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# A synthesis of the impact of Russian forests on the global carbon budget for 1961–1998

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ABSTRACT

An attempt is made to synthesize the current understanding of the impact of Russian forests on the global carbon (C) budget for the period 1961–1998 (37 years), based on a detailed inventory of pools and fluxes in 1988–1992, and a historical reconstruction of a full forest carbon budget for 1961–1998. All major intermediate indicators of the budget (phytomass, net primary production, impact of disturbances, soil respiration, etc.) were independently estimated and compared with earlier reported results. During the entire period, the C pools of Russian forest land (FL, $882.0 \times 10^6$ ha in 1998) increased by 433 Tg C $yr^{-1}$, of which 153 Tg C $yr^{-1}$ are accumulated in live biomass, 57 Tg C $yr^{-1}$ in above- and below-ground dead wood, and 223 Tg C $yr^{-1}$ are sequestered in soil. A significant part of this increase deals with land-cover changes. The annual average C uptake by the FL from the atmosphere, defined by a flux-based method, is estimated to be $-322$ Tg C $yr^{-1}$ for 1961–1998. The lateral transport to the lithosphere and hydrosphere comprised 47 Tg C $yr^{-1}$ (including charcoal), which is considered part of the "missing C sink." The uncertainties (excluding unrecognized biases) of averages for the entire period are estimated to be in the range of ±5–8% and ±24% for major fluxes out/into the atmosphere and for net ecosystem exchange, respectively (a priori confidential probability of 0.9). If the impact of land-cover change is excluded, the average annual sink in 1961–1998, estimated by both pool- and flux-based methods, was $268 \pm 94$ and $272 \pm 68$ Tg C $yr^{-1}$, respectively. The reported results are in line with recent estimates for Northern Eurasia made by inverse modeling at the continental scale, if land classes other than forests contribute to the total sink of terrestrial vegetation.

## 1. Introduction

The reported results on the impact by Russian forests (23% of the global growing stock) on the global carbon (C) budget are numerous and contradictory. During the last 15 years, the C sink of Russian forests was estimated to vary between 20–660 Tg C $yr^{-1}$ for the late 1980s to early 1990s (Melillo et al., 1988; Sedjo, 1992; Dixon et al., 1993; Krankina and Dixon, 1994; Kolchugina and Vinson, 1993a,b; 1995; Isaev et al., 1993; 1995; Krankina et al., 1996; Kokorin and Nazarov, 1994; Kokorin et al., 1996; Lelyakin et al., 1997; Isaev and Korovin, 1998; UN, 2000; Nilsson et al., 2000). There are four major reasons for this large variation. First, all of the studies [except Nilsson et al. (2000), who estimated the full carbon budget (FCB) for forested areas] did not present a full carbon account (FCA) for Russian forests. Second, the estimates of the most important parameters influencing the C balance [e.g., phytomass, detritus, net primary production (NPP), net ecosystem production (NEP), impact of major types of disturbances, etc.] reported by the above publications vary by two or more factors. Third, the assessments were often provided for different objects (forested areas, forest biome, etc.). Fourth, the C budget of boreal forests is by nature a complicated superimposition of stochastic processes that strongly depends on regional weather specifics for a definite year or period, as well as on the actual and historical regimes of disturbances, the extent and severity of which vary greatly. However, most of the

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above publications do not clearly identify the assessment year (or period).

In order to obtain a consistent estimate, a solid basis for comparison and validation of the other studies' results, as well as the major prerequisites for this assessment, should be explicitly identified. (1) We provided an FCA (we distinguish the FCB as a natural phenomenon and the FCA as an artificial accounting system) for Russian forests based on a systems approach (Nilsson et al., 2000). "A full C budget encompasses all components of all ecosystems and is applied continuously in time" (Steffen et al., 1998), and we tried to implement this approach in spatial, temporal and process terms. (2) Minimizing the uncertainties is a major scientific goal of the FCA. However, taking into account that the FCA at a national scale is a typical fuzzy system, no method or model is able to present results for which uncertainties could be reliably estimated. From a methodological point of view, this means that we should not consider, e.g., pool-based versus flux-based methods, or models versus inventory ("bookkeeping") approaches, but consider a system integrity of all appropriate methods as most relevant. From the information point of view, all appropriate sources and reported results should be used, at least for crosschecking and evaluating uncertainties. (3) The accounting requires an explicit structure including strict definitions and module (system) boundaries, as well as transparent and verifiable FCAs. (4) An FCB, as a non-stationary stochastic process, only makes sense if the assessment year or period is exactly indicated. (5) The estimation of uncertainties at all stages and for all modules of the FCA is mandatory. This means that accounting schemes, models and assumptions should be explicitly presented in algorithmic form.

Russia has a complicated forest land-cover classification. This study focuses on Russian forest lands (FL, $882.0 \times 10^6$ ha or some 52% of total Russian lands; data for 1998, FSFMR, 1999). FL is defined as lands that are designated for forests. The FL are divided into: (1) forested areas (FA), i.e., closed forests with relative stocking $\geq$0.4 for young stands and $\geq$0.3 for older forests ($774.3 \times 10^6$ ha); (2) unclosed forest plantations ($2.8 \times 10^6$ ha); and (3) unforested areas (UFA) that are designated for but are temporarily without forests ($105.0 \times 10^6$ ha, of which 68% consisted of sparse forests (open woodlands), 24% burned areas and dead stands, 5% unregenerated harvested areas, and 3% grassy glades, see also Tables 8 and 9 in the Appendix). About 25% of the FL has redundant humidity and soils with a peat layer of different depth. While the FL category in Russia was rather stable during 1961–1998 (increasing only by 4% since 1961), the changes in FA and UFA were more significant: 11.1% and −7.1%, respectively. These changes, which should be basically classified as land-cover changes (FAO, 2000) while land-use changes, i.e., converting forests to other land-use classes and versus versa, were limited by areas $\leq$ 0.1–0.2% of the FL, generated some methodological difficulties. In particular, it is practically impossible to strictly separate the impact of land-cover changes on the C budget from purely "ecological" impacts dealing with changes in the structure and functioning of forest ecosystems.

We used generally accepted terminology, although some influence of Russian forest inventory and forest management manuals was inevitable. Phytomass means all vegetation matter of live plants. It is divided in seven fractions: stem wood (over bark), branches (over bark), foliage, understory (undergrowth and bushes below canopy), green forest floor and roots, which were sometimes combined in above- and below-ground phytomass in the evident way. Dead vegetation organics include coarse woody debris (CWD), i.e., dead above-ground (standing dry trees, snags, dry branches of live trees, stumps) and on-ground wood or logs (downed wood, windbreak, etc.), larger than 1 cm in diameter at the thin end) and dead roots. Soil organic matter was divided in carbon into (1) "on-ground organic layer" which includes the organic layer, i.e., all organic material above the mineral soil horizons and well preserved or slightly decomposed plant surface residuals for organogenic (peat) soils; woody remains included in this pool were limited by 1 cm at the thin end; (2) organogenic (peat) soils and (3) humus of mineral horizons. This classification is caused by the structure of the information sources used. Here, "lithosphere" means deep crust layers outside the soil profile, i.e., from the bottom of the C horizon and below, and hydrosphere is presented by rivers and other surface water reservoirs. The (total) heterotrophic respiration consists of heterotrophic soil respiration and fluxes caused by decomposition of CWD. The lateral fluxes account for the carbon transported to the lithosphere and hydrosphere.

This paper deals with several aspects of the FCB. First, we estimated the most important intermediate FCB parameters and major C pools and fluxes of Russian forest lands for 1990, the base year for the Kyoto Protocol. Due to large inter-seasonal variations of fluxes and information limitations, the results were calculated as an annual average over five years (1988–1992); evidently, these results could serve as only an approximate estimate for the individually treated 1990. Second, we assessed the FCB for 1961–1998 by using data on forest dynamics and land-cover changes for this period. Some of the results, published earlier, are presented here in a very succinct form as a basis for the synthesis. Third, we tried to quantify the uncertainties of our results. Finally, a comparison was made with earlier results.

## 2. Methods and information base

The FCA presents an estimate of the annual change of organic C in forest ecosystems ($\Delta$C), and is considered here in terms of a pool-based approach:

$$[\mathrm{dC}/\mathrm{d}t] = \Delta \mathrm{C} = \Delta \mathrm{Ph} + \Delta \mathrm{D} + \Delta \mathrm{SOC}, \quad (1)$$

where $\Delta$Ph, $\Delta$D, and $\Delta$SOC are annual C changes in phytomass, dead vegetational organics and soils, respectively, and by using a flux-based approach, describing the C fluxes between ecosystems and atmosphere, lithosphere, and hydrosphere as:

$$\mathrm{NBP}_1 = \mathrm{NPP} - \mathrm{HR} - \mathrm{D}, \quad (2)$$

and

$$\mathrm{NBP}_2 = \mathrm{NPP} - \mathrm{HR} - \mathrm{D} - \mathrm{L}, \quad (3)$$

where NBP and NPP are net biome and net primary production, HR is ecosystem heterotrophic respiration, D is fluxes generated by disturbances, and L is lateral fluxes. Major C pools and fluxes, which are considered in this study, are presented in Fig. 1.

Schulze and Heimann (1998) initially introduced NBP as a net C exchange between terrestrial vegetation and the atmosphere [i.e., without L, see eq. (2)], aggregated for large territories and for long periods of time, and widely used elsewhere (e.g., Schulze et al., 1999; Schimel et al., 2001). However, following the logic of terms like NPP and NEP and to provide consistency between the estimates by flux- and pool-based methods, NBP should include lateral fluxes, which convey C from biological turnover. Both $\mathrm{NBP}_1$ and $\mathrm{NBP}_2$ are

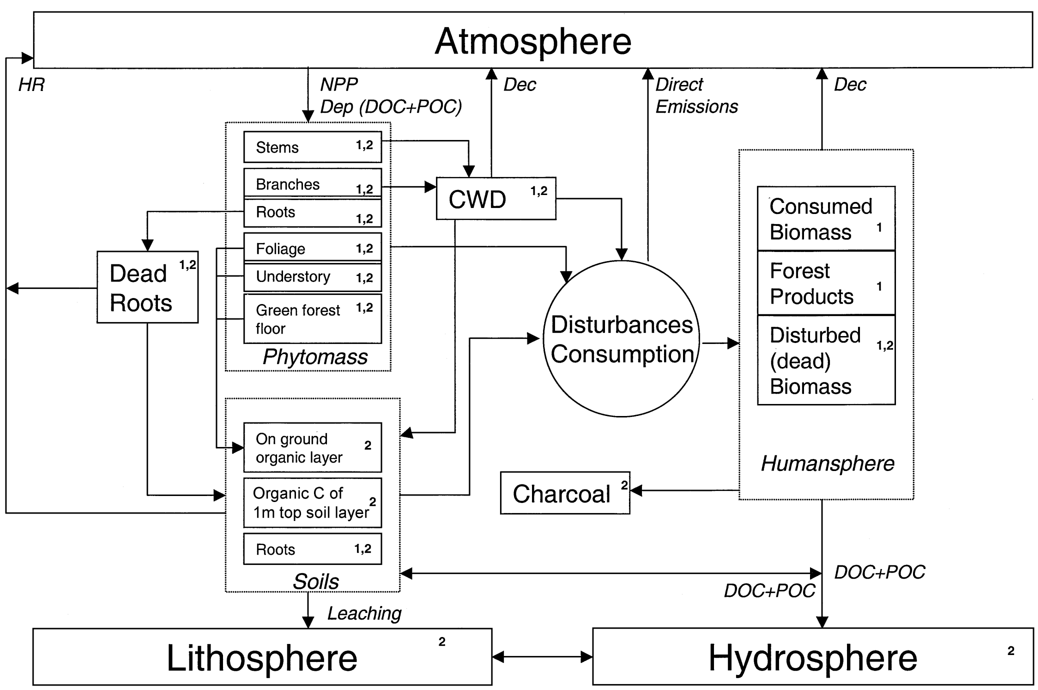


*Fig. 1.* Major C pools and fluxes presented in the FCA. Numbers 1 and 2 indicate basic sources used: (1) forest inventory data; (2) GIS approach; (1,2) use of both sources. Abbreviations: NPP, net primary production; HR, heterotrophic soil respiration; CWD, coarse woody debris; Dep, wet and dry C deposition; Dec, decomposition; DOC and POC, dissolved and particulate organic C.

assessed in this study. In addition, we considered fluxes caused by consumption of forest products, but those are not included in the final estimate of $NBP_2$.

As the basis for applying eqs. (1)–(3), we used an ecosystem approach in aggregated form. This approach means that a specific ecosystem (i.e., a soil–vegetation group with different aggregation levels) is considered as the primary object for modeling and assessment. In the most comprehensive methodological approach, the FCA should be based on a landscape–ecosystem approach that additionally includes the impact of landscape specifics on intra-ecosystem interactions. However, we were not able to use the latter in a sufficiently consistent fashion, and limited the application of the "landscape ideology" by simple large-scale substitutions, e.g., for assessing lateral fluxes.

Equations (1) and (3) are coupled, and should eventually present estimates of the same parameters of the FCB. However, in practical FCA applications at the national (continental) scale, these two approaches assess different processes, are applied to somewhat different objects, and do not give results that are strictly consistent over time. Evidently, the pool-based method cannot be sufficiently applied for a short period of time, which is why we limited our consideration for 1990 by estimating the fluxes and values of C pools. In the longrun, the main reason for the abovementioned inconsistency between eqs. (1) and (3) is the continuous transition of individual areas from one forest land-cover class to another (e.g., UFA in FA and vice versa), which cannot be comprehensively accounted in any applicable FCA version for large areas.

Two major information sources covering the entire country were used in this study: forest inventory data and an integrated land information system (ILIS), either independently or in different combinations (Fig. 1). Forest inventory data (called the forest inventory approach) were presented in aggregated form by the State Forest Account (SFA), which contains the comprehensive characteristics of forests (dominant species, age structure, levels of productivity, etc.) by forest enterprises (about 1800), administrative units (89), a number of large subdivisions (e.g., European and Asian Russia) and the whole country. The SFA data are available for 1961, 1966, 1973, 1978, 1983, 1988, 1993 and 1998 (SNKh SSSR, 1962; Gosleshoz SSSR, 1968, 1976, 1982, 1986; Goscomles SSSR, 1990, 1991; FSFMR, 1995, 1999; data for 1 January of each year). Details of the Russian forest inventory system and the specifics and reliability of the information are discussed in Shvidenko and Nilsson (2002). The ILIS consists of a multi-layer Geographical Information System (called GIS approach) for Russian land, developed by IIASA in 1993–2000 (Nilsson et al., 2000). The GIS components include digitized maps (forests, soils, landscapes, vegetation, land-use, on-ground organic layer, etc.), at the scale of 1:1–1:4 million, accompanied by attributive databases, numerous long-term statistical data, including wood harvest, disturbances, etc.; auxiliary modeling systems (e.g., for assessing phytomass, net and gross growth, and mortality); data on typified soil horizons; measurement results of CWD and dead roots; and relevant semi-empirical aggregations and scientific results. The classification of forests (FA) used 125 classes of forest vegetation applied to about 13 000 GIS polygons. Background data used for the GIS development were basically collected between the late 1970s to mid 1990s, and the major maps used were published about the 1990s. The forest map has been connected to the State Forest Account data for 1993. Thus we approximately reference the major GIS components for the 1990s.

Assessment results were aggregated by ecological regions (eco-regions) and bio-climatic zones. The eco-regions, totaling 141 (78 and 63 in European and Asian Russia respectively), were established as territorial entities, homogeneous in climatic, forest vegetation, and forest transformation aspects (Shvidenko et al., 2000). The bio-climatic (vegetation) zones are presented by a slightly aggregated classification used by Isachenko et al. (1990) and comprise eco-regions combined in tundra; forest tundra, northern and sparse taiga; middle taiga; southern taiga; temperate forests zone; steppe; and semi-desert and desert zones.

### 2.1. *The FCA for 1990*

The FA phytomass pool was assessed by two independent methods: forest inventory data and GIS tools (Nilsson et al., 2000). A special modeling system was developed for assessing forest phytomass by fractions based on forest inventory data. About 2250 sample plots and more than 250 regional studies were used for generating non-linear multidimensional regression equations of major forest-forming species for aggregated eco-regions of Russia. As forest inventory data is the only reliable source of forest phytomass inventories at the national scale, the regressions were presented by the ratio (conversion factor) between the mass of phytomass fractions and growing stock volume. The statistical accuracy of the equations, which included age, site index and relative stocking, was rather high. The multiple non-linear correlation coefficients and the regression coefficients of variables were, as a rule, statistically significant with a probability of $>0.95$; the adequacy was checked by analyzing the distribution of residuals and their correlations with the variables included in the equations. The calculation of phytomass (divided into stem wood, branches, bark, roots, foliage, understory and green forest floor) was provided by combining the inventory data for forest enterprises into data for the eco-regions and application of the regression equations to these aggregated data. The second estimate of phytomass was made by GIS tools. In this approach, we used the average densities of phytomass fractions for forest vegetation classes [calculated independently basically using the original database by Bazilevich (1993)] and areas from the forest map. The calculations were provided for initial polygons of the map and aggregated to eco-regions and seven bio-climatic zones. The regression equations mentioned above are available from Shvidenko et al. (2001a) and the aggregated conversion factors and carbon densities by major forest forming species are presented in Table 10 in the Appendix.

In order to estimate CWD we used forest inventory data aggregated for individual forest enterprises, data of CWD measurements on sample plots available in publications and archives, as well as our own measurements. The GIS was used for up-scaling the "point" measurements. The mass of dead roots was assessed in a similar way, based on IIASA's forest map and average estimates of dead roots by species and eco-regions (Shvidenko et al., 2000).

The GIS approach was used to assess the three C pools of soils (indicated in Fig. 1) by overlapping the soil map at the scale of 1:2.5 million (digitized by the Dokuchaev Soil Institute, Moscow) with the IIASA forest map that was connected to typified soil profiles and to an on-ground organic layer database (Nilsson et al., 2000). The reference data were collected for undisturbed soils. In order to take into account the impact of disturbances, we introduce an index of the severity of disturbance regimes (ISDR) as the ratio $\mathrm{ISDR} = 100\,(\mathrm{BA} + \mathrm{ASF} + \mathrm{GG})/\mathrm{FL}$, where BA, ASF and GG are area of burnt and dead forests, anthropogenic sparse forests, and grassy glades and barrens, respectively. Dependence between correction coefficients and the ratio (separately for on-ground organic layer and 1 m top soils) was parametrized based on fragmented regional data (Table 11 in the Appendix). The corrections for 1990 were $-10.9\%$ for C of on-ground organic layer and $-1.5\%$ for the 1 m soil.

The major C fluxes (NPP, HR and L) were also estimated based on the GIS approach. Because the fluxes should be assessed for definite areas and a given period of time, we applied statistical methods by using georeferenced polygons and the field measurement averages of the calculation indicators. The database, which was used for the NPP assessment, includes the results of measuring NPP on about 1600 sample plots by three aggregated fractions (total green parts, above-ground wood, and underground parts). We used the original database, developed by Bazilevich (1993), which was supplemented by measurements made during the last decade (e.g., Karelin et al., 1995; Gower et al., 1995, 2000; Schulze et al., 1999).

It is evident that NPP quantified in this way describes a certain "quasi-stable" state, as both GIS data and weather conditions used in the calculations are averaged over a certain period, data on forest disturbances are incomplete and have time lags, etc. To improve the accuracy of the results, the most important natural and anthropogenic factors that affected the forest ecosystems during the assessment period (1988–1992) were analyzed. In order to estimate NPP, we considered the change in forest productivity after disturbances (particularly after a fire on permafrost areas), impact of wetland amelioration, and loss of the actual NPP in areas affected by major types of disturbances during the assessment period; these corrections increased the total NPP on FL for about 7% (Shvidenko et al., 2001b). For many years, NPP located in wood has played and continues to play a crucial role in C sequestration and is an important component in assessing the NEP and NBP. For this reason, as well as for validating the results obtained by using GIS technologies, we estimated the gross and net increment on FA based on forest inventory data (Shvidenko et al., 1995a, 1997).

Heterotrophic respiration (HR) was assessed based on the soil map polygons and average annual accumulated fluxes calculated by soil types and aggregated land cover classes. The assessment was provided in two independent ways using different databases and methodological approaches. The first approach did not include winter fluxes, i.e., the period with an air temperature of $< 0\,^{\circ}\mathrm{C}$ and using a linear decrease in $CO_2$ soil evolution at the temperature interval of 5 to $0\,^{\circ}\mathrm{C}$ (Nilsson et al., 2000; Stolbovoi, 2002b). The second approach accounted for total yearly estimates of soil

C fluxes and attempted to include the specifics of different vegetation types (Kurganova, 2002).

UFA included burnt areas, dead stands, cutovers, glades and barrens, natural and anthropogenic sparse forests, and unstocked planted forests. The average densities of the main indicators (phytomass, NPP, CWD, SOC, etc.) were estimated by the UFA categories and bio-climatic zones and applied to these aggregated units.

A unified approach was used to assess the fluxes caused by major disturbances (fire, insect and disease outbreaks, site effects of harvest, and abiotic factors). The total carbon flux $TCF_{\rho,t_1}$ during year $t_1$ generated by a disturbance $\rho$ (for annual time steps) was calculated as:

$$TCF_{\rho,t_1} = DF_{\rho,t_1} + PDF_{\rho,t<t_1}. \tag{4}$$

where $DF_{\rho,t_1}$ is the direct flux during year $t_1$, and $PDF_{\rho,t<t_1}$ is the post-disturbance, as a rule biogenic, flux generated by disturbance $\rho$ that occurred during previous years $t < t_1$. The values of $DF_{\rho,t_1}$ and $PDF_{\rho,t<t_1}$, as well as the explicit form of eq. (4), depend on the type, strength and extent of $\rho$, the conditions under which $\rho$ occurs, the type and specifics of the ecosystem, and on the approach and structure of the model used. As an example, the direct flux due to forest fires (for the year of fire) is defined as (Shvidenko and Nilsson, 2000):

$$DF(t_1) = \sum_{ilkq} [C_{ilkq} S_{ilkq} (FC)_{ilkq}]_{t_1} \gamma, \tag{5}$$

where $C_{ilkq}$ are the coefficients for the consumed forest combustibles during a fire, $S_{ilkq}$ is the estimate of burned vegetation areas, $(FC)_{ilkq}$ is the storage of forest combustibles ($t$/ha, dry matter), and $\gamma$ is the coefficient for recalculating dry organic matter to C units [we used 0.5 for forest combustibles and 0.45 for the remaining vegetation (Vonsky, 1957; Filippov, 1968; Telizin, 1973)]. The indexes are: $i$ = territorial units for which the calculations are made; $l$ = aggregated land-use classes; $k$ = types of forest fire; and $q$ = types of forest combustibles.

Post-fire fluxes are caused by the decomposition of both incombustible (dead) residuals and post-fire die-back (mortality), as well as by changes in the structure and content of soil organic matter. Let $O_{ij}(t)$ be a function that describes the amount of dead organic matter entering a decomposition pool $j$ in year $t$, and $O_{ij}(t^*)$ be the value of this function in year $t^*$. Using a simple exponential model (the more advanced approaches available (Melillo et al., 1989; Aber et al., 1990) were not used due to the lack of data for the diversity of soil–vegetation groups of Russian forest lands), the process of decomposition of organic matter of pool j is described as:

$$G_{ij}(t^*, \tau) = O_{ij}(t^*)\exp(-\alpha_{ij}\tau), \tag{6}$$

where $G_{ij}(t^*,\tau)$ is the mass of organic matter that did not decompose during period $\tau$, $\alpha_{ij}$ is the constant of decomposition, and $\tau$ is the number of years between the year of the fire and the year of the PDF estimation, e.g., $\tau^* = t^* - t_1$. Evidently, for eq. (6), the time for decomposition of 95% of the decomposition C pool $T_{0.95}$ depends only on $\alpha_{ij}$, $T_{0.95} = \ln 20/\alpha_{ij}$. Thus, the post-fire biogenic flux to the atmosphere during year $t_1$ caused by fires during previous years can be estimated by:

$$(PDF)_{ij}(t_1) = 1.05\chi[\exp(\alpha_{ij}) - 1] \cdot \sum_{\tau=0}^{\varphi+1} O_{ij}(t-\tau)\exp(-\alpha_{ij}\tau) + \delta SOC, \tag{7}$$

where $\chi$, $0 < \chi < 1$, is the share of C from decomposed organic matter that is taken up by the atmosphere, $\varphi = \text{int}\,[T_{0.95}]$ (the integer part of $T_{0.95}$), and $\delta SOC$ is the post-fire change of heterotrophic soil respiration during year $t_1$. There is not sufficient data for regional estimates of $\chi$, so we used the average value of 0.88 [based on measurements by Chagina (1970) of 0.92 for old growth Siberian cedar (Pinus sibirica) forests, Vedrova (1995) of 0.75–0.92 and 0.77–0.88 for 25-yr-old coniferous and deciduous plantations, respectively, and Kurz et al. (1992) of 0.82 for Canadian forests]. To estimate the actual post-disturbance fluxes, a retrospective period of 200 yr is needed for the taiga and forest tundra zones. In the framework of the pool-based account, the changes of soil organic C were calculated as $(\Delta SOC)_{ijt_1} = 1.05(1-\chi)(PDF - \delta SOC)_{ijt_1} + Cch,t$, where the first component provides the change of soil C caused by the decomposition of post-fire die-back, and the second provides the input of charcoal during the year of the fire. Background data used in the calculation are presented in Tables 12 and 14 in the Appendix. More details on the topic can be found in Shvidenko et al. (1995b), Shvidenko and Nilsson (2000) and Nilsson et al. (2000).

Two independent attempts were made to assess the fluxes caused by the decay of forest products. The first attempt used the slightly modified approach described above for disturbances (Shvidenko, 1997), and the second used a specially developed model

(Obersteiner, 1999). Both approaches are based on as comprehensive as possible assessment of harvested wood (including domestic consumption), wood products and wastes, which are separated in three decay pools of different decomposition rates (fast, medium and slow).

The lateral fluxes were estimated in an aggregated form taking into account that the results of measurements are poor for big regions. Dissolved and particulate organic carbon (DOC and POC, for which the difference is defined by a boundary size of 0.45– 0.5 $\mu$m) is transported with surface and below-ground run-off to the hydrosphere (rivers and inner water reservoirs), lithosphere deposits on geochemical barriers, and to deep (outside soil profiles) below-ground water. Based on lysimetric measurements, the content of DOC and POC in soil water is rather high in forest soils of the boreal zone, and varies on average from 50 to 100 mg $L^{-1}$, sometimes significantly more (e.g., Ponomareva and Plotnikova, 1972; Djakonova, 1972; Glazovskaya, 1996). Concentrations of DOC and POC in rivers are significantly lower: from 10 to 30 mg $L^{-1}$ (Vinogradov et al., 1998; Kassens et al., 1999; Romankevich and Vetrov, 2001). This means that part of DOC and POC is absorbed by mellow deposits, where the content of organic matter is often high (0.5–1.5%); the results of direct estimates of this C flux to the lithosphere are fragmentary (Glazovskiy, 1983; Glazovskaya, 1996; Rapalee et al., 1998). The assessment of lateral fluxes were provided by aggregating the eco-region data. A part, which is supposedly transported to the hydrosphere by FL, was assessed based on areas of catchments and the corresponding share of FL. The C reaching ecosystems in dry and wet deposition (DOC + POC) was assessed by using published data (Meyback, 1982; Lychagin, 1983; Saet and Smirnova, 1983; Nilsson et al., 1998; Labutina and Lychagin, 1999; Russian official reports on the state of the environment for the last decade). We excluded from the consideration some processes due to their supposedly small impact on the forest FCB and/or contradictive opinions about the sign of fluxes generated by these processes (e.g., soil erosion, cf. Schlesinger, 1995; Smith et al., 2001).

### 2.2. *The FCA for 1961–1998*

The State Forest Account data for 1961–1998 were used as the basis for assessing the dynamics of vegetational C in phytomass, CWD, and dead roots. Because of the growing stock volume presented by the Russian forest inventory has a bias, which changes over time (Shvidenko and Nilsson, 2002), the calculations were carried out in two ways (for official forest inventory data and for "restored dynamics") in an endeavor to eliminate this bias. It should be noted that this correction was only done for growing stock on FA; the remaining forest inventory data do not have any significant biases in this respect. In order to estimate phytomass dynamics on FA, the above-mentioned modeling system on forest phytomass was applied to the forest inventory data for 1961–1998 by aggregated eco-regions. Phytomass on UFA was defined by major UFA land categories using average zonal values previously estimated for 1990 (Tables 8 and 9 in the Appendix). To check the consistency of our restored dynamics and to quantify the CWD input to decomposition pools, we calculated a wood balance based on forest inventory data, growth indicators, wood consumption data and the impact of disturbances (for details, see Shvidenko and Nilsson, 2002).

Fluxes due to litter dynamics were modeled based on a linear feedback theory (Olson, 1963) as:

$$\mathrm{d}M_j/\mathrm{d}t = L_j(t) - \alpha_j M_j(t), \qquad (8)$$

where $M_j(t)$ is the mass of the litter, $L_j(t)$ is the litter input during year $t$, $\alpha$ presents the zonal decomposition coefficients by four decomposition pools $j$ (foliage and green forest floor, two pools of CWD: medium-fast and slow pools, with a top diameter of wood residuals at $1 \le d \le 8$ cm and $d > 8$ cm, respectively, and roots). In order to calculate the integral of eq. (8), $L_j(t)$ were approximated by the polynomials at the intervals $[0 \le t \le 30]$ and $[0 \le t \le 8]$ for 1960–1990 and 1990–1998, respectively. Coefficients of analytical expressions for $L_j(t)$ and average zonal values of $\alpha_j$ are given in Table 12 in the Appendix.

The dynamics of soil organic C of FL were calculated based on land cover change, densities of soil C and the severity of disturbance regimes, expressed by ISDR. Supporting information is presented in Table 11 in the Appendix. For comparison, we examined a simple one-compartment model of soil C dynamics, which was presented as:

$$\mathrm{dCs}/\mathrm{d}t = \sum_j (1-\chi)\alpha_j \mathrm{M}_j(t) - \beta \mathrm{Cs}(t), \qquad (9)$$

where Cs($t$) is the mass of C in soil organic matter at time $t$, 1–$\chi$ is the share of C entering the soil (which is partially humified and partially transported out of forest ecosystems), and $\beta$ is the decomposition rate (mineralization) of the SOC compartment.

The basic idea of this simple approach has been used in a number of studies, e.g., for Canadian forests in the CFS-CBM (Kurz et al., 1992) and for arable crops in central Sweden in the ICBM (Andren and Katterer, 1997). In spite of its simplicity, the approach has evident advantages: outflows from the pools follow first-order kinetics, and eqs. (8) and (9) can be integrated analytically. However, while empirical regional data for $\alpha$ are rather numerous (e.g., Grishina, 1986; Kobak, 1988; Orlov, 1990), available estimates of $\beta$ are not sufficient and we considered the behavior of eq. (9) for $\beta$ varying from 0.006 to 0.002 (e.g., for $T_{0.95}$ from 500 to 1500 yr, cf. Andren and Katterer, 1997).

### 2.3. *Estimation of uncertainties*

We defined uncertainties as "an aggregation of insufficiencies of the FCA system outputs, regardless of whether these insufficiencies result from a lack of knowledge, the intricacies of the system, or other causes" (Nilsson et al., 2000). The reasons for uncertainties are numerous and different in nature (Shvidenko et al., 1996; Nilsson et al., 2000). Taking into account specifics of the FCA at the national scale, the uncertainties were estimated in several stages. (1) Precision (in terms of "summarized errors" as a function of random and systematic errors) was calculated for all accounting steps based on error propagation theory. (2) Because of the lack of sufficient statistical data, or steps where classical statistical analysis was not applicable, the use of expert estimates and subjective probabilities was employed. (3) The calculated precision was transformed into uncertainties based on standard sensitivity analysis and expert estimates of not accounted impacts and processes. The a priori confidential probability of 0.9 was used. (4) Finally, comparisons with independent estimations were provided (particularly with those where reliability was certified).

The approach suggested for estimating uncertainties includes subjective elements in the form of expert modifications of calculated precisions. In spite of the singularity of the approach, our experience shows that it contributes to understanding the completeness and strictness of the FCA structure used, and allows the improvement of conclusions. Unfortunately, other more formalized and practically applicable methods for assessing uncertainties of fuzzy systems have not yet been suggested.

## 3. Results

The distribution of Russian forest areas (1993) by dominant species and bio-climatic zones (Table 1) reflects a typical picture for boreal forests. There is limited diversity at the species level (five dominant coniferous species cover 71.1% of the total FA, and by adding two deciduous, species birch and aspen, the coverage is 87.4%), but a high ecological plasticity and adaptability of species (e.g., pine and birch forests occur in all bio-climatic zones, from the tundra to semi-desert and desert).

### 3.1. *Major carbon pools*

The phytomass of FA (Table 2), based on SFA-1993, comprises 32 862 Tg C [82.1% of the total phytomass of the Russian terrestrial ecosystems (Shvidenko et al., 2000)], of which 6.2% is in green parts, 71.8% in

Table 1. *Areas of Russian forests in 1993 ($\times$ $10^6$ ha)*

| | Distribution of forested areas (FA) by dominant species, $\times 10^6$ ha | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Zone[a] | pine | spruce | fir | larch | cedar | birch | aspen | other | shrub | total | UFA | FL |
| T | 0.1 | 0.0 | 0.0 | 0.2 | 0.0 | 0.5 | 0.0 | 0.0 | 3.0 | 3.8 | 3.5 | 7.3 |
| FT | 21.8 | 23.9 | 0.0 | 71.9 | 7.3 | 7.7 | 0.3 | 0.1 | 8.1 | 141.2 | 55.8 | 197.0 |
| MT | 60.4 | 43.1 | 11.1 | 191.0 | 29.9 | 46.7 | 7.9 | 10.3 | 54.7 | 455.0 | 34.5 | 489.5 |
| ST | 41.2 | 17.7 | 5.2 | 7.6 | 3.2 | 38.4 | 10.7 | 2.3 | 0.1 | 126.5 | 21.8 | 148.3 |
| TF | 3.1 | 0.2 | 0.1 | 0.0 | 2.6 | 4.4 | 2.4 | 14.4 | 0.2 | 27.4 | 5.7 | 33.1 |
| S | 2.1 | 0.0 | 0.0 | 0.0 | 0.1 | 2.9 | 1.1 | 1.8 | 0.4 | 8.3 | 1.6 | 9.9 |
| SDD | 0.1 | 0.0 | 0.0 | 0.0 | 0.0 | 0.3 | 0.0 | 0.7 | 0.2 | 1.3 | 0.1 | 1.4 |
| Total | 128.8 | 84.8 | 16.4 | 270.0 | 43.1 | 101.0 | 22.5 | 29.5 | 66.7 | 763.5 | 123.0 | 886.5 |

[a]T, tundra; FT, forest tundra, northern and sparse taiga; MT and ST, middle and southern taiga, respectively; TF, temperate forests zone; S, steppe; SDD, semi-desert and desert zones. The boundaries of the zones are given according to the vegetation map by Isachenko et al. (1990).

Table 2. *Carbon pools of Russian forest lands (1993)*

| Zone[a] | Phytomass[b] Tg C | | | | Density, kg C m$^{-2}$ | Coarse woody debris, Tg C | Dead roots, Tg C | Vegetation carbon | |
|---|---|---|---|---|---|---|---|---|---|
| | Above ground wood | Green parts | Below ground | Total | | | | Total, Tg C | Density, kg C m$^{-2}$ |
| T | 31 | 5 | 18 | 54 | 1.42 | 6 | 26 | 86 | 2.26 |
| FT | 2,176 | 334 | 865 | 3,375 | 2.39 | 726 | 587 | 4,628 | 3.32 |
| MT | 14,832 | 1,251 | 4,504 | 20,587 | 4.52 | 2,718 | 2,430 | 25,735 | 5.66 |
| ST | 5,081 | 379 | 1,372 | 6,832 | 5.40 | 827 | 623 | 8,282 | 6.55 |
| TF | 1,199 | 64 | 373 | 1,636 | 5.97 | 110 | 64 | 1,810 | 6.61 |
| S | 257 | 17 | 80 | 355 | 4.28 | 14 | 51 | 420 | 5.06 |
| SDD | 14 | 2 | 7 | 23 | 1.77 | 3 | 7 | 33 | 2.54 |
| Total on FA | 23,590 | 2,052 | 7,220 | 32,862 | 4.304 | 4,404 | 3,788 | 41,054 | 5.38 |
| UFA[c)] | 684 | 199 | 426 | 1,309 | 1.06 | 506 | 302 | 2,117 | 1.72 |
| Grand total | 24,274 | 2,251 | 7,646 | 34,171 | 3.855 | 4,910 | 4,090 | 43,171 | 4.870 |

[a]See Table 1.
[b]Zonal distribution of phytomass is presented for FA.
[c]Including unclosed forest plantations.

Table 3. *Soil carbon pool of 1 m top layer by main forest-forming species for FA (Tg C)*

| Zone[a] | Soil carbon pool by main forest forming species | | | | | | | | | Total |
|---|---|---|---|---|---|---|---|---|---|---|
| | pine | spruce | fir | larch | cedar | birch | aspen | other | shrubs | |
| T | 5 | 9 | <0.5 | 20 | 8 | 100 | 5 | 3 | 538 | 688 |
| FT | 5724 | 5693 | 1 | 14942 | 1704 | 1859 | 33 | 29 | 1573 | 31558 |
| MT | 10003 | 6063 | 977 | 24124 | 3004 | 7098 | 987 | 1743 | 7174 | 61173 |
| ST | 5848 | 2239 | 783 | 1547 | 1124 | 6706 | 1812 | 473 | 11 | 20543 |
| TF | 375 | 15 | 11 | 3 | 262 | 525 | 417 | 1738 | 34 | 3380 |
| S | 373 | <0.5 | 1 | 3 | 7 | 660 | 164 | 302 | 48 | 1558 |
| SDD | 16 | <0.5 | | <0.5 | <0.5 | 34 | 4 | 63 | 22 | 139 |
| Total | 22344 | 14019 | 1773 | 40639 | 6109 | 16982 | 3422 | 4351 | 9400 | 119039 |

[a]See Table 1.

above-ground wood and 22% is below ground. The phytomass density (an FA average of 4.304 kg C m$^{-2}$) has an evident zonal gradient: from 1.42 kg C m$^{-2}$ in forests running along valleys of rivers of the Arctic basin in the tundra zone to the highest values of 5.40–5.97 kg C m$^{-2}$ in the southern taiga and temperate forests, respectively, and decreasing to the south in the forests of arid steppes and semi-deserts. The share of below-ground phytomass is lower in southern taiga forests (20.1% of the total), increasing to the north and south. The phytomass estimate of FA, provided by the GIS approach, comprises 33 618 Tg C, or +2.3% to the SFA-based estimate (Nilsson et al., 2000). Based on the restored dynamics data, the estimate was 33 665 Tg C (+2.4% to the SFA estimate). Forests have a significant amount of CWD (0.577 and 0.411 kg C m$^{-2}$ on FA and UFA, respectively), which is explained by the wide distribution of disturbances, a large share of unmanaged and uneven-aged forests, particularly in the north, and the slow decomposition rate of dead organics. In total, the Russian FL contain 43 171 Tg C in vegetational organic matter, of which 79.2% are in phytomass, 11.3% in CWD and 9.5% in dead roots (Table 2).

In the 1990s, the top 1 m soil layer of FA contained 119 039 Tg of organic C (Table 3), or 15.59 kg C m$^{-2}$, and 11 423 Tg C are in the on-ground organic layer [the density is 1.50 kg C m$^{-2}$ (Table 4) (Nilsson et al., 2000)]. The corresponding C content for UFA is 15 442 Tg C and 1196 Tg C (the densities are 12.54

Table 4. *On-ground organic layer by main forest-forming species and bio-climatic zones for FA (Tg C)*

| | On-ground organic layer carbon pool by main forest forming species | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Zone[a] | pine | spruce | fir | larch | cedar | birch | aspen | other | shrubs | Total |
| T | 1 | 0.0 | 0.0 | 2 | 0.0 | 1 | 0.0 | 0.0 | 10 | 14 |
| FT | 367 | 525 | 0.0 | 984 | 141 | 141 | 5 | 1 | 70 | 2234 |
| MT | 1138 | 935 | 143 | 2602 | 485 | 771 | 146 | 170 | 808 | 7198 |
| ST | 575 | 292 | 81 | 93 | 46 | 516 | 156 | 24 | 1 | 1784 |
| TF | 15 | 2.0 | 1 | 0.0 | 21 | 32 | 14 | 92 | 2 | 179 |
| S | 2 | 0.0 | 0.0 | 0.0 | 1 | 2 | 6 | 1 | <0.5 | 12 |
| SDD | 1 | 0.0 | 0.0 | 0.0 | 0.0 | <0.5 | 0.0 | 1 | <0.5 | 2 |
| Total | 2099 | 1754 | 225 | 3681 | 694 | 1464 | 327 | 289 | 891 | 11423 |

[a]See Table 1.

and 0.972 kg C m$^{-2}$, respectively). This provides a ratio of soil to vegetation C to be 3:1, which is typical for the boreal forests. The density of C soil compartments are tree species and zone specific. Among the main forest-forming species, spruce has the highest average C content in the on-ground organic layer (2.07 kg C m$^{-2}$) followed by pine (1.63 kg C m$^{-2}$). The lowest value estimated is 1.36 kg C m$^{-2}$ for larch-dominated stands. With respect to zonal aspects, the highest C values in the on-ground organic layer are found in the northern parts of the boreal zone (excluding fir stands, where the highest density is observed in the southern taiga), due to increased areas of wetlands and peat soils on forest lands. A similar picture is observed for C in the 1 m top layer. However, many reasons impact the average densities of forests with different species and zones, for example, the differing share of peat lands and mountains, thereby smoothing the average densities.

### 3.2 *The major C fluxes for 1990*

The NPP on FA is estimated to be 1708 Tg C yr$^{-1}$, of which 49% is located in green parts, 26.4% in above-ground wood and 24.6% in below-ground phytomass (Table 5). The average density (224 g C m$^{-2}$ yr$^{-1}$) for forests (FA) slightly exceeds wetlands (219 g C m$^{-2}$ yr$^{-1}$), but have less compared to grasslands and shrubs [278 g C m$^{-2}$ yr$^{-1}$ (Nilsson et al., 2000)]. A zonal gradient of NPP on FA is strongly expressed, from about 165 g C m$^{-2}$ yr$^{-1}$ in the tundra and forest tundra zones to 347 g C m$^{-2}$ yr$^{-1}$ in temperate

Table 5. *Major carbon fluxes for forest lands in 1990, Tg C yr$^{-1}$*

| | Net Primary Production[b] | | | | Heterotrophic Respiration[c] | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Zone[a] | GR | AGW | BG | Total | (1) | (2) | Average | Disturbances | Lateral | Humification |
| T | 3.1 | 1.0 | 2.4 | 6 | 4 | 4 | 4 | 0.2 | 0.1 | 0.2 |
| FT | 107.6 | 46.4 | 80.5 | 232 | 138 | 148 | 143 | 11 | 4.7 | 14.1 |
| MT | 512.9 | 253.5 | 240.7 | 1004 | 752 | 826 | 789 | 102 | 20.4 | 72.8 |
| ST | 145.0 | 109.4 | 66.1 | 326 | 322 | 280 | 301 | 77 | 6.6 | 21.5 |
| TF | 45.9 | 28.9 | 16.2 | 92 | 74.5 | 74 | 74 | 8 | 1.8 | 5.2 |
| S | 20.0 | 11.6 | 13.4 | 45 | 27 | 28 | 28 | 1 | 0.4 | 1.3 |
| SDD | 2.0 | 0.4 | 0.8 | 3 | 3.5 | 3.2 | 3 | 0.5 | 0.0 | 0.1 |
| Total | 836.5 | 451.2 | 420.1 | 1708 | 1321 | 1363 | 1342 | 200 | 34 | 115 |
| UFA | 164.7 | 19.2 | 116.1 | 300 | | | 171 | 8 | 6 | 11 |
| Grand total | 1001.2 | 470.4 | 536.2 | 2008 | | | 1513 | 208 | 40 | 126 |

[a]See Table 1.
[b]GR, green parts; AGW, above-ground wood; BG, below-ground NPP.
[c](1) estimate by Kurganova (2002); (2) estimate by Nilsson et al. (2000) and Stolbovoi (2002b).

forests, and to 482 g C $m^{-2}$ $yr^{-1}$ in the steppe zone. The latter can be explained by the fact that most of the steppe forests are located in sites that are sufficiently supplied with water, and by an increased share of NPP allocated in the green forest floor. The NPP on UFA is about 11% higher than on FA, which is due to increased primary production (basically at the cost of green parts) after disturbances in the boreal zone, particularly on permafrost (Evdokimenko, 1989; Sedykh, 1997; Zimov et al. 1999).

To validate the results we independently calculated gross growth on forested areas, which is the above-ground woody part of NPP (over bark) allocated in stems. Based on SFA-1993 data and a modeling system developed for dominant species and eco-regions, we estimated the gross growth of Russian forests to be 1,880 × $10^6$ $m^3$ $yr^{-1}$, or 465.3 Tg C $yr^{-1}$, of which net growth comprises 52.2% and mortality 47.8%, respectively (Shvidenko et al., 1997). In order to make this result comparable with the GIS-based estimate (containing the NPP of all above-ground wood), the gross growth of crown wood must be added. Based on derivatives of the phytomass models for branches, the latter has been estimated at 5.2% to gross growth by stem wood. This assessment resulted in an annual above-ground wood NPP of 489.5 Tg C $yr^{-1}$, which exceeds the GIS-based estimate by 8.5%.

The above methods for estimating heterotrophic respiration (HR, C-$CO_2$) on FA resulted in 1363 Tg C $yr^{-1}$ (Nilsson et al., 2000) and 1321 Tg C $yr^{-1}$ (Kurganova, 2002) (Table 5). Both of the estimates include the fluxes caused by decomposition of CWD. While the total estimates are very close (the difference is only −3.1%, compared to the first estimate, which did not take into account winter fluxes), the estimates of the zonal HR densities differ by 10–15%. Analysis of the methods and information bases used lead us to conclude that the uncertainties of these two approaches are similar. Thus, we used the average of the two approaches in our estimation of the C budget for 1990.

The total disturbance (D) fluxes on FA was estimated to be 200 Tg C $yr^{-1}$, of this fires emitted about 37%, biogenic factors (mainly pests and diseases) 37%, harvest (only site effects, excluding forest products) 8% and abiotic factors 14%. The temporal input to the total D fluxes by individual types of D depends on the type of D, the severity of D during the assessment year and previous D history. For example, the ratio between direct fire emission and post-fire biogenic flux was 48:52% in 1990. The formation of charcoal was estimated to be 7 Tg C $yr^{-1}$. Details of these assessments and a discussion on data reliability are presented in Shvidenko et al. (1995a), Shvidenko and Nilsson (2000) and Nilsson et al. (2000).

The total lateral fluxes from ecosystems were estimated to be 40 Tg C $yr^{-1}$ on FL including 11 Tg C $yr^{-1}$ accumulated in the lithosphere, and the flux to the hydrosphere of 29 Tg C $yr^{-1}$. It should be pointed out that the amount of organic C, which is removed from the biological turnover, is big if we consider the geological time scale of this phenomenon, and presents part of the "missing sink". The major summarized fluxes on UFA, similarly calculated to the FA estimates, are presented in Table 5. Some intermediate results are given in Appendix.

Two close independent estimates of fluxes generated by the decay of forest products in 1990 comprise 87 Tg C $yr^{-1}$ (Shvidenko, 1997) and 81 Tg C $yr^{-1}$ (Obersteiner, 1999). The more recent and more detailed estimate is used in further considerations.

The total C uptake by FL from the atmosphere in 1990 (i.e., the average for 1988–1992) is estimated, for the flux-based method, to be −302 Tg C $yr^{-1}$ (in addition to the fluxes presented in Table 5, the C deposition flux of 15 Tg C $yr^{-1}$ was added), and −221 Tg C $yr^{-1}$ if the decay of forest products is included. FA provided about two-thirds of the total C sink. A relatively high C uptake on UFA is explained by increased post-disturbance NPP and intensive restoration processes in ecosystems, particularly in soils. Figure 2 illustrates carbon pools and fluxes for 1990, which were considered in this study.

### *3.3. The forest FCA for 1961–1998*

The major quantitative results of the FCA for 1961–1998 are presented in Table 6. By using the pool-based method, the final estimates in Table 6 are made based on the restored dynamics data, i.e., for the dynamics of FA growing stock with the eliminated bias. In spite of the fact that the differences between the official and restored growing stock did not exceed ±3% during the period considered (excluding the 5-yr period before 1966, where the difference is about 6%), the impact on the accumulation of phytomass is significant (Table 6). The average annual C accumulated in vegetational organics of Russian FL for restored dynamics comprised 210 Tg C $yr^{-1}$, of which 153 Tg C $yr^{-1}$ was accumulated in phytomass and 57 Tg C $yr^{-1}$ in dead wood. The contribution of FA is −227 Tg C $yr^{-1}$ (sink) and UFA 17 Tg C $yr^{-1}$ (source); where the latter is explained by decreased areas of UFA from 152.6 ×

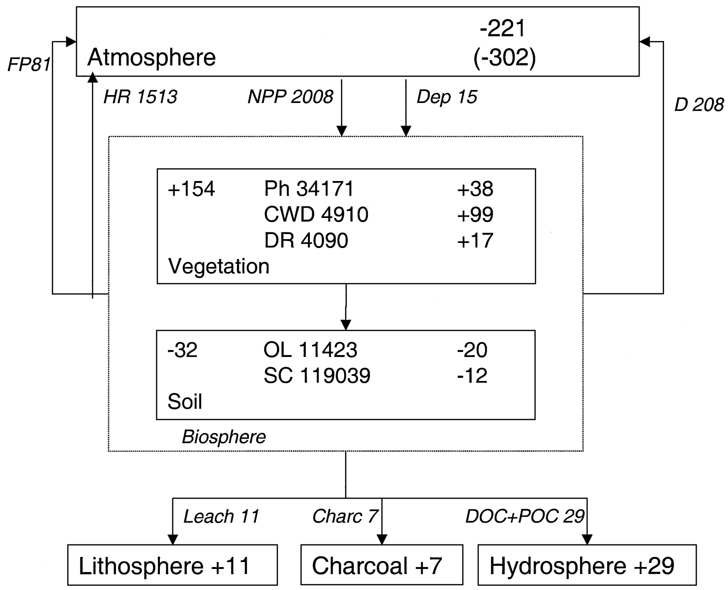


*Fig. 2.* Estimates of major C pools and fluxes for 1990 (annual averages for 1988–1992). Pools are presented in Tg C and fluxes and pool change in Tg C $yr^{-1}$. Changes of pools are derived from the long-term data (Table 6). Abbreviations: FP, forest products; D, disturbances; Ph, phytomass; DR, dead roots; OL, on-ground organic layer; SC, soil carbon of 1 m top layer. For the abbreviations of fluxes see Fig. 1.

$10^6$ to 107.7 × $10^6$ ha. The temporal variability of the sink is large, e.g., the C sequestration in phytomass varies from about 10 to about 280 Tg C $yr^{-1}$ (based on 5-yr averages).

The accumulated C in soil on FL is estimated to be 223 Tg C $yr^{-1}$ by the pool-based method. The accumulation of soil C is mainly explained by the increase of both FA and FL (78.7 × $10^6$ and 33.8 × $10^6$ ha, respectively). Alleviating the severity of disturbance regimes in 1961–1997 and the dynamics of forests led to the prevalence of the humification process over mineralization and lateral transport that resulted in an increased average content of organic C in soils on FL by about 1.8% per unit area for the entire country. This result is within the limits of uncertainty and only characterizes the trends. The average soil C accumulation on FL, defined by eq. (9), crucially depends on the values used for $\beta$, and the estimates significantly vary from C decline to accumulation for $0.006 \geq \beta \geq 0.002$ (see notes to Table 11 in the Appendix). We do not use these results below. In general, this method is very sensitive to varying coefficients of organic matter transformation, which does not allow considering uncertainties of the approach in a consistent way and hinders its application in aggregated soil C assessments.

By using official data of the Russian forest inventory, we change (comparatively with the above analysis for restored dynamics) only one component of the FCA, namely the dynamics of phytomass of FA. This results in a decrease of the C sequestration in phytomass of FL to 64 Tg C $yr^{-1}$, which comprises about 42% of the value estimated based on restored dynamics, and the total C sink, defined by restored dynamics, should in this case be decreased by 80 Tg C $yr^{-1}$. We consider the latter as a biased result and do not use it in the following analysis.

Major fluxes, assessed by the flux-based method, also changed during 1961–1998 (Table 6). The total NPP of FL increased by 6.4%, and the annual average NPP increased from 224 to 229 g C $yr^{-1}$ (or 2.2%). Two opposite major reasons impacted these estimates, namely the age structure and species composition of forests (FA) and the decrease of UFA in the total area of FL. The increase of the average ecosystem HR of 6.5% (from 162 to 173 g C $yr^{-1}$) and the changing structure of land-cover of FL are explained by the amount of dead wood. The total disturbance (D) fluxes, estimated as averages in 5-yr periods, do not vary much (from 167 to 232 Tg C $yr^{-1}$), while the variability of fluxes of individual types of D for separate years can reach 5–10

Table 6. *Dynamics of major indicators of the FCB of Russian forests for 1961–1998*

| Indicators | Dynamics of indicators in 1961–1998 | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1961 | 1966 | 1973 | 1978 | 1983 | 1988 | 1993 | 1998 | 1961–1998 |
| Official inventory data | | | | | | | | | |
| Forested Area (FA), $10^6$ ha | 695.5 | 705.6 | 729.7 | 749.5 | 766.6 | 771.1 | 763.5 | 774.2 | +78.7 |
| Unforested Area (UFA), $10^6$ ha | 152.6 | 149.2 | 132.4 | 122.8 | 113.9 | 113.0 | 123.0 | 107.7 | −44.9 |
| Growing stock, total, $10^9$ $m^3$ | 77.53 | 77.60 | 78.70 | 80.67 | 81.93 | 81.64 | 80.68 | 81.86 | +4.28 |
| European Russia | 16.29 | 16.74 | 17.37 | 18.74 | 19.34 | 20.28 | 21.11 | 22.10 | +5.81 |
| Asian Russia | 61.24 | 60.86 | 61.33 | 61.93 | 62.59 | 61.36 | 59.57 | 59.76 | −1.48 |
| Phytomass on FA (inventory), TgC | 30 662 | 30 831 | 31 486 | 32 430 | 33 078 | 33 104 | 32 862 | 33 362 | +2700 |
| Restored dynamics | | | | | | | | | |
| Growing stock, total, $10^9$ $m^3$ | 71.92 | 73.35 | 76.74 | 79.84 | 82.42 | 82.91 | 82.71 | 84.45 | +12.53 |
| European Russia | 16.79 | 17.27 | 17.98 | 19.49 | 20.44 | 21.34 | 22.40 | 23.67 | +6.88 |
| Asian Russia | 55.13 | 56.48 | 58.76 | 60.35 | 61.98 | 61.57 | 60.31 | 60.78 | +5.65 |
| Growing stock on UFA, $10^9$ $m^3$ | 2.66 | 2.58 | 2.46 | 2.36 | 2.23 | 2.23 | 2.53 | 2.22 | −0.44 |
| Pool-based approach | | | | | | | | | |
| Phytomass on FA, Tg C | 28415 | 29287 | 30687 | 32066 | 33265 | 33611 | 33665 | 34409 | +5994 |
| Phytomass on UFA, Tg C | 1497 | 1418 | 1321 | 1256 | 1177 | 1174 | 1309 | 1150 | −347 |
| Dead wood on FL, Tg C | 4074 | 4234 | 4578 | 4798 | 5082 | 5461 | 6043 | 6189 | +2115 |
| Soil C on FL, PgC | 140.33 | 141.98 | 144.04 | 146.34 | 148.17 | 148.88 | 148.72 | 148.60 | +8272 |
| $\Delta$C for FL, eq. (1), Tg C $yr^{-1}$ | | +526 | +526 | +765 | +648 | +286 | +135 | +122 | +433[b] |
| Flux-based approach | | | | | | | | | |
| NPP[a] on FL, Tg C $yr^{-1}$ | 1908 | 1929 | 1957 | 1989 | 2017 | 2031 | 2015 | 2034 | +126 |
| HR on FL, Tg C $yr^{-1}$ | 1376 | 1400 | 1433 | 1470 | 1495 | 1516 | 1511 | 1524 | +148 |
| Fluxes by D on FL, Tg C $yr^{-1}$ | 232 | 221 | 191 | 182 | 167 | 188 | 227 | 177 | −55 |
| $NBP_1$, eq. (2), Tg C $yr^{-1}$ | −300 | −308 | −331 | −337 | −355 | −327 | −277 | −333 | −322[b] |
| $NBP_2$, eq. (3), Tg C $yr^{-1}$ | −262 | −270 | −294 | −298 | −316 | −287 | −237 | −293 | −283[b] |

[a]NPP values include about 1% of the C wet and dry deposits.
[b]Averages for 1961–1998.

fold. For example, the direct fire emission on FL varied from 20 to 130 Tg C $yr^{-1}$ during the period 1961–2000 (Shvidenko and Nilsson, 2000; Kajii et al., 2002). The estimates of lateral fluxes (of the total amount of 37–40 Tg C $yr^{-1}$, including 29 Tg C $yr^{-1}$ to the hydrosphere and 11 Tg C $yr^{-1}$ to the lithosphere in 1990) are quite stable. For the entire period, Russian FL provided an average C uptake from the atmosphere (i.e., $NBP_1$) of 322 Tg C $yr^{-1}$, with a variation for individual years from about 170 to 450 Tg C $yr^{-1}$. The average $NBP_2$ is estimated to be 283 Tg C $yr^{-1}$ for 1961–1998.

Forest products are not considered in this study and their impact on the forest FCA is not included in the above analysis. Our estimate of the C fluxes caused by the decay of forest products results in a slight increase from 59 to 81 Tg C $yr^{-1}$ during 1961–1990 and decreasing to 72 Tg C $yr^{-1}$ by 1998. The forest products C pool changes in a similar way: increasing from 750 Tg C in 1961 to 1450 Tg C in the early 1990s and decreasing to 1354 Tg C by 1998. This provides an annual C pool increase of about 16 Tg C $yr^{-1}$ for the entire period. However, we should point out that (1) the uncertainty of these data cannot be estimated in any formal way, and (2) these results do not present a full account due to the missing impact of energy and matter consumption during the manufacturing processes of forest products.

### 3.4. *Estimation of uncertainties*

The FCA used in this study seems satisfactorily complete: the values of fluxes caused by recognized unaccounted processes (e.g., C fluxes due to erosion on FL, some types of D, etc.) are small, and the cumulative effect is estimated to be within ±1–3% of the estimated sink. The calculated precision of major C pools for 1990 are rather high, e.g., ±3.4% for forest phytomass and ±3.7% for soil organic C pools (the a priori confidential probability is 0.9). The estimates of

Table 7. *Uncertainties of carbon fluxes between forest ecosystems and the atmosphere in 1990*

| | Uncertainties (%) | | | | Fluxes, Tg C yr$^{-1}$ | |
|---|---|---|---|---|---|---|
| | Precision | | | | | |
| Carbon fluxes, Tg C yr$^{-1}$ | thematic | area | Modification[a] | Total | For 1990 | Uncertainty |
| Heterotrophic respiration | 5.2 | 1.5 | 4.0 | 7.0 | 1513 | ±106 |
| Disturbances - fire | 6.7 | 3.8 | 5.0 | 9.2 | 84 | ±8 |
| - abiotic | 8.9 | 5.8 | 7.0 | 12.7 | 30 | ±4 |
| - harvest | 2.7 | 1.0 | 2.0 | 3.5 | 16 | ±1 |
| - insects | 7.8 | 5.4 | 7.0 | 11.8 | 78 | ±9 |
| Total disturbances | | | | 6.2 | 208 | ±13 |
| Flux to the atmosphere | | | | 6.2 | 1721 | ±107 |
| NPP flux from the atmosphere | 4.2 | 1.0 | 2.0 | 4.7 | 2023 | ±96 |
| Net C exchange | | | | 47.7 | 302 | ±144 |
| Forest products decay | | | 15.0 | 15.0 | 81 | ±12 |
| Net carbon flux | | | | 65.2 | 221 | ±144 |

[a]Based on results of sensitivity analysis and expert estimates of unaccounted processes.

C in CWD and below-ground dead vegetation are less certain (calculated precision is ±7–12%).

The assessed uncertainties of major fluxes are higher. Table 7 presents an example of estimating uncertainties for major fluxes between FL and the atmosphere in 1988–1992 (average for 5 yr). While the most important fluxes, such as NPP and HR, are quite reliably estimated (uncertainties are in the range ±5–7%), the summarized C flux is estimated to have an uncertainty of about ±50%. If we take into account the flux, generated by decomposition of forest products, the uncertainty of net C sequestration by the forest sector increases to about ±65%. Here, we face a well recognized shortcoming of relative indicators of accuracy (in percent), which depends on both the dispersion of estimates and absolute values of averages. The estimates are presented for 5-yr averages, and if we considered the estimates for individual years the uncertainties would be dramatically higher.

The uncertainties of averages for the entire period 1961–1998 are smaller due to additional information derived from intra-series correlations and more observations. Over this period, the live vegetation of Russian FL is estimated to be a C sink of 153 ± 22 Tg C yr$^{-1}$, and the uncertainty is estimated to be ±14.3%. Most uncertain are the estimates on average C change in CWD and soils, comprising ±27% and ±29%, respectively. Assuming that the restored dynamics assessment of growing stock and the intra-series correlations are unbiased, the uncertainty of the overall average sink due to the pool-based method for 1961–1998 of −433 Tg C yr$^{-1}$ is ±16%. The estimate of uncertainties for the flux-based method is about 19%.

However, the danger of hidden biases remains because some assumptions cannot be exhaustively verified by strict formal methods. In some of our calculations (e.g., NPP and HR) we used average data, which were basically collected around the 1980–1990s. It means that our assessment did not take into account such environmental trends as $CO_2$ fertilization, N deposition and regional climate change during the period considered. Global terrestrial ecosystem models simulated an increase of NPP at a rate of 0.1–0.2% yr$^{-1}$ during the last decades (Cramer et al., 2001; Joos et al., 2001; McGuire et al., 2001), but there is doubt that this phenomenon is really observed in natural forests (Davidson and Hirsch, 2001; Scholes and Noble, 2001). Some attempts to find evidence of this in long-term inventory data were still unsuccessful (Caspersen et al., 2000). The increase of NEP is supposed to be less due to increasing soil respiration. Consequently, we could expect that our estimates of NPP and NEP are slightly overestimated for the 1960s to mid 1970s and underestimated for the last decade. Nevertheless, we did not consider it as relevant in this study to introduce any environmentally driven corrections due to the high level of natural and human-induced disturbances, air pollutions, specifics of boreal forests (none of the models takes into account permafrost), etc. The impact of this simplification on the final results is within the limits of calculated uncertainties, although it probably generates some underestimation of the actual sink for the period considered.

## 4. Discussion

"A consistent estimate" depends upon the reliability of estimating uncertainties. Even if we try to follow the explicitly formulated system requirements of a FCA, the available information and methods are sometimes not sufficient to verify some of the assumptions and hypotheses that cannot be avoided in accounting schemes. The intra- and inter-system consistency of the results remains the major evidence that the entire estimation procedure is valid. The results of the two methods (pool- and flux-based methods) characterize the different dimensions of the C account and cannot be directly compared. However, an approximate comparison can be done if we exclude the impact of land-cover changes. We present this analysis for $NBP_2$. Taking into account that the areas of Russian FL increased during the assessment period by about 4%, the average "ecological" C exchange with the atmosphere using the flux-based assessment is estimated to be 283 Tg C $yr^{-1} \times 0.96 = 272$ Tg C $yr^{-1}$ during 1961–1998. The uncertainty of this value cannot be strictly estimated because of the unknown and unaccounted partial replacement of different land cover categories during the period (e.g., FA $\rightarrow$ UFA and vice versa). However, bearing in mind that about 90% of FL did not change its land-cover status in 1961–1998 and that the uncertainty, calculated for the average of 283 Tg C $yr^{-1}$, is $\pm$19% we can conclude that the uncertainty of the average "ecological" C exchange does not exceed $\pm$25%, i.e., the estimate is $272 \pm 68$ Tg C $yr^{-1}$. The estimate for the pool-based calculations is done as follows. The stock of soil organic C increased during 1961–1998 by 8.3 Pg, of which the increase, due to land-cover change, is approximately 5.7 Pg C (we used the soil C density from 1961 for the increased area of $33.8 \times 10^6$ ha). This means that land cover changes increased the average C pool of FL by about 154 Tg C $yr^{-1}$, and the roughly estimated soil pool change due to "ecological" C exchange is $(433–154) \times 0.96 =$ 261 Tg C $yr^{-1}$. Again, based on the uncertainty of the average change of the soil organic C pool of $\pm$29%, we can assume that the uncertainty of the "ecological" C exchange is about $\pm$35%, or 94 Tg C $yr^{-1}$. Finally, the uncertainty of the average of these two estimates of 270 Tg C $yr^{-1}$ is $\pm$82 Tg C $yr^{-1}$, or $\pm$30%. This simple calculation illustrates the similarity of results, as well as the possibility of obtaing meaningful results in assessing the FCB by both pool- and flux-based methods.

A lack of system integrity, different methodologies and information availability generate a large variability of the major intermediate indicators of the FCB of Russian forests that are reported in numerous publications. A number of these publications clearly do not comply with the accounting requirements, which we outlined in the introduction. However, there is an evident temporal convergence of the estimates for major C pools and fluxes of Russian forests for the 1990s, and we present some of the latest examples. The differences of our three independent estimates on phytomass of the Russian FA for 1990 (i.e., official inventory data, restored dynamics and GIS approach) are within the limits of $\pm$2%. Our estimate differs by less than 1% from the arithmetic mean of three detailed assessments of forest phytomass reported by Alexeyev and Birdsey (1994), Isaev et al. (1995) and Isaev and Korovin (1998). All of the estimates published before 1994 differ by 1.5–2 times (for a review, see Shvidenko and Nilsson, 2002). The regional estimates of phytomass, calculated by using different Global Vegetation Models (GVM), overestimate the phytomass inventory stock by 2–2.5 times due to the fact that these models do not take into account disturbances and the succeeding transformation of vegetation, especially in forests.

We did not find any estimates of CWD and dead roots for all Russian forests in the available publications. For forests of the former USSR (of which Russian forests comprised 95%), the C density in the on-ground organic layer plus CWD was estimated to be 3.08 kg C $m^{-2}$ (Turner et al., 1998), which is 1.5 times higher than our estimate. However, the former result was derived from previous publications that used very rough methods and insufficient information (Kolchugina and Vinson, 1993a). A significant number of regional and local assessments of CWD in forests, reported during recent years [e.g., Syrjanen et al. (1999) for the Urals; Krankina et al. (2000) for the Northwest, European Center, East Siberia and Far East Russia; Shorokhova (2000) and Tarasov et al. (2000) for the Leningrad region, etc.) are in line with our data and aggregated estimates. The latest reported average soil C densities are $14.8 \pm 2.2$ and $13.3 \pm 2.2$ kg C $m^{-2}$ for FA and UFA, respectively (Chestnikh et al., 1999), and 11.3 kg C $m^{-2}$ for FA (Stolbovoi, 2002a). While the first estimate is close to our result, the second differs by over one-third. This difference is explained by

Table 8. *Areas (1993) and densities of phytomass and coarse woody debris for unforested areas by bioclimatic zones*

| Bioclimatic zones | Area by UFA categories, $\times 10^6$ ha[a] | | | | | | Phytomass density, kg C $m^{-2}$ | | | | | CWD, kg C $m^{-2}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | BD | SF | HA | GB | UP | Total | BD | SF | HA | GB | UP | |
| Tundra | 0.2 | 3.2 | — | 0.1 | — | 3.5 | 0.45 | 0.72 | — | 0.65 | — | 0.20 |
| Forest tundra, sparse and northern taiga | 8.3 | 46.1 | 0.6 | 0.8 | — | 55.8 | 0.51 | 1.19 | 0.97 | 0.85 | — | 0.45 |
| Middle taiga | 11.7 | 18.4 | 2.6 | 1.2 | 0.6 | 34.5 | 0.67 | 1.37 | 1.21 | 1.09 | 0.43 | 0.85 |
| Southern taiga | 7.1 | 6.9 | 5.0 | 1.6 | 1.2 | 21.8 | 0.71 | 1.44 | 1.35 | 1.18 | 0.67 | 0.96 |
| Temperate forests | 1.9 | 1.8 | 0.5 | 0.3 | 1.2 | 5.7 | 0.52 | 1.26 | 0.96 | 0.90 | 0.67 | 0.77 |
| Steppes | 0.1 | 0.3 | 0.1 | 0.2 | 0.9 | 1.6 | 0.42 | 0.88 | 0.77 | 0.71 | 0.33 | 0.13 |
| Semi-desert and desert | — | 0.1 | — | — | — | 0.1 | — | 0.62 | — | — | — | — |
| Total | 29.3 | 76.8 | 8.8 | 4.2 | 3.9 | 123.1 | 0.62 | 1.24 | 1.25 | 1.06 | 0.55 | 0.65 |

[a]BD, burnt areas and dead stands; SF, sparse forests; HA, unregenerated harvested areas; GB, grassy glades and barrens; UP, unclosed forest plantations; CWD, coarse woody debris.

Table 9. *Dynamics of unforested areas in 1961–1998, $\times 10^6$ ha*

| Categories of UFA | 1961 | 1966 | 1973 | 1978 | 1983 | 1988 | 1993 | 1998 |
|---|---|---|---|---|---|---|---|---|
| Burnt areas and dead stands | 57.6 | 54.6 | 45.3 | 36.5 | 31.3 | 30.6 | 29.3 | 25.6 |
| Including dead stands | 3.0 | 2.9 | 2.9 | 2.7 | 2.3 | 2.3 | 2.2 | 1.9 |
| Natural sparse forests | 60.9 | 60.8 | 59.7 | 59.6 | 59.0 | 60.2 | 61.5 | 61.5 |
| Anthropogenic sparse forest | 10.5 | 5.3 | 6.2 | 4.1 | 4.2 | 4.3 | 15.3 | 9.8 |
| Total sparse forests | 71.4 | 66.1 | 65.9 | 63.7 | 63.2 | 64.5 | 76.9 | 71.3 |
| Unregenerated clearcut areas | 10.7 | 12.6 | 9.9 | 10.4 | 8.9 | 8.8 | 8.8 | 4.9 |
| Grassy glades and barrens | 11.5 | 12.7 | 7.8 | 8.5 | 6.5 | 5.2 | 4.2 | 3.2 |
| Unstocked plantations | 1.5 | 3.2 | 3.5 | 3.7 | 3.9 | 3.9 | 3.9 | 2.7 |
| Total UFA | 152.7 | 149.2 | 132.4 | 122.8 | 113.9 | 113.0 | 123.0 | 107.7 |
| ISDR[a] | 10.6 | 10.3 | 8.4 | 7.2 | 5.8 | 5.5 | 6.5 | 4.9 |

[a]Index of severity of disturbance regimes.

different methods: we used the direct overlay of the forest and soil maps, and the cited publication used a spectrum of soils, which excludes a significant amount of peat soils, covered by forests. The area of the latter on FL could be approximately estimated to be about 200 $\times$ $10^6$ ha (Rojkov et al., 1997). There are other indirect evidences to support this estimate (e.g., Sabo et al., 1981). Previously reported estimates on the forest soil organic C pool vary between 106 and about 300 Pg C (Alexseyev and Birdsey, 1994; Kolchugina and Vinson, 1993b; Krankina and Dixon, 1994). The main reason for this large variability is the inconsistency in the methods used, although other reasons also contributed significantly, e.g., different forest definitions, incomplete data, lack of GIS-technologies, etc. The estimates for other boreal countries also vary. In sampling 934 forest sites in Norway, the median C stock in the whole profile was 22 kg C $m^{-2}$, and for productive forests without organic soils it was 14 kg C $m^{-2}$ (de Wit and Kvinsland, 1999); the estimate for coniferous forests in southern Finland by Liski and Westman (1995) is 11.6 kg C $m^{-2}$; the boreal forests in Canada are estimated to contain 12 kg C $m^{-2}$ in the 0–30 cm layer (Tarnocai, 1998), etc. Consequently, our estimate is in line with other studies for the boreal zone, if both mineral and organic soils covered by forests, are included in the assessment.

The NPP density for 1990 on FA (224 g C $m^{-2}$ $yr^{-1}$) is about 19% less than the average indicated in 12 published estimates on the NPP for the circumpolar boreal forests (e.g., see Bazilevich, 1993; Goldewijk et al., 1994; Schulze et al., 1999). This difference seems reasonable if we take into account the fact that large areas of Russian forests grow in severe

Table 10. *Carbon density and conversion factors by major forest forming species*

| Species and groups of species | Density by age groups, kg C $m^{-2}$ | | | | Conversion factor, Mg C $m^{-3}$ | | | |
|---|---|---|---|---|---|---|---|---|
| | Young | Middle-aged | Immature | Mature | Young | Middle-aged | Immature | Mature |
| European Russia | | | | | | | | |
| Coniferous | 1.66 | 5.83 | 7.60 | 6.01 | 0.3837 | 0.3538 | 0.3674 | 0.4188 |
| Pine | 1.76 | 5.64 | 7.09 | 5.11 | 0.3421 | 0.3304 | 0.3342 | 0.3783 |
| Spruce | 1.49 | 6.13 | 8.16 | 6.34 | 0.4910 | 0.4184 | 0.4063 | 0.4359 |
| Fir | 1.66 | 5.76 | 6.93 | 6.47 | 0.3029 | 0.2884 | 0.2939 | 0.2945 |
| Larch | 3.95 | 6.67 | 6.01 | 6.56 | 0.4584 | 0.4386 | 0.4459 | 0.4578 |
| Cedar | 1.75 | 8.41 | 8.23 | 7.76 | 0.4547 | 0.3969 | 0.3852 | 0.3686 |
| Hard deciduous | 3.41 | 7.49 | 8.46 | 9.79 | 0.6326 | 0.5201 | 0.4884 | 0.5043 |
| Oak | 3.47 | 6.66 | 7.48 | 7.86 | 0.6189 | 0.4838 | 0.4616 | 0.4641 |
| Soft deciduous | 1.30 | 5.18 | 6.86 | 6.77 | 0.4917 | 0.4288 | 0.3878 | 0.3688 |
| Birch | 1.20 | 4.93 | 7.05 | 6.42 | 0.5296 | 0.4230 | 0.3977 | 0.4025 |
| Aspen | 1.34 | 4.23 | 5.76 | 7.18 | 0.3705 | 0.3310 | 0.3141 | 0.2969 |
| Other species | 3.33 | 5.60 | 10.59 | 16.33 | 0.7515 | 0.5444 | 0.5416 | 0.5772 |
| Shrubs | 0.76 | 0.71 | 1.52 | 1.89 | 0.9458 | 0.9437 | 0.9452 | 0.9451 |
| Total | 1.60 | 5.60 | 7.34 | 6.25 | 0.4133 | 0.3972 | 0.3814 | 0.4099 |
| Asian Russia | | | | | | | | |
| Coniferous | 1.27 | 4.55 | 5.72 | 5.39 | 0.4075 | 0.3790 | 0.3728 | 0.4088 |
| Pine | 1.55 | 4.81 | 5.37 | 5.20 | 0.3514 | 0.3310 | 0.3297 | 0.3322 |
| Spruce | 1.44 | 5.25 | 6.90 | 6.25 | 0.4852 | 0.4102 | 0.3990 | 0.4114 |
| Fir | 1.18 | 4.56 | 5.54 | 6.03 | 0.3181 | 0.3023 | 0.3027 | 0.3002 |
| Larch | 0.93 | 3.76 | 5.29 | 5.03 | 0.4745 | 0.4615 | 0.4639 | 0.4698 |
| Cedar | 2.98 | 6.29 | 6.28 | 6.71 | 0.3655 | 0.3271 | 0.3126 | 0.3062 |
| Hard deciduous | 2.63 | 6.23 | 6.42 | 6.25 | 0.6164 | 0.5275 | 0.5190 | 0.5449 |
| Oak | 2.82 | 6.01 | 6.77 | 6.99 | 0.5953 | 0.4961 | 0.4860 | 0.4980 |
| Soft deciduous | 1.05 | 4.13 | 5.90 | 6.64 | 0.5058 | 0.4395 | 0.4125 | 0.4113 |
| Birch | 0.98 | 3.94 | 5.84 | 6.17 | 0.5362 | 0.4393 | 0.4222 | 0.4331 |
| Aspen | 1.13 | 4.14 | 5.89 | 7.94 | 0.4117 | 0.3774 | 0.3524 | 0.3562 |
| Other species | 1.98 | 1.37 | 5.97 | 15.28 | 0.7014 | 0.5044 | 0.5312 | 0.5764 |
| Shrubs | 0.54 | 1.29 | 1.25 | 1.48 | 0.7266 | 0.5841 | 0.5671 | 0.6008 |
| Total | 1.20 | 3.93 | 5.31 | 5.34 | 0.4336 | 0.4031 | 0.3867 | 0.4147 |

climatic conditions and the overestimation of NPP of Russian forests in some previous publications, particularly those that were based on GVMs. However, we could assume that a significant part of numerous (about 1600) Russian measurements underestimated the NPP of fine roots, although the value of this probable bias is not known (Shvidenko et al., 2001b). No estimates on HR and total D fluxes, other than those discussed in this study, were found for FA and FL of the entire country. While no system account for emissions generated by all types of D has been previously reported, a number of estimates for individual types of D differ greatly. Isaev and Korovin (1998) estimated the forest industry emissions to be 73.6 Tg C $yr^{-1}$ for 1993. Two abovementioned independent estimates by IIASA's Forestry Project resulted in 86 and 81 Tg C $yr^{-1}$, respectively, that were generated by forest products decay plus 16 Tg C $yr^{-1}$ for the impacts of logging on the harvested area in 1990. Isaev and Korovin (1998) estimated fire emissions to vary from 35 to 93 Tg C $yr^{-1}$ (including non-forest land of $296.6 \times 10^6$ ha in 1998). Our estimate for the same territory is $122 \pm 18$ Tg C $yr^{-1}$ (Shvidenko and Nilsson, 2000). Previous estimates (e.g., Dixon and Krankina, 1993; Kolchugina and Vinson, 1993a) are 1.5–2.0 times higher. The estimates of emissions caused by other types of D in Russian forests are omitted in publications.

There is considerable inconsistency in the available assessments of summarized C fluxes and annual C sequestration for Russian forests. Recent forest inventory-based estimates of the sequestered C in phytomass (for the early 1990s) vary from 240 Tg C $yr^{-1}$ (Isaev and Korovin, 1998) to 429 Tg C $yr^{-1}$ in above-ground wood (UN, 2000). The first estimate is

Table 11. *Auxiliary analytical expressions used in calculations*

| No. | Equations | Comments |
|---|---|---|
| A1 | $L_{11}$ $(t) = 174.82 - 1.8003t + 0.14865\ t^2$ | CWD1, $0 \leq t \leq 30$ for 1960–1990, $M_{11}(1961) = 3239$ Tg C |
| A2 | $L_{12}$ $(t) = 254.1 + 1.3621t - 08161t^2$ | CWD1, $0 \leq t \leq 8$ for 1990–1998, $M_{12}(1998) = 5001$ Tg C |
| A3 | $L_{21}$ $(t) = 20.68 - 0.2226\ t + 0.0183t^2$ | CWD2, $0 \leq t \leq 30$ for 1960–1990, $M_{21}$ (1961) = 304 Tg C |
| A4 | $L_{22}$ $(t) = 31.40 + 0.1684t - 0.1009t^2$ | CWD2, $0 \leq t \leq 8$ for 1990–1998, $M_{22}$ (1998) = 381 Tg C |
| A5 | $L_{31}$ $(t) = 338 + 1.9t$ | RL, $0 \leq t \leq 30$ for 1960–1990, $M_{31}$ (1961) = 3778 Tg C |
| A6 | $L_{32}$ $(t) = 395 - 3.25t$ | RL, $0 \leq t \leq 8$ for 1990–1998, $M_{32}$ (1998) = 4113 Tg C |
| A7 | $L_{41}$ $(t) = 740 + 5t$ | GPL1, $0 \leq t \leq 30$ for 1960–1990, $M_{41}(1961) = 1222$ Tg C |
| A8 | $L_{42}$ $(t) = 890 + 0.6t$ | GPL2, $0 \leq t \leq 8$ for 1990–1998, $M_{42}$ (1998 = 1490 Tg C |
| A9 | $H_1 = 150 + t$ | Input to soil C compartment, $0 \leq t \leq 30$ for 1960–1990 |
| A10 | $H_2 = 180 + t$ | Input to soil C compartment, $0 \leq t \leq 8$ for 1990–1998 |
| A11 | $\Delta_{\mathrm{OGL}} = 10.03 - 0.18x + 0.05667x^2$ | Correction for soil C (1 m layer) transformation, $x$ = ISDR |
| A12 | $\Delta_{\mathrm{1MTL}} = 0.836 + 0.120x$ | Correction for on-ground organic C transformation, $x$ = ISDR |
| A13 | $K_{\mathrm{NSF}} = 1.029 - 0.0052x$ | A13–A16 give corrections for C of 1 m top layer of UFA; see notes below |
| A14 | $K_{\mathrm{ASF}} = 1.02 - 0.00333x$ | |
| A15 | $K_{\mathrm{BA}} = 1.03 - 0.0048x$ | |
| A16 | $K_{\mathrm{GG}} = 1.025 - 0.0046x$ | |
| A17 | $K_{\mathrm{ONSF}} = 1.135 - 0.016x$ | A17–A20 give corrections for C of on-ground organic layer of UFA; see notes below |
| A18 | $K_{\mathrm{OASF}} = 1.044 - 0.0074$ | |
| A19 | $K_{\mathrm{OBA}} = 1.2 - 0.036x$ | |
| A20 | $K_{\mathrm{OGG}} = 1.23 - 0.028x$ | |

Equations A1–A8 present the input of litter to decomposition pools during 1960–1998 [eqs. (8)]: two pools of coarse woody debris (CWD1 and CWD2), root litter (RL) and green part litter (GP). All equations are calculated separately for two time periods, 1960–1990 and 1990–1998. The corresponding pools were calculated based on the integral of eq. (8), which for $L = a + bt + ct^2$ and decomposition rate $\alpha$ is:
$M = \mathrm{e}^{-\alpha t}(M_0 - \frac{a}{\alpha} + \frac{b}{\alpha^2} - \frac{2c}{\alpha^3}) + (\frac{a}{\alpha} - \frac{b}{\alpha^2} + \frac{2c}{\alpha^3}) + (\frac{b}{\alpha} - \frac{2c}{\alpha^2})t + \frac{c}{\alpha^2}t^2$. Equations A9 and A10 present the input to the soil compartment for forest land [the second component of eq. (9) for $1 - \chi = 0.12$)]. The amount of humified material by eq. A9 and A10 is in line with the results calculated by two other methods, which used aggregated zonal data (Table 13). However, the integral of these equations is very sensitive to $\beta$ due to the high value of the soil C pool, which limits practical applications of eq. (9) for only estimating some general tendencies of soil C dynamics. For the entire forest land, the soil C dynamics, close to equilibrium, is observed only for $\beta$ of about 0.001, which is probably explained by the significant amount of peat soil on FL; in some regional simulations for mineral soils the equilibrium was reached for $\beta$ of about 0.002–0.003. Available empirical information does not allow $\beta$ to be defined with acceptable accuracy. In addition, eq. (9) does not consider the impact of disturbances. For these reasons, we did not use the results calculated by eq. (9) in the final estimates of this paper. Equations A11 and A12 present the decrease of soil C content due to anthropogenic impact and natural disturbances ($\Delta_{\mathrm{OGL}}$ and $\Delta_{\mathrm{1MTL}}$ are expressed in percent to average value calculated for undisturbed soils). In eqs. A13–A20 $x$ = ISDR, $4.5 \leq$ ISDR $\leq 11$. These equations present the corrections of soil C in the 1 m top layer (A12–A16) and in on-ground organic layer (A17–A20) for unforested areas. Abbreviations: NSF/ASF = natural/anthropogenic sparse forests; BA = burnt areas; GG = grassy glades and barrens. The actual C stock is calculated as the product of corresponding coefficients and empirically defined values for 1990 (kg C m$^{-2}$): ASF 14.80, BA 14.65, dead stands (DS) 16.04, HA 14.80, UFP 12.80 for the top 1 m layer and ASF 1.197, BA 0.972, DS 1.091, HA 1.091, UFP 0.897 for the on-ground organic layer. The equations of Table 11 were calculated using regional measurements, different indirect sources, and professional judgments of regional experts, which is why we do not present formal statistic characteristics of the equations.

1.5 times higher than our estimate for 1961–1998 and about 20 times higher than our average estimate for the period 1988–1993. The reason for this inconsistency lies in the fact that the methods used by Isaev and Korovin do not relate to any definite year or period. The second assessment includes a serious error made in estimating natural losses in Russian forests (for details, see Shvidenko and Nilsson, 2002).

Based on AVHRR NOVA data, Myneni et al. (2001) estimated the C pool and sink in live woody biomass of Russian forests for 1995–1999 to be 24.39 Pg and 283.6 Tg C m$^{-2}$ yr$^{-1}$, respectively. Our estimates for

Table 12. *Coefficients of organic matter decomposition*

| Bioclimatic zones | Coefficients $a_{ij}$: Green parts | CWD2 (medium) | CWD1 (slow) | Roots | $T_{0.95}$, yr: Green parts | CWD-medium | CWD-slow | Roots |
|---|---|---|---|---|---|---|---|---|
| Tundra | 0.054 | 0.028 | 0.011 | 0.042 | 55 | 107 | 272 | 71 |
| Forest tundra, sparse and northern taiga | 0.145 | 0.043 | 0.024 | 0.066 | 21 | 70 | 125 | 45 |
| Middle taiga | 0.40 | 0.067 | 0.038 | 0.091 | 7.5 | 45 | 79 | 33 |
| Southern taiga | 0.75 | 0.093 | 0.048 | 0.15 | 4 | 32 | 62 | 20 |
| Temperate forests | 1.5 | 0.134 | 0.074 | 0.21 | 2 | 22 | 40 | 14 |
| Steppes | 2.7 | 0.188 | 0.120 | 0.27 | 1.1 | 16 | 25 | 11 |
| Semi-desert and desert | 4.5 | | 0.175 | 0.33 | 0.7 | | 17 | 7 |
| Total | 0.600 | 0.0680 | 0.0387 | 0.0920 | 5 | 44 | 77 | 33 |

Table 13. *Estimates of fluxes in 1990*

| Bioclimatc zones | Litter, Tg C: Total | Including AG | Including BG | Humification (1): % of litter | Humification (1): Tg C | Humification (2): Mg ha$^{-1}$ | Humification (2): Tg C | Lateral fluxes, Tg C yr$^{-1}$, to: H | L | Total |
|---|---|---|---|---|---|---|---|---|---|---|
| Tundra | 5 | 3 | 2 | 5 | 0.2 | 0.06 | 0.2 | 0.1 | — | 0.1 |
| Forest tundra, northern and sparse taiga | 210 | 138 | 72 | 6 | 12.6 | 0.10 | 14.1 | 3.3 | 1.4 | 4.7 |
| Middle taiga | 881 | 670 | 211 | 8 | 70.5 | 0.16 | 72.8 | 15.2 | 5.7 | 20.9 |
| Southern taiga | 274 | 196 | 78 | 8.5 | 23.3 | 0.17 | 21.5 | 4.8 | 1.8 | 6.6 |
| Zone of temperate forests | 83 | 68 | 15 | 9 | 7.5 | 0.19 | 5.2 | 1.2 | 0.4 | 1.6 |
| Steppe | 38 | 27 | 11 | 9.5 | 3.6 | 0.16 | 1.3 | 0.1 | 0.1 | 0.2 |
| Semidesert and desert | 1 | 1 | — | 10 | 0.1 | 0.05 | 0.1 | — | — | — |
| Total for FA | 1492 | 1103 | 389 | | 118 | | 115 | 25 | 9 | 34 |
| Unforested areas | 87 | | | | 12 | | 11 | 4 | 2 | 6 |
| Total | 1559 | | | | 130 | | 126 | 29 | 11 | 40 |

Humifications (1) and (2) are defined based on published zonal aggregations.

1993–1998 based on the official Russian long-term inventory data are 28.95 Pg C for the C pool and 85.2 Tg C m$^{-2}$ yr$^{-1}$ (3.3 times less) for the C sink. The restored dynamics picture is a little different: 29.78 Pg C (+22.0%) for the C pool and 125.4 Tg C m$^{-2}$ yr$^{-1}$ (or 2.3 times less) for the C sink in woody phytomass (for initial data on these calculations, see Shvidenko and Nilsson, 2002). There are different reasons for these inconsistencies. The underestimation of the C pool by Myneni et al. (2001) is caused by the rough resolution of the remote sensing tools (pixel size 64 km$^2$) resulting in missed large areas of forests (FA), particularly in regions with fragmented forest cover: FA were estimated by Myneni et al. to be 642.2 × 10$^6$ ha compared to the official inventory area of (763.5–774.2) × 10$^6$ ha during this period. The average densities of the C pools are very close: 3.80 kg C m$^{-2}$ by Myneni et al., 3.77 kg C m$^{-2}$ based on official inventory data and 3.87 kg C m$^{-2}$ for the restored dynamics. Probably, the difference in the C sink is partially caused by the not completely compatible period of the comparison. However, we suppose that the major reason stems from the fact that the accumulated values of the Normalized Difference Vegetation Index (NDVI) in the long-term should cause a bias comparatively with the real dynamics of live woody biomass due to the missing impact of disturbances. This conclusion is supported as follows: if we assume that the Myneni et al. (2001) estimate of 283.6 Tg C m$^{-2}$ yr$^{-1}$ is true, the consequence is that the growing stock volume in

Table 14. *Extent of major types of disturbances in Russian forests ($\times 10^6$ ha, average for 1988–1992)*

| | Wild fire by types | | | | Biotic factors | | | |
|---|---|---|---|---|---|---|---|---|
| | | | | | | Including | | |
| Bioclimatic zones | Crown fire | OGF | Peat fire | Total | Total | Insects | Diseases | Harvest |
| Tundra | — | 0.03 | 0.03 | 0.06 | — | — | | — |
| Forest tundra, sparse and northern taiga | 0.06 | 0.39 | 0.15 | 0.60 | 1.12 | 0.05 | 0.02 | 0.05 |
| Middle taiga | 0.12 | 0.62 | 0.11 | 0.85 | 0.35 | 0.57 | 0.31 | 0.74 |
| Southern taiga | 0.05 | 0.47 | 0.05 | 0.57 | 1.19 | 0.92 | 0.21 | 0.51 |
| Temperate forests | 0.01 | 0.10 | 0.01 | 0.12 | 0.90 | 0.31 | 0.37 | 0.34 |
| Steppes | — | 0.03 | — | 0.03 | 0.15 | 0.06 | 0.04 | 0.01 |
| Semi-desert and desert | — | — | — | — | 0.01 | — | | — |
| Total | 0.24 | 1.64 | 0.35 | 2.23 | 3.72 | 1.91 | 0.95 | 1.65 |

OGF, On-ground fire; peat fires include $0.012 \times 10^6$ ha underground fires.

Russian forests should increase by about $5 \times 10^9$ $m^3$ during the period considered. In reality, this increase is $1.00 \times 10^9$ $m^3$ according to official inventory data and $+1.74 \times 10^9$ $m^3$ based on our restored dynamics.

The recent results from a group of inverse modeling (Peylin et al., 2002; Schimel et al., 2001) indicate that the average $NBP_1$ for Eurasia (northwards from 30°N) is 39 g C $m^{-2}$ $yr^{-1}$ per land unit area and 46 g C $m^{-2}$ $yr^{-1}$ for vegetative land during 1990–1994. These publications estimated the range of NPP for vegetative lands to be 222–417 g C $m^{-2}$ $yr^{-1}$ (based on six different GVMs). This means that the share of net flux to the atmosphere is from 11 to 21% of the NPP, and would be even more if the abovementioned overestimation of NPP by GVMs is taken into account. By using the flux-based method, we estimate the average $NBP_1$ for 1988–1993 to be 32.9 g C $m^{-2}$ $yr^{-1}$ for Russian FL and 23.8 g C $m^{-2}$ $yr^{-1}$ if fluxes caused by forest products decay is included (81 Tg C $yr^{-1}$ for this period). These two estimates comprise 14.5% and 10.5%, respectively, of the NPP for this period.

Two other attempts were made, based on inverse modeling, taking a number of transport models into account [14 models (Maksyutov et al., 2001) and 16 models (Gurney et al., 2002)]. These publications report a net sink (for all land cover classes) of boreal Asia for 1992–1996 to be 0.4–0.6 Pg C $yr^{-1}$, and for Europe about 0.5 Pg C $yr^{-1}$. The uncertainties of these estimates are high and generally exceed 0.5 Pg C $yr^{-1}$. The uncertainties and different assessment regions hinder any direct comparison with our results. However, it can be concluded that recent inverse model regional estimates of the C sink are in line with the major conclusions of this study, if we take into account that vegetation other than forest land classes additionally contribute to the total C sink of terrestrial vegetation (Nilsson et al., 2000). The available results of direct measurements of net ecosystem exchange (NEE) in Russian forests are still very spatially and temporarily limited (Hollinger et al., 1998; Running et al., 1999; Schulze et al., 1999; Knohl et al., 2002), and could only be used for indicating probability ranges. In general, they blend with the quantification of the Russian forest FCB considered in this study.

## 5. Conclusion

Current analysis illustrates the possibilities and difficulties of the FCA at the national scale in two fundamental forms: the pool- and flux-based approaches. We showed that the combination of these two methods within a systems approach could be sufficient to reliably estimate the FCB for large territories. In spite of the high level of disturbances in Russian forests, the Russian FL served as a net sink of about 0.3 Pg C $m^{-2}$ $yr^{-1}$ during the period 1961–1998.

Bearing in mind that only the FCB completely corresponds to eventual goals of the UN FCCC, we suppose that the current international negotiation process will result in the transition from partial to full carbon account, and the methodology considered above could be useful for practical applications in the post-Kyoto world.

The study's results designated some ways in which the approach could be improved. (1) Increased assessment periods of the FCB decrease the uncertainties for any period of the assessed time interval. This supports the strategy of providing continuous reliable monitoring of the FCB that employs national information on greenhouse gas related systems, which combine multi-layer geographical information systems at appropriate scales, long-term forest inventory data, multi-sensor remote sensing, and regional ecological modeling. (2) Such a system could cover the other stakeholders' interests in ecological, environmental and resource monitoring as well as supply information for implementation of the sustainable forest management paradigm. (3) Soil processes cause the largest uncertainties in the FCB and should be a subject of special interests in further research. (4). There is a special need to develop new approaches for estimating the uncertainties.

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## 7. Appendix

Tables 8–14 provide additional detailed information of the estimations given in the text.

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