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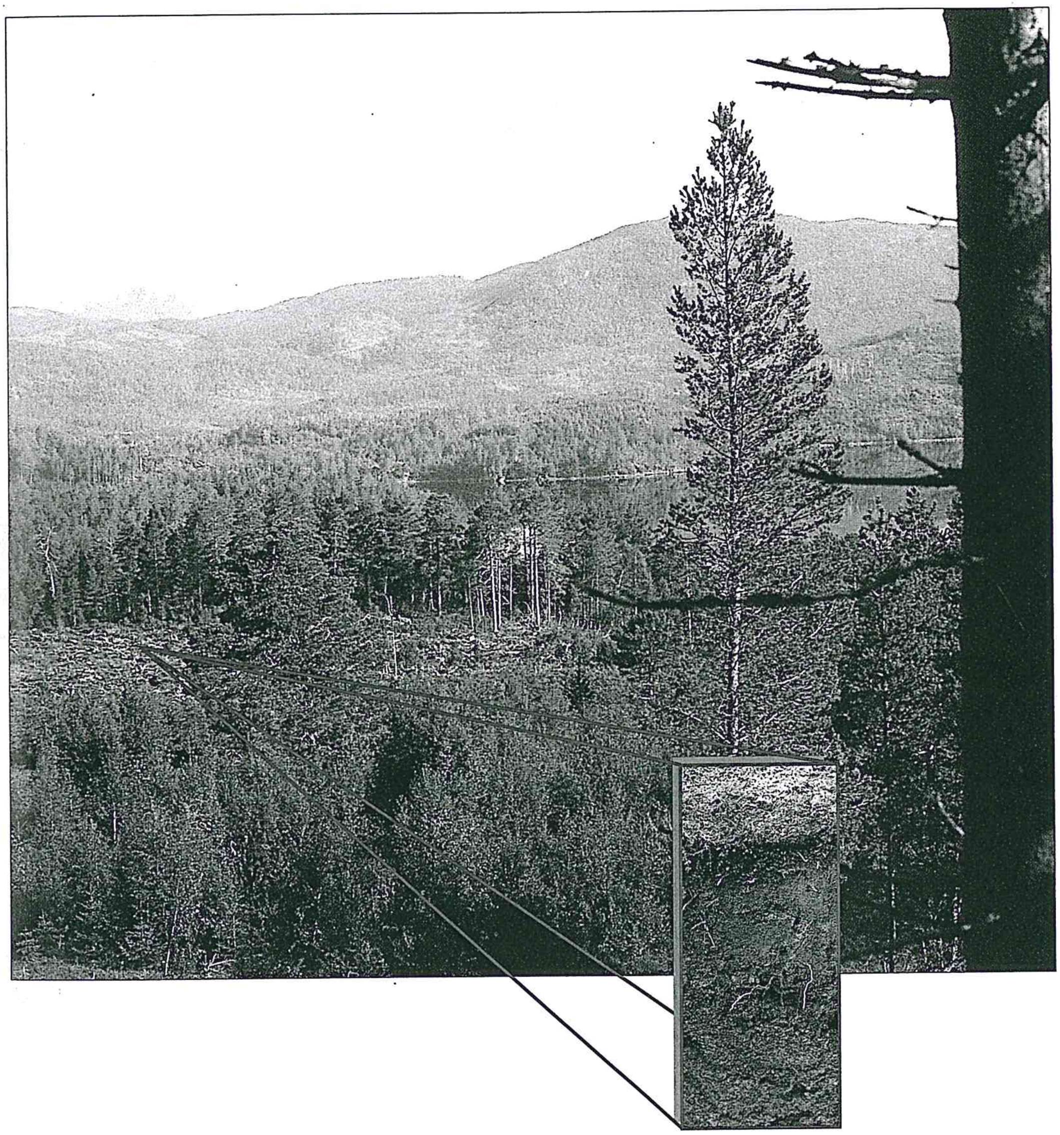
# Carbon stocks in Norwegian forest soils and effects of forest management on carbon storage



Helene A. de Wit and Sheila Kvindesland

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*Forsiden:* *Clearcut at Tingvoll peninsula, western Norway*
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## Abstract

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The Kyoto Protocol has set targets to reduce the emissions of greenhouse gasses to the atmosphere. One method to stabilise atmospheric $CO_2$ concentrations is to sequester carbon in terrestrial ecosystems. Forest management will possibly be included in the Protocol as an accountable measure for sequestration of C in forests. Soils in boreal forest are presently understood to be a large potential sink for atmospheric $CO_2$.

This report presents estimated C-stocks in Norwegian forest soils based on forest monitoring databases which cover forests from all counties except Finnmark. Uncertainty in the estimate of C stocks is ca 30 %, mainly due to missing soil bulk density data and inaccurate estimates of soil stoniness. The estimated C-stocks are higher than published C-stocks in boreal forest soils from Sweden and Finland, but in the same order as Canadian and Scottish estimates. Largest C-stocks are found in organic soils. Considering only mineral soils, pure spruce forest has lowest C-stocks whereas mixed forest and pure pine forest have highest C-stocks in the forest floor. This is in agreement with other studies and implies that planting mixed forest is a measure for increasing C-sequestration in forest soils. The C-stocks in the forest floor of pure pine and spruce forest were larger in higher cutting classes. This may be related to stand age and suggests that an increase in rotation length will increase C-accumulation in the forest floor. Forest management measures that increase C-sequestration in soils are planting or regenerating forest on cultivated, drained soils; N-fertilisation; and increasing rotation length. Measures that may affect soil C-stocks negatively are strongly mechanised site preparation, draining and planting of peatlands, and clearcutting. Organic soils and very sandy soils appear more sensitive to site preparation and harvesting practices, losing more C than other soils after these events.

*Key words: C storage, forest soils, forest management, tree species, Kyoto Protocol, Norway*

## Contents

# 1. Introduction

The expanding use of fossil fuels and large-scale land-use changes has led to an increase in concentrations of trace gasses in the atmosphere, affecting global climate. Predicted changes in temperature may be associated with changes in rainfall patterns and intensity. Land degradation, agricultural production, human health and terrestrial and aquatic ecosystems may be affected significantly by changes in global climate (Batjes 1999).

The Kyoto Protocol, signed in December 1997 by 174 countries, has set targets to reduce the emissions of greenhouse gasses to the atmosphere. Between 2008 and 2012, emissions should be reduced from 6 to 8% compared to 1990 levels. The aim of the Protocol is to stabilise greenhouse gas concentrations to a level that will limit the adverse impact on the Earth's climate (Batjes 1999).

One option to stabilise atmospheric $CO_2$ concentrations that is mentioned in the Kyoto Protocol is to sequester carbon in terrestrial ecosystems. Terrestrial ecosystems are presently thought to be a major sink for atmospheric carbon, estimated to about $1.8 \pm 1.5$ Pg C year$^{-1}$ (Pg = Gt = $10^{15}$ g) (Houghton et al., 1998). The Kyoto Protocol limits in Paragraph 3.3 the allowable terrestrial sources and sinks of carbon to be cases of 'afforestation, reforestation and deforestation' (Nabuurs et al. 1999). However, Paragraph 3.4 mentions 'additional activities by sources and removals by sinks in land-use change and forestry categories' (UNFCCC 1997, Nabuurs, et al. 1999). Negotiations in Kyoto showed that numerous parties, among them Norway, are interested in including activities such as forest management in the list of accountable sinks (Nabuurs et al. 1999).

Recent studies suggest that forest soils in the northern latitudes have a large potential for storing carbon under present-day and future climate conditions and can be important sinks for atmospheric carbon (Schimel 1995; Bird et al., 1996; Liski and Westman 1997).

To date, a limited set of studies has been published regarding soil organic matter stocks in Norwegian forest soils (e.g., Bruun and Frank 1994; Sogn et al., 1997; Sogn et al. 1999). Bruun and Frank (1994) based their estimation of soil C-stocks on a few published C contents of forest soils (from Norway and other countries) and extrapolated these results to cover the whole of Norway. Therefore it was a highly uncertain C stock estimation.

In this report, we calculate soil C stocks based on data from ca. 1000 soil profiles, obtained from Norwegian forest monitoring databases, and account for the uncertainties in the calculations (chapter 3). Results are compared with estimations from the Bruun and Frank report. Additionally, a extensive literature review was done on effects of forest management on soil C stocks (chapter 4), including both results from field experiments and model simulations. It concentrates on recent literature from 1994 to 1999. Bruun and Frank reviewed the literature until 1993 on this topic, and some of their conclusions are incorporated in chapter 4. We used the C-stock estimations in chapter 3 to look for correlations between the size of the C-pools and various factors such as tree species, cutting class, site index and regional distribution to relate them to the literature review on forest management. In addition, we compare the C stocks estimated in this report with published C-stocks in boreal forest soils from countries with similar forests and climate.

# 2. Soil organic matter in forest soils

## *2.1 Forests - the missing sink?*

The geochemical history of carbon cycling on Earth includes events like C release from volcanic activity, dissolution of calcareous sediments in the upper part of the Earth's crust, mineralisation of accumulated reserves of organic matter in the lithosphere and carbon sequestration in the newly forming sediments. Burning of fossil fuels can be considered as an anthropogenic modification of these geological processes and the global carbon cycle should be viewed against the background of these large-scale geological carbon releases (Konyushkov 1998).

In the global carbon cycle a large flux of carbon is not accounted for, which is referred to as 'the missing sink'. This sink amounts to almost 20% of the total annual $CO_2$ production.

The global C-cycle consists of several pools and fluxes. Uncertainties in the individual numbers reported are large and poorly quantified (Schimel 1995). The atmospheric C-pool is estimated to be 720 Gt (Gt = Pg = $10^{15}$ g), whereas the estimated size of the biomass pool ranges from 560 to 835 Gt. Soils may store between 1000 and 3000 Gt C. The C flux from fossil fuels and cement production to the atmosphere amounts to 5.5 Gt C/year. The C flux from terrestrial ecosystems due to changes in landuse - mainly deforestation in the tropics - has been estimated to be net 1.1 Gt C/year, which makes the net emission to the atmosphere 6.6 Gt C/year. The atmosphere gained 3.2 Gt C/year, leaving 3.4 Gt C/year to be explained by net sinks of atmospheric C. The oceans are believed to account for a net uptake of 2 Gt C/year. Thus, 1.4 Gt C/year are not accounted for and make up the so-called 'missing sink' (Schimel 1995). It is suggested that terrestrial ecosystems - vegetation *and* soils - may be the sought-for sink (Houghton et al., 1998). The size of the annual missing sink is ca 0.05% of the total C-pool in biomass and soils. Tans et al. (1990) suggested that the sink may be placed in forests of the northern hemisphere. Schlesinger (1991) estimated that boreal forests contain ca 10 % of the global terrestrial carbon pool. Recent studies, however, suggest that the missing sink may be found elsewhere than in the Northern Hemisphere (Nadelhoffer et al. 1999). Schindler (1999) suggests that the missing sink may be found in oceans, in northern wetlands or in stimulated forest growth due to warming of boreal regions. Schindler deems it most likely that the missing sink turns out to be the sum of many small sinks. Modern methods are not adequate to identify global sinks of sizes less than 0.5 Gt annually.

## *2.2 Carbon sequestration in forest soils*

The primary source of soil organic matter is plant biomass. A major mechanism for accumulation of C in the soil is the production of aboveground and belowground litterfall. Small changes in litter production and litter decomposition may have large consequences for the sink strength or source strength of the soil, and affect atmospheric $CO_2$ levels considerably. Climate and the chemical composition of the litter are important factors that determine decomposition rates (Swift et al., 1979).

Aboveground litterfall in forests is comprised of circa 70% leaf tissues. Global litterfall production follows a similar pattern as net primary production and declines with increasing latitude (Schlesinger, 1991). In addition to net primary production, also C storage in plant biomass within the terrestrial ecosystem declines with increasing latitude. In tropical and subtropical zones, the vegetation is the main store of C. Although litterfall production in the tropics is higher than in the boreal zone, relatively low soil C stocks are found due to favourable conditions for the decomposition of forest litter. In contrast, C-stocks in soils of boreal forests are larger than C-stocks in biomass even though litterfall production is lower than in the tropics. More unfavourable climatic conditions for mineralisation of litter, combined with a different quality of litter (coniferous forest versus tropical forest) result in a ca five times higher soil C stock than biomass C-stock for Norwegian conditions (Bruun and Frank 1994).

Litter is subject to decomposition by soil microfauna, bacteria and fungi. Most of the litter returns to the atmosphere as $CO_2$ (72 - 88%), a minor part is leaked from the soil as Dissolved Organic Carbon (DOC) (3 - 4 %) and a substantial part becomes humus and is highly resistant to further decomposition (8 – 25 %). Percentages for the fate of forest litter reported here have been found in young coniferous and deciduous stands in Siberia (Vedrova, 1995).

In boreal forests, DOC is the major input of C to the mineral soil and surface waters. DOC is mostly produced in the canopy and the forest floor, and is largely removed in the mineral soil by adsorption and to a smaller extent mineralization. Although the flux of DOC out of soil is small (estimated as 1 % of C in litter inputs for Sweden), the role of DOC in transporting C from the forest floor to lower mineral horizons is important, particularly in podzols.

Carbon that is allocated belowground by plants is used to build up root biomass. Additionally, a substantial part is used in root respiration. For annual plants, up to 46 % of the assimilated C has been reported to be transferred to the roots (Grayston et al. 1996). For trees, values as high as 60% (*Pinus sylvestris*; Ågren et al. 1980) and 73% (*Pseudotsuga menziesii*; Fogel and Hunt, 1983) have been reported.

Belowground litterfall consists mainly of fine dead roots. Fine root biomass and production were found to be inversely correlated with latitude on a global scale (Vogt et al. 1986). The C allocation within the plant to roots was found to be inversely correlated with nutrient availability across forest ecosystems (Ruess et al. 1996). However, *total* root growth is largest at nutrient rich sites (Raich and Nadelhoffer, 1989).

Steele et al. (1997) studied root dynamics in Canadian boreal forest stands. The net primary production of roots was compared with aboveground litterfall. The ratio of net root production to total (above- and belowground) litterfall was larger for coniferous forest stands than for deciduous forest stands, suggesting that relatively more C was allocated to roots in coniferous forest than in deciduous forest. For both types of forest, the net primary production of roots exceeded the annual fine root turnover, which implies an accumulation of soil organic matter originating from fine roots. In fact, the imbalance between net fine root production and fine root turnover was sufficient to explain the net accumulation of carbon in boreal forest soils, which demonstrates the importance of root dynamics for soil C sequestration.

Fogel and Hunt (1983) estimated that decomposition of fine roots lead to an annual input to the soil of 40-47% of C translocated to roots. They concluded that turnover of fine roots and mycorrhiza contributes 2-5 times more to soil organic matter than aboveground biomass does.

Root respiration is the C-cost of root maintenance, root growth and nutrient uptake. In annual plants, between one-quarter and two thirds of all photosynthates produced per day are respired in the same period, with a major part of the respiration occurring in the roots (Poorter et al., 1990; Van der Werf et al., 1994). Raich and Schlesinger (1992) estimated root respiration to be approximately 30% of total respiration in a boreal forest soil.

### 2.3 Factors controlling Soil Organic Matter dynamics

Soil Organic Matter (SOM) accumulation is a balance between production and decomposition. An understanding of factors that affect both processes are necessary for predicting changes in carbon accumulation.

#### 2.3.1 Climate and vegetation zone

Vogt et al. (1995) present a comprehensive study on biotic and abiotic factors that affect accumulation of SOM including boreal, temperate and tropical forests. They concluded that cold temperate and tropical climatic zones had the highest potential for sequestering C. The highest C-accumulation occurred when forest stands were composed of a mixture of evergreen and deciduous tree species. Evergreen forests had lower C accumulation in the mineral soil, but had higher total ecosystem C stocks than deciduous forests.

Post et al. (1982) found that soil carbon accumulation increased with increasing precipitation and decreasing temperature for a database of 2700 profiles from all climatic zones. By contrast, Vogt et al. (1995) could not explain variation in C in the mineral soil from 93 data sets from all climatic zones by climatic variables. Carbon accumulation in the forest floor, however, was strongly negatively related to temperature across climatic zones. In contrast, Liski and Westman (1997) found that C accumulation in the upper meter of the mineral soil was positively correlated with the temperature sum, suggesting that increasing temperatures may cause an enhanced C-sequestration in the soil. However, the study of Liski and Westman was done within one climatic zone, whereas Vogt et al. and Post et al. reported for relationships across climatic zones.

#### 2.3.2 Nitrogen availability

Nitrogen (N) is one of the main limiting nutrients in boreal forest ecosystems (Binkley and Högberg 1997). Fertilisation of the vegetation by atmospheric nitrogen deposition is considered to increase the $CO_2$ sink strength of temperate forests by enhancing forest growth and C accumulation in soils (Vogt et al., 1995; Liski & Westman, 1995).

In several field manipulation experiments that simulated effects of increased N deposition, the soil pool has been identified as the most important sink for the incoming N (Kahl et al., 1993; Aber et al., 1993; Christ et al., 1995; Magill et al., 1996; Kjønaas et al., 1998).

Recently, Nadelhoffer et al. (1999) examined C and N sequestration in $^{15}N$ tracer experiments in nine temperate forests. They argued that only 20% of the annual carbon uptake by forest growth could be explained by enhanced growth due to N-deposition. Most of the $^{15}N$ that was added was retained in the soil, immobilised by microbial activity. The addition of N in the $^{15}N$ tracer experiments on which Nadelhoffer et al. based their analysis were conducted beneath the tree canopy. However, it was pointed out that in some forests, the tree canopy plays an important role in the uptake of N and may increase the annual binding of C in vegetation (Jenksinson et al. 1999; Sievering 1999).

Nitrogen fertiliser experiments indicated an increased forest growth and nitrogen status of trees with increased nitrogen additions at initially N limited sites (e.g. Kenk and Fisher, 1988; Tamm, 1989). Additionally, an increase in soil C was observed in the majority of cases (e.g. Van Cleve and Moore 1978, Nohrstedt et al. 1989). The increase in soil C could not, however, be accounted for by increased litterfall and was attributed to reduced microbial activity and thus, reduced decomposition.

Enhanced N immobilization in the soil may affect the decomposition of SOM. N availability may change decomposition rates by i) affecting the diversity and activity of the microbial population, and ii) changing the quality of the organic matter, thus affecting the substrate of the decomposer community.

Low C/N ratios of the substrate (litter, fine roots, root exudates) have been recognized to increase decomposition rates (Alexander, 1977). However, N addition may in some cases actually reduce the microbial activity. Berg et al. (1996) suggest that initial decomposition rates are promoted by high N concentration, while N retards long-term decomposition through inhibition of lignin degrading enzymes and reactions producing recalcitrant aromatic compounds. Decreased microbial activity after nitrogen supplements has been confirmed by field studies (Söderström et al., 1983; Grayston et al., 1996).

Thus, the net ecosystem effect of N deposition on degradation of organic matter is not clear.

# 3. C storage in Norwegian forest soils

## 3.1 Calculation of C-stocks

C-stocks in soil horizons were calculated using the following equation:

$$\text{C-stock} = d \times BD \times \text{C-content} \times CF_{st}$$

C-stock ($kg/m^2$)
d: depth of horizon (m)
BD: bulk density ($kg/m^3$)
C-content ($g\ g^{-1}$)
$CF_{st}$: correction factor for stoniness and gravel content

The C-stock in a soil profile was obtained by summing of the C-stock of each horizon. No standardisation with respect to depth was undertaken. Bulk density data were not available in the database collected by the Norwegian Institute of Land Inventory (NIJOS) and for 35% of the Intensive Sites for Forestry Monitoring (OPS) database. Instead, we collected published and unpublished data on bulk density of Norwegian forest soils in a database. Two soil types were distinguished: organic soils which included fibrosols, humisols (Brække, 1981, 1987; Brække and Finer, 1991) and folisols (Birkenes, Hirkjølen: N. Clarke, pers. comm.) and all other soils, the mineral soil types, which included podzols, gleysols, brunisols and regosols (from intensive monitoring plots all over Norway (S. Solberg, pers. comm.), Nordmoen (T. A. Sogn, H. A. de Wit (pers. comm.), Birkenes, Hirkjølen (N. Clarke, pers. comm.) and Gårdsjön (O.J. Kjønaas, pers. comm.)). Averaged bulk densities for each horizon are presented in Tables 1 and 2.

The bulk density data from the non-organic soil types did not justify a differentiation between soil types. Therefore, the mean bulk density value for the specified horizon was used for all soil types. If no subdivision of the litter layer was described, we used the averaged bulk density for O horizons of all soil types except organic soils. If specified, we used different bulk densities for L, F and H layers or combinations of those.

For the organic soils, bulk density in fibrisols was assumed to be independent of depth. Fibrisols are organic soils often built up from sphagnum, which decomposes slowly. Thus, organic matter higher up in the profile is not very different in density from organic matter deeper in the profile. By contrast, humisols are organic soils that usually consist of different types of organic material, from hardly decomposed to well decomposed. We assumed that in most cases, the degree of decomposition increased with depth and therefore an increasing density was used for the lower layers of humisols. However, depending on drainage conditions, in some cases the more decomposed material is in the upper part of the profile. Folisols in Norway are generally layers of litter over 10 cm thick, overlying bedrock found on slopes and are formed under upland ecosystem development as opposed to wetland development (fibrosols and humisols).

Table 1 Bulk density for organic soils (kg $m^{-3}$). Stdev = standard devation. n = nr of observations

| Soil | horizon | mean | median | stdev | n |
|---|---|---|---|---|---|
| folisol | LFH, O | 145 | 140 | 47 | 22 |
| humisol | Of | 54 | 39 | 34 | 11 |
| humisol | Om | 83 | 79 | 58 | 9 |
| humisol | Oh | 142 | 131 | 59 | 9 |
| fibrisol | O | 59 | 62 | 8 | 7 |

Table 2 Bulk density for mineral soils: podzols, brunisols, gleysols, regosols and non-soils (kg $m^{-3}$). Stdev = standard devation. n = nr of observations

| Horizon | mean | median | stdev | n |
|---|---|---|---|---|
| LF, L, F | 87 | 85 | 31 | 8 |
| H | 154 | 122 | 91 | 122 |
| LFH, O | 151 | 139 | 67 | 84 |
| Ae, Ah, E | 992 | 966 | 262 | 46 |
| AB | 1021 | 1017 | 194 | 9 |
| Bf | 1008 | 1002 | 244 | 36 |
| Bh | 848 | 775 | 229 | 17 |
| other B horizons | 1006 | 1000 | 270 | 75 |
| BC | 1263 | 1272 | 262 | 13 |
| C, R | 1454 | 1460 | 155 | 31 |

A correction factor for stoniness (including gravel and stone content of a horizon) was calculated using the following equation:

$$CF_{st} = 1 - (\ \%gravel + \%stones)/100$$

The minimum value of the correction factor was set at 0.1, i.e. 90% gravel and stones.

The stone and gravel content in each soil horizon for NIJOS data has been judged visually in the field into different classes (see Table 3). The classes were converted to one value per class by choosing the mean value of the class or by setting a maximum value for the highest class. In the OPS database, only data on a visual field observation of % stones seen on the surface were available. The same classes as in Table 3 were used. Stoniness in the mineral soil was corrected in the mineral layers according to this Table.

Table 3 Conversion from classes of % gravel (2-20 mm) and stones (>20mm) to % for the calculation of $CF_{st}$

| gravel content of horizon | | mean | stone content of horizon | | mean |
|---|---|---|---|---|---|
| class | % | % | class | % | % |
| | | | | | |
| 1 | <20 | 10 | 1 | <0.1 | 0 |
| 2 | 20-50 | 35 | 2 | .1-2 | 1 |
| 3 | 50-90 | 70 | 3 | 2-5 | 3.5 |
| 4 | >90 | 90 | 4 | 5-10 | 7.5 |
| | | | 5 | 10-20 | 15 |
| | | | 6 | 20-40 | 30 |
| | | | 7 | >40 | 70 |

### *3.2 Uncertainty in estimations of C-stocks*

Uncertainty in the quality and accuracy of available data and missing data affect the estimations of C-stocks. The representativity of the soil profiles chosen for a forest area and sampling method are potential sources of error. For example, soil pits may have been dug where the soil was deepest, making upscaling of profile data to landscape scale highly uncertain. Additionally, soil sampling for classification purposes (sampling in the middle of a soil horizon) may lead to other results than soil sampling for determining pools (sampling the whole soil horizon). Although we do not know how the soils in the database were sampled, it might explain systematical differences between C-stocks from this report and from soil monitoring databases from other countries.

Data on bulk density and stoniness were missing. The NIJOS database lacked data on bulk density and in 40% of the horizons no stoniness was reported. The OPS database lacked in 35% of the soil horizons bulk density and did not have an estimate of stoniness per horizon, but only of stones at the surface.

We collected bulk density data for Norwegian forest soils and calculated mean and median bulk density per horizon with a standard deviation and assumed that the mean bulk density was a reasonable estimate to use where data were lacking. The variation in the bulk density estimates shows the variability in the data used, but does not reveal anything about the variability in bulk density at a specific site. The assumptions about bulk density may lead to considerable over- or underestimations of the soil C stocks, which may be larger than the variability in collected bulk density database.

For 40% of all soil horizons in the NIJOS database, no stone or gravel content was reported and therefore no correction was made. The average gravel content and stone content in the profiles with reported gravel and stone contents were respectively 15% and 19%. We assume that the profiles where these data were lacking had an average stoniness and gravel content of 20%, leading to a correction factor of 0.8. The lack of data on stone and gravel content may thus lead to a systematic overestimation of total C stocks by ca 25% (1/0.8). However, as only

40% of the horizons in the database lacked these data, the systematic overestimation is reduced to 10%.

Thus, the missing data are a considerable source of uncertainty but this is hard to account for quantitatively.

Regarding the available data including bulk density and stoniness, the uncertainty in the estimation of C-stocks was caused by errors in the estimation of bulk density, correction for stoniness- and gravel content, analysis of C-content and determination of the depth of the horizon (see equation).

Ideally, an account of uncertainty should be given by calculating uncertainty for every soil profile. However, this was not realistic within the time frame of the project. Therefore, we demonstrate which factors determine uncertainty in the outcome of simplified calculations. We calculated uncertainty in C-stocks separated according to soil type (organic and mineral soil types) and within the mineral soil type, according to O horizon and mineral soil horizons, per horizon and for a hypothetical profile composited of two soil horizons.

For bulk density, the standard error of the mean (standard deviation / $n^{0.5}$) for organic soils (Table 1) was 5% to 19%, with an average of 11%. The standard error in the organic horizons for the mineral soil types (Table 2) was 5 to 13%, with an average of 8%. For the mineral soil horizons, the standard error ranged between 0.3% to 6.3% with an average of 4% (Table 2).

The gravel and stone contents are a visual observation and thus only a qualitative indication of stoniness, rather than a physically measured quantitative determination of stoniness. An additional source of error is the conversion of classes of gravel content and stoniness to % (Table 3). In the range of gravel and stone contents from circa 10 to 70 %, an uncertainty exists of 5% (for low gravel and stone contents) to over 20% (for higher gravel and stone contents). This error may cause both under- and overestimation of C-stocks. We assume that stoniness in organic soils and in the O horizon did not exceed 20% in the majority of the profiles which implies a maximum uncertainty in the correction factor of 10%. For the mineral soil, stoniness and gravel content were generally higher than in the O horizon. We assume that uncertainty in the correction factor for stoniness and gravel content in the mineral soil was on average 20%.

The thickness of the horizon is a fairly accurate number and we assume that the uncertainty here was 5%, although accuracy in defining transitions between genetic soil horizons decreases with soil depth. Additionally, the depth of profile dug may have been the result of a number of random factors such as stoniness and weather conditions during establishment of the plot.

The C content was measured by a CHN element analyser (Perkin Elmer 2400 CHN elemental analyser) in both databases. This is a very accurate instrument, having 3 % uncertainty. However, the reliability of the measured C % for the O horizon depends to a large extent on (i) pre-treatment of the sample before analysis and on (ii) whether any mineral soil from the underlying horizon has been mixed with the organic material during field sampling. Mixing with mineral soil leads to an underestimation of the C content of the organic soil. We assume that the uncertainty in measuring the C content of the soil was 10% in the O horizon and 5% in the mineral soil. For organic soils, we assume that mixing with mineral soil was less likely and that the uncertainty for measuring the C content was 5%.

The equation used to calculate C-stocks in a soil horizon is as follows:

$$\text{C-stock} = d \times BD \times \text{C-content} \times CF_{st}$$

The C-stock in the whole profile was obtained by summing of the C-stock of each horizon.

The total uncertainty per soil horizon is calculated using the following equation:

$$se(\text{C-stock, horizon}) = (se^2(d)+se^2(BD)+se^2(C)+se^2(CF_{st}))^{-0.5}$$

se: standard error (in %)

For the whole profile, uncertainty is calculated by taking the root of the quadratic sum of *absolute* uncertainties in each horizon of which the profile consists.

$$se(\text{C-stock, profile}) = (se(\text{C-stock, horizon 1})^2 + se(\text{C-stock, horizon 2})^2 + ..)^{0.5}$$

We assume that covariance between factors was negligible compared with the sources of error mentioned above. The results are presented in Table 4.

Table 4 Standard error (se) in C-stock estimations (%) per horizon and per profile (consisting of two horizons which contribute equally to the total C-stock)

| | se(d) | se (BD) | se (C-content) | se (CFst) | total |
|---|---|---|---|---|---|
| organic soil types | | | | | |
| soil horizon | 5 | 11.5 | 5 | 10 | 17 |
| soil profile (2 horizons) | | | | | 25 |
| mineral soil types | | | | | |
| O-horizon | 5 | 8.2 | 10 | 10 | 17 |
| mineral soil | 5 | 4.2 | 5 | 20 | 22 |
| soil profile (2 horizons) | | | | | 28 |

The uncertainty in the C-stock of a soil horizon in organic soils was 17%, and was mostly determined by the uncertainty in bulk density and the correction for stoniness. Uncertainty in C-stocks of soil horizons in the mineral soil types was 17% for C-stocks in the O horizon and 22% for C-stocks a mineral soil horizon. Uncertainty in bulk density, C-content and stoniness contributed almost equally to the uncertainty in the O-horizon estimates. The uncertainty in the C-stock of the mineral soil was mostly determined by the uncertainty in stoniness.

The uncertainty in the C-stock of a soil profile, composited from several soil horizons, is determined by the root of the quadratic sum of absolute uncertainties in each horizon. Thus, small absolute stocks with a relatively high uncertainty contribute less to total uncertainty than large absolute stocks with a relatively low uncertainty.

Here, we simplify by assuming that a soil profile only consists of a mineral soil layer and an organic soil layer that contribute equally to the total soil C stock, leading to a total uncertainty of 25% to 28%, which is probably a conservative estimate as most profiles consist of more than two soil horizons. In organic soils, C stocks in the organic layer are usually more than twice as high as the mineral soil C-stock. In mineral soil types, the relative contribution of each soil layer depends on the soil type considered (see Fig. 1).

In conclusion for the NIJOS database, the uncertainty of calculated C-stocks in the O horizon is 17% and in the mineral soil is 22%, mainly due to uncertainty in C-content, stoniness and bulk density. The summed uncertainty for a profile consisting of an organic and a mineral horizon was estimated to be 28% but this is a conservative estimate. Due to lack of data on stoniness in 40% of the data, total C-stocks may have been overestimated by up to 10%. Other sources of error such as the sampling method and representativity of the data have not been quantified but may add a considerable uncertainty.

The uncertainty for stoniness in the OPS database is larger than the uncertainty in the NIJOS database, as the correction for stoniness in the OPS database was based on % stones on the surface and not on % stones and % gravel per horizon. However, as 65 % of the OPS horizons had measured bulk density data, uncertainty due to bulk density is lower. Therefore, the C-stock estimations in the OPS database can be assumed to have similar uncertainty as the NIJOS database.

## *3.3 C-stocks in NIJOS database*

### *3.3.1 Description of the NIJOS database*

The NIJOS database consists of 'level-1' forest-monitoring plots, defined as such in the international programme of forest monitoring (Lorenz, 1995). The database included 934 sites of which 192 were classified as non-productive woodlands (0.1 - 1 $m^3$/ha/year volume increment of wood). All sites were located below the tree limit. The majority of the sites were situated in the south-eastern part of Norway (Table 5).

The soils were classified according to the Canadian System of Soil Classification (Survey 1987) as podzols, brunisols, gleysols, regosols, organic soils and non-soils. Organic soils are typically poorly drained soils (Table 6) with a large accumulation of organic matter. Podzols, the most common soil type in Norway, are acidic soils. Brunisols usually have a slightly better nutrient status than podzols. Gleysols are soils that show signs of waterlogging. Regosols are soils that are weakly developed and that may in time develop into one of the other soil types. Non-soils are soils that are too shallow to be classified in other soil types and usually consist of a humus layer on bedrock. Organic soils usually have the largest C stocks of these soil types. Waterlogging, causing unfavourable conditions for decomposition leads to the development of thick layers of plant debris in organic soils.

The non-productive forest sites were characterised by a generally worse drainage status of the soil (Table 6) and a higher percentage of organic soils and gleysoils than productive forest sites. Scots pine was the dominating tree species on over 40% of the sites of non-productive forest. Additionally, birch appeared more frequently than in productive forests. Almost 60% of the productive forest sites were situated in the south-eastern part of Norway. Dominant soil types were podzols and brunisols.

Table 5. The distribution of productive and non-productive forest sites over different regions in Norway. n = nr of sites

| | All | Productive forest | Non-prod. forest |
|---|---|---|---|
| n | 934 | 742 | 192 |
| | n | % | % |
| north[1] | 100 | 9 | 18 |
| middle[2] | 155 | 15 | 21 |
| east[3] | 411 | 48 | 28 |
| west[4] | 84 | 8 | 13 |
| south[5] | 184 | 19 | 21 |

[1]counties of Troms and Nordland. [2] counties of Nord- and Sør-Trøndelag. [3] counties of Oppland, Hedmark, Buskerud, Østfold, Vestfold, Oslo and Akershus. [4] counties of Hordaland, Rogaland, Sogn og Fjordane, Møre og Romsdal. [5] counties of Telemark, Aust-Agder and Vest-Agder

Table 6. The distribution of forest sites on organic soils and mineral soils over drainage classes. n = nr of sites

| Drainage class | All | Organic soils | Mineral soils |
|---|---|---|---|
| n | 934 | 152 | 782 |
| | % | % | % |
| Poor to very poor | 34 | 61 | 28 |
| Moderately | 21 | 9 | 23 |
| Well to excessively well | 42 | 24 | 45 |
| Undefined | 4 | 6 | 3 |

In the productive forest sites, excluding the sites on organic soils (12 %), most forests were dominated by spruce, followed by pine-dominated forests and by forests mixed with birch (Table 7). Pure birch and spruce forest sites were equally abundant in northern Norway. Western and southern Norway were pine-dominated, but deciduous species were relatively more abundant in western Norway than in southern Norway.

Table 7 Dominating tree species in productive forest sites in different zones of Norway, excluding sites on organic soils. n = nr. of sites

| | All | no trees | spruce | spruce-pine | spruce-birch | pine | pine-birch | birch | aspen, grey alder | other deciduous |
|---|---|---|---|---|---|---|---|---|---|---|
| n | 648 | 27 | 235 | 46 | 52 | 168 | 37 | 42 | 10 | 31 |
| | n | % | % | % | % | % | % | % | % | % |
| north[1] | 64 | 0 | 38 | 0 | 8 | 9 | 8 | 33 | 2 | 3 |
| middle[2] | 101 | 5 | 50 | 6 | 10 | 11 | 4 | 5 | 1 | 9 |
| east[3] | 306 | 4 | 42 | 10 | 9 | 24 | 5 | 2 | 0 | 4 |
| west[4] | 52 | 6 | 17 | 0 | 2 | 46 | 8 | 12 | 4 | 6 |
| south[5] | 125 | 6 | 20 | 7 | 6 | 43 | 6 | 3 | 4 | 5 |

[1]counties of Troms and Nordland. [2] counties of Nord- and Sør-Trøndelag. [3] counties of Oppland, Hedmark, Buskerud, Østfold, Vestfold, Oslo and Akershus. [4] counties of Hordaland, Rogaland, Sogn og Fjordane, Møre og Romsdal. [5] counties of Telemark, Aust-Agder and Vest-Agder

### *3.3.2 Results*

C-stocks were highly variable and not normally distributed around the mean. Comparing mean with median values showed that the median was usually lower than the mean, which indicates that average values were influenced by a few very high values. We concluded that median values were more representative than mean values and comparisons are therefore done between median values of a class.

The C stocks were grouped according to soil types. Largest C-stocks were found in the organic soils (Fig. 1). The median C-storage in the organic horizons was over 20 kg C/m$^2$. In all other soils, C-stocks in the organic horizons varied between 3 (non-soils) and 9 (gleysols) kg/m$^2$. Poor drainage may account for the high value for the gleysols. More C was stored in organic horizons than in the mineral horizons, except for the podzols. The higher C-storage in the mineral soil of podzols may be related to the process of podzolization. Podzolization involves downward transport of organic compounds (DOC) from O horizon to B horizon. On its way through the mineral soil, DOC forms complexes with Al and Fe and precipitates in the B-horizon, causing an accumulation of organic matter that is rich in Al and Fe.

Separating the productive forest sites from the non-productive forest sites and organic soils from mineral soils showed (Fig. 2) that C-storage in organic soils in non-productive forest sites was considerably higher than in productive forest sites. Similarly, C-storage in the O horizon in the non-organic soils was highest in non-productive forests. This may be related to the generally worse drainage conditions of non-productive forest. Differences in mineral soil C stocks may be due to differences in soil depth.

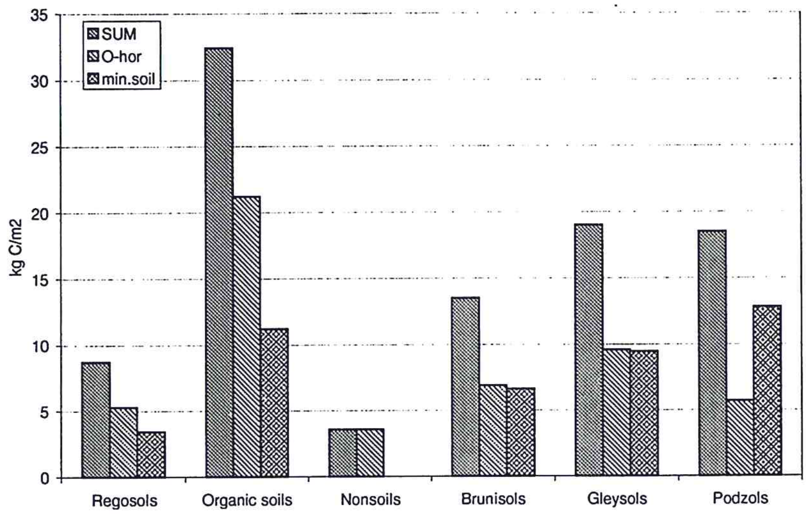


Fig. 1. Soil C stocks (median values) in O horizon and mineral soil for various soil types in NIJOS database

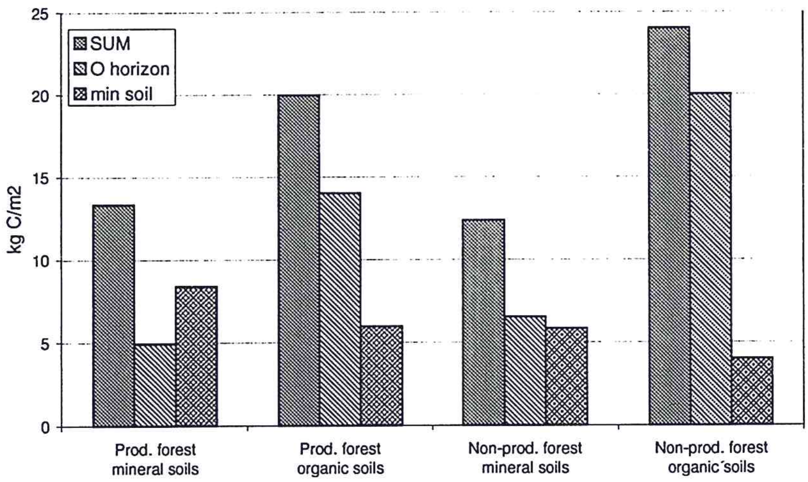


Fig. 2. Soil C stocks (median values) in O horizon and mineral soil for productive forest and non-productive forest on organic soil types and mineral soil types

Table 8 The total carbon stocks in forest soils in Norway on an area basis

| | Productive forest land | Non-productive forest land | Wooded mire (forest on organic soils) | Total |
|---|---|---|---|---|
| median C content (kg/m$^2$) | 13.2[a] | 12.5[a] | 24[a] | |
| Area, 1000 ha | 7273[b] | 1721[b] | 639[b] | 9633 |
| C stock (Gt = $10^{15}$ g) | 0.96 | 0.22 | 0.15 | 1.33 |

[a] NIJOS data, this report, figure 2.
[b] area data from NIJOS, 1996 (Tomter 1996)

The median C-stocks were used to calculate total C-stocks in forest soils in Norway (Table 8). The total C-stock is 1.3 Gt, most C being stored in productive forest land due to its large land coverage and least in wooded mire. The C-stocks may be overestimated because part of the land consists of rocks or very shallow soils. The total global C-storage in soils, including forest soils and agricultural soils, is estimated to be 1000 to 3000 Gt (Schimel, 1995).

To consider the effect of tree species, cutting class and site class on C-stocks, the low-productivity sites, which are of less interest for forest management, and the sites on organic soils were excluded from the database. The forest floor (O horizon) is the most dynamic part of the soil profile regarding the build-up of organic matter and effects of tree species are most likely to be detectable there.

C-stocks in the O horizon were highly variable (Fig. 3). The standard deviation varied between 60% and 105 % of the mean. Usually, mean values were larger than median values.

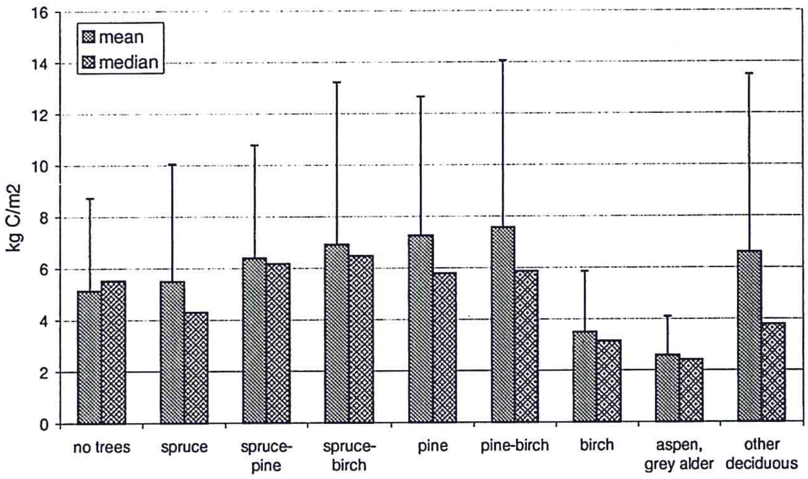


Fig. 3. Soil C stocks (mean and median values) in O horizon of mineral soil types for various tree species in productive forest. Error bar = standard deviation.

Lowest C-stocks were found in the sites dominated by birch and aspen-grey alder forest (2.5 - 3 kg C/m$^2$). Mixed forest and pure Scots pine forest carried the highest C-stocks (ca 6 kg C/m$^2$), whereas C-stocks in pure Norway spruce forest were circa 2 kg C/m$^2$ lower. This suggests that Norway spruce forests mixed with Scots pine or birch have a larger potential to store C than pure Norway spruce forests. Mixing Scots pine forest with other tree species did not affect the C-accumulation in the forest floor. However, other factors that potentially affect C-accumulation such as site index (Liski and Westman, 1995), stand age (Sogn et al. 1999) and climate were not considered here and may be able to explain part of the variability found. The high C-stocks found in forest floor in pine forest were somewhat surprising because a substantial part of pine forest in south-eastern Norway is located in areas where climate and soil nutrient status are not expected to stimulate accumulation of large soil C pools. Pine forest in south-eastern Norway is often located on dry and nutrient-poor soils (Norwegian: *A1, lavskog*) and has a low tree density and usually thin humus layers (Fremstad 1997). Pine forest in western, coastal Norway, however, is characterised by a more humid vegetation type than *lavskog* (Norwegian: *A3, røsslyng-blokkebærfuruskog*) with often a thick humus layer (Fremstad 1997). The majority of the productive pine forest sites in the NIJOS database are *røsslyng-blokkebærfuruskog*.

C-stocks in the mineral soil varied between 5 and 13.5 kg C/m$^2$ (Fig. 4). The mineral soils were not standardised with respect to depth, which complicates the comparison, as deep soils obviously potentially store more C than shallow soils. Median values for depth varied between 43 cm (pine forest) and 62 cm (birch forest). Spruce forest and aspen-grey alder forest had soils of similar (median) depth, but C-stocks were much higher in the latter forest type. This suggests a higher C-density in the mineral soil of aspen-grey alder forests.

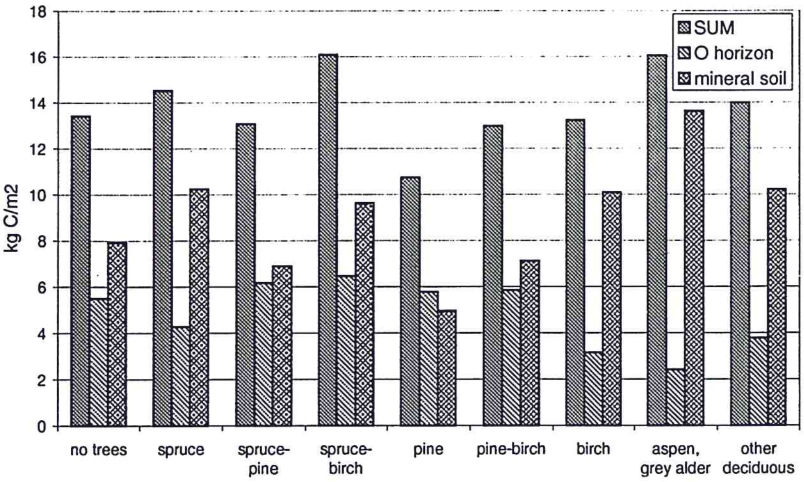


Fig. 4. Soil C stocks (median values) in O horizon and mineral soil of mineral soil types for various tree species in productive forest

C-stocks in the O horizon of pure spruce and pine forests were grouped according to cutting class II to V (Figures 5 and 6). Cutting class I is defined as having no trees, i.e. recently harvested forest. Cutting class is also termed 'development class', which is connected with tree age. However, cutting class V presumably contains a mixture of mature forest and very old forest. Additionally, tree age in a given cutting class is dependent on site index. Assuming that median values are more representative for each class than mean values, C-stocks increased with cutting class for both spruce and pine. For birch, no such relationship was observed (data not shown). No difference in forest floor C-stocks was observed between cutting class II and III, whereas a clear increase was observed going from cutting class III to V.

No relationship between site index and C-storage was found for spruce forest (Fig. 7).

The regional distribution of C-stocks in soils the productive spruce forest sites was calculated (Fig. 8). Highest median C-stocks were found in western Norway and lowest in northern Norway. C-stocks in the forest floor were ca 4 kg/m$^2$, except in the north where C-stocks were ca 1 kg/m$^2$ lower.

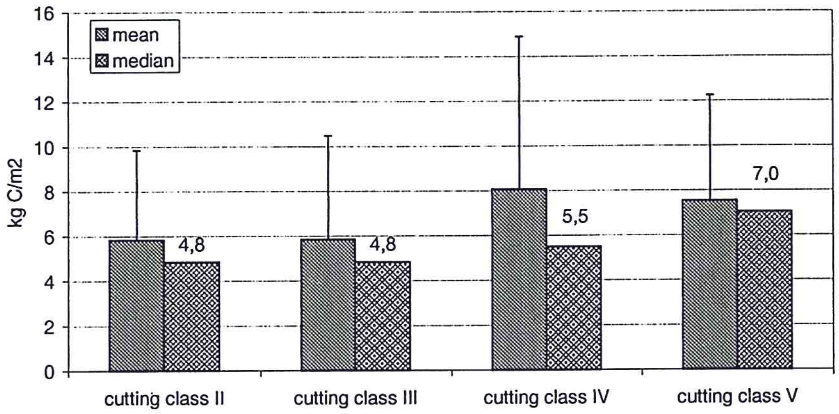


Fig. 5. Soil C stocks (mean and median values) in O horizon of productive pure spruce forest on mineral soil types as a function of cutting class. Error bar = standard deviation.

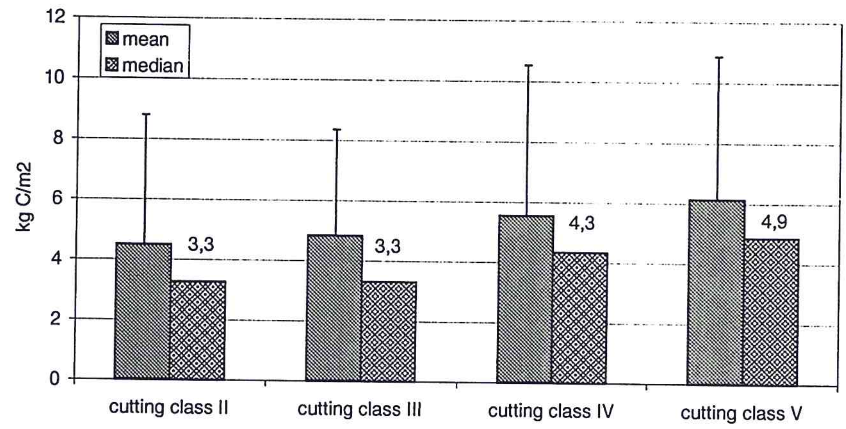


Fig. 6. Soil C stocks (mean and median values) in O horizon of productive pure pine forest on mineral soil types as a function of cutting class. Error bar = standard deviation.

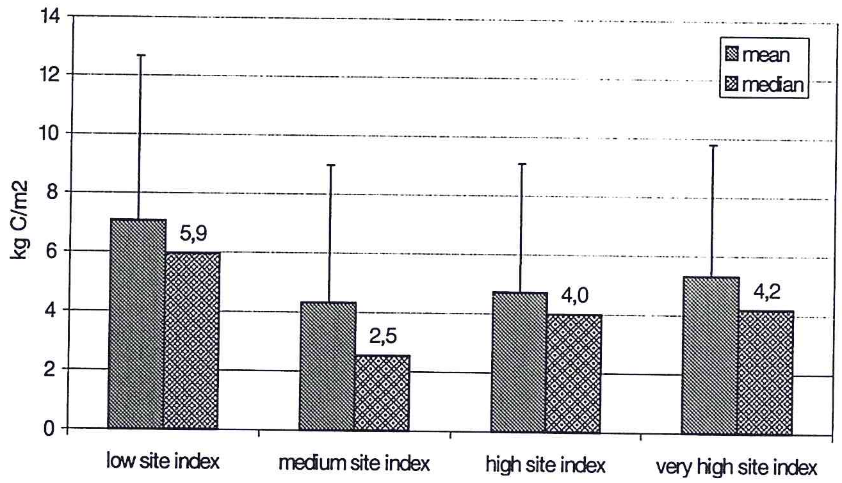


Fig.7. Soil C stocks (mean and median values) in O horizon of productive pure spruce forest on mineral soil types as a function of site index. Error bar = standard deviation.

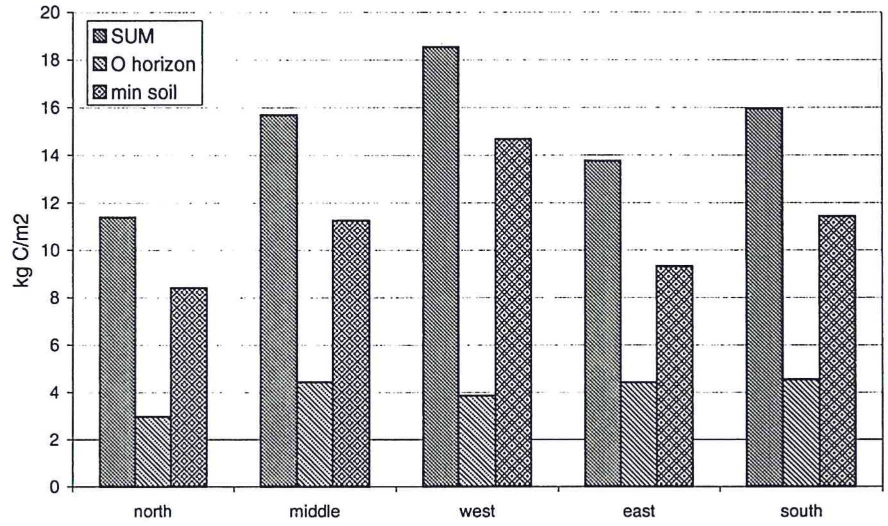


Fig. 8. Soil C stocks (median values) in O horizon and mineral soil of mineral soil types in productive forest for different regions in Norway

### *3.4 C-stocks in OPS database*

The original data for 24 soil profiles, sampled when the level-2 sites for Forestry Monitoring (Lorenz, 1995) were established between 1986 and 1989 (Horntvedt et al., 1992) are presented here. The sites are part of the Norwegian Programme for Monitoring of Forest (Norwegian: *Overvåkningsprogram for Skogskader* (OPS)). The sites were established in each county of Norway, generally in productive mature Norway spruce forest of the vegetation type *Eu-picetum myrtilletosum*. The dominant soil type is podzol and the site index ranges from G6 to G20.

The sites are arranged by descending C-stocks of the total profile (Table 9). The C-stocks of the total profile ranged from 6.5 to 25.5 kg/m$^2$. C-stocks in the forest floor and the mineral soil, respectively, ranged from 2.1 to 10.1 kg /m$^2$, and from 3.9 to 19.2 kg/m$^2$. The highest values were found in spruce forest on podzol soils. A multiple regression that related C-stocks in the forest floor to stand age and site index (only for spruce) failed to explain more than 10% of the variability found.

Table 9 Site details and soil C-stocks at the OPS intensive monitoring sites. n.d.=no data

| Site | Soil type | Stand age in 1987 | Species[1] and Site Index | C in O horizon (kg/m$^2$) | C in mineral soil (kg/m$^2$) | C in total profile (kg/m$^2$) | Total depth (cm) |
|---|---|---|---|---|---|---|---|
| | | | | | | | |
| Nedstrand | podzol | 54 | G17.5 | 8.8 | 16.7 | 25.5 | 70 |
| Valle | podzol | 116 | G11.5 | 10.1 | 15.4 | 25.5 | 108 |
| Kårvatn | podzol | 117 | F8.1 | 9.9 | 14.3 | 24.2 | 78 |
| Tustervatn | podzol | 139 | G5.6 | 3.9 | 19.2 | 23.1 | 71 |
| Søgne | podzol | 55 | G18.8 | 5.4 | 17 | 22.4 | 73 |
| Kårvatn | podzol | 36 | G19.9 | 2.9 | 18.4 | 21.3 | 98 |
| Fyresdal | podzol | 133 | G10.6 | 2.9 | 15.8 | 18.7 | 90 |
| Selbu | podzol | 150 | G5.9 | 2.6 | 14.8 | 17.4 | 91 |
| Ås | podzol | n.d. | n.d. | 3.9 | 13.4 | 17.3 | 86 |
| Mellesmo | gleysol | n.d. | G8.1 | 5.0 | 11.9 | 16.9 | 42 |
| Prestebakken | podzol | 84 | G20.1 | 3.5 | 13.3 | 16.8 | 81 |
| Naustdal | podzol | 43 | G23.1 | 6.7 | 9.5 | 16.2 | 72 |
| Lardal | podzol | 114 | G13.2 | 4.2 | 10.7 | 14.9 | 49 |
| Rana | podzol | n.d. | G6.7 | 4.5 | 9.7 | 14.2 | 86 |
| Birkenes | brunisol | 102 | G12.9 | 7.4 | 6.2 | 13.6 | n.d. |
| Osen | brunisol | 134 | F12.4 | 4.6 | 8.9 | 13.5 | n.d. |
| Høylandet | podzol | 148 | G9.1 | 2.4 | 9.5 | 11.9 | 46 |
| Dividalen | podzol | 53 | G11.1 | 5.0 | 6.5 | 11.5 | 58 |
| Voss | gleysol | 136 | G9.6 | 3.4 | 4.9 | 8.3 | 21 |
| Nordmoen | brunisol | 81 | G16.7 | 3.8 | 4.5 | 8.3 | 43 |
| Svanhovd | brunisol | 73 | G8 | 4.1 | 3.9 | 8.0 | 28 |
| Fagernes | podzol | 88 | G11.3 | 2.1 | 4.4 | 6.5 | 65 |
| | | | | | | | |
| | | | Mean | 4.9 | 11.3 | 16.2 | 67.8 |
| | | | Median | 4.2 | 11.3 | 16.5 | 71.5 |
| | | | Stdev | 2.3 | 4.9 | 5.8 | 23.6 |

[1] G denotes Norway spruce site, F denotes Scots pine site

### 3.5 C stocks and accumulation in Birkenes (Southern Norway)

Birkenes is a 42 ha spruce forested catchment, 30 km inland from Lillesand in Southern Norway. The Birkenes catchment has been estimated to consist of the following broadly defined soil types: 13 % peaty soils, 23 % podzols, 55 % thin humus soils and 9 % bare rock (Kvindesland et al., 1994).

A number of soil profiles first sampled in 1974 (Frank, 1980) were resampled in 1992, principally to establish changes in base cation pools (Kvindesland et al., 1994). The new soil profiles in 1992 were located as near as possible to the original profile to limit the influence of spatial variability. Soil layer thickness (1974 and 1992), bulk density (1992), stoniness (1974) and loss on ignition (1974 and 1992) were available to calculate C stocks, according to C stock equation (section 3.1). To obtain estimates for the C-content (g C $g^{-1}$ soil) from LOI (weight loss in g $g^{-1}$ soil), a conversion factor was used for the C: soil organic matter ratio of 0.58 for the organic layers, and 0.42 for the mineral layers (C/OM ratios calculated from Nordmoen soils, Kvindesland pers. comm.). When the difference in soil layer thickness between 1974 and 1992 was greater than 2 cm for any mineral layer, the profile was excluded in this comparison.

Total C-stocks in 1974 and 1992 and calculated C-accumulation rates are presented in Table 10. There is a large variability in C-accumulation rates. Even negative accumulation rates (losses of C) were calculated. The average C-accumulation rate for podzols at Birkenes is 60 g C/m$^2$/year. However, the standard deviation was 363 g C/m$^2$/year. For non-soils at Birkenes (thin humus layer) the average C-accumulation rate is 110 g C/m$^2$/year with a standard deviation of 90 g C/m$^2$/year. This shows the great uncertainty in these accumulation rates, due to both spatial variability in soil and poor C % data (due to the LOI :C conversion).

Table 10 C stocks (kg C/m$^2$) and accumulation for soils at Birkenes

| soil type | dominating tree genus | C in 1974 (kg/m$^2$) | C in 1992 (kg/m$^2$) | C accumulation (kg/m$^2$/year) |
|---|---|---|---|---|
| podzol | N.spruce | 15.0 | 22.2 | 0.40 |
| podzol | N.spruce | 21.7 | 18.3 | -0.19 |
| podzol | N.spruce | 17.8 | 20.5 | 0.15 |
| podzol | N.spruce | 17.5 | 22.8 | 0.29 |
| podzol | oak | 11.6 | 16.7 | 0.28 |
| podzol | N.spruce | 38.5 | 28.6 | -0.55 |
| | | | Mean (std) | 0,06 (0.36) |
| Non-soil | Scots pine | 2.2 | 2.9 | 0.04 |
| Non-soil | Scots pine | 1.1 | 4.9 | 0.21 |
| Non-soil | Scots pine | 1.7 | 2.9 | 0.07 |
| | | | Mean (std) | 0.11 (0.09) |

The forest floor of 85 plots distributed over the whole Birkenes catchment was sampled in 1993. Only thickness and loss on ignition data were available to calculate C stocks, so the bulk densities used are mean bulk density for these 3 soil types at Birkenes. The LOI data was converted from weight loss to C-content using the factors mentioned above. Combined results for the C content of the organic horizon (O) and mineral horizons for the 85 plots and resampled profiles and mean bulk densities used for the 85 soil plots are shown in Table 11. Using the C contents shown in Table 10, and the percentage area coverage described above, the Birkenes catchment contains ca. 460,000 kg C(+/- 40 %).

Table 11 Mean C contents and bulk densities of common soil types at Birkenes.
n = nr of observations

| soil type | Horizon | mean bulk density (kg/m$^3$) | mean C stock [n] (kg/m$^2$) |
|---|---|---|---|
| podzol | O | 160 | 6.1 [43] |
| | E | 1110 | 2.0 [7] |
| | B | 990 | 8.3 [8] |
| | C | 1340 | 3.6 [3] |
| thin humus | O | 145 | 4.2 [12] |
| peat | O | 120 | 30.6 [9] |

The case study of Birkenes suggested that soil C had been accumulated from 1974 to 1992 at a rate of 60 to 110 g C/m$^2$/year. However, there was an uncertainty of at least 100% in calculated C-accumulation rates. For comparison, a catchment N budget estimated that 1 g N/m$^2$/year accumulated in the soil (Kvindesland et al., 1993), which would imply a very high C/N ratio of over 60 for new OM accumulating in the profile. This suggests that the C accumulation calculated was probably too high, especially considering that other studies reported lower C accumulation rates. For example, 35 g C/m$^2$ was calculated to accumulate in the organic horizon under spruce forest in Scotland (Billett et al., 1990), but here the uncertainty was also over 100 %.

Sogn et al. (1999) showed that C-storage in the forest floor of homogeneous Norway spruce stands at Nordmoen in southern Norway increased significantly with stand age. No trend was found for the mineral soil. The calculated accumulation rate was 5.4 g C/m$^2$/year, substantially lower than in Birkenes. However, total C-stocks at Nordmoen were also much lower than in Birkenes (Fig. 9, Table 10). No data on stand age were available in the NIJOS database, but assuming that stand age increases with cutting class, our results agreed with the study of Sogn et al. (1999). Vogt et al. (1995) found a SOM accumulation rate in subalpine forests of the cold temperate zone of less than 40 g /m$^2$/year (equivalent to ca 20 g C/m$^2$/year).

## 3.6 Discussion

### *3.6.1 Comparison of NIJOS and OPS with international studies*

The C-stocks calculated for the OPS database were highly variable. The average whole profile C-stock (16.5 kg C/m$^2$) was slightly higher than the whole profile C-stock for productive forest without organic soils (14 kg C/m$^2$). However, podzols were the dominating soil type in the OPS database and Fig. 1 showed that C-stocks in podzols were among the highest for the mineral soil types. Whole profile C-stocks in Birkenes were ca 5 kg C/m$^2$ higher than the mean C-stocks in the other databases, but well within the variation observed. C-stocks in the forest floor were very similar in the OPS and NIJOS databases, ca 5 kg C/m$^2$.

The results of this report were compared with published C-stocks in boreal forest soils (Fig. 9).

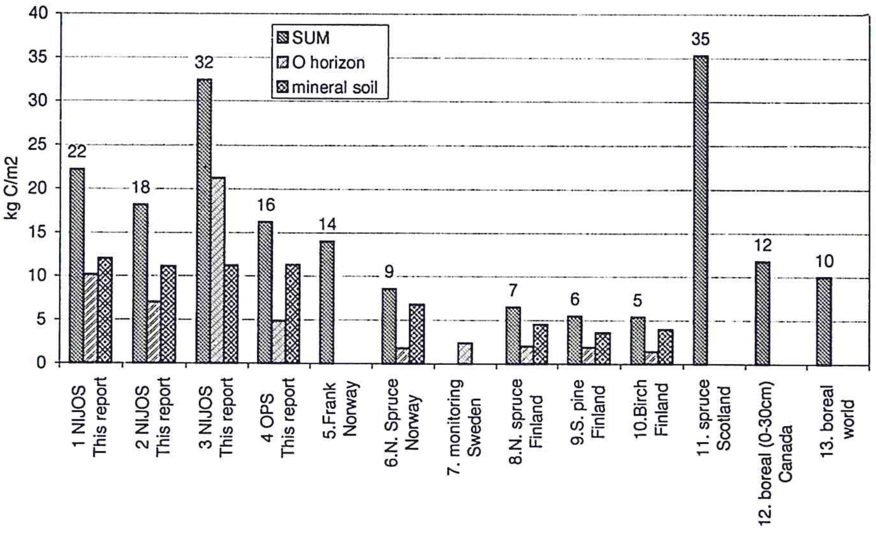


Fig. 9. Comparison of soil C stocks from this report with data from literature. References: 1. NIJOS, all 934 sites (median values); 2. NIJOS, 782 sites; organic soils excluded (median values); 3. NIJOS, 152 sites; only organic soils (median values); 4. OPS, 24 sites; 5. Bruun and Frank (1994); 6. De Wit, pers.com.; 7. Swedish National Vegetation and soil survey average C content for humus in forest soils ; 8. and 9. Liski and Westman (1995); 10. Mälkönen, 1977; 11. Billett et al.,1990; 12. Tarnocai, 1997. 13. Batjes, 1999.

Bruun and Frank (1994) estimated C stocks in Norwegian forest soils by dividing the country into climatic and vegetation zones and using a C stock from a reference profile per region. The estimated C-stocks were 4 and 2 kg/m$^2$ lower than the NIJOS estimate for forest on non-organic soils and the OPS estimate, respectively. Compared with the NIJOS estimate for all forests including those on organic soils, the discrepancy with Bruun and Frank increased to 6 kg C/m$^2$. Thus, our results suggest that the Bruun and Frank estimates of C-stocks in Norwegian forest soils were 2 to 4 kg C/m$^2$ too low.

Surprisingly, carbon stocks in the boreal and temperate forests of Finland and Sweden, particularly in the forest floor, were considerably lower than the C stocks for Norway. It is not clear whether this can be attributed to climatic variability or to methodological differences in for example sampling or analysis and clearly deserves more attention.

The NIJOS estimate for organic soils was similar to the reported C-stocks in Scotland, which consisted of 15 profiles of podzols, cambisols, gleysols and histosols (Billett et al., 1990). The global estimate for boreal forests was only 10 kg C/m$^2$, little over half of the C stocks in Norway, whereas the Canadian C-stocks (Tarnocai, 1998) were more similar to the estimates in this report, especially considering that only the top 30 cm were accounted for. This suggests that C-stocks in Norwegian forest soils are considerably higher than the average global C-stocks in boreal forests.

### 3.6.2 Effects of tree species, cutting class, site index and region on C-stocks

Gärdenäs (1998) related SOM storage in forest floors in European forest to site characteristics, and found that the best single explaining variable was tree genus. She found that Norway spruce stands had significantly higher SOM storage in the forest floor than birch forests and Scots pine. Gärdenäs suggested that Norway spruce accumulated more SOM (and C) in the forest floor than birch because of the litter quality, and thus the decomposability of the litter, produced by Norway spruce.

Interestingly, the effects of tree species in our study are partly in contrast with these findings. In our study, Norway spruce accumulated *less* C in the forest floor than Scots pine forest, but - similar to Gärdenäs - more than birch forest. Additionally, we found that mixed spruce-pine and spruce-birch forests had C stocks that were on average 2 kg C/m$^2$ larger than Norway spruce forests. This is in agreement with the study by Vogt et al. (1995), who found that mixed forests accumulate more than pure spruce forests.

The C stocks in the O horizon of birch and aspen-grey alder forests were clearly lower than for the other forest types. This may be partially explained by a higher biological activity in soils below this forest cover (Harrison and Harkness, 1993; Quideau et al., 1998). Litter may become transported down and incorporated in the upper part of the mineral soil due to the activity of soil fauna, like earthworms. The C-stocks in the mineral soil of aspen/grey alder forest was higher compared to spruce forest even though their soils had the same median depth. This is in agreement with the idea that lower C-stocks in the litter layer is caused by downward transport of organic material by biological activity under deciduous forest. Additionally, grey alder is an N-fixing species that tends to increase N and C stores in the soil (see section 4.3).

No relationship between site index and C-storage was found for spruce forest (Fig. 7). Comparison of non-productive forest and productive forest (Fig. 2) showed a slightly higher C-storage in the forest floor of non-productive sites. This is not in agreement with Liski and Westman (1995) who found a positive correlation between site productivity and C-storage in the forest floor in Finnish forests. Liski and Westman (1995) excluded many sources of variability by choosing sites that were homogeneous in geological parent material, climate, height over sea level and vegetation types. Thus, the lack of agreement between the Finnish study and results presented in this report may be due to the considerable variability in the Norwegian database regarding the parameters mentioned above.

Spruce sites in Western Norway had the highest total C stocks compared with other regions in Norway (Fig. 8). In western Norway, spruce forest was found on sites with a high or very high site index and the majority of the sites were in cutting class II and III. Planting spruce on former birch sites of high site index is a common forest management practice in Western Norway, but uncommon or absent in other regions of Norway. This may partly explain the differences in C-stocks, rather than climatic factors. Vogt et al. (1995) stressed the positive relationship between site nutrient status and soil organic matter accumulation.

The OPS data confirmed the trend of higher C stocks on sites in western Norway and generally lower C stocks in inland sites in eastern Norway. Bruun and Frank (1994) estimated C stocks in Norway by dividing the country into climate and vegetation regions. The regional results of C stocks (Fig. 8) from the NIJOS database do not support this division, but that might be due to the simple grouping criteria we used. The division into different regions was based on county borders rather than meteorological differences like precipitation and temperature data. To do a thorough analysis of the NIJOS database, C-stocks should be related to climatic variability, in addition to stand age, site index, tree species and other relevant factors.

## 4. Forest management effects on soil carbon

Forest management can have a profound effect on carbon storage in forest soils. Johnson (1992) reviewed the effects of forest management on soil C storage. A major problem for assessing effects of forest management was variability in soil chemical properties and various sampling methods used in different studies. Johnson (1992) wrote: 'Problems in this assessment arose due to differences in sampling intensity in the vertical, horizontal and temporal dimensions'. Additionally, effects of forest management on soil C in one climatic zone may not occur in other climatic zones. Extrapolation of results from studies in other countries to Norwegian conditions must be undertaken with caution, particularly when no supporting empirical or experimental evidence exists for Norway.

Most investigations focussed on changes in soil C pools under various forms of forest management, and did not concentrate on C cycling and C fluxes between C pools. This review will therefore concentrate on changes in soil C pools.

In Norway, forest management today is practised at a smaller scale, and often less intense and mechanised level than in countries like Sweden, Finland, UK, Canada and USA. In this review, we focussed on management practices that are more relevant for Norway. The main results from Johnson's global review and Bruun and Frank's Norwegian review on the effects of forest management on soil C will be summarised (Johnson 1992; Bruun and Frank 1994). More recent literature (1993-1999) will also be reviewed, including both experimental studies and model simulations. Whenever possible, empirical evidence of forest management in Norwegian forests will be supplied. A conclusion for each type of management's effect on soil C stock will close each section.

### *4.1 Cultivation*

Cultivation is the conversion of forested land to agricultural land, where the soil is often tilled and the crops be it cereal or pasture are managed far more intensively than under trees. In Norway, very little cultivation occurred in the 1990s and the trend is more towards afforestation.

Johnson (1992) in his review of cultivation concluded that soil C showed a slight gain to over 50 % loss. Mann (1986) reviewed 625 sites world-wide after cultivation and found 20 % decrease in soil C in sites initially high in C, while a slight gain in sites low in soil C. The conversion of forests to pasture has given mixed results in soil C balance (Batjes, 1999). Although initial decreases in soil C were detected after converting an amazon forest to grass, after 8 years the soil C was at the same level as under forest. In Russia, soil organic matter stocks of neighbouring woodlands and meadows were studied (Yakimenko, 1998). The soil organic matter was generally greater in the meadows, which the author attributes to more intensive humus production and greater amounts of fine roots.

*There is general scientific consensus that cultivation of forested lands tends to decrease soil C stocks. In Norway, the present trend is towards increased afforestation.*

### *4.2 Afforestation*

Here afforestation is defined as the covering of land, previously used for agriculture or non-forested e.g. moorland, with forest. However, the land may have been covered with forest previous to this agricultural land-use and this will be a large discussion topic in future Kyoto protocol negotiations. Major global groups have different definitions of afforestation dependent on how long ago areas were covered with forest before they were converted to agriculture (Nabuurs et al. 1999).

Bruun and Frank (1994) point out that for the last 50 years in Norway, an increasing amount of agricultural land has been left fallow, which through natural succession has become forested. This entails a shift from a shorter to a longer circulation time of carbon. They conclude that afforestation of former fallow agricultural soil will increase $CO_2$ fixation, not only in the tree biomass, but probably even more so in the soil. Here they point out that the humid climate along the coast of Norway from Rogaland to Troms/Finnmark can be particularly favourable for the formation of organic matter in the forest floor. As afforestation was recommended as a medium to cheap cost-effective way of binding a medium to large amount of $CO_2$ in trees (Lunnan et al. 1991), afforestation becomes even more attractive when its effects on increasing soil C are included.

Johnson (1992) concluded after reviewing 15 studies that soil C usually increases substantially when agricultural land is reverted to forest or where new soil undergoes afforestation. The majority of the studies show a 30-100% increase in soil C compared to the level before afforestation. Natural woodland regeneration on former arable land in England showed a 117 – 215 % increase in soil organic carbon over a 100 year period (Smith et al. 1997). In Scotland the changes in soil C caused by afforestation of Sitka spruce, Norway spruce and Scots pine were studied over a 40 year period (Billett et al., 1990). They report an average annual accumulation of 35 g $C/m^2$ in the soil.

Soils that are disturbed or undergoing a transition because of vegetation manipulation have different organic matter dynamics than soils in equilibrium with biological and environmental conditions. When cultivation ceases and soils become forested, it is unclear whether the native or cultivated mineralisation rates will apply. Therefore the rate of bomb radiocarbon incorporation was studied over four decades in 8 plots in S. Carolina loblolly pine forest (established 1957) that were previously cultivated for 150 years (Harrison et al. 1995; Richter et al. 1999). The initial C turnover rate in the 1960s of 12 years was found to be twice as fast as in undisturbed temperate forests and grasslands. This resulted in an initial rapid accumulation of C in the forest floor and upper 60 cm mineral soil. However by 1990, although the forest biomass remained a strong C sink (80 % of C in ecosystem), the forest floor only accounted for 20 % and mineral soil (upper 15 cm) (< 1%) of carbon stored in the ecosystem. This implies that the accumulation rates of C in soil during afforestation vary with time. The accumulation rates quoted previously and found in this report are average accumulation rates over a period of time.

*Afforestation in the form of planting or regenerating forests on cultivated drained soils (not on wetlands or peat bogs) will increase soil C sequestration.*

### 4.3 Tree species

The natural climax tree species at a site can be replaced by planting a new species after harvesting. The aim is to plant a tree species well suited to the site, which will give the highest production of timber on future rotations with a specific forest management regime. In Norway, especially on the West coast and in Northern Norway, the general trend has been to plant Norway spruce and Sitka spruce as a replacement of low productive Scots pine or birch. About 16.5 % of the productive forest area on the West coast have been converted to spruce forest over the last decades (Aalde et al., 1997).

Recent recommendations have been to include 30-50% birch in coniferous stands (Frank et al., 1998) to reduce soil acidification and stimulate biodiversity. Birch has a positive effect on the soil, counteracting to some degree acidification by coniferous trees and increasing nutrient levels in upper soil horizons. A recent study of mixed birch and Norway spruce stands concluded that a mixture of 10% birch would have a minimum effect on the timber production of Norway spruce (Granhus, 1996). The results from the NIJOS data suggest, in agreement with other studies (Vogt et al., 1995), that mixed spruce forests store more C than pure spruce forests.

A number of studies suggest that deciduous tree species tend to accumulate more C in the soil than coniferous species. Carbon cycling was studied in mature stands of sugar maple (*Acer saccharum* Marsh.) and jack pine (*Pinus banksiana* Lamb.) (Morrison et al. 1993). The C content of the forest floor was less in the maple compared to the pine, but the C content in the mineral soil (1 m depth) was four times as large in the maple. Calculations of residence time (taking into account litter, fine root and solution inputs) showed that the sugar maple had carbon turnover rates three times greater than the jack pine. This suggests that deciduous species not only have larger C pools, but also larger C fluxes. Vogt et al. (1995) reported a general trend found in many studies that deciduous forests accumulated more soil C than coniferous forests. They also found that mixed forest usually contained more soil C than pure deciduous or coniferous forests.

Several studies highlight the positive effect on soil C pools of N fixing tree species. In Johnson (1992), N fixing species such as Red alder (*Alnus rubra*) clearly increased soil C and N pools. In Norway, there are two N fixing tree species, grey alder (*Alnus incana* L.) and Black alder (*Alnus glutinosa* L.). The C stock results from the NIJOS database (section 3.3) for aspen/grey alder forest showed higher total C-stocks compared to other forest types, in agreement with Johnson. Additionally, a former Grey alder site at Kårvatn was planted with spruce (30 years old) and here the C/N ratio in the forest floor and top mineral soil was far lower than in any other monitored spruce site (Monitoring, 1991). Thus, planting of N-fixing species could be a tool for increasing C-sequestration in soils.

No significant effect of several tree species on soil C pools in the litter layer or top 15 cm of mineral soil was found by Alriksson and Eriksson (1998). The tree species, which included Norway spruce, Scots pine, Douglas fir, birch (*Betula pendula* Roth) and larch (*Larix sibirica* Ledeb) were grown for 27 years on set-aside farmland in N.E. Sweden. Possibly, more than 27 years are needed to be able to detect effect of tree species.

*Suitable tree species for planting to increase soil C-sequestration include planting of N fixing tree species and mixing pure spruce and pine forests with birch or other deciduous species. Planting only deciduous species is probably not as effective as planting both deciduous and coniferous species.*

### 4.4 Drainage

Forestry in Norway has been dramatically decreasing its drainage of wetland areas for tree planting over the last decades (Statistics Norway, 1998). This is due to economic conditions and the increased focus on preserving mires. The number of ditches maintained – cleaned out or added to- have halved since 1989 (Statistics Norway, 1998).

In 1991, drainage of bogs was considered a medium to cheap cost effective way of binding a medium to large amount of $CO_2$ in trees (Lunnan et al. 1991), but here the effects on soil C were not considered. Bruun and Frank (1994) conclude for Norwegian conditions that drainage of mires can be an effective method of binding carbon for a specified timespan, if certain conditions are met. These conditions include fertilising to establish a productive forest. If the forest owner does not follow up the drainage and planting of a bog, a poor stand of trees is the result and drainage will result in a net loss of $CO_2$.

It is clear from the NIJOS database (section 3 – Fig. 1) that forest sites with organic soils (peats and folisols) have the largest C stocks of all soil types. Non-productive forest sites have larger C stocks in their organic soils than productive forest sites (section 3- Fig. 2).

It is well known that effective drainage increases the aeration of the soil, creating aerobic conditions for decomposers, which can decompose at a rate far greater than anaerobic decomposition. This means that while one greenhouse gas (methane) production declines, $CO_2$ production increases and the peat can begin to subside. Therefore in the years before trees establish, the bogs will be a net source of carbon. However tree productivity increases as nitrogen is released through mineralisation (and helped by fertilisation of nutrients in short supply in bogs), creating greater aboveground and belowground litterfall to the soil. Trees also represent a greater C sink than normal bog vegetation. Therefore the total C balance in a drained afforested bog ecosystem after one generation of trees is of real interest.

In the UK, Cannell et al. (1993) investigated whether planting conifers on peat bogs would result in a net gain or loss of carbon. They estimated that the C store in peatlands was approximately 30 times larger than all carbon stored in vegetation in Britain. Using models, the amount of carbon sequestered by Sitka spruce systems (soil and vegetation) was simulated. Cannell et al. (1993) concluded that the total amount of C fixed reached a maximum after 2-3 rotations and was less than the amount of C contained in 15-40 cm of peat, depending on yield class and the type of peat and its oxidation rate. If rates of peat oxidation were <100g C /$m^2$/year then afforestation gave a net benefit of C accumulation for about 200 years. If oxidation rates were 100-200 g C/$m^2$/year then afforestation only gave a net benefit for C accumulation for 50 – 100 years. If oxidation rate were > 300 g C/$m^2$/year, afforestation was to be avoided. Extrapolating these model simulation results to deep peats and shallow peats (defined as peaty gley and peaty ironpan soils), the authors recommended that trees were not to be planted on peats that are deeper than 35 cm (deep peats) or 21 cm (shallow peats).

In Finland, $CO_2$ fluxes from mires were measured under varying temperature and moisture conditions (Silvola et al. 1996). The lowest $CO_2$ fluxes of 60-200 g C/$m^2$/year were from ombrotrophic sites dominated by *Sphagnum fuscum*, whereas the same sites with abundant understory vegetation had fluxes of 290-340 g C/$m^2$/year, both measured at 12 °C. Silvola et al. calculated that lowering the water table by 1 cm increased $CO_2$ fluxes by 9.5 g C/$m^2$/year. Another Finnish study estimated the carbon balance of a mire by studying soil profiles on a transect running from an undrained to a drained afforested area of the mire (Laine and Minkkinen, 1996). The average difference between the undrained and drained peat carbon stores averaged over 30 years after drainage was 35 g C/$m^2$ greater C accumulation in the undrained mire. However when tree C biomass was taken into account and the C balance was estimated over 300 years, they calculated that drainage increased the total carbon store of the mire if no tree harvesting or thinning was performed, but the C store was unchanged, if standard harvesting was performed in this period.

In a review of soil carbon in northern-forested wetlands, Trettin et al. (1995) concluded that carbon losses were highest in site-prepared and drained wetlands, where the aerated soil volume and water table fluctuations were greatest. They also pointed out that future research must study the effects of silvicultural practices on soil C balances in peatlands and determine the partitioning of the C flux between the aqueous and gaseous pathways.

*In conclusion, draining and planting trees on boggy areas with peaty soils is a risky affair, if trying to sequester C. The C accumulated in the trees and soil during the tree's rotation must exceed the C lost during tree establishment. If tree growth is poor due to unsuccessful drainage, fertiliser regime or general bad forest* *management, the site will become a net source of C during the forest rotation. In addition, there are indications that peat soils are more sensitive to forestry operations such as site preparation and harvesting, in that they are likely to lose more C during these disturbances than other soil types.*

### 4.5 Site preparation for next tree generation

The site preparation discussed here is mechanical soil disturbance (scarification) after harvesting, to promote good planting/regeneration conditions. There are many different types: patch scarification, trenching, mounding and ploughing representing increasing intensity of scarification. In patch scarification a patch of humus is inverted at a certain rate, e.g. 2500 patches per ha, to encourage the initial growth of planted seedlings or natural regeneration and decrease weed competition. Mounding, disc trenching or ploughing always invert the humus layer and in addition cover or mix it with the mineral soil. Mounding creates an elevated planting site and mitigates against increase water table levels following harvesting at wet sites.

In 1995, 57 % of harvested forests in Norway were replanted, while 43 % regenerated naturally (Statistics Norway, 1998). Only 15% of this planted and regenerated area was reported scarified, although not all site preparation is reported. This shows that there is still a lot of planting occurring on undisturbed vegetation in Norway. In contrast in Sweden, 90 % of regenerated forest was reported to be scarifed in the 1980s (Johansson, 1992). Patch scarification is the most usual type of site preparation in Norway and is principally used to encourage natural regeneration, either on shelterwood stands of Scots pine and Norway spruce or after clear-felling. It is particularly practised at sites with a thick humus layer prone to drought.

Johnson (1992) concludes in his world-wide review that there is a net loss of C, which increases with intensity of site preparation. However, the site preparations listed in his review are quite different from site preparations normally done in Norway.

A recent Swedish study of mechanical soil scarification methods (mounding, disc trenching and ploughing) showed that the average losses of C were related to the degree of disturbance and estimated this as a 7-16% loss (Eriksson and Alriksson, 1998). The report indicates that the fertility of the site affected the C loss, with sites with high pools of C and N suffering greater losses. On peat sites, site preparation, like mounding can lead to high levels of decomposition and hence C loss because of the increased soil temperature and aeration (Trettin et al. 1995).

*In Norway, intensive scarification methods are used to a very minor degree. The main site preparation is patch scarification. Although decomposition conditions will be altered drastically at the actual patch, on an area basis little change in carbon pools will occur. In addition the benefit of speeding up regeneration and the next tree cover could outweigh any C loss in the patches (Bruun and Frank 1994).*

### 4.6 Use of herbicides

Glyphosate is the active ingredient of the most commonly used herbicides to combat weeds, before and after planting or regenerating conifers. Weeds and deciduous vegetation are killed after application, allowing better growth conditions for the conifers and the former vegetation regenerates 2 to 3 years later. The use of chemical herbicides in forestry has decreased in Norway by ca. 70 % from 1989 to 1996, while the area being cleared and weeded has decreased by 45 % (Statistics Norway, 1998).

Very few studies have been done on the effects of soil C and herbicides. In Australia, Carlyle (1993) studied the effects of a weed-free row of tree plants versus a weedy row on the soil C status of sandy podsols. The topsoil C concentrations were consistently higher in weedy strips than the weed-free row. Input of above-and belowground biomass by weeds was clearly maintaining soil C levels in the 3 years after tree planting, when inputs from the developing plantation were absent or minimal. However the organic C in this Australian sandy podzol reacted quickly to all forms of forest management.

*In Norway, unless on a very unfertile, sandy site, prone to erosion, the use of herbicides should have no long-term and probably little short-term effect on soil carbon levels.*

### 4.7 Thinning

Thinning temporarily reduces the canopy, resulting in less litterfall from the standing biomass and more light, heat and rainfall to the openings. This may temporarily increase C mineralization of the forest floor. However, input of residues from the thinning may temporarily increase the input of C to the forest floor.

Carlyle (1993) studied the effects of residue management after thinning and clear felling conifers on sandy soils in S.E. Australia. Residues are the tree litter left after removing stems by clear felling. Carlyle estimated that the residues represented a considerable portion of the site's carbon reserves - 123% of soil C for harvesting residues and 67% of soil C for thinning residues. If the harvest residues were removed, there was a significant decline ($p<0.01$) in topsoil C the first 6 years after harvest, compared to when residues were retained. For thinning there was no significant decline, which was interpreted to be due to the continued input of plant litter and maintenance of pre-thinning decomposition rates. Bruun and Frank (1994) concluded that some thinning in a forest's rotation would probably not influence the accumulation of C in the soil.

The effects of thinning on soil and biomass C have been simulated (Nabuurs et al. 1999). Model simulations showed that thinning improved the growth of the remaining trees, but did not affect soil C. Carbon was removed by thinning and even if this was used in tree products, the model simulated no increase in C sequestered by thinning.

Increased tree stand density is recommended by Aalde et al. (1997) as a method of increasing $CO_2$ fixation and timber production in Norwegian forests. Tree litterfall (both above- and below-ground) would also increase, although the original ground vegetation may decrease because of shading. Assuming the same type of forest management as for less dense stands, soil C can be assumed to remain stable or increase under denser tree stands.

*Carlyle et al. (1993) reported that there was no response of soil C in sandy Australian soils to thinning and removal of thinning residue. It can be assumed that C in generally less sandy Norwegian forest soils will show no long-term response to thinning.*

### 4.8 Nitrogen fertilisation

A normal fertiliser dose of 150 kg N/ha causes an increase in stemwood of 30-50% during the following 10 years. This is mainly due to an increase in the canopy, which implies that needle litterfall also increases.

Fertilisation of Norwegian forests was more common in the 1960s and 1970s than today. Now, the use of fertiliser has been reduced for a number of different reasons. The increasing costs of fertilisers, little active forest management by small forest owners, increasing environmental awareness of the problems of nitrate leaching, and the fact that southern Norwegian forests receive approximately 10 kg N/ha/year in rainfall could all be mentioned as contributing factors. Forestry statistics show that the area of forest fertilised annually has halved between 1989 and 1996 (Statistics Norway, 1998). In 1996, 0.04 % of the productive forest was fertilised in Norway.

Johnson (1992) combined the presence of N fixing species and N and P fertilisation in his review into one category and concluded that there is a clear trend towards increased soil C pools with their use. He mentioned two possible mechanisms to explain this process; 1) increased productivity and thus greater organic matter input to soils and 2) stabilisation of soil organic matter, which results in decreased organic matter decomposition (see section 2.3.2).

It is well documented that an increase of N fertiliser to a forest ecosystem results in greater C pools. Nohrstedt (1990) found that the thickness of the humus layer increased with increasing N dosage in Scots pine stands. The weight and carbon content of the humus layer showed similar trends and the carbon content had doubled from the control plot to the 1500 kg N/ha plot. There was no significant trend in the eluvial layer, but both B horizons showed an increase in C content with increasing N dosage.

In Sweden and the UK, sewage sludge is often applied to forests and can be an excellent soil amendment in areas with reduced organic matter content. It can add 2-3 % organic matter to the soil at typical application rates (Wolsterholme and Dutsch, 1995), which increases water holding capacity and soil structure, encouraging better ground cover and thus reduced erosion.

Fertilising was considered a cost-effective method to increase $CO_2$ fixation in Norwegian forests in 1991, but here C fixation in biomass was the primary aim (Lunnan et al. 1991). In 1997, although fertilisation is mentioned as a management method increasing carbon accumulation in forests, it is not highly recommended.

The risks of N fertilisation are mentioned, including leaching to watercourses and the potential for acidification and eutrophication (Aalde et al., 1997).

*The addition of N to forests, be it by fertilisers or through N-fixing species, generally increases the amount of C stored in the soil. Forests in Norway are generally N limited. However, as stressed above, N fertilisation can result in nitrate leaching, especially if the timing and amounts are wrong. The site for N fertilisation must be selected from a hydrologic, fertility and tree age class perspective.*

## 4.9 Liming

Liming is the spreading of dolomite and/or lime (rich in the divalent cations $Ca^{2+}$ and $Mg^{2+}$) on the forest floor and is a method to counteract both forest soil and water acidification. Cation bridging of organic colloids causes condensation and stabilisation of organic matter. Johnson (1992) suggests that this could be the reason why four different studies found a positive effect of liming on soil C content. The long-term effects on soil chemistry of liming old Scots pine forest were studied in Norway (Nilsen 1998). Thirty years after liming, the C concentration in the humus layer had decreased with increasing lime doses. However as the bulk density of this layer also increased with liming, the amount of carbon on an area basis showed no effect with liming. A Finnish study of experiments involving both liming and fertilisation showed a 20 % increase in organic matter in soil with liming stands of Norway spruce and Scots pine (Derome et al., 1986). Using N fertilisers alone gave a greater increase in SOM than using both N fertilisers and liming. Stuanes and Abrahamsen (1994) found a decrease in organic material in soil 14 years after liming stands of contorta pine, Scots pine, Norway spruce and birch. Most of the decrease was registered in the humus layer. Concluding, studies on effects of liming report both increases and decreases in soil C. Possibly, the effect of liming on soil C is dependent on the nutrient status of the soil (Nilsson et al., 1996). On fertile soils with low C/N ratios, liming can induce an increase in earthworm activity, which increases decomposition and decreases the soil C content. On less fertile soils with higher C/N ratios and no earthworm activity, decomposition of SOM is unaffected whereas production of litter may increase, leading to accumulation of soil C (Nilsson et al., 1996).

*In Norway, liming is today not recommended as a means to counteract acidification of soils and water in Norway. International studies reported both increases and decreases in soil C after liming, but the few Norwegian studies indicated a decrease in soil C after large chalk doses.*

## 4.10 Rotation length

Rotation length is defined as the age when the trees are mature for harvest plus the time for new trees to be regenerated or planted. Here we use the biological definition of 'mature for harvest', that is the stand age when the annual average increment of a stand culminates, rather than the economical definition, which is generally shorter and depends on timber prices and interest rates. In Norway, stands are today often harvested at least 10 years after they are mature for harvest (Aalde et al., 1997).

In Norway, net increments of old stands are often positive for decades after forest is mature for harvest (Nelleman, 1992; Nilsen and Haveraaen, 1982). According to NIJOS data, soil C increased in the organic layer of both pine and spruce stands with cutting class, which is related to stand age. The highest increase was found going from cutting class IV to V in pine forest. Cutting class V may contain a large variation of stand ages: both mature for harvest stands and very old stands, which implies that the relation to stand age is not straightforward. However, the results from the NIJOS database suggest that an increase in rotation length of a forest will stimulate C-accumulation in the forest floor.

The C accumulation rate in the forest floor estimated from a chronosequence of Norway spruce stands (0-100 years old) was 5.4 g C/m$^2$/year (Sogn et al., 1999). This is a very low C accumulation compared to what has been found accumulating in the forest floor in Scotland (35.5 g C/m$^2$/year) (Billett et al., 1990) and Birkenes for the whole profile (60 g C/m$^2$/year) (this report). However, C-stocks at Nordmoen were substantially lower than at Birkenes and in Scotland (Fig. 9 and Table 10) which may partly explain the relatively low C accumulation occurring there.

Generally, the accumulation rate of soil C is expected to slow down as the stand matures but it is uncertain when the equilibrium level is reached under stable vegetation. In a study of a 5000-year chronosequence of boreal forest soils in coastal western Finland, the mean residence time of soil C in the mineral soil (0-30 cm) was 2000 years (Liski et al., 1998).

Although soil and biomass C pools might still increase under a mature forest, losses of C due to tree rot, insect attack, fire, wind and snow damage may outweigh this C gain. In Norwegian forests, 27% of Norway spruce trees were infected by root rot fungi at harvesting during the winter of 1992 (Solheim et al. 1994). The percentage of infected trees increases with stand age (Huse 1983). Tree rot increases the risk for trees felled by wind and decreases the sink strength for C in biomass ($CO_2$ emissions from decaying dead biomass). Pests and forest fire do not usually occur over large areas in Norway.

The C sequestration potential of unmanaged forest stands in Finland were simulated using gap type models both under current and changing climatic conditions (Karjalainen, 1998). According to their simulations for the current climate, the stands were C sources during the early and late phases of stand development, since emissions form decaying litter and soil organic matter in the humus layer exceeded the growth of vegetation. Stands became C sources earlier in their development under changed climate conditions (increased temperature, particularly in winter) than under current climatic conditions.

According to model simulations for Canadian forests (Price et al., 1997), C losses due to increased fire and insect induced stand mortality would outweigh any increase in ecosystem C caused by longer rotations. Model simulations in the Netherlands indicated that a 15% increase in rotation length increased biomass C pool slightly, but not the soil C pool (Nabuurs et al., 1999).

Fleming and Freedman (1998) studied carbon storage in conifer plantations versus natural undisturbed mixed forests in New Brunswick, Canada. Their study suggested that a landscape managed as a shifting mosaic of plantations on a 60 year rotation would store only about 22% of the above-ground carbon (in live trees, snags, coarse woody debris and the forest floor) as a landscape covered in older-growth natural forest.

*Although model simulations do not support that increasing rotation length increases C-sequestration in forest soils, there is contrasting empirical evidence. Several studies suggest that increased rotation length in managed forests generally leads to an increase in soil C and biomass C pools, although the rate of C accumulation will be much lower than in young stands. The increased risk of tree rot, however, potentially weakens the sink strength of the biomass C pool.*

### 4.11 Fires

In forestry, one can define two types of fire: 1) Controlled or prescribed burning, which in Norway is occasionally used on small areas with a thick humus layer to increase mineralisation and chances of natural regeneration or is used for game management (mostly in mountainous regions); 2) Natural, not-controlled, wildfires in dry seasons, which are either caused by humans or by lightening. Wildfires have been much rarer in Norway since the industrial revolution, due to more effective forest fire protection. However, wildfires initiated by man have been important for the last 10,000 years in Norway. Fire was an effective but uncontrollable way of clearing the land. In the Middle Ages, tactical burning of coastal forests was used by the Hanseatic League and others to destroy timber resources and prevent competition (Mysterud 1997). The history of forest fire influences the present soil organic carbon reserves at a site, but such historic information is generally unavailable.

Johnson (1992) as others (Conrad and Ivanona, 1997) pointed out that the intensity or severity of the fire determines the outcome of burning for soil C reserves. In addition postfire mortality, decomposition of fine fuels and changing postfire vegetation structure are important influences on the soil C pools, following fire (Conrad and Ivanona, 1997). A light or moderate burn may decrease soil C in the forest floor, but often results in no change or an increase in mineral soil C, i.e. a redistribution of soil C in the profile. It often causes a mobilisation of nutrients, allowing N fixation plants to establish and promoting growth in subsequent forests. Severe fires, however, deplete the soil of volatile nutrients (N, P and S), causing a long-term decrease in forest productivity and C sequestration.

Changes in carbon storage were measured in different stands in southern boreal forests in Minnesota 5 and 23 years after a large wildfire (Slaughter et al., 1998). They concluded that although there were gross fluxes of C following the fire, massive net losses of C occurred neither immediately after the fire or two decades after.

Modelling the effects of fire over several rotations of forests, however, often showed reduced carbon reserves in soil, which the real-life studies did not register after one rotation. In Finland, simulations of repeated fire reduced the amount of soil C by 25 % (Liski et al. 1998). Future carbon budgets of Canadian boreal forests (> 300 Mha land area) were simulated using the Carbon Budget Model of the Canadian Forest Sector (Kurz and Apps 1995). The modelling work showed that fires and insect-induced stand mortality largely influence the dynamics of this forest, while forest management had a smaller influence. The 1980s was the warmest decade on record in Canada and it was also the decade with the largest area of forest fire in the statistical record. This perhaps gives some indication of how climate change can affect wildfire frequency. They had two fire scenarios – a low fire run with fire at 50 % of the normal (60% of the area burnt in 1985-89) and a high fire run with fire at 200 % of the normal. The change in ecosystem C (Pg) over 50 years was +2.3 for the low fire run and – 1.4 for the high fire run.

*The intensity of fire, postfire mortality and postfire vegetation will all influence the outcome for soil C reserves. An out of control severe fire diminishes ecosystem C stocks. Controlled burning is less intense. Although C stocks will be reduced in the forest floor, these recover when new trees establish. Therefore forest fire protection should be encouraged, but the small amount of controlled burning that occurs in Norway for biodiversity reasons is of little consequence for soil C stocks.*

### 4.12 Harvesting

There are many intensities of harvesting. Whole tree harvesting (Norwegian: *heltrehogst*) means the removal of all aboveground biomass and is uncommon in Norway. Clear cutting or felling (Norwegian: *snauhogst/flatehogst*) involves the removal of only the stem or timber and is the usual form of harvesting in Norway. Shelter tree method (Norwegian: *frøtrestilling*) and selection system (Norwegian: *blendningshogst/ gruppehogst*) are less intensive forms of harvesting used in Norway due to regeneration success, economy and partly aesthetic reasons. According to a sample survey for forestry in 1995 (Statistics Norway 1998), 91 % of harvesting was clear cutting and shelter tree method, 7 % was thinning and 2 % less intensive kinds of harvesting. Felling and delimbing with a tree processor was used by 46%, 3% felled with power chain saw and then delimbed with a tree processor and 51% did the whole job with a power chain saw. The most mechanical method – the tree processor had increased in usage from 36 % in 1991 to 46 % in 1995.

The immediate consequences of mechanical tree harvesting are physical disturbance and relocation of soil materials. Physical disturbance appears to be correlated with low and high extremes in slope, with direct damage from machinery where the slope is gentle and drainage is poor and erosion where the slope is steep and runoff is high. With the tree canopy removed, the soil is exposed to new environmental conditions – increased light and greater variation in temperature and moisture.

Johnson (1992) reviewed 13 studies of whole-tree harvesting and conventional clear-cutting. He concluded that the majority of studies reported either no effect or very small changes (< 10%) in the long term, but that soil carbon could increase or decrease significantly the first few years after harvest (Johnson 1993). Bruun and Frank (1994) concluded that soil C under Norwegian forests on fertile sites would remain stable or decrease only slightly in the long-term compared to natural, undisturbed forests, when clear-cutting followed by fast natural regeneration or planting is performed. They mentioned the dangers of clear-cutting sites that are hard to regenerate, e.g. mountain forests or other areas with hard frost. Here, harvesting can decrease soil C over longer periods or perhaps permanently, as forest reestablishment takes such a long time. Bruun and Frank also suggested that planting rather than natural regeneration is more favourable for reducing the 'treeless' period, when soil C decreases.

Many studies of whole-tree harvesting and clear–cutting have been performed in Northern hardwoods in the USA. One study assessed the degree to which logging altered soil horizontation, bulk density and organic matter pools (Johnson et al. 1991). They found that while the thickness of the O horizon decreased, O horizon mass and OM content increased. One fourth of the soil pits exhibited a new Ap horizon that was not present prior to harvest, formed from soil of O, E and B horizons. Bulk density increased in the top mineral soil (5-15%) probably due to compaction from machinery. Overall, the total pool of organic matter in the soil profile did not change following harvest, as losses of C to stream water and respiration were compensated by inputs from decaying roots and leaf litter. Another study on whole tree harvesting (Huntington and Ryan 1990) echoed these conclusions, showing that although N and C concentrations decreased in the L and F litter layers, the mass of the H layer increased and compensated the reductions. Mechanical disturbance by logging displaced or buried some of the forest floor resulting in increased variability in soil C and N pools. Two much quoted studies (Covington 1981; Federer 1984) sampled soils in Northern hardwoods. Both studies plotted OM content of the forest floor against age of tree after clearcutting. Within 10-15 years after cutting, the OM content reached a minimum and it took 50 years for OM content to reach mature forest levels. A comprehensive study of litterfall dynamics was undertaken in different aged forests (Hughes and Fahey 1994), which confirmed the findings above. As the forest developed, the dynamics of litterfall changed which resulted in a reorganisation of the stand's nitrogen and carbon capital. Beginning about 15 years after large-scale disturbance, the forest received large pulses of wood-fall that was low in N but high in lignin and other structural carbohydrates. Leaf fall at this stage in forest development was also relatively high in structural carbohydrates but low in N, which resulted in reduced decomposition rates. The overall change in the lignin:N ratio of litter detritus is probably responsible in large part for the stabilisation of forest floor mass during this state of forest development.

The impact of conventional harvesting was studied in the mixedwood forest (aspen/white spruce) of Saskatchewan, Canada (Pennock and van Kessel 1997) comparing clear-cut sites 1-3 years after harvest and clear-cut sites 6-20 years after harvest with control sites of mature mixedwood forest. A few years after harvest, the

C content of soil (0-45cm) was significantly greater (0.04) compared to soil in the control plot, while soil in plots 6-20 years after harvest showed significantly lower C content (0.02), N content (0.00) and lower forest floor thickness compared to soil in control plots. The increase in soil C content one to three years after harvest was due to the addition of above- and below-ground harvest residues. After five years, the C content of soil began to decrease as the harvest residue effect was over and there was now minimal above- and below-ground litter deposition compared to under a mature mixed hardwood stand.

The effects of whole-tree harvesting and clearcutting were investigated in white birch stands in Newfoundland (Roberts et al. 1998). Four years after harvesting there were statistically significant decreases in the litter depth and forest floor depth at both levels of harvesting versus the control plots, but not much difference in depth between stem-only versus whole-tree harvesting. The same two types of harvesting were studied in a mixed oak stand in a warmer climate in Tennessee (Johnson and Todd 1997). They concluded that 15 years after harvest, there were no significant effects on the regeneration of biomass, species composition, soil bulk density, C, N, or C/N ratio in soil between the clear-cut and whole-tree harvested plots.

Trettin et al. (1995) suggest the impact of harvesting is far greater on peatland forests than on mineral soil forests, because of their thick O horizons. Changes in soil temperature, aeration and nutrient availability after harvesting increase organic matter decomposition. Few studies, however, have been performed on the long-term effects of different harvest intensities on peatland forests.

A study of residue management in sandy podzolised soils of S.E. Australia indicated that carbon concentrations in sandy soils are sensitive to forest operations, in particular management of residues after clearfelling/thinning and weed management (Carlyle 1993). Carlyle suggests that only a small fraction of the carbon added to a sandy soil will become chemically stabilised, the rest being lost fairly rapidly as $CO_2$ (rates governed by substrate quality and environmental conditions). This is in contrast to more clayey soils, where a significant portion of C, although readily decomposable, is rendered resistant to microbial attack by physical protection within soil aggregates (Greenland, 1997).

In Sweden, the effects of whole-tree harvesting (intensive), clear cutting (conventional) and windfall (clear-cut – no biomass removed) on soil C have been simulated for a spruce forest ecosystem over 300 years (Bengtsson and Wikström 1993). Although soil C was less after intensive forestry than after conventional forestry, the differences were small. The major effect on soil C was rather the decrease occurring with conventional harvesting compared to no biomass removal. A more recent study of the effects of whole-tree harvesting versus clear cutting in Norway spruce and Scots pine in Sweden (Olsson et al. 1996) agreed with these simulations. Clear cutting resulted in marked reductions in C and N contents in the humus layer the first 15 years after felling, with increases in the mineral layers. The intensity of harvesting had no significant effects on the C or N pools. A simulated clear-cutting of a boreal forest in Finland (Liski et al. 1998) caused a temporary decrease (5-10%) in soil C over a 20 year period. Over two one-hundred-year rotations, the long-term decrease in soil C was 14 %. A simple dynamic model of C decomposition in soil, parameterised with data from a 5000 year chronosequence in

Finland was used for the simulation. The amount of carbon added in harvest residues and from litterfall during growth were based on data from a Scots pine stand (myrtillus type) and the site's initial soil carbon content was calibrated for a previously unharvested stand.

A few studies have attempted to measure C cycling fluxes following harvesting. A northern spruce/fir forest in Maine, USA was clear-felled. Soil $CO_2$ flux, litter decay and root dynamics were followed in a unfelled reference plots and clear-felled plots during the first year after harvest (Lytle and Cronan 1998). Aboveground litterfall was greater in the unfelled plots, but soil $CO_2$ flux was 16% greater in the clear-felled plots. This appeared due to fine-root decomposition (estimated at 35% of the initial standing crop of fine root biomass and necromass) and live root biomass was measured decreasing sharply in the clear-felled plots during the growing season. Another study in Ontario compared carbon cycling in three mature forest types and estimated the effect of different harvest intensity on the stands biomass carbon reserves (Morrison et al. 1993). Clearcutting removed 20% and 33% on site carbon from the sugar maple and jack pine site respectively, while whole tree harvesting removed 27 % and 38 % from the same sites.

*In conclusion, the harvesting process clearly affects the soil C status. Assuming clearcutting, the forest floor receives a huge dose of branches, twigs, needles (harvest debris), and the soil receives a large dose of dying roots, which initially (1-3 years) can temporarily increase the C content. The Scandinavian studies indicate that the forest floor soil C content will decrease the first 15-20 years after harvest and most studies indicate a redistribution of soil C from the forest floor to the mineral layers, which can also be caused by tree processors/tractors disturbing the soil. The soil C recovers during the next tree rotation, but whether to the same level (as found in Northern hardwoods in USA), a lower level (7% lower was predicted by Finnish model simulations ) or a higher level is hard to predict. It is important to consider the soil type when choosing the harvest intensity. Peaty soils and sandy soils appear to be more sensitive to harvesting and removal of residues than other soil types. This sensitivity is due to the unprotected nature of the organic matter in sandy soils, which can be more quickly mineralised in exposed situations. Peat is sensitive because it consists of a thick layer of organic matter, which is often exposed to large fluctuations in water level after harvesting, again promoting faster mineralisation. Here less intensive harvesting and minimal removal of residues should be practised.*

## 5. Conclusions

Based on the NIJOS database of 934 Norwegian forest sites, the median C stock in the whole profile for all forest sites in Norway was 22 kg C/m$^2$. The organic soils had clearly the largest C stocks, followed by gleysols, then podzols. Excluding sites with organic soils, the median C stock in the forest floor was 7 kg C/m$^2$ and in the mineral soil was 11 kg C/m$^2$. The uncertainty in the C stock calculation was largely due to uncertainty in bulk density, soil stone content and analysis of soil C-content. This led to an estimated uncertainty in soil horizon C-stocks of 17% for organic soil types and for mineral soil types, 17 % for organic horizons and 22 % for the mineral horizons. For estimates of C-stocks in soil profiles, an uncertainty of 25 to 28% was calculated, but this was a conservative estimate. Other sources of error such as sampling method, missing data and representativity of the soil profiles probably add to the uncertainty calculated, but could not be accounted for quantitatively.

Compared with other published boreal forest C stock data, the C stock estimates from this report were high. The estimated global average soil C stocks for boreal forests was 50% of the Norwegian C-stocks, whereas soil C stocks in Sweden and Finland were ca 30% of Norwegian C-stocks. However, C-stocks estimated in this report were similar to soil C stocks in Canada and Scotland. Previously estimated Norwegian soil C stocks, based on the extrapolation of already published soil C content data (Bruun and Frank, 1994) were 2 - 4 kg C/m$^2$ lower than found in this report.

With regard to forest type and its effect on the organic horizon, the lowest C-stocks were found in the sites dominated by birch and aspen-alder forest (ca 3 kg C/m$^2$). Mixed forest and pure Scots pine forest carried the highest C-stocks (ca 6 kg C/m$^2$), whereas C-stocks in pure Norway spruce forest were circa 2 kg C/m$^2$ lower. A higher C density was found in the mineral soil under the aspen-alder forest compared to spruce, even through the forests had the same median depth. This was thought due to the N fixing properties of grey alder.

C-stocks in the forest floor of pure pine and spruce forest were larger in higher cutting classes. This may be related to stand age and implies that an increase in rotation length will increase C-accumulation in the forest floor. A very uncertain C accumulation rate of 60 g C m$^2$/year was calculated for podzols at Birkenes in southern Norway, which is very high compared to the 5.4 g Cm$^2$/year which has been calculated for spruce forest at Nordmoen (forest floor).

Forest management practices that increase the accumulation of carbon include afforestation in the form of planting or regenerating forests on cultivated drained soils. Suitable tree species for planting to increase soil C-sequestration include N fixing tree species, such as Grey alder and mixing pure spruce and pine forests with birch or other deciduous species. Nitrogen fertilisation will also increase the C accumulated in soil, but the site for N fertilisation must be selected from a hydrologic, fertility and tree age class perspective to avoid nitrate leaching to surface water. Increasing rotation length on managed forests will generally store more C in the tree biomass and soil for the years in question, although the rate of C accumulation will be much lower than in a young stand and this increases the risk of stand damage by tree rot, wind and snow.

Forest management practices that have little and no long-term effect on the C stock in soil include use of herbicides, thinning, patch scarification and controlled, less intensive fires. Forest management practices that affect C stocks negatively are cultivation (conversion of forest to agriculture), drainage of peatlands, highly mechanised site preparation, the intense forms of harvesting such as clearcutting and whole tree harvesting and intense wild fires. Organic soils and very sandy soils appear more sensitive to site preparation and harvesting practices, losing more C than other soils after these events.

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## **Rapport** fra skogforskningen

**Utkommet i 1999:**

1/99: Per Otto Flæte og Bohumil Kucera: Virkesegenskaper til mellomeuropeiske og norske granprovenienser plantet i Østfold.

2/99: Stein Magnesen: To proveniensforsøk med engelmannsgran på Vestlandet.

3/99: Halvor Solheim: Sporespredning hos rotkjuke (*Heterobasidion annosum*) i Rana og Saltdal.

4/99: Øystein Dale og Morten Nitteberg: Målenøyaktighet for diameter og lengderegistreringene på hogstmaskiner.

5/99: Kjell Vadla: Virkesegenskaper hos bjørk, osp og gråor i Troms.

6/99: Stein Magnesen: Forsøk med nordlig sitkagran-provenienser på Vestlandet.

7/99: Tron Eid og Svein Ola Moum: Bestandsuavhengig bonitering og nøyaktighet.

8/99: Tron Eid og Petter Økseter: Bestandsuavhengig bonitering og konsekvenser.

9/99: Ingvald Røsberg og Dan Aamlid: Program for terrestrisk naturovervåking. Overvåking av jordvann - Årsrapport 1998.

10/99: Ketil Kohmann: Overlevelse og utvikling av ulike plantetyper av gran under ulike forhold i Oppland, Hedmark, Sør- og Nord-Trøndelag.

11/99: Dan Aamlid, Svein Solberg, Gro Hylen, Kjetil Tørseth: Skogskader og skogovervåking i Norge. Årsrapport for Overvåkingsprogram for skogskader 1998.

12/99: Knut Solbraa: Barkdekking i eldre furuskog og tilføring av koakkslam og fullgjødsel i furuforyngelser.

13/99: Kjell Vadla: Utbytte av- og kvalitet hos finér fra stammekvistede- og ikke stammekvistede trær av furu, bjørk og osp.

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- **Supplement 7**: Vadla, K.: Verdiøkning og lønnsomhet ved stammekvisting. (En litteraturstudie).

- **Supplement 8**: Eid, T. og Hobbelstad, K.: AVVIRK-2000 - et Edb-program for langsiktige investerings-, avvirknings- og inntektsanalyser i skog.

- **Supplement 9:** Nersten, S., Eid, T., Heringstad, J.: Økonomiske tap på grunn av elgskader beregnet eiendomsvis.

- **Supplement 10:** Framstad, K.F. : Langsiktige miljøverknader av skogpolitiske verkemiddel - Simulert og optimert ved hjelp av modellen Gaya-JLP
*Long-run Environmental Effects of Forest Policy Measures - Simulated and Optimized by the Model Gaya-JLP.*

- **Supplement 11:** Nersten, S. og Hedegart, M.: Lønnsomheten av juletrehogst i hogstklasse II

- **Supplement 12:** Christiansen, E. and Kucera, B.: Resin pockets in Norway spruce wood are not caused by the bark beetle *Ips typographus. Kvaelommer i gran skyldes ikke angrep av granbarkbillen.*

- **Supplement 13:** Nilsen, P.: Skoggjødsling i Norge Et litteraturstudium over forsøksresultater fra fastmarksgjødsling

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### Rapport fra skogforskningen

- ✓ Manus skrives i Word 6.0 eller 7.0 - 12 pkt skrift (Times New Roman) med 1½ linjeavstand

- ✓ Forord
  Sammendrag
  Innledning
  Materiale og metode
  Resultat
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- ✓ I overskrifter brukes fet skrift (bold) og nummerering. Avsnitt markeres med dobbel linjeavstand.

- ✓ Alle tabeller og talloppsett skrives med tabellfunksjon i Word og plasseres bakerst i manuskriptet.
  Ikke bruk "tabulator"!

- ✓ Figurer gjøres helt ferdig - **NB! Velg gode størrelser i fontene (bokstavene) så de beholder sin lesbarhet når de skalleres / nedfotograferes.** Det er viktig at fontene er i samsvar med størrelsen på grafene/figurene!
  Figurene plasseres også bakerst i manuskriptet.

- ✓ Tenk lesbarhet i grafer - farger ser fint ut på skjermen, men er vanskelig lesbart i svart / hvit gjengivelse.

- ✓ Manuskriptet leveres ansvarlig avdelingssjef for godkjenning og påføres prosjektnummer.
  Deretter leveres det layout-ansvarlig for registrering.

- ✓ Redaksjonsrådet tar standpunkt til om manuskriptet er kvalifisert for utgivelse i serien og returnerer det til layout-ansvarlig for ombrekking/ klargjøring til trykking.

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