Dissertationes Forestales 50

Country-scale carbon accounting of the vegetation and mineral soils of Finland

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Academic dissertation

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ABSTRACT

The increase in global temperature has been attributed to increased atmospheric concentrations of greenhouse gases (GHG), mainly that of CO_2. The threat of severe and complex socio-economic and ecological implications of climate change have initiated an international process that aims to reduce emissions, to increase C sinks, and to protect existing C reservoirs. The famous Kyoto protocol is an offspring of this process. The Kyoto protocol and its accords state that signatory countries need to monitor their forest C pools, and to follow the guidelines set by the IPCC in the preparation, reporting and quality assessment of the C pool change estimates.

The aims of this thesis were i) to estimate the changes in carbon stocks vegetation and soil in the forests in Finnish forests from 1922 to 2004, ii) to evaluate the applied methodology by using empirical data, iii) to assess the reliability of the estimates by means of uncertainty analysis, iv) to assess the effect of forest C sinks on the reliability of the entire national GHG inventory, and finally, v) to present an application of model-based stratification to a large-scale sampling design of soil C stock changes. The applied methodology builds on the forest inventory measured data (or modelled stand data), and uses statistical modelling to predict biomasses and litter productions, as well as a dynamic soil C model to predict the decomposition of litter.

The mean vegetation C sink of Finnish forests from 1922 to 2004 was 3.3 Tg C a^{-1}, and in soil was 0.7 Tg C a^{-1}. Soil is slowly accumulating C as a consequence of increased growing stock and unsaturated soil C stocks in relation to current detritus input to soil that is higher than in the beginning of the period. Annual estimates of vegetation and soil C stock changes fluctuated considerably during the period, were frequently opposite (e.g. vegetation was a sink but soil was a source). The inclusion of vegetation sinks into the national GHG inventory of 2003 increased its uncertainty from between -4% and 9% to ± 19% (95% CI), and further inclusion of upland mineral soils increased it to ± 24%. The uncertainties of annual sinks can be reduced most efficiently by concentrating on the quality of the model input data. Despite the decreased precision of the national GHG inventory, the inclusion of uncertain sinks improves its accuracy due to the larger sectoral coverage of the inventory. If the national soil sink estimates were prepared by repeated soil sampling of model-stratified sample plots, the uncertainties would be accounted for in the stratum formation and sample allocation. Otherwise, the increases of sampling efficiency by stratification remain smaller.

The highly variable and frequently opposite annual changes in ecosystem C pools imply the importance of full ecosystem C accounting. If forest C sink estimates will be used in practice average sink estimates seem a more reasonable basis than the annual estimates. This is due to the fact that annual forest sinks vary considerably and annual estimates are uncertain, and they have severe consequences for the reliability of the total national GHG balance. The estimation of average sinks should still be based on annual or even more frequent data due to the non-linear decomposition process that is influenced by the annual climate. The methodology used in this study to predict forest C sinks can be transferred to other countries with some modifications. The ultimate verification of sink estimates should be based on comparison to empirical data, in which case the model-based stratification presented in this study can serve to improve the efficiency of the sampling design.

Keywords: carbon, biomass, forest inventory, greenhouse gas inventory, litter, model, Monte Carlo, soil, stratified sampling, uncertainty

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Vantaa, October 2007

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LIST OF ORIGINAL PUBLICATIONS

This thesis is based on the following publications:

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II. Peltoniemi, M., Mäkipää, R., Liski, J., Tamminen, P., 2004. Changes in soil carbon with stand age – an evaluation of a modelling method with empirical data. Global Change Biology 10, 2078-2091.

III. Peltoniemi, M., Palosuo, T., Monni, S., Mäkipää, R., 2006. Factors affecting the uncertainty of sinks and stocks of carbon in Finnish forests soils and vegetation. Forest Ecology and Management 232, 75-85.

IV. Monni, S., Peltoniemi, M., Palosuo, T., Mäkipää, R., Lehtonen, A. & Savolainen, I. 2007. Uncertainty of forest carbon stock changes - implications to the total uncertainty of GHG inventory of Finland. Climatic Change 81, 391-413.

V. Peltoniemi, M., Heikkinen, J. & Mäkipää, R. Stratification of regional sampling by model-predicted changes of carbon stocks in forested mineral soils. Silva Fennica 41, 527-539.

The publications are referred to in the text by their roman numerals.

AUTHOR'S CONTRIBUTIONS

I am solely responsible for the summary of this thesis. **I**) I participated in the planning, preparation and analysis of the results of this manuscript. I wrote parts of the text in the material and methods sections and commented on the manuscript. **II**) I was responsible for the planning of the study, the preparation and analyses of the results, and for writing the manuscript. **III**) I was responsible for the planning of the study, the preparation and analyses of the results, and writing the manuscript. **IV**) I participated in the planning of the study, prepared the methodology and results for the estimation of uncertainties in the forest sector and participated in the analysis and writing of the manuscript. **V**) I was fully responsible for the preparation of the results, and for most of the writing. I was partly responsible for most of the planning and analyses of the study.

ABBREVIATIONS

Abbreviation	Unit	Explanation
BEF	kg m^{-3}	Biomass expansion factor
A	kg C a^{-1}	Simulation uncertainty in Study V
b	kg	Biomass
C	kg	Carbon
CH_4		Methane
CO_2	kg	Carbon dioxide
CO_2-eq	kg	Carbon dioxide equivalent (in terms of radiative forcing)
COP		Conference of Parties (of UNFCCC)
DOC	kg	Dissolved organic carbon
dbh	m	Tree diameter at breast heigth
G		Number of strata in Study V
GHG		Greenhouse gas
GL		Guidelines (of IPCC)
GPG		Good practice guidance (of IPCC)
HFC		Hydrofluorocarbon
IPCC		Intergovernmental Panel on Climate Change
L	kg a^{-1}	Litter production
LULUCF		Land use, land-use change and forestry
m		Number of soil samples per plot in study V
N_20		Nitrous oxide
NDVI		Normalized difference vegetation index
NEP	kg C m^{-2} a^{-1}	Net ecosystem production
NFI		National forest inventory
NIR		Near infrared radiation
NPP	kg C m^{-2} a^{-1}	Net primary production
PAR	W m^{-2}	Photosynthetically active radiation
PCF		Perfluorocarbon
ppm		Parts per million
r	a^{-1}	Biomass turnover rate
RS		Remote sensing
SD		Standard deviation or standard deviation of the mean
SF_6		Sulphur hexafluoride
SOC	kg m^{-2}	Soil organic carbon
SOM	kg m^{-2}	Soil organic matter
T	a	Mean life span of biomass component
TR	a^{-1}	Turnover rate
UNFCCC		United Nations Framework Convention on Climate Change

CONTENTS

1 INTRODUCTION

1.1 Climate change and forests

Atmospheric CO_2 concentration has increased ~ 36% from pre-industrial times (280 ppm in 1750) to its highest in 420 000 years (380 ppm equivalent to 805 Pg C in 2005), and likely even its highest in 20 million years (Houghton et al., 2001, p. 185; Houghton, 2007). The average rate of increase in atmospheric CO_2 concentration in the 1990s was 1.5 ppma-1, equivalent to 3.2 Pg a^{-1} of C, and has shown considerable variability over the years (1.9–6.0 Pg a^{-1}).

The primary cause of this increase in atmospheric CO_2 concentration is the intensive use of fossil carbon. The use of fossil fuel (plus cement production) emitted on average 6.4 ± 0.3 Pg a^{-1} of C to atmosphere in the 1990s, of which oceans sequestered 2.2 ± 0.7 Pg a^{-1} of C (Houghton et al., 2001, p. 185; Houghton, 2007). Terrestrial ecosystems must have sequestered a net amount of 1.0 ± 0.8 Pg a^{-1} of C in the 1990s, otherwise the balance of the global C cycle fails to close. This net amount can be further divided into two components: one due to land-use change and the other due to terrestrial CO_2 sequestration elsewhere.

Changes in land-use have led to terrestrial emissions of CO_2, mainly as a result of the depletion of vegetation and soil C stocks due to deforestation. In the 1990s, global emissions due to the changed land-use were 1.6 ± 0.8 Pg a^{-1} C. This means that a sink of 2.6 ± 1.1 Pg a^{-1} C must be allocated elsewhere in the terrestrial ecosystem (Houghton, 2007). The residual terrestrial C sink is highly uncertain; many think that the residual sink resides in the enhanced growth of vegetation and enhanced sequestration of C into soil, but may also be due to uncertainties of the other estimates.

Studies have estimated that, in the early 1990s, forest ecosystems in the northern hemisphere were a sink for 0.6–0.7 Pg a^{-1} of C in the early 1990s, approximately half of which is in vegetation and half is in soil (Goodale et al., 2002). These estimates deviate considerably from the atmospheric inversion estimates for sinks in northern hemisphere mid-latitudes (2.1 ± 0.8 Pg C a^{-1}) and land-use change estimates (0.03 Pg C a^{-1}) (Houghton, 2007).

The balancing of the global C cycle is important for the scientific understanding of the global circulation of C. Land-based monitoring of forest C stocks thus serves this purpose, fulfilling this goal, requires more accurate estimates of forest C stock .

The sink of forests can be enhanced 1) by increasing the storage of carbon, either by increasing the area or mean carbon density of forests until the stock and available area reach their high limits; and 2) by producing bioenergy to substitute for the use of fossil carbon. A third means would be to store fresh biomass in geological reservoirs, a practice that could become feasible with future rises in CO_2 prices.

Increasing stocks of forest C and producing a maximal amount of bioenergy affect the structure of forests differently. When forests reach their maximum density, biomass production decreases. Monitoring forest C stocks on a regional scale is needed to use forests in a sustainable manner in order to mitigate climate change. The most attractive solution in the short term may not be the most sustainable in the long term.

1.2 Short history of forest-related climate treaties

The importance of accounting for the greenhouse gas emissions of countries was first acknowledged by the United Nations Framework Convention on Climate Change (UNFCCC) in Rio de Janeiro in the United Nations Conference on Environment and Development (UNFCCC, 1992). The monitoring and reporting of forest emissions/sinks were included in the Climate Convention. As a legally non-binding agreement, the Climate Convention preceded the later, legally-binding Kyoto Protocol approved in the 3rd COP (Conference of Parties [of UNFCCC]) meeting (UNFCCC, 1997). The Kyoto Protocol states that emissions should be reduced to 6-8% below 1990 levels during the period 2008-2012, and that forests could serve as a mitigation option for climate change in the first commitment period (2008-2012). Recalling this decision, the follow-up COP 6 meeting held in Bonn, sought to define a set of rules governing this decision (UNFCCC, 2001a). Annex I countries of the protocol can compensate for emissions with forest C sinks to a limited extent, meaning each Annex I country has a specific 'cap' that sets the upper limit for emissions compensation. For Finland, the 'cap' was set to a low value of 0.16 Gg C, a small percentage of the current forest sink (UNFCCC, 2001a; UNFCCC, 2001b; NIR, 2007). Since the credited sinks may compensate for emissions, a need exists for the reliable estimation of forest C sinks that are sufficiently accurate and precise to serve as a basis for emissions compensation. Consequently, at the COP 7 meeting held in Marrakech, Morocco, the member nations agreed to include uncertainty estimates (i.e. confidence intervals) in greenhouse gas inventories, including the forest C inventory (UNFCCC, 2001c).

The IPCC (International Panel on Climate Change) was established by the World Meteorological Organization (WMO) and the United Nations Environment Programme (UNEP) to provide a comprehensive, objective, open and transparent basis of scientific, technical and socio-economic information relevant to understanding the scientific basis of the risk of human-induced climate change, its potential impact and options for its adaptation and mitigation. The IPCC has provided guidelines (GL) for the estimation and reporting of GHG emissions and removals for all sectors (IPCC, 1996). The IPCC GL 1996 were later appended with general guidance for uncertainty management (IPCC, 2000) and with good practice guidance (GPG) for land use, land-use change and forestry sectors (LULUCF) (IPCC, 2003). The IPCC GPG 2003 provides more detailed guidance on how changes in five carbon pools (above- and belowground biomass, deadwood, litter and soil organic carbon) should be reported for the LULUCF sector. During the first commitment period, the IPCC GL and GPG will be used, but the IPCC has anticipated the need for new and revised guidelines for post-Kyoto commitment periods (IPCC, 2006).

According to the IPCC GPG, countries' forest carbon stock change estimates should be based on nationwide information on forest resources such as changes between forested and other area, estimates of growing stock, annual growth, commercial harvests and other losses. This information is often collected by the NFIs (UNECE, 2000). Furthermore, according to the IPCC GPG, other relevant information on nationwide soil surveys or forest soil monitoring programmes, if they exist, should be included in forest carbon inventories and sinks should be verified with independent methods.

This thesis builds foundations for forest sink estimation, uncertainty management and verification design with empirical sampling.

1.3 Overview of the forest ecosystem C cycle

The carbon cycle is an essential ecosystem process, providing the basis of life on earth. Plants serve as the primary producers of terrestrial ecosystems by using solar radiation, water and CO2 from the atmosphere to produce carbohydrates and oxygen (Campbell et al., 1999). In this way, the terrestrial C cycle functions as an integral part of the global C cycle between terrestrial ecosystems, oceans and the atmosphere (Houghton et al., 2001).

Photosynthesised CO_2 is stored in vegetation in various forms of carbohydrates in various organs of plants. During plant growth, proportions of their older organs die and become plant detritus, or 'litter'. During the events of natural mortality or natural disturbances (senescence due to forests fires, wind, insects, etc.), the plant becomes a part of the detritus pool via whole plant senescence. In managed forests, all or a proportion of the boles of stand's trees are removed from the stand during harvests, and other uncollected biomass components become a part of the detritus pool. During decomposition, a proportion of the plant detritus is rapidly emitted back to the atmosphere as heterotrophic respiration (CO_2), and a proportion is stored in the soil C pool as substances with long range of longevities.

Although photosynthesis and aerobic decomposition represent the major pathways of C in upland forest ecosystems, other flows of C also exist. Methane (CH_4) is an important part of the C cycle in (forested) peat-forming ecosystems (Segers, 1998). Methane is produced in the process of methanogenesis by anaerobic bacteria in infrequent conditions in well-drained mineral soils. On the other hand, CH_4 is captured in the process of methanotrophy by obligate aerobic microbes present in well-drained soils. Consequently, forest mineral soils often serve as sinks of CH_4 (Brumme et al., 2005). Methane has 23 times the global warming potential of CO_2 (Houghton et al., 2001, p. 244), but its role in large-scale forest C budgets has not yet been evaluated.

Minor flows of C, such as animal excrements and detritus, are typically neglected in ecosystem C budgets due to their small role in comparison to that of plant processes. The forest C budget also involves other gaseous fluxes of carbon, such as carbon monoxide and volatile organic compounds (VOC, isoprene, monoterpenes, oxygenated hydrocarbon), which are small in comparison to CO_2 fluxes, but play an important role in atmospheric chemistry (Monson and Holland, 2001).

The regional C cycle involves lateral flows (wind, water, animals) that transfer C within, in and out of defined geographical areas, which may be irrelevant to the stand scale C budget. Generally expressed, different processes tend to operate and shape the ecosystems at various levels of the landscape hierarchy (O'Neill et al., 1986; Holling, 1992).

Lateral fluxes of soil occur mostly in the form of dissolved organic compounds (DOC). Estimates indicate that between 30-80% of total organic C entering freshwater ecosystems is mineralized in lakes. Consequently, the mineralization of terrestrially derived C in lakes may affect the balance of CO_2 between boreal ecosystems and atmosphere (Algesten et al., 2003).

Erosional processes constitute another potentially important lateral flux in the regional C balance. The effect of erosion on forested boreal ecosystems is presumably small since most of the area is vegetated but should be accounted for in more vulnerable ecosystems, since it can move large quantities of C in or out of the system (Harden et al., 1999; Liu et al., 2003).

1.4 Building national-scale estimates of forest C stock changes

Estimating of forest vegetation and soil C stock changes over large geographical areas is a challenging task for three fundamental reasons: i) C stock changes cannot be measured directly, ii) changes occurring in C stocks within short periods are typically small in comparison to the stocks themselves, and iii) the natural variability of ecosystems impairs the measurement of small changes. To prepare reliable estimates of forest C stock changes on a large spatial scale, the data and the models used must be representative. Such a task may sound trivial, but it is difficult and laborious to achieve.

Ogle and Paustian (2005) describe the nationwide inventory development of soil organic carbon (SOC) in agricultural lands as a six-step approach. The steps are general, and can be adopted for forest C accounting as well. The approach to be applied to the nation wide inventory should first be selected or developed (step 1), then verified by case studies (2), followed by input data identification (3), uncertainty assessment (4), implementation (5), and finally validation by independent data (6). Some of the steps are itinerary, and seem self-evident, but identifying critical points emphasises the importance of each step for the whole framework and for the reliability of the results. The relationships of the separate publications of this thesis to the above-mentioned steps is as follows: steps 1, 3 and 5 (I), step 2 (II), step 4 (III, IV), and step 6 (V, as a part of potential validation design).

Soil C models, related datasets, and their applications are less frequent in forestry than in the agricultural sector. Consequently, few cases exist where the inventory process proposed by Ogle and Paustian (2005) has been or can be thoroughly applied. Step 1 is restricted by the availability of input data (identified in step 3), which affects the model's applicability. Similarly, steps 2 and 6 are restricted by either the availability of validation data availability, or the methodology for preparing independent estimates for comparison. Steps 4 and 5 are restricted, if for no other reason than the lack of society's will and devotion to share the costs of climate change.

Models of varying complexity and form serve to convert the input data sets into variables of interest. In the reporting context, these variables represent the changes in five carbon pools defined by the IPCC. Below, I describe the chief data sources used in large-scale C budgeting and how modelling has been used, in conjunction with forest inventory data and other data sources, to prepare these budgets.

1.5 Inventory data sources and methods in vegetation and soil C stock change estimation

National forest inventories (NFI) collect data on national forest resources and are common globally, though only 10–15 countries conduct thorough and representative sampling of all of its land area (FAO, 2005; Tokola, 2006). The traditional goal of these inventories has been to monitor timber resources for commercial use, but nowadays also alternative requirements have also emerged. Although inventory data sets suffer from the legacy of their original goal, they often still provide the best data sets for forest C budgeting on a national scale. In many countries (e.g. Finland, Sweden, Austria, Germany, Norway, USA), NFIs provide a representative sample of a nation's area and forest resources, and especially of timber resources (UNECE, 2000; Smith et al., 2001). Samples from tens of thousands of inventory sample plots can provide reasonably precise estimates of changes in forest resources.

The temporal resolution of inventory-based forest C budgets is generally about five years (Birdsey, 2004), depending on the inventory cycle of a country. Though possible, information on interannual forest growth variation (Henttonen, 1998) and harvests (Metla, 2006) has not been used to estimate the variability of a nation's annual vegetation (and soil) C stock changes. A trend in forest inventories seems to be apply a 'continuous' sampling of a nation's area rather than a spatially concentrated sampling: the whole country is surveyed each year rather than only a certain region(s), but with a smaller sampling density. This enables the preparation of annual estimates of forest resources based directly on measured data. NFIs in some countries have already changed the inventory cycle to a continuous one (e.g. Finland, Sweden, Norway, USA).

Forest inventories collect data on several variables strongly correlated with the carbon content of trees or a stand. Variables often used in the C stock estimation of trees include tree height, diameter at breast height, and stand-level variables such as tree density, volume or basal area. While the detailed geo-referenced plot- (or tree-) level data may be unavailable outside the inventory, forest inventories often provide summaries of such data (e.g. total growing stock, growth, forest area at given region). These data have frequently been used to estimate vegetation C stocks (Kurz and Apps, 1994; Kurz and Apps, 1999; Liski et al., 2002; Nabuurs et al., 2003; Smith et al., 2003).

As mentioned above, forest inventories provide the explanatory variables that are converted to biomass and then to carbon stock estimates. Changes in C stocks are estimated either as differences between C stocks surveyed in two consecutive inventories or by subtracting removals and senescence from the growth of vegetation (IPCC, 2003).

If tree data are available, the conversion should be done with tree-wise biomass equations (see reviews by Jenkins et al., 2003; Zianis et al., 2005). If the individual tree data are missing, one can use stand-level biomass expansion factors (BEF or BF) that convert the volume (or basal area) of a stand to biomass (Fang et al., 2001; Lehtonen et al., 2004a; Levy et al., 2004).

The majority of existing biomass models is based on case studies but some entail larger spatial coverage (Zianis et al., 2005; Somogyi et al., 2007). Biomass allocation is largely dependent on local conditions, thus the application of 'case-study' models is likely to lead to unreliable estimates of biomass. Generalized meta-models built by joining several local biomass models are likely to be better options if the representativeness of local models is uncertain (Jenkins et al., 2003; Muukkonen, 2007; Somogyi et al., 2007).

Wirth et al. (2004) proposed an elegant approach using mixed models that can utilize several empirical datasets of different origins and sizes to derive biomass estimates. For comprehensive guidelines on choosing a suitable method for biomass estimation, see Somogyi et al. (2007).

Soil inventories of forest soils are globally much less common than forest inventories. In the past, motivation to monitor forest soils has been weak. The rough classification of soils by fertility class or by mineral soil type with FAO classification has often sufficed. Current continental databases and maps of Europe's top soil C have joined several data sources (Jones et al., 2004). The limited number of soil C measurements related to soil type (texture, classification) have been scaled to continental domain with information on land cover, elevation, and temperature. Although valuable for other purposes, such as model input, this type of map is too imprecise to monitor the direct soil C change of a country. Rather, they present the current status of and spatial trends in soil C within a continent.

The difficulty of measuring small changes (relative to stocks) from material where the spatial variability is extremely high makes monitoring soil C changes with empirical data challenging. Detecting significant soil C changes requires a repeated soil sampling that is very intensive or, which spans a very long period of time (Conen et al., 2003; Conen et al., 2004; Smith, 2004). Few attemps have aimed to reduce the effort of regional soil sampling by stratification (Ståhl et al., 2004), and none by using pre-stratification based on simulation model predictions, although the benefits of stratified sampling are well known (e.g. Cochran, 1977).

Soil surveys capable of reporting nation wide estimates of soil resources are rare, and few have reported statistically significant changes in soil C stock on a national scale. Researchers detected losses of soil C in top soils of England and Wales during the period 1978–2003 (Bellamy et al., 2005). In Belgium, researchers detected, a statistically significant change in soil C on a few land scape units (Lettens et al., 2005). Sweden's soil survey has collected data, but the results are pending. On a European scale, repeated soil sampling was carried out in summer 2006 on plots measured previously in 1995, which may provide interesting results in the near future (BioSoil, 2006).

A straightforward way to prepare soil C stock change estimates is to use statistical models to upscale empirical material to larger scale. An example of such a statistical approach to monitor soil carbon stocks is the U.S. *Forcarb* model (2002), which uses data on percent C, soil texture, bulk density, and the content of large and small rock fragments in the STATSGO database, in conjunction with statistical models, to estimate the soil C stocks of a region (2002; Amichev and Galbraith 2004). Soil organic C stock changes at the national level are functions of changes in land cover and forest resources, including forest type and land use (US-EPA, 2006). However, land-use or environmental changes can have influences on soil C stocks that last for decades, centuries or even millennia. Usually, the structures of statistical models cannot represent such slow state-dependent dynamics. Statistical models can contribute to the upscaling and gap-filling of measurements, and thus to soil C monitoring if the models are continuously updated with newly measured data.

Another method to prepare regional soil C stock change estimates is to use process-based models of decomposition. Decomposition in the process-based models depends on the current C stock and on factors, such as temperature and moisture, that regulate the process of decomposition. The dependence of decomposition on the current stock allows the inclusion of slow dynamics, which are clearly present in soils. Furthermore, process-based models are generally considered better options for predictive purposes than are empirical models, since processes, rather than the states themselves, are primarily affected by the environment. Still, process-based models are also restricted by the measurements used in their calibration. The same principles of caution should govern when both of these model types are applied outside their calibration domains.

A popular method for using soil models in regional C budgeting is to link forest inventory data, biomass models and models of biomass turnover to a stand-alone process-based decomposition model (Kurz and Apps, 1999; Liski et al., 2002; Nabuurs et al., 2003; de Wit et al., 2006).

In process-based decomposition models, decomposition is mediated mainly by the activity of soil microbes, fungi and fauna, but their specific population dynamics and explicit contribution to decomposition is rarely described in soil models (McGill, 1996). Few exceptions exist, however (Eckersten and Beier, 1998; Rolff and Ågren, 1999; Chertov et al., 2001; Ågren and Hyvönen, 2003). Most models assume that the size of the microbial pool does not explicitly restrict decomposition, but rather that decomposition is limited by variables known to be correlated with microbial activity. Smith (2001; 2002) reviews the representation of decomposition processes in different SOM models.

In most models, microbial activity is expressed in the decomposition rates of model pools, which are typically first-order rate constants regulated by variables describing the ambient conditions and properties of the soil matrix. Compounds belonging to more stable fractions of SOM require higher activation energies to decompose (Davidson and Janssens, 2006). The complexity of degrading compounds creates a continuum of activation energies, which is usually approximated with several pools differing in turnover time. The effect of the soil matrix is often represented with soil clay content because small clay particles have a large surface area. SOM is protected from decomposition either chemically or physically by the occlusion of SOM in complexes with clay minerals and by encapsulation within soil aggregates (Oades, 1988; Christensen, 1996; Elliot et al., 1996; Six et al., 2002). Previous studies have implied that three or more pools are required for a realistic representation of the effect of temperature on the decomposition of SOM (Kätterer et al., 1998; Davidson et al., 2000; Knorr et al., 2005; Davidson and Janssens, 2006).

Besides temperature, the decomposition of litter or of SOM can be affected by litter quality, nitrogen or other macronutrients (Melillo et al., 1982; Prescott, 1995; Berg, 2000), heavy metals (Berg and McClaugherty, 2003), and chemical weathering (Sverdrup, 1990; Sverdrup et al., 1995). SOM decomposition may also be influenced by drought, flooding or freeze/thaw cycles (Davidson and Janssens, 2006). Many of these variables are affected by factors such as topography and past and future management (Jenny, 1941).

The complexity of the decomposition process and of large uncertainties in empirical data make it difficult to develop a completely accurate model as well as to parameterise exceedingly sophisticated models. Although more elaborate models (in term of process description) can, in principle, capture more of the natural variability, and thus provide more accurate stand-wise predictions of soil C stocks and soil C stock changes (McGill, 1996), their use is often challenged by larger input data requirements. For these reasons, researchers have developed simple (in terms of structure and input data requirements) soil models. Examples of such models include *RothC* (Coleman and Jenkinson, 1996), which requires data on litter input, clay content, and monthly PET and mean temperature, and *Yasso*, which requires data on litter production, estimates of annual temperature and rainfall (Liski et al., 2005); more examples of soil models can be found in published reviews (McGill, 1996; Peltoniemi et al., 2007). In practice, selection of a model (and an appropriate parameter set) is dictated by the availability of input data, and by the model's performance in a region's ecosystems, and by the region's climatic and environmental conditions.

As driving input, all soil models require an estimate of fresh detritus plant material (i.e. litter input). Litter input can be measured, but the measuring is tedious, especially for underground components. The use of statistical models of litter production can occasionally be useful (e.g. Starr et al., 2005) but such use breaks the functional link between living biomass and litter production if the models exclude the biomass as an explanatory variable.

More robust estimates of litter input (L_i) can be obtained by linking them to biomass, which is also closer to a process-based presentation of the issue. Litter is estimated separately for each functional component of a tree by multiplying the biomass estimate with a constant turnover rate:

$$L_i = b_i \cdot r_i \approx b_i \cdot \frac{1}{T_i}$$

This approach requires separate models of biomass (b) and biomass turnover (r) for each component of the tree (i) (stem, stump, needles, roots, fine-roots, bark). Biomass turnover models are generally based on the average life span (T) of each component. However, these estimates may be biased due to carbohydrate and nutrient resorption, especially in rapidly cycling components. Senescent needles and leaves are lighter than ones living, due to C and nutrient resorption to the branches and trunk. For example, in the material reviewed by van Heerwaarden et al. (2003), the average mass loss of leaves of various deciduous, broad-leaved, and some understorey species during senescence was 21% in comparison to the weight of living leaves. As a result of C and nutrient translocation, the turnover rates of Scots pine and Norway spruce needles are roughly 1/3 lower than these estimated without the resorption effect (Viro, 1955; Muukkonen and Lehtonen, 2004; Muukkonen, 2005; Muukkonen, 2006).

It would be reasonable assume that a similar process also occurs with other components of trees. In fact, comparison of senescent fine root to living ones has detected smaller proportions of N (2-26%) per unit length of fine-roots (kg•m^{-1}) (Kunkle et al., 2005). No such difference was found when the mass of N was expressed in relation to fine-root biomass (kgN•kg^{-1}). These findings could infer that C and N are resorbed in the same proportion, and that the effect of resorption would have an important effect on the fine-root turnover rate.

As one can see, the inventory-based approach builds on several consecutive models, and there are at least as many potential sources of uncertainties as there are parameters and inputs in these models, to say nothing of the structural uncertainties of the models. Inventory-based estimates of growing stocks in forests are generally considered reliable (Laitat et al., 2000), but little information exists on the reliability of the annual inventory-derived estimates of national forest C changes, and none on the magnitude of uncertainty in comparison to other sectors in green-house gas inventories.

1.6 Other methods to estimate forest C balance

In addition to the forest inventory-based method for estimating regional C balances, other methods also exist. Eddy-covariance measurements provide information on net ecosystem production (NEP = NPP - R_h, where NPP is the net primary production of vegetation and R_h is heterotrophic soil respiration) with high temporal resolution by measuring the net ecosystem gas exchange above forest canopies (Baldocchi, 2003). However, deriving vegetation and soil C stock change estimates comparable to the inventory approach is laborious. In addition, the ecophysiological definition of NPP is not directly comparable to the production that can be measured between two inventories (NPP_{inv} = change in biomass + senescent and removed biomass). However, the NPPs become parallel if harvests and senescence are properly accounted for between the inventory samplings, and the maintenance respiration of plant organs in the ecophysiological definition is assumed to be zero (Clark et al., 2001; Roxburgh et al., 2005). Moreover, the current density of eddy stations is far too modest (and biased towards ideal eddy covariance sites) to provide regional forest C accounting. Sparsely located monitoring stations also easily exclude disturbances such as harvests and forest fires.

Instead of using eddy covariance to compile regional C budgets directly, they can be used to calibrate ecosystem models. A recent study by Lagergren et al. (2006) estimated the C balance of the forested area of Sweden with the Biome-BGC ecosystem model calibrated with data from three eddy covariance stations in the central Sweden. Model initialisation used inventory measurements. The marked difference compared to an inventory-based methodology is that instead of using the measured inventory data of trees to estimate stock changes, process-based models predict stocks based on abiotic meteorological input (e.g. PAR, temperature, rainfall, vapour pressure deficit, day length).

Remote sensing has been mentioned as a tool to provide estimates of changes in forest resources for compliance purposes of measured biomass pool changes reported in national GHG inventories (IPCC, 2003). Remote sensing (RS) data have been used, for example, to prepare estimates of NPP (Running et al., 2004), and direct estimates of biomass stocks of vegetation based on ground calibration (Muukkonen and Heiskanen, 2005; Muukkonen, 2006). Remote sensing has also been used to improve the small region estimates of growing stocks in conjunction with inventory measurements of trees in multisource inventories (Tomppo, 2006). RS is also an ideal tool to detect rare events, at least when they are extensive enough for the measuring instruments (Li et al., 2000; Saksa et al., 2003). Rare events, such as intense forest fires and wind damage, can play an important role in the national forest C balance (Nilsson et al., 2004).

RS instruments vary not only in the spatial and temporal resolution of images they take, but also in the wavelengths they measure (Rosenqvist et al., 2003). RS images can be easily acquired for large areas, but require careful calibration with ground-based or modelled data when used to predict ecosystem variables such as NPP (Running et al., 2004). Furthermore, optical RS is limited to the assessment of upper canopy reflectance, and cannot take measurements below-canopy, nor do they take into account soil respiration. The latter is critical for the carbon budgeting of ecosystems since soil respiration is the major determinant of net ecosystem productivity (NEP) (Valentini et al., 2000). Change estimation based on RS is difficult, but when used jointly with forest inventory data the change estimates of biomass should we more precise than with either of the methods alone. RS-measured NDVI, land cover type, and season information has also been used jointly with the flux network using the data assimilation technique to predict daily NEP with temperature in continental Europe (Papale and Valentini, 2003).

The inverse modelling of concentrations of CO_2 in the atmosphere provides an opportunity to track chief sources and sinks of greenhouse gases reversely from the measured atmospheric concentrations, on the level of continents and oceans (Bousquet et al., 2000; Bousquet et al., 2006). Although inverse modelling cannot provide accurate and precise enough NEP estimates for local-scale greenhouse gas reporting, it does provide an important constraint for estimates prepared with other methodologies, such as those prepared with inventory data. However, inverse modelling cannot restrict comparison to forest sinks; all land-use categories must be addressed. European terrestrial sink estimates prepared with inverse modelling and land-based inventory have yielded similar results (Janssens et al., 2003). Still, considerable uncertainties remained in both estimates, and future comparisons will likely be limited to the continental scale.

Aeroplanes flying at low altitudes can be used to measure air momentum, CO_2, and latent and sensible heat fluxes in order to provide representative spatial estimates of gas fluxes and ecosystem production (Gioli et al., 2006). The temporal representativity of ecosystem production can be increased by combining these measurements with eddy covariance measurements, as was done for a small (16×16 km^2) region in Canada (Desjardins et al., 1997) and for a part of the Netherlands (~ 100×150 km^2) (Miglietta et al., 2007). Forest inventory data could either be joined to these estimate in order to build spatially representative national averages or for compliance purposes.

Several sources of data, measured variables, and the varying resolution and definitions of measured variables have led to various modelling approaches related to various approaches applied to forest C estimation, such as in the estimation of five IPCC-defined forest C pools. Consequently, the combination of different data types and models is laborious. Future estimates of regional or national forest C budgets will likely combine several data sources and models in sophisticated ways (Dolman et al., 2007).

The combination of inventory data with dynamic modelling is another example of joining two research traditions: i) statistically sound sampling and statistical modelling of target variables commonly applied in forest inventories, and ii) process-based modelling that aims at the calibration of processes affecting target variables (I).

2 OBJECTIVES

The overall objective of this thesis was to provide and assess the methodology that could be used for large-scale forest C stock change estimation, and to provide background information on the feasibility and reliability of the method for policy makers and scientists working in the field of carbon accounting.

The specific objectives of the separate publications of this thesis were:

- To demonstrate a method for estimating the total carbon balance of forests based on forest inventory data, and to prepare estimates of vegetation and soil C sinks for Finland's forests for the period 1922 to 2004 (I). To analyse the importance of natural and human-induced factors for the carbon balance of these managed forests, to analyse the effect of growing timber stocks of Finnish forests during the past century on soil carbon stocks, and to discuss the rationality of the reporting requirements of the UNFCCC.

- To evaluate the performance of the method developed in Study I, its predictions were evaluated against empirical data (II). Soil C stocks and the average accumulation rate of organic layer C were measured and estimated from a chronosequence of 64 stands in southern Finland

- To assess the factors that affect the uncertainties of sinks and stocks of carbon in the mineral soils and vegetation of Finnish forests during 1989–2004 (III). The information about the key factors can be used to improve the current system and the quality of the forest carbon inventory.

- To assess the effect of vegetation and mineral soil C sink uncertainties on the reliability of the national greenhouse gas inventory, and to identify the key sectors requiring the most attention (IV).

- To present a novel application of a stand model and a process-based forest soil model to sampling design for soil C changes. To estimate how much model-based stratification (with different sampling schemes) can be expected to improve sampling efficiency when both soil measurements and simulated predictions contain considerable uncertainties, and when they are accounted for in the stratification (V).

3 METHODOLOGY USED IN THIS THESIS

3.1 General methodology

The methodology in this thesis can be separated into four independent steps or modules, that estimate: 1) the preparation of input data (growing stock, forest area, drain, climate) , 2) the estimation of biomass, 3) the estimation of litter production, and 4) the estimation of the soil carbon stock and its changes with the Yasso soil model (see Figure 1 of Study II) . The process depicted to estimate forest carbon stocks and their changes is widely used (Liski et al., 2002; Nabuurs et al., 2003; de Wit et al., 2006; Liski et al., 2006). Still, each of the steps can involve considerable uncertainties (III, IV).

The calculation system is based on forest inventory data measured for the vegetation biomass C stock and stock changes. Vegetation biomass and soil C estimates are linked via litter input, but no feedback dynamics are presented between biomass and soil C pools, unlike in fully process-based models often used in ecosystem modelling. TThe development of this system has aimed to provide an easily applicable method that operates with commonly available forest inventory data, and that can be used to build estimates of forest C stock changes.

3.2 Study I

In Study I, aggregated forest inventory data on growing stock and area was utilized in the preparation of forest C stock change estimates. These data were grouped by combinations of classes of northern and southern Finland, tree species, and age of the stands. Estimates were prepared for the period 1922-2004 for vegetation (including understorey) on all forest land and for soil C in upland mineral soils of forest land within current national borders. Soil carbon was simulated with the Yasso decomposition model (Liski et al., 2005) by pine, spruce and deciduous species because their litter quality parameter differ, and in southern and northern Finland because the of the climatic differences between the regions. Changes in the area of Finland were accounted for during the preparation of inventory estimates for forest land.

Data on removals from forests originated from national statistics collected from major users of commercial wood, and the statistics based on questionnaires collected from households (Metla, 2005). These statistics provided data grouped by tree species, and by region of Finland (southern and northern).

Growth indices were used to estimate the interannual variation of tree growth and biomass C stocks, and are based on several hundred tree ring measurements taken on a spatially representative area by the NFI groups (Henttonen, 1998). The growth variability of the growing stock was estimated with growth indices applied on average growth derived from the stocks estimated in two consecutive inventories. The analysis of growth variability excluded understorey vegetation.

Biomass was estimated with BEFs based on the diameter distribution of Finnish stands in 1985 (data from NFI permanent sample plots) and Marklund's biomass equations (Marklund, 1988; Lehtonen et al., 2004a; Lehtonen, 2005a). BEFs convert aggregated growing stock data (m^3) to the biomass of its components (stem, bark, needles, branches, roots < 5 and > 5 cm in diameter, and stumps) and account for tree species and the age of the volume converted (refers to the age of the stand, but BEFs apply to more to an aggregated volume than to a specific stand). As such, they reflect the average growing stock and management regime that prevailed in Finnish forests prior to 1985. Marklund's equations are based on a large body of data from Sweden. Thus, Marklund's equations have been considered the most suitable for Finland

(Kärkkäinen, 2005), but new ones based on Finnish data are currently under development (Ojansuu, 2007, personal communication).

Fine-root biomass was estimated based on the assumption of a functional relationship between needles and fine-roots (Vanninen and Mäkelä, 1999). Understorey vegetation biomass was estimated with functions published in Study II, which represent preliminary versions of more elaborate functions (Muukkonen and Mäkipää, 2006). Estimates of biomass turnover (litter production) were obtained by multiplying the biomasses with turnover rates (see Table 1 in I).

Yasso is a dynamic seven-pool soil carbon model that takes litter as input, and simulates the decomposition of litter based on litter quality, temperature sum, and summer drought (precipitation – potential evapotranspiration) (Liski et al., 2005). Non-woody litter (fine roots, leaves, needles) entering the soil is divided directly into the decomposition compartments of extractives, celluloses and lignin-like compounds according to its chemical composition. Depending on its size or origin, woody litter enters either the fine (branches, roots, bark) or coarse (stems, stumps) woody litter compartment. Woody litter compartments retard the initial decomposition of woody detritus and represent the physical obstacle faced by invading microbes. A fraction of the woody compartments is transferred to the decomposition compartments (ext, cel, lig, hum1, hum2) in a time step of one year. Similarly, a fraction of the decomposition compartments is transferred to a subsequent compartment and a fraction to the atmosphere. The fractionation rates are controlled by temperature and drought. Extractives, cellulose and lignin-like compounds form a group that is responsible for rapid changes in the carbon stock; the residence time for these compounds is short. The Humus 2 compartment provides storage for carbon for centuries and millennia. The Humus 1 compartment lies between the rapid and slow compounds.

The mean effective annual temperature sum in southern and northern Finland (Tsum, $T > 0°C$) were used for *Yasso* based on the CRU TS 1.2 data set (Mitchell et al., 2003).

The effects of moisture were neglected because temperature alone explains more than 85% of the climatic effect on annual decomposition in Finland (Mikola, 1960; Liski et al., 2003).

The forest C sinks were compared to the emissions of other sectors reported in the National Inventory Report (NIR, 2007). This comparison was made only for this thesis to present the magnitude of interannual variability in comparison to the interannual variability of emissions. Variability was expressed as standard deviation after de-trending the time series.

3.3 Study II

The predictions of soil C with the methodology described was tested with empirical soil data from southern Finland (II). *MOTTI-Yasso* simulations of soil C were made for eight typical forest sites, which contained either Norway spruce or Scots pine on mesic or sub-xeric sites. These eight sites represented the extremes within the study area. The southern boreal region of Finland was selected as a study area from a larger number of sample plots since it was the only one to provide large enough material to compare simulations with a chronosequence of stands. The chronosequence approach was justified because the age of the stand explained most of the variance between the measured plots.

Since the history data of stands was limited, we had to assume that they are managed as recommended, but rotation length was adjusted to the maximum stand age measured in the empirical material. Clearly, this assumption adds a measure of uncertainty to the comparison, but it was considered minor in comparison to other uncertainties.

MOTTI (Hynynen et al., 2002; Matala et al., 2003; Hynynen et al., 2005; Salminen et al., 2005) has been developed to assess the effects of different forest management practices on stand development, on the profitability of forest management, on carbon sequestration and on biodiversity. *MOTTI* is based on extensive empirical data from Finnish forests; more than 68 000 trees on 4 400 sample plots with wide geographical coverage in Finland have been used to develop these models. The model can simulate all major tree species in Finland. *MOTTI* operates with a stand description in which trees are classified into tree classes characterised by tree species, number of trees in the class, diameter at 1.3 m, height, and crown ratio. Tree classes are updated every five years, growth is estimated with a distance-independent single tree growth model, and mortality is predicted a with single-tree survival model and stand-level self-thinning criteria (Hynynen and Ojansuu, 2003).

MOTTI uses tree-level predictions and Marklund's biomass equations to estimate the biomass of the trees' components. Fine-root biomass (< 2 mm) is estimated with a measurement-based relation between foliage and fine-root biomass (Vanninen and Mäkelä 1999). These estimates were used for both standing, dead and harvested trees.

The estimates of removals were stand-specific and simulated with the *MOTTI* stand simulator. The biomass of harvest residues equalled the sum of biomasses of all compartments, except that of bole. A fixed proportion of boles (tree tops) estimated with the *MOTTI* stand simulator remained in the forest after the harvest.

Data on the estimation of annual temperature sums ($T_{sum,\ T > 0°C}$) and drought indices (PET - precipitation) for *Yasso* were calculated with a climate model (Ojansuu and Henttonen, 1983). PET was estimated with Thornthwaite's model (Palmer and Havens, 1958). Mean estimates of climate variables during 1961-1990 were used for *Yasso*. Estimates of annual temperature sums ($T_{sum,\ T > 5°C}$) for *MOTTI* were estimated with the same climate model for the same period.

The sensitivity of soil C stock change estimates to those of turnover rates and initial soil C stock change was assessed by varying the turnover values.

3.4 Studies III and IV

Study III of this thesis assessed the key factors affecting national-scale estimates of forest sinks, and the uncertainties of the estimates. Uncertainties and key factors were assessed with Monte Carlo simulations. In Monte Carlo simulations, the calculation is repeated several times by drawing random samples from variable probability distributions (Morgan and Henrion, 1990). As a result, probability distributions are obtained for target variables. Key parameters affecting uncertainties were assessed based on correlations between input variables and target variables (Decisioneering, 2001).

For the purposes of Study III, the method presented in Study I was simplified in order to improve the comparability of variables included in the calculation, and its submodels were modified to operate on a national level with averaged inputs for all species (on mineral soils only). In Study IV, the calculation closely followed that of study I.

Annual sinks were estimated as differences between two consecutive stocks, and average sinks were calculated as a mean annual change using the carbon stocks of 1990 and 2004 (III). The IPCC (2003) defines the trend of net emissions as the change in net emissions between the base year (1990) and the latest inventory year (in our study, 2003) relative to net emissions in the base year (IV).

In the Monte Carlo simulations of these studies, we used reported uncertainties of input variables to define the probability distributions of input variables and model parameters. If no data existed or no reported confidence intervals were published, we relied on expert judgment.

The probability distributions and their data sources used appear in the appendices of Studies III and IV.

In Study III, the uncertainty and precision of sinks and stocks of C was defined as the standard deviation of simulated probability distributions of target variables. In Study IV, 95% confidence intervals of target variables were used, as the IPCC recommends for the GHG inventory estimates (IPCC, 2000).

In Study IV, the uncertainty estimates of source categories for the national GHG inventory were similarly assessed for the forestry sector, with the Monte Carlo simulations. Sectoral uncertainties were compared and key categories of the national GHG inventory were defined.

3.5 Study V

The purpose of this study was to determine how much the stratified sampling of sample plots (permanent NFI plots) could improve the sampling efficiency of litter and soil C stock changes on the national level.

The methodology of simulations in Study V was inherited from Study II. *MOTTI-Yasso* simulations were now made on 1 719 sample plots established by the NFI. These simulations were based on inventory-measured stand fertility, tree species, age of the stand, and stand location. Stands were presumably managed as suggested in silvicultural recommendations (Tapio, 2001). However, old-growth stands were not clearcut. For each of these plots, a number of predictions for litter and soil C stock changes over a ten-year period (y) was prepared with different assumptions of uncertainties. These simulated changes (y') (i.e. the predicted ys supplemented with uncertainties) served as a basis for the stratification of sample plots.

Estimates of the efficiency of stratified sampling were prepared with various sampling schemes including various assumptions of uncertainties. Different schemes composed all combinations of i) the number of strata G; ii) the allocation method [i.e. Neyman (optimal), proportional, or equal allocation], which defined how many samples were taken from each strata; iii) simulation uncertainty A (relative to the size of the predicted change + a constant proportion); iv) the anticipated measurement uncertainty of plots' litter and soil C stock change (related to soil sample number m); and v) scenario uncertainty represented by the accuracy of the timing of harvests and thinnings. The efficiency of stratified sampling was expressed as a standard error relative to that of simple random sampling.

Simulated changes in litter and soil C stock y' were grouped into a varying number of strata (G = 1–7) with the method originally presented by Dalenius and Hodges Jr. (1959). Because y' included the uncertainties in simulated changes of litter and soil C stocks on plots, and anticipated uncertainties in measurements of the litter and soil C stock changes on plots, they were accounted for before the formation of strata.

Uncertainty related to projected forest management (in fifth step above) was implemented by preparing a total of 50 y' for each plot by randomly varying the inventory-measured age of the stand with a standard deviation of 2.5, 5, or 10 years (Figure 1 of Study V). Each of the three sets of 50 y' (prepared for each plot with stand age uncertainties of 2.5, 5, or 10 years) included random noise due to model and measurement uncertainty. One of these 50 predictions in each set was the best estimate based on the measured stand age of the plot, which served as a basis for stratification, whereas the remaining 49 predictions were used to estimate the sampling efficiency from simulated samplings.

4 RESULTS

4.1 Carbon stock changes in the vegetation and mineral soils in Finland (I)

The mean annual vegetation sink during the period 1922–2004 was 3.3 Tg a^{-1} of C. The increase in tree C stock resulted from low initial stocking density and active forest management yielding, on average, 18% higher growth than removals during the same period, and was 33% higher after the 1970s.

The increase in soil C stock was small in comparison to the increase in tree C stock resulting from the slowness of soil C accumulation (Figure 1). With the 2004 litter input, climatic conditions and area, soil would still gain an additional 25% on top of the 2004 stock before saturation. The mean sink during the period 1922–2004 was 1.4 Tg a^{-1} when the net change in forest area transferred new soil C into the system, and 0.7 Tg a^{-1} when this transfer was ignored.

The increase in forest C stocks during the period 1922–2004 (trees 50%, ground vegetation 15%, soil and litter 13% or 7%, excluding the area increase) was accompanied by increases in mean stocking density (m3 ha^{-1}) of forests (32%), and an increase in total forest area (16%, 9% on mineral soils only) (I). The increase in ground vegetation biomass followed closely the net area change in the forests.

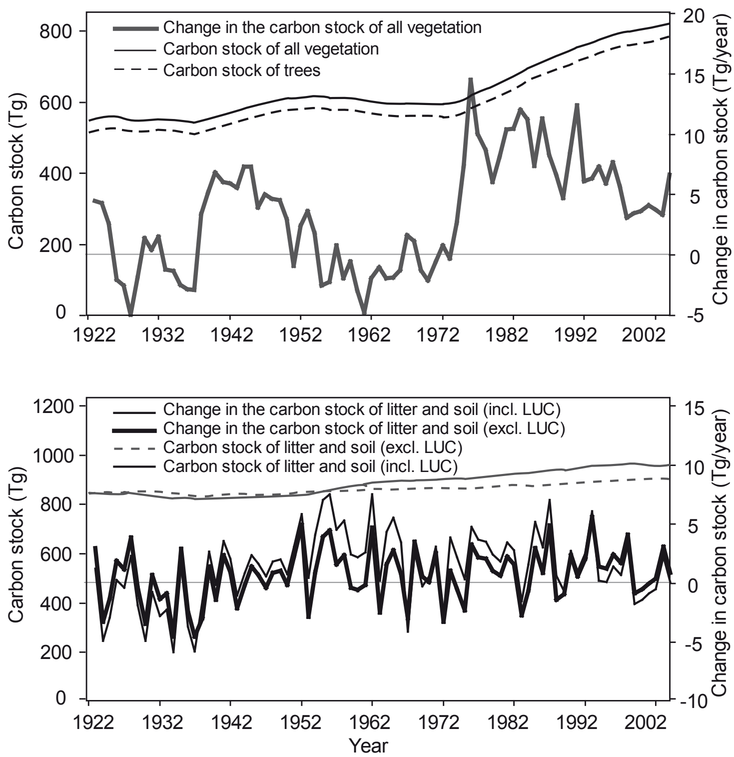

Figure 1. Carbon stocks and sinks of forest vegetation on all forest land (upper panel) and carbon stocks and sinks of upland mineral soil and litter (lower panel) (I).

Figure 2 represents the average C balance of forested mineral soils in Finland during the 1990s.

The annual vegetation sink fluctuated considerably (Figure 1). In this calculation, the fluctuation of the vegetation sink was caused by two factors: i) the interannual variability of growth and ii) the fluctuating removals from the forests. The former only weakly autocorrelated when all the tree species and regions of southern and northern Finland were pooled; the latter caused fluctuations with intervals of a few years, e.g. smaller removals during the World War II increased the tree C stock, whereas after the war, reconstruction of the nation required wood material from forests, which led to a decrease in biomass C stock. Subsequent effects, though weaker, can be found in the soil C stock as well.

The interannual variability in the soil C sink tended to be somewhat smaller than the corresponding variability in the trees (Figure 1). The factors responsible for most of the soil C stock variability included the removals of timber (harvest residues) and the temperature sum that affected the decomposition of litter and SOM. With a stable climate, the standard deviation of the sink estimates was 2.0 Tg C a^{-1}; the temperature variability explained 56% of the sink variance during the period 1922–2004 (see Figure 6 in I).

The variability of forest sinks was remarkable in comparison to the emissions and sinks of other sectors in the GHG inventory of Finland. In comparisons of sectoral emissions, the forest sinks are the second in magnitude to fuel combustion (NIR, 2007). The annual variability in forestry sinks is even larger than that in fuel combustion. The spline-detrended residual standard deviation of fuel combustion emissions between 1990 and 2004 was 0.6 Tg, compared to 1.6 Tg C a^{-1} in upland mineral soil stocks, and 1.4 Tg C a^{-1} in vegetation on forest land (Figure 3). Furthermore, it is noteworthy that our estimates of inter-annual variability (in Studies I, III and IV) cover only three major components: growth variation in gross increment, drain, and decomposition temperature. Furthermore, the growth index data did not cover the most recent years of the period, and an average growth estimate was assumed for these years.

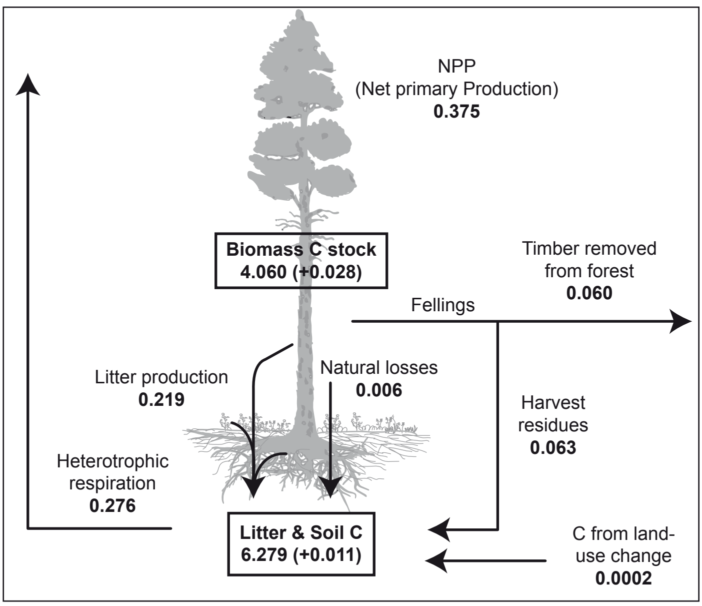

Figure 2. The flows of C (kg m^{-2} a^{-1}) in forested mineral soils in Finland. Lateral flows of C (in fact all possible flow including erosion) are implicitly accounted for in the heterotrophic respiration that represents the total outflux from the system, since the soil model calibration is based on measured stocks (I).

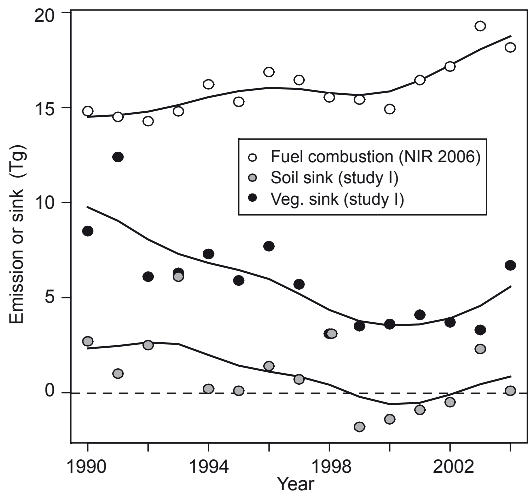

Figure 3. Absolute values of emissions from fuel combustion, and sinks of vegetation and soil (+litter) (Tg C a^{-1}). Spline was used to remove the trends from time series. Annual variability was estimated from residuals.

4.2 Method performance against empirical data (II)

The long-term average C accumulation rate in the organic layer estimated from a chronosequence of conifer plots (4.7 g m^{-2} a^{-1}) was parallel to the average long-term accumulation rate of soil carbon in simulated stands (5.8 g m^{-2} a^{-1} in young to mature stands between 20-125 a) within southern Finland, where the chronosequence data originated.

In the model simulations, the pine stands on mesic soils generally produced more litter than did the pine stands on sub-xeric soils; the same is true of the spruce stands on mesic soils in comparison to the pine stands on mesic soils. Consequent trends were reflected in the simulated soil C stocks as well. The simulated soil C stocks (F/H-100 cm) were ranged from 6.3 to 8.4 kg m^{-2}, being smallest in the subxeric pine stands and largest in the mesic spruce stands. The trend was also present in the empirical material (ranging from 6.3 to 8.0 kg m^{-2}), but only the difference between the mesic spruces and pines measured in the 0-40 cm soil layer was significant. The organic layer C stocks measured were not significantly different in any of the groups.

The sensitivity tests conducted for soil C initial state and turnover rates imply that these sources of error are unlikely to impair the increase in soil C simulated in the regenerating stands.

4.3 Uncertainty of forest C sinks and factors controlling their uncertainties (III)

The uncertainty of the mean national soil C stock in mineral soils (SD 540 Tg) was notable in comparison to the uncertainty of the vegetation C stock estimate (SD 15 Tg). The sink of vegetation was also more precise than the corresponding estimates for soil (III), but when the effect of the soil model initialisation was removed, the uncertainty of the annual soil sink nearly paralleled (SD 0.9 Tg a^{-1}) that of the vegetation sink uncertainty (SD 0.8 Tg a^{-1}). The uncertainties of the average sinks of vegetation and soil also paralleled and were approximately half of the uncertainties of corresponding annual sinks (vegetation SD 0.4 Tg a^{-1}, and soil SD

0.5 Tg a^{-1}). The uncertainties of the total annual and average forest C sink and total forest C stock were somewhat larger than those of vegetation or soil separately, indicating a lack of notable compensation of errors.

The uncertainty of the mean annual litter production estimate, given as input to the soil model, was considerably larger than that of the soil C stock change (5.5 Tg a^{-1} in comparison to a soil sink of 0.9 Tg a^{-1}). This is possible since, despite the large uncertainties of litter production, their estimates highly autocorrelated. Simulated consecutive low or high values showed no considerable effect on the annual differences in soil stock changes. To point out the importance of litter autocorrelation on sink precision, the calculation was modified to allow feeding the soil model with more weakly autocorrelated input: the results showed that inter-annually non-correlated uncertainties in litter input considerably increases the uncertainty of the soil C stock change (III).

The variables that fluctuated from year to year, with an independent estimate for each year (uncorrelated simulated samples), were the variables that most contributed to the uncertainties of sinks. The most important factors affecting the precision of the annual vegetation sinks included drain and growth variation whereas for the vegetation stocks, they included carbon density and stand BEF (Figure 4). Similarly, for soil, the soil model initialisation, temperature sum, and drain were the most important factors for contributing to the precision of the annual sink, whereas the soil model parameterisation, initialisation and biomasses and turnovers of ground vegetation and fine roots were the most important for soil C stock precision. For the average sink of vegetation, the most important factors included growing stocks at the beginning and end points of the period and the potential change in biomass models of trees due to the age structure change in the forests. For the average soil sink, the most important factors were the same as for the annual sinks, but the constant model parameters (e.g. soil model parameters, biomasses and turnover rates of trees) were slightly more pronounced,

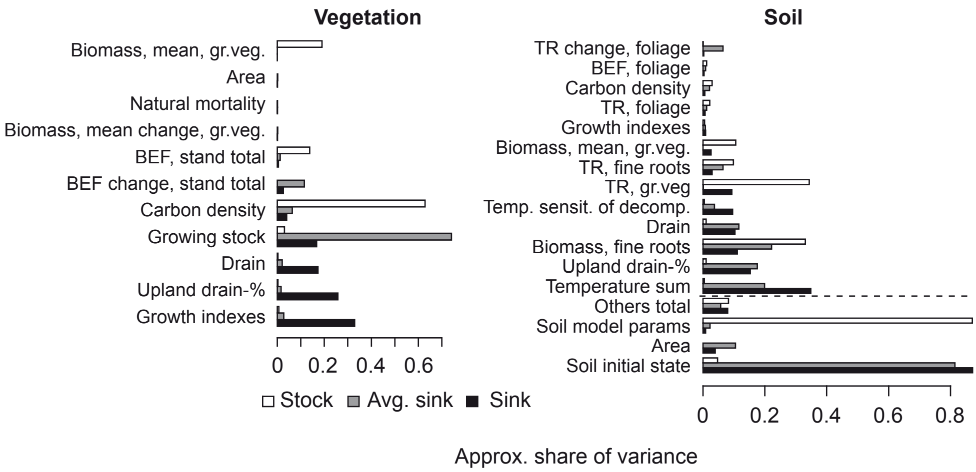

Figure 4. The most inportant factors affecting the uncertainties of vegetation (left) and soil (right) annual sink, average sink and stock of carbon (III). The “Others total” bars (in the right panel) sum the contributions of the variables above the dashed line.

as were the components that represented the potential change in these models due to the age structure change in the forests (e.g. TR change foliage, BEF change, stand total).

4.4 Uncertainties of forest sinks in comparison to uncertainties of other sectors (IV)

The precision of the carbon sink (CO_2) in the forestry sector was weaker than those of emissions from other sectors (Figure 5). Adding the vegetation biomass sink to the existing GHG inventory increased the total uncertainty to ±19% from the original -4% to +8%. The further addition of the soil sink increased the uncertainty to ±24%.

The uncertainties of forest sinks make them key categories influencing the reliability of Finland's net balance of emissions and sinks. It is also worth noting that the agriculture sector, which inherits many of the same challenges as the forestry sector, has uncertainty estimates of similar magnitude.

As was found in Section 4.1 and Study I of this thesis shows, the forest sinks fluctuated considerably over the years. In the years 1990 and 2003, forest sinks sequestered an estimated 60% and 20% of all emissions, respectively. Due to the large fluctuation in vegetation and soil sinks, they also had a fluctuating effect on the total GHG inventory estimate of Finland. However, the forest sinks did not affect the fluctuation of the uncertainties of the GHG inventory estimate, since forest sink confidence limits were rather constant (note, however, that the uncertainty of the 2003 vegetation sink was greater than that of the 1990 vegetation sink due to the more limited availability of data on growth variation).

The uncertainty of forest sinks of CO_2 parallels the uncertainties of the emissions of other gases in all other sectors (CH_4, N_2O, HFC, PFC, SF_6). The uncertainties of sinks and of the

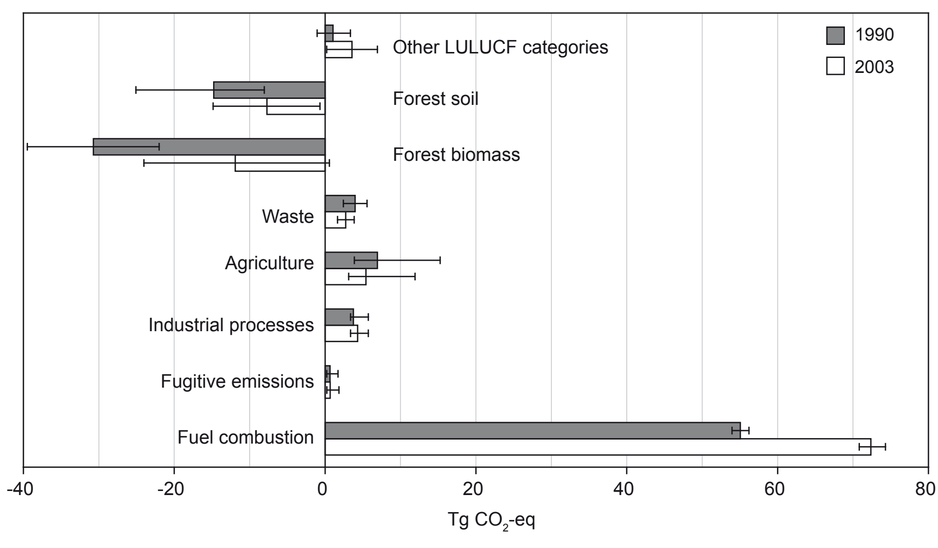

Figure 5. Sources and sinks by sector (IV). Net balances estimated from C stock changes in forest vegetation and soils are assumed to be fully consisted of CO_2 ((IV).

emissions of other gases have thus far not been estimated for forests. These estimates are likely to be highly uncertain.

4.5 Model-aided sampling design (V)

The large uncertainties that govern soil simulations and measurements make the stratification of soil C sampling a challenge. Based on the simulated sampling, however, it seems that a feasible number of soil samples (m = 10) can reduce the uncertainties of soil sampling enough to make the stratification beneficial. The Figure 6 (middle row) shows, for example, that with

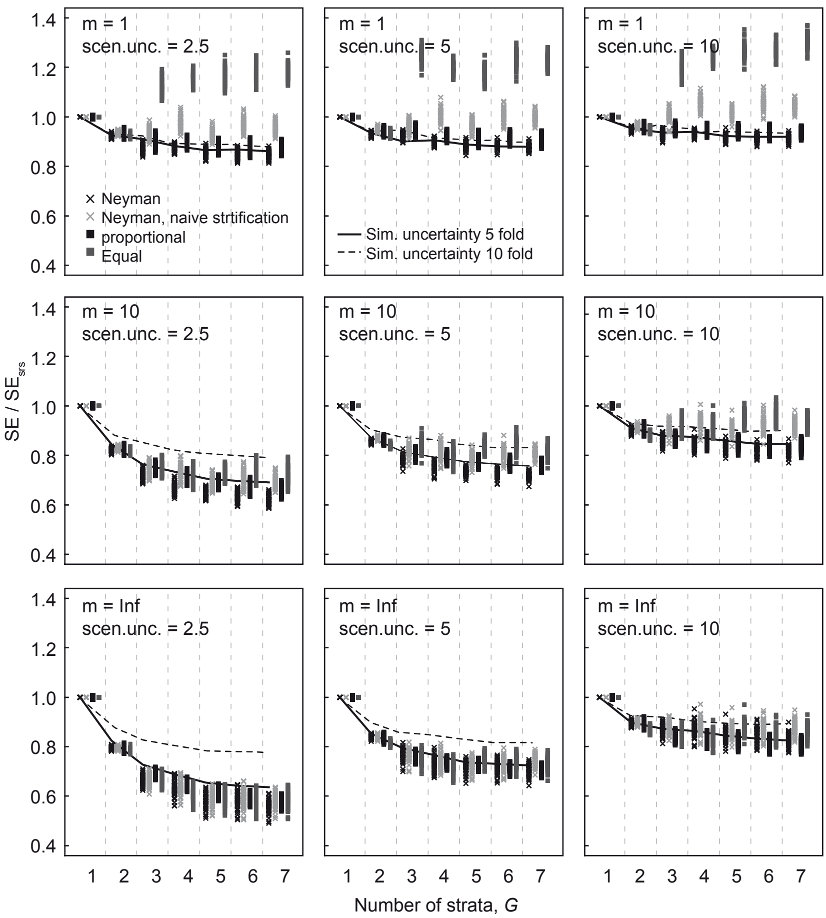

Figure 6. Efficiency of stratified samplings and allocations (see explanations in top left panel) relative to simple random sampling with different assumptions of scenario and measurement uncertainties (different panels), simulation uncertainties (lines), and with different numbers of strata, G. Measurement uncertainty is expressed with a number of soil samples collected from each of the plots (m = 1, 10 or Inf). Scenario uncertainty is expressed as the uncertainty (SD) of measured stand age that is reflected to simulated harvesting scenario. Default uncertainty of simulated predictions of y was (50 g C m^{-2} + 0.10 y), which was multiplied to five or ten fold, A=5 (solid line) and A=10 (dashed line). These lines are comparable to Neyman allocation (×) (V).

m = 10 and four strata (G = 4), the SE of the Neyman allocation is on average 67–91% that of simple random sampling, depending on the assumptions of scenario uncertainty (scen. unc.) and simulation uncertainty (A).

The uncertainty of model predictions of soil C change similarly affects stratification gain; due to its weak correlation with the soil C change measured, the stratification cannot be expected to decrease the group variances enough to improve sampling efficiency.

The stratification efficiency improved considerably when the uncertainties of the target material and predictions were accounted for. When the uncertainties were not accounted for in the division of the strata (naive stratification), efficiency improved less, and the stratification was even useless with only a few soil samples collected from each plot (Fig. 6, top row).

Despite the small relative improvements in sampling efficiency, the cost reductions in large-scale sampling may be considerable.

5 DISCUSSION

Forest C inventory involves several steps, some of which are represented in the separate papers of this thesis. The estimates of the vegetation and soil C budget and much of the methodology were developed in Study I, whereas Study II evaluated the approach by using independent empirical material. Study III analysed the precision of the approach, and identified the factors that require further development, while Study IV evaluated the role of the uncertainties of forest sinks in the uncertainty of the national GHG inventory. Study V investigated the potential of model-aided sampling to improve the efficiency of stratified sampling for soil C stock changes.

5.1 Country-scale forest sink estimates

A Recent inventory-based model assessment estimated that Swedish soils sequestered C at an average rate of 12-13 $gm^{-2}a^{-1}$ during the period 1926–2000, which is a somewhat higher rate than the 11 $gm^{-2}a^{-1}$ estimated for Finland's upland forest soils in the 1990s (I, Ågren et al., 2007). More intensive agricultural use of land in southern Sweden may have contributed to this somewhat larger soil sink.

Our estimates of vegetation sinks for the most recent years of the period (I, III, IV) were based on extrapolated growth estimates, since the results of the tenth inventory cycle had not yet been compiled for southern Finland. The mean year of the ninth inventory in southern Finland was 1998, leading to an extrapolation of 5–6 years, depending on the study. New estimates of gross increment are considerably higher than our extrapolations (Korhonen et al., 2006; Metla, 2006), and therefore our estimates of vegetation C sinks in the most recent years represent underestimates. Soil sinks are presumably little affected since the differences in litter estimates (based on biomass stocks rather than growth) are likely small.

Lateral fluxes of DOC have a considerable effect on the C balance of Finland's forests. Recently, Kortelainen et al. (2006) estimated that the lakes of Finland (based on 177 randomly selected lakes) emit an amount of CO_2 that corresponds to a 20% net change in the forest ecosystem C estimated in Study I. Most of this emitted CO_2 originates from forests of the surrounding catchments, and likely from surrounding peatlands and peatland forests that were not part of the calculation in Study I (due to the limitations of *Yasso*). DOC fluxes are small in comparison to heterotrophic respiration and they are not explicitly defined in *Yasso*. However, they seem considerable in comparison to changes in carbon stocks. The amount of C exported from rivers to lakes increases with increasing peat cover (Mattsson et al., 2005), suggesting that these outfluxes are even more important in peat soils. Peatlands cover 34% of the area of forestry land in Finland, and store 4.8 times the soil C of upland mineral soils (I, Kauppi et al., 1997; Metla, 2005).

5.2 Natural variability in forest C sinks

The fluctuation in the vegetation sink originated in the total amount of timber removed from the forests and in the variability of growth. The soil sink fluctuated as a consequence of the harvest residues and temperature that affected decomposition. As a consequence of the transfer of residue biomass from vegetation to soil, vegetation often sequestered C while soils emitted C, and vice versa. Our results support the conclusion that all ecosystem C pools should be accounted for in the preparation of forest C sink estimates are prepared (Kurz and Apps, 1999).

The natural fluctuation of vegetation sink estimates is an interesting feature of the national-level C budget, and it is also surprising when it occurs over such a large area. The variability of growth was expressed as indices based on measured tree rings collected during the field measurements of the NFIs, and thus have representative spatial coverage (Henttonen, 1998). The variability of the regional forest C balance has not been accounted for in inventory-based estimates before, although it is known that the productivities of different ecosystems vary (Knapp and Smith, 2001; Suni et al., 2003). The biome-level variability of NDVI, which is a strong correlate of NPP, has been detected with remote sensing and has been linked to mean temperature (Braswell et al., 1997).

Although the varying amount of harvest residues was the main determinant of soil C sink variability, the temperature control of decomposition accounted for ~ 1/4 of its variability (I). The effect of temperature alone, can render the carbon budgets of forest ecosystems negative, even though the vegetation sequesters C, as has been found at many EUROFLUX eddy covariance measurement sites (Valentini et al., 2000). The results suggest that careful calibration of the temperature dependence of decomposition is important for regional-scale forest C sink estimation in the boreal region.

5.3 Evaluation of the method

Study II provided empirical material on soil C to test the performance of the model's approach. The methodology applied in Study II parallels the method applied on the national scale. The most pronounced difference was the replacement of inventory estimates with stand simulations (*MOTTI*), and the replacement of BEFs with Marklund's biomass equations. However, these changes do not impair our conclusions of model performance or applicability: the stand growth models of *MOTTI* are based on broad empirical material from Finland, and the biomass equations originate from Sweden (Marklund, 1988). Marklund's equations have been assessed to suite Finnish conditions (Kärkkäinen, 2005).

Model predictions of litter production (II) were comparable to those of other studies presenting independent material, indicating that the input to the soil model was realistic (Viro, 1955; Mälkönen, 1974; Berg and Meentemeyer, 2001). Simulated soil C stocks with fertility and tree species are in line with soil C measurements within the region (Tamminen, 1991; Liski and Westman, 1995; Liski and Westman, 1997). The model predicted that, in southern

Finland, with increasing temperatures, the effect of increasing ecosystem productivity on the litter and soil C stock (via litter production) would overshadow the increasing respiration losses, and the soil C would increase from north to south. Empirical soil data from Finland support the trend predicted by the model (Liski and Westman, 1997).

The long-term average increase in organic layer C measured from a chronosequence (5 g m^{-2} a^{-1}) parallels the simulated average soil C accumulation rate during a rotation in stands older than 20 years (II), assuming that soil C was originally in steady state equilibrium. This average rate was parallel to or of the same magnitude as empirical data from conifer sites (Turner, 1975; Bormann et al., 1995; Wardle et al., 2003). Incorrect assumptions of the initial soil state in the model, however, can easily bias the comparison to the rates measured (II).

In Study I, the average C accumulation in the soil was approximately 2 g m^{-2} a^{-1} during the period 1922–2004. In recent years, soil C accumulated at a rate of 11 g m^{-2} a^{-1}. The rates in Studies II and I are not directly comparable because Study II is based on case studies and chronosequence, whereas the Study I is based mostly on structural changes in the forests of Finland, and on the consequent gravitation to new steady state of equilibrium that is higher than at the beginning of the calculation period. Recently, Ågren et al. (2007) have also estimated based on NFI data that Swedish soils will sequester C in future because current litter production is imbalanced with current soil C stocks.

The results of the comparison in Study II were encouraging, as they showed that the predictions are reasonable and plausible, but show little more. The available material is too scarce to permit comprehensive assessment of the model predictions. Therefore, while waiting more comprehensive test material, we must judge the plausibility of predictions with qualitative assessments of component models and their predictions. In theory, assessment of a model by components may be considered an even more rigorous test than a test of a model as a whole that permits error compensation.

Many of the applied biomass and litter production models are based on data originating southern Finland or from similar ecosystems and from similar climatic conditions. Marklund's biomass equations have been compared to recent Finnish biomass equations (Repola et al., 2007). The new Finnish biomass models, based on tree diameter and height, predict parallel results with Marklund's models of the same form, except for pine root biomass components

Two of the most important biomass components of forest ecosystems in terms of the C cycle are leaves and fine-roots (II). The preparation of leaf biomass and of biomass turnover models and their comparison to published material appear elsewhere (Lehtonen, 2005a; Muukkonen, 2006). One potential approach to predicting the biomasses of shoots in future is to apply the pipe model (Shinozaki et al., 1964), which performed best in the comparison of model approaches to empirical data from southern Finland (Lehtonen, 2005b). Exact estimation of fine-root biomass is difficult. Our estimates are based on the assumed functional interdependence of needles and fine roots; fine-root biomass is a constant proportion of needle (or leaf) biomass (Vanninen et al., 1996; Vanninen and Mäkelä, 1999). Fine roots have a reported range of longevities that has led to a range of turnover rates that may considerably affect the ecosystem C cycle (Gaudinski et al., 2001; Majdi, 2001; Matamala et al., 2003).

For soil, Palosuo et al. (2005) provided an independent evaluation of the performance of *Yasso*. In that comparison, *Yasso* accounted for most of the effects of temperature and initial litter quality on the short-term decomposition of a range of foliage litter types under varying climatic conditions. The comparison also showed that *Yasso* overestimated the overall decomposition rate of litter bags at Canadian sites, and is probably too sensitive to drought. The long-term accumulation or decomposition of slower soil C compounds remains untested. *Yasso*'s climatic dependence of decomposition is based on litter bag decomposition tests from a temperature gradient (mean annual T between -1.7 and 16.6°C) in Europe (Liski et al., 2003). Toposequence may not represent the effect of annual climatic variation iat a given location. However, no other suitable datasets existed to assess this.

The tests conducted in this thesis (II) and elsewhere (Palosuo et al., 2005) have increased our understanding of the model predictions of soil C: the model is not overly sensitive to its inputs and predicts comparable soil C stocks and stock changes for material measured in southern Finland, but may fail in distinctively different conditions. For example, in the very moist conditions of south east Norway, stocks seem largely underestimated, possibly due to the high downward transport of DOC to the subsoil, where it stabilises as physically or chemically protected organo-mineral complexes (de Wit et al., 2006). Similarly, stocks of soil C were underestimated in the southern Alpine region (Thürig, 2005; Thürig et al., 2005). Suggested reasons for this deviation include heavy but clustered rainfall during the year, and soil properties not accounted for in *Yasso*, which led to exceptionally stable soil organic matter in mild climatic conditions.

One potentially considerable source of uncertainty in large-scale studies stems from the aggregation of model inputs and from the use of models on these spatial areas. Evidently, the predictions become biased if the response of the process to input data is non-linear (Rastetter et al., 1992; Izaurralde et al., 2001; Rastetter et al., 2003). For practical reasons, however, some level of spatial and temporal aggregation is necessary to build national-level forest C budgets, since some data may exist and some models can be defined on a certain spatio-temporal domain. Some processes considered less important in the reporting context, such as within-day variation of temperature or tree-level estimates of biomass, can be represented with aggregate models and data since predictions are unlikely to improve. However, defining a proper level of aggregation with no detailed analysis may prove impossible. Ogle et al. (2006) demonstrated how soil C change estimates for US agricultural lands prepared with fine spatial resolution were considerably biased when the model was parameterised with broad scale information. Izaurralde et al. (2001) compared soil C change estimates made with three alternative input data aggregation schemes for two contrasting ecodistricts in Canada. The regional soil C change estimates prepared with different levels of input data aggregation were parallel for one ecodistrict, but were actually opposite for another ecodistrict (Izaurralde et al., 2001). Essentially, the scaling issues are ecosystem dependent, and the errors estimated are highly dependent on the selected models.

The aggregation approach was used in Studies I, III and IV, whereas the stand-wise approach was used in the other studies. To analyse the magnitude of aggregation bias qualitatively, I shortly review the model structure below.

The biomass and litter input were estimated by the classes of stand age, tree species, and region. The estimates are unbiased since these models are parameterisised for national level-estimations, or for estimations in northern or southern Finland (Lehtonen et al., 2004a; Lehtonen et al., 2004b; Muukkonen and Lehtonen, 2004; Lehtonen, 2005b; Muukkonen, 2005; Muukkonen and Mäkipää, 2006; Muukkonen et al., 2006).

The *Yasso* soil model, on the other hand, presents a non-linear decomposition process with respect to temperature and drought inputs. These two inputs affect decomposition rates, namely the *a* and *k* parameters, by modifying their values linearly with the temperature and drought (Liski et al., 2005).

In principle, more accurate results could be obtainable if the heterogenity of input data and its effects on soil C stock changes were captured by a classificating material into smaller groups with regard to factors affecting the estimate heterogenity, meaning classification with *Yasso* according to temperature sum and drought and subsequent division of litter input. On the other hand, harvest statistics strongly affecting litter input are collected only on the forest district level (19 in Finland). Assessment of the effect of aggregation on the accuracy of the estimates would require further study.

5.4 Uncertainty management

The Marrakech accords of the Kyoto protocol state that inventory estimates must be consolidated with uncertainty estimates. The IPCC GPG 2003 provides methodological guidance for uncertainty estimation with two methods: one using error propagation equations based on Taylor series expansions (Tier I), and another using Monte Carlo simulations (Tier II). In our case, the selection of a simulation-based method was evident because of the impossibility to estimate the uncertainties of dynamic model predictions with Tier I method. In addition to these methods, analytical solutions, fuzzy logic, Bayesian inversion and bootstrap simulations can be used for uncertainty assessment (Monni, 2005; van Oijen et al., 2005). None of these were deemed suitable for our purposes.

In Studies III and IV, we prepared a simulation based uncertainty estimates for vegetation and soil carbon sinks and stocks. The uncertainties estimated in Studies III and IV represent the uncertainties obtained with a certain model structure. It should be understood that no uncertainties attributable to missing factors or processes that could potentially affect on the resulting sinks have been estimated. Uncertainty due to other factors should be added to estimated uncertainties.

Model comparison could shed light on potential error due to model structure, which is often the greatest source of error (Chatfield, 1995). Model comparisons could serve as tools to help us understand the behaviour and possible limitations of the models and selected approach (Rastetter, 1996). Although, the model comparisons were beyond the scope of this thesis, a recent review of soil C models and a representation of existing forest soil datasets provides background data for such an exercise (Peltoniemi et al., 2007).

The definition of probability distributions for model parameters and input parameters is the most challenging task in this type of uncertainty analysis. Sufficient data are seldom available to prepare probability distributions for the purposes of uncertainty analysis. In these cases, the selection of an appropriate probability distribution revealed the quality of the initial data. With no plausible way to estimate a mean value of normal distribution, uniform distribution with low and high boundaries was set. On the other hand, normal distributions with cut points at low and high values were sometimes the most appropriate. The inherent subjectivity of the method requires that results be interpreted with a listing of the probability distributions of model parameters and input variables. The use of Bayesian methods would reduce the subjectivity of the uncertainty estimation since it relies heavily on the modelled process and on existing data (van Oijen et al., 2005). For large systems, however, the method becomes highly computer intensive. The Bayesian approach was not used because applied C model does not depict the process in sufficient detail and no measurements of target variables are available for use as a priori estimates of biomass and soil stock or change measurements. The lack of representative measurement data also inhibited the statistical derivation of uncertainty estimates based on modelled predictions and empirical data in case studies (Ogle et al., 2007).

Despite the fact that uncertainty analyses are rarely able to provide comprehensive estimates of the error, and that the analysis procedure often includes some subjectivity (in terms of expert evaluation of initial variable uncertainties), they are an integral and important part of inventory analysis and development (Ogle and Paustian, 2005). It is most reasonable to focus the future effort on the weak parts of the system.

The central location of the variable in the calculation system may have a considerable effect on its contribution to uncertainty estimates of the target variables. In Study III, we scaled the models of individual tree species and regions of Finland to composite models, in order to improve the comparability between the model parameters and input data. For example, instead of using the tree biomass model for branches for three tree species, we used one branch biomass model for all three tree species. The probability distributions for these composite models were simulated based on the original models, and are reported in the appendix of III.

Changes in C stocks and the C stocks themselves were affected by different types of variables. Changes in C were sensitive to input data, whereas the input data affected the C stock estimates less. Input data errors in consecutive years were uncorrelated (except in some test cases), which decreased the autocorrelation of C stocks, and which subsequently decreased the precision of C stock changes estimated as differences in consecutive C stocks (III). As Heath and Smith (2000) also concluded, different means are needed to improve the stock estimates and stock change estimates.

Many of the sub-models in the model framework are based on data from several years. They predict averages of, for example, biomass or biomass turnover for long periods, and thus by definition cannot represent year-to-year variability (I). They are better suited for predicting of averages of forest C sinks for more than one year (III). Previous studies have measured the high interannual variability of litter production in boreal forests (Flower-Ellis, 1985; Kouki and Hokkanen, 1991; Starr et al., 2005). Depending on the spatial correlation of annual estimates of litter production on a regional scale, these estimates can potentially increase the variability of vegetation and C stocks considerably. If this variability is not accounted for in the annual estimates, it should be considered uncertainty. Study III examined the effect of variable turnover rates by varying the correlation of interannual litter production estimates. The use of averaged (across time) turnover models will likely lead to underestimates of soil sink uncertainties (III).

Besides the annual variability of forest growth, forest resources also change in the long-term due to changes forest management or, for example, due to climatic shift. Except for age-structure changes represented in BEFs (Lehtonen et al., 2004a), none of the models explain these factors.

For more precise prediction of annual forest C sinks, it seems important to develop forest C accounting methods further by including models that express changes in forests with factors that vary from year-to-year and which account for long-term changes in forest structure and conditions. Some of these factors could be related to forest density, climatic conditions, and even to genotypic changes (III).

Soil initial state estimation is generally considered a problematic issue in soil C modelling, and many authors have used long initialisation periods for their soil C models (I, Chen et al., 2000; de Wit et al., 2006). Biased estimates results, especially when historical land-use is not properly accounted for in the model application, or if the model is calibrated with empirical soil material that is far from equilibrium (Wutzler and Reichstein, 2007). However, the uncertainty it brings to estimates should be separately assessed for each case, based on the purpose of the application. When we estimated annual soil C sinks, the uncertainty of soil C stock changes due to uncertain initial soil state levelled off rather soon due to uncertainty of other factors. This point is largely trivial but is of practical concern. Considering the other uncertainties that affect estimates of annual soil C stock changes, initialisation is a minor problem since its effect can be minimised with a rather short initialisation period. In this sense, the annual estimates of soil C changes (especially when evaluated relative to each other) in Study I some decades after the beginning of the period were little affected by the uncertainties of soil model initialisation. Records of forest resources and litter input spanning few decades should make annual soil C sink estimates prepared with any soil model nearly as precise as records spanning from the beginning of the 20th century, if soil model dynamics are close to the dynamics of *Yasso*.

5.5 Uncertain forest sinks in the national greenhouse gas inventory

The inclusion of forest sinks in the national GHG increased the uncertainty of the annual estimates of the national GHG budget considerably. Despite the considerable decrease in the precision of the inventory, new results increased the accuracy of the total GHG inventory with wider sectoral coverage (IV).

In the first commitment period, forest sinks were not part of the base year estimate (1990) but forest sinks, according to the Article 3.4 of Kyoto protocol, can nevertheless serve to a limited extent to compensate for emissions from other sectors. The limited extent for Finland means a compensation of 0.16 Gg C a^{-1}. When a sink of this magnitude is deducted from emissions, its contribution to the uncertainty of the entire GHG inventory remains small. However, in some other Annex I countries the cap for compensation was significantly higher, meaning that large forest sinks could be deducted from some nations' GHG emissions with a very uncertain basis. Countries that have negotiated large caps include Japan, Canada and Russia (UNFCCC, 2001b).

If forest sinks will be affect the assigned amounts of countries in the next commitment period after Kyoto, it seems reasonable that their contribution is based on the average C stock change of a few years rather than of a one year. Similarly, it seems reasonable to use averages as target period estimates.

5.6 Models in the empirical sampling design

Soil sampling is notoriously difficult and laborious. In this thesis, we proposed a method of using models as a way to stratify sampling on a national level (V). This type of approach would intimately join two research traditions, that of empirical research and that of ecosystem modelling, to achieve the joint goal of soil C monitoring.

Models provide an opportunity to capture a large share of the natural variability of soil C stock changes, since they quantify our most recent understanding of the process. Generally, the better the predictions of stratification correlate, the greater the increases in sampling efficiency. In Study V, we used the methodology applied in Study II to simulate soil C stock changes in permanent NFI-established sample plots.

The various assumptions we used for the reliability of our simulations, for the anticipated uncertainty of the measurements, and for the scenarios of the forest management regime, lead us to a conclude that despite these sources of error, stratification can be beneficial with a feasible number of soil samples. An important aspect in stratification is to anticipate the uncertainty of the measured material and simulated correlates. If this is not done, sample proportions for strata will not be optimal, and sampling efficiency will decrease. To our knowledge, the uncertainties of the target material have been anticipated previously only for an approach using a simple statistical model to provide stratification correlates (Godfrey et al., 1984; Kott, 1985). In our case, the model construct was considerably more elaborate, and providing a mathematical solution for the question is impossible. The use of more elaborate predictive models may play a role in the practical sampling designs of future soil C stock changes. Such an approach could be used for many other target variables as well.

The network of forest inventory sample plots that can provide initial material for the purposes of stratification is available in many countries and regions. Appropriate models are available for many regions as well. For example, a similar approach to ours could be implemented by replacing some of the models with region- or ecosystem- specific modules. In the total absence of inventory data, our approach may no longer be feasible (i.e. practically unachievable without great effort and high costs), although other types of modelling approaches may still work (e.g. in conjunction with remote sensing) (Prince and Steininger, 1999).

Change in soil C represents a time-dependent criterion for sampling stratification. Time-dependent criteria can be expected to be the most efficient criteria for stratifying the first sampling interval, but its efficiency will decrease with subsequent samplings unless new plots are selected. Therefore, some researchers have recommended using time-invariant criteria for change monitoring (Scott, 1998). In the case of forest soil C stock changes, however, other correlates can be expected to be very weak, which leaves stratification useless. The option of recycling plots could be considered, but its potential requires further study.

To build trust in model predictions of litter and soil C, models should be evaluated against empirical data, and when predictions are made for extensive spatial regions, verification should be made preferentially with a representative data set. Stratification of material can provide such data sets with less effort than standard sampling schemes. A measured sample from an existing population could also serve to provide measurement-based estimates reported to UNFCCC. If inventories are large enough, small decreases in sampling effort can also lead to large cost reductions.

6 CONCLUSIONS

The methodology presented here is easily transferable to new regions, and to practically any country collecting data on forest resources. Several biomass and biomass turnover models for various environmental conditions exist, as do several soil C models. Ultimately, estimates can be prepared for any region, but their reliability should always be evaluated.

Annual changes in the vegetation and soil C stocks of Finnish forests fluctuate considerable and are frequently contrary, thus requiring a full accounting of the forest C budget. Annual estimates are uncertain in comparison to emissions estimates from other sectors of the GHG inventory, thus requiring the acknowledgement of the different characters of ecosystem sinks and of anthropogenic emissions in the international inventory guidelines, reporting and use of data in emissions compensation, or in the potential estimations of countries' assigned amounts in future.

If forest C sink estimates are to be used in practice, it seems more reasonable to use long-term average estimates of ecosystem C sinks than annual estimates. The estimation of average sinks should still be based on annual or even more frequent data to avoid potential inaccuracies due to non-linear decomposition processes influenced by the climate.

In future, various data sources and methodologies will likely be joined for more accurate and precise ecosystem sink estimation. The increased spatial resolution of calculations is also likely to lead to the enhanced accuracy of sink estimates. More accurate and precise estimates are required in both international reporting and in global and regional climate modelling.

REFERENCES

Algesten, G., Sobek, S., Bergström, A.-K., Ågren, A., Tranvik, L.J., Jansson, M., 2003. Role of lakes for organic carbon cycling in the boreal zone. Global Change Biology 10, 141-147.

Amichev, B.Y., Galbraith, J.M., 2004. A revised methodology for estimation of forest soil carbon from spatial soils and forest inventory data sets. Environmental Management 33, S74-86.

Baldocchi, D.D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Global Change Biology 9, 479-492.

Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M., Kirk, G.J.D., 2005. Carbon losses from all soils across England and Wales 1978-2003. Nature 437, 245-248.

Berg, B., 2000. Litter decomposition and organic matter turnover in northern forest soils. Forest Ecology and Management 133, 13-22.

—, McClaugherty, C., 2003. Plant litter - decomposition, humus formation, carbon sequestration Springer, Berlin Heidelberg.

—, Meentemeyer, V., 2001. Litter fall in some European coniferous forests as dependent on climate: a synthesis. Canadian Journal of Forest Research 31, 292-301.

BioSoil, 2006. ICP Forests' Manual on methods and criteria for harmonized sampling, assessment, monitoring and analysis of the effects of air pollution on forests. Part IIIa Sampling and analysis of soil. United nations economic commission for Europe convention on long-range transboundary air pollution and the Regulation of the European parliament and of the council concerning monitoring of forests and environmental interactions in the community (Forest Focus).

Birdsey, R., 2004. Data gaps for monitoring forest carbon in the United States: an inventory perspective. Environmental Management 33, S1-S8.

Bormann, B.T., Spaltenstein, H., McClellan, M.H., Ugolini, F.C., Cromack JR, K., Nay, S.M., 1995. Rapid soil development after windthrow disturbance in pristine forests. Journal of Ecology 83, 747-757.

Bousquet, P., Ciais, P., Miller, J.B., Dlugokencky, E.J., Hauglustaine, D.A., Prigent, C., Van der Werf, G.R., Peylin, P., Brunke, E.G., Carouge, C., Langenfelds, R.L., Lathiere, J., Papa, F., Ramonet, M., Schmidt, M., Steele, L.P., Tyler, S.C., White, J., 2006. Contribution of anthropogenic and natural sources to atmospheric methane variability. Nature 443, 439-443.

—, Peylin, P., Ciais, P., Le Quere, C., Friedlingstein, P., Tans, P.P., 2000. Regional changes in carbon dioxide fluxes of land and oceans since 1980. Science 290, 1342-1346.

Braswell, B.H., Schimel, D.S., Linder, E., Moore III, B., 1997. The response of global terrestial ecosystems to interannual temperature variability. Science 278, 870-873.

Brumme, R., Verchot, L.V., Martikainen, P.J., Potter, C.S., 2005. Contribution of trace gases nitrous oxide (N_2O) and methane (CH_4) to the atmospheric warming balance of forest biomes. In: Griffiths, H., Jarvis, P.G. (Eds.), The Carbon balance of forest biomes. Taylor & Francis, Trowbridge, UK.

Campbell, N.A., Reece, J.B., Mitchell, L.G., 1999. Biology. Addison-Wesley, Menlo Park, CA.

Chatfield, C., 1995. Model uncertainty, data mining and statistical inference. Journal of the Royal Statistical Society A 158, 419-466.

Chen, J., Chen, W.J., Liu, J., Cihlar, J., Gray, S., 2000. Annual carbon balance of Canada's forests during 1895-1996. Global Biogeochemical Cycles 14, 839-849.

Chertov, O.G., Komarov, A.S., Nadporozhskaya, M., Bykhovets, S.S., Zudin, S.L., 2001. ROMUL - a model of forest soil organic matter dynamics as a substantial tool for forest ecosystem modeling. Ecological Modelling 138, 289-308.

Christensen, B.T., 1996. Matching measurable soil organic matter fractions with conceptual pools in simulation models of carbon turnover: Revision of model structure. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of soil organic matter models. NATO ASI Series I. 38. Springer-Verlag, Berlin Heidelberg.

Clark, D.A., Brown, S., Kicklighter, D.W., Chambers, J.Q., Thomlinson, J.R., Ni, J., 2001. Measuring net primary production in forests: concepts and field methods. Ecological Applications 11, 356-370.

Cochran, W.G., 1977. Sampling Techniques. John Wiley & Sons, Inc., United States of America.

Coleman, K., Jenkinson, D.S., 1996. RothC-26.3 - A Model for the turnover of carbon in soil. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of soil organic matter models - using existing long-term datasets. NATO ASI Series Vol. I 38 Springer-Verlag, Heidelberg.

Conen, F., Yakutin, M.V., Sambuu, A.D., 2003. Potential for detecting changes in soil organic carbon concentrations resulting from climate change. Global Change Biology 9, 1515-1520.

—, Zerva, A., Arrouays, D., Jolivet, C., Jarvis, P.G., Grace, J., Mencuccini, M., 2004. The carbon balance of forest soils: detectability of changes in soil carbon stocks in temperate and boreal forests. In: Griffith, H., Jarvis, P.G. (Eds.), The carbon balance of forest biomes. Garland Science/BIOS Scientific publishers

Dalenius, T., Hodges Jr., J.L., 1959. Minimum variance stratification. Journal of the American Statistical Association 54, 88-101.

Davidson, E.A., Janssens, I.A., 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440, 165-173.

—, Trumbore, S.E., Amundson, R., 2000. Biogeochemistry - soil warming and organic carbon content. Nature 408, 789-790.

de Wit, H.A., Palosuo, T., Hylen, G., Liski, J., 2006. A carbon budget of forest biomass and soils in southeast Norway calculated using a widely applicable method. Forest Ecology and Management 225, 15-26.

Decisioneering, 2001. Crystall Ball 2000.2.

Desjardins, R.L., MacPherson, J.I., Mahrt, L., Schuepp, P., Pattey, E., Neumann, H., Baldocchi, D., Wofsy, S., Fitzjarrald, D., McCaughey, H., Joiner, D.W., 1997. Scaling up flux measurements for the boreal forest using aircraft-tower combinations. Journal of Geophysical Research-Atmospheres 102, 29125-29133.

Dolman, A.J., Noilhan, J., Tolk, L., Lauvaux, T., van der Molen, M., Gerbig, C., Miglietta, F., Pérez-Landa, G., 2007. Regional measurements and modeling of carbon exchange. In: Dolman, H., Valentini, R., Freibauer, A. (Eds.), Observing the continental scale Greenhouse Gas Balance of Europe. Ecological Studies of Springer (In press).

Eckersten, H., Beier, C., 1998. Comparison of N and C dynamics in two Norway spruce stands using a process oriented simulation model. Environmental Pollution 102, 395-401.

Elliot, E.T., Paustian, K., Frey, S.D., 1996. Modelling the measurable or measuring the modelable: A hierarchical approach to isolating meaningful soil organic matter fractionations. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of soil organic matter models. NATO ASI Series I 38. Springer-Verlag, Berlin Heidelberg.

Fang, J.Y., Chen, A.P., Peng, C.H., Zhao, S.Q., Ci, L., 2001. Changes in forest biomass carbon storage in China between 1949 and 1998. Science 292, 2320-2322.

FAO, 2005. Global Forest Resources Assessment 2005. FAO Forestry Paper 147.

Flower-Ellis, J.G.K., 1985. Litterfall in an age series of Scots pine stands: summary of results for the period 1973-1983. In: Lindroth, A. (Ed.) Climate, photosynthesis and litterfall in pine forests on sandy soil - basic ecological measurement at Järdaås. SLU, Uppsala.

Gaudinski, J.B., Trumbore, S.E., Davidson, E.A., Cook, A.C., Markewitz, D., Richter, D.D., 2001. The age of fine-root carbon in three forest of the eastern US measured by radiocarbon. Oecologia 129, 420-429.

Gioli, B., Miglietta, F., Vaccari, F.P., Zaldei, A., De Martino, B., 2006. The Sky Arrow ERA, an innovative airborne platform to monitor mass, momentum and energy exchange of ecosystems. Annals of Geophysics 49, 109-116.

Godfrey, J., Roshwalb, A., Wrigth, R.L., 1984. Model-based stratification in inventory cost estimation. Journal of Business & Economic statistics 2, 1-9.

Goodale, C.L., Apps, M.J., Birdsey, R.A., Field, C.B., Heath, L.S., Houghton, R.A., Jenkins, J.C., Kohlmaier, G.H., Kurz, W., Liu, S.R., Nabuurs, G.J., Nilsson, S., Shvidenko, A.Z., 2002. Forest carbon sinks in the Northern Hemisphere. Ecological Applications 12, 891-899.

Harden, J.W., Sharpe, J.M., Parton, W.J., Ojima, D.S., Fries, T.I., Huntington, T.G., Dabney, S.M., 1999. Dynamic replacement and loss of soil carbon on eroding cropland. Global Biogeochemical Cycles 13, 885-901.

Heath, L.S., Smith, J.E., 2000. An assessment of uncertainty in forest carbon budget projections. Environmental Science & Policy 3, 73-82.

Henttonen, H., 1998. Puiden kasvunvaihtelu. In: Mälkönen, E. (Ed.) Ympäristömuutos ja metsien kunto, Metsien terveydentilan tutkimusohjelman loppuraportti. Metsäntutkimuslaitoksen tiedonantoja 691.

Holling, C.S., 1992. Cross -scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs 64, 447-502.

Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K., Johnson, C.A., 2001. Climate Change 2001: The Scientific Basis Intergovernmental Panel on Climate Change, Cambridge University Press.

Houghton, R.A., 2007. Balancing the global carbon budget. Annual Review of Earth and Planetary Sciences 35, 313-347.

Hynynen, J., Ahtikoski, A., Siitonen, J., Sievänen, R., Liski, J., 2005. Applying the *MOTTI* simulator to analyse the effects of alternative management schedules on timber and non-timber production. Forest Ecology and Management 207, 5-18.

—, Ojansuu, R., 2003. Impact of plot size on individual-tree competition measures for growth and yield simulators. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 33, 455-465.

—, Ojansuu, R., Hökkä, H., Siipilehto, J., Salminen, H., Haapala, P., 2002. Models for predicting stand development in MELA system. Finnish Forest Research Institute, Research Papers 835, 1-116.

IPCC, 1996. Intergovermental Panel for Climate Change. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, Vol. 1. Reporting Instructions.

—, 2000. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories.

—, 2003. Good practice guidance for land use, land-use change and forestry, Institute for Global Environmental Strategies (IGES), Japan.
—, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, IGES, Japan. www.ipcc-nggip.iges.or.jp/public/2006gl.
Izaurralde, R.C., Haugen-Kozyra, K.H., Jans, D.C., McGill, W.B., Grant, R.F., Hiley, J.C., 2001. Soil C dynamics: Measurement, Simulation and site-to-region scale-up. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), Assessment methods for soil carbon. Lewis Publishers, New York, NY.
Janssens, I.A., Freibauer, A., Ciais, P., Smith, P., Nabuurs, G.J., Folberth, G., Schlamadinger, B., Hutjes, R.W.A., Ceulemans, R., Schulze, E.D., Valentini, R., Dolman, A.J., 2003. Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic CO_2 emissions. Science 300, 1538-1542.
Jenkins, J.C., Chojnacky, D.C., Heath, L.S., Birdsey, R.A., 2003. National-Scale Biomass Estimators for United States Tree Species. Forest Science 49, 12-35.
Jenny, H., 1941. The factors of soil formation McGraw-Hill, New York.
Johnson, M.G., Kern, J.S., 2002. Quantifying the organic carbon held in forested soils of the United States and Puerto Rico. In: Kimble, J.M., Heath, L.S., Birdsey, R.A., Lal, R. (Eds.), The potential of U.S. forest soils to sequester and mitigate the greenhouse effect. Lewis, Boca Raton.
Jones, R.J.A., Hiederer, R., Rusco, E., Loveland, P.J., Montanarella, L., 2004. The map of organic carbon in topsoils in Europe, Version 1.2, September 2003: Explanation of Special Publication Ispra 2004 No.72 (S.P.I.04.72). European Soil Bureau Research Report 17, 1-26.
Kauppi, P.E., Posch, M., Hänninen, P., Henttonen, H., Ihalainen, A., Lappalainen, E., Starr, M., Tamminen, P., 1997. Carbon reservoirs in peatlands and forests in the boreal regions of Finland. Silva Fennica 31, 13-25.
Knapp, A.K., Smith, M.D., 2001. Variation among biomes in temporal dynamics of aboveground primary production. Science 291, 481-484.
Knorr, W., Prentice, I.C., House, J.I., Holland, E.A., 2005. Long-term sensitivity of soil carbon turnover to warming. Nature 433, 298-301.
Korhonen, K.T., Heikkinen, J., Henttonen, H., Ihalainen, A., Pitkänen, J., Tuomainen, T., 2006. Suomen metsävarat 2004-2005. Metsätieteen aikakauskirja 1B, 183-221.
Kortelainen, P., Rantakari, M., Huttunen, J.T., Mattsson, T., Alm, J., Juutinen, S., Larmola, T., Silvola, J., Martikainen, P.J., 2006. Sediment respiration and lake trophic state are important predictors of large CO2 evasion from small boreal lakes. Global Change Biology 12, 1554-1567.
Kott, P.S., 1985. A Note on model-based stratification (incl. reply from R. L. Wright). Journal of Business & Economic statistics 3, 284-288.
Kouki, J., Hokkanen, T., 1991. Long-term needle litterfall of a Scots pine Pinus sylvestris stand: relation to temperature factors. Oecologia 89, 176-181.
Kunkle, J., Walters, M., Kobe, R., 2005. Nutrient resorption from senescing fine roots of four northern temperate tree species. Presentation at ESA 2005 Annual meeting. http://abstracts.co.allenpress.com/pweb/esa2005/document/?ID=50925.
Kurz, W.A., Apps, M.J., 1994. The carbon budget of Canadian forests: a sensitivity analysis of changes in distrubance regimes, growth rates, and decomposition rates. Environmental Pollution 83, 55-61.
—, Apps, M.J., 1999. A 70-year retrospective analysis of carbon fluxes in the Canadia forest sector. Ecological Applications 9, 526-547.

Kärkkäinen, L., 2005. Evaluation of performance of tree-level biomass models for forestry modeling and analyses (dissertation). Metsäntutkimuslaitoksen tiedonantoja Metla, Helsinki, 940, pp. 123.

Kätterer, T., Reichstein, M., Andrén, O., Lomander, A., 1998. Temperature dependence of organic matter decomposition: a critical review using literature data analyzed with different models. Biology and Fertility of Soils 27, 258-262.

Lagergren, F., Grelle, A., Lankreijer, H., Molder, M., Lindroth, A., 2006. Current carbon balance of the forested area in Sweden and its sensitivity to global change as simulated by Biome-BGC. Ecosystems 9, 894-908.

Laitat, E., Karjalainen, T., Loustau, D., Lindner, M., 2000. Towards an integrated scientific approach for carbon accounting in forestry. Biotechnology, Agronomy, Society and Environment 4, 241-251.

Lehtonen, A., 2005a. Carbon stocks and flows in forest ecosystems based on forest inventory data. Faculty of Agriculture and Forestry, University of Helsinki, Helsinki, Dissertationes Forestales 11, pp. 51.

—, 2005b. Estimating foliage biomass for Scots pine (Pinus sylvestris L.) and Norway spruce (Picea abies (L.) Karst.) plots. Tree Physiology 25, 803-811.

—, Mäkipää, R., Heikkinen, J., Sievänen, R., Liski, J., 2004a. Biomass expansion factors (BEF) for Scots pine, Norway Spruce and birch according to stand age for boreal forests. Forest Ecology and Management 188, 211-224.

—, Sievänen, R., Mäkelä, A., Mäkipää, R., Korhonen, K.T., Hokkanen, T., 2004b. Potential litterfall of Scots pine branches in southern Finland. Ecological Modelling 180, 305-315.

Lettens, S., Van Orshoven, J., Van Wesemael, B., Muys, B., Perrin, D., 2005. Soil organic carbon changes in landscape units of Belgium between 1960 and 2000 with reference to 1990. Global Change Biology 11, 2128-2140.

Levy, P.E., Hale, S.E., Nicoll, B.C., 2004. Biomass expansion factors and root: shoot ratios for coniferous tree species in Great Britain. Forestry 77, 421-430.

Li, C.S., Aber, J., Stange, F., Butterbach-Bahl, K., Papen, H., 2000. A process-oriented model of N_2O and NO emissions from forest soils: 1. Model development. Journal of Geophysical Research-Atmospheres 105, 4369-4384.

Liski, J., Lehtonen, A., Palosuo, T., Peltoniemi, M., Muukkonen, P., Eggers, T., Mäkipää, R., 2006. Carbon accumulation in Finland's forests 1922-2004 - an estimate obtained by combination of forest inventory data with modelling of biomass, litter and soil. Annales of Forest Science 63, 687-697.

—, Nissinen, A., Erhard, M., Taskinen, O., 2003. Climatic effects on litter decomposition from arctic tundra to tropical rainforest. Global Change Biology 9, 575-584.

—, Palosuo, T., Peltoniemi, M., Sievänen, R., 2005. Carbon and decomposition model Yasso for forest soils. Ecological Modelling 189, 168-182.

—, Perruchoud, D., Karjalainen, T., 2002. Increasing carbon stocks in forest soils of western Europe. Forest Ecology and Management 169, 159-175.

—, Westman, C.J., 1995. Density of organic carbon in soil at coniferous forest sites in southern Finland. Biogeochemistry 29, 183-197.

—, Westman, C.J., 1997. Carbon storage in forest soil of Finland. 1. Effect of thermoclimate. Biogeochemistry 36, 239-260.

Liu, S.R., Bliss, N., Sundquist, E., Huntington, T.G., 2003. Modeling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. Global Biogeochemical Cycles 17, 1-24.

Majdi, H., 2001. Changes in fine root production and longevity in relation to water and nutrient availability in a Norway spruce stand in northern Sweden. Tree Physiology 21, 1057-1061.

Marklund, L.G., 1988. Biomassfunktioner för tall, gran och björk i Sverige. Sveriges Lantbruksuniversitet, Rapporter-Skog 45, 1-73.

Matala, J., Hynynen, J., Miina, J., Ojansuu, R., Peltola, H., Sievänen, R., Väisänen, H., Kellomäki, S., 2003. Comparison of a physiological model and a statistical model for prediction of growth and yield in boreal forests. Ecological Modelling 161, 95-116.

Matamala, R., Gonzalez Meler, M.A., Jastrow, J.D., Norby, R.J., Schlesinger, W.H., 2003. Impacts of fine root turnover on forest NPP and soil C sequestration potential. Science 302, 1385-1387.

Mattsson, T., Kortelainen, P., Räike, A., 2005. Export of DOM from boreal catchments: impacts of land use cover and climate. Biogeochemistry 76, 373-294.

McGill, W.B., 1996. Review and classification of ten soil organic matter (SOM) models. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of soil organic matter models - using existing long-term datasets. NATO ASI Series Vol. I 38 Springer-Verlag, Heidelberg.

Melillo, J.M., Aber, J.D., Muratore, J.F., 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. Ecology 63, 621-626.

Metla, 2005. Finnish statistical yearbook of forestry. Finnish Forest Reasearch Institute, Vammalan kirjapaino Oy, Vammala.

—, 2006. Finnish statistical yearbook of forestry. Finnish Forest Reasearch Institute, Vammalan kirjapaino Oy, Vammala.

Miglietta, F., Gioli, B., Hutjes, R.W.A., Reichstein, M., 2007. Net regional ecosystem CO2 exchange from airborne and ground-based eddy covariance, land-use maps and weather observations. Global Change Biology 13, 548-560.

Mikola, P., 1960. Comparative experiment on decomposition rates of forest litter in Southern and northern Finland. Oikos 11, 161-167.

Mitchell, T.D., Carter, T.R., Jones, P.D., Hulme, M., New, M., 2003. A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901-2000) and 16 scenarios (2001-2100). Tyndall Centre Working Paper 55, 1-25.

Monni, S., 2005. Estimation of country contributions to the climate change. Viewpoints of radiative forcing and uncertainty of emissions VTT Publications 577, Espoo.

Monson, R.K., Holland, E.A., 2001. Biospheric trace gas fluxes and their control over trophospheric chemistry. Annual Review of Ecology and Systematics 32, 547-576.

Morgan, M., Henrion, M., 1990. Uncertainty. A guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge University Press, Cambridge.

Muukkonen, P., 2005. Needle biomass turnover rates of Scots pine (Pinus sylvestris L.) derived from the needle-shed dynamics. Trees - Structure and Function 19, 273-279.

—, 2006. Forest inventory-based large-scale forest biomass and carbon budget assessment: new enhanced methods and use of remote sensing for verification. Department of Geography, Faculty of Science, University of Helsinki, Dissertitationes Forestales 30, Helsinki, pp. 49.

—, 2007. Generalized allometric volume and biomass equations for some European tree species. European Journal of Forest Research, In press.

—, Heiskanen, J., 2005. Estimating biomass for boreal forests using ASTER satellite data combined with standwise forest inventory data. Remote Sensing of Environment 99, 434-447.

—, Lehtonen, A., 2004. Needle and branch biomass turnover rates of Norway spruce (Picea abies). Canadian Journal of Forest Research 34, 2517-2527.

—, Mäkipää, R., 2006. Empirical biomass models of understorey vegetation in boreal forests according to stand and site attributes. Boreal Environment Research 11, 355-369.

—, Mäkipää, R., Laiho, R., Minkkinen, K., Vasander, H., Finer, L., 2006. Relationship between biomass and percentage cover in understorey vegetation of boreal coniferous forests. Silva Fennica 40, 231-245.

Mälkönen, E., 1974. Annual primary production and nutrient cycle in some Scots pine stands. Communicationes Instituti Forestalis Fenniae 84.5, 1-87.

Nabuurs, G.J., Schelhaas, M.J., Mohren, G.M.J., Field, C.B., 2003. Temporal evolution of the European forest sector carbon sink from 1950 to 1999. Global Change Biology 9, 152-160.

Nilsson, C., Stjernquist, I., Barring, L., Schlyter, P., Jonsson, A.M., Samuelsson, H., 2004. Recorded storm damage in Swedish forests 1901-2000. Forest Ecology and Management 199, 165-173.

NIR, 2007. Greehouse gas emissions in Finland 1990-2005. National inventory report to the UNFCCC.

O'Neill, R.V., DeAngelis, D.L., Waide, J.B., Allen, T.F.H., 1986. A hierarchical concept of ecosystems Princeton University Press, Princeton NJ, USA.

Oades, J.M., 1988. The retention of organic matter in soils. Biogeochemistry 5, 35-70.

Ogle, S.M., Breidt, F.J., Easter, M., Williams, S., Paustian, K., 2007. An empirically based approach for estimating uncertainty associated with modelling carbon sequestration in soils. Ecological Modelling 205, 453-463.

—, Breidt, F.J., K., P., 2006. Bias and variance in model results associated with spatial scaling of measurements for parameterization in regional assessments. Global Change Biology 12, 516-523.

—, Paustian, K., 2005. Soil organic carbon as an indicator of environmental quality at the national scale: inventory monitoring methods and policy relevance. Canadian Journal of Soil Science 85, 531-540.

Ojansuu, R., 2007. Personal communication, 15.5.2007.

—, Henttonen, H., 1983. Estimation of local values of monthly mean temperature, effective temperature sum, and precipitation sum from the measurements made by the Finnish Meteorological Office [In Finnish with English summary]. Silva Fennica 17, 143-160.

Palmer, W.C., Havens, A.V., 1958. A graphical technique for determining evapotranspiration by the Thornthwaite method. Monthly Weather Review, 123-128.

Palosuo, T., Liski, J., Trofymow, J.A., Titus, B.D., 2005. Litter decomposition affected by climate and litter quality–Testing the *Yasso* model with litterbag data from the Canadian intersite decomposition experimen. Ecological Modelling 189, 183-198.

Papale, D., Valentini, A., 2003. A new assessment of European forests carbon exchanges by eddy fluxes and artificial neural network spatialization. Global Change Biology 9, 525-535.

Peltoniemi, M., Thürig, E., Ogle, S., Palosuo, T., Schrumpf, M., Wutzler, T., Butterbach-Bahl, K., Chertov, O., Komarov, A., Mikhailov, A., Gärdenäs, A., Perry, C., Liski, J., Smith, P., Mäkipää, R., 2007. Models in country scale carbon accounting of forest soils. Silva Fennica 41, 575-602.

Prescott, C.E., 1995. Does Nitrogen Availability Control Rates of Litter Decomposition in Forests. Plant and Soil 169, 83-88.

Prince, S.D., Steininger, M.K., 1999. Biophysical stratification of the Amazon basin. Global Change Biology 5, 1-22.

Rastetter, E.B., 1996. Validating models of ecosystem response to global change. Bioscience 46, 190-198.

—, Aber, J.D., Peters, D.P.C., Ojima, D.S., Burke, I.C., 2003. Using mechanistic models to scale ecological processes across space and time. Bioscience 53, 68-76.

—, King, A.W., Cosby, B.J., Hornberger, G.M., O'Neill, R.V., Hobbie, J.E., 1992. Aggregating fine-scale ecological knowledge to model coarser-scale attributes of ecosystems. Ecological Applications 2, 55-70.

Repola, J., Ojansuu, R., Kukkola, M., 2007. Biomass functions for Scots pine, Norway spruce and birch in Finland. Working Papers of the Finnish Forest Research Institute 53, 1-28.

Rolff, C., Ågren, G.I., 1999. A model study of nitrogen limited forest growth. Ecological Modelling 118, 193-211.

Rosenqvist, A., Milne, A., Lucas, R., Imhoff, M., Dobson, C., 2003. A review of remote sensing technology in support of the Kyoto Protocol. Environmental Science & Policy 6, 441-455.

Roxburgh, S.H., Berry, S.L., Buckley, T.N., Barnes, B., Roderick, M.L., 2005. What is NPP? Inconsistent accounting of respiratory fluxes in the definition of net primary production. Functional Ecology 19, 378-382.

Running, S.W., Nemani, R.R., Heinsch, F.A., Zhao, M.S., Reeves, M., Hashimoto, H., 2004. A continuous satellite-derived measure of global terrestrial primary production. Bioscience 54, 547-560.

Saksa, T., Uuttera, J., Kolstrom, T., Lehikoinen, M., Pekkarinen, A., Sarvi, V., 2003. Clear-cut detection in boreal forest aided by remote sensing. Scandinavian Journal of Forest Research 18, 537-546.

Salminen, H., Lehtonen, M., Hynynen, J., 2005. Reusing legacy FORTRAN in the *MOTTI* growth and yield simulator. Computers and Electronics in Agriculture 49, 103-113.

Scott, C.T., 1998. Sampling methods for estimating change in forest resources. Ecological Applications 8, 228-233.

Segers, R., 1998. Methane production and methane consumption: a review of processes underlying wetland methane fluxes. Biogeochemistry 41, 23-51.

Shinozaki, K., Yoda, K., Hozumi, K., Kira, T., 1964. A quantitative analysis of plant form: the pipe model theory. I Basic analyses. Japanese Journal of Ecology 14, 133-139.

Six, J., Callewaert, P., Lenders, S., de Gryze, S., Morris, S.J., Gregoriche, E.G., Paul, E.A., Paustian, K., 2002. Measuring and understanding carbon storage in afforested soils by physical fractionation. Soil Science Society of America Journal 66, 1981-1987.

Smith, J.E., Heath, L.S., 2002. A model of forest floor carbon mass for United States forest types. Res. Pap. NE-722 USDA Forest Service, Northeastern Research Station, Newtown Square, PA.

—, Heath, L.S., Jenkins, J.C., 2003. Forest tree volume-to-biomass models and estimates for live and standing dead trees of U.S. forests. Gen. Tech. Rep. NE-298. U.S. Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, Pennsylvania, 57 p.

Smith, P., 2001. Soil organic matter modeling. In: Lal, R. (Ed.) *Encyclopedia of Soil Science*. Marcel Dekker Inc, NY.

—, 2002. Soil and the environment: Role of soil in models for climate change. In: Hillel, D., Rosenzweig, C., Powlson, D., Scow, K., Singer, M., Sparks, D. (Eds.), *Encyclopedia of Soils in the Environment*. Academic Press, London.

—, 2004. How long before a change in soil organic carbon can be detected? Global Change Biology 10, 1878-1883.

Smith, W.B., Vissage, J.S., Darr, D.R., Sheffield, R.M., 2001. Forest resources of the United States, 1997. Gen. Tech. Rep. NC-219. U.S. Department of Agriculture, Forest Service, North Central Research Station, St. Paul, Minnesota, 191 p.

Somogyi, Z., Cienciela, E., Mäkipää, R., Muukkonen, P., Lehtonen, A., Weiss, P., 2007. Indirect methods of large-scale forest biomass estimation. European Journal of Forest Research 126, 197-207.

Starr, M., Saarsalmi, A., Hokkanen, T., Merila, P., Helmisaari, H.-S., 2005. Models of litterfall production for Scots pine (Pinus sylvestris L.) in Finland using stand, site and climate factors. Forest Ecology and Management 205, 215-225.

Ståhl, G., Boström, B., Lindqvist, H., Lindroth, A., Nilsson, J., Olsson, M., 2004. Methodological options for quantifying changes in carbon pools in Swedish forests. Studia Forestalia Suecica 214, 1-46.

Suni, T., Berninger, F., Markkanen, T., Keronen, P., Rannik, U., Vesala, T., 2003. Interannual variability and timing of growing-season CO_2 exchange in a boreal forest. Journal of Geophysical Research 108 (D9), 4265.

Sverdrup, H., Warfvinge, P., Blake, L., Goulding, K., 1995. Modelling recent and historic soil data from the Rothamsted Experimental Station, UK using SAFE. Agriculture, Ecosystems & Environment 53, 161-177.

Sverdrup, H.U., 1990. The Kinetics of Base Cation Release Due to Chemical Weathering. Lund University Press, Lund.

Tamminen, P., 1991. Kangasmaan ravinnetunnusten ilmaiseminen ja viljavuuden alueellinen vaihtelu Etelä-Suomessa. Summary: Expression of soil nutrient status and regional variation in soil fertility of forested sites in southern Finland. Folia Forestalia 777, 3-29.

Tapio, 2001. Hyvän metsänhoidon suositukset (Recommendations for good silviculture). Metsätalouden kehittämiskeskus Tapio. Libris Oy, Helsinki, Finland (In Finnish).

Thürig, E., 2005. Carbon budget of Swiss forests: evaluation and application of empirical models for assessing future management impacts. PhD Thesis, Swiss Federal Institute of Technology, Zürich, pp. 125.

—, Palosuo, T., Bucher, J., Kaufmann, E., 2005. The impact of storm damage on carbon sequestration in Switzerland: a model-based assessment. Forest Ecology and Management 210, 337-350.

Tokola, T., 2006. Part III Cases: Europe, Asia, North America. In: Kangas, A., Maltamo, M. (Eds.), Forest inventory, Methodology and applications. Managing forest ecosystems 10. Springer, Dordrecht, The Netherlands.

Tomppo, E., 2006. The Finnish multi-source national forest inventory - small area estimation and map production. In: Kangas, A., Maltamo, M. (Eds.), Forest inventory, Methodology and applications. Managing forest ecosystems 10. Springer, Dordrecht, The Netherlands.

Turner, J., 1975. Nutrient cycling in a Douglas-fir ecosystem with respect to age and nutrient status, PhD thesis. University of Washington.

UNECE, 2000. Forest Resources of Europe, CIS, North America, Australia, Japan and New Zealand United Nations, Geneva.

UNFCCC, 1992. United Nations Framework Convention on Climate Change. http://www.unfccc.de/resource/convkp.html.

—, 1997. Kyoto Protocol. http://www.unfccc.de/resource.

—, 2001a. FCCC/CP/2001/5.

—, 2001b. Forest management activities under Article 3, paragraph 4, of the Kyoto Protocol: the Russian Federation. FCCC/CP/2001/13/Add.1. http://unfccc.int/.

—, 2001c. Matters relating to land use, land-use change and forestry. FCCC/CP/2001/L.11/ Rev.1. http://unfccc.int/resource/docs.html.

US-EPA, 2006. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2004. EPA 430-R-06-002 U.S. Environmental Protection Agency, Washington, D.C.

Valentini, R., Matteucci, G., Dolman, A.J., Schulze, E.-D., Rebmann, C., Moors, E.J., Granier, A., Gross, P., Jensen, N.O., Pilegaard, K., Lindroth, A., Grelle, A., Bernhofer, C., Grünwald, T., Aubinet, M., Celeumans, R., Kowalski, A.S., Vesala, T., Rannik, Ü., Berbigier, P., Loustau, D., Guðmundsson, J., Thorgeirsson, H., Ibrom, A., Morgenstern, K., Clement, R., Moncrieff, J., Montagnani, L., Minerbi, S., Jarvis, P.G., 2000. Respiration as the main determinant of carbon balance in European forests. Nature 404, 861-865.

van Heerwaarden, L.M., Toet, S., Aerts, R., 2003. Current measures of nutrient resorption efficiency lead to a substantial underestimation of real resorption efficiency: facts and solutions. Oikos 101, 664-669.

van Oijen, M., Rougier, J., Smith, R., 2005. Bayesian calibration of process-based forest models: bridging the gap between models and data. Tree Physiology 25, 915-927.

Vanninen, P., Mäkelä, A., 1999. Fine root biomass of Scots pine stands differing in age and soil fertility in southern Finland. Tree Physiology 19, 823-830.

—, Ylitalo, H., Sievänen, R., Mäkelä, A., 1996. Effects of age and site quality on the distribution of biomass in Scots pine (Pinus sylvestris L.). Trees 10, 231-238.

Wardle, D.A., Hörnberg, G., Zackrisson, O., Kalela-Brundin, M., Coomes, D.A., 2003. Long-term effects of wildfire on ecosystem properties across an island area gradient. Science 300, 972-975.

Viro, P.J., 1955. Investigations on forest litter. Communicationes Instituti Forestalis Fenniae 45 (6), 1-142.

Wirth, C., Schumacher, J., Schulze, E.D., 2004. Generic biomass functions for Norway spruce in Central Europe - a meta-analysis approach toward prediction and uncertainty estimation. Tree Physiology 24, 121-139.

Wutzler, T., Reichstein, M., 2007. Soils apart from equilibrium - consequences for soil carbon balance modelling. Biogeosciences 4, 125-136.

Zianis, D., Muukkonen, P., Mäkipää, R., Mencuccini, M., 2005. Biomass and stem volume equations for tree species in Europe. Silva Fennica Monographs 4, pp. 63.

Ågren, G.I., Hyvönen, R., 2003. Changes in carbon stores in Swedish forest soils due to increased biomass harvest and increased temperatures analysed with a semi-empirical model. Forest Ecology and Management 174, 25-37.

—, Hyvönen, R., Nilsson, T., 2007. Are Swedish forest soils sinks or sources for CO2 - model analyses based on forest inventory data. Biogeochemistry 82, 217-227.