Chapter 10
Monitoring Carbon Stock Changes in European Forests Using Forest Inventory Data

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10.1 Introduction

The forest carbon stock in Europe is large and changes in it may contribute notably to atmospheric CO_2 concentration. The area of forested land in Europe is about 1,000 million ha, which is about 47% of the land area (e.g. MCPFE 2003). The largest part, more than 800 million ha, of the forests in Europe is located in the Russian Federation, whereas the forested area of EU15 is 137 million ha. The percentage of forested land varies considerably between counties ranging from 68% in Finland and Sweden to 1% in Iceland. In the forests of Europe (excluding Russia), the carbon stock of the vegetation was estimated to be about 8,000 Tg (Nabuurs et al. 1997; Goodale et al. 2002). Estimates of the soil carbon stocks range from 5,000 to 14,000 Tg and are evidently more uncertain than the estimated carbon stocks of the vegetation (Goodale et al. 2002; Liski et al. 2002; Nabuurs et al. 2003). Current estimates of the changes in the carbon stock of vegetation in Europe (excluding Russia) range from 50 to 100 Tg C year^{-1}, and the changes in the soil range from 13 to 61 Tg C year^{-1}, depending on the methods applied as well as on the reference area and period (Goodale et al. 2002; Liski et al. 2002; Karjalainen et al. 2003; Nabuurs et al. 2003). In general, large carbon stocks of peatlands soils are not fully accounted in these studies, because of the lack of representative data and/or models.

The changes in the carbon stock of forested areas are partly due to annual variation in net primary production (NPP) and soil respiration, both resulting from climatic variation (Ciais et al. 2005). Furthermore, most of the annual changes in the forest carbon stock are the result of variation in commercial harvests and natural disturbances (Kauppi et al. 1992; UNECE 2000; Nabuurs et al. 2003). In general, the role of natural disturbance has been marginal in comparison to magnitude of commercial harvests in Europe. Exceptionally severe storms in 1999 reduced the carbon sink of biomass in that year (by 52 Tg C), but the increasing trend in the net carbon sink in Europe was only briefly interrupted by these large-scale disturbances (Nabuurs et al. 2003). Under future climatic conditions, the frequency of large-scale disturbances may, however, increase remarkably and they may also play a larger role in Europe's forests.

In addition to the annual variation, long-term trends can also be found in the forest carbon sequestration of Europe. In general, the forest carbon stock of a country tends to increase as a result of forest management and increased growing stock (UNECE 2000; Nabuurs et al. 2003; Lehtonen 2005; Liski et al. 2006). In Northern Europe, the increase in forest carbon stock may be partly a recovery from the period of extensive slash-and-burn cultivation and from overuse of timber resources in the 1800s and early 1900s, especially in Finland. In addition, the growth and biomass stock of trees have increased by drainage and amelioration of peatland forests (Hökkä et al. 2002; Minkkinen et al. 2002). In central Europe, increase in forest carbon stock is partly the result of increase in forested area (MCPFE 2003). The increase in forested area, as a result of afforestation projects and abandonment of agricultural fields, has been remarkable and has played a major role in forest carbon sequestration, for example, in the United Kingdom (Cannell and Dewar 1995). In Scandinavia, as much as 68% of the land area is forested and afforestation has played smaller role in carbon sequestration (Mäkipää and Tomppo 1998; IPCC 2000; UNECE 2000). However, change from other wooded land has increased the area of forested land, especially in Finland, where drainage of peatland has resulted in improved productivity and transformation of less productive peatlands to forested land. This transformation has evidently increased the carbon stock of trees, but the response of soil greenhouse gas (GHG) balance to drainage is more uncertain (Minkkinen et al. 2002).

On a European scale, monitoring of changes in the carbon stock of forest vegetation and soil is challenging because design of the monitoring system should be capable of detecting all these major trends as well as the annual variation that contributes to the carbon balance of forests. Because the factors that control forest carbon balance operate at different spatial and temporal scales, it is necessary to complement existing monitoring networks such as national forest inventories (NFIs) with statistical and dynamic modelling to establish a proper carbon-monitoring system for the forests of Europe.

10.2 Forest Inventories as a Source of Information for Assessment of Carbon

National forest inventories that are designed for monitoring of forest resources at national scales provide a firm basis for large-scale carbon assessments. Forest inventories that cover the entire country have been conducted in Scandinavia since the 1920s, while in France and in Germany the first NFIs were conducted in 1960 and in 1981, respectively (Laitat et al. 2000) (Table 10.1). Time span, sampling methods and quantity (and quality) of the information collected by NFIs vary among countries according to local conditions and information needs (UN-ECE/FAO 1985; UNECE 2000) (Table 10.1), but recent efforts of the research community are also aiming to harmonise inventory methods (COST E43 project, http://www.metla.fi/eu/cost/e43/). In Western European countries with substantial forest industries,

Table 10.1 National Forest Inventories as Data Sources (adopted and modified from Laitat et al. 2000). Country codes according to ISO 3166-1 Aplha-2 code elements, AT = Austria, BE = Belgium, CH = Switzerland, DE = Germany, DK = Denmark, ES = Spain, FI = Finland, FR = France, GB = United Kingdom, GR = Greece, IR = Ireland, IS = Iceland, NL = The Netherlands, NO = Norway, and SE = Sweden

Country	AT	BE	CH	DE	DK	ES	FI	FR	GB	GR	HU	IR	IS	NL	NO	SE
Start	1961	1980 1994[h]	1983	1986	1981	1964	1921	1960	1924	1963	1950	1998	1972	1940	1919	1923
Inventory type[a]	SB	SB	SB	SB	ST	SB	SB	SB	SA	SA	ST	–	–	SA	SA	SA
Periodicity	5	10	10	15	10	10–20	10	10	15–20	30	10	–	–	–	–	–
Forest area[b] (1,000 ha)	3,862	667	1,221	11,076	500	17,915	22,500	15,554	2,845	3,752	1,976	669	46	365	9,387	27,528
Grid	3.9×3.9	1 × 1	1.4×1.4	4×4	–	1 × 1	6 × 6	various	various	1 × 0.5	–	–	–	–	3×3	various
Plots	11,000	10,600	6,500	12,580	–	84,203	70,000	133,500	–	2,744	–	–	–	3,400	10,500	18,000
Percentage S.E. in area	1.2	0.42	0.3	1.1	–	–	0.48	0.71	–	0.2	–	1	–	–	0.96	0.5
Percentage S.E. in volume	1.6	5.1	1	0.8	–	1.13	0.57	0.54	–	2.6	–	–	–	5	1.36	0.6
Percentage S.E. in volume growth	–	–	0.9	–	–	–	0.8	0.59	–	–	–	–	–	–	1.4	0.4
Biomass, above ground[c,d]	E	E	E	E, F	E	E	E	–	O	E	E	E	O[e]	F	F	F
Biomass, below ground[c,d]	E	E	E	E, F	E	E	E	–	O	E	–	–	–	F	F	F
Origin of biomass functions/ expansion factors[c,d]	CS	CN	D	CS, D	CN	CS	CS	–	CS	D, CS	CS	CS	–	CS	CN	CS

(continued)

Table 10.1 (continued)

Country	AT	BE	CH	DE	DK	ES	FI	FR	GB	GR	HU	IR	IS	NL	NO	SE
Soil estimate in NIR	–	–	–	–	–	–	–	–	YES[f]	–	–	–	–	–	YES[f]	–
DOM estimate in NIR	–	–	–	–	–	–	–	–	YES[f]	YES	–	–	–	YES	YES[f]	–
Nationwide soil survey existing[g]	–	YES	–	–	YES	–	–	YES	YES	–	–	–	–	YES	–	YES
Repeated sampling in soil survey	–	–	–	–	–	–	–	–	YES	–	–	–	–	–	–	YES

[a]SB refers to sample-based inventory, while ST refers to stand-wise inventory (based on forest compartments)
[b]According to FAO forest definition, source of data: www.fao.org/forestry/
[c]*E* Biomass expansion factor, *F* Biomass function (tree-level accounting), *O* Other method, *CS* Country-specific model, *CN* model adopted from neighbouring countries, *D* IPCC default
[d]Information based on Annex I Party GHG Inventory Submissions, National Inventory Reports (NIR) 2005 from http://unfccc.int
[e]Only afforested land in calculation
[f]Modelled estimate
[g]Based on Jones et al. 2004
[h]Respectively for the Walloon and Flemish regions

systematic samples of forest sites have been collected several times, while Eastern European countries have conducted their first rounds of systematic NFI only recently. Before the current NFIs were established, forest resources were quantified based on regional data compilations that often under- or overestimated target variables, such as growing stock on a national level (UN-ECE/FAO 1985; UNECE 2000). Traditionally, an NFI covering a large country required almost a decade to carry out, while smaller countries were able to accomplish their NFIs in a few years. Because the most important target variables of the NFIs have been forested area, growing stock and increment, the inventories are designed and optimised for quantifying these variables. However, due to the long rotation times of NFIs, the annual estimates of increment were not directly available. Therefore, estimation of annual carbon stock changes is based on interpolation or on extrapolation.

10.3 Carbon Inventories Under the Climate Convention are Guided by IPCC

The United Nations Framework Convention on Climate Change (UNFCCC 1992), aims at stabilising the GHG concentrations in the atmosphere at a non-dangerous level and the Kyoto Protocol (UNFCCC 1997) under it, setting legally binding commitments to reduce emissions in the industrialised countries. In the Kyoto Protocol, forest carbon sinks were accepted as one of the mitigation options of climate change. The protocol allows that a limited part of the forest carbon sinks can compensate for the emission reductions of a country. Because the credited sinks may compensate for internationally agreed emission reductions, the need for more accurate sink estimates and more transparent reporting became evident.

Requirements concerning GHG reporting were already defined in the Climate Convention and in the Kyoto Protocol, but more detailed reporting rules were set in subsequent meetings (e.g. in Bonn and Marrakesh 2001). At the request of the Convention, the Intergovernmental Panel on Climate Change (IPCC) has developed guidance for estimation and reporting of GHG emissions and removals of land use, land-use change and forestry sectors including all different carbon pools (above- and belowground biomass, deadwood, litter and soil organic carbon) (IPCC 2003, 2006).

IPCC guidance (2003, 2006) stated that forest carbon inventories should be based on representative nationwide information on forest resources, such as forested area, area of other land-use classes and exchanges between them, estimate of growing stock, annual growth, commercial harvests and other losses. In general, this information is collected by the NFIs (UNECE 2000). Furthermore, relevant information of nationwide soil surveys or forest soil monitoring programmes can be used in the carbon inventories (IPCC 2003).

Verification of reported forest carbon stock change estimates must be conducted with independent methods and material. Verification of the forest carbon inventory can be based on remote sensing, ecosystem modelling or combinations of these (IPCC 2003). Furthermore, inverse modelling that calculates fluxes from the concentration measurements and atmospheric transport models can be used for verification on a continental scale and direct measurements of ecosystem fluxes on a local scale (IPCC 2003) (Chap. 17).

10.4 Estimation of Forest Carbon Balance Based on Inventory Data and Modelling of Litter and Soil

10.4.1 Biomass

On a national scale, changes in the biomass and carbon stocks of trees can be estimated based on NFI data. The change can be estimated based on the difference between two consecutive inventories or on estimated increment and losses (drain). These two approaches are also recommended by the IPCC (2003). The default method of the IPCC, in which estimates of both annual losses and growth are needed, assumes that countries are able to quantify all components of the drain (losses), such as natural mortality, fuelwood gathering and loggings, as well as growth on an annual basis. For example, in Finland, data on commercial drain and domestic fuel wood use are aggregated by species and gathered at the national level by questionnaires sent to wood users (both forest industry and households), while estimates of natural mortality have been derived from permanent sample plots of the NFI. In the case of Finland, the forest industry is obliged to report its wood use and therefore the estimate of the annual drain resulting from commercial harvests is considered and tested to be reliable (Kuusela 1979). In Sweden, the carbon stock change is based on consecutive measurements from permanent sample plots of NFI (Ståhl et al. 2004). The use of permanent sample plots gives a more reliable estimate of stock change than the use of temporary sample plots (with the same number of plots), since the high covariance between the following measurements increases the accuracy of the change estimate.

Countries have annually reported changes in the biomass carbon stock based on the NFI to the United Nations Framework Convention on Climate Change (UNFCCC) according to the guidelines developed by the IPCC. Usually the net changes in carbon stock have been obtained by deducting losses from the increment of growing stock, and thereafter this net change of volume has been converted to biomass by biomass expansion factors (BEFs). These BEFs were developed to obtain whole-tree biomass, including canopy and roots (e.g. IPCC 2003; Lehtonen et al. 2004; Levy et al. 2004; Lehtonen et al. 2007) or the biomass of roots can be estimated separately with shoot–root ratios available from the literature (IPCC 2003; Levy et al. 2004). The values of these BEFs and the methodology developed vary considerably among countries (Löwe et al. 2000; Somogyi et al. 2007). The main disadvantages of current BEFs are that often the uncertainty estimates are lacking or that even the origin of the estimates is unknown. Furthermore, most of the BEFs are constant and developed for conversion from stem volume to whole-tree biomass, but they are often applied for increment and losses.

Biomass equations are applied at the tree-level and tree-wise biomasses are predicted as a function of tree dimensions (such as diameter and height) and other predictors. Most of the existing biomass equations are based on a couple of sites with only a few felled sample trees (Zianis et al. 2005; Muukkonen and Mäkipää 2006a). Therefore, the main limitation of biomass equations is poor representativeness in national-scale inventories. However, a few exceptions exist, for example Marklund (1988) in which uniform sampling was conducted for the whole of Sweden. Recently there have also been other works that compiled data from various studies and derived new biomass equations, as Wirth et al. (2004) did for Norway spruce in central Europe.

In addition to the biomass equations and constant BEFs, there are other options for regional biomass estimation. Fang et al. (2001) modelled BEFs as a function of stem volume based on direct field measurements in China, while Jenkins et al. (2003) grouped biomass equations and developed generalised functions for species groups in the USA and Muukkonen (2007) developed generalised functions for the major tree species in Europe. Levy et al. (2004) modelled BEFs as a function of tree height in the UK, while Lehtonen et al. (2004) modelled BEFs in Finland as a function of stand age. In general, these biomass models use measurements or existing biomass models from single trees for development of more robust biomass estimation methods on regional or national scales.

Liski et al. (2006) estimated changes in biomass and soil carbon stocks for Finland, based on NFI data and modelling. Stem volume estimates reported by NFIs were converted to biomass components (foliage, branches, stem and roots of different sizes) with BEFs. These BEFs with uncertainty estimates were developed by age-class and dominant species (Lehtonen et al. 2004). Tree biomass is the main component of the carbon stock of vegetation, but the understorey vegetation may still play an important role, especially in litter input. The biomass of understorey vegetation was modelled as a function of stand age. The models that were based on nationwide vegetation data were complemented with models on the relationship between biomass and coverage of understorey vegetation (Muukkonen and Mäkipää 2006b; Muukkonen et al. 2006).

10.4.2 Deadwood

The amount of carbon in deadwood is highly variable between stands across the landscape, both in managed and unmanaged forests. Unmanaged or natural forests contain considerably larger quantities of deadwood than managed forests (Siitonen 2001; Jonsson et al. 2005). Coarse woody debris (deadwood) may be a notable carbon stock in forests, especially after major disturbances (Harmon et al. 1986; Krankina et al. 2001). Furthermore, changes in management practices may show wide and relatively rapid influence on the carbon stock of deadwood. Current management practices are already aiming to increase the amount of deadwood because of its value for the maintenance of biodiversity. In addition to the management practices that control the amount of deadwood left on a site, the amount of deadwood is dependent on the time and severity of the disturbances, the rate of natural mortality in general and the decay rate of deadwood (Harmon et al. 1986; Siitonen 2001; Stokland 2001).

In carbon inventories, changes in the carbon stock of deadwood are usually calculated using the stock change method or by calculating the difference between transfer of carbon into deadwood and transfer out of the deadwood pool that is modelled with a decomposition model (IPCC 2003).

The stock change method can be applied by countries in which measurements of the carbon stock of deadwood are included in the forest inventories and where such an inventory is repeated. Measurements of the volume of deadwood are conducted in the NFIs, for example, in Sweden, Norway, Switzerland and Finland, but repeated nationwide inventories of deadwood are rare. Deadwood volume is measured according to decay classes (Ståhl et al. 2001; Stokland 2001), and the volume estimates can be converted to biomass and carbon estimates by decay classes (inventory measure) using species-specific statistical models (Kruys et al. 2002; Mäkinen et al. 2006).

Change in the carbon stock of deadwood can also be calculated based on the difference between input and output of carbon in this pool. Carbon input into the deadwood pool includes harvest residues left at the site and natural mortality (both self-thinning and biomass of trees killed by natural disturbances). Carbon transfer out of the deadwood pool is carbon emitted in decomposition. The rate of decomposition of deadwood under conditions representing conditions of boreal forests was modelled by Tarasov and Birdsey (2001), Kruys et al. (2002) and Mäkinen et al. (2006), and decomposition of deadwood is also included in some soil carbon models, for example in Yasso (Liski et al. 2005).

10.4.3 Litter and Soil Carbon

The changes in litter and soil carbon are difficult to monitor (Chap. 9) because the changes in upland forest soils (forests remaining as forests) are very small in comparison to the size of carbon stocks (Chap. 9). Forest soils are also highly heterogeneous; most of the variance in mean stock is already present over very short distances (Conen et al. 2004). Consequently, large numbers of samples and monitoring sites, or a long-time interval, are required until a significant change in soil carbon can be detected (Conen et al. 2003; Smith et al. 2004)(Chap. 9). For organic soils, such as peatlands drained for forestry, changes in the soil carbon may be fast during the first decades after drainage, but uncertainty of change estimates is still high even after 30 years of drainage (Minkkinen et al. 1999). Even though the expected rate of change in soil carbon stock may be small, the vast areas make them important components of national and global carbon budgets (Chap. 12). Furthermore, change in the soil carbon stock may be in the direction opposite to that of the vegetation and overall change in the ecosystem (Liski et al. 2006).

If no repeated measurements of soil are available or measuring is deemed impractical due to high costs, modelling of soil carbon can be used to close the entire carbon budget of forests. In this approach, biomass models are applied to forest inventory data to obtain biomass estimates and the biomass estimates are further converted to litter (foliage, branches, fine roots, etc.) with biomass turnover rates. Each biomass component has a distinctive lifespan (*a*) that can be used to estimate the biomass turnover rate (a^{-1}). Various estimates of biomass turnover rates are available from literature and for some ecosystem types and some components, the estimates of biomass turnover are rather comprehensive (Afman and Kaipainen 2005).

The litter input is then decomposed with a dynamic soil carbon model. The inventory-based approach was previously presented by Kurz and Apps (1994) and Liski et al. (2002, 2006). Another modelling option for estimating soil carbon stock changes is to link the soil carbon or its changes directly to NFI variables with statistical models (Johnson and Kern 2002; Smith and Heath 2002; Amichev and Galbraith 2004). A large variety of applicable dynamic soil models is available, with a variety of input demands, parameterisations for different types of soils and climatic ranges (e.g. Powlson et al. 1996; Smith et al. 1997; Peltoniemi et al. 2007). In the nationwide inventories, selection of a soil model is often determined by the availability of representative input data covering the entire country.

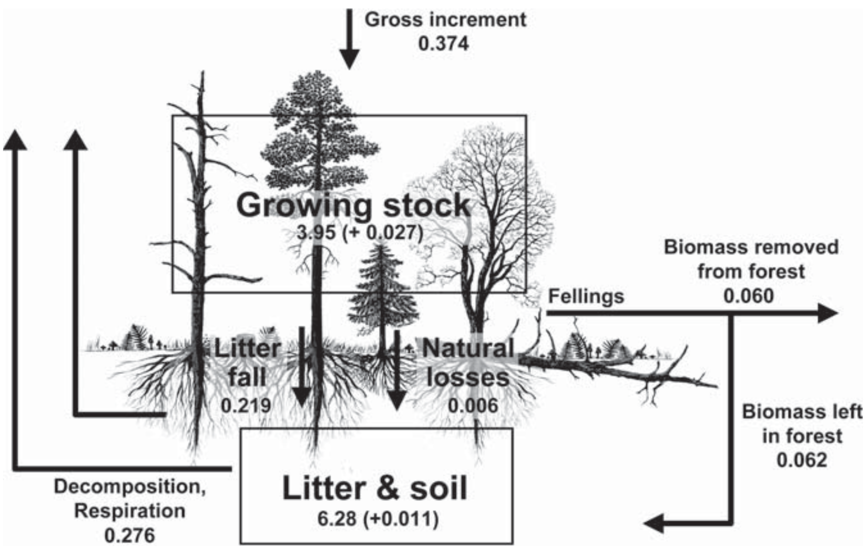

Fig. 10.1 Average carbon budget of the Finnish forests in the 1990s (according to Liski et al. 2006). The average carbon stocks (kg C m^{-2}) and the annual changes in carbon stocks (kg C m^{-2} $year^{-1}$) are shown in boxes and annual carbon fluxes (kg C m^{-2} $year^{-1}$) are illustrated by arrows (graph by Essi Puranen).

In general, the dynamic soil models that are applicable for mineral soils and for the litter layer of peatlands are not capable for modelling of decomposition of peatland soils. Furthermore, organic soils may also be sources of non-CO_2 GHG emissions and, for example, the emissions of N_2O may be significant in their relative GHG impact (Alm et al. 2007b). In the GHG inventory of the organic peatland soils, fluxes of CO_2, CH_4 and N_2O need to be estimated (IPCC 2003, 2006). For that, flux measurements must be available for different peatland ecosystems (Alm et al. 2007a) and the flux estimates have to be added to the modelled litter decomposition rate.

The benefit of the dynamic modelling approach is that it takes into account the decomposition of carbon that has accumulated in the soil over time. However, initialisation of soil carbon models with measured data is problematic because the data on soil are inaccurate and imprecise, and measured pools rarely correspond to model pools. The problem in determining the initial state of carbon can be partly avoided by initiating the calculation as far from the past as the data allow (Chen et al. 2000) and letting the model find the initial state with corresponding inputs. When the calculation is initiated far enough from the past, the error in the upward or downward trend due to initialisation is likely to be smaller than other uncertainties governing the modelled sink estimates (Peltoniemi et al. 2006).

10.4.4 Inventory-Based Soil Carbon Budget of Finland

In an application that was run for Finland, input data provided by the NFI was grouped into classes of tree species, region and age-class (Liski et al. 2006). Grouping by age was included, since age-class distribution of forests in Finland has changed notably in the last century and biomass allocation is known to be age-dependent (Satoo and Madgwick 1982; Lehtonen 2005; Metla 2006).

In addition to the above- and belowground litter produced by standing vegetation, natural disturbances (severe storms or pests) and harvestings leave decomposing material into forests. In Finland, natural disturbances have played a small role and most of the trees killed are collected from the disturbed sites. Commercial harvests in Finland are remarkable (currently about 70% of the growth is used) and a large amount of litter is left in the forest in the form of harvest residues (Metla 2006). Because the data related to timber collection are tabulated with high precision and accuracy, the carbon input of harvest residues to the soil was derived based on this information (Liski et al. 2006). Currently, Finland is targeting to higher efficiency in biofuel production with harvesting of logging residues. The effect of the harvesting of branches and root system to the forest carbon balance can be accounted with this modelling approach. Ground vegetation also produces a significant amount of litter, but little is known of its annual variability. In the application for Finland, this flow of litter to the soil was represented with mean estimates of biomass and with published estimates of its growth, which were used for its turnover.

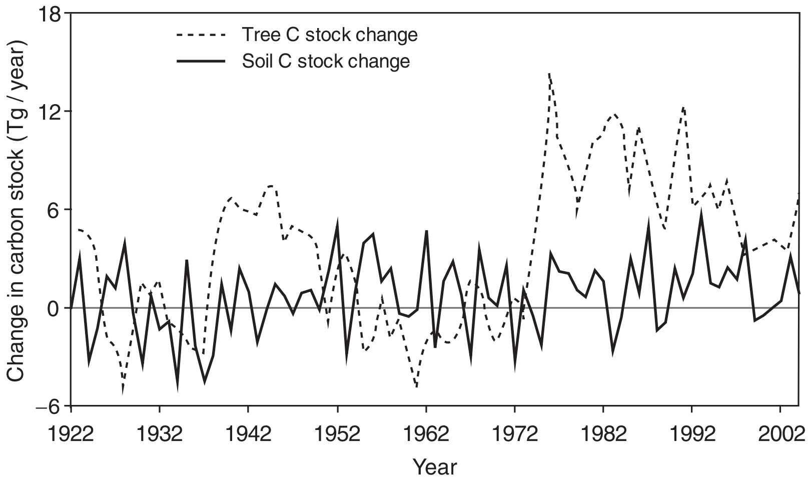

Fig. 10.2 Tree and soil carbon stock changes between 1922 and 2004 in Tg of carbon (tree stock from all soils, while soil stock comprised mineral soils) (adopted from Liski et al. 2006).

In addition to the estimates of litter input, the Yasso decomposition model used in this study required information on temperature sum and drought that was calculated as a difference between potential evaporation and precipitation (Liski et al. 2006). The soil model predicts the carbon content of litter and soil down to 1 m (on mineral soils), which was the depth used in its parameterisation. Yasso is based on the concept that chemical compounds existing in litter determine the rate at which the litter decomposes (Liski et al. 2005). This process was parameterised and tested only for mineral soils with no peat cover (Liski et al. 2005; Palosuo et al. 2005), and other models should be used for peatland soils.

The above-described approach allowed calculation of the carbon budget of forest soils in Finland for the period between 1922 and 2004 (Fig. 10.2). This study shows that all pools must be accounted for in the forest carbon inventory to avoid biased conclusions. At the same time, forest soils may be a source of carbon while trees act as a sink.

10.5 Evaluation of Inventory-Based Methods

10.5.1 Measures for Assessing Accuracy and Precision

The error sources in large-scale carbon budget compilations are numerous. The NFI data and other input data involve errors that are characteristic of measurements and sampling protocol. Models that convert measured variables to target variables are often developed with inadequate sampling of the population or the data quality may be poor (see Sect. 10.4.1). The model structure used may not represent adequately the phenomenon we are interested in (for general discussion on uncertainties, see Morgan and Henrion 1990 and Cullen and Frey 1999). Uncertainty analysis of the modelled results should be a part of any carbon budget calculation. Here we will first examine the uncertainty in the structure and submodels of the inventory-based carbon balance calculation, mostly in the form of subjective assessment. Secondly, we will view the results of a Monte Carlo analysis of uncertainty and discuss the differences in factors that affected the uncertainty of carbon sink and carbon stock estimates of the inventory-based carbon balance. Subjective assessment of the uncertainty of carbon stocks and their changes provides insight on factors that may affect the accuracy (bias) and Monte Carlo analysis the precision (random error) of the carbon sink and stock estimates.

10.5.2 Assessment of Applicability and Accuracy of the Model in Estimating Forest Carbon Budgets

Forest carbon inventories have usually been prepared with aggregated input data, that is by summing or averaging the inputs over large spatial or temporal domains before they are fed to a model (UNFCCC, National Inventory Reports, http://unfccc.int/). This was also the case in the application for Finland, where the NFI data were grouped into several classes and into the two regions of southern and northern Finland. Depending on the structure of the model, aggregation can bias the results markedly or only to a limited extent, or may not affect the results (Rastetter et al. 1992; Paustian et al. 1997). In the application for forests in Finland, aggregation does not affect the results of biomass sinks, because the summation of the biomass estimates is linear. For soil carbon stocks and sinks, however, summation may have some effect due to the non-linear effect of climate on decomposition.

A justified application of models requires that the models are used in a population similar to that for which they were developed. In practice, this is rarely the case and subjective assessment must be used in model selection. Here, the biomass of trees was estimated with BEFs (Lehtonen et al. 2004), which were developed based on Swedish biomass equations (Marklund 1987; Marklund 1988) and tested for their applicability for conditions in Finland and Sweden (Jalkanen et al. 2005; Lehtonen 2005). The major limitation of both these biomass equations and consequently BEFs was possible underestimation of the root biomass (Petersson and Ståhl 2006).

Biomass turnover models are based on data of varying quality (Afman and Kaipainen 2005). Typically, the best data are available for temperate and boreal zone countries. Similarly the data on the aboveground components of foliage and branch litter are generally better known than belowground components. Belowground turnover estimates are often lacking, which is most problematic for the rapidly regenerating component of fine roots. Because fine roots constitute a substantial percentage of the total net primary production (NPP) of forests (Jackson et al. 1997), quantification of their contribution to the carbon budget is crucial. However, the estimation of fine-root turnover is challenging because the fine-root longevity varies widely, for example, with tree species and the size of fine roots (Matamala et al. 2003). Furthermore, different methods that have been used to assess the turnover produce different results (Gaudinski et al. 2001; Tierney and Fahey 2002).

Component-wise tests of empirical data reveal whether the components give accurate and precise estimates, but they do not assess the system as a whole. Therefore, system-level tests are required but such tests are often limited by data availability and quality. The modelling approach that was used to estimate the carbon budget for forests in Finland was tested on empirical data from mineral soils in the region of southern Finland (Peltoniemi et al. 2004). Despite the difficulties in comparisons of model predictions of soil with measured material, the results showed similar trends and the same magnitude of change, that is the simulated rate of change in soil carbon stock was similar to the rate derived from chronosequence of Scots pine and Norway spruce stands.

10.5.3 Assessment of Precision of the Forest Carbon Budget with Monte Carlo Analysis of Uncertainty

Analysis of random error with the Monte Carlo simulations does not take into account failures of the model structure to explain the phenomenon, but assesses only the random part of the error as related to model parameters and input variables. In this analysis, the calculation is repeated thousands of times, each time drawing random samples from the probability density distributions of the model parameters and input variables (Morgan and Henrion 1990). As a result, probability densities are obtained for the resulting variables.

Sensitivity and uncertainty analysis of the carbon budget for forests in Finland (excluding soils of peatlands) showed that the data on annual growth variation and harvests were the most critical components of the biomass carbon sink (Peltoniemi et al. 2006). The biomass carbon stocks were mostly affected by carbon content, inventory data on growing stock and BEFs. In the Finnish application, the random error in simulated soil carbon stocks was large and mostly the result of uncertain soil model parameters. However, the random error of the soil carbon sink was only somewhat larger than that of the biomass sink. Most of the uncertainty in the soil carbon sink was caused by soil model initialisation, a source of uncertainty that decreased in effect with the time elapsed from initialisation (Fig. 10.3). The decrease was rather rapid after which two inputs, harvest residues and temperature, began to control the soil sink uncertainty. Model parameters of the major litter fractions (biomass and turnover of ground vegetation and fine roots) also contributed notably to the uncertainty in the soil sink.

Many of the variables that contributed largely to the uncertainties in the carbon budget had features in common. The input data played an important role in the sink uncertainties of vegetation and soil, because of uncorrelated error estimates for each year of the calculation period (e.g. temperature, growth variation and drain).

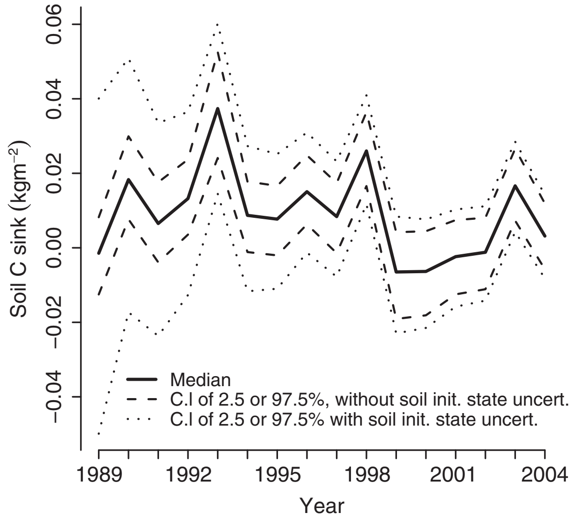

Fig. 10.3 Mean changes in the carbon stock of forest soils in Finland from 1989 to 2004 (mean and S.D. in kg C m^{-2} year^{-1}, calculated with and without soil initial state uncertainty) (adopted from Peltoniemi et al. 2006).

Use of temporal autocorrelations estimated from the input data series did not significantly affect the uncertainty estimates. The central position of the variables was another important feature in common with those variables that were important for the uncertainty in sinks or stocks of carbon. For example, the effect of carbon content was emphasised because it was a shared parameter for all tree species and biomass components.

10.5.4 Land-Use Changes

The rate of change in the soil carbon resulting from land-use shifts is large in comparison to the rate of changes in the forested areas remaining as forests (Schlesinger 1990; Post and Kwon 2000). Therefore, these land-use changes may release large quantities of carbon, which are detected in atmospheric measurements. The role of forest inventories in this assessment is to provide data on areas of different land-use categories and on their mean carbon stocks. However, sampling of forest inventories may not be sufficient to cover small changes in land use and must be appended with other data sources such as survey and satellite data. Furthermore, exact assessment of the effects of land-use change requires that each parcel of land is tracked through history, since the previous land use affects current soil carbon. Following the modelling approach for estimating regional carbon sinks and sources, the simulations should be performed for each of the land parcels, with a tool suitable for modelling the soil carbon stocks in various land-use classes.

The inventory-based approach as applied for Finland used aggregated inventory data, which provided only information on net changes in the forested areas. As such, the method is not optimised to account for the effects of land-use changes on forest carbon sinks. Land-use changes in Finland are relatively small, and originally their effect was assumed to be small. However, uncertainty analysis revealed that even small changes in area are important for soil carbon budgets. This indicates a need for more detailed accounting of the effects of land-use changes. A complementary accounting of land-use changes should be performed by preparing a land-use transition matrix that classifies and aggregates different types of land and applies conversion factors for a period of 20 years, as suggested by the IPCC 2003. For example, Heath et al. (2002) presented the methodology applied in the Forest Carbon model (FORCARB) in the USA, and Tate et al. (2003) used a modified IPCC method to estimate the effect of land-use change for soils in New Zealand.

10.6 Use in Carbon Cycle Research

Forest inventories themselves are able to provide little ecological interpretation of carbon cycle research. Forest inventories measure the realised stocks rather than the process fluxes themselves. Factors affecting the fluxes are implicitly recorded in measured stocks of timber and growth. Comparison of the estimates of large-scale forest carbon sinks obtained with different methods is challenging, and forest inventory-based carbon budgets of forests provide valuable reference data for comparisons, especially in a situation where the carbon stocks of tree biomass are facing trend-like changes. Long-term changes in biomass stocks may be related to silviculture, age-structure of forests or climate change. Recurrent inventories with sound sampling are able to detect these changes in the tree biomass stocks.

10.6.1 Ecological Equivalences can be Estimated Based on the Inventory Data

Data collection in forest inventories is typically slow and may require up to 10 years until the entire country has been surveyed. Some countries are using or have recently begun to use (Finland, Hungary, Sweden and USA) continuous inventory of forest resources, in which the entire country is surveyed each year, but with smaller sampling density. This improves the prospects of monitoring the annual variation in carbon stocks in a representative area, based on forest inventory data. However, even this leads to a minimum time step in inventory data of 1 year with which atmospheric measurements can be compared.

Before annually updated nationwide inventory data can be made available, periodic mean estimates of growing stock and growth need to be processed into annual estimates that are comparable with results obtained with other methods (such as annual flux measurements based on the eddy covariance technique). In the Finnish application, the annual estimates of timber volume were interpolated using data on removal of timber from forests and on annual variation in growth, which were considered the most important factors affecting the annual variation in forest carbon stocks (Liski et al. 2006). The estimates of annual growth were based on growth indices measured from tree rings collected from several hundred NFI sample plots. This variable conveys multiple factors that affect annual variation in growth, such as early start of growing season or otherwise favourable growing conditions during the previous or current season.

In biological science, the main ecosystem processes have been named and defined according to their function. Net primary production (NPP) refers to the net production of carbon by plants and it equals the gross primary production (GPP) minus the carbon respired by plants, where GPP refers to the total amount of carbon assimilated by plants. Net ecosystem production (NEP) refers to NPP minus the carbon losses in heterotrophic respiration, which is the carbon lost by organisms other than plants (e.g. microbes). Net biome production (NBP) equals NEP minus sudden carbon losses from the ecosystem, such as forest fires or harvestings. These terms are commonly used when carbon fluxes of well-measured ecosystems are reported. In forest inventories, the terminology is different although the ecosystems are the same. The fact that the inventory-based approach is able to produce estimates for ecological equivalences, such as NPP, NEP and NBP, makes comparison with other approaches possible (see, however, Roxburgh et al. 2005).

In conjunction with carbon balance of Finnish forests, Liski et al. (2006) estimated NPP, NEP and NBP based on NFI data and modelling for upland soils, by defining:

$$\mathrm{NPP} = \Delta C + L + M + F \tag{10.1}$$

where ΔC is the change in carbon stocks, L is annual litter production by vegetation, M represents amount of natural losses and F fellings.

The estimate of NEP was obtained by subtracting from NPP the outflows of carbon, R_h (most of it occurs as heterotrophic respiration), that were simulated using the soil model Yasso (Liski et al. 2005; Palosuo et al. 2005):

$$\mathrm{NEP} = \mathrm{NPP} - R_h \tag{10.2}$$

The NBP was calculated by subtracting from NEP removals RE, which represented felled roundwood removed from the forests. In the case of Finland, forest fires are negligible and the impact of windthrows is included in the NFI growing stocks:

$$\mathrm{NBP} = \mathrm{NEP} - \mathrm{RE} \tag{10.3}$$

Liski et al. (2006) showed that the NBP of forests in Finland growing in mineral soils has been positive since the 1970s, that is, the ecosystem has been a net sink of

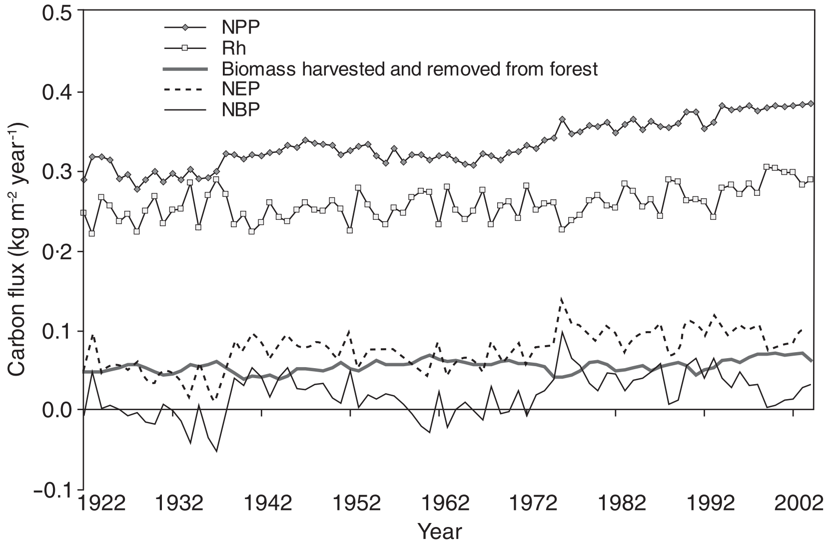

Fig. 10.4 Mean carbon fluxes (kg C m^{-2} year^{-1}) of forests in Finland from 1922 to 2004 in terms of NPP, heterotrophic respiration, NEP, removals and NBP (adopted from Liski et al. 2006).

carbon. These forests were a source of carbon during periods of heavy harvesting in the 1960s and in the 1930s, when NBP was negative (Fig. 10.4). These estimates include other main carbon fluxes occurring at large scale, but exclude estimates for the peatland soils as well as all non-CO_2 fluxes.

10.6.2 Future Scenarios of Forest Carbon Balance

Future scenarios for forest carbon sequestration are required by the Climate Convention as well as by national policy processes, especially as a part of climate and energy policy and strategic planning. In the national communications under the Climate Convention, countries report projections of GHG emissions and removals (sinks) for the next 20 years (e.g. Kuusisto and Hämekoski 2001). In general, national and regional scenarios of forest growth and harvests are available and have been updated for the European countries, because these provide basic information for the strategic planning of the forest industry. The scenario of forest carbon balance for the coming decades can be derived with inventory-based methods using information on projected growth and harvests as input data (Karjalainen et al. 2003). The findings of the scenario may vary considerably, depending on the assumptions put forward in both the input data and model simulations. Models that are not validated to conditions of changing climate (and changes in litter quality) are, however, limited to predictions under current climatic variation.

10.7 Conclusions

Forests are considered as an important mechanism for mitigation of climate change, because they can accumulate carbon from the atmosphere to biomass and soil carbon stock. The carbon inventory for the forestry sector should cover the annual changes in the carbon stocks of biomass, deadwood, litter and soil organic matter (IPCC 2003). Methods and datasets needed for the estimation of changes in the carbon stock of tree biomass are somehow available, especially those concerning the aboveground parts. On European scale, however, variation in local conditions and in the tradition of forest inventories between countries makes it challenging to compile a complete GHG inventory for the entire continent with coherent methods. In general, the major challenge in forest carbon inventories is monitoring of the carbon pools of litter and soil organic matter. Because repeated measurements of soil carbon stock on a national scale are generally lacking (Chap. 9), preliminary estimates of soil carbon sink/source can be provided by modelling only. The magnitude of soil carbon changes is such that it should be accounted for in the national forest carbon budget estimates (Liski et al. 2006). Furthermore, the carbon sequestration potential of forests can be affected by forest management practices and management may have opposing influences on biomass and soil carbon stocks. Therefore, forest carbon monitoring must cover all pools to avoid biased conclusions and mitigation strategies.

The inventory guidance (IPCC 2003, 2006) suggested that countries should accommodate their GHG inventories with uncertainty and key category analysis that show the most important sector for the overall GHG inventory as well as the major sources of uncertainties within the sectors. These types of analyses aid in prioritising efforts to develop and improve the quality of the GHG inventories (Peltoniemi et al. 2006; Monni et al. 2007). Furthermore, verification of the inventory by independent methods can provide insights to unknown factors or excluded components that may contribute notably to GHG balance (IPCC 2003). For example, inverse modelling that calculates fluxes from concentration measurements and atmospheric transport models can be used for verification of compiled inventory-based carbon balance on a continental scale (Bousquet et al. 2000). Janssens et al. (2003) compared inverse modelling and inventory-based approaches on a European scale by calculating separate estimates for the forest and agricultural sectors. Verification of the entire inventory may be impossible, but several methods can be used to test the validity of some parts of the inventory; for example, remote-sensing data can be used for verification of sampling-based estimates of the land-use changes on a regional scale. In addition to regional scale, direct measurements of ecosystem fluxes on a local scale can be used for evaluation of the annual variation in the inventory-based carbon balance (IPCC 2003).

The default methods used in the national carbon inventories do not detect annual variations in the carbon balance very accurately, because they interpolate changes over longer time spans (e.g. Tomppo 2000). Based on forest inventory data, it is possible to calculate more detailed carbon balance with the help of growth indices that indicate interannual variations in growth as a result of variation in the climatic conditions measured from sample trees of the NFI (Lehtonen 2005; Liski et al. 2006). In this application for Finland, the effects of the annual variation in growth and harvests on carbon balance were accounted for, but the litter input was calculated as constant proportions of the biomass of each component. It is, however, evident that annual biomass production and litter input from biomass to soil vary from year to year according to climatic conditions, and the accuracy of the annual carbon balance could be improved by taking this variation into account.

Most of the European countries have reported increases in the biomass carbon stock during recent decades (UNECE 2000; Liski et al. 2003) but have not reported estimates of the soil carbon sink/source. Uncertainty estimates for soil are often lacking, or if they are reported, are larger than those of the biomass (Ståhl et al. 2004; Peltoniemi et al. 2006; Monni et al. 2007). The complete carbon balance calculated for forests in Finland showed that the biomass carbon pool increased by a mean of 27 g C m^{-2} annually in the 1990s, while the carbon pool of upland forest soils increased by 11 g C m^{-2} (Liski et al. 2006). However, both the tree and soil carbon pools are known to vary widely on an interannual basis. Because the carbon sinks of forest soils have been smaller (and close to zero in some years) and the uncertainty of their estimate is larger than that of vegetation, they may have acted as sources as well as sinks during the last decade.

The importance of soils in the forest GHG balance indicates that all European countries should develop their inventories in a way that soil carbon balance is included before continental-scale inventory-based carbon balance covering the entire forest ecosystem can be compiled. Methods applicable for such an inventory of upland forests are already available (e.g. Lehtonen 2005; Liski et al. 2005, 2006).

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