Chapter 2

The effects of forest management on the soil carbon stocks in a Sitka spruce (*Picea sitchensis*) chronosequence on peaty gley soil.

2.1 Introduction

Soils constitute a significant reservoir of carbon in both organic and mineral forms and can play an important role in the greenhouse effect, by mitigating it through removing CO_2 from the atmosphere, or conversely contributing carbon to the atmosphere. The mineral forms (mostly calcite and dolomite) are relatively stable, whereas the organic forms are more readily reactive (Rosenweig and Hillel, 2000). About 75% of total terrestrial C is stored in the world's soils (Eswaran *et al*., 1993). There are various estimates of global soil carbon with numbers between 700 and 2946 Pg C (1 Pg = 10^{15} g) (Post *et al*., 1982). Additional estimates and further details are given in Chapter 1. Accurate global estimates are difficult to obtain as soil C content exhibits very high spatial variability. Also, reliable data (particularly for bulk density) are generally unavailable to compute volumetric composition and the effects of vegetation and land use changes are often confounded (Eswaran *et al*., 1993).

The amount of C stored in the soil is the balance between inputs of organic material from the biota, which depends on the type of vegetation and its productivity at a particular site, and losses primarily through soil respiration (Post *et al*., 1982). Forests are continuously recycling C through photosynthesis and respiration; and the rate of sequestration is dependent on species, site conditions, disturbance and management practices (Dixon *et al*., 1994). Soil organic matter content has a large

influence on many soil physical properties, such as water holding capacity, aeration, drainage and bulk density. Changes in these soil properties by site preparation or following timber harvesting and loss of organic matter may reduce the overall site productivity and stability (Jurgensen *et al.*, 1997). For the maintenance of the long-term fertility of a site it is important to know the fate of the organic matter after clearfelling. Soil respiration and stream water export cause permanent losses of soil organic matter that will affect the site fertility in the long term, while relocation into the mineral soil by mixing or leaching preserves the total organic matter pool (Johnson *et al.*, 1991).

Currently, forest plantations globally occupy an area of 187×10^6 ha; however, they account for less than 5% of the global forest cover (FAO, 2000). Recent trends towards harvesting younger stands raise concerns on how such forest management will impact on soil processes and global carbon sequestration as well as site productivity and forest biodiversity (Harmon *et al.*, 1990; Johnson, 1992). Forest plantations, particularly in temperate regions, are usually planted in areas that did not have forest before, such as grasslands or abandoned agricultural land. In order to establish a forest preparing the site (burning of remaining vegetation, creation of drainage ditches and ploughing of the soil for the planting of trees) is sometimes necessary and that can cause decreases in soil C. These losses can be attributed to the mechanical disturbance for tree planting, which can accelerate decomposition by disturbing soil structure and breaking soil aggregates (Turner and Lambert, 2000; Guo and Gifford, 2002). However, Paul *et al.* (2002) suggest that soil C decrease following forest establishment may be attributed mainly to the lack of plant growth and thus lack of C input into the soil, rather to soil disturbance during site preparation. Johnson (1992) concluded that while site preparation prior to the establishment of a new forest leads to decreases in soil C, the magnitude of these losses is dependent upon the severity of the disturbance. Mechanical site preparation will invariably increase spatial variability in soil C, making it difficult to detect any slight changes in soil C (Paul *et al.*, 2002).

Guo and Gifford (2002) conducted a meta analysis of the literature on the effects of land use changes on soil C stocks. They concluded that changing from pasture (including natural grassland) to conifer plantation decreases soil C stocks by 12% average. As already mentioned, Jobbagy and Jackson (2000) found that the relative distribution of soil organic C in the top metre of soil was deeper in grasslands than in forests. That indicates that more organic matter is available for decomposition in the upper part of the soil in a forest than in grassland. It also means that lowering the water table by afforestation brings more C into the aerated zone where it can be decomposed. This is also supported by Guo and Gifford (2000), who estimated that although trees deposit a larger fraction of inputs in the surface soil they are subject to higher decomposition that might involve less formation of soil organic matter. Post and Kwon (2000), in a literature review of the effects of land use change on soil C accumulation, concluded that a change from cultivated land to pine-dominated forests in the cool temperate zone resulted in a net loss of soil organic matter. They concluded that the growth of woody plants, in some environments, can result in a decrease in total soil organic C, despite the greater production of recalcitrant material, as the inputs are in the surface soil, where decomposition conditions are generally more favourable. However, although land use change can lead to soil C losses, the growth of trees can compensate by C accumulation in the living biomass. Laine and Vasander (1991) evaluated the effects of drainage and forest establishment on the C balance of a peat bog in Finland and found an overall ecosystem C increase of 9% due to increases in tree, litter and peat C, which compensated for any loss of peat C due to increased decomposition rate.

Substantial losses of C from vegetation and soils can also be caused by clearfelling (Houghton, 2003). Soil carbon storage is likely to decline initially after clearfelling, because of low C inputs from plant production. From an ecosystem perspective, there are three major pathways for C loss during and following logging. Carbon is exported from the ecosystem in biomass for use in wood products. The significance

of this flux depends on the harvesting methods used (i.e. selective cutting, whole tree harvesting, etc.) and the relative amount of C in the forest biomass and soil. Accelerated loss of C via soil respiration also can be significant when post-harvest conditions stimulate microbial activity. Finally, the loss of forest canopy leads to increased water fluxes in soils, which may result in increased export of particulate and dissolved C in drainage waters (Johnson *et al*., 1995).

On the other hand, regenerating forests and plantations may represent important carbon sinks as a result of carbon storage in both plant biomass and soils (IPCC, 1996). In the initial phases during re-growth, C is sequestered in the ecosystem through photosynthetic assimilation. For a given site, forest management may thus result in net accumulation or loss of C, depending on the balance between net photosynthesis and the various processes resulting in C export (Johnson *et al*., 1995). Carbon accumulation rates during afforestation depend on tree species and the length of the rotation (Thuille *et al.*, 2000). Johnson and Curtis (2001) conducted a meta-analysis of the effects of forest harvesting on soil C storage. This updated the previous review by Johnson (1992). They concluded that losses of soil C after harvesting and reforestation are generally negligible, with either no effects or very small changes of around ± 10%. Johnson and Curtis (2001) also concluded that forest harvesting on average had little effect on soil C. However, while sawlog harvesting increased soil C by 18%, whole-tree harvesting caused a decrease of 9%. A limitation of both reviews was that an analysis of organic horizons was not included and the conclusions were based solely on A horizons. Pennock and van Kessel (1997) studied the effects of clear felling on soil organic carbon in mixed hardwood (*Populus tremuloides - Picea glauca*) stands in Canada. They found that in the short term (1 to 5 years) soil organic C increased compared to mature uncut stands, probably due to increased additions and decomposition of slash and incorporation of the surface organic layer into the mineral horizon.

In Britain, about 315,000 ha of shallow peatlands (peaty gley) have been planted with coniferous forests, mostly Sitka spruce (Cannell *et al*., 1993). Afforestation on peaty gley soils causes an increase in the rates of oxidation of the peat layer due to improved aeration brought about by land drainage and a lowering of the water table under the tree canopy (e.g., King *et al*., 1986). Growing of trees sequesters carbon in the above-ground biomass as well as in the litter layer and soil. However, whether afforestation results in a net C sequestration depends on the rate of peat oxidation. Cannell *et al*. (1993) estimated that if the rates of oxidation in drained peat are in the range of 1-2 t C ha^{-1} y^{-1}, afforestation will give a benefit for one or two rotations (they considered a rotation of 59 years), but if rates of oxidation are more than 3 t C ha^{-1} y^{-1}, there might not be a net benefit from afforestation.

The common methods for measuring soil C are wet chemical oxidation, the gravimetric Loss on Ignition (L.O.I.) method and dry combustion by C/N analysers (Allen, 1989). The wet oxidation method is not considered to give reliable results, as it tends to underestimate soil C (Sollins *et al*., 1999). The L.O.I., which measures C content by measuring mass loss following high temperature combustion at approximately 500 °C, is a relatively inexpensive method, as it requires only a muffle furnace, which is easy to operate and maintain. However, L.O.I may not be the true measure of organic matter, because at the temperature of ashing some bound water is lost from the clay minerals and is included in the overall loss, but this error is more serious in soils low in organic matter (Allen, 1989). Dry combustion by C/N analysers is recommended by Sollins *et al*., (1999) as the most suitable method of total C determination in the soil as it gives precise and accurate measures. Dry combustion analysers oxidise samples at high temperature (approximately 1000 °C) then measure the CO_2 gas evolved by infrared gas absorption analysis (IRGA) or gas chromatography (GC). The use of C/N analysers is expensive and the high temperature oxidation liberates C from carbonate minerals, thus a separate analysis to correct for carbonate-C is required. The ignition temperature at L.O.I. is below that at which carbonate C decomposes (Howard, 1965). Sollins *et al*. (1999)

suggested the use of L.O.I., as long as it is checked against a dry-combustion method.

Soil carbon storage is an important factor in the long term ecosystem stability but is difficult to detect small changes in a large pool because of very high spatial variability, which increases with increasing C content (Conen *et al*., 2003). Conen *et al*. (2003) found a significant positive relationship between mean carbon content and variance (R^2=0.49) and the relationship became stronger (R^2=0.71) when specific sites with specific disturbance history and microtopographical features were isolated from the regressions. They estimated that in order to detect a change of 5 t C ha^{-1} over time, the sample size varied from 7 to almost 4000 samples for the various sites they examined; the number of samples depending on the mean C content of the sites. For Harwood Forest, the sample size required was 2065 (3729) at the plot (forest) scale respectively.

The objective of the study was to examine the effects of afforestation with Sitka spruce, and forest management, on the C stocks in peaty gley soils. This was perceived as a way of gaining some insight into the long-term effects of afforestation, forest harvesting, reforestation and forest growth on soil C stocks.

2.2 Materials and Methods

2.2.1. Site description

A full site description was given in the Introduction. Briefly, in Harwood Forest (Northumberland, N.E. England) Sitka spruce (*Picea Sitchensis*) stands varying in age between 40-years-old (first rotation), 12-years-old, 20-years-old and 30-years-old (all second rotation), together with a clearfelled site and a natural unplanted

grassland as the control site, all on peaty gley soil, were selected to provide a chronosequence.

2.2.2 Soil sampling

Soil sampling took place during the summers of 2000 and 2001. Two different sampling procedures were followed in the field. The sampling carried out in 2000, conducted as part of my MSc dissertation, had the primary objective of determining the soil C stocks of the chronosequence of sites chosen for eddy covariance measurements. Therefore site selection was largely constrained to the sites already chosen as part of the CARBO-AGE project. The sampling conducted in 2001 had the primary objective expanding the investigation to a wider area inside the forest by a more careful selection of paired sites and by increasing the number of replicate stands.

2.2.2a The 2000 study

In 2000, soil sampling was done according to a nested factorial design (Anderson and McLean, 1974). Soil samples were taken from five stands of different age, approximately: 40-years-old (40-yr), 30-years-old (30-yr), 20-years-old (20-yr), 12-years-old (12-yr), a recently clearfelled site, where young trees had just been replanted and a control site (UN). The clearfelled site was subsequently excluded from this analysis, as it was discovered that the soil type was not peaty gley. Samples were also collected from an unplanted "control" site. All the sites were second-rotation stands, except for the 40-years-old one, which was a first-rotation stand. In each stand five plots were randomly selected and in each plot a soil core was taken from each of eight points at randomly selected distances of 0 to 10 m from the centre of the plot, i.e., so that the points fell within a circle of 10 m radius from the centre of

the plot. The eight points were located in clockwise order, each one at an angle of 45° from the previous one. The samples were taken with a soil auger 45 cm long and of about 2 cm in diameter. The samples were visually separated into three layers: litter, organic and mineral. The samples were put in polyethylene bags and stored in a cold room pending preparation and analysis.

2.2.2b The 2001 study

The 2001 study sites were selected to be as close to each other as possible in order to avoid variability due to topographic effects. In some cases, this resulted in the selection of stands adjacent to each other of the same or different age. In other cases, distances of about 1 km or so were tolerated if it could be reasonably assumed that the topography and soil type were very similar. The sites sampled were: One unplanted site located close to one of the 40-yr first rotation stands, three 40-year-old stands, a clearfelled site (CF, clearfelled 18 months prior sampling), three 12-year-old, two 20-year-old and two 30–year-old stands. All sites were second-rotation stands, except the 40-years-old one, which was a first-rotation stand. Only one plot was sampled in the unplanted grassland surrounding one of the 40-yr old stands and one in the clearfelled site because comparable sites similar conditions and soil type were unavailable. Furthermore, three plots were also placed in the unplanted rides inside the 40-yr stands, to test whether a difference existed between relatively undisturbed soil and ploughed soil very close to each other. Rides are unplanted strips regularly spaced within mature forests. They are usually 8 to 10 metres wide and are separated from the adjacent stands by drainage ditches on both sides. It was recognised that unplanted strips do not represent a perfect control for the planted stands, because of partial shading and litterfall from the forest. The depth of the water table and the soil temperature were also different from larger unplanted areas. Random plots of 15 x 4 m were selected using the same co-ordinates in paired sites in order to get parallel plots under similar topography. In each plot 9 points were

randomly sampled. The samples were taken using a manually driven soil corer with a slide hammer attachment (Giddings Machine Company, Inc., U.S.) (5.5 cm diameter) to a depth of about 50 cm. The samples were kept in the corer liners and transferred to the lab, where the depth of the total core and of each layer (litter, organic and mineral layer) was measured, and the layers separated and kept in polythene bags in the freezer (-4 °C) till further analysis.

The sampling depth difference of about 5 cm between the two corers (the soil auger sampled to a depth of 45 cm and the soil corer sampled to a depth of 50 cm) was not adjusted for since this depth involves soil in the mineral layer, the C content of which is very much lower than that of the upper organic horizons. Since the objective was not to measure the total soil C in the three layers, but rather to a fixed depth, this discrepancy results in only a minor difference. Tests were also carried out to compare the estimates obtained with the auger with those obtainable with the 5.5 cm corer. Twenty-eight individual cores were taken adjacent to one another with both corers in several different plots. The cores were separated into three layers as above and a t-test conducted to test whether significant differences existed in the estimation of soil C stocks by the two instruments. The values given by the two methods were not significantly different ($P>0.05$).

2.3 Sample Analysis

2.3.1 Sample preparation

The samples were oven-dried at 105 °C for 24 hours (Jackson 1958; Allen 1989). Coarse fragments were removed by hand and then the soil was ground to pass a 0.5 mm mesh. The samples were also tested for moisture after grinding. Sub-samples of about 5 g were oven-dried at 105 °C (Allen 1989) for three hours and the moisture

content was estimated. The moisture content was found to vary between 0.95 and 4%.

2.3.2 Measurements for the determination of C

All the samples from the first two plots of each stand sampled in 2000 were analysed both by C/N analyser and by loss on ignition (L.O.I.). The remaining three plots from the 2000 sampling as well as the samples from the 2001 study were analysed by L.O.I. only. These values were then corrected using the calibration functions obtained from the comparison of methods.

2.3.2a Measurements of C by C/N analyser

Sub-samples of about 4 mg were taken from the litter and the organic layer samples and 10 mg sub-samples were taken from the mineral layer samples. They were weighed into cups made of tin foil and then they were combusted in a Carlo-Erba NA 2500 C/N analyser to determine their C concentration. Each combustion run contained 50 samples of soil and 7 samples of standards of known concentration. Total C was assumed to equal organic C, as the soils from Harwood are not calcareous.

The effect of sub-sampling on the accuracy of the measurements was evaluated by measuring % C on repeated sub-samples. The results showed differences of no more than 1% among the sub-samples.

A C/N analyser measures the percent of C (%) in the combusted material. The mass of C in the core sample was calculated from the equation:

$$M_C = \frac{M_d \times C(\%)}{100}$$ (1)

where: M_C is the total mass of C in the sample, M_d is the dry mass of the sample, and C (%) is the percentage C given by the C/N analyser.

2.3.2b Measurement of C by Loss on ignition (L.O.I.)

Samples of approximately 1 g (Allen, 1989) were weighed and then ignited in a furnace at 500 °C for 5 hours. After cooling the samples were weighed again and the percentage loss of mass (L%) was calculated. The estimated mass of carbon in each sample (M_C) was given by the equation

$$M_C = M_d \times \frac{L}{100} \times \frac{1}{X}$$ (2)

where L is the % mass decrease after loss-on-ignition, and X represents the estimated ratio of % mass loss on ignition to % organic carbon present as determined by C/N analyser, determined empirically from the parallel analyses of samples by C/N analyser and L.O.I.

The total mass of C (t C ha^{-1}) estimated for each sample and for both methods was estimated from the equation:

$$M_{total} = M_C \times \frac{10^4}{A} \times \frac{1}{10^6} = \frac{M_C}{A \times 100}$$ (3)

where M_{total} is the total estimated mass of C (t C ha^{-1}), A is the auger cross-sectional area (m^2) (0.00037 for the auger and 0.002826 for the corer) and 100 converts mass into tonnes.

The C (%) obtained from the C/N analyser and the mass loss (L%) obtained from the L.O.I. for the two first plots of each stand (in total 230 samples), were found to be highly significantly related to each other (see Results section). Also a significant effect of soil layer was found. Therefore, separate regression equations were calculated for each layer and were used to estimate the C (%) for the remaining plots of each stand. The same regression equations were used in the summer 2001 study as well.

2.3.2c Soil bulk density

Bulk density is a key soil physical property for determining the degree of compactness; it is also used as an indicator of soil aeration status (together with soil water content), and it is dependent on the composition as well as the structural conditions. It therefore tends to vary with the soil texture, which determines the packing of the soil particles (Chan, 2002).

Soil bulk density was calculated for the samples from the 2001 study only, because no layer depths had been measured during the 2000 study. Bulk density was calculated with the following formula:

$$P_b = M/V \tag{4}$$

Where P_b is the bulk density (g cm^{-3}), M is the dry mass of a given soil sample (g) and V its fresh volume (cm^3).

2.3.3 Statistical analysis

The data from both study years were first analysed separately to determine whether significantly different patterns emerged as a result of the different methodologies employed. Because the patterns were similar, they were eventually combined and the mean was used as the soil C stock for each site. Differences among age classes were tested using one-way ANOVA, using plot averages as the unit for analysis. The slopes of the regression equations for the relationship between the % C obtained from C/N analyser and the % mass loss from L.O.I. methods for each of three layers (litter, organic, mineral) was tested by ANCOVA (Analysis of Covariance) in SPSS (1998). A Mann-Whitney U test was employed for the comparison between the 40-yr stands and the ride plots, as the ride values did not follow a normal distribution. One-way ANOVA (Analysis of Variance) was performed using SAS and equations were fitted using Sigma Plot. All the probabilities were tested at the 5% significance level.

2. 4 Results

## 2.4.1 Equations for predicting soil C using the L.O.I. method

When the mass loss (%) by L.O.I was plotted against the C concentration (%) obtained by the C/N analyser, a significant linear relationship was obtained (R^2=0.98, Figure 1). The ANCOVA analysis revealed that there were significant differences in the slopes of the regression equations among the 3 layers (P=0.04) and thus the use

of the common regression equation would not be appropriate. Thus, separate regression equations were fitted for each layer:

$C (\%) = 0.513 L(\%) - 0.092$, $R^2 = 0.99$ for the litter layer (5)

$C(\%) = 0.542 L(\%) + 0.184$, $R^2 = 0.99$ for the organic layer (6)

$C(\%) = 0.533 L(\%) - 0.700$, $R^2 = 0.99$ for the mineral layer (7)
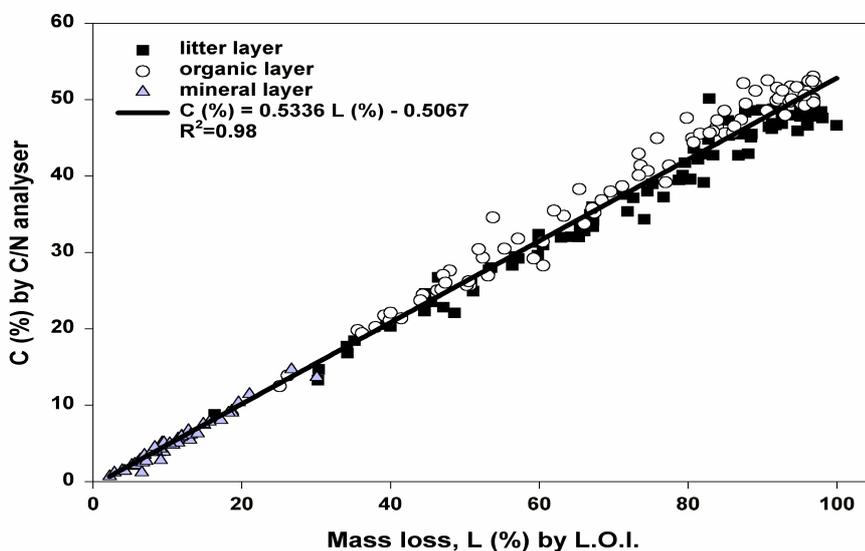
Figure 1: Linear regression ($R^2 = 0.98$) between mass loss (%) from L.O.I and C (%) by C/N analyser for 230 samples from litter, organic and mineral layers.
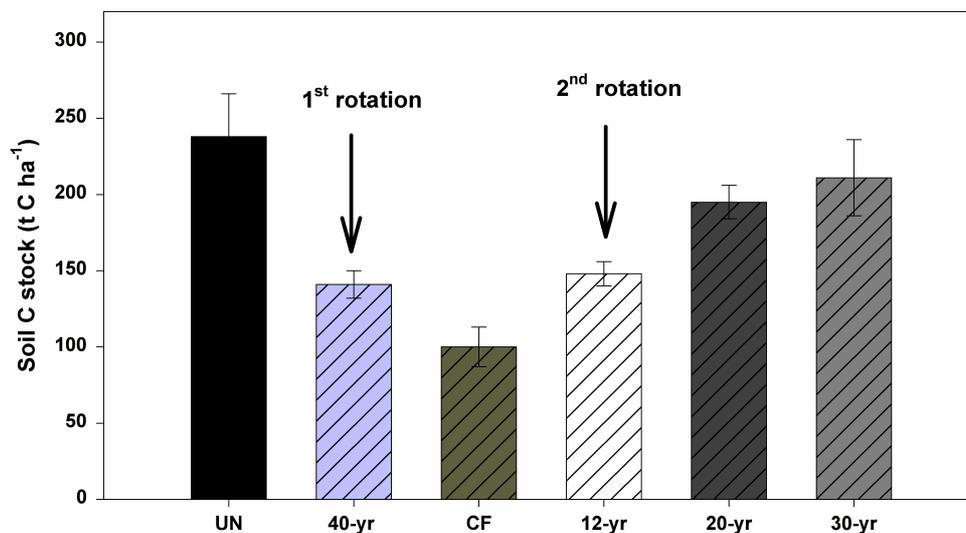
2.4.2 Soil C stocks along the chronosequence

The changes in soil C stocks along the chronosequence are shown in Figure 2 as well as in Table 1. The planting of trees in the grassland (UN) led to a significant decrease in soil C stocks from 238 ± 28 t C ha^{-1} to 141 ± 9 t C ha^{-1} at the end of the first

rotation (40-yr stand, P<0.001). Clearfelling led to a further small but significant decrease (P=0.02) in soil C to 100 ± 13 t C ha^{-1}. C started accumulating again during the second rotation with soil stocks increasing curvilinearly (y= 98.67 +((374.61* x)/(66.85 +x), R^2=0.98, Figure 3) from 148 ± 8 to 195 ± 11 and 211 ± 25 t C ha^{-1} in the 12-yr, 20-yr and 30–yr stands, respectively (Figure 3). The soil C stocks in the 30-yr were not significantly different from the UN site (P=0.5). However, although the total was the same, there were differences between layers, with the 30-yr old stand having a higher amount of C in the litter and mineral layer, while the natural grassland contained higher amounts of C in the organic layer (Table 1). Overall, soil C stocks along the whole chronosequence were significantly different (P<0.001). Soil C stocks were significantly different (P<0.001) when trees grew in the second rotation. The C stock in the 12-yr stand was not significantly different from CF or 20-yr (P>0.05), but is was significantly different from the 30-yr stand (P=0.003). The 20-yr was not significantly different from the 30-yr (P=0.07).

Table 1: Soil C stocks (t ha^{-1}) in the litter, organic and mineral layer and total along the Sitka spruce chronosequence. The numbers in the brackets indicate the standard error of the mean.

site	Litter layer C (t ha^{-1})	Organic layer C (t ha^{-1})	Mineral layer C (t ha^{-1})	Total C (t ha^{-1})
UN	24.5 (4.8)	204.6 (27.5)	8.9 (2.9)	238 (28)
40-yr	33.0 (5.1)	54.5 (12.5)	54.0 (9.0)	141 (9)
CF	7.4 (0.6)	31.6 (15.9)	64.4 (15)	100 (13)
12-yr	16.0 (0.5)	97.7 (20.5)	34 (12.5)	148 (8)
20-yr	22.3 (1.4)	108.3 (19.6)	64.5 (13.1)	195 (11)
30-yr	37.8 (6.3)	116.2 (40.5)	57.3 (14.2)	211 (25)

Figure 2: Soil C stocks (t ha^{-1}) along the Sitka spruce chronosequence. The planting of the trees on natural grassland (UN) leads to a decrease in soil C, which is further enhanced by clearfelling of the trees (CF). Soil C starts accumulating again when trees grow in second rotation. The vertical bars represent the standard error of the mean.
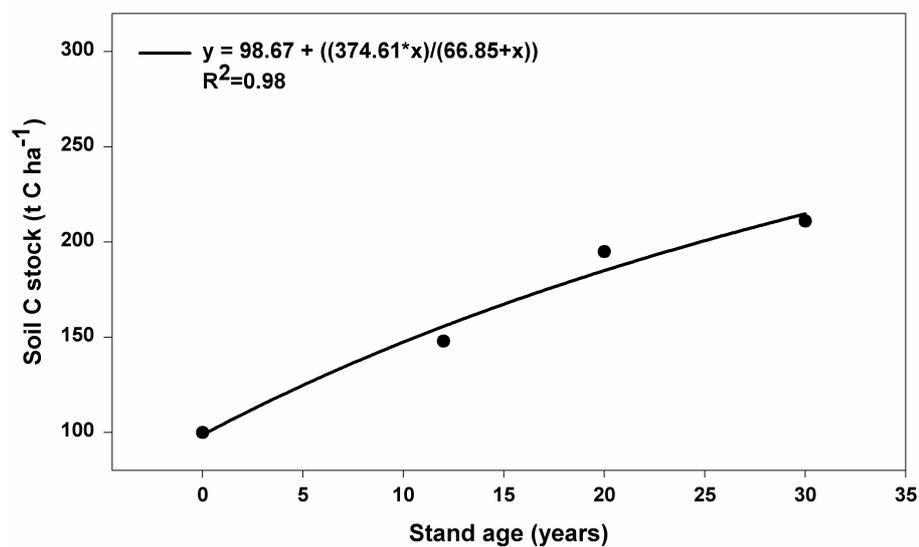
Figure 3: Increase in soil C stocks (t ha^{-1}) in stands growing in second rotation after clearfelling. Year 0 since clearfelling refers to the CF site.

Table 1 also shows the soil C stocks in the litter, organic and mineral layers at each site. The UN had 24.5 ± 4.8 t C ha^{-1} in the litter layer and the planting of the trees and the production of litter led to an increase to 33.6 ± 5.1 t C ha^{-1} at the 40-yr (P=0.02). Clearfelling of the trees and the absence of above-ground litterfall reduced the litter C to 7.4 ± 0.6 t ha^{-1} (P<0.0001) and thereafter litter layer C increased with age and increased litterfall (16 ± 0.5, 22.3 ± 1.4 and 37.8 ± 6.3 t ha^{-1}, in the 12, 20 and 30-yr stands respectively, P<0.05).

Table 2 shows the depth of litter and mineral layer in the chronosequence. Planting of trees increased the litter layer depth from 3.8 ± 0.9 cm in the UN site to 4.6 ± 0.5 cm in the 40-yr stand, however, the increase was not significant (P=0.4). Litter layer in the ride was also shallower than in the 40-yr stand (1.8 ± 0.3 cm, P<0.001). Clearfelling caused a significant decrease the litter layer depth to 2.7 ± 0.4 cm (P=0.01) and with the re-growth of trees in second rotation it started increasing again (3.6 ± 0.2, 4.7 ± 0.6 and 7 ± 0.3 cm, in the 12-yr, 20-yr and 30-yr stands, respectively, all P<0.05). Planting of trees in the grasslands caused a significant decrease in the depth of the organic layer from 20.3 ± 1.3 in the UN to 12.8 ± 0.7 cm in the 40-yr (P<0.001). The ride had also a significantly deeper organic layer (16.1 ± 0.8 cm, P=0.006). Clearfelling caused a significant decrease (6.3 ± 1.4 cm, P=0.01). Organic layer depth started increasing in second rotation again and the 12-yr stand had significantly deeper organic layer than the CF site (17.6 ± 1.7 cm, P<0.001), although not significant different from the 20-yr stand (17.4 ± 1.5, P=0.9). The organic layer depth in the 30-yr stand was 20.9 ± 1.4 cm and it was significantly deeper than the 20-yr stand (P=0.04).

Table 2: Depth (in cm) in the litter and organic layer across the chronosequence sites in Harwood. The numbers in the brackets indicate the standard error of the mean.

site	Litter layer depth (cm)	Organic layer depth (cm)
UN	3.8 ± 0.9	20.3 ± 1.3
40-yr	4.6 ± 0.5	12.8 ± 0.7
Ride	1.8 ± 0.3	16.1 ± 0.8
CF	2.7 ± 0.4	6.3 ±1 .4
12-yr	3.6 ± 0.2	17.6 ± 1.7
20-yr	4.7 ± 0.6	17.4 ± 1.5
30-yr	7 ± 0.3	20.9 ± 1.4

The average soil C stock in the rides next to the 40-yr stands was 130 ± 14 t C ha^{-1}, while the average in these 40-yr stands was 127 ± 17 t C ha^{-1}. Figure 4 shows the pairs of plots in the rides and the adjacent 40-yr stand that were sampled. Statistical analysis revealed no significant differences (P=0.73) between rides and 40-yr stands. However, there were different patterns in the distribution of C. The C in the litter layer of the 40-yr stands was significantly higher than the C contained in the litter layer of the rides (19.6 ± 1.9 and 6.7 ± 1.1 t C ha^{-1} respectively, P<0.001). Soil C stocks in the organic layer of the 40-yr stands were lower but not significantly different (73.4 ± 5.8 and 92.1 ± 10.8 t C ha^{-1}, in the 40-yr stands and rides respectively, P=0.14). No significant differences were observed in the mineral layer, either (38 ± 3.3 and 31.8 ± 6 t C ha^{-1}, P=0.35, Figure 5).
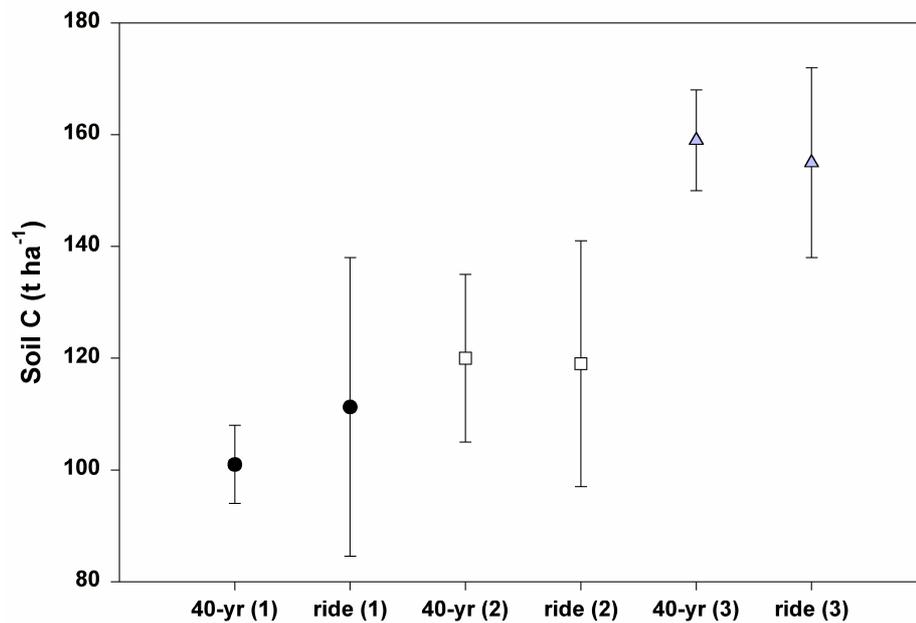
Figure 4: Soil C stocks (t ha^{-1}) in 40-yr stands and adjacent rides. The rides are original pieces of grassland that were not planted with trees. The vertical bars indicate the standard error of the mean.
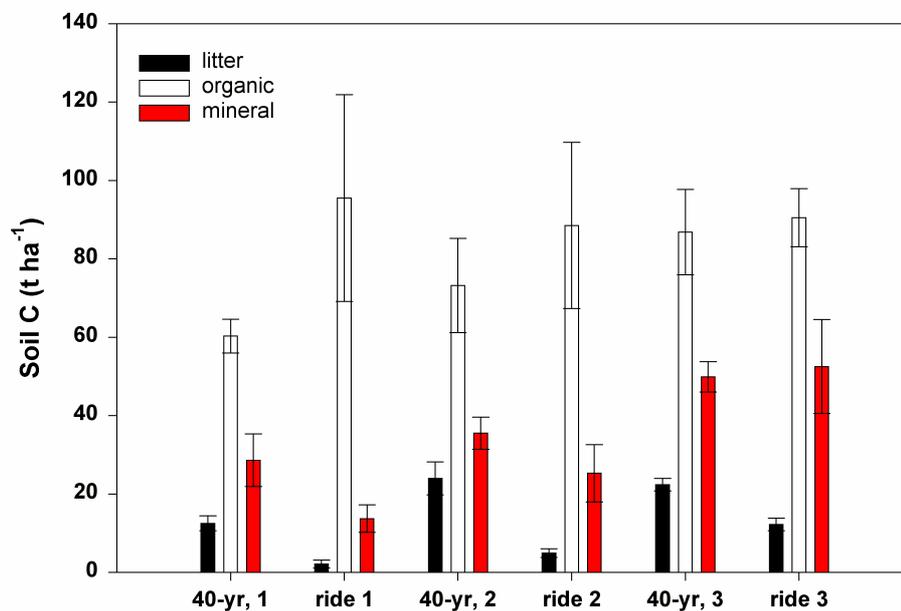
Figure 5: Soil C (t ha^{-1}) in litter, organic and mineral layer in the pairs 40-yr stands and rides. The vertical bars represent the standard error of the mean.

The concentration of soil C (%) in each layer is shown in Table 3. C concentration in the litter layer was not significantly different across the chronosequence and ranged between 33.9 and 45.3 %, increasing slightly in the second-rotation stands (33.9 ± 3.8, 36.8 ± 1.4 and 40.2 ± 3.5% in the 12-yr, 20-yr and 30-yr-old stands respectively). The unplanted grassland had the highest C concentration in the organic layer (39.9 ± 3.4 %), while planting of trees resulted in a significantly lower C concentration of 15.5 ± 2.9 % at the end of first rotation (40-yr stand) (P<0.001). Clearfelling led to a further small but not significant reduction in the organic layer C concentration of 13.8 ± 2.1% (P=0.5). C concentration increased again significantly with stand age during the second rotation, from 15.2 ± 5.1 % in the 12-yr, to 30.6 ± 1.9% in the 20-yr and 41.3 ± 1.3 % in the 30-yr stand (P<0.001). The C concentration in the mineral layer ranged from 1.8 ± 0.4 % to 2.9 ± 0.2 % and there were no significant differences between sites (P=0.2).

Table 3: C (%) concentration in the litter, organic and mineral layers across the chronosequence. The numbers in the brackets represent the standard error of the mean.

	C concentration (%)		
Site	**Litter layer**	**Organic Layer**	**Mineral Layer**
UN	45.3 (1.2)	39.9 (3.4)	1.6 (0.5)
40-yr	33.9 (3.2)	15.5 (2.9)	1.7 (0.4)
CF	33.7 (0.8)	13.8 (2.1)	2.5 (0.7)
12-yr	33.9 (3.8)	15.2 (5.1)	1.8 (0.4)
20-yr	36.8 (1.4)	30.6 (1.9)	1.7 (0.04)
30-yr	40.2 (3.5)	41.3 (1.3)	2.9 (0.2)

Soil bulk density for each layer along the chronosequence is shown in Table 4. The litter layer at all sites had a bulk density between 0.1 and 0.2 g cm^{-3}. The bulk density of the organic layer ranged between 0.34 and 0.86 g cm^{-3}. Planting of trees increased bulk density from 0.55 ± 0.1 g cm^{-3} (in the UN site) to 0.86 ± 0.2 g cm^{-3}, while the bulk density of the organic layer of the rides adjacent to the 40-yr old stands was 0.75 ± 0.05 g cm^{-3}. The soil at the clearfelled site had a bulk density of 0.49 ± 0.04 g cm^{-3}. Within the stands growing in second rotation, bulk density decreased from 0.62 ± 0.06 g cm^{-3} in the 12-yr stand to 0.40 ± 0.04 g cm^{-3} in the 20-yr stand and 0.34 ± 0.03 g cm^{-3} in the 30-yr stand. The bulk density in the mineral layer was very similar at all sites, ranging between 1.30 and 1.52 g cm^{-3}.

Table 4: Bulk densities (g cm^{-3}) in the litter, organic and mineral layer, across the chronosequence. The numbers in the brackets represent the standard error of the mean. The data are from the 2001 study only.

Bulk density (g cm^{-3})			
Site	**Litter layer**	**Organic layer**	**Mineral layer**
UN	0.11 (0.01)	0.55 (0.1)	1.42 (0.04)
40-yr (first rotation)	0.15 (0.02)	0.86 (0.2)	1.52 (0.03)
Ride	0.18 (0.03)	0.75 (0.05)	1.48 (0.04)
CF	0.14 (0.03)	0.49 (0.04)	1.35 (0.08)
12-yr	0.16 (0.01)	0.62 (0.06)	1.39 (0.05)
20-yr	0.14 (0.02)	0.40 (0.04)	1.36 (0.05)
30-yr	0.11 (0.01)	0.34 (0.03)	1.30 (0.04)

2.5 Discussion

The strong linear relationship between the mass loss obtained by L.O.I. and the C (%) by C/N analyser (R^2=0.98) for the three layers, litter, organic and mineral, indicates that L.O.I can be used as a very precise method for estimating organic C from peaty gley soils, with the significant advantage of a lower cost compared with the C/N analyser method. The coefficient of determination (R^2) of the separate equations for each layer (litter, organic, mineral) was 0.99, and analysis of covariance (ANCOVA) indicated that the slopes of the equations were significantly different (P=0.04); thus a unique equation was used for each layer. The reason for the necessity of a different equation for each layer could be the different composition of the organic matter present in each layer, with the litter and organic layers containing more C per unit of organic matter and therefore per unit of mass loss than the mineral layer. Konen *et al*. (2002) developed equations for predicting organic C content as determined by C/N analyser from L.O.I, from analysis of 255 noncalcareous samples from selected major land resource areas in the north central USA. They found significant differences in the slopes of equations between each major land resource area, indicating a unique predictive equation for each area. They concluded that possible reasons for this difference were the varying organic matter composition, the clay content, and clay mineralogy. The strong predictive equations found here indicate that L.O.I can be a precise method that can be used successfully to predict accurately C (%) in peaty gley soils in the Harwood forest area.

The total soil C in Harwood forest soils significantly differed between sites (P<0.001). The soil of the unplanted natural grassland (UN) in the forest area contained 238 ± 28 t C ha^{-1}, which is in agreement with the estimation by Cannell *et al*. (1993) of 200-240 t C ha^{-1} in shallow peat (peaty gleys and peaty ironpan soils) in Britain. The planting of trees on the natural grassland caused a decrease in soil C stocks to 141 ± 9 t C ha^{-1} at the end of first rotation (40-yr). Although there is the

possibility of natural variation contributing to the difference between the C content of the unplanted grassland and the 40-yr stand, this decrease could be due to the accelerated decomposition caused by the site preparation for drainage for the planting of trees. Indeed, the soil C content in the organic layer in the UN site was 204.6 ± 27.5 t C ha^{-1} and the layer had a depth of 20.8 ± 1.3 cm, while in the 40-yr stand the organic layer contained 54.5 ± 12.5 t C ha^{-1} in a depth of 12.8 ± 0.7 cm, indicating the subsidence of the organic layer of the previous grassland presumably due to accelerated decomposition. From the data in Table 1, a loss of about 2.5 t C ha^{-1} yr^{-1} can also be estimated due to conversion of grassland to forest, which is in agreement with Armentano and Menges (1986), who estimated that the drainage of peat for afforestation accounted for mean losses in the temperate region of 2.81 t C ha^{-1} y^{-1}. Braekke (1987) estimated that after 27 years of peatland afforestation in Norway with spruce there was a loss of 1.2 t C ha^{-1} y^{-1} in the top 40 cm (he assumed that all the dry matter loss occurred in the top 40 cm). Guo and Gifford (2002), in their meta-analysis found that land use change overall decreases soil C stocks by 9%. Furthermore, they concluded that the tree type affected the magnitude of soil C stocks after forest was planted onto pasture (where pasture includes natural grassland), with conifer trees significantly reducing soil C stocks by 12%, while broadleaves had little effect.

Jobbagy and Jackson (2000) indicated that the relative distribution of SOM in the top metre of soil is deeper in grasslands than in forests. They attributed this difference to the different relative aboveground allocation for trees and grasses, with grasses having an average root: shoot ratio of 3 or 4, and the temperate forest a ratio around 0.26, and the different rates of decomposition due to different lignin content. Tree roots are less decomposable than grass roots, hence contributing less to annual turnover.

Post and Kwon (2000) concluded that woody plants may be less effective than grasses in some environments in storing soil C, as they store a larger fraction of total

inputs than grasses on the surface, where decomposition might involve less formation of organic matter. Paul *et al.* (2002) found a significant effect of previous land use on soil C change after afforestation. After afforestation soil C tended to decrease on ex-pasture sites and increase on ex-cropping sites. The effect of former land use was even greater in the short-term (<10 years) than in the long term (>10 years).

When the total soil C stocks in the 40-yr stands were compared with the C stocks in adjacent unplanted rides, no significant difference was found (127 ± 17 t C ha^{-1} and 130 ± 14 t C ha^{-1} respectively). The litter layer in the rides had much less C content and it was much shallower, than the corresponding layer in the forest (6.7 ± 1.1 t C ha^{-1} in a depth of 1.8 ± 0.3 cm in the rides litter layer and 33.6 ± 5.1 t C ha^{-1} in a depth of 4.6 ± 0.5 cm in the 40-yr stands litter layer, P<0.001). In contrast, the organic layer in the rides was thicker and had more C (91.2 ± 10.8 t C ha^{-1} in a depth of 16 ± 0.7 cm the rides organic layer and 73.4 ± 5.8 t C ha^{-1} in a depth of 12.8 ± 0.7 cm in the 40-yr stand organic layer). The extra C in the litter in the forest is a characteristic feature of this system. The decrease in C in the organic layer in the forest was not significant at the 5% significance level (P>0.05) but the trend was in the same direction as for the comparison between the UN grassland and the 40-yr stand. The trend was also in the same direction for the the litter layer, with significantly higher C content in the litter layer of the 40-yr stand (P=0.02) than the corresponding litter layer in the UN.

The total soil C stocks in the CF site were significantly lower than in the 40-yr stand (100 ± 13 t C ha^{-1} compared to 141 ± 9 t C ha^{-1}, respectively). Clearfelling also caused a decrease in the C stocks in the litter layer (7.4 ± 0.6 t C ha^{-1} and 33.6 ± 5.1 t C ha^{-1} in the CF and 40-yr respectively) as well as in the depth of the litter layer (2.7 ± 0.4 cm and 4.6 ± 0.5 cm respectively), due to absence of litter inputs from the trees. A significant decrease was also observed in the content and depth of the

organic layer: 31.6 ± 15.9 t C ha^{-1} and average depth of 6.3 ± 1.4 cm compared to the 54.5 ± 12.5 t C ha^{-1} and 12.9 ± 0.7 cm of the 40-yr stand. Again, the decrease in the soil C could be due to increased decomposition and the lack of litter inputs. Ballard (2000) indicated that clearfelling causes changes in the temperature regime, which are likely to be significant in contributing to increased biological activity and increased rates of organic matter decomposition and in some cases increased net mineralization of nitrogen. The greater the diurnal swing in temperature in a clearfelled (un-shaded) area the greater the decomposition is likely to be, because of an overall increase in microbial activity (Smith *et al.*, 2003).

At Harwood, the stands growing in second rotation after clearfelling showed a strong curvilinear increase in soil C stocks (R^2=0.96) with time since planting. The soil C increased from 100 ± 13 t ha^{-1} in the CF to 148 ± 8 t ha^{-1} in the 20-yr, 195 ± 11 t ha^{-1} in the 20-yr and to 211 ± 25 t ha^{-1} at the 30-yr. In several North American studies, changes in soil C have been similarly studied along chronosequences, and it is clear that the presence of slash in the topsoil may have altered some of the measurements. Black and Harden (1995) studied the soil carbon storage along a mixed conifer forest chronosequence in California and found a loss of 30 t ha^{-1} of organic C from the soil within 1-7 years after harvesting. Knoepp and Swank (1997) found that soil C declined slightly in the 0 to 10 cm depth following clearfelling in a white pine plantation in the southern Appalachians while the soil C in the 10 to 30 cm depth showed no response to the tree harvesting (the measurements took place from 1979 to 1994). Smerthurst and Nambiar (1995) found that clearfelling of a *Pinus radiata* plantation in South Australia caused a decrease of about 14 t C ha^{-1} during the first 3 years after planting. However, Gholz and Fisher (1982) found that the A horizon of a 2 yr-old stand of slash pine in Florida contained approximately twice as much soil C as the other stands of the chronosequence (up to the age of 34 years). This was attributed to slash incorporated in the soil after harvesting of the stand. The effect of slash was short-lived, so by the age of 5 years, the soil C decreased to 50% of to pre-harvest levels. Ewel *et al.* (1987a) found that a clearfelled site of slash pine (*Pinus*

elliotti) in Florida contained more below-ground carbon (150 t ha^{-1}) than a 29-yr-old stand (103 t ha^{-1}). Johnson *et al*. (1995) investigated the effects of clearfelling on soil carbon dynamics in a northern hardwood forest, in New Hampshire, with dominant vegetation of American beech (*Fagus grandifolia* EHRH.), sugar maple (*Acer saccharum* Marsh.) and yellow birch (*Betula allegheniensis* Britt.) and at high elevation spruce-fir, with red spruce (*Picea rubens* Sarg.), balsam fir (*Abies balsamea* (L.) Mill) and white birch (*Betula papyrifera* var. cordifolia (Marsh) Regel). A decrease of 20 t C ha^{-1} eight years after logging was estimated, as well as a redistribution of carbon within the mineral soil, with an increase in C in the A and E horizons. Pennock and van Kessel (1997) observed that clearfelling mixed hardwood stands (*Populus tremuloides- Picea glauca*) in Canada increased the soil organic C in short term (1 to 5 years) compared to mature uncut stands (62.4 ± 0.4 s.d. t ha^{-1} and 57.7 ± 5.6 s.d. t ha^{-1} in the clearfelled and mature stands respectively), probably due to increased addition and decomposition of slash and incorporation of the surface organic layer into the mineral horizon. However, in the medium term (6-20 years after clearfelling) the soil organic carbon decreased significantly (44.1 ± 5.4 s. d. t ha^{-1}). In the CF site there was slash that was left behind after clearfelling, but the extent to which slash contributed C to the soil and prevented a greater loss of C from the site is not known. Cooper (1983) suggested that if a forest is re-established soon after clearfelling the loss of soil organic C is relatively slight. Johnson (1992) reviewed the literature on the effects of forest management on soil carbon storage and came to the conclusion that the majority of studies indicated little or no change in soil C (around 10%) following harvesting and reforestation. Schiffman and Johnson (1989) studied a second-growth plantation chronosequence of loblolly pine (*Pinus taeda* L.) in Canada, established after clearing and site preparation on clayey soils and observed that soil surface C stocks (defined as those in the upper 10 cm of the soil) were not significantly different along the chronosequence; the plantation maintained about 185 t C ha^{-1} over a 26-year-old chronosequence. Covington (1981) studied a secondary succession of northern hardwood stands in the White Mountains of New Hampshire, and concluded that during the first 15 years following

clearfelling (degrading phase), the forest floor organic matter decreased by 30.7 t ha^{-1} (50% decline) but increased by 28 t ha^{-1} during the age of 15-64 years (rapidly aggrading phase). After that and up to the age of 200 years the rate of increase was slow, with a total further addition of only 2.7 t ha^{-1} of organic matter. Yanai *et al.* (2000) re-sampled the same stands as Covington (1981) and found that the changes in stands did not agree with the pattern of Covington (1981) and although young stands had less forest floor than old stands, this pattern could be equally well explained by the year of harvest, which is related to the method of tree harvesting as well as to the age of the stands. This is a common problem in the use of chronosequence, sites can differ for reasons other than just the time since disturbance.

In the stands at Harwood growing in second rotation, the litter layer and the organic layer C and depths increased , as the litter inputs increased with stand age. The same pattern in the litter layer C and depth was observed also by Griffiths and Swanson (2001) in a chronosequence of harvested Douglas fir (*Pseusotsuga menziesii* (Mirb.) Franco) forests in Oregon (ages 5, 15, 40 years and an old-growth stand), by Covington (1981) in northern hardwood forests in New Hampshire and by Black and Harden (1995) in mixed conifer forests in California. Krause (1998) studied the organic matter accumulation after clearfelling in the forest floor of two chronosequences (2-16 yr) of jack pine (*Pinus banksiana* Lamb.) and black spruce (*Picea mariana* [Mill] B.S.P.) in Canada. For the jack pine chronosequence the organic matter after clearfelling (2 and 4 years) was as low as 28% of the value in natural forests (50 t ha^{-1}). It rose to about 46 t ha^{-1} by the age of 12 years and remained at this level in the older plantations (14-16 years), suggesting a steady-state condition. However, for the black spruce chronosequence, organic matter accumulation was far more irregular, with a slower organic matter decomposition after clearfelling and minimum levels of organic matter between the ages of 6 and 8 years (16 t ha^{-1}). The stock of organic matter increased exponentially with age, up to 60 t ha^{-1} at 16 years. They also observed that at sites where decay-resistant organic

matter was left from the previous forest cover, the organic matter accumulation was much higher than in the natural forest.

The soil C stock in the 30-yr stand, although slightly lower than the UN stock (211 ± 25 t ha^{-1} and 238 ± 28 t ha^{-1}, respectively) was not significantly different from it. The carbon accumulation rates exceeded rates of loss after 80 years of re-growth, although the rate of increase in carbon storage is probably now declining.

Boone *et al.* (1988) estimated that it took more than 100 years for the organic layer in a Mountain hemlock (*Tsuga mertensiana* (Bong.) Carriere) forest, in Oregon, to recover to the pre-disturbance values in thickness and carbon storage. Switzer *et al.* (1979) found a steady state in the organic layer carbon pools after about 70 years of old field succession to oak-hickory-pine forests in Mississippi.

Cannell *et al*. (1993) estimated that if afforestation with conifers on shallow peatlands (peaty gley soils) causes a loss of 12 cm of peat by decomposition, then the carbon loss will exceed the amount of carbon that can ever be added to the soil by growing conifer plantations at Yield Class 12 (12 m^3 ha^{-1} y^{-1}). From the equilibrium storage graph in that paper, it was estimated that approximately the same numbers should also be representative of Yield Class 10 (10 m^3 ha^{-1} y^{-1}), applicable to Harwood Forest. From Table 2 it can be estimated that the loss of organic layer from the UN to the 40-yr stand was 7.5 cm, much lower than the limit values that Cannell *et al*. (1993) provide. However, this number has been derived as the net change from UN to 40-yr and discounts the possibility that the initial total loss may have been larger, followed by a recovery during first rotation because of the input of carbon through litterfall for many years. Finally, it seems that in a second rotation soil C accumulates again, and by the age of 40 years at the end of the second rotation the soil C stock will be approximately 238 ± 19 t C ha^{-1} (value estimated from the equation in Figure 3), approximately the same as the soil C originally in the

grassland. However, it cannot be determined at this time if this trend will continue with subsequent management of the forest, or for how many rotations.

Changes in the soil C stocks across the chronosequence were accompanied by changes in the concentration of C in the soil. C concentration in the litter layer was not significant different across the chronosequence (P=0.5). Schiffman and Johnson (1989) also found no significant differences in the C concentrations with stand age in the L layer in a chronosequence of lobolly pine plantations in Canada, between 1 and 47 years old and established on previous agricultural land or cleared land. However, afforestation in Harwood caused a significant decrease in the C concentration in the organic layer, from 39.9 ± 3.4 % in the unplanted grassland to 15.5 ± 2.9 % in the 40-yr stand at the end of the first rotation (P<0.001). The clearfelled site had a lower but not significantly different concentration (13.8 ± 2.1 %), while the C concentration in the organic layer significantly increased (P<0.001) with stand age during the second rotation. The same pattern was observed by Wilde (1964), who found that soil organic matter concentration (%) increased linearly with stand age (13 to 48 years) in red pine plantations in Wisconsin. He also found a faster increase with age (12 to 30 years) in a jack pine plantation in the same area. However, Griffiths and Swanson (2001) found no significant differences in the C concentration down to a soil depth of 10 cm in harvested stands of 5-yr, 15-yr, 40-yr old and old-growth Douglas fir stands in Canada. Smethurst and Nambiar (1995) observed that clearfelling decreased C concentration in the 0–15 cm depth from an initial value of 2.7% to 1.9%, 3 years after replanting a *Pinus radiata* plantation, in South Australia. C concentrations in the mineral layer were not significantly different across the chronosequence in Harwood (P=0.2); however, the C concentration in the mineral layer of the clearfelled site was higher (although not significantly), compared with the corresponding one of the 40-yr stand, indicating that mixing of the soil during clearfelling can result in some transfer of organic material deeper into the soil. Gholz and Fisher (1982) also found no significant differences (or trend) in the organic

matter concentrations in the mineral horizons of a slash pine chronosequence (stand ages between 2 and 34 years old) in USA.

Afforestation on natural grassland, clearfelling and replanting also caused changes in the soil bulk densities, mainly in the organic layer. Bulk densities in the litter layer of all sites ranged between 0.1 and 0.2 g cm^{-3} and increased with increasing depth. Planting of trees caused an increase in the organic layer bulk density from 0.55 $\pm$ 0.06 in the natural grassland to 0.86 $\pm$ 0.2 g cm^{-3} in the 40-yr stand at the end of the first rotation (P<0.001), accompanied by a decrease in soil C. Soil bulk density is strongly related to organic matter (Tamminen and Starr, 1994) and increases inorganic matter result in decreases in bulk density. The organic layer in the 40-yr stand did not have a significantly different bulk density from the organic layer of the ride (0.76 $\pm$ 0.05 g cm^{-3}, P>0.05), but had significantly higher bulk density than that of the CF organic layer (0.49 $\pm$ 0.04), although the opposite would be expected. Bock and Van Rees (2002) studied the effects of clearfelling on soil physical properties in white spruce forests in Canada. They found that, three years after harvesting, soil bulk densities in the LFH and the mineral layers were higher by 12 and 7%, respectively compared with the bulk density in the respective layers of an uncut stand. That was due to the accelerated decomposition of organic matter. Johnson *et al*. (1991) also found that clearfelling of hardwood and spruce forests in USA caused an increase of between 5 and 14% in bulk density in the top 20 cm of mineral soil. The increase depended on the severity of the disturbance. In our clearfelled site the low bulk density probably represents a small effect of the harvesting practices on soil compaction, and recovery since clearfelling (the site was sampled 18 months after clear felling). Frazer *et al*. (1990) also found similar bulk densities between clearfelled and uncut mixed conifer stands (0.8 and 0.9 t m^{-3} respectively) in Sierra Nevada. They attributed it to cumulative action of frost and incorporation of residues into the soil, which was indicated by higher C concentration in the clearfelled site.

Wood *et al*. (2003) investigated the effect of harvesting operations on peaty soils in England and Scotland. The operation of harvesting machines on slash roads (where logging residues are placed on the ground to form a protective layer and over which the machines travel) limited soil disturbance, since direct contact of the machine with the ground was avoided. Also, Pennock and Van Kessel (1997) found no significant differences in the soil bulk densities between mature mixed-wood stands of trembling aspen and white spruce (*Populus tremuloides - Picea glauca*) in Canada and clearfelled sites (in the surface or the sub-soil). Soil bulk density in the organic layer decreased with stand age during second rotation in Harwood, from 0.62 ± 0.06 in the 12-yr-old, to 0.40 ± 0.04 in the 20-yr-old and to 0.34 ± 0.03 g cm^{-3} in the 30-yr-old stand. This decrease in soil bulk density during the second rotation reflects increases in the soil C with age. Switzer *et al*. (1979) also found that soil bulk density decreased from 1.16 g cm^{-3} in the early stage to 0.85 g cm^{-3} in the late stage in oak-hickory-pine forest growing on abandoned agricultural land, in the South-eastern USA. They also found that bulk density decreased with organic matter concentration (R^2=0.69).

As was mentioned before, bulk density increased with increasing depth, therefore the mineral layers had the highest bulk densities. They were similar between sites (range between 1.30 and 1.5 g cm^{-3}, P>0.05), indicating that the mineral layer is not directly affected by land use changes and stand growth. Bulk density also reflects the distribution of organic matter with depth as well as soil compaction (Tamminen and Starr, 1994). Tamminen and Starr (1994) also found that soil bulk density increased with depth and remained uniform in depths of more than 20 cm. Bock and Van Rees (2002) found no significant differences in soil bulk densities of mineral layers between the clearfelled and uncut white spruce stands in Canada. Gholz and Fisher (1982) also observed no significant differences in soil bulk densities, although there was a slight increase in the soil bulk density of the A1 horizon in older stands in slash pine chronosequence (stand ages between 2 and 34 years old) in Florida.

2.6 Conclusions

Soil C content in soils with high organic matter can be determined in a cheap and accurate way by measuring weight loss by the Loss on Ignition method, making it possible to analyse many more samples than is possible with the C/N analyser method (dry combustion).

The establishment of Sitka spruce (*Picea sitchensis*) forests on former grasslands on peaty gley soils can lead to net accumulation of soil carbon during a second rotation. Seventy years after the initial afforestation, soils in Harwood Forest have recovered to essentially the same levels of C stock as in the pre-existing unplanted grassland (grazed moorland). However, the distribution of the stocks has changed, with more C stored as litter and in the mineral layers, but less inside the organic layers. It is estimated that by the age of clearfelling of the second rotation stands soil C stocks will have equalled those of the unplanted grassland. However, at this stage it cannot be anticipated whether and for how long this accumulation will continue during a third rotation.