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Effects of afforestation and deforestation on boreal soil carbon stocks—Comparison of measured C stocks with Yasso07 model results

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ABSTRACT

Land use changes like afforestation and deforestation are known to affect stocks of carbon in soils. We measured changes in soil carbon stocks in afforested and deforested sites. Repeated measurements were made at six sites which had been afforested with three different tree species 17–18 years before this sampling. The deforestation sites consisted of six field soils that were taken to cultivation 1–200 years before the sampling and adjacent forest sites representing the same soil types as the fields. The performance of the Yasso07 model in predicting the soil carbon stock changes in afforestation and deforestation was evaluated by simulating the changes in the carbon stocks and comparing the measured and simulated results for these sites. The mean observed 20-year carbon stock change after the land use change was −9% in the afforested sites and −19% in the deforested sites. The decrease in the mean carbon stock after afforestation was most pronounced during the first 9–10 years and was probably due to low rates of litter production in the early growth phase of the forests. The stock change in deforestation was lowest in fields with grasses as the main crop and highest in cereal monoculture. The simulation results were well in accordance with the measured carbon stocks on most sites.

1. Introduction

Taking virgin soils (native grassland or forest) to cultivation has usually resulted in a decline in soil organic carbon (C) and net release of CO_2 to the atmosphere (Davidson and Ackerman, 1993; Ellert and Gregorich, 1996; Guo and Gifford, 2002; Johnson, 1992; Paul et al., 2002; Post and Kwon, 2000). Due to the increased aeration of the ploughed layer and changes in the carbon input in plant residues, the surface layers of cultivated agricultural soils may have 30–35% less C than forest soil, as was observed e.g. in Ontario and Quebec (Ellert and Gregorich, 1996; Martel and Deschenes, 1976). On the other hand, converting arable land to forest has been shown to increase soil C storage (Jandl et al., 2007; Johnson, 1992; Post and Kwon, 2000). The magnitude of the changes in soil C stocks after land use conversions depends on environmental, biogeochemical and management factors (Franzluebbers et al., 2001; Lal, 1997; Post and Kwon, 2000). Climatic factors play an important role in the long-term development of soil C pools (Paul et al., 2002). Generally, the influence of climate on soil C content is reflected in the balance between C inputs from vegetation (litterfall and rhizodeposition) and C losses via decomposition. The turnover of soil organic matter depends on the chemical quality of the C compounds, site conditions, and soil properties.

In Finland, almost all agricultural land has been cleared from forests. Both clearing of new agricultural land from forests, and afforestation of agricultural land, have been major land use changes. Between the years 1970–2005, altogether 250 600 ha of forests were cleared for agricultural fields. During the same time span, 244 300 ha of agricultural fields were afforested. Therefore, both processes could have a significant effect on the national soil carbon budget. Relatively little information is available on the effects of land use change on soil carbon stocks in the boreal climatic zone.

The aim of this study was to quantify C stock changes in the cases of land use conversions from forest to agriculture and afforestation of former agricultural fields in Finland. At the afforested sites, the effect of tree species on the measured C stock change over the 17–18 years period after afforestation was also studied. The measured soil C stocks were compared with the results of the Yasso07 soil carbon model, in order to test the applicability of the model for predicting effects of land use changes on soil C stocks in boreal conditions. Yasso07 is a dynamic soil carbon model developed from the earlier Yasso model (Liski et al., 2005). In Finland, the Yasso model is used for estimating the C stock changes in forest soils as part of the Finnish greenhouse gas inventory (Statistics Finland, 2010). Yasso07 calculates soil carbon stock based on litter input (quantity and chemical quality of litter) and climate (mean annual temperature, mean temperature of the

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coldest and warmest month, and mean annual precipitation) (Tuomi et al., 2009). Because the model is developed for forest soils, model validation against measurements from afforested and deforested sites is needed, before application to land use change situations in practice. We hypothesized that although there are differences in the soil conditions and management between forest and agricultural soils, climate and quality of C inputs would be the main factors determining decomposition rates also in land use change situations.

2. Materials and methods

2.1. Study sites

The six afforestation experiments were located in southern and central Finland (Fig. 1). The mean effective temperature sum (threshold 5 °C) of the study sites varies from 1000 °C days (central Finland) to 1300 °C days (southern Finland), and the mean annual precipitation varies from 561 mm (central Finland) to 620 mm (southern Finland). The altitude of the sampling sites varies slightly, between 20 and 150 m above the sea level. The soils were either fine-textured, consisting of clayey soils, or medium-textured, consisting of fine sandy soils (Table 1). The soils were derived from glacial deposits. Until the establishment of the experiments, the sites had been in agricultural use for decades, and were cultivated for cereals. The experiments were established as a randomized block design with two or three replications in 1990 or 1991. The plot size ranged from 100 m^2 to 625 m^2. Bare-rooted 3- or 4-year-old Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* L.) seedlings and 1-year-old containerized silver birch (*Betula pendula* Roth) seedlings were planted in these experiments to a density of 3000 seedlings ha^{-1}. Due to complete soil preparation with rotavation and harrowing before planting, the soil was free from vegetation when planting the seedlings. After planting, herbicides were applied for weed control.

For the study on the effects of deforestation on soil carbon stock, we chose eight fields cleared from forest in southern Finland (Fig. 1). We aimed at finding sites with some of the original forest on similar soil type still available next to the fields. However, there were some site pairs where the soil texture differed between the forest and field. The sites represented different soil types and cultivation history (Table 2) and sites 7–9 were adjacent fields of different age. The farmers were interviewed for the cultivation history of the fields.

2.2. Sampling

The afforestation experiments were sampled three times during the experiment: at the time of the establishment, in 1999 and in 2008. Thus, at the time of the sampling the plantations were 0, 8–9 or 17–18 years old. At the time of the establishment, the soil samples were taken from circular sample plots (50 m^2) from soil depths of 0–10 cm and 30–40 cm. At three sites (sites 2, 3 and 6) the soil samples were taken from five points on each three plots planted with Norway spruce and bulked into one composite sample for each soil depth and sample plot. At the other three sites, the soil samples were taken from nine points and bulked into one composite sample for each soil depth and experiment. In 1999 and in 2008 the soil samples were taken from five points on each plot (one in the centre and four at the perimeter) from mineral soil to a depth of 40 cm. In the laboratory, the soil cores were divided into the 0–10 cm, 10–20 cm, 20–30 cm and 30–40 cm mineral soil layers and then bulked to form one series of composite samples representative for each plot and soil layer. In 1999 the soil samples were taken with a hammer driven soil

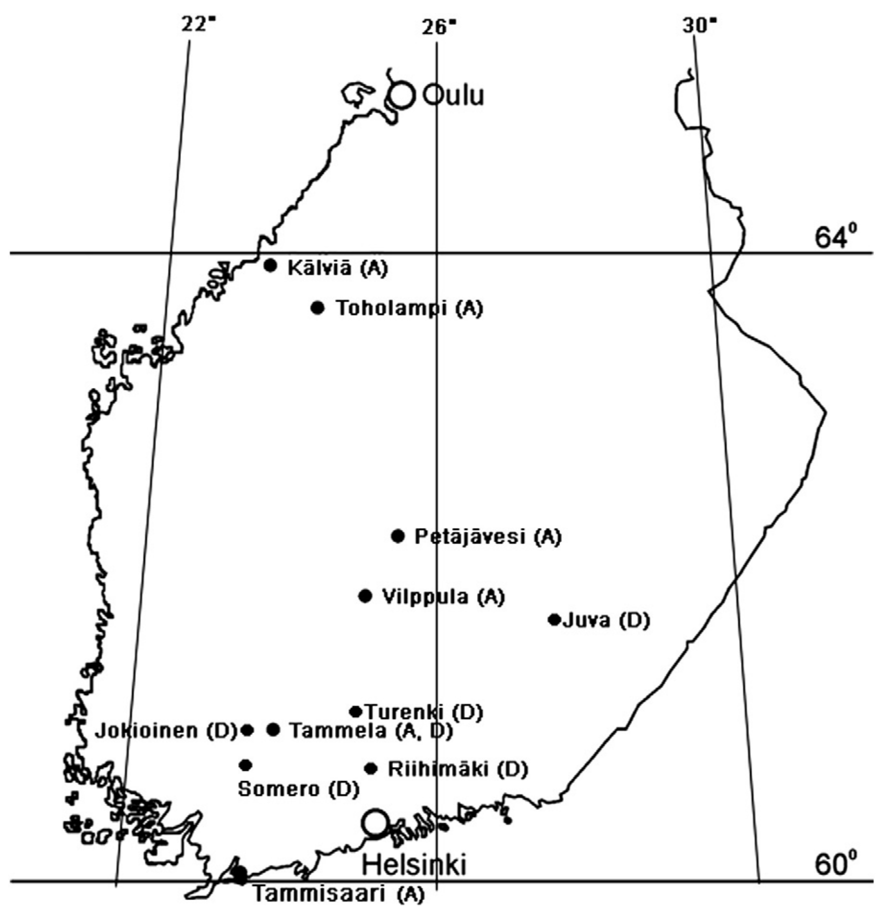

Fig. 1. Location of the afforestation (A) and deforestation (D) sites.

Table 1
Site characteristics of the afforested sites.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Location	Tammisaari	Tammela	Vilppula	Petäjävesi	Kälviä	Toholampi
Afforestation year	1991	1991	1991	1990	1990	1990
Former crop	Barley	Oats	Barley	Oats	Oats	Barley
Tree species[a]	S, P, B	S, B	S, P, B	S, P, B	S	S, P, B
Soil type (FAO)	Humic Cambisol	Haplic Podzol	Haplic Podzol	Haplic Podzol	Haplic Podzol	Eutric Regosol
Clay[b,c] (%)	9.5	4.8	18.8	8.5	3.2	28.8
Silt (%)	16.5	14.4	29.5	15.9	9.4	53.6
Sand (%)	74.0	80.8	51.7	75.6	87.4	17.6
Dry bulk density [c] ($g\ cm^{-3}$)	1.14	1.22	1.42	1.38	1.13	1.17
C/N[c]	14.6	15.6	15.7	14.4	23.1	11.7

Proportion of fine earth was ~100% in each soil.
[a] Tree species: S = Spruce, P = Pine, B = Birch.
[b] Clay < 2 µm, silt 2–20 µm, sand 20–2000 µm.
[c] In the 0–40 cm layer.

Table 2
Site characteristics of the deforested sites.

	Site 7	Site 8	Site 9	Site 10	Site 11	Site 12	Site 13	Site 14
Location	Jokioinen	Jokioinen	Jokioinen	Somero	Tammela	Riihimäki	Juva	Turenki
Years since clearing	~200	28	1	18	24	8	13	10
Main crop	Grass	Cereals	Cereals	Cereals	Grass	Cereal-grass rotation	Cereal-grass rotation	Cereal-grass rotation
Soil type (FAO)	Vertic Cambisol	Vertic Cambisol	Vertic Cambisol	Vertic Cambisol	Haplic Podzol	Vertic Cambisol	Haplic Podzol	Eutric Regosol
Clay[a] (%)								
Forest	26.3	26.3	26.3	70.8	3.4	24.1	3.2	9.3
Field	51.9	42.5	38.5	67.8	5.6	13.4	3.9	8.1
Silt (%)								
Forest	15.6	15.6	15.6	13.0	11.2	34.4	13.2	18.1
Field	22.9	21.8	18.4	9.0	12.5	23.8	13.0	17.5
Sand (%)								
Forest	58.2	58.2	58.2	15.7	85.4	41.5	83.0	72.6
Field	25.2	35.7	43.1	22.7	81.9	62.8	83.0	74.4
Fine earth (%)[b,c]	98.9 ± 1.7	99.0 ± 1.7	98.6 ± 1.8	99.8 ± 0.2	89.9 ± 8.2	98.8 ± 1.6	86.1 ± 5.1	97.9 ± 0.7
Dry bulk density [b] ($g\ cm^{-3}$)								
Forest	1.32	1.32	1.32	0.86	1.32	1.37	1.41	1.50
Field	1.49	1.55	1.58	1.21	1.51	1.67	1.60	1.64
C/N[b]								
Forest	12.1	12.1	12.1	11.1	18.7	13.6	17.1	16.1
Field	9.9	9.4	11.6	13.4	14.8	15.3	14.9	15.5

[a] Clay < 2 µm, silt 2–20 µm, sand 20–2000 µm.
[b] In the 0–40 cm layer.
[c] Mean of the forest and field.

Table 3
Biomass carbon stocks of afforested arable land 17–18 years after afforestation (mean ± standard deviation).

C ($Mg\ ha^{-1}$)	Scots pine		Norway spruce		Silver birch	
	Trees	Vegetation	Trees	Vegetation	Trees	Vegetation
Site 1	69.5 ± 11.1	0.1 ± 0.0	49.7 ± 4.3	0 ± 0	26.3 ± 35.5	1.7 ± 0.3
Site 2	–	–	64.2 ± 10.9	0.0 ± 0.0	62.3 ± 17.6	0.7 ± 0.2
Site 3	50.1 ± 11.3	0.1 ± 0.1	36.3 ± 13.5	0.0 ± 0.0	47.3 ± 13.0	0.0 ± 0.0
Site 4	61.0 ± 18.7	0.2 ± 0.1	57.1 ± 23.7	0.0 ± 0.0	47.5 ± 5.4	1.6 ± 0.8
Site 5	–	–	43.1 ± 9.8	0.0 ± 0.0	–	–
Site 6	37.4 ± 2.7	0.3 ± 0.1	51.6 ± 12.1	0.0 ± 0.0	44.5 ± 5.1	0.4 ± 0.1
Overall	54.5 ± 13.9	0.2 ± 0.1	50.3 ± 9.9	0.0 ± 0.0	45.6 ± 12.8	0.9 ± 0.7

corer (5.5 cm × 4.4 cm) and in 2008 with cylindrical corer 4.6 cm in diameter (Westman, 1995).

To estimate the carbon content of the forest floor, samples from the organic layer on top of the mineral soil (litter, fermentation and humus layer) were collected using a frame 0.02 m^2 in size from the same five points as the soil samples were taken from in 1999 and 2008. To estimate carbon content of the surface vegetation, biomass samples from surface vegetation were taken by clipping all above ground parts of vegetation using a frame 0.25 m^2 in size from the same five points as the soil samples in 2008.

All trees of >1.3 m height were recorded and their diameter at 1.3 m height was measured. For every second tree, the height was

Table 4
The climate data (means of 1971–2000) used as input in the modelling.

Site	Weather station	Precipitation (mm)	Annual temperature (°C)	Temperature coldest month (°C)	Temperature warmest month (°C)	Amplitude (°C)
1	Tvärminne	620	5.7	−4.2	16.6	10.4
2	Jokioinen, observatorio	607	4.3	−6.5	16.1	11.3
3	Pirkkala	601	4.2	−7.0	16.5	11.8
4	Jyväskylä	638	2.9	−8.7	16.0	12.3
5–6	Nivala	561	2.2	−9.2	15.8	12.5
7–11	Jokioinen, observatorio	607	4.3	−6.5	16.1	11.3
12	Hyvinkää, Mutila	647	4.3	−6.6	16.4	11.5
13	Mikkelin mlk. Suonsaari	611	3.4	−8.3	16.3	12.3
14	Hattula	575	4.3	−6.9	16.5	11.7

measured, and for the other trees, the height was estimated using functions based on tree diameter. The biomass of trees (Mg ha^{-1}) was calculated using biomass functions (Repola et al., 2007). Biomass values for surface vegetation and trees were converted to C pools assuming that they contain 50% organic carbon (Table 3).

On the deforestation sites, the samples were taken in late summer 2008 from the fields and the adjacent forests. Three replicate sampling circles (50 m^2) on both the forest and field soil were located along a ca. 75 m transect. The sampling scheme and device were similar to those described above for the afforestation sites in 2008. The tree biomass was estimated as described above but the surface vegetation was not sampled from these sites.

2.3. Soil analyses

The bulked samples of each mineral soil layer were dried at 60 °C and weighed for the determination of bulk density. The samples from the organic horizon were ground (2 mm) and the mineral soil samples were sieved through a 2 mm mesh (live plant remnants were discarded), air dried and stored at room temperature. The sample was crushed to powder size in a mechanical crucible and the total carbon concentration of the <2 mm fraction was measured using a LECO® CHN 600 analyser (Leco Corp., St. Joseph, Mich., USA). A subsample of dried soil was oven dried at 105 °C to correct for moisture content. The C stock for each soil depth was calculated by multiplying the concentration of carbon by the bulk density. To estimate the carbon pool of the upper 40 cm soil layer at the time of the establishment of the experiments in 1991, the missing values of the 10–20 cm and 20–30 cm soil layers were substituted using values of the soil samples taken in 1999 from the respective soil layers. To correct for the error caused by differences in bulk densities in forest and agricultural soil, the carbon content was calculated per an equivalent soil mass by adding to the depth of the deepest layer if needed (Ellert and Bettany, 1995). On average this correction increased the C stock of the forest soil profiles by 4%. Stone content (>2 mm) was taken into account in the calculation of C stock in the

Table 5
Soil carbon stocks of the afforested sites (mean of all tree species ± standard deviation).

Year	C, Mg ha^{-1}						
	Forest floor L	Forest floor FH	0–10 cm	10–20 cm	20–30 cm	30–40 cm	Total
Site 1							
1991	0 ± n.a.	0 ± n.a.	64.8 ± n.a.	49.3 ± n.a.	42.4 ± n.a.	22.6 ± n.a.	179.1 ± n.a.
1999	0 ± n.a.	3.1 ± 1.8	44.3 ± 7.5	50.3 ± 6.1	34.7 ± 14.9	12.3 ± 13.9	144.7 ± 32.0
2008	2.8 ± 1.4	2.2 ± 1.4	50.4 ± 6.7	49.3 ± 6.4	33.1 ± 12.3	14.4 ± 15.9	152.2 ± 28.8
Site 2							
1991	0 ± n.a.	0 ± n.a.	54.6 ± 5.4	39.4 ± 5.7	16.0 ± 4.1	16.3 ± 1.2	126.3 ± 15.6
1999	0.5 ± 1.3	3.0 ± 0.9	40.0 ± 9.0	41.4 ± 4.3	17.3 ± 3.7	9.2 ± 3.1	111.4 ± 14.2
2008	1.1 ± 0.3	1.2 ± 1.2	43.1 ± 4.1	41.7 ± 4.7	18.1 ± 4.8	8.6 ± 1.8	113.8 ± 12.7
Site 3							
1991	0 ± n.a.	0 ± n.a.	35.3 ± 8.7	33.7 ± 5.6	16.5 ± 2.5	6.9 ± 0.3	92.4 ± 10.8
1999	0.2 ± 0.7	2.6 ± 1.0	33.1 ± 7.5	34.8 ± 3.8	17.6 ± 7.3	9.8 ± 16.4	98.1 ± 31.0
2008	1.6 ± 1.3	1.1 ± 0.7	40.6 ± 11.3	36.7 ± 8.7	18.5 ± 7.2	10.0 ± 16.3	108.5 ± 42.7
Site 4							
1990	0 ± n.a.	0 ± n.a.	40.4 ± n.a.	33.7 ± n.a.	20.0 ± n.a.	14.0 ± n.a.	108.1 ± n.a.
1999	0 ± n.a.	2.9 ± 0.5	33.5 ± 6.1	33.7 ± 2.7	17.4 ± 7.5	7.3 ± 2.4	94.8 ± 12.4
2008	2.1 ± 0.5	0.8 ± 0.6	38.7 ± 3.7	32.6 ± 4.7	15.8 ± 7.2	6.7 ± 2.3	96.7 ± 15.0
Site 5							
1990	0 ± n.a.	0 ± n.a.	47.5 ± n.a.	42.8 ± n.a.	38.5 ± n.a.	20.7 ± n.a.	149.5 ± n.a.
1999	0 ± na	4.2 ± 0.2	44.0 ± 12.8	42.8 ± 14.0	38.5 ± 17.8	30.1 ± 15.6	159.6 ± 56.1
2008	1.3 ± 0.6	4.0 ± 0.9	38.7 ± 13.4	39.1 ± 12.2	33.2 ± 6.0	16.3 ± 6.7	132.6 ± 25.1
Site 6							
1990	0 ± n.a.	0 ± n.a.	39.1 ± 8.1	34.3 ± 4.4	30.0 ± 2.9	11.1 ± 1.2	114.5 ± 5.0
1999	0 ± n.a.	3.3 ± 1.4	32.1 ± 3.0	35.2 ± 6.8	25.4 ± 7.3	10.2 ± 4.4	106.2 ± 19.8
2008	2.8 ± 1.2	3.1 ± 0.8	31.2 ± 6.9	32.8 ± 6.7	22.6 ± 6.5	9.5 ± 5.2	102.0 ± 22.0

n.a. = not available.

Table 6
Soil carbon stocks (mean ± standard deviation) of the afforested sites 17–18 years after afforestation with different tree species.

Year	C (Mg ha^{-1})						
	Forest floor L	Forest floor FH	0–10 cm	10–20 cm	20–30 cm	30–40 cm	Total
Pine	3.2 ± 0.6	1.9 ± 1.2	45.3 ± 9.6	39.4 ± 7.9	20.9 ± 5.2	10.1 ± 7.4	120.7 ± 23.0
Spruce	1.5 ± 0.9	2.7 ± 1.2	37.9 ± 6.9	36.8 ± 6.5	24.1 ± 9.1	10.8 ± 4.9	113.7 ± 26.0
Birch	1.9 ± 0.9	0.9 ± 0.8	39.8 ± 7.1	39.6 ± 8.6	22.3 ± 9.0	10.2 ± 7.2	114.7 ± 28.7

deforestation sites. In the afforestation sites the amount of mineral soil particles >2 mm was negligible and was not taken into account.

Four sites (the afforestation sites 4 and 6 planted with spruce and the deforestation sites 8 and 14) were selected for SOC fractionation for a closer look of changes in different SOC pools. An air-dried soil sample of 20 g (<2 mm) was suspended in 60 ml water with glass beads and shaken overnight (17 h, 200 rpm) to disrupt soil aggregates. The fractions >0.063 mm were recovered by wet sieving and separated at 0.63, 0.2 and 0.063 mm (modified from Balesdent et al., 1998). Light particulate organic matter fraction >0.63 mm (light coarse POM) was separated from the minerals by flotation in water. This separation was not possible from the smaller size fractions. All fractions were dried at 60 °C, ground and their C contents measured with the LECO® CHN 600 analyser. Fractionation was conducted in triplicates, and the C contents were measured in triplicates from some samples to determine the accuracy of the method. For most samples the analytical replicates were combined for C measurement to obtain a representative sample. SOC content of the fraction <0.063 mm was not measured, but was assumed to be TOT C minus C in the >0.063 mm fractions. On sites 4 and 6 this fractionation was done for soil samples from the 0 to 10 cm layers, on sites 8 and 14 for layers between 0 and 30 cm.

2.4. Model simulations

Yasso07 is a dynamic soil carbon model developed from the earlier Yasso model (Liski et al., 2005). It is based on three assumptions of litter decomposition: (1) Non-woody litter consists of four compound groups, i.e. compounds soluble in a non-polar solvent, ethanol or dichloromethane (denoted using E), or in water (W), and compounds hydrolysable in acid (A) and neither soluble nor hydrolyzable at all (N). Each group has its own mass loss rate independent of the origin of the litter. (2) The mass loss rates of the compound groups depend on the climatic conditions that can be described simply by using temperature and precipitation. (3) Decomposition of the compound groups results in mass loss from the system and in mass flows between the compound groups. In addition, the mass loss of the four compound groups results in formation of more recalcitrant humus (H) (Tuomi et al., 2009). Coarse woody litter decomposition depends on its chemical quality (EWAN), but also on its physical size (diameter) (Tuomi et al., 2011).

The afforested sites had been cleared for cultivation on average 40 years ago. When initializing the Yasso07 model, it was assumed that before this forest clearance, there has been continuous forests cover for long enough for the soils to be in steady state with the litter input. This steady-state C stock was calculated by using average litter input of spruce forest over rotation period in these Finnish conditions (Liski et al., 2005). Spruce is the dominating tree species on fertile sites in Finland, and was thus assumed to have been the main tree species on the sites cleared for agriculture. All litter compartments were multiplied with a similar factor, so that after simulating 40 years with field vegetation, the modeled soil C stocks were equal to the measured mean C stock in the beginning of the afforestation (1990/1991). For the 40 years period in cultivation, average litter input and litter quality of the different cereals was used (see description for deforested sites below).

After afforestation the quality and the quantity of the yearly litter input to the soil were calculated based on tree and ground vegetation

Table 7
Carbon content of the different particle size fractions (Mean ± S.E.).

Site	Size f. (mm)	2–0.63 POM	2–0.63 Sand	0.63–0.2	0.2–0.063	Sum of >0.063	2–0.63 POM	Sum of >0.063	Sum of >0.063
layer		(mg C g soil d.w.$^{-1}$)					(% of Tot C)		(% of soil mass)
Site 4									
0–10 cm	1991	0.5 ± 0.1	0.02 ± 0.003	1.6 ± 0.1	4.2 ± 0.3	6.3 ± 0.6	1	18	52
0–10 cm	1999	0.8 ± 0.1	0.1 ± 0.02	1.3 ± 0.02	3.0 ± 0.3	5.2 ± 0.4	3	18	52
0–10 cm	2008	1.3 ± 0.2	0.1 ± 0.02	1.9 ± 0.2	3.8 ± 0.3	7.1 ± 0.7	4	22	52
Site 6									
0–10 cm	1991	1.4 ± 0.5	0.2 ± 0.06	3.5 ± 1.4	6.4 ± 2.5	11.5 ± 4.4	3	22	5
0–10 cm	1999	2.2 ± 0.4	0.2 ± 0.03	1.9 ± 0.3	3.0 ± 0.1	7.3 ± 0.6	6	20	5
0–10 cm	2008	3.1 ± 0.3	0.2 ± 0.03	2.5 ± 0.5	3.5 ± 0.3	9.3 ± 0.9	9	28	5
Site 8									
0–10 cm	Forest	4.4 ± 1.4	1.5 ± 0.6	4.9 ± 2.1	9.6 ± 1.7	20.3 ± 3.0	8	38	44
10–20 cm		n.a.	2.1 ± 0.1	1.8 ± 0.1	1.0 ± 0.06	4.9 ± 0.2	n.a.	29	39
20–30 cm		n.a.	0.5 ± 0.1	0.4 ± 0.06	0.3 ± 0.04	1.2 ± 0.2	n.a.	19	29
0–10 cm	Field	n.a.	0.3 ± 0.05	1.2 ± 0.1	1.8 ± 0.2	3.4 ± 0.4	n.a.	21	25
10–20 cm		n.a.	0.2 ± 0.03	0.8 ± 0.1	1.3 ± 0.2	2.3 ± 0.3	n.a.	17	23
20–30 cm		n.a.	0.04 ± 0.01	0.2 ± 0.04	0.4 ± 0.1	0.6 ± 0.1	n.a.	9	6
Site 14									
0–10 cm	Forest	5.6 ± 1.1	1.5 ± 0.6	8.8 ± 1.6	6.1 ± 1.4	22.0 ± 4.6	12	48	45
10–20 cm		n.a.	1.8 ± 0.2	2.4 ± 0.3	1.7 ± 0.2	5.8 ± 0.5	n.a.	28	48
20–30 cm		n.a.	0.3 ± 0.04	0.5 ± 0.1	0.3 ± 0.04	1.2 ± 0.2	n.a.	17	48
0–10 cm	Field	2.6 ± 1.0	0.1 ± 0.02	3.9 ± 1.9	3.6 ± 1.5	10.2 ± 4.4	10	35	44
10–20 cm		n.a.	1.4 ± 0.6	2.6 ± 1.3	2.8 ± 1.5	6.8 ± 3.4	n.a.	29	43
20–30 cm		n.a.	0.4 ± 0.1	0.8 ± 0.4	0.7 ± 0.3	1.9 ± 0.9	n.a.	18	41

n.a. = not available.

Table 8
Soil carbon stocks of the fields and the adjacent forests (mean+-standard deviation).

Site		C (Mg ha^{-1})					
		Forest floor FH	0–10 cm	10–20 cm	20–30 cm	30–40 cm	Total
7	Forest	8.9 ± 2.7	40.4 ± 3.3	26.4 ± 1.7	10.3 ± 1.9	12.7 ± 2.0	98.7 ± 4.2[a]
	Field	–	35.0 ± 2.6	32.8 ± 3.1	15.2 ± 2.4	8.5 ± 1.0	91.5 ± 2.7
8*	Field	–	25.4 ± 3.1	23.8 ± 3.0	11.3 ± 2.5	7.0 ± 0.5	67.5 ± 8.1[b]
9*	Field	–	52.3 ± 6.4	28.5 ± 6.8	14.6 ± 5.8	10.5 ± 3.0	106 ± 21.8
10	Forest	39.4 ± 3.2	36.2 ± 8.9	48.4 ± 8.0	22.2 ± 4.8	18.6 ± 3.9	165 ± 13.7
	Field	–	51.1 ± 7.5	56.0 ± 5.8	25.1 ± 13.9	10.3 ± 4.4	142 ± 14.6
11	Forest	23.1 ± 13.3	30.4 ± 1.0	16.5 ± 1.9	10.8 ± 2.2	11.3 ± 2.5	92.1 ± 1.8[a]
	Field	–	41.7 ± 2.0	29.9 ± 2.9	8.1 ± 2.1	4.8 ± 1.9	84.5 ± 3.9[b]
12	Forest	18.6 ± 7.5	42.8 ± 3.1	23.2 ± 7.9	8.5 ± 4.8	14.3 ± 10.0	107 ± 15.2
	Field	–	37.1 ± 7.7	29.1 ± 7.3	18.1 ± 7.0	7.2 ± 1.6	91.5 ± 21.1
13	Forest	22.3 ± 3.7	41.8 ± 4.4	23.8 ± 3.4	15.8 ± 2.4	10.3 ± 1.7	114 ± 5.7[a]
	Field	–	37.9 ± 1.8	33.7 ± 3.2	14.0 ± 2.1	7.1 ± 0.7	92.7 ± 5.4[b]
14	Forest	7.4 ± 7.4	41.3 ± 7.5	31.4 ± 9.1	12.5 ± 1.4	8.0 ± 1.2	100 ± 23.2
	Field	–	35.8 ± 14.4	32.3 ± 17.6	17.8 ± 7.8	6.3 ± 2.7	92.2 ± 42.5

Total amounts marked with different letters indicate statistically significant differences between the forest and field of a site.
* Results can be compared with the forest of site 7.

measurements in 2008. We assumed that the tree biomass grew linearly from 1990/1991 to 2008. The biomass was divided to 7 compartments: roots, fine roots, stem and bark, living branches, needles and leaves, ground vegetation, and roots of the ground vegetation. The biomasses of the compartments were calculated based on the measured tree diameters at 1.3 m height and tree heights, using the biomass functions of Repola et al. (2007). Fine root biomass was estimated to be 50% of foliage biomass for pine and birch and 30% for spruce (Liski et al., 2006). Herbs and grasses were assumed to make up 90% of the biomass of ground vegetation. Root biomass of the ground vegetation was estimated to be 85% of the above ground biomass (Törmälä and Raatikainen, 1976). All biomass estimates were converted to carbon by multiplying by 0.5. Litter production of trees and ground vegetation was calculated by multiplying the biomass estimates by compartment-specific biomass turnover rates (Liski et al., 2006). Litter chemical quality (i.e. proportions of compounds soluble in dichloromethane or ethanol (E), or in water (W), hydrolysable in acid (A), and neither soluble nor hydrolysable (N) (Tuomi et al., 2009)) for each litter type was taken from literature (Berg et al., 1991a, 1991b, 1993; Gholz et al., 2000; Trofymow and CIDET Working Group, 1998). The uncertainty in the quality and quantity of litter inputs rising from the uncertainty in biomass models and biomass turnover rates was taken into account by using the coefficients of variation from Peltoniemi et al. (2006). The litter inputs were first calculated separately for each of the three plots and the average of these plots was then used as input into the model.

The steady-state C stock of the deforested sites was calculated using the measured biomasses of the adjacent forest sites in year 2008, which were turned into estimates of litter inputs similarly as described for the afforested sites. The biomass of the ground vegetation was not measured, but estimated as a function of stand

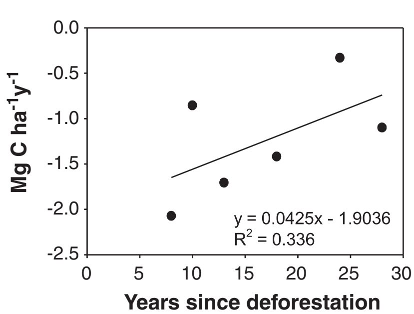

Fig. 2. The effect of the age of the field on the amount of annual C loss. The results of the one year and 200 years old fields were left out.

age using the statistical models of Muukkonen and Mäkipää (2006). Similar assumptions of the below ground biomass of ground vegetation and litter production from ground vegetation were used as for the afforested sites. Each litter compartment was multiplied by a same constant to produce a litter input that produced a steady-state C stock corresponding to the measured C stock in the beginning of the simulation.

After clearance for agriculture, the litter input was calculated based on the cultivation history (plant species) and the mean yields and harvest indices (MMM, 2009). The chemical quality of wheat and barley litter was measured by fractionating it into the compound soluble in ethanol (E), water (W), hydrolysable with acid (A) and a non soluble- non hydrolysable residue (N) (Berg et al., 1991b). For rye and oats an average of these values was used because all these cereals have a rather similar chemical quality. The quality of grass litter was estimated based on Van Soest extractions (Jensen et al., 2005) transformed to correspond to the proximate carbon fractions (EWAN) with the regression models of Ryan et al. (1990). The variation in these litter quality estimates were derived from the measurements. Coefficient of variation for cereal litter input was assumed to be the same as for forest ground vegetation (0.2). Estimates of C input from grass were considered to be much more uncertain and thus a coefficient of variation of 0.5 was used.

The used climate input data (monthly and annual mean temperature and annual precipitation) was an average from years 1970 to 2000 from the nearest meteorological station of the study area (Table 4) (Drebs et al., 2002).

2.5. Statistical analyses

To compare changes in the C stocks in the soil layers and combined carbon pool of the upper 40 cm soil layer over time in the afforested sites, the differences were tested over all sites with repeated-measurement analysis of variance. In the mixed models, time was considered to be a fixed effect and the location of the site a random effect as well as their interaction. In the analysis, the choice of the best covariance structure was based on Akaike's information criterion (Littel et al., 1996). The effect of tree species on carbon pools was tested separately for all soil layers and combined carbon pool of the upper 40 cm soil layer with the analysis of variance as a multilocation trial at the plantation age of 17–18 years. In the case of deforestation, differences in the total carbon stock in the 0–40 cm soil layer were studied with *t*-test as well as the differences in the SOC fractions between the forest and agricultural soils. The correlation between the observed changes in C stock and the C/N ratio of the soil layers was studied with Pearson correlation analysis.

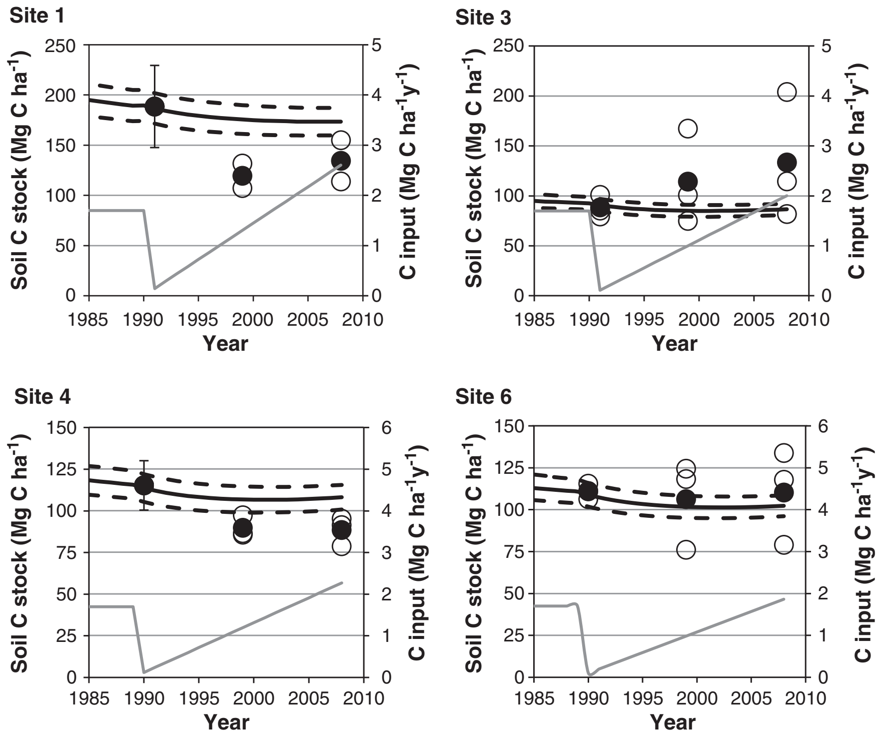

Fig. 3. Measured and modelled C stocks of the afforested sites planted with pine (first y-axis). C input used in modelling is presented on the second y-axis. Figures present measured values (open circles), measured mean (black circles), modelled mean (black line), 95% confidence limits of the modelled mean (dotted lines), C input (grey line). Error bars for 1990/1991 measurements represent standard deviation.

Regression analyses were made to see whether model residuals (difference between modeled and measured mean C stock) depended on soil texture (clay%), or time since deforestation. We studied whether the measured soil C stocks correlated with soil texture (clay%) between sites. Correlations of the SOC fractions with the modelled C input were also studied (Pearson correlation analysis). Data analysis was done using the SAS statistical software (SAS Institute, Cary, NC).

3. Results

3.1. Afforestation

At the time of establishment of the plantations, the mean carbon pool of the upper 40 cm soil layer was 128.3 Mg $ha^{-1}$and varied from 92.4 to 179.1 Mg ha^{-1} (Table 5). Averaged over all sites, the mean carbon stock in the 0–10 cm soil layer and carbon pool of the upper 40 cm soil layer decreased significantly over 18 years ($p<0.001$, Table 5) whereas in the 30–40 cm soil layer there was no significant change. The variance component estimates of the site-by-time interaction indicated that there were site specific differences in the change of soil carbon pools over time ($p<0.05$). The mean 20-year change in carbon pools of the upper 40 cm soil layer varied from an increase of 18.9 Mg ha^{-1} to a decrease of 31.6 Mg ha^{-1}. The mean change in carbon pools across all sites was a decrease of 9%. This decrease corresponds to a mean annual decrease of 0.6 Mg ha^{-1} over 20 years.

The effect of tree species on the carbon pool of the upper 40 cm soil layer was not significant at the plantation age of 17–18 years. However, the C stock of the forest floor was significantly highest when the site was planted with spruce ($p<0.05$, Table 6) whereas in the 0–10 cm soil layer the C stock was highest when planted with pine ($p<0.05$). The build-up of the forest floor litter was fastest when the afforested sites were planted with Scots pine and slowest when planted with silver birch. The mean annual rate of carbon accumulation in the forest floor over 18 years was 0.28 and 0.15 Mg ha^{-1} for pine and birch, respectively.

The effect of tree species on the carbon pool of the trees was not significant at the plantation age of 17–18 years. At that age, the trees had accumulated carbon on average 50.1 Mg ha^{-1} (Table 3). This corresponds to a mean annual rate of accumulation of carbon of 2.9 Mg ha^{-1}. Thus, the mean annual rate of accumulation of carbon in trees offsets the mean annual decrease of carbon pool of the upper 40 cm soil layer. The carbon pool of the surface vegetation was significantly higher under silver birch ($p<0.05$) compared to that of the other tree species at the plantation age of 17–18 years (Table 3).

The content of C in the coarse light POM fraction (2–0.63 mm) in the 0–10 cm layer correlated with the modelled C input on site 4 ($r=0.87$, $p<0.01$) and site 6 ($r=0.81$, $p<0.01$). The increase in the C content of this fraction over the 18 years was statistically significant only at site 4 ($p<0.05$), but its proportion of total C increased statistically significantly at both sites 4 and 6 (Table 7). There was a decreasing trend in the average C amount in 0.63–0.2 mm and 0.2–0.063 mm sized fractions and sum of C in all >0.063 mm fractions from 1991 to 1999, and an increasing trend from 1999 to 2008 on both sites. However, these changes were not statistically significant. There were no visible soil aggregates on the sieves after the disruption of soil by shaking.

3.2. Deforestation

The carbon content of the soil profiles varied from 92 to 165 Mg ha^{-1} in the forest soils and from 68 to 142 Mg ha^{-1} in the agricultural soils (Table 8). On the average, the carbon stock was smaller in field soils than in forest soils, although statistically significant differences were found only in three cases. In the case of the field cleared one year before the sampling (site 9), the forest floor apparently was mixed with the mineral topsoil which increased the carbon content of that layer. To be able to compare the change in the

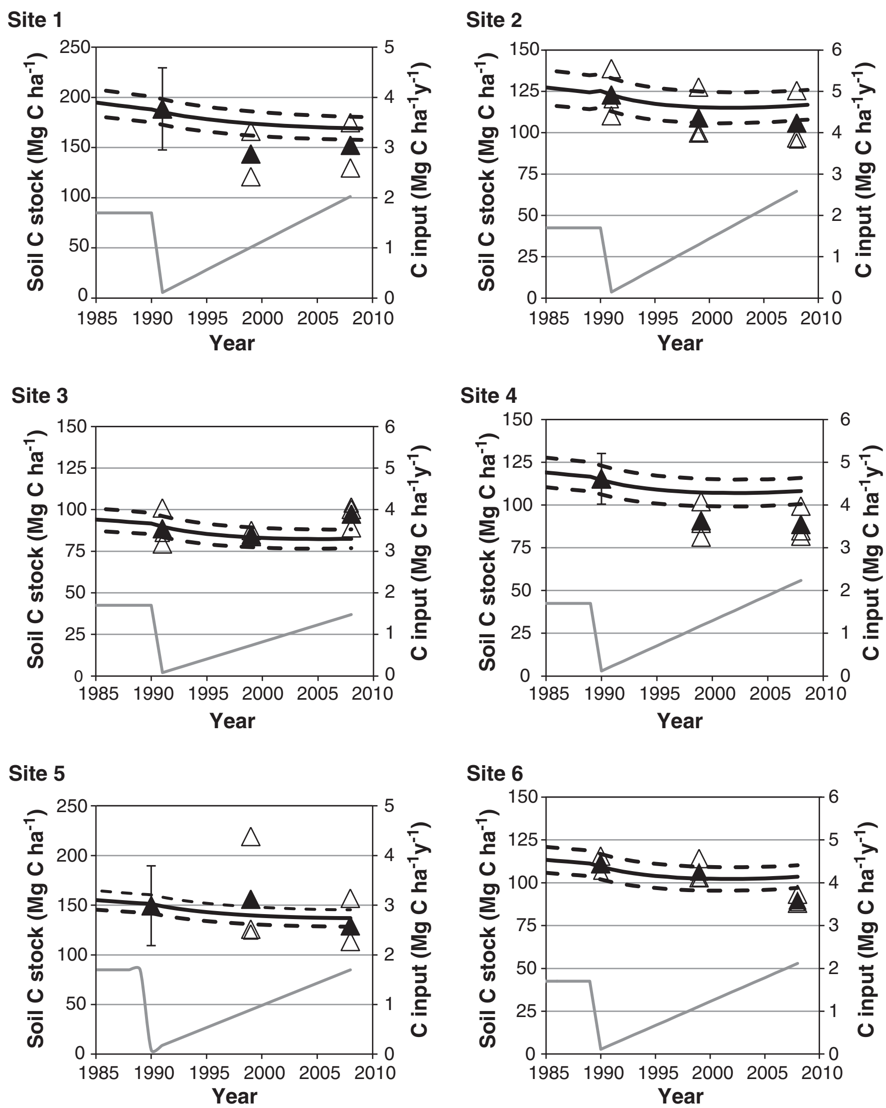

Fig. 4. Measured and modelled C stocks of the afforested sites planted with spruce (first y-axis). C input used in modelling is presented on the second y-axis. Figures present measured values (open triangles), measured mean (black triangles), modelled mean (black line), 95% confidence limits of the modelled mean (dotted lines), C input (grey line). Error bars for 1990/1991 measurements represent standard deviation.

carbon stock in each of the field-forest pairs to the IPCC default value of −29% per 20 years (IPCC, 2003) we calculated the change in each site per 20 years (results not shown). The average 20-year change with the one-year-old field excluded was −19%. The smallest change was observed on fields with grass as the main crop (−7%). In the fields with cereals as the main crop the average change in 20 years was 24%. The loss was fastest in the first years after the clearance of the forest, since there was a weak linear relationship between the time since clearance and the annual change in carbon stock (Fig. 2). The C/N ratio of the soil was on the average lower in field soils compared to forest soils (Table 2). There was more variation in the values of bulk density in the deforestation sites than in the afforestation sites. The bulk density was always higher in the field than in the forest. The exceptionally low values on site 10 were probably a result from the high content of organic matter which can be caused by the observed high ground water level on the site and thus slow mineralization. The bulk density was highest on sites with coarse soil.

The surface layers of forest soils had generally higher amounts of C in the >0.063 mm particle size fractions than the field soils, the clearest differences being in the topmost layers and in the coarsest (2–0.63 mm) fractions of site 8 (Table 7). There were measurable amounts of coarse light POM only in the 0–10 cm layer of the forest soil. The percentage of total soil C in the >0.063 mm sized fractions was statistically significantly higher in forest than in field soil ($p<0.05$) in all the studied soil layers of site 8. This was partly due to higher percentages of total soil weight in the >0.063 mm size fractions reflecting the visibly higher amount of stable aggregates left after soil disruption in the forest soil compared to the field soil, and partly due to higher C concentration of these fractions in forest soil compared to field soil. On site 14, there were no statistically significant differences in the amounts of the SOC fractions between forest and field soil. There were no statistically significant differences between forest and field soil in the proportion of soil weight in the >0.063 mm fractions, although there were little visible aggregates left in the forest soil fractions, but not in the field soil fractions.

3.3. Model simulation

The modeled carbon stock fit generally well with the measurements. Model residuals (modeled mean C stock—measured mean C stock) both on afforested and deforested sites were close to normal distribution and did not deviate from zero (95% confidence level).

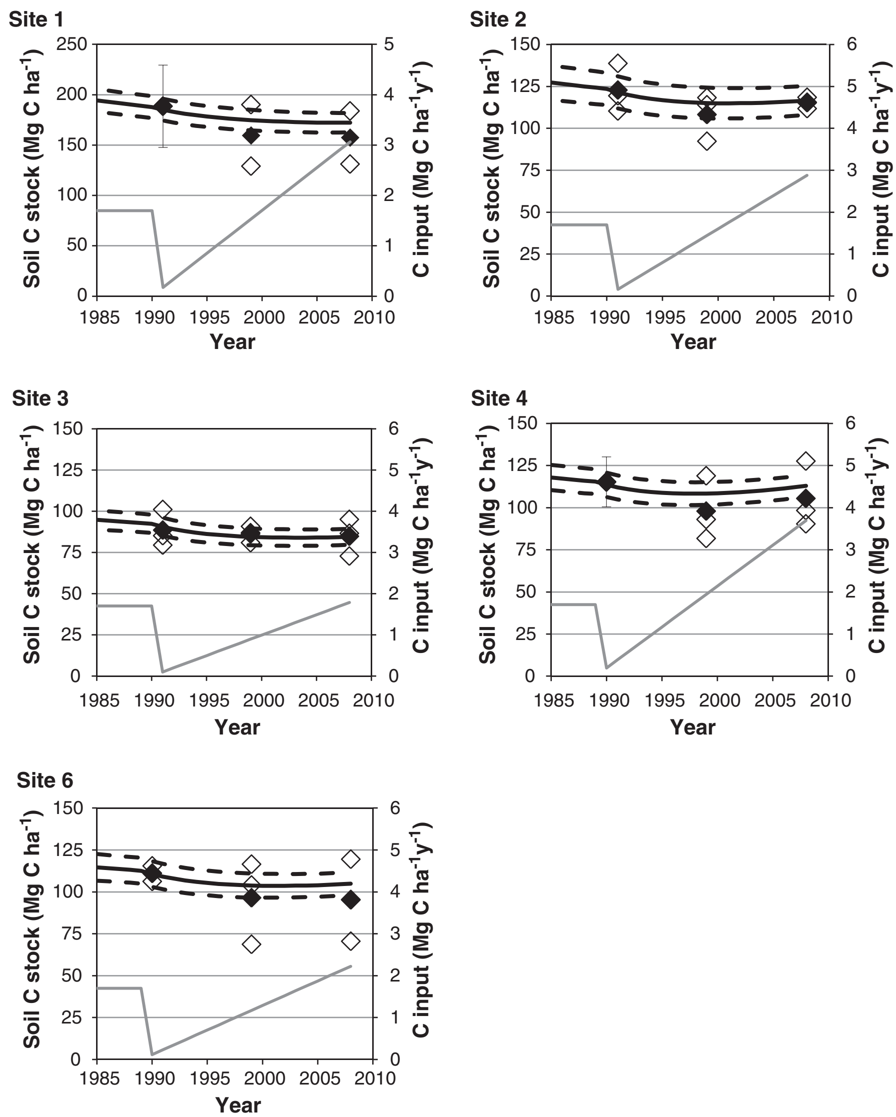

Fig. 5. Measured and modelled C stocks of the afforested sites planted with birch (first y-axis). C input used in modelling is presented on the second y-axis. Figures present measured values (open diamonds), measured mean (black diamond), modelled mean (black line), 95% confidence limits of the modelled mean (dotted lines), C input (grey line). Error bars for 1990/1991 measurements represent standard deviation.

On the afforested sites, the estimated rate of C input diminished remarkably at the start of the afforestation but in most cases eventually increased to a level that was higher than during the period of cultivation (Figs. 3–5). The 95% confidence limits of the modeled mean and the range of measurements overlapped on most cases. For sites 2, 3, and 6, the replicate measurements for years 1990/1991 were taken from the plots that were planted with spruce. For sites 1, 4 and 5, there was only one pooled sample for year 1990/1991, representing the whole experimental area. The error bars in the figures of these sites (standard deviation) were estimated by calculating an average coefficient of variation (CV) of the C stock of all plots on years 1998 and 2008, and multiplying the 1990/1991 average with this CV. Only on the afforested sites 1 and 4, planted with pine, the model slightly overestimated the C stock after afforestation (years 1999, 2008) (Fig. 3), but this difference between modeled and measured means was not statistically significant. Model residuals were not correlated with soil texture (clay%) in either year (Fig. 6a, b) or soil C/N ratio (results not shown). Initial measured C stocks (1990/1991) were not correlated with soil texture either (Fig. 6c). The change in C stock (Mg C ha^{-1} y^{-1}) between 1990/1991–1999–2008) was not related to soil texture (clay%, data not shown).

In the deforested sites the estimated annual C input varied depending on the crop (Fig. 7). In the case of grass species the input was higher in the year of renewal. The measured and modeled mean agreed well and the model did not systematically over- or underestimate the stocks and changes. The 95% confidence limits of the modeled mean and the range of measurements overlapped for all cases, except site 7, which had been cultivated for centuries. Model residuals did not depend on soil texture (clay%) (Fig. 6d) or time since deforestation (Fig. 6e, excluding site 7) or soil C/N ratio (results not shown). When the site 10 with highest C stock, and highest clay% was removed, there was no correlation between clay% and C stock (Fig. 6f). The change in C stock (Mg C ha^{-1} yr^{-1}) was not related to soil texture (clay%, data not shown).

4. Discussion

The rates of carbon accumulation in the forest floor of the afforestation sites (from 0.15 to 0.28 Mg ha^{-1} yr^{-1}) were similar to rates reported on afforested agricultural lands in Denmark for Norway spruce over 29 years (Vesterdal et al., 2002) and in USA for loblolly pine over 50 years (Schiffman and Johnson, 1989) but lower than those (0.9 Mg ha^{-1}) in USA for loblolly pine over 40 years (Richter et al.,

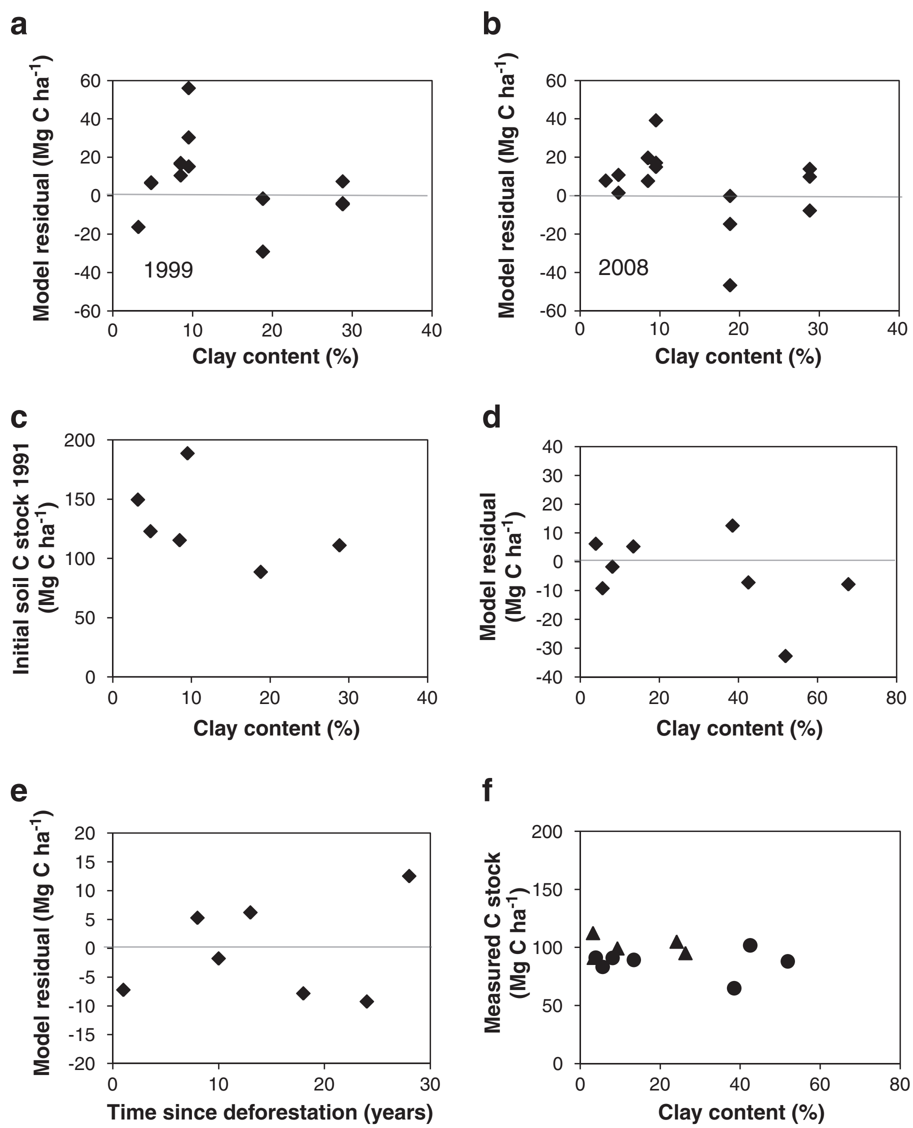

Fig. 6. Correlations of model residuals (modelled-measured mean) with clay content on the afforested sites on a) year 1999, b) year 2008, c) correlation of initial soil C content with clay content on the afforested sites, d) correlation of model residuals with clay content on the deforested sites, e) correlation of model residuals with time since deforestation and f) correlation of measured stocks with clay content on the deforestation sites (black triangles: forest soils, black dots: field soils, Site 10 was left out of the picture).

1999). However, the build-up of the forest floor in the afforestation experiments did not fully compensate for the decrease in the C stock of the upper 10 cm soil layer and as a result, 18 years from afforestation, the average soil carbon pool of the upper 40 cm soil layer was lower compared to that at the time of establishment of the plantations. This contrasts to many studies in which increased carbon pool has been a common consequence of afforestation of agricultural land even though also loss of carbon pool and no change have been reported (Jandl et al., 2007; Johnson, 1992; Post and Kwon, 2000). The decline in carbon pools over 20 years found in our study can be attributed to reduced carbon inputs to the surface soil following afforestation. Similar results have been reported previously from a compilation of studies (Laganière et al., 2010; Paul et al., 2002). In a Swedish study, the C stock of afforested sites increased by more than 90 Mg ha^{-1} between 18 and 91 years of afforestation (Cerli et al., 2006). Thus, the prediction that carbon pools will increase after afforestation may be realized in the longer term, as our study period was only 17–18 years. The results from the SOM fractionation support this conclusion: the coarse light POM fraction (0.63 to 2 mm sized), which is considered most labile, and responsive to land use and management changes (Buyanovsky et al., 1994; Cambardella and Elliot, 1992; Christensen, 1992; Del Galdo et al., 2003), increased with the increasing C inputs since the start of afforestation. In 2008 this fraction comprised only 4 or 9% of total C in the sites 4 and 6, respectively. Thus, the 109–128% increase in the amount of this C fraction from 1990/1991 did not yet show as an increase in total C amounts. This increase, however, can be seen as a trajectory showing the direction of the development of the bulk C storage and more slowly cycling SOC fractions. It will take a longer time until the C in the other fractions with slower turnover will also reach a new steady state with the litter inputs.

As regards the forest soils studied in the deforestation cases, the measured carbon stocks are higher than reported by e.g. Liski et al. (2005) and Peltoniemi et al. (2004), which may result from the fact that the most fertile sites have originally been taken for agricultural use. There are no published estimates of the average carbon stocks of agricultural soils in Finland to allow similar comparison. Site 10 has an exceptionally high carbon stock both in the forest and field which may be caused by a high water table slowing down the mineralization. Excluding this site, there was no correlation between clay content and C stock. The average change in the carbon stock following forest clearance for agriculture was less than the 30–40% observed e.g. in Canada (Carter et al., 1998; Ellert and Gregorich, 1996) or Alaska

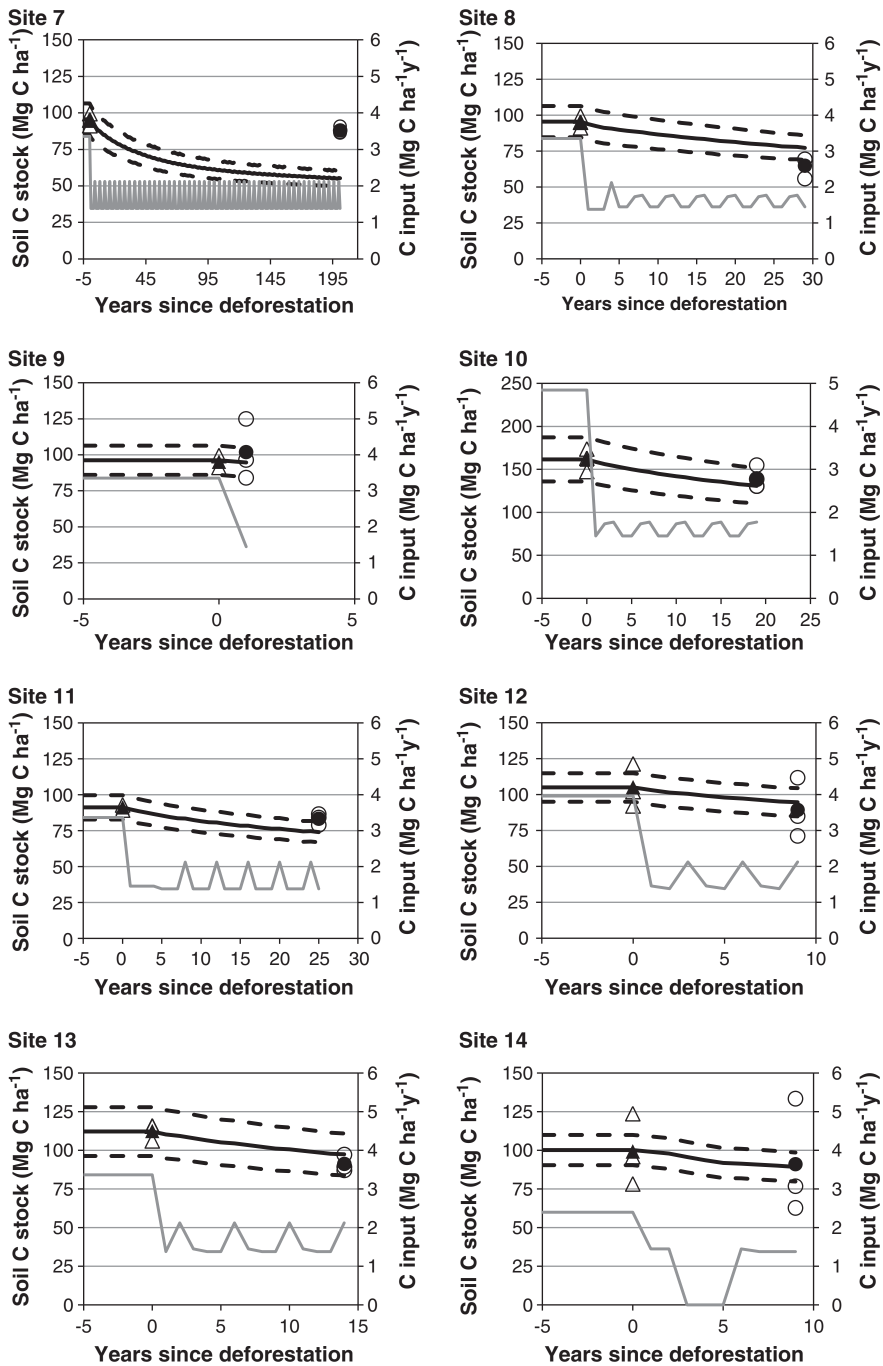

Fig. 7. Measured and modelled stocks of the deforested sites. Measured values forest (open triangles), mean values forest (closed triangles), measured values field (open circles), mean values field (closed circles), modelled mean (black line), 95% confidence limits of the modelled mean (dotted lines). C input used for modelling is presented on the second axis (grey line).

(Grünzweig et al., 2004). However, variation in these estimates is high and in some cases even a greater soil C storage has been found after the conversion due to improved fertility or drainage of soils (Ellert and Gregorich, 1996). The differences in carbon stock changes in cultivation between Finland and North America may result e.g. from different residue management practices. It seems that removal or burning of residues (Chen et al., 2005) and fallowing (Liang et al., 2005) are more common in North America than in Finland. On the basis of the results from the deforested sites, the IPCC default value of −29% determined for wet temperate regions (IPCC, 2003) seems a too high value to describe the change in carbon stock following forest clearance for agriculture in Finnish conditions. The fact that the results were not based on an experiment with repeated measurements adds some uncertainty to the results. Although we aimed at having exactly the same soil type for the forest-field pairs there were some differences in the soil texture between the field and the adjacent forest in sites 7–9 and 12 which also may have affected the results. As found before e.g. by Compton and Boone (2000) and Murty et al. (2002) the C/N ratio of the mineral soil was lower in the field soils compared to the forest soils indicating that even in such a short period as in this study the differences in the composition of agricultural vs. forest litter has an effect on the C/N ratio of the mineral soil.

Results from the SOM fractionation support the earlier observations that changes in soil C stock after clearing forest for agriculture occur mostly from the more labile, coarser size C fractions from the top soils (Balesdent et al., 1998; Buyanovsky et al., 1994; Del Galdo et al., 2003). However, the variation in the amounts of these C fractions was also large, and no statistically significant differences were found if the differences in bulk C amounts did not differ statistically. Thus, we support the view of Leifeld and Kögel-Knabner (2005) that the time-consuming SOM fractionation is not useful alone as a verification tool for detecting changes in C stocks due to land use change, although it can help understanding SOM turnover. On the deforested site 8 there was a clear difference in the weights of the fractions separated by size, indicating a higher stability of soil aggregates in the forest soil compared to field soil. On this site, decreased physical protection of SOM inside soil aggregates after deforestation might have partly affected the decrease in soil C storage. However, we hypothesize that in the studied soils the physical protection is of smaller importance for the soil C storage than the changed amount and quality of litter inputs of C. On the afforested sites there were no visible soil aggregates left after soil disruption (before or after afforestation) and thus afforestation did not increase soil aggregate stability.

The Yasso07 model could well predict the changes in C stocks due to land use change. The observation that the model could equally well predict C stocks in soils with different texture is important for the applicability of the model in land use change situations. The model was originally developed for forest soils, which often have coarse texture, whereas agricultural soils are located on the more fertile sites and include finer-textured soils. If texture had a strong effect on decomposition rates in the studied soils it would be expected that the most extreme sites would be systematically different. Based on the idea that clay stabilizes SOC on its surfaces and slows down decomposition, overestimation of C losses would be expected on sites with high clay content. The lack of systematic error means that, on average, the description of decomposition rates in the model is valid and adequate also for predicting decomposition on afforested and deforested sites in Finland. The error in the results is thus due to random variation, i.e. possible higher deviation in microclimate, management and C inputs. Average (30 years) climate data and average litter inputs from agricultural statistics were used in the modeling. On the other sites the C inputs probably were estimated with better accuracy but on site 7 the long cultivation history, management and inputs were difficult to estimate based on the recent agricultural statistics. Thus on this site a larger error in estimating the C inputs and frequency of tillage probably caused the modeled results to differ from the measured. This was a site with a long history of grass cultivation and it is known that in grasslands the soil carbon is better protected from decomposition than in more frequently ploughed soils. It is possible that the discrepancy between measured and simulated values of the soil C stock indicates that the effect of less frequent tillage on soil C stocks should be taken into account in the formulation of the model if it is to be used for estimating C stocks on grasslands.

Soil mineralogy and clay content may be important for the long-term development of soil C stock over centuries (Torn et al., 1997), but shorter term changes are likely to occur from other fractions than the mineral associated fractions that are considered more stabilized (Balesdent et al., 1998; Cambardella and Elliot, 1992; Eusterhues et al., 2003; Gregorich and Janzen, 1996). Several studies have found soil C storage to positively correlate with clay content (e.g. Homann et al., 1995; Jastrow and Miller, 1997). Some soil carbon models include clay % as input information affecting the stabilization of soil organic matter at higher clay contents (Coleman and Jenkinson, 1996; Parton et al., 1987). However, we did not find correlation between clay content and C storage which is in agreement with some other studies (Callesen et al., 2003; Fissore et al., 2008). Although high clay contents can stabilize C, at lower clay contents increase in clay% can increase the decomposition rate due to higher capacity to retain water and nutrients (Callesen et al., 2003). Clay content may be a more important SOC stabilizing mechanism in warmer climates since the magnitude of the effect that clay has on C stabilization may increase with temperature (Conen et al., 2008; Leifeld et al., 2009; Thornley and Cannell, 2001). In the boreal climate, the SOC accumulation to soil is most likely due to the climatic conditions being sub-optimal (low temperature) (Conen et al., 2008), i.e. temperature is the most important factor limiting decomposition rates in these soils. Thus, soil texture, e.g. clay%, is not necessarily needed in the model when predicting C stock changes due to land use change in boreal conditions.

Although there are changes in soil conditions due to taking soil into cultivation (e.g. ploughing, changes in soil structure, fertilization), their effect on C stock changes is less important than the effects of quality and quantity of litter inputs and climate. These main factors controlling C dynamics included in the Yasso07 model seem sufficient for predicting decomposition rates also in land use change situations in boreal conditions. Thus, Yasso07 model can be used for modeling the short-term effects of land use changes on C stocks in Finland.

5. Conclusions

Instead of increasing as expected, the average soil C stock decreased slightly after afforestation, because decomposition exceeded C input during the early years of afforestation. A longer time than the 20-year period in this study is probably needed before an increase in the mineral soil C stocks becomes apparent. Soil carbon stock on the deforested sites decreased less than estimated using the IPCC default value for reporting. The decrease was also smaller when compared to results from North America, which could result e.g. from differences in farming practices. On the basis of these results the decrease in C stocks in deforestation in boreal conditions is faster than the recovery of the C stock in afforestation. Yasso07 model predicted the measured changes in C stocks well both at afforested and deforested sites. The tests of validity suggest that Yasso07 model can be applied for estimating soil C stock changes due to land use change in boreal conditions. However, further validation using more long-term data sets would be valuable.

Acknowledgements

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References

Balesdent, J., Besnard, E., Arrouays, D., Chenu, C., 1998. The dynamics of carbon in particle-size fractions of soil in a forest-cultivation sequence. Plant and Soil 201, 49–57.

Berg, B., Booltink, H., Breymeyer, A., Ewertsson, A., Gallardo, A., Holm, B., Johansson, M.-B., Koivuoja, S., Meentemeyer, V., Nyman, P., Olofsson, J., Pettersson, A.-S., Reurslag, A., Staaf, H., Staaf, I., Uba, L., 1991a. Data on needle litter decomposition and soil climate as well as site characteristics for some coniferous forest sites, Part I, Site characteristics. Report 41. Swedish University of Agricultural Sciences, Department of Ecology and Environmental Research, Uppsala.

Berg, B., Booltink, H., Breymeyer, A., Ewertsson, A., Gallardo, A., Holm, B., Johansson, M.-B., Koivuoja, S., Meentemeyer, V., Nyman, P., Olofsson, J., Pettersson, A.-S., Reurslag, A., Staaf, H., Staaf, I., Uba, L., 1991b. Data on needle litter decomposition and soil climate as well as site characteristics for some coniferous forest sites, Part II, Decomposition data. Report 42. Swedish University of Agricultural Sciences, Department of Ecology and Environmental Research, Uppsala.

Berg, B., Berg, M.P., Bottner, P., Box, E., Breymeyer, A., De Anta, R.C., Couteaux, M., Mälkönen, E., McClaugherty, C., Meentemeyer, V., Munoz, F., Piussi, P., Remacle, J., De Santo, A.V., 1993. Litter mass loss in pine forests of Europe and Eastern United States: some relationships with climate and litter quality. Biogeochemistry 20, 127–159.

Buyanovsky, G.A., Aslam, M., Wagner, G.H., 1994. Carbon turnover in soil physical fractions. Soil Science Society of America Journal 58, 1167–1173.

Callesen, I., Liski, J., Raulund-Rasmussen, K., Olsson, M.T., Tau-Strand, L., Vesterdal, L., Westman, C.J., 2003. Soil carbon stores in Nordic well-drained forest soils—relationships with climate and texture class. Global Change Biology 9, 358–370.

Cambardella, C.A., Elliot, E.T., 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. Soil Science Society of America Journal 56, 777–783.

Carter, M.R., Gregorich, E.G., Angers, D.A., Donald, R.G., Bolinder, M.A., 1998. Organic C and N storage, and organic C fractions, in adjacent cultivated and forested soils of eastern Canada. Soil Tillage Research 47, 253–261.
Cerli, C., Celi, L., Johansson, M.B., Kögel-Knabner, I., Rosenqvist, L., Zanini, E., 2006. Soil organic matter changes in a spruce chronosequence on Swedish former agricultural soil I. Carbon and lignin dynamics. Soil Science 171, 837–849.
Chen, Y., Tessier, S., Cavers, C., Xu, X., Monero, E., 2005. A survey of crop residue burning practices in Manitoba. Applied Engineering in Agriculture 21, 317–323.
Christensen, B.T., 1992. Physical fractionation of soil and organic matter in primary particle size and density separates. Advances in Soil Science 20, 1–90.
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. NATO ASI Series I 38. Springer-Verlag, Heidelberg, pp. 237–246.
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.
Conen, F., Zimmermann, M., Leifeld, J., Seth, B., Alewell, C., 2008. Relative stability of soil carbon revealed by shifts in delta N-15 and C:N ratio. Biogeosciences 5, 123–128.
Davidson, E.A., Ackerman, I.L., 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20, 161–193.
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.
Drebs, A., Nordlund, A., Karlsson, P., Helminen, J., Rissanen, P., 2002. Climatological statistics of Finland 1971–2000. Finnish Meteorological Institute, Edita Prima, Helsinki. 99 pp.
Ellert, B.H., Bettany, J.R., 1995. Calculation of organic matter content and nutrients stored in soils under contrasting management regimes. Canadian Journal of Soil Science 75, 529–538.
Ellert, B.H., Gregorich, E.G., 1996. Storage of carbon, nitrogen and phosphorus in cultivated and adjacent soils of Ontario. Soil Science 161, 587–603.
Eusterhues, K., Rumpel, C., Kleber, M., Kögel-Knabner, I., 2003. Stabilization of soil organic matter by interactions with minerals as revealed by mineral dissolution and oxidative degradation. Organic Geochemistry 34, 1591–1600.
Fissore, C., Giardina, C.P., Kolka, R.K., Trettin, C.C., King, G.M., Jurgensen, M.F., Barton, C.D., McDowell, S.D., 2008. Temperature and vegetation effects on soil organic carbon quality along a forested mean annual temperature gradient in North America. Global Change Biology 14, 193–205.
Franzluebbers, A.J., Haney, R.L., Honeycutt, C.W., Arshad, M.A., Schomberg, H.H., Hons, F.M., 2001. Climatic influences on active fractions of soil organic matter. Soil Biology and Biochemistry 33, 1103–1111.
Gholz, H.L., Wedin, D.A., Smitherman, S.M., Harmon, M.E., Parton, W.J., 2000. Long-term dynamics of pine and hardwood litter in contrasting environments: toward a global model of decomposition. Global Change Biology 6, 751–765.
Gregorich, E.G., Janzen, H.H., 1996. Storage and soil carbon in the light fraction and macro-organic matter. In: Carter, M.R., Stewart, B.A. (Eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC Press, Boca Raton, pp. 167–190.
Grünzweig, J.M., Sparrow, S.D., Yakir, D., Chapin, F.S., 2004. Impact of agricultural land-use change on carbon storage in boreal Alaska. Global Change Biology 10, 452–472.
Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345–360.
Homann, P.S., Sollins, P., Chappel, H.N., et al., 1995. Soil organic matter in a mountainous, forested region: relation to site characteristics. Soil Science Society of America Journal 59, 1468–1475.
IPCC, 2003. Good Practice Guidance for Land use, Land use change and Forestry. In: Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (Eds.), . Hayama, Japan. on the web at http://www.ipccnggip.iges.or.jp/public/gp/gpgaum.htm.
Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D., Minkkinen, K., Byrne, K.A., 2007. How strongly can forest management influence soil carbon sequestration? Geoderma 137, 253–268.
Jastrow, J.D., Miller, R.M., 1997. Soil aggregate stabilization and carbon sequestration: feedbacks through organo-mineral association. In: Lal, R., Kimble, J.M., Follet, R.F., Stewart, B.A. (Eds.), Soil Processes and the Carbon Cycle. CRC Press, New York, pp. 207–230.
Jensen, L.S., Salo, T., Palmason, F., Breland, T.A., Henriksen, T.M., Stenberg, B., Pedersen, A., Lundström, C., Esala, M., 2005. Influence of biochemical quality on C and N mineralization from a broad variety of plant materials in soil. Plant and Soil 273, 307–326.
Johnson, D.W., 1992. Effects of forest management on soil carbon storage. Water, Air, and Soil Pollution 64, 83–120.
Laganière, J., Angers, D.A., Parė, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Global Change Biology 16, 439–453.
Lal, R., 1997. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO_2-enrichment. Soil & Tillage Research 43, 81–107.
Leifeld, J., Kögel-Knabner, I., 2005. Soil organic matter fractions as early indicators for carbon stock changes under different land-use? Geoderma 124, 143–155.
Leifeld, J., Zimmermann, M., Fuhrer, J., Conen, F., 2009. Storage and turnover of carbon in grassland soils along an elevation gradient in the Swiss Alps. Global Change Biology 15, 668–679.
Liang, B.C., Campbell, C.A., McConkey, B.G., Padbury, G., Collas, P., 2005. An empirical model for-estimating carbon sequestration on the Canadian prairies. Canadian Journal of Soil Science 85, 549–556.
Liski, J., Palosuo, T., Peltoniemi, M., Sievänen, R., 2005. Carbon and decomposition model Yasso for forest soils. Ecological Modelling 189, 168–182.
Liski, J., Lehtonen, A., Palosuo, T., Peltoniemi, M., Eggers, T., Muukkonen, P., Mäkipää, R., 2006. Carbon accumulation in Finland's forests 1922–2004—an estimate obtained by combination of forest inventory data with modeling of biomass, litter and soil. Annales of Forest Science 63, 1–12.
Littel, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., 1996. SAS® System for Mixed Models. SAS Institute, Cary, NC.
Martel, Y.A., Deschenes, J.M., 1976. Effects of cultivation and age of cultivated meadows on evolution of carbon, nitrogen and soils in Quebec. Canadian Journal of Soil Science 56, 373–383.
MMM, 2009. Yearbook of Farm Statistics 2008. Ministry of Agriculture and Forestry. on the web at http://www.mmmtike.fi/en/index/statistics_information_services/agricultural_statistics.html.
Murty, D., Kirschbaum, M.U.F., McMurtrie, R.E., McGilvray, H., 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. Global Change Biology 8, 105–123.
Muukkonen, P., Mäkipää, R., 2006. Empirical biomass models of understorey vegetation in boreal forests according to stand and site attributes. Boreal Environment Research 11, 355–369.
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.
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.
Peltoniemi, M., Mäkipää, R., Liski, J., Tamminen, P., 2004. Changes in soil carbon with stand age—an evaluation of a modelling method with empirical data. Global Change Biology 10, 2078–2091.
Peltoniemi, M., Palosuo, T., Monni, S., Mäkipää, R., 2006. Factors affecting the uncertainty of sinks and stocks of carbon in Finnish forest soils and vegetation. Forest Ecology and Management 232, 75–85.
Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6, 317–328.
Repola, J., Ojansuu, R., Kukkola, M., 2007. Biomass functions for Scots pine, Norway spruce and birch in Finland. Working Papers of the Finnish Forest Research Institute, 53, pp. 1–28.
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.
Ryan, M.G., Melillo, J.M., Ricca, A., 1990. Comparing methods for determining proximate carbon fractions of forest litter. Canadian Journal of Forest Research 20, 166–171.
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.
Statistics Finland, 2010. Greenhouse gas emissions in Finland 1990–2008. National Inventory Report under the UNFCCC and the Kyoto Protocol. Submission to the UNFCCC April 15 2010 on the web at http://www.stat.fi/greenhousegases.
Thornley, J.H.M., Cannell, M.G.R., 2001. Soil carbon storage response to temperature: a hypothesis. Annals of Botany 87, 591–598.
Törmälä, T., Raatikainen, M., 1976. Primary production and seasonal dynamics of the flora and fauna of the field stratum in a reserved field in Middle Finland. Journal of the Scientific Agricultural Society of Finland 48, 363–385.
Torn, M.S., Trumbore, S.E., Chadwick, O.A., Vitousek, P.M., Hendricks, D.M., 1997. Mineral control of soil organic carbon storage and turnover. Nature 389, 170–173.
Trofymow, J.A., CIDET Working Group, 1998. The Canadian Intersite Decomposition ExperimenT (CIDET). Project and site establishment report. Information report BC-X-378. Pacific Forestry Centre, Victoria, Canada.
Tuomi, M., Thum, T., Järvinen, H., Fronzek, S., Berg, B., Harmon, M., Trofymow, J.A., Sevanto, S., Liski, J., 2009. Leaf litter decomposition—estimates of global variability based on Yasso07 model. Ecological Modelling 220, 3362–3371.
Tuomi, M., Laiho, R., Repo, A., Liski, J., 2011. Wood decomposition model for boreal forests. Ecological Modelling 222, 709–718. doi:10.1016/j.ecolmodel.2010.10.025.
Vesterdal, L., Ritter, E., Gundersen, P., 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecology and Management 169, 137–147.
Westman, C.J., 1995. A simple device for sampling of volumetric forest soil cores. Silva Fennica 29, 247–251.