above ground in the center and four cardinal points of one circular subplot (10 m radius) per stand. Litter from the five traps was collected every month and dried to constant weight (at 60°C) before weighing the litter produced during each of the two years. A subsample of the litter was subsequently ground for chemical analysis. Ground samples of litter material were analyzed for total C by dry combustion (Dumas method) in a Leco CSN 2000 Analyzer (Matejovic 1993). In Sweden, total litterfall was measured in four stands at Tönnersjöheden. Five litter traps (diameter 64 cm) were randomly placed within a 25 m × 25 m plot in each stand, and litter was collected two times during the sampling period. In other respects the methods were similar to those applied in Denmark. In the Netherlands litterfall was measured in four of the oak stands. Three litter traps (100 × 100 cm) were randomly placed 0.3 m above ground in each of the stands. Litter was collected bi-monthly from the traps and dried to constant weight at 40°C before weighing. A subsample of the litter was subsequently ground and analyzed for total C by dry combustion and measurement of the CO_2 gasses using an IR-cell.

2.5. Soil sampling and analyses

Stand sizes were different in the three countries and slightly different sampling designs and sampling methods were used for soil C while analyses and calculations were performed similarly in the three countries. The sampling design and methods are summarized in Table 2.2. More detailed descriptions of the sampling and analysis of soils in Denmark and Sweden may be found in Vesterdal et al. (2002) and Rosenqvist & Johansson (2005), respectively.

In Denmark and Sweden, C concentrations in the forest floor and soil were determined by dry combustion (Dumas method) in a Leco CSN Analyzer (Matejovic 1993). The Dutch soil samples were analyzed for C by wet oxidation with potassium dichromate (Kurmies 1949).

Carbon sequestration is reported as the relationship between C contents and stand age. Forest floor C contents were calculated by multiplying C concentrations with forest floor mass. For the mineral soil C contents for the fraction $\geq$ 2mm (small stones and gravel) were neglected (McNabb et al. 1986; Homann et al. 1995), and soil organic carbon (SOC) contents in [Mg ha^{-1}] for each of the three soil layers were calculated via

$$SOC_i = \rho_i \bullet (1 - (\delta_{i,\,2mm}/100)) \bullet d_i \bullet C_i \qquad (1)$$

where ρ_i is the bulk density of the < 2 mm fraction in g cm^{-3}, $\delta_{i,\,2\,mm}$ is the relative volume of the fraction $\geq$ 2 mm (%), d_i denotes the thickness of layer i in cm, and C_i denotes the C concentration of layer i.

Table 2.2. *Summary of soil sampling designs for estimation of C stocks.*

	Denmark	Sweden	The Netherlands
Time of sampling	Vestskoven: Sept. 1998 Gejlvang: June 2001	Sept. 1998	May 2003
No. of sampling plots per stand	3 circular plots of 10 m ∅	1 plot 25 × 25 m	1 plot 15 × 20 m
Sampling points per stand	12 (4 cardinal points × 3 plots)	In grid intersections Soil: 20 Forest floor: 16	In grid intersections Soil: 20 Forest floor: 10
Forest floor sampling area	25 × 25 cm	Auger ∅ 10 cm or 25 × 25 cm	10 × 10 cm
Mineral soil sampling method and depth	Auger (50 mm ∅) to 25 cm (Westman 1995)	Auger (44 mm ∅) to bottom of Ap (ca. 25 cm)	Auger (44 mm ∅) to 30 cm
Subdivisions of mineral soil for analysis	0-5, 5-15, 15-25 cm	0-5 cm, 5 cm - bottom of Ap	0-10, 10-30 cm
Bulk density (< 2 mm fraction)	Measured, 6 cores per stand	Measured on composite sample of 16 subsamples	Estimated from equations by Hoekstra & Poelman (1982)

2.6. Calculations and statistics

Soil C stocks were assessed for the forest floor and upper 25 cm of the mineral soil in all chronosequences. Relationships between stand age and soil, biomass and ecosystem C contents were explored by simple linear regression while the influence of tree species in Denmark was tested by analysis of covariance. No transformations were necessary to fulfil the requirements regarding normally distributed residuals and homogeneity of variances. All statistical tests were carried out using the procedure GLM in SAS (SAS Institute 1993). The significant regression models were used to calculate the changes in soil C stores for the time span of the chronosequence. The 200-yr old stand in Denmark was not included in regressions, but was included in figures for comparison.

3. RESULTS

3.1. Carbon in biomass

The basic data for estimation of carbon sequestration in biomass was the accumulated biomass measured as the volume of merchantable wood. Thus, measured heights, stem diameters and stand densities (stem number per ha) determine the amount of carbon sequestered after afforestation. The development in stand height with age was surprisingly similar across all chronosequences (Figure 2.1a). The low mean height of the ca. 200-year-old Ledøje Plantage is due to multi-layered structure, i.e. with beech and sycamore maple forming a subcanopy. The heights of dominant oak trees were within 25-30 m at Ledøje Plantage.

Stand densities were more variable in the younger stands reflecting different planting densities and different thinning intensities (Figure 2.1b). Especially the Dutch oak stands and the Danish spruce stands at Gejlvang had high densities, the latter because they were planted for Christmas tree production.

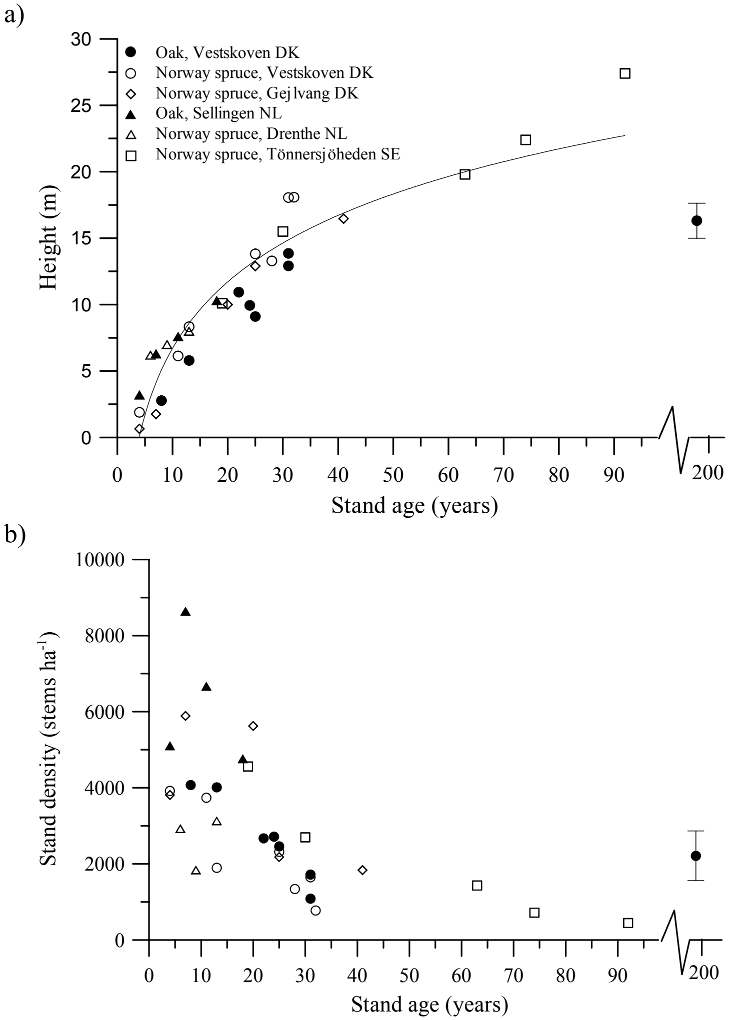


Figure 2.1. *a) Stand height and b) number of trees per hectare as a function of age in chronosequence stands.*

The C content of the total biomass is shown in Figure 2.2. There was a strong relationship with age, and the rates of biomass C sequestration for different chronosequences were quite similar for the first 40-50 years following afforestation (Table 2.3). For all stands younger than 45 years, the rate of C sequestration was about 3.7 Mg C ha^{-1} yr^{-1}. However, the Swedish chronosequence, which included older stands, had a clearly lower rate of biomass C sequestration that was not quite significantly different from 0 (Table 2.3). The older Swedish stands have a relatively lower C storage and do not continue the rate of C sequestration depicted by the Swedish stands younger than age 40. Oak stands in Denmark also tended to have a lower rate of biomass C sequestration, and differed from spruce at the same site. As indicated by stand heights, there was little difference in rate of C sequestration between stands of spruce on contrasting soil types within Denmark. Spruce stands at the nutrient-rich Vestskoven site sequestered carbon only at a slightly higher rate than at the sandy, nutrient-poor Gejlvang site (Table 2.3). The few spruce stands in the Netherlands prevent a general comparison between species.

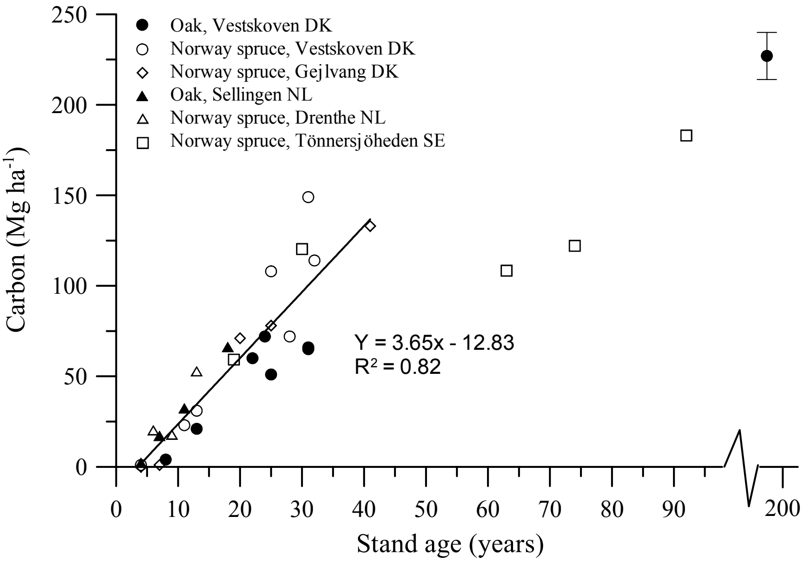


Figure 2.2. *Carbon content in above- and belowground biomass in the chronosequence stands. The general regression for all stands younger than 45 years is shown.*

Table 2.3. *Rates of biomass C sequestration (SE of regression slope) in the AFFOREST chronosequences.*

Chronosequence	Rate (Mg C ha^{-1} yr^{-1})	P*
Oak, Vestskoven, DK	2.72 (0.53)	0.004
Norway spruce, Vestskoven, DK	4.61 (0.82)	0.002
Norway spruce, Gejlvang, DK	3.76 (0.31)	0.001
Norway spruce, SE	1.20 (0.47)	0.082
Oak/Norway spruce, NL	4.55 (0.56)	<0.001
All stands <45 years	3.65 (0.34)	<0.001

**P values < 0.05 indicate whether slopes of regressions (y = ax + b) are significantly different from 0.*

3.2. Aboveground input of C to soils with litterfall

The carbon in annual litterfall represents the aboveground input of C to the soil. Litterfall is the source of organic matter for development of forest floors following afforestation. There was no clear relationship between stand age and the rate of litterfall C over the age span of the chronosequences. However, especially for the Danish stands there were increasing rates of litterfall C during the first 20 years until the annual C content of litterfall appear to level off. Litterfall tended to be higher in oak than in spruce stands in Denmark, and Dutch oak stands had comparable litterfall C contents to Danish oak stands. However, litterfall C contents representative of closed stands were reached at a younger age than in Denmark. In the oldest Swedish stands, litterfall C contents appear to decrease.

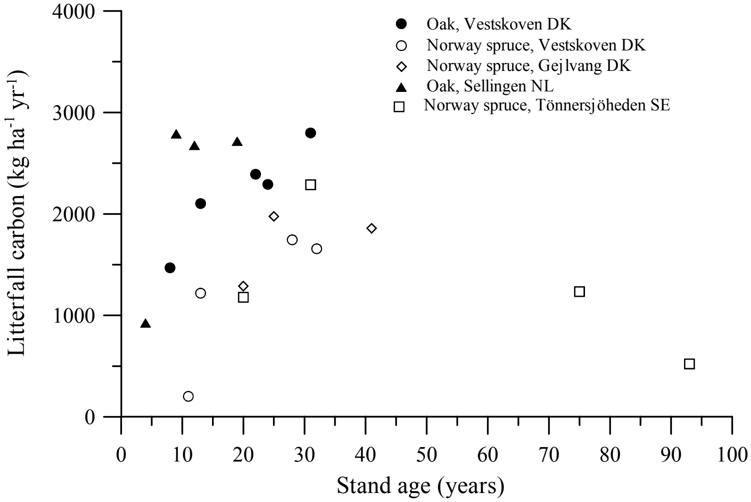


Figure 2.3. *Rates of litterfall C measured over two years in Denmark and for one year in Sweden and the Netherlands.*

3.3. Carbon in soils

The forest floor is the layer of dead organic matter, i.e. leaves, needles, twigs, branches and fruits, that blankets the mineral soil of a forest. Immediately after afforestation there is no forest floor present, but it developed rapidly (Figure 2.4) when the input of C from litterfall increases concurrently with canopy closure (Figure 2.3). The development in forest floor C sequestration was at a first glance quite similar between tree species and sites. However, this similarity is probably an artefact of the different age spans represented by the chronosequences. For instance oak stands in Denmark are in the low range of forest floor C storage at 25-30 years, and the carbon storage in forest floors of the 200-year-old Ledøje Plantage is comparable to that of the 25-30-year-old oak stands. Rates of C sequestration ranged from 0.08 (oak at Vestskoven, DK) to 0.65 Mg C ha^{-1} yr^{-1} (spruce in Sweden, Table 2.4.). Oak stands in Denmark had significantly lower (P=0.005) rate of C sequestration than spruce at the same site. Danish spruce stands and Dutch oak and spruce stands were quite comparable in forest floor C sequestration rate (0.34-0.4 Mg C ha^{-1} yr^{-1}, Figure 2.4 and Table 2.4).

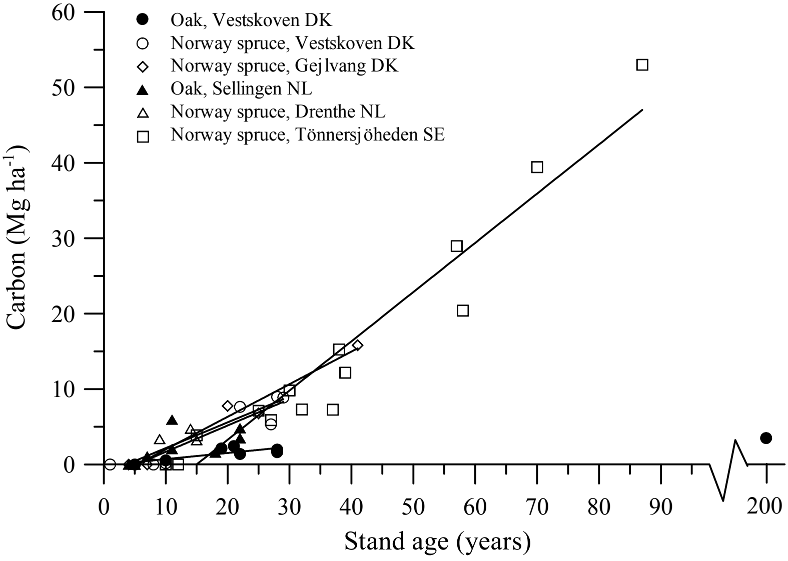
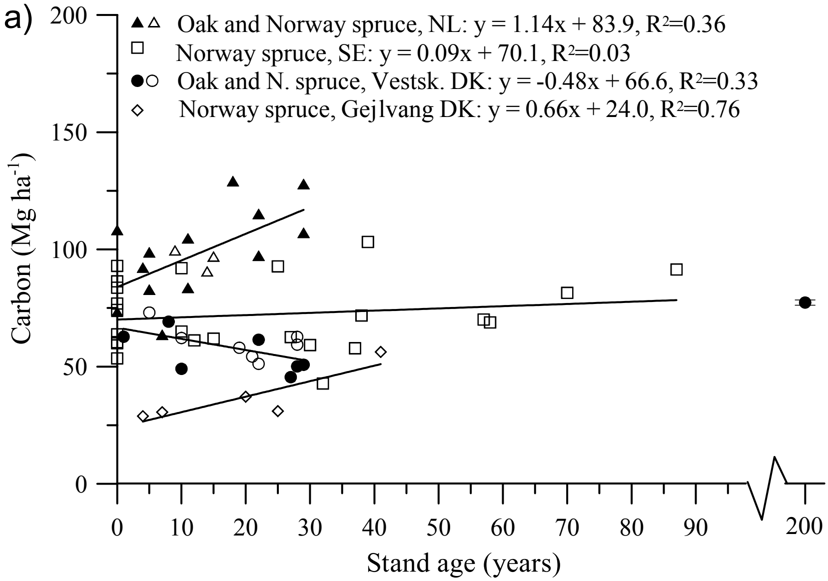


Figure 2.4. *The C content of forest floors in the chronosequence stands.*

In the mineral soil, i.e. the former plow layer 0-25 cm, the pattern was more diverse and the relative change less dramatic than for forest floors. The different chronosequence sites were clearly showed different behaviour with respect to the initial level of soil C and its development with age since afforestation (Figure 2.5a). There was no difference between the Danish spruce and oak chronosequences at Vestskoven, so all data from Vestskoven were combined in the analysis of soil C sequestration rates (Vesterdal et al. 2002). Similarly the few Dutch spruce stands did not differ from the pattern of the oak chronosequence and these data were consequently combined. Dutch sites had the highest C contents in the mineral soil and the Danish site Gejlvang was lowest in mineral soil C content. Only in the Dutch oak chronosequence and to some extent also the Danish Gejlvang chronosequence did the mineral soil C contents increase significantly with increasing stand age. For the studied soil compartments as a whole, i.e. the forest floor and the plow layer, there was a general pattern of constant or increasing soil C stores over 30-90 years after afforestation (Figure 2.5b).

The allocation of sequestered soil C was quite different for the studied chronosequences (compare Figures 2.5a-b). For the Swedish spruce chronosequence and the Danish spruce and oak chronosequences on fertile soils, the change in C was solely due to development of forest floors on top of the mineral soil. The amount of C in the mineral soil was unchanged (SE) or even decreased slightly (Vestskoven, DK). However, in the Danish spruce chronosequence on poor soil in Gejlvang the former plow layer also increased significantly in C content. The allocation of sequestered C within the soil was again different for the Dutch oak chronosequence. Here the plow layer sequestered almost four times more C than the forest floor.



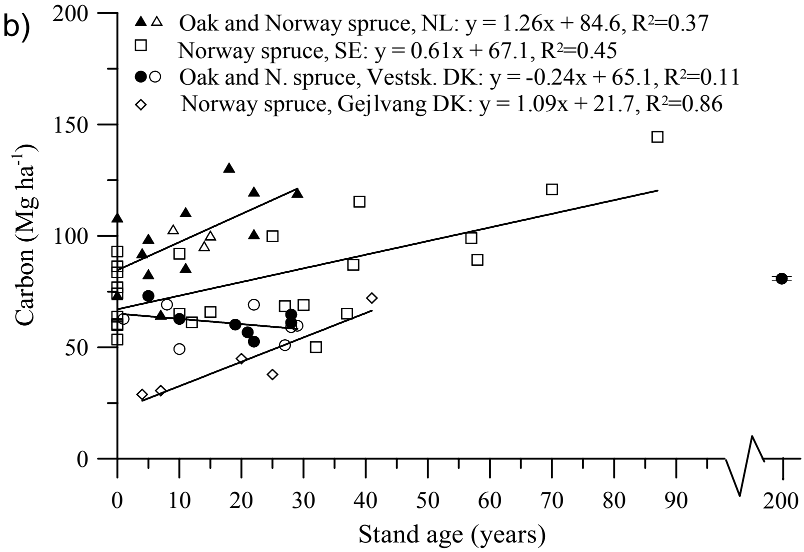


Figure 2.5. *a) Mineral soil C content in the former plow layer (0-25 cm). b) Total soil C content (forest floor + former plow layer).*

The rates of soil C sequestration based on linear relationships are given in Table 2.4. For the mineral soil, rates varied from a significant loss of 0.5 Mg C ha^{-1} yr^{-1} in the Danish chronosequences on rich soil to a gain of 1.1 Mg C ha^{-1} yr^{-1} in the Dutch chronosequence (Table 2.4). There was also a significantly positive rate of mineral soil C sequestration in the Danish spruce chronosequence on poor soil but this was not the case in the Swedish spruce chronosequence.

Rates of C sequestration were higher for both former plow layer and forest floor and were not significantly different from 0 in the Danish oak and spruce chronosequence at Vestskoven. The highest rates were 1.3 Mg C ha^{-1} yr^{-1} in the Dutch chronosequence. The chronosequences with a significant positive change in soil C were all situated on relatively poor, sandy soils. The Danish spruce chronosequence on poor soil was quite similar to the Dutch chronosequence in C sequestration rate while the Swedish spruce chronosequence indicated a lower rate as C was only sequestered in the forest floor. It is notable that there was quite large variation in the Swedish chronosequence and no net C sequestration would have been detectable within 30-40 years as in Denmark and the Netherlands.

Table 2.4. *Rates of soil C sequestration (SE of regression slope) over 30-90 years in the AFFOREST chronosequences.*

Chronosequence	Forest floor C		Mineral soil C*		Total soil C*	
	Rate (Mg ha^{-1} yr)	P value	Rate (Mg ha^{-1} yr)	P value	Rate (Mg ha^{-1} yr)	P value
Oak, Vestskoven, DK	0.08 (0.03)	0.032	-0.48 (0.19)	0.031	-0.24 (0.20)	0.25
Spruce, Vestskoven, DK	0.35 (0.06)	0.003				
Spruce, Gejlvang, DK	0.43 (0.05)	0.003	0.66 (0.21)	0.056	1.09 (0.25)	0.022
Spruce, SE	0.65 (0.05)	<0.001	0.09 (0.13)	0.461	0.61 (0.14)	<0.001
Oak/spruce, NL	0.34 (0.08)	0.001	1.14 (0.40)	0.014	1.26 (0.46)	0.017

**Rates of C sequestration for mineral soil and total soil C at Vestskoven are combined for oak and spruce.*

The Swedish chronosequence included nine cropland plots. Each cropland plot was adjacent to one or two afforested stands. Under the assumption that the cropland soil C stocks of today represent conditions prior to afforestation we conducted an analysis of pairwise differences between cropland plots and afforested stands. The paired plot design reduces spatial variability attributable to differences in soil properties. Such spatial variability may possibly mask the influence of changed land use when using the chronosequence approach. The differences in soil C stocks at different stand ages are shown in Figure 2.6 for mineral soils alone and for mineral soils + forest floors. Differences in soil C stocks between afforested and cropland plots were quite variable, and differences (forest minus cropland) were in some cases negative for the plow layer. In line with the similar absolute mineral soil C stocks found along the Swedish chronosequence, there was no significant relative change in mineral soil C stocks along the chronosequence (P=0.732). When forest floor C was included there was always most soil C in afforested plots after about 30 years, and differences in soil carbon significantly increased with increasing age of the afforested stand (P=0.009)

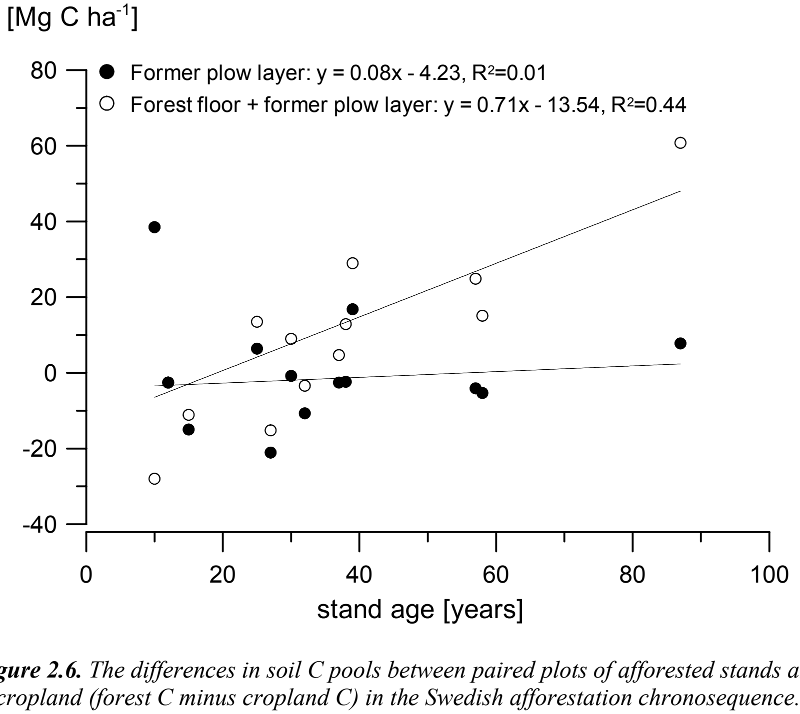


Figure 2.6. The differences in soil C pools between paired plots of afforested stands and cropland (forest C minus cropland C) in the Swedish afforestation chronosequence.

3.4. Carbon in the afforested ecosystem

The change in ecosystem C storage, i.e. in C contents of soil plus biomass, and the allocation of C to woody biomass and soil, respectively, is shown in Figure 2.7a for all chronosequences. It must be noted that while all Danish stands had data for both soil C and biomass C, biomass C was only assessed in the intensively studied stands in the Netherlands and in Sweden (Table 2.1). As a result, the data set for analysis of biomass and total ecosystem C sequestration did not cover all stands in these two countries.

There was a good relationship between C storage and age across all chronosequences with a general rate of C sequestration of about 2.8 Mg ha^{-1} yr^{-1} (Figure 2.7a) The relationship with age indicated sequestration of 250 Mg C ha^{-1} over the chronosequence of 90 years as a general result for all tree species and sites. The rate of C sequestration is expected to level off as stands mature as also indicated by the 200-year-old oak stand. Nevertheless, there was no basis for non-linear relationships. The age distribution of stands was skewed towards the age 0-30 years, so there is a better picture of the variability in C content within this age span.

The change in ecosystem C content was primarily caused by the growth of trees, but soils also contributed significantly. A general assessment based on regression slopes indicates that about 70% of the total C sequestered was allocated to biomass following afforestation, and the remaining 30% was sequestered in soils. As already reported for soils (Figure 2.7) there were different levels of soil C at the chronosequence sites. Therefore the general relationship between soil C and stand age was weak (Figure 2.7a). However, when the more wet Dutch sites with relatively high C contents were excluded, the relationship between soil C and age clearly improved (Figure 2.7b). Among individual chronosequences, the contribution of soils ranged from none at Vestskoven over ca. 20% in the Netherlands and at Gejlvang to 31% in Sweden.

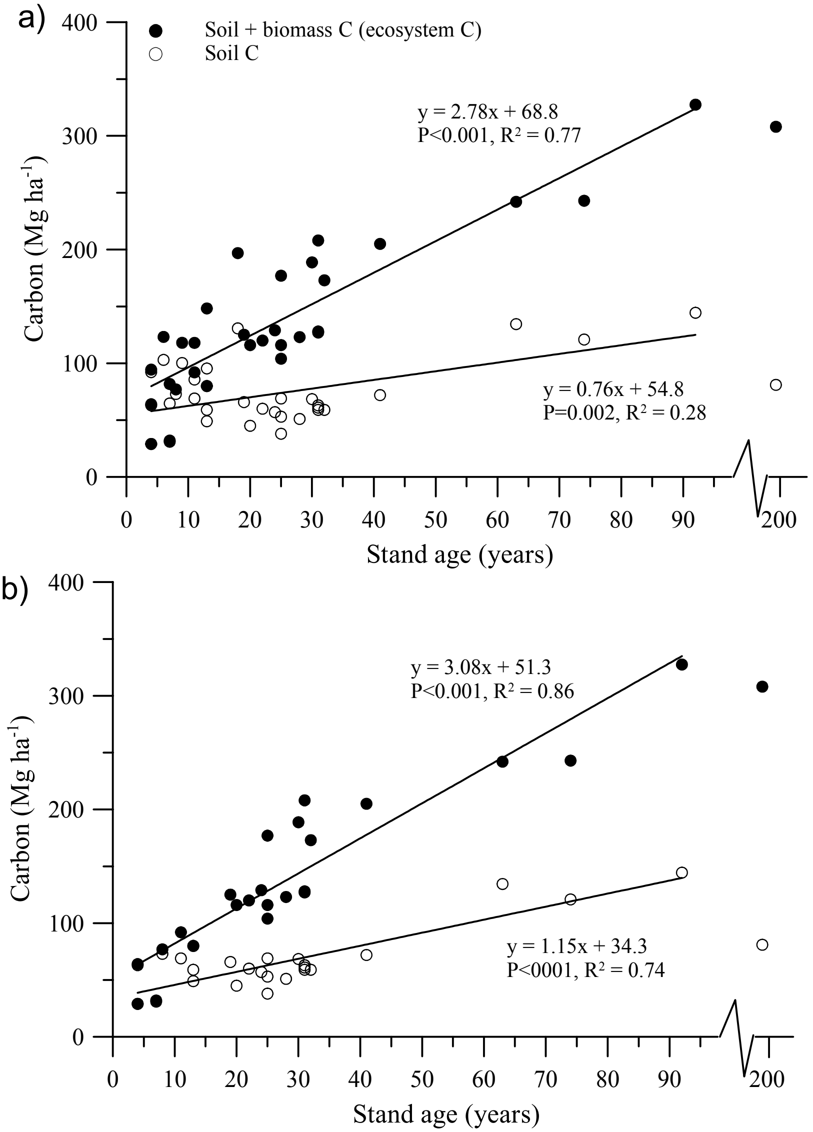


Figure 2.7. *Changes in total ecosystem C storage with time since afforestation. The graphs show the relative contribution of soils and vegetation to total ecosystem C sequestration following afforestation. The difference between the regression lines indicates the relative contribution of biomass C sequestration. a) all chronosequence sites in Denmark, the Netherlands and Sweden. b) chronosequence sites without the more wet Dutch sites.*

4. DISCUSSION

4.1. Biomass carbon

Rates of biomass carbon sequestration were estimated at 3.7 (range 2.7-4.6) Mg C ha^{-1} yr^{-1} for AFFOREST stands younger than 45 years. This range is fairly well in line with the median sequestration rate of 4.1 Mg C ha^{-1} yr^{-1} reported in a review of potential C storage in temperate regions over 25 years (Winjum & Schroeder 1997). The AFFOREST chronosequence studies only included three stands in Sweden and one stand in Denmark older than 40 years. Thus, there is little basis for conclusions regarding biomass C sequestration over a full rotation. However, from an age of 40-50 years there appears to be some differentiation between stands of different species and locations. The Swedish stands older than 45 years had sequestered less C than would be expected from the trajectories of the other chronosequences. This is mainly attributable to the fact that the Swedish chronosequence cover almost a full rotation, and current rates of increment and thus C sequestration decrease when trees mature. The sequestration rate in the Swedish chronosequence covering 90 years is well in line with the modelled sequestration rates (0.8-1.2 Mg C ha^{-1} yr^{-1}) reported by Nabuurs & Mohren (1995) for the first 100-year rotation of afforested Norway spruce in boreo-nemoral climate. Another reason for differences between chronosequences is an increasing effect of different management practices (e.g. thinning) as stands grow older. Information on thinned volumes for the older stands would have provided a more complete picture of C sequestration in biomass, but unfortunately such data were not available. Furthermore, old stands probably have a different legacy of former agriculture than the younger stands. Afforestation more than 50 years ago was carried out on soils that had not been subject to the intensive farming practices of today which include frequent liming and fertilization. The fertilized soils abandoned for afforestation today would probably also lead to higher rates of C sequestration, at least in the first decades until the organic nitrogen pool is in steady state in the forest ecosystem.

It was expected to find lower rates of C sequestration in biomass in the same tree species at poor soil compared to nutrient-rich soil. But there was little difference in the rate of C sequestration between stands of spruce on contrasting soil types within Denmark. Spruce stands at Vestskoven sequestered carbon at a slightly higher rate than at Gejlvang. Within Denmark, a difference between oak and Norway spruce was more evident with rates of C sequestration of 2.7 and 4.6 Mg C ha^{-1} yr^{-1}, respectively. This is consistent with the generally higher growth rates in spruce than in oak.

Changes in biomass C were mainly estimated for comparison with concurrent changes in soil C. As in national reporting under United Nations Framework Convention on Climate Change (e.g. Illerup et al. 2005), there are uncertainties with regard to conversion of biomass to C stock in the total biomass. These uncertainties may in part influence differences between chronosequences. However, the estimated sequestration rates are fairly conservative as we only considered on site C stocks. If thinned tree biomass and resulting storage in wood products were included, the contribution of biomass would potentially be higher in the longer chronosequences.

4.2. Forest floor carbon

When cropland is afforested, soils experience a marked reduction in management intensity. Tillage of the upper 20-30 cm ceases, as does also the application of lime and fertilizers. In addition, the vegetation changes from annual to perennial, thus accumulating a large amount of biomass C. This may change the production of dead organic matter and its quality in terms of decay. Such differences between the two types of land use would result in a phase of net accumulation of soil C until a new equilibrium level of soil C is established according to the rates of input and decomposition in a forest.

The most visible change in soil C stock at the AFFOREST sites was caused by development of a forest floor on top of the mineral soil. At a first sight, forest floors showed a quite similar development over time (Figure 2.4). Rates of C sequestration were most comparable for Danish and Dutch spruce stands and the Dutch oak stands (Table 2.4). However, as mentioned before this apparent general pattern is probably an artefact of the different age spans represented by the chronosequences. The low amount of C in the forest floor of the 200-year-old stand strongly suggests that more mature oak stands will differ from spruce stands in forest floor C storage. The Danish oak chronosequence also had a much lower rate of C sequestration (0.08 Mg C ha^{-1} yr^{-1}, Table 2.4) than that of other chronosequences. This difference in forest floor C between oak and spruce is in line with general observations in Denmark (Vesterdal & Raulund-Rasmussen 1998) and a study of litter decomposition rates in the two species (Dziadowiec 1987). In the Netherlands, however, there was no clear difference between oak and spruce, but comparison is limited by the low number of relatively young spruce stands.

The Swedish spruce stands have sequestered quite high amounts of C in forest floors, and this development may possibly be representative also for Danish and Dutch spruce stands on poor sandy soils. However, forest floor C contents rarely exceed 40 Mg ha^{-1} in Denmark (Vejre et al. 2003).

In contrast to forests with a longer history, soil type did not seem to affect forest floor development in the first decades after afforestation. Sequestration of forest floor C differed little between the Danish Gejlvang and Vestskoven sites in spite of very contrasting mineral soils. Sandy, podsolized soils usually develop thick more-like forest floors (Vesterdal & Raulund-Rasmussen 1998; Vejre et al. 2003) because of slow decomposition and presence of few macrofauna species. The similar rates of C sequestration at the two Danish sites may be attributed to the high pH and nutrient availability in recently abandoned cropland soils. The legacy of cropland management in terms of liming and fertilization can probably maintain an active microflora and fauna in both soil types during the first decades. For instance, Jussy et al. (2002) and Compton & Boone (2000) found high rates of N mineralization several decades after agricultural abandonment and afforestation. This legacy of agriculture probably offsets the differences in forest floor accumulation that would otherwise have developed at sites with such difference in parent materials (Vesterdal & Raulund-Rasmussen 1998).

The afforestation literature confirms that forest floor development contributes most to sequestration of C during the first decades . Up to 38 Mg C ha^{-1} was sequestered over 40 years (0.94 Mg C ha^{-1} yr^{-1}) in *Pinus taeda* forest floors in warm-temperate climate (Richter et al. 1999). Comparable amounts of C accumulated (rates about 0.48 Mg C ha^{-1} yr^{-1}) in cooler temperate regions under Norway spruce and mixed broadleaves following 60-80 years of stand development (Hamburg 1984; Leth & Breuning-Madsen 1992).

The rate of forest floor development partly depends on aboveground litterfall. In the AFFOREST chronosequences, annual litterfall C content increased over the first decades following afforestation (Figure 2.3), and the development in litterfall C is consistent with canopy closure usually being complete after 20 years. After canopy closure, foliar litterfall tends to be constant, and foliar litterfall constitutes about 90% of total litterfall in young stands (Pedersen & Bille-Hansen 1999). Litterfall tended to be higher in oak than in spruce stands in Denmark, and Dutch oak stands had comparable litterfall C contents to Danish oak stands. However, rates of litterfall C representative of closed stands were reached at an earlier age than in Denmark. The much higher stand densities used in oak stands in the Netherlands (Fig. 1b) may explain this, as canopy closure is attained at an earlier age. In the oldest Swedish stands, litterfall C contents are relatively low which partly may be attributed to reduced production in these mature stands.

Afforested stands reached a relatively constant level of aboveground carbon input to the soil after about 20 years. However, the composition of litterfall may change with age, e.g. there may be more woody litter (branches and twigs) as stands mature. Another important source of carbon to soils is root litter. This source of carbon may be especially important for the C pool of the mineral soil. An assessment of root litter C was not included in the project, but in cool temperate forests, root litter was estimated to contribute between 20 and 77% of the total organic matter input to forest floors (Vogt et al. 1986).

4.3. Mineral soil carbon

The obvious accumulation of C in forest floors may well lead to expectations of increases in the mineral soil too. Nevertheless, there was no general pattern in change in mineral soil C stocks in the AFFOREST chronosequences (Figure 2.5a). In fact, there was a slight net C loss in mineral soils in chronosequences (30 years) at Vestskoven in Denmark, and not even the Swedish chronosequence covering 90 years exhibited a change. For the Swedish chronosequence it was also possible to do a paired plot analysis (Figure 2.6) to see if spatial variability possibly masked the influence of changed land use when using the chronosequence approach. However, there was still no net change detectable when comparing adjacent arable and forest soils along the chronosequence. This result for Sweden suggests that there is little reason to expect mineral soil C sequestration at all sites, even over a full rotation. At Vestskoven in Denmark the soil C contents in Ledøje Plantation (200-years-old) nevertheless indicate that the mineral soil may start to sequester C on a net basis in later stages following afforestation (Figure 2.5a). There was also evidence that soil C had started to build up in the 0-5 cm of the mineral soil at Vestskoven, however, not enough to compensate a decrease in soil C in 5-25 cm (Vesterdal et al. 2002).

On the other hand, mineral soil C storage increased with age in the Netherlands and at Gejlvang, Denmark already within 30-40 years. The mechanisms behind the site-dependent response to afforestation remain elusive, but high decomposition rates in the nutrient-rich soil at Vestskoven could be responsible for the small rate of C sequestration. Furthermore the allocation of the dead organic matter in soil profiles probably differ due to different activities of macrofauna species that mix forest floor material into the mineral soil. The large forest floor C sequestration rate and lack of C allocation to mineral soils in the Swedish chronosequence (Figures 5 and 6) could be a result of limited macrofauna activity.

Other chronosequence studies also show mixed evidence of mineral soil net C sequestration. In a recent review of soil C sequestration following afforestation by Post & Kwon (2000), most studies indicated net sequestration of C in mineral soils. Post & Kwon (2000) found an average rate of 0.34 Mg C ha^{-1} yr^{-1} for all studies regardless of climate, which is fairly similar to the rates estimated for chronosequences in the Netherlands and Gejlvang, Denmark (Table 2.4.). A meta-analysis by Guo & Gifford (2002) revealed that mineral soil C stocks increased by 18% following afforestation of former cropland. In other studies, however, the mineral soil did not function as a C sink, at least initially. In a review of global data on soil C sequestration after afforestation, Paul et al. (2002) found that soil C in 0-30 cm decreased during the first three decades and finally recovered to the pre-afforestation level by age 30. The dynamics in soil C at the Danish Vestskoven site might conform to a comparable pattern where it takes longer for mineral soil C to build up. Within a time perspective of a few decades losses of C derived from agriculture may possibly more than make up for the beginning accretion of forest-derived C (Bashkin & Binkley 1998; Binkley & Resh 1999; Paul et al. 2002; Vesterdal et al. 2002). In many cases there appears to be a transient period of decreasing soil C stores following afforestation. This period is characterized by redistribution of C within the soil profile rather than a net accretion in C stores. Such a pattern was found at Vestskoven, where there was a positive change in C in 0-5 cm and a negative change in 5-25 cm (Vesterdal et al. 2002). The reason for this initial development may be the placement of produced organic matter (Post & Kwon 2000). Compared to arable land use, inputs of C to the lower parts of the plow layer are reduced due to ceased tillage. At the same time the legacy of organic material from agriculture may decompose fast in the homogenized plow layer. In the very beginning of afforestation, litter input may also be quite small to the upper soil layers, thereby contributing to a temporal decline in mineral soil C (Romanya et al. 2000).

The former agricultural land use in the AFFOREST chronosequences was mainly cropland (cereals, potatoes) or rotation between cropping and pasture phases. This is important in comparisons with reviews where data were compiled from both afforested pastures and cropland. Pastures are known to have higher mineral soil C contents than cropland (Post & Kwon 2000). Consistently with this, Paul et al.

(2002) and Guo & Gifford (2002) both found in reviews that afforestation of former cropland resulted in mineral soil C sequestration while afforestation of pastures resulted in unchanged or decreasing C stocks. The mineral soil C sequestration found at Gejlvang, Denmark is in line with that for former cropland reported by Paul et al. (2002), whereas the rates for the other chronosequences are both lower (Vestskoven, DK and SE) and higher (NL). Obviously, other factors than former agricultural land-use such as soil type, climate and site preparation may influence the magnitude of change in soil C.

4.4. Total soil carbon

For the total soil compartment studied, i.e. forest floor and mineral soil 0-25 cm, we found that afforestation as a minimum resulted in unchanged soil C contents and in most cases led to net C sequestration (Figure 2.5b, Table 2.4). The review of global data by Paul et al. (2002) included 34 sites at which changes in C were measured in both forest floor and mineral soil. For these sites, the contribution of mineral soils 0-30 cm was limited (0.15 Mg C ha^{-1} yr^{-1}), but inclusion of forest floor C resulted in a mean rate of C sequestration of 0.46 Mg C ha^{-1} yr^{-1} for 0-30 cm. In the AFFOREST study from cool temperate climate all chronosequences but one indicated C sequestration rates that were 1.3 to 2.7 times higher for the total soil compartment studied.

The rate of input of tree litter and the rate of decomposition of the litter together determine the C content of the soil. The organic matter input to arable soils is often said to be lower than to forest soils because a greater amount of the produced biomass is harvested (Bouwman & Leemans 1995). However, in conventional Danish cropland (barley and wheat) the input from aboveground biomass is about 2.4 Mg C ha^{-1} yr^{-1} (Olesen et al. 2001), whereas the annual input of C to soils with litterfall approaches about 2 Mg C ha^{-1} yr^{-1} in many cool temperate forests (Bastrup-Birk et al. 2003; Vogt et al. 1986), assuming 50% C in litterfall. Comparable amounts of C were also transferred to the soil with aboveground litterfall in the AFFOREST chronosequences (Figure 2.3), although litterfall C varied quite a lot between chronosequences. Belowground litter input from root systems is an important contributor to mineral soil C pools, but little is known about general differences between cropland and forestry. Even if the input of organic matter may be comparable, rates of decomposition in arable soils can be higher due to higher soil temperatures and moisture (Bouwman & Leemans 1995), more decomposable litter types (Schimel et al. 1985), and increased accessibility of organic matter to microbial attack because of frequent soil cultivation (Voroney et al. 1981; Denef et al. 2004).

The AFFOREST chronosequence studies did not reveal large differences in total soil C sequestration between oak and Norway spruce although oak sequestered less C in forest floors than Norway spruce. This may partly be attributed to the short period where it was possible to compare species. To our knowledge no other studies addressed species effects in afforestation using a chronosequence approach. General studies of species effects on forest floors usually agree that conifers store more C in forest floors than do broadleaves (France et al. 1989; Vesterdal & Raulund-Rasmussen 1998), but results for mineral soil C are scarce and often contradictory (Binkley 1995; Hagen-Thorn et al. 2004). Paul et al. (2003) also found no clear difference in mineral soil C sequestration after 25-50 years between afforestation with deciduous and coniferous species using isotopic C dating.

4.5. Carbon sequestration of the afforested ecosystem

One of the objectives of AFFOREST was to determine the total C sequestration and the relative contributions of soil and biomass components of the new forest ecosystems. Roughly one third of the total C was sequestered in the soil and two thirds were sequestered in biomass; thus the allocation of C to woody biomass and soil occurred approximately at a ratio of 2:1 (woody biomass : soil) for all chronosequences combined (Figure 2.7). However, this ratio was largely driven by data from the long Swedish chronosequence, and in individual chronosequences, the contribution of soils ranged from 0 to 31% (Table 2.5.). The change in ecosystem C content was primarily caused by the growth of trees as supported by other studies (Post & Kwon 2000; Paul et al. 2002).

Studies of both biomass and soil C sequestration following afforestation are scarce, and no other European studies on former cropland were available for comparison with the AFFOREST chronosequences. Several studies have been conducted in the eastern United States on afforestation and natural succession following abandonment of agriculture since this land-use change was widespread after the western United States was opened up in late 19th century. Table 2.5 therefore lists rates of C sequestration in forest floors, mineral soils and biomass and the relative contribution of soils for the AFFOREST chronosequences for comparison with similar studies in the eastern United States and a single European study from the Italian Alps. As for the AFFOREST chronosequences, most of the other studies found that simple linear relationships best described the development in ecosystem C stocks. This facilitates the comparison of sequestration rates between studies.

Table 2.5. *Rates of C sequestration following afforestation in the temperate region. Rates of C sequestration may not add up to ecosystem total due to different treatment of non-significant regressions and unlisted pools (e.g. woody debris).*

Forest type	Time period (yr)	Rate of C sequestration (Mg ha^{-1} yr^{-1})				Soil contrib. to ecosystem C seq. (%)	Mineral soil contrib. to total soil C seq. (%)	Source
		Plant biomass	Forest floor	Mineral soil (sample depth in cm)	Eco-system total			
Vestskoven, DK, Oak	31	2.72	0.08	-0.47 (25)	2.36	~0	~0	This study
Sellingen, NL Oak, N. spruce	29	4.55*	0.34	1.14 (25)	7.14*	20	77	This study
Vestskoven, DK Norway spruce	32	4.61	0.35	-0.47 (25)	4.46	~0	~0	This study
Gejlvang, DK Norway spruce	41	3.76	0.43	0.66 (25)	4.84	23	61	This study
Tsjh, SE Norway spruce	92	1.20	0.65	0.09 (25)	2.36	31	14	This study
Rhode Island, US White pine	115	1.53	0.37	0.15 (70)	2.10	24	29	Hooker & Compton (2003)

Table 2.5. Continued.

New Hampshire Mixed dec./con.	65	1.67	0.40	-0.05 (50)	1.95	18	~0	Hamburg (1984) in Hooker & Compton (2003)
South Car., US Loblolly pine	35	4.26	0.95	0.04 (60)	5.24	19	4	Richter et al. (1999)
Minn., US Oak	39	0.96	0.12	0.13 (25)	1.21	21	52	Johnston et al. (1996)
Virginia, US Loblolly pine	47	3.03	0.23	0.20 (100)	3.52	12	47	Schiffman & Johnson (1989)
Michigan, US Red pine	38	$1.91^{\dagger}$	0.28	~0 (10)	2.18	13	~0	Pregitzer & Palik (1997)
Northern Italy, Norway spruce	62	$2.73^{\ddagger}$	0.36	$\sim 0^{\S}$ (A horizon)	3.18	13	~0	Thuille et al. (2000)

‡ Stem biomass only.

§ Former pasture soil, no linear trend.

**Biomass and ecosystem rates apply to 18-year period only.*

†Aboveground biomass only.

Other studies largely agree with the relative contribution of soils in spite of various methods applied for estimating changes in C stocks, including the soil depth considered. A contribution of soils of between 15 and 20% of total ecosystem storage appears to be the common picture. However, several studies in eastern United States had lower relative soil contribution because there were no changes in mineral soil C stock. The Swedish chronosequence was outstanding in terms of its large soil C sequestration almost solely caused by forest floor buildup. It follows from Table 2.5 that the relative storage by mineral soils – like in the AFFOREST chronosequences – also varied tremendously among other studies. The contribution of mineral soils relative to the total soil ranged from 0 to 52% in other studies and from 0 to about 80% in the AFFOREST chronosequences. The relative contribution of soils and also the partitioning of C between forest floor and mineral soil are probably very influenced by factors like climate, soil type, tree species, former agricultural land use, and sampling methodology. These factors all vary between studies and with this in mind the relatively stable contributions of soils to ecosystem sequestration are remarkable.

The soil type of former cropland may influence the relative effect of afforestation on ecosystem C sequestration. For biomass, soils with high nutrient availability and fine texture would build up the highest C stocks, but it is less easy to predict which soil types will be most conducive to C sequestration in the long term. Mineral soils are potentially a better storage medium than forest floors or biomass with respect to ecosystem disturbance. A large amount of C in mineral soils is bound in organo-mineral complexes that protect C from microbial oxidation (Six et al. 2002; Sollins et al. 1996; Van Veen & Kuikman 1990) whereas forest floor C is more labile. Therefore it would be particularly interesting to select soil types with high probability of increasing the mineral soil C stock. Clay soils have been suggested to be more conducive to C sequestration because of formation of such stable clay-organic matter complexes (Hagedorn et al. 2001), but the synthesis of field studies by Paul et al. (2002) did not support this in general. Several authors suggested that changes in soil C stocks are affected by the organic matter input in coarse-textured soils, and by clay mineralogy in fine textured soils (Christensen 1992; Hassink 1995; Quiroga et al. 1996). If afforestation increases the input of organic matter to soils, this hypothesis suggests that coarse textured soils would experience a greater relative increase in C content in response to afforestation.

In fact, the two Norway spruce chronosequences on contrasting soils in Denmark indicated that sandy, poor soils would sequester more C. The rate of C sequestration was somewhat higher in forest floor at the sandy soil, but the difference was most marked in the mineral soil (Figure 2.5a). Also the Dutch chronosequences with sandy texture was remarkable in the relatively large C sequestration in mineral soils. This pattern suggests that sandy, poor soils with slow decomposition are more conducive to C sequestration in the north European region. The rate of C sequestration for the ecosystem was also high for these sites, as biomass C sequestration was not lower at the soils with poor parent materials for stands younger than 45 years. According to parent materials (see Appendix 1), tree growth within the Danish chronosequences would be expected to be faster on the calcareous, nutrient-rich loamy till at Vestskoven than at the nutrient-poor, acidic glaciofluvial sand at Gejlvang. The legacy of former fertilization and liming in afforested ecosystems is probably responsible for larger nutritional homogeneity than expected based on parent materials. Thus, differences in biomass production are leveled out during the first decades, but sequestration of C in soils is still higher at the poor site, keeping ecosystem C sequestration high. The starting point in soil C stocks possibly also influence the sequestration potential. Former cropland with low initial soil C content as in Gejlvang might be more prone to increase in C stock, which could explain the difference in mineral soil C sequestration to the other Danish chronosequences. The Dutch soils, however, increased in mineral soil C in spite of high initial C stocks. The Dutch sites are less well drained than other sites as the groundwater table can be within 50 cm depth during winter months. Both the current drainage regime and pre-drainage hydrological conditions may have contributed to the high mineral soil C contents and C sequestration rates encountered in the Dutch chronosequences. The more wet soil conditions in the Dutch stands may also be the reason why the general relationship between stand age and soil C improved when Dutch stands were excluded (Figures 2.7a-b).

The AFFOREST chronosequence studies have contributed to quantification of C sequestration following afforestation in north-western Europe. No other studies within the region previously provided similar comprehensive quantitative data on C sequestration following afforestation. Still, conclusions must be tempered by some facts. For instance, sampling and analysis methods differed among the three countries, however, less than seen in compilations of independent studies. Moreover, only one chronosequence included stands older than 40 years and there was one single stand representing an age of about 200 years. As discussed in Chapter 1, there are certain drawbacks to chronosequence studies. Other sources of variation between stands cannot be distinguished from stand age, e.g., changes in agricultural practices since the oldest stand was planted and spatial variability. As pointed out by Yanai et al. (2000), these other sources of variability may lead to erroneous conclusions about the influence of stand age. Keeping the shortcomings in mind, chronosequence studies as in AFFOREST provide a first valuable estimate of changes in soil C before resampling of soils is possible. Repeated sampling at least ten years after the first sampling is planned to test the results obtained in the project. This will provide real evidence of the directional change for each stand in the chronosequences and will further test the predictive value of the chronosequence approach. Chronosequence studies also reveal little about processes responsible for changes in soil C stocks. Chronosequence studies combining pool changes and assessment of isotopic C fractions (e.g. Richter et al. 1999; Del Galdo et al. 2003) enable more insight in the dynamics of old agricultural C and new forest-derived C in soils. Lastly, to address the sustainability of sequestered soil C, i.e. the quality of C pools in sense of stability, there is a need for further studies of C fractions in the mineral soil.

5. CONCLUSIONS

The rate of biomass C sequestration was relatively similar in all chronosequences. A possible effect of parent material was not evident during the first 40-50 years. Parent material effects may be masked by the soil enrichment, which is a legacy of former agriculture. Biomass C sequestration differed more after 40-50 years, probably due to different management, tree species-specific growth patterns and less influence of former fertilization.

For soils there was good evidence that afforestation of former cropland leads to constant or increasing total soil carbon storage. Forest floor build up, and the associated C sequestration, was a general feature, but the contribution of mineral soil to total soil C sequestration differed among chronosequences. In the short term (30 years), tree species had little influence on total soil C sequestration, but in stands younger than 40 years, C sequestration in the forest floor was higher under spruce than under oak. Afforestation of nutrient-poor sandy soils results in larger forest floor C sequestration and larger total soil C sequestration than afforestation of nutrient-rich, clayey soils.

For the afforested ecosystem the general contribution of soils to C sequestration was about one third of the total C sequestration in biomass and soil. The contribution of soil varied among chronosequences from none to 31%. Rates in similar studies from eastern United States were around 15-20%. Total C sequestration was higher in afforested Norway spruce than in afforested oak in the short term (30 years). Soil type did not clearly influence the rate of ecosystem C sequestration in the short term (30-40 years).

We conclude that while biomass C sequestration initially was relatively comparable, there is still little support for generalizations regarding the potential for soil C sequestration. However, based on the AFFOREST chronosequence studies and results in the literature it is safe to conclude that soils play a minor quantitative role relative to biomass of trees in the short and possibly also the long term. Still, the contribution of soils around 20% warrants quantification of this C pool.

As a contribution to mitigation of atmospheric CO_2 concentrations, countries should report changes in C stocks in living biomass, dead organic matter and mineral soils following land-use change in national greenhouse gas budgets (IPCC 2003). The work on land use change in AFFOREST primarily focused on improving the basic knowledge of contributions of afforested cropland to sequestration of C. These results will hopefully help to bridge the gap between current knowledge and policy demands.

6. REFERENCES

Bashkin, M. A., & Binkley, D. (1998). Changes in soil carbon following afforestation in Hawaii. *Ecology, 79,* 828-833.

Bastrup-Birk, A., Hansen, K., Ro-Poulsen, H., Jørgensen, B. B., Mikkelsen, T., Pilegaard, K., & Bille-Hansen, J. (2003). Biomasse og produktion. Næringsstofkredsløb i skove - Ionbalanceprojektet (Ed. K. Hansen). *Forest and Landscape Research, 33,* 69-96. Ministry of the Environment, Danish Forest and Landscape Research Institute, Hørsholm, Denmark. (In Danish).

Billings, W. D. (1938). The structure and development of shortleaf pine stands and certain associated physical properties of soil. *Ecological Monographs, 8,* 437-499.

Binkley, D. (1995). *The influence of tree species on forest soils: Processes and patterns.* Proceeding of the Trees and Soil Workshop. Lincoln University, 28 February-2 March 1994 (Eds. D. J. Mead & I. S. Cornforth), pp. 1-33. Canterbury, New Zealand: Lincoln University Press.

Binkley, D., & Resh, S. C. (1999). Rapid changes in soils following Eucalyptus afforestation in Hawaii. *Soil Science Society of America Journal, 63,* 222-225.

Bouwman, A. F., & Leemans, R. (1995). The role of forest soils in the global carbon cycle. Carbon forms and functions in forest soils (Eds. W. F. McFee & F. M. Kelly), pp. 503-525. Madison, Wisconsin, USA: Soil Science Society of America.

Brandel, G. (1990). Volume functions for individual trees. Scots pine (Pinus sylvestris), Norway spruce (Picea abies) and birch (Betula pendula & Betula pubescens). Swedish University of Agricultural Sciences, Department of Forest Yield Research, Report No. 26. 183 pp. (In Swedish with English summary).

Christensen, B. T. (1992). Physical fractionation of soil and organic matter in primary particle size and density separates. *Advances in Soil Science, 20,* 1-89.

Compton, J. E., & Boone, R. D. (2000). Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. *Ecology, 81,* 2314-2330.

Del Galdo, I., Six, J., Peressotti, A., & Cotrufo, M. F. (2003). Assessing the impact of land-use change on soil C sequestration in agricultural soils by means of organic matter fractionation and stable C isotopes. *Global Change Biology, 9,* 1204-1213.

Denef, K., Six, J., Merckx, R., & Paustian, K. (2004). Carbon Sequestration in Microaggregates of No-Tillage Soils with Different Clay Mineralogy. *Soil Science Society of America Journal, 68,* 1935-1944.

Dewar, R. C., & Cannell, M. G. R. (1992). Carbon sequestration in the trees, products and soils of forest plantations: an analysis using UK examples. *Tree Physiology, 11,* 49-71.

Dick, W. A. (1983). Organic carbon, nitrogen, and phosphorus concentrations and pH in soil profiles as affected by tillage intensity. *Soil Science Society of America Journal, 47,* 102-107.

Dziadowiec, H. (1987). The decomposition of plant litter fall in an oak-linden-hornbeam forest and an oak-pine mixed forest of the Bialowieza National Park. *Acta Societatis Botanicorum Poloniae, 56,* 169-185.

France, E. A., Binkley, D., & Valentine, D. (1989). Soil chemistry changes after 27 years under four tree species in southern Ontario. *Canadian Journal of Forest Research, 19,* 1648-1650.

Garcia-Oliva, F., & Masera, O. R. (2004). Assessment and measurement issues related to soil carbon sequestration in land-use, land-use change, and forestry (LULUCF) projects under the Kyoto Protocol. *Climatic Change, 65,* 347-364.

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology, 8,* 345-360.

Hagedorn, F., Maurer, S., Egli, P., Blaser, P., Bucher, J. B., & Siegwolf, R. (2001). Carbon sequestration in forest soils: effects of soil type, atmospheric CO2 enrichment, and N deposition. *European Journal of Soil Science, 52,* 619-628.

Hagen-Thorn, A., Callesen, I, Armolaitis, K., & Nihlgård, B. (2004). The impact of six European tree species on the chemistry of mineral topsoil in forest plantations on former agricultural land. *Forest Ecology and Management, 195,* 373-384.

Hamburg, S. P. (1984). Effects of forest growth on soil nitrogen and organic matter pools following release from subsistence agriculture. Forest Soils and Treatment Impacts (Ed. E. L. Stone), pp. 145-158. University of Tennessee, Knoxville.

Hassink, J. (1995). Decomposition rate constants of size and density fractions of soil organic matter. *Soil Science Society of America Journal, 59,* 1631-1635.

Hoekstra, C., & Poelman, J. N. B. (1992). Dichtheid van gronden gemeten aan de meest voorkomende bodemeenheden in Nederland. Stiboka Rapport 1582, Wageningen, The Netherlands. (In Dutch).

Homann, P. S., Sollins, P., Chappell, H. N., & Stangenberger, A. G. (1995). Soil organic carbon in a mountainous, forested region: Relation to site characteristics. *Soil Science Society of America Journal, 59,* 1468-1475.

Hooker, T. D., & Compton, J. E. (2003). Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. *Ecology Applications, 13,* 299-313.

Illerup, J. B, Lyck, E., Nielsen, M., Winther, M., Mikkelsen, M. H., Hoffmann, L., Gyldenkærne, S., Sørensen, P., Vesterdal, L., Fauser, P., & Thomsen, M. (2005). Denmark's National Inventory Report 2005. Submitted under the United Nations Framework Convention on Climate Change 1990-2003. Ministry of Environment, Denmark, National Environmental Research Institute. http://cdr.eionet.eu.int/dk/Air_Emission_Inventories/Submission_UNFCCC/colql9meg/envql9mfg/NIR_April_2005_DK.doc

IPCC. (2003). Good Practice Guidance for Land Use, Land-Use Change and Forestry. http://www.ipcc-nggip.iges.or.jp/public/gpglulucf/gpglulucf.htm

Jansen, J. J., Sevenster, J., & Faber, P. J. (1996). Opbrengsttabellen voor belangrijke boomsoorten in Nederland. IBN-rapport 221, Wageningen. (In Dutch).

Johannsen, V. K., Jørgensen, B. B., & Morsing, M. (2004). Skovstatistik feltinstruks 2004. Skov & Landskab, KVL. 173 pp. (In Danish).

Johnson, D. W. (1992). Effects of forest management on soil carbon storage. *Water, Air, and Soil Pollution, 64,* 83-120.

Johnston, M. H., Homann, P. S., Engstrom, J. K., & Grigal, D. F. (1996). Changes in ecosystem carbon storage over 40 years on an old-field/forest landscape in east-central Minnesota. *Forest Ecology and Management, 83,* 17-26.

Jug, A., Makeschin, F., Rehfuess, K. E., & Hofmann-Schielle, C. (1999). Short-rotation plantations of balsam poplars, aspen and willows on former arable land in the Federal Republic of Germany. III. Soil ecological effects. *Forest Ecology and Management, 121,* 85-99.

Jussy, J. H., Koerner, W., Dambrine, E., Dupouey, J. L., & Benoit, M. (2002). Influence of former agricultural land use on net nitrate production in soils. *European Journal of Soil Science, 53,* 367-374.

Karlsson, K. (1998). A strategy for reducing the number of sample trees when estimating stand volume on sample plots. Swedish University of Agricultural Sciences, Department of Forest Yield Research, report 43, 18 pp.

Krogh, L., Nørgaard, A., Hermansen, M., Greve, M. H., Balstrøm, T., & Breuning-Madsen, H. (2003). Preliminary estimates of contemporary soil organic carbon stocks in Denmark using multiple datasets and four scaling-up methods. *Agriculture, Ecosystems & Environment, 96,* 19-28.

Kurmies, B. (1949). Humusbestimmung nach dem Biochromatverfahren ohne Kaliumjodid. *Zeitschrift für Pflanzenernährung, Dungung und Bodenkunde, 44,* 121-125.

Leth, S., & Breuning-Madsen, H. (1992). Changes in soil profile development and nutrient status due to afforestation of agricultural land. *Danish Journal of Geography, 92,* 70-74.

Lettens, S., Van Orshoven, J., van Wesemael, B., & Muys, B. (2004). Soil organic and inorganic carbon contents of landscape units in Belgium derived using data from 1950 to 1970. *Soil Use and Management, 20,* 40-47.

Madsen, S. F. (1987). Vedmassefunktioner ved forskellige aflægningsgrænser og nøjagtighedskrav for nogle vigtige danske skovtræarter. *Det Forstlige Forsøgsvæsen i Danmark, 41*, 47-242. (In Danish with English summary: Volume equations for some important Danish tree species. Standard and form class equations. Total and merchantable volumes.)

Madsen, S. F., & Heusérr, M. (1993). Volume and stem-taper functions for Norway spruce in Denmark. *Forest & Landscape Research, 1,* 51-78.

Malhi, Y, Baldocchi, D. D., & Jarvis, P. G. (1999). The carbon balance of tropical, temperate and boreal forests. *Plant, Cell and Environment, 22,* 715-740.

Marklund, L. G. (1988). Biomass functions for pine, spruce and birch in Sweden. Swedish University of Agricultural Sciences, Umeå. Dept. of Forest Survey, Report 45, 73 pp. ISSN 0348-0496. (In Swedish with English summary.)

Matejovic, I. (1993). Determination of carbon, hydrogen, and nitrogen in soils by automated elemental analysis (dry combustion method). *Communications in Soil Science and Plant Analysis, 24,* 2213-2222.

McNabb, D. H., Cromack, K., & Fredriksen, R. L. (1986). Variability of nitrogen and carbon in surface soils of six forest types in the Oregon Cascades. *Soil Science Society of America Journal, 50,* 1037-1041.

Nabuurs, G.-J., & Mohren, G. M. J. (1995). Modelling analysis of potential carbon sequestration in selected forest types. *Canadian Journal of Forest Research, 25,* 1157-1172.

Näslund, M. (1936). Skogsförsöksanstaltens gallringsförsök i tallskog. *Meddelanden från Statens Skogsförsöksanstalt, 29,* 1-169. (In Swedish).

Olesen, J. E., Andersen, J. M., Jacobsen, B. H., Hvelplund, T., Jørgensen, U., Schou, J. S., Graversen, J., Dalgaard, T., & Fenhann, J. V. (2001). Kvantificering af tre tiltag til reduktion af landbrugets emission af drivhusgasser. DJF Rapport Markbrug nr. 48, Danish Institute of Agricultural Sciences, Foulum, Denmark. (In Danish with English summary).

Paul, E. A., Morris, S. J., Six, J., Paustian, K., & Gregorich, E. G. (2003). Interpretation of soil carbon and nitrogen dynamics in agricultural and afforested soils. *Soil Science Society of America Journal, 67,* 1620-1628.

Paul, K. I., Polglase, P. J., Nyakuengama, J. G., & Khanna, P. K. (2002). Change in soil carbon following afforestation. *Forest Ecology and Management, 168,* 241-257.

Pedersen, L. B., & Bille-Hansen, J. (1999). A comparison of litterfall and element fluxes in even aged Norway spruce, sitka spruce and beech stands in Denmark. *Forest Ecology and Management, 114,* 55-70.

Post, W. M., & Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology, 6,* 317-327.

Powlson, D. S., Smith, P., Coleman, K., Smith, J. U., Glendining, M. J., Körschens, M., & Franko, U. (1998). A European network of long-term sites for studies on soil organic matter. *Soil and Tillage Research, 47,* 263-274.

Pregitzer, K. S., & Palik, B. J. (1997). Change in ecosystem carbon 46 years afte establishing red pine (Pinus resinosa Ait.) on abandoned agricultural land in the Great Lakes Region. In Soil organic matter in temperate agroecosystems: long-term experiments in North America (Eds. E. A. Paul, K. Paustian, E. T. Elliott & C. V. Cole), pp. 263-270. New York: CRC Press.

Quiroga, A. R., Buschaiazzo, D. E., & Peinemann, N. (1996). Soil organic matter particle size fractions in soils of the semiarid Argentinian pampas. *Soil Science, 161,* 104-107.

Richter, D. D., Markewitz, D., Trumbore, S. E., & Wells, C. G. (1999). Rapid accumulation and turnover of soil carbon in a re-establishing forest. *Nature, 400,* 56-58.

Rodriguez-Murillo, J. C. (2001). Organic carbon content under different types of land use and soil in peninsular Spain. *Biology and Fertility of Soils, 33,* 53-61.

Romanya, J., Cortina, J., Falloon, P., Coleman, K., & Smith, P. (2000). Modelling changes in soil organic matter after planting fast-growing Pinus radiata on Mediterranean agricultural soils. *European Journal of Soil Science, 51,* 627-641.

Römkens, P. F. A. M., van der Pflicht, J., & Hassink, J. (1999). Soil organic matter dynamics after the conversion of arable land to pasture. *Biology and Fertility of Soils, 28,* 277-284.

Rosenqvist, L., & Johansson, M.-B. (2005). Carbon sequestration and fluxes of dissolved organic carbon in a chronosequence of Norway spruce forest planted on former arable land. Manuscript in preparation.

SAS Institute (1993). SAS/STAT User's Guide Rel. 6.07. Cary, North Carolina, USA: SAS Institute Inc.

Schiffman, P. M. & Johnson, W. C. (1989). Phytomass and detrital carbon storage during forest regrowth in the southeastern United States Piedmont. *Canadian Journal of Forest Research, 19,* 69-78.

Schimel, D. S., Coleman, D. C., & Horton, K. A. (1985). Soil organic matter dynamics in paired rangeland and cropland toposequences in North Dakota. *Geoderma, 36,* 201-214.

Six, J., Conant, R. T., Paul, E. A., & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil, 241,* 155-176.

Smith, K.A. (1999). After the Kyoto Protocol: Can soil scientists make a useful contribution? *Soil Use and Management, 15,* 71-75.

Soil Survey Staff (1992). Keys to Soil Taxonomy, 5th Edition. SMSS technical monograph no. 19. Blacksburg, Virginia: Pocahontas Press, Inc.

Sollins, P., Homann, P., & Caldwell, B. A. (1996). Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma, 74,* 65-105.

Thuille, A., Buchmann, N., & Schulze, E.-D. (2000). Carbon stocks and soil respiration rates during deforestation, grassland use and subsequent Norway spruce afforestation in the Southern Alps, Italy. *Tree Physiology, 20,* 849-857.

Van Veen, J. A., & Kuikman, P. J. (1990). Soil structural aspects of decomposition of organic matter by micro-organisms. *Biogeochemistry, 11,* 213-233.

Vejre, H., Callesen, I., Vesterdal, L., & Raulund-Rasmussen, K. (2003). Carbon and nitrogen in Danish forest soils - contents and distribution determined by soil order. *Soil Science Society of America Journal*, *67,* 335-343.

Vesterdal, L., & Raulund-Rasmussen, K. (1998). Forest floor chemistry under seven tree species along a soil fertility gradient. *Canadian Journal of Forest Research*, *28,* 1636-1647.

Vesterdal, L., Ritter, E., & Gundersen, P. (2002). Change in soil organic carbon following afforestation of former arable land. *Forest Ecology and Management, 169,* 137-143.

Vogt, K. A., Grier, C. C., & Vogt, D. J. (1986). Production, turnover, and nutrient dynamics of above- and belowground detritus of world forests. *Advances in Ecological Research, 15,* 303-377.

Voroney, R. P., Van Veen, J. A., & Paul, E. A. (1981). Organic C dynamics in grassland soils. 2. Model validation and simulation of the long-term effects of cultivation and rainfall erosion. *Canadian Journal of Soil Science, 61,* 211-224.

Westman, C. J. (1995). A simple device for sampling of volumetric forest soil cores. *Silva Fennica, 29,* 247-251.

Winjum, J. K., & Schroeder, P. E. (1997). Forest plantations of the world: their extent, ecological attributes, and carbon storage. *Agricultural and Forest Meteorology, 84,* 153-167.

Yanai, R. D., Arthur, M. A., Siccama, T. G., & Federer, C. A. (2000). Challenges of measuring forest floor organic matter dynamics: Repeated measures from a chronosequence. *Forest Ecology and Management, 138,* 273-283.