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Methodological options for quantifying changes in carbon pools in Swedish forests

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Abstract

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This report presents and discusses different methodological options for Sweden's reporting of emissions and removals of carbon dioxide by forests, under the United Nations' framework convention of climate change and its Kyoto Protocol. The main emphasis is on methods for estimating changes in different carbon pools, *i.e.* aboveground biomass, belowground biomass, dead wood, litter, and soil organic carbon. The report does not treat the issue of estimating areas of land-use transfers, such as afforestation, reforestation and deforestation.

The main methodological options considered in the report are repeated measurements in ground-based inventories, eddy-covariance measurements, remote sensing, modelling, and methods combining different data sources. The feasibility of using different methods is discussed, as well as the likely size of errors in applications. It is concluded that the existing field-based national inventory of forests forms a good basis for covering most of the reporting requirements. Remote sensing has a potential for further improving the precision of the estimates, although this method must be applied with care. Eddy-covariance measurements constitute an excellent basis for furthering the understanding of processes as well as for calibrating models. However, this method alone cannot be applied in fulfilment of the reporting requirements.

Keywords: carbon balance, carbon pools, climate change, eddy covariance, forest inventory, Kyoto Protocol, national forest inventory, remote sensing

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Executive summary

Background

Carbon uptake by terrestrial sinks is important in the global carbon balance, hence for limiting the concentration of CO_2 in the atmosphere. During recent decades, terrestrial ecosystems overall have served as a net sink for CO_2, in spite of net emissions to the atmosphere from land-use change, primarily in the tropics. Northern forests appear to account for a major portion of the terrestrial sink, partly as a result of improved forest management. Signatories to the Kyoto Protocol have had different views on the extent to which sinks should be used to meet the commitments. The agreement reached in Marrakesh represents a complex compromise, in which anthropogenic sinks can be used to a limited and carefully regulated extent. The agreement requires thorough monitoring and complex accounting for sinks. For Sweden, the sink in forests is of major potential importance.

The Parties to the Kyoto Protocol must report what areas have been used to meet commitments. Areas that have been included in one commitment period also need to be monitored, reported and accounted for in future commitment periods. The pools of carbon included are aboveground biomass, belowground biomass, litter, dead wood, and soil organic carbon.

The intergovernmental panel on climate change (IPCC) will elaborate methods for estimating, measuring, monitoring, and reporting changes in carbon stocks and anthropogenic greenhouse gas emissions by sources and removals by sinks resulting from land use, land-use change, and forestry activities, and prepare a report on good-practice guidance and uncertainty management. IPCC further will develop practicable methodologies to factor out direct human-induced changes from natural effects, and indirect human-induced effects, such as fertilisation, from deposition of nitrogen or increased CO_2 concentrations and the dynamic effects of age structure, resulting from activities and practice before 1990.

Objectives

The objective of this study was to evaluate Sweden's methodological options for quantifying changes in forest carbon pools on a national scale, to meet important reporting requirements of the Kyoto Protocol. Our treatment of different methodological options has been general, since there will be some freedom for individual countries to develop suitable national methods. In addition, since Sweden is a country with rather stable land-use, we have focussed on Article 3.4 in the Kyoto Protocol, rather than on the activities afforestation, reforestation, and deforestation under Article 3.3. Our ambition has thus been to study methods for quantifying changes in carbon pools in forest management areas. No attempt has been made to factor out natural and indirect human-induced effects or to estimate fluxes of greenhouse gases other than CO_2.

The approach of the study has been to compile and synthesise current knowledge and practices in Sweden, and to evaluate the advantages and disadvantages of various methods, in terms of requirements for and availability of data, the carbon pools included, uncertainty of the outcome, *etc.* The work has considered:

- Which forest definition to use;
- How to define different forest carbon pools in the Swedish case;
- What methods are available for national accounting, and the corresponding accuracy of estimates at national level;
- What methods are available for purposes of verification;
- Needs for research and improvement of ongoing national monitoring programmes to enhance the accuracy of estimates of carbon pool change.

Forests and forest monitoring in Sweden

Since the mid-20th Century, Sweden has undergone rather limited changes in land use. Currently, there are no major transfers between different land-use categories, although some land is being exploited for the expansion of infrastructure, and some agricultural land has been converted to forest. Moreover, there is currently an increase in the proportion of protected forests.

Most Swedish forests are managed. Exceptions are the tree-covered lands on mires and rock outcrops, which are forest according to FAO's definition, but not according to the Swedish

definition. By FAO's definition, the forested area in Sweden is *ca.* 27 Mha, while it is slightly less than 23 Mha by the national definition, which states that the potential productivity should exceed 1 m^3 of stemwood per hectare and year. We suggest that the Swedish definition of forest should be used for reporting forest management under the Kyoto Protocol (Article 3.4). It remains to be discussed whether or not protected forest areas should be included.

As regards the monitoring of forest carbon pools, Sweden is well provided with basic data, thanks to the annual surveys within the National Forest Inventory (NFI) and the Swedish Forest Soil Inventory (MI). These two monitoring programmes share the same plots, and thus data can easily be integrated. The NFI includes measurements of trees and dead wood. Within the MI, soil samples are taken and analysed for carbon. During the past decade, the NFI has comprised *ca.* 12–15 000 sample plots annually. About 50% of these were situated in forests. Soil samples within the MI are taken only on a small fraction of the NFI plots, owing to the high cost of sampling and analysis.

In contrast to the situation in most other countries, the two national forest surveys cover the entire country on an annual basis. Thus there is no great need for data extrapolation or interpolation when reporting for a specific period (some interpolation might be needed in the case of soil carbon, since soil data are collected in 10-year cycles). Moreover, permanent sample plots were established in the 1980s, and provide a good foundation for estimating change by repeated measurements.

In the past decade, the annual growth in managed forests in Sweden has been estimated at *ca.* 100 Mm^3, while fellings have been *ca.* 70–80 Mm^3. The annual sequestration of carbon in tree biomass has been 5–10 Mtonne C – a substantial proportion of the anthropogenic emissions of carbon. There are indications that some soil types are carbon sinks and that others are sources. Currently there is a lack of data for a full evaluation.

Methods for assessing changes in carbon pools

When considering methods for assessing carbon pools in Swedish forests, it is judged appropriate to make use of the data available from the NFI and the MI. Eddy-covariance measurements are being carried out at some places in Sweden; these sites can be very useful for purposes of verification. With the national inventory programmes, it is convenient to separate out the detailed measurements of the carbon pools at plot level, and then to study the problem of scaling-up for estimating total changes at national level.

At plot level, it is important to have clear definitions of the different carbon pools. Currently, aboveground biomass in trees can be calculated from detailed tree measurements on the sample plots of the NFI. On the basis of these measurements, and contrary to the practice in many other countries – where biomass expansion factors are applied – individual tree biomass functions are used. By means of these functions, tree biomass (aboveground) can be estimated for the fractions stem, branches, and needles and fine twigs. To convert from biomass to carbon, the factor 0.49 has been found to be rather stable in several studies. Currently, there are no procedures available for assessing the quantities of carbon in aboveground vegetation other than trees. However, the latter carbon pool is judged to be small in comparison to the tree-carbon pool.

Tree stumps and roots are considered to be part of the belowground biomass. Belowground biomass of trees is estimated at individual tree level by means of the same kind of functions as aboveground biomass. However, owing to the procedures used in developing these functions, belowground tree biomass is most probably slightly underestimated. No procedures are available for assessing the amount of belowground biomass other than living tree roots.

Dead wood is measured on a regular basis within the NFI. From these measurements, the volume of dead wood (with a diameter exceeding 10 cm) is obtained. Studies on the conversion of these volume estimates to carbon are in progress. Information is lacking about the finer fractions of dead wood (aboveground) and fine-root necromass.

As regards the quantity of litter in Swedish forests, information is available only from literature sources and from a limited number of research sites. Measurement of this carbon pool today is included neither in the NFI nor in the MI.

Samples for determining soil organic carbon are regularly taken within the MI. These comprise humus layer cores as well as samples of fine fractions of mineral soil at different depths. Among other analyses, the samples are analysed for carbon concentration. Since the inventory was originally not designed for quantifying the amounts of different compounds in the soil, some problems are encountered in the estimation of changes in soil carbon pools, although in principle, the survey is adequate for this purpose. The problems involve the principles for selecting humus cores, how to estimate the bulk density of the finer soil fractions (*i.e.* those analysed), lack of methods for assessing the amount of stones and boulders in the soil, and a lack of samples from land with a deep peat layer.

Turning from the measurements at plot level to the problem of scaling-up to estimates at national level, our main studies comprise the following methods as regards tree biomass (above- and belowground):

(1) Summation of growth minus removals over the years in a reporting period, according to the principles used for current Swedish reporting under the United Nations' framework convention on climate change (UNFCCC);

(2) Direct estimation of changes between the end and starting point of a reporting period, based on *(i)* temporary sample plots and *(ii)* permanent sample plots;

(3) Combined estimation, using (1) and (2) above. In this case, the two original estimates are weighted, in order to obtain a single estimate with a lower variance than for either of the two original estimates;

(4) Change estimation aided by satellite data. In this case, satellite data were used as auxiliary data to the field data, and principles of post-stratification were applied.

A summary of the results regarding biomass in trees is given in Table S1 below. It may be seen that many methods result in quite reasonable sizes of the standard error of estimated changes. The approach to 'sum annual growth-minus-cuttings', which is that used in Sweden for reporting under the UNFCCC, results in fair levels of standard error. Methods based on direct-change estimation – averaging results over three-year periods at the start and end of the reporting period – result in quite low standard errors when all of the NFI plots are permanent (the plan is that two-thirds will be permanent in the period 2008–2012). With temporary plots, the standard error of the change estimate would be high, and there would also be a serious risk of systematic error, owing to the difference being taken between two large pools. Slight variations in systematic error levels between the start and the end of the reporting period could in this case lead to a large systematic error in the change estimate. This risk is reduced if permanent plots are used, since a tree-list from the first inventory is available in the field at the time of the second inventory. To avoid systematic error, it is important to carry out the inventories very carefully, and to have separate control teams that repeat the measurements on a random sample of the original plots.

A challenge in the application of remote sensing in the Swedish case is to improve the estimates obtained when using field data alone. Since these have quite high accuracy, as described above, a

Table S1. *Summary of important features of the proposed methods for estimating changes in carbon pools in trees (above- and belowground carbon). The estimates are based on data from the Swedish NFI, and to some extent on assumptions*

Method	Standard error of an estimate of five years change, Mtonne C*	Risk of systematic error	Ready for application?
Summation of growth minus cuttings ('IPCC')	11–(25%)	Moderate	Yes
Direct change (temporary plots)	18–(40%)	High	Yes
Direct change (permanent plots)	6–(13%)	Moderate	Yes
Combined estimation ('IPCC' and permanent plots)	5–(11%)	Moderate	Almost
Remote sensing aided change estimation (post-stratification)	5–(10%)	Moderate	No

*The total change of carbon in biomass for a 5-year period is set to be 43 Mton C, which is based on the change during the period 1990–2000. Approximate 95% confidence intervals are obtained as 43 $\pm$ 1.96*Standard error.

major issue is to avoid introducing systematic errors when remote sensing data are incorporated into the estimation procedures. One important basis is that remote sensing alone seldom is very useful for biomass estimation and similar purposes. There is almost always a need for field reference data, so that the specific spectral signatures can be interpreted.

Theoretically, the combination of remote sensing and field data is not complicated. Many methods exist, *e.g.* stratification, multi-phase or multi-stage sampling. However, when working with satellite imagery to derive estimates for forests, we face problems concerned with the poor geometry of images, as well as poor precision of geo-referenced field plots. Mismatches between field plots and satellite pixels may lead to biased estimates. Moreover, classification procedures, *e.g.* to estimate the forest area from satellite imagery, seldom are unbiased. Care must therefore be taken when applying remote sensing to estimate changes in carbon pools. The principle identified in the present study was post-stratification, using medium-resolution imagery.

The above discussion is specific to countries in which the quality of field data is high. When field data are sparse or lacking, remote sensing plays an important part in providing at least some basic information. Specifically, deforestation can often be identified with rather high accuracy.

Leaving aside carbon in biomass and turning to carbon in soils, soil organic carbon is measured in the MI, and expressed as concentration and amount. The concentration is expressed, *e.g.*, as C as a percentage of the dry mass of fine soil (<2 mm). The values are obtained from laboratory analyses of collected soil samples. The C content per unit volume is calculated as the concentration multiplied by the bulk density of the fine soil. To obtain amounts of C per unit area, the density of C is multiplied by the soil volume down to a certain reference depth or within a certain layer. Layers are defined either as genetic soil horizons (such as O, E and B horizons) or as set depth layers (such as 0–5, 5–10 cm, *etc.*). For soils containing mineral matter larger than 2 mm, *i.e.* coarse material, the volume has to be reduced with the volume of coarse material.

An important question that affects estimation of carbon in mineral soils, is the assessment of coarse material. Another important question is to what depth the C stock should be estimated. We suggest that, for Sweden, 0.5 m would be suitable in mineral soils.

The MI-dataset on carbon in the humus layer was used to determine changes in this carbon pool, and the corresponding uncertainties. The results, based on 1000 plots, indicate that this carbon pool has increased in southern Sweden during recent decades, by *ca.* 0.5% annually. In central Sweden, an increase is indicated, although it is not significant. For northern Sweden, a slight decrease in humus-layer carbon is indicated.

As regards carbon in thick peat layers, data and methods are still lacking for making a thorough analysis. However, it is stressed that a large proportion of all soil carbon in Sweden occurs in peatland, and that peatland under forest may be a potential carbon source. Hence, it is important that methods to survey peat carbon stocks are developed.

Owing to the wide variation in carbon concentration in mineral soils, and to a lack of data on stones and boulders, no reliable estimate of changes in the carbon stock in mineral soils has so far been made. If we assume an annual increase of 0.5%, and apply the known variation in concentration, bulk density and stone and boulder content, the change over a five-year period for mineral soil material to a depth of 0.5 m would be 21 Mtonne C, with a standard error of 8.5 Mtonne C. The estimate is based on the present number of analysed samples (3500).

In Table S2, a summary of important features is given regarding the possibilities for using the MI to monitor soil-carbon pools.

A prominent feature, in general, of using sample-based approaches for reporting under the Kyoto Protocol, is that standard errors can be estimated on the basis of the data collected. Thus, in contrast to many other cases of reporting of removals or emissions of greenhouse gases, there is no need to make qualified guesses concerning the precision of the data.

The Kyoto Protocol states that the estimated changes in the carbon pools shall be verifiable, *i.e.* that the changes are true. The closest we can come to verifying an estimate of a change in carbon pools is probably to compare estimates made by independent methods, designed to measure the same thing. The eddy-covariance

Table S2. *Summary of important features of soil monitoring for estimating changes in soil carbon pools. In this case, NSFSV data (and assumptions) are used for direct change estimation between two time points*

Method	Standard error of an estimate of five years change, Mtonne C*	Risk of systematic error	Ready for application?
Humus layer, south Sweden	0.75–(25%)	High	Yes, with minor modifications
Mineral soils to 0.5 m	8.5–(40%)	High	Yes, with modifications

*Approximate 95% confidence limits are obtained as the estimate ± 1.96*Standard error.

method provides one possible means for such comparisons. With the eddy-covariance method, the net flux of CO_2 between the atmosphere and a forest can be measured with a half-hourly time resolution. The integral over time of the flux, plus the transport of dissolved organic carbon (DOC) out of the system, shall be comparable to the change in the pools estimated over the same time period, provided that all pools are measured. Since the DOC transport is known to be fairly small in most systems, it can be neglected in a first approximation; thus integrated fluxes measured by eddy-covariance can be directly compared with measured changes in the pools at the same site (if the export of carbon by harvesting is also included). At present, eddy-covariance measurements are in progress in several forests in Sweden, as part of different research projects, but there exist no sites for continuous long-term monitoring. We suggest that a number of eddy-covariance sites be set up in a monitoring and verification program. At these sites, the methods applied for monitoring carbon pools should be used in exactly the same way as for the purpose of Kyoto Protocol reporting. Results from these measurements can be used for testing and development of methods based on modelling.

Conclusions

It is concluded that the current Swedish national forest monitoring programmes, the NFI and the MI, provide adequate bases for reporting changes in forest carbon pools under the Kyoto Protocol (Article 3.4). Also, with the permanent plots of the NFI it is likely that most of the requirements regarding reporting of afforestation, reforestation and deforestation (Article 3.3) can be fulfilled. Several methodological options are available for reporting changes in tree biomass; to some extent these methods are complementary and can be used for verification or for combining estimates. The most promising approach is judged to be direct-change estimation, based on permanent plots (although this might imply a need to recalculate the annually reported figures once the reporting period has ended, which is allowed). This principle might be combined with post-stratification, using remote sensing auxiliary data, to improve the estimates somewhat more. However, the routines for check assessments within the NFI call for improvement.

The existing soil survey (MI) can be used, after some modification, to monitor changes in the soil carbon stock in mineral soils to a depth of 0.5 m. To include peatlands, new methods would be needed. An alternative to basing the reporting on repeated measurement of soils, would be to develop soil-carbon models. Since annual reporting to the Kyoto Protocol is required, such models might assist in deriving extrapolated values from the ten-year estimates of change obtained from the MI.

Dead wood is covered by the NFI, and once new functions for conversion from volume to carbon have been established, it will be possible to include this pool in the reporting.

No measurements are currently made to estimate the carbon pool in litter. One possibility could be to expand the humus sampling of the MI to include litter. In particular, new methods would be required to survey fine woody debris (< 10 cm), which is included in this pool.

Eddy-covariance measurements provide a valuable means of verification, and for further improving our understanding of greenhouse gas fluxes (including model calibration), for example regarding the temporal variability of fluxes. However, currently there is no possibility of using these measurements to improve estimates of

carbon pool changes at the national level. The reason is that the network of measurement stations is very sparse, and that there are no methods available for scaling-up from site level to the national level.

Overall, the precision of the estimates of carbon change obtained in this synthetic study are reasonable by comparison with what has been reported in other studies.

Finally, although this report focusses on methods for estimating change on the basis of current and past data, it is important to point out that scenario analyses are important tools for enabling proactive engagement with issues related to carbon sources and sinks in forests. Whether or not the forest ecosystem is a source or a sink is a result of complex ecological and socio-economic interactions, where tree growth and soil processes act on the one side, and the processes that lead to management decisions, such as harvesting, act on the other side. In Sweden, joint SLU-MISTRA funded projects (Heureka, LUSTRA, and others) aim at developing tools that can be applied to this type of integrated analysis.

Introduction

Carbon uptake in terrestrial sinks is an important process in the global carbon balance, hence in limiting the concentration of CO_2 in the atmosphere. The agreement reached in Marrakesh (UNFCCC, 2001), where some remaining issues in the Kyoto Protocol were agreed upon, means that anthropogenic sinks can be used to a limited and carefully regulated extent by signatory parties to meet commitments. The agreement requires thorough monitoring and accounting of sinks and sources.

The intergovernmental panel on climate change (IPCC) will elaborate methods to estimate, measure, monitor, and report changes in carbon stocks and anthropogenic greenhouse gas emissions by sources and removals by sinks resulting from land use, land-use change and forestry activities. Furthermore, IPCC is preparing a report on good-practice guidance and uncertainty management relating to the measurement, estimation assessment of uncertainties, monitoring and reporting of net carbon-stock changes and anthropogenic greenhouse gas emissions by sources and removals by sinks in the land use, land-use change and forestry sector, by the ninth conference (COP 9; held in 2003) of the Parties to the United Nations' framework convention on climate change (UNFCCC). Moreover, IPCC will develop practicable methodologies to factor out direct anthropogenic changes from natural effects and indirect anthropogenic effects, such as fertilisation, from atmospheric deposition of nitrogen or increased CO_2 concentration, and the dynamic effects of age structure resulting from activities and practices before 1990, by COP 10 in 2004.

Various methods can be used to account for changes in carbon stocks of different pools in forest ecosystems. All of those methods have advantages and disadvantages; the usefulness of the results depends strongly on the objectives, the database, and the geographical scale. In general, it is easy to obtain detailed information on greenhouse-gas balances at a local level. The big challenge, however, is to develop and apply methods, which produce estimates on the national scale, with acceptable accuracy in a cost-efficient way.

The aim of this report was to evaluate Sweden's methodological options for quantifying changes in forest carbon pools on a national scale, to meet the requirements in the Kyoto Protocol. Our treatment of different methodological options has been quite general, since there will be some freedom for individual countries to develop suitable national methods for reporting. Also, since Sweden is a country with rather stable land-use, our focus has been on Article 3.4 in the Kyoto Protocol, rather than on the activities afforestation, reforestation, and deforestation under Article 3.3. Thus, our focus has been on methods for quantifying changes in carbon pools in forest-management areas. No attempt has been made to factor out natural and indirect human-induced effects, which is likely to be required in future reporting. Nor have we made any attempt to study methods for estimating fluxes of greenhouse gases other than CO_2.

The approach of the study has been to compile and synthesise current knowledge and practices in Sweden, and to evaluate the advantages and disadvantages of various methods in terms of the requirement for and availability of data, included carbon pools, uncertainty of the outcome, *etc.* The work has considered:

- Which forest definition to use;
- How to define different forest carbon pools in the Swedish case;
- What methods are available for national accounting, and the corresponding accuracy of estimates at national level;
- What methods are available for purposes of verification;
- Needs for research and improvement of on-going national monitoring programmes to enhance the accuracy of estimates of carbon pool change.

International agreements

The issue

Carbon uptake in terrestrial sinks is important in the global carbon balance, hence in limiting the concentration of CO_2 in the atmosphere. During the past two decades, terrestrial ecosystems have served as a small net sink for CO_2. This terrestrial uptake has occurred in spite of net emissions into the atmosphere from land-use change, primarily in the tropics, with 1.6 ± 0.8 Gtonne C yr^{-1} (1 Gtonne $= 10^9$ tonnes). The uptake in terrestrial sinks was estimated at 2.3 ± 1.3 Gtonne C yr^{-1} (IPCC, 2000a). The magnitude of the net carbon flows (in Gtonne C yr^{-1}) between the atmosphere, the oceans and the terrestrial ecosystems, averaged for 1989–1998, is shown in Fig. 1 (IPCC, 2000a).

The uptake in terrestrial ecosystems has been calculated as a residual term from emissions from fossil fuels and cement production, emissions from land-use change, storage in the atmosphere and uptake in the oceans, and is therefore often referred to as the residual terrestrial sink.

The terrestrial sink mainly is a result of *(i)* growth of forest and other vegetation and the accumulation of organic matter in soils, including the effects of improved land and forest management, particularly in temperate and boreal forests of the northern hemisphere; *(ii)* indirect effects of human activities, such as atmospheric CO_2 fertilisation and nutrient deposition; and *(iii)* changing climate, derived from both natural and anthropogenic causes. It is at present not possible to determine the relative importance of these different processes (IPCC, 2000a).

Satellite data, combined with forest inventory data, have indicated that forests in Europe, Russia and North America stored nearly 700 Mtonnes C yr^{-1} in forest biomass during the 1980s and 1990s (Myneni, Dong, Tucker, Kaufmann, Kauppi, Liski, Zhou, Alexeyev & Hughes, 2001). Northern forests may account for a major portion of the residual terrestrial sink if other pools, such as the soils, are also included. The study indicated that most northern forests – with the exception of Canada's boreal forests – were storing carbon. Russia accounted for almost

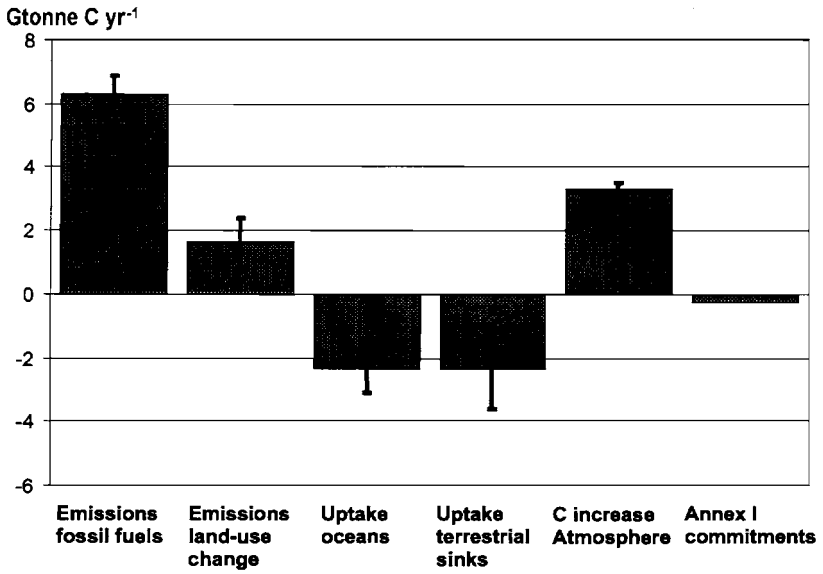

Fig. 1. Annual carbon flows for the period 1989–1998 compared with Annex I commitments in the Kyoto Protocol of 250 Mtonne C yr^{-1} for the period 2008–2012. (The vertical bars in the diagram show the uncertainty at an estimated 90% confidence interval).

40% – or 280 Mtonne C yr^{-1} – of this forest carbon sink, declining harvests being the most probable cause. In other areas, such as the Nordic countries and the USA, current management practices are probable reasons for the net forest carbon sink. Forest fires and infestations are possible reasons for the net carbon loss from boreal forests in Canada (Myneni *et al.*, 2001). This study reports very high figures for the case of Sweden, as compared to previous studies (*e.g.* Eriksson, 1991). Recently, carbon sequestration in European forests was estimated at 880 Mtonnes C yr^{-1}, from inventory data (Liski, Palosuo & Sievänen, 2003a). Janssens *et al.* (2003) also reports large removals by the terrestrial biosphere.

Karjalainen, Pussinen, Liski, Nabuurs, Eggers, Lapveteläinen & Kaipainen (2003) predict that the European forests will continue to act as sinks of greenhouse gases.

In addition to the sources and sinks shown in Figure 1, also the sum of the commitments of the 'Annex I countries' of the Kyoto Protocol (countries that have quantified emission reduction or limitation commitments are listed in Annex I to the Kyoto Protocol, basically the industrialised countries, including Russia and other Eastern European countries with economies in transition) is shown (250 Mtonne C yr^{-1}). The agreement in Kyoto concerning commitment levels was reached on the understanding that there should be only a limited inclusion of sinks during the first commitment period, and that the focus should be on the reduction of emissions. An unlimited inclusion of sinks could result in claims corresponding to a major part of the residual terrestrial uptake.

The terrestrial carbon uptake at present plays an important part in the global carbon cycle. The net uptake, resulting from improved forest management, may significantly increase the amount of carbon in a forest over a rotation period as compared with that of natural forests. The crux of the matter is, however, the changes in a longer perspective. The net loss of carbon from terrestrial ecosystems since pre-industrial times is probably considerably less than 100 Gtonne C, and the total net gains through deliberate human efforts to increase uptake by terrestrial ecosystems will probably be of the same magnitude during the 21st Century. The foreseen accumulated emissions resulting from the of burning of fossil fuels, according to IPCC's scenarios during this period, are, however, predicted to be an order of magnitude larger, 800–1500 Gtonne C, and emphasis must therefore be placed on the reduction of emissions, in order to achieve the stabilisation targets for atmospheric CO_2.

Decisions on sinks in the Kyoto Protocol and the Marrakesh Accords

The Parties to the Kyoto Protocol have had different views on the extent to which sinks should be used in the first budget period, 2008–2012, to meet commitments. The agreement reached in Marrakesh (UNFCCC, 2001), which makes the Kyoto Protocol operational, represents a fairly complex compromise, in which anthropogenic sinks can be used to a limited and carefully regulated extent by Parties to meet their commitments. The agreement requires thorough monitoring and complex accounting of sinks in Annex I countries.

The issue of carbon sinks received attention only at a late stage in the negotiations of a protocol under the Climate Convention, and diverging views among Parties led to a fairly complicated compromise that was agreed in the Kyoto Protocol at the third conference of the Parties to UNFCCC (COP 3), held in Kyoto in 1997. It was agreed that sinks could be used, but only to a limited extent, to meet commitments during the first period.

Article 3.3 in the Protocol states that the net changes in carbon stock from the activities afforestation, reforestation and deforestation since 1990 shall be used to meet commitments. Article 3.4 states that Parties may choose to apply additional human-induced activities to be agreed within the agricultural and forestry sectors. Further negotiations were needed to make these clauses operational, *e.g.* the definition of forest, the definition of activities under Article 3.3, and on the modalities, rules and guidelines for inclusion of new activities under Article 3.4. Agreement in principle was reached at COP 6 in Bonn in June 2001, which was made operational in the Marrakesh Accords, agreed in Marrakesh at COP 7 in November 2001 (UNFCCC, 2001). The Marrakesh Accords also set the rules for accounting and reporting of the carbon sinks.

The activities in Article 3.3 – afforestation, reforestation and deforestation – represent a change in land use, and Parties have to account for the changes in carbon stock due to these activities. Reforestation activities for the first commitment period are limited to reforestation occurring on lands that were not forested on 31 December 1989. Reforestation activities in the normal harvesting–reforestation cycle are thus not included under Article 3.3. This means that, for most Parties, including Sweden, Article 3.3 activities will only have a limited quantitative impact. Article 3.3 will have a large impact only in countries with plantation forestry, such as New Zealand, and in countries where extensive afforestation programmes occur. Annex I Parties would, of course, also be responsible for decreases in carbon stock due to deforestation.

Under Article 3.4, the broadly defined activities forest management, cropland management, grassland management, and the more narrowly defined revegetation, can be used by Parties to meet commitments during the first commitment period. The three latter activities all concern agriculture, and carbon stock changes are compared on an annual basis with the corresponding change in carbon stock during the base year (1990). Thus credit is given to improvements relative to the base year ('net–net accounting'). The agricultural activities will not be further analysed in the present report.

Forest management is broadly defined as 'a system of practices for stewardship and use of forest land aimed at fulfilling relevant ecological (including biological diversity), economic and social functions of the forest in a sustainable manner'. The definition would basically include all forest areas under management in Sweden. For the forest definition, Parties have to select a forest definition from ranges given for tree crown cover, tree height and minimum land area, and keep that definition for the first commitment period. Parties need to justify the choice and show that its values are consistent with information that has historically been given to FAO and other international bodies or, if they differ, explain why and how the definition was chosen.

Accounting for forest management is based on the actual carbon uptake, not the uptake relative to the base year as for the agricultural activities. It was decided in the Marrakesh Accords that accounting of sinks should be guided by a number of principles, *e.g.* accounting and crediting of carbon uptake should exclude effects of indirect effects such as fertilisation from atmospheric deposition of nitrogen or increased CO_2 concentration and the dynamic effects of age structure resulting from activities and practices before 1990.

Such adjustments should be based on scientific data, but for the first commitment period it was agreed to use a pragmatic, simplified basis for accounting: Numbers were calculated for each Annex I Party, based on an 85% discount factor and a 3% (of 1990 greenhouse gas emissions) cap on forest management, using a combination of data provided by Parties and by the FAO. Some Parties were given higher numbers based on 'national circumstances'. This applied primarily to Canada and Japan, but also to Russia. The amount for Sweden is 0.58 Mtonne C yr^{-1}. These numbers are in addition to a compensation for a net debit that may have been incurred under Article 3.3.

Parties shall report the areas that have been used to meet commitments. It is stated in the Decision 11/CP7, Land-use, land-use change and forestry (LULUCF decision) that 'National inventory systems ... shall ensure that areas of land subject to land use, land-use change and forestry activities under Article 3 paragraphs 3 and 4 are identifiable, and information about these areas should be provided by each Party included in Annex I in their national inventories...'. The accounting provision states that information should be given on 'the geographical location of the boundaries of the areas that encompasses *(i)* units of land subject to activities under Article 3, paragraph 3; *(ii)* units of lands subject to activities under Article 3, paragraph 3, which would otherwise be included in lands subject to elected activities under Article 3, Paragraph 4...; and land subject to elected activities under Article 3, Paragraph 4.'

The text indicates that the exact geographical location of individual units of land with afforestation, reforestation or deforestation does not need to be identified, but statistical sampling within normal forest inventories can be employed. The key part of the decision is 'geographical location of the boundaries of the areas that encompasses' ... 'units of lands subject to activities under Article 3, paragraph 3'.

Areas that have been accounted for and used to meet commitments during the first commitment period need to be monitored, reported and accounted for also in future commitment periods. The carbon pools included are aboveground biomass, belowground biomass, litter, dead wood and soil organic carbon. A Party may choose not to account for a given pool in a commitment period, if transparent and verifiable information is provided that the pool is not a source.

Forests and forest monitoring in Sweden

Definition of forest

The Swedish definition of forest implies that all areas – with no other major land use – where the potential production of wood is at least 1 m^3 ha^{-1} yr^{-1}, are considered to be forests. The minimum patch size for an area to be a forest is 0.25 ha. This is different from the definition used by FAO in their last world forest resource assessment, FRA2000 (Anon., 2001). According to FAO, all land areas not predominantly used for other purposes, where the crown cover is at least 10% and the tree height is at least 5 m, are forests. Areas temporarily below these limits, *e.g.* owing to clearfelling or fire, are also included. The minimum patch size is 0.5 ha.

In comparison with the FAO definition, the Swedish definition is narrower, *i.e.* fewer land areas are included. The forest area in Sweden is *ca.* 23 Mha according to the national definition (Anon., 2003), while the corresponding area according to FAO's definition is *ca.* 27 Mha. Areas added are mires and rock outcrops sparsely covered with trees, and alpine birch areas in the north-western part of the country.

For Kyoto Protocol reporting in general, each country's definition of forest must be selected within certain limits. The suggestion is that Sweden should choose a forest definition identical to FAO's definition. However, for reporting under Article 3.4, an additional definition of 'forest management areas' must be given. Our proposal is that the Swedish definition of forest be used in this case. Whether or not productive forest areas within reserves should be reported under Article 3.4, remains an open question.

Forest and soil conditions in Sweden

The total stem volume in Swedish forests is *ca.* 3000 Mm^3 (average 126 m^3 ha^{-1}); 39% is Scots pine, 42% Norway spruce, and 11% birch. The remaining 5% is composed of other deciduous tree species such as aspen, oak and beech. The mean site capacity is *ca.* 5.3 m^3 ha yr^{-1}, while the annual increment is 104 Mm^3 (Anon, 2003). This is almost 70% higher than the growth at the beginning of the 20th Century. The areas of forest reserves are currently increasing, owing to an increasing ambition to preserve biological diversity.

The total carbon pool in tree biomass is *ca.* 1000 Mtonne C, which corresponds to an average of 45 tonne C ha^{-1} forest. There is a strong gradient, from an average of about 20 tonne C ha^{-1} in the north to more than 60 tonne ha^{-1} in the south. The gross annual accumulation in biomass is estimated at about 35 Mtonne C yr^{-1}. The annual net accumulation of C in biomass is lower than the gross accumulation, owing to harvest removals and natural mortality. During the past decade, the annual net increase has varied between 5–10 Mtonne C yr^{-1}, depending mainly on the harvesting level.

Of the forest land, 66% can be classified as well-drained upland soils with a groundwater table below 1 m, and 34% as poorly drained, with a groundwater table closer than 1 m to the soil surface. The poorly-drained soils are often peatlands, with an organic horizon (histic) thicker than 30 cm. The forest area with peat amounts to about 1.8 Mha, which is 8% of the total forest area.

In the upland (drained) soils of Sweden, the soil organic carbon generally occurs in an upper humus-rich layer (O- or A horizon) and in the mineral soil down to at least 2 m (E, B and C horizons). Approximately 25% of the total soil organic C stock occurs in the O or A horizon, 50% in the mineral soil to 1 m and the remaining 25% from 1 to 2 m. Additional small amounts may occur below 2 m (LUSTRA, 2002). The knowledge of the carbon distribution pattern below 2 m is very poor.

The average soil carbon pool for upland forest soils in Sweden is approximately 8.5 kg C m^{-2} to a reference depth of 1 m (LUSTRA, 2002).

Under wet or moist conditions, *i.e.* a high groundwater table (0–1 m), decomposition of

organic matter is hampered and soil-carbon pools therefore have accumulated to an extent that largely exceeds the soil-carbon pool of the drained upland soils. The carbon can accumulate as a thick humus layer as well as in the underlying mineral soil. Peat layers may develop. In Sweden, peat is commonly classified as an organic horizon that is at least 30 cm thick.

The size of the carbon pool on wet and moist forest land is less well known than the carbon pools under drained upland conditions, but may amount on average from *ca.* 15–50 kg C m^{-2}. Thus a substantial part of the total carbon pool in forest soils in Sweden occurs at sites with a high groundwater table.

In assessing the impact of wet or moist land on greenhouse gases, carbon losses as methane (CH_4) should also be considered (*e.g.* Olsson, Lundin & Lode, 2002). A wet soil may emit a small quantity of methane, together with a net accumulation of carbon. Because methane has a much higher global warming potential (GWP) than CO_2 (*ca.* 21:1), the net effect might be an emission of greenhouse gases, in terms of GWP, despite a net accumulation of carbon.

Large-scale inventories of Swedish forests

Sweden is well equipped with national forest monitoring programs that can be used for assessing changes in forest carbon pools. During the past decade, the National Forest Inventory (NFI) annually has comprised about 12–15 000 sample plots, about half of these in forest. Carbon-related measurements within the NFI involve measurement of individual trees on sample plots, and measurement of coarse woody debris. The inventory is composed of a combination of permanent and temporary plots, the permanent plots being revisited at 5–10 year intervals. On these plots, the Swedish Forest Soil Inventory (MI) is conducted. This inventory is closely related to the NFI, and the two inventories share the same infrastructure. Among other things, the MI collects soil samples that are analysed for carbon. In addition to these two programmes, a National Survey of Landscapes in Sweden (NILS) started in 2003. This inventory covers all parts of Sweden, and – from the point of view of carbon monitoring – contributes data from some types of area that are currently not covered within the NFI and the MI, *e.g.* mountain birch forests. The NFI and the MI are described in more detail below.

The Swedish NFI started in 1923 as a county-wise survey that focussed mainly on forests from the point of view of timber production (Segebaden, 1998). Over the years, there have been many changes, both with regard to the scope and the design of the inventory. Especially during recent decades, many new variables have been introduced, as a result of new requirements. From 1953 onwards, the inventory has been based on measurements on sample plots allocated in clusters ('tracts'). In 1983, permanent sample plots were introduced in addition to temporary plots, allowing for more efficient estimation of changes. At present, the time-interval between re-measurements of permanent plots is about ten years; but in future this will be adjusted to five years for basic variables, including these needed to derive the carbon pools in trees.

From the point of view of carbon monitoring, the careful measurement of trees on plots with a radius of 7 or 10 m is important. From these measurements, the biomass of trees, hence also the quantity of carbon, can be assessed. In addition, specific measurements of growth, mortality, and fellings are made, as well as measurements of dead wood. This involves both standing and fallen dead trees.

In total, the MI comprises about 20 000 permanent plots, corresponding to the permanent plots (in forests) of the NFI. The first soil assessments on these plots were made in the period 1983–1987. Currently, the re-measurement interval is ten years for the soil assessments. In general, soil pits and soil and site descriptions are made on one to two plots per tract. On each plot, general site properties are described, such as vegetation type and occurrences of different species, type of soil parent material, and hydrological conditions. Specific variables are, *e.g.*, thickness of the humus layer, humus form, soil unit (Swedish system and according to the FAO-Unesco legend) and thickness of the E horizon. The occurrence of stones and boulders has been included from 2003 onwards.

The soil is sampled in O or A, E-, B-, B/C or C-horizons. For the O horizon, *ca.* 1–5 samples are taken per plot (depending on O horizon thickness). The A to C horizons are sampled in one pit per plot. Samples are stored in a soil bank and analysed, *e.g.* for *p*H and concentration of

C, N, aluminium, and exchangeable base cations. Bulk density is not determined.

The MI measures the carbon pool in the humus layer to a depth of 30 cm. Thus, peat layers below 30 cm are not sampled. However, from 2003 onwards, the total depth of peatlands is measured. Peat thickness and bulk density have also been measured by the Swedish Geological Survey, although not in a way that permits the evaluation of changes.

Until now, the NFI and the MI have been entirely field-based. In the plans for the period starting in 2003, estimates aided by remote sensing data will be introduced, mainly for enhancing the precision of estimates within smaller areas.

Methods for assessing changes in forest carbon pools

The underlying problem

An important part of the implementation of the Kyoto Protocol is to develop methods, which are well suited for estimating changes in forest carbon pools as prescribed by the agreements. The pools included are aboveground biomass, belowground biomass, litter, dead wood and soil organic carbon. When discussing different methods, it is important to note that it is not the change in the pools as such that is of interest from the point of view of climate, but rather the net change in the amount of CO_2 in the atmosphere. The change in the amount of CO_2 in the atmosphere is mainly determined by the flux of gaseous CO_2 into and out of the atmosphere at the interface between the earth's surface and the atmosphere (omitting the direct effect of airborne vehicles' CO_2 emissions). This flux is the result of different processes in the ecosystem, with photosynthesis, autotrophic and heterotrophic respiration as the dominant ones (Fig. 2). The net flux between the atmosphere and the ecosystem is called 'Net Ecosystem Exchange' (NEE), and consists of the sum of the fluxes originating from the different processes. Typically, the flux is directed downwards from the atmosphere to the ecosystem during daytime, when photosynthesis is larger than respiration, and in the opposite

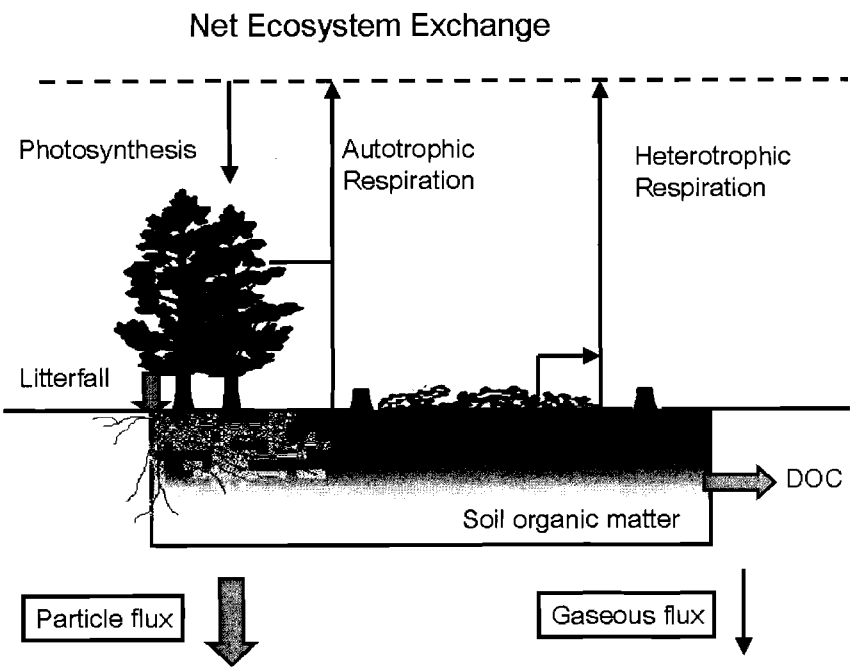

Fig. 2. Schematic figure showing the main compartments and fluxes in a forest ecosystem. (Removals due to harvesting is an additional component which is not included in this overview.)

direction during the night, when only respiration occurs.

The term 'carbon balance' often refers to a certain longer time period, and if the integrated flux is downwards during this period, the ecosystem is a sink, and if the net flux is directed upwards, the system is a source. The NEE over a certain time period can in principle be measured in two ways: Either by measuring the fluxes and integrating them over time, or by measuring the change in the amount of carbon in all pools over the same time period. The latter has been chosen as a primary method for reporting according to the international agreements. An assumption underlying both methods is that the transport of dissolved organic carbon from the system is small, which may not always be the case (Siemens & Janssens, 2003).

Below, different techniques for estimating carbon balances are presented. First, methods based on estimation of changes in the different carbon pools are treated. This is followed by a discussion of flux measurements.

Measurement of changes in carbon pools

Assessment of changes in different forest carbon pools is the main methodological option identified in the international agreements (*e.g.* IPCC, 1997). As pointed out above, this is an indirect way of estimating the NEE. Changes in carbon pools can be measured by many different methods. Default methods proposed by IPCC generally are based on the separate assessment of emission factors and of areas, to which the emission factors apply. When, as in the case of Sweden, detailed data are available from the national forest inventories, there is no need to make use of IPCC's default data. Nor is it judged to be a relevant approach to elaborate national default values, which is another methodological option. Instead, a relevant approach is to make use of existing data from the NFI and the MI. Even with these or similar inventories, there remain many methodological options for assessing changes in the different carbon pools. However, regardless of method, the procedures generally comprise two steps: The first step involves detailed measurement of the pools at a local scale (*e.g.* of the trees on a sample plot). The second step involves scaling-up from the measurements made at the local level to estimates for the entire country.

Below, the five forest carbon pools are first described in more detail, and the measurements related to these pools are outlined. Then the general aspects of sampling-based methods for the scaling-up procedure are given.

In all cases, our only ambition has been to quantify changes in the forest carbon pools between two points in time. Thus we do not consider the problem of what happens to harvested wood products, *etc.*, once trees have been felled and extracted from the forest.

The pools

According to the Kyoto Protocol, the following carbon pools are identified. Assessment of changes should be made for each one, unless it can be established that the pool is not a source of carbon.

- Aboveground biomass
- Belowground biomass
- Litter
- Dead wood
- Soil organic carbon

The boundaries between the different pools are not always distinct. Below, we discuss which carbon components might be part of each pool, given current inventory practices in Sweden. The methods for measuring the pools are described, as well as potential need for improvement, as regards both definitions and measurement procedures.

Aboveground biomass This pool consists of tree biomass – above stump height – including stem, bark, branches, and needles/twigs. For the majority of Swedish tree species, the biomass (dry matter) can be estimated by means of individual-tree regression functions derived by Marklund (1987, 1988) and Petersson (1999). These functions separate the biomass into stem, bark, branches, and needles/leaves. An alternative would be to use biomass expansion factors (*e.g.* IPCC, 1997), which convert volume estimates to biomass estimates. Since such factors distinguish in a few cases only between trees of different age (*e.g.* Lehtonen, Mäkipää, Heikkinen, Sievänen & Liski, 2004) or trees in stands of different

density, the application of individual-tree biomass functions is judged to be superior to the use of biomass expansion factors, provided that data with adequate resolution are available. The data needed for using the functions are typically tree species, diameter, height, and geographical location. Different kinds of function exist, so that the same principle can be applied under different conditions.

Currently, the biomass in stumps is not considered to be part of this carbon pool, owing to the way in which the biomass functions are constructed. Moreover, current knowledge about the amount of carbon in aboveground vegetation other than trees is poor (*e.g.* Anon., 2000). Work to include this part of the aboveground pool could be justified, although the quantity involved is probably limited in comparison to the amount of carbon in trees. However, increasing density of the tree stand generally implies decreasing density of other vegetation; thus the exclusion of carbon contained in other vegetation might lead to the overestimation of the build-up of carbon in aboveground biomass when the density of the tree stand increases (and *vice versa*). This would justify the inclusion of carbon in other vegetation as well.

Belowground biomass This pool consists of biomass in living roots of trees, and of the biomass in tree stems below 1% height (stump height). As for the aboveground biomass, individual-tree regression functions for estimating the belowground tree biomass exist (Marklund (1987, 1988) and Petersson (1999)). The functions are applied by means of the same kind of basic data as for aboveground biomass. However, functions are currently available for Scots pine and Norway spruce only, which is a limitation. In many applications, these functions are also used for other Swedish tree species.

Owing to the procedures used for uprooting trees when the material for these functions was collected (Marklund, 1987), it is likely that the functions slightly underestimate belowground tree biomass.

Living roots of vegetation other than trees are currently not assessed as part of this pool, according to Swedish practice. Potentially, new procedures for soil sampling could be introduced, in which these roots could be sampled and measured separately.

Litter This pool generally refers to the dead organic debris that is supplied to the soil by litterfall, and as root litter. It can be separated into three different pools – fine debris such as leaves and needles, fine woody debris (< 10 cm diameter) aboveground, and root litter (root necromass). Litter is generally distinguished from soil organic matter by its low degree of decomposition or fragmentation. Litter at least occasionally accumulates on top of the soil, but litter may also include newly dead roots in the soil. Currently, the litter pool is rather poorly defined in Swedish large-scale monitoring practices. No measurements are carried out in the national inventories. It is often difficult to separate the lower parts of the litter layer from the upper parts of the humus layer (which are part of the soil organic carbon).

Dead wood The definition of dead wood used in the NFI is that the diameter should be larger than 10 cm, and – in the case of highly decomposed material – the stem form should be clearly distinguishable. Both standing and fallen dead trees are included. On intact trees, measurements of breast height diameter and height are made, and ordinary functions are applied to derive the volume (*e.g.* Näslund, 1947). When trees are not intact, sectioning techniques are applied (*e.g.* Loetsch & Haller, 1973). To convert from volume to amount of carbon, there is a need for conversion factors or functions. Research is in progress to establish such results.

Soil organic carbon The soil organic carbon pool is referred to as a mixture of dead plant and animal residues in various stages of decomposition, of substances synthesised microbially or chemically from the breakdown products, and of the remains of soil microorganisms in a more or less decomposed state. Soil organic carbon occurs in the form of a distinct organic layer (*ca.* 35–45% C) on top of the mineral soil (O horizon) or blended with mineral matter (A or B horizons). Soil organic carbon is usually determined for the size fraction < 2 mm. Under wet conditions, soil organic matter may occur in the form of peat layers.

The soil organic pool is difficult to delimit both upwards and downwards. Upwards, the boundary generally is set between the humus layer and the litter layer (a boundary that is not

always clearly distinguishable). Downwards, a limit must be set from a practical point of view.

The decision as to the lower depth limit for quantification of the carbon pool is crucial. The turnover rate of soil carbon is much higher in the upper than in the lower soil horizons. The upper part of the soil will therefore respond in the short term more to forest management than will the lower parts. On the other hand, the biggest carbon stocks are found in the underlying mineral soil, *i.e.* in the B horizon. The deeper is the soil reference depth considered, the more accurate is the assessment of the total soil carbon stock or of stock change, but at the same time, the more expensive a high-precision inventory will be. Deep sampling enables a fair comparison between countries and sites with different vertical distribution of soil carbon. This can be illustrated by, on the one hand, thin soils (Leptosols), in which all organic carbon occurs in a 10–30 cm thick soil layer, and on the other hand, deep soils in which the same amount of soil carbon is distributed down to 1–2 m. Arguments against deep sampling are the high sampling costs, and statistical uncertainties depending on the measurement of low concentrations, including small changes in concentration.

For Sweden, sampling of the humus layer and of the mineral soil to 0.5 m is suggested to be a good compromise. This depth would cover more than half of the total carbon pool, and the difficult sampling of the compact underlying and compact basement till is avoided.

Carbon in the humus layer can be directly sampled by means of humus cores, which are weighed and analysed for their carbon content. A problem is that large roots and stumps may prevent cores from being taken at certain points. Quantification of carbon in the mineral soil is generally more difficult, owing to the presence of boulders and stones in this layer. Thus to measure the carbon quantity in this soil layer, a two-step procedure must be followed. First, soil core samples are taken at certain depths for analysis of carbon content and bulk density. Secondly, the amount of stones and boulders must be determined.

Determination of carbon in peat soils gives rise to specific problems. It is not possible to delimit an upper part of the peat layer, *e.g.* 50 cm. Instead, the carbon in the whole peat layer must be determined.

Sampling methods

The NFI and the MI are based on sampling. A powerful feature of sampling is that is possible to estimate the uncertainty of the estimates, based on the data obtained (*e.g.* Thompson, 1992). This contrasts with many of the methods used in connexion with the reporting of greenhouse gas removals or emissions under other conditions. Some basic aspects and requirements concerning sampling methods are summarised below.

Objectivity

Two conceptually different kinds of sampling practice exist: one where measurement locations are selected subjectively – subjective sampling – and one where the locations are determined following randomisation procedures – random sampling (*cf.* Thompson, 1992; Schreuder, Gregoire & Wood, 1993). Although subjective procedures can be appropriate in some cases, it should be stressed that there is a serious risk of major systematic error if such procedures are chosen. When many different stakeholders are involved, and when there is a need for results in which all can feel confidence, objective sampling is generally the solution (*cf.* Thompson, 1992). Moreover, most of the existing sampling theory is applicable only when random sampling is applied. Subjective sampling may still be appropriate when very limited funds are available, in order to obtain rough estimates of likely ranges of emissions or removals.

Well-defined population

Where carbon in trees is concerned, it is clear that the trees make up the target population. As regards carbon in soils, the population can no longer be enumerated in distinct units, but a quadrat defined by area must be used for the sampling procedure. In both cases, however, the important decision to be made is what areas should be defined as forest (if the results should apply only to forests). Many different definitions exist, and it is important that a clear, operational definition is used. Not least, it is important that the definition is capable of practical application in fieldwork, to avoid a time-lag as regards the type of areas considered as forest and non-forest. Such time-lags would lead to biased estimates of changes in carbon pools.

Strict measurement procedures

Just as it is important to have strict definitions, it is also very important to have strict procedures for performance of the measurements. As regards the carbon pools in trees, generally these are derived from basic measurements on trees, *e.g.* of diameter at breast height (1.3 m). From the humus layer and from the mineral soil, samples are taken, and the quantity or concentration of carbon within these is determined in the laboratory. It this context, it is very important that the same definition, *e.g.* of the boundary between the humus layer and the mineral soil, is maintained on different occasions. Moreover, it is important that the procedures used in the laboratory at different points in time are comparable.

Cost-efficiency

Many different sampling procedures exist. Most of them have arisen because, in a particular situation, they have resulted in estimates with better precision than 'traditional' procedures. Thus there is a vast number of potential sampling procedures to choose between for carbon monitoring (*e.g.* Schreuder *et al.*, 1993). However, carbon monitoring often is only one component of a multipurpose forest inventory, like most current national forest inventories (Lund, 1998). In this case, the design of the NFI will limit the freedom to choose a sampling procedure that is particularly efficient for carbon monitoring. Nevertheless, within the framework of an NFI, specific samples taken solely for carbon estimation can be distributed in such a way that maximum precision is attained within the available budget.

Existing information of various kinds can often be used to improve the precision of estimates. One such example is the use of remote-sensing data.

Accurate estimates and check assessments

It must be emphasised that it is estimates of *change* in different pools, rather than estimates of state, that are of prime importance in the monitoring of carbon pools in forests according to the Kyoto Protocol. Consequently, sampling procedures must be set up that are efficient for estimating change rather than state. Generally, this involves the use of permanent plots. For soil sampling, the use of permanent plots has some limitations, since exactly the same plots cannot be used, owing to the destructive character of soil sampling.

Flux measurements

Direct estimation of NEE can be made by means of flux measurements. Techniques for the measurement of gaseous fluxes from different types of surface have developed rapidly during recent decades (*e.g.* Aubinet *et al.*, 2000). The flux is often expressed as a flux density, *i.e.* it is expressed per unit area of an object. For flux measurements of separate components of an ecosystem, *e.g.* a leaf of a tree, it is most common to use some kind of chamber, which encloses the entire object studied. For measurements from whole ecosystems, micrometeorological methods are most commonly used. In the following, a brief outline is given of the basic principles of these two types of flux-measurement method.

Fluxes by chamber techniques

The basic principle for almost all chamber measurements is simple; the object under study is enclosed in a chamber, transparent or dark depending on the objective. Then the concentration of CO_2 in the air entering the chamber, and leaving the chamber, is measured by a gas analyser. From measurements of the mass flow of air passing the object, the flux of CO_2 can be calculated. These types of system are called dynamic chambers, and can be 'open' or 'closed' depending on the technical solution chosen. The main disadvantage is that the environment within the chamber is often quite strongly affected, which in turn can affect the object itself and therefore create bias. In advanced systems, the environment is controlled to avoid such bias. In static chambers, there is no air circulation, and the flux is measured by measuring concentrations at different times; from the rate of change of concentration per unit time, and a knowledge of air volume, the flux can be calculated. This type of system is commonly used for measuring the flux of CO_2 from the soil (soil efflux). All soil efflux chamber measurements are hampered by the adverse effect of pressure on the measurements, in addition to the fact that the soil is disturbed by the installation of the chamber.

All methods based on measurement of component fluxes must be scaled-up to achieve an

estimate of NEE. This is a serious constraint, and soil fluxes in particular are difficult to scale up because of the heterogeneity of soil. It is, however, a relatively cheap method, and can be widely used to give important information about the variability of fluxes with local conditions.

Fluxes by micrometeorological methods

For studies of the planetary boundary layer, *i.e.* the part of the atmosphere that is in contact with the earth's surface, different methods have been developed for measurement of fluxes of heat, water vapour (evaporation) and momentum (wind friction). The sub-discipline of meteorology concerned with the development of such methods is called micrometeorology. Micrometeorological methods can, in principle, be used to measure the flux of any gas that exists in the air, *e.g.* CH_4 or CO_2. The methods are based on the fact that it is the wind that is the bearer of the 'properties' (*e.g.* the concentration of a certain substance) that are transported between the earth's surface and the atmosphere. When the wind sweeps over the surface, mechanical turbulence develops, because of the friction between the air and the underlying surface. Turbulence (which can also be caused by thermal effects) consists of seemingly chaotic wind movements in all three dimensions. This turbulent movement can be seen as parcels of air, 'eddies' of different size and moving with different speed and rotation. It is the vertical component of the movement of these 'eddies' that can cause a net upward or downward transport of a 'property'. However, for the net transport of a 'property' to take place, there must exist a difference in concentration of this 'property' in the vertical direction. If, for instance, the concentration of CO_2 is lower at the earth's surface compared with that at a given height above the surface, an eddy that is moving downwards will, on the average, carry a higher concentration as compared to an eddy that is moving upwards. This will result in a net transport downwards of CO_2. The difference in concentration with height can be maintained only as long as there is a sink or source somewhere. In the case of fluxes of CO_2, the sink is caused by photosynthesis and the source is caused by respiration.

Various micrometeorological methods have been developed on the basis of the principles described above. The most direct of these methods is the 'eddy-correlation' or 'eddy-covariance' method (*e.g.* Kramer *et al.*, 2002). In this method, the vertical velocity of the respective eddies must be known, together with their CO_2 concentration. This sounds fairly simple, but is rather complicated in practice. One problem is that the eddies have different sizes, from less than metres to hundred of metres – *e.g.* above a forest canopy – and they sometimes move with high velocity. This implies specifically that the sensors must be fast in response, down to typically 10 Hz, and must also be sufficiently sensitive to detect small differences. The wind-speed is typically measured by a sonic anemometer, which has no moving parts and which is very fast in response. With such sensors, the 3-dimensional movement of the wind can be measured very accurately. The CO_2 concentration is normally measured by an infrared gas analyser, *i.e.* an instrument that measures the absorption of light by CO_2 molecules in the infrared region of the spectrum. Recent technical development has made such instruments fast enough to be used routinely for this type of measurement. With this methodology, fluxes can be measured with a time resolution of about 30 minutes. This means that the method gives very detailed information about the processes concerned, and is therefore especially suitable for research aimed at providing a better understanding of the interaction between climatic parameters and the NEE.

With the method described above, it is in principle possible to measure a flux of CO_2 at any point in space; but since in our case, the purpose is to measure the NEE, it must be placed above the forest concerned. The higher the instruments are situated above the forest, the larger is the area covered by them. It is also important to understand that the area measured by the instruments is some distance upwind of the sensors (because of the horizontal movement of the wind), and that the size of that area varies with conditions in the atmosphere and depends on the height of the sensors. The higher the sensors are placed above the forest, the more extensive becomes the measured area. At night, when the atmosphere is stable, the measured area is larger than in daytime. The area representing each measured flux is called a 'footprint'. In a typical setup, the sensors are placed, say, 5 m above the canopy, and the footprint is in the

range 50–300 m upwind of the sensors. This will restrict the type of forest that can be measured by this method. It must be fairly homogeneous in all directions, and quite large in extent.

Scaling-up from individual sites to the national level would require that a large number of micro-meteorological sites be established. Owing to the cost, this is currently not a feasible approach to obtaining national-level carbon budgets.

Modelling

Although models of various kinds are involved in obtaining change estimates, both from sample surveys and from flux measurements, we consider those methods that are based on actual measurements of the quantities to be of prime interest; the measurement parts dominate over the model parts in the assessments.

With modelling approaches, measurement data are generally utilised only for obtaining the initial conditions. Models are then used to predict the changes in carbon stock for a given period of time (*e.g.* Mollicone, Matteucci, Koble, Masci, Chiesi & Smits, 2003), perhaps with the incorporation of easily obtained weather data during this period as additional input data.

Model approaches can be of many different kinds. Examples of different kinds of model are given in the next section.

Methodological options – analyses for the case of Sweden

In this section, different methods for determining the changes in carbon pools in Swedish forests are presented and discussed. It is shown what can be achieved with these methods, on the basis of the data currently available in Sweden. A separation is made between methods for estimating changes in carbon stocks in *tree biomass* and corresponding methods for estimating *soil carbon*. The carbon pools in litter and dead wood are only briefly discussed. In addition to the methods for assessment of change in carbon pools, eddy-covariance methods for measuring fluxes are discussed in the context of verification.

Uncertainties in estimates of change in carbon pools

Uncertainties in estimates of change in carbon pools are of two different kinds. *Systematic errors* express the average difference between the true change and the estimated change, if an inventory is repeated many times. This kind of error typically stems from incorrect measurements, subjectively located sites, model errors, *etc.* Some sources of this kind of error can be checked (*e.g.* measurement errors), and be corrected for. However, in practice it is generally impossible to determine the size of systematic errors. Instead, procedures must be chosen in order to avoid this type of error as far as is practicable (*e.g.* IPCC, 2000b).

Random errors express the variation around the average estimate, when an inventory is repeated many times. These errors occur in sampling because only a sample of the entire population is measured. Moreover, they may be a result of variable quality in the measurements. Random errors commonly are expressed as standard errors (*i.e.* the estimated standard deviation of an estimate).

If an estimate is unbiased (*i.e.* it has no systematic error), a 95% confidence interval can be set up as the change estimate ± 1.96 times the standard error. This means that about 95 times of 100 the confidence interval will comprise the true value of the change.

According to the recommendations from IPCC (IPCC, 2000b), systematic errors and random errors can be combined into a single uncertainty estimate, which is treated as a random error (although this is somewhat inadequate from a statistical point of view). Confidence intervals are then constructed which include both random and systematic errors. In this case, the size of systematic errors generally has to be assessed subjectively.

When dealing with carbon estimates, different kinds of model are used, *e.g.* biomass functions to calculate the biomass of individual trees from basic measurements. As long as these functions provide unbiased results at the plot level, the estimated standard errors will reflect the model errors as well as the errors due to sample selection (*e.g.* Bondesson, 1990). This is an important example in which the approach suggested by IPCC may be misleading, and in which standard

sampling procedures for estimating standard errors should instead be used.

In the analyses which follow, we concentrate on a formal analysis of the random errors of the estimates of change, and make only subjective assessments of the likely size of systematic errors.

Sampling for tree biomass change estimation

With forest inventory data, changes in the total quantity of carbon in living trees between two points in time can be assessed in different ways. Below, different methods are treated in greater detail for the case of Sweden. The examples shown are based on data from the period 1990–2000. During this period, the annual increase of carbon in tree vegetation (including roots) has been 8.7 Mtonne C yr^{-1} (in forests outside protected areas). The total amount of carbon in forest trees in Sweden is *ca.* 1 Gtonne C. The figures for standard errors, *etc.*, have been obtained from analysis of the NFI material.

The procedures for estimating changes in tree biomass, based on field sampling, to be described and discussed, are:

(1) Summation of biomass growth minus removals, along the basic principles of IPCC (1997).

(2) Direct estimation of change between the end and the start of a reporting period.
   (a) using temporary sample plots
   (b) using permanent sample plots

(3) Combined estimation; using (1) and (2).

Summation of growth-minus-removals

In forest inventories, growth generally is estimated from measurement of annual rings on increment cores (Svensson, 1988) as well as from repeated measurement of trees on permanent plots. This typically provides a growth estimate for the latest five-year period. Fellings may be estimated from measurements on the stumps of harvested trees. This procedure is used in Sweden, but is less common in other countries. However, most countries have industrial consumption statistics, as well as figures on export and import, whereby annual harvested volumes can be estimated. In the latter case, only growth and natural mortality is estimated on a sampling basis.

In principle, this is the procedure used for Sweden's past reporting under the UNFCCC. The approach is suitable for annual reporting, although a problem (seldom addressed) is that the growth figures implicitly will be based on assumptions of steady state, since they are generally taken as averages for a certain period of time before the actual reporting year.

In formal terms, the basic principle is to estimate the change for a five-year period, Δ, as:

$$\hat{\Delta} = \sum_{t=1}^{5} Grôwth_t - Cuttîngs_t - Morîality_t \quad (1)$$

Every year in the evaluation period, growth, fellings and natural mortality are estimated. Since these estimates commonly are obtained as volumes, they must be converted to amounts of carbon by means of expansion and conversion factors (*e.g.* IPCC, 1997). Then a sum is calculated according to equation (1). The different components can be based on the separate assessments of growth, fellings and natural mortality that often are part of NFIs. Owing to the likelihood of underestimation of fellings in the NFI (*e.g.* Daamen, 1980), it is advised that Swedish industrial consumption statistics (Anon., 2002) be used for estimating harvests.

As regards the precision of this method, the standard error of estimates of annual fellings is generally poor; it is of the order of 5–10% in NFI data. The uncertainty of industrial consumption statistics is not known, but this estimate is likely to have a somewhat smaller standard error. Growth estimates have better precision; usually, the standard error is of the order of 1–3% of the annual growth. When equation (1) is used to estimate the total change in carbon quantity, the variance will be a sum of the variances of the estimates for the single years (*e.g.* Cochran, 1977). The standard error of the estimate of change for a five-year period is likely to be of the order of 25%. In this case, the standard error of the consumption statistics has been set at a level slightly better than the precision of the corresponding NFI estimate.

A conclusion is that this method results in reasonable precision, but that there is a risk of systematic error, which may be quite large. This depends partly on the many conversion and expansion factors that must be applied in obtaining biomass and carbon estimates from the basic volume estimates on which the method

relies. Another methodological problem is the time-lag between the growth estimate and the estimate of fellings. For a five-year period, one solution would be to use the growth figures from the inventory at the end-point of the period, while the annual harvest estimates should be summed. This would require recalculation at the end of the reporting period, of the figures reported for the single years.

Direct estimation of change between two time points

If the main objective of the reporting of change is a certain evaluation period – *e.g.* of duration five years – rather than the provision of annual figures, forest inventory data can be used to estimate the change in carbon pools directly, instead of estimating and summing annual changes. The total carbon pool is simply estimated at two different points in time, and the change is taken as the difference. With this approach, the use of permanent sample plots is superior to the use of temporary plots (*e.g.* Ranneby, Cruse, Hägglund, Jonasson & Swärd, 1987), although when the number of plots is large, this procedure can be meaningfully applied also with temporary plots (*i.e.* plots visited once only).

This approach can be used for assessing changes in all the carbon pools, not only those in trees.

In formal terms, an estimator of the change, Δ, is set up according to equation (2), where Y_2 is the carbon quantity at time point 2 and Y_1 the quantity at time point 1.

$$\hat{\Delta} = \hat{Y}_2 - \hat{Y}_1 \qquad (2)$$

The estimators of Y_1 and Y_2 will depend on the sampling procedures adopted. A generally applicable estimator of the carbon quantity at either of the two points in time is the Horvitz-Thompson estimator (*e.g.* Thompson, 1992), which is obtained as a sum over all sampled trees, of the carbon quantity in a tree divided by the tree's probability of inclusion in the sample. With plot-based inventories such as the Swedish NFI, all trees (within a certain stratum) will have the same probability of inclusion.

Use of temporary plots If temporary plots are used, the variance of (2) will be:

$$Var(\hat{\Delta}) = Var(\hat{Y}_2) + Var(\hat{Y}_1) \qquad (3)$$

If all plots in the Swedish NFI were temporary, and if the estimate of change for the five-year evaluation period were based on three-year averages at the starting point and the end point, the standard error would be 18 Mtonne C, or about 40% of the change (an annual increase of 8.7 Mtonne C is assumed), which is relatively large.

As expected, a conclusion is that the use of temporary plots for assessing the changes in carbon pools in forest trees over a five-year period will result in quite unreliable estimates. However, precision can be improved by increasing the number of sample plots.

In addition to large random errors, there is also a risk of substantial systematic errors when this approach is used. This is the case, since small systematic errors in assessment of the initial and end-point carbon stock are likely to result in a large systematic error in the estimate of change.

Use of permanent plots In estimating changes, it is well known that permanent plots generally will result in higher precision than temporary plots. The background principle for this is shown below, in estimating the same quantity as above. When permanent plots are used, the variance of the estimator (2) will be:

$$Var(\hat{\Delta}) = Var(\hat{Y}_2) + Var(\hat{Y}_1) - 2\,Cov(\hat{Y}_2, \hat{Y}_1) \qquad (4)$$

Compared to the case when only temporary plots are used, the covariance between the estimates at time point 1 and time point 2 will lead to considerably reduced variance of the change estimator. If the same variance is assumed on both occasions (which will generally be a good approximation), the formula can be simplified to:

$$Var(\hat{\Delta}) = 2\,Var(\hat{Y})(1 - \rho) \qquad (5)$$

Here, ρ is the correlation between the two estimates, which generally is very high in applications of this kind. Typically, in estimates of the amount of biomass, the correlation will be at least 0.9 over a five-year period; thus the variance of the estimate of change will be only 10% of what it would have been, had the plots been temporary.

If it is assumed that all NFI plots were permanent, and that the change for a five-year

period is calculated on the basis of three-year averages at the beginning and the end of the evaluation period, the standard error of the change estimator would be only 6 Mtonne C, or about 13% of the change. This is judged to be quite sufficient precision.

By comparison with the use of temporary plots only, the risk of systematic error in the estimate of change due to variable systematic errors at the two points in time is reduced. This is the case, since the trees are marked at the initial inventory, and revisited at the second inventory. The problem of missed trees at one of the two points in time can thus be avoided. On the other hand, there is a risk that the management of permanent plots is different from the management of the rest of the forest, if those plots are too clearly marked. It is therefore important that permanent plots be marked very discreetly, so that forest managers are unable to detect them.

A conclusion from this is that if all NFI plots were permanent, the precision of estimates of change in carbon mass over a five-year period would be quite high.

Combination of permanent and temporary plots
Combination of permanent and temporary plots The NFI in the period 2008–2012 will most probably be based on a combination of permanent and temporary sample plots. With such a combination, the precision of the estimates will be intermediate to the ones presented under (*2a*) and (*2b*) above. However, since the majority of the plots (two-thirds) probably will be permanent, the figures will be close to the ones presented under (*2b*), *i.e.* the standard error can be expected to be about 15% of the change.

Combined estimation, by means of a mix of permanent and temporary plots (*e.g.* Scott & Köhl, 1994) is rather complicated from a theoretical point of view, especially as regards the estimation of standard errors. However, approximate methods exist for estimating the standard errors.

Concluding remarks on direct-change estimation
Some concluding remarks on the estimation of change from repeated measurements are given below:

- With permanent sample plots, the size of random errors will be reasonably small, while with temporary plots, the errors will be much larger. The calculation of changes for a five-year period on the basis of three-year averages at the starting and end points in time, will result in quite high precision when NFI data are used.
- It is important that the re-measurement interval for permanent plots matches the evaluation period, otherwise the gain from using permanent plots is somewhat reduced.
- The change is estimated as a difference between two large numbers. It must be ensured that the size of systematic errors will not differ between the two points in time. (Systematic errors of equal size at both points in time are, however, not a problem.) The risk of change in systematic error levels probably is much larger when temporary plots are used, than when permanent plots are used, since the risk for varying levels of systematic error is reduced in the latter case, due to the fact that the coordinates of all trees are recorded.
- Check assessments of the size of systematic errors should be carried out. To emphasise the importance of check assessments, a 1% difference in systematic error between the two points in time would correspond to 10 Mtonne C, *i.e.* almost 25% of the expected change over a five-year period. Results from check assessments within the Swedish NFI (*e.g.* Daamen, 1980), indicate systematic error levels for biomass estimates of the order of 1–2%.
- Detailed biomass conversion factors can be applied, *i.e.* functions that give the amount of carbon for single trees from basic measurements on the trees. The application of biomass functions is straightforward in this case, while it is impossible to apply when growth minus fellings is summed.

As was assumed in the examples above, there is sometimes a need to use data from more than one year to obtain reliable averages. For example, for the reporting period 2008–2012, data might be needed for the period 2007–2009 to assess the state in 2008, and from the period 2011–2013 to assess the state in 2012. In such cases, there will be a delay in reporting, because of the need to wait for data from the year after the end of the reporting period. Another possibility would

instead be to assess changes on the basis of data between 2006–2008 and 2010–2012, and to correct by extrapolation for the slight mismatch.

Finally, the size of random errors can always be reduced by increasing the number of sample plots. However, there is a limit beyond which further decrease in the size of random errors is not meaningful, since the systematic errors will then tend to be very large in comparison with the random errors. The dimensioning of inventories should thus not aim solely at reducing the statistical errors (random errors), but also at controlling the size of systematic errors by adequate use of control procedures, such as control inventories.

Combined estimation

When different estimates are available for the same parameter – in this case the total change in carbon stocks over a certain period – the different estimates can be used in two principally different ways:

- Verification
- Combined estimation

In the first case, one of the figures is considered superior, and the other(s) used for verifying the plausibility of the first figure.

In combined estimation, different estimates of a certain quantity are consolidated into one single estimate. For example, if estimates are obtained both using the 'growth-minus-removals' approach and direct estimation of change, the two estimates can be combined into a single estimate in a way that minimises the variance (*cf.* Raj, 1968; Ståhl, 1992). For combined estimation to be meaningful, the different single estimates should be reasonably unbiased, and it should be possible to estimate their variances. With sampling-based approaches, these criteria generally are met.

A combined estimator is set up according to (example for the case with two different available estimates):

$$\hat{\Delta}_c = a\hat{\Delta}_1 + (1-a)\hat{\Delta}_2 \qquad (6)$$

With uncorrelated estimates of equal precision, the variance of the combined estimator will be only 50% of the variance of any of the two separate estimators (*e.g.* Raj, 1968).

The details of how to set up a combined estimator have not been explored in the case of estimation of forest carbon pools. The application of combined estimation – using estimates from methods 1 and 2 as input – would require some further development before it could be implemented. However, with combined estimation there is a potential for further improvement of the precision of estimates of carbon change. If a situation in which two-thirds of the NFI plots are permanent is considered, it is likely that the increased precision from combined estimation would be quite limited. A rough estimate is that the standard error would be about 5 Mtonne C or 10% of the change.

Sampling for estimation of soil carbon change

Assessment of changes in soil carbon pools by sampling approaches follows the principles outlined above for assessing changes in biomass. The pools are assessed repeatedly, and the change is taken as the difference between two successive assessments. However, for soils there exists no counterpart to the 'growth-minus-removals' approach for biomass, since inputs and outputs of carbon from the soil system cannot be measured without applying very advanced and expensive methods.

Depending on the soil layer involved, different methodological approaches are used for assessing the carbon stock. The approaches apply to the methods used in the Swedish Forest Soil Inventory (MI).

- For carbon in the humus layer, sample cores are taken by random sampling. The dry mass of humus per unit area of forest land is determined and from laboratory analyses the mass of carbon per unit dry mass of humus is estimated.
- The amount of carbon in peat soils currently cannot be determined, since no cores are taken from different depths of the peat soils. Furthermore, measurement of the depth of peat layers (below 1 m) in the MI started only in 2003.
- The amount of carbon in mineral soils is assessed by taking soil samples at different depths from the soil fraction <2 mm. The carbon concentration in these fractions is determined in the laboratory.

In the following, the accuracy of estimates of changes in carbon in humus and mineral soils,

based on MI data, is discussed. No analyses are provided for carbon in peatlands, owing to lack of data.

Changes in the carbon stock in the humus layer

Changes in humus-layer carbon can be analysed on the basis of repeated sampling of plots. The re-measurement interval currently is ten years. In total, *ca.* 3000 plots in the whole country (and *ca.* 1000 plots in southern Sweden) have been included in the analyses presented here. Each basic sample plot has a radius of 10 m. About 3–5 small sample cores per plot were acquired and pooled to give a composite sample, which was then analysed for carbon concentration in the laboratory.

The humus-layer carbon stock for an individual plot is obtained as:

$$C_{stock} = C_{conc} \cdot D \cdot L \cdot A \qquad (7)$$

Here, C_{stock} is the stock of carbon on a sample plot, C_{conc} the concentration of carbon in the humus layer, D the bulk density of the humus, L the average depth of the humus layer on the plot, and A the area of the sample plot. Roots and similar material thicker than 2 mm are removed from the cores, and thus not included in the analyses. To derive an overall national-level estimates from the plot values, standard sampling theory is applied to the specific design of the MI (*e.g.* Ranneby *et al.*, 1987).

However, no formal analyses of changes in the carbon stock, covering the entire country, have been made for the humus-layer carbon. Instead, results from investigations limited to certain soil types are used, to determine what sample sizes would be necessary to detect changes in the humus-layer carbon stock.

The standard deviation of values of changes by plots was found to be 1.2 kg C m^{-2} for the whole country and 1.4 kg C m^{-2} in southern Sweden – equal to a coefficient of variation of *ca.* 50%. From these values, approximate 95% confidence intervals were constructed (Fig. 3). Half the (expected) length of a confidence interval was plotted versus the number of plots.

We can assume an annual increase in humus-layer C content that might amount to 0.5–1.0% of the present stock, in southern Sweden at least. This value is based on repeated measurements in the MI, modelling by COUP and on results from the soil model YASSO in Finland (LUSTRA, 2002). The mean content is *ca.* 2–3 kg C m^{-2}. Consequently, the possible change that should be verified is *ca.* 0.02 kg C m^{-2} yr^{-1}, or 0.10 kg C m^{-2} for a five-year monitoring period. From Fig. 3 it can be deduced that 700 plots would result in a confidence interval of *ca.* $\pm$0.099 kg C m^{-2}, *i.e.* the change would be significantly larger than zero.

On the basis of 3000 plots for 3 Mha in southern Sweden, for a five-year-period we can assume a change of 3 ± 1.5 Mtonne C. This conclusion requires that sampling procedures and analytical methods are constant between the two sampling periods, so that there are no distortions due to varying levels of systematic error. Potential sources of systematic error in this case are:

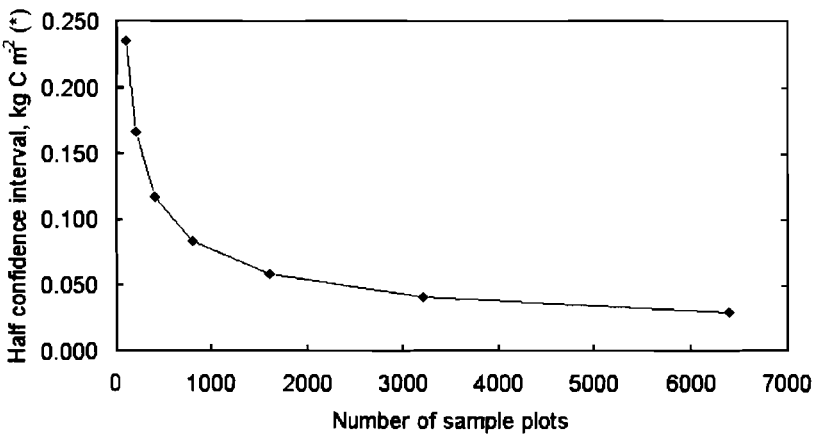

Fig. 3. The approximate relation between the confidence interval and the number of plots used (for a STD of 1.2 kg m^{-2}).

- The selection of humus sample cores may not be entirely objective. Specifically, some sample locations (*e.g.* close to stumps, large roots or boulders) may be impossible to assess.
- The boundaries between the litter layer and the humus layer, as well as between the humus layer and the mineral soil layer, are difficult to determine. The principles for determining these boundaries may vary between surveyors as well as between years.
- Difficulties in separating dead and living roots during sample preparation.
- Erroneous estimates of bulk densities, particularly in compact subsoil horizons.
- Carbon analyses – particularly change in method during the monitoring period, sometimes due to small sample volumes and poorly homogenised samples.

Carbon stock changes in the mineral soil

On *ca.* 3500 plots within the MI survey, soil pits are dug and soil samples taken from various depths and layers in the mineral soil. However, the analysis of these samples comprises only soil fractions finer than 2 mm. Measurements from the same plots are repeated at ten-year intervals; the new soil pit is dug a few metres from the old one on any plot.

The C stock on a plot, in a certain layer or horizon, can be obtained as:

$$C_{stock} = C_{conc} \cdot D \cdot f \cdot L \cdot A \qquad (8)$$

In this equation, C_{stock} is the stock of carbon per unit area, C_{conc} the concentration of carbon in the <2 mm fraction, D the bulk density of the <2 mm fraction, f the proportion of soil volume <2 mm in the layer studied, L the depth of the layer, and A the area of the sample plot.

So far, no comprehensive analysis of changes in the carbon stock in the mineral soils has been made on the Swedish MI data. This is because many problems remain in determining the bulk density of the fractions analysed, and in inferring the total carbon stock at a site on which samples have been taken only from a few mineral soil horizons. The sample size discussion below is thus largely hypothetical. It is based on information about the variation in C concentration from one measurement only.

Since L and A in equation (8) are constants, the coefficient of variation (*e.g.* Cochran, 1977) for the plot-level values of carbon in the mineral soil can be approximated as:

$$CV = \sqrt{CV_C^2 + CV_D^2 + CV_f^2} \qquad (9)$$

Here, CV_C is the coefficient of variation for carbon concentration, CV_D is the coefficient of variation of the bulk density of the finer fractions, and CV_f is the coefficient of variation of the proportion of soil volume finer than 2 mm.

In general, with a CV of 30% for bulk density, 40% for the proportion of fine material, and 80% for carbon concentration, the total CV at the plot level would be *ca.* 95%.

A change of 0.5% annually in the amounts of C in the mineral soil is assumed. This change is slightly less than the one in the humus layer (see above), and reasonable in southern Sweden at least. The present mean content in mineral soils to a depth of 0.5 m is *ca.* 4 kg C m^{-2}. Consequently, the possible change that should be verified is 0.02 kg C m^{-2} yr^{-1}. During a five-year monitoring period, this corresponds to 0.10 kg C m^{-2}. These assumptions were inserted into equation (4), assuming the covariance between the estimates at points in time 1 and 2 to be 0.7 when estimating a change over a five-year period For a sampling density of 3200 samples, the 95% confidence interval would be the estimate ± 0.095 kg C m^{-2}, thus enough to verify that the change in carbon stock is larger than zero. A current sampling density of *ca.* 3500 samples in Sweden gives a change of 0.10 ± 0.08 kg C m^{-2}.

However, this calculation has not considered systematic errors between sampling occasions, by which the uncertainties might be greatly increased.

Sources of systematic error in carbon stock estimation in mineral soils

Many problems remain regarding the assessment changes in carbon stock from repeated measurements in the mineral soil.

Determination of the proportion of coarse material is a major uncertainty in countries in which glacial tills or similar soil types dominate. Few data on this issue seem to exist in the Nordic countries. In Norway, NISK roughly estimate the stone and boulder content at 30%. However, the variation between sites was extremely large. In Sweden, the stoniness index according to Viro or Tamminen (Viro, 1952) is commonly used.

Determination of bulk density is less of a problem. However, knowledge and techniques for assessing bulk density in deep layers are poor.

A further problem is the definition and recognition of dead organic matter under decomposition. Living roots of any size are not included in the soil organic pool. However, it is generally difficult to separate living roots from recently dead roots, when the tissue structure is intact. Different scientists have different traditions concerning the separation of living-root-C from dead-organic-matter C. The practice of including coarse organic matter also seems to differ among scientists. By definition, it should be included in the C-pool of soil organic matter, but sometimes it is excluded in the sieving process, *i.e.* before analysis, the soil is passed through a 2-mm sieve, and coarser material is excluded.

To conclude, some major potential sources of systematic error are:

- Erroneous assessment of the volume of coarse material, *i.e.* stones and boulders that are not included in the soil samples, but either neglected, guessed or estimated by other techniques.
- Inconsistent identification of soil horizons or reference depth – particularly if there is a change in field staff between measurements.
- No sampling beneath stones and large roots (slight underestimate of C concentration).
- Erroneous estimate of bulk density, particularly in compact subsoil horizons.
- Overestimation of organic C in calcareous soils (a minor problem in Sweden).
- Carbon analyses – particularly change in method during the monitoring period, sometimes due to small sample volumes and poorly homogenised samples.

Spatial variation and sampling density

It is of interest to know the spatial autocorrelation in data when designing sampling surveys. A study carried out within the LUSTRA research programme at Knottåsen in Sweden (Fig. 4), indicated that autocorrelation was very low at distances >3 m for O horizons, and >20 m for mineral soil horizons (B horizon). For the mineral soil horizon, autocorrelation was limited even at short distances. However, these results depend on the scale at which the processes are observed, and on the reliability of data obtained in single samples. For example, measurement errors will decrease autocorrelation.

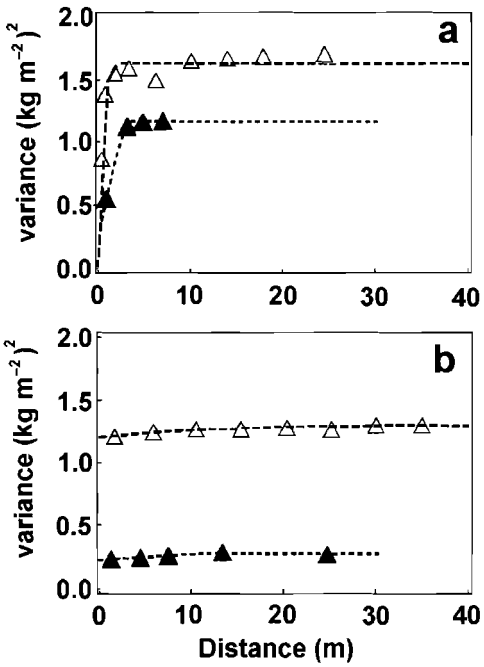

Fig. 4. Variograms for moist and fresh classes: (*a*) O-horizon; (*b*) B-horizon. Moist and fresh classes are indicated by empty and solid symbols, respectively.

Apart from the spatial variation of carbon concentration in fine soil (<2 mm), one should also consider the spatial variation in bulk density and content of coarse mineral material, *i.e.* stones and boulders. However, this variation is probably smaller than for the concentration of C.

There is little information on the spatial variation of stones and boulders. A study from glacial till in central Sweden (Eriksson & Holmgren, 1996) showed that the CV was in the range 25–75%, with a mean of *ca.* 40%. The study was conducted by measuring the percentage by volume of material >2 cm on five sample plots 0.5×0.5 m at each of ten different sites.

Stratified sampling for improving carbon stock change estimation in soils

Within the MI, soil samples are taken on rather few plots because of the costs associated with the fieldwork and the subsequent analyses in the

laboratory. Thus in this case it could be wise to consider the opportunity for allocating the soil sampling effort to the plots, within the MI, where they are most useful, *i.e.* where it contributes most to the precision of the assessment of change in soil carbon over a given period. One approach to this would be to use the principle of stratification (or two-phase sampling for stratification, *e.g.* Thompson (1992)), to allocate the soil sampling efforts optimally. The basic principle would be to stratify the existing MI plots on the basis of characteristics that are believed to covary with soil carbon change, and then to allocate the sampling effort to the different strata in proportion to the expected variability of carbon changes. Below, a general presentation of the principle is first given. Secondly, an example is given that is based on rather crude assumptions about the variability of soil carbon changes between different kinds of soil.

Allocating observations to strata

Given a total sample size n, one may choose how to allocate it among the strata. The interest lies in finding the best choice of sample sizes, n_h, in each stratum to obtain the best precision. If each stratum has the same size, and if there is no prior information about the population, a reasonable choice would be to assign equal sample sizes to the strata, so that for stratum h the sample size would be

$$n_h = \frac{n}{L} \tag{10}$$

This corresponds to *proportional allocation.* If the strata differ in size, proportional allocation could be used to maintain a constant sampling fraction throughout the population. If stratum h has N_h units, the sample size allocated to it would be

$$n_h = \frac{nN_h}{N} \tag{11}$$

where N is the population size. When the strata are large enough, the population standard deviation under proportional allocation is usually smaller than the standard deviation achieved with ordinary simple random sampling.

The objective in sampling generally is to obtain the maximum information for the least cost. Let C represent the total cost, c_0 represent overhead costs, and c_h represent the cost of taking an observation in stratum h. The objective is to allocate observations to strata in order to minimise the variance for a given total cost C or to minimise C for a given variance. If we suppose that the costs $c_1, c_2, \ldots, c_L$ are fixed, than the allocation scheme, which estimates the population mean or total, with the lowest variance for a fixed sample size n and fixed cost under stratified random sampling, is *optimum allocation*:

$$n_h = \frac{nN_hS_h/\sqrt{c_h}}{\sum_{k=1}^{L} N_kS_k/\sqrt{c_k}} \tag{12}$$

where s_h is the population standard deviation in stratum h. This leads to the following rules: In a given stratum, take a larger sample if

(1) The stratum is larger.
(2) The stratum is more variable internally.
(3) Sampling is cheaper in the stratum.

If the standard deviations are more or less equal across all the strata, proportional allocation is probably the best allocation for increasing precision. When the S_h varies greatly, optimal allocation can result in increased precision. Optimal allocation is, of course, possible only when there is prior information about the standard deviation in each stratum. In the case of monitoring of soil carbon stocks, it is likely that the standard deviation of change in carbon stock varies between different soil types, *e.g.* peat soils and upland soils.

The problem of estimating the carbon stock changes in Swedish forest soils

A case study was conducted to discover what could be achieved by using stratified sampling for estimating change in the soil carbon pool in Swedish forests. The population (all soils) can, for instance, be stratified into podsols on well-drained soils, brown forest soils under well-drained conditions, thin well-drained soils, and soils under wet or moist conditions. However, to keep the example simple, only two strata were identified: well-drained soils and peatlands.

There are 591 measurements for the carbon stock changes on well-drained soils from 1983–1987 to ten (or 11) years later from the MI.

The mean changes in carbon stock for these measurements are 0.29 kg m^{-2}, with standard deviation 2.77.

Two datasets were constructed, each with 745 simulated measurements for the changes in carbon stock on peatland during the same period of time as data were available for well-drained soils. The mean changes in carbon stock in these two datasets were 3.10 kg m^{-2}, with standard deviation 4.15 (dataset 1) and 9.12 kg m^{-2}, with standard deviation 2.70 (dataset 2).

Assume a random sample of 100 observations of carbon stock changes (kg C m^{-2}) with proportional allocation. Since $N_1 = 591$ and $N_2 = 745$, we obtain $n_1 = 44$ and $n_2 = 56$. The estimated mean value of the changes in carbon stock ($\bar{C}$) and the estimated standard deviation (s) of $\bar{C}$ for the two different datasets with simple random sampling (SRS) and stratified sampling became those shown in Table 1.

As may be seen from Table 1, stratified sampling gives a lower standard deviation, 15% for dataset 1 and 52% for dataset 2, compared to SRS. Thus, the case study indicates that stratified sampling would be a means of considerably improving precision within soil monitoring programmes.

Modelling approaches based on survey data

The preceding sections have treated cases in which changes in carbon stock were estimated by repeated measurements, or by separately assessing growth and removals of biomass. Sampling data have been used to assess the changes in the carbon pools. To some extent, models have also been involved, for example for converting basic measurements on trees to biomass (*e.g.* Marklund, 1987). However, these models have generally played a minor part in the analysis.

Moving one step further, models can be used as a major component in the analysis of changes in carbon stock. Moreover, the models can be applied directly to data which describe the present state of forests; hence there is no need to make repeated measurements.

Table 1. *The precision in change estimates using stratified sampling and simple random sampling*

	$\bar{C}$ (SRS)	s (SRS)	$\bar{C}$ (Strat.)	s (Strat.)
Dataset 1	1.7562	0.3947	1.5127	0.3356
Dataset 2	4.7926	0.4389	4.8216	0.2512

In general, it can be argued that very precise repeated measurements will always give the best estimates. However, the cost of applying models may be much lower, and some potential sources of systematic error in repeated measurements can be avoided. The choice between the application of models or repeated measurements should be made in consideration of the cost and accuracy of the approaches (*cf.* Ståhl, 1994).

Below, separate sections are devoted to model-based assessment of changes in the five carbon pools to be reported under the Kyoto Protocol.

Model-based estimation of change in aboveground biomass carbon

The baseline approach here would be to estimate separately gross growth, natural mortality, and removals due to harvesting. Overall changes in biomass would then be converted to change in carbon. The basic data to which the models would be applied, would be data from the NFI.

Gross growth can be estimated by many different models (*e.g.* Ekö, 1985; Söderberg, 1986). If models for single trees are applied, static biomass functions for individual trees (Marklund, 1987) can be used to obtain growth directly in terms of biomass rather than volume. Most Swedish growth functions have been established to provide growth for a five-year period. Even if such functions are known to be rather accurate on the average, large fluctuations in growth appear between different years.

Specific functions (*e.g.* Söderberg, 1986; Fridman & Ståhl, 2001) can be used for estimating natural mortality. For natural mortality, there is an even larger variation between years, compared to gross growth.

Removals due to harvesting would typically be taken from industrial consumption statistics (Anon., 2002), rather than from models. The accuracy of such figures is unknown, but probably reasonably high.

The overall accuracy of a modelling approach of the above kind is difficult to assess. It might result in quite large systematic errors for a five-year evaluation period.

An advantage of modelling approaches is that annual values can easily be derived. Such values are needed, even though the reporting period for the Kyoto Protocol is five years.

Model-based estimation of change in belowground biomass carbon

At present, no specific growth functions are available for estimating the growth of belowground biomass. However, by applying individual tree-growth models (Söderberg, 1986) together with biomass functions (Marklund, 1987), the gross growth of the belowground biomass of trees can be estimated. Alternatively, a fixed ratio between above- and belowground biomass can be assumed, and any kind of model for the growth of aboveground biomass be used for assessing also the growth of belowground biomass.

Any assessment of this pool also needs to consider both the natural mortality of trees, and harvests. In both cases, the roots will die, and thus become part of another pool. These components need to be assessed along the same lines as for aboveground biomass.

Model-based estimation of change in dead wood

The creation of new dead wood can be modelled by means of mortality functions (*e.g.* Söderberg, 1986; Fridman & Ståhl, 2001). The likely amount of dead wood left in the forest after harvesting can be obtained from the NFI.

However, as regards the decomposition rate of dead wood, at present only very crude models are available.

Model-based estimation of change in litter and soil organic carbon

Models for predicting changes in litter and soil C are either ecosystem models that consider different C fluxes in the entire ecosystem, or decomposition models that focus on the turnover of the soil C pool. Examples of ecosystem models are the CO2Fix model developed at EFI in Finland (Masera, Garza-Caligaris, Kanninen, Karjalainen, Liski, Nabuurs, Pussinen, de Jong & Mohren, 2003) or the Coup model, developed in the research program LUSTRA in Sweden (Jansson & Moon, 2001). The CO2Fix V.2 is a multi-cohort, ecosystem-level model, based on carbon accounting of forest stands, including forest biomass, soils and products. Carbon stored in living biomass is estimated with a forest cohort model that allows for competition, natural mortality, logging, and mortality due to logging damage. Soil carbon is modelled for five stock pools, three for litter and two for humus. The Coup model is a process-based model, driven by abiotic factors.

Examples of decomposition models are Yasso (*e.g.* Liski *et al.*, 1999; Liski *et al.*, 2003b) and the Q model (Ågren & Hyvönen, 2003). However, decomposition models need to be supplemented by data on biomass and litter production (*e.g.* Liski *et al.*, 2002). The potential use of models in combination with micro-meteorological measurements is further described in the section 'Carbon fluxes – eddy-covariance measurements and modelling'.

A limitation of most of the models is that they do not pay full attention to the distribution of carbon fluxes to and from the various soil horizons. This reduces the reliability and usefulness of models for predicting changes. The reason is that the carbon pools in different horizons are affected by different conditions, and thus respond to management and environmental changes in different ways.

Concluding remarks on modelling

Many different kinds of model can be applied to the assessment of change in carbon pools. Some of them were outlined above. In addition, modelling approaches, in connection with eddy-covariance measurements, will be treated in a later section.

Moreover, although this report focusses on methods for estimating change on the basis of current and past data, it is important to point out that scenario analyses are important tools, in that they can be applied proactively to issues related to carbon sources and sinks in forests. To make such analyses, forecasting models of various kinds are needed.

Whether or not the forest ecosystem is a source or a sink, is a result of complex ecological and socio-economic interactions, where tree growth and soil processes act on the one side, and the processes that lead to management decisions, such as harvesting, act on the other. In Sweden, projects such as Heureka (*e.g.* Lämås & Eriksson, 2003) and LUSTRA (*e.g.* LUSTRA, 2002) aim at developing tools that can be applied to this type of integrated analysis.

Remote-sensing-aided carbon change estimation

The use of remote-sensing data is often proposed as a means to improve the accuracy of

estimates of carbon pools and fluxes. In this section, some procedures for estimation based on remote sensing are outlined, and possibilities and problems are discussed. The challenge in the Swedish case is to improve the estimates that are obtained using field sampling only. Since these have reasonably high accuracy, as described above, a major issue is to avoid introducing systematic errors when remote-sensing data are incorporated into the estimation procedures.

Use of remote sensing offers many possibilities; a comprehensive coverage would require a separate report. Thus the treatment is limited to a few cases in which medium-resolution satellite remote sensing (SPOT, Landsat, Terra MODIS) is combined with field sampling in a statistical manner. In addition, comments on other kinds of application are given.

One important basis is that remote sensing alone seldom is very useful for biomass estimation and similar purposes. Almost always, there is a need for field reference data, so that the specific spectral signature can be given an interpretation (*e.g.* Tomppo, 1993; Olsson, 1994; Nilsson, 1997).

Possibilities and problems

Theoretically, the combination of remote sensing and field data is by no means complicated. Many methods exist, *e.g.* stratification, multi-phase, or multi-stage sampling (*e.g.* Cochran, 1977). However, when working with satellite imagery for deriving estimates for forests, we face a number of practical problems:

- The geometry of images, as well as the level of precision of geo-referencing with GPS-equipment, implies problems in tying an individual pixel in the image to a specific field reference plot. Mismatches may in some cases lead to biased estimates.
- In pixel-based image estimation approaches, we also face problems with mixed pixels, especially when some of the features in a pixel have much higher reflectance than others. Field reference data for mixed pixels seldom are available to cover all cases that may be encountered.
- Our existing digital maps provide only rough estimates of what areas are forest and what areas are non-forest. Thus, to derive estimates for forests from image data, classification procedures must be adopted. In general, it is problematic (and impossible without a proper set of field reference data!) to set up classification procedures that lead to unbiased area estimates. This is a common problem, and source of error, when satellite imagery is used for deriving statistics for some larger region (*cf.* Bååth, Gällerspång, Hallsby, Lundström, Löfgren, Nilsson & Ståhl, 2002).

Different methods

Three different approaches to utilising satellite data for change estimation will be discussed:

(1) Stratification,
(2) Ratio- or Regression estimation,
(3) kNN estimation.

Stratification A simple and robust approach to using remote-sensing auxiliary data is to perform stratified sampling or post-stratification (*e.g.* Thompson, 1992). In this case, all pixels or segments of pixels from the image are classified as belonging to a certain stratum, and the field-work may then be allocated in an efficient manner to these. The general principle is that strata with large internal variation receive large samples.

If a network of plots already exists, as is the case with the Swedish NFI, post-stratification can be applied. Post-stratification will generally not be as efficient as stratification, since there is no longer an opportunity to allocate the plots in an efficient manner. Nevertheless, post-stratification may lead to major enhancements in the precision, as compared to the use field data only.

A pilot study was conducted (Ståhl *et al.*, 2002) to evaluate to what extent post-stratification increases the accuracy of estimates in comparison to the use of field data only. Three different procedures were compared in a region corresponding roughly to the county of Västerbotten:

- Stratification using image segments (clustered pixels with equal properties) with 1 ha minimum size (derived from Landsat TM);
- Stratification using segments with 5 ha minimum size (derived from Landsat TM);
- Stratification based on Terra MODIS pixels, 25 ha large.

With small segments, the theoretical efficiency can be expected to be high, whereas the number of practical problems arising from the assignment of plots close to a boundary of a certain segment can also be expected to be high. With larger segments, the practical problems will be less pronounced. The motivation for using entire MODIS pixels as units in the stratification was the simplicity of such a procedure, especially the simplicity with which the stratification for a large region can be established. However, since a certain reflectance value does not imply the same forest conditions in different eco-regions, stratification based on MODIS should also be based on some broad delimitation of eco-regions which comprise similar forest conditions.

Compared to the use of NFI field data only for estimation within the region, the standard errors for key parameters were reduced by 20–50% by the use of post-stratification. The best results were obtained when segmentation was based on Landsat TM data, and when segments down to 1 ha in size were allowed. Examples of results are shown in Table 2.

The above study concerned state estimation, in contrast to change estimation. If the figures in the table are taken as valid for the entire country, the estimates of change would also be improved by using post-stratification. However, at the national scale, the gain in precision would be less pronounced. Thus the use of remote-sensing auxiliary data can mainly be justified from the point of view of obtaining accurate estimates for limited regions.

The relative efficiency of using permanent plots will be somewhat reduced when stratification is applied. This is because the strata are similar, and consequently the correlation between the states on a plot at two different points in time will be lower. The magnitude of this depends on how homogeneous the strata are.

Table 2. *Standard errors of biomass estimates for all tree species, for deciduous trees, and for dead wood using NFI data only and using different kinds of post-stratification*

	Total	Deciduous	Dead wood
Method	St err (%)	St err (%)	St err (%)
NFI	5.4	9.2	11.5
MODIS*	3.6	6.7	9.5
Landsat, 5 ha	2.9	6.6	8.5
Landsat, 1 ha	2.8	6.6	8.3

Ratio- or regression estimation Satellite data could also be used for ratio- or regression estimation, in a similar way to their use in stratification (*e.g.* Cochran, 1977). A rough estimate of the total carbon quantity in each segment should also be derived, and used as auxiliary information. Although such procedures have not been studied, it is likely that they would yield some improvement compared to stratification, without the risk of (major) bias.

In the context of regression estimation, a study by Myneni *et al.* (2001) will be used as the basis for further discussion of potential risks involved in estimating carbon change by satellite-aided methods. In the study in question, satellite data were used to estimate the carbon sinks in forest vegetation in northern regions; the study showed that the sink is much larger than previously expected.

In the study, regression models were developed that linked satellite spectral values to certain values of carbon pools on the ground. Regression functions of this kind typically show quite large random variability. This variability is by no means entirely random; the errors are likely to be very much spatially correlated – implying that estimates for a certain region may have systematic errors. To estimate the total carbon pool over a large area, the regression models were applied to all pixels covering that area. In the specific study, the target area was a major part of the northern hemisphere.

If state estimates are made at two successive points in time, the change in carbon pools can be estimated. This was the approach adopted in the study. Even small systematic errors in estimates of the carbon stock may have a large impact on estimates of change. For example, in Sweden the carbon stock in the forest vegetation is *ca.* 1 Gtonne C. A 10% systematic error implies 100 Mtonne C. If an estimate of change is based on the difference between two state estimates, one of which has a 10% bias, the estimate of change will be extremely biased.

In addition to the general problems outlined above, *e.g.* that we do not know from the satellite images which areas are forest, certain other problems are also present in an application of this kind:

- A certain spectral value in a satellite image means different things in different parts of a region, a country, or the world. In the study, in addition to spectral values, latitude was incorporated into the regression models for assessing total biomass. It was thus implicitly assumed that, given a certain spectral value, the biomass in forest vegetation is the same in different parts of the world at the same latitude. If field samples were objectively distributed in a balanced way over the entire region to be assessed, this might not be a problem. However, it is a major problem when conditions in areas where field data are available, differ from those where field data are not available.
- With large pixels, such as those used in the study (*ca.* 1×1 km), most pixels will be mixed, *i.e.* they will contain both forest and other land-cover classes. Thus land-cover classes other than forest contribute to the spectral values in a manner which often is poorly known. In consequence, the regression functions become still more uncertain. In addition, the impact of land-cover classes other than forest may differ in different parts of the world, which adds to the unknown bias of the estimates.

kNN estimation In recent years, kNN estimation (Tomppo, 1990, Nilsson, 1997) has become a popular method for various purposes of estimation by means of medium-resolution images. The principle is that field data are assigned to all pixels in an image, based on nearness in the spectral feature space, between the pixel of interest and all pixels for which field reference data are available.

The kNN method is straightforward to apply, but it is quite difficult to assess the formal properties (bias, standard error) of the resulting estimates. Especially since it is difficult to control the systematic error component, the use of kNN for purposes of monitoring change in carbon stocks is not recommended when there is an option to derive high-precision, unbiased estimates from field data only. When only limited amounts of field data are available, the use of kNN or similar techniques would, of course, be meaningful.

Modelling NEE with satellite data

Today, no method based on remote sensing exists, that can estimate NEE from satellite data only. In this context, satellite data have mainly been used to estimate leaf-area index (LAI), a key parameter in most causal models. LAI may be estimated from remotely sensed data, for agricultural vegetation and grasslands in particular, but also to some extent for forests. The relationship between a reflectance index, NDVI, and LAI is fairly strong and general for agricultural crops and deciduous forests, but does not work well for coniferous forests. However, there are other indices, which are promising also for conifers (Eklundh, Harrie, & Kuusk, 2001). It is realistic to assume that LAI can be obtained from remote sensing, although it is known that images are saturated at rather low index values, and it is difficult to separate high values from intermediate values of LAI.

Another option is to estimate NPP (Net Primary Productivity) using satellite data, then combine this with a model for heterotrophic soil respiration to obtain NEE. It is well known that photosynthesis is strongly correlated to the amount of light absorbed in the visible part of the spectrum. On the basis of this, Monteith (1972, 1977) introduced the 'Production Efficiency Method' (PEM), which is very simple, and is well suited for exploiting reflectance data from satellites. In the PEM, NPP is estimated as the product of an efficiency coefficient, times the amount of absorbed radiation (APAR), minus the maintenance respiration. The APAR can be estimated from satellite reflectance data, while the efficiency coefficient must be determined from empirical data. The latter are probably dependent on stress factors, such as soil-moisture deficit and high and low temperatures. Attempts have been made to correlate the efficiency coefficient with threshold values of vapour-pressure deficit and temperature, and there is some promise in these attempts. However, more work must be done to develop this concept, before it can be used operationally.

Conclusions regarding the use of remote sensing

In countries such as Sweden, where a network of permanent forest inventory plots exists, the basic estimates of change based on field data will generally have reasonably high precision. In this case, rough regression-type estimates based

on satellite data are not very useful, since they are likely to have lower accuracy. For example, the size of systematic errors is generally unknown.

Satellite data may nevertheless be used to improve the quality of the estimates under these conditions, but care must be taken to use procedures that do not lead to deteriorated estimates. One possibility for improving estimates in this case is post-stratification, whereby image data are first segmented to reduce the problem of mismatches between satellite and field data.

In countries that lack other forest inventory data, the use of satellite information can, of course, be very valuable for obtaining estimates, *e.g.* of afforestation, reforestation, and deforestation.

Carbon fluxes – eddy-covariance measurements and modelling

Even if the NEE can be measured very accurately for a single stand by the eddy-covariance method, it is still not feasible to have a sufficient number of systems so that scaling-up to national level becomes meaningful. The costs for micro-meteorological measurements of this kind are too high.

It is, however, of interest to use such systems in combination with modelling. The fluxes measured by a number of eddy-covariance systems can be used to test, develop and calibrate different models, which then in turn are used in conjunction with other available data, to scale-up to the required area. This will give a high degree of flexibility, and the models can be adapted to the type of data available. It also allows the development of a prediction system, in which, for instance, different scenarios for management aimed at increasing the sink strength can be tested, or the effect of future climate change on the carbon balance can be analysed and quantified. A further advantage of a modelling system is that the carbon balance can be estimated with a far higher time resolution, as compared to an inventory-based system. This means that the annual development, or even monthly or daily progress, can be followed.

A scheme showing how such a model-based system could be built up is illustrated in Fig. 5. It is recommended to use different types of model, and to base the scaling-up on different types of data. This will provide a range of estimates for the NEE of Swedish forests, which will give an indication of the uncertainty of the estimates. In a first step, the models should be tested against measured NEE data by eddy-covariance at as many sites as possible. Here, historical data can be used, as well as data from sites other than Swedish ones, especially those from other Nordic countries, where site conditions are comparable. Flux measurements are currently in progress in *ca.* 15 Nordic forests, and many of them have multi-annual time series of data. In this step of the scheme, local weather data are used if the model requires such an input. In the second step, after the models have been verified and calibrated, scaling-up to the Swedish forest will take place. Depending on the type of model, either forest inventory data or remotely sensed data can be used. In both cases, additional information about soil conditions will be needed. If the model requires weather data as input, such data can be provided by a regional climate model driven by analysed data. Such a model is available in Sweden through the Rossby centre at the Swedish Meteorological and Hydrological Institute (SMHI). The model gives weather data with a spatial resolution of 44 km and a temporal resolution of six hours. The choice of model must be made after consideration of the availability of data. Below, some examples of models are given, which could potentially be suitable for this type of application. There are, however, several other models that could be used too.

An example of a causal model

Biome-BGC is a process-based model that simulates the storage and fluxes of water, carbon and nitrogen between the different components of the ecosystem and the atmosphere. The model was initially developed for forests, as Forest-BGC (Running & Coughlan, 1988) but has now been developed into a more general 'biome' model. The time step is daily, which is a good compromise between time resolution and time demand for simulations when applications are large-scale. Models with a daily time-step are recognised as describing day-to-day variation satisfactorily (Kimball, Thornton, White & Running, 1997). Daily maximum, daily minimum and daylight temperature, precipitation, daylight vapour-pressure deficit, daylight radiation and day-length are input variables. The soil is treated as one layer, and so is the canopy. The canopy is, however, divided into shaded and sunlit fractions. Photosynthesis is based on the

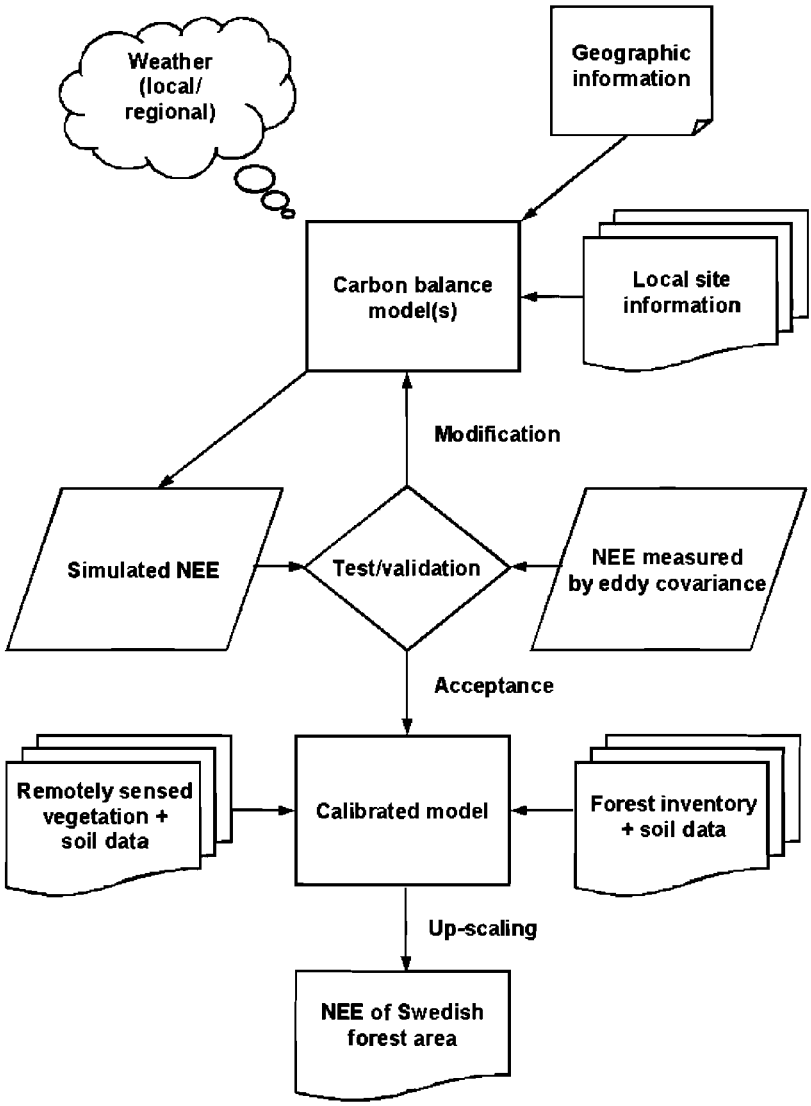

Fig. 5. A scheme of how a model-based system for estimation of NEE for the Swedish forest area could be organised.

Farquhar model (Farquhar, von Caemmerer & Berry, 1980). Canopy conductance to CO_2 and water vapour is controlled by air temperature, vapour-pressure deficit, radiation and soil-water potential. Stocks of carbon in the various compartments must be given as initial values. The amount of nitrogen is given by the C:N-ratios of the compartments.

Below, an example is given of a process-oriented model, which could be used as a tool for scaling-up. It should be emphasised that many models are available, and that it is a good strategy to use several models in assessments such as this, in order to grasp the range of uncertainties produced by the different model approaches. The Biome-BGC model can be found at:

http://www.ntsg.umt.edu

An advantage of this model is that it is widely used and will be continuously developed. A comprehensive literature study has been made to identify values for ecophysiological parameters that are representative for the major biomes on the earth (White, Thornton, Running & Nemani, 2000). Work is in progress in Sweden

to parameterise and test this model for boreal conditions. Although this is a fairly simple model, 43 parameters must be determined for each biome. Most of these are biome-specific, and need not be changed for different stands, although *e.g.* LAI is a key parameter, which is needed for each stand.

Current status of eddy-covariance measurements in Swedish forests

All eddy-covariance measurements going on today are part of different research projects. These are normally short-term, *i.e.* 2–3 years at most. Two sites have so far been able to operate for as long as eight and six years, respectively. These are Norunda in northern Uppland and Flakaliden in Västerbotten. NEE is at present measured at an additional five sites, all of which are situated in coniferous forests of different age and species composition. The measurements at Norunda, a drained site, show, interestingly, that forests can also be carbon sources over long periods of time. A recent analysis of the total greenhouse gas-exchange of drained forest shows that even if the drained area is small compared to the total forest area, the global warming effect is large, and needs to be considered (Klemedtsson, Weslien, von Arnold, Ågren, Nilsson & Hånell, 2002).

The measurements at Norunda also show that variation between years is quite large. During the period 1995–2000, the mean annual NEE was $+55.9$ g C m^{-2}, with a standard deviation of ± 48.6 g C m^{-2} (Lankreijer, Lindroth, Grelle & Vesala, 2003). Variation between years was of similar size at Flakaliden and Hyytiälä in Finland, although these two forests showed mean uptakes of -139 and -235 g C m^{-2}, respectively. The first preliminary results from NEE measurements in a chronosequence of stands, show that stand age is a key parameter in determining annual NEE. It is, therefore, important that stands used for calibrating the models are representative not only as regards age distribution, but also as regards soil properties, which is another sensitive factor in the simulations.

Errors in eddy-covariance measurements

As with all methods, there are uncertainties and errors associated also with eddy-covariance measurements. It is quite complex to perform a full error analysis, without introducing the theoretical background of the measurements; therefore, a more qualitative outline is given here. Errors in eddy-covariance measurements can be classified into systematic and random errors, as summarised by Grelle (1997); see Table 3.

The division into random and systematic errors is not as clear as it might appear from Table 3. Some of the errors can be of both types: for instance, inadequate sensor height can cause a systematic error if the sensor is above the constant flux layer, and thus 'sees' an area other than that which is intended to be measured; whereas if the sensor is positioned within the constant-flux layer, but too close to the surface, *i.e.* is in the roughness sub-layer, then the error will be of the random type. For long-term measurements, such as are discussed here, the random errors will diminish; for CO_2 flux on an annual basis, a typical random error above forest will be 2% (Grelle, 1997). If, for instance, a five-year value is sought for, the random error will be practically negligible. This relates to the errors for the estimates at the single location.

As in all types of measurement, systematic errors are always more difficult to handle, and are also more serious, since they can cause an unknown bias in the estimate. An unresolved problem concerning flux measurements is the 'night-time problem'. It appears during stable night-time conditions, when the flux measurements can indicate a zero flux even when respiration is fully active, and should produce a positive flux. Part of this bias between the observed flux above the canopy and the production rate of CO_2, can be explained by increasing storage of CO_2 below the measurement level. Part of the discrepancy has been suggested to be caused by a katabatic flow in

Table 3. *Errors in eddy covariance measurements (after Grelle, 1997)*

Random errors	Systematic errors
One-point measurement	Instrumental calibration error
Surface heterogeneity	Inadequate correction function
Inadequate average time	Inadequate functioning during stable conditions
Random signal noise	Inadequate sensor height
Non-stationarity	Sensor separation
	Flow distortion

the trunk space, but this has not so far been convincingly demonstrated. Our own results from an analysis of the 'night-time problem', indicate that the method functions adequately during stable conditions, and that the problem may have been exaggerated. However, care should be taken and more research is needed to resolve this matter.

Grelle (1997) estimated the systematic instrumental error for the CO_2 flux at 5.3% for a typical forest set-up. Assuming that the errors are independent, the total error on an annual basis would typically be of the order of 6%. This estimate is without consideration of the potential night-time error, and it should probably be considered a minimum error.

Accuracy of different methods for determining changes in forest carbon pools – a summary

In this section, the qualitative and quantitative aspects of the various methods for estimating changes in forest carbon pools are summarised. The summary comprises most of the methods outlined in the earlier parts of this section. Statements are given from the point of view of the currently available monitoring programs and systems. The summary is shown in Table 4.

Discussion and conclusions

General issues

The study shows that there are several options available for assessing changes in forest carbon pools in Sweden, and that there are many possibilities for improving the current Swedish reporting. Considering the different pools of the Kyoto Protocol, adequate data and methods are available for assessing changes in above- and belowground biomass. For these pools, only minor improvement of the methodology is needed. Furthermore, most of the data required for this purpose are available from the Swedish National Forest Inventory (NFI). The traditional method for reporting changes in this case has been to estimate growth and removals of tree volume separately, and to convert the difference thus obtained to amount of carbon by applying various conversion and expansion factors. An interesting alternative to this approach is considered to be direct estimation of change, based on data from repeated measurement of permanent sample plots. This method probably is more accurate than the one currently used. Ideally, both methods can be applied and combined into a single estimate, by appropriate weighting of the two different estimates.

Table 4. *Summary of important features of the proposed methods for estimating changes in carbon pools in Swedish forests. The estimates are based on data from currently available monitoring programmes and systems in Sweden, and to some extent on assumption*

Method	Standard error of an estimate of five years change, Mtonne C*	Risk of systematic error	Ready for application?
Tree biomass – Summation of growth minus cuttings ('IPCC')	11–(25%)	Moderate	Yes
Tree biomass – Direct change estimation (temporary plots)	18–(40%)*	High	Yes
Tree biomass – Direct change estimation (permanent plots)	6–(13%)*	Moderate	Yes
Tree biomass – Combined estimation ('IPCC' and permanent plots)	5–(11%)*	Moderate	Almost
Tree biomass – Remote sensing aided change estimation (post-stratification)	5–(10%)*	Moderate	No
Humus layer (example for south Sweden) – Direct change estimation	0.75–(25%)	High	Yes, after minor modifications
Mineral soils to 0.5 m – Direct change estimation	8.5–(40%)	High	Yes, after major modifications
Eddy covariance measurements – all pools	Not assessable	Moderate-High	No

*The total change of carbon in biomass for a 5-year period is set to be 43 Mton C, which is based on the change during the period 1990–2000. 95% confidence intervals are obtained as 43 ± 1.96*Standard error.

As regards changes in soil carbon, further improvement of methods and monitoring principles are needed, if these changes are to be based on repeated measurements. Current practices and methods are promising, but it is unclear whether the systematic-error component of change estimates can be kept at a sufficiently low level. In an international perspective, very few countries have a long-term monitoring programme like the Swedish Forest Soil Inventory (MI). Sweden thus has an opportunity to set standards concerning how soil-sampling programmes should be set up to assess changes in soil organic carbon. An alternative to repeated soil sampling would be to base the reporting on modelling. However, since the Swedish MI will continue to operate, regardless of Kyoto reporting, data on stock changes will continue to be available.

Although an assessment of dead wood volume is conducted within the NFI, currently there is a lack of knowledge for assessing changes in this carbon stock. However, work is in progress, and within a few years, functional relationships that relate the volume and decomposition status of dead trees to amounts of carbon, will be available.

As regards the carbon pool in litter, no national-level data are currently available.

Eddy-covariance measurements are a valuable means for verification and for further improving our understanding of fluxes of greenhouse gases, for example with respect to the temporal variability of fluxes. In addition, these measurements provide information about gases other than CO_2. However, at present it is not possible to use this kind of measurement for improving the national-level estimates of change in carbon pools. The reasons for this are that the network of measurement stations is very sparse, and that no methods are available for scaling-up from site level to the national level.

The accuracies found in our studies of changes in forest biomass and soils are generally lower than those found for other sources and sinks of greenhouse gases in national inventories (*e.g.* Rypdal & Winiwarter, 2001). Whether or not these accuracies are sufficient, is a difficult question. Currently, there are no recommendations from IPCC concerning a given level of accuracy. Nor are such recommendations likely to be presented in the near future. Instead, each country is requested to provide as accurate estimates as possible, considering national circumstances (*cf.* IPCC, 2000b).

Although this can be considered to be reassuring, we wish to propose an alternative approach, whereby information about the uncertainty of estimates is actually utilised. This approach would be that only sinks that are 'certain' would be permitted for inclusion in the reporting. This could be based on 95% confidence intervals of estimated changes. If such an interval overlaps zero, the conclusion would be that the pool cannot be accounted for as a sink. Otherwise, the lower limit of the confidence interval could be used as the accountable part of the sink. For example, if a country reports a confidence interval 5 ± 3 Mtonne C as the change in forest management areas, only 2 Mtonne C – instead of 5 Mtonne – would be allowed in the accounting. A prerequisite for this is that careful checks are made to ascertain that there are no systematic errors in data.

Need for improvement of the national monitoring programmes

For estimating changes in the forest carbon pools in Sweden, the NFI and the MI are of prime importance. However, no major changes in these inventories are foreseen as resulting from the requirements to improve the reporting of changes in carbon pools under the Kyoto Protocol. The basic measurements will continue to be very useful. However, a number of minor modifications and developments would contribute to making the inventories better suited for carbon reporting. For the NFI, such modifications include:

- The reporting of changes in biomass carbon pools would benefit from a higher share of permanent plots. Such a shift from temporary plots to permanent plots is, however, already scheduled for the inventory that starts in 2003. This shift is planned to meet the demands for change estimates also in areas other than carbon reporting.
- New routines for control assessment need to be established, in order to obtain scientifically sound estimates of the systematic errors of

the estimates. To achieve high precision in the control, an increased number of sample plots should be checked.

- The estimates of harvested volumes need to be improved, by introducing stricter routines for these assessments. The ambition must be to obtain unbiased harvest estimates, although the random errors in estimates for single years may continue to be high.
- The biomass functions used to calculate the biomass of individual trees from the basic measurements should be improved, and be developed for all major species. Currently, there is a lack of functions, especially for root biomass.
- Functions need to be developed for the assessment of carbon in dead wood. Currently, only the volume of dead wood is assessed. Estimation of the carbon mass in dead wood is somewhat complicated, since the density of carbon varies depending on decomposition stage of the wood. However, it is judged that it will be sufficient to use rather crude conversion functions.
- New routines could potentially be introduced to quantify changes in the carbon pools in aboveground vegetation other than trees. This also would require the development of a set of regression functions for this purpose.
- If necessary, coarse litter could be assessed by specific local line-intersect sampling techniques. However, such measurements would be rather expensive.

The Forest Soil Inventory can be used for monitoring carbon stocks and stock changes in forest land at a national and regional level in Sweden. However, the system is not optimised for meeting the Kyoto Protocol demands, and some modifications in field-sampling procedures should be undertaken.

A stratified sampling design of permanent plots is suggested as one option. The number of plots should be optimised with respect to expected changes. However, since the MI is conducted for a number of reasons other than carbon-change assessment, our recommendation is to retain the current allocation of plots on which soil samples are taken.

It is suggested that the amount of carbon be monitored in the humus layer and in the mineral soils down to a reference depth of 50 cm. However, in the case of peat soils with an organic horizon deeper than 50 cm, it is suggested that the entire peat layer be included in the monitoring.

Concrete suggestions for modification of the MI are:

- Humus layer cores should be taken in a more objective manner, to avoid potential systematic errors from purposive selection of core locations. Litter samples might be taken simultaneously with the humus samples, using the same basic approach.
- Mineral-soil samples would continue to be taken from certain predefined layers and depths in randomly situated soil pits. However, inclusion of a new soil sample at a randomly located depth would improve the possibility of quantifying various compounds in the soil.
- The amount of stones and boulders should be measured according to the principles of Viro.
- The total thickness of peat layers should be measured. In addition, peat samples should be taken at different depths on peatlands.

Intercomparison of methods – a tool for verification

The Kyoto Protocol states that the estimated changes in carbon pools shall be verifiable. To verify means to make certain that something is true, and the closest we can come to verifying an estimate of a change in carbon pools is probably to compare estimates made by independent methods, designed to measure the same characteristic. The eddy-covariance method provides a means for such comparisons. With the eddy-covariance method, the net flux of CO_2 between the atmosphere and a forest can be measured with a half-hourly time resolution. The integral over time of the flux, plus the transport of dissolved organic carbon (DOC) out of the system, should be comparable to the change in the pools estimated over the same time period, provided that all pools are measured. Since DOC transport is known to be fairly small in most systems, it can be neglected in a first approximation; thus, integrated fluxes measured by eddy-covariance can be directly compared to measured changes in the pools on the same site.

Eddy-covariance measurements are currently in progress in several forests in Sweden, as part of different research projects, but no sites exist for continuous long-term monitoring.

We suggest that a number of eddy-covariance sites are set up in a monitoring and verification programme. At these sites, the methods applied for monitoring of carbon pools should be used in exactly the same way as is done for the purpose of reporting. Results from these measurements can be used for testing and development of other methods based on modelling and remote sensing.

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Annex – Summary of the Nordic workshop '*Monitoring and reporting of carbon stock changes in forest land*'

The workshop was held at Krusenberg, Sweden, 31 October–1 November 2002. The aim was to discuss a preliminary version of the study presented in this report. Conclusions from the workshop sessions are provided below.

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Background to the workshop

The Swedish Energy Agency has funded a study on 'Quantification of changes in forest carbon pools – methodological options for Sweden'. As an integrated part of the project, a workshop was organised between 31 October–1 November, 2002. The aim of the workshop was to evaluate the conclusions in a background report, and to put the results into an international perspective.

Most of the time was spent in three working groups, which focussed on: *(i)* evaluation of available methods and systems – their strengths and weaknesses in terms of accuracy, reliability, regional applicability, costs, verifiability, *etc.*; *(ii)* possible improvements and the need for future research.

The working group reports are summarised below.

Working group 1: Inventory of carbon stock changes – biomass and dead wood

Q 1. Which methods are suitable for application in national accounting systems? The WG focused on Tier 3-like methods (*cf.* IPCC reports on Good Practice Guidance) and the situation in the northern part of Europe.

National Forest Inventories (NFIs)

Are based either on estimation of growth minus drain or measurements of the stocks at two different times. NFIs can be used for above-ground and belowground biomass. Both pools can be covered either through BEFs (Biomass expansion factors) or individual-tree biomass functions.

Dead wood is included in some inventories, but the methodology needs improvement, particularly in terms of conversion from volume to mass of carbon.

Resolution in space and time depends on the sampling scheme, but in general the NFIs can be used only for larger areas (regions), and through interpolations and extrapolations over several years (3–5 years).

NFI, in combination with remote sensing, can improve resolution in time and space.

Remote sensing, models and flux measurements

These methods should not be used alone, if an NFI exists, owing to difficulty in achieving a sufficient degree of accuracy, but they are complementary tools in using the results from inventory methods. (See further Working Group 3)

Q 2. What is the accuracy and reliability of the methods? What is acceptable accuracy? This question was addressed in relation to measurement of stock changes (NFI).

The accuracy for stem volume is high, and can be estimated (in Nordic NFIs, typically of the order of 1% standard error). The estimates for non-stemwood biomass (belowground biomass, ground vegetation) are less accurate, since BEFs (Biomass expansion factors/functions) are used. These could be a function of tree age, diameter, site fertility and other factors. This pool is sometimes underestimated, but it can be improved through (labour-intensive) studies.

Dead wood estimates can be very inaccurate. In terms of acceptable accuracy, anything is acceptable as long as the best available methods and data are used (according to IPPC-guidelines), which is a matter of national circumstances. Requirements focus more on documentation. Verifiers need to be able to track what has been done.

Q 3. Which are the main sources of error (random and systematic)?

The main errors differ in part, depending on whether one uses: *(i)* NFI-change estimates from two measurements; *(ii)* NFI-growth minus drain. Some of the problems listed below make a smaller difference in approach *(ii)* than in *(i)*, because they cancel out when two occasions are compared. It is often good to combine the two methods, since this will improve the understanding of why a certain change has occurred.

Some major errors in using NFIs-growth minus drain were discussed:

- Harvested trees are of different age, and biomass expansion factors are averaged over the whole growing stocks.
- Uncertainty in the harvest statistics (*e.g.* under-reporting for tax reasons). Non-commercial (*e.g.* fuelwood) uses of wood are only guessed.
- Import and export are difficult to handle in terms of accuracy, which will affect the harvest statistics.

Q 4. Which methods are applicable in which regions?

Some major conclusions: Direct measurement methods can only be applied where frequent NFI is carried out. The systems used are *(i)* continuous inventories (the Nordic countries) or *(ii)* rotating inventories (over 2–3 years and then a break for some years (as applied in Germany and Austria), which has implications for the results and the accuracy.

In areas without NFI data, the use of flux measurements, remote sensing and modelling

approaches may be useful in national accounting systems. The choice of method will depend on the density of activities and on the size of the country.

Q 5. What are the visions/potentials for improvements of the outcome in a 4–10-year perspective?

It is crucial to meet the forthcoming reporting requirements and their timescales. Further R&D should address two different time frames: *(i)* short term issues (*e.g.* IPCC Good Practice Guidance). It is important to feed in Nordic experiences; *(ii)* long-term issues with the aim of addressing the needs in a second commitment period, from 2012 onwards.

Some of the conclusions regarding R&D needs in a 4-year perspective were:

- Improved biomass expansion factors and biomass functions through research projects.
- Methods for estimating rate of stock changes in dead wood and litter.
- More useful remote sensing methods, both for scaling-up and for detection of change.
- Better knowledge of all types of error, and particularly systematic errors.
- Improved information on removals (harvest and fuelwood).
- Established links to international monitoring systems (*e.g.* FAO, ICP Forest).
- A closer link between the UNFCCC inventory people, national forest inventory people, and the research community.
- Those issues were also identified as urgent topics for research, aiming at an improvement of the monitoring and reporting system.

Some conclusions in a ten-year perspective were:

- Possibility of controlling systematic errors in general.
- Independent verification methods are available.
- Other environmental effects of LULUCF activities (*e.g.* biodiversity, groundwater) are considered.

Working group 2: Inventory of carbon stock changes – soil and litter

Q 1. Which methods are suitable for application in national accounting systems?

There are no harmonised methods used to estimate changes in carbon stock in soils over time. The ideal way to plan a suitable inventory is *(i)* to determine the expected change rate, and *(ii)* to determine the spatial variability of the change rate. With this information, and a given or required degree of precision, the level of density of an inventory grid can be determined. However, today we are confronted with existing soil-monitoring systems, which represent various methodologies and purposes. One approach to improving accuracy and to increasing cost-efficiency, is to apply a stratification scheme.

Repeated intervals of inventory of 10–30 years are necessary to detect soil changes. The reporting system requires an accounting system by the end of 2006. Soil inventories do not fulfil that requirement, because of a common repeated inventory period of ten years. However, interpolation and extrapolation can be used.

With stratification, sampling might be adapted to reduce the time interval required to detect changes.

Peat soils, which cover large areas in some countries, are not included in inventories, not even in the Nordic countries.

The sampling procedure needs standardisation in terms of *(i)* soil depth to be sampled; *(ii)* sampling of litter, which should be included in the inventory of soils; *(iii)* approaches to deal with coarse organic material and stoniness.

Q 2. What is the accuracy and reliability of the methods? What is acceptable accuracy?

For Sweden, according to the background report, the existing inventory (given a return interval of ten years) is able to detect C stock changes in a reliable manner. The standard error is estimated at 25–40%. A similar assessment is not known from other countries. Other conclusions are:

- An integrated approach should be investigated. Additional information beyond the scope of national soil inventory (*e.g.* models, scenarios long-term experimental sites, *etc.*) is necessary to provide the necessary information.
- No soil inventory is available which distinguishes the various sources of error. Rather, the overall statistics are determined for the various strata (soil types, production class, forest stand type, climate region).
- Repetitive sampling should be carried out on permanent plots

Q 3. What are the visions/potentials for improvement of the outcome in a 4–10-year perspective?

- Initiate systematic cooperation within the Nordic countries, to improve monitoring methods, particularly how to deal with stones and coarse organic material.

- Develop methods to stratify according to the reporting requirements, in order to improve the accuracy of regional soil carbon estimates.
- Integrate different approaches (*e.g.* modelling of humus dynamics) to meet the requirements.
- There is a great need to monitor changes in the carbon stock in peatlands, since these might change from sinks to sources (drainage, climate change).

Working group 3: Other methodologies – their role in the implementation of accounting systems

The working group focussed its discussions on the role of non-statistical methods (modelling, remote sensing, flux measurements and manipulation experiments), and how the results could be used for accounting, verification or both, as targeted information with the aim of being complementary to those inventory methods discussed in working groups 1 and 2.

Q 1. What are the characteristics of these methods – their role and usefulness?

Modelling

This cannot be used for reporting alone, but could be applied for scaling-up and updating of inventory data, through interpolation and extrapolation in space and in time (fill gaps between inventories). Other uses are:

- System studies – understanding the processes.
- Scenario analysis and sensitivity analysis.
- Help in sampling design (statistical sampling, choice of pools, *etc.*).
- Inclusion of other trace gases.
- Models are used in combination with flux measurements, remote sensing and manipulation experiments. Different models are used in different regions in Europe.

Flux measurements

These cannot be used to cover regions or countries, but they are useful for:

- Understanding the processes (*e.g.* interannual variation), particularly if other aspects are studied at the same time; and for scaling-up from compartment studies.
- Validation of pool changes (inventories); quality control.
- Quantification of the impact of various silvicultural measures (*e.g.* thinning).
- Verification of whether changes are natural or human-induced.
- Test and development basis for modelling efforts.
- Results could be used in data assimilation (Flux, NFI, Remote sensing, Modelling).

Remote sensing

This should not be used to estimate carbon directly if NFIs are available, but it is useful for:

- Pre- and post-stratification to enhance the accuracy of inventory data. It might not help to estimate NFI tree biomass, but it can be used for updating of accounting results to annual intervals.
- Stratification of landscapes for soil inventories.
- Updating of field-measurement based estimates (together with growth models).
- Area estimation, including independent estimates of clearfelled areas.
- Accounting of stock changes in areas where NFIs are not available (estimation of Leaf Area Index and direct measurement of changes in carbon stock, but accuracy is often low).

Manipulation experiments

These do not aim at national reporting, but could help to:

- Quantify effects of different silvicultural measures and the impact of environmental changes (*e.g.* Nitrogen and CO_2-fertilisation).
- Factor out direct and indirect human effects.
- Understand processes.

Q 2. Are these methods suitable for application in national accounting systems?

In regions with an NFI, these methods should not be used alone, because of difficulties in achieving sufficiently high accuracy in scaling-up to national level, but the current inventories alone are not themselves sufficient. These other methods could be used to improve the quality of inventory data. National accounting systems need to have an integrated approach, whereby these methods can be complementary (sampling strategy, understanding processes, factoring-out natural and direct and indirect human-induced effects, verification, *etc.*).

In regions without NFI data, the use of satellite information, flux measurements and modelling approaches may be useful in national accounting systems. Remote sensing can be used if it is combined with field sampling. A combination of different inventories with models is also a way of using available sources more efficiently.

Q 3. Which are the major R&D challenges?

- Development of better integration between disciplines and approaches is needed. It is also urgent to have a close link between scientists and practical people, working with national inventories.

- Creation and maintenance of common databases and common protocols (harmonisation of data and methods).
- Develop better biomass expansion factors and equations; better information on deadwood, soil carbon and litter; more input data for models with spatial and temporal resolution.
- Use of common test sites for validation of different methods and approaches.
- Nordic countries should collaborate to cover, *e.g.* main forest site types, belowground biomass, and to making a joint contribution for the whole boreal region.